A GEOGRAPHICAL APPROACH TO TRACKING *ESCHERICHIA COLI* AND NUTRIENTS IN A TEXAS COASTAL PLAINS WATERSHED

A Thesis

by

CARA HARCLERODE

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

December 2009

Major Subject: Agronomy

A GEOGRAPHICAL APPROACH TO TRACKING *ESCHERICHIA COLI* AND NUTRIENTS IN A TEXAS COASTAL PLAINS WATERSHED

A Thesis

by

CARA HARCLERODE

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Approved by:

Chair of Committee,	Jacqueline Aitkenhead-Peterson
Committee Members,	Terry Gentry
	B. L. Harris
Head of Department,	David Baltensperger

December 2009

Major Subject: Agronomy

ABSTRACT

A Geographical Approach to Tracking *Escherichia coli* and Nutrients in a Texas Coastal Plains Watershed. (December 2009)

Cara Harclerode, B.S.; B.S., LeTourneau University

Chair of Advisory Committee: Dr. Jacqueline Aitkenhead-Peterson

Carters Creek in Brazos County, Texas, like many surface water reaches in the Texas Gulf Coast region, has been identified for bacteria and nutrient impairment on the Texas Commission on Environmental Quality (TCEQ) 303(d) List. Carters Creek drains a rapidly urbanizing watershed and has been found to carry high concentrations of dissolved organic carbon (DOC), nitrate, phosphate and sodium. These constituents have a severe impact on the creek's capacity for healthy aquatic life and increase the potential for eutrophication downstream. The creek has also had chronic high *Escherichia coli* counts, making the creek unsuitable for contact recreation according to the accepted standard for surface water quality, which is a geometric mean of 126 CFU per 100 ml. In this study, grab samples were taken twice monthly from fifteen sites on Carters Creek and its subcatchments from July 2007 to June 2008. The samples were analyzed for *E. coli*, DOC, total N, NO_3-N, NH_4-N, Na^+, K^+, Mg^{2+}, Ca^{2+}, F^-, Cl^-, Br^-, NO_2^-, SO_4^{2-} and PO_4^{-3}. Mean annual DOC concentrations varied from 24.8 mg/L in Carter at Boonville Road to 55.5 mg/L in Wolfpen Creek; sodium varied from 33 mg/L in Carter at Old Reliance Road to 200 mg/L, also in Wolfpen Creek. Burton 4, the subcatchment with the highest geometric mean for *E. coli* with 2547 CFU/100 mL, was also sampled with greater geographical intensity for *E. coli* and optical brightener fluorescence at 445 nm to identify any leaking sewer pipes, but no evidence of defective pipes was found. During both the spring

season and annual high flow (storm events), *E. coli* counts were positively correlated with total urban land use, probably caused by storm runoff carrying residues from impervious surfaces into the stream. High flow *E. coli* also had a negative relationship with potassium and a positive relationship with calcium, possibly suggesting a bioflocculation effect. Sites downstream of wastewater treatment plants (WWTPs) showed higher nitrate, phosphate, sodium, potassium, chloride and fluoride than other urban subcatchments. Creeks with golf courses carried more phosphate, sodium and fluoride than subcatchments without golf courses or WWTPs.

DEDICATION

To the One in whom I can do all things, and without whom I can do nothing

And to my grandparents, who always told me and my sister not to swim

in the catfish pond, but never seemed to mind when we did it anyway

ACKNOWLEDGEMENTS

I gratefully acknowledge the IMC Fertilizer Corporation for funding my research, as well as the National Garden Club Scholarship Fund and the TWRI Mills Scholarship. Texas AgriLife Research supported the sample collection, laboratory equipment and supplies.

I would like to thank my committee members, Dr. Aitkenhead-Peterson, Dr. Gentry and Dr. Harris for their guidance, wisdom and patience throughout the course of this research.

Acknowledgement also goes to the City of Bryan and the City of College Station for allowing me to use their watershed data and current zoning data. Thanks to William Shaw and Miriam Olivarez for their much-needed and useful help with the GIS analysis.

I owe many thanks to Nurun Nahar, Heidi Mjelde, Nina Stanley, Meredith Steele, Leon Holgate, Sarah Turner and Elizabeth Evetts for their help in the lab and in field sampling.

And finally, thanks to my family and friends who always knew that I could do it even when I didn't and kept encouraging me to "just get it done". Thanks to my father, who never missed a chance to teach me math and science, and to my mother, who helped me see every stream as an opportunity.

TABLE OF CONTENTS

Page

ABSTRACT iii

DEDICATION v

ACKNOWLEDGEMENTS vi

TABLE OF CONTENTS vii

LIST OF FIGURES x

LIST OF TABLES xiii

1. INTRODUCTION 1

1.1 Contaminant Sources 1
1.2 Wastewater Treatment 2
1.3 Urbanization 5
1.4 *Escherichia coli* 6
1.5 Dissolved Organic Carbon 9
1.6 Nitrogen 11
1.7 Phosphorus 12
1.8 Cations 13
1.9 Anions 14
1.10 Study Objectives 16

2. MATERIALS AND METHODS 17

2.1 Soil and Geology 17
2.2 Sampling Sites 17
2.3 Monitoring/Observation 21
2.4 Fluorometry 22
2.5 Geographical Analysis 23
2.6 Statistical Analysis 24

3. RESULTS 26

3.1 Stream Water Constituents 26
3.1.1 *E. coli* 26
3.1.1.1 Low Flow Conditions 26
3.1.1.2 High Flow Conditions 26
3.1.1.3 All Flow Conditions 28

Page

3.1.1.4 Seasonal Variation 28
3.1.2 Dissolved Organic Carbon 29
3.1.2.1 Low Flow Conditions 29
3.1.2.2 High Flow Conditions 30
3.1.2.3 All Flow Conditions 30
3.1.2.4 Seasonal Variation 32
3.1.3 Nitrogen Species: Nitrate, Nitrite, Ammonium and Dissolved Organic Nitrogen 32
3.1.3.1 Low Flow Conditions 32
3.1.3.2 High Flow Conditions 35
3.1.3.3 All Flow Conditions 35
3.1.3.4 Relative Proportion of N Species 39
3.1.3.5 Seasonal Variation 39
3.1.4 Orthophosphate 41
3.1.4.1 Low Flow Conditions 41
3.1.4.2 High Flow Conditions 42
3.1.4.3 All Flow Conditions 42
3.1.4.4 Seasonal Variation 43
3.1.5 Cations: Sodium, Potassium, Magnesium and Calcium 44
3.1.5.1 Low Flow Conditions 45
3.1.5.2 High Flow Conditions 48
3.1.5.3 All Flow Conditions 51
3.1.5.4 Seasonal Variation 52
3.1.6 Anions: Chloride, Fluoride, Bromide and Sulfate 55
3.1.6.1 Low Flow Conditions 55
3.1.6.2 High Flow Conditions 59
3.1.6.3 All Flow Conditions 61
3.1.6.4 Seasonal Variation 64
3.2 Geographical Distribution of *E. coli* in Carters Creek 65
3.2.1 *E. coli* Accumulation with River Miles 65
3.2.2 Fluorescence in Burton Creek 66
3.3 Correlations among *E. coli* and Nutrients without WWTPs 69
3.3.1 *E. coli* 69
3.3.2 Dissolved Organic Carbon 69
3.3.3 Nitrogen Species: Nitrate, Nitrite, Ammonium and Dissolved Organic Nitrogen 69
3.3.4 Orthophosphate 70
3.4 Correlations among Stream Water Constituents and Land Cover 72
3.4.1 *E. coli* 72
3.4.2 Dissolved Organic Carbon 72
3.4.3 Nitrogen Species: Nitrate, Nitrite, Ammonium and Dissolved Organic Nitrogen 73
3.4.4 Orthophosphate 73
3.4.5 Cations: Sodium, Potassium, Magnesium and Calcium 74
3.4.6 Anions: Chloride, Fluoride, Bromide and Sulfate 74

Page

4. DISCUSSION 76

4.1 *E. coli* 76
4.2 Dissolved Organic Carbon 79
4.3 Nitrogen Species: Nitrate, Nitrite, Ammonium and Dissolved Organic Nitrogen 81
4.4 Orthophosphate 82
4.5 Cations and Anions 85

5. CONCLUSIONS 88

REFERENCES 90

APPENDIX I 98

APPENDIX II 107

APPENDIX III 116

APPENDIX IV 125

VITA 126

LIST OF FIGURES

FIGURE		Page
1	Map of sampling sites on Carters Creek subcatchments	20
2	Rain events and sampling dates over the one-year sampling period	25
3	Base-10 logarithm transformations of annual *E. coli* counts during low flow conditions	27
4	Seasonal dissolved organic carbon	32
5	Mean annual nitrate-N concentrations in low flow conditions	33
6	Mean annual nitrite concentrations during low flow conditions	34
7	Mean annual nitrate-N for all flow	37
8	Mean annual total dissolved nitrogen (TDN) concentrations during all flow conditions	37
9	Percentage of total N concentrations in the form of DON	40
10	Seasonal nitrate-N	40
11	Mean annual orthophosphate concentrations during low flow	41
12	Mean orthophosphate concentrations during high flow conditions	42
13	Mean annual orthophosphate concentrations for all flow	43
14	Seasonal orthophosphate	44
15	Mean annual sodium concentrations at low flow	46
16	Mean annual potassium concentrations at low flow	46
17	Mean annual magnesium concentrations at low flow	47
18	Mean annual sodium concentrations at high flow	49
19	Mean annual potassium concentrations at high flow	49

FIGURE Page

20 Mean annual magnesium concentrations at high flow. 50

21 Mean annual sodium concentrations for all flow. 52

22 Mean annual potassium concentrations for all flow.. 53

23 Mean annual magnesium concentrations for all flow 53

24 Mean annual chloride concentrations at low flow.. 56

25 Mean annual fluoride concentrations at low flow 57

26 Mean annual bromide concentrations at low flow 58

27 Mean annual sulfate concentrations at low flow 58

28 Mean annual chloride concentrations during
high flow conditions.. 60

29 Mean annual bromide concentrations during
high flow conditions.. 60

30 Mean annual sulfate concentrations during
high flow conditions.. 61

31 Mean annual chloride concentrations during
all flow conditions... 62

32 Mean annual fluoride concentrations during
all flow conditions... 63

33 Mean annual bromide concentrations during
all flow conditions... 63

34 Mean annual sulfate concentrations during
all flow conditions... 64

35 *E. coli* with river miles upstream of Carter 5,
during dry conditions. 67

36 *E. coli* with river miles upstream of Carter 5,
during wet conditions.. 67

37 *E. coli* logarithmic transformations upstream of Burton 2,
during dry conditions 68

FIGURE Page

38 Fluorescence at 445 nm upstream of Burton 2, during dry conditions. 68

39 Seasonal linear regression of mean DOC vs. SAR 70

40 Seasonal linear regression of mean orthophosphate vs. fluoride 71

41 Seasonal linear regression of mean orthophosphate vs. SAR 71

42 Seasonal linear regression of mean *E. coli* counts 73

43 Seasonal linear regressions of mean magnesium concentrations 75

44 Seasonal regressions of mean calcium concentrations 75

LIST OF TABLES

TABLE		Page
1	Coordinates of sample locations and percent land use in the individual watersheds	19
2	Annual geometric means for *E. coli* in CFU/100 mL during low flow	27
3	Annual geometric means for *E. coli* in CFU/100 mL during high flow	28
4	Annual geometric means for *E. coli* in CFU/100 mL during all flow	29
5	Mean annual concentrations of dissolved organic carbon during low flow conditions	30
6	Mean annual dissolved organic carbon during high flow	31
7	Mean annual dissolved organic carbon during all flow	31
8	Mean annual ammonium-N concentrations during low flow	34
9	Mean annual dissolved organic nitrogen concentrations during low flow	35
10	Mean annual nitrite concentrations during high flow	36
11	Mean annual ammonium-N concentrations during all flow	36
12	Mean annual nitrite concentrations during all flow	38
13	Mean annual dissolved organic nitrogen concentrations during all flow	38
14	Mean annual calcium concentrations during low flow	47
15	Mean annual calcium concentrations during high flow	50
16	Mean annual calcium concentrations for all flow	54

TABLE Page

17 Mean annual fluoride concentration during high flow. 59

18 A comparison with other water quality studies conducted
in various parts of the US 87

1. INTRODUCTION

Water quality constituents interact in complex ways because of the many factors operating in the watershed at any given time. Some of these factors are temporally stable, but some can change instantaneously, providing a constantly shifting in-stream environment. Before analyzing water quality parameters and their interactions, it is important to understand the context of the surface water body in terms of contaminant sources, land use and the functions of the parameters themselves in a watershed.

1.1 Contaminant Sources

Surface water contamination sources are divided into two broad categories: point sources and nonpoint sources. The easier category to identify and mitigate is the point source, in which a pipe directly discharges effluent into a surface water body at a single location. Urban-industrial plants, for example, use water for a variety of functions, such as steam-powered energy, cooling, washing and diluting wastes (Aziz et al., 1998). If the water is contaminated, it is then treated to comply with a discharge permit for water quality and released back into natural waters. Waste water treatment plants (WWTPs) centralize municipal sewage and treat it to minimize particulates, fecal pathogens and dissolved nutrients that can cause toxicity to aquatic life at high concentrations. If these usually limiting nutrients are no longer limiting, they stimulate rapid growth of primary aquatic organisms. When these organisms eventually die they consume more dissolved oxygen during their decomposition than the water body can supply (Al Bakri et al., 2008). Point source discharges are often a relatively constant source of contamination, as the industrial plant or WWTP usually operate at a steady capacity (Cotman et al., 2008). Nonpoint

---

This thesis follows the style of the Soil Science Society of America Journal.

sources, on the other hand, contribute nutrients and other constituents to surface waters diffusely over a large area, sometimes without immediate or obvious signs of contamination. Common nonpoint sources include agricultural and lawn fertilizer leaching into groundwater, topsoil erosion, irrigation water signatures, farm animal wastes, wildlife, septic systems and runoff from impervious surfaces. These sources generally have a varying influence based on frequency and intensity of precipitation events (Cotman et al., 2008). Two notable exceptions to this trend are the impacts of irrigation water seen more during dry weather and the contribution of septic systems, which yield a steady state nutrient enrichment and also have more of an impact on water quality during low flow when dilution effects are minimal (Clinton and Vose, 2006).

1.2 Wastewater Treatment

Of all the sources listed above, wastewater treatment plant effluent has a great impact on stream water quality (Petersen et al., 2006; Carey and Migliaccio, 2009). Wastewater treatment is imperative for maintaining clean cities, preserving healthy surface waters and lowering the incidence of devastating water-borne diseases. Since the Clean Water Act of 1972, a growing percentage of total wastewater produced is cleaned and clarified at municipal wastewater treatment plants (WWTPs), making positive impacts on human and aquatic health (Carey and Migliaccio, 2009). During the wastewater treatment process, primary treatment screens out large debris and solids, then secondary treatment greatly reduces the organic carbon content, biochemical oxygen demand and other nutrients by aerobic microbial respiration (Sanin et al., 2006). Tertiary treatment is optional and reduces N, P, and targeted heavy metals through several different methods or combinations of methods, depending on the unique characteristics of the wastewater and its receiving body. After that, the water is sent through a disinfection step to reduce the pathogen load to the natural receiving waters.

For the typical municipal waste water treatment plant employing tertiary treatment, the parameters of greatest interest are phosphate, nitrogen and to a certain extent pathogens, though greater microbial reduction occurs in the following disinfection step. A few common approaches to reducing phosphate in particular include chemical precipitation, biological phosphate reduction or constructed wetlands (Oehmen et al., 2007, Carey and Migliaccio, 2009). Denitrification and biological P removal can be achieved in a single step, as the bacteria can utilize nitrate as an electron acceptor in anaerobic conditions. Although nitrate is 40% less efficient than oxygen for P removal, the input costs for oxygen are reduced, as well as the amount of sludge to be disposed (Oehmen et al., 2007). A phosphate removal rate of 70% was observed in a full-scale wastewater treatment plant with a single-sludge tertiary treatment, while approximately half of that removal was accomplished through denitrification processes (Kuba et al., 1997). In a two-sludge system, nitrogen was removed at an efficiency of 88% and phosphate at 99%, though with a two-sludge system the processing cost and complexity also increase (Kuba et al., 1996). A pilot study of slow sand filtration (with a flow rate of 2 L min^{-1}) as a tertiary treatment reduced bacteria plate counts 88-93%, nitrate by 17-30% and phosphate as much as 84% (Farooq et al., 1994). Chemical precipitation of phosphorus by alum or iron salts followed by sand filtration also produces satisfactory results, but does not reduce effluent nitrogen (Vigneswaran et al., 1997). These chemical treatments also have higher operating costs because of the quantities of reagents used, and produce larger amounts of sludge that must be disposed of (Kuba et al., 1997).

The final step in waste water treatment is disinfection, once the effluent has been through sufficient nutrient reduction. Commercial disinfection agents include chlorine, hypochlorite, ozone, peracetic acid and UV radiation (Liberti et al., 2002; Gehr et al., 2003). If disinfection is accomplished through chlorination, high organic matter or ammonium present in the effluent can react to form disinfection byproducts (DBPs) that do not have the same disinfecting properties, requiring a larger amount of disinfection agent to have the same effect. Chlorine disinfection combined with elevated organic carbon concentrations produce trihalomethanes and haloacetic acids (Chow et al., 2008), while an elevated ammonium or organic nitrogen concentrations will produce chloramines and organochloramines (Yu et al., 2009). In addition, after the disinfection process is complete the residual chlorine must be removed to prevent injury to aquatic life in the receiving body (Yu et al., 2009). At high UV intensities, the photodecomposition of hydrocarbons and halohydrocarbons (as in the case of many pesticides) may produce more harmful byproducts; however, at lower intensities, UV radiation has a lower incidence of DBPs than ozone or chlorine while remaining an effective disinfection method. Liberti et al. (2002) found that a sequence of sand filtration, additional clarification and then UV disinfection was the most effective method in meeting a total coliform bacteria count of 2 CFU/100 mL and reducing protozoan cysts by 60%, a reduction that would render the effluent suitable for unrestricted irrigation water but would not meet the drinking water standard for *Giardia lamblia* (USEPA 1998).

Effluent reuse for agricultural or urban irrigation in water-stressed areas is an attractive alternative to stream discharge, as current standards for irrigation water are less stringent and the nutrient rich effluent lowers the demand for fertilizer on the receiving land (Carey and Migliaccio, 2009). However, the high salt content of secondary-treatment wastewater may have a detrimental effect on vegetation as receiving lands become more saline or sodic over time. In a wastewater-irrigated area near Hyderabad, India, salinity exceeded the salt tolerance of rice in 38-83% of field soils sampled, with electrical conductivity values 6.2-8.5 times higher than soils irrigated with groundwater (Biggs and Jiang, 2008). When a historically sodic soil was leached with highly sodic wastewater in a study by Jalali et al. (2008), there was no appreciable change in soil sodicity or soil properties, but when the wastewater treatment was followed by water with low sodicity (such as rain or groundwater), porosity and hydraulic conductivity rapidly declined. Receiving lands with high traffic, such as golf courses, may also be more sensitive to marginal sodic wastewater than other soils, exhibiting sodic properties well below the accepted threshold (Qian and Mecham, 2005). In Kuwait, where the main source of domestic and agricultural water is desalinization from the Persian Gulf, advanced tertiary treatment to remove salts for reuse is a very appealing alternative at a quarter of the cost, after conventional secondary treatment of the wastewater (Abdel-Jawad et al., 1999).

1.3 Urbanization

Before analyzing water quality parameters of urban streams, it is important to understand stream dynamics and how a stream's urban setting differentiates it from a rural or undisturbed stream. In general, urbanized streams typically have lower groundwater base flow per unit area than rural streams, due to less infiltration and soil storage under impervious surfaces (Rose, 2007). As urban areas develop and percent impervious surface increases, the stream channels are often deepened, widened and concrete-lined to drain the increased overland flow more quickly. However, the diminished habitat eliminates much of the streamside vegetation that provides food, shelter, shade, bank stabilization and natural baffles to slow the velocity and energy of storm flow (Alberti et al., 2007). When storm flow velocity is unhampered, as in a lined, straightened channel, the peak flow of the flood as well as the sediment load, nutrients and bacteria it transports is carried farther downstream (Alberti et al., 2007). Road runoff is often directly channeled by concrete gutters and culverts to the nearest stream channel, bypassing the filtering capacity of the riparian zone. Road intensity is directly correlated with total impervious surface, which in turn affects the quality of urban streams (Alberti et al., 2007). For urban areas of the Texas Gulf Coast and Coastal Plains regions, the slight slope and paucity of groundwater recharge to streams exacerbate the effect of impervious surfaces, producing surface waters dominated by waste water treatment plant (WWTP) effluent during base flow (Petersen et al., 2006; Cotman et al., 2008; Al Bakri et al., 2008). Both base flow and potential storm runoff increases during the fall and winter until March, at the end of the wet season when the exposed urban soil is most saturated (Rose, 2007). Then base flow begins to decrease as the weather becomes drier and summer temperatures stimulate evapotranspiration. During dry summer weather, groundwater contributions are at their lowest, masked by inputs from WWTP effluent and the probable addition of residential irrigation water, car washes, etc. According to a study conducted on two bayous in Houston, Texas, two-thirds to three-fourths of median flow is WWTP effluent, causing effluent quality to directly affect bayou water quality (Petersen et al., 2006). Precipitation runoff (and subsequent overflows and bypasses of sanitary sewers) is the primary nonpoint source (NPS) of pollution into surface waters (Rose, 2007).

1.4 *Escherichia coli*

In 2002, only 19% of US rivers and streams had been evaluated for designated uses, and 39% of this evaluated river mileage had bacterial impairments (Arnone and Walling, 2007). In response to watersheds across the country with fecal bacteria impairments, watershed managers are seeking methods for diagnosing the problems affecting surface waters. A large bacterial load to a stream may come from a number of source vectors, including wildlife, birds, pets, livestock, or humans. Although bird and wildlife contributions cannot be readily changed and in many cases are encouraged in natural riparian areas (Tufford and Marshall, 2002), good planning, research and management of the watershed can reduce contributions from human and domesticated animal waste through the implementation of best management practices, or BMPs

(Dickerson et al., 2007). In Rhode Island urban areas, storm water was sent through a structural BMP separator with a grit chamber that traps suspended solids and debris as well as 39-86% of suspended bacteria before releasing the filtered water to the stream (Zhang and Lulla, 2006). Suburban residential BMPs include leaving clippings on the lawn for mulch, diverting roof runoff onto lawns to minimize runoff and removing pet waste from yards and driveways (Dietz and Clausen, 2004).

Escherichia coli, an easily-culturable, gram-negative coliform bacteria, grows abundantly in the lower intestines of warm-blooded animals and is expelled in feces (Winfield and Groisman, 2003). A presence of human-origin fecal indicator bacteria such as *E. coli* also means that there is a potential for human pathogens as well. Besides pathogenic strains of *E. coli* (e.g., 0157:H7), other water-borne pathogenic bacteria include *Salmonella* spp., *Shigella* spp., *Vibrio cholerae* and *Legionella pneumophilia* (Arnone and Walling, 2007). Reducing the amount of indicator bacteria by eliminating human sources into the water will also reduce these pathogens, and thus reduce the potential for water-borne diseases (Clinton and Vose, 2006).

One emerging challenge to maintaining bacterial standards in surface waters is the issue of regrowth. Given a favorable environment, an individual bacterial cell can regrow and divide several times in a matter of hours, propagating into a colony. In an estuarine microcosm with chlorine-injured *E. coli*, Bolster et al. (2005) observed 2.8-fold to 50-fold increases of culturable cells over a 74 hour period. Although regrowth was greater in microcosms with high DOC and nitrogen concentrations, the cells regrew even in low nutrient solutions. Through disinfection in wastewater treatment plants, the bacteria in effluent are killed, injured or pass through unharmed if shielded by other cells and large molecules such as DOC. ‘Viable but nonculturable’ (VBNC) cells first described by Xu et al. (1982) are bacteria cells that are injured beyond the ability to form colonies when cultured by known methods, but are still alive and take on altered enzymatic and metabolic function. In recent years there has been much debate over whether VBNC cells can resuscitate, or regain the ability to form colonies once the cause of the injury has been removed (Arana et al., 2007).

Weather patterns also play a major role in fluxes of fecal bacteria. Muirhead et al. (2004) postulated that during dry periods, fecal bacteria in a watershed were stored on the land surface and in the stream channel itself, in bottom sediments. These two pools became sources during a rain event, causing a dramatic increase in *E. coli* numbers for the stream at high flow. During this "first flush," the colony forming units (or CFU) per 100 ml of *E. coli* increased by two orders of magnitude compared to base flow values (Muirhead et al., 2004). For the two Houston bayous, fecal indicator bacteria decreased slightly over several days without rain, but bacterial numbers were still elevated four days or more later, exceeding TCEQ standards for contact recreation (Petersen et al., 2006). Subsequent flooding of the stream channel, even to the same flood stage of the stream, yielded a much lower *E. coli* peak because the bacteria have already been flushed out in the recent high flow (Muirhead et al., 2004).

To reduce human bacterial sources most effectively, watershed managers must be able to identify the sources of fecal contamination. There are many methods of bacterial source tracking (BST) that have been developed, but many are time and cost prohibitive for interested small researchers and municipalities (McDonald et al., 2006). One less expensive (and less precise) method detects fluorescing chemicals as a quick, in-field proxy for human-origin fecal bacteria. Optical brighteners used in laundry detergents emit blue light (415-445 nm), making clothes appear whiter and masking gradual fabric yellowing (Hartel et al., 2007). Because most modern sewage systems combine the "gray water" from washing machines, sinks, bathtubs and showers with the "black water" from the toilet into one sewer line, a leak of human fecal material from a breached sewer line will often be associated with a relative spike in fluorescence from optical brightener concentrations (Hartel et al., 2007).

1.5 Dissolved Organic Carbon

In addition to harboring a broad spectrum of bacteria and other microorganisms, surface waters carry dissolved organic matter that has been washed from overland runoff, leached from soil, or discharged from anthropogenic sources. Organic matter is an important contributor to a soil's capacity to retain nutrients that can be made available to plant uptake. Phosphorus and sulfur are also bound up within organic matter, in addition to the base cations that attach to the numerous electronegative or negatively charged sites from carboxylic, phenolic and alcoholic functional groups (Brady and Weil, 2002). This organic material is cleaved into smaller organic molecules by soil microbes and then randomly reassembled into humic and fulvic acids that may become dissolved in the soil solution. Dissolved organic carbon (DOC) is that material which passes through a 0.45 μm filter (Thurman, 1985) and is derived from various organic materials. As microbes degrade the organic matter, nutrients are released into the soil solution or stream, making them available for other organisms (Fellman et al., 2008). Different soils and vegetation will produce a different chemical composition and fraction of biodegradable organic matter. For example, 50-75% of throughfall DOC, or DOC carried to the soil through a leaf canopy, is biodegradable compared to ~10-30% of DOC derived from the forest floor solution (Yano et al., 2000; McDowell et al., 2006). Land use and biome type have been shown to be important for DOC and nutrients in surface waters (Aitkenhead and McDowell, 2000; Aitkenhead-Peterson et al., 2007, 2009; Alexander et al., 2004, 2008). Composition of DOC can change, as Izbicki et al. (2007) observed in a temporal increase in specific UV absorbance at 254 nm, an indication of aromaticity, during the course of a southern California rainy season. DOC fractionation change can also occur within a single rain event; as storm runoff increases, in-stream DOC composition shifts from infiltrated ground water DOC to reflect the higher lignin content of younger plant litter on the soil surface (Vidon and Smith, 2007). Leaf litter with high lignin, aromaticity and C:N ratio, as with coniferous litter, is much more difficult to break down than “softer,” higher-quality litter (Prescott and Preston, 1994). In following DOC movement through soil and into surface waters, understanding hydrology is particularly important (Aitkenhead-Peterson et al., 2005). Though wetland cover and poorly drained soils develop better hydrologic connections to the stream network during wetter-than-normal seasons, connections to other landscape features with lower DOC concentrations and the general dilution factor from the abundance of water complicate DOC predictability. On the other hand, dry conditions lower predictability also by hydrologically isolating wetlands and poorly drained soils from the stream network and limiting water availability for transport (Wilson and Xenopoulos, 2008).

High concentrations of DOC in distribution water sources can have negative impacts on drinking water quality. When DOC concentrations in drinking water sources are high, disinfection chemicals used in water treatment such as chlorine and chloramines can combine with DOC to form carcinogenic trihalomethanes and haloacetic acids (Chow et al., 2008). When the disinfection agents react with DOC instead of the bacterial targets, the disinfectant is rendered ineffective and infectious bacteria can still remain (Crump et al., 2004). In source waters with high DOC content, disinfection agents ozone and ClO_2 break up the large DOC molecules to form smaller, more labile aldehydes and carboxylic acids. These oxygenated functional groups and smaller molecule size make the dissolved organic matter more biodegradable for any remaining bacteria in the distribution system (Swietlik et al., 2004).

1.6 Nitrogen

In the process of microbial mineralization of DOC, various other nutrients such as nitrogen are released into the soil solution and eventually into the stream network. Organic nitrogen is bound up in amide groups in organic matter, in a form much more resistant to leaching than inorganic N forms because the organic molecules are so large (Naidu and Rengasamy, 1993). The nitrogen is retained in the soil until it is mineralized by soil microbes, thus making nitrogen slowly available to plants. If the microbes mineralize more nitrogen than the plant needs immediately or other sources of inorganic nitrogen are added to the soil, the excess can be leached into the groundwater and consequently any nearby surface waters. In-stream nitrate, for example, was found to be significantly correlated to soil texture and drainage capability; a soil with better drainage will allow the soil solution through soil pores more quickly, taking with it more soil nitrate, and producing higher nitrate concentrations downstream (Meynendonckx et al., 2006). River and stream bottom sediment provides ideal conditions for denitrification as nitrate diffuses into the anoxic conditions and higher nutrient content of the stream bed (Mulholland et al., 2008). However, this capacity can be easily overloaded with high nitrate concentrations. Though biological uptake of nitrate increases somewhat with increased concentration, uptake efficiency drops, leading to greater nitrate export downstream (Mulholland et al., 2008). Nitrogen sources in the landscape may include lawn and agricultural crop fertilizer, plant residues, animal wastes, percolation from rural septic systems and wastewater treatment plant effluent. According to Mulholland et al. (2008), in stream nitrate concentrations were significantly higher from streams under agricultural and urban land use than in reference streams. Lewis et al. (2007) found that nitrate, dissolved organic nitrogen (DON) and total dissolved nitrogen (TDN) concentrations were significantly higher downstream of a wastewater treatment plant than upstream. A stream with some raw sewage contamination often carries a high nitrate load as well (Phillips et al., 2007). In extreme cases such as the Bagmati River in Kathmandu Valley, Nepal, low dissolved oxygen and high organic matter from raw sewage caused reduced nitrification, and the predominant nitrogen form was ammonium (Bhatt and McDowell, 2007). Though ammonium is usually identified as the preferred nitrogen form for plant uptake, high in-stream ammonium concentrations can become toxic, particularly in conjunction with hypoxia (Beutel, 2001). In marine environments, nitrogen is the major limiting nutrient, making near-coastal zones particularly susceptible to nitrate-caused eutrophication (Alexander et al., 2008). A particularly large hypoxic zone, just to the south and west of the Mississippi River delta, is attributed to leached nitrate fertilizer from the breadbasket states (Alexander et al., 2008). On a smaller scale, the use of fertilizer on golf courses, residential lawns and urban green spaces can also contribute to nitrate concentrations in urban streams (King et al., 2007). In rural areas even with low density development, the density of septic systems may exceed the capacity of soil microorganisms and chemical denitrification to reduce the inputs of nitrate into the watershed (Cunningham et al., 2009). The US drinking water standard calls for <10 mg L^{-1} nitrate-N (USEPA 2009), but much less nitrate can be enough to cause algal blooms and degradation of the aquatic habitat.

1.7 Phosphorus

Though plentiful in marine environments, phosphorus is a limiting nutrient for freshwater systems. When abundant in lentic surface waters, phosphates cause rapid eutrophication and consequent degradation of freshwater systems. Fitzpatrick et al. (2007) found that in the Muskegon River Watershed in Michigan, USA, nutrient concentrations of nitrogen and particularly phosphorus were more closely correlated with urban land use rather than the often-blamed agricultural use. Also, phosphate loss from agricultural cropland is primarily attached to eroded soil in the particulate form, while phosphate from wastewater is in dissolved form and more mobile (Olli et al., 2008). Unlike nitrogen, phosphorus is negatively correlated with sandy soil textures, which allow for greater infiltration and adsorption into the soil profile and produce less runoff (Meynendonckx et al., 2006). Forested land cover also has a negative relationship with in-stream phosphate concentrations because of better infiltration and a thick cover of plant litter on the forest floor to retard runoff and erosion. In fact, Zampella et al. (2007) found no phosphate correlation with agricultural or urban land use intensity in the sandy, nutrient-poor New Jersey pinelands, reporting that the bulk of phosphate contamination came from wastewater treatment plant effluent. Though point sources, especially municipal wastewater effluents, contribute heavily to the urban footprint, nonpoint sources still contribute significant amounts of P to the watershed. In Portland, Oregon, in less nutrient-poor soil, Sonoda and Yeakley (2007) found that phosphate was correlated with mixed urban land use and multi-family residential areas ($p=0.003$, $p=0.009$, respectively; $R^2=0.739$), while total phosphorus flux was correlated with single-family residential land use ($p=0.019$; $R^2=0.409$). Nonpoint sources of phosphate, including leaky septic systems, agricultural runoff, and impervious surface runoff from urban areas are often more variable and difficult to control. Most strategies addressing these P sources attempt to trap and store the contaminants present in the water, such as in tertiary-treatment lagoons in concentrated animal feeding operations (CAFOs) or wastewater treatment plants (Schussler et al., 2007).

1.8 Cations

In addition to dissolved organic carbon, nitrogen and phosphorus, base cations are also very common in lotic surface waters and higher concentrations are typically observed during base flow. These base cations are namely sodium, potassium, magnesium and calcium. The divalent cations (Ca^{2+} and Mg^{2+}) form ionic "bridges" with two negatively charged clay particles, chelation sites in DOC, or metal hydroxide colloids, aiding in their flocculation and settling (Sanin et al., 2006). On the other hand, monovalent cations such as Na^{+} and K^{+} serve as ionic "caps" to the negative charge, dispersing their counterparts and keeping them suspended (Bourgeois et al., 2004). As monovalent concentrations increase, it requires an increasingly large divalent to monovalent ratio to achieve flocculation (Bourgeois et al., 2004). Lambrakis (2006) describes the salinization of groundwater as the displacing of soil calcium with large quantities of sodium ions, which are commonly associated with monovalent anions such as chloride, bicarbonate, or nitrate. This can occur through saltwater intrusion from the coast, or from sodic irrigation water as in the case of the Carters Creek watershed in south-central Texas (Aitkenhead-Peterson et al., in press). The reverse reaction, groundwater freshening, occurs when calcium ions replace the soil sodium, for example after precipitation dissolves calcium-rich minerals and replenishes groundwater stores (Lambrakis, 2006). In urban areas, sodium, potassium and the anion chloride tend to increase in concert as common electrolytes found in human wastes as well as chemicals used for water treatment (Rose, 2007). Zampella et al. (2007) discovered that at a threshold of 10% altered land use, the water quality of the Mullica River Basin in New Jersey experienced higher calcium and magnesium, probably caused by liming in upland agriculture.

1.9 Anions

As negatively charged counterparts to the cations, anions are more easily leached from the soil and into groundwater because they lack attachment to negatively charged soil particles. The anions fluoride, chloride, bromide and sulfate, also found in lotic waters, are typically indicators of anthropogenic activity when observed in high concentrations. Although nitrate, nitrite and orthophosphate are also anions, they are treated separately because of their roles as primary nutrients and are not conserved like the anions discussed here. The first anion, fluoride, is a common component of groundwater when the groundwater source has high concentrations of sodium, high pH, and high bicarbonate-alkalinity. Silica is also commonly associated with such waters (Hem, 1959). Natural, unpolluted surface waters generally carry fluoride concentrations of 0.3 mg/L or less (Meenakshi and Maheshwari, 2006). Another halide, chloride, is a ubiquitous, highly soluble anion that can be compared to other constituents to yield useful information about the sources and quality of natural water. In shallow groundwater, both bromide and chloride typically occur in low concentrations and are considered conservative in soil solution because they are rarely adsorbed onto soil minerals. This conservative behavior makes them good candidates for tracers in hydrological studies (Davis et al 1998). Also, since chloride behaves conservatively, the ratio of chloride to nitrogen or chloride to phosphorus can be used to differentiate reductions from dilution or from nutrient uptake, assuming no additional chloride inputs (Clinton and Vose, 2006). Chloride concentrations are often viewed as an indicator of the degree of human disturbance in a watershed. Even in rural areas, chloride was found to have a significant linear relationship with impervious surface cover in both summer and winter in New York, reflecting groundwater storage and slow release into streams throughout the year (Cunningham et al., 2009). Other studies have shown increases in both chloride and sulfate with increasing land use intensity (Zampella et al., 2007; Rose, 2007). In the greater Atlanta area, sulfate concentrations were not significantly different between stream channels that conveyed sewer lines and streams that had direct WWTP effluent discharge, suggesting possible leaks in the aging sewer network (Rose, 2007). In contrast, Zampella et al. (2007) found upland agriculture to be the only significant sulfate predictor, highlighting the fact that the same parameter may have different nonpoint sources in different regions.

1.10 Study Objectives

The general objectives of this study were to identify the geographical source of bacterial contamination and to offer a rough characterization of nutrient contaminants for use in remediating impaired surface waters. Although there are many bacterial source tracking (BST) methods that are more species- and source-specific, they are almost always more time-consuming and costly. The approach taken here is not meant as a substitute for these higher resolution methods, but rather a preliminary step to geographically narrow the field of investigation to a smaller scale. From that narrowed field the various BST methods may be used to more precisely identify the source without the time and expense of testing a much broader area.

The specific objectives of this study are as follows: (1) to quantify bacteria and nutrient levels in nested subcatchments of Carters Creek and to identify any relationships between them; (2) to measure fluorescence and *E. coli* in a selected subcatchment; and (3) to determine the relationship between measured bacterial counts and nutrient levels with geographical areas or land use.

Working Hypotheses

H_0: There is no significant difference in *E. coli* counts and nutrients in Carters Creek sub-catchments

H_1: Nutrients N and P are positively and significantly correlated to *E. coli* counts

H_2: There is a significant correlation between creek water fluorescence and *E. coli* counts

H_3: Nutrient concentrations and *E. coli* counts among Carters Creek sub-catchments are significantly related to sub-catchment land use

2. MATERIALS AND METHODS

2.1 Soils and Geology

Carters Creek (Segment 1209C) is a tributary of the Navasota River situated in Brazos County, Texas. In the Brazos Valley above the confluence of the Brazos and Navasota Rivers, the interfluves are dominated by dark-colored alfisols with a very slowly permeable, clayey argillic horizon in the Hydrologic Soil Group (HSG) of D (NRCS, 2009). The creek floodplain soils are slightly loamier, with a few in the HSG class C. The parent material is predominantly residuum from sandstone and shale or loamy/clayey alluvium over Yegua geologic formation from the Eocene age.

2.2 Sampling Sites

Carters Creek has been identified by the Texas Commission on Environmental Quality (TCEQ) as impaired for bacteria (Category 5a) since 1999; in addition, Burton Creek (Segment (1209L), a tributary of Carters Creek, has been identified as impaired for bacteria (Category 5c) since 2006 (TCEQ, 2007). Category 5a on a 303(d) list of impaired water indicates that a TMDL is underway, while Category 5c indicates that more data needs to be collected before TMDL is introduced for that water body. Both Carters Creek and Burton Creek were also listed as concerns for water use attainment and screening levels in June 2007 for nitrate and orthophosphate (TCEQ, 2007).

The fifteen Carters Creek subcatchments drain land varying from agricultural rangeland to developed urban and residential areas, as represented in Table 1. One rural subcatchment and two urban subcatchments were sampled in this study using a nested scheme, while three more subcatchments were sampled at a single site before their confluence with Carters Creek main stem. The main stem was sampled at two locations for a total of 15 sampling sites (Figure 1).

Most samples were taken from the upstream side of bridges, with the exception of Carter 3 (taken on the downstream side), Wolfpen and Burton 2 (taken from the creek bank in city parks).

The most rural of the subcatchments sampled (Carter 3) formed the headwaters of the main stem at the crossing of Old Reliance Road, and was sometimes not flowing during periods of drought. Carter 1 at Booneville Road and Carter 2 at Austin's Colony Parkway drained more developed/residential land, but both had significant riparian zones that may have lent a degree of protection from contaminant loading.

On the other end of the spectrum, Burton Creek had headwaters in residential neighborhoods and commercial districts from the city of Bryan, with some remaining riparian zone upstream but much of the lower creek channelized in a concrete stream bed for flood control. Three samples were taken from the different first order branches: Burton 3 at Texas Avenue near Hensel Avenue, Burton 4 on Villa Maria Drive near Wayside Drive and Burton 5 on Villa Maria near Cavitt Drive. One sample was taken from the second-order segment (Burton 2 at Tanglewood Park) and one from the third-order segment (Burton 1 at E 29th Street) to allow for nested monitoring. Briar Creek was also nested with two sampling sites within Bryan city limits, Briar 1 at Broadmoor Drive and Briar 2 on Villa Maria near Blinn College. Both Burton and Briar Creeks joined the Carters Creek main stem upstream of the Carter 4 sampling site.

Three additional subcatchments, Hudson, Wolfpen and Bee Creeks, were sampled at a single site each. Hudson Creek joined Carters Creek just downstream of the Carter 4 sampling site on the main stem, (but sampling was before the confluence, on University Drive) followed by Wolfpen Creek at Wolfpen Creek Park. Bee Creek, sampled at Appomattox Drive, flowed from highly urbanized/residential areas in College Station. Wolfpen Creek was somewhat protected by a riparian zone corridor and city park at the sampling site. Bee Creek, on the other hand, had very little riparian zone and was undergoing an improvement project upstream during the year it

Table 1. Coordinates of sample locations and percent land use in the individual watersheds. Land use categories are based on the Anderson Land Use/ Land Cover (LULC) classification system.

