A comparison of watershed nitrogen loading and watershed nitrogen exports from on-site wastewater treatment systems and centralized sewer systems in the North Carolina Coastal Plain

by

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July 2013

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Elevated nitrogen (N) concentrations in groundwater may cause adverse effects to adjacent surface water bodies. In North Carolina, half of the residences use on-site wastewater treatment systems (OWTS), yet they are typically not regulated beyond the permitting process. The overall goal of this study was to determine if OWTS affect groundwater N loading and surface water N export at the watershed scale. Eight sub-watersheds were monitored monthly for physical and chemical parameters in Greenville, NC. Four watersheds used OWTS and four watersheds used a centralized sewer system (CSS) that transported wastewater from these watersheds and discharged the treated wastewater to the Tar River. To evaluate the effects of wastewater management on groundwater quality, groundwater was monitored at 10 residential sites, five in an OWTS watershed and five in a CSS watershed. Groundwater samples were collected quarterly for a year (August 2011 to August 2012) and analyzed for dissolved N species (ammonium, nitrate + nitrite, and dissolved organic N) and chloride. Surface water samples were collected monthly and analyzed for the same physical and chemical parameters, including turbidity and particulate N. Groundwater and surface water samples were collected and sent to the Stable Isotope Facility at UC Davis for $\delta^{15}N$ and $\delta^{18}O$ of nitrate analysis. Groundwater TDN concentrations and loads at OWTS sites were significantly greater than at CSS sites, with mean TDN concentrations in OWTS groundwater up to two times greater and loads up to five times greater than CSS TDN concentrations and loads. Groundwater and surface water stable isotopes, ^{15}N and ^{18}O in nitrate, suggested that N sources in OWTS watersheds were wastewater derived, while CSS sources were fertilizer derived. Mean total nitrogen (TN) concentrations in surface water at OWTS watersheds were approximately two times greater than for CSS watersheds during baseflow and storm conditions. Streams draining OWTS watersheds exported significantly greater TN masses than CSS watersheds. Assuming average measured OWTS loads to the soil were representative of each residence in OWTS watersheds, on average OWTS watersheds were found to attenuate 81% (± 14%) of OWTS TN loads to the soil prior to TN export from the watershed. The results from this study illustrate a need for inclusion among nutrient management strategies by North Carolina Department Environment and Natural Resources and other state, federal, and international agencies.

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ACKNOWLEDGMENTS

I would like to thank North Carolina Department of Environment and Natural Resources (319 Grant) for funding this research, without their support it would not have been possible to meet our study objectives and goals. Furthermore, thank you Michael O'Driscoll and Charles Humphrey for not only being outstanding advisors, but for also being a friend. I cherish the relationship we have built over the years; I would not have been successful if not for your unwavering support, guidance, and knowledge. Thank you Alex Manda and Richard Spruill, your counsel provided me with the tools for success. Thank you Jim Watson and John Woods not only for the assistance you provided me in the field, but for all of your hard efforts. This project would not have been possible without the help of the anonymous homeowners who volunteered their homes. I also wish to thank Eliot Anderson-Evans, AJ Finley III, Katie Supler Del Rosario, Jonathan Harris, Matthew Smith, and Sarah Hardison for their tireless efforts both in the field and in the laboratory; my life would have been much more complicated without you. Colleen Rochelle, Laura McKenna, and Central Environmental Laboratory staff thanks for all of your hard work, especially Colleen and Laura; I am glad the late night hand syringe filtering days are behind us. I would also like to acknowledge East Carolina University Geological Sciences, Pitt County Planning, City of Greenville, and Collin Ferebee.

Last, but certainly not least, I lovingly thank Hannah Postma for her stoic fortitude. Her eternal love, advice, and patience provided with me the drive to persevere in the most difficult of times. I am forever grateful.

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LIST OF SYMBOLS OR ABBREVIATIONS

BELL – Bell Branch
CEL- Central Environmental Laboratory
CHOK – cherry oaks
Cl^- – chloride
CSS – centralized sewer system
DIN – dissolved inorganic nitrogen
DKN – dissolved kjeldahl nitrogen
DON – dissolved organic nitrogen
EP-1 – Eastern Pines sub-basin 1
EP-O – Eastern Pines outlet
FT-1 – Firetower sub-basin 1
FT-2 – Firetower sub-basin 2
FT-O – Firetower outlet
GUC – Greenville Utilities Corporation
MHB – Meeting House Branch
MILL – mill branch
N – nitrogen
NC DENR – North Carolina Department of Environment and Natural Resources
NH_4^+ – ammonium
NO_2^- – nitrite
NO_3^- – nitrate
O - oxygen
OWTS – on-site wastewater treatment systems
PN – particulate nitrogen
TDN – total dissolved nitrogen
TN – total nitrogen
WWTP – wastewater treatment plant

PREFACE

This thesis is composed of 5 chapters that focus on: Introduction, Methodologies, Groundwater Analysis, Surface Water Analysis, and Management Implications. Chapters 1 and 2 are comprehensive chapters to discuss background information pertinent to this study (Introduction) and the necessary tools and procedures to meet study objectives and goals (Methodologies). Chapters 3 and 4 are designed to be manuscripts for publication. Therefore, there is redundancy within the introductory and methodology information between Chapters 3 and 4 and Chapters 1 and 2. Chapter 5 synthesizes data from Chapters 3 and 4 and provides suggestions for how these data may help watershed managers and regulators to deal with nitrogen loading in nutrient-sensitive watersheds.

CHAPTER 1: INTRODUCTION

Excess nitrogen (N) concentrations pose a significant risk to both surface water and groundwater. Human activities have doubled the amount of available reactive N on a global scale (Jordan and Weller, 1996; Vitousek *et al*., 1997; Asner *et al*., 1997). Over the last two centuries, anthropogenic activities have increased global N, and inputs have accelerated since the 1950s (UNEP, 2005). Elevated N concentrations may adversely affect human health and/or aquatic habitats.

Water containing nitrate-N above the maximum contaminant level for drinking water of 10 mg/L (US EPA, 2002) may cause adverse health effects to infants and expectant mothers (Baird, 1997). In addition, total nitrogen (TN) concentrations of as low as 1 mg/L can promote eutrophication in surface water bodies (Osmond *et al.*, 2003). In a recent study, Dodds et al. (2009) found elevated nutrient concentrations in 90% of streams in 12 of 14 ecoregions in the US and estimated annual costs of eutrophication of US freshwaters at approximately 2.2 billion dollars annually. Intensive agriculture, fossil fuel combustion, extensive cultivation of leguminous crops, and wastewater discharge can cause significant additional concentrations of N in terrestrial and aquatic ecosystems (Smil, 2001). On-site wastewater treatment systems (OWTS) are a potential source of nutrient discharges to groundwater and surface water.

On-Site Wastewater Treatment Systems

OWTS collect, treat, and release wastewater to the subsurface. In the United States, OWTS serve 26 million homes, businesses, and recreational facilities (US EPA, 2002). Domestic wastewaters contain elevated concentrations of TN ranging from 26-75 mg/L (US EPA, 2002), although wastewater may contain TN values >75 mg/L (WERF, 2007). These wastewaters are potential sources of nutrients to surface water and groundwater systems. For example, in the northeastern US, Driscoll *et al*. (2003) found wastewater effluent can contribute 36% to 81% of N loadings to estuaries.

There are many types of OWTS technology in use, but conventional OWTS are most common. Conventional OWTS typically utilize two forms of wastewater treatment: primary and secondary. Primary treatment occurs in the tank where three layers of different densities develop: scum, liquid, and sludge layers (NC DHHS, 2007). Primary treatment facilitates the stratification of waste to develop a prominent liquid effluent layer and drives the conversion of organic N to ammonium (NH_4^+). The scum layer consists of fats, oils, and greases that accumulate above the liquid layer. The sludge layer consists of the heavier solids that settle from the liquid effluent. The liquid layer is the liquid wastewater that moves through the tank and enters the drainfield. Secondary treatment occurs during the percolation of wastewater into the subsurface. Aerobic conditions in the soil beneath the drainfield trenches helps facilitate oxidation of ammonium and die off of anaerobic pathogens in wastewater effluent. Ideally, wastewater encounters an anaerobic, carbon-rich environment before discharging to groundwater, thus, facilitating TN reductions through the nitrification-denitrification processes (Fig. 1). If primary and secondary treatments are not enough to reduce TN concentrations adequately, tertiary treatment may be implemented (NC DHHS, 2007).

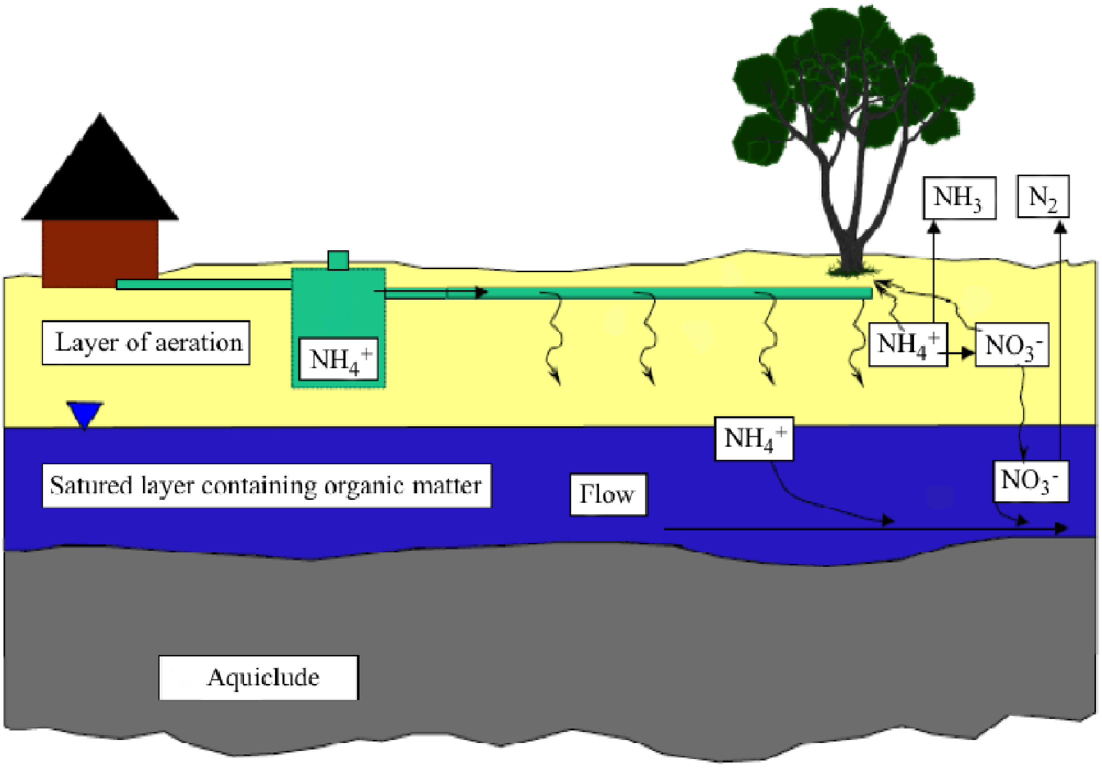


Figure 1. Nitrogen cycling from OWTS to surrounding soil profile. This scenario is common in the North Carolina Coastal Plain. Studied sites are underlain by unconsolidated sediments with a lower permeability unit (aquiclude) beneath which facilitates lateral flow to adjacent surface waters. Ideally, wastewater migrates from tank to subsurface layers to facilitate nitrification-denitrification reactions, thereby attenuating TDN concentrations (Modified from Cardona, 2006).

North Carolina has one of the largest rural, non-farm per capita populations in the country (Hoover *et al*., 1998). People residing in rural areas predominantly rely on OWTS as a means of wastewater treatment. Approximately 50% of North Carolina residents use OWTS as their primary source of wastewater treatment (Pradhan *et al*., 2007). Pradhan *et al*. (2007) conducted a study to estimate the amount of N loading to major North Carolina river basins caused by OWTS. The Neuse and Cape Fear River Basins had some of the highest densities of OWTS in the state. The Cape Fear River Basin had the largest N loading, followed by the Neuse Basin (a portion of the current study area resides in this basin). The White Oak Basin had the greatest OWTS density in the state (15 OWTS/km^2). The Tar-Pamlico River Basin (a portion of the current study area resides in this basin) had a low OWTS density, but had the 3rd highest estimated OWTS N load in NC (Pradhan *et al*., 2007). This study estimated OWTS N loads to watersheds assuming no treatment occurred, which provides an estimate for the maximum possible N load to watersheds from OWTS use. This may not be the case in many settings, where denitrification or biological uptake may reduce N loads prior to surface water discharge. More information is needed on OWTS N loss/attenuation in soils and surficial aquifers to help better quantify the OWTS N inputs to surface waters.

Water Quality Problems from Excess Nitrogen

In North Carolina, water quality degradation has been documented due to high concentrations of nutrients from point and non-point sources of pollution (Fear *et al*., 2004; NC DENR, 2010). North Carolina Department of Environment and Natural Resources (NC DENR) devised Nutrient Sensitive Waters Management Strategies for the Neuse and Tar-Pamlico River Basins, first in place in 1994, (15A NCAC 2B .0232-.0240; 15A NCAC 2B .0255-.0259 and 15A NCAC 2B .0263-.0272) to reduce fish kills, nutrient pollution (Reay, 2004), and eutrophication (NC DENR, 2009; 2010; Humphrey, 2010). Further explanation of the aforementioned rules is available at NC OAH (2013). The strategies required implementation of agricultural best management practices and nutrient management training for fertilizer use, caps on nutrient discharges from centralized sewer system (CSS), and engineered stormwater runoff controls for new developments (NC DENR, 2009; 2010). However, reduction of nutrient contributions to surface waters by OWTS was not among the strategies. The current study was designed to help determine if OWTS in the Neuse and Tar-Pamlico watersheds contribute N loads that affect surface water quality at the watershed-scale. This information can help decision-makers determine if OWTS warrant inclusion in nutrient-sensitive watershed planning efforts.

Nitrogen Cycling by On-site Wastewater Systems (OWTS)

The anaerobic decomposition of organic matter in wastewater occurs within the septic tank (Wilhelm *et al*., 1994b). Septic wastewater is dominated by NH_4^+ and organic N (NC DENR, 2003), and may contain low concentrations of NO_3^- (Wilhelm *et al*., 1994a). After discharge from the tank, the effluent migrates to the drainfield. As the effluent migrates through the drainfield and into the subsurface, aerobic oxidation of organic carbon to carbon dioxide and oxidation of NH_4^+ to NO_3^- occurs in the unsaturated zone beneath the drainfield trenches (nitrification) (Fig. 1). The oxidation of NH_4^+ requires nitrifying bacteria, oxygen, and produces NO_3^- and increases acidity (Wilhelm *et al*., 1996; Pradhan *et al*., 2007). Aquifers with high permeability can contain concentrated plumes of NO_3^- from sources such as OWTS (Wilhelm *et al*., 1996), which may travel long distances (up to 170 m) throughout the subsurface (Robertson *et al*., 1991). The reduction of NO_3^- requires organic carbon, denitrifying bacteria, anaerobic environment, and presence of NO_3^- (Wilhelm *et al*., 1996). In subsurface environments where nitrification and denitrification do not occur, N-transport to local waterways may become significant.

Wastewater NO_3^- plumes have been documented to receive limited transformations and dilution in aerobic, unconfined sand aquifers (Robertson *et al*., 1991; Harman *et al*., 1996; Ptacek, 1998) and in limestone aquifers (Keeney, 1986; Dillon *et al*., 1999). However, NO_3^- plumes may potentially exhibit rapid reductions over about 3 m, if denitrification hotspots exist along the plume flowpath (Groffman *et al*., 2009). In areas where wastewater converges with carbon-enriched media, Robertson *et al*. (1991) observed large decreases in dissolved N concentrations along shorelines with carbon-enriched deposits. Additionally, organic carbon and pyrite-rich aquifers may exhibit NO_3^- removal in anaerobic plumes (Pederson *et al*., 1991;

Korom, 1992; Postma *et al*., 1992; Robertson and Cherry, 1992; Aravena and Robertson, 1998). Furthermore, riparian wetlands, lakes and reservoirs, and headwater streams may act as NO_3^- sinks (Kellogg *et al*., 2010).

NH_4^+ is another N species that can persist in the layer of aeration and surficial aquifer depending on site conditions. Typically in sandy, aerobic sediments, NH_4^+ transforms (nitrification) almost entirely to NO_3^- (Robertson *et al*., 1991; Harman *et al*., 1996; Ptacek, 1998). However, if the trench bottom resides within the water table throughout the year or separation distances are inadequate, NH_4^+ may not adequately nitrify (Cogger and Carlile, 1984; Cardona, 2006; Humphrey *et al.*, 2010; Humphrey *et al.*, 2012). While some NH_4^+ may adsorb to soil, once the cation exchange capacity of the soil is reached, NH_4^+can migrate through the shallow aquifer towards nearby surface waters (Carlile *et al*., 1981; Corbett, 2002; Humphrey *et al*., 2012). NH_4^+ is a plant available form of N and can contribute to eutrophication in surface waters.

Nitrogen Cycling by Centralized Sewer Systems (CSS)

CSS utilize a multitude of differing treatment processes. However, the following explanation relates specifically to the Greenville Utilities Corporation wastewater treatment plant (GUC WWTP) (Tar River-Greenville, NC) and those using similar technologies. The GUC WWTP utilizes a 3-stage process: primary, secondary, and tertiary treatment. During primary treatment, wastewater passes through filtration screens to remove foreign objects. The filtered wastewater moves to an aeration tank, where microbes grow and nitrify wastewater. The aeration tanks are shut off to create an anaerobic condition and through use of biological N removal the NO_3^--rich effluent denitrifies. Wastewater migrates to a secondary clarification tank where microbes and solid wastes form particles and settle out. Some residue can be reused in the aeration tank by remixing with oxygen gas. After the solids settle, wastewater percolates through a deep-bed sand filter. Harmful microbes are killed off through ultraviolet disinfection, chlorination, and deep-bed sand filters. In the current study, the treated water is discharged to the Tar River via a drainage canal (GUC, 2012).

Anaerobic Ammonium Oxidation

Anaerobic ammonium oxidation (anammox) is another potential treatment process. This occurs when NH_4^+ oxidizes to N_2 gas in the presence of carbon dioxide and low oxygen conditions (Jetten *et al*., 2009). Anammox has been primarily researched in marine and freshwater (lakes) systems. Anammox in soil ecosystems has not yet been thoroughly investigated (Jetten *et al*., 2009). Despite this, Penton *et al*., (2006) showed that anammox sequences could be retrieved from several soil samples and Clark *et al.* (2008) showed the same for groundwater.

Influence of Centralized Sewer Systems on Water Resources

In a review of WWTP treatment performance, the US EPA found that TN in discharge from CSS systems can range from approximately 0.13 – 13.6 mg/L based on the treatment technology (US EPA, 2008). Based on sampling from March 2012 – August 2012, the GUC WWTP had slightly higher mean N concentrations (5.23 ± 1.08 mg/L; n=6) than other WWTP in North Carolina. The annual average for Johnston County, NC CSS using similar technologies as GUC WWTP was 2.14 (± 0.36 mg/L), while a North Cary, NC WWTP found an annual average of 3.67 (± 0.51 mg/L) (US EPA, 2008).

Influence of On-site Wastewater Treatment Systems on Water Resources

OWTS pose a potential risk for NO_3^--N pollution to shallow groundwater (Table 1). Previous studies (Table 1) have shown that OWTS can contribute N concentrations and loads significant enough to promote eutrophication in surface water. The risk increases in areas of high population density (Hallberg, 1989; Gold *et al*., 1990; Bouchard *et al*., 1992; Hantzche and Finnemore, 1992; County of Butte, 1998; Gold and Sims, 2000; Rich, 2005) because higher population density leads to increased OWTS density, thus more N inputs to nearby surface waters from OWTS use. Conventional OWTS absorption systems are not primarily designed to treat N, therefore they can potentially cause high N loadings to nearby surface waters due to low (between 10-20%) N reductions (Keeney, 1986; Siegrist and Jenssen, 1989; Lamb *et al*., 1990). Newer technologies have since been developed over the past 20 years to improve N treatment (Whitmyer *et al*., 1991; Brooks, 1996; CRWQCB, 1997; Ayres Associates, 1998; County of Butte, 1998; Rich, 2005; Scholes, 2006), such as aerobic treatment units, media filters, sequencing batch reactors, drip dispersal, low-pressure distribution, media filters used as a drainfield, pumps, timers, or controls (URI, 2006). However, OWTS with advanced technologies have greater initial start-up costs and increased overhead when compared to conventional OWTS. Some of these technologies require operators and/or monitoring to be paid by users of the advanced OWTS.

As shown in Table 1, wastewater inputs have been shown to impact N loading to groundwater and surface water in a variety of settings across the United States, especially OWTS. Attenuation of N in surficial aquifers is controlled by a variety of factors, including cation exchange capacity, denitrification potential, plant uptake, presence of riparian buffers, anammox, and perhaps others. Therefore, it is not clear if OWTS N is always translated to surface waters and how much OWTS contributes to watershed N-exports in the nutrient-sensitive watersheds of the North Carolina Coastal Plain. In this study, the goal was to quantify groundwater OWTS N loading and determine if this N is transported to Coastal Plain streams, thereby affecting N exports from these watersheds.

Study Objectives

The goal of this study was to determine if OWTS affect groundwater and surface water nutrient loading at the watershed-scale in nutrient sensitive watersheds of the North Carolina Coastal Plain. The objectives of the groundwater study were to (1) determine if significant differences existed between groundwater TDN concentrations and loads at the residential yard scale in OWTS and CSS watersheds and (2) compare the N treatment efficiencies of OWTS versus CSS. The study objectives of the surface water study were to determine if (1) TN concentrations in surface water and (2) watershed TN exports were affected by wastewater management approaches at the watershed scale. To achieve these objectives a variety of field hydrogeological, geochemical, and data analysis techniques were used that will be documented in the following chapter.

Table 1. Wastewater influences to water resources in differing physiographic settings using OWTS and CSS technologies.

Source	**Findings**	**Physiographic Setting**
Wang *et al*. (2013)	NO_3^- loads to surface waters were estimated using ArcNLET (ArcGIS-based model). NO_3^- loads were approximately 1.4 and 8.6 kg/day.	Southeastern Coastal Plain (USA)
Oakley *et al*. (2010)	Single pass sand filter with denitrification bed of solid carbon may match advanced CSS N effluent and use less energy than other OWTS. [a]	Multiple regions (Southern Florida Coastal Plain, Eastern Cascades Slopes and Foothills (USA), and New Zealand)
Kroeger *et al*. (2006)	CSS load between 295 and 10,729 kg-N/yr to nearby coastal water, despite significant subsurface TN attenuation.	Atlantic Coastal Pine Barrens (USA)
Fagergren *et al*. (2004)	Hypoxic problems of Hood Canal, WA resulted from OWTS inputs of N.	Puget Lowland (USA)
Castro *et al*. (2003)	Nearly half of TN inputs in developing watersheds originated from OWTS sources. Also, developed/urbanized watersheds had the lowest TN retention.	Atlantic and Gulf Coasts (USA)
Moore *et al*. (2003)	OWTS and CSS contributed significant concentrations and loads of nutrients to nearby lakes. However, at the urban-rural fringe, high eutrophication at the urban-rural fringe was more likely related to OWTS discharges than CSS.	Lakes in the Seattle region of Washington – Pacific Province (USA)
Ray *et al*. (2000)	N from OWTS contributed to eutrophication of New Zealand Lakes, up to 25% of TN input to lakes may be from OWTS discharges.	New Zealand
Ricker *et al*. (1994)	55-60% of N-load to San Lorenzo River, CA originated from OWTS in summer months as baseflow discharge.	West of Central California Valley (USA)
Horsley Witten Hegeman (1991)	N inputs from OWTS contributed to nuisance algal growth and diminished eelgrass beds.	Atlantic Coastal Pine Barrens (USA)
Robertson *et al*. (1991)	NO_3^- plumes (>10 mg/L) persisted up to 25-35m before declining below 10 mg/L.	Ontario, Canada
Valiela and Costa (1988)	Nearly half of TN inputs in developing watersheds originated from OWTS sources.	Atlantic Coastal Pines Barrens (USA)
Cogger and Carlile (1984)	Consistently high water tables correlated with high NH_4^+, while NO_3^- dominated for lower water tables.	North Carolina Coastal Plain (USA)

[a]= Study did not collect and compare groundwater, only wastewater within the tank.

CHAPTER 2: METHODOLOGY

Site Selection

Eight sub-watersheds located in the Greenville, NC (Fig. 2; Appendix A) area were selected for physical and chemical water quality assessment that contained topography and land-use (Table 2). Soils were also similar between sub-watersheds as evident from soil data from each groundwater-sampling site (Table 3). These watersheds were analyzed for TN characteristics based on wastewater treatment approach. The major difference across the watersheds was the wastewater disposal/treatment approach. In addition, because lot sizes are allowed to be smaller in CSS watersheds, there was a tendency for CSS watersheds to have greater impervious area and greater population density. An unnamed tributary to Hardee Creek, within the Tar River Basin in the Eastern Pines Road section of Simpson, NC was selected because it exclusively used OWTS to dispose of household wastewater. This sub-watershed is referred to as EP-O (Fig. 2; Fig. 3). In addition, sites along 2 tributaries to the Tar River, Hardee Creek and Mill Branch, which used OWTS, were selected. The Hardee Creek site was located in the Cherry Oaks neighborhood in Greenville and this site is referred to as CHOK hereafter. The Mill Branch watershed site is south of the confluence of Mill Branch stream and the Tar River and is referred to as MILL (Appendix A).

The Firetower Watershed, which is a tributary to Fork Swamp that later drains into the Neuse River Basin, (FT-O; Fig. 2; Fig. 3) exclusively used the GUC WWTP for wastewater treatment. FT-O is twice the size of EP-O and is subdivided into FT-1 and FT-2 sub-watersheds. These sub-watersheds are comparable in size to the EP-O watershed (Table 2). In addition, Meeting House Branch (MHB) and Bell Branch (BELL) watersheds were served by CSS and were selected. MHB and CHOK are the largest watersheds of all the CSS and OWTS watersheds. Water quality and nutrient loading was compared between FT-1, FT-2, MHB (CSS), and EP-O, CHOK, and MILL (OWTS).

Five residences were selected in both the FT-O and EP-O watersheds for monitoring groundwater quantity and quality. Sites 100-500 were located within the EP-O watershed (Fig. 3; Table 3). Sites 600-1000 were located within the FT-O watershed (Fig. 3; Table 3). OWTS permits (5 permits for sites 100-500) were obtained from Pitt County Environmental Health Department and scanned for each site in that EP-O watershed. Permit data that were acquired included tank size, drainfield area, system installation date, and design loading rate (Table 4).

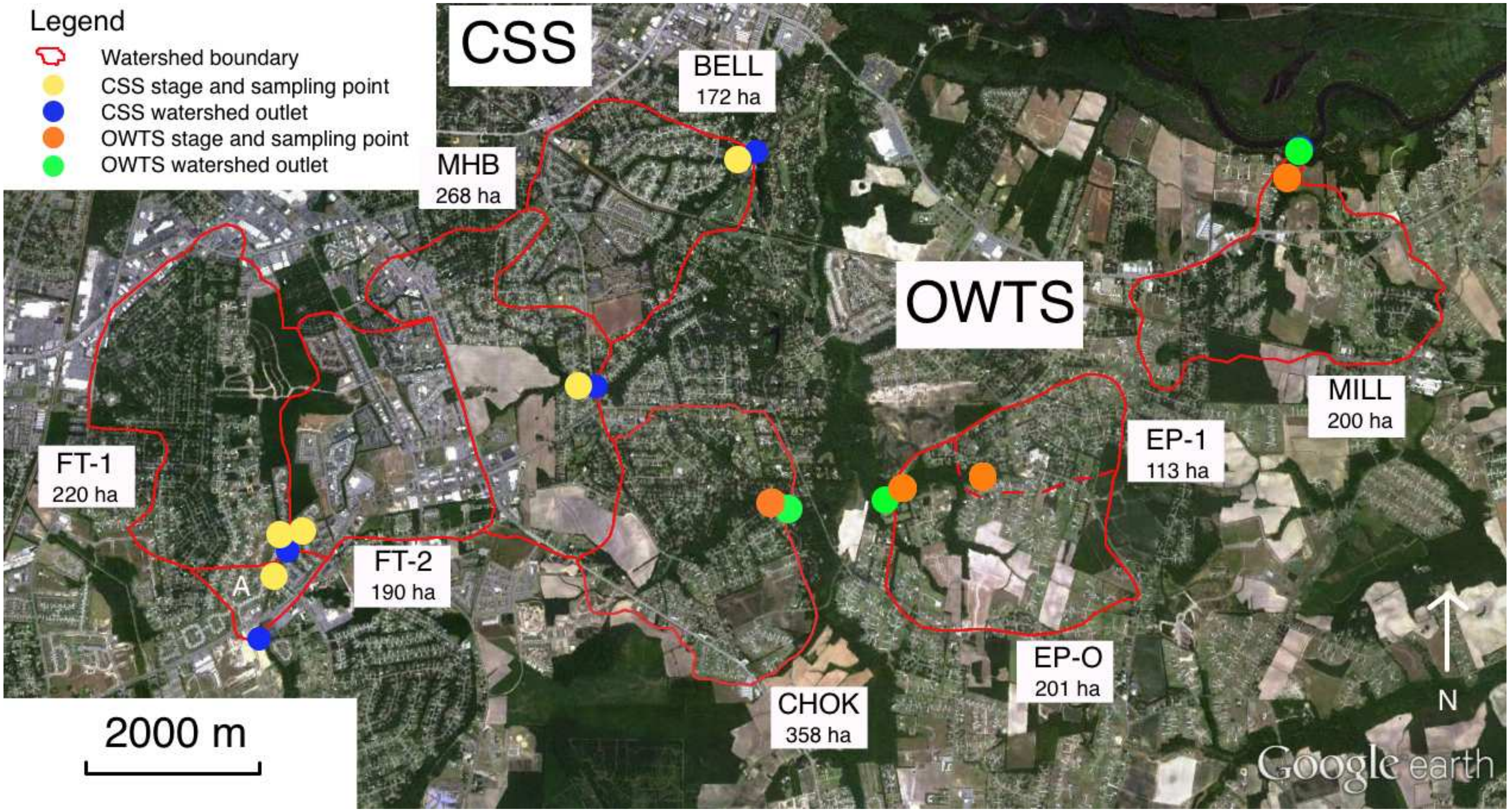
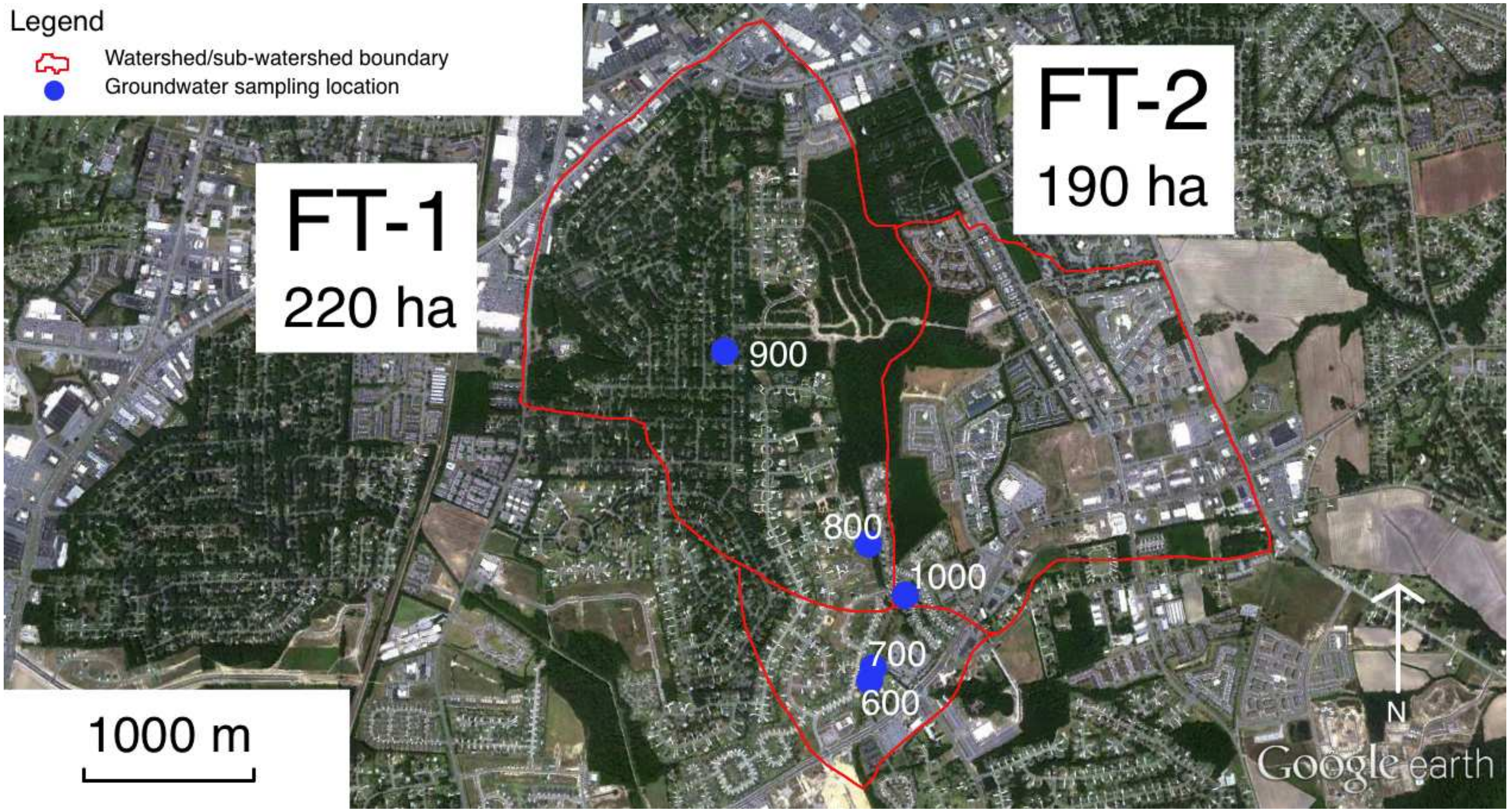
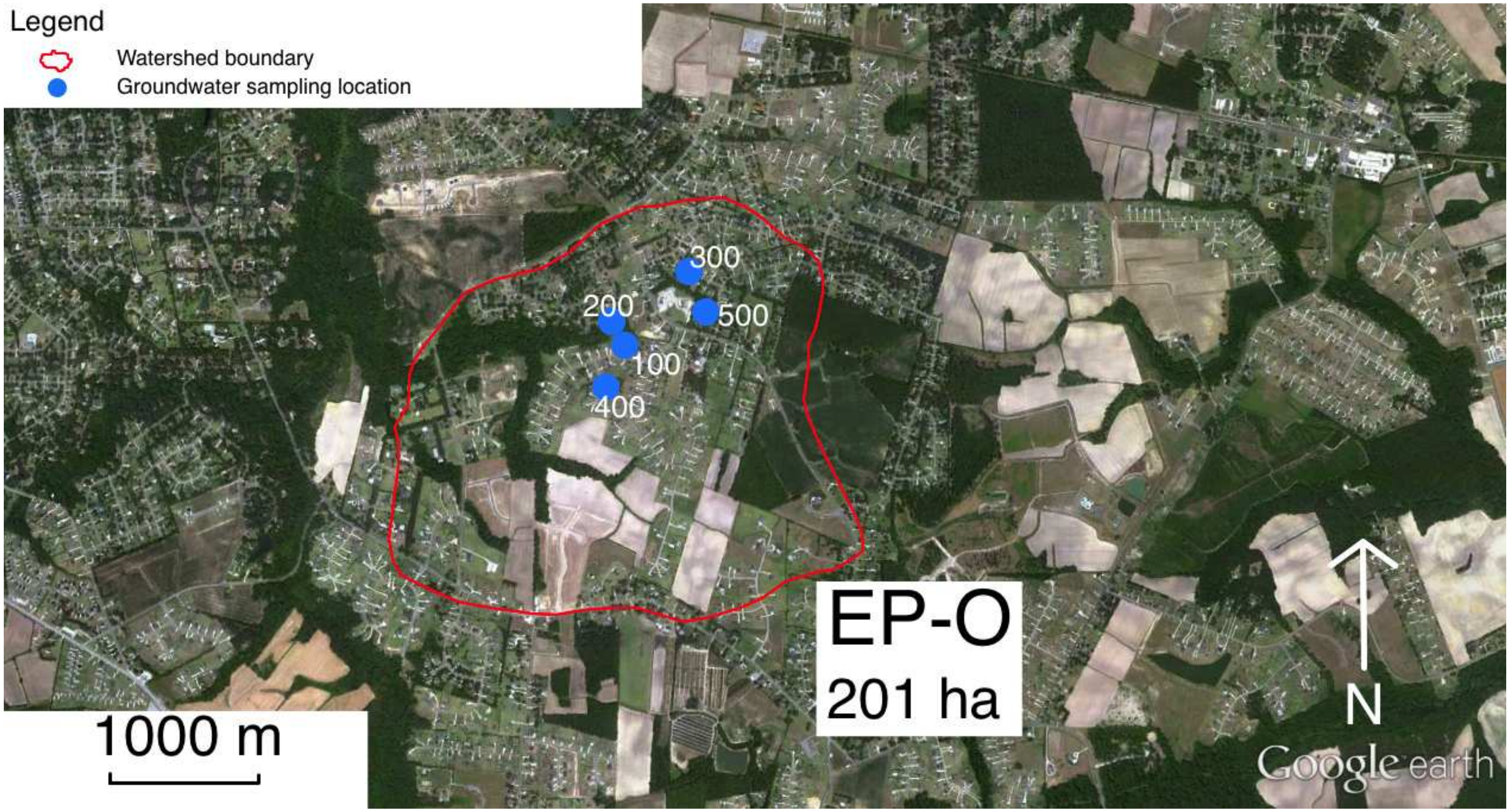


Figure 2. Surface water watershed delineation maps, each watershed is labeled by its name and area (in hectares). Additional watershed information is available in Appendix A. Basemap was acquired and modified from Google Earth (2013) based on satellite imagery updated on April 6, 2013.



Legend
Watershed/sub-watershed boundary
Groundwater sampling location
FT-1
220 ha
FT-2
190 ha
900
800
1000
700
600
1000 m
N
Google earth

A



B

Figure 3. A) FT-O watershed showing groundwater monitoring sites at each residence (blue circle). B) EP-O watershed showing groundwater monitoring sites at each residence (blue circle). Additional information at these watersheds is available in Appendix A.

Table 2. Watershed site classification based on area, wastewater treatment, topography, land-use, percentage of impervious surface in watershed, and number of samples collected from August 2011 to August 2012.

Watershed Name	Watershed Area (ha)	Wastewater Treatment	# Of OWTS/ha (total)	Stream Slope	Dominant Land-class	Total Impervious Surface (%)	No. Of Samples
FT-1[a]	199	CSS	-	0.0004	Residential	25%	13
FT-2[a]	140	CSS	-	0.0003	Residential	34%	13
MHB	268	CSS	-	0.0008	Residential	32%	13
BELL	172	CSS	-	0.0019	Residential	~30%[c]	13
EP-O	201	OWTS	1.63 (328)	0.0018	Residential	10%	13
EP-1[b]	113	OWTS	Structure count unavailable	0.0014	Residential	13%	13
MILL	200	OWTS	1.95 (389)	0.0033	Residential	12%	13
CHOK	280	OWTS	1.28 (358)	0.0018	Residential	12%[d]	13

[a]= These streams together represent the FT-O sampling location
[b]= Tributary to EP-O stream
[c]= Estimated from EEG (2012)
[d]= Estimated from Hardison *et al*. (2009)

Table 3. Groundwater site classification based on watershed location, wastewater treatment, predominant soils, and number of samples. Land use at these sites was residential. Water table depth was an average from four groundwater-sampling events in September 2011, November 2011, January 2012, and May 2012.

Site Name	Watershed Location	Wastewater Treatment	Soil Series	Soil Texture (Group I, II, III, or IV)	Average Water Table Depth (m)	No. Of Samples
100	EP-O	OWTS	Lynchburg and Goldsboro	Sandy clay loam (Group III)	0.89	69
200	EP-O	OWTS	Goldsboro	Sandy clay loam (Group III)	0.80	84
300	EP-O	OWTS	Ocilla	Sandy clay (Group III)	1.18	59
400	EP-O	OWTS	Ocilla	Sandy clay loam (Group III)	2.80	17
500	EP-O	OWTS	Ocilla	Sandy loam (Group II)	0.91	12
600	FT-O	CSS	Portsmouth	Sandy loam (Group II)	1.67	14
700	FT-O	CSS	Portsmouth	Sandy loam (Group II)	2.19	12
800	FT-1	CSS	Ocilla	Sandy loam (Group II)	2.10	16
900	FT-1	CSS	Pantego	Sandy clay loam (Group III)	2.19	17
1000	FT-2	CSS	Lynchburg	Sandy loam (Group II)	3.01	11

Table 4. Physical characteristics of each OWTS per site from original permitting information. Actual loading rates were calculated based on water use records. Vertical separation to the water table is between bottom of the drainfield trench and the water table based on average water depth at the drainfield during the four groundwater-sampling events. Water use and loading rate was not attained from site 500 because the tank was inaccessible. All tank sizes were similar between at OWTS sites at 3785 L (3.79 m^3).

Site	Drainfield Area (m^2)	Approx. System Installation Date	Water Use (L/day)	Design Loading Rate ($L/d/m^2$)	Actual Loading Rate ($L/d/m^2$)	Vertical Separation to Water Table (m)
100	156	1998	656	8.73	4.20	0.00
200	84	1977	515	16.30	6.16	0.21
300	111	1989	383	12.22	3.44	0.48
400	111	1999	905	12.22	8.12	2.22
500	84	1987	N/A	16.30	N/A	0.21

Local Hydrology

The Tar-Pamlico River Basin drains an area of 14,000 km^2. The major land-use is forest and wetlands (54%), followed by cultivated cropland (22%), open water area (20%), pasture and other controlled herbaceous area (3%) and urban (1%) (O'Driscoll *et al*., 2010). Approximately 80% of the basin resides in the Coastal Plain, distinguished by flat topography, blackwater streams, low-lying swamps, and estuarine areas. Tributaries generally have low gradients and stream velocities. Vast swamps and bottomland hardwood forests dominate these tributary floodplains (NC DENR, 2003). Unconfined surficial aquifers provide the majority of stream discharge to low-order streams (Winner and Coble, 1996).

The Neuse River Basin drains an area of 16,108 km^2. The major land-uses are agriculture (35%) and forestry (34%), with the remainder being urban areas (5%), wetlands (12%), scrub (4%), and open-water (10%) (NC DENR, 1993). The Neuse River originates just north of Durham, NC at the confluence of the Flat and Eno Rivers. It opens up into the Neuse Estuary outside New Bern, NC and extends 70 km before joining the Pamlico Sound (Qian, 2000). Similar to the Tar River, most of the Neuse River resides within the Coastal Plain of North Carolina. The mean annual precipitation and mean air temperature is analogous to that of the Tar Basin (at 126 cm of precipitation and 15.1°C mean air temperature) (Southeast Regional Climate Center, 2011). Observed precipitation at the FT-O and EP-O watersheds showed that from August 2011 – August 2012 there was more precipitation (145 ± 2.12 cm) than the long-term average of 126 cm.

Generally, stream flow is greatest in March and lowest in October. Baseflow separation estimates found 60% of discharge originates from groundwater and 40% from stormwater runoff in the Tar Basin (O'Driscoll *et al*., 2010). Monthly baseflow is largest between December and

March, suggesting that the greatest groundwater inputs occur during the winter and spring months. The late summer to early fall months record the lowest baseflow and thereby indicate little groundwater inputs (O'Driscoll *et al*., 2010). It is also important to note that during this period evapotranspiration is elevated due to warmer temperatures (Southeast Regional Climate Center, 2011) and increased plant water uptake.

Regional Geology of Eastern North Carolina

Thick deltaic and marine siliciclastic sediments exist within the Coastal Plain (Horton and Zullo, 1991). These sediments range from clay to gravel with inclusions of smaller quantities of marine limestone (Eocene to Oligocene). Typically, these siliciclastic beds with limestone inclusions dip and thicken towards the east and range in age from Holocene to Cretaceous (Winner and Coble, 1996). These beds terminate near the approximately 130 km west of Greenville, NC (study area) just outside Raleigh, NC. This termination marks the Piedmont-Inner Coastal Plain interface. Greenville, NC is underlain by a surficial aquifer composed mainly of Holocene to Pleistocene aged unconsolidated sand, silty, and clay sediments. The base of the unconfined aquifer is typically 3 to 6 m below the land surface. However, it can occur as deep as 15 m (Sumsion, 1970). Based on the locations, sediment characteristics and depths of piezometers, the current study focused on the surficial aquifer where OWTS systems are typically located in the Greenville, NC area. The surficial aquifer in the Greenville area is dominantly by the Yorktown Formation.

The Yorktown Formation, composed of Pliocene fossiliferous clay with varying amounts of fine-grained sand is present near the surface. In the Greenville area, the Yorktown Formation is underlain by the Black Creek, Pee Dee, and Beaufort Formations (Winner and Coble, 1996).

Late Cretaceous lagoonal and marine sediments compose the Black Creek Formation. Generally, these sediments are grey to black muds interbedded with fine-grained tan sands. A confining unit composed of clay and silty- to sandy-clay overlies the Black Creek aquifer (Winner and Coble, 1996). The Pee Dee aquifer consist of Late Cretaceous fine- to medium-grained sand interbedded with grey to black muds. The Pee Dee confining unit consists of low permeable clay and silty- to sandy-clay (Winner and Coble, 1996). The Beaufort Formation aquifer consists of Paleocene fine- to medium-grained dark green (glauconitic) and grey sand and marine clay (Winner and Coble, 1996). The Beaufort confining unit is typically clay to sandy clay (Winner and Coble, 1996).

Site Installation

Each stream site was instrumented with a staff gauge at each sampling point attached to a PVC stilling well that was anchored to a nearby tree or stream bank. HOBO water level loggers were housed in the stilling wells and recorded water levels every 30 minutes (August 2011 - August 2012). The staff gauge showed a snapshot of the stream water height on 13 discrete dates, while the logger data showed a more representative scenario throughout the year. Furthermore, the staff gauge data allowed for calculation of discharge rating curves, which allowed for the estimation of stream discharge based on the logger data.

