ASSESSMENT OF THE VIABILITY OF A NATURAL URBAN WETLAND IN THE TREATMENT OF STORMWATER

by

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Dedication Page

This work is dedicated both to the esteemed *Watershed Hydrology, Modeling, Microbial Source Tracking, and Urban Stormwater Wetlands…With Some Remote Sensing… Research Group*, also known as WHMMSTUSWWSRSRG, as well as to my large, loving and supportive family. Mom, Dad, Jill, Paul, Maria, Lucas, Clara, Janice, Rob, Kailey, Jake, Julie, Norman, Scout, Finn, Kristy, big girl Lily and tiny baby Eve, who some say coming into this world as I was finishing my research was 'perfect timing', I could not have done this without you.

When I heard the learn'd astronomer,

When the proofs, the figures, were ranged in columns before me,

When I was shown the charts and diagrams, to add, divide,

and measure them,

When I sitting heard the astronomer where he lectured with

much applause in the lecture-room,

How soon unaccountable I became tired and sick,

Till rising and gliding out I wander'd off by myself,

In the mystical moist night-air, and from time to time,

Look'd up in perfect silence at the stars.

-Walt Whitman

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ABSTRACT

Stormwater runoff generated from urban areas can be a source of contamination and may negatively impact receiving waters. Best management practices, including the use of treatment wetlands, are recommended to minimize impacts and maintain the quality of water bodies receiving stormwater discharge. This study focuses on the viability of a natural wetland in the treatment of urban runoff. Kuhn Marsh is a natural urban wetland located in Dartmouth, NS. The wetland is approximately 2 ha in size and the primary inlet is a stormwater outfall servicing a 28 ha urban drainage area. Kuhn Marsh has been receiving stormwater generated from the urban drainage area for decades. A wetland drainage area of approximately 9 ha contributes to surface runoff downstream of the wetland inlet. Project objectives are defined as: (i) characterization of the hydrology and hydraulics of the wetland system, (ii) characterization of contaminant fluxes within the wetland system, and (iii) analysis of the treatment performance of Kuhn Marsh. Research strategies used to achieve project objectives include physical and hydrologic characterization of the wetland and contributing watersheds as well as surface and ground water quality analysis. Monitoring was conducted in the wetland during both baseflow and stormflow conditions from May 2011 through October 2012, with the exception of November 2011 to January 2012. Surface water samples were analyzed in the laboratory for TSS, TOC, TN, TP, turbidity, *E.coli*, and a suite of heavy metals including Fe, Pb, Cu, Cd and Zn. In-situ surface water monitoring included DO, temperature, conductivity and pH. Groundwater samples were analyzed for *E.coli* and microbial source tracking was performed on all well samples in addition to samples from the inlet and outlet of the wetland. Results from the well samples and the wetland outlet were inconclusive, however the wetland inlet showed human source bacteria indicating potential sewer cross connections within the stormwater system. It was determined that the wetland is an area of groundwater discharge, with groundwater accounting for an average of 50% of the volume discharging through the outlet control structure. Largely due to groundwater influence, Kuhn Marsh shows no peak flow dampening or volume reduction between inlet and outlet. Minimal hydraulic retention times, between 2 and 4 hours, were calculated during stormflow conditions, indicating potential short circuiting of flows through the wetland. Wetland treatment performance was analyzed on a concentration and mass reduction basis and on the number samples that exceeded parameter guidelines at the outlet of the wetland. Guideline exceedances were reported for the majority of samples taken and increases in concentration between inlet and outlet resulted in a larger number of samples exceeding guidelines at the outlet. Despite dilution from groundwater discharge, minimal to no concentration reduction was reported between the inlet and outlet of the wetland. Mass reduction did not occur between the inlet and outlet and Kuhn Marsh was found to be a source of all contaminants sampled. Results of this study show that Kuhn Marsh is no longer acting as a reservoir for stormwater contaminants and, based on the fact that the wetland has been receiving stormwater input on the order of decades, study results may be indicative of the long-term treatment capacity of a stormwater treatment wetland. In the future, comprehensive sampling of groundwater is recommended to determine if contaminants are entering the wetland via groundwater discharge, and if possible, surface water sampling should be conducted on a finer scale to better estimate mass fluxes and contaminant loading rates.

LIST OF ABBREVIATIONS AND SYMBOLS USED

Abbreviations

AASHTO	American Association of State Highway and Transportation Officials
ASTM	American Society for Testing and Materials
BMP	Best Management Practice
CCME	Canadian Council of Ministers of the Environment
CFU	Colony Forming Unit
CSO	Combined Sewer Overflow
CWCS	Canadian Wetland Classification System
DEM	Digital Elevation Model
DNA	Deoxyribonucleic Acid
E.coli	*Escherichia coli*
EGSPA	Environmental Goals and Sustainable Prosperity
FAC	Facultative Plants
FACU	Facultative Upland Plants
FACW	Facultative Wetland Plants
FWS	Free Water Surface
GCN	Gene Copy Number
GIS	Geomatics Information System
HDR	High Density Residential
HRM	Halifax Regional Municipality
HSSF	Horizontal Sub-Surface Flow
ICP-MS	Inductively Coupled Plasma Mass Spectrometry
IDF	Intensity-Duration-Frequency
LiDAR	Light Detection and Ranging

MDR	Medium Density Residential
MPN	Most Probable Number
NCDIA	National Climate Data and Information Archive
NURP	National Urban Runoff Program
NSWCP	Nova Scotia Wetland Conservation Policy
OBL	Obligate Wetland
qPCR	Quantitative Polymerase Chain Reaction
RTK-GPS	Real-Time Kinematic Global Positioning System
SI	International System of Units
UPL	Upland
VF	Vertical Flow

Symbols

A	Wetland Area (m^2)
As	Arsenic
Be	Beryllium
Bw	Bridgewater Soil Series
C	Carbon
C_i	Inlet Concentration (mg/L; CFU/100 mL)
C_o	Outlet Concentration (mg/L; CFU/100 mL)
Cd	Cadmium
Cl	Chlorine
ЄOM	Meguma Group
ЄOMh	Halifax Bedrock Formation
Cr	Chromium
C_t	Outlet Tracer Concentration (g/L)

Cu	Copper
d	Estimated Wetland Depth (m)
DO	Dissolved Oxygen
EDC	Endocrine-Disrupting Chemical
EtOH	Ethanol
FC	Fecal Coliform
Fe	Iron
FeO	Ferrous Iron
Fe_2O_3	Ferric Iron
h_1	Up-Gradient Well Head (m)
h_2	Down-Gradient Well Head (m)
H_0	Initial Well Drawdown Time
H_t	Instantaneous Well Drawdown Time
HAH	Halogenated Aliphatics
HNO_3	Nitric Acid
H_3PO_4	Phosphoric Acid
I_a	Initial Abstraction (mm)
K	Hydraulic Conductivity (m/day)
$K_2S_2O_8$	Potassium Persulfate
L	Well Point Distance (m)
L_b	Basin Length (m)
L_e	Length of Well Screen (m)
M_i	Mass of Tracer In (g)
Mn	Manganese
n	Dimensionless Shape Parameter

N	Nitrogen
N_2	Nitrogen gas
NH_3	Un-ionized Ammonia
NH_4	Ammonium
NO_2	Nitrite
Ni	Nickel
NO_3	Nitrate
NO_3-N	Nitrate as Nitrogen
NO_x	Nitrate-Nitrite
NTU	Nephelometric Turbidity Units
P	Phosphorous
PAH	Polycyclic Aromatic Hydrocarbon
Pb	Lead
PCB	Polychlorinated Biphenol
PO_4	Orthophosphate
P_r	Precipitation (mm)
PVC	Polyvinyl Chloride
Q_i	Inlet Flow (L/s)
Q_o	Outlet Flow (L/s)
Q_u	Unit Hydrograph Flow (m^3/s)
Q_{up}	Unit Hydrograph Peak Flow (m^3/s)
R	Radius of Well Gravel Pack (m)
r_c	Radius of the Well Casing (m)
R_e	Effective Radial Distance (m)