			Residential	Commercial	Industrial	Transport	Ind/Comm	Other Urban	Crop/Pasture	Shrub Range	Mixed Range	Forest	Water	Barren
Site Name	**N**	**W**	**11**	**12**	**13**	**14**	**15**	**17**	**21**	**32**	**33**	**41**	**53**	**76**
Bee	30°36'34.88"	96°16'53.45"	61.26	13.10	1.79	1.40	0.65	4.26	3.84		14.20	0.02		
Briar 1	30°39'42.52"	96°20'23.58"	72.86	13.96				13.08			0.11			
Briar 2	30°39'56.21"	96°20'53.97"	87.96	3.69				8.17			0.18			
Burton 1	30°38'26.87"	96°19'44.72"	56.78	31.65	5.61		0.09	4.93	0.61		0.21			0.11
Burton 2	30°38'26.13"	96°20'03.76"	52.96	31.17	8.28			6.79	0.49		0.31			
Burton 3	30°37'58.62"	96°20'28.28"	67.04	29.55				1.49	1.92					
Burton 4	30°38'48.62"	96°20’59.71"	59.32	38.86	0.22			1.37			0.24			
Burton 5	30°38'32.50"	96°21'13.53"	37.74	22.97	21.17			16.27	1.27		0.57			
Carter 1	30°40'14.34"	96°19'17.19"	20.48	9.75	4.96	1.81		2.19	18.05	0.71	41.43	0.22		0.39
Carter 2	30°40'50.43"	96°20'09.54"	43.23	24.11	0.93	3.82		5.58	0.15	0.09	20.60	0.61		0.87
Carter 3	30°41'49.28"	96°20'21.74"	14.16	7.80	32.28	4.35			2.08	3.31	36.02			
Carter 4	30°38'26.11"	96°18'33.52"	38.06	18.34	3.81	1.61	0.03	5.31	8.27	0.32	23.89	0.20		0.22
Carter 5	30°35'19.06"	96°13'28.54"	33.76	13.64	2.60	1.21	0.26	6.90	8.51	0.14	28.76	4.70	0.01	0.11
Hudson	30°38'38.49"	96°18'09.69"	7.90	1.57				54.69	0.51		33.27	1.86		0.19
Wolfpen	30°37'07.14"	96°18'28.24"	43.30	48.74			0.03	2.43			0.01	5.49		

was sampled. Because some residential plots were constructed too close to the creek's flood level, segments of the creek were channelized with loosely-interlocking concrete blocks with the intent to aid infiltration and increase the capacity of flow.

The two remaining sampling sites, Carter 4 and Carter 5, were on the main stem of the creek. Both sites have a gravelly, rocky bed and visible flow. Carter 4 on University Drive, College Station was downstream of the confluence of Briar Creek, Burton Creek and the three upper Carter sites (1-3), while Carter 5 on William D. Fitch Parkway was the farthest downstream of all sites and encompassed all of the subcatchments studied (Figure 1).

Figure 1. Map of sampling sites on Carters Creek subcatchments.

2.3 Monitoring/Observation

The fifteen sites on the subcatchments of Carter Creek were sampled twice monthly from July 2007 to June 2008. The water samples were collected in sterile 500 mL Nasco Whirlpak bags and transported back to the lab on ice for analysis within 6 hr of collection. To enumerate *E. coli*, samples were filtered through a sterile 0.45 µm Millipore filter and incubated on modified mTEC agar for 2 hr at 35 ^{o}C and 22-24 hr at 44.5^{o}C according to EPA Method 1603 (USEPA, 2002).

For the water chemistry, samples were tested for pH and electrical conductance, prior to filtration through ashed 0.7 µm Whatman GF/F filters and frozen until analysis. Concentrations of non-purgeable organic carbon (NPOC) and total dissolved nitrogen (TDN) were determined using a Shimadzu TOC-$V_{CSH.}$ This combustion method mineralizes DOC and the resulting CO_2 is detected with a NDIR detector. Total dissolved N is detected by chemiluminesence. Nitrate-N was determined by Cd-Cu reduction according to USEPA Method 353.3, while ammonium-N was quantified by phenate hypochlorite with sodium nitroprusside enhancement according to USEPA Method 350.1, using a Westco Scientific Smartchem Discrete Analyzer for both. A Dionex ICS Ion Chromatograph was used to quantify Na^{+}, K^{+}, Mg^{2+}, Ca^{2+}, F^{-}, Cl^{-}, Br^{-}, NO_2^{-}, SO_4^{2-} and PO_4^{-3}. The anions were separated using an Ionpak AS20 analytical and Ionpak AG20 guard columns, with 35 mM KOH as eluent, a flow rate of 1 mL min^{-1} and an injection volume of 25 µL (DIONEX ICS 2000). Cations were separated on an Ionpac CS16 analytical and Ionpac CG16 guard columns, with 20 mM methanosulfonic acid as eluent, a flow rate of 1 mL min^{-1} and an injection volume of 10 µL (DIONEX ICS 1000).

For some chemical parameters some calculation was necessary. Check standards and NIST traceable standards were run every twelfth sample to ensure QA/QC of the sample run. The coefficient of variance between replicates was typically less than 2% for colorimetric analysis and ion chromatography and less than 5% for DOC and TDN by combustion. The DOC:DON ratio, DON:TDN ratio and Cl:Br ratio were all calculated by mass. Sodium Adsorption Ratio (SAR) was calculated according to the following formula (in units of mEq/L):

$$SAR = \frac{[Na^{+}]}{\sqrt{\frac{1}{2}([Ca^{2+}]+[Mg^{2+}])}}$$

Thus SAR takes into account not only the concentration of dispersing sodium-derived positive charge, but also the opposing flocculent effect of the divalent cation charge.

2.4 Fluorometry

Burton 4, a subcatchment with routinely high *E. coli* counts, was chosen for analysis using *E. coli* Method 1603 and fluorescence. The fluorometer used was the Aquafluor handheld fluorometer and turbidimeter (Turner Designs, Sunnyvale, CA). Sampling up the stream reach gave a rough indication of possible seepage from leaky sewage pipes or septic systems in Bryan’s older neighborhoods. The underlying premise is that fluorescence should increase with greater human inputs of optical brighteners in wastewater, along with a qualitative increase of *E. coli* counts. As outlined by Hartel et al. (2007), using a microbial source tracking method with fecal bacteria counts and optical brightener fluorescence, the following scenarios are possible: (1) strong relative fluorescence and high bacterial counts, indicating human fecal contamination from a sewer line or septic system; (2) strong relative fluorescence and low bacterial counts, pointing to a gray water leak; (3) weak relative fluorescence and high bacterial counts, suggesting a black water leak or a non-human fecal source, such as pets or wildlife; and (4) weak relative fluorescence and low bacteria counts, indicating no significant contaminating factor.

Calibration of the fluorometer was performed according to the protocol described in Hagedorn et al. (2002), in which 100 μg L^{-1} Tide (Proctor & Gamble, Cincinnati, OH) was calibrated to 100 RFU (relative fluorescence units), and a field measurement over 100 RFU was considered a positive result. In the field, samples were taken at intervals of varying distances beginning at 350 m upstream of the Burton 2 sampling site. Because of the rapidity of the fluorometer measurements, fluorescence was taken approximately every 50-100 m in selected reaches of the creek, while an *E. coli* sample was taken at significant confluences, suspected bacteria-contributing sites or sites with fluorescence spikes. Samples for *E. coli* and DOC analysis were kept cold until returning to the lab, where they were filtered and processed for *E. coli* within 6 hr of collection according to EPA Method 1603 (USEPA, 2002). As before, DOC was quantified using a Shimadzu TOC-V_{CSH} analyzer by the combustion method.

2.5 Geographical Analysis

The relationship of nutrient loadings to different land use types were explored using geographical information systems (GIS) in ArcView 9.3 (ESRI Inc., Redlands, CA). It has been shown that a particular land use may produce a specific nutrient footprint which could be used for general predictions in other watersheds. To determine the polygon shapes of the subcatchments, a Stratmap DEM (TNRIS, 2009a) was overlaid with the National Hydrography Dataset, or NHD (TNRIS, 2009b) to find the ridges of the watershed boundaries. Zoning data from 2008 were obtained from the City of Bryan and the City of College Station to provide more recent land use data in this analysis. For the land area outside of Bryan or College Station city limits, the USGS National Land Cover Dataset (NLCD) from 2001 was used. This land cover dataset was clipped with the watershed polygons upstream of each sampling site, and the land use percentages for each subcatchment were calculated for comparison with nutrient and *E. coli* data. Because the sample sites were chosen according to a nested scheme for pinpointing a more precise geographical source of different constituents, several lower nested sites are omitted from this portion of the analysis to avoid counting the same watershed area twice. Also, the sites downstream of wastewater treatment plants are omitted because the dominance of effluent masks the effects of land use on stream chemistry. Thus this portion of the project will only compare a subset of nine headwater sites from the tributary watersheds, namely Burton 3, Burton 4, Burton 5, Briar 2, Carter 2, Carter 3, Hudson, Bee and Wolfpen Creeks.

2.6 Statistical Analysis

The logarithmic transformations of bacterial counts and the concentrations of all nutrients analyzed were compared using SPSS through a one-way ANOVA by sampling site and a post-hoc Tukey Test to determine which sites were significantly different from each other. The data was also divided according to low-flow, high-flow, all-flow and seasonal patterns. A sample was considered high flow if there was precipitation within two days before the sample date (Figure 2). Pearson bi-variate correlation was then used to determine any significant relationship between microbial and chemical variables and watershed land use using the nine-creek subset to minimize sample dependence. Significant relationships were plotted to find linear regression models explaining the majority of the variance for a particular water quality parameter.

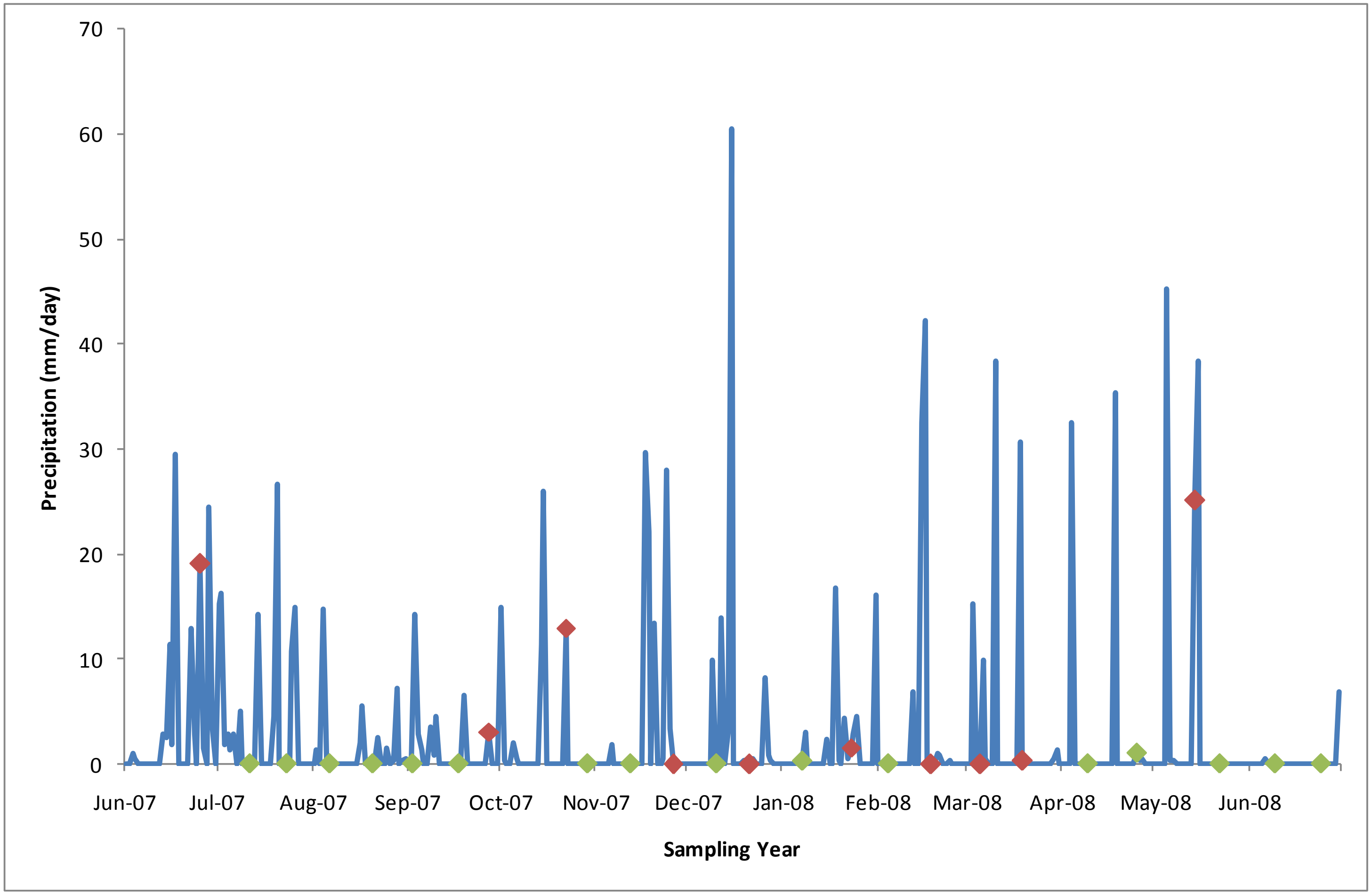

Figure 2. Rain events and sampling dates over the one-year sampling period. Precipitation measurements were taken at the Easterwood Airport (code KCLL) in College Station, TX (weatherunderground.com, 2009).

3. RESULTS

3.1 Stream Water Constituents

Using one-way analysis of variance followed by a Tukey honestly significant difference post hoc test I was able to reject or accept my initial null hypotheses that all the streams monitored were not significantly different in terms of *E. coli* counts and chemical constituents. The ANOVA was applied to low flow, high flow, all flow and seasonal data in turn.

3.1.1 *E. coli*

3.1.1.1. Low Flow Conditions

In the Carters Creek watershed, all sites during the sample period displayed low minimum counts but nevertheless all sites exceeded the state requirements for secondary contact recreation, namely the geometric mean standard of 126 CFU/100 mL and the single-sample standard of 394 CFU/100 mL (TCEQ). Mean annual *E. coli* counts during low flow conditions were significantly different among the sub-catchments sampled when a one-way analysis of variance was performed on base-10 logarithmic transformations of the grab sample *E. coli* counts (Figure 3). Burton 4 had significantly higher *E. coli* during low flow than Burton 1, Carter 1, Carter 3 and Wolfpen sub-catchments (Table 2). Burton 2 was also significantly higher than Burton 1, Carter 1 and Carter 3, while Burton 5 was higher than Burton 1 and Carter 1.

3.1.1.2. High Flow Conditions

Under high flow conditions, all creeks displayed *E. coli* counts well over the established single-sample standard for contact recreation, but unlike in the previous low flow subsection, no statistical differences were found among the creeks sampled (Table 3).

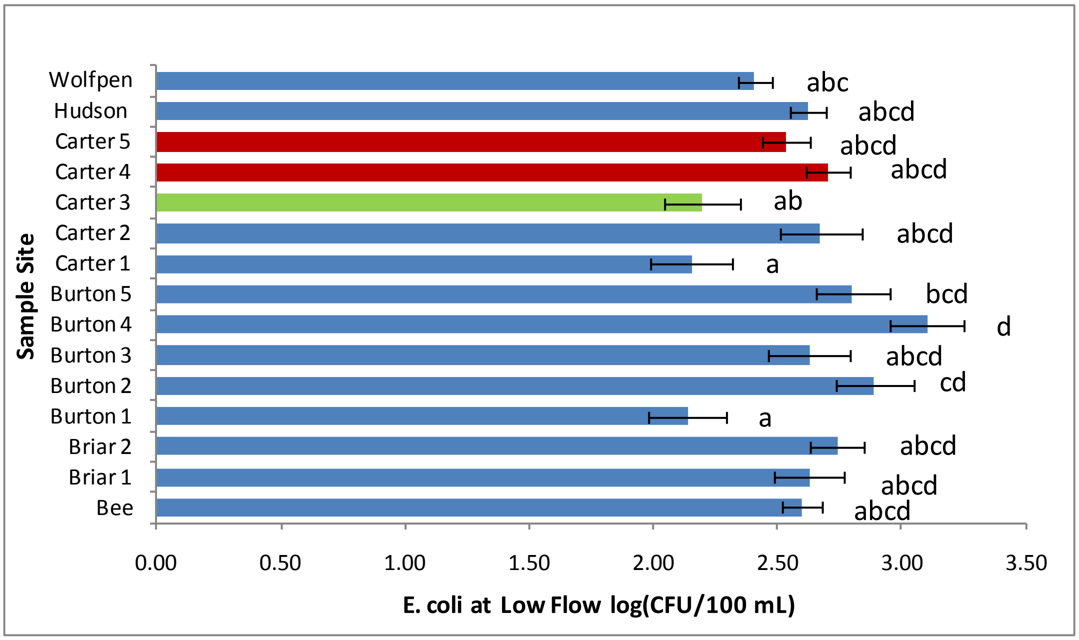

Figure 3. Base-10 logarithm transformations of annual *E. coli* counts during low flow conditions. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$. Red bars indicate sample sites downstream of wastewater treatment plants, while the green bar indicates a site at the top of the Carters Creek watershed in a rural open area.

Table 2. Annual geometric means for *E. coli* in CFU/100 mL during low flow. Different letters indicate significant difference at $p \leq 0.05$.

Site Name	10^x	Std Dev	Std Error	Geo Mean	Minimum	Maximum
Bee	[abcd]2.60	0.33	0.08	397	80	1380
Briar 1	[abcd]2.63	0.56	0.14	426	40	4700
Briar 2	[abcd]2.74	0.44	0.11	550	110	6600
Burton 1	[a]2.14	0.63	0.16	137	9	1170
Burton 2	[cd]2.89	0.63	0.16	776	20	6600
Burton 3	[abcd]2.63	0.66	0.17	424	10	3500
Burton 4	[d]3.10	0.60	0.15	1265	170	17000
Burton 5	[bcd]2.80	0.59	0.15	635	10	5600
Carter 1	[a]2.15	0.69	0.17	142	10	9200
Carter 2	[abcd]2.67	0.68	0.17	471	20	9700
Carter 3	[ab]2.20	0.58	0.15	157	9	1090
Carter 4	[abcd]2.71	0.36	0.09	507	140	2500
Carter 5	[abcd]2.53	0.40	0.10	341	30	1180
Hudson	[abcd]2.62	0.30	0.08	419	130	1190
Wolfpen	[abc]2.41	0.29	0.07	257	90	710

3.1.1.3. All Flow Conditions

For all flow conditions, Carter 1 and Carter 3 were significantly lower than Burton 4 when the base-10 logarithm values were analyzed in the ANOVA test (Table 4). However, no other significant differences were found. Particularly of interest is the fact that Carter 4 and 5, sample sites downstream of wastewater treatment plants, were not significantly higher or lower than any of the other sites.

3.1.1.4. Seasonal Variation

The only significant difference in *E. coli* among the creeks during individual seasons was that Burton 2 and 4 had a higher average base-10 logarithm values than Carter 3 in the summer. This difference, however, apparently drove much of the difference in mean annual counts, as described in the above subsections.

Table 3. Annual geometric means for *E. coli* in CFU/100 mL during high flow. Equivalent letters indicate no significant difference at $p \leq 0.05$.

Site Name	N	10^x	Std Dev	Std Error	Geo Mean	Minimum	Maximum
Bee	10	[a]3.58	0.42	0.13	3791	810	14400
Briar 1	10	[a]3.50	0.71	0.23	3172	200	31000
Briar 2	10	[a]3.68	0.69	0.22	4820	500	45000
Burton 1	10	[a]3.71	0.64	0.20	5149	900	50000
Burton 2	10	[a]3.74	0.71	0.23	5441	600	64000
Burton 3	10	[a]3.78	0.61	0.19	6014	650	46000
Burton 4	10	[a]3.89	0.55	0.17	7803	1200	55000
Burton 5	10	[a]3.74	0.61	0.19	5561	670	41000
Carter 1	9	[a]3.56	0.57	0.19	3643	400	28000
Carter 2	9	[a]3.59	0.40	0.13	3932	900	12400
Carter 3	9	[a]3.18	0.61	0.20	1494	150	11100
Carter 4	10	[a]3.35	0.67	0.21	2241	400	19000
Carter 5	9	[a]3.24	0.63	0.21	1716	170	17200
Hudson	10	[a]3.35	0.53	0.17	2228	230	37000
Wolfpen	9	[a]3.67	0.51	0.17	4707	1310	64000

Table 4. Annual geometric means for *E. coli* in CFU/100 mL during all flow. Different letters indicate significant difference at p ≤ 0.05.

Site Name	N	10^x	Std Dev	Std Error	Geo Mean	Minimum	Maximum
Bee	26	$^{ab}3.0$	0.6	0.1	946	80	14400
Briar 1	26	$^{ab}3.0$	0.7	0.1	922	40	31000
Briar 2	26	$^{ab}3.1$	0.7	0.1	1268	110	45000
Burton 1	26	$^{ab}2.7$	1.0	0.2	553	9	50000
Burton 2	26	$^{ab}3.2$	0.8	0.2	1642	20	64000
Burton 3	26	$^{ab}3.1$	0.9	0.2	1177	10	46000
Burton 4	26	$^{b}3.4$	0.7	0.1	2547	170	55000
Burton 5	26	$^{ab}3.2$	0.8	0.1	1463	10	41000
Carter 1	26	$^{a}2.6$	0.9	0.2	437	10	28000
Carter 2	26	$^{ab}3.0$	0.7	0.1	981	20	12400
Carter 3	24	$^{a}2.6$	0.8	0.2	365	9	11100
Carter 4	26	$^{ab}3.0$	0.6	0.1	898	140	19000
Carter 5	26	$^{ab}2.8$	0.6	0.1	597	30	17200
Hudson	26	$^{ab}2.9$	0.5	0.1	797	130	14800
Wolfpen	26	$^{ab}2.8$	0.7	0.1	704	90	37000

3.1.2 Dissolved Organic Carbon

3.1.2.1 Low Flow Conditions

Dissolved organic carbon is derived from various natural and anthropogenic sources in the watershed which might include leachate from lawns or wooded areas; plant and insect exudates; trash and plant litter fallen into the stream; or various surfactants and residues that are washed into the drainage network. For the Carters Creek watershed, concentrations of dissolved organic carbon were significantly different among the fifteen study sites ($p = 0.008$; Table 5). Wolfpen creek had the highest mean annual concentration of DOC during low flow conditions and it was significantly higher than Briar 2 and Carter 1. The lowest DOC concentration was found at Carter 1 and the highest at Wolfpen Creek (Table 5).

3.1.2.2. High Flow Conditions

In high flow conditions, dissolved organic carbon concentrations showed no significant difference among subcatchments (Table 6). The range of high flow creek averages varied from 22.0 mg/L in Carter 1 to 49.0 mg/L in Wolfpen.

3.1.2.3. All Flow Conditions

Mean annual DOC produced similar results to high flow DOC concentrations and lack of significance. Dissolved Organic Carbon in Wolfpen Creek was significantly higher than the upper Carter sites (Carter 1-3), Briar 2, Burton 3 and Burton 5 (Table 7).

Table 5. Mean annual concentrations of dissolved organic carbon during low flow conditions. Different letters indicate significant difference at $p \leq 0.05$.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	16	^{ab}32.9	18.44	4.61	11.68	75.6
Briar 1	16	^{ab}46.8	27.36	6.84	15.89	97.89
Briar 2	16	^{a}26.0	13.17	3.29	10.4	51.51
Burton 1	16	^{ab}37.5	22.02	5.51	11.86	86
Burton 2	16	^{ab}45.3	28.62	7.15	12.76	121.56
Burton 3	16	^{ab}30.2	21.91	5.48	10.3	88.14
Burton 4	16	^{ab}36.5	20.61	5.15	15.69	85.9
Burton 5	16	^{ab}34.6	20.25	5.06	9.35	90.76
Carter 1	17	^{a}26.3	10.71	2.60	8.29	41.53
Carter 2	17	^{ab}43.3	24.73	6.00	8.79	90.04
Carter 3	15	^{ab}31.5	11.73	3.03	12.01	47.67
Carter 4	16	^{ab}48.1	29.10	7.28	9.72	84.83
Carter 5	17	^{ab}39.0	26.95	6.54	9.42	82.2724
Hudson	16	^{ab}42.4	24.15	6.04	12.51	73.65
Wolfpen	17	^{b}58.9	44.83	10.87	10.99	141.35

Table 6. Mean annual dissolved organic carbon during high flow. Equivalent letters indicate no significant difference at $p \leq 0.05$.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	10	[a]27.35	18.76	5.93	11.25	70.2
Briar 1	10	[a]27.41	25.38	8.03	10	95.45
Briar 2	10	[a]24.13	13.96	4.41	7.89	50.41
Burton 1	10	[a]27.44	22.43	7.09	9.84	79.11
Burton 2	10	[a]25.64	16.15	5.11	11.03	52.2
Burton 3	10	[a]27.21	14.81	4.68	11.02	48.53
Burton 4	10	[a]32.63	28.28	8.94	11.04	102.83
Burton 5	10	[a]26.83	13.63	4.31	10.57	52.43
Carter 1	9	[a]22.01	10.22	3.41	10.95	41.35
Carter 2	9	[a]26.14	12.52	4.17	2.81	39.85
Carter 3	9	[a]23.56	10.83	3.61	10.71	44.79
Carter 4	10	[a]32.95	25.4	8.03	7.8	86.54
Carter 5	9	[a]35.94	20.09	6.7	13.96	82.76
Hudson	10	[a]36.02	28.87	9.13	10.06	110.86
Wolfpen	9	[a]48.96	42.03	14.01	13.24	130.34

Table 7. Mean annual dissolved organic carbon during all flow. Different letters indicate significant difference at $p \leq 0.05$.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	26	[a]30.74	18.39	3.61	11.25	75.6
Briar 1	26	[ab]39.34	27.82	5.46	10	97.89
Briar 2	26	[a]25.25	13.23	2.59	7.89	51.51
Burton 1	26	[ab]33.64	22.30	4.37	9.84	86.00
Burton 2	26	[ab]37.72	26.08	5.11	11.03	121.56
Burton 3	26	[a]29.04	19.22	3.77	10.3	88.14
Burton 4	26	[ab]35.03	23.38	4.58	11.04	102.83
Burton 5	26	[a]31.63	18.11	3.55	9.35	90.76
Carter 1	26	[a]24.83	10.55	2.07	8.29	41.53
Carter 2	26	[a]37.39	22.61	4.43	2.81	90.04
Carter 3	24	[a]28.55	11.84	2.42	10.71	47.67
Carter 4	26	[ab]42.26	28.23	5.54	7.8	86.54
Carter 5	26	[ab]37.94	24.42	4.79	9.42	82.76
Hudson	26	[ab]39.94	25.69	5.04	10.06	110.86
Wolfpen	26	[b]55.48	43.30	8.49	10.99	141.35

3.1.2.4. Seasonal Variation

Dissolved Organic Carbon concentrations showed significant differences among creeks only in the fall (Figure 4). In comparing mean fall concentrations, Wolfpen was significantly higher than Briar 2, Carter 1, Carter 3, Burton 3 and Burton 5. This pattern of results was almost identical to the pattern of significant differences in all flow conditions, with the omission of Carter 2 for the fall means.

3.1.3 Nitrogen Species: Nitrate, Nitrite, Ammonium and Dissolved Organic Nitrogen

3.1.3.1. Low Flow Conditions

Nitrate-N was significantly higher at those sites sampled downstream of a WWTP (p = 0.001; Figure 5). There was no significant difference in nitrate-N concentrations during low

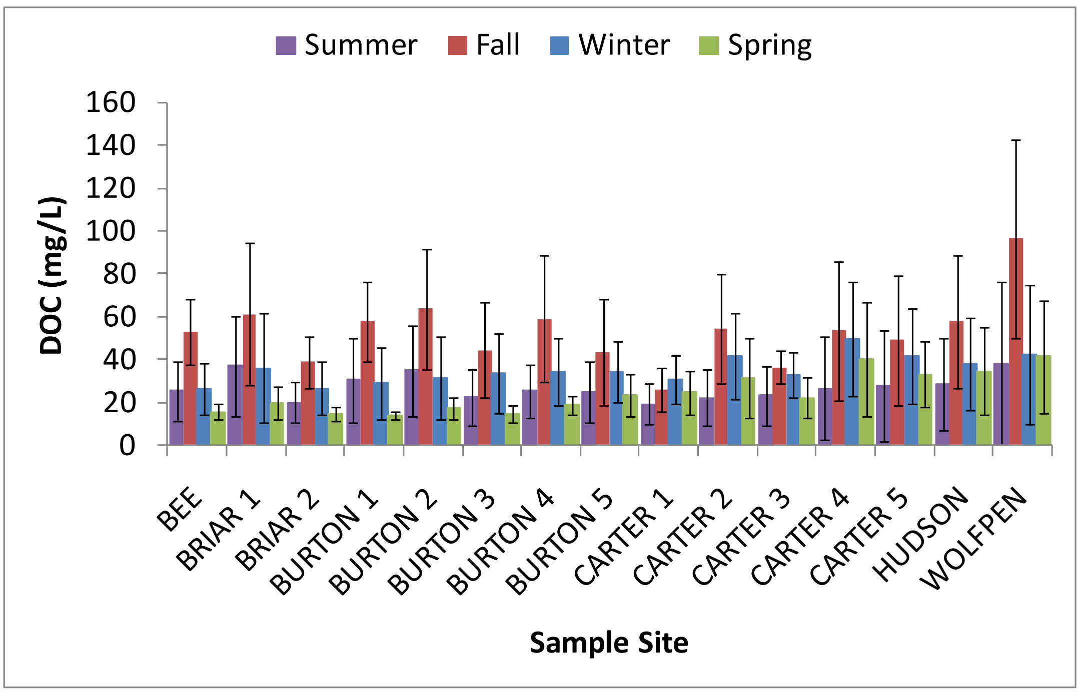

Figure 4. Seasonal dissolved organic carbon. Error bars are standard error of the mean.

flow conditions among those sites without a WWTP. Nitrite was significantly higher at those sites downstream of a WWTP and in addition higher at Briar 1, Carter 1 and 2 and Burton 4 ($p < 0.001$; Figure 6). Nitrite is an indicator of raw sewage and is typically oxidized readily to nitrate. There was no significant difference in ammonium-N concentrations among study sites ($p = 0.07$; Table 8). However, there was a significant difference in dissolved organic nitrogen concentrations among sites during low flow conditions. Significantly lower mean low flow concentrations were found at Carter 1 and 2, Burton 3 and Wolfpen, while the highest mean low flow concentration was found at Carter 4 downstream of Burton Creek WWTP (Table 9).

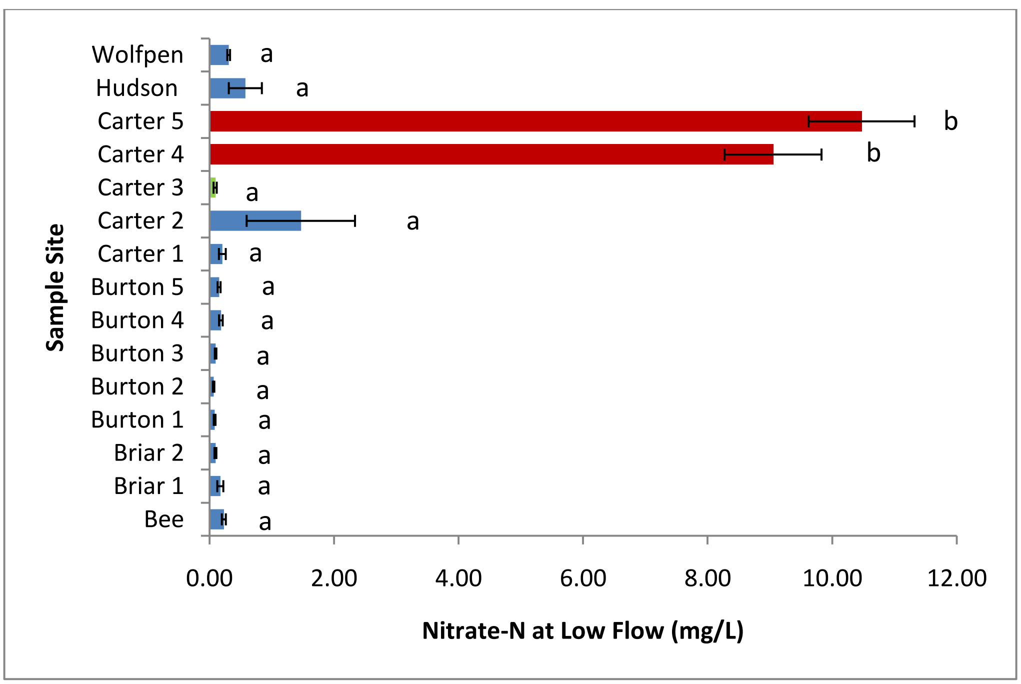

Figure 5. Mean annual nitrate-N concentrations in low flow conditions. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

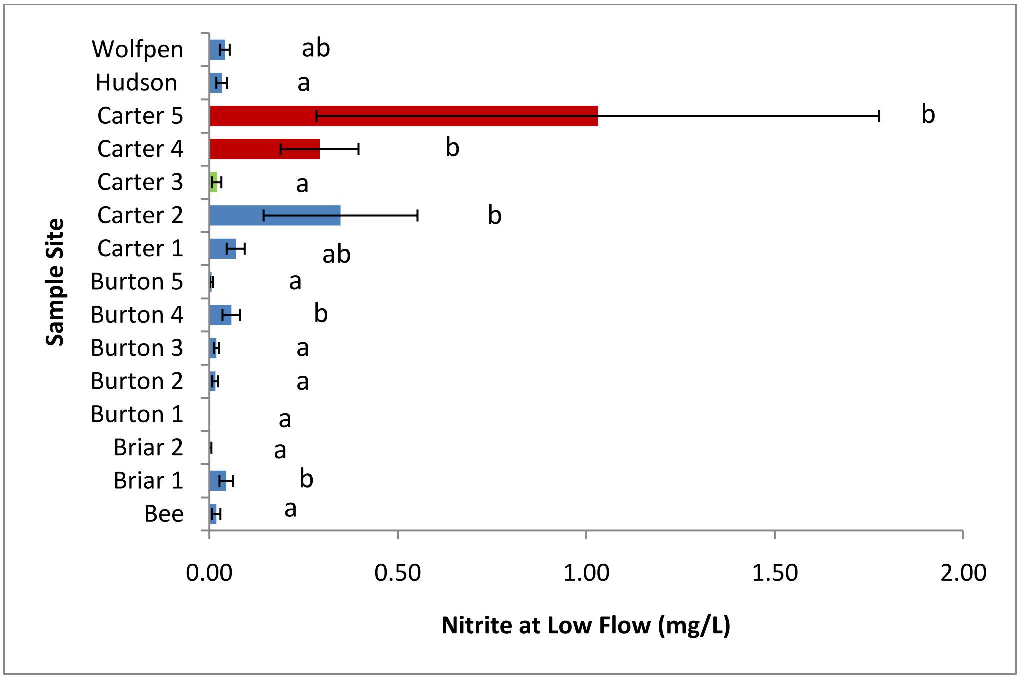

Figure 6. Mean annual nitrite concentrations during low flow conditions. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

Table 8. Mean annual ammonium-N concentrations during low flow. Equivalent letters indicate no significant difference at $p \leq 0.05$.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	16	[a]0.132	0.20	0.05	0.01	0.84
Briar 1	16	[a]0.142	0.38	0.09	0.01	1.55
Briar 2	16	[a]0.076	0.05	0.01	0.01	0.2
Burton 1	16	[a]0.034	0.02	0.00	0.01	0.08
Burton 2	16	[a]0.031	0.02	0.00	0.01	0.07
Burton 3	16	[a]0.043	0.04	0.01	0	0.16
Burton 4	16	[a]0.063	0.05	0.01	0.01	0.19
Burton 5	16	[a]0.048	0.04	0.01	0.01	0.14
Carter 1	17	[a]0.070	0.05	0.01	0.02	0.21
Carter 2	17	[a]0.165	0.24	0.06	0.01	0.81
Carter 3	15	[a]0.069	0.06	0.02	0.02	0.25
Carter 4	16	[a]0.138	0.12	0.03	0.04	0.45
Carter 5	17	[a]0.082	0.05	0.01	0.05	0.25
Hudson	16	[a]0.062	0.04	0.01	0.02	0.18
Wolfpen	17	[a]0.051	0.03	0.01	0.02	0.12

Table 9. Mean annual dissolved organic nitrogen concentrations during low flow. Different letters signify a significant difference at $p \leq 0.05$.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	16	[ab]0.91	0.29	0.07	0.53	1.74
Briar 1	16	[ab]1.11	0.35	0.09	0.61	2.1
Briar 2	16	[ab]0.76	2.48	0.62	0.23	10.61
Burton 1	16	[ab]0.97	0.42	0.10	0.36	2.09
Burton 2	16	[ab]1.27	0.71	0.18	0.57	3.23
Burton 3	16	[a]0.61	0.40	0.10	0.19	1.91
Burton 4	16	[ab]0.97	0.39	0.10	0.53	2.01
Burton 5	16	[ab]0.81	0.27	0.07	0.36	1.6
Carter 1	17	[a]0.54	0.30	0.07	0	1.15
Carter 2	17	[a]0.61	0.45	0.11	0	2.04
Carter 3	15	[ab]0.85	0.27	0.07	0.22	1.17
Carter 4	16	[c]3.85	3.70	0.92	-0.86	12.72
Carter 5	17	[b]2.13	1.97	0.48	-0.03	6.55
Hudson	16	[ab]1.67	0.61	0.15	0	2.89
Wolfpen	17	[a]0.60	0.21	0.05	0.26	1.13

3.1.3.2. High Flow Conditions

Both Total Dissolved Nitrogen and nitrate-N concentrations were significantly higher in the two sample sites downstream of WWTPs at the $p<0.05$ level, while there was no significant difference among sites without WWTPs. Dissolved organic nitrogen and ammonium showed no difference among all fifteen sites. The nitrite concentration in Carter 4 also remained significantly higher than Briar 1 & 2, Burton 1,3 & 5, Carter 2, and Wolfpen (Table 10).

3.1.3.3. All Flow Conditions

Nitrogen during all flow showed no difference among subcatchments in ammonium-N (Table 11), but Carter 4 and 5 had significantly higher nitrate-N (Figure 7) and TDN (Figure 8) concentrations than the other sites. Nitrite concentrations in Carter 5 were similar to Carter 2 and Carter 4, but higher than the remaining watersheds (Table 12). Carter 4 and 5 had significantly higher DON than Carter 1, and Carter 5 was higher than all but Carter 4 (Table 13).

Table 10. Mean annual nitrite concentration during high flow. Different letters signify a significant difference at $p \leq 0.05$.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	10	^{ab}0.07	0.09	0.03	0	0.23
Briar 1	10	^{a}0.02	0.03	0.01	0	0.1
Briar 2	10	^{a}0.03	0.05	0.02	0	0.12
Burton 1	10	^{a}0.01	0.02	0.01	0	0.06
Burton 2	10	^{ab}0.05	0.07	0.02	0	0.21
Burton 3	10	^{a}0.03	0.04	0.01	0	0.08
Burton 4	10	^{ab}0.07	0.05	0.02	0	0.15
Burton 5	10	^{a}0.03	0.03	0.01	0	0.07
Carter 1	9	^{ab}0.08	0.09	0.03	0	0.26
Carter 2	9	^{a}0.03	0.06	0.02	0	0.17
Carter 3	9	^{ab}0.04	0.07	0.02	0	0.22
Carter 4	10	^{b}0.23	0.4	0.13	0	1.32
Carter 5	9	^{ab}0.15	0.1	0.03	0.04	0.32
Hudson	10	^{ab}0.05	0.08	0.03	0	0.24
Wolfpen	9	^{a}0.03	0.05	0.02	0	0.12

Table 11. Mean annual ammonium-N concentrations during all flow. Equivalent letters indicate no significant difference at the $p \leq 0.05$ level.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	26	^{a}0.12	0.16	0.03	0.01	0.84
Briar 1	26	^{a}0.12	0.30	0.06	0.01	1.55
Briar 2	26	^{a}0.08	0.05	0.01	0	0.20
Burton 1	26	^{a}0.04	0.03	0.01	0.01	0.10
Burton 2	26	^{a}0.05	0.05	0.01	0.01	0.20
Burton 3	26	^{a}0.08	0.19	0.04	0	0.99
Burton 4	26	^{a}0.09	0.09	0.02	0.01	0.41
Burton 5	26	^{a}0.07	0.07	0.01	0.01	0.34
Carter 1	26	^{a}0.07	0.05	0.01	0.02	0.21
Carter 2	26	^{a}0.13	0.20	0.04	0.01	0.81
Carter 3	24	^{a}0.08	0.09	0.02	0.02	0.41
Carter 4	26	^{a}0.15	0.17	0.03	0.04	0.80
Carter 5	26	^{a}0.10	0.07	0.01	0.05	0.34
Hudson	26	^{a}0.08	0.07	0.01	0.02	0.36
Wolfpen	26	^{a}0.06	0.03	0.01	0.02	0.15

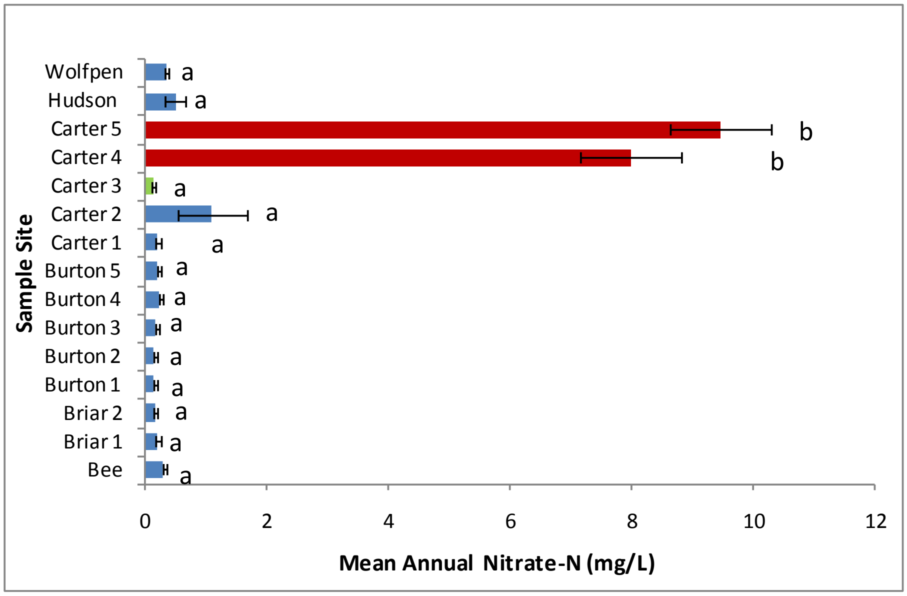

Figure 7. Mean annual nitrate-N for all flow. Error bars are standard error of the mean.