Three sites (100-300) located in EP-O (Fig. 3) were intensively instrumented for groundwater monitoring, with 15 piezometers at site 100, 18 piezometers at site 200, and 12 piezometers at site 300 (Appendix B). Piezometers within the drainfield and near-stream area were installed at varying depths (Fig. 4) to capture the full extent of the effluent plume. Based on the upward flow component of groundwater (Fig. 4), these regions were located in discharge areas. The remaining 7 sites located in both FT-O (500-1000) and EP-O (400 and 500) (Fig. 3) watersheds were non-intensively instrumented (Appendix B), with approximately 3 piezometers at each location, for a total of 65 piezometers at the 10 sites. Near-stream piezometers at intensive sites were nested to determine the vertical extent of the OWTS plume. Most piezometers were installed by boring holes using hand augers. One piezometer was located adjacent to a highly incised stream reach and had a deep water table (mean depth: 12.7 ± 0.58 m). In this case, a GeoProbe was used for installation of the piezometer. A GeoProbe is a direct push machine that pushes a probe rod into the subsurface and displaces the sediment. During the boring process, soils were characterized and samples were taken for lab analyses at each site. Soil samples were analyzed for cation exchange capacity, pH (analyzed in 2011 at the North Carolina Department of Agriculture and Consumer Services Agronomics Division) and particle size distribution (analyzed in 2011 at North Carolina State University Soil Science Department) using the hydrometer method (Appendix C). Soils from sites 100, 200, and 400 soils were sandy clay loam, whereas the soils from sites 300 and 500 were sandy clays and sandy loams, respectively (Table 3; Appendix C).

Piezometers were installed in June 2011 and piezometer depth ranged from 1.04 m to 5.73 m. Piezometers were constructed using 3.18 cm or 5.08 cm PVC pipe and 0.9 m screen intervals. After setting the piezometer in the boring, well pack sand was used to fill the annular spaces surrounding the piezometer screen to anchor the pipe without inhibiting groundwater flowpaths. A mixture of well pack sand, natural soils, and bentonite was used to anchor the piezometer and seal the boring. Irrigation boxes were installed flush with ground surface around the top of the piezometer casing. Geographic coordinates were collected for each piezometer using a GPS unit. Piezometers were surveyed using a laser level to determine relative piezometer elevation. This was determined using a fixed point approximately 15.2 meters above mean sea level at each yard (e.g., septic tank at OWTS and a tree or other yard marker at CSS)

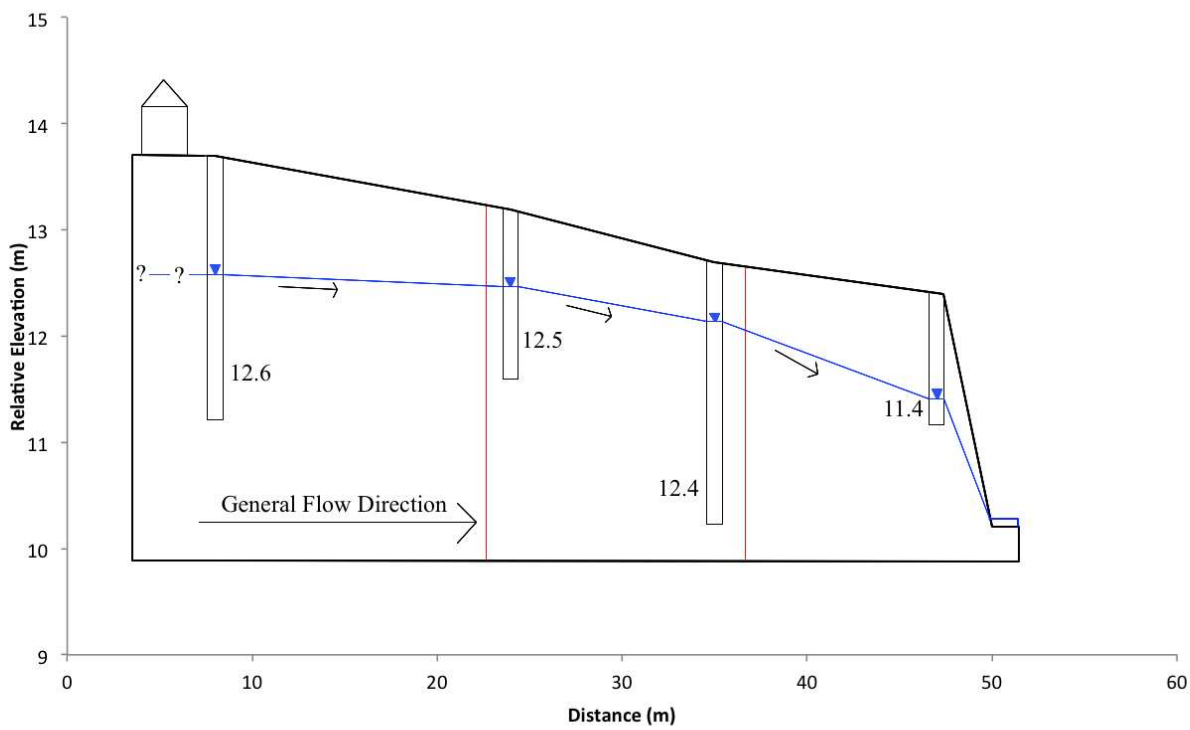


Figure 4. Cross-sectional view of site 100 showing a transect of piezometers 101, 103, 110s, and 108s, which is the prominent flow path of groundwater. The red lines denote piezometers within the drainfield. The small arrows between piezometers show the local groundwater flow direction and vertical flow component. The large arrow shows the general flow direction. The numbers beside piezometers reference the total hydraulic head relative to mean sea level.

Characteristics of On-site Wastewater Treatment Systems

At each OWTS site, tank volume was 3785 L (3.79 m^3). At sites 200 and 500, the drainfield area was 84 m^2. At sites 300 and 400, the drainfield area was 111 m^2. Site 100 had the largest drainfield area at 156 m^2 and had 2 systems (Table 4). One of the OWTS was installed in the 1970s (site 200), 2 in the late 1980s (sites 300 and 500), and 2 in the late 1990s (sites 100 and 400). Water use ranged between 383 and 905 L/day. Water use records were obtained from Eastern Pines Water Corporation. These water use data represent a per day average from

September 2011 to May 2012, this daily average was scaled up to the year to represent the potential wastewater inputs to the drainfield. Assuming all household water drains through the drainfield, this average can be used to estimate the loading to the groundwater system. June 2012 through August 2012 was left out to attempt to reduce error associated with potential losses occurring from irrigation and potential ET losses occurring during summer months. All loading rates that were measured in OWTS fell under their designed loading rates. Vertical separation distances from bottom of the trench and top of the water table were less than 1 m for all sites excluding site 400, ranging from 0-2.22 m (Table 4), based on mean annual water table depth from 0.60 to 2.76 m.

Sample Collection and Analysis

Stream water was sampled monthly for a year (August 2011 - August 2012). Physical water quality parameters were collected in the field at surface water monitoring locations (Fig. 2). Groundwater sampling events occurred quarterly during the year of study (September 2011, November 2011, January 2012, and May 2012). Piezometers were purged prior to sampling, using a disposable PVC bailer. Prior to purging, depth to groundwater was measured using a *Solinst* Temperature, Water Level, and Conductivity meter. After purging, groundwater readings were measured in the field for pH, temperature, specific conductivity (μS/cm), and dissolved oxygen (mg/L) using a *YSI-556 MultiProbe Meter*.

The same physical water quality parameters were measured in the streams. In addition turbidity was measured using a *Hach* turbidity meter. Stream width and average depth were measured each month during sampling. A flow meter (*Global Water* FP101) was initially used to gauge stream velocity. However, due to drought conditions during the summer of 2011 stream flow was too low for the meter to record velocity. Therefore, the floating object method (WV

DEP, 2013) was used instead and for consistency this method was continued throughout the study after drought conditions subsided. Three trials of the floating object method were conducted and the average velocity from these three readings was used to estimate the stream velocity. The results from the floating object method was compared to the results from flow meters at each of the 6 major watersheds. A correction factor was determined and multiplied to each floating object discharge to correct for overestimates made by the floating object method (Appendix D). These correction factors were similar to Brooks *et al.* (2003) factor of 0.8 or US EPA (2012) factor of 0.9 for muddy-bottom streams (similar to some of streams in this study). Stream discharge was calculated by multiplying average stream depth and velocity by stream width, and reported in L/day for the monthly sampling events. Average stream depth was based on depth data collected across the stream channel at 15 cm intervals. Based on a comparison of stream velocity measurements between the floating object method and the flow meter method, our estimates show that floating object velocity measurements were within 27% of flow meter velocity measurements (Appendix D).

Sample bottles used for surface water samples were rinsed 3 times in stream water and before samples were collected. Stream flow, environmental readings, and stream samples were collected before, during, and after 2 storm events using the same methods. One storm occurred in the dormant season (November) and another in the growing season (May). Groundwater and septic tank samples were collected in polypropylene sample bottles using a clean, new bailer that was discarded after each sampling event. Groundwater, wastewater, and surface water samples were stored in an iced cooler for transport. On the same date, samples were taken to the Central Environmental Laboratory at East Carolina University for nutrient analysis. Approximately 10% of samples were replicates and blanks. Over the course of the study, 64 replicates and 10 deionized water blanks were collected and submitted to the CEL. Results (Appendix E) showed that replicates were on average within 0.03 (± 0.08 mg/L) for TN and within 0.01 (± 0.37 mg/L) for Cl^- and blank samples had a mean of 0.09 (± 0.03 mg/L) TN and 0.26 (± 0.35 mg/L) Cl.

Laboratory Analysis

Filtering Process

Either ashed 934-AH (1.5 micron) or GF/C (1.2 micron) filters were combusted in a muffle furnace at 500^oC for 3-4 hours to burn off any organics from the filter. If groundwater filtrate had visibly high silt/clay loads additional filtration was performed with 0.45-micron membrane filters. Approximately 100 mL of sample was filtered. Filtrate was collected in a flask, and then transferred back to a washed (at least 3 times with deionized water) field bottle. Filtered samples were stored in the cold room overnight until analysis occurred the next day. If analysis was not immediate, samples were frozen and thawed no more than 24 hours prior to analysis.

Surface water samples were filtered similarly. Approximately 300 mL of sample was measured using a graduated cylinder and filtered. Foil packets with filters were stored in a freezer until PN analysis began.

Nitrate/Nitrite, Ammonium, and Chloride Analyses

Nitrate (NO_3^-) plus nitrite (NO_2^-) (henceforth called NO_3^-), ammonium (NH_4^+), and chloride (Cl) analyses shared similar procedures and these parameters were analyzed simultaneously using the *SmartChem 200* (WestCo, 2008). Sample cups were labeled according to field bottle label. Cups were washed using sample filtrate and discarded at least one time.

Sample cups were filled with a pipette and placed in a *SmartChem* rack. Each analysis was run with deionized water blank and 2 or 3 quality controls (solution samples with known concentrations of NO_3^- and NH_4^+) at the start and end of each rack. Reagents used for NO_3- analysis were NH_4Cl-EDTA and a color reagent. Reagents for NH_4^+ analysis were sodium-EDTA, sodium phenolate, sodium nitroprusside, and sodium hypochlorite. Sodium nitroprusside and sodium hypochlorite were made fresh daily. Chloride analysis used a composite chloride color reagent that consisted of mercuric thiocyanate, methanol, nitric acid, deionized water, ferric nitrate nonahydrate, and polyoxyethylene lauryl ether.

Dissolved Kjeldahl Nitrogen Analysis

Digestion Set-Up

Kjeldahl tubes were baked in a drying oven (105^oC) for at least 4 hours. Between 8 and 10 boiling chips were added to each tube. A graduated cylinder was rinsed 15 times with deionized water. Then, 25 mL of filtered sample was measured and added to a kjeldahl tube. This process was repeated until all samples were added or 35 samples were in the rack. Sample was used to wash the graduated cylinder between pouring. The last five spaces were saved for two water blanks, two quality controls, and one standard. Water blanks were fresh deionized water and quality controls and standards were less than a week old (or made fresh). After all kjeldahl tubes were filled, 10 mL of digestive reagent (mixture of cupric sulfate, potassium sulfate, and sulfuric acid) was added to each tube. The digestion rack was inspected for cracks, chips, external liquid, and boiling chips.

Digestion and Post-Digestion

The rack was placed into the digestion block before digestion began. Samples were heated at 210^{o}C for 1.8 hours, and then heated at 385^{o}C for 1.5 hours. The digestion rack was removed from the block and allowed to cool for 9 minutes. Afterwards, 25 mL of deionized water was added to each kjeldahl tube. The digestion rack was removed from the fume hood, placed on the bench top, and all tubes were capped with rubber stoppers. Using a *Vortex Genie*, samples were mixed for approximately 30 seconds each then allowed to rest for 1 hour. Mixes were repeated 2 times (3 total) each mix lasting approximately 30 seconds and allowing 1-hour rest between each mix. If a second digestion occurred, the digestion block was allowed to cool to 180^{o}C before beginning the digestion.

SmartChem Analysis

After the final mix, samples remained undisturbed overnight. Sample cups were labeled according to kjeldahl tube identification number. Using a 5 mL automatic pipet, approximately 3 mL of digested sample were used to wash sample cups at least one time. Sample cups were filled with a pipette and then placed into the *SmartChem* rack. This procedure was repeated until all samples were filled. At the start and end of each analysis were a deionized water blank, digested water blank, and 2 quality controls. DKN reagents used were working stock buffer, sodium salicylate, sodium hypochlorite, and sodium nitroprusside. The 200 uM-N standard was used to generate a standard curve to determine DKN micromolar concentrations. Sodium hypochlorite and sodium nitroprusside were made fresh daily.

Particulate Kjeldahl Nitrogen

Digestion Set-Up

Digestion set-up was synonymous to the DKN set-up. However, filters were torn up and placed into individual labeled kjeldahl tubes, rather than pouring up sample. Similar quality controls were used, but including 2 filter blanks and a higher uM spike concentration at 500 uM N.

Digestion and Post-Digestion

Digestion occurred similarly as well, however the phase 1 digestion only ran for 1.5 hours, rather than 1.8 hours. Mixing times were doubled to ensure the filter completely broke apart.

Particulate Nitrogen Analysis

SmartChem setup was nearly identical to the DKN analysis. At the start and end of each analysis were a deionized water blank, digested water blank, digested filter blank, and 2 quality controls. The required reagents were the same as the DKN process, but required a 500 uM N standard (used as spike) rather than 200 uM N.

Isotopic Analysis

Groundwater isotopic samples were collected from drainfield and near-stream piezometers at intensive sites in November 2011 and May 2012. Surface water isotopic data were collected in 8 of the 9 watersheds (excluded EP-1 because EP-O sample was collected downstream) during baseflow, storm, and after-storm conditions (Appendix S). Surface water and groundwater samples were sent to UC Davis for isotopic analysis (UC Davis Stable Isotope Facility). The Stable Isotope Facility at UC Davis analyzed water samples for $\delta^{15}N$ and $\delta^{18}O$ in

NO_3^-. The facility uses a ThermoFinnigan GasBench plus PreCon trace gas concentrations system interfaced to a ThermoScientific Delta V Plus isotope-ratio mass spectrometer (Bremen, Germany). More information regarding isotope analysis methods can be found in Sigman *et al*. (2001), Casciotti *et al*. (2002), and Granger and Sigman (2009).

Stable (non-radioactive) isotopes differ by their number of neutrons in the atomic nucleus, thereby affecting the isotopic weight. Physical and biological processes affect relative concentrations of light and heavy isotopes (McQuillan, 2004), thereby leading to isotopic fractionation, which is the relative enrichment or depletion of one stable isotope over another. For example, when water evaporates, ^{16}O preferentially enters the vapor phrase over ^{18}O due to its lower mass, thus enriching ^{18}O in the residual water (McQuillan, 2004). Two stable isotopes can be present in the NO_3^- molecule: ^{15}N (Mariotti, 1986; Mariotti *et al*., 1988; Smith *et al*., 1991; Böhlke and Denver, 1995) and ^{18}O (Böttcher *et al.*, 1990). These isotopes have been shown to be indicators of denitrification in groundwater (Aravena and Robertson, 1998). ^{15}N and ^{18}O are enriched during the denitrification process (Kreitler, 1975; Bates and Spalding, 1998). This enrichment occurs because biological organisms prefer ^{14}N for respiration and assimilation. This is because ^{14}N bonds are weaker than ^{15}N, thus the bonds are generally broken easier (Bates and Spalding, 1998). Therefore, ^{15}N enriches in the residual N source (Kreitler, 1975). For example, when NO_3^- is denitrified in the groundwater, more ^{14}N is extracted from the source (i.e. human and animal waste), thereby enriching ^{15}N in the residual source relative to ^{14}N (McQuillan, 2004).

Application of Isotopic Analysis

Generally, when denitrification occurs, enriched $\delta^{15}N$ and $\delta^{18}O$ values are observed (Böhlke and Denver, 1995) in the remaining NO_3^-. Field and laboratory studies indicate an enrichment ratio of $\delta^{18}O$ to $\delta^{15}N$ close to 1:2 (Olleros, 1983; Amberger and Schmidt, 1987; Böttcher *et al.*, 1990; Voerkelius and Schmidt, 1990; Kendall and McMahon, unpublished data). Therefore, denitrification creates a discernible isotopic signature of $\delta^{15}N$ versus $\delta^{18}O$ plots (i.e. slopes of roughly 0.5) (Kendall and McDonnell, 1998). Heaton (1984) found that as little as 20% of total NO_3^- removal via denitrification could increase $\delta^{15}N$ values by 8‰ of the remaining NO_3^- relative to original values.

Analyzing $\delta^{15}N$ alone will not provide conclusive evidence of the N source. In a case study by Kendall and McDonnell (1998), groundwater $\delta^{15}N$ values downgradient from a heavily fertilized (KNO_3) orchard ranged from +5 to +6‰. They hypothesized that fertilizer upgradient from the well would cause elevated $\delta^{15}N$ values. However, the analyzed range (+5 to +6‰ $\delta^{15}N$) was greater than the expected isotopic signature for fertilizer sources (0 ± 2‰). Therefore, the $\delta^{15}N$ values could be higher due to additional sources of NO_3^- (i.e. leaking OWTS or local manure sources, both of which range from 0 to +25‰), or the enriched values could be due to denitrification of NO_3^- within the fertilizer as it travels through the subsurface (Kendall and McDonnell, 1998). Through analysis of $\delta^{18}O$ values, the answer becomes clear if additional sources are present or if denitrification occurred (Heaton, 1986; Durka *et al.*, 1994; Kendall and McDonnell, 1998). If denitrification occurred, then the $\delta^{18}O$ values should range from +2.5 to +3‰. Additionally, both $\delta^{15}N$ and $\delta^{18}O$ values should become enriched as denitrification occurs and NO_3^- reduces (Aravena and Robertson, 1998). In the current study, isotopic analysis of NO_3 was used to determine potential sources of N in the groundwater samples and sources of N in surface waters compared between OWTS- and CSS-served watersheds and to indicate if denitrification was important along groundwater flowpaths from OWTS drainfields.

Data Analysis and Loading Calculations

Data were organized into comparison groups and summary statistics were used to distinguish potential trends using Microsoft Excel and Minitab v16. Mean, median, and standard deviation for all available water quality parameters were calculated for all comparison groups. Data for groundwater comparison groups were acquired from septic tanks, drainfield piezometers, near-stream nested piezometers, background piezometers, residential drainage ditches/streams, piezometers in CSS-served watersheds, and GUC inflow and outflow. There was one comparison group for surface water data: wastewater treatment approach (OWTS or CSS).

The N-speciation of surface water was reported as percent of TN that is dissolved organic nitrogen (DON), particulate nitrogen (PN), dissolved ammonium (NH_4^+), and dissolved nitrate (NO_3^-). DON was estimated by subtracting DKN from NH_4^+. Dissolved groundwater N-speciation includes the aforementioned, excluding PN. It was assumed that PN remains relatively immobile within the subsurface.

TN export was calculated by multiplying groundwater/stream discharge by groundwater/stream TN concentration. For surface water TN export, to normalize for the difference between watershed area, TN export was divided by watershed area. TN export was calculated under both baseflow and storm flow conditions for streams. Surface water TN export was reported as kg/yr/ha.

OWTS treatment efficiency was calculated as

$$TE = \frac{TDNt - TDNp}{TDNt} * 100\% \quad \text{(Eq 1.)}$$

Where TE is treatment efficiency, TDN_t is the tank TDN value, and TDN_p is the drainfield/near-stream piezometer TDN value.

Treatment efficiencies were calculated between the tank and the groundwater beneath the drainfield. Additionally, treatment efficiencies were calculated between the tank and near-stream groundwater. These were calculated quarterly at all OWTS sites and at the CSS. At sites 100-300, treatment efficiencies were calculated between septic tank and drainfield groundwater as well as between septic tank and near-stream groundwater. OWTS and CSS treatment efficiencies were compared to determine if significant differences existed between OWTS and the GUC WWTP. Nutrient concentrations in groundwater samples within the plumes and near-stream piezometers were used to show overall impacts of OWTS at each site. Treatment efficiency between the tank and the individual piezometer with the highest TDN was calculated to show the area within the plume that is most influenced, the plume core.

Groundwater TDN was multiplied by drainfield and near-stream groundwater discharge to determine TDN loading to groundwater and loading to streams. Groundwater TDN loads to streams were only calculated at sites 100 and 200. Site 300 did not have nested piezometers, thus it was not possible to determine the plume depth using similar methods as sites 100 and 200. Therefore, groundwater TDN loads to adjacent surface waters was not calculated for site 300. Additionally, TDN load leaving the tank was calculated by multiplying tank TDN by average water used per system, then divided by number of people per household to determine kg/yr/person. Load reductions were calculated between the tank, drainfield, and near-stream piezometers. These reductions were calculated similar to treatment efficiencies.

Mixing models were used to estimate the effects of dilution on groundwater N. The percentage of Cl^- contributed from natural groundwater and OWTS sources was estimated by the following equation.

$$GW\% = \frac{Clt - Clp}{Clt - Clb} * 100\% \qquad \text{(Eq. 2)}$$

Where GW% = Cl^- % from natural groundwater, Clt = tank chloride, Clp = drainfield/near-stream piezometer chloride, Clb = background chloride.

The percentage of Cl^- contributed from OWTS sources was calculated by subtracting the result from Eq. 2 from 100%. Cl^- is a good tracer because it remains relatively conservative from OWTS septic discharges from the tank to nearby surface waters (Corey and Fenimore, 1968). Mixing models were conducted by using the highest mean Cl^- data for the intensive sites in the drainfield and near-stream comparison groups. Based on visual inspection, most water quality data sets were not normally distributed. Mann-Whitney non-parametric statistical tests were run to determine if significant differences existed between comparison groups.

Overall, this field study was developed to help answer the broader management question: Are N-exports from OWTS influencing surface water quality in the NC Coastal Plain? Answering this question can help NC regulators to determine if OWTS management should be included in watershed nutrient management planning and regulations.

CHAPTER 3: EFFECTS OF ON-SITE WASTEWATER TREATMENT SYSTEMS ON GROUNDWATER NITROGEN LOADING IN THE NORTH CAROLINA COASTAL PLAIN

Abstract

Elevated nitrogen (N) concentrations in groundwater may cause adverse effects to adjacent surface water bodies. In Coastal North Carolina, management efforts have been focused on reducing nutrient exports to the Albemarle-Pamlico estuary system by implementing nutrient-sensitive management in the Neuse and Tar-Pamlico River Basins. Although half of North Carolina residences use on-site wastewater treatment systems (OWTS) these systems are typically not regulated beyond the permitting process. In the Tar-Pamlico and Neuse River Basins OWTS may contribute significant total dissolved nitrogen (TDN) concentrations to groundwater and adjacent nutrient-sensitive surface waters but this potential N input has not yet been adequately quantified. In this study, groundwater quantity and quality was monitored at 10 residential sites, 5 using OWTS and 5 using centralized sewer system (CSS). Three OWTS sites were intensively instrumented, with approximately 16 piezometers per site. The remaining 7 sites were instrumented non-intensively with approximately 3 piezometers per site. Groundwater samples were collected quarterly for a year (August 2011 to August 2012) and analyzed for TDN constituents (ammonium, nitrate + nitrite, and dissolved organic N) and chloride. Groundwater loadings from OWTS were calculated and compared amongst sites to determine if there was a difference in N loading between OWTS and CSS sites. TDN concentration (at OWTS and CSS groundwater monitoring sites) and load reductions (OWTS sites only) were calculated. Treatment efficiencies were compared between OWTS sites and the Greenville Utilities Commission wastewater treatment plant (GUC WWTP). Average OWTS groundwater TDN concentrations in the drainfields were 12.3 ± 15.5 mg/L, significantly elevated above the

background groundwater (3.46 ± 2.63 mg/L) and the groundwater TDN concentrations of 0.97 ± 1.00 mg/L at residential CSS sites. Groundwater loads downgradient from OWTS to adjacent streams at OWTS sites (0.62 ± 0.13 kg/yr) were approximately 5 times greater than groundwater loads to adjacent streams at CSS sites (0.13 ± 0.11 kg/yr). Groundwater TDN concentrations and loads to adjacent streams were significantly greater in OWTS watersheds in contrast to watersheds served by CSS. The results from this study suggest that OWTS N inputs should be considered among nutrient management strategies for nutrient-sensitive watersheds.

Introduction

The general approach for wastewater nutrient management strategies has been to target point sources of pollution (NC DENR 2009, 2010; NC DWQ 2010; 2013; Pradhan *et al*., 2007). On-site wastewater treatment systems (OWTS) may contribute significant loads of total dissolved nitrogen (TDN) to nearby groundwater and surface water (Pradhan *et al*., 2007), until recently these loads have rarely been accounted for in watershed nutrient management strategies (NC DENR 2009; 2010; Oakley *et al*., 2010; CBP, 2012).

In North Carolina, about 50% of residences utilize OWTS as a wastewater treatment approach (Pradhan *et al*., 2007). OWTS disperse domestic wastewater to buried drainfield trenches where soil and microorganisms treat the effluent. Septic tank effluent total nitrogen concentrations typically range between 26-124 mg/L, and therefore can be a significant source of nitrogen (N) (US EPA, 2002, WERF 2007). Conversely, many centralized sewer system (CSS) wastewater treatment plants (WWTP) treat wastewater off-site from the source and discharge the treated wastewater directly into surface water bodies. Regulatory agencies require CSS to monitor and report nutrient loads from wastewater discharges. Regulatory monitoring after the initial permitting process is not required for most OWTS. OWTS require regular maintenance (i.e. pumping the tank), although this responsibility falls on the homeowner. Failure to maintain OWTS may result in system failure and/or reduced performance, facilitating increased nutrient discharges to nearby surface water bodies. More field research that quantifies OWTS contributions of N to groundwater is needed to provide a scientific basis for N management in basin-wide planning efforts in the Tar-Pamlico and Neuse River basins and other nutrient-sensitive watersheds (NC DENR 2009, 2010; NC DWQ 2010; 2013).

Previous studies identified that, given adequate separation distance between the bottom of drainfield trenches and the water table, ammonium-rich (NH_4^+) effluent from the tank converts to nitrate (NO_3^-) as it moves through the vadose zone (Cogger and Carlile, 1984; Robertson *et al.*, 1991; Wilhelm *et al.*, 1996; Pradhan *et al.*, 2007; Oakley *et al*., 2010; Humphrey *et al*., 2010). Cogger and Carlile (1984), and Cardona (2006) via literature review, showed that the shorter separation distance between the bottom of the trench and the water table, the less oxygen available for nitrification. Therefore, if NH_4^+ does not readily convert to NO_3^- and if the cation exchange sites of the local soils are filled, NH_4^+ may disperse in the groundwater (Cogger and Carlile, 1984; Cardona, 2006). In the surficial aquifer of eastern North Carolina, Humphrey *et al.* (2010) found that 60 cm of separation distance provided adequate aeration for NH_4^+ transformations to NO_3^-. Assuming ideal conditions for nitrification, wastewater NO_3^- plumes can extend great distances (typically 10-100 m) (Robertson *et al*., 1991; Harman *et al*., 1996; Ptacek, 1998). Sandy aquifers enriched with organic matter and pyrite-rich deposits may significantly reduce NO_3^- in anaerobic plumes via denitrification (Pederson *et al*., 1991; Korom, 1992; Postma *et al*., 1992; Robertson and Cherry, 1992; Aravena and Robertson, 1998).

Most prior studies (e.g., Robertson *et al.*, 1991; Wilhelm *et al*., 1996; Pradhan *et al.*, 2007; Oakley *et al*., 2010) focused on OWTS N treatment have been performed at the labcolumn or lot-scale and do not attempt to quantify N mass loadings to groundwater and nearby surface water at the watershed scale. However, recently several studies have begun to address this shortcoming (Table 1). Modeling approaches have been helpful to approximate watershed-scale N inputs from OWTS sources (e.g., Pradhan *et al.* 2007; Oakley *et al*., 2010; Harrison *et al.*, 2012; Wang *et al*., 2013). However, these studies may overestimate the N inputs if adequate N attenuation is not characterized in the vadose zone and along groundwater flow paths.

Several studies have found significant N concentrations in and/or N loads from OWTS to surficial aquifers downgradient from OWTS (Table 1). These studies (Table 1) have shown that OWTS can influence groundwater N concentrations, which can also affect surface waters. Although several studies have modeled N inputs from OWTS to surface water (Pradhan *et al.*, 2007; Harrison *et al.*, 2012; Wang *et al*., 2013), to evaluate watershed-scale influence of OWTS on N-inputs to surface water, it is important to consider N reductions along groundwater flowpaths. To quantify the effects of wastewater management approach on water quality at the watershed-scale it is necessary to compare wastewater technologies between OWTS and CSS treatment methods. Several studies have shown that up to 60% of TN inputs to surface waters have been linked to OWTS use (Table 1). Furthermore, studies have shown OWTS use to be linked with increased eutrophication to surface waters (Table 1). However, these studies do not show groundwater inputs to surface waters at the watershed scale in the North Carolina Coastal Plain. Additionally, these studies do not compare OWTS to CSS treatment efficiencies using groundwater data downgradient of OWTS (Table 1).

The study objectives were to (1) determine if significant differences existed between groundwater TDN concentrations and loads at the residential yard scale in OWTS and CSS watersheds and (2) compare the N treatment efficiencies of OWTS vs. CSS. The research hypotheses were: (1) TDN concentrations and loads to groundwater and surface water are greater in OWTS-served watersheds than in CSS-served watersheds and (2) CSS TDN treatment efficiency is greater than OWTS TDN treatment efficiency.

This approach can help answer the broader management question: Are N-exports from OWTS impacting surface water quality in the NC Coastal Plain? Answering this question can help regulators and watershed managers to determine if OWTS N management should be included in basinwide nutrient management planning and regulations.

Methodology

Site Selection and Instrumentation

In the OWTS watersheds, three sites (100, 200, and 300) located in the Eastern Pines Road section of Simpson, NC were instrumented for detailed groundwater monitoring. At site 100, 15 piezometers were installed, 18 piezometers at site 200, and 12 piezometers at site 300. Piezometer depth varied within the drainfield and near-stream to ensure the plume core was captured (Fig. 4). Intensive sites included near-stream piezometers, but sites 100 and 200 were the only sites with nested near-stream piezometers. The remaining 7 sites located in both the Firetower watershed (FT-O) (CSS sites 600, 700, 800, 900, and 1000) and Eastern Pines watershed (EP-O) (OWTS sites 400 and 500) watersheds were non-intensively instrumented, with approximately 3 piezometers at each location, for a total of 65 piezometers at the 10 sites. Piezometer installation included hand augering a borehole and driving piezometer drivepoints to depths below the water table. Well pack sand was used to fill the borehole adjacent to the screened interval (0.9m screens) and piezometers were sealed from surface runoff using bentonite, natural soils, and well pack sand. Most piezometers were installed 1 m or greater beneath the water table using hand augers (depth range 1.04 to 5.73 m).

Groundwater Sampling and Analysis

Groundwater sampling events occurred quarterly for a year. Physical water quality data were collected with a calibrated handheld *YSI-556 MultiProbe Meter* and groundwater samples were taken using disposable PVC bailers. Samples were analyzed for NH_4^+, NO_3^-, dissolved kjeldahl N, and chloride (Cl^-) using a *SmartChem 200* color spectrometer (WestCo, 2008) at the East Carolina University Central Environmental Laboratory. Groundwater samples from November and May were sent to UC Davis for isotopic analysis of $\delta^{15}N$ and $\delta^{18}O$ of NO_3^-. The Stable Isotope Facility at UC Davis uses a ThermoFinnigan GasBench plus PreCon trace gas concentrations system interfaced to a ThermoScientific Delta V Plus isotope-ratio mass spectrometer (Bremen, Germany).

Groundwater Characterization

Groundwater discharge was estimated using Darcy's Law:

$$Q = K * A * \frac{dh}{dl} \qquad \text{(Eq. 1)}$$

Where Q = discharge, K = hydraulic conductivity; A = area; dh = change in head; dl = distance between piezometers.

The hydraulic conductivity approach was estimated via slug test approach (Bouwer and Rice, 1976). On average, there were 11 slug tests conducted for intensive sites and 2 for non-intensive sites. K estimates were based on median due to large variability between all tests and assumed to represent the K value for the site. K estimates were compared to K data from the National Cooperative Soil Survey (1971) and the USDA Pitt County Soil Survey (1974). K saturated for similar soils ranged from 0.10 to 1.22 m/d (National Cooperative Soil Survey, 1971; USDA, 1974). These data were calculated using constant head permeameter tests from field collected soil samples. OWTS and CSS groundwater K saturated values fell within this range. OWTS site mean K values were 0.24 (± 0.12) m/d, while CSS K values were 0.69 (± 0.51) m/d (Appendix I). The cross-sectional area of the OWTS plume was determined at 2 locations within each of the intensive OWTS residences. The plume cross-sectional area was estimated directly adjacent to the drainfield and in an area next to piezometers that were adjacent to the stream. The drainfield plume area was determined using OWTS permitting information and tile drain probing to identify drainfield width (as an estimate for plume width) and the plume depth was estimated using piezometer depth and groundwater quality data. The near-stream plume cross-sectional area was determined using the water quality and hydraulic head data from near-stream piezometers. Groundwater flow direction and hydraulic gradient ($\frac{dh}{dl}$) were determined using the 3-point solution method (Heath, 1983).

Data Analysis

TDN treatment efficiency was calculated for OWTS sites and the Greenville Utilities Corporation (GUC) WWTP. Soil N loading rates were calculated based on water use records and tank wastewater TDN concentrations. Groundwater N exports were calculated at both OWTS and CSS watersheds by multiplying groundwater discharge rate by groundwater TDN concentrations. To help identify sources of N in groundwater at each piezometer, isotopic data (UC Davis, 2013) and mixing models (Genereux and Hemond, 1990; Eq. 2) were used. Mixing models were used to evaluate the effects of dilution on groundwater N concentrations. The models estimated the Cl^- contributions from background groundwater and wastewater sources

(Eq. 2). The highest mean groundwater Cl^- was selected in the drainfield and near-stream piezometers at each intensive site to represent the core of the plume.

$$\%Cl_{BG} = 100\% * \frac{Cl_T - Cl_P}{Cl_T - Cl_{BG}} \quad \text{(Eq. 2)}$$

Where BG= background, T= tank, P= selected piezometer

Results

Groundwater Nitrogen Speciation, Concentrations, and Loads

Wastewater and Groundwater Nitrogen Speciation

Wastewater in the tank was predominantly NH_4^+ at each OWTS site. Similarly, GUC-wastewater influent was more than 80% NH_4^+ (Fig. 5). NH_4^+ was the dominant N-species at the tank, near-stream groundwater and stream at OWTS sites. NO_3^- and NH_4^+ made up a similar percentage of groundwater TDN at pooled OWTS sites. However, at sites 100 and 200, NH_4^+ was the dominant groundwater TDN species (Appendix G). Mean N-speciation was 70.7% (± 35.3%) NO_3^- (Fig. 5) in background groundwater. Similarly, at sites 300-500, drainfield groundwater was predominantly NO_3^-, with a mean of 64.6% (± 29.6%) (Appendix G). Dissolved organic nitrogen (DON) represented the mean dominant species in groundwater at CSS sites, with an average of 51.1% (± 27.6%) (Fig. 5; Appendix G).

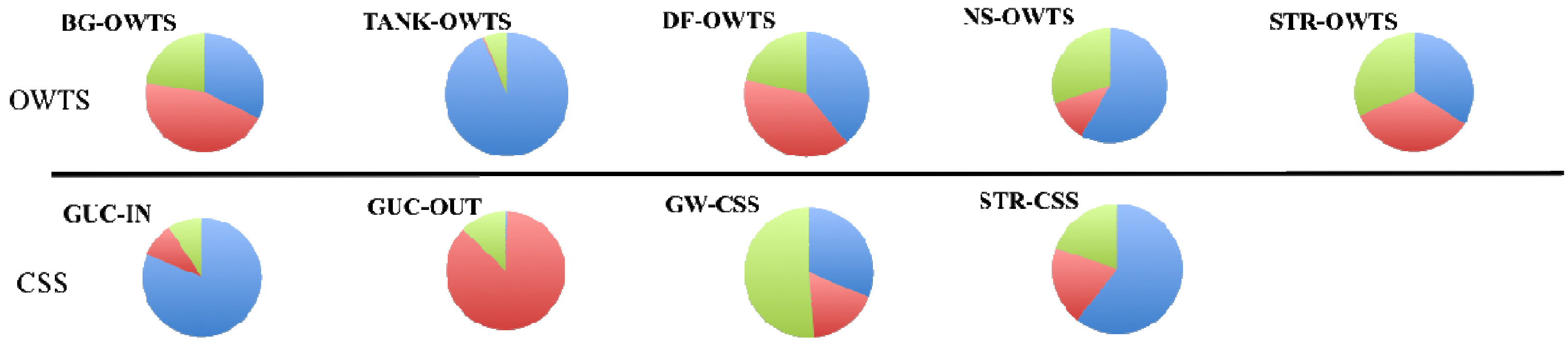


Figure 5. N speciation for each comparison group. Blue represents NH_4^+, red represents NO_3^-, and green represents DON. BG= background, DF= drainfield, NS= near-stream, STR= stream, IN= influent, OUT= effluent, and GW= groundwater. Tank and GUC-IN is raw wastewater, while GUC-OUT is treated wastewater. BG, DF, and NS represent groundwater that may be influenced from wastewater use.

Wastewater and Groundwater Dissolved Nitrogen Concentrations

OWTS wastewater TDN concentrations varied at each site. Site 200 had the highest mean wastewater TDN concentrations (81.8 ± 17.8 mg/L), while site 400 had the lowest (35.1 ± 5.00 mg/L) (Fig. 6). Site 100 and 200 background groundwater TDN concentrations were elevated relative to sites 300-500 (Fig. 6). Site 200 had the greatest mean background TDN at 6.69 (± 2.03 mg/L), while site 300 had the lowest (0.28 ± 0.22 mg/L). Concentrations gradually decreased from drainfield trenches towards near-stream piezometers. Drainfield TDN concentrations also varied between each site. Site 200 exhibited the greatest mean drainfield groundwater TDN concentration at 33.0 (± 20.2 mg/L). Site 500 recorded the lowest mean drainfield groundwater TDN concentrations (2.99 ± 3.41 mg/L (Fig. 6).

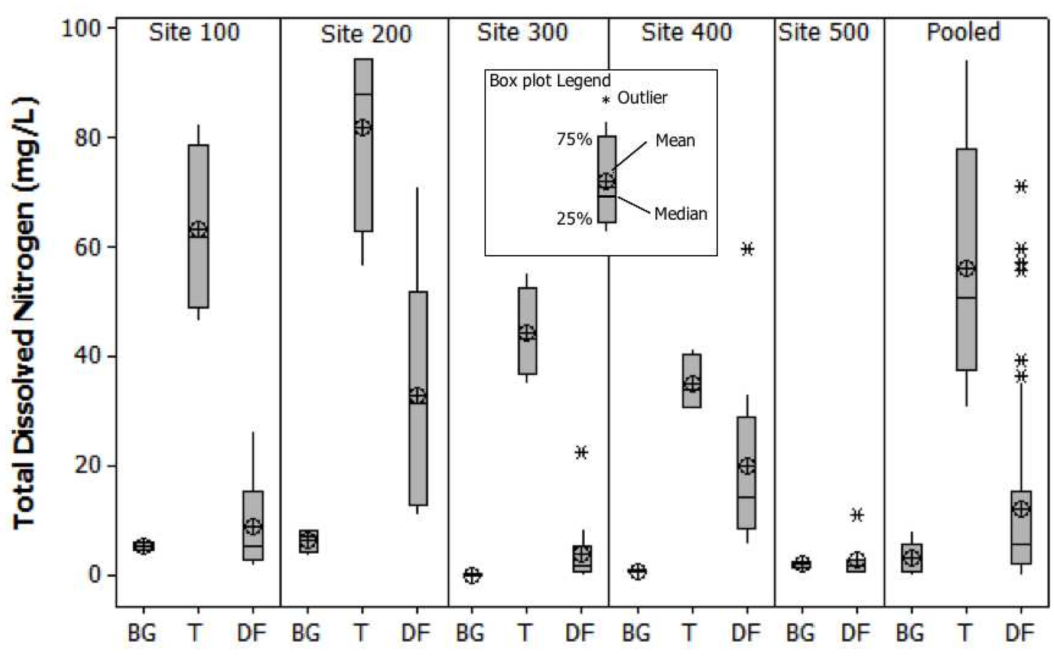


Figure 6. OWTS groundwater TDN concentrations at background and drainfield piezometers and tank TDN concentrations. BG= background; DF= drainfield. Mann-Whitney tests were conducted between comparison groups to determine if significant differences exist.

OWTS mean wastewater TDN concentration was 56.1 (± 21.8 mg/L) (Fig. 6). Mean TDN concentration in background groundwater at OWTS yards was 3.46 (± 2.63 mg/L), which was significantly higher than CSS groundwater mean TDN concentrations at 0.97 (± 1.00 mg/L) (Fig. 6). TDN concentrations in background groundwater at OWTS sites were more concentrated and variable, ranging from 0.07 to 8.37 mg/L. TDN concentrations at CSS groundwater ranged from 0.08 to 2.68 mg/L (Fig. 6). Pooled drainfield and near-stream OWTS groundwater TDN was also higher than CSS groundwater (Fig. 6). Mean TDN concentration of the pooled drainfield and near-stream groundwater was 8.02 (± 10.9 mg/L) (Fig. 7), significantly elevated ($p= 0.00$) relative to both OWTS background groundwater and CSS groundwater. The pooled OWTS drainfield and near-stream groundwater were also more variable, ranging between 0.10 and 16.4 mg/L with outliers (Fig. 7).

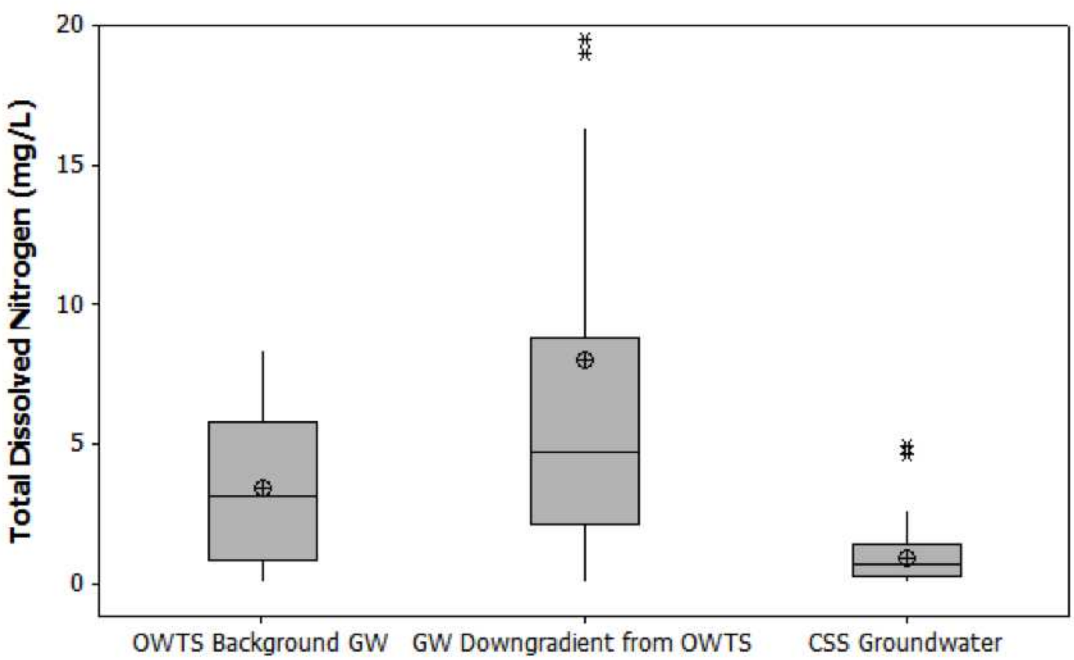


Figure 7. OWTS groundwater compared to CSS groundwater. Not plotted in this figure, but shown on Figure 6 is OWTS mean tank TDN concentrations (56.1 ± 21.8 mg/L).

Near-Stream Groundwater and Stream Dissolved Nitrogen Concentrations

Groundwater TDN concentrations in near-stream piezometers were generally higher and more variable than stream and background TDN concentrations (Fig. 8). On average, TDN concentrations (5.21 ± 4.49 mg/L) in near-stream piezometers at intensive OWTS sites were elevated relative to background (3.46 ± 2.63 mg/L) and residential streams (2.07 ± 1.27 mg/L) (Fig. 8). Groundwater TDN concentrations in near-stream piezometers at sites 100 and 300 were greater than background TDN concentrations. At a 95% confidence interval, this was significantly different at site 100 ($p = 0.02$), but not at site 300 ($p = 0.22$). Conversely, TDN concentrations in near-stream groundwater at site 200 were lower than background TDN concentrations (Fig. 8), but were not significantly different ($p = 0.20$). However, sites 100-300, TDN concentrations in the plume core were elevated relative to background groundwater. At sites 100 and 200, these differences were significant ($p = 0.01$ and $p = 0.03$), while at site 300 this difference was not significantly different at a 95% confidence interval ($p = 0.06$). TDN concentrations in residential streams at sites 100 and 200 were higher than CSS groundwater, but were not significantly different at a 95% confidence interval ($p = 0.08$). TDN concentrations in the residential stream at site 300 were higher, which was significantly different from CSS groundwater (Fig. 8).