S	Potential Maximum Retention (mm)
S_b	Slope of Basin (m/m)
τ	Hydraulic Retention Time (hours)
t	Time (s)
t_p	Time to Peak Flow (s)
TC	Total Coliform
TDS	Total Dissolved Solids
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorous
TSS	Total Suspended Sediments
Zn	Zinc

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1.0 INTRODUCTION

Historically, stormwater infrastructure design focused on conveying runoff from an area quickly and efficiently to the nearest body of water. In recent decades, with the noted degradation of receiving waters and the characterization of stormwater quality, stormwater infrastructure design has evolved in attempts to mitigate adverse environmental impacts. Stormwater can be a source of pollutants such as heavy metals, road salt, pathogenic bacteria and can contribute to nutrient overloading of receiving waters. The proper management of runoff from a water quality perspective is vital in maintaining the quality of receiving waters. An increased focus on 'green infrastructure' design has developed, where design approaches involve enhancement or replication of natural environmental features. From a stormwater treatment perspective, this would include diverting polluted runoff to areas which would allow for infiltration or filtration prior to reaching a body of water. Treatment wetlands are an example of proposed green infrastructure. Although the bulk of existing research into the function and performance of treatment wetlands has been conducted in the context of treating municipal wastewater, preliminary studies on pollutant attenuation in stormwater treatment have had positive results. These studies, however, tend to focus on the function of newly developed treatment wetlands and little is known concerning the function of these wetlands over the long term. This study focuses on Kuhn Marsh, a natural urban wetland located within the Halifax Regional Municipality, in Dartmouth, Nova Scotia. Kuhn Marsh encompasses an area of approximately 2 ha and has been receiving stormwater runoff from a 28 ha urban residential area for decades. The purpose of this proposed Master's project is to characterize the function of Kuhn Marsh as a treatment wetland by looking at a selection of physical, chemical and biological parameters. As Kuhn Marsh is a natural wetland, it lacks the control inherent in the design of constructed systems. To best categorize the treatment function of the natural system, the following project objectives were developed:

(i) Characterization of the hydrology and hydraulics of the wetland system
(ii) Characterization of contaminant fluxes within the wetland system
(iii) Analysis of the treatment performance of Kuhn Marsh

2.0 BACKGROUND

2.1 Stormwater: Characterization, Impacts and Management

2.1.1 Stormwater Hydrology

The subject of stormwater hydrology arises from the development of natural landscapes through urbanization. Land use alteration through urbanization has a direct impact on the quantity and quality of surface runoff during storm events. Land use alterations associated with urban development include vegetation removal, increase in percentage of impervious surfaces, and drainage channel alteration (Goonetilleke, 2004). These land use alterations change established rainfall-runoff relationships, with the two most common effects being: 1) reduced infiltration and 2) reduced time of concentration (Maidment, 1993). Increasing the percentage of impervious area in a watershed leads to an increase in surface runoff and a proportional decrease in infiltration. Reduced time of concentration in an urban watershed means a reduced travel time of water from the furthest point in a watershed to the point of discharge. The effect that urbanization has on the flow hydrograph of a stream receiving stormwater discharge is illustrated in Figure 2-1. There is a noted increase in peak flow and overall volume for urbanized conditions. Lag time, the time between peak precipitation and peak outflow, is also decreased in urbanized conditions. This indicates rainfall is reaching the discharge point, in this case a stream, more rapidly post-urbanization.

Traditionally, conventional urban drainage systems have been designed to quickly move water off the landscape during a storm event. These drainage systems consist of street-level inlet drains that collect surface runoff and route the runoff into an underground piping system which then discharges into a body of water or treatment facility (Chin, 2006a).

Burns *et al.* (2012) refer to the conventional urban drainage system as a 'drainage-efficiency' based system. Based on a review of the literature, Burns *et al.* (2012) have compiled a list of changes to flow regimes which arise from urbanization and the use of a drainage-efficiency based system. They are as follows: 1) Increased frequency, magnitude and volume of storm flow due to impervious surfaces being connected directly to storm drainage systems, 2) Increased volume of total runoff resulting from reduced evapotranspiration due to loss of vegetation, 3) Reduction in magnitude of seasonal baseflow, as a result of reduced infiltration, 4) Increased magnitude of low-frequency flows, and 5) Reduced storm recession time.

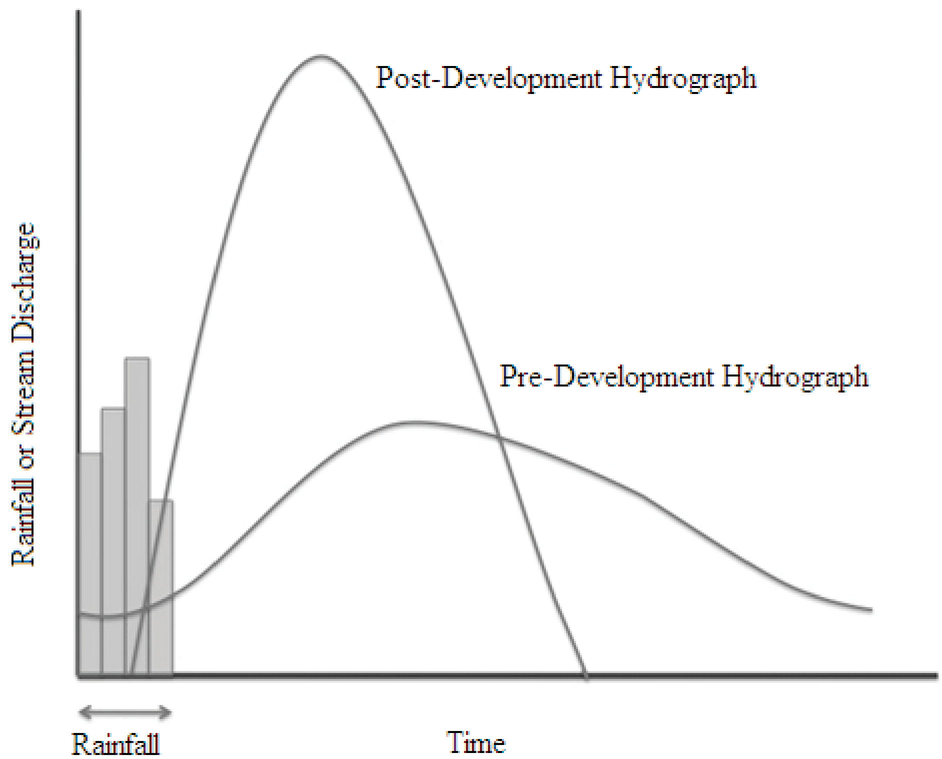

Figure 2-1 Pre and post-urbanization hydrograph (Adapted from White, 2010)

Chocat *et al.* (2007) suggest the optimal drainage solution for urban areas is the combination of traditional and non-traditional stormwater infrastructure. Traditional piping systems are necessary to move water quickly and efficiently from an urban centre but these systems do not have the capacity to provide stormwater treatment or reduction in volume and peak flow. Best Management Practices (BMPs) are noted as the primary way to manage both quality and quantity of stormwater runoff (Chin, 2006b). The implementation of BMPs in an urban setting focus on two main outcomes: 1) the replication of the natural runoff and infiltration patterns of an undeveloped system and 2) the reduction or prevention of water quality degradation caused by urban development (Dillon Consulting, 2006). BMPs can be used in series to improve treatment and management efficiency and are commonly categorized into source control, conveyance control and end-of-pipe control practices. Stormwater wetlands are considered an end-of-pipe best management practice. The use of traditional systems in combination with stormwater best management practices can mitigate changes to flow regimes by promoting infiltration and evapotranspiration and increasing time of concentration, while providing the added benefit of contaminant removal.