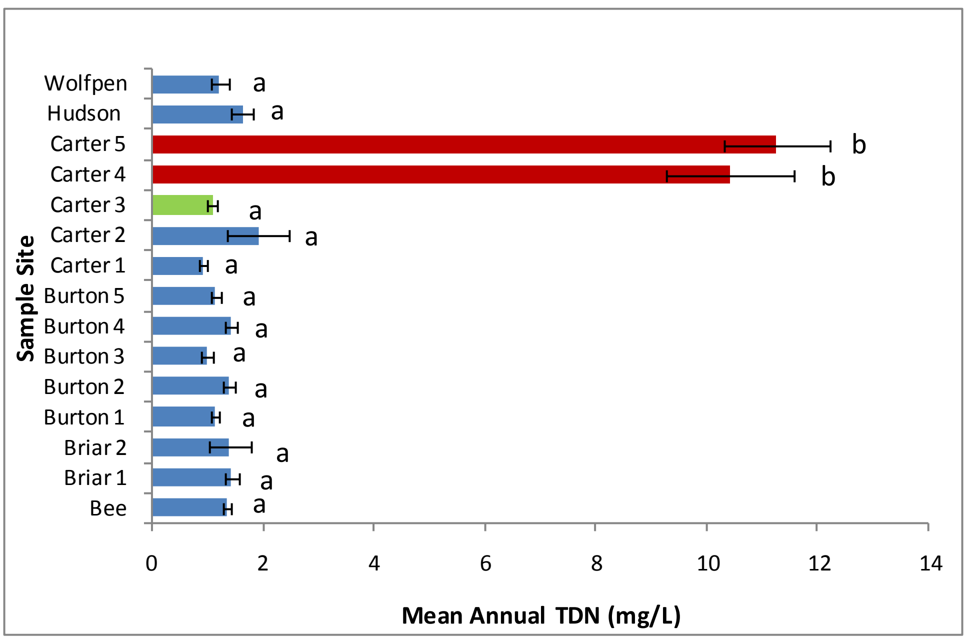

Figure 8. Mean annual total dissolved nitrogen (TDN) concentrations during all flow conditions. Error bars are standard error of the mean.

Table 12. Mean annual nitrite concentrations during all flow. Different letters signify a significant difference.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	26	[a]0.04	0.07	0.01	0	0.23
Briar 1	26	[a]0.03	0.06	0.01	0	0.22
Briar 2	26	[a]0.01	0.03	0.01	0	0.12
Burton 1	26	[a]0.00	0.01	0.00	0	0.06
Burton 2	26	[a]0.03	0.05	0.01	0	0.21
Burton 3	26	[a]0.02	0.03	0.01	0	0.08
Burton 4	26	[a]0.06	0.08	0.02	0	0.29
Burton 5	26	[a]0.01	0.03	0.01	0	0.07
Carter 1	26	[a]0.07	0.10	0.02	0	0.28
Carter 2	25	[ab]0.23	0.67	0.13	0	3.08
Carter 3	24	[a]0.03	0.06	0.01	0	0.22
Carter 4	26	[ab]0.27	0.40	0.08	0	1.54
Carter 5	26	[b]0.73	2.50	0.49	0.04	12.95
Hudson	26	[a]0.04	0.07	0.01	0	0.24
Wolfpen	26	[a]0.04	0.05	0.01	0	0.19

Table 13. Mean annual dissolved organic nitrogen concentrations during all flow. Different letters signify a significant difference.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	26	[ab]0.92	0.29	0.06	0.49	1.74
Briar 1	26	[ab]1.12	0.37	0.07	0.6	2.1
Briar 2	26	[ab]1.16	1.94	0.38	0.23	10.61
Burton 1	26	[ab]0.94	0.36	0.07	0.36	2.09
Burton 2	26	[ab]1.18	0.59	0.12	0.57	3.23
Burton 3	26	[ab]0.73	0.39	0.08	0.19	1.91
Burton 4	26	[ab]1.08	0.38	0.07	0.53	2.01
Burton 5	26	[ab]0.86	0.33	0.07	0.36	1.95
Carter 1	26	[a]0.64	0.29	0.06	0	1.15
Carter 2	26	[ab]0.76	0.39	0.08	0	2.04
Carter 3	24	[ab]0.89	0.29	0.06	0.22	1.41
Carter 4	26	[c]2.36	3.14	0.62	-0.86	12.72
Carter 5	26	[bc]1.74	1.75	0.34	-0.03	6.55
Hudson	26	[ab]1.19	0.93	0.18	0	4.59
Wolfpen	26	[ab]0.82	0.81	0.16	0.26	4.6

3.1.3.4. Relative Proportion of N Species

The proportion of total nitrogen in surface waters as DON is typically a good indicator of the degree of disturbance in a watershed. A higher percentage of total N being transported as DON indicates a relatively undisturbed watershed, while a lower percentage is indicative of a watershed with a high level of disturbance. During low flow conditions all but three of our sites had > 60% of the total N in the form of DON and four of our sites had more than 80% of total N in the form of DON (Figure 9), indicating a predominantly healthy riparian system.

3.1.3.5. Seasonal Variation

Dissolved Organic Nitrogen, as in the case of Dissolved Organic Carbon, only showed significant differences among creeks during the fall. Carter 5 was higher in DON than Carter 1, Carter 2, Wolfpen, Briar 2, Burton 1 and Burton 3, while Carter 4 was higher than every other creek except Carter 5. Curiously, the site highest in DOC (i.e. Wolfpen) was not one of the highest in DON, even though DOC and DON are intrinsically linked.

Nitrate concentrations analyzed seasonally showed no shift in pattern among the creek sites: Carter 4 and 5 were significantly higher than all other sites during all four seasons, though the mean value varied depending on the dilution effect of rain and groundwater contributions (Figure 10). Total Dissolved Nitrogen also indicated no shift in pattern, since it is predominantly driven by high nitrate in sites downstream of the WWTP as well. The proportion of DON to TDN remained lowest in these sites, followed by Carter 2 in the fall when the creek was dredged and Wolfpen in the winter, possibly due to the adjacent construction of high density housing units.

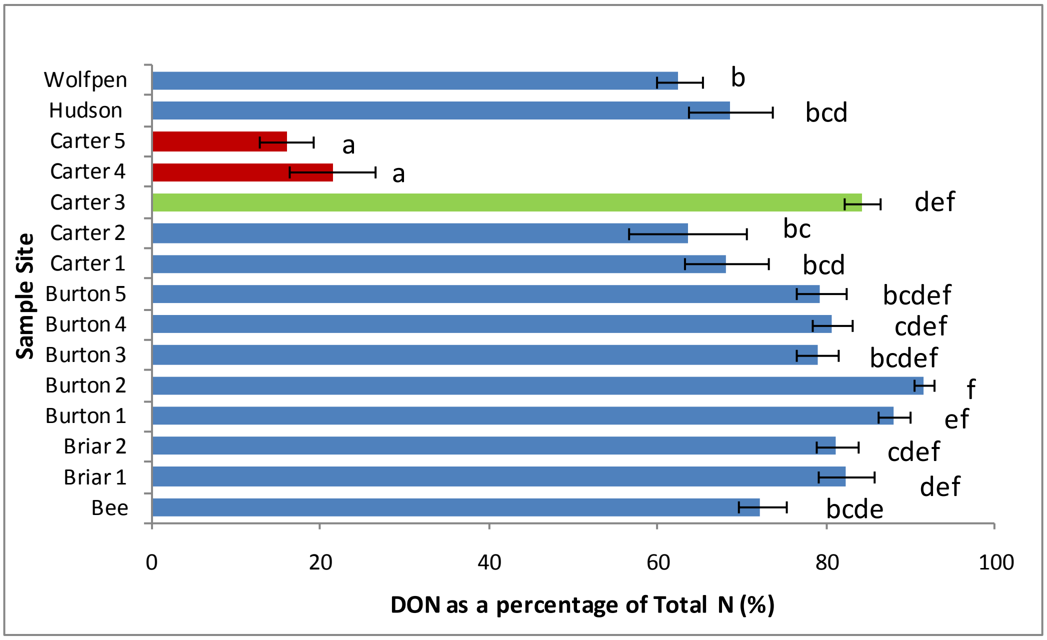

Figure 9. Percentage of total N concentrations in the form of DON. Error bars are standard error of the mean.

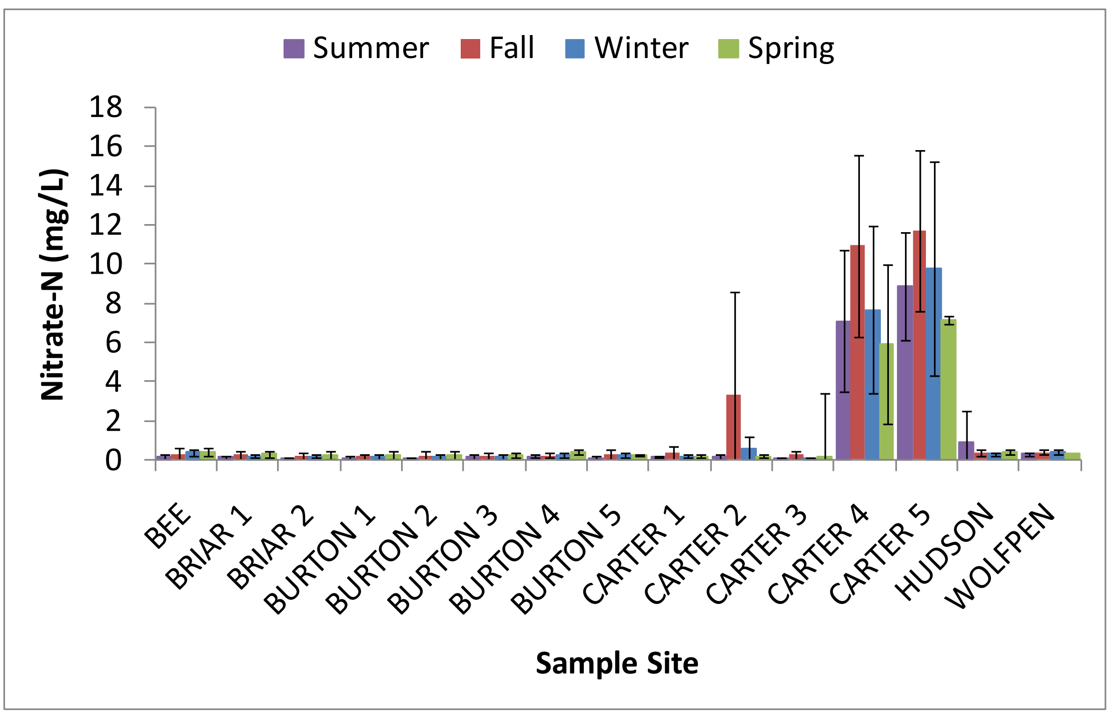

Figure 10. Seasonal nitrate-N. Error bars are standard error of the mean.

Among creeks analyzed seasonally there were no statistical differences in ammonium for summer, fall or winter, but Carter 4 was higher than all the other creeks in the spring.

There was no significant difference among seasonal concentrations of nitrate, nitrite, dissolved organic nitrogen or total dissolved nitrogen. There was, however, a significantly higher ammonium average in spring than in summer.

3.1.4. Orthophosphate

3.1.4.1. Low Flow Conditions

During low flow conditions mean annual orthophosphate concentrations were significantly different among the study sites ($p < 0.001$; Figure 11). The two highest mean concentrations were found at the two sites that were downstream from WWTPs. Wolfpen also had higher phosphate than Bee; Burton 1, 2, 3 and 5; the upper Carter sites (Carter 1-3); and Briar 2.

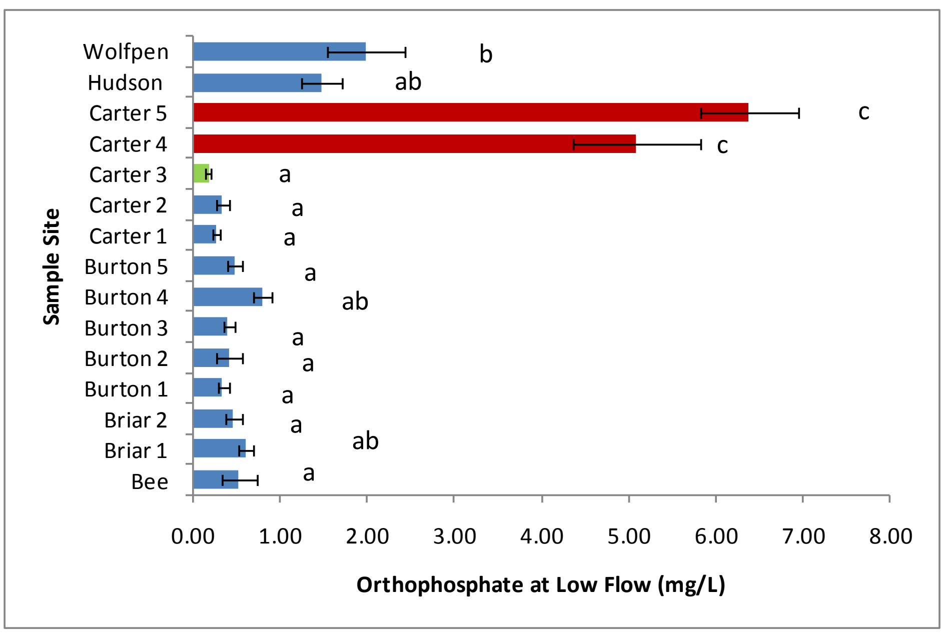

Figure 11. Mean annual orthophosphate concentrations during low flow. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

3.1.4.2. High Flow Conditions

Carter 5, at the bottom of the watershed studied, showed a significantly higher concentration of orthophosphate during high flow compared to all sample sites except Carter 4. Carter 4 was shown to be statistically similar to Carter 5, Hudson, Wolfpen, Burton 4 and Briar 1, but higher than the remaining nine watersheds (Figure 12).

3.1.4.3. All Flow Conditions

Mean annual orthophosphate concentrations showed a similar pattern, with the significantly highest concentration in Carter 5, second highest in Carter 4, and next highest in Wolfpen. Although Wolfpen Creek was not significantly different from Bee, Burton 4 or Hudson, it was higher than the remaining nine watersheds (Figure 13).

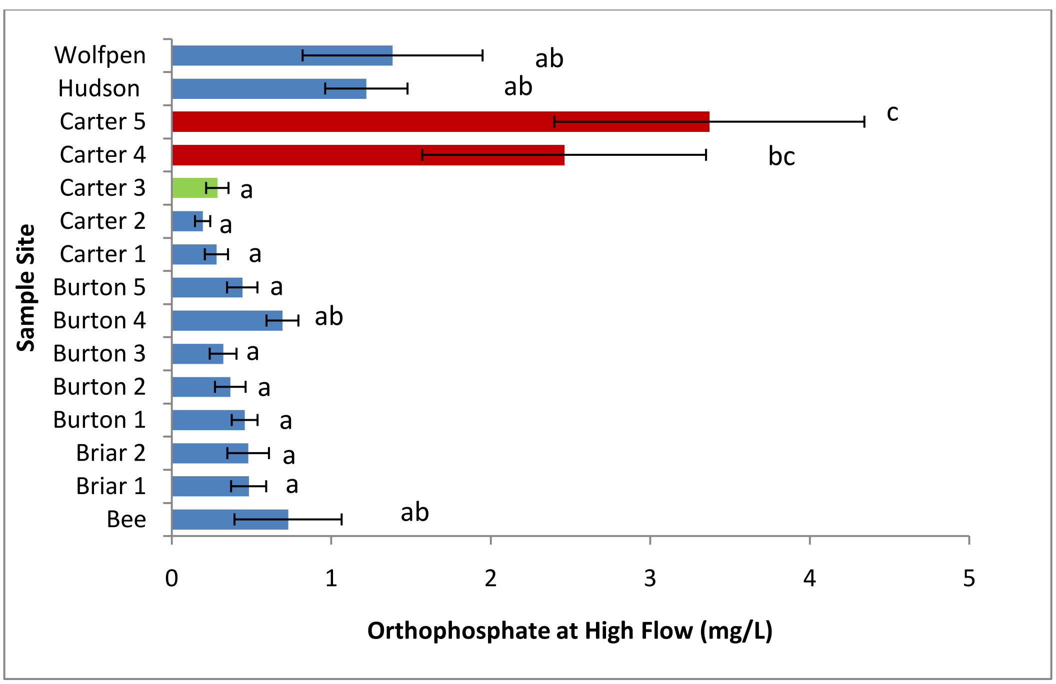

Figure 12. Mean orthophosphate concentrations during high flow conditions. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

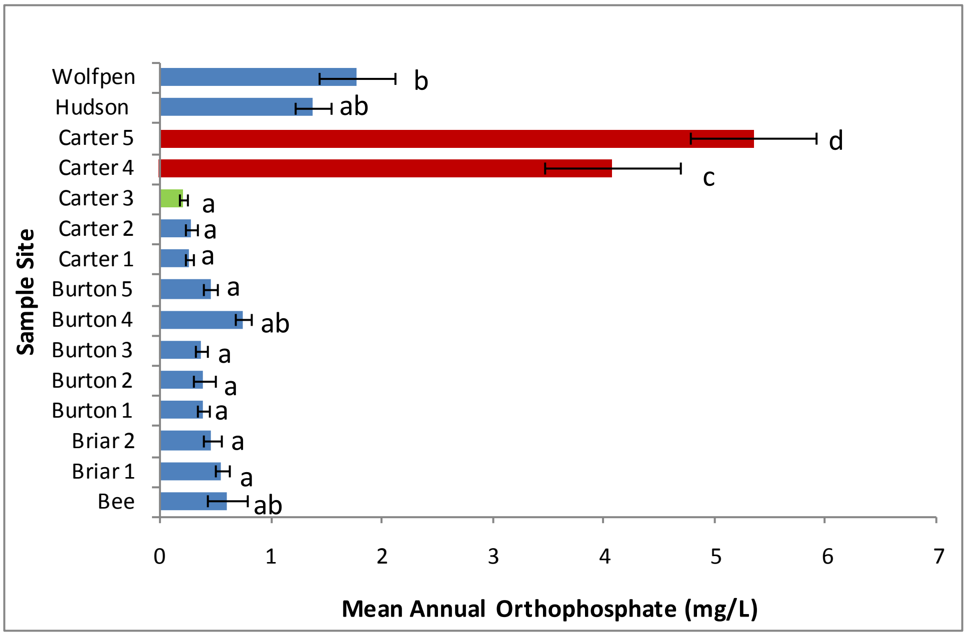

Figure 13. Mean annual orthophosphate concentrations for all flow. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

3.1.4.4. Seasonal Variation

Seasonal means of orthophosphate among creeks vary somewhat, but still indicate higher concentrations downstream of WWTPs, as reflected in low, high and all flow above. In spring and summer, Carter 4 and 5 were significantly higher than any of the other creeks, while in winter Carter 5 was higher than all others except Carter 4 and Wolfpen. Carter 4 and 5 were also highest in the fall, with the addition of higher concentrations in Hudson than in Carter 3 (Figure 14).

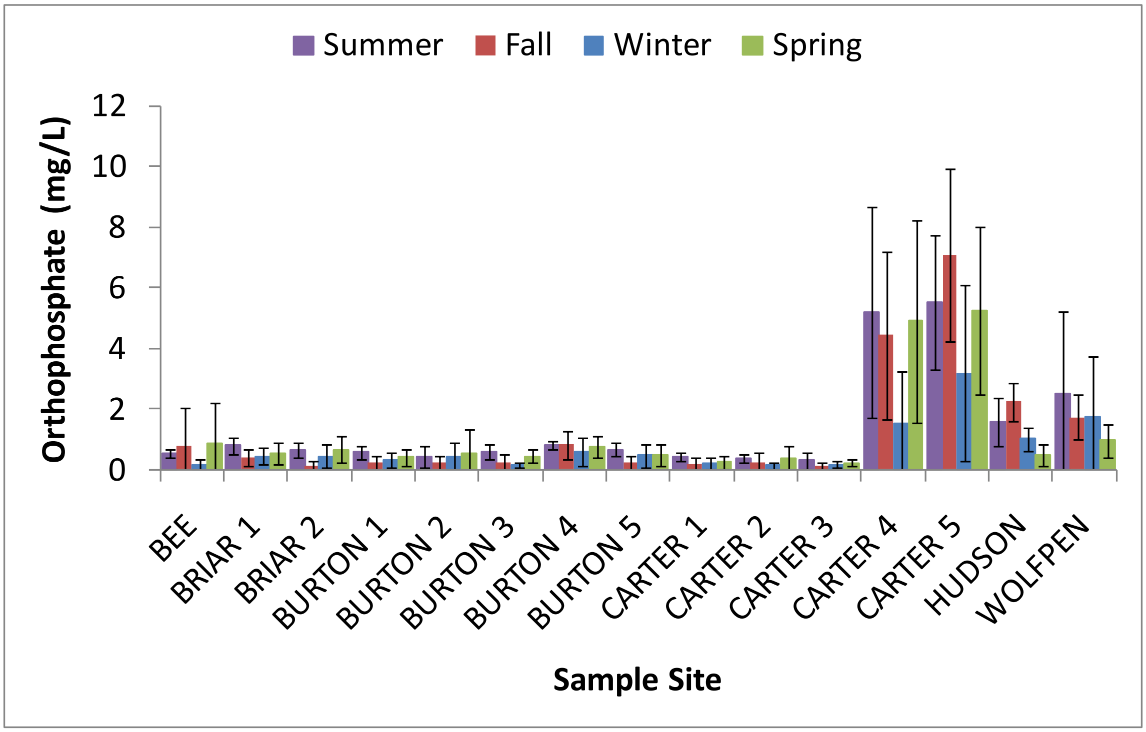

Figure 14. Seasonal orthophosphate. Error bars are standard error of the mean.

3.1.5. Cations: Sodium, Potassium, Magnesium and Calcium

One of the major characteristics of the local irrigation water is the high concentration of sodium. Because the local distribution water source is a deep groundwater well containing high bicarbonate-alkalinity and high sodium, soil irrigated with this water has a high sodium saturation. When the sodium has no more cation exchange sites available, the excess sodium leaches into the shallow groundwater and leaves a variable irrigation water "signature" in the surface water streams. Though in less abundance, the other cations, namely potassium, magnesium and calcium, serve an important purpose in both soil and water systems as nutrients necessary for plant and microbial growth.

3.1.5.1. Low Flow Conditions

During low flow conditions, there were several differences in sodium concentrations among the creeks. Wolfpen was significantly higher in sodium than all other watersheds besides Carter 5, probably from its high amount of irrigation water from the Texas A&M golf course (Figure 15). Carter 4 and 5 were higher than Briar 2, Burton 4, Carter 1 and Carter 3, all sites closer to the top of their respective watersheds that have not perhaps collected as much of the tap/irrigation signature. Carter 3, with the lowest amount, was the most rural site and thus the least impacted by the sodic tap water.

As a major nutrient and electrolyte for humans, potassium has a high turnover rate in urban environments and is often seen to increase in wastewater. True to form, potassium concentrations were significantly higher in Carter 4 and 5 than in Wolfpen and in Burton 1, 3 and 4 (Figure 16). Carter 5 was significantly higher than all Burton and Briar sample sites as well as Hudson and Wolfpen Creeks.

Low flow magnesium concentrations, unlike nitrate, orthophosphate, sodium and potassium concentrations, showed Carter 4 with significantly lower concentrations than Carter 3, Burton 3 and Bee Creek (Figure 17). On the other end of the scale, Bee Creek was higher than all other sites besides Burton 3, which was higher than all remaining sites besides Carter 3.

Calcium concentrations were significantly higher in Bee Creek at 20.4 mg/L than in Burton 2 at 9.4 mg/L, but no other differences among the creeks were observed (Table 14).

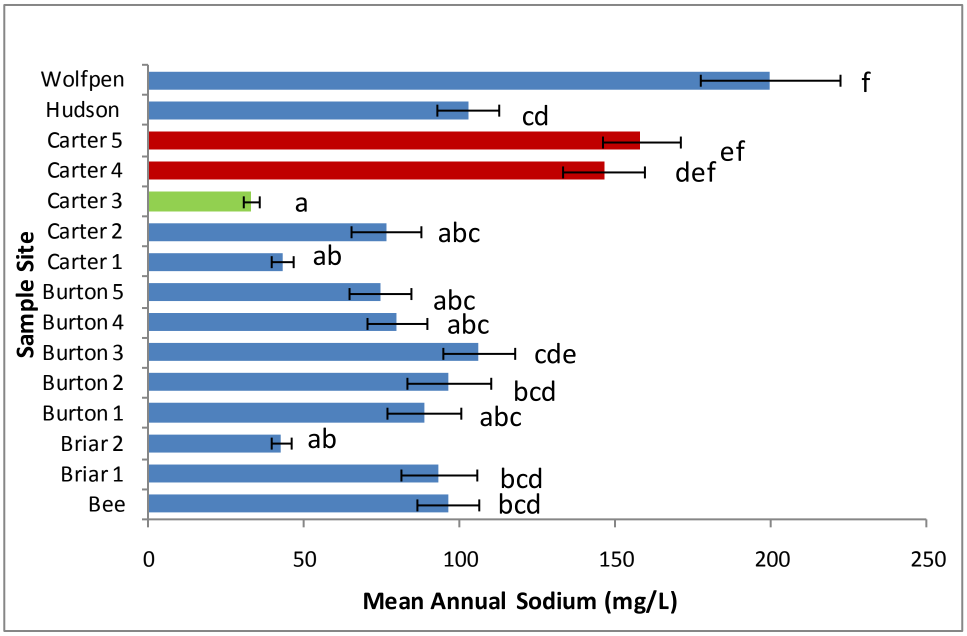

Figure 15. Mean annual sodium concentrations at low flow. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

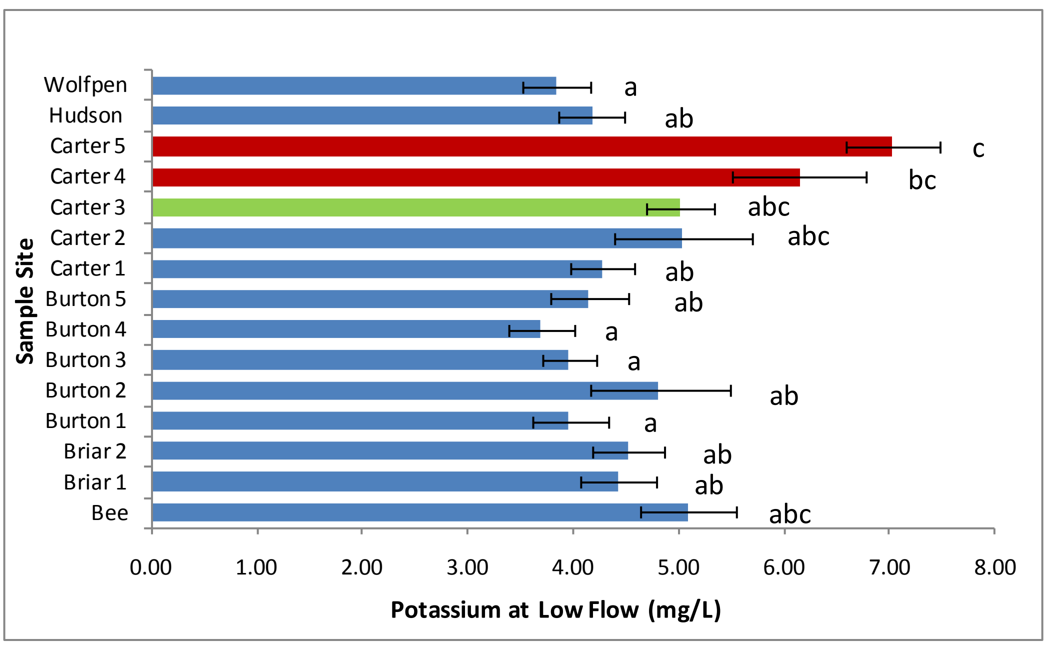

Figure 16. Mean annual potassium concentrations at low flow. Different letters indicate significant difference at $p \leq 0.05$.

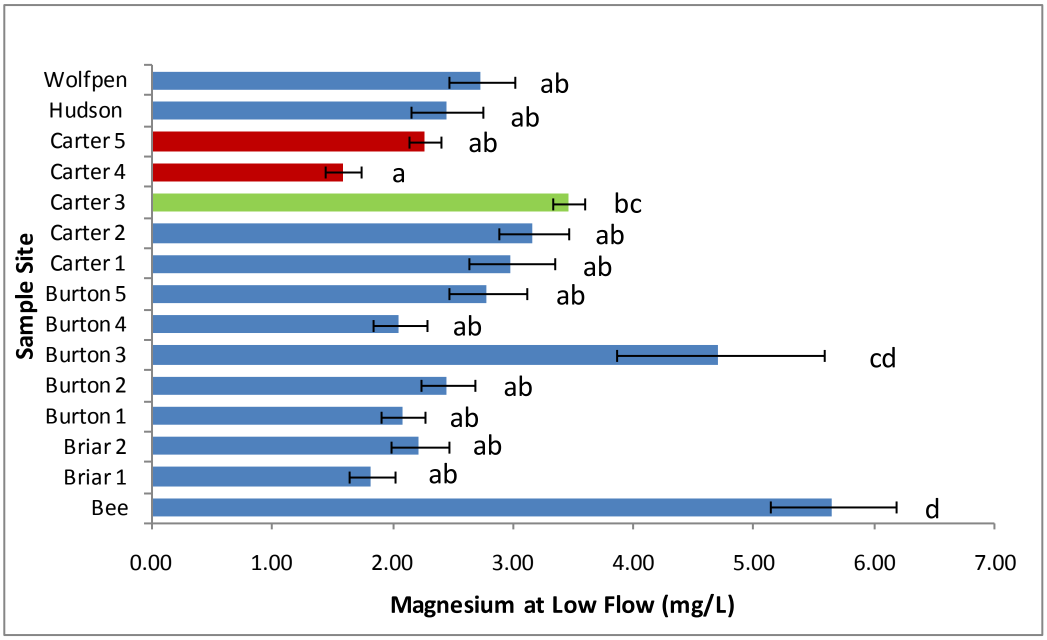

Figure 17. Mean annual magnesium concentrations at low flow. Error bars are standard error of the mean. Different letters indicate significant difference at p ≤ 0.05.

Table 14. Mean annual calcium concentrations during low flow. Different letters signify a significant difference.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	16	[b]20.38	15.22	3.81	7.00	70.48
Briar 1	16	[ab]10.52	7.03	1.76	3.41	30.99
Briar 2	16	[ab]13.35	7.18	1.79	5.09	32.21
Burton 1	16	[ab]11.17	6.99	1.75	4.94	33.25
Burton 2	16	[a]9.35	6.80	1.70	3.28	31.51
Burton 3	16	[ab]16.71	20.74	5.19	3.27	92.31
Burton 4	16	[ab]12.26	7.37	1.84	6.73	36.15
Burton 5	16	[ab]13.09	6.63	1.66	5.42	30.12
Carter 1	17	[ab]14.50	7.72	1.87	6.01	32.65
Carter 2	16	[ab]11.67	9.75	2.44	3.16	38.71
Carter 3	15	[ab]15.97	6.94	1.79	7.3	34.86
Carter 4	16	[ab]10.51	3.09	0.77	5.55	17.71
Carter 5	17	[ab]11.56	3.87	0.94	5.27	22.42
Hudson	16	[ab]13.62	5.88	1.47	5.34	30.28
Wolfpen	17	[ab]11.20	5.93	1.44	5.27	30.4

3.1.5.2. High Flow Conditions

During high flow conditions, cations exhibited a diluted effect from the greater quantity of flowing water added to base flow concentrations. Wider standard error values from the varying precipitation amounts, intensities and intervals between precipitation and sampling generally decreased the significant differences among the sample sites.

For sodium during high flow conditions, Wolfpen Creek maintained the highest concentration and was still significantly higher than the Burton 1, 2, and 5; the Briar sites; and the upper Carter sites (Figure 18). Carter 4 and 5, both statistically similar to Wolfpen, were also higher than the upper Carter sites and Briar 2.

Potassium concentrations were higher in Carter 4 and 5 than in Briar 2; Burton 1, 2, 4 and 5; Carter 2 and Wolfpen (Figure 19). In addition, Carter 4 was significantly higher than Burton 3 and Hudson Creeks.

High flow magnesium concentrations remained greatest in Bee Creek, but were not significantly greater than Burton 3, the upper Carter sites, Carter 5 and Wolfpen (Figure 20). Bee Creek did have significantly more magnesium than both Briar sites; Burton 1, 2, 4 and 5; Carter 4 and Hudson. Burton 3 and Carter 5 were significantly higher than Briar 2, Burton 1, Burton 2 and Carter 4.

There were no significant differences among the creeks from high flow calcium concentrations. The mean annual concentrations ranged from 10.17 mg/L in Hudson Creek to 15.64 mg/L in Burton 3 (Table 15).

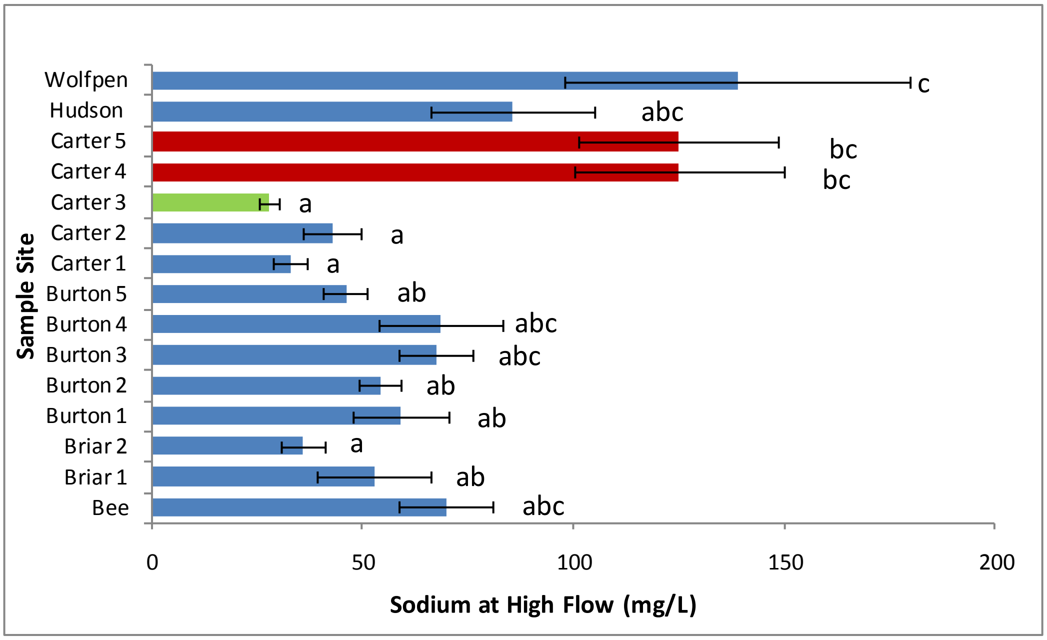

Figure 18. Mean annual sodium concentrations at high flow. Different letters indicate significant difference at p ≤ 0.05.

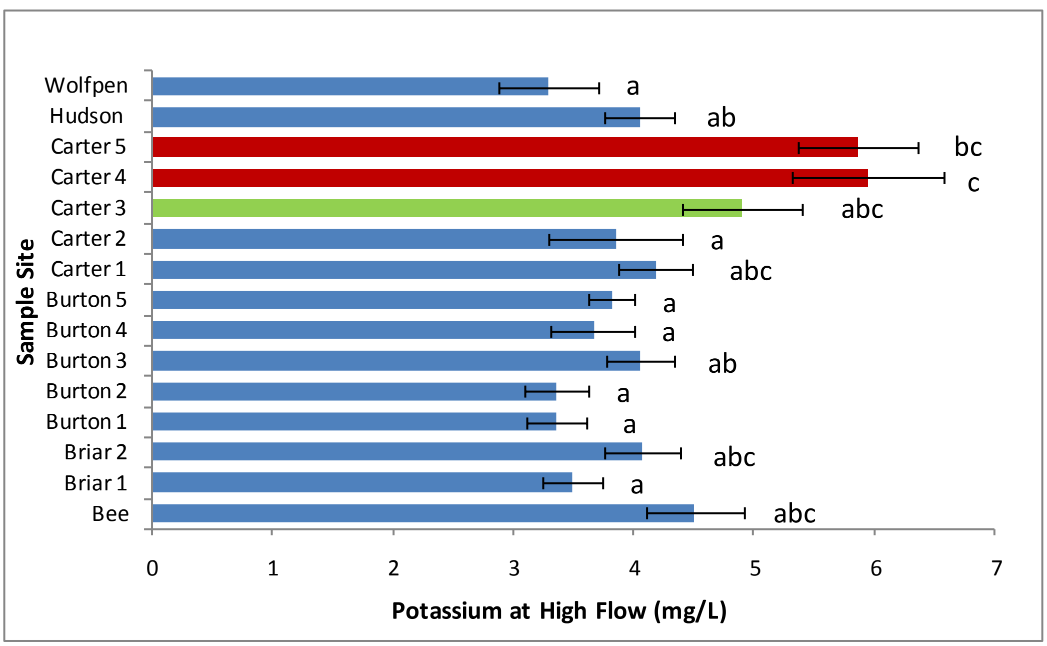

Figure 19. Mean annual potassium concentrations at high flow. Different letters indicate significant difference at p ≤ 0.05.

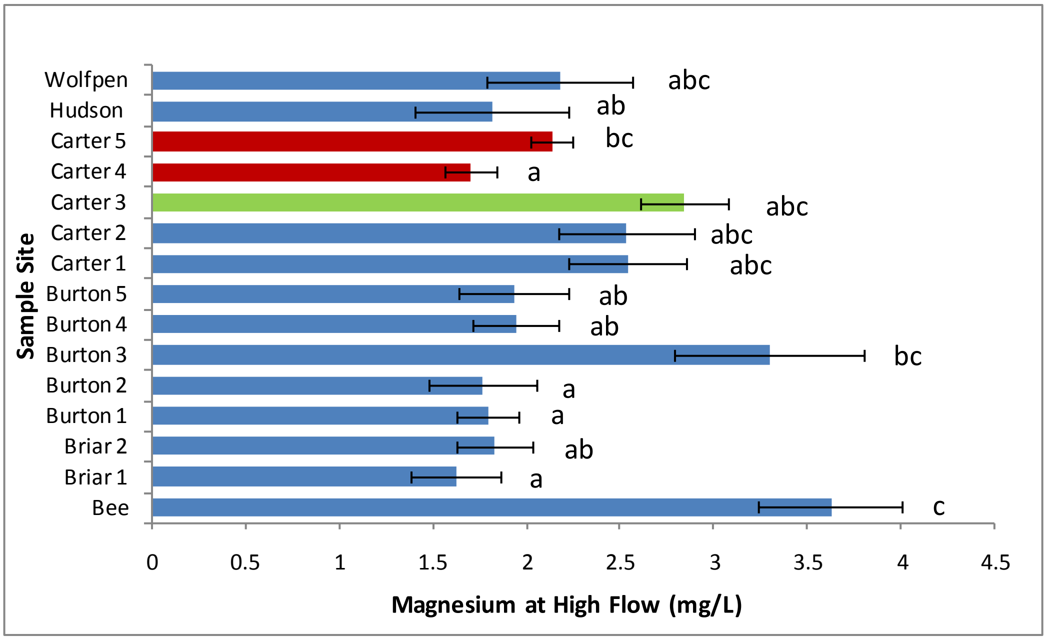

Figure 20. Mean annual magnesium concentrations at high flow. Different letters indicate significant difference at $p \leq 0.05$.

Table 15. Mean annual calcium concentrations during high flow. Equivalent letters indicate no significant difference.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	10	[a]14.79	6.25	1.98	3.61	24.67
Briar 1	10	[a]11.75	4.18	1.32	5.35	17.46
Briar 2	10	[a]12.79	4.84	1.53	5.59	18.63
Burton 1	10	[a]10.68	3.50	1.11	4.75	15.38
Burton 2	10	[a]11.00	3.52	1.11	5.91	16.91
Burton 3	10	[a]15.64	5.68	1.80	8.26	23.69
Burton 4	10	[a]13.33	4.43	1.40	7.88	20.56
Burton 5	10	[a]13.23	2.37	0.75	7.83	15.45
Carter 1	9	[a]12.73	4.25	1.42	8.24	22.15
Carter 2	9	[a]12.39	5.88	1.96	5.08	23.46
Carter 3	9	[a]14.57	3.30	1.10	7.93	18.89
Carter 4	10	[a]10.90	3.51	1.11	6.01	14.95
Carter 5	9	[a]12.09	2.57	0.86	7.62	15.16
Hudson	10	[a]10.17	4.94	1.56	2.59	20.00
Wolfpen	9	[a]10.67	2.98	0.99	6.55	15.28

3.1.5.3. All Flow Conditions

Mean annual cation concentrations showed similar patterns to high and low flow conditions. Sodium once again showed the highest concentrations in Wolfpen, followed by Carter 5 and Carter 4 (Figure 21). Burton 3 and Hudson also had higher concentrations of sodium than did Briar 2, Carter 1 and Carter 3.

Potassium concentrations in all flow conditions were significantly higher at Carter 4 and 5 than at the other sites, with the exceptions of Carter 2, Carter 3 and Bee (Figure 22).

Concentrations of magnesium during all flow were greatest in Bee Creek and Burton 3 (Figure 23). Magnesium concentrations in Carter 3 were significantly higher than Briar 1, Burton 1, Burton 4 and Carter 4. Carter 4 was the lowest, with a mean significantly lower than Carter 2, Carter 3, Burton 3 and Bee Creek.

Mean annual calcium concentrations were significantly greater in Bee Creek than in Burton 2 and Carter 4, but no other significant differences were found (Table 16).

3.1.5.4. Seasonal Variation

Even in summer, Wolfpen maintained a higher sodium level than Carter 3. Fall sodium concentrations were higher in Wolfpen than all other sites except Carter 5. Carter 5 was higher than Briar 2, Burton 5, Carter 1 and Carter 3; Carter 4 was also higher than Carter 3. In winter, Carter 4, Carter 5 and Wolfpen were higher than Carter 3, Carter 1 and Briar 2. In addition, Carter 5 was higher than Carter 2; Burton 1, 2, 4 and 5; and Briar 1; while Wolfpen was higher than these plus Hudson and Bee Creeks. In spring, Carter 4, Carter 5 and Wolfpen were higher than Carter 3 and Briar 2, while Carter 4 was also higher than Carter 1 and Wolfpen was higher than Carter 1 and Burton 1, 2, 4 and 5.   Carter 3 mean sodium concentration varied little seasonally, from 30.2 mg/L in spring to 37.6 mg/L in the fall.

Potassium was higher in Carter 4 and 5 than in Wolfpen during spring. Spring concentrations in Carter 5 were also higher than all the Burton sites and the upper Carter sites (1-3). In summer there was no significant difference among any of the sample sites. In winter, Carter 5 was higher than the other sites except for Carter 3 and 4, and those two sites were higher than Wolfpen and Burton 4. Fall potassium concentrations were significantly higher in Carter 5 than Hudson and Burton 1; Carter 3 was also higher than Burton 1.