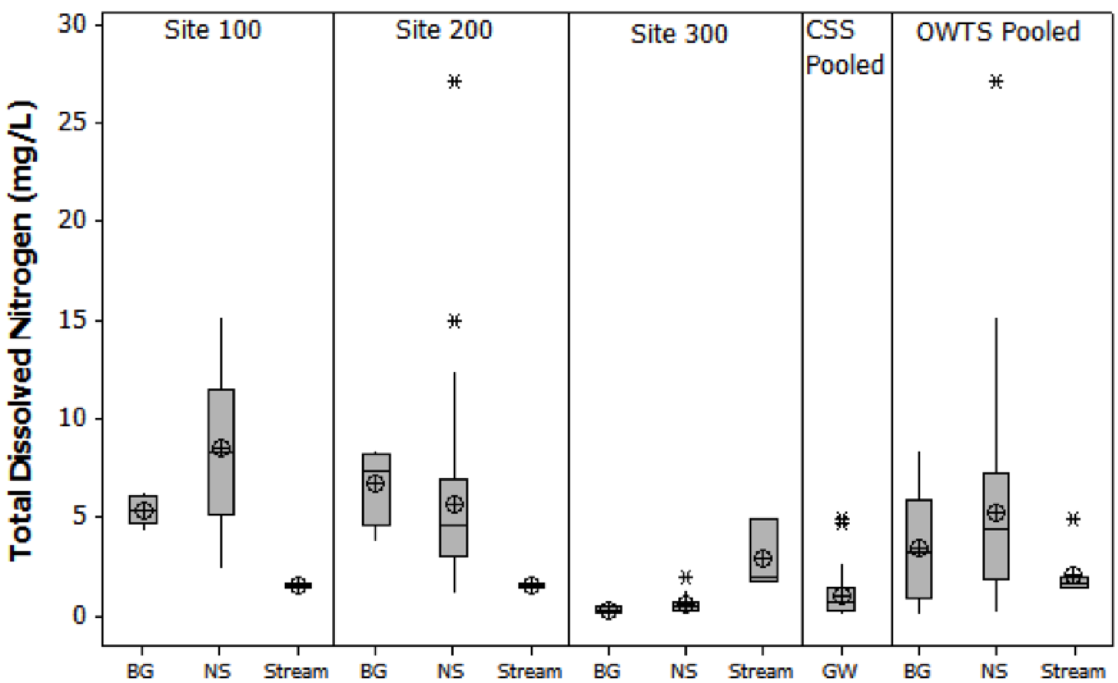


Figure 8. Near-stream (NS) groundwater TDN concentrations compared to background (BG) groundwater and residential surface water (stream) at intensive OWTS sites. CSS groundwater TDN concentrations are also provided to illustrate concentrations for sites not containing OWTS.

OWTS vs. CSS Treatment Efficiency

OWTS treatment efficiencies in the plume core were variable between each intensive site. The site with the highest treatment efficiency was site 300 (Table 5). However, as previously mentioned, piezometers were not nested at this site, so it is possible that the plume core was not fully characterized. Mean TDN reduction between tank and drainfield groundwater at all sites was 55% (± 21%). Mean treatment efficiency improved significantly between tank and near-stream groundwater and TDN declined by 88% (± 8%) (Table 5). The average distance between drainfield and near-stream piezometers at sites 100, 200, and 300 was approximately 17.3 m (range: 16-20 m).

CSS treatment efficiency was measured using mean TDN influent and effluent concentrations at the GUC WWTP. Treatment efficiency at the GUC WWTP (81% ± 3.17%) was slightly lower than all three intensive OWTS sites (Appendix H). At sites 100 and 200, treatment efficiencies were similar at 83% (site 100) and 85% (site 200). However, treatment efficiency at site 300 was greater than CSS treatment efficiency by 16% (mean treatment efficiency: 97%) (Table 5). This study's estimates for 2011 were similar to the treatment efficiency reported by GUC, which was approximately 77% (GUC, 2012).

Table 5. OWTS treatment efficiencies calculated. Drainfield reduction is defined as the percentage change of TDN from the tank to the groundwater in the drainfield. Near-stream reduction is defined as the percentage of TDN reduction from that tank to the near-stream groundwater. Site 500 is not included because the OWTS tank was inaccessible. Site 400 did not have a near-stream reduction because there were no adjacent surface waters.

Site	Tank (mg/L)	Drainfield Groundwater (mg/L)	Near-Stream Groundwater (mg/L)	Stream (mg/L)	Drainfield Reduction	Near-Stream Reduction
100	63.2	21.5	10.8	1.49	66%	83%
200	81.8	55.0	12.3	1.49	33%	85%
300	44.3	9.81	1.43	2.84	78%	97%
400	35.1	20.3			42%	
Average	56.1	26.6	8.16	2.17[a]	55%	88%
STDEV	21.8	19.6	5.88	0.96[a]	21%	8%

[a] = Mean and standard deviation were estimated from stream 100 and 300, since sites 100 and 200 share the same stream.

Groundwater TDN Loads

Three TDN loading estimates were calculated (Fig. 9). OWTS tank (A) is where wastewater from home is collected and stratification occurs. Wastewater migrates from A to B and enters the subsurface here through porous media; this zone represents OWTS TDN loads to soils. OWTS wastewater eventually percolates to the water table (C); this zone represents soil TDN loads to groundwater. After wastewater enters the saturated zone, wastewater and groundwater mixes and migrates downgradient toward adjacent surface waters (D); this zone represents groundwater TDN loads to surface water (Fig. 9).

The OWTS TDN load to soils (Fig. 9) varied between each OWTS site based on water usage, TDN concentration in the tank, and number of occupants in the household. Mean OWTS load to the soil was highest at site 200 at 16.7 ± 2.09 kg-N/yr. Site 300 showed the lowest mean soil load per person at 6.15 ± 1.64 kg-N/yr (Appendix I). Groundwater TDN loads were not available for site 500; the OWTS tank was not accessible due to an obstruction in the yard.

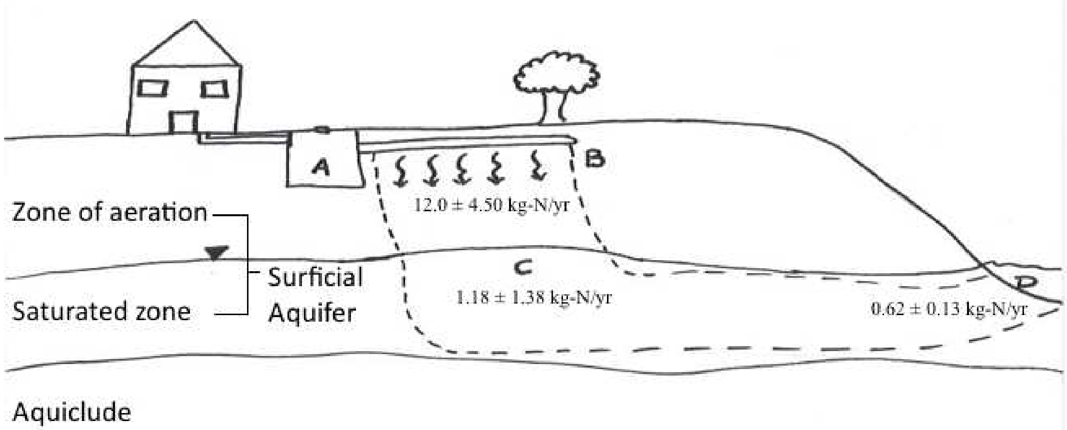


Figure 9. Idealized cross-section of OWTS use in a coastal plain setting showing the TDN loads from different OWTS components. The OWTS TDN load to soil (A to B) and soil TDN load to groundwater (B to C) are based on sites 100-400. The groundwater TDN load to adjacent surface waters (C to D) was based on sites 100 and 200.

Soil TDN loads to groundwater decreased substantially from the OWTS TDN loads to soil at each OWTS site (Fig. 10; Appendix I). Site 200 had the highest mean soil TDN load to groundwater at 3.49 ± 0.24 kg-N/yr (Fig. 10), which was a 79.6% ± 1.68% mean TDN load reduction from soil loadings (Table 6). Site 100 was the next highest at 1.84 ± 0.47 kg-N/yr. This corresponded to an average 86.7% ± 6.45% TDN load reduction. Mean soil TDN loading to groundwater at site 400 (0.38 ± 0.25 kg-N/yr) was slightly greater than at sites 300 and 500. Sites 300 and 500 had similar tank TDN loading to the groundwater at 0.11 ± 0.03 and 0.08 ± 0.08 kg-N/yr (Fig. 10). This resulted in a 98.1% ± 0.77% and 96.7% ± 2.20% load reduction at sites 300 and 400 (Table 6). At sites 100 and 200 estimates for groundwater loading to the stream were calculated. Both sites had similar groundwater TDN loads to the stream. At site 100, mean groundwater TDN loading to the stream (kg-N/yr) was 0.58 ± 0.17 kg/yr and site 200 was 0.66 ± 0.17 kg-N/yr (Fig. 10). At site 100, there was a load reduction of 95.8% ± 2.23% between TDN loading to the tank and groundwater TDN loading to the stream. Site 200 yielded a similar load reduction at 96.0% ± 1.69% (Table 6). Load reduction at site 300 to the stream was not estimated because near-stream piezometers were not nested.

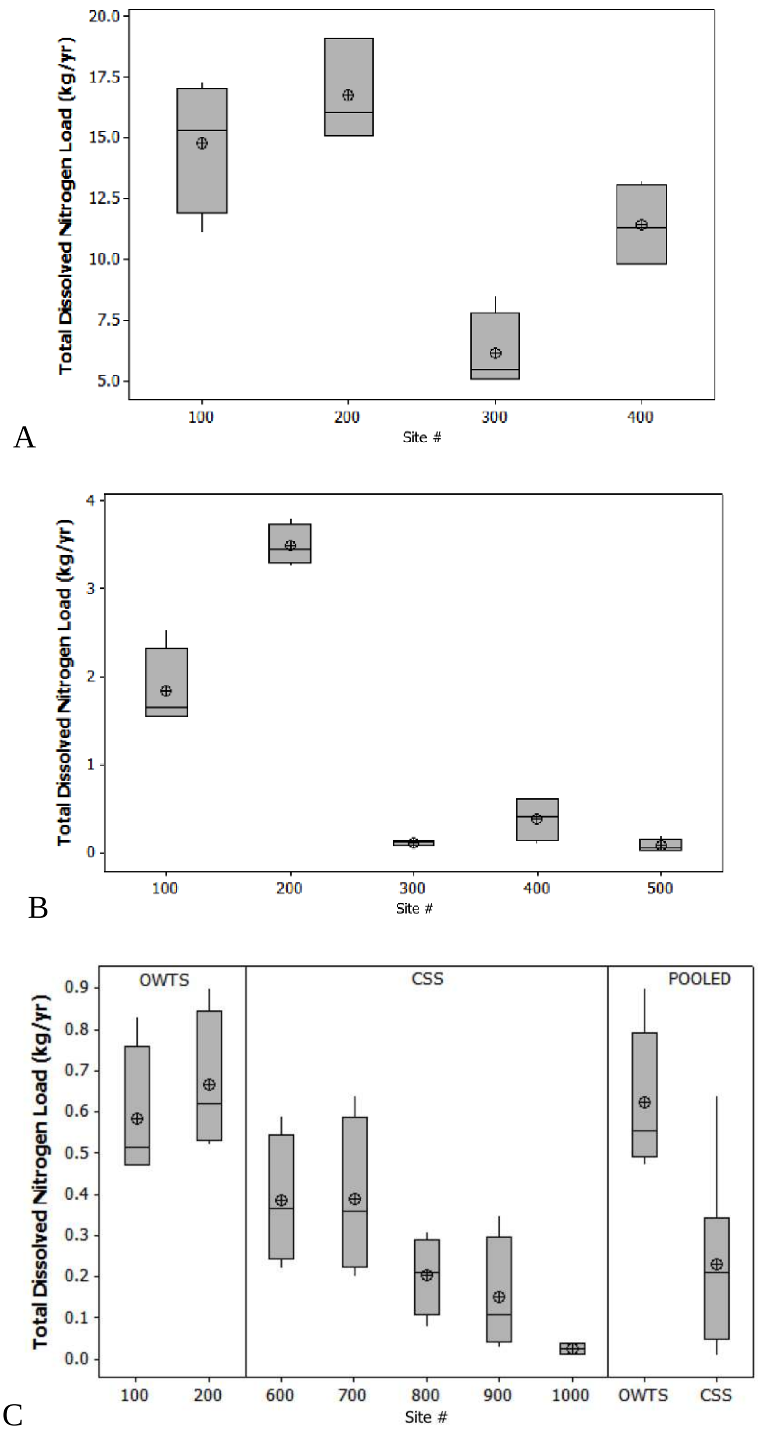


Figure 10. A) Soil loadings from wastewater entering the subsurface via drainfield trenches, the TDN load prior to entering groundwater. B) Soil loadings to groundwater after diffusing through the zone of aeration and mixing with the groundwater. C) Groundwater loadings to adjacent surface waters (this situation does not occur at all OWTS sites) compared to CSS groundwater loadings. CSS groundwater loads to adjacent surface waters were calculated under the assumption that the site 100 and 200 conditions occur at sites 600, 700, 800, 900, and 1000. These assumptions are based on the near-stream piezometer plume width and depth. There is an approximate distance from near-stream piezometers to adjacent streams of 1-2m prior to groundwater upwelling to these streams, thus the results are conservative estimates.

Table 6. TDN loading reductions between OWTS tank and groundwater beneath the drainfield (denoted Tank to Drainfield) and between tank and near-stream groundwater (denoted Tank to Near-Stream). Site 400 has no nearby streams. Site 500 load reductions were not calculated because the tank was inaccessible for sampling.

Site	Date	Load Reduction (Tank to Drainfield)	Load Reduction (Tank to Near-Stream)
100	Sep-2011	77.0%	92.5%
	Nov-2011	89.4%	96.2%
	Jan-2012	90.7%	97.3%
	May-2012	89.6%	97.1%
	Average:	86.7%	95.8%
	Median:	89.5%	96.6%
	STDEV:	6.45%	2.23%
200	Sep-2011	N/A	N/A
	Nov-2011	81.6%	97.1%
	Jan-2012	79.0%	96.8%
	May-2012	78.4%	94.0%
	Average:	79.6%	96.0%
	Median:	79.0%	96.8%
	STDEV:	1.68%	1.69%
300	Sep-2011	98.6%	N/A
	Nov-2011	97.5%	N/A
	Jan-2012	98.9%	N/A
	May-2012	97.3%	N/A
	Average:	98.1%	N/A
	Median:	98.0%	N/A
	STDEV:	0.77%	N/A
400	Sep-2011	94.3%	N/A
	Nov-2011	95.3%	N/A
	Jan-2012	98.1%	N/A
	May-2012	98.9%	N/A
	Average:	96.7%	N/A
	Median:	96.7%	N/A
	STDEV:	2.20%	N/A

Mechanisms of TDN Reduction

Dilution

Dilution, plant uptake, denitrification, cation exchange, and anammox are some potential means of TDN reduction. Denitrification was the targeted method of removal based on OWTS design. However, due to limited nitrification at some sites, denitrification rates may also have been limited. Mixing models show that dilution may significantly reduce TDN concentrations (Table 7). Cl^- concentrations in drainfield piezometers occurred mostly from wastewater sources. However, background groundwater contributed more Cl^- in near-stream piezometers. These patterns suggest that dilution can explain nearly 70% of TDN concentration reductions between tanks and some near-stream piezometers (Table 7).

Between tanks and drainfield and near-stream piezometers, mixing model estimates suggested that dilution could have accounted for most of TDN concentration reductions at intensive OWTS sites (Table 7). At site 200, mixing model estimates suggested that up to approximately 90% of TDN reductions may have occurred from dilution. At site 100, dilution could have been the dominant TDN reduction mechanism between tanks and drainfield piezometers. However, it is likely other attenuation mechanisms (plant uptake, denitrification, and anammox) also occurred to reduce TDN. Similar to site 200, site 100 mixing model estimates could potentially reduce nearly all the TDN between the tank and near-stream piezometers. These sites are in discharge areas near streams, so dilution is expected.

Table 7. Results from mixing models for the highest mean Cl^- at drainfield and near-stream piezometer in each intensive site. The model shows the percentage of Cl^- occurring from groundwater and wastewater sources. This percentage represents the percent dilution of Cl^- between drainfield and near-stream piezometers. DF= drainfield, NS= near-stream, and ED= estimated dilution. TDN from ED is a predicted value based on TDN dilution estimates suggested from the conservative Cl^- ion. If actual TDN falls below this value, it suggests that those declines in TDN are accounted for by mechanisms other than dilution.

Piezometer	Mean Dilution	Tank Concentration (mg/L)	TDN estimate from ED (mg/L)	Actual TDN (mg/L)	Total Actual Decline (mg/L)	Reduction by dilution mg/l (and % of total reduction)	Reduction by other sources mg/l (and % of total reduction)
			Drainfield				
110-s	45.40%	63.2	34.5	23.5	39.7	28.7 (72.3%)	11.0 (27.7%)
203	29.50%	81.8	57.7	55	26.8	24.1 (89.9%)	2.70 (10.1%)
303	44.40%	44.3	24.6	9.81	34.5	19.7 (57.1%)	14.8 (42.9%)
			Near-stream				
108-s	59.60%	63.2	25.5	9.22	53.98	37.7 (69.8%)	16.3 (30.2%)
212-s	58.50%	81.8	33.9	13.1	68.7	47.9 (69.7%)	20.8 (30.3%)
310	60.00%	44.3	17.7	1.43	42.87	26.6 (62.0%)	16.3 (38.0%)

Sources of Groundwater

Isotopic analyses of groundwater for $\delta^{15}N$ and $\delta^{18}O$ of NO_3^- data were used to confirm sources of N and determine if denitrification was an important mechanism of TDN attenuation. The $\delta^{15}N$ data suggest that during both November 2011 and May 2012 the primary source of NO_3^--N was manure and septic effluent at OWTS sites (Fig. 11; Appendix J). No livestock farms were located near OWTS sites, thus the dominant source of $\delta^{15}N$ and $\delta^{18}O$ in groundwater was likely from septic effluent alone. One OWTS groundwater sample fell within the ammonia fertilizer at site 200. This occurred at piezometer 204 in May 2012. It is possible the homeowner may have fertilized the lawn prior to sampling, thereby influencing the $\delta^{15}N$ and $\delta^{18}O$ signatures at this piezometer. A vegetable garden is present at site 200, although piezometer 204 was upgradient from the garden. At CSS watershed sites, data show that in November 2011 sources of NO_3^--N were likely from fertilizer or soils (Fig. 11; Appendix J).

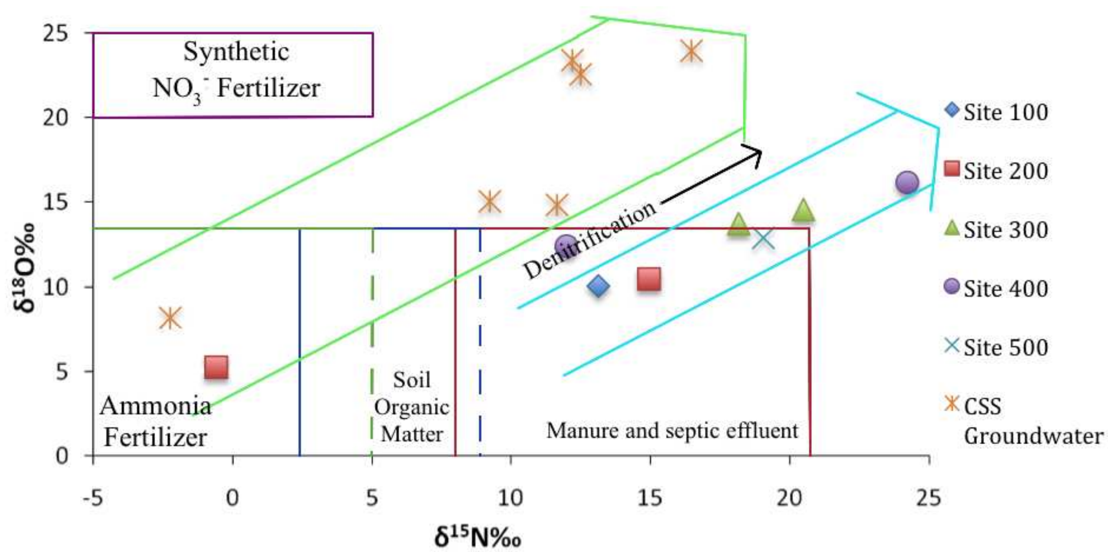


Figure 11. OWTS groundwater δ15N vs. $\delta^{18}O$ signatures in drainfield piezometers and CSS groundwater $\delta^{15}N$ vs. $\delta^{18}O$ signatures at all CSS piezometers as compared to Silva *et al*. (2002). The green box shows ammonia fertilizer, blue box shows soil organic matter, and the red box shows manure and septic effluent. The dashed line represents where fields may overlap. The black arrow shows the line that enrichment of $\delta^{15}N$ and $\delta^{18}O$ signatures from denitrification should follow. The teal lines show the extrapolated source of CSS groundwater, while the yellow line shows the same for OWTS groundwater

Discussion

Hypothesis 1: Groundwater TDN concentrations and loads to groundwater and surface water were greater in OWTS-served watersheds than in CSS-served watersheds

Results from the current study support hypothesis 1, which stated that OWTS sites have significantly greater groundwater TDN concentrations ($p = 0.00$) and TDN loads ($p = 0.00$) than CSS sites.

Groundwater TDN Concentrations

The similarity between TDN concentrations in background and drainfield groundwater at site 100 and 500 suggested that TDN concentrations in background piezometers might not reflect actual background TDN concentrations (Fig. 6). Background piezometers were upgradient of the studied OWTS drainfield, although some piezometers (specifically sites 100 and 200) were downgradient from other OWTS. At site 100, background piezometers were installed downgradient of the original OWTS drainfield, which may still be releasing wastewater. At site 500, TDN concentrations were similar between background and drainfield piezometers. This could be indicative of either sufficient treatment to reflect TDN concentrations in background piezometers or the plume core may have been missed due to the insufficient piezometers at this site.

TDN concentrations in groundwater at OWTS sites were significantly higher than those at CSS sites. This relationship was expected since OWTS discharge and treat wastewater effluent on site, whereas CSS, treat the household waste offsite. Nevertheless, these data and interpretation supported hypothesis 1. TDN concentrations within OWTS tanks fell within the WERF (2007) expected range of TDN concentrations for wastewater (Fig. 6). The differences in TDN concentrations within OWTS tanks between each site were likely due to differing uses and resident lifestyles (e.g., full-time workers or stay home parents). Sites 100, 200, and 400 typically had residents at home throughout the day. While residents at site 300 were not home as frequently, and thus the water use was lower. Tank TDN was predominantly rich in NH_4^+, which was in agreement with the literature (Anderson, 2006; Cardona, 2006).

At sites 100 and 200, mean separation distance was 0.10 (± 0.15 m), and did not appear adequate to facilitate significant nitrification. Groundwater beneath the drainfield and in near-stream piezometers was mostly NH_4^+ rich at sites 100 and 200. Cogger and Carlile (1984) found that NH_4^+ prevailed in conditions of consistently high water tables, while NO_3^- was most dominant at sites with consistently lower water tables. At sites 100, 200, and 500, separation distance (depth between bottom of trench and water table) did not meet the required 30 cm of vertical separation for group II – IV soils as required by the NC DHHS (2007). Despite this, at site 500, adequate nitrification appeared to occur. This may be attributed to differing resident lifestyle, which affected how often the OWTS was used. A sites 300 and 400, separation distance consistently exceeded 30 cm of separation (Table 4). The inadequate separation distance at sites 100 and 200, explains why these sites were mostly NH_4^+ in groundwater due to limited nitrification potentials. Conversely, the adequate separation at sites 300 and 400 explain why these sites were mostly NO_3^-.

Generally, the sites with lower nitrification had higher mean TDN concentrations than those that significantly nitrified wastewater effluent. In Carteret County, Humphrey *et al*. (2010) found that sites with similar geology and soils (group III soils) had average DIN concentrations of 25.8 mg/L (tank), 4.0 mg/L (drainfield), and 0.6 mg/L (background). All OWTS sites, except 500, had higher groundwater DIN concentrations than those found by Humphrey *et al*. (2010). The current study soils were predominantly group III soils (Table 3). Greater drainfield concentrations were likely due to increased DIN concentrations in effluent loads from the tank.

This may be due to differing loads to the tank and/or DIN reductions within the tank. Increased DIN at some sites relative to Humphrey *et al*. (2010) was likely due to limited nitrification for sites 100 and 200, which may adversely affect treatment. However, sites 100 and 200 soils were predominantly Goldsboro. When comparing sites 100 and 200 DIN concentrations to group II Goldsboro series soils from Humphrey *et al*. (2010) a different trend was observed. Site 100s and 200 DIN concentration in the drainfield (18.5 ± **ADD** mg/L), which was similar (within a standard deviation) to the mean group II soil DIN (17.1 mg/L) from Humphrey *et al*. (2010).

At sites 300 and 400, where nitrification appears to occur, findings were consistent with the literature. According to Anderson (2006), denitrification occurs while wastewater percolates through the saturated layer and N reductions due to denitrification can range between 10 and 75% (Sikora and Corey, 1976; Reneau, 1977; Harkin *et al*., 1979; Jenssen and Siegrist, 1988; Stewart and Reneau, 1988; Alhajjar *et al*., 1989; Siegrist and Jenssen, 1989; Stolt and Reneau, 1991; Mote and Buchanan, 1994; Duncan *et al*., 1994; Anderson *et al*., 1994; Chen and Harkin, 1998; Anderson, 1998; Anderson and Otis, 2000; US EPA, 2002). Studies that found denitrification rates of the upper end of the range typically utilized more advanced OWTS technologies rather than conventional OWTS. The current study found denitrification rates might attenuate NO_3^- concentrations up to 40% (Table 7), falling in the 10 to 75% range of denitrification. Therefore, at site 300, and perhaps site 400, denitrification reduced NO_3^- concentrations significantly. Denitrification was observed at sites 100 and 200 but rates may have been hindered due to limited nitrification of NH_4^+. This suggests that at sites where adequate nitrification is observed, denitrification may significantly reduce NO_3^-.

Other studies have shown that NO_3^- concentrations consistently match or exceed 10 mg/L NO_3^--N directly below conventional OWTS (Star and Sawhney, 1980; Cogger and Carlile, 1984;

Robertson *et al*., 1989; Ayres Associates, 1989; Converse *et al*., 1991; Converse *et al*., 1994; McNeillie *et al*., 1994; Anderson *et al*., 1994). This was the case at the plume core for sites 300 and 400. This was expected since adequate nitrification appeared to occur at these sites. If separation distance was adequate for sites 100 and 200, it is likely that NO_3^- concentrations would exceed the 10 mg/L directly below the drainfield. Site 500 exceeded the 10 mg/L for NO_3^--N on one occasion. However, it was possible drainfield piezometers at this site reside outside of the plume core based on groundwater flow data (Appendix B).

Dilution could have accounted for most of TDN concentration reductions at intensive sites, ranging between 57.1% and 89.9%. On average dilution reduced approximately 73.1% (± 16.4%) of groundwater TDN concentrations at piezometers at OWTS sites (Table 7). Dilution does not remove TDN masses from the subsurface system since it represents the mixing of wastewater with elevated N concentrations with background groundwater that has lower N concentrations, not a mass removal from the system. Mixing models showed less N mass reduction than other methods (e.g., groundwater load reductions and isotopic data) possibly because sites 100 and 200 background piezometers may have been influenced from upgradient OWTS.

The dominant reason for TDN attenuation varied based on OWTS performance at each site. All sites exhibited TDN concentration reductions greater than those estimated by dilution alone. Therefore, other factors affected the TDN attenuation. At sites 100 and 200, denitrification may have been limited (due to limited nitrification). Therefore, plant uptake, cation exchange, and/or anammox could account for the additional TDN attenuation observed. Drainfield and near-stream data suggest that cation exchange may significantly reduce NH_4^+ concentrations (Appendix K). Although nitrification was subdued at other sites, NO_3^- was elevated at sites 300 and 400, suggesting that nitrification was not inhibited. Therefore, plant uptake, denitrification, and dilution (Table 7) were potential attenuation sources for sites 300 and 400. Isotopic analysis suggested that denitrification occurred within groundwater beneath the drainfield (Fig. 11).

Isotopic analysis can provide insight on sources of NO_3^- in water resources (Kreitler, 1975; Fogg *et al*., 1998; McQuillan, 2004). Isotopic fractionation patterns of $\delta^{15}N$ and $\delta^{18}O$ in NO_3^- can help infer the original NO_3^- source (McQuillan, 2004). Biological organisms prefer ^{14}N for respiration and assimilation due to the lighter chemical bonds, which can more readily break down than heavier isotopes (i.e. ^{15}N) (Bates and Spalding, 1998). Therefore, ^{15}N accumulates in the residual N source and in human and animal wastes (Kreitler, 1975). For example, microbes utilize the ^{14}N from human and animal waste causing denitrification, which leads to an enrichment of ^{15}N, paired with NO_3^- concentration declines (Kreitler, 1975). Silva *et al*. (2002) found that $\delta^{15}N$ and $\delta^{18}O$ signatures differed based sources of NO_3^-. They found that for manure and septic effluent $\delta^{15}N$ signatures ranged from approximately +8‰ to +20‰ and $\delta^{18}O$ signatures ranged from approximately -5‰ to 13‰. Soil organic matter $\delta^{15}N$ signatures ranged from approximately +2.5‰ to +9‰, while ammonia fertilizers $\delta^{15}N$ signatures ranged from -5‰ to +5‰ (Fig. 11). The $\delta^{18}O$ signatures for both soil organic matter and ammonia fertilizers shared the same range as manure and septic effluent.

In the current study, isotope data showed that denitrification occurred at both OWTS and CSS groundwater sites. Most of the OWTS groundwater plots within the manure and septic effluent range according to Silva *et al*. (2002) (Fig. 11). At sites 300 and 400, where adequate nitrification was observed, denitrification enriched $\delta^{15}N$ and $\delta^{18}O$ signatures beyond the manure and septic effluent range. Through extrapolation these points plot within the manure and septic effluent range. Manure did not contribute to $\delta^{15}N$ and $\delta^{18}O$ signatures at OWTS yards. There were no livestock present near these residences, although pet waste could be a minor source of N. Based on the observed patterns, it is apparent that septic effluent from OWTS use contributed to the observed $\delta^{15}N$ and $\delta^{18}O$ signatures in groundwater. Groundwater at CSS-served yards showed more denitrification relative to groundwater at OWTS-served yards. Upon extrapolation, the data suggested that the primary source of NO_3^- in groundwater at CSS-served yards was ammonia fertilizer.

Groundwater TDN Loads

The current study's groundwater TDN loading to surface waters was consistent with other studies in the literature (Table 8). These studies showed that groundwaters loaded TDN masses of approximately 4.28 (± 2.39 kg/yr/ha) on average, ranging from 1.14 to 7.14 kg/yr/ha (Table 8). In the current study, the estimate was 2.48 kg/yr/ha, which was normalized from the 0.62 kg-N/yr by dividing by average OWTS lot size area (0.26 ha). Anderson (2006) studied OWTS influences to local groundwater and adjacent surface water in Apopka, FL. Anderson (2006) used model estimates to quantify OWTS loadings to soil and to groundwater using conservative reduction estimates. Anderson (2006) estimated 9.57 kg-N/home/yr exits the OWTS and loads the soil. Mean observed soil loading from the OWTS in the current study was similar to these estimates at 12.0 (± 4.51) kg-N/home/yr. Anderson (2006) estimated a 25% N reduction in the unsaturated zone based on previous literature (Ayres Associates, 1993; Anderson *et al.*, 1994), which leads to a soil N load to groundwater of 7.17 kg/home/yr. This was approximately 7 times larger than the mean soil TDN load to groundwater (1.18 ± 1.38 kg/home/yr) found in the current study. This large difference may be due to differing geological and groundwater recharge conditions at Anderson (2006) field sites since these estimates are based on OWTS in Florida.

Pradhan *et al*. (2007) estimated that 4.57 kg-N/yr/person (based on data from Alhajjar *et al.*, 1989 and Buetow, 2002) was generated from OWTS effluent. This was identical to the mean observed OWTS effluent load to soils in the current study (3.90 ± 1.60 kg-N/yr/person based on sites 100, 200, 300, and 400). However, N attenuation observed during the current study suggests that all of the N generated from OWTS loads will not likely make it to surface waters. Observed load reductions in the current study at 2 sites, ranged from 91% - 98% at sites 100 and 200. Therefore, these data suggest that an attenuation factor is important to consider for modeling efforts to estimate the actual loading to surface waters. Dilution accounted for a significant TDN reduction (up to 70%), which does not remove N from the groundwater system. Therefore, the observed attenuation factor at these yards is likely between 30% (losses from denitrification) (Table 7) and 98% (upper limit of load reductions) (Table 6). It was also possible the plume cross-sectional area did not capture the entire area where even diluted N migrates, thereby overestimating the actual attenuation.

For the 100 and 200 study sites in the current study, mean OWTS TDN loadings to adjacent surface waters (2.48 ± 0.77 kg/yr/ha) were nearly 2 times greater than mean TDN loadings from groundwater to surface water (1.52 ± 1.34 kg/yr/ha) at sites using CSS. Therefore, OWTS use affected the N inputs to adjacent surface water at the residential scale.

Table 8. TN input (kg/day) to watersheds from wastewater management reported by various studies on eastern United States watersheds. App= approximately, RC= residential and commercial, DNR= did not report, R= range, WWTP= wastewater treatment plant. For DNR, Bowen *et al*. (2007) conducted a study in the same watershed as previous studies; therefore OWTS characteristics may be similar. ACPB= Atlantic Coastal Pine Barrens, NC= North Carolina, APP= Atlantic Plain Providence, SCP= Southern Coastal Plain.

Reference	# Of OWTS	TN Input (kg/day)	Est. TN Input (kg/yr/ha)	Est. TN Input (kg/yr/person)	Watershed Name	Watershed Area (ha)	Physiographic Province
Valiela and Costa (1988)	app. 2000	56.2[b]	4.44	4.70[d]	Buttermilk Bay	4620	ACPB (MA)
Horsley Witten Hegeman Inc, (1991)	app. 3088	83.2[b]	6.57	4.51[d]	Buttermilk Bay	4620	ACPB (MA)
Sham *et al*. (1995)	app. 4230	36.2[b]	2.64	1.43[d]	Waquoit Bay	app. 5000	ACPB (MA)
Valiela *et al*. (1997)	app. 4230	316	2.20	1.19[d]	Waquoit Bay	app. 5000	ACPB (MA)
Bowen and Valiela (2001)	> 4000 houses	28.3[a]	2.07	1.12[d]	Waquoit Bay	app. 5000	ACPB (MA)
Kroeger *et al*. (2006)	33 WWTP (**not OWTS**)	8.51 (± 9.58) (R: 0.81-29.4)			Green Pond and West Falmouth Harbor	DNR	ACPB (MA)
Bowen *et al*. (2007)	DNR	86.8[a]	6.34		Waquoit Bay	app. 5000	ACPB (MA)
Pradhan *et al.* (2007)	app. 1.4 million	39353 (3557)[c]	1.14	4.54	All Major NC Watersheds	1.26E+07 (1.31E+06)	NC
Anne Arundel County (2008)	app. 41,000	1031	2.47	3.49[d]	Portion of Chesapeake Bay	152300	APP (MD)
Harrison *et al*. (2012)	app. 420,000	4384	1.48	1.45[d]	Chesapeake Bay	1160100	APP (MD)
Wang *et al*. (2013)	app. 5495	1.4 and 8.6[a]			Lower St. James River Basin	DNR	SCP (FL)

[a]: Estimated TN input calculated from models, Bowen used data from Valiela *et al*. (1997). While Wang *et al*. (2013) collected groundwater data.
[b]: Estimated TN input based on literature derived TN concentrations reaching water table
[c]: Estimated TN input assuming no treatment beyond OWTS tank concentrations, 39353 represents the total TN load for all of North Carolina's major basins, while the 3557 represents the average TN load between the Neuse and Tar-Pamlico basins.
[d]: Based on US Census (2013a) data, these data may be an overestimate if the average people per household changed from 1988 to 2010.

Hypothesis 2: CSS treatment efficiency is greater than OWTS treatment efficiency

Mean treatment efficiency at OWTS groundwater sites were more efficient than CSS treatment that occurred at the GUC WWTP (Fig. 12), which did not support hypothesis 2. Treatment efficiency differences between OWTS and CSS were not significantly different at a 95% confidence interval ($p = 0.09$). Average OWTS tank TDN concentration was double that of CSS influent TDN concentration, yet TDN concentrations in the plume core at near-steam piezometers were similar ($p = 0.34$) to TDN concentrations in GUC WWTP effluent. Oakley *et al*. (2010) stated that N treatment for the majority of OWTS could not match the stability or reliability of advanced CSS technologies. However, groundwater data downgradient from OWTS were not collected and compared to advanced CSS N treatment technologies in that study. The soil beneath the OWTS represents the most important component of the OWTS because it is where most of the treatment occurs (Cogger and Carlile, 1984; Hoover *et al.*, 1996; Cardona, 2006; Humphrey *et al.*, 2010). Furthermore, TDN reductions are limited to up to approximately 20% within the tank itself (US EPA, 1980; Laak, 1982; Pell and Nyberg, 1989) in conventional OWTS. Advanced OWTS yield increased reductions (Oakley *et al*., 2010). However, to ascertain a full-scale understanding of treatment efficiency between OWTS and CSS, the effluent discharge techniques must be considered.

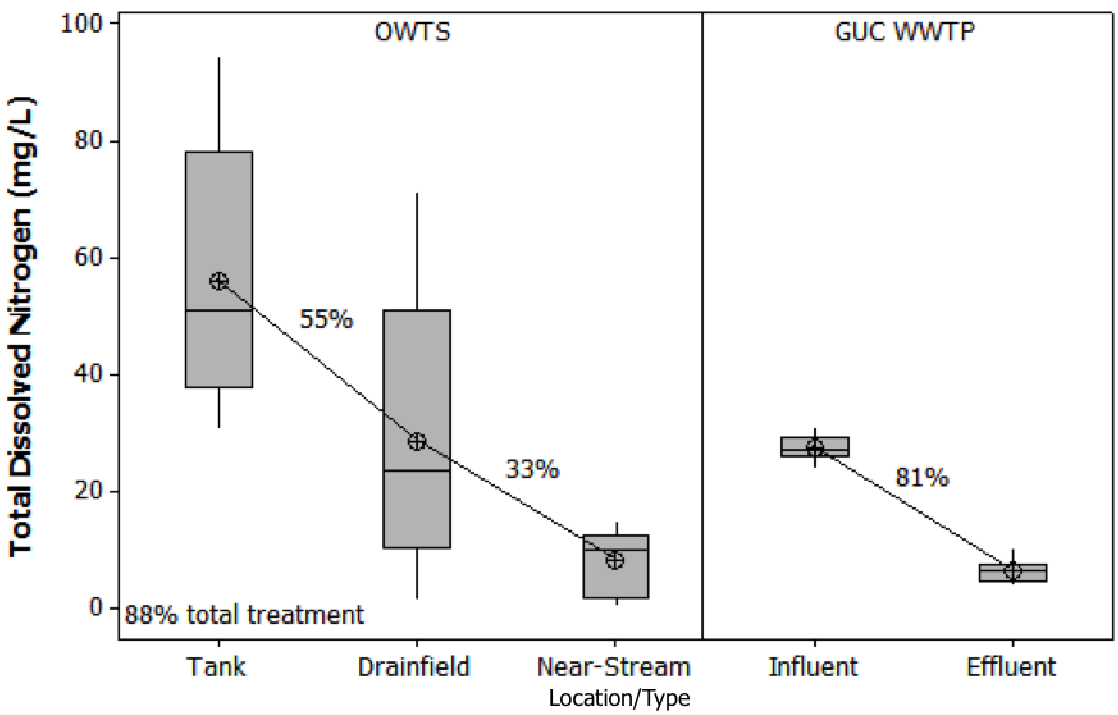


Figure 12. OWTS and CSS treatment efficiencies. OWTS treatment efficiency was pooled from sites 100, 200, and 300 compared to CSS influent and effluent, which was pooled from March 2012 to August 2012.

Oakley *et al*. (2010) found CSS with biological N removal to have influent N concentrations between 28-56 mg/L, with effluent between 1.6-5.3 mg/L. They found that OWTS using single pass sand filter with denitrification bed treatment was similar to advanced CSS, with mean N concentrations in influent at 66 mg/L and effluent at 1.8-3 mg/L. However, when comparing conventional OWTS to CSS, the TDN attenuation that occurs in the soil must be considered to ensure a complete comparison. The GUC WWTP used biological N removal and was found to have an 81% (± 3.17%) treatment efficiency (Fig. 12) and effluent TDN concentrations of 5.23 (± 1.08 mg/L). In the current study, all OWTS sites used conventional technologies and had higher pre-treatment wastewater TDN than CSS and better plume core treatment efficiency at 88% (Fig. 12), between tanks and near-stream piezometers. However, at sites 100 and 200, mean TDN concentrations in groundwater at the near-stream was approximately 2 times greater than mean TDN concentrations in GUC WWTP effluent.

The data for these coastal plain sites revealed that OWTS treatment efficiency between OWTS tank and near-stream piezometers approximately 17 m away was greater than GUC WWTP treatment efficiency. This difference was not significant at a 95% confidence interval at p= 0.09. However, TDN concentration reductions at OWTS sites were predominantly related to dilution so assumptions associated with plume width may result in some error associated with OWTS load reductions, possibly resulting in overestimates of load reductions. Observed TN concentrations at GUC WWTP effluent (5.23 ± 1.08 mg/L at GUC WWTP) were greater than the Johnston County, NC WWTP (2.14 ± 0.36 mg/L) and North Cary, NC WWTP (3.67 ± 0.51 mg/L). The GUC WWTP mean was within range of the North Cary WWTP (1.8 to 7.0 mg/L), but was elevated relative to the Johnston County WWTP (0.47 to 3.76 mg/L) (US EPA, 2008). OWTS near-stream groundwater TDN concentrations ranged from 0.11 and 27.2 mg/L across all near-stream piezometers. Average TDN concentrations in the plume core at near-stream piezometers were 8.16 (± 5.44 mg/L). These TDN concentrations were within range of many different CSS technologies across the US (US EPA, 2008). Therefore, mean OWTS treatment efficiencies and near-stream groundwater TDN concentrations were similar to that of CSS technologies.

Furthermore, OWTS plume core near-stream groundwater TDN was not significantly different (p= 0.34) from TDN concentrations in effluent at the GUC WWTP. Load reductions suggested that OWTS concentration reductions alone did not show the full treatment picture. Approximately 95% of the OWTS TDN load to soils reduced prior to discharging to adjacent surface waters. However, OWTS TDN concentration reductions were more likely to be influenced by dilution.

Conclusions

There were some limitations with the current study. At site 100, the TDN data suggested that the plume width along the near-stream could be wider, which could have lead to underestimation of groundwater TDN loads to surface water at this site. The surficial aquifer is generally anisotropic and heterogeneous, which requires a significant number of piezometers that are sampled frequently to characterize the spatiotemporal variability of hydraulic head and water quality. However, these studied sites were all volunteered by homeowners, which limited the number of piezometers that could be installed. Furthermore, labor and supply and sample costs also limited the number of piezometers installed at each site.

Excessive N in groundwater poses a risk to private water supplies and surface water ecosystems. In this study it was shown that TDN concentrations and loads to groundwater in OWTS watersheds were greater than those in CSS watersheds. OWTS TDN treatment efficiency at these coastal plain sites was approximately 9% greater on average than treatment efficiency at the GUC WWTP. This difference in treatment efficiency was not consistent with the literature, which found CSS to be most effective. However, some studies neglect to collect groundwater data downgradient of the OWTS drainfield, which overestimates the effluent exports from the OWTS. This overestimation occurs because the soil and surficial aquifer can play large roles in reducing TDN concentrations and must be considered. The findings of this study supported hypothesis 1, which stated that OWTS concentrations and loads are greater than CSS. However, it did not support hypothesis 2, which stated that CSS treatment efficiency is greater than OWTS treatment efficiency. Conventional OWTS can be as effective as advanced wastewater treatment technologies but site soils and hydrogeological characteristics play an important role in system function.

The results from this study suggested that because of potential nutrient inputs from OWTS use; they should be considered for inclusion in NC DENR’s and other state and international agency’s nutrient management strategies. While some sites may significantly reduce TDN concentrations beyond that of CSS standards, many sites have failing OWTS or TDN concentrations significant enough to contribute to eutrophication potentials in adjacent and downstream surface waters. OWTS density and distance from surface water bodies play a role in risk assessment. Future work can include setting up broader monitoring zones to assess more sites in the NC Coastal Plain, determining if long-term trends exist via long-term assessment, and installing OWTS mitigation strategies to curtail current TDN attenuation issues.

CHAPTER 4: THE EFFECTS OF WASTEWATER TREATMENT APPROACH ON SURFACE WATER NITROGEN CONCENTRATIONS AND EXPORTS IN COASTAL PLAIN WATERSHEDS

Abstract

Excess watershed nitrogen (N) loading can pose a significant threat to water supplies and aquatic ecosystems. The United States Environment Protection Agency (US EPA) regulates N concentrations in groundwater used for drinking purposes and provides guidelines for N in surface waters to reduce the potential for eutrophication. In North Carolina, half of the residents use on-site wastewater treatment systems (OWTS) for wastewater management. OWTS may contribute significant total nitrogen (TN) concentrations and loads to surface waters at the watershed scale but there is a lack of published studies focusing on surface water quality effects in nutrient sensitive Coastal Plain watersheds. In this study conducted in the Neuse and Tar-Pamlico River basins (North Carolina), surface water and groundwater N concentration and discharge information was collected over a year from 8 sub-watersheds. In sub-watersheds using OWTS, the surface water mean TN concentrations were approximately 2 times greater than in centralized sewer system (CSS) watersheds during baseflow and storm conditions. Streams draining OWTS sub-watersheds exported TN masses (kg/yr/ha) greater than 2 times that of CSS watersheds on an annual basis. It was estimated that TN export from wastewater sources in the OWTS-served watershed was approximately 2.2 kg/yr/ha. A watershed-scale TN attenuation factor of 81% (± 14%) was estimated for OWTS watersheds. These data show that CSS and OWTS can provide different outcomes for watershed-scale nutrient loading and consideration of wastewater management approach effects on surface water N is important in nutrient sensitive watersheds.

Introduction

Nitrogen (N) inputs to Atlantic and Gulf Coast estuaries have increased to between 2 and 20 times greater than pre-industrialized conditions (Boynton *et al*., 1995; Howarth *et al*., 1996; Jaworski *et al*., 1997; Goolsby, 2000). These inputs have increased over the past 2 centuries and have accelerated since the 1950s (UNEP, 2005). Excessive N inputs to surface waters potentially degrade aquatic ecosystems because primary production within estuaries is typically N-limited (Ryther and Dunstan, 1971; Nixon, 1986, 1995; Fisher and Openheimer, 1991; D'Elia *et al.*, 1992; Howarth *et al*., 2000). Increased N loading to surface waters may lead to greater frequencies of harmful algal blooms, hypoxic and anoxic bottom waters, loss of emergent plants, and reduced fish stocks (Valiela and Costa, 1988; Paerl, 1988, 1995, 1997; Valiela *et al*., 1990; Hallegraeff, 1993; Boynton *et al*., 1995). The main causes of these increases in N include intensive agriculture, fossil fuel combustion, extensive cultivation of leguminous crops (Smil, 2001), and wastewater management (Table 1). Recent work has shown that on-site wastewater treatment system (OWTS) may be a source of N to groundwater and surface waters in nutrient sensitive watersheds (Table 1).