2.1.2 Stormwater Characterization

As defined by Chin (2006b), point source pollution is a localized discharge of pollutants derived from a specific source, whereas nonpoint pollution is non-specific in source and can originate from multiple point sources. Point sources, or localized discharges, are more easily identified and therefore better regulated. Although a stormwater discharge pipe represents a point source of pollution for receiving waters, stormwater runoff is designated as a nonpoint pollution source as it originates from diffuse urban runoff (Chin, 2006b). Pitt *et al.* (1995) identified industrial and commercial areas as pollutant sources contributing to diffuse urban runoff. Characterization of water quality is accomplished through the assessment of various chemical, biological, and physical water quality parameters. Parameters that are known to directly or indirectly impact the quality of water are used as indicators of water quality. Depending on the source water and the purpose of testing, different sets of parameters may be selected characterize water quality. Exceptionally, if a body of water receiving stormwater discharge is also used as a source of drinking water, parameters which characterize risks to human health are chosen. Otherwise, parameters used to indicate the quality of stormwater are generally selected to characterize impacts to aquatic life in receiving waters. Davis and McCuen (2005) divide water quality parameters into two categories, (1) factors necessary for aquatic life and (2) water pollutants.

Table 2-1 Water Quality Parameters (Adapted from Davis and McCuen, 2005)

Factors Necessary For Aquatic Life	**Water Pollutants**
Dissolved Oxygen	Suspended Solids
Temperature	Oxygen Demanding Substances
pH	Nutrients
	Microbial Pathogens
	Heavy Metals
	Oil and Grease
	Toxic Organic Compounds

2.1.2.1 Water Quality Guidelines

There are no current guidelines in effect pertaining directly to stormwater quality within the province of Nova Scotia. The following are Canadian guidelines that may be used as metrics to determine the quality of stormwater in Nova Scotia. It is noted that guidelines are considered recommendations and are not able to be legally enforced.

2.1.2.1.1 Canadian Water Quality Guidelines for the Protection of Aquatic Life

The Canadian Water Quality Guidelines for the Protection of Aquatic Life were proposed by the Canadian Council of Ministers of the Environment (CCME) and are based on the original Canadian Water Quality Guidelines published by the then Canadian Council of Resources and Environment Ministers in 1987 (CCME, 1999). The guidelines contain maximum allowable concentrations based on short term and long term exposure for numerous organic and inorganic substances, and are updated and added to as contaminant information emerges and evolves based on continuing research.

2.1.2.1.2 Guidelines for Canadian Recreational Water Quality

The Guidelines for Canadian Recreational Water Quality are published by Health Canada and pertain to the safety of surface waters for primary contact (eg. swimming) and secondary contact (eg. boating) recreational activities (Health Canada, 2012b). The third edition of the guidelines was published in April, 2012 and contains recommended guidelines for physical, chemical and aesthetic water parameters as well as maximum accepted levels of pathogen indicator organisms.

2.1.2.1.1 Guidelines for Canadian Drinking Water Quality

The Guidelines for Canadian Drinking Water Quality have been published by Health Canada since 1968 and pertain to the safety of water for human consumption. The latest edition of the guidelines was published in August, 2012. The guidelines provide benchmarks for drinking water quality and cover a wide range of microbiological, chemical and physical parameters. The Guidelines for Canadian Drinking Water Quality are the only guidelines pertaining to the direct and intentional human consumption of surface and ground water, but are not legally enforceable (Health Canada, 2012a).

2.1.2.2 Water Quality Impacts

2.1.2.2.1 Nitrogen

Nitrogen (N) is a required nutrient for the growth and development of both plants and animals. Nitrogen gas (N_2) is found in abundance in the atmosphere, however only a small number of microorganisms are able to use N in a gaseous state. For use as a nutrient, plants require N in the form of either nitrate (NO_3) or ammonium (NH_4) and fixation processes are required to change atmospheric N into a useable form (Vaccari *et al.*, 2006). Nitrogen fixation of N_2 is the primary natural source for N. Vaccari *et al.* (2006) outline the five primary steps in the nitrogen cycle as follows: 1) Fixation: the conversion of N_2 to NH_4 by bacteria located on plant roots, 2) Ammonification: the conversion of organic N to NH_4, primarily occurring in soils, 3) Nitrification: oxidation of NH_4 first to nitrite (NO_2) then to NO_3, 4) Denitrification: conversion of NO_2 and NO_3 back to N_2 and 5) Assimilation: conversion of NO_3 and NH_4 to organic compounds, such as amino acids.

Both NO_3, and to a lesser extent NH_4, can be easily transported via surface and groundwater. Nitrate is considered harmful to humans as it can interfere with oxygen transfer in the blood, and is of special concern in children under the age of six months (Chin, 2006b). The Guidelines for Canadian Drinking Water Quality published by Health Canada state a maximum allowable concentration of 45 mg/L NO_3, equivalent to 10 mg/L of nitrate as nitrogen (NO_3-N) in drinking water, and specifies concentrations of NO_2 should not exceed 3.2 mg/L (Health Canada, 2010). Un-ionized ammonia (NH_3), NO_2 and NO_3 also pose risk to aquatic organisms. CCME has proposed guidelines to minimize harm to aquatic life from these nitrogen species. These guidelines refer to maximum allowable concentrations and are presented in Table 2-2.

Table 2-2 CCME Freshwater Aquatic Life guidelines for specific nitrogen species (CCME, 1999)

	Concentration	
Nitrogen Species	**Short Term**	**Long Term**
Ammonia (un-ionized)	---	19 µg/L
Nitrate	550,000 µg/L	13,000 µg/L
Nitrite	---	60 µg/L NO_2-N

In addition to the aquatic and human health impacts of certain nitrogen species, high levels of N can contribute to eutrophication and plant growth in surface waters. Eutrophication refers to the excess accumulation of nutrients within a body of water and is attributed to both nitrogen and phosphorous. Increase in plant growth due to eutrophication causes the accumulation and sedimentation of biomass which eventually results in loss of water storage capacity within a body of water (Vaccari *et al.,* 2006). It is suggested that human interference with the natural nitrogen cycle has doubled the rate at which N is introduced into the terrestrial cycle (Vitousek *et al.*, 1997). Anthropogenic activities that contribute to increased N include: the use of N-based fertilizers, fossil fuel combustion, the increase in planting of N-fixing crops and the mobilization of N from biological storage through land use alteration and biomass burning (Vitousek *et al.*, 1997).

2.1.2.2.2 Phosphorous

Phosphorous (P) is an essential nutrient required for the growth and development of both plants and animals. Phosphorous is generally considered a limiting nutrient as it is the only nutrient that is not readily available through atmospheric deposition or natural presence in surface waters (Davis and Masten, 2009). The primary source of P is through the weathering of rock; with decay of plants and animals considered secondary sources (Campbell *et al.,* 1999). P cycles between organic and inorganic states through microbiological activity, with soluable inorganic orthophosphate (PO_4) being the form of P that is required by most plants and microorganisms (Tchobanoglous and Schroeder, 1985).