Magnesium showed no significant difference among the creeks during the spring. In summer, Bee Creek was higher than the other sites except for Burton 3, Burton 5 and the upper Carter sites, but no other differences were observed. In the fall, Bee was higher than all the others except Carter 2, Carter 3 and Burton 3. Fall Burton 3 concentrations were higher than in

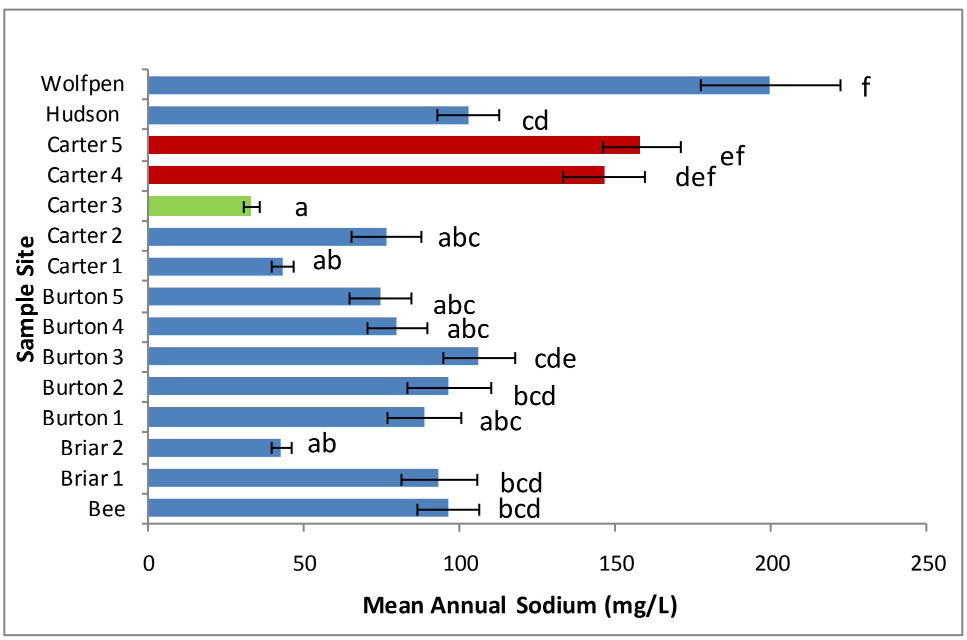

Figure 21. Mean annual sodium concentrations for all flow. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

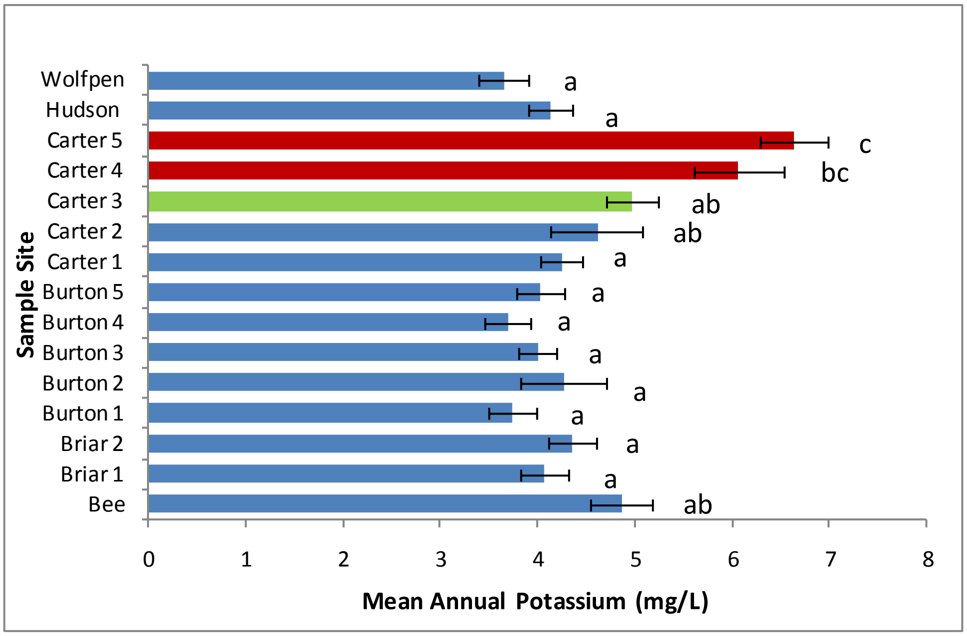

Figure 22. Mean annual potassium concentrations for all flow. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

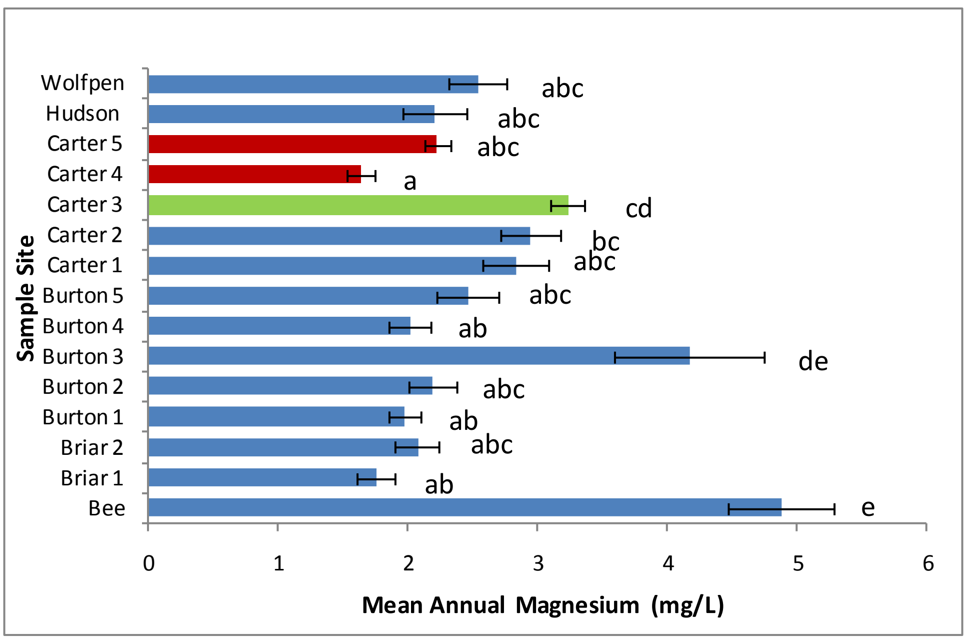

Figure 23. Mean annual magnesium concentrations for all flow. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

Table 16. Mean annual calcium concentrations for all flow. Equivalent letters indicate no significant difference at the $p \leq 0.05$ level.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	10	[a]14.79	6.25	1.98	3.61	24.67
Briar 1	10	[a]11.75	4.18	1.32	5.35	17.46
Briar 2	10	[a]12.79	4.84	1.53	5.59	18.63
Burton 1	10	[a]10.68	3.50	1.11	4.75	15.38
Burton 2	10	[a]11.00	3.52	1.11	5.91	16.91
Burton 3	10	[a]15.64	5.68	1.80	8.26	23.69
Burton 4	10	[a]13.33	4.43	1.40	7.88	20.56
Burton 5	10	[a]13.23	2.37	0.75	7.83	15.45
Carter 1	9	[a]12.73	4.25	1.42	8.24	22.15
Carter 2	9	[a]12.39	5.88	1.96	5.08	23.46
Carter 3	9	[a]14.57	3.30	1.10	7.93	18.89
Carter 4	10	[a]10.90	3.51	1.11	6.01	14.95
Carter 5	9	[a]12.09	2.57	0.86	7.62	15.16
Hudson	10	[a]10.17	4.94	1.56	2.59	20.00
Wolfpen	9	[a]10.67	2.98	0.99	6.55	15.28

Carter 4, Carter 5, Hudson, Burton 1, Burton 4 and the Briar sites, while Carter 3 was higher than Carter 4, Burton 1, Briar 1 and Hudson. Bee and Burton 3 were also highest among the sites in winter, while Carter 2 and 3 were only significantly higher than Carter 4.

Calcium concentrations in spring and summer showed no significant difference. In the fall, Carter 3 was higher in calcium than Briar 1, Burton 1 and 2, Carter 2 and Hudson. Bee Creek had the second highest fall mean value, but was not significantly different from any of the sites. Winter calcium concentrations were greater in Bee Creek than in Carter 4.

3.1.6. Anions: Chloride, Fluoride, Bromide and Sulfate

3.1.6.1. Low Flow Conditions

The previous section reported concentrations of cations, which are attracted to the negatively charged cation exchange sites on the surface of soil particles. Anions, on the other hand, do not attach very strongly to soil or particulate matter because of their negative charge, and are often observed in leachate water.

Under low flow conditions, chloride concentrations were highly variable among creeks. Chloride in Burton 3 exceeded that in Carter 1, Carter 3 and Briar 2 (Figure 24), all sites close to the top of their respective subcatchments with the lowest chloride concentrations. In addition, Carter 5 was higher than Burton 4, Burton 5 and Hudson, and Carter 4 was higher than these three plus Carter 2. There was no significant accumulation from Carter 4 to Carter 5, possibly because the wastewater treatment plant upstream of Carter 5 uses UV-radiation for disinfection rather than an additional amount of chloride. As the site with the greatest chloride concentration, Wolfpen Creek was significantly higher than in Hudson; the upper Carter sites (Carter 1-3); both Briar sites; and Burton 1, 2, 4 and 5.

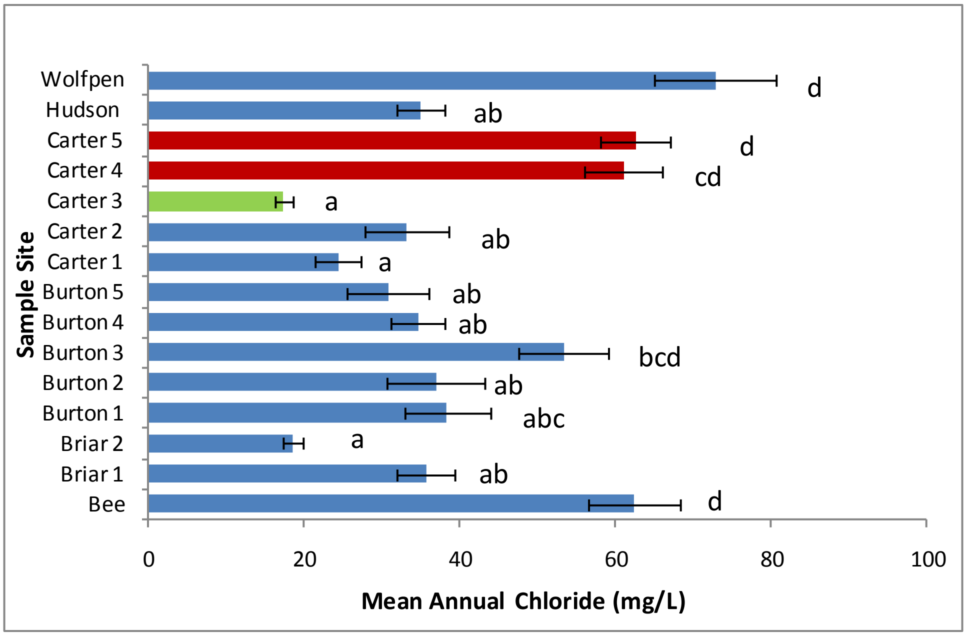

Figure 24. Mean annual chloride concentrations at low flow. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

Fluoride concentrations during low flow were highest in Carter 4 and 5 downstream of wastewater treatment plants (Figure 25). Hudson Creek was next highest, with higher concentrations than the upper Burton sites (3-5), upper Carter sites (1-3), and Briar 2. Wolfpen was also higher in fluoride than Carter 3 and Briar 2.

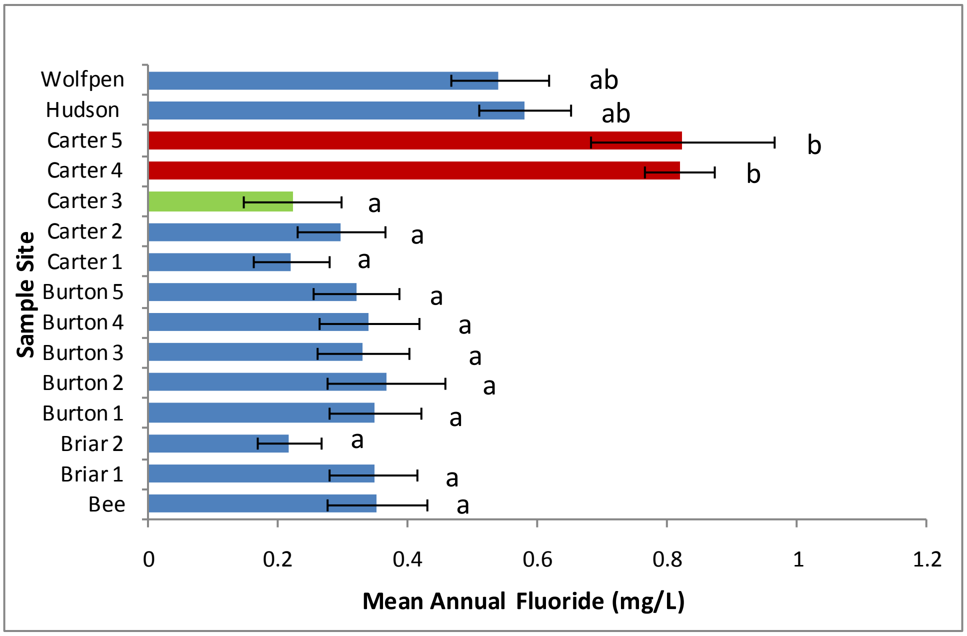

Figure 25. Mean annual fluoride concentrations at low flow. Different letters indicate significant difference at $p \leq 0.05$.

Unlike the low flow fluoride stream concentrations, bromide was higher in Wolfpen than in any other watershed sampled (Figure 26). Burton 5 was also higher than Briar 2 and Carter 3, possibly the effect of industry in the Burton 5 drainage basin. There was no significant difference between sites below wastewater treatment plants and most other urban watersheds.

Low flow sulfate concentrations were also highly variable. Bee creek sulfate was greater than in every other site besides Wolfpen (Figure 27). Hudson and Burton 3 were significantly higher than Briar 1 and 2, Burton 4 and the upper Carter sites. Carter 5 and Burton 5 followed, with higher concentrations than the Briar sites, Burton 4 and Carter 1 and 3. Carter 4 was higher than Briar 2, Burton 4 and Carter 1 and 3, while Burton 1 was higher than Briar 2 only.

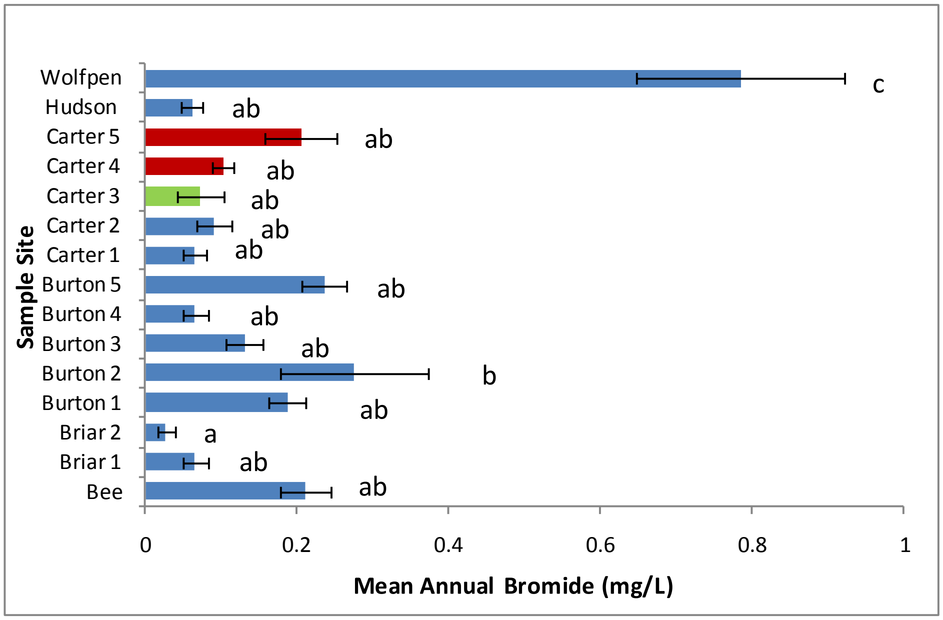

Figure 26. Mean annual bromide concentrations at low flow. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

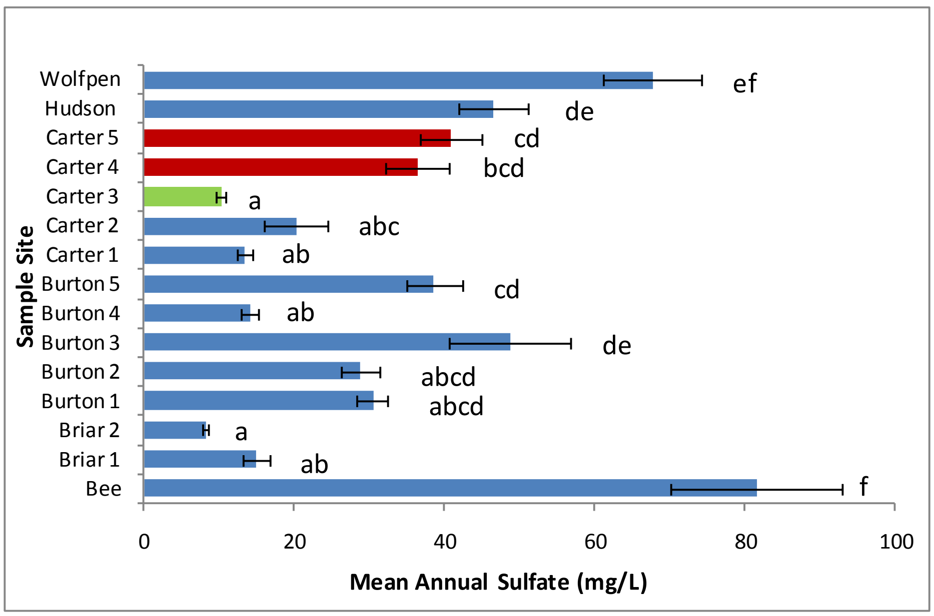

Figure 27. Mean annual sulfate concentrations at low flow. Error bars are standard error of the mean. Different letters indicate significant difference at $p \leq 0.05$.

3.1.6.2. High Flow Conditions

There were no significant differences in high flow fluoride concentrations among the creeks sampled. Concentrations ranged from 0.23 mg/L in Carter 1 to 0.68 mg/L in Carter 4 (Table 17).

Chloride concentrations during high flow were higher in Carter 5 than in Briar 2, Burton 5 and the upper Carter sites (Figure 28). Wolfpen was next highest, with significantly higher values than Briar 2, Burton 5 and Carter 3. Carter 4 was higher than Burton 5.

High flow bromide concentrations were significantly higher in Wolfpen than in Burton 3 and Burton 4; both Briar sites; Carter 1, 2, and 4; and Hudson (Figure 29).

Sulfate high flow concentrations were significantly higher in Bee Creek than in any of the other sites except Burton 3 and Wolfpen (Figure 30).

Table 17. Mean annual fluoride concentrations during high flow. Equivalent letters signify no significant difference at the $p \leq 0.05$ level.

Site Name	N	Mean	Std Dev	Std Error	Minimum	Maximum
Bee	10	[a]0.31	0.35	0.11	0.1	1.27
Briar 1	10	[a]0.43	0.55	0.17	0.06	1.61
Briar 2	10	[a]0.32	0.39	0.12	0.05	1.21
Burton 1	10	[a]0.43	0.55	0.18	0.1	1.74
Burton 2	10	[a]0.50	0.71	0.22	0.1	2.04
Burton 3	10	[a]0.45	0.56	0.18	0.07	1.68
Burton 4	10	[a]0.51	0.60	0.19	0.06	1.71
Burton 5	10	[a]0.43	0.53	0.17	0.11	1.44
Carter 1	9	[a]0.23	0.35	0.12	0.06	1.17
Carter 2	9	[a]0.39	0.52	0.17	0.03	1.55
Carter 3	9	[a]0.38	0.59	0.20	0.06	1.74
Carter 4	10	[a]0.68	0.32	0.10	0.1	1.01
Carter 5	9	[a]0.65	0.49	0.16	0.21	1.82
Hudson	10	[a]0.54	0.49	0.15	0.07	1.73
Wolfpen	9	[a]0.54	0.61	0.20	0.17	2.11

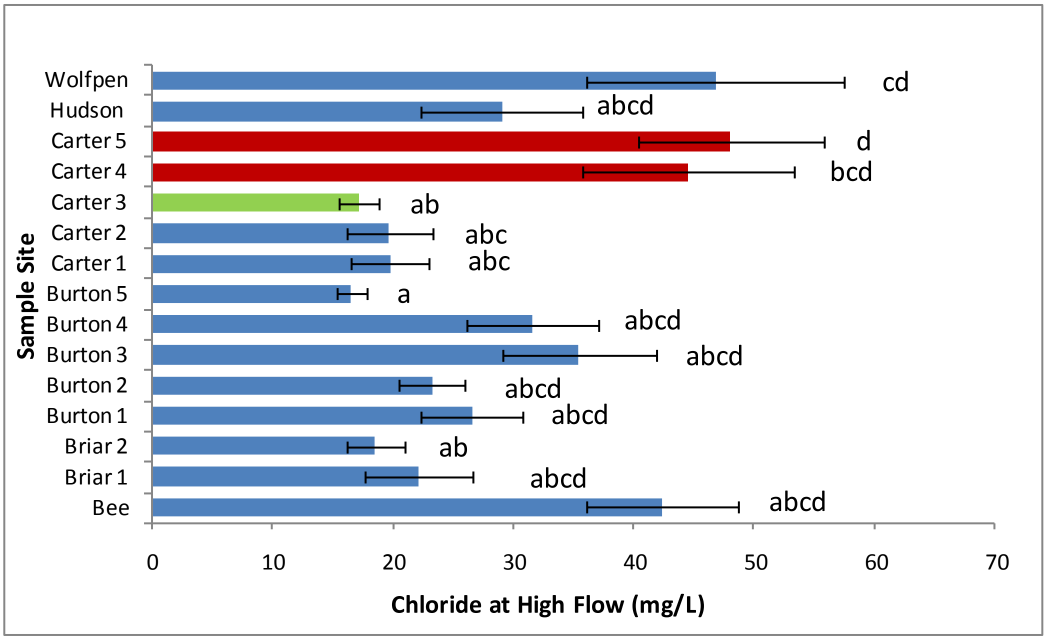

Figure 28. Mean annual chloride concentrations during high flow conditions. Error bars are standard error of the mean. Different letters signify a significant difference between mean concentrations.

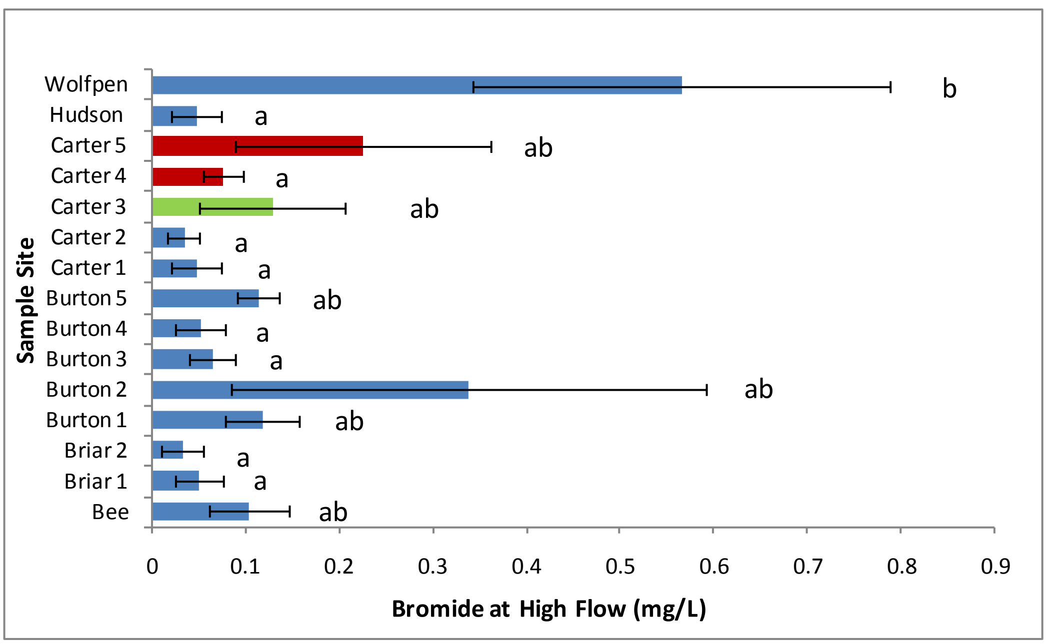

Figure 29. Mean annual bromide concentrations during high flow conditions. Error bars are standard error of the mean. Different letters signify a significant difference between mean concentrations.

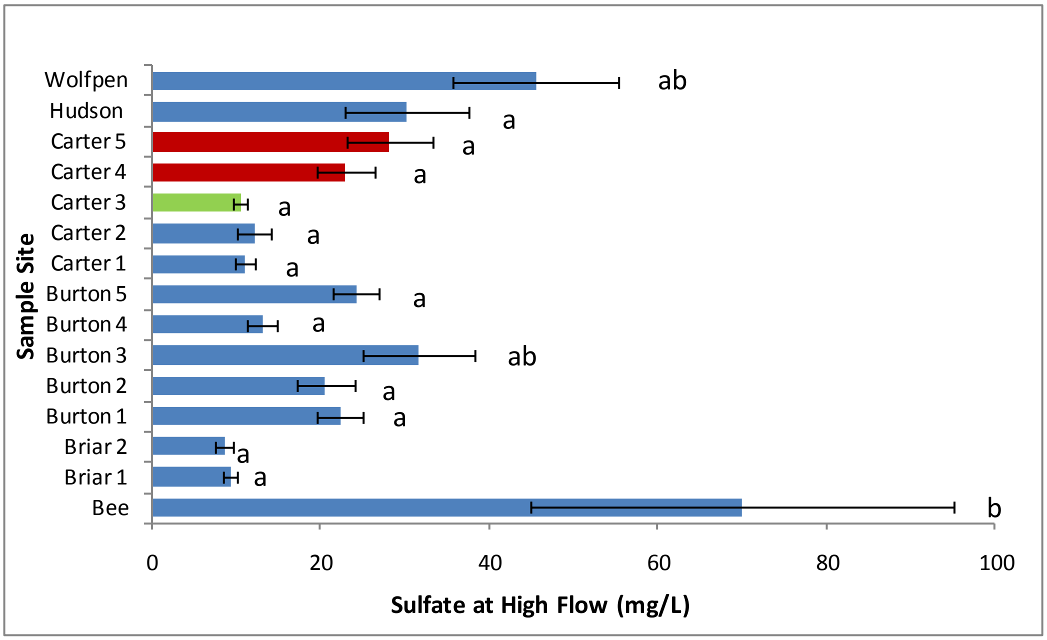

Figure 30. Mean annual sulfate concentrations during high flow conditions. Error bars are standard error of the mean. Different letters signify a significant difference between mean concentrations.

3.1.6.3. All Flow Conditions

When combining high and low flow data, chloride concentrations in all flow were greater in Carter 5, Bee and Wolfpen than in all the sites except for Burton 3 and Carter 4 (Figure 31). Carter 4 was significantly higher than both Briars; Burton 2, 4 and 5; Hudson; and the upper Carter sites. Burton 3 was higher than Briar 2, Carter 1 and Carter 3.

Fluoride concentrations were significantly higher in Carter 4 and 5 than the other sites, with exception of Hudson and Wolfpen, reflecting the irrigation water footprint (Figure 32).

Mean annual bromide concentrations remained higher in Wolfpen than in any other creek sampled. Burton 2 was also significantly higher than Briar 2, but no other significant differences were discovered (Figure 33).

All flow sulfate concentrations were statistically higher in Bee Creek than in any other site besides Wolfpen (Figure 34). Burton 3 and Hudson were not significantly different from Wolfpen, but were higher than the Briar sites, the upper Carter sites and Burton 4. Burton 5 and Carter 5 were also higher than the Briar sites, Burton 4, Carter 1 and Carter 3, while Carter 4 was only higher than Briar 2 and Carter 3. Interestingly, the sites downstream of the wastewater treatment plant were not significantly different from most of the Burton sites, Carter 2 or Hudson, suggesting that sulfate may be an irrigation-water characteristic but is not significantly elevated by wastewater treatment in this watershed.

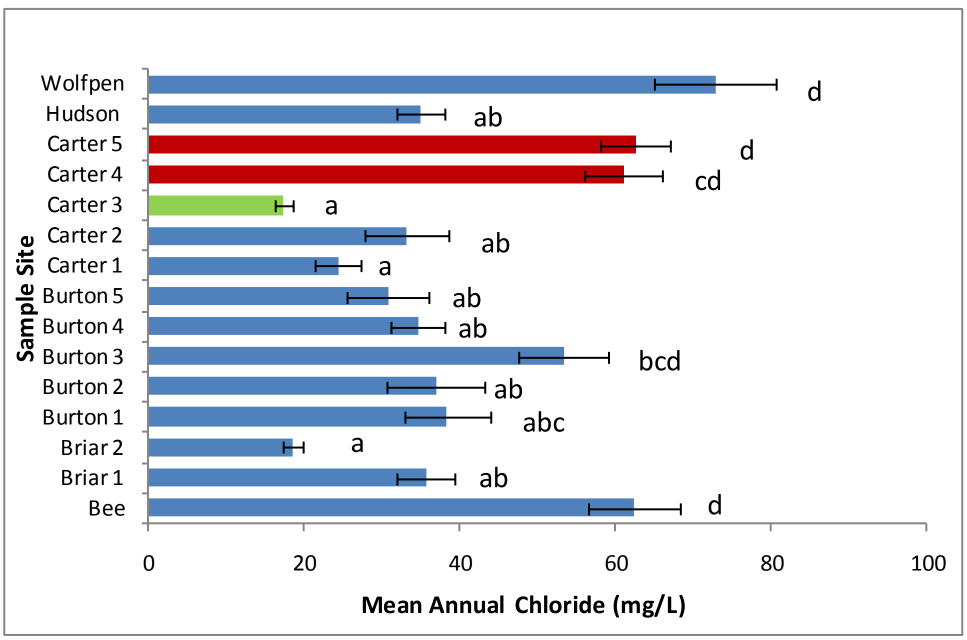

Figure 31. Mean annual chloride concentrations during all flow conditions. Error bars are standard error of the mean. Different letters signify a significant difference between mean concentrations.

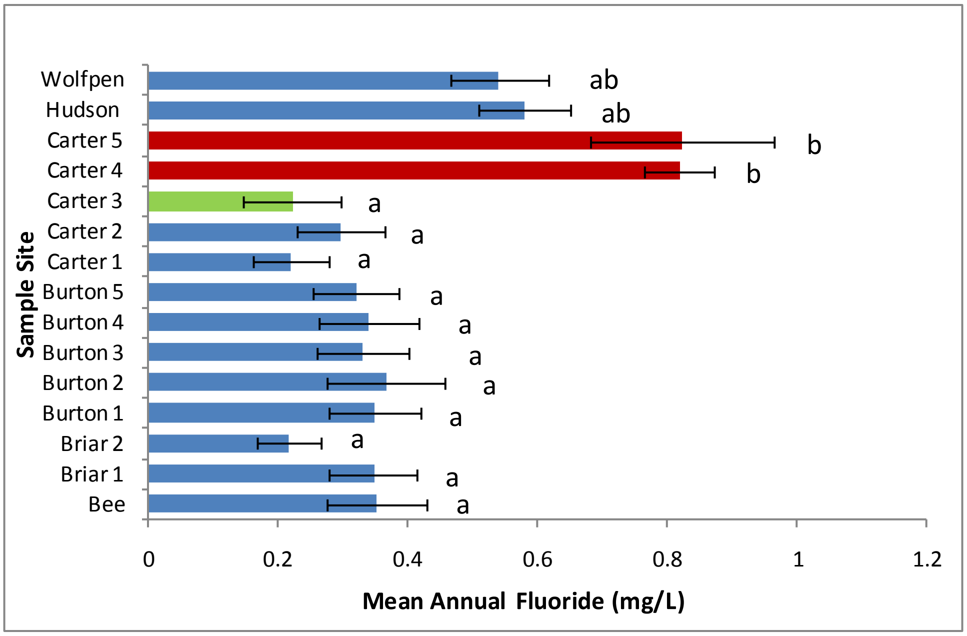

Figure 32. Mean annual fluoride concentrations during all flow conditions. Error bars are standard error of the mean. Different letters signify a significant difference between mean concentrations.

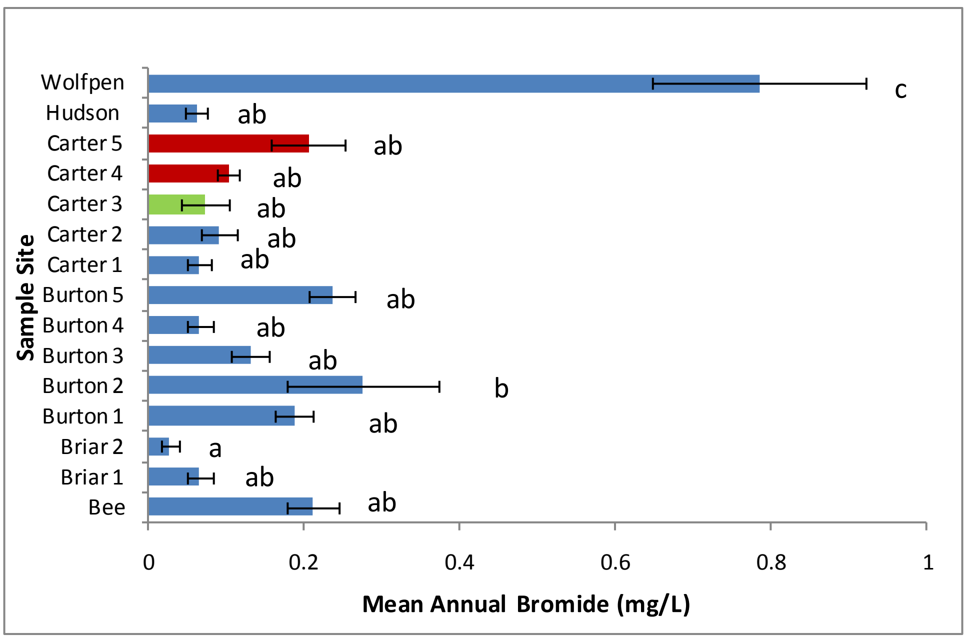

Figure 33. Mean annual bromide concentrations during all flow conditions. Error bars are standard error of the mean. Different letters signify a significant difference between mean concentrations.

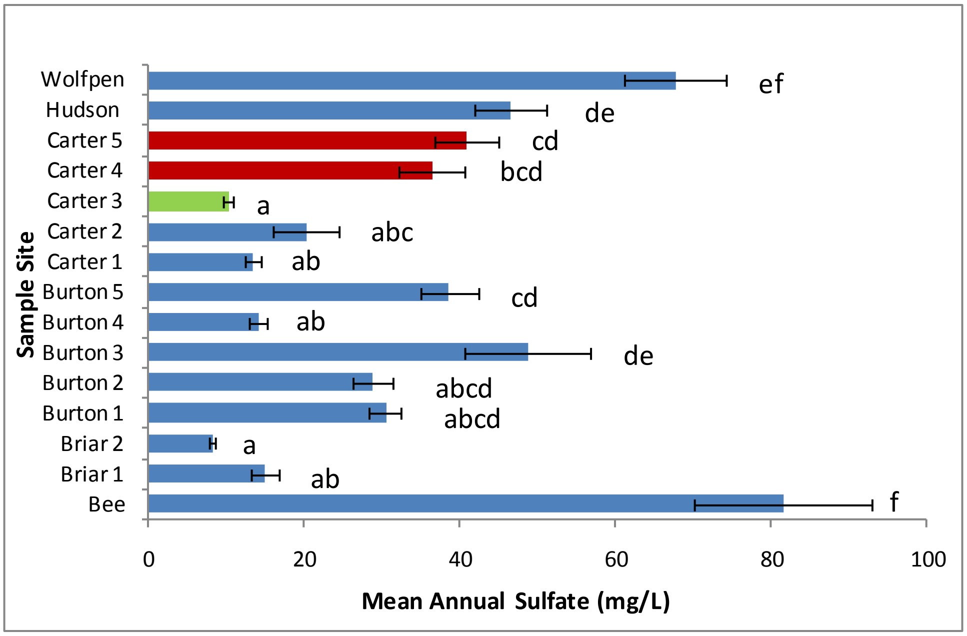

Figure 34. Mean annual sulfate concentrations during all flow conditions. Error bars are standard error of the mean. Different letters signify a significant difference between mean concentrations.

3.1.6.4. Seasonal Variation

Fluoride concentrations in summer showed significantly higher levels in Carter 4 and 5 than in the upper Carter sites (1-3) and Briar 2. Hudson Creek was also higher than Carter 3 in summer. In the fall, Carter 4, Carter 5 and Hudson were significantly higher than all other sites except Wolfpen, which was in turn higher than the upper Carter sites, both Briar sites, and Burton 2, 3, 4 and 5. Winter fluoride concentrations were higher in Carter 4 and 5 than in all others except Hudson and Wolfpen, with Wolfpen showing higher concentrations than Carter 1, Carter 3 and Briar 2, and Hudson only higher than Carter 3. There were no significant differences in spring, but Carter 4 and 5 maintained the highest mean values.

Chloride concentrations for the summer sample times showed no significant difference. In the fall, however, there were several differences, with Wolfpen containing higher concentrations than Hudson, Briar 1, Briar 2, Burton 2, Burton 5, Carter 1 and Carter 3. Carter 5 was also higher than Carter 1, Carter 3 and Briar 2, while Bee Creek was higher than Briar 2 and Carter 3, and Carter 4 was higher than Carter 3. In winter, Wolfpen, Burton 3 and Carter 5 were higher than all the other creeks except for Bee Creek and Carter 4. These two were higher than Briar 2 and Carter 3. Spring chloride values were greater in Wolfpen than in Burton 5 or Briar 2.

Wolfpen maintained higher bromide concentrations than any other site throughout summer, fall and winter seasons, with mean values more than twice as high as the next highest site. In spring, Wolfpen remained significantly higher than Burton 4, Carter 1 and the Briar sites, but was not significantly different from the other creeks sampled.

Summer sulfate concentrations were higher in Bee Creek than in all the other creeks except for Burton 5, Hudson and Wolfpen. Burton 5 and Hudson were higher than Burton 4, Carter 3 and Briar 2, while Wolfpen was only higher than Carter 3 and Briar 2. In the fall, Wolfpen and Bee were higher than Carter 1, Carter 3, Burton 2 and the Briar sites, while Wolfpen alone was also higher than Burton 1, Burton 2 and Carter 2. In winter, Bee was higher than all the other sites except Wolfpen and Burton 3. Wolfpen winter concentrations were higher than Carter 1, Carter 3 and the Briar sites; Burton 3 was also higher than Briar 2. Bee Creek was also higher in the spring than Burton 1, 2, 4, and 5; the upper Carter sites; and the Briar sites.

3.2. Geographical Distribution of *E. coli* in Carters Creek

3.2.1. *E. coli* Accumulation with River Miles

Counts of *E. coli* were found to be greatest in the Burton Creek subcatchments in both dry and wet conditions, as shown in Figures 35 and 36 (please note different *E. coli* scales). In dry conditions, Burton 4 has conspicuously high *E. coli* counts which likely become diluted, killed or settled out of suspension by the time the stream reaches Burton 2, and again before Burton 1.

In wet conditions the nested Burton sample sites are much more clustered together, indicative of the greater hydrologic connectivity and the flush of *E. coli* that reaches much farther downstream than in dry conditions. Also of note is the diluting effect of WWTP effluent on *E. coli* counts, especially during wet weather. Although there is no significant difference among any of the creeks during high flow, the high flow geometric mean for Carter 4 (downstream of a WWTP) was lower than that of any of the measured sites immediately upstream and less than half of the Burton 1 geometric mean.

3.2.2. Fluorescence in Burton Creek

Given the geographical pattern of elevated *E. coli* counts, the objective for the next part of the project was to walk the length of the sub-catchment producing the highest number of *E. coli* that did not appear to be affected by upstream counts, i.e. Burton 4, and identify any leaky sewer pipes. Although the target source for fluorescence at 445 nm was optical brighteners from laundry detergent, dissolved organic carbon has also been shown to fluoresce over a broad wavelength range that includes 445 nm. This caused some interference in Burton Creek because of the high concentrations of DOC in the stream. In fact, there was no apparent relationship (even qualitatively) between fluorescence and bacterial counts and no sudden jumps in *E. coli* and fluorescence together, as demonstrated by Figure 37 and 38. As a result, no evidence of leaking sewer pipes were found.