Numerous studies in North America have shown that OWTS can affect groundwater TDN concentrations through TDN loading from OWTS to soil, groundwater, and streams downgradient from OWTS (Cogger and Carlile, 1984; Robertson *et al*., 1991; Wilhelm *et al*., 1996; Cardona, 2006; Pradhan *et al*., 2007; Oakley *et al*., 2010). Due to the interconnectivity between surficial aquifers and surface waters, surface waters located in OWTS-served watersheds can be affected by OWTS effluent (US EPA, 1980; Valiela and Costa, 1988; Meybeck *et al*., 1989; Howarth *et al*., 1996; Castro *et al*., 2003). This can be a problem in nutrient sensitive watersheds, because limited concentrations of N can increase eutrophication potentials in surface waters (Osmond *et al*., 2003).

Residences using centralized sewer systems (CSS) are within a network of infrastructure that transports wastewater to wastewater treatment plants. CSS treat wastewater offsite from the source and commonly discharge the treated wastewater directly into nearby surface water bodies. OWTS dispose, treat, and discharge wastewater directly into the subsurface, which later discharge to adjacent surface waters via groundwater transport. Regulatory agencies require CSS operators to monitor and report effluent discharges. However, most OWTS do not require regular monitoring after the initial permitting process. Additionally, homeowners are responsible for facilitating regular maintenance to ensure optimal OWTS performance. Failure to conduct regular maintenance typically causes reduced OWTS performance or complete malfunction. Regulatory monitoring and improved understanding of OWTS contributions of N to groundwater can improve basin-wide planning efforts and nutrient management in the Tar-Pamlico and Neuse River basins (NCDWQ, 2010; 2013). These basins are currently being managed to reduce N and phosphorus inputs due to current and past issues related to fish kills, algal blooms, and eutrophication occurrences in the receiving Albemarle-Pamlico estuary system (NCDWQ, 2010; 2013).

Previous studies to determine the effects of wastewater treatment approaches on surface water quality have found that OWTS and CSS discharges may potentially contribute significant concentrations and loads to adjacent surface waters, which may adversely impact aquatic ecosystems (Table 1; Table 9). Moore *et al.* (2003) found OWTS discharges to be more likely to contribute increased total nitrogen (TN) to lakes relative to CSS. There is an absence of research documenting the effects of OWTS on surface water quality at the watershed-scale in southeastern Coastal Plain settings. OWTS are the predominate means of wastewater treatment in rural watersheds. Castro *et al*. (2003) noted that trends regarding N retention based on watershed land-use is largely unknown and requires further study. Fertilizer and manure were the dominant sources of TN in the agricultural watersheds studied (Castro *et al*., 2003). TN concentrations in undeveloped or non-agricultural watersheds have been shown to be predominantly derived from atmospheric deposition (Valiela and Costa, 1988; Castro *et al*., 2003) but fertilizer and waste sources may also play a role.

Several studies (e.g., Ricker *et al*., 1994; Castro *et al*., 2003; Kroeger *et al*., 2006; Oakley *et al*., 2010; Wang *et al*., 2013) have applied model approaches to estimate OWTS and/or CSS inputs to adjacent surface waters. However, these models may not always be verified with field samples that show actual surface water impairments. Kroeger *et al*. (2006) found that their model estimates typically underestimated measured load to receiving waters downgradient from a CSS wastewater treatment plant (WWTP). This study attempted to show if any actual surface water N impairment occurs from OWTS and CSS use at the watershed scale in the North Carolina Coastal Plain.

The study objectives were to determine if surface water TN concentrations and watershed TN exports were affected by watershed wastewater management approaches. To achieve these objectives, several hypotheses were tested: (1) Surface water TN concentrations in OWTS watersheds are greater than those in CSS-served watersheds; and (2) Surface water TN loads are greater in OWTS watersheds than those in CSS-served watersheds. This work will help improve understanding of non-point source contributions of N to Coastal Plain surface waters and help regulators in the decision-making process to determine if OWTS should be included in nutrient-sensitive watershed management approaches.

Methods

Site Selection

Eight sub-watersheds located in Greenville, NC were selected based on wastewater management approaches (Fig. 2; Appendix A). Four watersheds used OWTS (EP-O, EP-1, MILL, and CHOK), while the other 4 used CSS (FT-1, FT-2, BELL, and MHB) were chosen and compared based on physical and chemical water quality parameters. The main comparison groups were: EP-O, MILL, and CHOK vs. FT-1, FT-2, and MHB (Table 10). Additional data were collected from 3 other watersheds; 2 of these watersheds used CSS (FT-O and BELL) and the last used OWTS (EP-1). FT-O and EP-1 were not included among the comparison groups because FT-O represents the confluence of FT-1 and FT-2, while EP-1 drains into EP-O. Including FT-O and EP-1 among the main comparison groups would skew the data since these data were accounted for in FT-1 and FT-2 and EP-O. BELL was only used for the collection of N concentrations data.

Table 9. Wastewater exports from watersheds shown in TN export (kg/day) and approximated TN export (kg/yr/ha) based on estimated watershed area for watersheds in Eastern USA. ACPB= Atlantic Coastal Pine Barrens, NC= North Carolina

Reference	# Of OWTS	TN Export (kg/day)	Approximated TN Export (kg/yr/ha)	Watershed Name	Watershed Area (ha)	Physiographic Province
Valiela *et al*. (1997)	app. 4230	30.4[a]	2.22	Waquoit Bay	app. 5000	ACPB
Bowen and Valiela (2001)	app. 4230	28.8	2.10	Waquoit Bay	app. 5000	ACPB
Bowen *et al*. (2007)	app. 4230	30.1	2.20	Waquoit Bay	app. 5000	ACPB
Pradhan *et al*. (2007)	app. 1.4 mil	39353 (3557)[b]	1.14	All Major NC Watersheds	1.26E+07 (1.31E+06)	NC

[a]= Estimated using an 65% attenuation factor reported in Valiela *et al*. (1997)
[b]= Assumes no N attenuation from OWTS technologies, parentheses data shows Neuse and Tar-Pamlico River Basin average

Table 10. OWTS and CSS watershed characteristics at each site showing wastewater management approach, major river basin location, watershed area, and total impervious surface. Total impervious surface (%) and area data was calculated by the Pitt County Planning Department (2011), excluding CHOK TIA.

Watershed Name	Wastewater Management Approach	River Basin	Area (ha)	Impervious Surface (%)
FT-1	CSS	Neuse	220	25.50%
FT-2	CSS	Neuse	190	33.60%
MHB	CSS	Tar	268	32.00%
EP-O	OWTS	Tar	201	9.60%
MILL	OWTS	Tar	200	11.70%
CHOK	OWTS	Tar	280	12.4%[a]

[a]= Estimated from Hardison *et al*. (2009)

Stream Instrumentation, Sample Collection and Analysis

Each stream was instrumented with a staff gauge. *HOBO* water level loggers were installed in each PVC stilling well programmed to record stream height every half-hour from August 2011 – August 2012. The logger data allowed for a long-term window of data, whereas staff gauges only showed a snapshot of stream height. Furthermore, the stream stage allowed for development of discharge rating curves, which were used to determine discharge data using logger data.

Stream water quality and discharge sampling events occurred monthly for 1 year from August 2011 - 2012. Physical water quality parameters were collected in field near staff gauge locations (Fig. 2). The *YSI-556 MultiProbe Meter* was used to determine the pH, dissolved oxygen (DO), electrical conductance (EC), temperature, and a *Hach* turbidity meter was used for turbidity measurements for each stream. The stream stage was read each month during sampling. A flow meter was initially used to gauge stream velocity. However, due to drought conditions (August 2011-November 2011) shortly after sampling began stream flow was too low for the meter to record velocity. Therefore, the floating object method (WV DEP, 2013) was used instead and for consistency this method was continued throughout the study after drought conditions subsided. Three trials of the floating object method were conducted and the average was accepted as the stream velocity. Stream discharge was calculated by multiplying average stream depth and velocity by stream width. Since the floating object method does not account for stream velocity differences with depth, velocities were multiplied by a coefficient calculated based on multiple floating object and flow meter trials (Appendix D). These data revealed that on average the float method estimated velocities were 27% greater than the flow meter. To correct for this overestimate the coefficients (0.76-1.00) in Appendix D were used.

New polypropylene sample bottles were rinsed 3 times in stream water, collected, and transported to the Central Environmental Laboratory at East Carolina University. In addition to stream samples, the GUC WWTP was sampled monthly from March 2012 to August 2012. Approximately 10% of the samples were replicates and blanks. Replicates differed from original samples by an average of 0.03 for TN and 0.01 for Cl^- (n=44). Blank samples had a mean of 0.09 (± 0.03 mg/L) TN and 0.26 (± 0.35 mg/L) Cl^- (n=10). Stream flow, environmental readings, and stream samples were collected before, during and after two storm events using the same methods. One storm occurred in the wet season (November 5-7, 2011- 3.43 cm) and in the dry season (May 9-10, 2012- 0.58 cm). Physical water quality data for each watershed and sub-watershed is characterized in Appendix L.

Samples were filtered the day of collection and stored overnight in a refrigerated storage room or were frozen (Avanzino and Kennedy, 1993) until analysis could be conducted. Samples were analyzed for ammonium (NH_4^+), nitrate and nitrite (reported as nitrate: NO_3^-), dissolved kjeldahl N, particulate nitrogen (PN), and chloride (Cl^-) using a *SmartChem 200* color spectrometer (WestCo, 2008). Storm and baseflow samples from November and May were sent to the Stable Isotope Facility at UC Davis for isotopic analysis. This facility uses a ThermoFinnigan GasBench plus PreCon trace gas concentrations system interfaced to a ThermoScientific Delta V Plus isotope-ratio mass spectrometer (Bremen, Germany).

Data Analysis

Annual discharge data were calculated using discharge rating curves (Appendix M) created for each of the 6 main watersheds based on monthly monitoring events and during storms. Using 2 different trend lines (low-flow and higher-flow) (Appendix M), discharge was calculated every 30 minutes over a year for each of the 6 primary watersheds based on stream height from logger data. BELL did not show a strong R^2 value and was not among the primary watersheds. Therefore, BELL was not included among these estimates. Due to channel instability, the stage-discharge relationship was unreliable. Therefore, BELL was not utilized for discharge and loading estimates. Logger and manually measured stage readings were compared to ensure data were similar (Appendix N). Discharge data were plotted against time (Appendix O) and correspond to precipitation data (Appendix P).

Hydrograph separation (using web analysis hydrograph tool: Lim *et al*. (2005)) was conducted by uploading 30-minute interval discharge data. A recursive digital filter using the aquifer type "perennial streams with porous aquifers" was selected. The filter parameter was 0.98 and the baseflow index maximum was 0.80 for each of the 6 primary watersheds. More information about the recursive digital filter method is available in Eckhardt (2004). Using hydrograph separation, the percent of annual flow (calculated based on measured monthly sampling events) that occurs as baseflow and storm flow was derived. The annual flow was multiplied by these percentages to determine total annual baseflow and storm flow discharges. Baseflow and storm flow TN concentrations were multiplied by annual baseflow and storm flow discharges to determine baseflow and storm flow TN exports. These exports were summed to determine total TN exports for each main watershed.

Discharge was multiplied by stream TN concentration to estimate stream TN export. Since watershed size varied, the estimated TN export was divided by watershed size to show kg-N/yr/ha per each main watershed. Using a normalized TN export, which allowed for a more uniform comparison between watersheds of different extent. Due to the lack of a consistent stage-discharge relationship, the normalized TN export for BELL could not be reported. If BELL surface water TN concentrations are similar to MHB, due to their proximity and stream characteristics, it is possible that TN export (kg/yr/ha) may be similar.

CSS technologies treat waste from residences served and then discharge treated waste into a river. The studied CSS watersheds were not directly influenced by these point-source discharges because the GUC WWTP discharges the treated effluent directly to the Tar River downstream of the studied watersheds. Therefore, the effects of CSS effluent discharges are not observed in these watersheds. In order to estimate the N exports from the CSS watersheds that are discharged at the GUC WWTP outfall on the Tar River a per capita N loading estimate was scaled up using watershed population data and water use. The number of residential structures was calculated from 2011 satellite imagery using GIS at the Pitt County Planning Department (2011; Appendix A). US Census (2013b) for Greenville, NC data were used to determine the average people per household. The number of people in the watershed was calculated by multiplying the structure count and average person per household. Total water use in the watershed was determined by multiplying the number of people in the watershed by 190 L/d, which was based on Eastern Pines Water Corporation for OWTS sites and compared to US EPA (2002) estimates. Using the TN concentration of the GUC WWTP effluent, the TN export was calculated as if that waste were discharged to the watershed that generated it.

Watershed TN export at CSS sites was assumed to be the estimated TN mass from non-wastewater sources (i.e. fertilizer, soils, and atmospheric deposition of N). TN export from OWTS watersheds was subtracted from TN export from CSS watersheds to determine an approximate watershed-scale export of TDN from OWTS sources. This allows for a watershed scale attenuation estimate between estimated N that is loaded from the OWTS tank and N that exports from the 3 primary OWTS watersheds after per capita scaling. This model assumes that all residential sites have similar OWTS loads to soils and groundwater loads to adjacent surface waters. Additionally, the model assumes that OWTS N characteristics are similar in the MILL and CHOK watersheds.

Results

OWTS and CSS Stream Discharge

OWTS average monthly and annual discharge was greater than CSS discharges (Table 11). This difference was significant at $p = 0.01$. Baseflow accounted for most of OWTS and CSS discharges. CSS watersheds had approximately 10% more discharge from storm events compared to OWTS watersheds. Total precipitation was similar at both CSS (126 ± 1.77 cm/yr) and OWTS (125 ± 0.81 cm/yr) watersheds. Total annual discharge represented approximately 14% at CSS and 21% at OWTS of total precipitation data. Therefore, evapotranspiration at CSS watersheds was approximately 86%, while at OWTS watersheds it was 79% (Table 11). Hydrograph separation showed that baseflow contributed 64% of total annual flow at CSS watersheds, whereas at OWTS watersheds baseflow contributed 75% (Appendix M). Stormwater runoff contributed 36% and 25% of annual flow at CSS and OWTS watersheds, respectively (Appendix M).

Table 11. OWTS and CSS discharge measurements compared to annual precipitation in each of the watersheds and pooled. Q= average annual discharge, mil= million, BF= baseflow, SF= storm flow, and Tot. Precip.= total precipitation, ET= evapotranspiration, ET shows the volume of water from August 2011 to August 2012 that evaporated and transpired, while ET represents the percentage of total precipitation that evaporated or transpired.

Site	Avg Annual Q (mil m^3/yr)	BF (%)	SF (%)	BF Q (mil m^3/yr)	SF Q (mil m^3/yr)	Tot. Precip. (cm/yr)	Total ET (cm/yr)	Total Q (cm/yr)	ET (%)
FT-1	0.41	63	37	0.26	0.15	146	127	19	87.3
FT-2	0.4	58	42	0.23	0.17	146	125	21	85.7
MHB	0.56	64	36	0.36	0.2	145	124	21	85.8
EP-O	0.67	77	23	0.51	0.15	143	110	33	76.7
MILL	0.53	74	26	0.39	0.14	145	118	27	81.4
CHOK	0.87	71	28	0.62	0.24	145	113	31	78.3
CSS	0.45 (0.09)	62 (3.32)	38 (3.32)	0.28 (0.07)	0.17 (0.03)	146 (0.87)	126 (1.77)	20 (1.26)	86.2 (0.92)
OWTS	0.69 (0.17)	71 (2.89)	29 (2.40)	0.51 (0.11)	0.18 (0.06)	144 (0.87)	125 (0.81)	21 (0.25)	78.7 (4.59)

Baseflow and Storm Surface Water N Quality

Baseflow and Storm Surface Water N Speciation

Surface water TN concentrations for CSS and OWTS watersheds were dominantly NO_3^- rich. N speciation was similar at all sites except the FT-1 and FT-2 sub-watersheds (Fig. 13). During baseflow and storm conditions, FT-1 and FT-2, surface water TN was mostly composed of dissolved organic nitrogen (DON), accounting for approximately 40% of TN. NH_4^+ also contributed about a quarter, if not more, to TN concentrations at these watersheds during both baseflow and storm conditions (Fig. 13). NO_3^- was the dominant N species at each of the other watersheds during baseflow conditions. Under baseflow, NO_3^- contributed approximately half of total TN. During storm conditions, this trend prevailed for MHB and BELL. However, during storm events at MILL, DON contributed a substantial portion of TN. At MILL, DON was the dominant N species during storms, although NO_3^- still represented approximately a third of TN (Fig. 13).

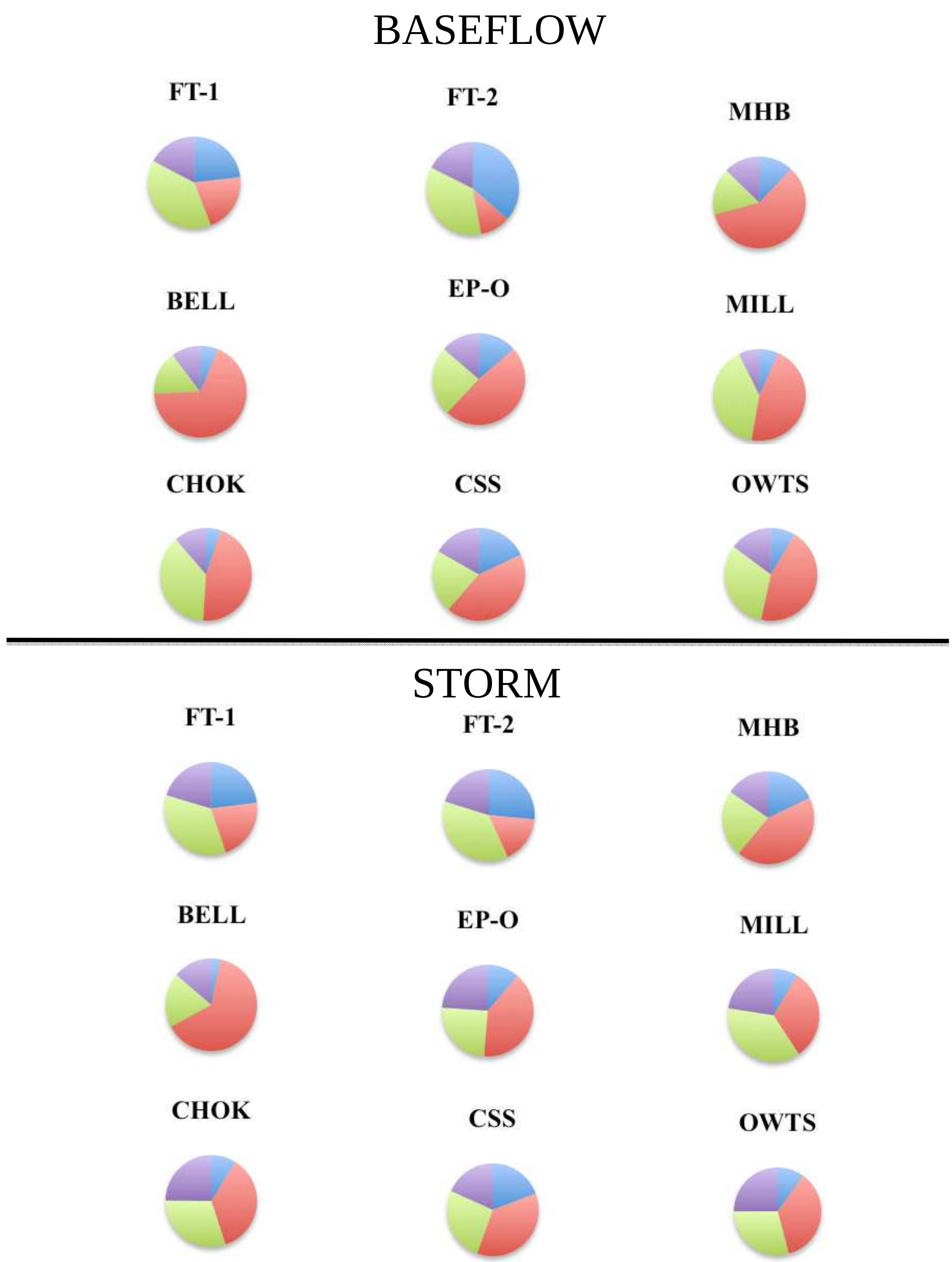


Figure 13. A) Baseflow N speciation at the 6 main watersheds and BELL. B) Storm N speciation at the 6 main watersheds and BELL. Blue represents NH_4^+, red represents NO_3^-, green represents DON, and purple represents particulate nitrogen (PN).

Baseflow and Storm Surface Water N Concentrations

TN concentrations in surface water during baseflow conditions differed between CSS and OWTS watersheds (Fig. 14). DIN comprised more than 50% of TN for both OWTS and CSS watersheds. DON accounted for approximately 30% of surface water TN for both OWTS and CSS watersheds. Less than 20% of TN surface water concentrations during baseflow occurred as PN (Appendix Q). The mean surface water TN concentration for CSS watersheds during baseflow conditions was 0.86 (± 0.50 mg/L). Surface water TN concentrations during baseflow conditions for OWTS watersheds (1.27 ± 0.46 mg/L) were approximately 2 times greater than CSS watersheds (Fig. 14), which was significantly different ($p = 0.00$).

During storm conditions, the mean concentration for surface water DIN, DON, TDN, PN, and TN slightly increased from baseflow conditions. DIN remained as the dominant contributor to TN concentrations, contributing approximately 50% of surface water TN to CSS and OWTS watersheds. DON remained the same, accounting for 30% of TN. Surface water PN concentrations increased during storms relative to baseflow conditions at both CSS and OWTS streams, contributing approximately 20% of TN (Appendix R). Mean surface water TN concentrations increased from baseflow to storm conditions, and then mean TN concentrations decreased after storms at both OWTS and CSS sites. OWTS watersheds had greater mean surface water TN concentrations during storms (1.43 ± 0.44 mg/L) and after storms (1.33 ± 0.32 mg/L) than CSS watersheds (storms: 0.97 ± 0.32 mg/L, after: 0.72 ± 0.31 mg/L).

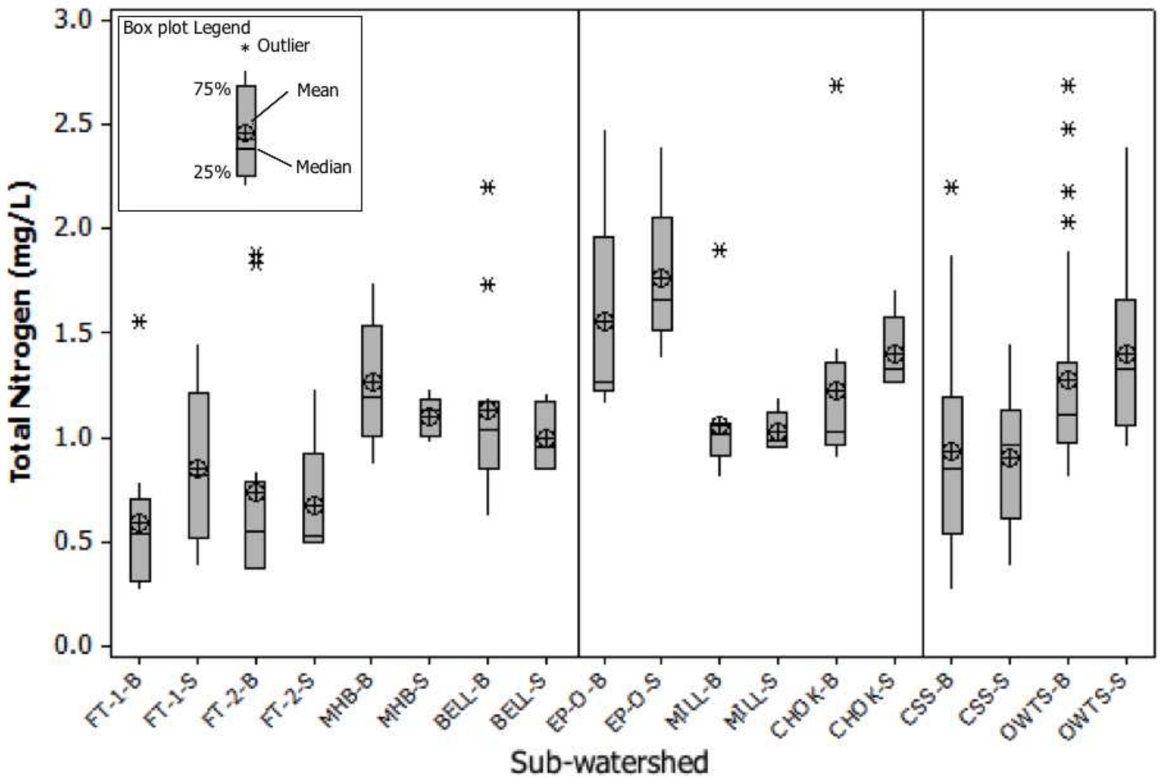
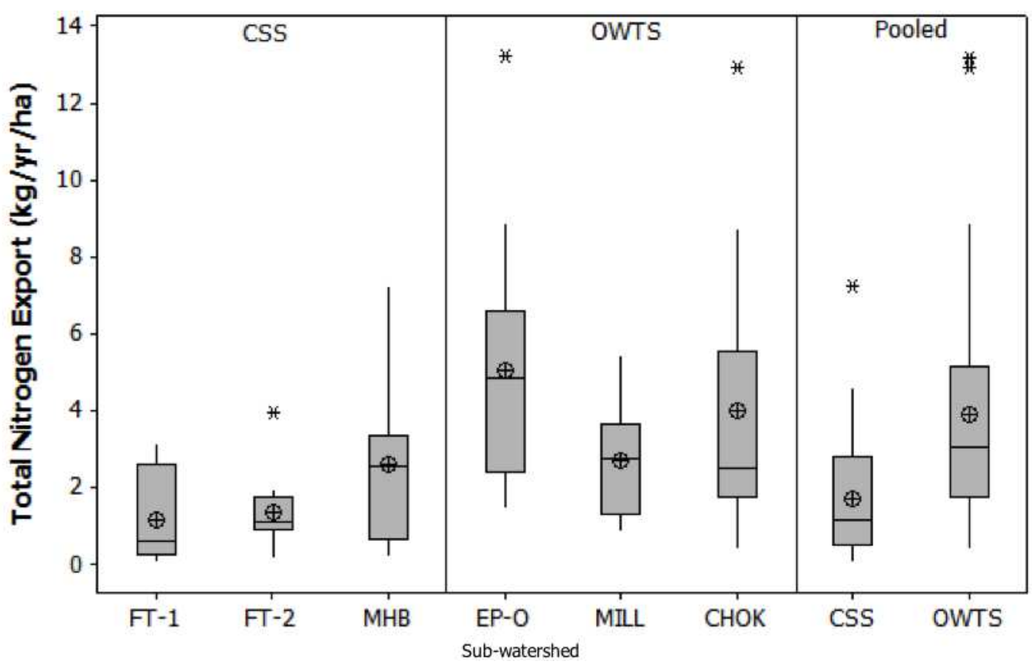


Figure 14. Baseflow and storm surface water TN concentrations at individual watersheds and pooled. Concentrations remained similar between baseflow and storm conditions, although a slight increase in median and mean values occurred from baseflow to storm conditions. B= baseflow, S= storm flow.

Baseflow and Storm Surface Water N Exports

Mean TN export from OWTS (3.92 ± 2.80 kg/yr/ha) watersheds was approximately two times greater than that of CSS (1.73 ± 1.56 kg/yr/ha) TN watershed export (Fig. 15). OWTS and CSS TN export were significantly different at a 95% confidence interval at p= 0.00. EP-O exported the highest mean TN mass at 5.08 (± 3.37 kg/yr/ha). CHOK and MILL exported lower, though similar, mean TN masses at 3.99 (± 3.62) and 2.71 (± 1.43) kg/yr/ha, respectively (Fig. 15). TN export was similar among CSS watersheds. MHB exported the highest mean TN among the CSS watersheds at 2.63 (± 1.98) kg/yr/ha. FT-2 and FT-1 exported lower mean TN masses at 1.37 (± 0.99) and 1.18 (± 1.21) kg/yr/ha (Fig. 15). Assuming CSS TN export represents non-wastewater sources, then approximately 2.2 kg/yr/ha of TN export was resultant from wastewater N contributions in watersheds served by OWTS.

Unlike baseflow conditions, OWTS (0.08 ± 0.14 kg/day/ha) streams exported similar mean TN masses during storm events to CSS streams (0.06 ± 0.07 kg/day/ha) (Fig. 15). CHOK (0.14 ± 0.23 kg/day/ha) and MHB (0.09 ± 0.11 kg/day/ha) exported the greatest mean storm event loads of TN. FT-1 exported the lowest mean TN storm event loads at 0.03 (± 0.03) kg/day/ha (Fig. 15). DIN, DON, TDN, and PN loading are also available in Appendix R.



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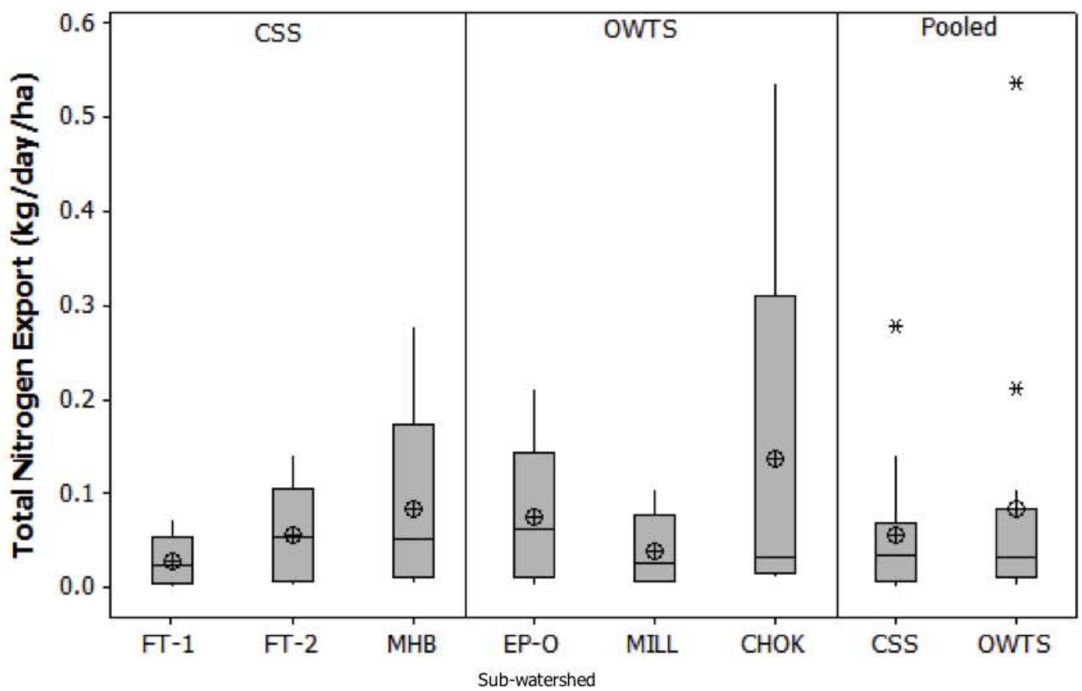


B

Figure 15. A) TN export (kg/yr/ha) at individual watersheds and pooled during baseflow conditions. B) TN export (kg/day/ha) at individual watersheds and pooled during storm conditions. OWTS TN export was increased relative to CSS during baseflow conditions, while TN export was similar during storm conditions.

<u>GUC WWTP Net TN to Tar River from Average CSS Watershed Compared to Isolated OWTS Watershed TN Export</u>

TN export in surface water within the CSS-served watersheds did not account for the wastewater that was piped out of the watersheds and treated at the GUC WWTP, which was later discharged in downstream reaches of the Tar River. Therefore, this section shows estimated TN input to the GUC WWTP scaled up based on average CSS watershed population and water use. The TN input to influent tanks at the GUC WWTP was 7434 (± 4113 kg/yr) on average. Average TN export from the GUC WWTP to the Tar River was 1402 (± 776 kg/yr), which showed 81% TN attenuation (Fig. 16). OWTS watershed TN attenuation was the same as at the GUC WWTP. OWTS TDN loads to soils were 3176 (± 270 kg/yr). Average TN export from surface water in OWTS-served watersheds was 605 (± 322 kg/yr) (Fig. 16).

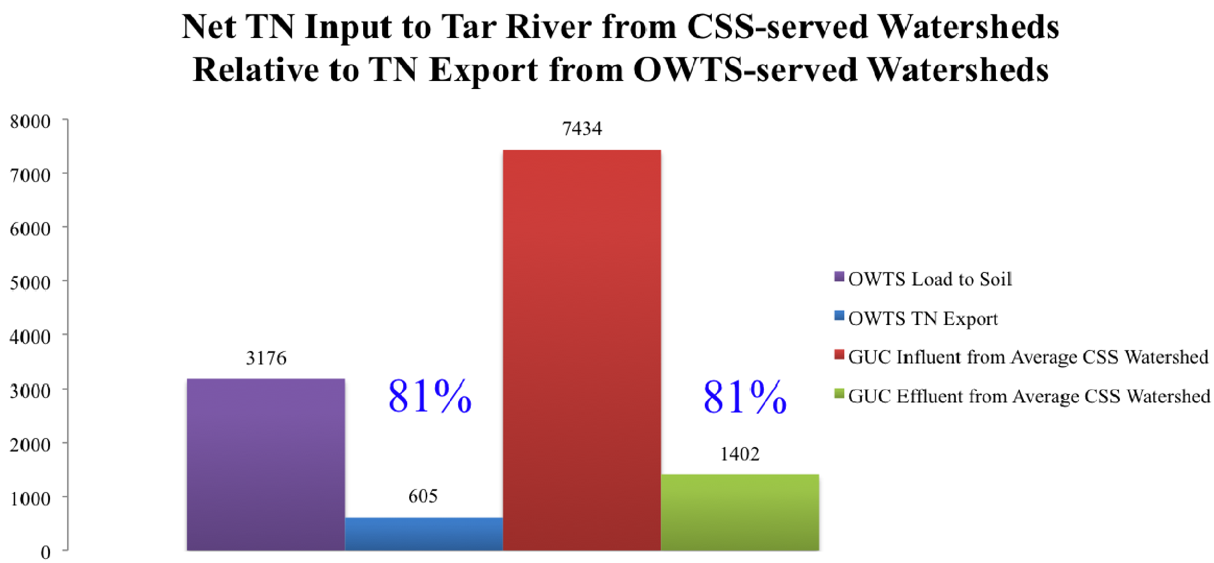


Figure 16. TN (kg-N/yr) load from OWTS to soil (purple), surface water TN export from OWTS-served watersheds (blue), TN inputs to influent at GUC WWTP (red), and effluent TN inputs to Tar River from GUC WWTP (green). GUC WWTP inputs were based on the average population and water use among the 3 major CSS watersheds. TN concentrations were based on average GUC influent and effluent TN concentrations. The blue text shows the attenuation between the OWTS and CSS treatment approaches.

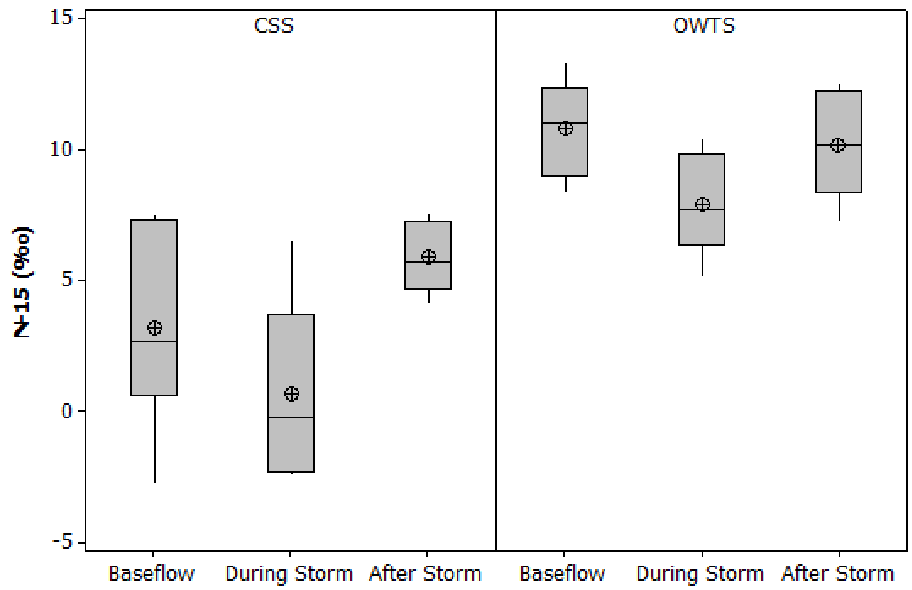
Stable Isotope Indicators of the Sources of Nitrogen

Baseflow Stable Isotope Indicators

Baseflow $\delta^{15}N$ (‰) stable isotope indicators showed that $\delta^{15}N\text{-}NO_3^-$ values differed in OWTS vs. CSS watersheds. Baseflow surface water $\delta^{15}N\text{-}NO_3^-$ values (mean: +3.20 ± 3.83‰) in CSS watersheds were depleted relative to surface water $\delta^{15}N$ (mean: +10.8 ± 1.84‰) in OWTS watersheds. Baseflow $\delta^{15}N\text{-}NO_3^-$ signatures in OWTS watersheds ranged from +8.35 to +13.3‰. Baseflow $\delta^{15}N\text{-}NO_3^-$ signatures in CSS watersheds ranged from -2.71 to +7.30‰ (Fig. 17; Appendix S). OWTS and CSS $\delta^{15}N\text{-}NO_3^-$ signatures were significantly different ($p= 0.00$) from each other during baseflow, during storms, and after storm conditions. OWTS baseflow NO_3^- concentrations were significantly greater than CSS NO_3^- concentrations. However, during storms and after storms NO_3^- concentrations were similar when comparing CSS and OWTS watersheds.

Storm Stable Isotope Indicators

A trend of depleted $\delta^{15}N\text{-}NO_3$ signatures during storms relative to baseflow conditions occurred at both CSS and OWTS watersheds. During storm events surface water $\delta^{15}N\text{-}NO_3$ values (mean: +0.70 ± 3.44‰) in CSS watersheds were depleted relative to those from OWTS watersheds ($\delta^{15}N\text{-}NO_3$ mean: +7.92 ± 1.93‰). During storm events surface water $\delta^{15}N\text{-}NO_3^-$ values ranged from -2.43 to +6.56‰ in CSS watersheds, while surface waters had $\delta^{15}N\text{-}NO_3^-$ values ranging from +5.15 to +10.5‰ (Fig. 17) in OWTS watersheds during storm events. At the OWTS watersheds, after storm $\delta^{15}N\text{-}NO_3$ values (mean: 10.2 ± 2.00‰) transitioned closer to pre-storm conditions. However, CSS watersheds contained after storm $\delta^{15}N\text{-}NO_3$ values (mean: +5.90 ± 1.38‰) that were enriched relative to pre-storm conditions (Appendix S).



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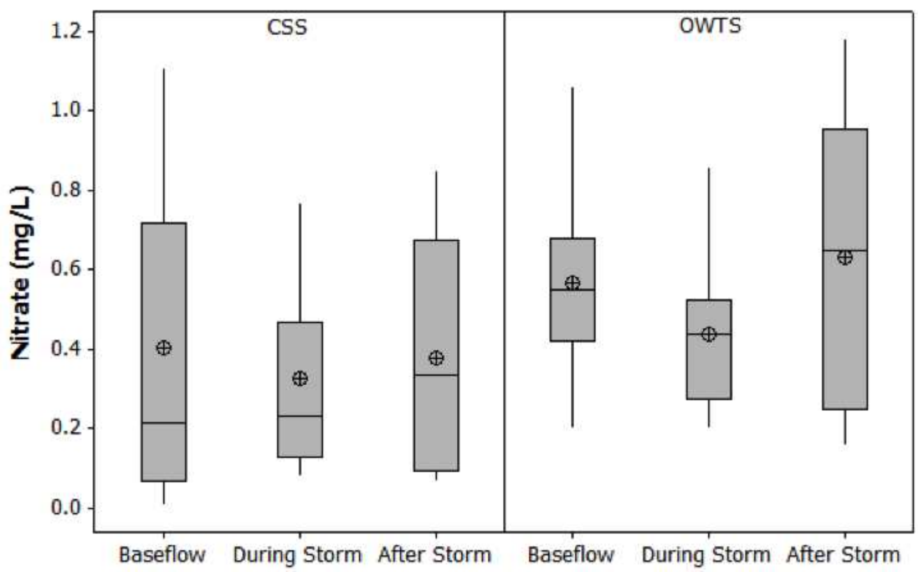

B

Figure 17. A) $\delta^{15}N$ stable isotope indicators for CSS and OWTS during baseflow conditions, storm conditions, and post-storm conditions (falling limb of the storm hydrograph). OWTS sources show depletion during storm and a return to pre-storm conditions after storms, while CSS show depletion but no return to post-storm conditions immediately following the peak of storm activity. B) NO_3^- concentrations during baseflow, during storm, and after storm conditions. NO_3^- concentrations decrease during storms, and then increase to conditions similar to baseflow conditions.

Discussion

Hypothesis 3: Surface water TN concentrations in OWTS watersheds were greater than CSS

OWTS watersheds had significantly higher surface water TN concentrations than CSS watersheds. This difference in TN concentrations (0.41 mg/L in OWTS during baseflow conditions) suggested that OWTS use affected surface water quality during baseflow conditions. The similarity between baseflow and storm flow TN concentrations suggested that groundwater sources of TN were more dominant than atmospheric sources, potentially suggesting that OWTS contributed most of the TN in OWTS watersheds. Individual and pooled TN concentrations in surface water were similar or less than Dodds *et al*. (2009) estimated value of 1.2 mg/L of southern coastal plain rivers. Dodds *et al*. (2009) estimated mean TN concentrations based on 90 gauging stations that collected TN data across the southern coastal plain. Smith *et al*. (2003) found that median TN concentrations in undisturbed streams in the Eastern Coastal Plain region were approximately 0.52 mg/L, ranging from approximately 0.30 to 0.70 mg/L. The current study found mean OWTS surface water TN to be approximately 3 times greater than median TN reference data (Smith *et al.*, 2003), but were similar to surface water TN concentrations determined by Dodds *et al*. (2009). Mean TN concentrations in surface water served by CSS watersheds were less than TN concentrations in surface water served by OWTS watersheds and findings by Dodds *et al*. (2009), but were elevated compared to TN reference data (Smith *et al*., 2003).

Previous studies have shown OWTS to contribute significant N concentrations to surface waters. Surface water DIN (0.67 ± 0.24 mg/L) concentrations during baseflow and storm flow conditions in OWTS watersheds were similar to those found in watersheds in Massachusetts by Valiela and Costa (1988). However, they reported N concentrations as DIN, which excludes

DON data. They found that DIN concentrations ranged from approximately 0.10 to 2.55 mg/L based on field data in Buttermilk Bay (Cape Cod, MA). Inclusion of DON data may expand this range. Neilson and Cronin (1981) showed that in some coastal embayments and estuaries across the world DIN concentrations can be as high as 23 mg/L.

Surface water $\delta^{15}N$-NO_3^- signatures in OWTS watersheds were similar to those found by previous studies. Kendall and McDonnell (1998) showed that $\delta^{15}N$ signatures in manure and septic effluent ranged from +10 to +20‰. They also showed that $\delta^{15}N$ signatures ranged from +4 to +7‰ for soil organic matter and -5 to +5‰ for ammonia fertilizer. Silva *et al*. (2002) redefined these ranges and found that $\delta^{15}N$ signatures for manure and septic effluent ranged from +8 to +20‰ and +2.5 to +9‰ for soil organic matter. They found similar $\delta^{15}N$ signature ranges for ammonia fertilizer as Kendall and McDonnell (1998).

The current study found $\delta^{15}N$ significantly differed ($p= 0.00$) between surface water in OWTS-served watersheds and surface water in CSS-served watersheds. Most of the surface water $\delta^{15}N$ data in OWTS-served watersheds plotted within the manure and septic effluent range defined by Silva *et al*. (2002) (Fig. 17). Based on surface water $\delta^{15}N$ data, the dominant source of NO_3^- in OWTS-served watersheds was septic effluent. Residential land classes prevailed throughout the watershed. Furthermore, livestock was not observed within close proximity of any surface water sampling location. Most of the surface water $\delta^{15}N$ data in CSS-served watersheds plotted within the ammonia fertilizer range defined by Kendall and McDonnell (1998) and Silva *et al*. (2002) (Fig. 17). Therefore, OWTS use contributed to the elevated N concentrations in surface water within OWTS-served watersheds.

Hypothesis 4: Surface water TN loads were greater in OWTS watersheds than CSS

OWTS watersheds exported significantly more N than CSS watersheds during baseflow conditions. Furthermore, when comparing the mean watershed TN export on an individual watershed basis, each OWTS stream exported significantly greater TN loads than CSS streams during baseflow. However, this trend was not observed during storm conditions. This is likely due to the total impervious area differences between watersheds. The increased total impervious area at CSS watersheds (approximately 30%), compared to OWTS watersheds (approximately 10%), generated more runoff during storms and these watersheds were flashy (Appendix M; Appendix O). The influx of greater runoff and the similar (relative to baseflow conditions) TN concentrations during storms in CSS-served watersheds caused increased TN loads. TN export at the MHB watershed tended to be nearly two times greater than the FT-1 and FT-2 watersheds. This trend could be related to leaky CSS infrastructure. Upgradient from the sampling point was an exposed sewer pipe directly above the stream channel that may have eroded and leaked wastewater into the stream. In addition, upstream there are two large farm fields, which could contribute fertilizer runoff. Comparing $\delta^{15}N$ signatures at MHB seem to support this explanation (Appendix S). Average $\delta^{15}N$ signatures at MHB were 6.31 (± 2.06‰), which were slightly out of the septic effluent range. Most points plotted near the septic effluent range. However, one point (3.26‰) suggested fertilizer as a source of NO_3^-, which could be related to the large farm fields upstream from the sampling location.