As P is available in limited quanities naturally, excess P loading is typically attributed to anthropogenic activities. Improper disposal of wastewater and use of fertilizers containing P are the primary sources for excess P loading (Tchobanoglous and Schroeder, 1985). CCME trophic status is a determination of water productivity based on phosphorous concentration. Productivity, in this sense, is defined as the ability to support plant and algae growth. Where light is a limiting factor, in the case of shaded waterbodies, productivity may be supressed despite high P levels. In cases of un-shaded water bodies, nutrient availability is considered a limiting factor and high P levels can result in high productivity (Hill and Knight, 1988). Trophic status for freshwater and the corresponding P concentrations can be found in the following table, as defined by CCME (1999):

Table 2-3 CCME Trophic Status for P concentrations (CCME, 1999)

Phosphorous	
Trophic Status	**Concentration Range (µg/L)**
Ultra-Oligotrophic	<4
Oligotrophic	4-10
Mesotrophic	10-20
Meso-Eutrophic	20-35
Eutrophic	35-100
Hyper-Eutrophic	>100

Oligotrophic waterbodies have low nutrient concentrations and are therefore considered low in productivity with insignificant plant growth. Eutrophic waterbodies are high in productivity due to an excess of nutrients and have significant plant growth as a result. Meso-Eutrophic waterbodies have nutrient concentrations between that of Oligo and Eutrophic waterbodies. Hyper-Eutrophic waterbodies are characterized by severe excess nutrient loading and as a result are usually overcome by excess plant growth (Davis and Masten, 2009). Excess plant growth is undesirable in a body of water supporting aquatic life as it can lead to fluxuation in dissolved oxygen levels on a diurnal and seasonal basis through photosynthesis, respiration and biomass oxidation requirements. Excess plant growth can also impart an odour and colour to water, which is undesirable from an aesthetic perspective (Tchobanoglous and Schroeder, 1985). Excess growth of algae, or 'algal blooms', can be produced through high availability of nutrients in surface water and are of special concern as toxins produced by certain species of algae and cyanobacteria may be harmful to human and aquatic health (O'Neil *et al.*, 2012; Hudnell, 2008)

2.1.2.2.3 Microbial Pathogens

Marsalek and Rochfort (2004) identified urban wet-weather pollution as a source of microbiological contamination for receiving waters. Urban wet-weather pollution is an overarching term used to define stormwater, combined sewer overflows (CSOs) and sanitary overflows. Levels of *Escherichia coli* (*E.coli*) in stormwater alone were found to range from 10^3-10^4 Colony Forming Units (CFU)/100mL, with higher levels indicating the possible presence of sewer cross-connections (Marsalek and Rochfort, 2004). *E.coli* is a bacterium found in the digestive tract of warm-blooded animals and is used as an indicator organism to detect possible fecal contamination of water bodies (Tortora *et al.*, 2004). The presence of indicator organisms, which also include fecal coliforms (FC) and fecal enterococci, suggest the possibility of presence of disease-causing pathogenic microorganisms (Simpson *et al*., 2002). Pathogenic microorganisms in stormwater include various bacteria, viruses, protozoa and helminthes (Chin, 2006b). The most common laboratory bacterial analysis provides information only on the presence or absence of fecal bacteria within a water body. Source tracking of bacteria provides a laboratory method to determine the origin of fecal bacteria within a water source. Source tracking to determine the origin of fecal contamination may be done through biochemical or molecular means. Depending on the method selected, source determination may be specific (eg. determination of actual animal source) or non-specific (eg. determination of human vs. non-human source) (Santo Domingo *et al.*, 2002). Source tracking is beneficial from a human health standpoint as the determination of origin of contamination allows for mitigation measures to reduce or remove the source of contamination from the water area. The Guidelines for Canadian Drinking Water Quality specify the maximum allowable concentration for *E.coli* in drinking water is none detectable per 100 mL (Health Canada, 2012a). The Canadian Recreational Water Quality Guidelines for *E.coli* specify the geometric mean of a minimum of 5 samples have no more than 200 CFU per 100 mL, with no single sample exceeding 400 CFU per 100 mL (Health Canada, 2012b).

2.1.2.2.4 Metals

Metals exist naturally in the environment and are considered contaminants only when the concentrations of a metal species exceed the background levels of the natural environment (Kaufman *et al.*, 2011). Based on an extensive review of stormwater contaminant literature, Makepeace *et al.*(1995) concluded that copper (Cu), lead (Pb), zinc (Zn) and cadmium (Cd) were the primary metals studied and stated that these metals, along with nickel (Ni), arsenic (As) and beryllium (Be), were of greatest concern as stormwater contaminants. Anthropogenic sources, human health concerns, and aquatic life guidelines for various metals can be found in Table 8-5, in the appendix.

2.1.2.2.5 Organic Compounds

Organic compounds are compounds that contain carbon (C) and can be either natural or synthetic in origin. Natural organic compounds include proteins, carbohydrates and lipids and the decay of plant materials is the primary natural source of carbon in surface waters (Tchobanoglous and Schroeder, 1985). Quantification of organic compounds within a water body may be approached in several manners but according to Manahan (2009), the determination of Total Organic Carbon (TOC) in a water sample is the best method used to assess the organic content of an aquatic environment. Organic carbon may serve as a source of nutrients for microorganisms and is associated with reduced aesthetic value and production of disinfection byproducts in drinking water (Clesceri *et al*., 1998).

A vast number of synthetic organic chemicals are considered toxic organic compounds, including pesticides, polychlorinated biphenols (PCBs), polycyclic aromatic hydrocarbons (PAHs), phthalate esters, halogenated aliphatics (HAHs) and phenols (Chin, 2006). These compounds are anthropogenic in nature, and water contamination pathways include industrial discharges, accidental spills, agricultural and urban runoff and the chlorination of drinking water (Chin, 2006). Toxic effects, combined with environmental persistence of synthetic organic compounds and degradation products, are of great concern. A large number of these chemical compounds are carcinogenic and have also been studied for their effects on the endocrine systems of mammals. Endocrine-disrupting chemicals (EDCs) can mimic natural hormones and affect reproduction and development. Toxic organic compounds that are known EDCs include PCBs, phthalates esters, PAHs, and many pesticide and ester compounds (Davis and Masten, 2009). The Freshwater Aquatic Life guidelines published by CCME contain a large selection of guidelines pertaining to organic compounds, including PAHs, PCBs, pesticides and halogenated aliphatics.

2.1.2.2.6 Total Suspended Solids & Turbidity

Total Suspended Solids (TSS) is a measure of the solid matter suspended in a water sample. It is typically measured as the mass of solids retained on a standard 1.5 µm pore-size filter and is reported as milligrams of suspended matter per liter of sample filtered. Turbidity is a measure of the intensity of the scattering of light projected through a water sample and is generally reported in nephelometric turbidity units, or NTU (Standard Methods, 1998). While the concentration of TSS pertains solely to particles larger than 1.5 μm, the determination of turbidity includes microscopic particles as well. Correlation between TSS and turbidity may be determined on a site-specific basis, allowing for the estimation of TSS concentrations from turbidity readings (Packman *et al.,* 1999; Suk *et al.*, 1998). The CCME has established Guidelines for the Protection of Aquatic Life for both turbidity and TSS. These guidelines are based on limiting increases in relation to natural background conditions for both clear flow and turbid waters, and are presented in Table 2-4.

High levels of sedimentation in surface water may negatively affect fish and fish habitat. Sediment adherence to fish eggs may reduce the ability of the eggs to attach to rocks and vegetation. Certain fish species lay eggs in void spaces along the bottom of stream beds for protection. Increased sedimentation may fill void spaces, reducing the number of protected areas for fish eggs. If sediment is deposited on top of eggs laid in a stream bed there can be interference with oxygen transfer to and waste disposal from the eggs (Ward, 1992). Suspended sediment may also be a source of contamination, as nutrients, metals and microorganisms adsorb onto sediment particles. The removal of suspended matter associated with TSS and turbidity may also provide removal of contaminants associated with sediment sorption (Characklis *et al.,* 2005; USEPA, 1995a).

2.1.2.2.7 Dissolved Oxygen

Dissolved Oxygen (DO) is a measure of the quantity of oxygen molecules dissolved in water (Chin, 2006b). Dissolved oxygen levels are presented in either direct concentrations (mg/L) or a percent saturation; which is the percent ratio of measured dissolved oxygen in the water to the maximum dissolved oxygen capacity of the water, for a specific temperature. Surface water DO levels are directly related to temperature and atmospheric pressure, with percent saturation decreasing with increasing temperature and pressure. Low levels of DO can affect both the health and aesthetics of surface water ecosystems (Chin, 2006b). Dissolved oxygen is required for the decomposition of organic matter as well as aquatic life respiration. The CCME Freshwater Aquatic Life guidelines provide lowest acceptable concentrations of DO for cold and warm water biota (Table 2-5).