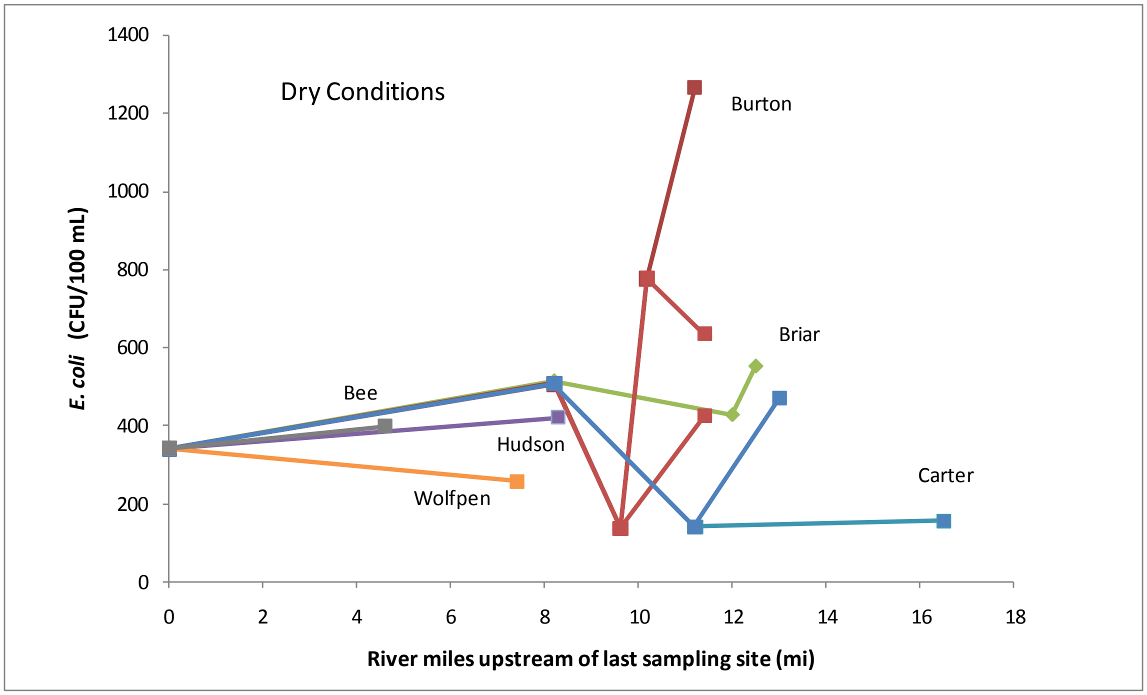

Figure 35. ***E. coli*** **with river miles upstream of Carter 5, during dry conditions.**

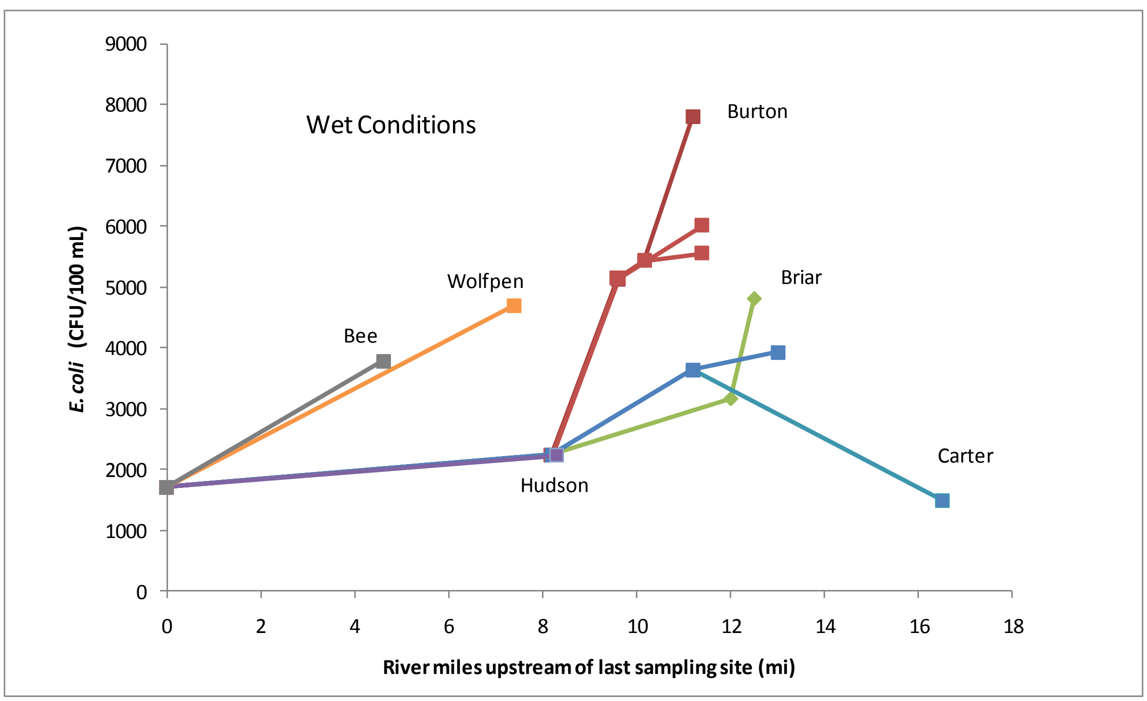

Figure 36. ***E. coli*** **with river miles upstream of Carter 5, during wet conditions.**

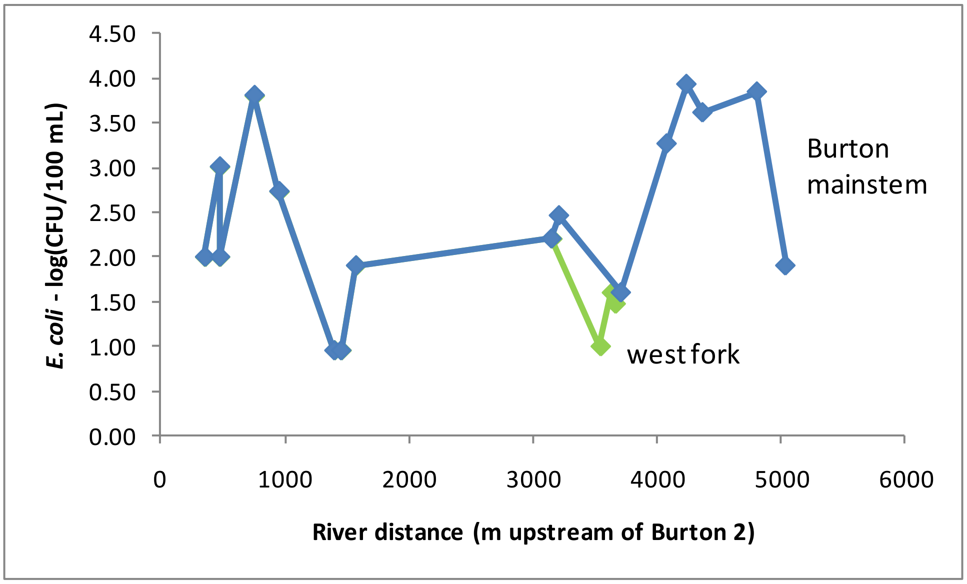

Figure 37. ***E. coli*** **logarithmic transformations upstream of Burton 2, during dry conditions.**

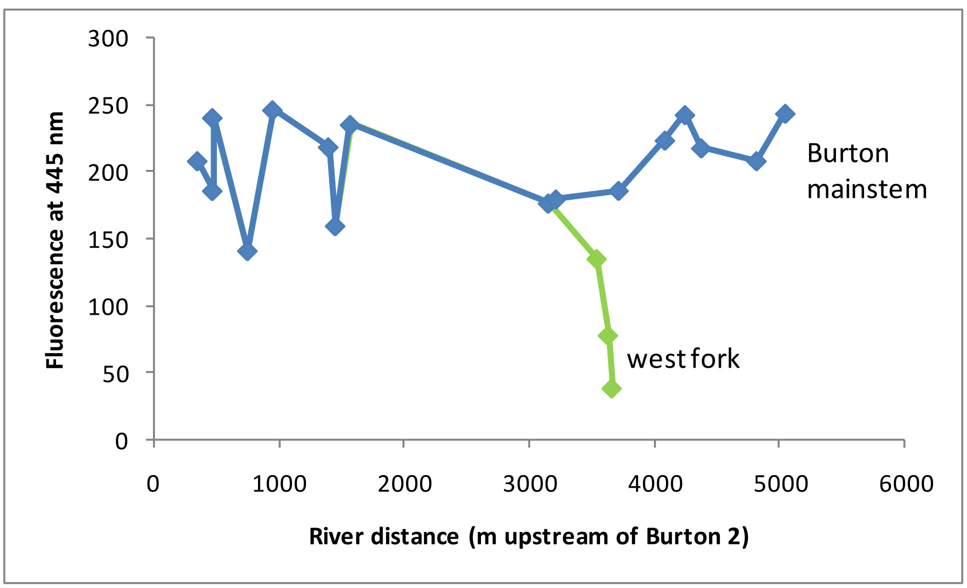

Figure 38. Fluorescence at 445 nm upstream of Burton 2, during dry conditions.

3.3 Correlations among *E. coli* and Nutrients without WWTPs

3.3.1. *E. coli*

During high flow, $\log_{10}$ values of *E. coli* counts could be predicted by a model of potassium and calcium as independent variables. The regression with the highest significance and greatest R^2 value (p=0.002; R^2=0.86) was discovered as:

$$\log_{10}(E.\ coli) = 0.777[\mathrm{Ca}] - 1.123[\mathrm{K}] + 4.517$$

There were no significant nutrient regressions found among the nine creek subset for any of the four seasons. However, this high-flow model indicates a positive in-stream association with calcium and a negative association with potassium following rain events.

3.3.2. Dissolved Organic Carbon

Dissolved organic carbon was significantly correlated with sodium in both summer (p=0.002) and fall (coincidentally, when DOC concentrations were highest; p<0.0001) and with the sodium adsorption ratio in both summer (p=0.002) and fall (p<0.0001). As shown in Figure 39, SAR explains 75% of DOC variance in summer and 93% in the fall.

3.3.3. Nitrogen Species: Nitrate, Nitrite, Ammonium and Dissolved Organic Nitrogen

No significant correlations were found between any nitrogen species and *E. coli*, anions, cations or DOC.

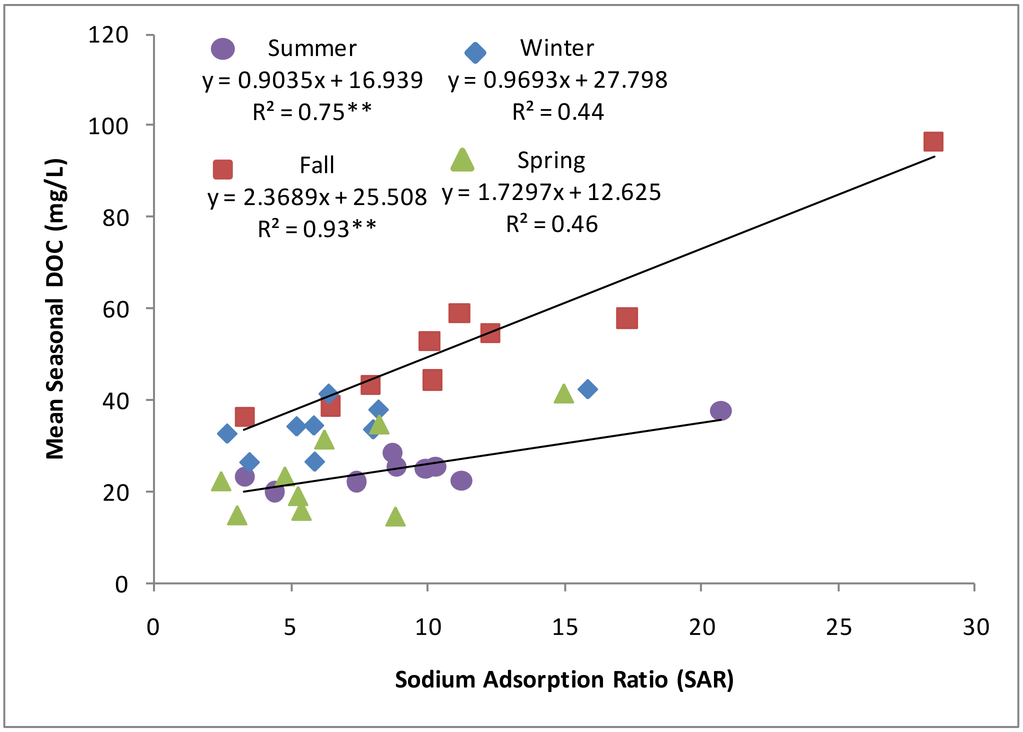

Figure 39. Seasonal linear regression of mean DOC vs. SAR. Two asterisks (**) indicate significance at the $p \leq 0.01$ level. Trend lines are not shown for non-significant regressions.

3.3.4. Orthophosphate

During summer, fall and winter, orthophosphate shows a significant correlation to fluoride concentrations, with p-values of 0.003, <0.0001 and 0.001, respectively (Figure 40). Spring phosphate also had a correlation with fluoride (p=0.007) if the outlier Briar 2 (circled in Figure 40) is left out. Phosphate is also correlated to SAR in summer, fall and winter (p=0.007, p=0.011, p=0.007, respectively; Figure 41); sodium in summer and fall (p=0.010, p=0.023, respectively); and DOC in summer (p <0.001).

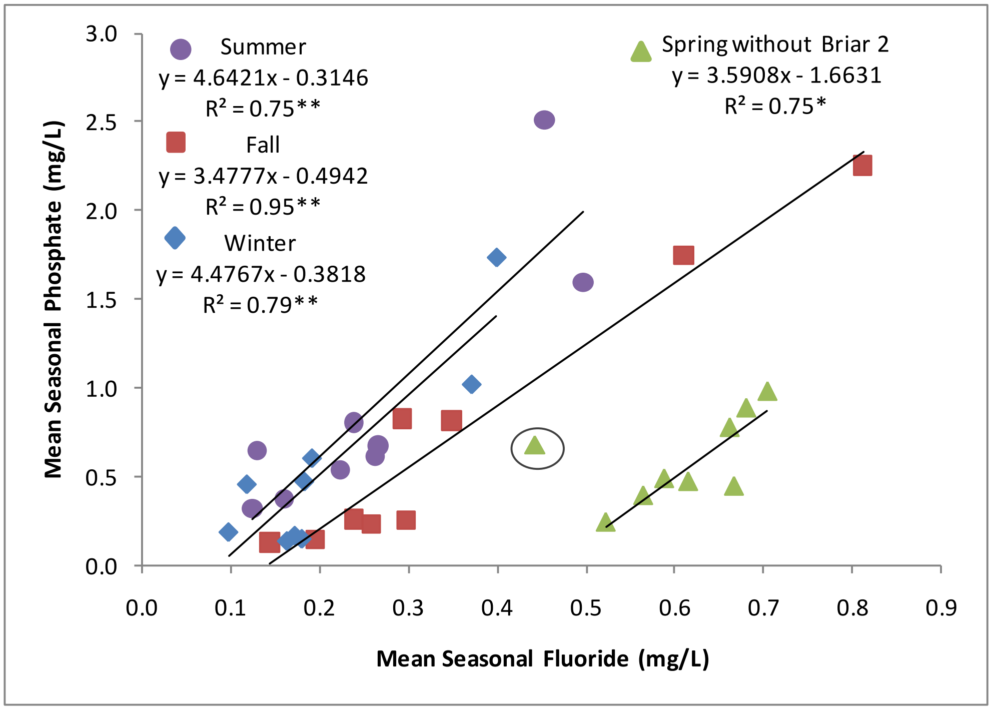

Figure 40. Seasonal linear regression of mean orthophosphate vs. fluoride. One asterisk (*) indicates significance at the $p \leq 0.05$ level; two asterisks (**) indicates significance at the $p \leq 0.01$ level. Trend lines are not shown for non-significant regressions.

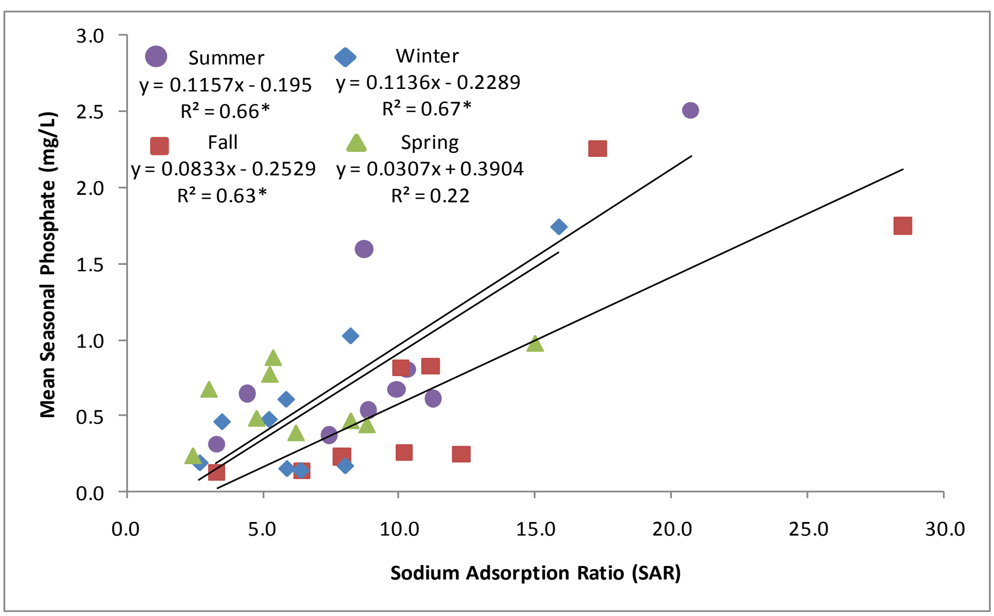

Figure 41. Seasonal linear regression of mean orthophosphate vs. SAR. One asterisk (*) indicates significance at the $p \leq 0.05$ level. Trend lines are not shown for non-significant regressions.

3.4. Correlations among Stream Water Constituents and Land Cover

Chemical and bacterial constituents in Carters Creek varied with the level of flow in the stream and with seasonal changes, as discussed above. Another causal factor in surface water chemistry was the different land uses established on the subcatchment watersheds.

3.4.1. *E. coli*

In spring, *E. coli* ($\log_{10}$ values) showed a significant positive correlation to total urban land use ($p=0.009$; Figure 42) and a significant negative correlation to total range land use ($p=0.012$). It is probable that these two correlations are merely inverses of each other, as urban and range land uses together account for 94.5-100% of the included subcatchments, but it was unclear which land use had a greater influence on *E. coli* in the stream until $\log_{10}$ values of *E. coli* were modeled by percent land use during high flow, as follows ($p<0.001$; $R^2=0.89$):

$$\log_{10}(E.\ coli) = 0.947(\%\ \text{total urban}) + 2.491$$

This equation showed *E. coli* to be significantly related to total urban land use when runoff from impervious surfaces into the stream was at its greatest.

3.4.2. Dissolved Organic Carbon

In summer and fall, dissolved organic carbon had a significant correlation ($p<0.001$ for both seasons) with deciduous forest LULC 41, which was represented almost entirely by urban riparian strips in the watersheds studied.

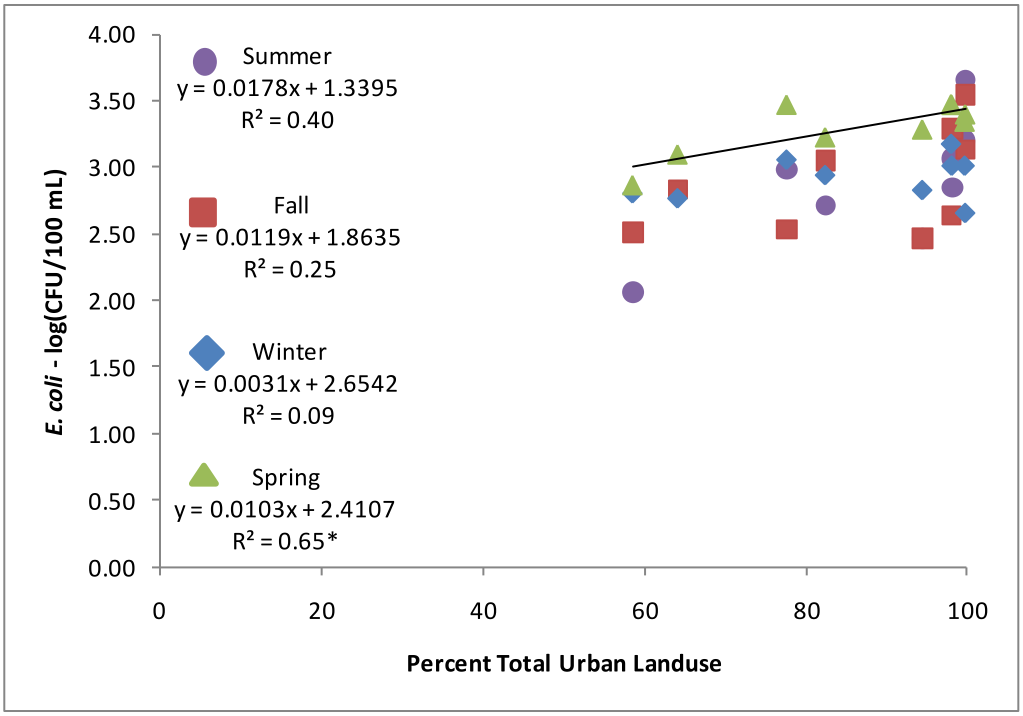

Figure 42. Seasonal linear regression of mean *E. coli* counts. One asterisk (*) indicates significance at the p ≤ 0.05 level. Trend lines are not shown for non-significant regressions.

3.4.3. Nitrogen Species: Nitrate, Nitrite, Ammonium and Dissolved Organic Nitrogen

There were no correlations between land use and nitrogen-related species in any season.

3.4.4. Orthophosphate

Like Dissolved Organic Carbon, orthophosphate manifested a correlation to deciduous forest LULC 41, but unlike the DOC relationship observed in the summer and fall, phosphate was significant in summer and winter ($p<0.001$ for both).

3.4.5. Cations: Sodium, Potassium, Magnesium, and Calcium

Sodium was also significantly related to the deciduous riparian strips (LULC 41) in all seasons ($p=0.011$ in summer; $p<0.001$ in fall; $p=0.005$ in winter; and $p=0.008$ in spring). There were no correlations discovered between any land use and potassium.

Both magnesium and calcium were correlated with pasture/crop land use (LULC 21), as indicated in Figures 43 and 44, respectively. Magnesium had a significant relationship with pasture land in summer, fall and spring ($p=0.001$, $p=0.006$, $p=0.006$, respectively), while calcium had a significant relationship in winter and spring ($p=0.004$, $p=0.003$, respectively).

3.4.6. Anions: Chloride, Fluoride, Bromide and Sulfate

Fluoride, another ion generally associated with distribution water, shows a significant relationship to deciduous buffer strip LULC 41 in summer and winter ($p=0.017$, $p=0.003$, respectively). No significant land use correlations were found for chloride, bromide or sulfate during any season.

Overall from these results we can identify the increasing effect of wastewater treatment on nitrate, phosphate and potassium and the irrigation water signature of the local distribution water on sodium, fluoride and phosphate, along with the mobilized dissolved organic carbon released into the irrigation water from the soil. Also, the pattern of *E. coli* in the watershed does not appear to be stimulated by either of these factors or by sewage leaks, but rather is influenced by total urban land use and a complex nutrient balance.

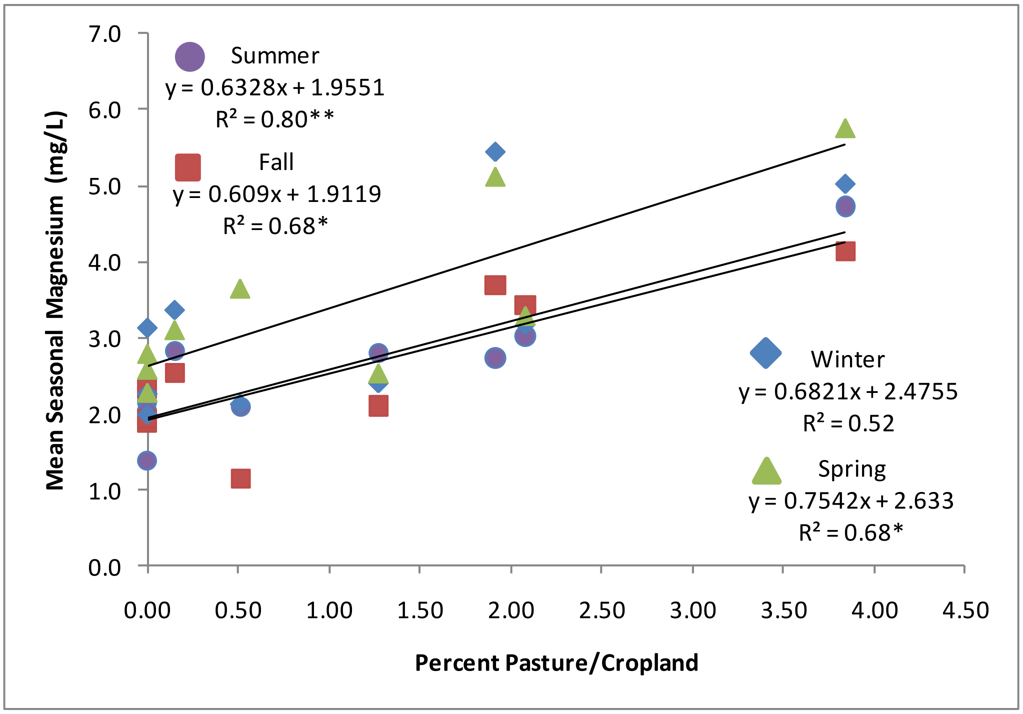

Figure 43. Seasonal regressions of mean magnesium concentrations. One asterisk (*) indicates significance at the $p \leq 0.05$ level; two asterisks (**) indicate significance at $p \leq 0.01$. Trend lines not shown for non-significant regressions.

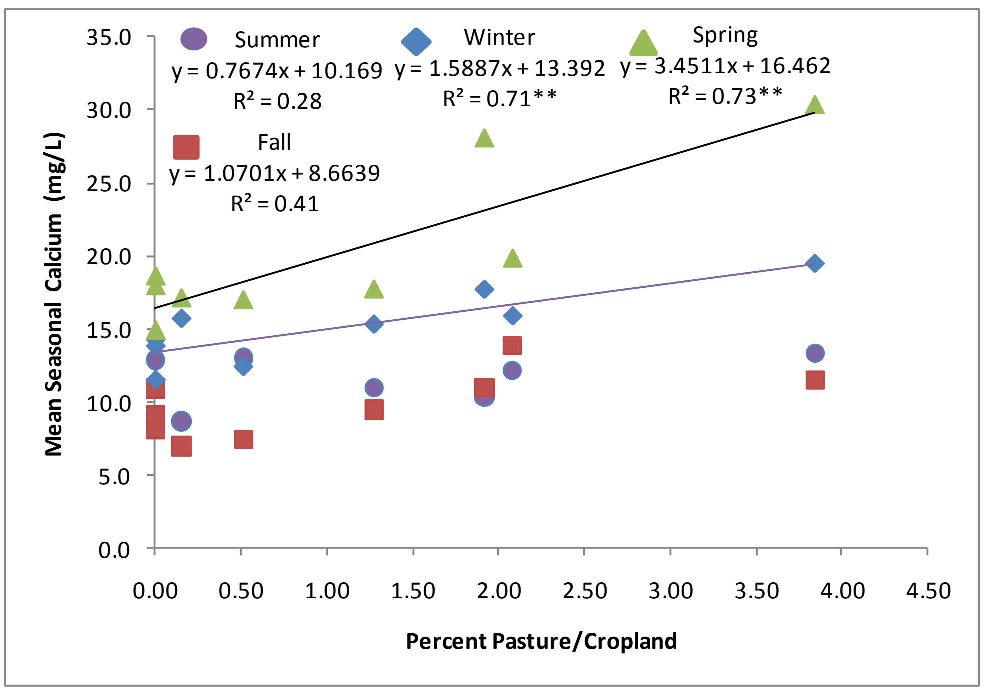

Figure 44. Seasonal regressions of mean calcium concentrations. Two asterisks (**) indicate significance at $p \leq 0.01$. Trend lines not shown for non-significant regressions.

4. DISCUSSION

This study investigated the geographical patterns of bacteria and nutrients as they varied with hydrologic flow, season and land use. "Targeted sampling," a method developed by Kuntz et al. (2003), used repeated, geographically narrowing sample collecting and visual observation to pinpoint the source of fecal contamination. The nested sampling strategy of this study discussed earlier had a similar aim, which is to reduce the area of interest contributing the most to bacteria and nutrient impairment of Carters Creek. Although no nutrient in a watershed is likely to have only one source, the pattern of concentrations and land uses in the various subcatchments can help pinpoint major contributors as the first step to mitigation, if indeed mitigation is needed for meeting standards for human and aquatic health.

4.1. *E. coli*

Although *E. coli* counts in all the Carters Creek subcatchments studied were found to be higher than the contact recreation standard of 126 CFU/100 mL, some sites consistently exceeded this threshold more than others. Burton 4 and Burton 5, for example, were both significantly higher in $\log_{10}$ values than Burton 1 (a site 1.6 river miles downstream of Burton 4 and 1.8 river miles downstream of Burton 5) during low flow, suggesting dilution, die-off or both before the Burton 1 sample location. A pilot study in Portugal showed 10^6 *E. coli* counts decreased by four orders of magnitude in one hour of exposure to sunlight and by 5 orders of magnitude in 4 hours of sunlight (Gomes et al. 2009), following a negative log pattern. Other causes of *E. coli* reduction in surface waters suggested by Whitman et al. (2008) include predation and settling. Burton 4 was also significantly higher in *E. coli* counts than Carter 1, Carter 3 and Wolfpen during low flow, suggesting an additional source of *E. coli* or a source of greater impact in this watershed in dry weather. There was a lower percentage of residential land use in Carter 1, Carter 3 and Wolfpen compared to Burton 4, but also wider riparian zones that would separate manicured lawns and pets from the stream channel compared to Burton 4. These wider riparian zones may have been conducive to filtering and retaining more bacteria before reaching the stream. Muirhead et al. (2006) found that a 5 m long plot of grass reduced *E. coli* counts in 2 L/min runoff by 27%, while a cultivated 5 m plot had a reduction of 41% due to the increase in surface roughness, infiltration and adsorption capacity. Likewise, increasing runoff contact time in riparian zones and increasing infiltration in the local heavy clay soils may help to mitigate *E. coli* inputs from storm water and irrigation runoff.

The *E. coli* geometric mean values in this study were all higher than the contact recreation standard and all means were within one order of magnitude of each other, indicating no pristine or model watershed to use as a possible springboard for mitigation. In a similar study conducted in southern California, both non-urban creek and urban creek headwaters exceeded contact recreation standards for fecal coliform counts and were within an order of magnitude of each other and the watershed outlets, indicating a diffuse, widespread source for bacteria in both subcatchments (Schiff and Kinney, 2001). Tufford and Marshall (2002) found commercial and urban open land uses contributed more heavily to fecal coliform loads because of increased impervious surface, compacted soil such as athletic fields, higher runoff and a tendency to attract urban birds and rodents. In a runoff experiment Muirhead et al. (2006) found that tap water accumulated 26000 MPN/100 mL *E. coli* as it travels the length of a 5 m turf grass plot, which is comparable to the maximum counts of 11100 – 64000 CFU/100 mL observed in the Carters Creek watershed during high flow. This supports the relationship found in this study between $\log_{10}$ values of *E. coli* and total urban land use in the spring, when the temperature increased, precipitation and runoff were high and soils were already saturated at the end of the wet season.

A strong significant relationship was also found between fecal coliform geometric means and impervious cover in Houston's Buffalo Bayou (Petersen et al., 2006). Schoonover and Lockaby (2006) developed similar predictive models, showing the log-transform of fecal coliform counts to be predicted by impervious surface cover (r^2=0.69; $p < 0.0001$). Their model supports the understanding that urbanization increases bacterial and pollutant loads to surface waters. Concrete channelization and reduced soil storage described by Alberti et al. (2007) may be responsible for pushing peak floodwaters and the flush of *E. coli* rapidly downstream during a rain event, when there is more quantity and velocity of flow from Burton 4 downstream to Burton 2 and Burton 1. Schiff and Kinney (2001) argued that even if stormwater runoff was eliminated as a source of indicator bacteria, inputs from non-urban areas and from non-human sources would continue to exceed contact recreation standards during storm events.

Regrowth of *Escherichia coli* is linked to dissolved organic carbon in some studies (e.g. Bolster et al., 2005; Boualam et al., 2003). In-stream *E. coli* could be weakly described by select nutrient concentrations. During high flow, there was a significant positive relationship between $\log_{10}$ values of *E. coli* and calcium concentrations and a significant negative relationship to potassium (r^2=0.86; p=0.002). Muirhead et al. (2006) explained that *E. coli* is typically transported in one of three configurations: single-celled, as when washed loose from feces; attached to finer soil particles, usually $\leq$20 μm in diameter; or in flocs. As a strong flocculating agent, calcium is frequently used to aggregate both DOC and bacteria and to induce settling in drinking source water (Crump et al., 2004), in wastewater treatment before disinfection (Meric et al., 2002) and in bioreactors before membrane filtration (Kim and Jang, 2006). On the other hand, excess sodium or potassium disperses the sludge and produces weak flocs with poor settling attributes (Sanin et al., 2006). During bioflocculation, a gel-like sludge made of extracellular polymeric substances (EPS) provided protection, hydration, humic carbon sources and various nutrients that became anchored into the gel along with the bacteria cells (Sanin et al., 2006). Under conditions with excess dissolved organic carbon and limiting nitrogen, the EPS produced by the bacteria has a higher DOC:DON ratio (Sanin et al., 2006), as seen in the Carters Creek watershed upstream of WWTPs. When precipitation events are more frequent and groundwater contribution is higher, it is reasonable to suppose that these bioflocs might become dislodged from the creek bottom and float closer to the surface, acting as a carrier for the *E. coli* sampled.

The fluorometric part of the study, conducted on the Burton 4 subcatchment in spring during dry weather, yielded no related spikes of *E. coli* and fluorescence. The high dissolved organic carbon concentration in the creek was a probable interference (Hartel et al., 2007), as it also fluoresces. More data, particularly in wet weather, would be needed to assess relational spikes between *E. coli* and fluorescence over time, but the very limited data I obtained did not directly detect any leaky sewer lines in the area tested. The fact that I found no related spikes between *E. coli* and fluorescence may suggest that the source of bacterial impairment is not human waste, though the cause may be from anthropogenic development. However, with the interference from dissolved organic carbon this method may not yield a result that could be confidently used without first eliminating the DOC background.

4.2. Dissolved Organic Carbon

Because of the senescence and dropping of deciduous leaves during the autumn, more leaves get flushed into the stream and accumulate in slow-moving areas or during times of low flow, allowing time for organic material to dissolve into the stream (McDowell and Fisher, 1976; Bernhardt and McDowell, 2008). One of the dominant features in the urban watershed of Carters Creek is the ubiquitous elevated concentration of dissolved organic carbon, reaching a maximum of 96 mg/L from Wolfpen Creek in the autumn. Consequently, watersheds with more deciduous trees close to the stream, such as in the wooded walking trails in the Wolfpen Creek corridor, or contributing storm drains would have a potential for higher DOC concentrations.

Carters Creek is also characterized by its irrigation water signature with high bicarbonate-alkalinity, sodium and SAR, which disperse and mobilize large quantities of DOC into the stream network (Aitkenhead-Peterson et al., 2009). Negatively charged clay minerals and organic matter attract base cations to their surface to neutralize the molecular charges. Furthermore, if base cation composition is dominated by flocculating divalent and trivalent cations, these cations hold multiple molecules in a large "clump" (Bourgeois et al., 2004). If these multivalent cations are not present in sufficient concentration, there are not enough bridges anchoring the organic material to the mineral soil particles, and DOC is lost through leaching or runoff in saturated conditions (Naidu and Rengasamy 1993). Sodic conditions also produce a high pH, which may be responsible for solubilization of humic acids. These humic acids are less soluble in water at lower pH ranges compared to fulvic acids, which are readily soluble in water under all pH conditions (Stevenson 1994). At the same time, organic N mineralization is decreased, reflected in the findings of Aitkenhead-Peterson et al. (2009), in which in-stream proteins explained DON concentrations. This often results in a soil carbon content of <1% and C:N ratios of <12 in alkaline sodic soils with insufficient calcium (Naidu and Rengasamy 1993). Once in solution, humic and fulvic acids can be leached out as the soil water moves through the root zone or in runoff if the soil is saturated. When this occurs, the sodium and dissolved organic carbon are eventually carried to the stream channel in concert, reflected in the linear regressions of SAR vs. DOC observed in the Carters Creek watershed.

Deciduous forest land use from urban riparian zones was surprisingly correlated with several nutrients including DOC, Na, Br, PO_4 and F; however, these results are likely driven by the high

DOC, Na and urban land uses in the Wolfpen Creek watershed. Because storm water runoff from impervious surfaces is conducted by concrete storm drains directly into the stream, it bypasses the riparian strips that could help filter and detain the nutrients (Alberti et al. 2007). Thus, the majority of dust and residues from city streets, parking lots and roofs are conveyed with little impediment to the nearest stream channel. Other contributing factors may include irrigation from residential lawns, public green spaces and the Texas A&M golf course at the headwaters, most of which are located upstream of the Wolfpen riparian forest corridor. Though it is possible that the deciduous forest land cover, soil and underlying geology may contribute to these nutrient concentrations, it is unlikely that only the deciduous land cover would display all these nutrient correlations. Considering the high sodium content of the local distribution water which serves as the predominant irrigation source, it is likely that the sodium is transmitted to the creek either through overland flow of excess irrigation water or through irrigation water that has infiltrated past the root zone and is now moving in lateral interflow towards the creek. This irrigation water is not typically applied to the deciduous riparian zones themselves, however. More investigation is needed to resolve this ambiguity concerning the source of these nutrients correlated with deciduous land cover.

4.3. Nitrogen Species: Nitrate, Nitrite, Ammonium and Dissolved Organic Nitrogen

Unlike dissolved organic carbon, nitrogen appeared to act independently of the high sodium concentrations. For ammonium-N, average spring concentrations were significantly higher than summer ammonium concentrations, probably due to cooler temperatures and less available soil air. For example, after the winter-spring wet season the watershed soil would have a higher water content, leaving less pore space available for air to oxygenate soil ammonium into nitrite or nitrate. Trojan et al. (2003) found groundwater concentrations of nitrate were 0.6 mg/L in undeveloped areas, indicating a background nitrate-N value of 0.14 mg/L. Nitrate-N concentrations according to Stackelberg et al. (1997) showed 0.07 mg/L in undeveloped areas, 2.6 mg/L for new urban areas, 3.5 mg/L for old urban areas, and 13 mg/L for agricultural land use. Several sites in the Carters Creek watershed during low flow had comparable background values, between 0.07 mg/L NO_3-N at Burton 2 to 0.23 mg/L NO_3-N at Bee Creek, indicating healthy background levels of nitrate. Carters Creek also had lower nitrate concentrations for urban areas than the average from Stackelberg et al. (1997). Slow-flowing waters with high temperatures, such as Wolfpen, Hudson, Bee and Briar Creeks from this study, are likely to have significant denitrification potential, moderating the effect of urbanization somewhat (Schaffner et al., 2009). However, Carter 4 and 5 had concentrations derived from nutrient rich WWTP effluent discharged upstream of the sampling sites that approached the high-nitrate-leaching agricultural croplands. In an undisturbed old growth forest watershed the in-stream DON:TDN ratio may be 0.60-0.95, while urban ratios are closer to 0.35 (Pellerin et al., 2006). Surprisingly, the DON:TDN ratio for most of the urban subcatchments is closer to the undisturbed bracket, perhaps because of the higher nitrogenous organic matter content in the stream. Sites downstream of the WWTP are more consistent with the urban bracket or even lower at 0.15-0.25, due to the high nitrate content of the effluent.

4.4. Orthophosphate

In addition to nitrate, Carter 4 and 5 sites also had the largest orthophosphate concentrations in the Carters Creek basin. The sites with the next highest phosphate concentrations were those with headwaters in golf courses. According to King et al. (2007), a golf course in Austin, Texas produced 1.2 kg NO_3-N ha^{-1} yr^{-1} and 0.51 kg PO_4-P ha^{-1} yr^{-1}, or the equivalent of 3.3% and 6.2% applied N and P, respectively. Though the nitrate inputs from storm water did not cause concentrations that threaten the stream aquatic habitat, the phosphate contribution posed a threat according to the USEPA standard of 0.1 mg/L (King et al., 2007). Greater phosphate inputs to the stream were measured during fall and winter, when the turf grass metabolic rate was slowing down (King et al., 2007). The Carters Creek watershed also showed a significant phosphate contribution from subcatchments with golf courses, especially in fall and winter. Interestingly, Burton 5 also has golf course headwaters, but does not show any increase in phosphate compared to the other sample sites, possibly because all golf course runoff is diverted to a small lake so that particulates and the associated contaminants are allowed to settle out of the stream suspension.

Fertilizer application of superphosphate to pasture in New Zealand led to an accumulation of contaminant fluoride in the top 200 mm of soil, in a mobility pattern similar to that of phosphate (Loganathan et al., 2001). Triple superphosphate fertilizer has a F:PO_4 ratio of 0.085, whereas single superphosphate (SSP) has a ratio of 0.20 (Loganathan et al., 2001). In this study, fluoride concentration during low flow was significantly higher in Hudson than in Briar 2, Burton 3-5 and the upper Carter sites, while Wolfpen had higher fluoride concentrations than Briar 2 and Carter 3. Both Hudson and Wolfpen had headwaters from golf courses. Annual fertilizer application to turf grass in golf courses, residential lawns and urban green spaces at the beginning of the growing season would also explain why the spring fluoride values are higher in the watershed as a whole than in the rest of the year.

In addition to being a commonly added ingredient in toothpaste (Buzalaf et al., 2008), sodium monofluorophosphate has been used in the construction industry as a corrosion inhibitor to steel reinforcements in concrete structures for the last twenty years (Chaussadent et al., 2006). An aqueous solution of the compound is applied to the concrete surface and diffuses into the porous concrete matrix, where it coats the steel reinforcements, reacts with $Ca(OH)_2$ to form insoluble apatites or hydrolyzes into phosphate and fluoride ions (Ngala et al. 2003). Phosphate and fluoride adsorb to soil particles more strongly than other anions commonly found in soil solution, greatly reducing ion mobility. In an experiment on a northern hardwood spodosol soils, Nodvin et al. (1986) reported that phosphate has a linear adsorption rate of 0.99 of initial mass in the soil solution, followed by the fluoride adsorption rate of 0.80. The strong relationship between phosphate and fluoride seen in the Carter Creek watershed may also indicate erosion of surface sediment, carried into the stream along with the adsorbed anions. Nevertheless, fluoride concentrations were maintained safely below the Secondary Maximum Contaminant Level (SMCL) of 2 mg/L for drinking water (USEPA, 2009).

In-stream orthophosphate also revealed a relationship with in-stream SAR in this study. As sodium is adsorbed onto soil cation exchange sites under high pH conditions and replaces divalent and trivalent cations, the increasing negative charge of the soil particle repels any nearby phosphate, which becomes much more soluble (Naidu and Regasamy, 1993). Once in the soil solution, phosphate is easily assimilated by plants or lost to groundwater or runoff and then the stream channel. Curtin et al. (1995) found an SAR of 20 significantly decreased the binding ability of clay minerals and greatly increased the water-extractable fraction of total phosphorus. By comparison, mean annual SAR in the Carters Creek watershed varied from 2.9 in Carter 3 to 20.3 in Wolfpen. In Results Section 3.4, orthophosphate showed a strongly significant relationship with SAR in summer, fall and winter, but not in spring. This may reflect the opposing factors of the flushing of Na-PO_4 complexes during rain events and the rapid turf assimilation of solubilized phosphate when grass and other vegetation is coming out of winter dormancy (King et al., 2007).

4.5. Cations and Anions

Wastewater effluent adds significant amounts of N and P that can have a severe impact on streams (Lewis et al., 2007; Phillips et al., 2007; Fitzpatrick et al., 2007; Zampella et al., 2007; Murdock et al., 2004). One of the significant sources for increases in both cations and anions within the urban area of Bryan/College Station was WWTP effluent discharge to the creeks. Wastewater effluent can provide the hydrologic benefit of stable flow even during periods of drought, when the creek might otherwise dry up or be reduced to a mere trickle (Cotman et al., 2008). For the creeks sampled downstream of a wastewater treatment plant in this study, the effluent-dominated creeks were enriched with calcium and magnesium but not enough to counterbalance the highly dispersive characteristics of sodium. Wastewater effluent was not found to significantly contribute to either *E. coli* counts or DOC concentrations in the catchment, and in fact diluted excessive in-stream *E. coli* during storm flow, though counts still grossly exceeded TCEQ standards. Several electrolytes, namely sodium, potassium and chloride, were found by Rose (2007) to be higher in municipal wastewater effluent than in other urban streams. These electrolytes were also higher in the creeks sampled in this study, with the exception of Carter 4, which was lower in sodium than Wolfpen during low flow. It is surprising that sodium was not correlated to any land use in the summer; perhaps this is because irrigation was implemented almost universally throughout the watershed, eliminating the dominance of one land use over the others. The high sodium in Wolfpen was undoubtedly caused by the highly sodic irrigation water signature from the local distribution water. Yet there is a stark contrast between the ion concentrations found in Georgia (Rose 2007) and South Carolina (Lewis et al., 2007) and those found in the Carter Creek watershed (Table 18). The urban site at Burton 1 contained at least triple the concentrations of chloride, sulfate and sodium as those observed in the southeast US. Whether these differences were because of the ion-rich irrigation water signature or the geologic input of inorganic solutes during low flow (Aitkenhead-Peterson et al., submitted), is unknown. Both Burton 1 and Carter 5 downstream of the WWTP had lower chloride than in Duchess County, New York, but this effect is likely best explained by the frequent use of road salt (NaCl or $CaCl_2$) during the winter in northern climates, which will maintain high chloride concentrations into the summer (Cunningham et al., 2009).