Recent studies (Table 9) have shown model estimates of TN exports from watersheds to larger basins or coastal waterways in eastern USA. Comparing the OWTS kg-N/yr/ha watershed N export data to the literature, these values were similar to those described in Table 9. The similarity showed that despite these model estimates using conservative treatment estimates or no treatment estimates, the results of the current study were similar to these models. CSS watersheds exported TN masses lower than all the described studies excluding Pradhan *et al*. (2007) and Harrison *et al*. (2012). Pradhan *et al*. (2007) estimated OWTS TN loads (1.30 kg/yr/ha) for a sub-basin in which the current study sites reside; this approximation is similar to the TN export estimate of 2.2 kg/yr/ha that only accounted for wastewater sources of N. Despite the similarities between the current study and Pradhan *et al*. (2007), they did not include attenuation factors. This similarity could be attributed to the inclusion of non-septic areas upon normalizing TN export based on area.

Although OWTS-served watersheds export greater TN compared to CSS-served watersheds, the N generated in these CSS-served watersheds reaches downstream segments of the Tar River from the GUC WWTP treated effluent discharges. Some studies (Valiela and Costa, 1988; Horsley Witten Hegeman Inc., 1991; Sham *et al*., 1995; Valiela *et al*., 1997; Bowen and Valiela, 2001; Kroeger *et al*., 2006; Bowen *et al*., 2007; Pradhan *et al*., 2007; Wang *et al*., 2013) estimated the N concentrations and/or loads to adjacent surface waters and/or at the watershed scale. The current study found an average attenuation factor between the 3 OWTS watersheds to be 81% (± 14%). These N attenuation factors at the watershed scale were slightly elevated compared to Valiela *et al.* (1997), which found N inputs to watersheds from wastewater sources attenuated approximately 65% of N prior to discharging into Waquoit Bay. This attenuation factor was calculated based on model estimates of N inputs from OWTS to watersheds and N exports from these watersheds to Waquoit Bay. A model that does not utilize an attenuation factor could overestimate N loads to surface water by up to 96% in watersheds of similar geological, physiographical, topographical, soil, and OWTS characteristics.

Conclusions

There were some limitations with the current study. This study was based in relatively small (< 400 ha) watersheds using predominantly the same land class. Therefore, replicating this study in watersheds of greater extent may not be possible due to labor needs and cost of supplies. Furthermore, the land classes may become more diverse in larger watersheds. In these cases, model estimates may be a more cost feasible option of estimating TN exports from watersheds of greater extent. However, replicating the current study in watersheds of similar size or similar land use in differing physiographic provinces, topographical, geological, and OWTS characteristics could provide additional TDN attenuation factors at the watershed scale. These data could help constrain model estimates in regions that differ from the study area of the current study. However, replication of the current study may be difficult to apply to some regions, such as the Piedmont and Blue Ridge regions of NC due to significant differences in geological settings.

Field data collected over a yearlong study period revealed that watershed wastewater management approaches affected surface water N concentrations and loads in the selected Coastal Plain watersheds. Surface waters within OWTS watersheds had greater TN concentrations and exports compared to those within CSS watersheds. OWTS watersheds annually exported approximately 2 times more TN than CSS watersheds during baseflow conditions. CSS can be useful in areas adjacent to nutrient sensitive waters, since effluent can be better controlled. CSS effluent, rather than discharging directly into nutrient sensitive waters, can be diverted to another less vulnerable watershed, can receive advanced tertiary treatment, or can be directly injected into the subsurface allowing for additional treatment prior to discharge.

The current study found that OWTS use could contribute up to 3.13 kg-N/yr/ha to adjacent surface waters in watersheds located in the Coastal Plain of North Carolina. This study estimated that OWTS watersheds attenuated up to 96% of TN exports, with an average of 81% (± 14%). The current study estimates can help provide an attenuation factor that may be used in future studies to help account for attenuation in watershed-scale N models. Since these attenuation factors were derived in Inner Coastal Plain settings, future work should aim to develop these factors in different hydrogeological settings to help improve understanding of N inputs from OWTS to surface waters. The results from the current study and future work should be considered among nutrient management strategies for the North Carolina Department of Environment and Natural Resources, as well as other state, federal, and international agencies in their planning nutrient management efforts.

CHAPTER 5: MANAGEMENT IMPLICATIONS

Results from this study indicate that OWTS use influences groundwater and surface water quality both locally and at the watershed scale. Meanwhile, CSS infrastructures have minimal impact on groundwater and nearby surface water at the residential scale. However, they may have local effects if the sewer infrastructure leaks. Groundwater and surface water N data showed that OWTS use has influenced groundwater and surface water quality and increased recharge from subsurface wastewater disposal led to increased discharge in OWTS watersheds. The increased TDN and TN concentrations in OWTS groundwater and surface water relative to CSS groundwater and surface water show that OWTS use affected water resources.

Based on the studied OWTS sites, there are potential suggestions that could mitigate the influence of these OWTS upon water resources. At sites where separation distance was inadequate (sites 100 and 200), it is possible to reinstall drainfield trenches at a shallower depth using shallow depth or low-profile chambers (CULTEC, 2010) to prevent the seasonal high water table from submerging the trench bottom. This could potentially facilitate increased nitrification at these sites, which may lead to increased TDN attenuation prior to discharge to nearby streams. Denitrification trenches are another potential strategy to mitigate nitrate inputs to surface water. These trenches could be installed at sites similar to sites 300 and 400 where most of the septic effluent occurs in the form of nitrate. A limiting factor for all of these potential management strategies is cost. Altering or moving OWTS components are costly, without grant funding these costs would otherwise be the responsibility of the homeowner. Planting of vegetated buffers along stream banks (where present) at these sites is a potentially cost feasible management strategy. Vegetation with deep root zones may be able to attenuate TDN significantly in areas where the depth to water is shallow.

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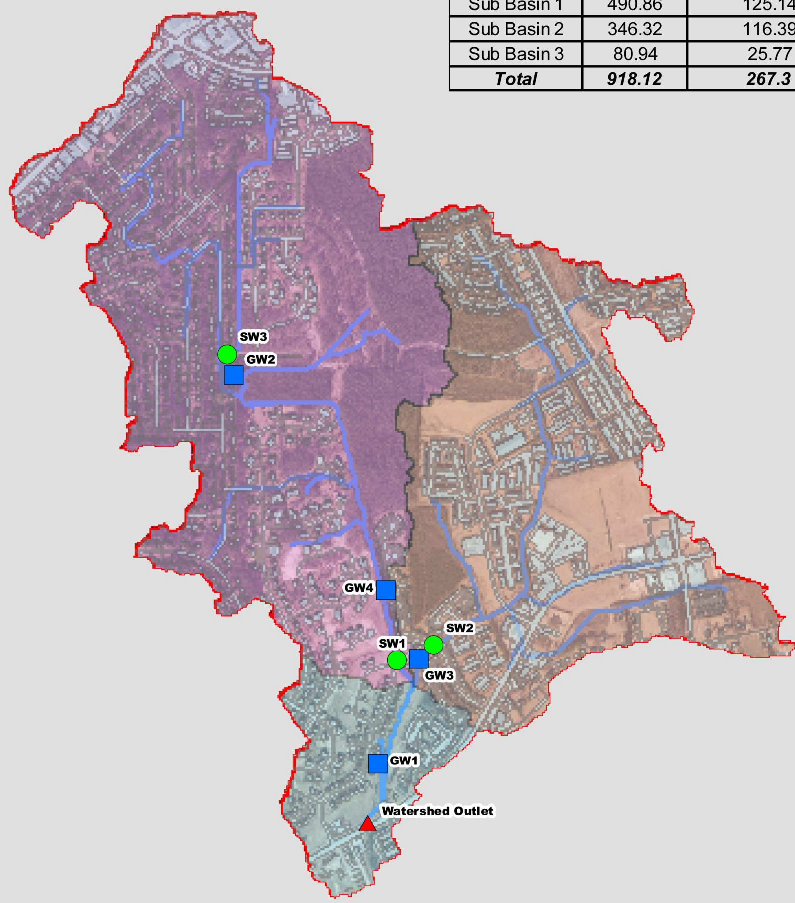
APPENDIX A: WATERSHED DELINEATION MAPS

Fire Tower Total Impervious Surface Area

	Acreage	Impervious Surface	**Percent**
Drainage Basin			
Sub Basin 1	490.86	125.14	**25.49%**
Sub Basin 2	346.32	116.39	**33.61%**
Sub Basin 3	80.94	25.77	**31.84%**
Total	***918.12***	***267.3***	***29.11%***



N

Legend

Monitoring Sites / Type

Ground Water

Outlet

Surface Water

impervious_surfaces

Study Basin

Drainage Features

Sub Basins

1

2

3

PittCounty Planning & Development

0 625 1,250 Feet

1 inch = 1,250 feet

PITT COUNTY NORTH CAROLINA

Aerial Image Source - January 2009

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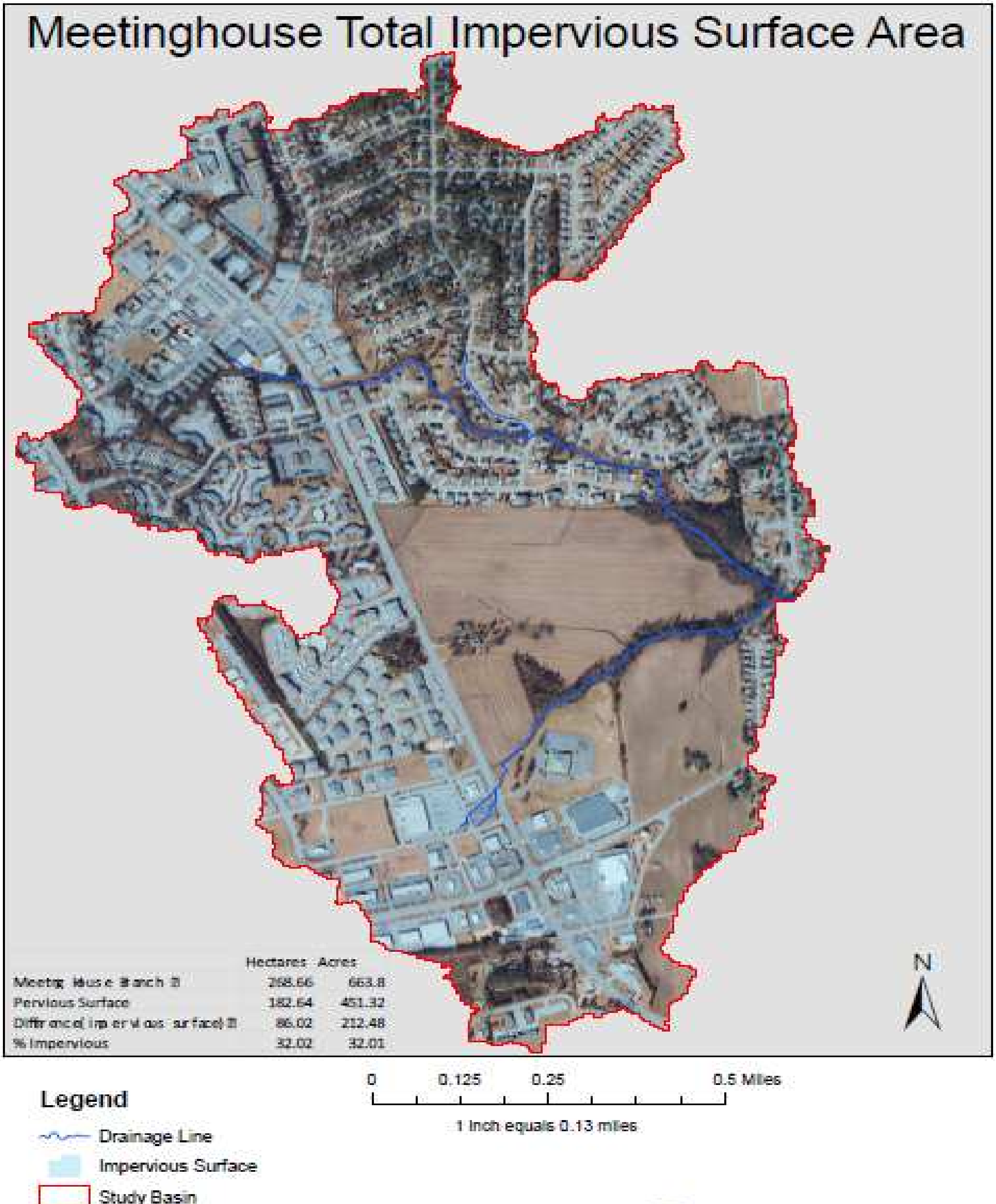

	Hectares	Acres
Meeting House Branch	268.66	663.8
Pervious Surface	182.64	451.32
Difference (impervious surface)	86.02	212.48
% Impervious	32.02	32.01

Aerial Image Source - January-March 2010

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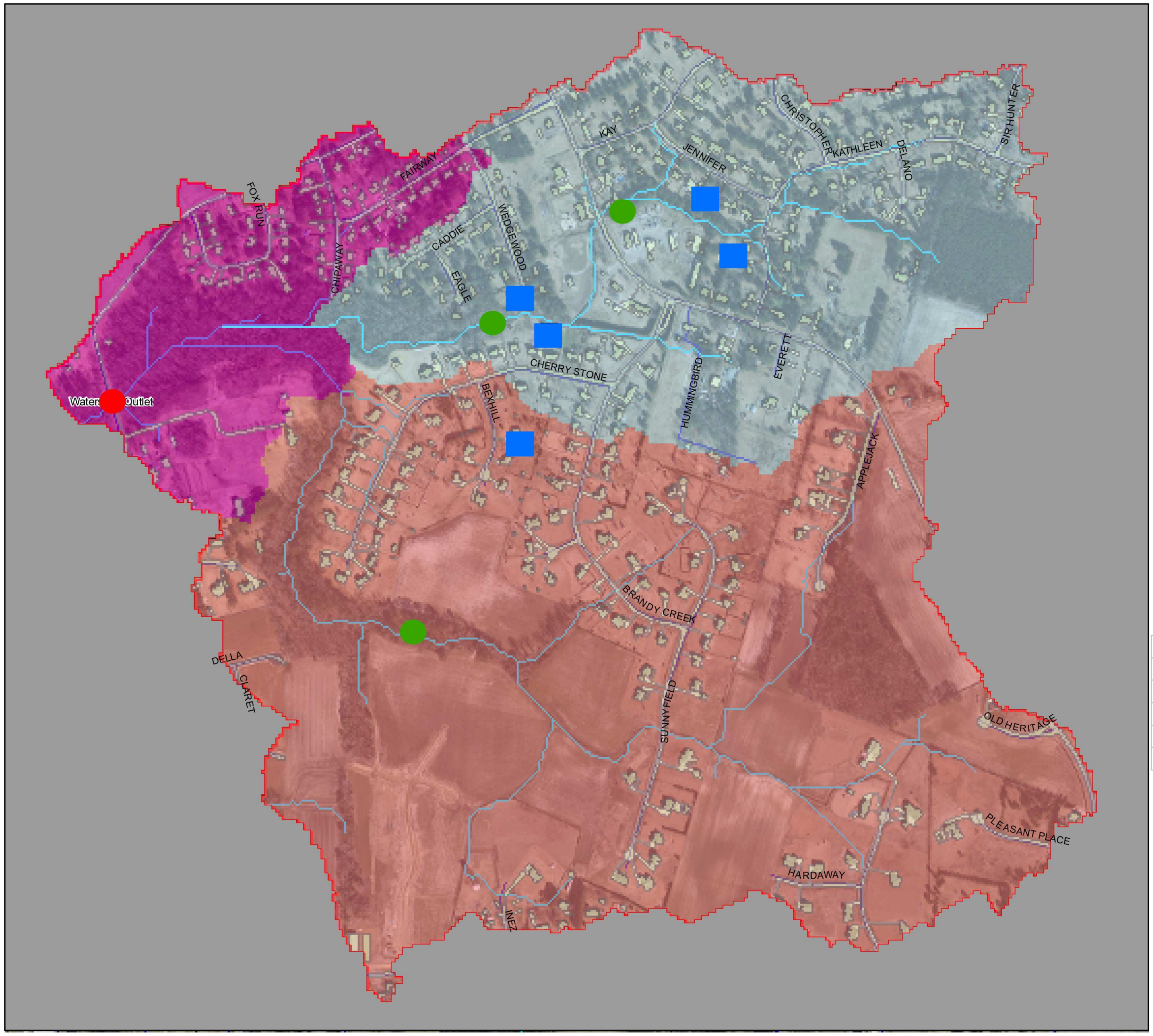

Eastern Pines
Total Impervious Surface Area

Legend

Monitoring Sites /Type

- Ground Water
- Outlet
- Surface Water
- Impervious Surfaces

SubBasins

- 1
- 2
- 3
- Drainage Features
- Roads
- Drainage Basin

	Acreage	Impervious Surface	Percent
Drainage Basin			
Sub Basin 1	145.14	18.94	13.05%
Sub Basin 2	276.23	20.59	7.45%
Sub Basin 3	58.18	6.49	11.16%
Total	**479.55**	**46.02**	**9.60%**

0 275 550 Feet

1 inch = 550 feet

PittCounty

Map Produced By:
Pitt County Planning Department

N

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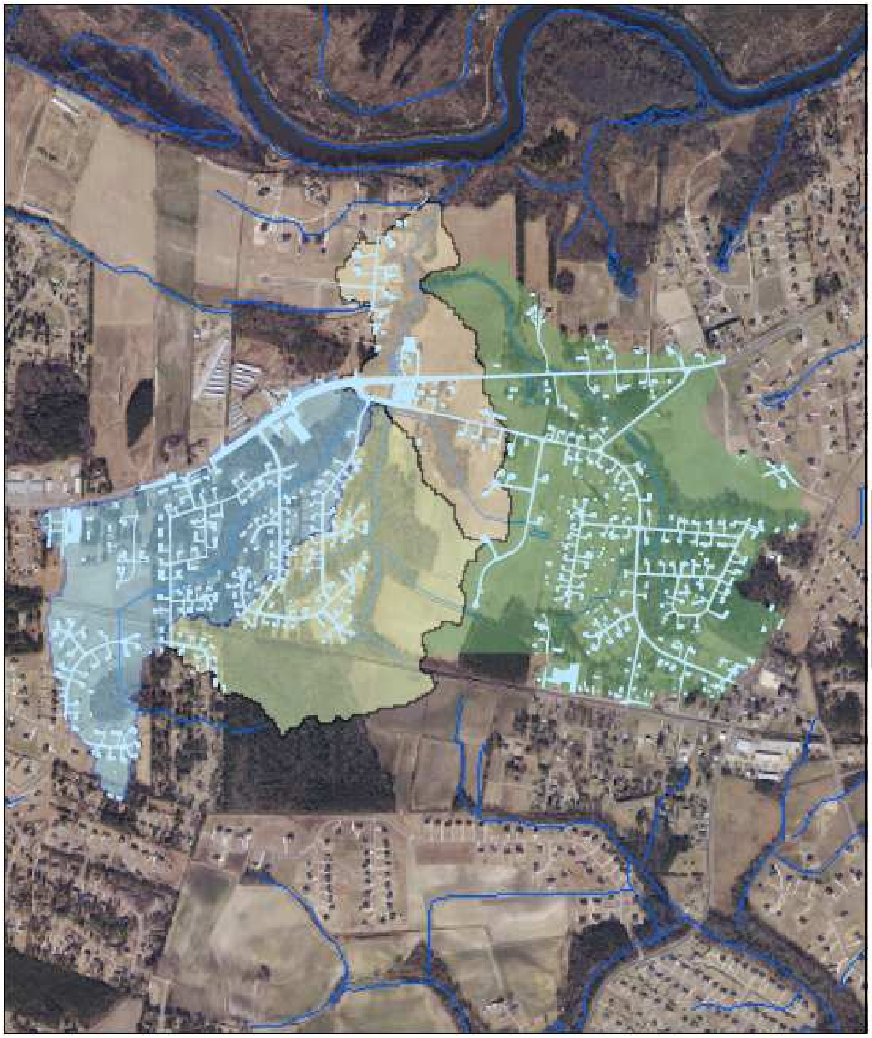

Simpson Total Impervious Surface Area

Legend

Impervious Surface

Hydrology

Study Basin

Sub Basins

Basin 1

Basin 2

Basin 3

Basin 4

Drainage Basin	Acreage	Imp. Surf. (acres)	Hectare	Imp. Surf. (Ha)	Percent
Sub Basin 1	*210.22*	*23.92*	*85*	*75*	11.38%
Sub Basin 2	*119*	*20.02*	*48*	*40.05*	16.82%
Sub Basin 3	*102.62*	*6.53*	*41*	*38.88*	6.36%
Sub Basin 4	*62.61*	*7.44*	*25*	*22.32*	11.88%
Total	**494.45**	**57.91**	**199**	**176.25**	11.71%

0 165 330 Meters

0 625 1,250 Feet

N W E S

PittCounty Planning & Development

Map Produced By:
Pitt County Planning Department
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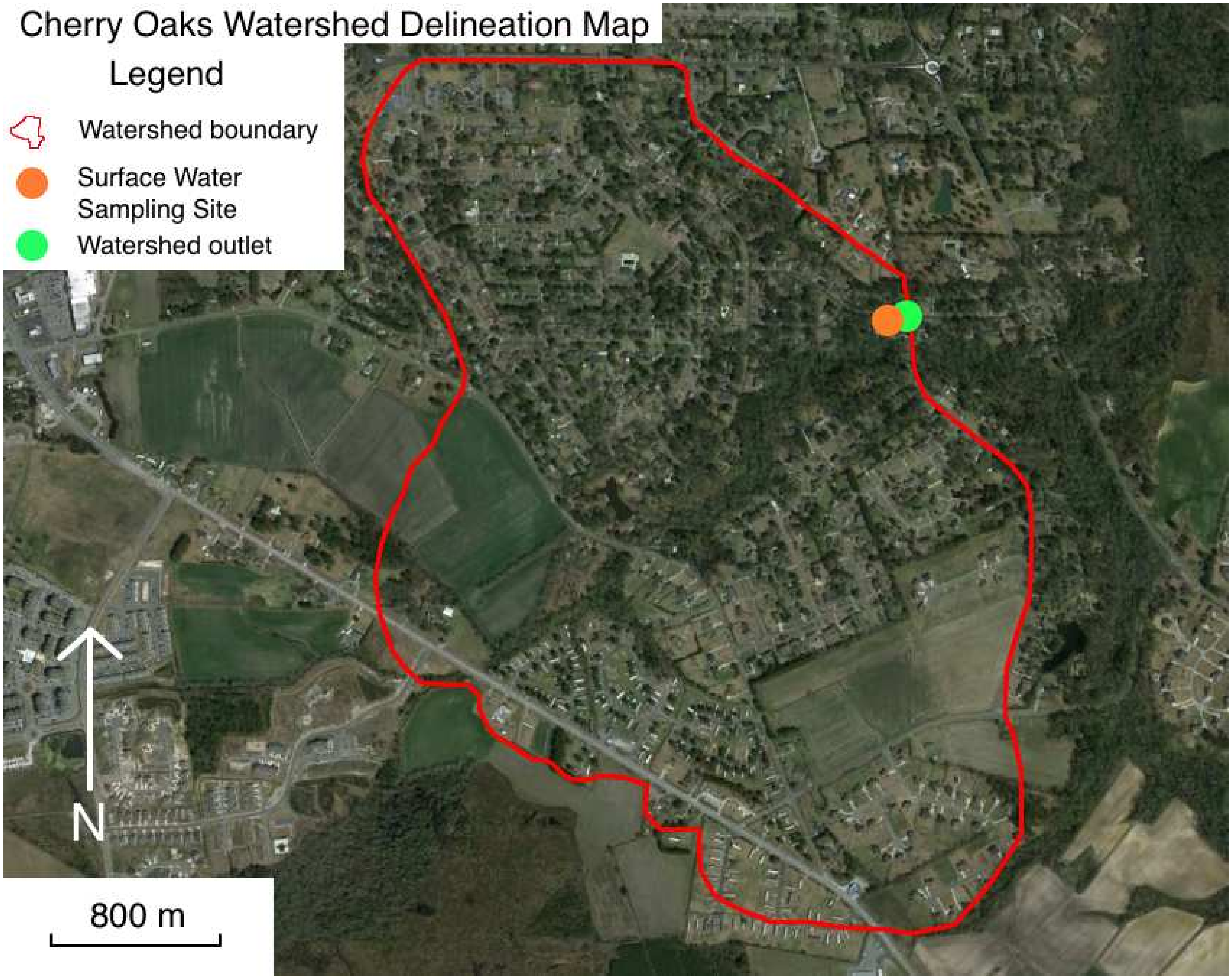
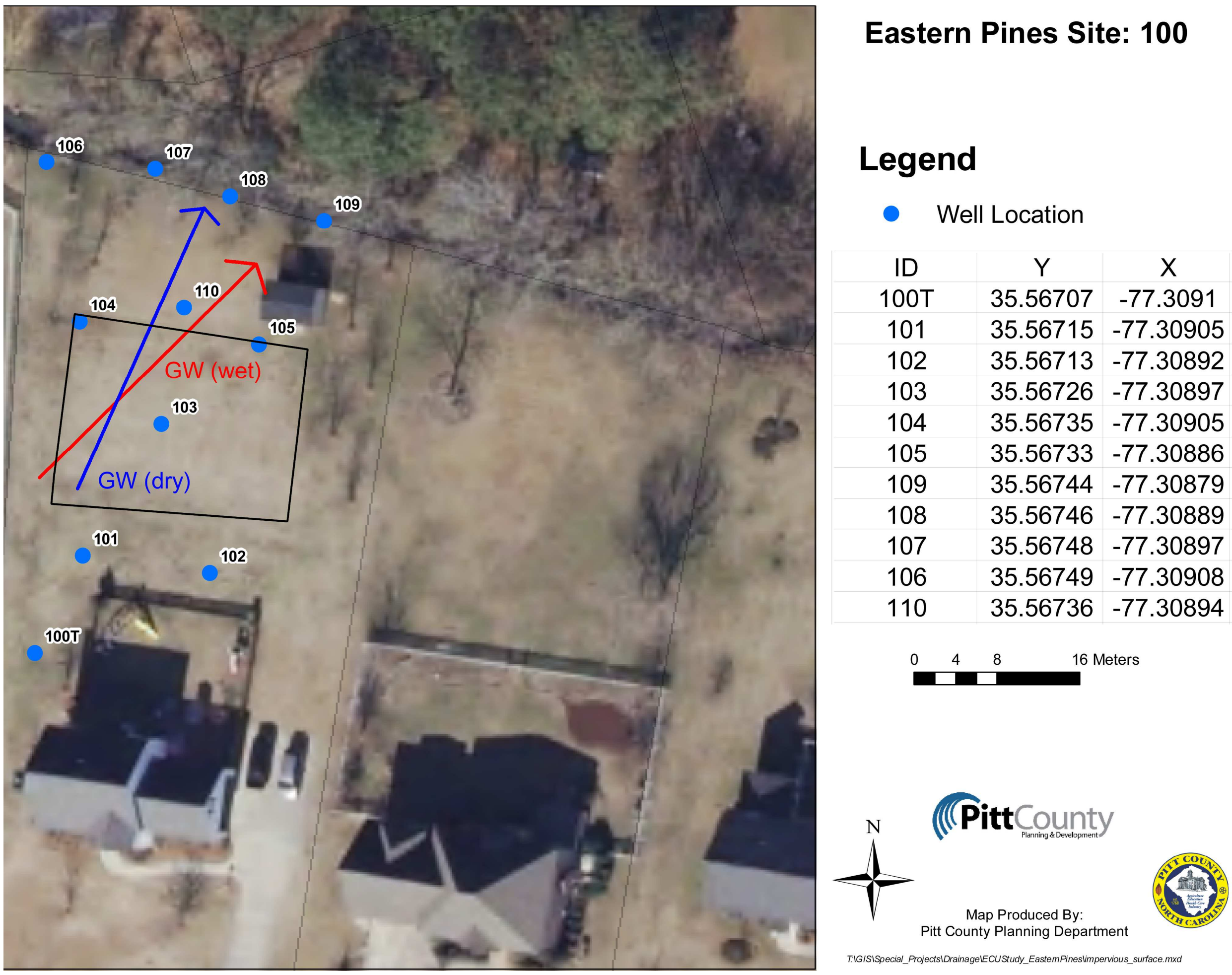

Cherry Oaks Watershed Delineation Map
Legend
Watershed boundary
Surface Water Sampling Site
Watershed outlet
N
800 m

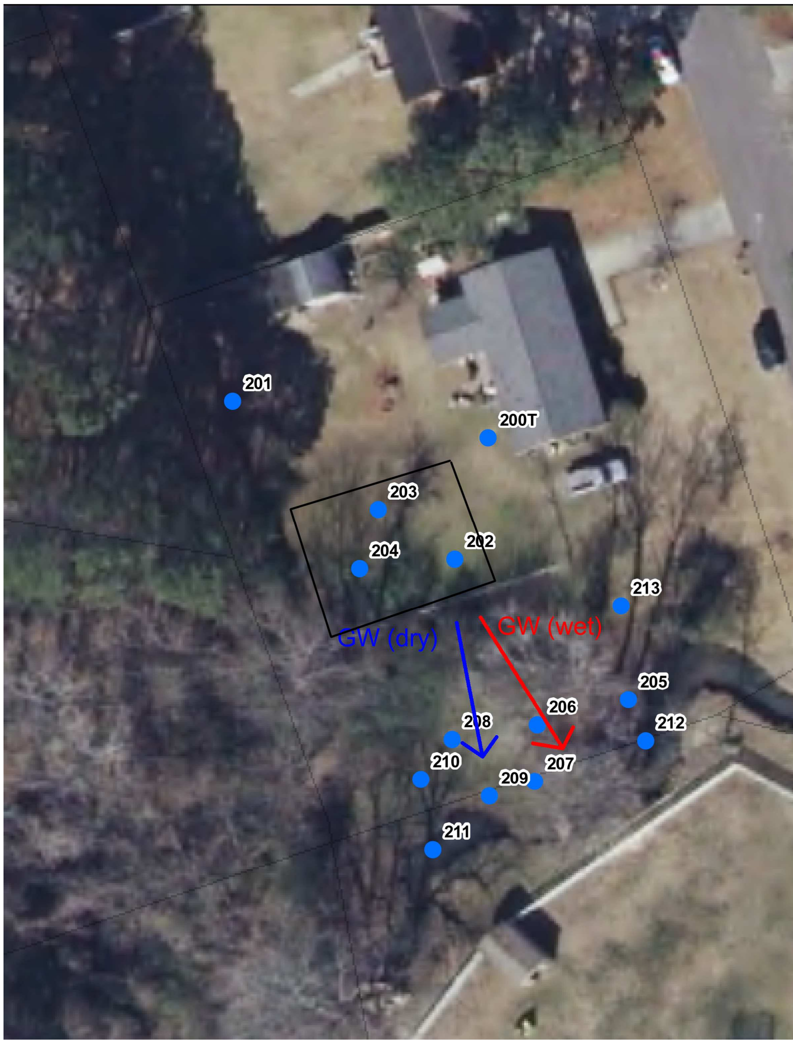
APPENDIX B: GROUNDWATER SITE MAPS AND FLOW DIRECTION

Groundwater monitoring sites and their flow directions during wet (red arrow) and dry (blue arrow) conditions of the year. The black box denotes the drainfield area. Figures B1-B5 are OWTS residential sites. Sites B1-B3 are intensive sites. Sites B4-B9 are non-intensive sites. Sites A5-A10 are CSS residential sites. Site B9 does not include flow direction because there were not enough piezometers to conduct a 3-point solution. However, it is estimated that groundwater flows from 1002 to 1001. All maps are courtesy of Pitt County Planning Department located in Greenville, NC.



ID	Y	X
100T	35.56707	-77.3091
101	35.56715	-77.30905
102	35.56713	-77.30892
103	35.56726	-77.30897
104	35.56735	-77.30905
105	35.56733	-77.30886
109	35.56744	-77.30879
108	35.56746	-77.30889
107	35.56748	-77.30897
106	35.56749	-77.30908
110	35.56736	-77.30894

Figure B1. Site 100 residential map showing piezometer network and flow direction.



Eastern Pines Site: 200

Legend

● Well Location

ID	Y	X
211	35.56743	-77.30951
210	35.56749	-77.30952
208	35.56752	-77.30949
209	35.56747	-77.30945
207	35.56748	-77.30941
206	35.56753	-77.3094
205	35.56755	-77.30931
212	35.56751	-77.3093
213	35.56762	-77.30932
204	35.56766	-77.30958
202	35.56766	-77.30948
203	35.5677	-77.30956
201	35.56779	-77.3097
200T	35.56776	-77.30945

0 4 8 16 Meters

N

PittCounty Planning & Development

Map Produced By:
Pitt County Planning Department

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Figure B2. Site 200 residential map showing piezometer network and flow direction.

Eastern Pines Site: 300

Legend

● Well Location

ID	Y	X
300BG	35.56948	-77.30611
301	35.5695	-77.30621
302	35.56954	-77.30626
303	35.56945	-77.30631
304	35.56938	-77.30631
305	35.56942	-77.30642
306	35.56931	-77.30637
307	35.56932	-77.30643
307	35.56935	-77.30649
307	35.56935	-77.30649
309	35.56937	-77.30654
310	35.56949	-77.30651
311	35.56959	-77.30644

0 4 8 16 Meters

N

Pitt County

Map Produced By:
Pitt County Planning Department

Figure B3. Site 300 residential map showing piezometer network and flow direction.

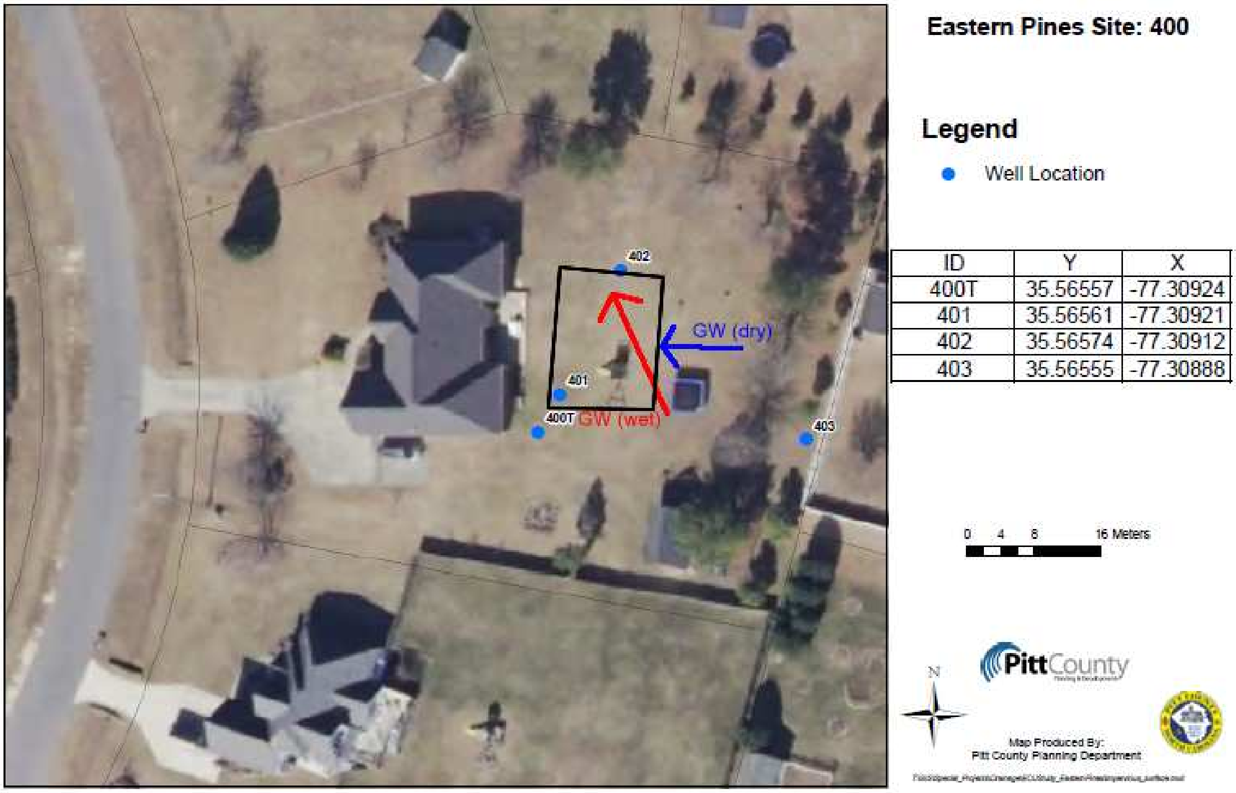

Figure B4. Site 400 residential map showing piezometer network and flow direction.

Figure B5. Site 500 residential map showing piezometer network and flow direction.

ID	Y	X
601	35.55965	-77.36782
602	35.55964	-77.36756
603	35.55949	-77.3676
ID	Y	X
703	35.55997	-77.36748
702	35.56012	-77.36745
701	35.56017	-77.36766

0 4 8 16 Meters

PittCounty Planning & Development

Aerial Image Source - January 2009

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Figure B6. Site 600 and site 700 residential map showing piezometer network and flow direction.

0 3 6 12 Meters

ID	Y	X
801	35.56515	-77.36737
803	35.56519	-77.36751
802	35.56507	-77.36752

PittCounty Planning & Development

Aerial Image Source - January 2009

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Figure B7. Site 800 residential map showing piezometer network and flow direction.

Fire Tower Well Locations: FT900

ID	Y	X
901	35.57195	-77.37302
902	35.57192	-77.37281
903	35.57158	-77.37279

Aerial Image Source - January 2009

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Figure B8. Site 900 residential map showing piezometer network and flow direction.

ID	Y	X
1002	35.56273	-77.36629
1001	35.56262	-77.36617

0 4 8 16 Meters

Aerial Image Source - January 2009

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Figure B9. Site 1000 residential map showing piezometer network. Flow direction was not determined from lack of installed piezometers, it is inferred to travel from 1002 to 1001.

APPENDIX C: GROUNDWATER SOILS DATA

Particle size distribution is shown for sites 100 through 500 in the table below. Particle size distribution was not conducted on CSS soil samples. The tables following particle size distribution show the soil texture, matrix and mottles (if present) color, cation exchange capacity (CEC), pH, and ion species and their concentrations (P, K, Ca, Mg, Mn, Zn, Cu, S, Na) for each site.

Sample ID	Sand %	Silt %	Clay %	USDA Class.
103	68.9	6.4	24.6	sandy clay loam
203	65.3	10.0	24.7	sandy clay loam
301	49.4	15.3	35.3	sandy clay
400	66.2	7.9	25.9	sandy clay loam
502	77.6	5.2	17.3	sandy loam

Site 100

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
101A	0-30IN	SANDY LOAM	10YR4/2			5.6	6	293	33	67	9	61	253	125	26	0.1
101B	30-40	SANDY LOAM	2.5Y7/6			6.5	5.5	9	153	43	21	10	18	25	60	0.4
101C	40-48	SANDY LOAM	10YR7/6			4.3	4.7	27	62	23	9	12	23	15	169	0.2
101D	48-57	SILT LOAM	2.5Y6/1			5.1	4.6	44	50	27	7	19	30	20	150	0.1
101E	57-61	SILTY CLAY	2.5Y7/2			3.9	4.5	9	45	19	9	7	15	15	113	0.1
101F	61-94	SILTY CLAY	2.5Y7/6 2.5Y7/2			3.7	4.6	3	41	17	9	6	17	42	114	0.1
102A	0-21	SANDY LOAM	10YR4/2			5	6	201	25	65	14	46	168	135	29	0.1
102B	21-27	SANDY LOAM	10YR6/1			3.1	5.9	88	29	59	15	21	93	80	29	0.1
102C	27-34	SANDY CLAY LOAM	2.5Y7/6			4.7	5.1	6	70	28	14	9	20	15	95	0.2
102D	34-66	SANDY CLAY	2.5Y7/6 2.5Y7/1			5.7	4.8	3	69	26	14	8	12	10	131	0.1
103A	0-15	SANDY LOAM	10YR3/2			4.9	5.6	221	49	57	14	57	170	109	56	0.1
103B	15-17	SANDY CLAY LOAM	10YR5/6	10YR6/1		6.6	5.4	92	79	37	16	22	185	441	83	0.4
103C	17-25	SILT LOAM	10YR3/1			7.6	5.3	62	35	48	18	36	202	73	101	0.2
103D	25-27	SANDY LOAM	10YR6/1			5	5.2	8	74	31	17	12	43	52	60	0.3
103E	27-47	SANDY LOAM (MORE ORGANIC MATTER)	7.5YR2.5/1													
103F	47-53	SANDY LOAM (HIT THE WATER TABLE)	10YR4/3													
103G	53-66+	SANDY CLAY	10YR5/8													
104A	0-8	SANDY LOAM (MORE ORGANIC)	10YR3/2			5.4	5.9	90	17	70	7	44	129	73	29	0.2
104B	8-28IN	SANDY LOAM (WHITISH COLOR)	10YR6/2			2.2	5.6	41	11	51	8	19	45	36	31	0.1
104C	28-34	SANDY LOAM	10YR6/1			2.2	5.2	52	22	41	11	12	62	25	51	0.1
104D	34-45	SANDY CLAY LOAM (DARK)	2.5Y2.5/1			11	5.6	134	112	57	13	70	630	28	112	0.4
104E	WATER TABLE	?	?													
105A	0-8	SANDY LOAM	10yr3/1			7.9	6.4	168	19	80	11	66	203	153	43	0.1
105B	8-24in	SANDY LOAM	10yr4/2			4.1	5.9	177	14	70	9	44	167	75	32	0.1
105C	24-39	SANDY LOAM	10yr4/1			2.8	5.6	103	11	66	8	22	74	36	77	0.1
105D	39-	SANDY LOAM	10yr7/1			2.1	6.1	56	19	68	11	13	32	37	52	0.1
106A	0-23	SANDY LOAM	10YR3/2			5.1	5.9	177	50	60	13	35	146	64	43	0.1
106B	23-42	SANDY	2.5Y6/4			3.3	6	110	32	60	15	18	67	53	30	0.1
106C	42-61	SILTY CLAY	2.5Y2.5/1			8.5	5.4	91	88	53	15	55	93	14	101	0.2

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
107A	0-24	SANDY LOAM	10YR3/2			3.4	5.7	65	29	49	13	30	80	30	20	0.1
107B	21-47	SILTY CLAY LOAM	2.5Y2.5/1			4.9	4.9	44	22	22	6	24	103	5	50	0.1
107C	47-66	SANDY CLAY LOAM	2.5Y7/1			4.8	4.6	82	20	25	9	23	43	6	40	0.1
108A	0-6	SANDY LOAM	10YR5/1			8.8	5.8	188	70	53	23	48	203	53	26	0.1
108B	6-18IN	SANDY LOAM	10YR6/2			3.8	6.1	106	33	56	18	17	59	47	23	0.1
108C	18-32	SILTY CLAY LOAM	2.5Y2.5/1			7.2	5	60	157	25	15	27	545	4	114	0.5
108D	32-55	SILTY CLAY LOAM	2.5Y2.5/1			8	5.3	71	91	46	7	69	974	3	93	0.3
109A	0-12	SANDY LOAM	10YR4/3			8.3	6	263	39	70	14	51	289	115	39	0.1
109B	12-24IN	CLAY LOAM	10YR4/2			9.9	5.6	146	83	56	20	34	101	68	62	0.1
109C	24-67	SILTY CLAY	2.5Y2.5/1	2.5Y7/5		9.5	5.3	113	77	54	19	43	114	24	194	0.2

Site 200

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
201A	0-12	LOAMY SANDY	2.5Y4/3			3	5	89	21	37	7	16	146	11	25	0.1
201B	12-24	LOAMY SANDY	2.5Y6/4			1.9	5.9	10	11	71	7	6	15	6	20	0.1
201C	24-40	SANDY CLAY	10YR5/6			4.5	4.8	2	16	51	4	3	10	5	146	0.1
201D	40-60	SANDY CLAY	10YR5/4			3.8	5	3	27	23	10	2	7	10	106	0.3
201E	60-+	SANDY LOAM	10YR6/6			2.7	5.1	2	26	22	9	3	7	15	44	0.2
202A	0-17	SANDY LOAM	10YR4/2			5.3	5.7	50	17	61	7	15	44	29	30	0.4
202B	17-21	LOAM; SILTY CLAY LOAM	10YR2/1			4.2	6.1	34	30	43	5	19	22	5	22	0.9
202C	21-36	SANDY LOAM	2.5Y5/2			2.1	6.2	30	33	37	8	10	17	20	28	0.4
202D	36-+	SANDY CLAY	10YR5/6 WET	10YR6/2		3.8	5.1	7	56	29	15	12	12	18	87	0.7
202A						3.9	5.5	106	19	48	6	12	145	22	29	0.2
202B						3.2	5.3	160	18	41	6	7	167	36	61	0.3
202C						2.1	5.6	143	9	42	10	6	146	56	30	0.2
202D						4	4.6	94	11	37	7	13	103	11	174	0.2
203A	0-6	SANDY LOAM	10YR4/2			7.3	5.7	25	17	64	12	35	148	72	23	0.4
203B	6-14	SANDY LOAM	10YR4/3			3.5	6	25	18	55	11	27	51	21	24	0.7
203C	14-21	SANDY LOAM	2.5Y4/1			2.5	6.3	32	20	47	6	16	25	14	20	0.6
203D	21-26	SANDY LOAM	2.5Y5/3			1.6	6.3	65	22	58	10	13	14	21	21	0.4
203E	26-50	SANDY CLAY	2.5Y5/2			2.9	6	5	69	54	14	22	16	18	13	0.7
203F	50+	SANDY LOAM	GLEY1-62			2.7	5.8	46	39	47	25	20	13	14	21	0.5
204A	0-12	SANDY LOAM	2.5Y3/2			5.4	5.9	76	21	62	10	26	109	29	21	0.2
204B	12-24	SANDY LOAM	2.5Y3/1			3.6	5.9	59	37	50	9	17	45	22	40	0.4
204C	24-38	SANDY CLAY LOAM	10YR5/4			3.5	5.5	3	69	28	10	15	28	16	53	0.8
204D	38-+	SANDY CLAY LOAM	2.5Y5/1			4	5.5	5	54	28	13	14	25	22	47	0.8