Table 2-4 CCME guidelines for maximum allowable increase of TSS and Turbidity

TSS	Turbidity
Clear Flow	
Maximum increase of 25 mg/L from background levels for any short-term exposure (e.g., 24-h period). Maximum average increase of 5 mg/L from background levels for longer term exposures (e.g., inputs lasting between 24 h and 30 d).	Maximum increase of 8 NTU from background levels for a short-term exposure (e.g., 24-h period). Maximum average increase of 2 NTU from background levels for a longer term exposure (e.g., 30-d period).
High Flow	
Maximum increase of 25 mg/L from background levels at any time when background levels are between 25 and 250 mg/L. Should not increase more than 10% of background levels when background is ≥ 250 mg/L.	Maximum increase of 8 NTU from background levels at any one time when background levels are between 8 and 80 NTU. Should not increase more than 10% of background levels when background is >80 NTU.

Table 2-5 CCME guidelines for lowest acceptable DO concentration (CCME, 1999)

Dissolved Oxygen Lowest acceptable concentration			
Warm Water Biota		**Cold Water Biota**	
Early life stages:	6.5 mg/L	Early life stages:	9.5 mg/L
Other life stages:	5.5 mg/L	Other life stages:	6.5 mg/L

Dissolved oxygen is introduced to a body of water through diffusion across the water surface, with wind or gravity-induced turbulence greatly increasing diffusion rates. Photosynthesis by plants and algae also contribute DO during daylight hours, however plant respiration during the night depletes oxygen levels. These biological processes can cause major fluctuations in DO levels on a diurnal basis (Kalff, 2002). Oxidation reactions are a major source of oxygen depletion in water bodies, most notably through the aerobic decomposition of biomass caused by excess plant growth (Tchobanoglous and Schroeder, 1985).

2.1.2.2.8 pH

For water quality purposes, pH is reported as the negative log of the molar concentration of hydrogen ions in a water sample (Manahan, 2009). Values of pH are reported on a scale ranging from <1 to 14, with low concentrations of hydrogen ions relating to high values on the scale. Samples with low-range values are considered acidic, mid-range values are considered neutral, and high-range values are considered basic, or alkaline (Kalff, 2002). The CCME (1987) sets an ideal pH range of 6.5 to 9.0 for maintaining quality of life for aquatic species. The concentration of hydrogen ions in soil and water can have a large effect on precipitation and other important chemical reactions. Values of pH greater than 9 cause the conversion of NH_4 to NH_3, which is toxic to both aquatic species and humans (Chin, 2006b). Lower values of pH can cause the precipitation of metals into solution allowing for ingestion and uptake by aquatic species and vegetation (Kalff, 2002). Surrounding geology may contribute to changes in pH values within water bodies, specifically in the case of acid rock drainage which may be exacerbated by anthropogenic activities. Acid rain is also a contributing factor to lowering pH of surface water; especially in locations were the native soil has a low buffering capacity (USEPA, 2012a).

2.1.2.2.9 Temperature

Temperature directly affects the solubility of DO and several biological processes (e.g. denitrification), and is of importance to the quality of aquatic life in a water body (Chin, 2006b; Kadlec, 2012). Rates of chemical and biochemical reactions and the solubility of minerals increase with temperature, whereas gas solubility, in the case of DO, decreases with temperature (Tchobanoglous and Schroeder, 1985). The CCME sets temperature guidelines for thermal additions to water bodies pertaining to thermal stratification, maximum weekly average temperature and short-term exposure to extreme temperature (CCME, 1999). The guidelines are as follows: 1) Thermal Stratification: Thermal additions to receiving waters should be such that thermal stratification and subsequent turnover dates are not altered from those existing prior to the addition of heat from artificial origins, 2) Maximum Weekly Average Temperature: Thermal additions to receiving waters should be such that the maximum weekly average temperature is not exceeded, 3) Short-term Exposure to Extreme Temperature: Thermal additions to receiving waters should be such that the short-term exposures to maximum temperatures are not exceeded. Exposures should not be so lengthy or frequent as to adversely affect species of importance. In addition to thermal additions, anthropogenic temperature changes may be caused by removal of shading vegetation along waterways and through sedimentation which results in shallower water bodies, allowing for solar penetration through the entire water column (Rutherford *et al.*, 1997).

2.1.2.2.10 Conductivity

Conductivity pertains to the 'ability of an aqueous solution to carry and electric current' and is generally reported by the International Systems of Units (SI) as millisiemens per meter (mS/m) (Clesceri *et al.*, 1998). The conductivity of a water body is dependent on both the concentration of ions present in the water as well as the temperature, with conductivity increasing with temperature (Clesceri *et al.*, 1998; USEPA, 2012b). Ions contributing to conductivity may be positively charged cations, such as sodium, magnesium, calcium, iron and aluminum or negatively charged anions, such as chlorides, nitrates, sulphates and phosphates (USEPA, 2012b). Generally, water high in inorganic compounds will have high conductivity, and water high in organic compounds will have low conductivity (Clesceri *et al.,* 1998).

2.1.2.3 Stormwater Management Guidelines and Regulations

The Halifax Regional Municipality (HRM) is currently working on a Stormwater Management Functional Plan. According to a Regional Municipal Planning Strategy document printed by HRM in 2006, a Stormwater Management Functional Plan is necessary for the Halifax area and should consider, in part, methods to reduce increased stormwater flows, the reduction of site disturbance and impervious surfaces in new developments, the employment of naturally occurring soils and native plant species in stormwater management plans, the employment of methods to reduce sediments and contaminants being discharged into watercourses, the application of emerging technologies to improve stormwater system performance and the establishment of BMPs and criteria concerning the quality and quantity of stormwater discharge

(HRM, 2006). Publications currently in use within HRM concerning stormwater management and infrastructure include both the HRM Stormwater Management Guidelines and Halifax Water Design and Construction Specifications.

2.1.2.3.1 Halifax Regional Municipality Stormwater Management Guidelines

The HRM Stormwater Management Guidelines were produced by Dillon Consulting in March 2006. The majority of the report covers BMP implementation. Information concerning BMPs ranges from defining and identifying BMPs, to design criteria, selection of appropriate BMPs, and operation and maintenance. Although the guidelines regarding BMPs are not enforceable by law, the document covers a section concerning Legislative Authority and identifies areas within the NS Environment Act and the Stormwater Drainage Works Approval Policy which work to ensure the health of aquatic ecosystems and limit the degradation of water quality (Dillon Consulting, 2006).

2.1.2.3.2 Halifax Water Design and Construction Specifications

The current Halifax Water Design and Construction Specifications, also known as the "White Book", was updated in April 2012 by the Halifax Regional Water Commission (HWRC, 2012). The purpose of the specifications is to standardize the design and construction of municipal water systems, including stormwater drainage systems. While largely comprised of construction standards, the design requirements for a stormwater system within HRM also specify that 'any stormwater drainage system within the Municipality shall be designed to achieve the following objectives' which include the preservation of natural water courses, the minimization of long term effects to receiving waters from development, and the mitigation of adverse effects on downstream properties caused by storm flow, such as flooding or erosion (HWRC, 2012). All designs based on the specifications are required to be approved by an engineer and require proper construction permitting from Nova Scotia Environment (NSE), a provincial department under the Minister of Environment.

2.2 Wetlands: Natural and Constructed

2.2.1 Natural Wetlands

The Nova Scotia Environment Act (1994a), established by the Nova Scotia provincial government, defines a wetland as:

> Land commonly referred to as a marsh, swamp, fen or bog that either periodically or permanently has a water table at, near or above the land's surface or that is saturated with water, and sustains aquatic processes as indicated by the presence of poorly drained soils, hydrophytic vegetation and biological activities adapted to wet conditions.

2.2.1.1 Natural Wetland Types

The Canadian Wetland Classification System (CWCS) was established in 1997 through a partnership with the University of Waterloo Wetlands Research Centre, the North American Wetlands Conservation Council and the Canadian Wildlife Service. The CWCS has standardized natural wetland classification into five categories: bog, fen, swamp, marsh, and shallow water wetland. The following sections describe the wetland classes as defined in the CWCS (National Wetlands Working Group, 1997).