In addition to having the highest concentrations of sodium and chloride, Wolfpen also had the highest bromide concentrations. In commercial areas where impervious surfaces are prevalent, storm runoff from urban streets often flushes bromide-containing gasoline residues into surface waters, increasing bromide concentrations and reducing the Cl:Br ratio (Davis et al 1998). Other sources of bromide may include private pool maintenance chemicals, rainwater and irrigation water residues (Aitkenhead-Peterson et al, submitted). Although mean spring and fall concentrations follow the same trend, the correlations are not significant, perhaps because of the more frequent rain events that dilute bromide concentrations and keep streets from building up large residues. Surprisingly, no significant correlation was found between chloride and any urban land use, in contrast to other studies (Cunningham et al., 2009; Zampella et al., 2007; Rose, 2007).

Table 18. A comparison with other water quality studies conducted in various parts of the US. *Actually measured as Total Phosphorus (TP), but included here for comparison.

Comparison Studies	DOC	TDN	NO_3-N	PO_4	Cl	SO_4	Na	K	Mg	Ca	Location	Sample Type
Cunningham et al., 2009			0.64		90.6						Duchess Co., NY	suburban to rural
Stein and Yoon 2008	2.68		0.05	0.03							Los Angeles, CA	urban
Dietz and Clausen 2004		2.7	1.6	0.07*							Branford, CT	urban residential
Lewis et al., 2007	6.2	0.74	1.16	<0.10	3.41	1.9	3.74	1.63	1.15	5.24	Newberry, SC	upstream of WWTP
Lewis et al., 2007	6.1	2.96	3.11	1.15	9.84	6.37	12.47	3.36	1.16	5.57	Newberry, SC	downstream of WWTP
Rose 2007					12.3	5.28	8.14	2.46	2.12	9.64	Atlanta, GA	urban
Rose 2007					32.2	22.4	18.3	4.81	3.66	18.4	Atlanta, GA	WWTP effluent
current study, Burton 1	33.6	1.15	0.16	0.39	38.3	30.4	88.4	3.73	1.97	11	Brazos Co., TX	urban
current study, Carter 5	37.9	11.3	9.46	5.34	62.6	40.9	158	6.63	2.23	11.8	Brazos Co., TX	downstream of WWTP

5. CONCLUSIONS

Based on the hypotheses stated in the beginning of this work, the patterns of *E. coli* and nutrient concentrations measured in Carters Creek elucidated some relationships that could be useful for understanding contaminant concentrations in other urban watersheds. These observed patterns led to the following conclusions:

1. There are significant differences in both *E. coli* counts and nutrient concentrations in the various subcatchments of Carters Creek. Mean annual *E. coli* was significantly higher in Burton 4 than in Carter 1 or Carter 3. Sites downstream of wastewater treatment plants showed higher nitrate, phosphate, sodium, potassium, chloride and fluoride than other urban subcatchments. Creeks with golf courses tended to carry more phosphate, sodium and fluoride than subcatchments without golf courses and without WWTPs. Wolfpen Creek had significantly higher mean annual DOC concentrations, but whether this is a result of the golf course or urban irrigation runoff in general remains ambiguous.

2. In-stream *E. coli* counts were not found to be correlated directly with nitrogen or phosphorus concentrations. However, high-flow $\log_{10}$ values of *E. coli* had a negative relationship with potassium and a positive relationship with calcium, suggesting a bioflocculation effect when the higher base flow and more frequent rain events might dislodge and suspend the floc back in the stream.

3. No significant correlation between 445 nm fluorescence and *E. coli* was observed, possibly due to interference from DOC which also has fluorescence properties.

4. Nutrients and *E. coli* demonstrated some significant correlations to subcatchment land use. Magnesium had a significant relationship with pasture/crop land use in summer, fall and spring, while calcium was significantly related with the same land use in winter and spring. In both

annual high flow and spring seasonal sample means, $\log_{10}$ values of *E. coli* were positively correlated with urban commercial land use, probably caused by storm runoff carrying residues from impervious surfaces into the stream.

REFERENCES

Abdel-Jawad, M., Ebrahim, S., Al-Tabtabaei, M., and Al-Shammari, S. 1999. Advanced technologies for municipal wastewater purification: technical and economic assessment. Desalinization 124:251-261.

Aitkenhead, J.A., and W.H. McDowell. 2000. Soil C : N ratio as a predictor of annual riverine DOC flux at local and global scales. Global Biogeochemical Cycles 14:127-138.

Aitkenhead-Peterson, J.A., J.E. Alexander, and T.A. Clair. 2005. Dissolved organic carbon and dissolved organic nitrogen export from forested watersheds in Nova Scotia: identifying controlling factors. Global Biogeochemical Cycles 19:8.

Aitkenhead-Peterson, J.A., R.P. Smart, M.J. Aitkenhead, M.S. Cresser, and W.H. McDowell. 2007. Spatial and temporal variation of dissolved organic carbon export from gauged and ungauged watersheds of Dee Valley, Scotland: effect of land cover and C : N. Water Resources Research 43:11.

Aitkenhead-Peterson, J. A. M. K. Steele, N. Nahar and K. Santhy 2009. Dissolved organic carbon and nitrogen in urban and rural watersheds of south central Texas: land use and land management influences. Biogeochemistry 96:119-129.

Aitkenhead, J.A., N. Nahar, C.L. Harclerode, and N. Stanley. 2010. Effect of urbanization on surface water chemistry in south-central Texas. Urban Ecosystems (submitted).

Al Bakri, D., S. Rahman, and L. Bowling. 2008. Sources and management of urban stormwater pollution in rural catchments, Australia. Journal of Hydrology 356:299-311.

Alberti, M., D. Booth, K. Hill, B. Coburn, C. Avolio, S. Coe, and D. Spirandelli. 2007. The impact of urban patterns on aquatic ecosystems: an empirical analysis in Puget lowland sub-basins. Landscape and Urban Planning 80:345-361.

Alexander, R.B., R.A. Smith, G.E. Schwarz, E.W. Boyer, J.V. Nolan, and J.W. Brakebill. 2008. Differences in phosphorus and nitrogen delivery to the gulf of Mexico from the Mississippi river basin. Environmental Science & Technology 42:822-830.

Alexander, R.B., R.A. Smith, and G.E. Schwarz. 2004. Estimates of diffuse phosphorus sources in surface waters of the United States using a spatially referenced watershed model. Water Science and Technology 49(3):1-10.

Arnone, R.D., and J.P. Walling. 2007. Waterborne pathogens in urban watersheds. Journal of Water and Health 5:149-162.

Bernhardt, E.S. and W.H. McDowell. 2008. Twenty years apart: comparisons of DOM uptake during leaf leachate releases to Hubbard Brook Valley streams in 1979 vs. 2000. Journal of Geophysical Research, 113, G03032, doi:10.1029/2007JG000618.

Beutel, M.W. 2001. Oxygen consumption and ammonia accumulation in the hypolimnion of Walker Lake, Nevada. Hydrobiologia 466:107-117.

Bhatt, M.P., and W.H. McDowell. 2007. Evolution of chemistry along the Bagmati drainage network in Kathmandu valley. Water Air and Soil Pollution 185:165-176.

Biggs, T.W., and B.B. Jiang. 2009. Soil salinity and exchangeable cations in a wastewater irrigated area, India. Journal of Environmental Quality 38:887-896.

Bolster, C. H. J. M. Bromley and S. H. Jones (2005) Recovery of chlorine-exposed *Escherichia coli* in estuarine microcosms. Environ. Sci. Technol. 39:3083-3089.

Boualam, M., S. Fass, S. Saby, V. Lahoussine, J. Cavard, D. Gatel, and L. Mathieu. 2003. Organic matter quality and survival of coliforms in low-nutritive waters. Journal American Water Works Association 95:119-126.

Bourgeois, J.C., M.E. Walsh, and G.A. Gagnon. 2004. Treatment of drinking water residuals: comparing sedimentation and dissolved air flotation performance with optimal cation ratios. Water Research 38:1173-1182.

Brady, N.C, and R.R. Weil. 2002. The nature and properties of soils, 13th ed. Prentice Hall: Upper Saddle River, New Jersey.

Buzalaf, M.A.R., A.L. Leite, N.T.A. Carvalho, M.H.C. Rodrigues, E.R. Takamori, D.B. Niconielo, F.M. Levy, and V.E.S. Cardoso. 2008. Bioavailability of fluoride administered as sodium fluoride or monofluorophosphate to humans. Journal of Fluorine Chemistry 129:691-694.

Carey, R.O., and K.W. Migliaccio. 2009. Contribution of wastewater treatment plant effluents to nutrient dynamics in aquatic systems: a review. Environmental Management 44:205-217.

Chaussadent, T., W. Nobel-Pujol, F. Farcas, I. Mabille, and C. Fiaud. 2006. Effectiveness conditions of sodium monofluorophosphate as a corrosion inhibitor for concrete reinforcements. Cement and Concrete Research 36:556-561.

Chow, A.T., D.M. Leech, T.H. Boyer, and P.C. Singer. 2008. Impact of simulated solar irradiation on disinfection byproduct precursors. Environmental Science & Technology 42:5586-5593.

Clinton, B.D., and J.M. Vose. 2006. Variation in stream water quality in an urban headwater stream in the southern appalachians. Water Air and Soil Pollution 169:331-353.

Cunningham, M.A., C.M. O'Reilly, K.M. Menking, D.P. Gillikin, K.C. Smith, C.M. Foley, S.L. Belli, A.M. Pregnall, M.A. Schlessman, and P. Batur. 2009. The suburban stream syndrome: evaluating land use and stream impairments in the suburbs. Physical Geography 30:269-284.

Curtin, D., H. Steppuhn, A.R. Mermut, and F. Selles. 1995. Sodicity in irrigated soils in saskatchewan - chemistry and structural stability. Canadian Journal of Soil Science 75:177-185.

Davis, S.N., D.O. Whittemore, and J. Fabryka-Martin. 1998. Uses of chloride/bromide ratios in studies of potable water. Ground Water 36:338-350.

Dickerson, J.W., C. Hayedorn, and A. Hassall. 2007. Detection and remediation of human-origin pollution at two public beaches in Virginia using multiple source tracking methods. Water Research 41:3758-3770.

Dietz, M.E., J.C. Clausen, and K.K. Filchak. 2004. Education and changes in residential nonpoint source pollution. Environmental Management 34:684-690.

Farooq, S., A.K. Alyousef, R.I. Allayla, and A.M. Ishaq. 1994. Tertiary-treatment of sewage effluent via pilot-scale slow sand filtration. Environmental Technology 15:15-28.

Fellman, J.B., D.V. D'Amore, E. Hood, and R.D. Boone. 2008. Fluorescence characteristics and biodegradability of dissolved organic matter in forest and wetland soils from coastal temperate watersheds in southeast Alaska. Biogeochemistry 88:169-184.

Fitzpatrick M.L., D.T. Long, and B.C. Pijanowski. 2007. Exploring the effects of urban and agricultural land use on surface water chemistry, across a regional watershed, using multivariate statistics. Applied Geochemistry 22:1825-1840.

Gehr, R., M. Wagner, P. Veerasubramanian, and P. Payment. 2003. Disinfection efficiency of peracetic acid, UV and ozone after enhanced primary treatment of municipal wastewater. Water Research 37:4573-4586.

Gomes, A.I., J.C. Santos, V.J.P. Vilar, and R.A.R. Boaventura. 2009. Inactivation of bacteria *E-coli* and photodegradation of humic acids using natural sunlight. Applied Catalysis B-Environmental 88:283-291.

Hagedorn, C., Reneau, R, Saluta, M, Chapman, A. 2002. Impact of onsite wastewater systems on water quality in coastal regions. 50312-01-13-PT. Virginia Department of Conservation and Recreation: Richmond, Virginia.

Hartel, P.G., J.L. McDonald, L.C. Gentit, S.N.J. Hemmings, K. Rodgers, K.A. Smith, C.N. Belcher, R.L. Kuntz, Y. Rrvera-Torres, E. Otero, and E.C. Schroder. 2007. Improving fluorometry as a source tracking method to detect human fecal contamination. Estuaries and Coasts 30:551-561.

Hem, J.D. 1959. Study and interpretation of the chemical characteristics of natural water. Water Supply Paper 2254. U.S. Geological Survey: Washington, D.C.

Izbicki, J.A., I.M. Pimentel, R. Johnson, G.R. Aiken, and J. Leenheer. 2007. Concentration, UV-spectroscopic characteristics and fractionation of DOC in stormflow from an urban stream, Southern California, USA. Environmental Chemistry 4:35-48.

Jalali, M., H. Merikhpour, M.J. Kaledhonkar, and S. Van der Zee. 2008. Effects of wastewater irrigation on soil sodicity and nutrient leaching in calcareous soils. Agricultural Water Management 95:143-153.

Kim, I.S., and N. Jang. 2006. The effect of calcium on the membrane biofouling in the membrane bioreactor (MBR). Water Research 40:2756-2764.

King, K.W., J.C. Balogh, K.L. Hughes, and R.D. Harmel. 2007. Nutrient load generated by storm event runoff from a golf course watershed. Journal of Environmental Quality 36:1021-1030.

Kuba, T., M.C.M. van Loosdrecht, and J.J. Heijnen. 1996. Phosphorus and nitrogen removal with minimal cod requirement by integration of denitrifying dephosphatation and nitrification in a two-sludge system. Water Research 30:1702-1710.

Kuba, T., M.C.M. van Loosdrecht, and J.J. Heijnen. 1997. Biological dephosphatation by activated sludge under denitrifying conditions: pH influence and occurrence of denitrifying dephosphatation in a full-scale waste water treatement plant. Water Scence Technology 36(12):75-82.

Kuntz, R.L., P.G. Hartel, D.G. Godfrey, J.L. McDonald, K.W. Gates, and W.I. Segars. 2003. Targeted sampling protocol as prelude to bacterial source tracking with *Enterococcus faecalis*. Journal of Environmental Quality 32:2311-2318.

Lambrakis, N. 2006. Multicomponent heterovalent chromatography in aquifers. Modelling salinization and freshening phenomena in field conditions. Journal of Hydrology 323:230-243.

Lewis, G.P., J.D. Mitchell, C.B. Andersen, D.C. Haney, M.K. Liao, and K.A. Sargent. 2007. Urban influences on stream chemistry and biology in the Big Brushy Creek watershed, South Carolina. Water Air and Soil Pollution 182:303-323.

Loganathan, P., M.J. Hedley, G.C. Wallace, and A.H.C. Roberts. 2001. Fluoride accumulation in pasture forages and soils following long-term applications of phosphorus fertilisers. Environmental Pollution 115:275-282.

McDowell, W.H., A. Zsolnay, J.A. Aitkenhead-Peterson, E.G. Gregorich, D.L. Jones, D. Jodemann, K. Kalbitz, B. Marschner, and D. Schwesig. 2006. A comparison of methods to determine the biodegradable dissolved organic carbon from different terrestrial sources. Soil Biology and Biochemistry 38:1933-1942.

McDowell, W.H. and S.G. Fisher. 1976. Autumnal processing of dissolved organic matter in a small woodland stream ecosystem. Ecology 57:561-569.

Meenakshi, and R.C. Maheshwari. 2006. Fluoride in drinking water and its removal. Journal of Hazardous Materials 137:456-463.

Meric, S., M. Guida, A. Anselmo, M.L. Mattei, G. Melluso, and G. Pagano. 2002. Microbial and cod removal in a municipal wastewater treatment plant using coagulation flocculation process. Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering 37:1483-1494.

Meynendonckx, J., G. Heuvelmans, B. Muys, and J. Feyen. 2006. Effects of watershed and riparian zone characteristics on nutrient concentrations in the River Scheldt Basin. Hydrology and Earth System Sciences 10:913-922.

Muirhead, R.W., R.J. Davies-Colley, A.M. Donnison, and J.W. Nagels. 2004. Faecal bacteria yields in artificial flood events: quantifying in-stream stores. Water Research 38:1215-1224.

Muirhead, R.W., R.P. Collins, and P.J. Bremer. 2006. Numbers and transported state of *Escherichia coli* in runoff direct from fresh cowpats under simulated rainfall. Letters in Applied Microbiology 42:83-87.

Mulholland, P.J., A.M. Helton, G.C. Poole, R.O. Hall, S.K. Hamilton, B.J. Peterson, J.L. Tank, L.R. Ashkenas, L.W. Cooper, C.N. Dahm, W.K. Dodds, S.E.G. Findlay, S.V. Gregory, N.B. Grimm, S.L. Johnson, W.H. McDowell, J.L. Meyer, H.M. Valett, J.R. Webster, C.P. Arango, J.J. Beaulieu, M.J. Bernot, A.J. Burgin, C.L. Crenshaw, L.T. Johnson, B.R. Niederlehner, J.M. O'Brien, J.D. Potter, R.W. Sheibley, D.J. Sobota, and S.M. Thomas. 2008. Stream denitrification across biomes and its response to anthropogenic nitrate loading. Nature 452:202-U46.

Naidu, R., and P. Rengasamy. 1993. Ion interactions and constraints to plant nutrition in australian sodic soils. Australian Journal of Soil Research 31:801-819.

Ngala, V.T., C.L. Page, and M.M. Page. 2003. Corrosion inhibitor systems for remedial treatment of reinforced concrete. Part 2: sodium monofluorophosphate. Corrosion Science 45:1523-1537.

Nodvin, S.C., C.T. Driscoll, and G.E. Likens. 1986. Simple partitioning of anions and dissolved organic-carbon in a forest soil. Soil Science 142:27-35.

NRCS. 2009. Web Soil Survey. National Soil Survey Center, Lincoln, Nebraska.

Oehmen, A., P.C. Lemos, G. Carvalho, Z. Yuan, J. Keller, L.L. Blackall, M.A.M. Reis. 2007. Advances in enhanced biological phosphorus removal: from micro to macro scale. Water Research 41:2271-2300.

Olli, G., A. Darracq, and G. Destouni. 2009. Field study of phosphorous transport and retention in drainage reaches. Journal of Hydrology 365:46-55.

Pellerin, B.A., S.S. Kaushal, and W.H. McDowell. 2006. Does anthropogenic nitrogen enrichment increase organic nitrogen concentrations in runoff from forested and human-dominated watersheds? Ecosystems 9:852-864.

Petersen, T.M., M.P. Suarez, H.S. Rifai, P. Jensen, Y.C. Su, and R. Stein. 2006. Status and trends of fecal indicator bacteria in two urban watersheds. Water Environment Research 78:2340-2355.

Phillips, P., F.A. Russell, and J. Turner. 2007. Effect of non-point source runoff and urban sewage on Yaque del Norte River in Dominican Republic. International Journal of Environment and Pollution 31:244-266.

Qian, Y.L., and B. Mecham. 2005. Long-term effects of recycled wastewater irrigation on soil chemical properties on golf course fairways. Agronomy Journal 97:717-721.

Rose, S. 2007. The effects of urbanization on the hydrochemistry of base flow within the Chattahoochee River Basin (Georgia, USA). Journal of Hydrology 341:42-54.

Sanin, F.D., A. Vatansever, I. Turtin, F. Kara, B. Durmaz, and M.L. Sesay. 2006. Operational conditions of activated sludge: influence on flocculation and dewaterability. Drying Technology 24:1297-1306.

Schaffner, M., H.P. Bader, and R. Scheidegger. 2009. Modeling the contribution of point sources and non-point sources to Thachin River water pollution. Science of the Total Environment 407:4902-4915.

Schiff, K., and P. Kinney. 2001. Tracking sources of bacterial contamination in stormwater discharges to Mission Bay, California. Water Environment Research 73:534-542.

Schussler, J., L.A. Baker, and H. Chester-Jones. 2007. Whole-system phosphorus balances as a practical tool for lake management. Ecological Engineering 29:294-304.

Sonoda, K., and J.A. Yeakley. 2007. Relative effects of land use and near-stream chemistry on phosphorus in an urban stream. Journal of Environmental Quality 36:144-154.

Stackelberg, P.E., J.A. Hopple, L.J. Kauffman, National Water-Quality Assessment Program (U.S.). Long Island-New Jersey Study Unit., and Geological Survey (U.S.). 1997. Occurrence of nitrate, pesticides, and volatile organic compounds in the Kirkwood-Cohansey Aquifer system, southern New Jersey U.S. Geological Survey ; Branch of Information Services [distributor], Denver, CO.

Stein, E.D., and V.K. Yoon. 2008. Dry weather flow contribution of metals, nutrients, and solids from natural catchments. Water Air and Soil Pollution 190:183-195.

Swietlik, J., A. Dabrowska, U. Raczyk-Stanislawiak, and J. Nawrocki. 2004. Reactivity of natural organic matter fractions with chlorine dioxide and ozone. Water Research 38:547-558.

Thurman E.M. 1985. Organic geochemistry of natural waters. Martinus Nijhoff/Dr. W. Junk: Dordrecht, The Netherlands.

TNRIS. 2009a. Stratmap Elevation. Texas Natural Resources Information System, Austin, Texas.

TNRIS. 2009b. Stratmap National Hydrography Dataset. Texas Natural Resources Information System, Austin, Texas.

Trojan, M.D., J.S. Maloney, J.M. Stockinger, E.P. Eid, and M.J. Lahtinen. 2003. Effects of land use on ground water quality in the Anoka sand plain aquifer of Minnesota. Ground Water 41:482-492.

Tufford, D.L., and W.D. Marshall. 2002. Fecal coliform source assessment in a small, mixed land use watershed. Journal of the American Water Resources Association 38:1625-1635.

USEPA. 2002. Method 1603: *Escherichia coli* (*E. coli*) in water by membrane filtration using modified Membrane-Thermotolerant *Escherichia coli* Agar (modified mTEC). Office of Water, United States Environmental Protection Agency: Washington DC.

USEPA. 2009. National primary drinking water regulations, Washington D. C.

USEPA. 1998. National primary drinking water regulations: interim enhanced surface water treatment. Federal Register 63(241):69478-69521.

Vidon, P., and A.P. Smith. 2007. Upland controls on the hydrological functioning of riparian zones in glacial till valleys of the Midwest. Journal of the American Water Resources Association 43:1524-1539.

Vigneswaran, S., H.H. Ngo, and J. Jeyaseelan. 1997. Application of non-conventional filtration technologies in pollution control. Environmental Monitoring and Assessment 44:231-240.

Weather Underground, Inc. 2009. “History for College Station, TX.” 26 Sept. 2009 <http://www.wunderground.com/history/airport/KCLL/2007/6/1/MonthlyHistory.html>

Wilson, H.F., and M.A. Xenopoulos. 2008. Ecosystem and seasonal control of stream dissolved organic carbon along a gradient of land use. Ecosystems 11:555-568.

Winfield, M.D., and E.A. Groisman. 2003. Role of nonhost environments in the lifestyles of *Salmonella* and *Escherichia coli*. Applied and Environmental Microbiology 69:3687-3694.

Whitman, R.L., K. Przybyla-Kelly, D.A. Shively, M.B. Nevers, and M.N. Byappanahalli. 2008. Sunlight, season, snowmelt, storm, and source affect *E. coli* populations in an artificially ponded stream. Science of the Total Environment 390:448-455.

Xu, H.R., N; Singleton, FL; Attwell, RW; Grimes, DJ; Colwell, RR. 1982. Survival and viability of non-culturable *E. coli* and *Vibrio cholerae* in the estuarine and marine environment. Microbial Ecology 8:313-323.

Yano, Y., W.H. McDowell, and J.D. Aber. 2000. Biodegradable dissolved organic carbon in forest soil solution and effects of chronic nitrogen deposition. Soil Biology & Biochemistry 32:1743-1751.

Yu, R.F., H.W. Chen, W.P. Cheng, and Y.C. Shen. 2009. Application of pH-ORP titration to dynamically control the chlorination and dechlorination for wastewater reclamation. Desalination 244:164-176.

Zampella, R.A., N.A. Procopio, R.G. Lathrop, and C.L. Dow. 2007. Relationship of land-use/land-cover patterns and surface-water quality in the Mullica River basin. Journal of the American Water Resources Association 43:594-604.

Zhang, X., and M. Lulla. 2006. Distribution of pathogenic indicator bacteria in structural Best Management Practices. Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering 41:1421-1436.

APPENDIX I

E. COLI COUNTS AND C AND N IN SUBCATCHMENTS

Date	Site	*E. coli* CFU/100mL	NPOC mg L^{-1}	TDN mg L^{-1}	NO_3-N mg L^{-1}	NH_4-N mg L^{-1}	DON mg L^{-1}	DON:TDN
9/3/2007	Bee	550	45.45	1.22	0.10	0.18	0.94	0.77
9/17/2007	Bee	720	57.50	2.72	0.14	0.84	1.74	0.64
9/27/2007	Bee	3900	70.2	2.14	0.70	0.08	1.36	0.64
10/22/2007	Bee	7500	38.56	1.59	0.21	0.17	1.21	0.76
10/29/2007	Bee	330	43.39	1.43	0.24	0.18	1.00	0.70
11/12/2007	Bee	80	75.6	1.31	0.15	0.02	1.14	0.87
11/26/2007	Bee	7500	38.81	1.97	0.68	0.04	1.24	0.63
6/25/2007	Bee	3500	13.73	1.16	0.20	0.09	0.88	0.76
7/11/2007	Bee	500	19.51	1.25	0.05	0.19	1.02	0.81
7/23/2007	Bee	1380	42.99	1.05	0.10	0.07	0.88	0.83
8/6/2007	Bee	250	11.68	0.85	0.11	0.01	0.73	0.86
8/20/2007	Bee	760	46.00	1.09	0.27	0.10	0.73	0.67
6/9/2008	Bee	200	15.8	1.2	0.17	0.03	0.97	0.83
6/24/2008	Bee	110	27.0	1.6	0.36	0.03	1.21	0.76
3/5/2008	Bee	10800	11.66	1.15	0.56	0.10	0.49	0.42
3/19/2008	Bee	14400	18.74	1.31	0.42	0.13	0.76	0.58
4/9/2008	Bee	610	19.61	1.40	0.29	0.11	1.00	0.71
4/25/2008	Bee	370	15.61	1.29	0.39	0.127	0.77	0.60
5/14/2008	Bee	1190	11.25	1.21	0.55	0.10	0.55	0.46
5/22/2008	Bee	540	16.72	1.01	0.29	0.05	0.68	0.67
12/10/2007	Bee	260	38.97	1.03	0.21	0.03	0.80	0.77
12/21/2007	Bee	1400	17.26	1.86	0.52	0.18	1.17	0.63
1/7/2008	Bee	890	36.65	1.01	0.35	0.01	0.65	0.64
1/23/2008	Bee	810	36.78	1.15	0.39	0.03	0.74	0.64
2/4/2008	Bee	450	13.27	1.18	0.52	0.13	0.53	0.45
2/18/2008	Bee	3800	16.53	1.13	0.22	0.09	0.82	0.72
6/25/2007	Briar 1	31000	10.00	0.86	0.23	0.03	0.60	0.70
7/11/2007	Briar 1	200	35.52	1.39	0.15	0.04	1.20	0.86
7/23/2007	Briar 1	3700	76.77	1.28	0.17	0.05	1.07	0.83
8/6/2007	Briar 1	1510	20.79	0.74	0.09	0.04	0.61	0.83
8/20/2007	Briar 1	280	60.29	1.17	0.07	0.01	1.09	0.93
9/3/2007	Briar 1	40	76.39	1.66	0.62	0.04	1.00	0.60
9/17/2007	Briar 1	580	78.79	1.19	0.08	0.02	1.10	0.92
9/27/2007	Briar 1	200	95.45	1.99	0.16	0.02	1.81	0.91
10/22/2007	Briar 1	6500	19.17	1.50	0.30	0.05	1.15	0.77
10/29/2007	Briar 1	120	24.95	1.21	0.04	0.03	1.14	0.94
11/12/2007	Briar 1	70	97.89	1.24	0.15	0.02	1.07	0.86
11/26/2007	Briar 1	2500	36.53	1.84	0.45	0.02	1.37	0.74
12/10/2007	Briar 1	380	80.51	3.08	0.10	1.55	1.43	0.46
12/21/2007	Briar 1	700	22.75	2.06	0.26	0.30	1.49	0.73

1/7/2008	Briar 1	790	49.79	1.01	0.02	0.01	0.98	0.97
1/23/2008	Briar 1	650	32.25	1.15	0.24	0.02	0.90	0.78
2/4/2008	Briar 1	360	15.89	1.09	0.21	0.07	0.80	0.74
2/18/2008	Briar 1	4000	14.36	1.27	0.20	0.07	0.99	0.78
3/5/2008	Briar 1	1900	15.17	0.92	0.17	0.08	0.67	0.73
3/19/2008	Briar 1	18500	14.68	1.22	0.31	0.10	0.81	0.66
4/9/2008	Briar 1	4700	20.03	1.79	0.17	0.13	1.49	0.83
4/25/2008	Briar 1	390	34.54	2.95	0.69	0.160	2.10	0.71
5/14/2008	Briar 1	16000	13.75	1.31	0.47	0.12	0.72	0.55
5/22/2008	Briar 1	710	20.60	0.86	0.06	0.04	0.75	0.88
6/9/2008	Briar 1	190	28.6	1.6	0.06	0.03	1.49	0.94
6/24/2008	Briar 1	720	27.4	1.4	0.15	0.03	1.21	0.87
9/17/2007	Briar 2	1380	39.63	0.84	0.12	0.02	0.70	0.84
10/29/2007	Briar 2	1130	15.79	1.05	0.08	0.06	0.90	0.86
11/12/2007	Briar 2	130	51.51	0.59	0.13	0.01	0.45	0.76
9/3/2007	Briar 2	680	42.17	0.66	0.07	0.05	0.54	0.82
9/27/2007	Briar 2	1400	50.41	0.84	0.10	0.00	0.73	0.87
10/22/2007	Briar 2	19000	34.20	1.31	0.30	0.10	0.91	0.69
11/26/2007	Briar 2	2400	37.17	1.74	0.47	0.06	1.21	0.70
7/11/2007	Briar 2	300	20.38	10.73	0.03	0.09	10.61	0.99
7/23/2007	Briar 2	460	29.28	0.40	0.04	0.13	0.23	0.58
8/20/2007	Briar 2	1700	34.72	1.03	0.07	0.12	0.84	0.82
6/9/2008	Briar 2	670	16.2	1.1	0.11	0.10	0.87	0.81
6/24/2008	Briar 2	540	20.5	1.2	0.15	0.03	1.01	0.85
6/25/2007	Briar 2	45000	7.89	0.73	0.19	0.03	0.50	0.69
8/6/2007	Briar 2	6600	10.40	0.91	0.01	0.08	0.81	0.89
4/9/2008	Briar 2	630	14.94	1.39	0.11	0.20	1.08	0.77
4/25/2008	Briar 2	360	14.06	1.04	0.11	0.06	0.87	0.84
5/22/2008	Briar 2	450	12.87	0.78	0.14	0.05	0.59	0.75
3/5/2008	Briar 2	5400	10.94	0.88	0.21	0.10	0.57	0.65
3/19/2008	Briar 2	22000	20.62	1.38	0.50	0.15	0.73	0.53
5/14/2008	Briar 2	20000	14.26	1.22	0.45	0.10	0.67	0.55
1/7/2008	Briar 2	200	33.18	0.80	0.03	0.03	0.74	0.93
2/4/2008	Briar 2	410	15.82	1.13	0.27	0.15	0.71	0.63
12/10/2007	Briar 2	110	43.86	0.99	0.10	0.03	0.85	0.87
12/21/2007	Briar 2	500	19.03	1.75	0.23	0.14	1.38	0.79
1/23/2008	Briar 2	620	33.80	1.19	0.21	0.02	0.96	0.81
2/18/2008	Briar 2	3200	12.97	0.85	0.18	0.10	0.57	0.67
9/17/2007	Burton 1	150	49.71	1.05	0.02	0.02	1.01	0.96
10/29/2007	Burton 1	190	56.98	0.47	0.06	0.05	0.36	0.77
11/12/2007	Burton 1	60	86.00	1.15	0.12	0.02	1.00	0.87
9/3/2007	Burton 1	30	56.08	0.92	0.07	0.01	0.84	0.91
9/27/2007	Burton 1	900	79.11	1.24	0.12	0.01	1.12	0.90
10/22/2007	Burton 1	21000	37.55	1.57	0.42	0.10	1.05	0.67
11/26/2007	Burton 1	1600	38.55	1.66	0.34	0.02	1.30	0.78
7/11/2007	Burton 1	100	17.56	1.46	0.04	0.02	1.40	0.96
7/23/2007	Burton 1	540	39.01	0.59	0.01	0.03	0.55	0.93

8/20/2007	Burton 1	600	55.36	1.40	0.06	0.01	1.33	0.95
6/9/2008	Burton 1	9	24.1	1.4	0.07	0.03	1.30	0.93
6/24/2008	Burton 1	1090	55.3	2.3	0.15	0.03	2.09	0.92
6/25/2007	Burton 1	50000	9.84	0.83	0.21	0.09	0.53	0.64
8/6/2007	Burton 1	1170	11.86	1.05	0.16	0.06	0.83	0.79
4/9/2008	Burton 1	240	13.58	0.88	0.12	0.04	0.72	0.82
4/25/2008	Burton 1	30	15.77	1.32	0.09	0.05	1.18	0.90
5/22/2008	Burton 1	50	15.55	0.80	0.08	0.03	0.69	0.87
3/5/2008	Burton 1	11800	13.32	1.29	0.24	0.10	0.96	0.74
3/19/2008	Burton 1	6000	10.78	0.84	0.26	0.07	0.51	0.61
5/14/2008	Burton 1	26000	14.90	1.48	0.59	0.10	0.78	0.53
1/7/2008	Burton 1	30	50.61	0.85	0.00	0.03	0.82	0.96
2/4/2008	Burton 1	550	15.60	0.83	0.11	0.08	0.64	0.77
12/10/2007	Burton 1	140	37.2	1.04	0.21	0.03	0.80	0.77
12/21/2007	Burton 1	1500	13.10	1.37	0.38	0.02	0.97	0.71
1/23/2008	Burton 1	950	44.11	0.85	0.04	0.01	0.80	0.93
2/18/2008	Burton 1	3300	13.18	1.26	0.24	0.05	0.97	0.77
3/5/2008	Burton 2	2600	14.13	1.43	0.10	0.12	1.21	0.85
3/19/2008	Burton 2	5700	13.90	1.04	0.34	0.07	0.63	0.61
4/9/2008	Burton 2	780	12.76	0.99	0.10	0.04	0.85	0.86
4/25/2008	Burton 2	20	23.93	1.58	0.10	0.07	1.41	0.89
5/14/2008	Burton 2	40000	14.58	1.54	0.61	0.15	0.77	0.50
5/22/2008	Burton 2	580	23.54	0.91	0.05	0.04	0.82	0.89
6/25/2007	Burton 2	64000	11.03	0.99	0.21	0.03	0.75	0.75
7/11/2007	Burton 2	2600	18.97	1.61	0.01	0.02	1.58	0.98
7/23/2007	Burton 2	1800	51.25	1.35	0.01	0.04	1.31	0.96
8/6/2007	Burton 2	2200	13.72	0.94	0.01	0.06	0.88	0.93
8/20/2007	Burton 2	3000	55.75	2.24	0.05	0.01	2.18	0.97
6/9/2008	Burton 2	5000	61.1	3.3	0.09	0.03	3.23	0.96
6/24/2008	Burton 2	500	31.1	1.8	0.16	0.04	1.64	0.89
9/3/2007	Burton 2	430	71.80	0.81	0.07	0.01	0.73	0.90
9/17/2007	Burton 2	330	58.10	0.83	0.05	0.02	0.76	0.92
9/27/2007	Burton 2	26000	52.20	2.11	0.76	0.08	1.26	0.60
10/22/2007	Burton 2	10000	43.26	1.84	0.33	0.20	1.31	0.71
10/29/2007	Burton 2	550	63.00	0.64	0.06	0.01	0.57	0.90
11/12/2007	Burton 2	6600	121.56	2.18	0.13	0.02	2.03	0.93
11/26/2007	Burton 2	1570	36.53	1.53	0.19	0.02	1.33	0.87
12/10/2007	Burton 2	120	43.45	0.98	0.13	0.02	0.83	0.84
12/21/2007	Burton 2	600	14.84	1.41	0.37	0.02	1.02	0.73
1/7/2008	Burton 2	530	58.04	1.04	0.00	0.01	1.03	0.99
1/23/2008	Burton 2	680	43.45	0.94	0.00	0.01	0.92	0.99
2/4/2008	Burton 2	760	16.14	0.93	0.09	0.06	0.78	0.84
2/18/2008	Burton 2	3600	12.48	1.22	0.23	0.05	0.95	0.77
6/25/2007	Burton 3	44000	11.02	1.12	0.43	0.04	0.65	0.58
7/11/2007	Burton 3	300	15.98	2.08	0.12	0.05	1.91	0.92
7/23/2007	Burton 3	670	45.23	0.79	0.10	0.05	0.64	0.81
8/6/2007	Burton 3	1080	12.10	0.79	0.07	0.03	0.69	0.88

8/20/2007	Burton 3	1000	36.75	0.57	0.08	0.00	0.49	0.85
9/3/2007	Burton 3	340	88.14	0.44	0.08	0.01	0.35	0.80
9/17/2007	Burton 3	20	47.26	0.76	0.04	0.16	0.55	0.73
9/27/2007	Burton 3	8400	45.1	0.85	0.18	0.06	0.60	0.71
10/22/2007	Burton 3	12700	39.91	1.18	0.39	0.09	0.70	0.59
10/29/2007	Burton 3	210	14.80	0.98	0.06	0.03	0.89	0.90
11/12/2007	Burton 3	10	33.20	0.47	0.14	0.02	0.32	0.68
11/26/2007	Burton 3	2000	41.9	1.92	0.42	0.01	1.49	0.78
12/10/2007	Burton 3	3500	51.32	0.29	0.09	0.01	0.19	0.66
12/21/2007	Burton 3	8300	19.75	2.72	0.23	0.99	1.50	0.55
1/7/2008	Burton 3	260	52.4	0.5	0.00	0.01	0.49	0.97
1/23/2008	Burton 3	650	48.53	0.91	0.19	0.02	0.70	0.78
2/4/2008	Burton 3	1700	13.21	0.79	0.16	0.06	0.56	0.71
2/18/2008	Burton 3	1360	16.56	1.36	0.26	0.05	1.05	0.77
3/5/2008	Burton 3	3100	15.57	0.84	0.09	0.06	0.69	0.82
3/19/2008	Burton 3	6300	21.46	1.25	0.34	0.04	0.87	0.70
4/9/2008	Burton 3	670	10.30	0.93	0.19	0.06	0.68	0.73
4/25/2008	Burton 3	1700	11.40	1.10	0.12	0.05	0.93	0.85
5/14/2008	Burton 3	46000	12.29	1.43	0.72	0.14	0.57	0.40
5/22/2008	Burton 3	680	15.10	0.48	0.12	0.04	0.31	0.65
6/9/2008	Burton 3	420	18.8	0.8	0.07	0.09	0.64	0.79
6/24/2008	Burton 3	710	17.0	0.7	0.18	0.02	0.47	0.70
6/25/2007	Burton 4	55000	11.04	0.92	0.21	0.02	0.68	0.74
7/11/2007	Burton 4	700	26.11	1.32	0.05	0.03	1.24	0.94
7/23/2007	Burton 4	900	41.00	1.16	0.17	0.05	0.94	0.81
8/6/2007	Burton 4	7300	15.69	1.48	0.31	0.07	1.10	0.74
8/20/2007	Burton 4	17000	43.37	1.34	0.09	0.02	1.23	0.92
9/3/2007	Burton 4	6400	45.70	0.71	0.09	0.02	0.60	0.84
9/17/2007	Burton 4	3400	85.90	1.01	0.13	0.15	0.73	0.72
9/27/2007	Burton 4	13900	102.83	1.88	0.37	0.41	1.10	0.58
10/22/2007	Burton 4	29000	25.55	1.84	0.28	0.26	1.29	0.70
10/29/2007	Burton 4	910	26.95	1.25	0.10	0.01	1.14	0.91
11/12/2007	Burton 4	670	72.86	1.28	0.14	0.02	1.12	0.87
11/26/2007	Burton 4	1200	53.22	2.26	0.40	0.03	1.83	0.81
12/10/2007	Burton 4	280	54.05	0.69	0.14	0.01	0.53	0.77
12/21/2007	Burton 4	2500	20.90	1.80	0.32	0.05	1.42	0.79
1/7/2008	Burton 4	170	46.07	0.95	0.00	0.02	0.93	0.98
1/23/2008	Burton 4	2100	44.98	1.30	0.37	0.02	0.91	0.70
2/4/2008	Burton 4	1530	21.20	1.40	0.30	0.09	1.00	0.72
2/18/2008	Burton 4	3100	19.63	1.79	0.26	0.06	1.47	0.82
3/5/2008	Burton 4	11200	14.16	1.86	0.39	0.16	1.31	0.70
3/19/2008	Burton 4	7500	21.28	1.56	0.52	0.10	0.94	0.60
4/9/2008	Burton 4	840	21.62	2.09	0.32	0.13	1.64	0.79
4/25/2008	Burton 4	200	23.03	2.60	0.40	0.19	2.01	0.77
5/14/2008	Burton 4	23000	12.68	1.44	0.59	0.14	0.71	0.50
5/22/2008	Burton 4	350	20.28	1.25	0.29	0.06	0.90	0.72
6/9/2008	Burton 4	1770	20.8	0.9	0.13	0.07	0.67	0.77