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
205A	0-18	SANDY LOAM	10YR4/3													
205B	18-25	SANDY LOAM	10YR3/2													
205C	25-33	SANDY LOAM	10YR4/2													
205D	33-48+	SANDY CLAY LOAM	10YR3/1													
206A	0-18	SANDY LOAM	10YR3/2			6.8	5.6	86	34	67	10	39	181	43	36	0.1
206B	18-26	SANDY LOAM	10YR4/2			4	5.3	157	18	50	7	24	192	59	46	0.2
206C	26-35	CLAY LOAM	10YR2/1			7.5	4.8	112	42	46	8	36	168	37	111	0.3
206D	35-48+	CLAY	10YR2/1			7.9	4.6	98	41	42	10	37	140	30	141	0.3
207A	0-18	SANDY LOAM	10YR3/2													
207B	18-24	SANDY CLAY	10YR2/1													
207C	24-36+	SANDY CLAY	10YR3/1													
208A	0-13	LOAM	10YR3/2			8.2	5.3	170	40	64	8	49	273	45	99	0.4
208B	13-23	CLAY LOAM	10YR3/1			6.9	5.4	151	35	56	8	33	214	78	67	0.4
			10YR3/4 ROOT CHEMLS										266	18	604	0.5
208C	23-36+	CLAY	7.5YR2.5/1			8.6	4.5	96	44	46	8	59				
209A	0-18	SANDY LOAM	10YR4/2			7.1	5.5	106	32	66	11	34	207	54	105	0.6
209B	18-27	SANDY LOAM OXIDIZED ROOT ZONE (7.5YR3/4)	10YR3/2			6.3	5.8	140	12	73	8	25	226	33	51	0.1
209C	27-37	LOAMY SAND/SANDY LOAM	2.5Y4/2			3.2	4.7	111	8	53	8	16	168	45	221	0.1
209D	37-+	CLAY	2.5Y2.5/1			6.2	4.9	121	26	41	10	16	132	14	138	0.1
210A	0-12	SANDY LOAM	10YR4/2													
210B	12-24	CLAY (OXIDIZED ROOT ZONE 10YR3/6)	10YR3/2													
210C	24-36+	CLAY	10YR2/1													
211A	0-12	SANDY LOAM	10YR5/3			4.7	5.6	118	11	65	8	30	192	36	35	0
211B	12-24	SANDY LOAM	10YR4/2			6.3	5.7	138	10	70	6	34	207	47	39	0.1
211C	24-35	CLAY	10YR3/2			6.6	5.2	205	20	55	8	34	123	40	88	0.2
211D	35-44+	CLAY	10YR3/1			8.7	4.9	87	36	45	11	31	724	671	253	0.2
212A	0-12	SANDY LOAM	10YR4/3			3.9	4.5	114	40	21	7	21	168	123	78	0.1
212B	12-18	SANDY LOAM	10YR3/3			3.9	5	146	25	30	4	12	158	75	57	0.2
212C	18-35	SANDY LOAM	10YR4/3			2.5	5.6	143	13	39	7	8	106	100	38	0.2
212D	35-48+	SANDY CLAY LOAM	10YR3/2			4.1	5	95	13	50	10	15	85	31	101	0.3
213A	0-6	SANDY LOAM	10YR3/2													
213B	6-18	LOAMY SAND	10YR5/2													
213C	18-24	SANDY LOAM	10YR3/2													
213D	24-36	SANDY LOAM	10YR3/2													
213E	36-+	CLAY LOAMY	10YR2/2													

Site 300

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
300A	0-14	SANDY LOAM	10YR4/3			5.4	5.4	248	17	56	8	14	83	30	25	0
300B	14-25	SANDY LOAM	10YR3/2			6.9	5.3	75	16	58	8	10	35	25	31	0.1
300C	25-38	SANDY CLAY LOAM/SANDY CLAY	10YR5/6	10YR6/2		4	4.5	1	22	32	11	3	15	15	151	0.1
300D	38-68	SANDY CLAY LOAM	10YR6/2	10YR5/6		4.6	4.1	2	33	35	14	8	30	20	290	0.1
300E	68+	SANDY LOAM	GLEYL 4/N			9.9	2.6	2	24	9	9	36	259	19	3279	0
301A	0-9	SANDY LOAM	10YR4/3			5.2	6	85	12	73	10	9	20	16	37	0.1
301B	9-15	CLAY	10YR5/4			8.5	5.7	1	24	69	22	2	5	5	118	0.2
301C	15-24	CLAY	10YR5/4	7.5YR5/8	7.5YR6/1 19'	4.9	4.9	1	18	35	14	2	5	6	177	0.2
301D	24-30	CLAY	10YR5/32	7.5YR5/8	10YR6/1	4.2	5	1	18	40	12	2	6	7	187	0.3
301E	30-42	SANDY CLAY	10YR5/2			3.7	5.4	1	24	51	14	2	7	9	79	0.5
301F	42-52	SANDY CLAY	10YR5/6	10YR 6/2												
301G	52-67	SANDY CLAY	2.5Y5/6													
301H	67-+	SANDY CLAY	6/10Y GLEY 1													
302A	0-4	SANDY LOAM	2.5Y4/2			5	5.4	230	22	61	9	14	85	25	29	0.1
302B	4-22	CLAY	2.5Y4/3			7.3	5	6	19	55	19	3	15	5	63	0.1
302C	22-34	CLAY	2.5Y4/3	2.5Y5/2	10YR5/8	5.6	5	5	19	43	13	9	8	5	133	0.2
302D	34-46	SANDY CLAY	2.5Y5/4	2.5Y5/2		4.6	5.2	17	35	45	13	8	10	5	71	0.4
302E	46-64	SANDY CLAY LOAM	2.5Y6/1			3.4	5.2	10	29	37	12	17	14	10	91	0.5
303A	0-9	SANDY LOAM	2.5Y4/3			4.3	5.6	70	9	62	10	10	28	20	31	0
303B	9-28	SANDY CLAY	10YR5/4			8.4	5.1	1	16	51	18	2	10	15	83	0.1
303C	28-38	SANDY CLAY	10YR5/4	10YS5/2	10YR5/6	6.1	4.8	0	15	42	12	2	8	10	140	0.1
303D	38-50	SANDY CLAY	10YR4/3	10YR6/1	10YR5/6	4.5	5.2	2	17	26	8	2	13	15	127	0.4
303E	50-+	LOAMY SAND	10YR6/8	10YR 6/1		0.8	5.7	0	7	53	21	2	11	8	40	0.2
304A	0-15	CLAY	2.5Y5/3CONCENTRATIONS			5.6	4.9	5	16	43	14	3	6	4	189	0.2
304B	15-30	CLAY	2.5Y4/3	2.5YR6/2	2.5YR5/2	5.2	5.3	2	17	41	13	6	5	4	136	0.4
304C	30-40	CLAY	2.5Y5/2	2.5Y4/3	2.5Y6/2	4.5	5.2	33	35	38	12	15	6	4	95	0.4
304D	40-50	LOAMY SAND	2.5Y 6/1			2.6	5.8	19	41	41	15	9	7	10	47	0.5
304E	50-60+	LOAMY SAND	2.4Y 6/8			1.5	6	3	27	43	17	4	6	8	30	0.4
305A	0-10	SANDY LOAM	10YR4/3													
305B	10-20	SANDY CLAY LOAM	10YR5/4													
305C	20-28	SANDY CLAY	10YR5/4	10YR5/2	5YR5/8											
305D	28-46	SANDY CLAY LOAM	2.5Y6/8	2.5Y5/2	2.5Y5/8											
305E	46-51	SANDY LOAM	2.5Y6/8 5/2 2.5/8 MOTTLES	2.5Y5/2	2.5Y5/8											
305F	51-60+	LOAMY SANDY	2.5Y6/8													
306A	0-26	CLAY	2.5Y5/4													
306B	26-38	SANDY CLAY LOAM	2.5Y5/6 MOTTLES +CONCENTRATED													
306C	38-50+	SANDY LOAM	2.5Y6/6 MOTLES+CONCENTRATED													
306D																
306E																

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
307A	0-15	SANDY LOAM	10YR4/3													
307B	15-27	SANDY CLAY	10YR5/4													
307C	27-38	SANDY CLAY LOAM														
307D	38-50+	LOAMY SAND	10YR6/8													
308A	0-23	SANDY LOAM	10YR4/3													
308B	23-35	SANDY LOAM	10YR3/3													
308C	35-48	SANDY CLAY	10YR4/3	10YR5/8	10YR6/1											
308D	48-60+	SANDY CLAY	10YR6/1	10YR5/8		5.1	5	5	11	52	16	2	12	15	96	0.4
308E						4	4.8	7	29	37	10	7	24	28	143	0.2
309A	0-36	SANDY LOAM	10YR3/2													
309B	36-65	LOAMY	10YR2/1													
310A	0-9	SANDY LOAM	10YR4/2													
310B	9-30	SANDY LOAM	10YR3/3													
310C	30-36	LOAMY SANDY	10YR6/2													
310D	36-48+	SANDY LOAM	10YR6/8	10YR6/2												
311A	0-18	SANDY LOAM	10YR4/3													
311B	18-29	CLAY	10YR4/3	10YR4/2	10YR5/6	4.6	5.5	35	8	58	12	3	26	29	42	0.1
311C	29-36	SANDY CLAY LOAM	10YR4/3			2.6	4.9	7	14	30	12	4	18	18	86	0.1
311D	36-50	SANDY CLAY LOAM	10yr5/6			5.1	4.5	1	21	23	7	4	8	14	202	0.1
311E	50-60+	CLAY	10YR5/6?			6.9	4.6	5	45	37	15	76	26	46	93	0.1

Site 400

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
401A	0-16	SANDY LOAM	2.5Y5/4			3.9	5.4	316	36	53	10	40	171	185	28	0.1
401B	16-36	SANDY CLAY LOAM	10YR5/6	10YR5/8		5.2	5.1	6	90	29	10	4	22	32	256	0.4
401C	36-50+	SANDY CLAY	10YR6/2	10YR 5/6		4.2	5.8	15	87	41	16	6	16	15	43	1
402A	0-12	SANDY LOAM	2.5Y4/3			4.7	5.5	583	39	58	7	66	435	330	26	0.1
402B	12-23	SANDY LOAM	2.5Y5/4			3	5.9	404	19	64	6	64	273	290	19	0.1
402C	23-36	SANDY CLAY LOAM	10YR5/4	10YR6/2		4.3	6	44	61	59	13	16	18	80	35	0.7
402D	36-60	SANDY CLAY	10YR5/3	10YR6/2		3.8	5.5	8	60	38	14	6	13	20	95	0.4
402E	60-84	SANDY CLAY	10YR7/1	10YR6/8 M		4	5.3	1	76	20	11	3	10	10	120	0.6
402F	84-120	SANDY LOAM	2.5Y8/1			1.9	4.9	3	22	15	8	2	8	15	117	0.2

Site 500

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
501A	0-12	SANDY LOAM	10YR4/4			4.6	5.1	18	31	34	14	3	15	15	36	0.1
501B	12-18	SANDY CLAY LOAM	2.5YR7/3	10YR6/8		4.3	5	3	24	29	13	3	8	5	63	0.1
501C	18-26	SANDY CLAY LOAM	10YR6/1	10YR7/6		3.5	5	3	20	24	11	2	5	5	54	0.1
501D	26-36	LOAMY SANDY	10YR6/1			2	6.1	19	30	43	18	4	8	10	27	0.3
501E	36-END	SANDY LOAM	10YR6/1			1.4	5.9	22	23	48	17	6	7	8	22	0.2
WL--->4																
503A	0-36	SANDY LOAM	10YR3/2			4.8	5.3	59	25	47	11	14	45	40	29	0.1
503B	36-END	SANDY LOAM	2.5YR2/1			3.8	6.1	21	19	25	7	8	16	21	22	0.4
WL-->3																

Site 600

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
601A	0-7	SANDY CLAY LOAM	10YR5/4			12.7	7.4	31	18	94	5	48	30	27	32	0.1
601B	7-15	SANDY CLAY LOAM	10YR5/6			5.1	5.2	3	16	67	10	11	16	16	124	0
601C	15-28	SANDY CLAY	2.5Y5/3			6.5	5	52	28	36	8	34	206	6	49	0
601D	28-54	LOAMY	10YR2/1			3.6	4.7	1	12	23	8	9	17	15	122	0
601E	54-65	SANDY LOAM	10YR4/1			2.8	5	18	5	40	8	3	11	11	18	0
603A	0-11	SANDY LOAM	10YR4/2			2.5	5.1	25	12	35	9	6	33	5	17	0.1
603B	11-32	SANDY LOAM	10YR4/2			2.1	5.2	12	11	51	9	8	18	5	16	0.1
603C	32-76	LOAMY SANDY	10YR6/2			2.2	5.3	14	13	60	10	14	18	5	24	0.1
603D	76-+	LOAMY SANDY	10YR7/1													

Site 700

Well #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
701A	0-53	TOP SOIL (SL)	10YR6/1			4.5	5.2	44	27	40	12	26	48	5	50	0
701B	53-65	LOAMY SANDY	102.5Y2.5/1			3.2	5	28	12	25	16	3	8	5	21	0
701C	65-83	CLAY	10YR3/2			5.7	4.4	8	8	32	10	3	8	5	63	0.1
701D	83-107	LOAMY SANDY	10YR7/1	10YR2/1		1.9	4.3	10	7	37	11	4	15	5	116	0.1
701E	107-END	SANDY CLAY LOAMY	10YR4/2			4.4	4	10	13	49	10	7	20	11	451	0.1
W.T--> 8.4FT																
702A	0-27	MOTTLE, TOP SOIL (SL)				4.4	4.6	77	28	13	5	5	30	10	41	0
702B	27-54	SANDY LOAM	10YR2/1			3.4	4.6	15	15	20	6	2	15	15	24	0.1
702C	54-68	SANDY	10YR7/1			2.6	4.2	7	7	44	10	5	23	10	173	0.1
702D	68-END	SANDY	10YR6/1			2.5	6.3	9	14	79	15	12	17	20	33	0.1
W.T---> 6.1FT																

Site 800

WELL #	DEPTH	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn		Cu	S	Na
801A	0-6	TOP SOIL (SLIGHTLY SL)	10YR6/9													
801B	6-24	SANDY LOAM	10YR6/9			1.8	4.9	1	11	18	8	3	17	27	52	0.1
801C	24-END	LOAMY SAND	10YR7/1			1.2	4.6	1	5	23	10	2	12	11	45	0
W.L--->2FT																

Site 900

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
901A	0-20	TOP SOIL (L)	2.5YR6/1			6.1	4.6	62	15	23	7	10	33	27	30	0.1
901B	20-31	SANDY CLAY LOAM	7.5YR5/8	7.5YR5/1		4	4.5	8	8	31	5	4	13	10	85	0.1
901C	31-51	SANDY CLAY LOAM	7.5YR5/8	10YR4/1		4.7	4.5	14	6	44	5	10	26	11	74	0.1
901D	51-68	SANDY CLAY LOAM	2.5YR8/1	7.5YR5/8	10YR5/2	2.9	4.4	0	6	10	5	1	8	10	39	0.1
901E	68-88	SANDY CLAY LOAM	2.5YR8/1	10YR5/1		4.8	4.4	4	11	36	5	4	14	15	63	0.1
901F	88-108	SANDY CLAY LOAM	2.5YR8/1	7.5YR5/8	10YR6/6	4.6	4.5	1	26	11	7	4	16	19	48	0.1
901G	108-END	CLAY	10YR4/2			7.5	4.2	3	63	17	11	18	20	64	82	0.2
WL-->9FT																
902A	0-60	LOAMY SANDY	10YR7/1	10YR6/6	10YR4/2	5	4.5	3	22	13	7	7	18	20	46	0.2
902B	60-BOTTOM	CLAY	10YR7/1	10YR4/2		5.1	4.5	1	43	25	14	10	15	118	27	0.2
WL-->5																

Site 1000

WELL #	DEPTH (IN)	SOIL TEXTURE	MATRIX	MOT1	MOT2	CEC	pH	P	K	Ca	Mg	Mn	Zn	Cu	S	Na
1001 A	0-8	SANDY LOAM	10YR 6/1			3.5	4.7	11	14	31	7	38	67	22	68	0
	8-16"	SANDY LOAM	2.5Y 4/2			3.7	4.1	13	11	16	2	1	14	6	37	0
	16-48"	SANDY LOAM	2.5Y3/2			4.1	4.1	95	10	6	2	1	11	17	51	0
	48-84	SANDY CLAY LOAM	10YR4/1	7.5YR 3/2		4.5	4.3	0	12	8	2	6	24	9	105	0
	84-96	SANDY LOAM	7.5YR 3/2			2	4.5	3	5	19	5	3	13	11	16	0
	96-108	SANDY LOAM	2.5Y 5/2			3.4	4.1	10	18	37	10	15	17	31	228	0
	WT96"															

APPENDIX D: FLOW METER VS. FLOATING OBJECT METHOD DATASET

Flow meter vs. floating object method at each of the 6 primary watersheds and FT-O (used as a check for FT-1 and FT-2), the calculated correction factors were applied to baseflow and storm flow velocities to correct for floating object methods overestimation of flow. FT-2 streambed was not uniform similar to the other 5 watersheds. Therefore, there were 4 tests conducted within the same stream. Each of these tests occurred approximately within 3 m of each other. W= width, D= depth, Q= discharge, V= velocity, Diff.= difference, AVG= average, STDV= standard deviation, CF= correction factor.

FT-O								
Trial #	Stream W (ft)	Stream D (ft)	Float Method (ft/s)	Flow Meter (ft/s)	Q Float (cfs)	Q Flow (cfs)	V Diff. %	Q Diff. %
1	6.7	0.65	0.33	0.24	1.43	1.04	27.4%	27.4%
2	6.7	0.65	0.29	0.23	1.25	1.00	19.8%	19.8%
3	6.7	0.65	0.27	0.22	1.19	0.96	19.7%	19.7%
4	6.7	0.65	0.32	0.25	1.40	1.09	22.3%	22.3%
5	6.7	0.65	0.31	0.25	1.33	1.09	18.2%	18.2%
6	6.7	0.65	0.26	0.24	1.15	1.04	9.2%	9.2%
7	6.7	0.65	0.25	0.23	1.08	1.00	7.4%	7.4%
8	6.7	0.65	0.25	0.24	1.10	1.04	5.3%	5.3%
9	6.7	0.65	0.23	0.22	0.99	0.96	3.7%	3.7%
10	6.7	0.65	0.20	0.22	0.86	0.96	11.2%	11.2%
AVG:			0.27	0.23	1.18	1.02	14.4%	14.4%
STDV:			0.04	0.01	0.18	0.05		
CF:			0.86	AVG Q/ w/ CF:	1.02			

FT-1								
Trial #	Stream W (ft)	Stream D (ft)	Float Method (ft/s)	Flow Meter (ft/s)	Q Float (cfs)	Q Flow (cfs)	V Diff. %	Q Diff. %
1	2.5	0.27	1.06	1.03	0.71	0.69	2.57%	2.57%
2	2.5	0.27	0.59	1.01	0.39	0.68	72.0%	72.0%
3	2.5	0.27	1.19	1.01	0.80	0.68	15.4%	15.4%
4	2.5	0.27	1.06	0.99	0.71	0.66	6.35%	6.35%
5	2.5	0.27	1.03	1.02	0.69	0.68	0.76%	0.76%
6	2.5	0.27	1.32	0.93	0.89	0.62	29.6%	29.6%
7	2.5	0.27	0.88	0.97	0.59	0.65	10.1%	10.1%
8	2.5	0.27	0.93	1.00	0.62	0.67	8.11%	8.11%
9	2.5	0.27	1.28	1.09	0.85	0.73	14.6%	14.6%
10	2.5	0.27	1.48	1.01	0.99	0.68	31.8%	31.8%
AVG:			1.08	1.01	0.72	0.67	6.90%	6.90%
STDV:			0.25	0.04	0.17	0.03		
CF:			0.93	AVG Q/ w/ CF:	0.67			

	FT-2								
	Trial #	Stream W (ft)	Stream D (ft)	Float Method (ft/s)	Flow Meter (ft/s)	Q Float (cfs)	Q Flow (cfs)	V Diff. %	Q Diff. %
Test 1	1	2.9	0.46	0.97	0.34	1.29	0.45	65.1%	65.1%
	2	2.9	0.46	0.95	0.33	1.26	0.44	65.2%	65.2%
	3	2.9	0.46	1.03	0.36	1.36	0.48	65.0%	65.0%
	4	2.9	0.46	1.06	0.36	1.40	0.48	65.9%	65.9%
	5	2.9	0.46	0.88	0.37	1.17	0.49	58.0%	58.0%
	6	2.9	0.46	0.84	0.37	1.11	0.49	56.0%	56.0%
	7	2.9	0.46	0.88	0.38	1.17	0.50	56.9%	56.9%
	8	2.9	0.46	0.86	0.36	1.14	0.48	58.2%	58.2%
	9	2.9	0.46	0.90	0.37	1.20	0.49	59.0%	59.0%
	10	2.9	0.46	0.90	0.39	1.20	0.52	56.8%	56.8%
Test 2	1	2	0.38	0.97	0.52	0.73	0.39	46.6%	46.6%
	2	2	0.38	0.86	0.50	0.65	0.38	41.9%	41.9%
	3	2	0.38	0.98	0.52	0.73	0.39	46.9%	46.9%
	4	2	0.38	1.00	0.49	0.75	0.37	51.0%	51.0%
	5	2	0.38	1.00	0.48	0.75	0.36	52.0%	52.0%
	6	2	0.38	1.19	0.50	0.90	0.38	58.1%	58.1%
	7	2	0.38	1.32	0.55	0.99	0.41	58.4%	58.4%
	8	2	0.38	1.32	0.51	0.99	0.38	61.4%	61.4%
	9	2	0.38	1.03	0.47	0.77	0.35	54.3%	54.3%
	10	2	0.38	1.19	0.47	0.90	0.35	60.6%	60.6%
Test 3	1	1	0.35	1.28	1.39	0.45	0.49	8%	8%
	2	1	0.35	1.19	1.47	0.42	0.51	23%	23%
	3	1	0.35	1.61	1.68	0.56	0.59	4%	4%
	4	1	0.35	1.54	1.53	0.54	0.54	1%	1%
	5	1	0.35	1.28		0.45			
	6	1	0.35	1.76		0.62			
	7	1	0.35	1.54		0.54			
	8	1	0.35	1.16		0.40			
Test 4	1	1	0.35	1.61	1.39	0.56	0.49	14%	14%
	2	1	0.35	1.40	1.47	0.49	0.51	5%	5%
	3	1	0.35	1.54	1.68	0.54	0.59	9%	9%
	4	1	0.35	1.54	1.53	0.54	0.54	1%	1%
	5	1	0.35	1.48		0.52			
	AVG1:			0.93	0.36	1.23	0.48	61%	61%
	STDV1:			0.07	0.02	0.10	0.02	4%	4%
	CF1:			0.39		AVG Q w CF	0.48		
	AVG2:			1.09	0.50	0.82	0.38	53%	53%
	STDV2:			0.16	0.03	0.12	0.02	7%	7%
	CF2:			0.46		AVG Q w CF	0.38		
	AVG3:			1.42	1.52	0.50	0.53	9%	9%
	STDV3:			0.22	0.12	0.08	0.04	10%	10%
	CF3:			1.07		AVG Q w CF	0.53		
	AVG4:			1.51	1.52	0.53	0.53	7%	7%
	STDV4:			0.08	0.12	0.03	0.04	5%	5%
	CF4:			1.00		AVG Q w CF	0.53		
	AVG CF:			0.73					

MHB								
Trial #	Stream W (ft)	Stream D (ft)	Float Method (ft/s)	Flow Meter (ft/s)	Q Float (cfs)	Q Flow (cfs)	V Diff. %	Q Diff. %
1	6	0.21	0.95	0.69	1.20	0.87	27.3%	27.3%
2	6	0.21	0.62	0.71	0.78	0.90	15.1%	15.1%
3	6	0.21	0.63	0.68	0.79	0.86	8.4%	8.4%
4	6	0.21	0.79	0.69	1.00	0.87	12.4%	12.4%
5	6	0.21	0.70	0.72	0.88	0.91	3.1%	3.1%
6	6	0.21	0.67	0.69	0.85	0.87	2.6%	2.6%
7	6	0.21	0.70	0.70	0.88	0.89	0.3%	0.3%
8	6	0.21	0.69	0.71	0.87	0.90	3.6%	3.6%
9	6	0.21	0.71	0.71	0.90	0.90	0.2%	0.2%
10	6	0.21	0.73	0.74	0.92	0.94	2.0%	2.0%
AVG:			0.72	0.70	0.91	0.89	7.5%	7.5%
STDV:			0.09	0.02	0.12	0.02		
CF:			0.98	AVG Q/ w/ CF:	0.89			

EP-O								
Trial #	Stream W (ft)	Stream D (ft)	Float Method (ft/s)	Flow Meter (ft/s)	Q Float (cfs)	Q Flow (cfs)	V Diff. %	Q Diff. %
1	3.12	0.27	1.06	1.01	0.90	0.86	4.46%	4.46%
2	3.12	0.27	1.09	1.02	0.92	0.87	6.27%	6.27%
3	3.12	0.27	1.00	1.04	0.85	0.88	4.00%	4.00%
4	3.12	0.27	1.19	1.06	1.01	0.90	11.2%	11.2%
5	3.12	0.27	1.00	1.04	0.85	0.88	4.00%	4.00%
6	3.12	0.27	1.03	1.04	0.87	0.88	1.19%	1.19%
7	3.12	0.27	0.90	1.08	0.77	0.92	19.7%	19.7%
8	3.12	0.27	0.90	1.00	0.77	0.85	10.8%	10.8%
9	3.12	0.27	1.03	1.01	0.87	0.86	1.73%	1.73%
10	3.12	0.27	1.09	1.01	0.92	0.86	7.19%	7.19%
AVG:			1.03	1.03	0.87	0.87	7.05%	7.05%
STDV:			0.09	0.03	0.07	0.02		
CF:			1.00	AVG Q/ w/ CF:	0.87			

MILL								
Trial #	Stream W (ft)	Stream D (ft)	Float Method (ft/s)	Flow Meter (ft/s)	Q Float (cfs)	Q Flow (cfs)	V Diff. %	Q Diff. %
1	1.7	0.42	1.54	1.15	1.11	0.83	25.4%	25.4%
2	1.7	0.42	1.37	1.32	0.98	0.95	3.7%	3.7%
3	1.7	0.42	1.09	1.29	0.78	0.93	18.5%	18.5%
4	1.7	0.42	1.16	1.15	0.83	0.83	0.5%	0.5%
5	1.7	0.42	1.32	1.21	0.95	0.87	8.4%	8.4%
6	1.7	0.42	1.28	1.31	0.92	0.94	2.7%	2.7%
7	1.7	0.42	1.42	1.24	1.02	0.89	12.9%	12.9%
8	1.7	0.42	1.42	1.24	1.02	0.89	12.9%	12.9%
9	1.7	0.42	1.23	1.23	0.89	0.88	0.3%	0.3%
10	1.7	0.42	1.54	1.29	1.11	0.93	16.3%	16.3%
AVG:			1.34	1.24	0.96	0.89	0.10	0.10
STDV:			0.15	0.06	0.11	0.04	0.08	0.08
CF:			0.93	AVG Q w/ CF:	0.89			

CHOK						
Trial #	Float Method (ft/s)	Flow Meter (ft/s)	Q Float (cfs)	Q Flow (cfs)	V Diff. %	Q Diff. %
1	2.59	2.37	1.24	1.14	8.6%	8.6%
2	3.18	2.42	1.53	1.16	23.9%	23.9%
3	3.04	2.5	1.46	1.20	17.9%	17.9%
4	3.33	2.43	1.60	1.17	27.1%	27.1%
5	3.50	2.48	1.68	1.19	29.1%	29.1%
6	2.92	2.42	1.40	1.16	17.0%	17.0%
7	3.50	2.45	1.68	1.18	30.0%	30.0%
8	3.18	2.28	1.53	1.09	28.3%	28.3%
9	3.18	2.42	1.53	1.16	23.9%	23.9%
10	3.33	2.49	1.60	1.20	25.3%	25.3%
AVG:	3.18	2.43	1.52	1.16	23.1%	23.1%
STDV:	0.28	0.06	0.13	0.03	0.067	0.067
CF:	0.76	AVG Q w/ CF:	1.16			

CHOK sampling and discharge data were collected using a culvert, therefore stream W and D were not recorded.

APPENDIX E: REPLICATES AND BLANKS DATA

Table E1. Surface and groundwater original and replicate TN and Cl^- (mg/L) concentrations. Blank sample TN and Cl^- concentrations are also shown.

Surface Water				
Sample ID	**Original TN**	**Replicate TN**	**Original Cl**	**Replicate Cl**
FT-O	0.42	0.36	11.47	11.36
	0.27	0.47	14.13	11.99
	0.64	0.62	20.91	21.04
	0.54	0.59	11.75	12.01
FT-1	0.42	0.30	11.26	10.83
	0.25	0.27	13.33	13.34
	0.76	0.81	8.26	8.46
	1.45	1.23	5.42	3.88
	0.73	0.62	12.82	13.45
FT-2	0.37	0.35	13.69	11.72
	0.84	0.83	9.36	9.22
MHB	0.99	0.87	15.57	15.58
	1.11	1.17	52.45	52.41
	0.89	0.86	11.50	11.71
Bell	1.00	0.98	12.59	13.68
	1.18	1.08	13.99	12.86
	0.95	0.94	9.69	9.94
EP-O	1.26	1.15	18.23	17.67
	1.28	3.68	16.94	20.19
	1.60	1.90	16.23	14.77
	1.71	1.42	14.82	14.93
	1.32	1.12	14.47	13.76
	1.34	1.19	13.05	13.30
EP-1	1.08	0.89	16.32	17.84
	2.10	2.19	21.88	24.17
	1.07	1.07	17.90	16.85
	2.59	2.47	15.43	13.31
Mill	0.96	1.11	16.92	14.85
	0.92	0.91	15.67	16.89
	1.05	1.04	14.52	13.56
	0.99	0.99	12.42	11.86
	1.22	1.17	2.52	2.81
CHOK	0.99	0.95	16.95	17.48
	1.35	1.33	14.48	14.47
	1.07	1.05	22.21	21.44
GUC-I	3.96	3.60	56.01	55.40
	5.08	5.17	53.20	53.36
	10.17	10.37	63.37	65.83
	27.35	28.81	50.85	50.37
	27.38	26.86	40.05	41.70
GUC-O	7.88	8.02	56.22	58.70
	7.55	6.87	56.11	53.15
	6.68	6.07	45.54	47.63
	4.24	4.47	50.06	51.33
Average:	3.11	3.14	22.74	22.75
STDEV	5.80	5.88	16.92	17.29

Ground Water				
Sample ID	**Original TDN**	**Replicate TDN**	**Original Cl**	**Replicate Cl**
802	0.19	0.18	7.29	7.79
803	0.35	0.31	7.27	7.87
800-stream	0.37	0.22	11.68	12.55
901	2.54	2.83	13.45	14.73
902	0.22	0.15	19.77	19.63
903	0.30	0.27	22.73	22.75
900-stream	0.14	0.21	11.51	10.54
1000	0.62	0.29	16.15	17.31
1001	0.69	0.35	17.22	11.35
101	4.57	4.69	11.24	10.88
110	5.25	5.24	12.88	12.40
201	6.97	6.77	18.16	19.66
204	13.26	13.04	23.71	23.94
302	9.14	8.58	28.82	31.78
402	34.54	32.03	25.59	24.22
101	6.38	5.79	16.74	16.08
110	15.49	15.57	31.99	35.63
201	7.74	7.71	23.67	21.37
204	12.69	12.60	32.36	33.39
302	1.52	1.60	33.60	30.17
Average	6.15	5.92	19.29	19.20
STDEV	8.29	7.84	8.23	8.59

Blank Samples		
	Blank TN Conc	Blank Cl^- Conc
	0.07	0.05
	0.12	0.72
	0.15	0.05
	0.06	0.55
	0.07	0.05
	0.07	0.05
	0.07	0.05
	0.11	0.05
	0.13	0.95
	0.07	0.05
Average	0.09	0.26
STDEV	0.03	0.35

APPENDIX F: GROUNDWATER PHYSICAL AND CHEMICAL WATER QUALITY SUMMARY

Sample Point	DTB (m)	Piezometer Elevation (m)	DTW (m)	Temp. (°C)	EC (µS/cm)	pH	DO (mg/L)	Total Hydraulic Head (m)	NH_4 (mg/L)	NO_3^- (mg/L)	DKN (mg/L)	Cl^- (mg/L)	TDN (mg/L)
101	2.49	13.7	1.12	19.9	137	5.89	0.85	12.6	0.21	3.72	0.39	12.8	4.15
102	2.33	13.7	1.05	20.1	137	5.89	2.65	12.6	0.23	4.37	0.41	12.5	4.86
103	5.21	43.4	2.39	20.8	235	5.93	2.40	12.5	5.65	1.01	6.92	24.4	7.69
104	1.78	12.9	0.59	20.4	238	6.03	3.66	12.3	1.70	0.49	2.13	27.8	2.25
105	2.23	13.0	0.60	20.5	338	6.18	3.88	12.4	2.86	0.46	4.73	15.2	4.98
106	2.42	12.3	0.73	19.6	94.2	5.84	2.47	11.6	1.30	0.16	1.68	12.9	1.70
107-s	1.52	12.5	1.09	18.7	155	5.89	2.05	11.4	9.27	0.06	9.08	13.4	10.8
107-d	2.49	12.5	1.08	18.9	109	5.86	1.73	11.4	2.94	0.25	3.83	10.8	3.87
108-s	1.23	12.4	0.98	20.6	244	5.77	2.43	11.5	3.97	1.96	7.26	52.7	9.22
108-m	1.98	12.5	1.25	18.6	241	5.76	3.20	11.3	10.7	0.65	7.75	22.9	9.56
108-d	2.37	12.5	1.22	18.4	198	5.90	2.01	11.3	9.82	0.03	8.14	20.2	10.3
109-s	1.64	12.4	1.06	17.6	345	5.81	2.08	11.9	6.02	0.09	7.15	14.6	7.25
109-d	2.25	12.4	0.87	18.0	269	6.01	3.42	11.6	7.53	0.24	7.30	15.9	9.23
110-s	1.58	12.9	0.55	22.6	501	5.71	2.94	12.4	22.5	0.01	20.59	50.2	23.5
110-d	2.46	12.9	0.52	20.3	381	6.21	3.07	12.4	8.47	0.34	7.72	22.2	8.96
100-pipe				18.1	206	6.33	7.58		0.15	3.58	0.53	14.6	4.11
100-tank				24.6	883	6.61			61.3	0.07	62.52	85.3	63.2
201	2.93	13.5	1.61	19.1	247	6.24	2.61	11.9	0.48	4.78	0.86	16.2	6.69
202	1.46	13.0	0.74	19.4	603	5.94	2.17	12.3	30.6	0.79	30.30	78.3	31.6
203	2.72	13.3	1.28	19.7	872	6.08	2.33	12.0	47.8	0.39	53.81	96.4	55.0
204	2.31	13.0	1.15	18.3	641	5.96	2.56	11.8	9.44	1.37	9.40	29.4	12.4
205	2.51	12.0	0.47	18.2	72.0	5.74	2.04	11.5	1.87	0.06	2.42	17.6	2.48
206	2.07	12.0	0.62	18.8	169	5.84	2.12	11.3	2.58	0.24	5.22	24.9	5.30
207-s	1.45	11.9	0.71	18.5	437	5.71	3.02	11.2	2.29	0.05	4.68	33.3	4.72
207-d	2.27	12.0	0.94	18.6	70.4	5.65	2.32	11.0	3.05	0.12	3.54	12.8	3.69
208-s	1.05	11.8	0.56	18.8	537	5.42	2.37	11.2	1.35	0.07	2.56	28.7	2.62
208-d	1.67	11.8	0.60	18.9	385	5.37	2.91	11.2	3.61	0.35	4.65	29.2	4.74
209-s	1.46	11.9	0.67	19.1	295	5.72	2.29	11.2	11.8	0.03	12.23	17.9	12.3
209-d	2.07	11.8	0.51	19.3	138	6.06	2.46	11.3	1.84	0.34	2.89	17.1	3.22
210-s	1.48	11.6	0.49	18.7	295	5.85	3.05	11.1	3.70	0.41	4.34	26.8	4.95
210-d	1.65	11.7	0.57	19.2	283	5.62	2.14	11.1	3.97	0.42	5.39	26.9	5.47
211	2.06	11.9	0.76	20.1	139	5.94	3.54	11.1	3.76	0.29	4.69	21.2	4.76
212-s	1.31	12.0	0.80	17.3	715	5.72	1.94	11.2	11.1	0.17	12.95	52.9	13.1
212-d	2.13	12.1	0.59	17.9	141	5.92	2.05	11.5	7.60	0.25	8.01	15.5	8.15
213	2.12	12.2	0.96	16.7	150	5.93	1.99	11.3	2.54	0.20	3.05	14.5	3.42
100/200-stream				16.5	161	6.87	5.78		0.65	1.00	0.94	21.1	1.49

Sample Point	DTB (m)	Well Elevation (m)	DTW (m)	Temp. (°C)	EC (µS/cm)	pH	DO (mg/L)	Total Hydraulic Head (m)	NH_4 (mg/L)	NO_3^- (mg/L)	DKN (mg/L)	Cl^- (mg/L)	TDN (mg/L)
200-tank				17.9	1066	6.12			67.5	0.09	81.7	108	81.8
300BG	3.47	13.8	1.17	19.6	65.4	6.31	2.59	12.6	0.09	0.12	0.22	4.16	0.28
301	2.16	13.6	1.24	19.6	48.0	6.59	2.49	12.4	0.72	0.13	2.00	9.69	2.14
302	2.39	13.6	1.11	19.5	220	6.22	3.68	12.5	1.15	6.58	1.89	30.2	9.81
303	2.51	13.5	1.18	19.1	149	6.02	2.57	12.6	0.10	3.40	0.39	20.5	4.40
304	2.65	13.3	0.98	18.4	116	5.87	2.24	12.4	0.27	1.27	0.92	13.6	2.30
305	2.64	13.2	1.10	18.3	114	5.65	3.12	12.1	0.06	0.65	0.23	10.3	0.92
306	2.26	13.3	1.07	17.9	257	5.53	3.17	12.3	0.05	0.20	0.27	27.3	0.29
307	2.65	13.3	1.24	17.8	226	5.63	2.58	12.0	0.05	0.11	0.23	35.3	0.27
308	2.80	13.3	1.30	17.7	195	5.22	2.91	12.0	0.07	0.24	0.12	22.6	0.35
309	2.13	13.3	1.28	18.0	415	5.19	2.71	12.0	0.10	0.15	0.74	57.9	0.77
310	2.28	13.2	1.28	17.5	188	5.58	3.22	11.9	0.09	0.10	0.43	21.5	0.46
311	2.14	13.3	1.15	18.3	115	5.12	2.52	12.2	0.18	0.91	0.34	21.2	1.43
300-stream				11.9	164	6.39	4.74		1.72	0.54	2.30	27.9	2.84
300-tank				18.2	611	5.81			42.4	0.55	39.2	45.0	44.3
401	4.24	16.7	2.86	19.7	262	5.90	2.84	13.8	0.81	7.91	1.60	28.8	9.19
402	4.05	16.5	2.68	20.2	379	6.19	3.11	13.8	1.41	22.8	3.19	26.0	31.4
403	4.13	16.9	2.87	19.1	164	5.58	3.21	14.0	0.40	0.37	0.62	6.09	0.97
400-tank				23.1	567	6.18	1.30		33.8	0.08	32.9	60.9	35.1
501	1.65	13.8	0.97	18.2	312	6.32	2.04	12.8	1.62	0.30	2.12	38.3	2.15
502	2.04	13.9	0.91	18.8	300	6.20	2.13	13.0	0.81	3.32	1.13	32.2	4.88
503	1.42	13.7	0.85	14.2	111	6.20	3.61	12.9	0.51	0.59	0.77	24.0	1.10
601	2.21	14.0	2.07	19.4	156	6.28	2.38	11.9	0.38	0.34	1.35	12.0	1.58
602	1.66	12.5	1.34	20.3	153	6.52	2.28	11.2	0.25	0.55	1.19	20.3	1.35
603	2.56	12.9	1.59	19.7	148	6.37	2.91	11.3	0.17	0.45	0.61	20.5	0.75
701	2.99	14.0	2.29	19.3	221	5.44	1.95	11.8	0.50	0.23	1.42	18.6	1.53
702	2.87	13.3	2.09	19.7	133	5.30	4.00	11.2	0.40	0.24	1.62	17.5	1.81
703	2.77	13.4	2.20	20.0	110	5.67	2.54	11.2	0.23	0.26	0.64	14.4	0.80
600/700-stream				19.7	93.2	5.93	7.10		0.16	0.44	0.26	8.44	0.44
801	5.73	14.2	3.87	20.0	104	5.38	2.15	10.3	0.05	0.11	0.13	7.77	0.26
802	1.37	10.7	1.38	17.9	56.4	4.39	2.70	9.36	0.09	0.19	0.10	6.84	0.18
803	1.39	10.8	1.40	18.4	70.2	4.37	3.07	9.36	0.08	0.16	0.07	7.46	0.24
800-stream				17.7	78.5	5.34	5.32		0.48	1.03	0.23	10.6	0.32
901	3.38	13.9	2.21	17.6	83.4	4.84	2.21	11.6	1.44	0.11	2.18	7.07	2.21
902	3.29	12.2	2.49	17.8	147	4.03	2.67	9.72	0.10	0.10	0.18	20.1	0.21
903	2.88	12.5	1.87	18.0	2009	2.83	2.49	10.6	0.13	0.20	0.45	19.7	0.52
900-stream				18.9	65.0	5.43	5.84		0.08	0.01	0.17	11.0	0.17

Sample Point	DTB (m)	Well Elevation (m)	DTW (m)	Temp. (°C)	EC (μS/cm)	pH	DO (mg/L)	Total Hydraulic Head (m)	NH_4 (mg/L)	NO_3^- (mg/L)	DKN (mg/L)	Cl^- (mg/L)	TDN (mg/L)
1001	3.64	13.7	3.04	20.6	73.8	5.88	2.90	10.9	0.37	0.04	1.06	8.73	1.09
1002	3.72	13.3	2.97	21.0	128	5.41	2.50	10.4	0.54	0.04	0.96	9.30	1.00
1000-stream				20.4	117	6.15	5.70		0.21	0.03	0.16	10.9	0.24
OWTS GW	2.28	13.6	1.07	18.9	264	5.86	2.62	12.0	4.92	1.44	5.55	25.8	7.37
CSS GW	2.70	12.1	2.05	18.0	239	4.85	2.45	10.0	0.32	0.20	0.80	12.7	0.90
OWTS T				21.0	781	6.18	1.30		51.3	0.20	54.1	74.8	56.1
CSS I									20.5	2.40	22.8	49.2	25.2
OWTS RS				14.2	162	6.63	5.26		1.18	0.77	1.62	24.5	2.17
CSS RS				19.2	88.3	5.71	5.99		0.23	0.38	0.20	10.2	0.30

GW= groundwater, T= tank, I= influent tank, RS= residential stream, DTB= depth to bottom of casing, DTW= depth to water. Piezometer elevation was calculated based on the relative elevation above sea level at a fixed point (septic tank for OWTS and a yard marker at CSS) approximately 15.2 meters above mean sea level.

APPENDIX G: GROUNDWATER NITROGEN SPECIATION

Nitrogen (N) speciation is shown below in tables G1-G3. NH4, NO3, and DON percentage is shown at each groundwater site. G1 focuses on N speciation at intensive OWTS sites, while G2 and G3 shows non-intensive OWTS and CSS sites.

Table G1. N speciation at intensive OWTS sites.

N-Speciation by Comparison Group				
Treatment Approach	**Site**	**NH4**	**NO3**	**DON**
OWTS Groundwater Intensive Sites	100			
	Tank	97.6 (± 3.82%)	0.10 (± 0.11%)	2.31 (± 3.7%)
	Drainfield	54.9 (± 32.6%)	4.55 (± 9.4%)	40.6 (± 30.9%)
	Near-Stream	82.2 (± 19.1%)	2.64 (± 6.2%)	15.8 (± 16.5%)
	Stream	25.2 (± 7.75%)	37.3 (± 18.9%)	37.5 (± 12.8%)
	Background	1.9 (± 1.39%)	92.6 (± 2.5%)	5.51 (± 2.7%)
	200			
	Tank	84.4 (± 18.1%)	0.10 (± 0.13%)	15.5 (± 18.1%)
	Drainfield	91.8 (± 13.1%)	4.25 (± 11.1%)	3.90 (± 5.37%)
	Near-Stream	75.8 (± 21.5%)	2.80 (± 7.31%)	21.4 (± 19.3%)
	Stream	25.2 (± 7.75%)	37.3 (± 18.9%)	37.5 (± 12.8%)
	Background	10.2 (± 15.5%)	82.0 (± 23.2%)	7.86 (± 8.00%)
	300			
	Tank	95.8 (± 8.16%)	1.25 (± 2.31%)	2.92 (± 5.85%)
	Drainfield	21.2 (± 24.7%)	60.3 (± 31.5%)	18.4 (± 28.4%)
	Near-Stream	17.0 (± 15.4%)	30.3 (± 30.9%)	52.8 (± 34.8%)
	Stream	51.7 (± 21.5%)	28.4 (± 24.4%)	19.9 (± 4.15%)
	Background	38.3 (± 39.5%)	15.9 (± 14.7%)	45.8 (± 53.0%)
	Average:			
	Tank	92.6 (± 12.2%)	0.48 (± 1.34%)	6.91 (± 12.0%)
	Drainfield	50.5 (± 37.4%)	25.5 (± 34.2%)	23.9 (± 29.7%)
	Near-Stream	58.4 (± 33.7%)	11.2 (± 21.9%)	30.4 (± 28.7%)
	Stream	36.6 (± 19.6%)	33.5 (± 20.0%)	29.9 (± 13.2%)
	Background	13.1 (± 24.5%)	70.7 (± 35.3%)	16.2 (± 29.9%)

Table G2. N speciation at non-intensive OWTS sites.

N-Speciation by Comparison Group				
Treatment Approach	**Site**	**NH4**	**NO3**	**DON**
OWTS Groundwater Non-Intensive Sites	400			
	Tank	96.5 (± 4.37%)	0.22 (± 0.26%)	3.31 (± 4.18%)
	Drainfield	8.24 (± 4.93%)	83.4 (± 8.88%)	8.39 (± 5.70%)
	Background	22.0 (± 16.9%)	35.6 (± 3.81%)	42.4 (± 16.9%)
	500			
	Tank	No data available; tank inaccessible		
	Drainfield	19.1 (± 19.2%)	45.9 (± 34.0%)	35.0 (± 28.8%)
	Background	88.7 (± 4.88%)	1.04 (± 0.51%)	10.4 (± 4.54%)
	Average:			
	Tank	96.5 (± 4.37%)	0.22 (± 0.26%)	3.31 (± 4.18%)
	Drainfield	13.7 (± 9.66%)	64.6 (± 29.6%)	21.7 (± 25.1%)
	Background	55.3 (± 37.4%)	18.3 (± 18.7%)	26.4 (± 20.5%)

Table G3. N speciation at CSS sites.