2.2.1.1.1 Bog Wetland

A bog is a peat wetland that is unique in the fact that it is ombrogenous, receiving water solely from direct rainfall, fog and snowmelt. The ombrotrophic nature of bogs is due to terrain position. The bog surface is generally raised or level with the surrounding topography and thus receives no runoff or groundwater inflow. With rain being the primary hydrologic source, bogs are generally low in mineral content and pH. Vegetation is primarily Sphagnum moss and plants adapted to acidic soil conditions (National Wetlands Working Group, 1997).

2.2.1.1.2 Fen Wetland

A fen is a peat wetland that receives inflow from surface and groundwater as well as precipitation. Fens generally have higher mineral content than bogs due to water inflow. The water level is generally found at the surface of a fen, with some surface flow. Fen vegetation is classified based on mineral content, from poor to extremely rich in content. Poor fens have low mineral content and are similar to bogs in that mosses and plants adapted to acidic conditions are the primary vegetation. Extremely rich fens are high in mineral content and the vegetation consists of sedges, brown mosses and shrubs (National Wetlands Working Group, 1997).

2.2.1.1.3 Swamp Wetland

A swamp wetland may contain peat or mineral soils and has a water table that is at, or below, the surface of the wetland. Swamps are fed by both surface and groundwater, but according to the CWCS, are found to be drier than marshes and fens. The primary vegetation consists of tall shrubs and trees causing the peat or soils to be largely comprised of decaying woody vegetation (National Wetland Working Group, 1997).

2.2.1.1.4 Marsh Wetland

A marsh wetland is largely comprised of mineral soils and is characterized by a fluctuating water table with shallow areas of ponded water. Groundwater and surface runoff are the primary hydrologic sources and marshes are most often found in areas of low elevation which allow for water inflow. Marshes are generally comprised of mineral soils, with fluctuating water levels allowing for aerobic decay of plant matter. Marsh vegetation typically follows a decreasing moisture gradient with emergent aquatic vegetation found in ponded areas, shrubs, reeds and sedges found in areas of saturated soil and tall shrubs and trees found along the drier outer edges of the marsh wetland (National Wetland Working Group, 1997).

2.2.1.1.5 Shallow Water Wetland

A shallow water wetland is defined as a transitional zone between a marsh wetland and a permanently ponded water body, such as a lake or pond. Shallow water wetlands are characterized by ponded water areas of less than 2 metres in depth covering more than 75% of the surface area of the confined basin or saturated zone. Rooted emergent vegetation may cover up to 25% of the surface area of the shallow water wetland (National Wetland Working Group, 1997).

2.2.1.2 Natural Wetland Delineation

Proper delineation of wetland boundaries is required for both research and regulatory purposes. Within Nova Scotia, wetland alteration approvals are not required if the wetland area is less than 100 m^2, however if the wetland is large enough to require an alteration approval the exact wetland boundaries must be determined as part of the approval process (Nova Scotia Environment, 2011; Nova Scotia Wetland Conservation Policy, 2011). The US Army Corps of Engineers published a wetland delineation manual in 1987, with the three primary wetland delineation measures being vegetation, soil conditions and hydrology (US Army Corps of Engineers, 1987). Mitsch and Gosselink (2000) suggest the US Army Corps Wetland Delineation Manual is the standard for wetland delineation, with Berkowitz (2011) verifying the accuracy of the manual in the majority of delineations completed in a study of 232 wetlands in the United States.

2.2.1.2.1 Vegetation

Wetland vegetation is defined as hydrophytic vegetation, or vegetation that has ‘anatomical or physiological adaptations that allow them to survive and thrive in saturated or inundated soils, where oxygen depletion is the primary factor limiting vegetation occurrence’ (Tammi, 2000). For the purpose of wetland delineation, the US Army Corps Wetland Delineation Manual (1987) divides vegetation into five classes, defined by probability of vegetation occurrence in wetlands and non-wetlands; 1) obligate wetland plants (OBL) (>99% occurrence in wetlands, <1% occurrence in non-wetlands), 2) facultative wetland plants (FACW) (>67-99% occurrence in wetlands, 1-33% occurrence in non-wetlands), 3) facultative plants (FAC) (33-67% occurrence in both wetlands and non-wetlands), 4) facultative upland plants (FACU) (1-33% occurrence in wetlands, >67-99% occurrence in non-wetlands), and 5) obligate upland plants (UPL) (<1% occurrence in wetlands, >99% occurrence in non-wetlands). For determination of wetland conditions based on vegetation, more than 50% of the dominant wetland species must fall into the OBL, FACW or FAC category (US Army Corps, 1987).

2.2.1.2.2 Soils

Hydric soils can be either mineral or organic in composition. Mineral soils are soils with less than 20-35% organic matter; organic soils are considered peat (1/3 decomposed) or muck (2/3 decomposed) depending on the level of decomposition of organic matter present (Vaccari, 2006). All organic soils, with the exception of folists, are considered hydric soils (Tammi, 2000; Mitsch and Gosselink, 2000). Mineral soils that exist in hydric, or saturated, conditions exhibit gleying and mottling due to periodic or sustained lack of oxygen (Soil Classification Working Group, 1998). During reduced conditions ferric iron (Fe_2O_3) is reduced to soluble ferrous iron (FeO) and is removed from the soil resulting in gray or bluish coloured soil (Tammi, 2000). In areas of fluctuating water tables, soil may be subject to periods of aerobic conditions in which oxidation of iron (Fe) and manganese (Mn) can occur. These areas of oxidation are called ‘mottles’ and are characterized by red or black spots (>1mm in diameter) within the soil matrix (Vaccari, 2006; Soil Classification Working Group, 1998). The presence of hydric soils is an indication of wetland soil conditions; however areas with hydric soils must also support wetland vegetation and have hydrologic characteristics of a wetland to be officially considered a wetland (US Army Corps, 1987). The Canadian System of Soil Classification gives detailed information regarding soil classification in Canada, including information on organic and gleyed soils (Soil Classification Working Group, 1998).

2.2.1.2.3 Hydrology

The US Army Corps Wetland Delineation Manual (1987) defines 'wetland hydrology' as encompassing 'all hydrologic characteristics of areas that are periodically inundated or have soils saturated to the surface at some time during the growing season.' Standing water or saturated soil is an obvious indicator of wetland hydrology. Signs of previous saturation or presence of standing water may also indicate recent wetland conditions. These include the presence of drift lines, water marks, and water-stained leaves (Burton and Tiner, 2009). Topography, stratigraphy, vegetation, and soil characteristics all influence wetland hydrology (US Army Corps, 1987). The US Army Corps Wetland Delineation Manual (1987) defines wetlands as areas with inundation of less than 2 metres during 12.5-100% of the growing season.

2.2.1.3 Regulations Protecting Natural Wetlands

The Nova Scotia Wetland Conservation Policy (NSWCP) was released in September 2011 with the intent to clarify and consolidate provincial wetland regulations and formulate future objectives for the conservation of natural wetland areas. The NSWCP states that the Environment Act and the Environmental Goals and Sustainable Prosperity Act (EGSPA) both hold regulatory information concerning wetlands. Of importance, Activities Designation Regulations made under Section 66 of the Environment Act state that wetland alterations require approval through the Minister of Environment (Nova Scotia Environment Act, 1994b), and Section 4 of the EGSPA stated that a policy for the prevention of net loss of wetland be established by 2009 (EGSPA, 2007). NSE holds primary responsibility for enforcement of regulations concerning natural wetlands (NSWCP, 2011), and also requires permitting for storm drainage works based on legislation found in the Environment Act. This permitting requirement is in place to ensure, among other things, protection of natural wetlands from direct stormwater discharge (Nova Scotia Environment, 2002).