6/24/2008	Burton 4	5500	19.8	1.1	0.32	0.07	0.69	0.64
6/25/2007	Burton 5	31000	10.57	0.92	0.18	0.06	0.68	0.73
7/11/2007	Burton 5	600	16.11	1.73	0.09	0.04	1.60	0.93
7/23/2007	Burton 5	1100	39.24	0.60	0.14	0.09	0.36	0.61
8/6/2007	Burton 5	1590	9.35	0.83	0.06	0.03	0.73	0.89
8/20/2007	Burton 5	1200	46.79	1.08	0.10	0.01	0.97	0.90
9/3/2007	Burton 5	2000	54.03	1.02	0.32	0.02	0.68	0.67
9/17/2007	Burton 5	520	30.46	0.92	0.08	0.02	0.82	0.90
9/27/2007	Burton 5	12300	52.43	3.09	0.79	0.34	1.95	0.63
10/22/2007	Burton 5	19000	28.01	1.62	0.52	0.12	0.98	0.60
10/29/2007	Burton 5	360	17.88	1.13	0.11	0.03	0.99	0.88
11/12/2007	Burton 5	530	90.76	0.95	0.14	0.02	0.79	0.83
11/26/2007	Burton 5	2300	28.69	1.36	0.22	0.09	1.04	0.77
12/10/2007	Burton 5	270	41.76	1.15	0.32	0.09	0.73	0.64
12/21/2007	Burton 5	900	11.89	1.40	0.34	0.11	0.95	0.68
1/7/2008	Burton 5	5600	52.99	0.85	0.10	0.01	0.74	0.87
1/23/2008	Burton 5	670	42.85	1.28	0.11	0.02	1.14	0.89
2/4/2008	Burton 5	920	29.00	1.19	0.38	0.14	0.67	0.56
2/18/2008	Burton 5	3700	27.18	1.20	0.27	0.08	0.84	0.71
3/5/2008	Burton 5	3500	21.70	0.68	0.11	0.10	0.46	0.68
3/19/2008	Burton 5	5300	32.56	1.17	0.30	0.11	0.77	0.65
4/9/2008	Burton 5	780	26.10	0.92	0.13	0.07	0.72	0.78
4/25/2008	Burton 5	640	35.01	1.09	0.14	0.08	0.87	0.80
5/14/2008	Burton 5	41000	12.37	1.04	0.44	0.08	0.52	0.50
5/22/2008	Burton 5	1690	11.16	0.77	0.17	0.05	0.55	0.72
6/9/2008	Burton 5	240	25.3	0.9	0.07	0.04	0.82	0.88
6/24/2008	Burton 5	10	28.1	1.3	0.17	0.02	1.08	0.85
6/25/2007	Carter 1	6200	11.04	0.84	0.13	0.04	0.67	0.80
7/11/2007	Carter 1	99	17.79	0.56	0.10	0.05	0.42	0.74
7/23/2007	Carter 1	310	30.54	0.60	0.23	0.06	0.31	0.52
8/6/2007	Carter 1	9200	8.29	0.79	0.19	0.07	0.53	0.67
8/20/2007	Carter 1	500	28.88	0.63	0.21	0.07	0.35	0.55
9/3/2007	Carter 1	70	35.78	0.68	0.13	0.11	0.44	0.65
9/17/2007	Carter 1	300	23.57	0.67	0.12	0.06	0.48	0.72
9/27/2007	Carter 1	60	41.53	0.61	0.10	0.03	0.48	0.79
10/22/2007	Carter 1	400	15.17	1.19	0.12	0.16	0.91	0.76
10/29/2007	Carter 1	310	21.76	1.20	1.06	0.21	0.00	0.00
11/12/2007	Carter 1	10	13.23	1.06	0.17	0.04	0.86	0.81
11/26/2007	Carter 1	8600	29.99	1.70	0.60	0.02	1.08	0.63
12/10/2007	Carter 1	40	40.67	1.16	0.18	0.04	0.94	0.81
12/21/2007	Carter 1	1600	15.08	1.29	0.30	0.03	0.96	0.74
1/7/2008	Carter 1	80	38.29	0.65	0.07	0.03	0.56	0.86
1/23/2008	Carter 1	1010	41.35	1.07	0.23	0.02	0.82	0.77
2/4/2008	Carter 1	670	28.12	1.11	0.28	0.07	0.76	0.68
2/18/2008	Carter 1	4300	19.92	1.01	0.14	0.07	0.80	0.80
3/5/2008	Carter 1	28000	26.56	0.61	0.27	0.05	0.29	0.48
3/19/2008	Carter 1	10900	28.01	1.10	0.24	0.05	0.81	0.74

4/9/2008	Carter 1	350	33.25	0.96	0.14	0.11	0.71	0.74
4/25/2008	Carter 1	200	35.44	1.39	0.19	0.13	1.07	0.77
5/14/2008	Carter 1	2500	10.95	0.84	0.16	0.15	0.52	0.62
5/22/2008	Carter 1	60	12.61	0.62	0.19	0.06	0.37	0.60
6/9/2008	Carter 1	40	11.5	0.4	0.06	0.03	0.36	0.80
6/24/2008	Carter 1	20	26.3	1.3	0.14	0.02	1.15	0.88
6/25/2007	Carter 2	8100	2.81	0.58	0.08	0.08	0.41	0.72
7/11/2007	Carter 2	1200	16.65	1.07	0.06	0.03	0.97	0.91
7/23/2007	Carter 2	470	38.50	0.39	0.17	0.05	0.17	0.44
8/6/2007	Carter 2	810	8.79	0.95	0.39	0.06	0.49	0.52
8/20/2007	Carter 2	9700	34.52	0.92	0.19	0.06	0.67	0.73
9/3/2007	Carter 2	360	80.95	13.01	13.99	0.81	0.00	0.00
9/17/2007	Carter 2	50	60.15	1.11	0.21	0.16	0.75	0.67
9/27/2007	Carter 2	350	90.04	9.17	6.53	0.61	2.04	0.22
10/22/2007	Carter 2	6500	39.85	2.52	1.32	0.06	1.14	0.45
10/29/2007	Carter 2	390	19.04	1.33	0.73	0.06	0.54	0.41
11/12/2007	Carter 2	20	58.88	1.01	0.04	0.02	0.96	0.95
11/26/2007	Carter 2	1700	32.8	1.84	0.73	0.01	1.10	0.60
12/10/2007	Carter 2	8000	78.65	3.15	1.83	0.57	0.75	0.24
12/21/2007	Carter 2	900	17.48	1.70	0.65	0.03	1.02	0.60
1/7/2008	Carter 2	230	39.21	0.45	0.00	0.01	0.44	0.97
1/23/2008	Carter 2	2600	39.60	0.96	0.28	0.02	0.66	0.69
2/4/2008	Carter 2	330	37.85	1.03	0.25	0.09	0.69	0.67
2/18/2008	Carter 2	1560	35.57	1.03	0.20	0.06	0.77	0.74
3/5/2008	Carter 2	5900	29.47	0.96	0.28	0.07	0.62	0.64
3/19/2008	Carter 2	9400	21.63	0.76	0.21	0.06	0.49	0.64
4/9/2008	Carter 2	530	47.79	0.82	0.11	0.10	0.62	0.75
4/25/2008	Carter 2	780	59.63	1.24	0.15	0.06	1.03	0.83
5/14/2008	Carter 2	12400	16.08	1.09	0.29	0.08	0.72	0.66
5/22/2008	Carter 2	2300	12.95	0.60	0.09	0.05	0.47	0.78
6/9/2008	Carter 2	180	24.3	1.1	0.07	0.03	1.04	0.91
6/24/2008	Carter 2	120	29.0	1.3	0.18	0.04	1.11	0.83
6/25/2007	Carter 3	4600	10.71	0.98	0.14	0.10	0.75	0.76
7/11/2007	Carter 3	99	18.94	1.15	0.01	0.02	1.12	0.97
7/23/2007	Carter 3	190	34.84	0.33	0.07	0.04	0.22	0.67
8/6/2007	Carter 3	9	12.01	0.79	0.02	0.08	0.70	0.88
8/20/2007	Carter 3	340	45.84	1.28	0.08	0.03	1.17	0.91
9/17/2007	Carter 3	100	47.67	0.87	0.06	0.06	0.76	0.87
9/27/2007	Carter 3	530	37.1	1.47	0.32	0.03	1.12	0.76
10/22/2007	Carter 3	800	27.99	2.36	0.54	0.41	1.41	0.60
10/29/2007	Carter 3	140	37.93	1.48	0.33	0.11	1.05	0.71
11/12/2007	Carter 3	170	39.56	1.44	0.03	0.25	1.16	0.81
11/26/2007	Carter 3	1190	27.94	1.6	0.30	0.03	1.27	0.79
12/10/2007	Carter 3	610	39.27	1.15	0.14	0.04	0.97	0.84
12/21/2007	Carter 3	1300	15.54	1.16	0.12	0.04	1.01	0.87
1/7/2008	Carter 3	1090	39.11	0.85	0.00	0.02	0.83	0.98
1/23/2008	Carter 3	360	44.79	0.81	0.08	0.02	0.71	0.88

2/4/2008	Carter 3	150	31.27	1.02	0.09	0.07	0.85	0.84
2/18/2008	Carter 3	1470	26.19	0.78	0.09	0.04	0.65	0.83
3/5/2008	Carter 3	11100	16.60	1.60	0.26	0.08	1.26	0.79
3/19/2008	Carter 3	7400	29.88	1.03	0.22	0.04	0.76	0.74
4/9/2008	Carter 3	310	35.17	1.21	0.06	0.11	1.04	0.86
4/25/2008	Carter 3	190	26.19	0.92	0.08	0.10	0.75	0.81
5/14/2008	Carter 3	150	12.44	0.67	0.14	0.08	0.45	0.68
5/22/2008	Carter 3	220	12.25	0.54	0.05	0.04	0.44	0.82
6/9/2008	Carter 3	9*	15.9	1.0	0.07	0.03	0.88	0.90
6/25/2007	Carter 4	19000	7.80	0.83	0.28	0.06	0.49	0.60
7/11/2007	Carter 4	500	14.59	11.37	7.04	0.04	4.29	0.38
7/23/2007	Carter 4	1130	62.01	18.13	5.33	0.08	12.72	0.70
8/6/2007	Carter 4	1280	9.72	8.92	7.95	0.06	0.91	0.10
8/20/2007	Carter 4	1220	61.84	11.91	11.07	0.06	0.78	0.07
9/3/2007	Carter 4	420	84.83	14.98	13.11	0.05	1.82	0.12
9/17/2007	Carter 4	270	75.62	13.78	8.18	0.05	5.56	0.40
9/27/2007	Carter 4	620	86.54	16.37	14.60	0.06	1.71	0.10
10/22/2007	Carter 4	400	22.38	17.46	14.27	0.11	3.08	0.18
10/29/2007	Carter 4	270	69.50	17.16	9.97	0.09	7.10	0.41
11/12/2007	Carter 4	240	16.92	17.35	14.44	0.12	2.79	0.16
11/26/2007	Carter 4	12400	18.71	3.32	1.99	0.09	1.24	0.37
12/10/2007	Carter 4	170	74.00	22.58	13.69	0.05	8.84	0.39
12/21/2007	Carter 4	400	16.85	8.83	8.79	0.05	0.00	0.00
1/7/2008	Carter 4	310	78.65	12.84	10.67	0.24	1.93	0.15
1/23/2008	Carter 4	660	65.20	8.85	6.59	0.07	2.19	0.25
2/4/2008	Carter 4	2500	39.82	4.94	4.27	0.15	0.53	0.11
2/18/2008	Carter 4	5600	22.99	1.52	2.05	0.08	0.00	0.00
3/5/2008	Carter 4	2200	47.08	7.81	8.68	0.09	0.00	0.00
3/19/2008	Carter 4	14100	28.14	1.35	0.45	0.22	0.67	0.50
4/9/2008	Carter 4	610	56.08	6.87	5.01	0.13	1.73	0.25
4/25/2008	Carter 4	370	81.51	13.61	9.65	0.45	3.51	0.26
5/14/2008	Carter 4	1190	13.81	6.29	5.13	0.80	0.36	0.06
5/22/2008	Carter 4	540	14.35	6.92	6.51	0.35	0.06	0.01
6/9/2008	Carter 4	140	14.9	6.9	7.36	0.21	0.00	0.00
6/24/2008	Carter 4	1300	15.0	9.8	10.59	0.07	-0.86	-0.09
6/25/2007	Carter 5	99	12.43	5.34	4.83	0.05	0.47	0.09
7/11/2007	Carter 5	700	15.27	13.47	10.00	0.06	3.41	0.25
7/23/2007	Carter 5	840	57.36	13.12	7.79	0.07	5.26	0.40
8/6/2007	Carter 5	1180	9.42	6.16	5.95	0.06	0.15	0.02
8/20/2007	Carter 5	810	72.74	13.09	10.23	0.07	2.79	0.21
9/3/2007	Carter 5	320	75.73	14.05	12.18	0.08	1.79	0.13
9/17/2007	Carter 5	30	82.2724	15.0426	8.41	0.09	6.55	0.44
9/27/2007	Carter 5	170	82.76	17.53	15.84	0.09	1.60	0.09
10/22/2007	Carter 5	700	22.31	14.80	12.28	0.08	2.43	0.16
10/29/2007	Carter 5	600	20.45	20.56	16.61	0.10	3.85	0.19
11/12/2007	Carter 5	560	17.84	12.77	12.04	0.06	0.66	0.05
11/26/2007	Carter 5	3500	41.98	6.46	4.72	0.34	1.40	0.22

12/10/2007	Carter 5	300	72.83	19.90	15.80	0.11	3.99	0.20
12/21/2007	Carter 5	600	20.31	9.94	8.91	0.11	0.92	0.09
1/7/2008	Carter 5	280	22.20	16.12	15.46	0.07	0.58	0.04
1/23/2008	Carter 5	1120	40.64	6.44	5.96	0.08	0.40	0.06
2/4/2008	Carter 5	410	63.93	14.31	10.86	0.25	3.20	0.22
2/18/2008	Carter 5	5000	28.52	1.96	1.77	0.09	0.10	0.05
3/5/2008	Carter 5	6700	38.68	6.75	6.58	0.11	0.06	0.01
3/19/2008	Carter 5	17200	34.31	1.86	0.44	0.24	1.18	0.63
4/9/2008	Carter 5	850	43.79	7.38	5.80	0.07	1.51	0.20
4/25/2008	Carter 5	240	52.00	9.69	7.41	0.104	2.17	0.22
5/14/2008	Carter 5	800	13.96	11.54	11.29	0.07	0.17	0.02
5/22/2008	Carter 5	360	16.32	12.15	11.44	0.05	0.66	0.05
6/9/2008	Carter 5	170	12.7	9.6	10.72	0.05	0.00	0.00
6/24/2008	Carter 5	150	15.7	12.6	12.53	0.05	-0.03	0.00
6/25/2007	Hudson	14800	10.06	0.75	0.12	0.04	0.59	0.78
7/11/2007	Hudson	1000	19.51	2.15	0.39	0.06	1.70	0.79
7/23/2007	Hudson	620	39.79	1.07	4.54	0.05	0.00	0.00
8/6/2007	Hudson	590	12.51	1.02	0.28	0.09	0.66	0.64
8/20/2007	Hudson	350	72.18	1.26	0.30	0.05	0.91	0.72
9/3/2007	Hudson	400	71.41	1.27	0.25	0.04	0.98	0.77
9/17/2007	Hudson	270	73.10	1.65	0.48	0.04	1.13	0.69
9/27/2007	Hudson	230	110.86	1.39	0.15	0.05	1.18	0.85
10/22/2007	Hudson	10800	25.16	1.95	0.52	0.11	1.32	0.68
10/29/2007	Hudson	1190	59.40	1.26	0.52	0.06	0.69	0.54
11/12/2007	Hudson	500	22.00	1.26	0.22	0.04	0.99	0.79
11/26/2007	Hudson	1600	43.60	1.86	0.39	0.06	1.41	0.76
12/10/2007	Hudson	140	73.65	1.22	0.20	0.02	1.00	0.82
12/21/2007	Hudson	1200	16.78	1.59	0.38	0.06	1.14	0.72
1/7/2008	Hudson	190	18.63	1.37	0.27	0.02	1.08	0.79
1/23/2008	Hudson	830	50.73	2.98	0.30	0.03	2.65	0.89
2/4/2008	Hudson	1090	39.62	1.21	0.39	0.07	0.76	0.62
2/18/2008	Hudson	1410	28.13	0.95	0.36	0.07	0.52	0.55
3/5/2008	Hudson	3000	22.82	1.02	0.80	0.11	0.11	0.11
3/19/2008	Hudson	4500	29.85	1.68	0.38	0.36	0.94	0.56
4/9/2008	Hudson	850	44.76	3.56	0.49	0.18	2.89	0.81
4/25/2008	Hudson	440	71.72	0.90	0.19	0.06	0.65	0.72
5/14/2008	Hudson	2700	22.19	5.02	0.33	0.10	4.59	0.91
5/22/2008	Hudson	280	16.08	1.09	0.29	0.04	0.76	0.70
6/9/2008	Hudson	280	20.4	1.6	0.24	0.13	1.22	0.77
6/24/2008	Hudson	130	23.5	1.3	0.21	0.04	1.02	0.81
6/25/2007	Wolfpen	99	11.30	1.07	0.36	0.04	0.66	0.62
7/11/2007	Wolfpen	400	15.12	1.29	0.13	0.03	1.13	0.87
7/23/2007	Wolfpen	710	107.88	1.04	0.20	0.06	0.78	0.75
8/6/2007	Wolfpen	280	10.99	0.87	0.27	0.09	0.52	0.59
8/20/2007	Wolfpen	340	75.10	1.25	0.38	0.12	0.75	0.60
9/3/2007	Wolfpen	430	141.35	1.15	0.49	0.04	0.61	0.54
9/17/2007	Wolfpen	310	129.44	1.01	0.36	0.03	0.62	0.61

9/27/2007	Wolfpen	2200	130.34	0.85	0.51	0.03	0.31	0.37
10/22/2007	Wolfpen	3600	111.52	1.06	0.30	0.07	0.69	0.65
10/29/2007	Wolfpen	310	99.27	0.80	0.28	0.06	0.46	0.58
11/12/2007	Wolfpen	90	21.79	1.23	0.34	0.03	0.85	0.70
11/26/2007	Wolfpen	4200	42.29	1.99	0.56	0.03	1.40	0.70
12/10/2007	Wolfpen	340	90.32	0.72	0.44	0.02	0.26	0.36
12/21/2007	Wolfpen	1400	21.40	1.50	0.57	0.08	0.85	0.57
1/7/2008	Wolfpen	130	74.69	0.73	0.33	0.02	0.37	0.51
1/23/2008	Wolfpen	3800	38.91	1.47	0.57	0.15	0.75	0.51
2/4/2008	Wolfpen	310	15.76	0.82	0.25	0.08	0.48	0.59
2/18/2008	Wolfpen	1310	13.24	0.85	0.26	0.08	0.51	0.60
3/5/2008	Wolfpen	4900	28.95	1.14	0.65	0.04	0.45	0.40
3/19/2008	Wolfpen	27000	31.79	1.21	0.29	0.08	0.84	0.70
4/9/2008	Wolfpen	430	77.55	1.07	0.32	0.06	0.69	0.64
4/25/2008	Wolfpen	120	71.72	0.90	0.19	0.06	0.65	0.72
5/14/2008	Wolfpen	37000	22.19	5.02	0.33	0.09	4.60	0.92
5/22/2008	Wolfpen	200	16.08	1.09	0.29	0.06	0.74	0.68
6/9/2008	Wolfpen	100	17.3	0.7	0.27	0.03	0.43	0.59
6/24/2008	Wolfpen	670	26.3	1.3	0.39	0.03	0.89	0.68

APPENDIX II

CATIONS AND SAR IN CARTERS CREEK SUBCATCHMENTS

Date	Site	Na mg L^{-1}	K mg L^{-1}	Mg mg L^{-1}	Ca mg L^{-1}	SAR
9/3/2007	Bee	72.17	4.04	3.79	7.00	7.7
9/17/2007	Bee	130.44	5.72	5.92	16.17	10.0
9/27/2007	Bee	154.45	5.74	4.46	9.88	14.5
10/22/2007	Bee	93.60	6.63	3.99	8.67	9.3
10/29/2007	Bee	101.14	4.22	3.14	8.71	10.6
11/12/2007	Bee	162.29	5.80	4.94	15.55	13.0
11/26/2007	Bee	59.79	4.96	2.69	14.79	5.3
6/25/2007	Bee	31.37	3.61	2.95	3.61	4.2
7/11/2007	Bee	73.54	2.24	4.49	15.46	6.0
7/23/2007	Bee	61.07	4.36	4.59	13.22	5.2
8/6/2007	Bee	65.97	4.35	4.75	18.26	5.0
8/20/2007	Bee	77.46	4.08	3.57	12.16	7.1
6/9/2008	Bee	185.53	3.73	4.29	9.75	17.6
6/24/2008	Bee	258.18	8.93	8.43	21.16	17.0
3/5/2008	Bee	58.91	4.00	4.68	24.67	4.0
3/19/2008	Bee	35.57	3.89	2.35	17.46	3.0
4/9/2008	Bee	99.19	6.79	10.94	70.48	4.1
4/25/2008	Bee	97.66	8.67	6.12	37.28	5.5
5/14/2008	Bee	65.76	2.87	1.83	12.12	6.6
5/22/2008	Bee	136.60	5.44	8.67	20.56	9.0
12/10/2007	Bee	81.56	4.08	4.96	13.97	6.7
12/21/2007	Bee	83.57	5.90	5.88	18.23	6.2
1/7/2008	Bee	90.01	3.81	6.28	24.75	5.9
1/23/2008	Bee	57.83	2.97	3.53	18.36	4.6
2/4/2008	Bee	107.26	5.22	5.66	21.60	7.5
2/18/2008	Bee	56.70	4.46	3.86	20.10	4.3
6/25/2007	Briar 1	20.63	3.30	0.92	9.37	2.4
7/11/2007	Briar 1	89.55	2.29	1.02	7.21	11.7
7/23/2007	Briar 1	87.65	2.90	1.16	7.83	10.9
8/6/2007	Briar 1	73.60	3.44	1.39	12.01	7.6
8/20/2007	Briar 1	73.43	2.93	1.10	5.89	10.3
9/3/2007	Briar 1	134.34	5.35	1.50	3.41	21.6
9/17/2007	Briar 1	118.53	3.23	1.16	6.67	15.7
9/27/2007	Briar 1	167.78	4.17	0.85	6.08	23.9
10/22/2007	Briar 1	46.95	2.25	0.54	5.35	7.3
10/29/2007	Briar 1	121.60	3.82	1.42	4.72	17.8
11/12/2007	Briar 1	139.30	3.22	1.28	8.66	16.5
11/26/2007	Briar 1	35.53	4.76	1.91	13.20	3.4
12/10/2007	Briar 1	134.32	5.08	1.89	6.11	17.2
12/21/2007	Briar 1	71.94	3.61	3.03	15.27	6.2

1/7/2008	Briar 1	61.83	4.10	2.63	16.84	5.2
1/23/2008	Briar 1	27.34	3.04	1.48	9.17	3.1
2/4/2008	Briar 1	31.23	4.20	2.25	18.95	2.6
2/18/2008	Briar 1	43.60	4.57	2.45	17.46	3.7
3/5/2008	Briar 1	41.72	3.17	1.87	11.10	4.3
3/19/2008	Briar 1	23.96	3.05	1.62	15.65	2.2
4/9/2008	Briar 1	55.67	6.50	3.42	30.99	3.6
4/25/2008	Briar 1	162.54	6.71	2.08	11.28	16.5
5/14/2008	Briar 1	46.87	2.95	1.53	14.82	4.4
5/22/2008	Briar 1	132.84	5.32	2.59	14.41	12.0
6/9/2008	Briar 1	212.10	5.45	2.94	7.95	23.1
6/24/2008	Briar 1	263.36	6.31	1.41	5.44	36.8
9/17/2007	Briar 2	43.35	5.18	2.46	9.88	4.5
10/29/2007	Briar 2	64.74	3.84	1.75	7.98	7.6
11/12/2007	Briar 2	65.35	3.71	1.92	7.81	7.7
9/3/2007	Briar 2	43.34	4.66	1.82	5.09	5.9
9/27/2007	Briar 2	73.11	5.48	2.45	5.95	9.0
10/22/2007	Briar 2	42.68	3.96	0.84	5.59	6.3
11/26/2007	Briar 2	43.45	4.73	1.93	14.39	4.0
7/11/2007	Briar 2	72.72	3.51	1.49	9.19	8.3
7/23/2007	Briar 2	34.54	2.55	1.20	9.13	4.0
8/20/2007	Briar 2	27.14	3.63	1.56	7.91	3.3
6/9/2008	Briar 2	65.44	6.91	4.10	21.86	4.8
6/24/2008	Briar 2	67.48	7.42	4.54	18.71	5.1
6/25/2007	Briar 2	26.56	2.56	0.75	7.28	3.5
8/6/2007	Briar 2	20.55	3.95	1.46	15.98	1.9
4/9/2008	Briar 2	30.32	6.46	3.14	32.21	1.9
4/25/2008	Briar 2	37.14	4.18	2.47	17.23	3.1
5/22/2008	Briar 2	47.50	4.49	2.53	15.90	4.1
3/5/2008	Briar 2	22.42	3.20	1.79	17.29	1.9
3/19/2008	Briar 2	27.52	4.46	2.09	16.83	2.4
5/14/2008	Briar 2	45.65	3.14	1.60	12.43	4.6
1/7/2008	Briar 2	36.32	2.69	1.50	9.36	4.1
2/4/2008	Briar 2	23.11	4.43	1.82	18.42	1.9
12/10/2007	Briar 2	65.91	4.80	1.83	6.96	8.1
12/21/2007	Briar 2	31.30	5.41	2.62	15.42	2.7
1/23/2008	Briar 2	24.30	3.20	1.67	14.09	2.3
2/18/2008	Briar 2	20.68	4.54	2.51	18.63	1.7
9/17/2007	Burton 1	85.42	3.04	1.42	5.58	11.8
10/29/2007	Burton 1	118.96	2.24	1.09	4.94	17.8
11/12/2007	Burton 1	163.75	2.27	1.67	9.81	18.0
9/3/2007	Burton 1	87.19	2.83	1.68	5.73	11.6
9/27/2007	Burton 1	152.50	4.51	1.85	4.75	21.3
10/22/2007	Burton 1	54.87	3.61	1.46	7.52	6.8
11/26/2007	Burton 1	63.75	4.13	1.89	10.12	6.8
7/11/2007	Burton 1	62.38	2.61	1.06	7.90	7.8
7/23/2007	Burton 1	79.61	3.37	1.28	6.66	10.5

8/20/2007	Burton 1	79.86	3.71	1.90	6.89	9.8
6/9/2008	Burton 1	218.23	5.76	2.49	6.05	26.7
6/24/2008	Burton 1	298.60	8.11	2.23	9.31	32.3
6/25/2007	Burton 1	16.45	2.20	0.79	8.93	2.0
8/6/2007	Burton 1	76.06	3.76	2.18	17.34	6.5
4/9/2008	Burton 1	61.92	4.68	2.94	33.25	3.9
4/25/2008	Burton 1	65.71	4.59	2.71	11.98	6.3
5/22/2008	Burton 1	87.22	4.04	2.85	15.58	7.5
3/5/2008	Burton 1	49.48	3.21	2.56	15.38	4.4
3/19/2008	Burton 1	33.10	2.13	1.32	14.43	3.2
5/14/2008	Burton 1	66.12	2.89	1.54	7.94	8.0
1/7/2008	Burton 1	104.71	3.97	3.43	13.38	9.3
2/4/2008	Burton 1	59.02	4.22	2.55	12.51	5.6
12/10/2007	Burton 1	60.55	4.32	1.90	11.85	6.1
12/21/2007	Burton 1	53.13	4.10	2.29	11.81	5.2
1/23/2008	Burton 1	53.50	3.22	2.13	11.12	5.4
2/18/2008	Burton 1	45.07	3.58	2.08	14.76	4.1
3/5/2008	Burton 2	74.34	3.02	3.81	16.91	6.0
3/19/2008	Burton 2	46.38	3.14	1.81	15.68	4.2
4/9/2008	Burton 2	56.23	4.06	3.22	31.51	3.6
4/25/2008	Burton 2	78.73	6.24	0.97	6.26	10.9
5/14/2008	Burton 2	45.90	2.44	1.01	10.98	5.0
5/22/2008	Burton 2	84.82	4.56	3.82	9.96	8.2
6/25/2007	Burton 2	23.78	2.20	0.75	7.80	3.1
7/11/2007	Burton 2	91.49	3.41	2.52	6.77	10.8
7/23/2007	Burton 2	92.21	3.08	2.02	9.70	10.0
8/6/2007	Burton 2	77.33	3.04	1.76	12.71	7.6
8/20/2007	Burton 2	101.95	3.16	3.15	5.01	12.4
6/9/2008	Burton 2	338.83	11.13	1.00	5.07	50.9
6/24/2008	Burton 2	237.97	7.51	2.20	4.59	32.3
9/3/2007	Burton 2	123.05	2.72	2.53	4.74	16.1
9/17/2007	Burton 2	106.38	3.55	2.11	6.03	13.4
9/27/2007	Burton 2	62.79	5.14	0.88	5.91	9.0
10/22/2007	Burton 2	64.60	4.04	1.30	7.47	8.1
10/29/2007	Burton 2	140.44	3.20	1.82	3.28	21.8
11/12/2007	Burton 2	199.16	10.48	3.65	8.99	20.0
11/26/2007	Burton 2	43.04	3.01	1.60	12.73	4.3
12/10/2007	Burton 2	76.30	4.10	1.80	7.49	9.2
12/21/2007	Burton 2	68.24	3.64	1.96	10.37	7.2
1/7/2008	Burton 2	84.29	3.15	3.72	12.36	7.6
1/23/2008	Burton 2	65.08	3.55	2.58	9.88	6.7
2/4/2008	Burton 2	71.57	3.80	3.01	15.16	6.2
2/18/2008	Burton 2	46.16	3.38	1.90	12.31	4.6
6/25/2007	Burton 3	17.96	2.98	1.14	9.22	2.1
7/11/2007	Burton 3	74.69	2.69	3.92	9.64	7.2
7/23/2007	Burton 3	60.98	4.25	4.40	11.75	5.4
8/6/2007	Burton 3	71.88	3.58	2.91	11.82	6.9

8/20/2007	Burton 3	62.82	3.57	2.95	10.80	6.2
9/3/2007	Burton 3	169.92	4.42	2.68	3.27	23.9
9/17/2007	Burton 3	66.70	3.51	3.03	10.85	6.5
9/27/2007	Burton 3	74.79	4.95	3.59	9.90	7.3
10/22/2007	Burton 3	68.63	3.63	1.29	8.26	8.3
10/29/2007	Burton 3	76.17	3.63	2.65	8.68	8.2
11/12/2007	Burton 3	176.32	4.51	9.35	14.05	12.6
11/26/2007	Burton 3	58.08	4.36	3.21	21.59	4.4
12/10/2007	Burton 3	127.79	4.18	5.46	10.14	11.4
12/21/2007	Burton 3	124.15	5.89	6.37	21.81	8.5
1/7/2008	Burton 3	140.13	4.51	6.17	14.16	11.1
1/23/2008	Burton 3	69.12	3.42	3.57	14.90	5.9
2/4/2008	Burton 3	85.59	4.08	6.47	26.31	5.5
2/18/2008	Burton 3	75.58	4.52	4.66	18.95	5.7
3/5/2008	Burton 3	83.04	3.30	3.98	15.85	6.8
3/19/2008	Burton 3	41.01	4.14	3.39	23.69	3.0
4/9/2008	Burton 3	207.80	7.26	15.27	92.31	7.5
4/25/2008	Burton 3	123.75	2.71	3.32	10.57	12.0
5/14/2008	Burton 3	59.64	3.36	1.72	12.22	6.0
5/22/2008	Burton 3	199.23	3.29	3.07	14.22	17.7
6/9/2008	Burton 3	192.83	3.51	1.89	13.00	18.7
6/24/2008	Burton 3	246.45	3.73	1.91	5.79	32.1
6/25/2007	Burton 4	18.59	1.88	0.65	7.88	2.4
7/11/2007	Burton 4	42.49	1.39	1.26	9.29	4.9
7/23/2007	Burton 4	54.19	3.28	1.31	7.14	6.9
8/6/2007	Burton 4	51.37	3.46	1.65	13.94	4.9
8/20/2007	Burton 4	41.18	2.74	1.22	7.41	5.2
9/3/2007	Burton 4	61.80	2.94	1.48	7.93	7.5
9/17/2007	Burton 4	127.70	5.38	2.99	10.95	12.5
9/27/2007	Burton 4	181.77	4.81	1.93	9.76	19.7
10/22/2007	Burton 4	106.00	4.93	1.62	8.07	12.6
10/29/2007	Burton 4	65.32	2.81	1.16	7.03	8.5
11/12/2007	Burton 4	117.70	4.24	2.17	14.07	10.9
11/26/2007	Burton 4	78.44	4.44	2.17	18.64	6.5
12/10/2007	Burton 4	81.44	2.30	1.28	7.61	10.2
12/21/2007	Burton 4	58.28	4.06	2.63	15.05	5.2
1/7/2008	Burton 4	57.27	2.36	1.97	11.36	5.8
1/23/2008	Burton 4	55.67	2.62	2.50	13.44	5.2
2/4/2008	Burton 4	50.18	4.00	2.36	17.45	4.2
2/18/2008	Burton 4	57.58	4.38	2.86	20.56	4.5
3/5/2008	Burton 4	42.49	3.24	2.18	14.68	3.9
3/19/2008	Burton 4	46.08	4.03	2.03	15.78	4.1
4/9/2008	Burton 4	73.26	5.81	4.41	36.15	4.3
4/25/2008	Burton 4	88.71	5.45	3.04	15.17	7.7
5/14/2008	Burton 4	39.45	2.19	0.85	9.47	4.7
5/22/2008	Burton 4	81.48	4.51	2.94	16.61	6.9
6/9/2008	Burton 4	157.30	4.16	1.51	7.34	19.5

6/24/2008	Burton 4	230.96	4.39	2.14	6.73	28.1
6/25/2007	Burton 5	42.08	3.42	1.49	11.40	4.4
7/11/2007	Burton 5	54.02	1.85	1.13	8.57	6.5
7/23/2007	Burton 5	71.07	3.91	2.20	7.83	8.2
8/6/2007	Burton 5	70.23	3.75	1.87	10.04	7.6
8/20/2007	Burton 5	76.13	3.70	2.28	5.98	9.5
9/3/2007	Burton 5	94.78	4.46	2.38	5.42	12.1
9/17/2007	Burton 5	46.02	2.34	1.48	10.20	5.0
9/27/2007	Burton 5	26.37	5.13	1.25	14.35	2.5
10/22/2007	Burton 5	27.02	3.02	1.06	7.83	3.4
10/29/2007	Burton 5	99.19	4.16	3.00	7.24	11.1
11/12/2007	Burton 5	170.40	5.04	3.87	8.02	17.5
11/26/2007	Burton 5	38.68	4.19	1.68	13.14	3.8
12/10/2007	Burton 5	57.58	3.86	1.93	12.62	5.6
12/21/2007	Burton 5	63.10	3.76	1.75	14.33	5.9
1/7/2008	Burton 5	85.16	3.32	3.58	20.09	6.5
1/23/2008	Burton 5	57.12	4.27	2.62	13.31	5.3
2/4/2008	Burton 5	57.14	4.12	2.64	16.08	4.9
2/18/2008	Burton 5	32.78	3.70	1.91	15.44	3.0
3/5/2008	Burton 5	78.45	3.56	4.30	15.20	6.5
3/19/2008	Burton 5	46.86	3.32	1.80	15.45	4.3
4/9/2008	Burton 5	49.95	4.52	2.84	30.12	3.3
4/25/2008	Burton 5	51.60	3.70	2.05	16.08	4.6
5/14/2008	Burton 5	45.82	3.74	1.41	11.86	4.7
5/22/2008	Burton 5	64.13	3.83	2.77	17.82	5.3
6/9/2008	Burton 5	186.29	5.35	3.90	13.62	16.2
6/24/2008	Burton 5	243.19	8.65	6.72	19.69	17.1
6/25/2007	Carter 1	20.74	3.40	1.68	9.10	2.3
7/11/2007	Carter 1	33.48	1.86	1.53	11.56	3.5
7/23/2007	Carter 1	45.89	3.80	2.25	8.23	5.2
8/6/2007	Carter 1	42.34	3.61	1.33	12.47	4.3
8/20/2007	Carter 1	33.43	3.63	1.98	10.80	3.5
9/3/2007	Carter 1	50.83	4.92	2.14	6.39	6.3
9/17/2007	Carter 1	25.26	2.75	2.16	13.81	2.4
9/27/2007	Carter 1	52.64	5.26	3.37	7.51	5.7
10/22/2007	Carter 1	44.68	5.18	2.25	8.24	5.0
10/29/2007	Carter 1	41.18	1.87	0.88	6.01	5.9
11/12/2007	Carter 1	66.28	5.26	3.04	7.93	7.2
11/26/2007	Carter 1	16.95	5.57	2.56	16.43	1.5
12/10/2007	Carter 1	75.30	4.94	2.76	9.86	7.7
12/21/2007	Carter 1	34.96	4.31	2.24	11.42	3.5
1/7/2008	Carter 1	52.29	4.03	3.45	16.19	4.4
1/23/2008	Carter 1	29.14	4.62	3.49	10.93	2.8
2/4/2008	Carter 1	16.79	5.27	2.89	20.03	1.3
2/18/2008	Carter 1	25.73	4.48	2.39	13.23	2.4
3/5/2008	Carter 1	48.19	4.12	4.55	22.15	3.4
3/19/2008	Carter 1	23.75	3.13	1.59	12.16	2.4

4/9/2008	Carter 1	40.49	5.40	4.24	32.65	2.5
4/25/2008	Carter 1	40.13	4.70	3.46	25.96	2.8
5/14/2008	Carter 1	49.67	2.79	2.07	10.92	5.1
5/22/2008	Carter 1	44.85	4.70	3.75	16.53	3.7
6/9/2008	Carter 1	83.98	4.63	4.46	14.34	7.0
6/24/2008	Carter 1	78.96	6.06	7.06	26.23	5.0
6/25/2007	Carter 2	4.57	0.70	0.42	5.08	0.7
7/11/2007	Carter 2	80.80	4.40	3.93	6.94	8.6
7/23/2007	Carter 2	68.30	3.83	2.61	6.12	8.2
8/6/2007	Carter 2	35.06	4.31	2.34	14.33	3.2
8/20/2007	Carter 2	34.29	3.37	2.10	6.00	4.3
9/3/2007	Carter 2	167.25	14.08	2.32	3.26	24.5
9/17/2007	Carter 2	98.59	4.72	2.80	8.27	10.7
9/27/2007	Carter 2	174.33	5.72	2.17	4.62	23.7
10/22/2007	Carter 2	63.79	5.95	2.76	10.44	6.4
10/29/2007	Carter 2	30.50	2.27	1.54	7.01	3.8
11/12/2007	Carter 2	104.29	4.98	3.28	4.65	12.8
11/26/2007	Carter 2	40.98	5.24	2.87	10.76	4.1
12/10/2007	Carter 2	147.34	4.70	2.90	3.16	20.4
12/21/2007	Carter 2	58.20	5.50	3.97	15.36	4.8
1/7/2008	Carter 2	39.33	2.82	2.87	17.62	3.2
1/23/2008	Carter 2	40.31	3.89	2.48	6.48	4.8
2/4/2008	Carter 2	34.23	4.72	4.11	28.18	2.3
2/18/2008	Carter 2	38.55	4.53	3.88	23.46	2.7
3/5/2008	Carter 2	38.19	3.20	2.51	15.19	3.4
3/19/2008	Carter 2	27.94	3.12	2.20	16.98	2.4
4/9/2008	Carter 2	84.41	5.46	5.09	38.71	4.8
4/25/2008	Carter 2	119.13	4.50	2.52	9.47	12.6
5/14/2008	Carter 2	71.28	2.48	1.69	7.74	8.6
5/22/2008	Carter 2	67.80	4.29	4.57	14.77	5.6
6/9/2008	Carter 2					
6/24/2008	Carter 2	237.19	6.45	5.54	13.58	19.4
6/25/2007	Carter 3	15.01	3.89	1.99	13.11	1.4
7/11/2007	Carter 3	37.96	3.74	2.81	10.85	3.8
7/23/2007	Carter 3	44.41	2.97	2.44	8.32	4.9
8/6/2007	Carter 3	19.95	3.80	3.85	21.89	1.5
8/20/2007	Carter 3	28.86	4.12	3.36	7.30	3.1
9/17/2007	Carter 3	37.43	7.01	3.74	16.11	3.1
9/27/2007	Carter 3	42.63	6.14	3.86	15.20	3.6
10/22/2007	Carter 3	19.78	7.96	2.99	13.15	1.8
10/29/2007	Carter 3	48.56	6.35	3.27	11.22	4.6
11/12/2007	Carter 3	47.19	6.78	4.41	13.76	4.0
11/26/2007	Carter 3	29.77	5.68	2.32	13.85	2.8
12/10/2007	Carter 3	27.96	5.76	2.92	12.27	2.6
12/21/2007	Carter 3	28.11	5.70	3.11	16.22	2.4
1/7/2008	Carter 3	42.00	5.37	3.64	18.37	3.3
1/23/2008	Carter 3	34.62	5.15	3.29	7.93	3.7

2/4/2008	Carter 3	13.95	5.39	3.28	22.71	1.0
2/18/2008	Carter 3	36.43	4.33	2.82	17.81	3.0
3/5/2008	Carter 3	29.05	4.63	3.91	18.89	2.2
3/19/2008	Carter 3	25.09	2.69	1.71	13.36	2.4
4/9/2008	Carter 3	19.01	4.56	3.94	34.86	1.2
4/25/2008	Carter 3	22.47	3.76	3.02	16.86	1.9
5/14/2008	Carter 3	32.74	4.01	3.42	16.84	2.7
5/22/2008	Carter 3	53.15	4.20	3.71	18.47	4.2
6/9/2008	Carter 3	54.13	5.25	3.67	11.31	5.1
6/25/2007	Carter 4	15.45	3.11	0.67	6.16	2.2
7/11/2007	Carter 4	121.97	3.77	1.35	8.91	14.2
7/23/2007	Carter 4	116.59	4.25	1.48	10.37	12.7
8/6/2007	Carter 4	147.85	5.97	1.88	14.04	13.9
8/20/2007	Carter 4	141.65	5.46	1.44	10.30	15.5
9/3/2007	Carter 4	53.12	2.03	0.53	9.62	6.4
9/17/2007	Carter 4	96.82	3.55	1.19	9.46	11.2
9/27/2007	Carter 4	235.67	8.92	1.40	6.59	30.8
10/22/2007	Carter 4	208.64	8.70	1.42	6.01	28.1
10/29/2007	Carter 4	171.37	5.37	1.02	5.55	24.8
11/12/2007	Carter 4	148.72	5.64	0.96	7.40	19.3
11/26/2007	Carter 4	66.62	5.92	2.04	13.09	6.4
12/10/2007	Carter 4	165.91	5.47	1.48	9.68	18.6
12/21/2007	Carter 4	146.58	5.56	1.81	9.64	16.1
1/7/2008	Carter 4	153.70	4.96	1.35	10.39	16.9
1/23/2008	Carter 4	157.94	6.15	1.84	13.49	15.1
2/4/2008	Carter 4	98.98	4.77	1.33	6.28	13.2
2/18/2008	Carter 4	63.82	4.43	2.11	13.13	6.1
3/5/2008	Carter 4	122.56	5.93	2.12	14.95	11.1
3/19/2008	Carter 4	26.54	3.25	1.74	14.19	2.5
4/9/2008	Carter 4	193.26	8.76	2.53	17.71	16.1
4/25/2008	Carter 4	193.81	9.63	2.64	14.55	17.4
5/14/2008	Carter 4	205.43	7.44	1.82	11.75	20.8
5/22/2008	Carter 4	229.21	7.77	2.31	12.80	21.9
6/9/2008	Carter 4	251.19	10.27	2.01	10.81	26.0
6/24/2008	Carter 4	264.99	10.70	1.95	10.30	28.1
6/25/2007	Carter 5	80.94	5.27	2.56	5.27	10.2
7/11/2007	Carter 5	123.71	4.33	1.95	9.65	13.4
7/23/2007	Carter 5	144.92	5.58	2.57	13.70	13.3
8/6/2007	Carter 5	119.87	5.21	2.12	15.14	10.8
8/20/2007	Carter 5	168.19	6.79	2.41	13.57	15.6
9/3/2007	Carter 5	208.02	8.32	1.99	9.36	22.8
9/17/2007	Carter 5	107.09	3.88	1.24	10.24	11.9
9/27/2007	Carter 5	252.60	9.09	2.25	8.74	27.9
10/22/2007	Carter 5	166.22	6.34	1.47	7.62	20.4
10/29/2007	Carter 5	214.69	8.99	1.90	7.66	25.4
11/12/2007	Carter 5	197.11	6.56	1.57	6.96	24.8
11/26/2007	Carter 5	67.01	5.58	2.10	12.84	6.5