N-Speciation by Comparison Group				
Treatment Approach	**Site**	**NH4**	**NO3**	**DON**
CSS Groundwater	600			
	Groundwater	21.3 (± 10.1%)	15.2 (± 17.4%)	63.5 (± 16.1%)
	Stream	45.0 (± 25.0%)	29.4 (± 11.4%)	25.7 (± 36.3%)
	700			
	Groundwater	31.5 (± 20.0%)	10.0 (± 8.51%)	58.5 (± 24.8%)
	Stream	45.0 (± 25.0%)	29.4 (± 11.4%)	25.7 (± 36.3%)
	800			
	Groundwater	26.6 (± 30.8%)	42.1 (± 24.5%)	31.3 (± 31.2%)
	Stream	79.8%	20.2%	0.00%
	900			
	Groundwater	41.0 (± 35.2%)	12.4 (± 14.4%)	46.8 (± 33.7%)
	Stream	47.7%	4.6%	47.7%
	1000			
	Groundwater	36.8 (± 18.2%)	6.36 (± 7.25%)	56.9 (± 15.6%)
	Stream	85.6%	14.4%	0.00%
	Average:			
	Groundwater	31.3 (± 25.2%)	17.6 (± 25.2%)	51.1 (± 20.0%)
	Stream	60.6 (± 23.8%)	19.6 (± 12.0%)	19.8 (± 27.2%)

APPENDIX H: CSS TREATMENT EFFICIENCY

CSS treatment efficiency at the WWTP. Based on TN/TDN (PN here is negligible) concentration reductions from the influent and effluent tanks. These data were collected monthly from March 2012 to August 2012 (n=6) from the influent receiving tank (GUC Influent Concentration) and effluent exiting tank (GUC Outflow Concentration).

Site	**GUC Influent Concentration (mg/L - TN)**	**GUC Outflow Concentration (mg/L - TN)**	**Treatment Efficiency**
GUC WWTP	31.0	6.81	78.0%
	24.1	4.36	81.9%
	27.4	6.13	77.6%
	28.8	3.96	86.3%
	27.4	4.88	82.2%
	27.7	5.23	81.1%
AVERAGE:	27.7 (± 2.24)	5.23 (± 1.08)	81.2 (± 3.17%)

APPENDIX I: SOIL, GROUNDWATER, AND SURFACE WATER LOADING FROM OWTS VS. GROUNDWATER CSS LOADING TO SURFACE WATER

OWTS wastewater loads to the soil are shown in table I1. Table I2 shows TDN loading to the groundwater from OWTS wastewater discharges. Table I3 shows OWTS groundwater TDN loads to adjacent surface waters. Table I4 shows CSS groundwater TDN loads to adjacent surface waters.

Table I1. OWTS wastewater TDN loadings to the soil immediately underneath drainfield trenches.

Soil Loadings from the Tank						
Site	Date	Usage (L/mo)	Tank TDN (mg/L)	Soil Load (kg/yr)	Household Residents	Soil Load (kg/yr/per)
100	Sep-11	19871	46.5	11.1	4	2.77
	Nov-11	21196	57.0	14.5	4	3.62
	Jan-12	17411	82.6	17.3	4	4.31
	May-12	20250	66.7	16.2	4	4.05
	Average:	19682	63.2	14.8		3.69
	Median:	20061	61.8	15.4		3.84
	Standard Deviation:	1613.3	15.3	2.70		0.67
200	Sep-11		56.7			
	Nov-11	16843	94.4	19.1	2.50	7.63
	Jan-12	14194	94.3	16.1	2.50	6.42
	May-12	15329	81.9	15.1	2.50	6.02
	Average:	15455	81.8	16.7		6.69
	Median:	15329	88.1	16.1		6.42
	Standard Deviation:	1329.2	17.8	2.10		0.84
300	Sep-11	12869	55.3	8.5	2.00	4.27
	Nov-11	10030	42.0	5.1	2.00	2.53
	Jan-12	10977	44.8	5.9	2.00	2.95
	May-12	12112	35.1	5.10	2.00	2.55
	Average:	11497	44.3	6.2		3.08
	Median:	11544	43.4	5.5		2.75
	Standard Deviation:	1249.4	8.37	1.64		0.82
400	Sep-11	26684	31.6	10.11	4.00	2.53
	Nov-11	30280	36.5	13.28	4.00	3.32
	Jan-12	25170	41.6	12.6	4.00	3.14
	May-12	26495	30.8	9.79	4.00	2.45
	Average:	27157	35.1	11.43		2.86
	Median:	26590	34.1	11.33		2.83
	Standard Deviation:	2188.0	5.0	1.74		0.44

Table I2. OWTS wastewater discharge loading to the groundwater beneath the drainfield trenches.

TDN Loading from OWTS Tank to Groundwater Beneath Drainfield Trenches per Site								
Site	**Date**	**Hydraulic Gradient**	**Plume A (m^2)**	**K (m/d)**	**TDN (mg/L)**	**GW Loading (kg-TDN/yr)**	**Household Residents**	**GW Loading (kg/yr/person)**
100	Sep-11	0.043	49.4	0.30	9.53	2.23	4.0	0.56
	Nov-11	0.038	42.7	0.30	7.05	1.24	4.0	0.31
	Jan-12	0.028	39.1	0.30	10.85	1.32	4.0	0.33
	May-12	0.030	37.7	0.30	10.69	1.34	4.0	0.33
	Average:	0.035	42.2	0.30	9.53	1.53		0.38
	Median:	0.034	40.9	0.30	10.11	1.33		0.33
	STDEV:	0.007	5.22	0.00	1.75	0.47		0.12
200	Sep-11	0.041	45.1	0.18	24.8	3.80	2.5	1.52
	Nov-11	0.039	36.2	0.18	30.6	3.52	2.5	1.41
	Jan-12	0.039	30.2	0.18	40.1	3.37	2.5	1.35
	May-12	0.039	29.3	0.18	36.5	3.26	2.5	1.30
	Average:	0.039	35.2	0.18	33.0	3.49		1.40
	Median:	0.039	33.2	0.18	33.5	3.45		1.38
	STDEV:	0.001	7.30	0.00	6.73	0.24		0.09
300	Sep-11	0.027	61.8	0.09	2.26	0.12	2	0.06
	Nov-11	0.018	58.1	0.09	3.53	0.12	2	0.06
	Jan-12	0.022	56.2	0.09	1.58	0.06	2	0.03
	May-12	0.033	51.1	0.09	2.39	0.13	2	0.06
	Average:	0.025	56.8	0.09	2.44	0.11		0.05
	Median:	0.025	57.1	0.09	2.32	0.12		0.06
	STDEV:	0.006	4.45	0.00	0.81	0.03		0.01
400	Sep-11	0.006	76.3	0.26	14.52	0.60	4	0.15
	Nov-11	0.006	53.1	0.26	20.80	0.61	4	0.15
	Jan-12	0.003	26.5	0.26	34.75	0.25	4	0.06
	May-12	0.003	29.3	0.26	11.01	0.10	4	0.03
	Average:	0.005	46.3	0.26	20.27	0.39		0.10
	Median:	0.005	41.2	0.26	17.66	0.42		0.11
	STDEV:	0.002	23.3	0.00	10.47	0.25		0.06
500	Sep-11	0.015	12.2	0.39	11.14	0.28	2	0.14
	Nov-11	0.002	12.2	0.39	2.95	0.01	2	0.00
	Jan-12	0.012	12.2	0.39	2.21	0.05	2	0.02
	May-12	0.020	12.2	0.39	3.21	0.11	2	0.05
	Average:	0.012	12.17	0.39	4.88	0.11		0.06
	Median:	0.014	12.17	0.39	3.08	0.08		0.04
	STDEV:	0.008	0.00	0.00	4.20	0.12		0.06

Table I3. OWTS groundwater loading to adjacent streams at site 100 and 200. Site 300 was not calculated because the extent of the plume depth was unknown because near-stream piezometers were not nested. Sites 400 and 500 did not have any adjacent streams.

Groundwater TDN Loading to Nearby Streams per Site								
Site	**Date**	**Hydraulic Gradient**	**Plume A (sq. m)**	**K (m/d)**	**TDN (mg/L)**	**Stream Loading (kg-TDN/yr)**	**Household Residents**	**Stream Loading (kg/yr/person)**
100	Sep-11	0.043	26.1	0.304	7.85	0.97	4	0.24
	Nov-11	0.038	21.1	0.304	6.78	0.59	4	0.15
	Jan-12	0.028	21.2	0.304	7.79	0.51	4	0.13
	May-12	0.030	20.1	0.304	7.49	0.50	4	0.12
	Average:	0.035	22.1	0.304	7.47	0.642		0.16
	Median:	0.034	21.2	0.304	7.64	0.551		0.14
	STDEV:	0.007	2.70	0.000	0.49	0.221		0.06
200	Sep-11	0.041	35.4	0.18	5.71	0.54	2.5	0.21
	Nov-11	0.039	30.8	0.18	5.71	0.44	2.5	0.18
	Jan-12	0.039	29.5	0.18	6.33	0.47	2.5	0.19
	May-12	0.039	27.7	0.18	10.7	0.74	2.5	0.29
	Average:	0.039	30.8	0.18	7.12	0.55		0.22
	Median:	0.039	30.1	0.18	6.02	0.50		0.20
	STDEV:	0.001	3.29	0.00	2.41	0.13		0.05

Table I4. CSS residential yard groundwater TDN loading to adjacent streams.

Groundwater TDN Loading to Nearby Streams per Site									
Site	**Date**	**Hydraulic Gradient**	**Plume A* (sq. m)**	**K (m/d)**	**Q (L/d)**	**TDN (mg/L)**	**Stream Loading (kg-TDN/yr)**	**Household Residents**	**Stream Loading (kg/yr/person)**
600	Sep-11	0.032	30.8	1.06	1109	1.82	0.27	2.27	0.12
	Nov-11	0.043	25.9	1.06	1224	0.73	0.15	2.27	0.06
	Jan-12	0.028	25.4	1.06	762	1.80	0.24	2.27	0.10
	May-12	0.024	23.9	1.06	663	0.70	0.08	2.27	0.03
	Average:	0.032	26.5	1.06	940	1.26	0.18		0.08
	Median:	0.030	25.6	1.06	935	1.27	0.19		0.08
	STDEV:	0.008	2.97	0.00	270	0.63	0.09		0.04
700	Sep-11	0.033	30.8	1.06	995	0.73	0.11	2.27	0.05
	Nov-11	0.046	25.9	1.06	1204	1.09	0.23	2.27	0.10
	Jan-12	0.026	25.4	1.06	654	0.76	0.09	2.27	0.04
	May-12	0.026	23.9	1.06	643	4.64	0.57	2.27	0.25
	Average:	0.033	26.5	1.06	874	1.81	0.25		0.11
	Median:	0.029	25.6	1.06	824	0.93	0.17		0.08
	STDEV:	0.009	2.97	0.00	274	1.90	0.22		0.10
800	Sep-11	0.130	30.8	1.07	4330	0.18	0.11	2.27	0.05
	Nov-11	0.029	25.9	1.07	786	0.26	0.04	2.27	0.02
	Jan-12	0.099	25.4	1.07	2671	0.23	0.11	2.27	0.05
	May-12	0.097	23.9	1.07	2574	0.14	0.06	2.27	0.03
	Average:	0.089	26.5	1.07	2590	0.20	0.08		0.04
	Median:	0.098	25.6	1.07	2622	0.21	0.09		0.04
	STDEV:	0.043	2.97	0.00	1448	0.05	0.04		0.02
900	Sep-11	0.201	30.8	0.16	907	0.28	0.04	2.27	0.02
	Nov-11	0.032	25.9	0.16	125	0.44	0.01	2.27	0.00
	Jan-12	0.129	25.4	0.16	515	0.55	0.05	2.27	0.02
	May-12	0.077	23.9	0.16	399	0.35	0.02	2.27	0.01
	Average:	0.110	26.5	0.16	486	0.41	0.03		0.01
	Median:	0.103	25.6	0.16	457	0.40	0.03		0.01
	STDEV:	0.073	2.97	0.00	325	0.12	0.02		0.01
1000	Sep-11	0.071	30.8	0.11	246.4	0.52	0.02	2.27	0.01
	Nov-11	0.017	25.9	0.11	48.7	0.91	0.01	2.27	0.00
	Jan-12	0.034	25.4	0.11	98.1	0.22	0.00	2.27	0.00
	May-12	0.005	23.9	0.11	13.94	2.35	0.01	2.27	0.00
	Average:	0.032	26.5	0.11	102	1.00	0.01		0.00
	Median:	0.025	25.6	0.11	73	0.71	0.01		0.00
	STDEV:	0.029	2.97	0.00	102	0.95	0.01		0.00

***= Plume area is assumed based on the average of the 100 and 200 near-stream plume dynamics**

APPENDIX J: GROUNDWATER NITROGEN ISOTOPE DATA

Table J1 shows the raw data for each groundwater isotopic monitoring event. Figure J1 shows the data plotted as compared to Kendall and McDonnell (1998) suggested N sources based on $\delta^{15}N$ vs. $\delta^{18}O$ values

Table J1. Raw $\delta^{15}N$ and $\delta^{18}O$ values for transections at varying groundwater residential sites. The first sampling event occurred in November 2011, while the second occurred in May 2012.

	Sample	ID	$\delta^{15}N$ vs. Air	$\delta^{18}O$ vs. V-SMOW
November 2011	1	103	28.05	22.42
	2	108S	23.76	11.68
	3	108M	9.37	13.10
	4	202		
	5	206	11.78	22.28
	6	212D	20.97	17.62
	7	200 Stream	12.20	8.63
	8	302	20.49	14.55
	9	401	24.24	16.20
	10	400BG	10.64	7.10
	11	500BG	13.53	33.33
	12	502	19.05	12.91
	13	701	11.68	14.85
	14	702	9.23	15.04
	15	902	13.25	25.22
	16	1001	12.49	22.60
May 2012	1	101	15.97	9.61
	2	103	13.15	10.03
	3	108M	3.13	6.14
	4	203	14.94	10.45
	5	204	-0.54	5.27
	6	212S	5.30	9.54
	7	200 Stream	12.11	6.81
	8	300	9.85	21.17
	9	302	18.19	13.73
	10	402	11.99	12.42
	11	403	11.87	3.76
	12	502	8.15	28.21
	13	701	12.22	23.41
	14	801	16.46	23.96
	15	901	-2.22	8.16
	38	201	41.92	68.26
	40	110S	9.42	28.92

APPENDIX K: NH_4^+ AND DO DRAINFIELD AND NEAR-STREAM CONCENTRATIONS

The table shows the NH_4^+ and DO concentrations (mg/L) from the drainfield to the near-stream. The average CEC for all of site 100 and 200 is also shown. DF= drainfield, NS= near-stream, and CEC= cation exchange capacity.

Site 100	Site 200	Site 100				Site 200			
Average CEC	Average CEC	DF NH_4	NS NH_4	DF DO	NS DO	DF NH_4	NS NH_4	DF DO	NS DO
5.57	4.78	5.41	9.46	0.96	1.71	27.7	2.16	1.54	1.05
		2.93	3.45	1.57	1.27	20.9	1.72	1.43	2.1
		3.65	10.79	5.15	2.9	33.7	1.41	3.9	3.2
		6.85	13.39	1.93	2.3	38.8	2.17	1.8	1.8
		0.17	2.96	1.33	1.34	29.9	2.51	1.66	1.68
		0.85	0.72	1.65	1.27	56.0	3.51	1.57	1.93
		0.82	3.42	10	2.1	64.3	1.34	4.4	3.6
		0.96	4.19	1.66	2.2	56.9	3.16	1.7	1.28
		0.24	3.44	5.83	2.43	5.9	1.00	1.63	3.28
		0.20	4.51	1.6	1.34	12.8	0.93	1.9	1.66
		2.56	7.84	5.29	1.41	12.4	3.48	5.3	4.87
		4.50	7.36	2.78	7.96	12.4	3.77	1.4	2.25
		26.66	7.56	2.94	2.09		2.91		1.69
		19.46	13.34	1.78	1.36		2.63		1.57
		21.43	5.63	7.07	1.08		2.48		4.12
		0.76	14.15	1.72	3.4		4.91		1.88
		3.89	4.68	1.7	2.19		0.89		1.72
		14.77	14.80		1.28		1.22		2.09
		18.98	5.32		1.65		1.37		3.3
		3.12	3.80		3.3		1.93		1.7
		0.40	7.73		1.56		3.89		2.31
			7.24		1.62		3.65		5.6
			8.17		7.07		3.33		2.04
			3.65				4.21		1.47
			7.53				11.69		1.71
			15.17				8.86		4.26
							11.64		1.71
							14.93		1.95
							1.56		1.48
							0.57		4.5
							2.30		1.91
							2.95		1.9
							4.79		2.04
							1.34		2.7
							4.51		5.55
							4.16		1.86
							4.07		1.6
							3.70		3.2
							4.31		1.9

Site 100	Site 200	Site 100				Site 200			
Average CEC	Average CEC	DF NH4	NS NH4	DF DO	NS DO	DF NH4	NS NH4	DF DO	NS DO
							5.79		2.84
							3.86		1.45
							5.82		7.3
							3.17		2.55
							4.07		2.2
							6.74		1.02
							6.44		2.45
							9.86		2.1
							21.23		2.07
							6.31		1.86
							7.72		2.75
							6.51		1.5
							10.44		1.48
							2.83		1.63
							1.59		3.3
							0.90		1.55
							6.77		
		Average				Average			
	With outliers:	6.60	7.32	3.23	2.38	31.0	4.5	2.4	2.4
	Without outliers:	12.40	7.32	2.89	2.38	No outliers			

APPENDIX L: SURFACE WATER PHYSICAL AND CHEMICAL WATER QUALITY SUMMARY

Mean baseflow and storm flow surface water physical water quality this data is summarized from August 2011 to August 2012. Mean storm data occurred from 2 storm events that occurred between Nov. 5-7, 2011 and May 9-10, 2012. Temp= temperature in degrees Celsius, EC= electrical conductance in microsiemens per cm, DO= dissolved oxygen, turb= turbidiy, ntu= nephelometric turbidity units, Q= discharge, WA= watershed area, d= day, and cfs= cubic feet per second.

	Sample Point	pH	Temp. (°C)	EC (µS/cm)	DO (mg/L)	Turb. (ntu)	Q (cfs)	WA (ha)	NH_4 (mg/L)	NO_3^- (mg/L)	DKN (mg/L)	PN (mg/L)	Cl^- (mg/L)	TN (mg/L)	TN Load (kg-TN/yr/ha)
Baseflow	FT-O	6.43	17.4	86.4	6.10	11.6	0.98	364	0.11	0.13	0.33	0.13	11.9	0.59	1.42
	FT-1	6.23	17.1	78.2	6.24	11.1	0.46	220	0.14	0.12	0.33	0.12	11.1	0.58	1.18
	FT-2	6.07	17.2	102	4.47	14.4	0.44	190	0.31	0.06	0.39	0.16	9.46	0.73	1.37
	MHB	6.03	17.3	114	6.10	16.7	0.62	269	0.15	0.69	0.33	0.18	18.0	1.26	2.63
	BELL	5.93	17.4	91.2	5.85	5.34		172	0.05	0.72	0.25	0.13	11.4	1.13	
	EP-O	6.08	17.1	104	5.62	17.0	0.75	201	0.19	0.70	0.60	0.24	15.5	1.55	5.08
	EP-1	5.67	17.2	98.2	4.80	18.2	0.27	113	0.08	0.76	0.32	0.34	15.9	1.41	3.01
	MILL	6.10	17.5	89.6	5.82	16.8	0.60	200	0.07	0.49	0.46	0.09	13.3	1.05	2.71
	CHOK	5.97	17.2	88.1	4.96	12.6	0.97	368	0.06	0.54	0.47	0.20	15.5	1.22	3.12

	Sample Point	pH	Temp. (°C)	EC (µS/cm)	DO (mg/L)	Turb. (ntu)	Q (cfs)	WA (ha)	NH_4 (mg/L)	NO_3^- (mg/L)	DKN (mg/L)	PN (mg/L)	Cl^- (mg/L)	TN (mg/L)	TN Load (kg/ha/d)
Storm	FT-O	6.61	16.3	71.3	4.98	27.8	10.5	364	0.14	0.17	0.44	0.24	5.67	0.79	0.05
	FT-1	6.53	16.2	59.0	5.02	40.7	3.20	220	0.24	0.20	0.53	0.21	6.85	0.85	0.03
	FT-2	6.28	16.1	71.3	4.16	20.2	5.25	190	0.22	0.12	0.45	0.14	5.51	0.67	0.06
	MHB	6.02	15.8	80.0	4.87	26.5	7.04	269	0.22	0.50	0.49	0.19	8.29	1.10	0.09
	BELL	5.72	13.9	53.0	6.91	13.1		172	0.03	0.62	0.29	0.14	6.58	1.06	
	EP-O	5.94	15.2	114	3.81	93.6	3.70	201	0.22	0.70	0.67	0.50	11.9	1.76	0.07
	EP-1	5.75	13.9	78.3	5.25	40.8	1.86	113	0.11	0.58	0.49	0.25	9.68	1.31	0.03
	MILL	6.19	15.8	77.3	6.20	50.1	3.01	200	0.09	0.33	0.46	0.24	8.48	1.02	0.04
	CHOK	6.01	15.9	72.0	4.17	23.7	11.0	368	0.12	0.53	0.55	0.34	9.92	1.40	0.11

APPENDIX M: ANNUAL TN LOADS

Annual discharge was calculated based on the average monthly discharge measurements taken from August 2011 to August 2012. These were then adjusted based on the web-based hydrograph analysis tool (Lim *et al.*, 2005) estimated percent of storm flow (SF) and baseflow (BF). Baseflow TN export estimates were based on monthly discharge and N concentration data collected from August 2011-August 2012. Storm N concentration data were based on 2 storms that occurred from November 5-7, 2011 and May 9-10, 2012.

Site	**Average Stream Q (cf/s)**	**SF %**	**BF %**	**Storm Q (L/yr)**	**Baseflow Q (L/yr)**
FT-1	0.46	37%	63%	148733154	258384250
FT-2	0.44	42%	58%	166812768	230360490
MHB	0.62	36%	64%	199856667	355300741
EP-O	0.75	23%	77%	154947212	512411771
MILL	0.59	26%	74%	137980078	392712529
CHOK	0.97	28%	71%	242937457	616019980
CSS	0.51	38%	62%	171800863	281348494
OWTS	0.77	26%	74%	178621582	507048093

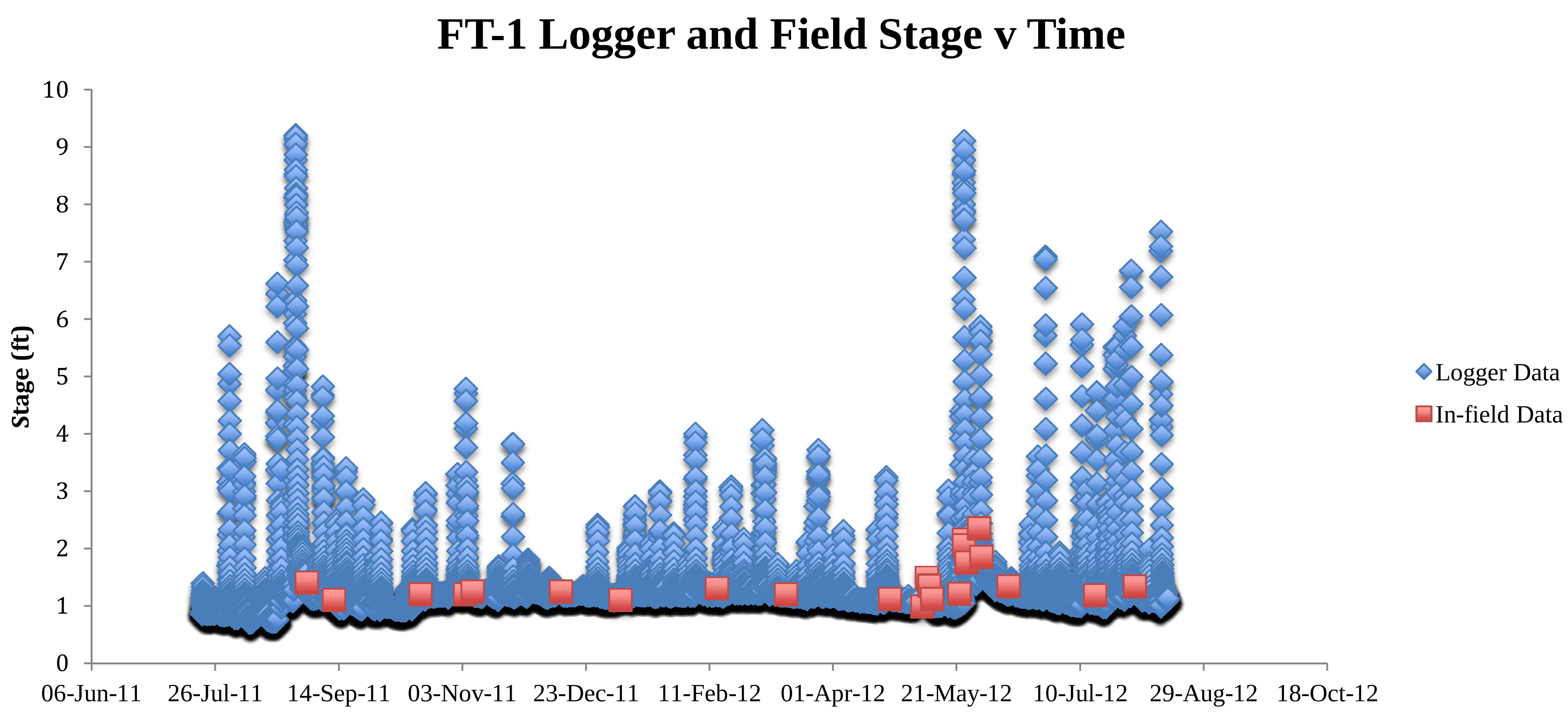
Site	**SF TN (mg/L)**	**BF TN (mg/L)**	**SF TN Export (kg/yr)**	**BF TN Export (kg/yr)**	**Total TN Export (kg/yr)**	**Total TN Export (kg/yr/ha)**
FT-1	0.85	0.58	127	150	127	150
FT-2	0.67	0.73	111	168	111	168
MHB	1.10	1.26	219	448	219	448
EP-O	1.76	1.55	272	793	272	793
MILL	1.02	1.05	141	413	141	413
CHOK	1.40	1.22	339	753	339	753
CSS	0.87	0.86	152	255	408 (± 224)	1.74 (± 0.66)
OWTS	1.39	1.27	251	653	904 (± 303)	3.71 (± 1.39)

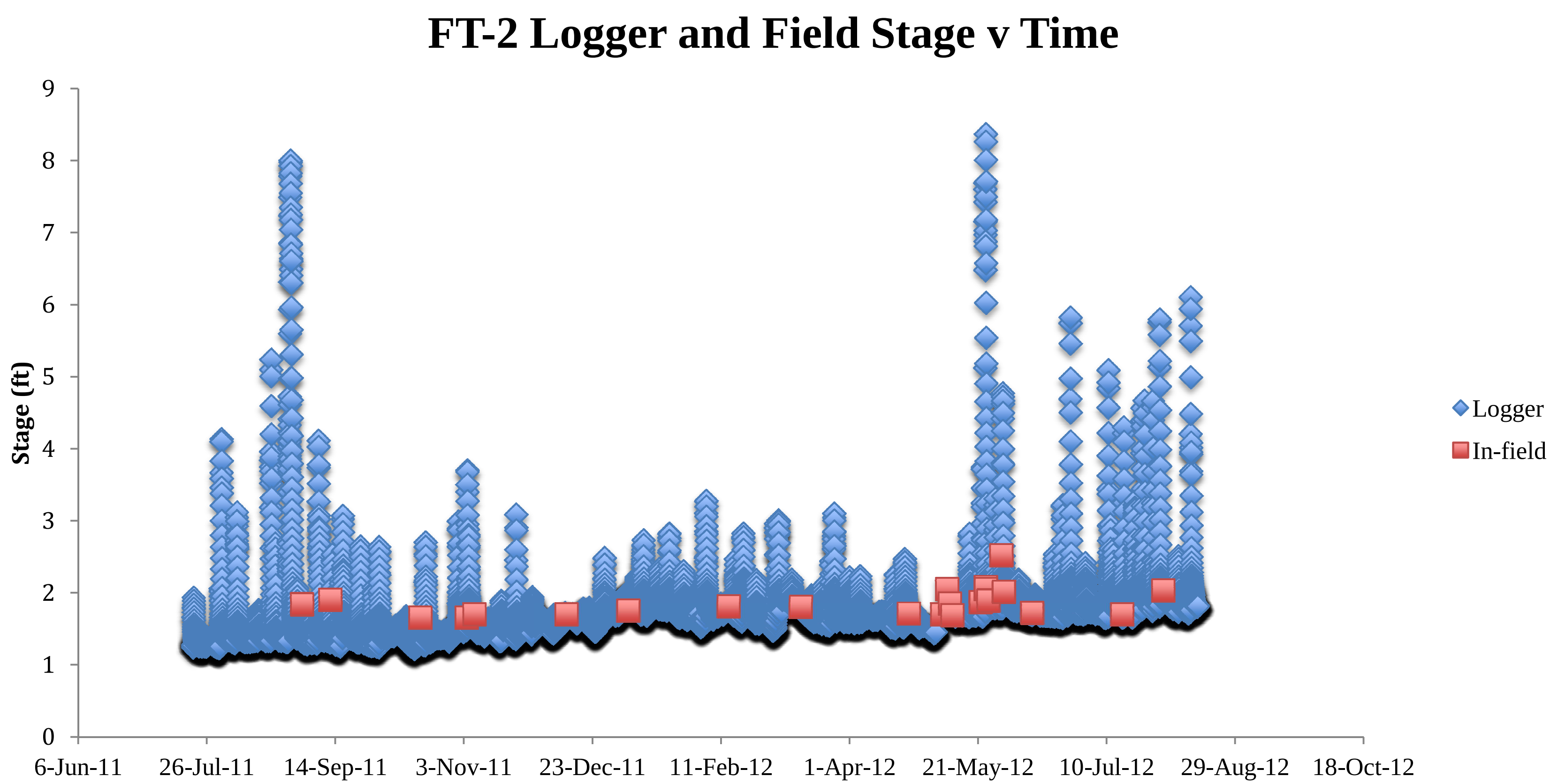
Discharge data that were input into the hydrograph separation model were estimated using discharge-rating curves and their equations are listed below. BF= baseflow and SF= storm flow.

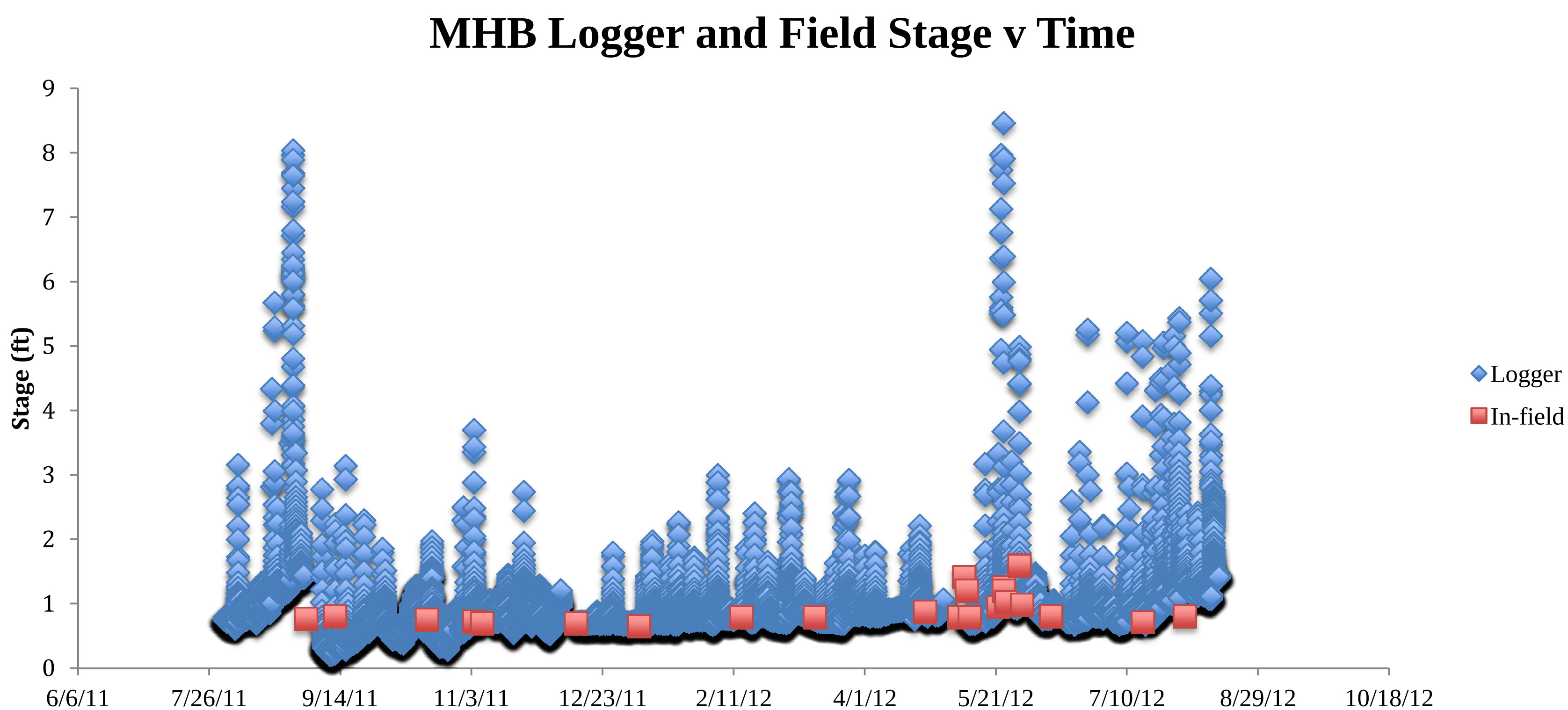
Site	Eq. 1 (Lower flow rating curve)	Eq. 2 (Higher flow rating curve)	BF R^2	SF R^2	Explanation
FT-1	$y = 0.1446x^{4.9882}$	y = 16.04ln(x) - 5.5622	0.70	0.88	Eq. 1 was used from 0-2.45 ft. and Eq. 2 was used >2.45 ft.
FT-2	$y = 0.001x^{10.949}$	y = 58.176ln(x) - 35.712	0.74	0.94	Eq. 1 was used from 0-2.5 ft. and Eq. 2 was used >2.5 ft.
MHB	$y = 0.0148e^{4.831x}$	y = 64.083ln(x) - 6.0909	0.72	0.67	Eq. 1 was used from 0-1.61 ft. and Eq. 2 was used >1.61 ft.
EP-O	$y = 0.0909e^{1.5327x}$	y = 22.891ln(x) - 13.239	0.62	0.94	Eq. 1 was used from 0-2.6 ft. and Eq. 2 was used >2.6 ft.
MILL	$y = 7E\text{-}08x^{16.011}$	y = 112.94ln(x) - 115.89	0.66	0.66	Eq. 1 was used from 0-3.1 ft and Eq. 2 was used >3.1 ft.
CHOK	$y = 8E\text{-}08x^{16.011}$	y = 311.44ln(x) - 363.09	0.78	0.92	Eq. 1 was used from 0-3.71 ft. and Eq. 2 was used >3.71 ft.

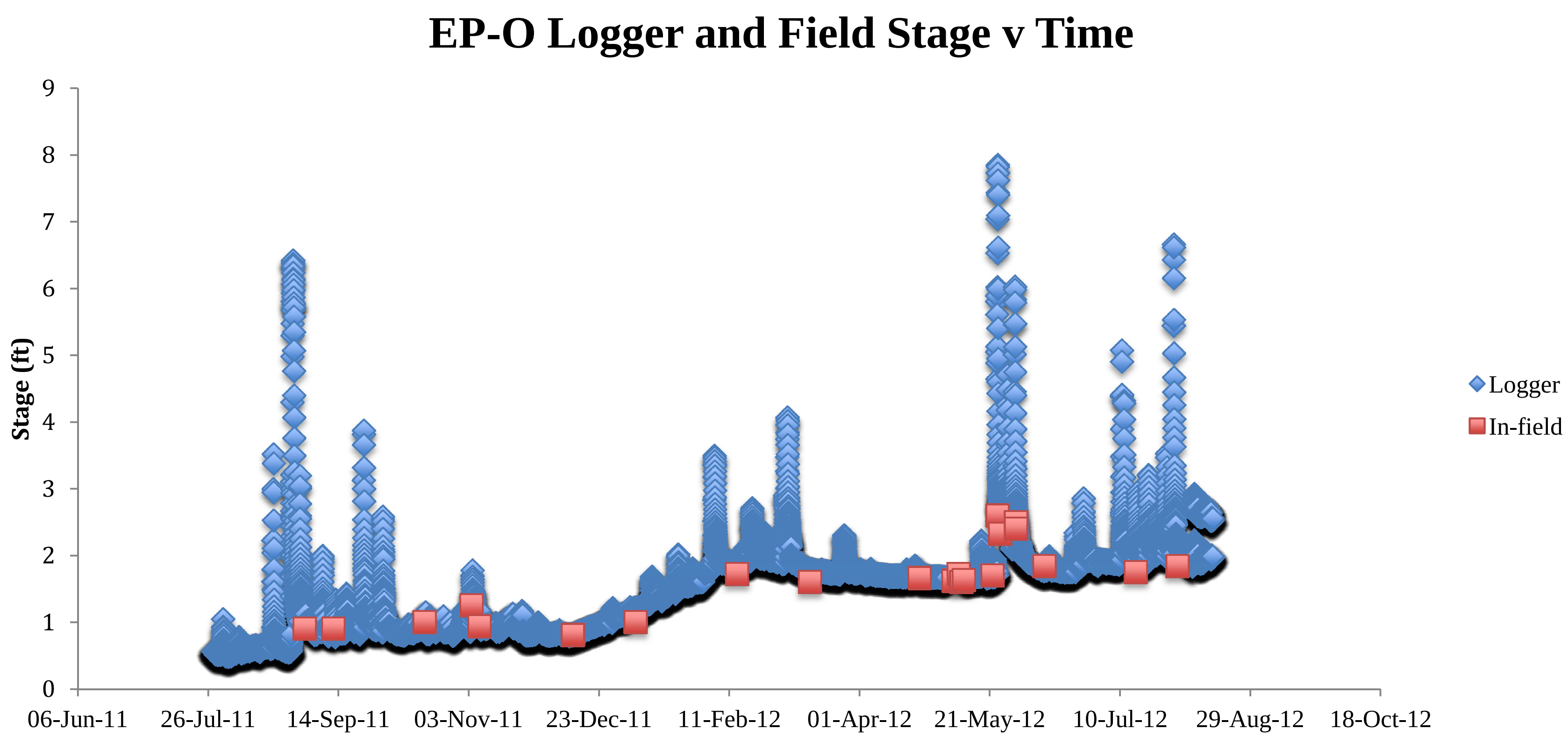
APPENDIX N: LOGGER STAGE VS. MONTHLY MEASURED STAGE DATA PER MAIN WATERSHED

Logger data are shown in blue diamonds, while monthly measured stage is shown as red squares.



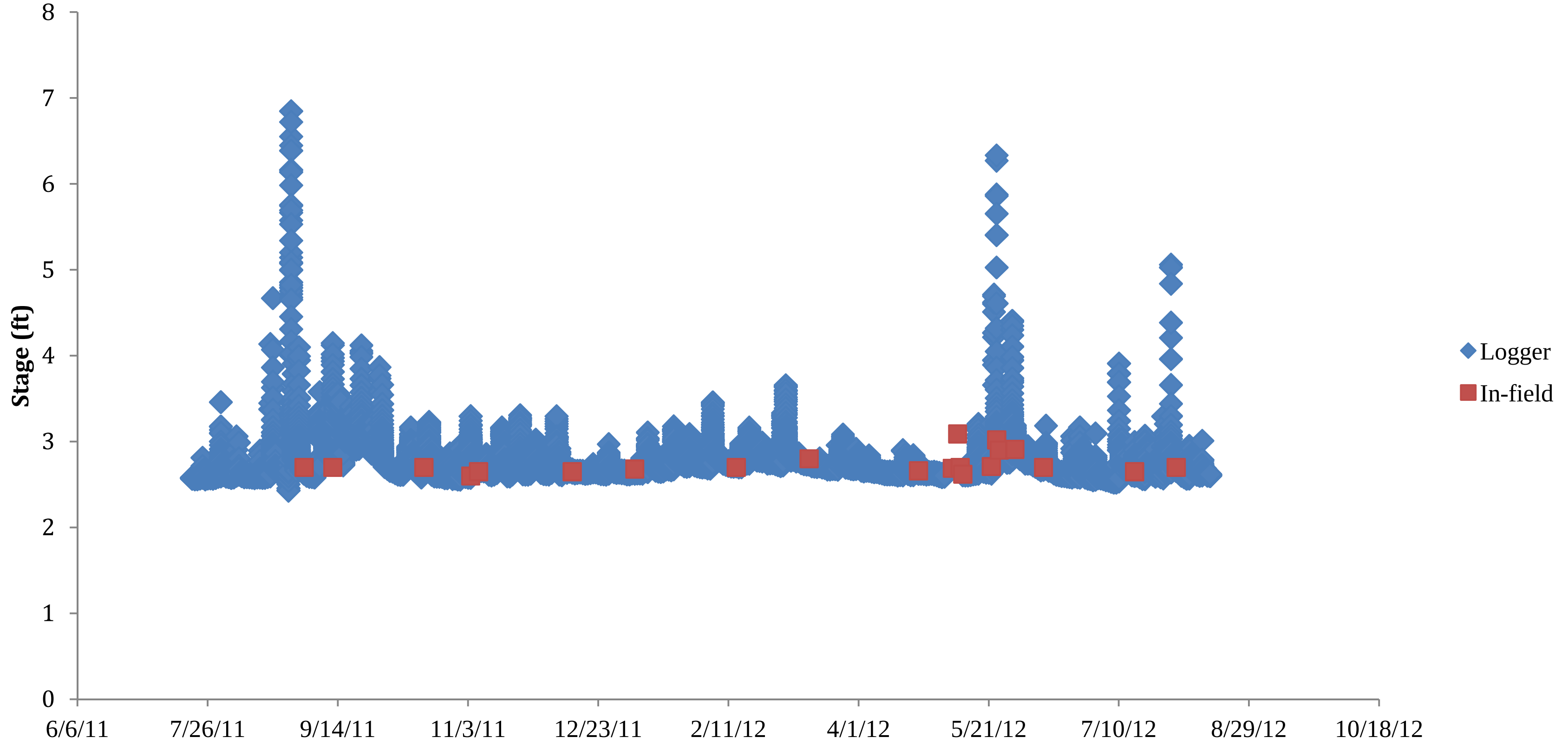

FT-2 Logger and Field Stage v Time
Stage (ft)
0
1
2
3
4
5
6
7
8
9
6-Jun-11
26-Jul-11
14-Sep-11
3-Nov-11
23-Dec-11
11-Feb-12
1-Apr-12
21-May-12
10-Jul-12
29-Aug-12
18-Oct-12
Logger
In-field

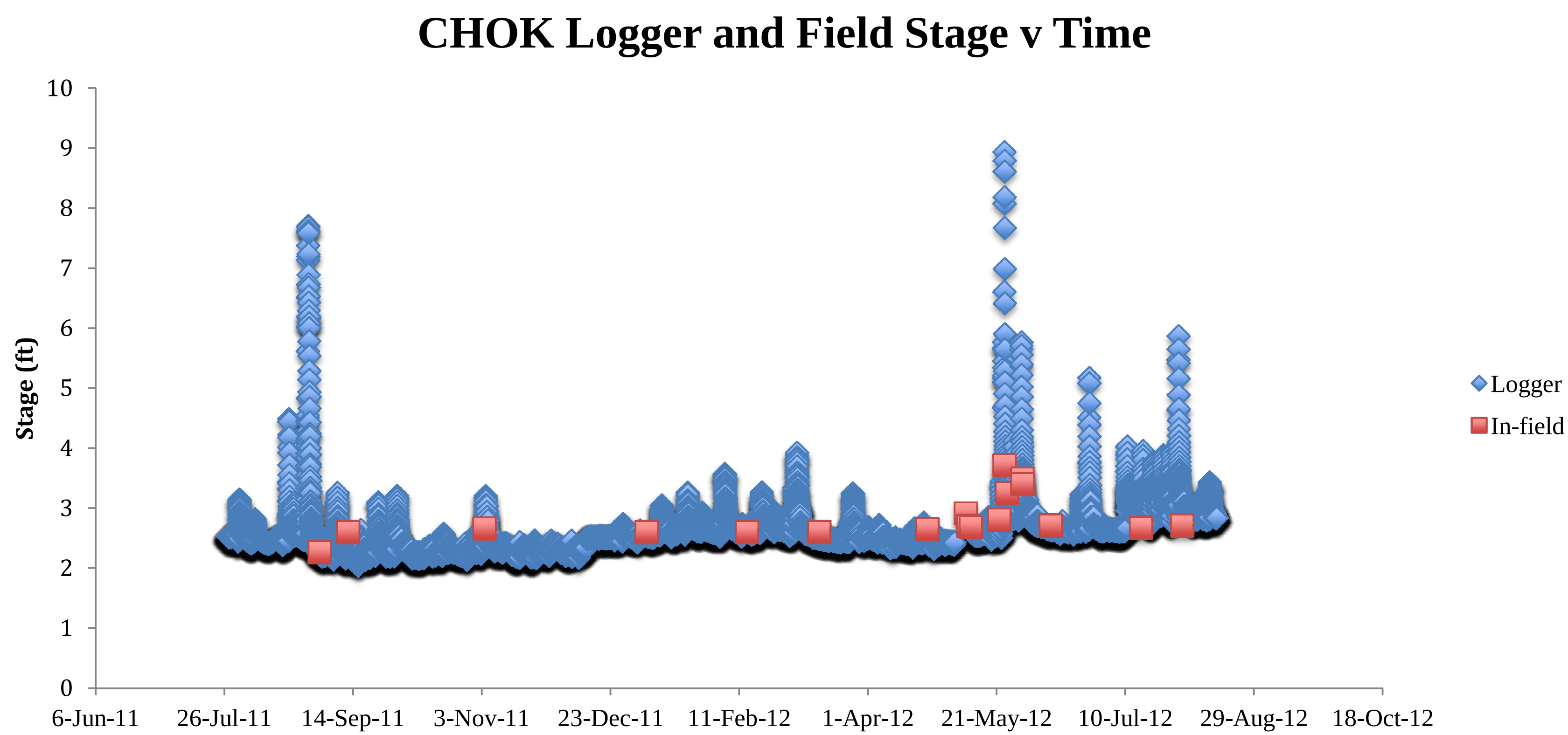

MHB Logger and Field Stage v Time
Stage (ft)
0
1
2
3
4
5
6
7
8
9
6/6/11
7/26/11
9/14/11
11/3/11
12/23/11
2/11/12
4/1/12
5/21/12
7/10/12
8/29/12
10/18/12
Logger
In-field