2.2.2 Constructed Wetlands

Research on the benefits of natural wetlands and resulting conservation efforts has been in place since the 1970's (Ramsar Convention Secretariat, 2011; US Army Corps, 1976). Research on design of constructed wetlands for treatment purposes began in the early 1990's (Buchberger and Shaw, 1995; Brix, 1994; Reed and Brown, 1992) as an attempt to improve on the treatment benefit of natural wetlands through standardized wetland design processes. Constructed wetlands are thought to be more appropriate for stormwater treatment than natural wetlands due to the fact that constructed wetlands may be designed to achieve specific treatment goals and hydrologic function may be predicted and controlled based on design. Natural wetlands may vary in treatment efficiency and wetland treatment function may be difficult to assess (Kennedy and Mayer, 2002). Constructed wetland design minimizes the use of natural wetlands for treatment purposes while also creating the ability to benefit from the treatment function of wetlands in upland areas where natural wetlands do not exist (Kadlec and Wallace, 2009; Kennedy and Mayer, 2002).

2.2.2.1 Constructed Wetland Types

Kadlec and Wallace (2009) outline three basic types of constructed wetlands used for the purpose of effluent treatment; vertical flow wetlands, horizontal sub-surface flow wetlands and free water surface wetlands. These three constructed wetland types are considered to be widespread in usage and acceptance.

2.2.2.1.1 Vertical Flow Wetlands

Vertical flow (VF) wetlands consist of wetland vegetation planted atop sand and subsequent layers of graded gravel. These types of wetlands operate via the downward movement of influent through layers of substrate. Influent is dosed across the top vegetation layer and treatment occurs as the effluent percolates through layers of sand and gravel (Kadlec and Wallace, 2009).

2.2.2.1.2 Horizontal Sub-surface Flow Wetlands

Horizontal sub-surface flow (HSSF) wetlands consist of wetland vegetation planted atop substrate consisting of gravel or soil. Influent is dosed first into a layer of coarse gravel media which precedes the main bed substrate. Treatment occurs as the influent moves horizontally through the bed media, under the vegetation surface (Kadlec and Wallace, 2009).

2.2.2.1.3 Free Water Surface Wetlands

Free water surface (FWS) wetlands largely resemble natural marshes in function and appearance. These wetlands consist of a bottom layer of substrate with emergent, submerged or floating vegetation. To avoid short-circuiting, influent is typically dosed from an inlet pipe into a deep inlet channel before flowing over the entire wetland area. The outlet has a water level control function to control the water level within the FWS wetland (Kadlec and Wallace, 2009).

2.2.2.2 Design Uses and Challenges

Kadlec and Wallace (2009) suggest FWS wetlands are most efficiently used for stormwater treatment as the design is capable of handling intermittent and erratic flow patterns. The use of FWS wetlands for primary wastewater treatment is not desirable as there is the potential for wildlife and human contact with the influent (Kadlec, 2009). HSSF wetlands are typically used for primary treatment of wastewater, as the influent is kept below the surface of the wetland preventing direct contact with contaminated water. Clogging of gravel inflow channels and unintended surface flow in HSSF wetlands may be an issue (USEPA, 1995b; Kadlec and Wallace, 2009). VF wetlands are of especial use in treatment situations requiring either anaerobic or oxidative conditions. The surface may be flooded to prevent oxygen transfer into the substrate creating ideal conditions for anaerobic reactions. Under non-flooded circumstances, VF wetlands have improved oxygen transfer capacities compared to HSSF wetlands and are able to better promote oxidation reactions (Kadlec and Wallace, 2009). Hydraulic efficiency must be taken into consideration in the design of any constructed wetland, as short-circuiting of influent can greatly reduce treatment efficacy (Reed *et al.*, 2006). Based on a review of literature, Kennedy and Mayer (2002) noted that the use of constructed wetlands is more cost-effective than the use of hard infrastructure, however in areas where land acquisition is difficult this may be a barrier to construction. Doku and Heinke (1995) suggest constructed wetlands be used in Northern Canada, as there is adequate land space and a need for efficient and inexpensive treatment in many small Northern communities. FWS wetlands may be used in cold climates, and TSS removal is improved under ice conditions, however treatment is only effective provided the wetland water column does not completely freeze (Reed *et al.*, 2006; Kadlec and Wallace, 2009). Treatment challenges in cold-climates include high volumes salt and sand in storm runoff from roadway ice treatments, high runoff volumes and pollutant load concentrations in spring melt water and reduced treatment capacity due to freezing of wetland substrate and ponded areas.

Freezing of wetland substrate and ponded areas is of particular concern as complete thawing may not occur before spring melt runoff moves through the wetland, thus allowing for highly polluted runoff to flow over the surface of the wetland and discharge without any treatment benefit (USEPA, 2012c). Some suggestions for wetland design in cold climates include creating a large ponded area at the inlet to capture and store spring melt runoff for gradual release into the wetland, having a wetland volume deep enough to avoid freezing the entire wetland column, and selecting salt-tolerant wetland vegetation for planting (USEPA, 2012c).

2.2.2.3 Regulations and Guidelines Governing Design

Within HRM, the use of wetlands for the treatment of stormwater is not specifically regulated. The Halifax Water Design and Constructions Specifications (HRWC, 2012) give authorization to the project engineer to implement alternative approaches to design provided the alternative approach will produce the results desired by the conventional method of design. The HRM Stormwater Management Guidelines (HRM, 2006) have an appendix section covering the implementation and design of stormwater wetlands, however it is not comprehensive.

2.3 Function of Surface Flow Wetlands As Stormwater Treatment Systems

2.3.1 Review of Current Literature

Wadzuk *et al.* (2010) studied a constructed stormwater treatment wetland located in Pennsylvania, US, during both 2003-2004 and 2007-2008. The wetland area is 0.4 ha and the drainage watershed is 18.2 ha, giving a watershed-to-wetland ratio of 0.02. During the duration of study, 19 storm events were captured and 30 baseflow samples were taken. Samples were analyzed for TSS, total dissolved solids (TDS), total N, total and reactive P, Cl, Pb, Cu and *E. coli*. The retention time of the wetland during storm flow was between 1-3 hours. For the majority of seasons sampled the wetland was able to decrease both the mass and concentration of contaminants during storm events. Retention time during base flow was approximately 58 hours. There was a reduction in mass and concentration for total and reactive P, total N, TSS, Cu and *E. coli* for all seasons sampled during base flow with the exception of Spring 2007-2008. It is noted that while there may have been contaminant reduction between the inlet and outlet, certain parameters still exceeded state guidelines at the outlet during both base and storm flow conditions.

Birch *et al.* (2004) studied an urban wetland receiving stormwater discharge in Sydney, Australia. The constructed wetland is 0.007 ha in size and services an urban residential area of 48 ha, giving a watershed-to-wetland ratio of 0.0015. Six storm events were captured during the months of April to June, 2000, with eight samples taken per storm event. Removal efficiencies for chromium (Cr), Cu, Pb, Ni and Zn were computed in terms of concentration reduction between inlet and outlet and were found to be 64%, 65%, 65%, 22% and 52% respectively. The wetland was found to be a source of Fe and Mn during most events sampled. Removal of TSS varied between 9% and 46% during four storm events; however the wetland was a source of TSS during two extremely high flow events. Inflow FC counts were found to be high and the wetland removed 76% of bacteria, on average, during four events monitored. The wetland was summarized as having a moderate ability to remove contaminants during storm events. The authors noted wetland size restrictions due to the urban location may have been a factor in not achieving higher removal rates.