12/10/2007	Carter 5	202.88	8.62	1.97	9.23	22.4
12/21/2007	Carter 5	145.13	6.30	2.70	15.16	12.8
1/7/2008	Carter 5	227.25	8.13	2.29	12.73	21.8
1/23/2008	Carter 5	103.63	4.36	1.84	13.06	10.1
2/4/2008	Carter 5	168.29	8.10	2.73	12.77	15.8
2/18/2008	Carter 5	60.73	4.53	2.23	11.90	6.0
3/5/2008	Carter 5	87.84	4.89	2.39	14.44	8.0
3/19/2008	Carter 5	39.71	4.75	2.09	14.05	3.7
4/9/2008	Carter 5	187.10	8.80	3.54	22.42	13.7
4/25/2008	Carter 5	132.10	5.24	2.15	13.59	12.4
5/14/2008	Carter 5	199.89	6.87	2.13	10.99	20.4
5/22/2008	Carter 5	210.89	7.95	3.10	12.10	19.8
6/9/2008	Carter 5	226.07	7.91	2.04	10.50	23.7
6/24/2008	Carter 5	270.80	9.86	2.52	11.68	26.5
6/25/2007	Hudson	13.90	2.59	0.51	2.59	2.9
7/11/2007	Hudson	33.67	1.80	2.07	11.78	3.4
7/23/2007	Hudson	75.24	3.59	2.75	15.14	6.6
8/6/2007	Hudson	104.82	4.63	2.50	17.20	8.8
8/20/2007	Hudson	100.36	4.37	2.22	13.42	9.5
9/3/2007	Hudson	76.03	2.90	1.31	10.42	8.3
9/17/2007	Hudson	131.05	4.15	1.27	8.32	15.8
9/27/2007	Hudson	217.21	5.16	1.59	7.24	26.9
10/22/2007	Hudson	157.54	5.20	1.24	5.22	22.8
10/29/2007	Hudson	127.07	2.31	0.73	5.34	19.3
11/12/2007	Hudson	146.20	2.91	0.81	7.36	19.3
11/26/2007	Hudson	68.53	3.41	1.02	8.15	8.5
12/10/2007	Hudson	143.98	4.91	2.71	12.91	13.5
12/21/2007	Hudson	83.12	4.40	1.88	11.33	8.5
1/7/2008	Hudson	89.89	4.46	2.51	14.14	8.2
1/23/2008	Hudson	70.49	4.77	1.85	12.39	7.0
2/4/2008	Hudson	70.94	4.82	2.62	14.99	6.3
2/18/2008	Hudson	48.24	3.14	1.21	8.63	5.8
3/5/2008	Hudson	58.87	3.84	1.97	11.80	5.9
3/19/2008	Hudson	27.73	3.31	1.55	14.34	2.6
4/9/2008	Hudson	98.70	6.46	5.32	30.28	6.2
4/25/2008	Hudson	170.52	4.74	3.46	7.20	18.5
5/14/2008	Hudson	108.86	4.66	5.26	20.00	7.9
5/22/2008	Hudson	107.46	4.37	4.30	18.46	8.3
6/9/2008	Hudson	188.88	6.10	2.53	16.32	16.2
6/24/2008	Hudson	147.81	4.31	2.03	14.62	13.6
6/25/2007	Wolfpen	69.29	4.88	2.04	14.36	6.4
7/11/2007	Wolfpen	74.88	1.45	1.20	5.30	10.8
7/23/2007	Wolfpen	203.20	3.69	2.94	6.79	23.2
8/6/2007	Wolfpen	166.00	4.00	1.55	10.83	17.7
8/20/2007	Wolfpen	91.48	2.48	1.38	5.27	13.0
9/3/2007	Wolfpen	339.47	6.89	3.22	9.41	34.5
9/17/2007	Wolfpen	311.76	3.70	2.51	7.88	35.0

9/27/2007	Wolfpen	380.51	4.70	2.43	8.32	42.2
10/22/2007	Wolfpen	267.68	4.35	2.26	7.82	30.7
10/29/2007	Wolfpen	275.36	3.05	2.12	7.16	32.8
11/12/2007	Wolfpen	144.80	2.24	1.74	13.12	14.1
11/26/2007	Wolfpen	97.38	4.18	2.00	10.39	10.2
12/10/2007	Wolfpen	224.04	3.83	2.56	8.28	24.7
12/21/2007	Wolfpen	223.64	4.81	4.80	12.73	19.2
1/7/2008	Wolfpen	212.06	2.43	2.80	11.45	20.6
1/23/2008	Wolfpen	42.71	1.60	1.31	9.26	4.9
2/4/2008	Wolfpen	223.58	4.14	4.95	13.01	18.9
2/18/2008	Wolfpen	73.96	2.60	2.38	14.19	6.8
3/5/2008	Wolfpen	72.73	3.07	2.36	11.51	7.2
3/19/2008	Wolfpen	24.48	2.12	1.25	15.28	2.3
4/9/2008	Wolfpen	292.55	4.28	4.47	30.40	18.5
4/25/2008	Wolfpen	289.12	3.84	3.19	8.51	30.3
5/14/2008	Wolfpen	64.89	2.14	0.76	6.55	9.0
5/22/2008	Wolfpen	289.71	4.12	4.66	17.27	22.6
6/9/2008	Wolfpen	355.45	4.63	2.41	9.19	38.2
6/24/2008	Wolfpen	376.87	5.63	2.81	12.17	35.8

APPENDIX III

ANIONS IN CARTERS CREEK SUBCATCHMENTS

Date	Site	F mg L^{-1}	Cl mg L^{-1}	NO_2 mg L^{-1}	Br mg L^{-1}	SO_4 mg L^{-1}	PO_4 mg L^{-1}
9/3/2007	Bee	0.31	62.18	0.00	0.24	66.99	0.36
9/17/2007	Bee	0.42	105.43	0.09	0.22	90.54	0.36
9/27/2007	Bee	0.47	78.43	0.23	0.40	64.97	3.61
10/22/2007	Bee	0.26	57.12	0.00	0.17	67.12	0.21
10/29/2007	Bee	0.29	50.88	0.05	0.14	49.42	0.29
11/12/2007	Bee	0.49	91.64	0.00	0.27	82.73	0.12
11/26/2007	Bee	0.20	30.98	0.00	0.00	37.55	0.78
6/25/2007	Bee	0.10	22.82	0.00	0.00	29.63	0.62
7/11/2007	Bee	0.23	93.90	0.00	0.45	160.38	0.27
7/23/2007	Bee	0.16	37.13	0.00	0.15	60.42	0.64
8/6/2007	Bee	0.14	48.85	0.00	0.19	64.20	0.59
8/20/2007	Bee	0.27	49.92	0.00	0.15	61.03	0.67
6/9/2008	Bee	0.29	93.11	0.00	0.72	102.34	0.55
6/24/2008	Bee	0.37	150.82	0.00	0.50	172.60	0.43
3/5/2008	Bee	0.19	44.01	0.14	0.13	59.66	0.93
3/19/2008	Bee	1.27	18.69	0.00	0.00	21.80	0.11
4/9/2008	Bee	1.95	82.29	0.00	0.37	106.44	0.05
4/25/2008	Bee	0.23	65.94	0.00	0.21	98.99	3.48
5/14/2008	Bee	0.13	30.66	0.00	0.00	28.57	0.51
5/22/2008	Bee	0.30	90.30	0.00	0.31	119.92	0.21
12/10/2007	Bee	0.20	48.17	0.00	0.14	0.00	0.04
12/21/2007	Bee	0.19	68.53	0.16	0.22	291.84	0.00
1/7/2008	Bee	0.16	61.46	0.00	0.21	101.37	0.12
1/23/2008	Bee	0.15	40.83	0.13	0.11	52.92	0.00
2/4/2008	Bee	0.21	64.79	0.16	0.19	80.86	0.18
2/18/2008	Bee	0.16	30.44	0.05	0.00	45.65	0.53
6/25/2007	Briar 1	0.06	8.23	0.00	0.21	5.29	0.80
7/11/2007	Briar 1	0.42	49.11	0.00	0.18	16.01	1.29
7/23/2007	Briar 1	0.20	35.41	0.07	0.10	12.24	1.01
8/6/2007	Briar 1	0.15	24.35	0.00	0.00	38.76	0.48
8/20/2007	Briar 1	0.22	30.81	0.00	0.00	20.11	0.60
9/3/2007	Briar 1	0.28	45.15	0.22	0.00	28.65	0.24
9/17/2007	Briar 1	0.41	52.34	0.00	0.00	14.01	0.43
9/27/2007	Briar 1	0.48	55.70	0.00	0.14	9.21	0.16
10/22/2007	Briar 1	0.17	16.29	0.00	0.00	5.77	0.16
10/29/2007	Briar 1	0.26	41.65	0.10	0.00	12.20	0.27
11/12/2007	Briar 1	0.37	42.30	0.12	0.00	8.75	0.56
11/26/2007	Briar 1	0.10	13.25	0.10	0.00	10.34	0.96
12/10/2007	Briar 1	0.31	45.08	0.00	0.11	10.86	0.61
12/21/2007	Briar 1	0.19	38.17	0.05	0.14	13.13	0.34

1/7/2008	Briar 1	0.17	30.07	0.04	0.00	14.87	0.23
1/23/2008	Briar 1	0.14	15.33	0.00	0.00	8.77	0.07
2/4/2008	Briar 1	0.12	18.33	0.00	0.00	11.53	0.71
2/18/2008	Briar 1	0.15	16.97	0.00	0.00	9.68	0.79
3/5/2008	Briar 1	1.61	23.55	0.00	0.00	11.48	0.57
3/19/2008	Briar 1	1.26	14.51	0.00	0.00	8.29	0.85
4/9/2008	Briar 1	0.18	30.56	0.00	0.09	18.81	0.21
4/25/2008	Briar 1	0.40	56.74	0.18	0.11	20.20	0.98
5/14/2008	Briar 1	0.13	18.86	0.00	0.00	10.09	0.13
5/22/2008	Briar 1	0.29	47.99	0.00	0.14	15.06	0.55
6/9/2008	Briar 1	0.41	79.89	0.00	0.28	44.35	0.90
6/24/2008	Briar 1	0.52	74.41	0.00	0.19	11.22	0.60
9/17/2007	Briar 2	0.20	17.09	0.00	0.00	6.43	0.23
10/29/2007	Briar 2	0.25	26.07	0.00	0.00	9.27	0.07
11/12/2007	Briar 2	0.21	26.43	0.00	0.00	7.13	0.04
9/3/2007	Briar 2	0.17	19.06	0.00	0.00	8.25	0.00
9/27/2007	Briar 2	0.20	35.50	0.00	0.13	6.88	0.06
10/22/2007	Briar 2	0.17	18.81	0.00	0.00	5.51	0.08
11/26/2007	Briar 2	0.16	24.97	0.12	0.00	16.83	0.51
7/11/2007	Briar 2	0.14	11.68	0.00	0.00	7.34	0.99
7/23/2007	Briar 2	0.09	7.47	0.00	0.00	4.93	0.83
8/20/2007	Briar 2	0.14	12.08	0.00	0.00	6.37	0.50
6/9/2008	Briar 2	0.22	27.91	0.00	0.13	9.30	0.42
6/24/2008	Briar 2	0.19	29.49	0.00	0.15	7.02	0.25
6/25/2007	Briar 2	0.05	7.36	0.00	0.19	4.62	0.73
8/6/2007	Briar 2	0.07	9.34	0.00	0.00	7.22	0.80
4/9/2008	Briar 2	0.18	18.02	0.00	0.10	13.26	1.01
4/25/2008	Briar 2	0.11	16.85	0.00	0.00	8.01	0.66
5/22/2008	Briar 2	0.14	18.56	0.00	0.00	8.62	0.56
3/5/2008	Briar 2	1.21	16.32	0.00	0.00	8.30	0.05
3/19/2008	Briar 2	0.88	12.49	0.04	0.00	8.03	1.29
5/14/2008	Briar 2	0.13	21.43	0.00	0.00	8.26	0.49
1/7/2008	Briar 2	0.10	18.29	0.00	0.00	6.95	0.06
2/4/2008	Briar 2	0.10	13.45	0.05	0.00	8.01	0.86
12/10/2007	Briar 2	0.15	23.99	0.00	0.00	7.56	0.22
12/21/2007	Briar 2	0.10	16.77	0.11	0.00	8.22	0.61
1/23/2008	Briar 2	0.14	16.69	0.00	0.00	7.91	0.12
2/18/2008	Briar 2	0.11	14.57	0.00	0.00	11.30	0.86
9/17/2007	Burton 1	0.30	31.95	0.00	0.36	38.86	0.00
10/29/2007	Burton 1	0.55	51.29	0.00	0.13	14.87	0.11
11/12/2007	Burton 1	0.41	66.31	0.00	0.21	26.42	0.19
9/3/2007	Burton 1	0.28	40.11	0.00	0.00	31.54	0.38
9/27/2007	Burton 1	0.36	57.75	0.00	0.41	35.64	0.12
10/22/2007	Burton 1	0.20	30.85	0.00	0.00	17.97	0.14
11/26/2007	Burton 1	0.17	23.89	0.00	0.09	22.71	0.63
7/11/2007	Burton 1	0.33	22.16	0.00	0.29	49.96	0.75
7/23/2007	Burton 1	0.17	22.05	0.00	0.32	47.02	0.72

8/20/2007	Burton 1	0.23	28.73	0.00	0.15	34.53	0.34
6/9/2008	Burton 1	0.44	121.71	0.00	0.28	33.32	0.18
6/24/2008	Burton 1	0.63	124.05	0.00	0.43	33.20	0.63
6/25/2007	Burton 1	0.10	11.21	0.00	0.00	5.56	0.74
8/6/2007	Burton 1	0.23	23.53	0.00	0.33	42.70	0.67
4/9/2008	Burton 1	0.13	29.59	0.00	0.20	35.97	0.20
4/25/2008	Burton 1	0.16	31.79	0.00	0.21	38.32	0.43
5/22/2008	Burton 1	0.23	39.48	0.00	0.17	33.93	0.02
3/5/2008	Burton 1	1.74	28.21	0.00	0.09	22.78	0.65
3/19/2008	Burton 1	1.14	12.32	0.00	0.00	15.16	0.67
5/14/2008	Burton 1	0.18	28.42	0.06	0.17	18.67	0.59
1/7/2008	Burton 1	0.26	47.11	0.00	0.26	40.70	0.11
2/4/2008	Burton 1	0.17	29.57	0.00	0.21	39.25	0.15
12/10/2007	Burton 1	0.21	22.00	0.00	0.14	27.22	0.64
12/21/2007	Burton 1	0.11	26.60	0.00	0.14	31.42	0.64
1/23/2008	Burton 1	0.16	29.50	0.00	0.16	30.05	0.19
2/18/2008	Burton 1	0.15	16.02	0.00	0.11	23.10	0.21
3/5/2008	Burton 2	2.04	38.17	0.00	2.61	37.86	0.05
3/19/2008	Burton 2	1.61	17.87	0.00	0.12	24.29	0.04
4/9/2008	Burton 2	0.12	15.71	0.00	0.11	22.81	0.28
4/25/2008	Burton 2	0.19	35.69	0.06	0.18	32.89	2.18
5/14/2008	Burton 2	0.15	17.52	0.06	0.00	9.45	0.37
5/22/2008	Burton 2	0.20	38.43	0.00	0.20	37.42	0.22
6/25/2007	Burton 2	0.10	8.87	0.00	0.00	5.24	0.92
7/11/2007	Burton 2	0.27	19.94	0.00	0.27	49.86	0.75
7/23/2007	Burton 2	0.23	29.41	0.00	0.24	40.85	0.68
8/6/2007	Burton 2	0.24	24.95	0.00	0.30	37.65	0.50
8/20/2007	Burton 2	0.26	44.47	0.00	0.28	40.50	0.00
6/9/2008	Burton 2	0.84	151.22	0.00	0.58	53.21	0.00
6/24/2008	Burton 2	0.56	111.20	0.00	0.43	23.22	0.28
9/3/2007	Burton 2	0.30	44.50	0.00	0.00	34.24	0.00
9/17/2007	Burton 2	0.28	35.36	0.00	0.35	36.69	0.00
9/27/2007	Burton 2	0.25	25.81	0.00	0.11	18.57	0.13
10/22/2007	Burton 2	0.17	32.13	0.00	0.09	11.93	0.44
10/29/2007	Burton 2	0.00	0.18	0.00	0.00	0.00	0.06
11/12/2007	Burton 2	0.46	84.51	0.04	0.34	37.56	0.29
11/26/2007	Burton 2	0.16	16.57	0.21	0.00	15.22	0.64
12/10/2007	Burton 2	0.21	27.29	0.00	0.16	28.13	0.19
12/21/2007	Burton 2	0.19	25.20	0.11	0.12	26.52	0.27
1/7/2008	Burton 2	0.18	35.68	0.09	0.16	32.79	0.03
1/23/2008	Burton 2	0.14	30.58	0.06	0.23	36.65	0.12
2/4/2008	Burton 2	0.19	29.33	0.07	0.18	35.74	1.29
2/18/2008	Burton 2	0.16	18.92	0.07	0.09	19.24	0.70
6/25/2007	Burton 3	0.07	9.12	0.00	0.00	8.25	0.93
7/11/2007	Burton 3	0.16	51.00	0.00	0.27	86.40	0.32
7/23/2007	Burton 3	0.20	40.47	0.00	0.15	51.68	0.69
8/6/2007	Burton 3	0.22	31.45	0.00	0.00	20.15	0.43

8/20/2007	Burton 3	0.17	30.66	0.00	0.00	29.40	0.40
9/3/2007	Burton 3	0.38	59.72	0.00	0.00	25.14	0.63
9/17/2007	Burton 3	0.18	34.45	0.00	0.00	30.16	0.49
9/27/2007	Burton 3	0.19	41.55	0.00	0.13	33.32	0.05
10/22/2007	Burton 3	0.24	29.21	0.00	0.00	13.75	0.01
10/29/2007	Burton 3	0.15	34.98	0.05	0.12	27.84	0.05
11/12/2007	Burton 3	0.24	127.96	0.05	0.42	179.09	0.26
11/26/2007	Burton 3	0.30	34.78	0.00	0.00	37.09	0.32
12/10/2007	Burton 3	0.17	81.98	0.04	0.25	91.90	0.07
12/21/2007	Burton 3	0.20	84.31	0.06	0.22	81.25	0.22
1/7/2008	Burton 3	0.18	85.90	0.05	0.25	75.33	0.10
1/23/2008	Burton 3	0.15	38.39	0.08	0.09	34.59	0.16
2/4/2008	Burton 3	0.16	60.03	0.07	0.18	85.00	0.17
2/18/2008	Burton 3	0.18	41.09	0.00	0.11	42.31	0.28
3/5/2008	Burton 3	1.68	34.10	0.07	0.09	29.39	0.31
3/19/2008	Burton 3	1.29	18.23	0.00	0.00	17.93	0.54
4/9/2008	Burton 3	0.20	120.43	0.00	0.43	138.79	0.14
4/25/2008	Burton 3	0.27	59.88	0.05	0.16	39.43	0.46
5/14/2008	Burton 3	0.19	23.11	0.06	0.00	18.14	0.41
5/22/2008	Burton 3	0.37	78.37	0.00	0.18	32.19	0.79
6/9/2008	Burton 3	0.54	62.97	0.00	0.16	21.28	0.91
6/24/2008	Burton 3	0.47	69.67	0.00	0.16	15.76	0.60
6/25/2007	Burton 4	0.06	7.69	0.00	0.00	3.90	0.97
7/11/2007	Burton 4	0.23	27.34	0.00	0.13	25.61	0.90
7/23/2007	Burton 4	0.14	17.34	0.00	0.00	11.33	0.75
8/6/2007	Burton 4	0.20	22.97	0.00	0.00	10.50	0.94
8/20/2007	Burton 4	0.15	18.12	0.00	0.00	10.41	0.69
9/3/2007	Burton 4	0.13	18.68	0.00	0.00	10.12	0.64
9/17/2007	Burton 4	0.31	69.09	0.06	0.22	23.83	0.80
9/27/2007	Burton 4	0.46	69.16	0.05	0.21	11.20	1.16
10/22/2007	Burton 4	0.27	44.68	0.00	0.12	13.23	0.47
10/29/2007	Burton 4	0.20	21.00	0.00	0.00	5.64	0.48
11/12/2007	Burton 4	0.31	47.70	0.05	0.14	11.35	1.77
11/26/2007	Burton 4	0.37	36.86	0.09	0.00	22.37	0.48
12/10/2007	Burton 4	0.29	31.85	0.00	0.00	8.49	0.47
12/21/2007	Burton 4	0.16	31.62	0.11	0.00	19.35	0.25
1/7/2008	Burton 4	0.15	28.17	0.16	0.00	12.39	0.16
1/23/2008	Burton 4	0.13	35.95	0.09	0.18	18.80	0.40
2/4/2008	Burton 4	0.17	26.09	0.29	0.00	14.31	1.34
2/18/2008	Burton 4	0.24	29.11	0.11	0.00	13.43	1.00
3/5/2008	Burton 4	1.53	23.32	0.15	0.00	9.72	0.81
3/19/2008	Burton 4	1.71	22.41	0.00	0.00	10.45	0.95
4/9/2008	Burton 4	0.21	44.54	0.24	0.12	24.58	1.31
4/25/2008	Burton 4	0.20	46.61	0.06	0.12	21.02	0.81
5/14/2008	Burton 4	0.13	14.43	0.06	0.00	8.25	0.46
5/22/2008	Burton 4	0.19	39.37	0.08	0.10	18.70	0.30
6/9/2008	Burton 4	0.36	49.27	0.00	0.12	10.61	0.63

6/24/2008	Burton 4	0.52	78.11	0.00	0.24	14.46	0.76
6/25/2007	Burton 5	0.16	13.31	0.00	0.20	29.63	0.96
7/11/2007	Burton 5	0.24	16.92	0.00	0.33	53.50	0.87
7/23/2007	Burton 5	0.16	23.40	0.00	0.36	54.53	0.81
8/6/2007	Burton 5	0.17	21.52	0.00	0.35	45.71	0.67
8/20/2007	Burton 5	0.23	26.52	0.00	0.39	51.41	0.40
9/3/2007	Burton 5	0.26	34.77	0.00	0.40	51.95	0.40
9/17/2007	Burton 5	0.22	23.87	0.00	0.38	41.83	0.21
9/27/2007	Burton 5	0.24	14.16	0.00	0.10	19.08	0.10
10/22/2007	Burton 5	0.11	14.91	0.05	0.00	10.69	0.10
10/29/2007	Burton 5	0.30	55.25	0.04	0.43	61.24	0.09
11/12/2007	Burton 5	0.47	84.87	0.00	0.44	57.40	0.08
11/26/2007	Burton 5	0.21	20.05	0.00	0.14	28.86	0.63
12/10/2007	Burton 5	0.18	19.31	0.06	0.14	27.12	1.08
12/21/2007	Burton 5	0.22	22.79	0.07	0.12	26.39	0.82
1/7/2008	Burton 5	0.20	36.22	0.00	0.26	51.07	0.16
1/23/2008	Burton 5	0.16	22.62	0.07	0.22	42.57	0.35
2/4/2008	Burton 5	0.18	23.47	0.00	0.22	38.30	0.21
2/18/2008	Burton 5	0.15	12.00	0.00	0.11	22.15	0.20
3/5/2008	Burton 5	1.43	12.56	0.00	0.10	22.13	0.23
3/19/2008	Burton 5	1.44	16.46	0.00	0.13	25.11	0.59
4/9/2008	Burton 5	0.13	14.56	0.00	0.10	20.58	0.69
4/25/2008	Burton 5	0.19	23.94	0.00	0.15	27.18	0.95
5/14/2008	Burton 5	0.17	15.99	0.06	0.00	15.02	0.45
5/22/2008	Burton 5	0.18	26.20	0.00	0.18	30.49	0.00
6/9/2008	Burton 5	0.44	81.22	0.00	0.31	54.03	0.57
6/24/2008	Burton 5	0.46	121.23	0.00	0.55	95.20	0.43
6/25/2007	Carter 1	0.12	11.68	0.00	0.00	7.65	0.49
7/11/2007	Carter 1	0.11	8.29	0.00	0.11	6.97	0.47
7/23/2007	Carter 1	0.13	11.64	0.00	0.00	11.65	0.48
8/6/2007	Carter 1	0.12	10.87	0.12	0.00	11.06	0.57
8/20/2007	Carter 1	0.14	15.82	0.00	0.00	20.47	0.18
9/3/2007	Carter 1	0.17	21.77	0.00	0.00	17.58	0.37
9/17/2007	Carter 1	0.11	25.95	0.00	0.00	16.18	0.31
9/27/2007	Carter 1	0.13	40.26	0.00	0.14	16.32	0.01
10/22/2007	Carter 1	0.19	27.16	0.04	0.08	16.22	0.12
10/29/2007	Carter 1	0.19	13.01	0.18	0.00	6.52	0.03
11/12/2007	Carter 1	0.21	48.64	0.28	0.15	19.55	0.07
11/26/2007	Carter 1	0.06	11.07	0.00	0.00	9.58	0.48
12/10/2007	Carter 1	0.18	35.30	0.20	0.09	18.84	0.18
12/21/2007	Carter 1	0.16	18.13	0.26	0.00	10.51	0.10
1/7/2008	Carter 1	0.14	29.01	0.00	0.09	16.99	0.06
1/23/2008	Carter 1	0.11	26.45	0.05	0.13	13.60	0.03
2/4/2008	Carter 1	0.10	11.91	0.05	0.00	9.32	0.32
2/18/2008	Carter 1	0.11	12.10	0.11	0.00	8.71	0.51
3/5/2008	Carter 1	0.09	38.64	0.20	0.22	15.98	0.00
3/19/2008	Carter 1	1.17	10.15	0.08	0.00	6.34	0.31

4/9/2008	Carter 1	1.28	16.20	0.23	0.00	10.28	0.35
4/25/2008	Carter 1	0.14	25.24	0.14	0.11	16.96	0.42
5/14/2008	Carter 1	0.10	21.83	0.00	0.00	9.84	0.49
5/22/2008	Carter 1	0.13	24.83	0.00	0.09	13.36	0.00
6/9/2008	Carter 1	0.17	49.20	0.00	0.17	13.13	0.46
6/24/2008	Carter 1	0.15	67.65	0.00	0.28	24.37	0.27
6/25/2007	Carter 2	0.03	2.61	0.00	0.00	2.18	0.15
7/11/2007	Carter 2	0.15	16.36	0.00	0.14	18.49	0.49
7/23/2007	Carter 2	0.14	15.47	0.00	0.00	12.71	0.47
8/6/2007	Carter 2	0.14	15.87	0.07	0.00	15.93	0.45
8/20/2007	Carter 2	0.10	10.42	0.00	0.00	16.12	0.26
9/3/2007	Carter 2	0.60	88.73	3.08	0.00	116.82	0.30
9/17/2007	Carter 2	0.32	45.03	0.19	0.21	34.57	0.16
9/27/2007	Carter 2	0.51	75.64	1.47	0.23	30.01	0.94
10/22/2007	Carter 2	0.17	30.41	0.17	0.10	18.35	0.01
10/29/2007	Carter 2	0.13	11.14	0.10	0.00	6.90	0.03
11/12/2007	Carter 2	0.23	50.93	0.00	0.16	31.95	0.10
11/26/2007	Carter 2	0.12	18.00	0.08	0.00	13.27	0.22
12/10/2007	Carter 2	0.27	56.10	0.48	0.22	19.34	0.09
12/21/2007	Carter 2	0.19	33.06	0.00	0.11	21.82	0.21
1/7/2008	Carter 2	0.12	18.42	0.00	0.00	11.60	0.11
1/23/2008	Carter 2	0.13	23.21	0.00	0.00	16.23	0.00
2/4/2008	Carter 2	0.13	21.59	0.12	0.10	15.84	0.31
2/18/2008	Carter 2	0.13	16.95	0.00	0.00	11.82	0.09
3/5/2008	Carter 2	1.55	14.94	0.00	0.00	10.60	0.36
3/19/2008	Carter 2	1.02	7.13	0.00	0.00	5.51	0.36
4/9/2008	Carter 2	0.15	34.19	0.00	0.14	15.74	0.00
4/25/2008	Carter 2	0.30	42.45	0.07	0.12	14.56	1.16
5/14/2008	Carter 2	0.20	30.43	0.00	0.09	9.22	0.35
5/22/2008	Carter 2	0.18	35.07	0.00	0.15	17.33	0.11
6/9/2008	Carter 2						
6/24/2008	Carter 2	0.40	115.96	0.00	0.49	17.49	0.41
6/25/2007	Carter 3	0.06	14.20	0.00	0.10	8.43	0.73
7/11/2007	Carter 3	0.10	7.00	0.00	0.00	7.20	0.52
7/23/2007	Carter 3	0.10	12.09	0.00	0.00	13.72	0.34
8/6/2007	Carter 3	0.15	12.32	0.00	0.00	6.32	0.16
8/20/2007	Carter 3	0.14	12.50	0.00	0.00	6.31	0.00
9/17/2007	Carter 3	0.15	13.74	0.00	0.00	14.96	0.14
9/27/2007	Carter 3	0.16	22.16	0.05	0.08	8.18	0.05
10/22/2007	Carter 3	0.10	13.96	0.07	0.00	11.08	0.23
10/29/2007	Carter 3	0.14	19.67	0.18	0.00	12.86	0.06
11/12/2007	Carter 3	0.16	23.10	0.07	0.00	9.29	0.05
11/26/2007	Carter 3	0.14	14.32	0.00	0.00	9.09	0.24
12/10/2007	Carter 3	0.09	13.38	0.00	0.00	6.55	0.23
12/21/2007	Carter 3	0.09	16.04	0.06	0.00	12.83	0.23
1/7/2008	Carter 3	0.10	21.93	0.00	0.00	14.95	0.18
1/23/2008	Carter 3	0.08	21.03	0.00	0.10	12.12	0.00

2/4/2008	Carter 3	0.09	15.41	0.00	0.00	10.41	0.12
2/18/2008	Carter 3	0.12	12.91	0.00	0.00	8.45	0.35
3/5/2008	Carter 3	0.96	21.98	0.22	0.70	11.54	0.12
3/19/2008	Carter 3	1.74	12.96	0.00	0.00	6.83	0.46
4/9/2008	Carter 3	0.09	17.42	0.00	0.09	8.40	0.18
4/25/2008	Carter 3	0.09	14.89	0.00	0.08	7.23	0.31
5/14/2008	Carter 3	0.11	27.02	0.00	0.25	13.40	0.22
5/22/2008	Carter 3	0.15	27.78	0.00	0.14	17.89	0.14
6/9/2008	Carter 3	0.20	28.00	0.00	0.17	7.74	0.15
6/25/2007	Carter 4	0.10	7.17	0.00	0.00	5.33	0.61
7/11/2007	Carter 4	1.06	49.01	0.08	0.13	44.90	4.06
7/23/2007	Carter 4	0.81	64.97	0.06	0.16	38.51	3.72
8/6/2007	Carter 4	0.53	56.05	0.05	0.14	39.52	3.77
8/20/2007	Carter 4	0.78	79.61	0.09	0.00	83.47	4.38
9/3/2007	Carter 4	0.87	88.72	0.35	0.00	90.41	5.17
9/17/2007	Carter 4	0.90	92.58	0.14	0.00	88.25	7.04
9/27/2007	Carter 4	0.73	41.95	0.00	0.11	26.92	2.58
10/22/2007	Carter 4	0.89	83.35	0.00	0.12	31.63	8.41
10/29/2007	Carter 4	0.79	62.71	0.12	0.10	22.65	5.09
11/12/2007	Carter 4	1.24	57.64	0.13	0.10	24.10	1.98
11/26/2007	Carter 4	0.48	39.30	0.11	0.00	27.65	0.83
12/10/2007	Carter 4	0.98	60.22	0.00	0.22	38.03	3.62
12/21/2007	Carter 4	0.82	63.33	0.41	0.16	28.95	0.64
1/7/2008	Carter 4	0.78	61.51	0.19	0.11	26.67	3.87
1/23/2008	Carter 4	0.85	61.82	0.26	0.10	31.50	0.64
2/4/2008	Carter 4	0.60	38.64	0.09	0.00	19.78	0.02
2/18/2008	Carter 4	0.16	11.38	0.07	0.00	9.09	0.25
3/5/2008	Carter 4	0.79	44.85	0.07	0.09	23.68	3.64
3/19/2008	Carter 4	1.01	11.33	0.06	0.00	10.17	0.80
4/9/2008	Carter 4	0.88	79.23	0.00	0.19	42.49	2.37
4/25/2008	Carter 4	1.07	79.05	0.29	0.17	39.48	10.19
5/14/2008	Carter 4	0.92	80.06	1.32	0.17	34.63	6.21
5/22/2008	Carter 4	1.15	85.15	0.67	0.21	41.04	6.26
6/9/2008	Carter 4	1.13	92.39	1.54	0.18	37.71	10.31
6/24/2008	Carter 4	0.95	93.66	0.89	0.19	39.30	9.52
6/25/2007	Carter 5	0.23	35.16	0.08	0.10	28.12	2.64
7/11/2007	Carter 5	0.60	45.88	0.16	0.19	46.24	4.91
7/23/2007	Carter 5	0.53	57.27	0.19	0.19	46.22	4.11
8/6/2007	Carter 5	0.35	46.00	0.09	0.20	30.63	3.99
8/20/2007	Carter 5	0.89	71.81	0.17	0.18	84.66	6.35
9/3/2007	Carter 5	1.03	81.33	12.95	0.00	71.70	6.84
9/17/2007	Carter 5	1.14	91.71	0.51	0.36	103.05	10.64
9/27/2007	Carter 5	0.65	75.53	0.29	1.29	50.51	5.88
10/22/2007	Carter 5	0.94	64.70	0.07	0.20	31.06	8.09
10/29/2007	Carter 5	1.23	82.21	0.18	0.17	34.68	10.04
11/12/2007	Carter 5	0.90	72.03	0.13	0.21	32.80	5.90
11/26/2007	Carter 5	0.30	28.10	0.17	0.00	22.90	2.21

12/10/2007	Carter 5	0.93	78.03	0.50	0.16	41.30	6.40
12/21/2007	Carter 5	0.58	69.66	0.32	0.14	44.27	3.02
1/7/2008	Carter 5	0.69	89.21	0.62	0.24	46.43	6.86
1/23/2008	Carter 5	0.33	40.27	0.13	0.09	0.00	0.27
2/4/2008	Carter 5	0.65	78.07	0.25	0.17	47.30	2.36
2/18/2008	Carter 5	0.21	27.43	0.09	0.00	24.37	0.21
3/5/2008	Carter 5	0.44	31.43	0.08	0.09	22.80	3.36
3/19/2008	Carter 5	1.82	19.00	0.04	0.00	18.70	0.63
4/9/2008	Carter 5	3.89	42.94	0.19	0.14	28.02	5.66
4/25/2008	Carter 5	0.52	54.38	0.35	0.15	30.96	7.91
5/14/2008	Carter 5	0.55	76.04	0.19	0.21	38.87	6.67
5/22/2008	Carter 5	0.52	84.53	0.12	0.26	53.01	7.22
6/9/2008	Carter 5	0.77	83.64	0.56	0.20	38.24	8.19
6/24/2008	Carter 5	0.71	100.56	0.48	0.38	47.25	8.53
6/25/2007	Hudson	0.07	7.18	0.00	0.00	5.11	0.63
7/11/2007	Hudson	0.28	26.14	0.06	0.12	54.59	0.91
7/23/2007	Hudson	0.29	28.04	0.00	0.00	58.89	0.98
8/6/2007	Hudson	0.56	28.38	0.00	0.00	80.45	1.77
8/20/2007	Hudson	0.60	37.58	0.00	0.00	59.69	1.76
9/3/2007	Hudson	0.63	35.47	0.00	0.00	64.05	2.11
9/17/2007	Hudson	1.17	38.21	0.00	0.00	83.43	3.00
9/27/2007	Hudson	0.72	58.07	0.00	0.24	46.64	1.94
10/22/2007	Hudson	0.91	49.59	0.06	0.11	61.80	2.26
10/29/2007	Hudson	0.68	31.83	0.07	0.00	27.67	1.11
11/12/2007	Hudson	1.06	36.09	0.00	0.11	45.99	2.81
11/26/2007	Hudson	0.51	17.68	0.00	0.00	20.98	2.55
12/10/2007	Hudson	0.75	53.33	0.04	0.12	48.08	1.34
12/21/2007	Hudson	0.35	29.44	0.00	0.00	26.31	0.62
1/7/2008	Hudson	0.33	31.52	0.06	0.00	34.99	0.72
1/23/2008	Hudson	0.21	30.91	0.06	0.00	25.11	1.48
2/4/2008	Hudson	0.36	30.43	0.21	0.16	37.57	0.75
2/18/2008	Hudson	0.22	11.01	0.24	0.00	7.38	1.20
3/5/2008	Hudson	0.24	11.41	0.13	0.00	12.81	0.98
3/19/2008	Hudson	1.73	11.29	0.00	0.00	20.54	0.22
4/9/2008	Hudson	0.36	42.38	0.10	0.12	77.99	0.00
4/25/2008	Hudson	0.46	60.20	0.00	0.13	51.48	0.63
5/14/2008	Hudson	0.41	63.63	0.00	0.12	74.40	0.33
5/22/2008	Hudson	0.49	43.59	0.00	0.13	58.70	0.66
6/9/2008	Hudson	0.89	54.24	0.00	0.13	74.70	2.62
6/24/2008	Hudson	0.78	42.24	0.00	0.09	49.63	2.49
6/25/2007	Wolfpen	0.17	30.00	0.00	0.20	37.93	0.72
7/11/2007	Wolfpen	0.39	53.91	0.00	0.73	58.71	0.66
7/23/2007	Wolfpen	0.60	88.15	0.00	0.86	105.90	1.28
8/6/2007	Wolfpen	0.42	53.24	0.00	1.11	42.83	2.08
8/20/2007	Wolfpen	0.10	10.86	0.00	0.00	13.84	0.22
9/3/2007	Wolfpen	0.78	120.06	0.13	0.00	117.47	3.16
9/17/2007	Wolfpen	0.90	146.62	0.00	0.00	118.98	2.19

9/27/2007	Wolfpen	0.56	56.41	0.00	2.07	48.90	1.59
10/22/2007	Wolfpen	0.64	101.86	0.00	1.15	77.47	1.16
10/29/2007	Wolfpen	0.50	77.70	0.05	1.27	91.12	1.59
11/12/2007	Wolfpen	0.59	48.26	0.05	0.60	76.28	1.59
11/26/2007	Wolfpen	0.29	47.95	0.11	0.55	53.24	0.97
12/10/2007	Wolfpen	0.67	85.75	0.00	0.59	92.37	1.75
12/21/2007	Wolfpen	0.41	94.45	0.12	0.64	101.70	0.89
1/7/2008	Wolfpen	0.38	84.77	0.05	1.05	94.82	0.66
1/23/2008	Wolfpen	0.21	19.73	0.00	0.10	22.61	5.72
2/4/2008	Wolfpen	0.45	94.05	0.00	1.69	81.43	1.17
2/18/2008	Wolfpen	0.27	35.95	0.08	0.19	43.57	0.24
3/5/2008	Wolfpen	0.19	27.91	0.00	0.26	32.63	0.00
3/19/2008	Wolfpen	2.11	12.92	0.00	0.00	11.70	0.89
4/9/2008	Wolfpen	0.60	120.77	0.06	2.41	101.30	1.55
4/25/2008	Wolfpen	0.59	98.54	0.19	0.57	93.21	1.46
5/14/2008	Wolfpen	0.17	23.50	0.00	0.12	17.03	1.01
5/22/2008	Wolfpen	0.55	113.45	0.06	1.76	106.76	0.96
6/9/2008	Wolfpen	0.84	116.89	0.08	0.59	57.06	7.49
6/24/2008	Wolfpen	0.66	127.51	0.04	1.89	62.73	5.11

APPENDIX IV

E. COLI AND FLUOROMETRY IN BURTON CREEK,

APRIL 11, 2009

Meters upstream from Burton 2	Fluorescence at 445 nm	E. coli CFU/100 mL	NPOC mg L^{-1}	TDN mg L^{-1}
350	208.5	100	38.91	2.74
470	186.2	1020	35.79	2.33
470	240.8	100	41.36	2.59
750	141.3	6400	21.24	1.43
950	246.6	540	14.95	5.33
1395	218.9	9	52.93	3.78
1450	159.9	9	18.50	3.77
1570	235.6	80	87.55	5.40
3150	176.8	160	15.18	1.21
3210	179.9	290	15.53	1.33
3710	186.3	40	13.96	1.40
4080	223.9	1840	14.07	1.34
4240	243.0	8500	24.53	1.54
4370	218.1	4100	18.31	1.83
4810	208.5	7000	26.91	1.83
5040	244.0	80	18.11	1.48
3540	135.2	10	10.04	0.89
3630	77.9	40	5.66	0.79
3660	38.3	30	4.54	0.89

VITA

Name:	Cara Harclerode
Address:	c/o Dr. Jacqueline Aitkenhead-Peterson Texas A&M Department of Soil and Crop Sciences 370 Olsen Blvd. Mailstop #2474 College Station, TX 77843-2474
Email Address:	caraharclerode@tamu.edu

Education:	B.S., Biology, and B.S., Chemistry, LeTourneau University, 2004 M.S., Soil Science, Texas A&M University, 2009