EP-O Logger and Field Stage v Time
Stage (ft)
0
1
2
3
4
5
6
7
8
9
06-Jun-11
26-Jul-11
14-Sep-11
03-Nov-11
23-Dec-11
11-Feb-12
01-Apr-12
21-May-12
10-Jul-12
29-Aug-12
18-Oct-12
Logger
In-field

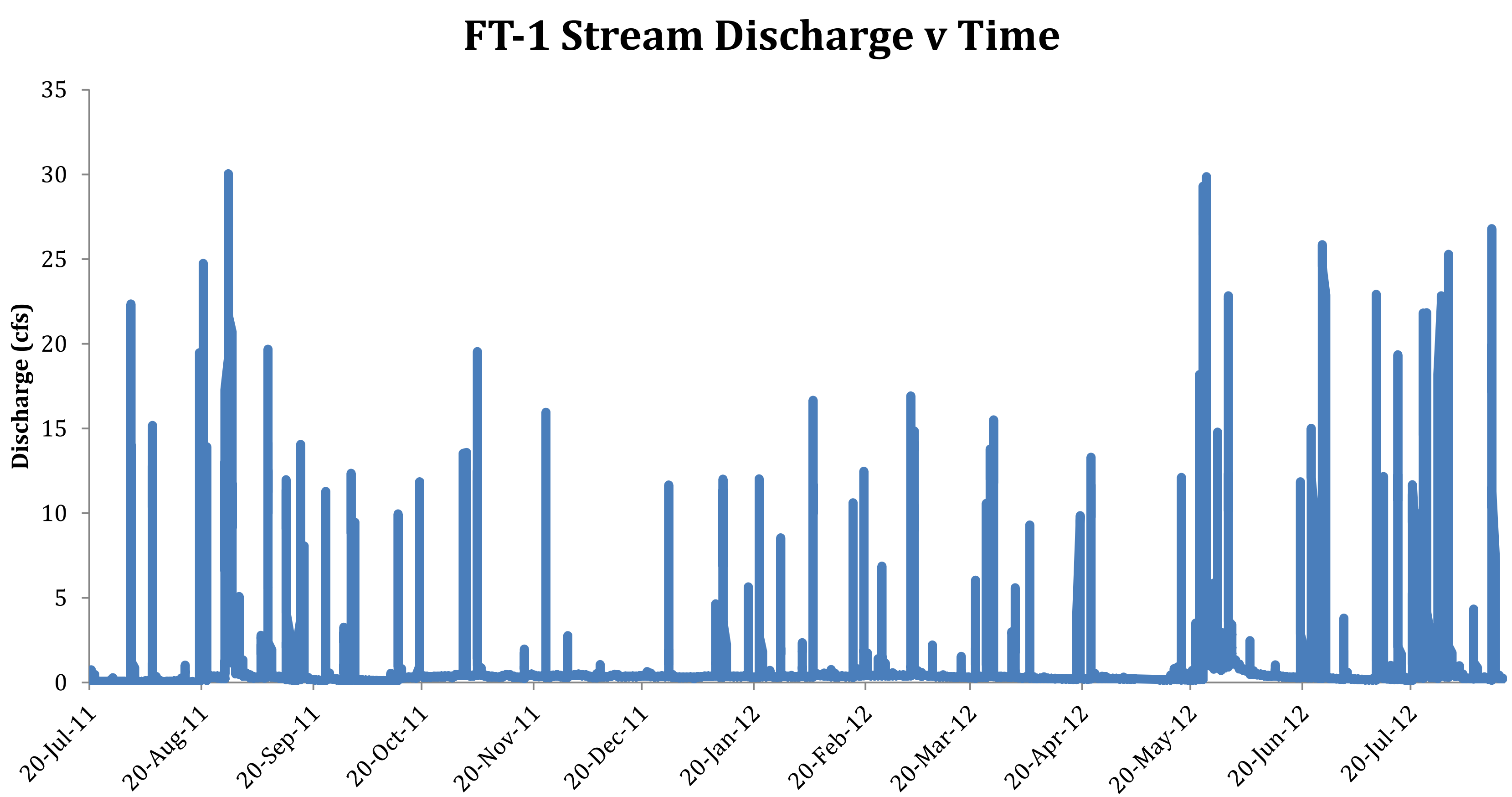
MILL Logger and Field Stage v Time

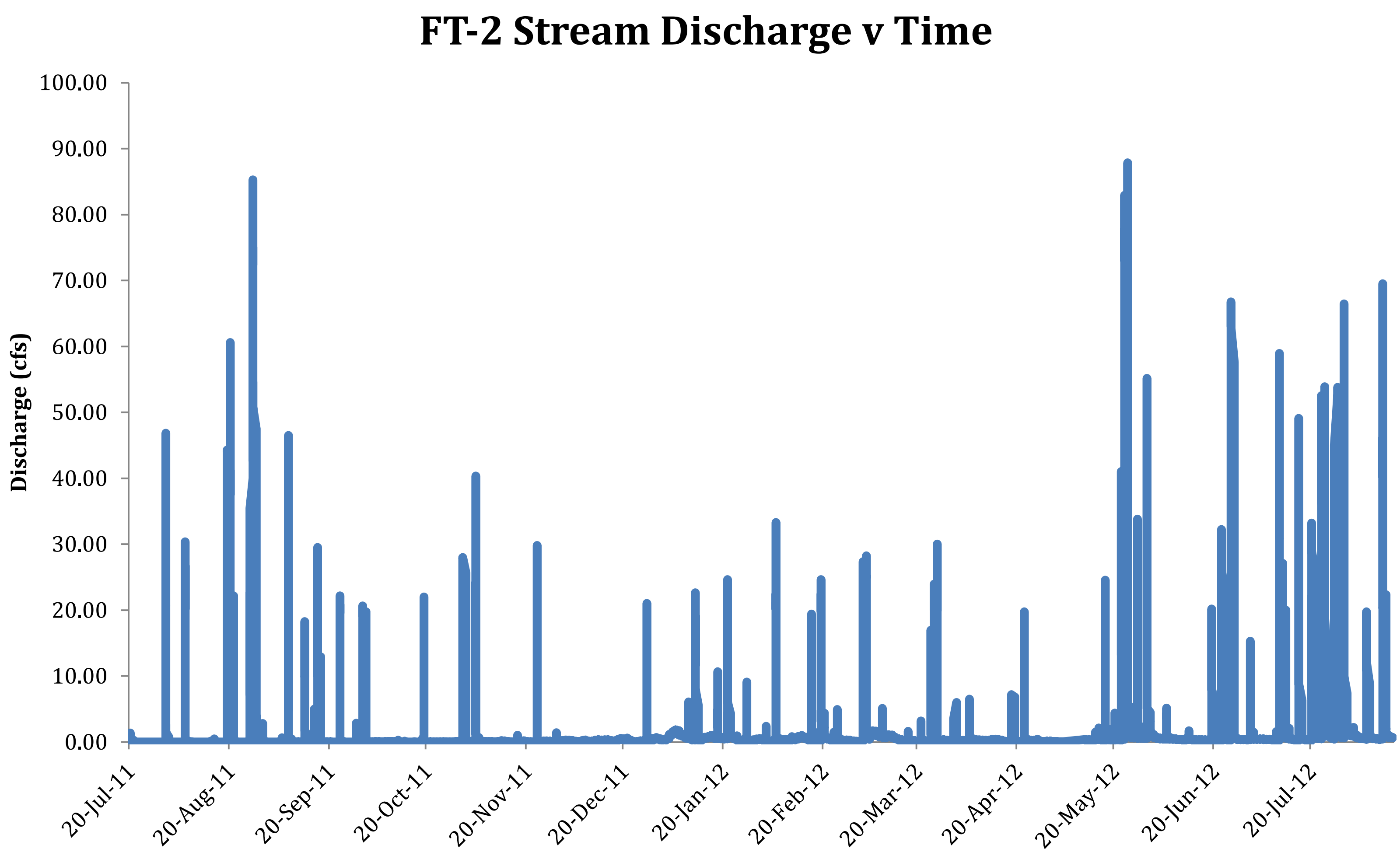




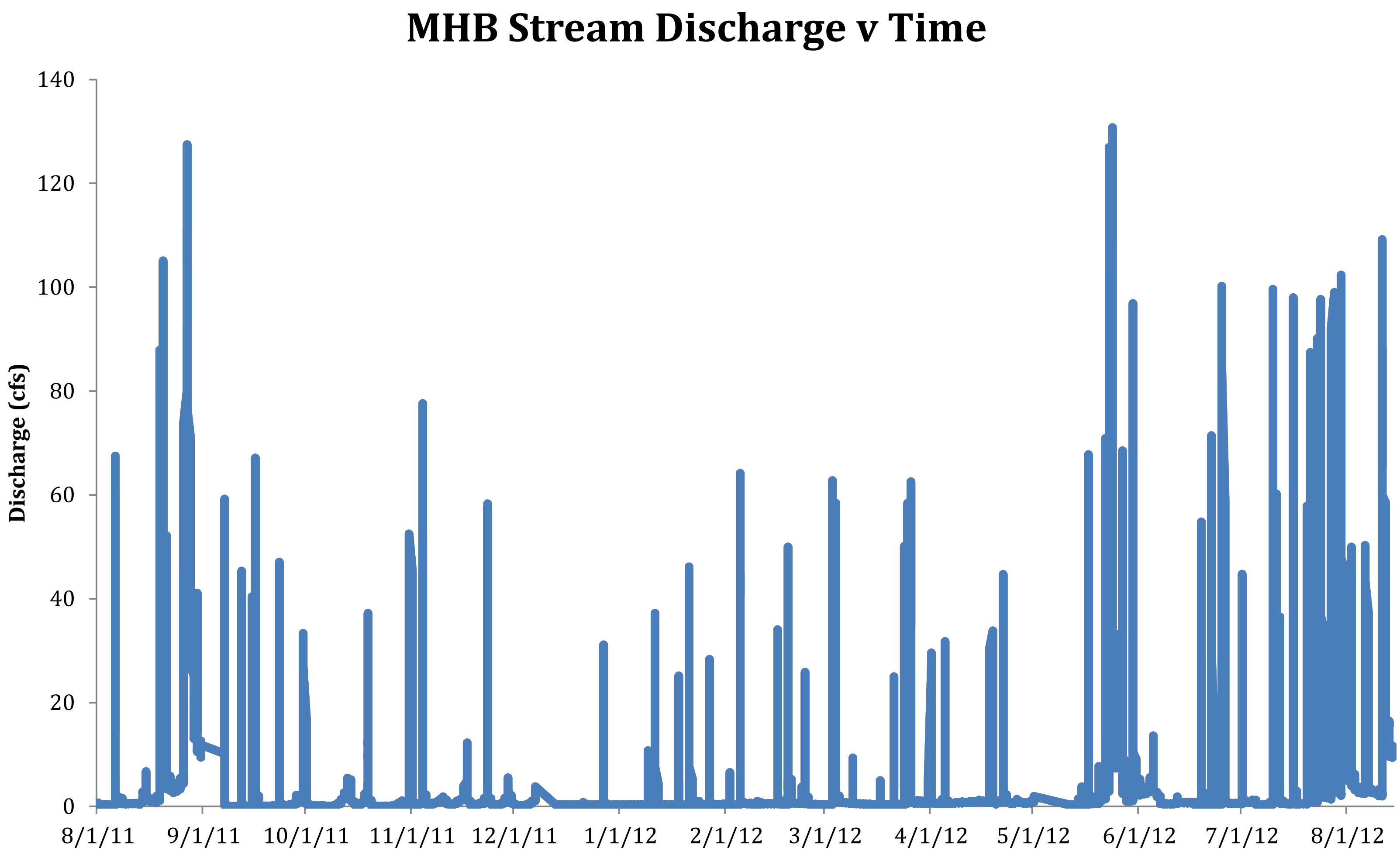
CHOK Logger and Field Stage v Time
Stage (ft)
10
9
8
7
6
5
4
3
2
1
0
6-Jun-11
26-Jul-11
14-Sep-11
3-Nov-11
23-Dec-11
11-Feb-12
1-Apr-12
21-May-12
10-Jul-12
29-Aug-12
18-Oct-12
Logger
In-field

APPENDIX O: SURFACE WATER DISCHARGE VS. TIME PER MAIN WATERSHED

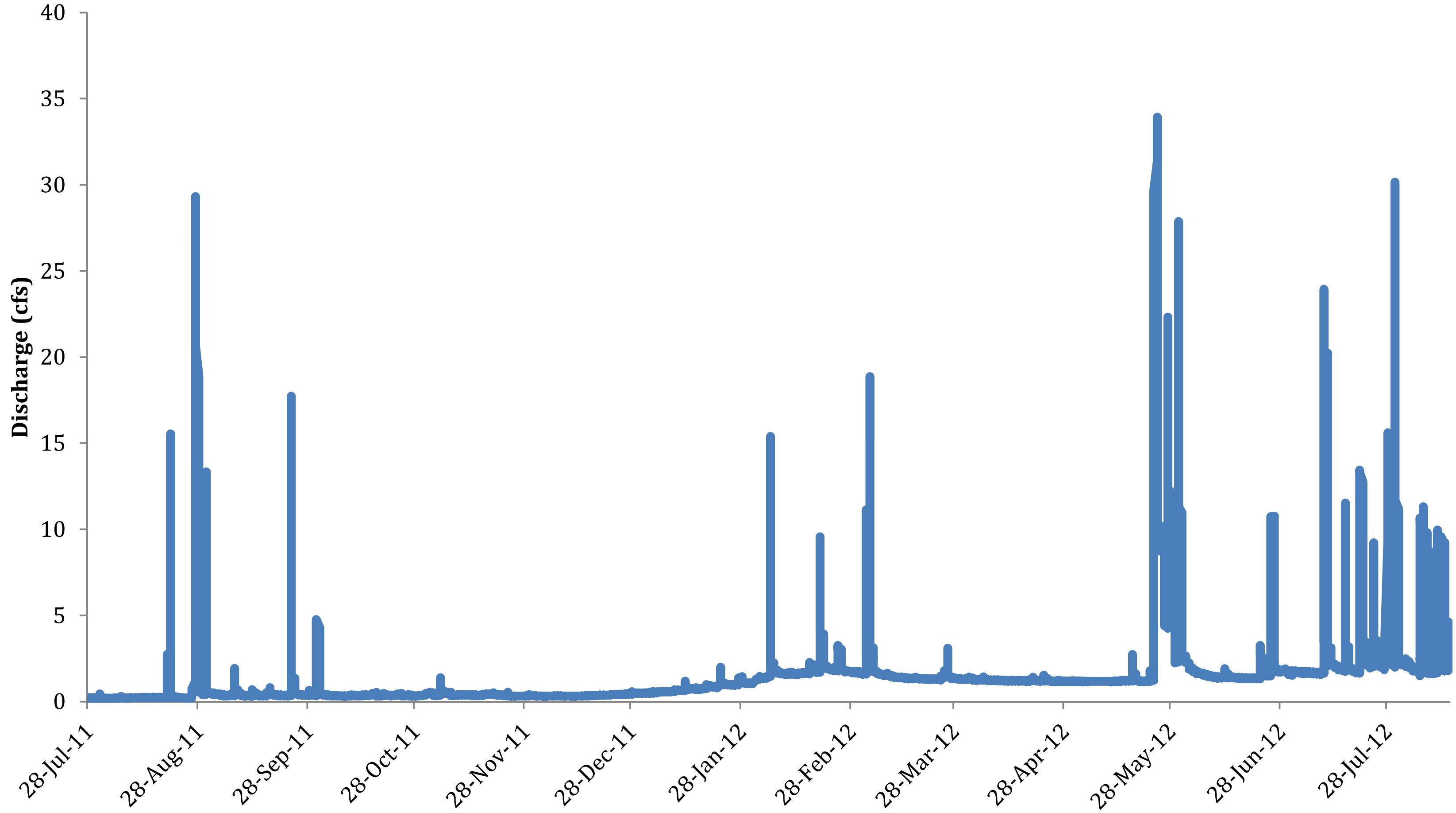




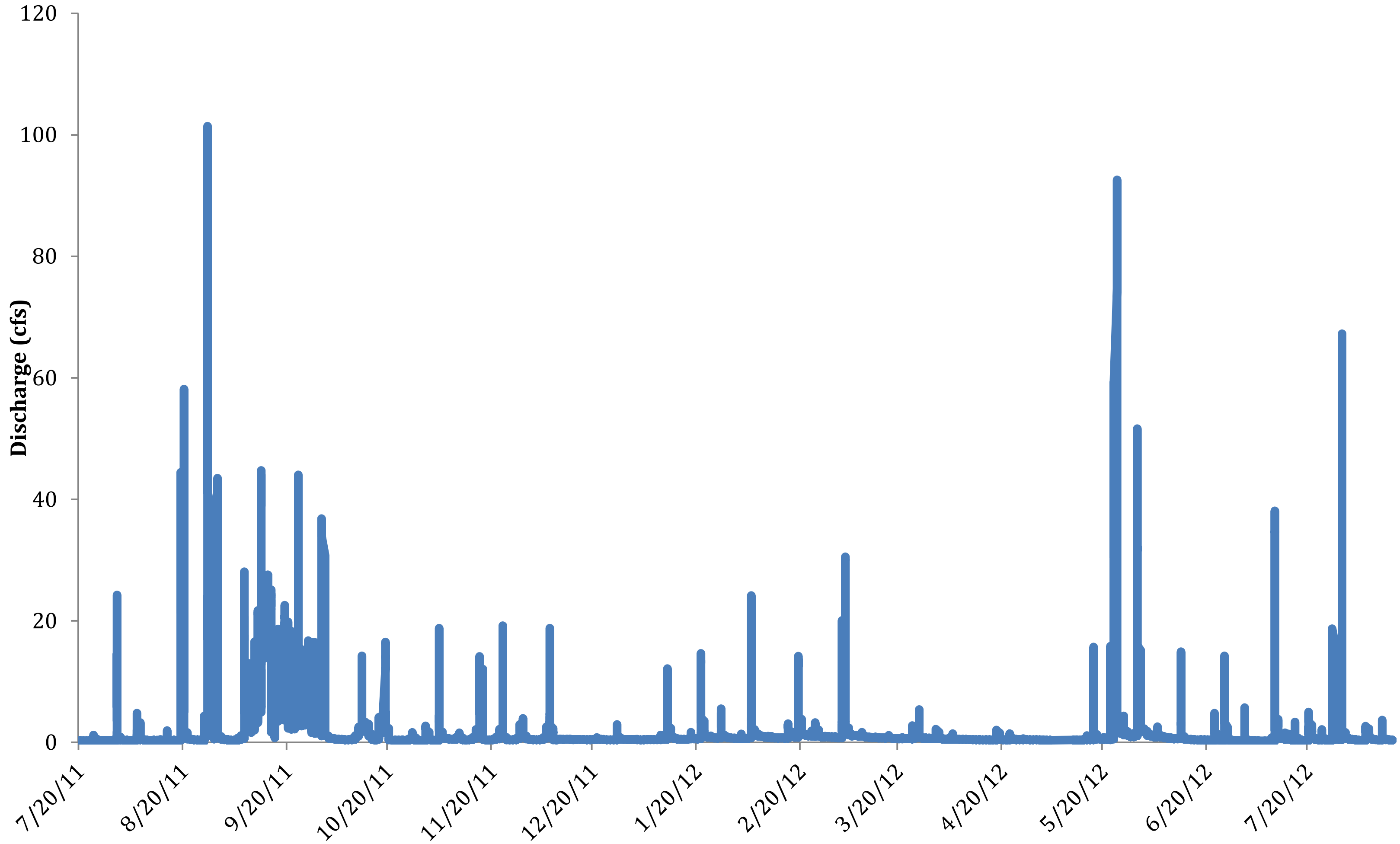
FT-2 Stream Discharge v Time
Discharge (cfs)
100.00
90.00
80.00
70.00
60.00
50.00
40.00
30.00
20.00
10.00
0.00
20-Jul-11
20-Aug-11
20-Sep-11
20-Oct-11
20-Nov-11
20-Dec-11
20-Jan-12
20-Feb-12
20-Mar-12
20-Apr-12
20-May-12
20-Jun-12
20-Jul-12

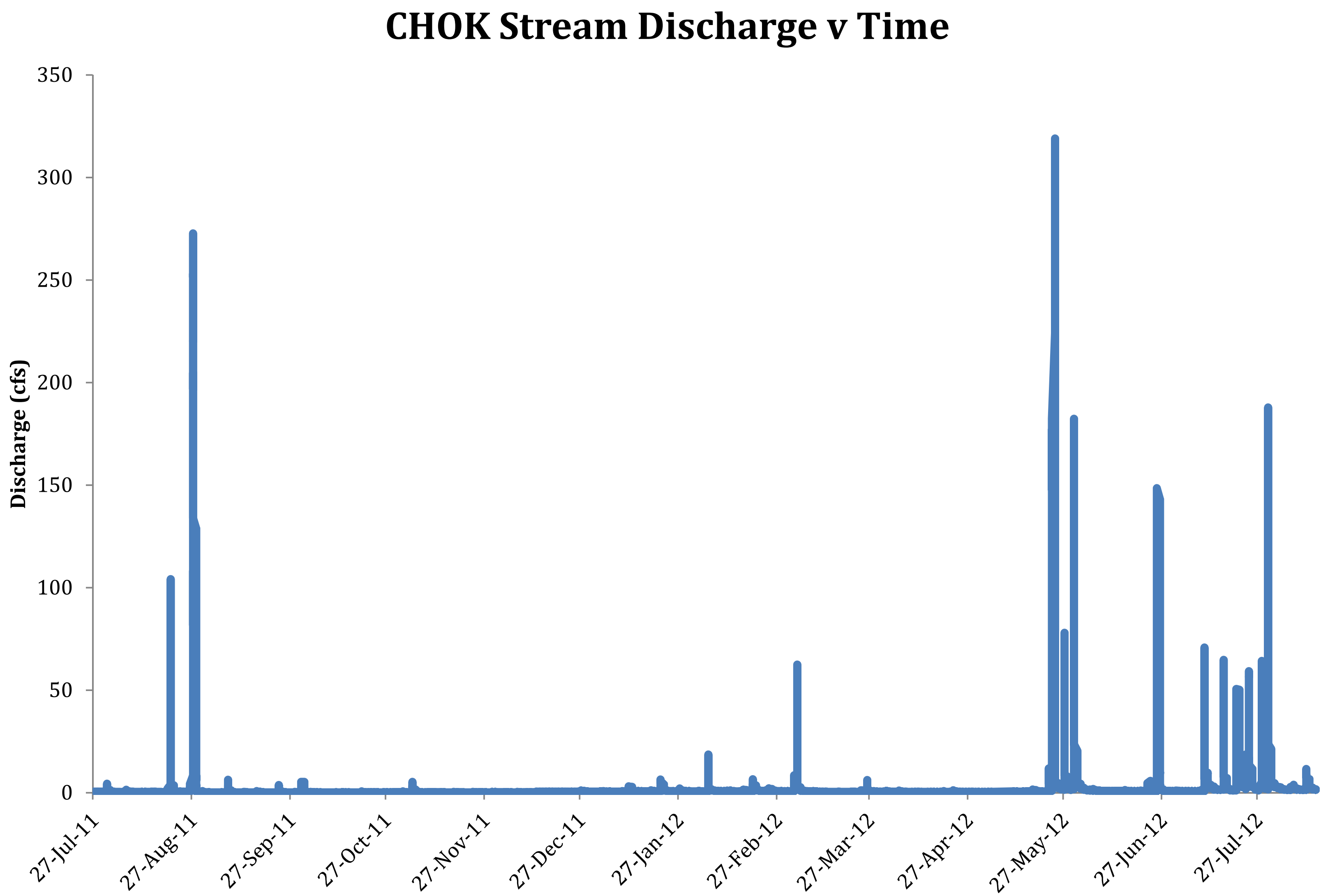

MHB Stream Discharge v Time
Discharge (cfs)
140
120
100
80
60
40
20
0
8/1/11
9/1/11
10/1/11
11/1/11
12/1/11
1/1/12
2/1/12
3/1/12
4/1/12
5/1/12
6/1/12
7/1/12
8/1/12

EP-O Stream Discharge v Time



MILL Stream Discharge v Time



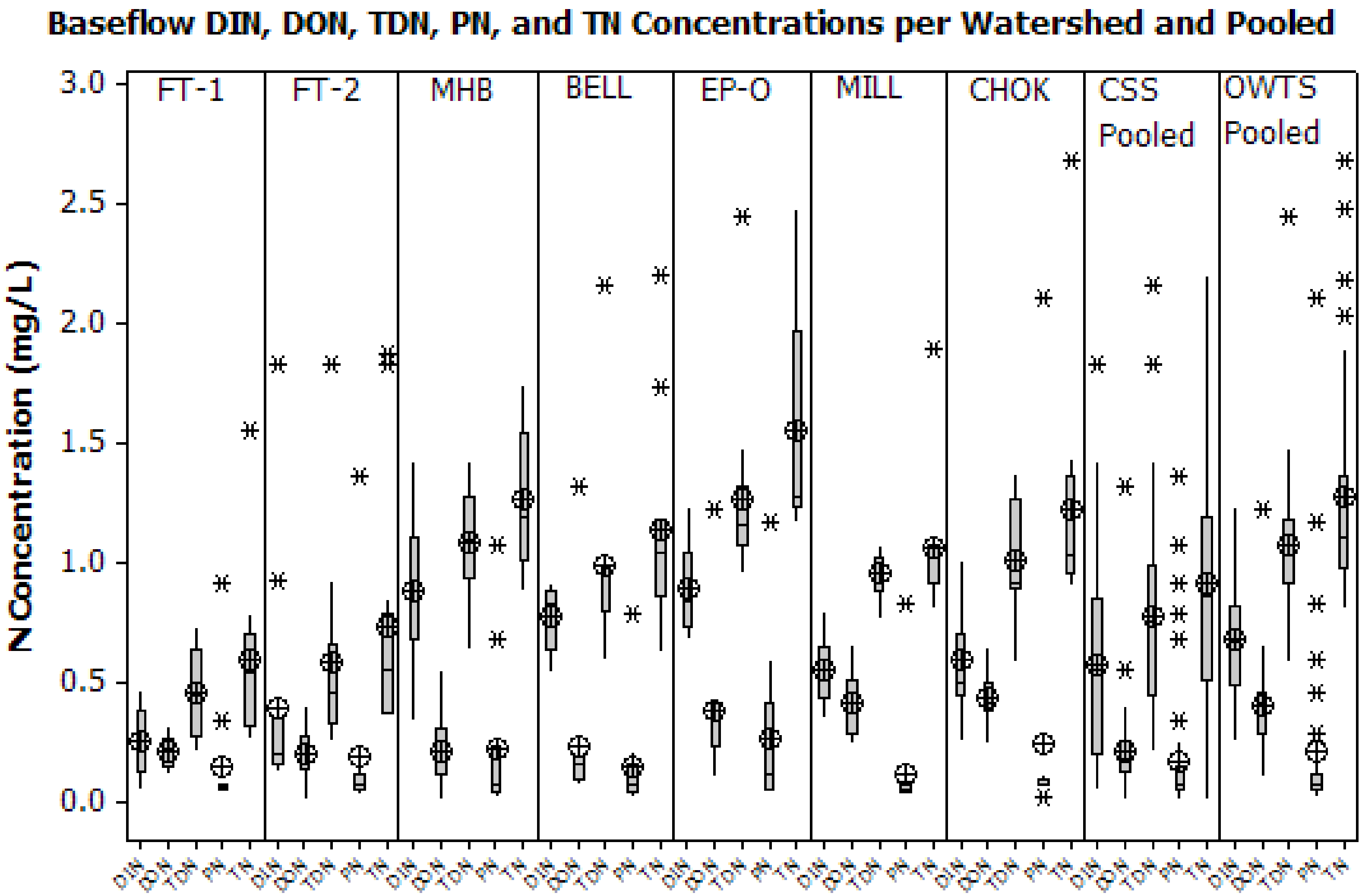

CHOK Stream Discharge v Time
Discharge (cfs)
350
300
250
200
150
100
50
0
27-Jul-11
27-Aug-11
27-Sep-11
27-Oct-11
27-Nov-11
27-Dec-11
27-Jan-12
27-Feb-12
27-Mar-12
27-Apr-12
27-May-12
27-Jun-12
27-Jul-12

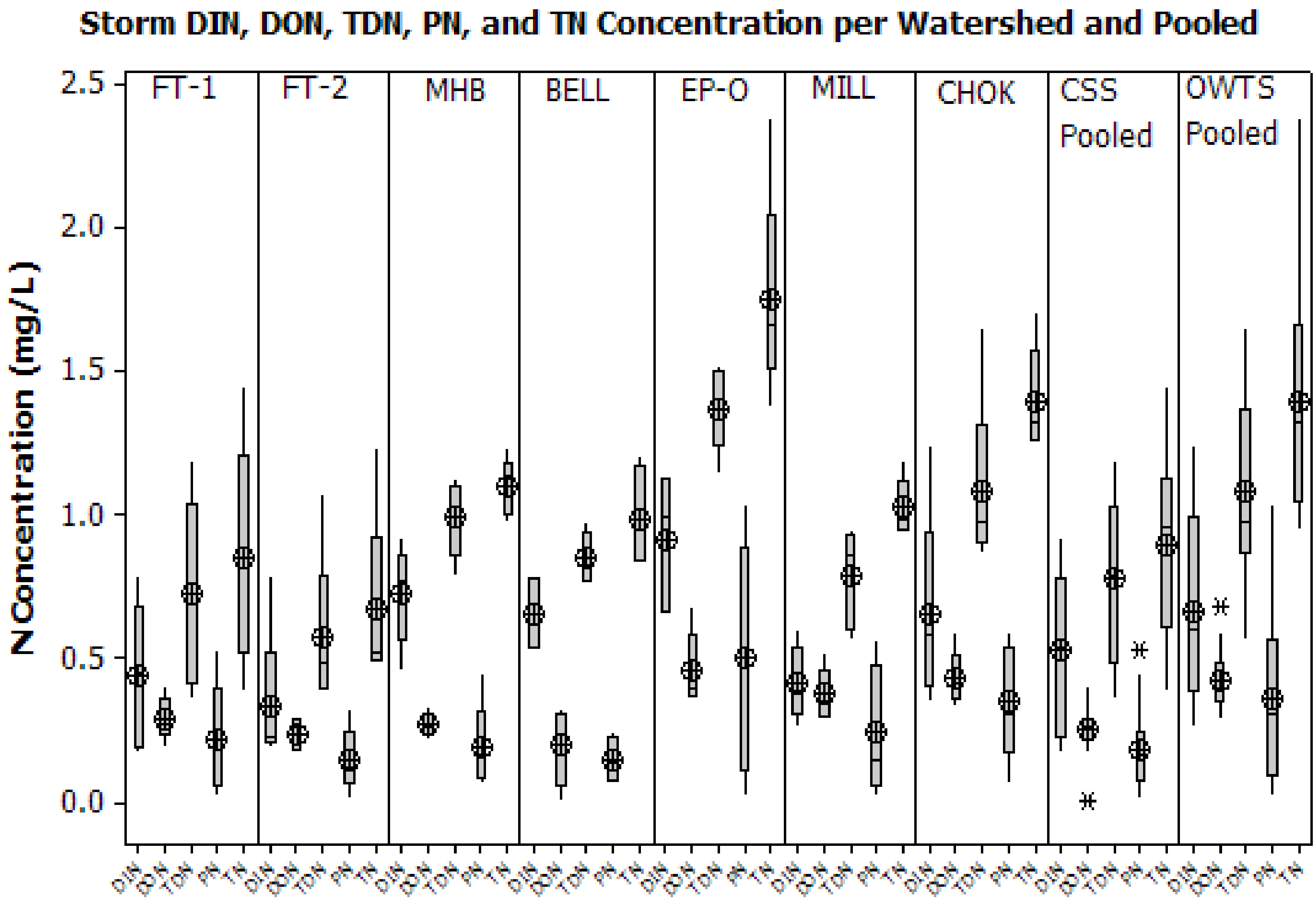
APPENDIX P: EASTERN PINES AND FIRETOWER WATERSHED MONTHLY PRECIPITATION SUMMARY

Precipitation data is summarized by summing up the total daily rainfall from the Eastern Pines and Firetower watersheds. Rain gauges collect data at midnight for each day and gauging began on August 17, 2011 and ended August 13, 2012.

	Eastern Pines Watershed	**Firetower Watershed**
Month	**Total Precipitation (cm)**	**Total Precipitation (cm)**
Aug-11	37.5	35.4
Sep-11	9.91	13.0
Oct-11	4.90	4.90
Nov-11	7.14	7.14
Dec-11	2.03	2.29
Jan-12	8.28	7.47
Feb-12	8.94	8.53
Mar-12	10.2	11.0
Apr-12	4.50	4.70
May-12	16.0	18.3
Jun-12	5.54	5.54
Jul-12	19.0	19.0
Aug-12	8.59	8.59
Total Annual:	143	146
Average Monthly:	11.0 (± 9.20)	11.2 (± 8.83)

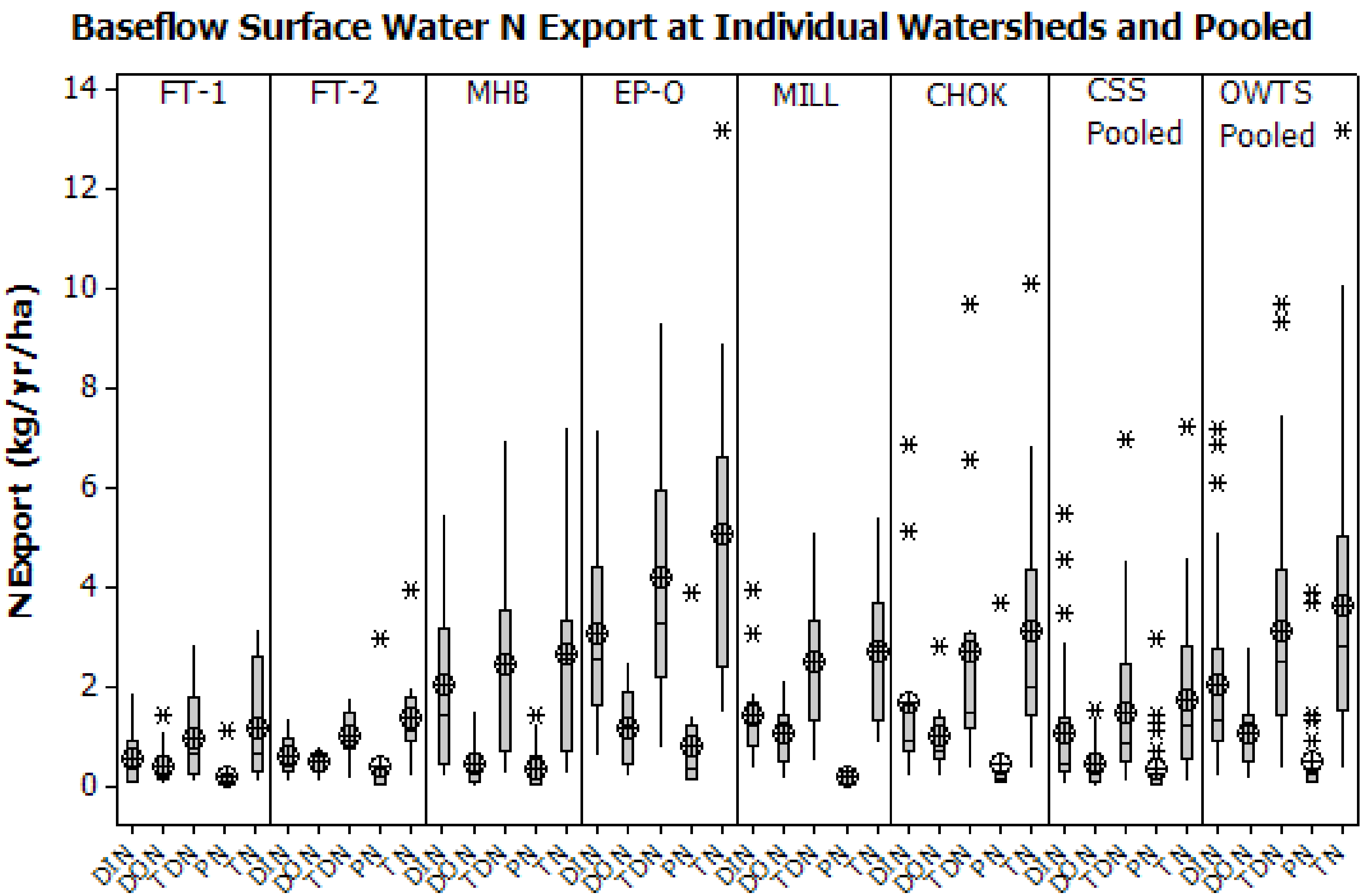
APPENDIX Q: BASEFLOW AND STORM SURFACE WATER N CONCENTRATIONS PER WATERSHED AND POOLED

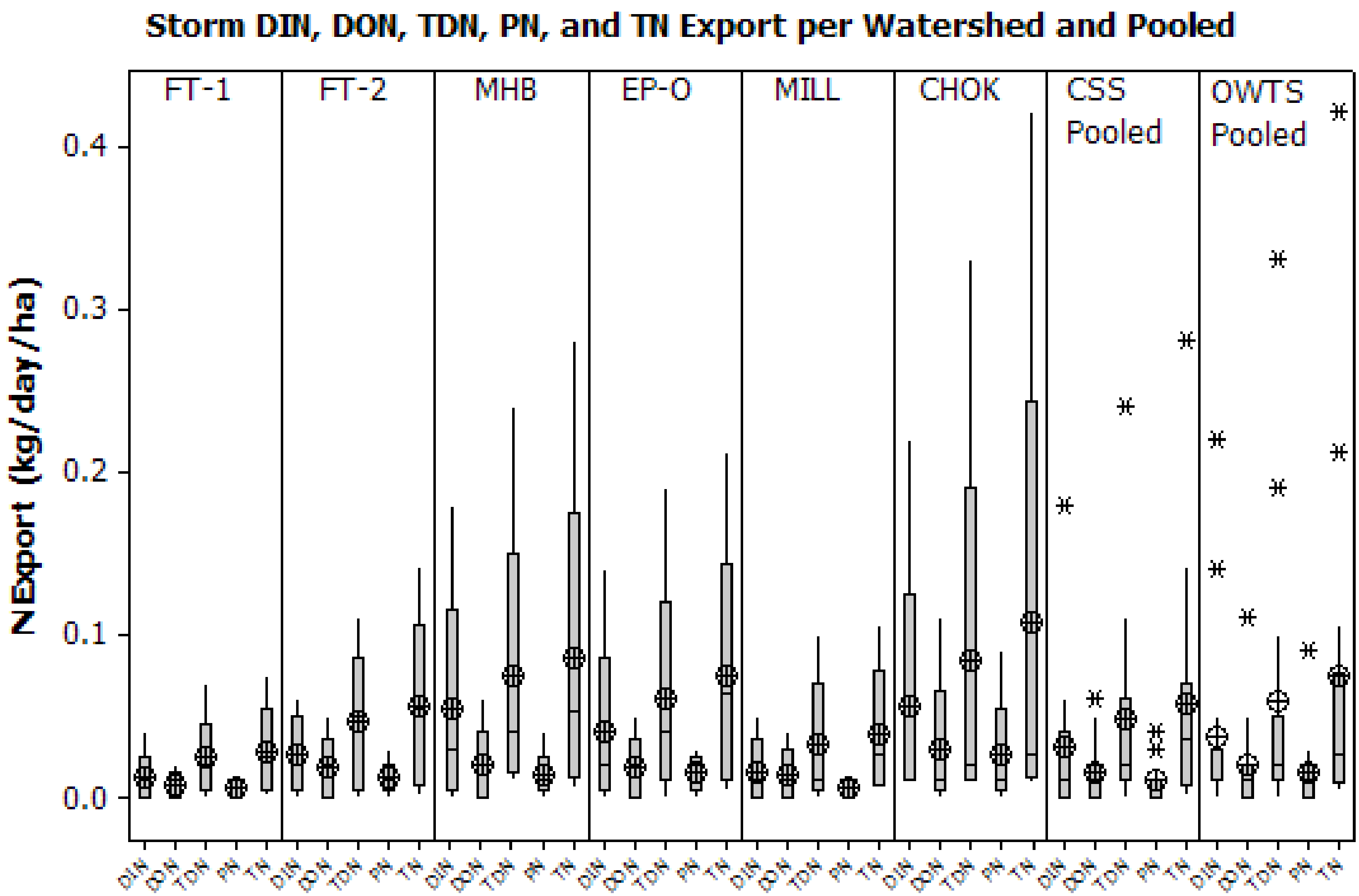




Storm DIN, DON, TDN, PN, and TN Concentration per Watershed and Pooled
N Concentration (mg/L)
2.5
2.0
1.5
1.0
0.5
0.0
FT-1
FT-2
MHB
BELL
EP-O
MILL
CHOK
CSS Pooled
OWTS Pooled
DIN DON TDN PN TN

APPENDIX R: BASEFLOW AND STORM SURFACE WATER N EXPORT PER WATERSHED AND POOLED





Storm DIN, DON, TDN, PN, and TN Export per Watershed and Pooled
FT-1
FT-2
MHB
EP-O
MILL
CHOK
CSS Pooled
OWTS Pooled
NExport (kg/day/ha)
0.0
0.1
0.2
0.3
0.4
DIN
DON
TDN
PN
TN

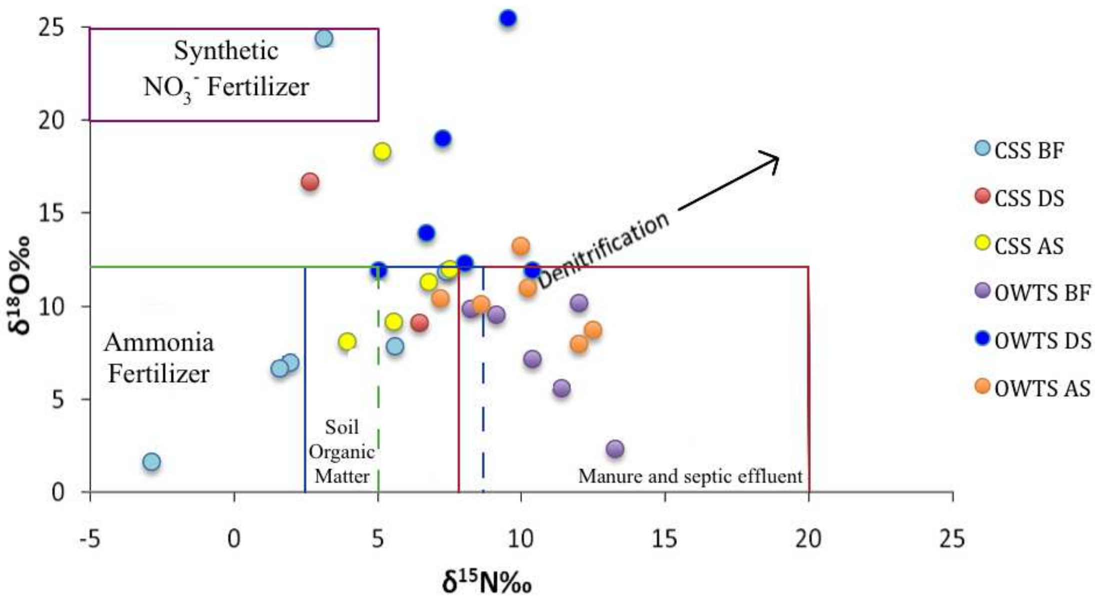
APPENDIX S: SURFACE WATER N ISOTOPE DATA

Table S1 shows the raw data for each surface water isotopic monitoring event. Figure S1 shows the data plotted as compared to Kendall and McDonnell (1998) suggested N sources based on δ^{15}N and δ^{18}Ovalues. Data came from two before, during, and after storm events from November 5-7, 2011 and May 9-10, 2012. BS= before storm, DS= during storm, and AS= after storm.

Table S1. Raw data for surface water δ^{15}N and δ^{18}O values. The data was collected during pre-storm, during storm, and post-storm conditions in November 2011 and May 2012.

Site	**δ^{15}N vs. Air**	**δ^{18}O vs. V-SMOW**	**δ^{15}N vs. Air**	**δ^{18}O vs. V-SMOW**
	Nov-11		May-12	
FT-1-B	5.70	7.30	1.72	6.18
FT-1-D	-0.70	39.19	-2.22	34.52
FT-1-A	5.69	8.57	No data	
FT-2-B	2.10	6.48	-2.71	1.48
FT-2-D	0.19	32.40	-2.43	26.75
FT-2-A	4.08	7.54	5.26	17.15
MHB-B	7.50	11.08	3.26	22.84
MHB-D	2.78	15.62	6.56	8.48
MHB-A	7.61	11.17	6.87	10.57
CSS-B	**5.10**	**8.29**	**0.76**	**10.17**
CSS-D	**0.76**	**29.07**	**0.64**	**23.25**
CSS-A	**5.79**	**9.09**	**6.06**	**13.86**
EP-O-B	12.06	9.50	11.48	5.18
EP-O-D	8.13	11.50	10.45	11.15
EP-O-A	10.30	10.26	7.28	9.74
MILL-B	8.35	9.21	10.47	6.70
MILL-D	5.15	11.11	7.35	17.76
MILL-A	8.69	9.42	12.57	8.14
CHOK-B	9.23	8.90	13.31	2.15
CHOK-D	6.79	13.01	9.63	23.82
CHOK-A	10.06	12.37	12.07	7.45
OWTS-B	**9.88**	**9.20**	**11.75**	**4.68**
OWTS-D	**6.69**	**11.87**	**9.14**	**17.58**
OWTS-A	**9.68**	**10.68**	**10.64**	**8.44**

Figure S1. Surface water δ^{15}N and δ^{18}O values collected from November 5-7, 2011 and May 9-10, 2012 as compared to Kendall and McDonnell's (1998) suggested N sources. OWTS drainfield average δ^{15}N was +16.3 ± 8.75‰, while CSS groundwater δ^{15}N was +10.4 ± 5.98‰.


Synthetic
NO_3^- Fertilizer
Ammonia
Fertilizer
Soil
Organic
Matter
Manure and septic effluent
Denitrification
CSS BF
CSS DS
CSS AS
OWTS BF
OWTS DS
OWTS AS
δ18O‰
δ15N‰
25
20
15
10
5
0
-5
0
5
10
15
20
25