Reinelt and Horner (1995) studied two natural FWS wetlands during a two year period during 1988-1990, located in Washington, US. The first wetland is approximately 2 ha in size and receives drainage from a highly urbanized watershed of 187 ha, giving a watershed-to-wetland ratio of 0.011. The second wetland is approximately 1.5 ha in size and inflow primarily consists of natural runoff from a heavily forested watershed of 87 ha in size, giving a watershed-to-wetland ratio of 0.017. Sampling was performed at the inlet and outlet of both wetlands, during both base flow and storm flow conditions. Surface water samples were tested for TSS, TP, total coliform (TC) and Zn. It was noted by the authors that the wetlands responded as expected to storm events; with the forested watershed having a slow return to baseflow conditions and very little inflow during events in the summer months and the urban watershed producing higher peak flows, low time of concentration, rapid return to baseflow conditions, and continuous piped inflow on a yearly basis. The forested-watershed wetland possessed retention times of 20 hours during baseflow conditions and 9 hours during storm flow. The urban-watershed wetland possessed retention times of 3.3 hours during baseflow conditions and 1.9 hours during storm flow. Contaminant removal rates were calculated on a mass basis and reported as percent removal of mass per year. The mass of contaminant inflow from the urbanized watershed was higher for all contaminants tested, in comparison with the forested watershed. Removal of TP, TSS, TC and Zn for the forested watershed wetland was 7.5, 13.6, 49.1 and 30.6% respectively. Removal of TP, TSS, TC, Zn for the urban watershed wetland were 82.4, 56.5, 29.0, 23.2% respectively. Groundwater and rainwater were also sampled and groundwater was found to be a source of TP within the forested watershed. Although the study reported positive removal percentages on a yearly basis, the urban watershed wetland was found to be a source of contaminants during certain seasons and flow regimes. The urbanized watershed wetland produced removal of TSS and TP solely during storm events, with the wetland becoming a source of both contaminants during baseflow. Removal of Zn within the wetland occurred primarily during storm events; however removal of FC was high and consistent during all flow regimes and seasons. The forested watershed wetland saw positive removal rates for all seasons and no discrepancies were reported between flow regimes. It was noted by the authors that the wetland would have been considered a source of P had it not been discovered that P was leaching from the groundwater into the wetland.

While evaluating P retention in wetlands, Reddy *et al.* (1999) outline the importance of considering both the short term assimilation of P into vegetative tissues as well as long term assimilation into soil and sediment. The authors suggest that studies which focus solely on short term P assimilation fail to fully assess the P retention capacity of a wetland. It is suggested by the authors that FWS wetlands with long retention times may be able to store P on a long-term basis.

Kadlec (2010) studied N dynamics in seven wetlands over the course of four years as part of the Des Plaines River Wetlands Demonstration Project, based in Illinois, US. The wetlands were receiving river water as inflow. Mass removal of NO_3-N ranged from 17-100% and concentration reduction ranged from 10-99% over the course of the study.

Fink and Mitsch (2004) studied nutrient dynamics in an emergent wetland located in Ohio, US. The wetland is 1.2 ha in size and receives runoff from a 17 ha agricultural and forested watershed, giving a watershed-to-wetland ratio of 0.17. Bi-monthly sampling was performed over 2 year period beginning in October, 1998. Over the 2 year study period, concentrations of nitrate-nitrite (NO_x) and TP were found, on average, to be 40 and 59% lower at the outlet than the inlet. There was an increase in P discharging from the wetland during storm events; however no increase in NO_x was reported. By mass, the wetland retained 41% of NO_x and 28% of TP during storm events. Although the wetland was still considered a phosphorous sink, it is noted by the authors that mass retention of P dropped by over 30% during the second year of sampling. Phosphorous loading into the wetland on a yearly basis is 7.1 g/m^2, or 71 kg/ha. Based on the literature, the authors suggest that the ideal loading rate to ensure a wetland performs as a phosphorous sink over the long-term is between 1-5 g/m^2 , or 10-50 kg/ha, of phosphorous per year (Mitsch, 1992; Mitsch *et al*., 1995; Richardson *et al.*, 1997).

Walker and Hurl (2002) studied suspended sediment and metal removal in a constructed ponded wetland system in Adelaide, Australia. The ponded wetland system covered an area of 172 ha and the primary drainage area contributing stormwater to the system was 2050 ha in size, giving a wetland-to-watershed ratio of 0.08. Walker and Hurl sampled suspended sediment on a biweekly basis from July through October, 1998. It was found that sediment deposition occurred as water moved through the system and metal concentrations of Zn, Pb and Cu were reduced by 57, 71 and 48% respectively. No removal of Cr was noted and the wetland was found to be a source of As. Walker and Hurl concluded that although sedimentation was the primary process contributing to contaminant removal, biological and chemical processes also played a role in contaminant removal. It was noted that concentrations of metals sorbed to suspended sediment was reduced as flow moved through the system and the authors attributed this reduction to biological and chemical removal processes.

Scholes *et al.* (1998) completed a 2 year study on metal retention in a constructed wetland located in the UK. The wetland is a 250 m long, sinusoidal FWS wetland receiving substantial stormwater inflow. Five sample sites were chosen along the length of the wetland, and water samples were taken on a bi-monthly basis and analyzed for Pb, Zn, Cu, Cd, Cr and Ni. During dry weather conditions, metal mass removal efficiencies were given as 13% for Zn, 53% for Cd, -180% for Pb, -171% for Cu, 52% for Ni, and 43% for Cr. Removal efficiencies during storm events were given as 100% for Zn, 100% for Cd, 79% for Pb, 92% for Cu, 17% for Ni, and 99% for Cr. Temporal Variability in metal removal efficiency was highlighted by the authors as a difficulty arising from the large variation in water quality and inflow volume when dealing with stormwater. Stormwater quality and inflow volume can change on an event and/or seasonal basis, making stormwater treatment wetland design difficult in comparison to those treating wastewater.

Goulet *et al.* (2001) studied a constructed FWS wetland receiving runoff from a mixed agricultural and residential watershed in Kanata, Ontario. The watershed area was 3.13 ha in size and receiving runoff from a drainage area of 637 ha in size, giving a wetland-to-watershed ratio of 0.005. Inlet and outlet sampling was done once every three days from April 1997 to March 1999, with interruptions in sampling during both Winter and Fall 1998. Samples were analyzed for total Fe and Mn and dissolved Fe, Mn, Zn, As and Cu. Treatment efficiency was calculated based on percent mass removal between inlet and outlet. On average, the wetland retained all metals tested with the exception of dissolved As. Seasonal variations were found to be significant, with the wetland functioning as both a source and sink for dissolved metals depending on the time of year. The wetland was found to be a source of total Mn for every season with the exception of fall; however removal of dissolved Mn occurred in every season with the exception of winter. Dissolved Zn was the only metal to be retained in the wetland for every season tested. The wetland was found to be a source of dissolved Cu during the spring, and removal rates varied from 7-25% for the other three seasons. The wetland had total Fe removal rates of 21 and 72% in the summer and fall, respectively, but was found to have very low removal in the spring and was a source of total Fe during the winter months. Dissolved Fe removal rates were found to be high in the spring and summer months (50-60%), but low in the fall and the wetland was found to be discharging dissolved Fe during the winter months. It is noted that due to measurement difficulties, inlet flow rates in this study were estimated based on the outlet flow rate and an assumption of a constant wetland water level within the wetland, and these estimated inlet flows were used to calculate the mass of contaminants moving into the wetland.

2.3.2 Conclusions

Small retention times of between 1-3 hours during stormflow conditions are noted in several studies. This is an indication that flow is rapidly moving through the study wetland, which may negatively impact treatment capacity and reduce contaminant removal capabilities, as noted by Kadlec and Knight (1996). Large variations in watershed-to-wetland ratios, from 0.005 to 0.17, make it difficult to compare treatment capacity between studies. For the most part, contaminant reduction is presented on a mass removal basis; however, concentration reduction data is also used as a metric to evaluate performance. The use of concentration reduction data without comment on potential groundwater influence makes it difficult to ascertain if the study wetland is removing contaminants or if dilution is a factor in apparent concentration reduction. Wetlands that are under the influence of groundwater may see contaminants introduced to the wetland via groundwater discharge, as noted by Reinelt and Horner (1995), thus interfering with the calculation of contaminant removal rates. Negative contaminant removal rates were noted in several studies, meaning the study wetlands became a source of contaminants. Scholes *et al.* (1998) noted a discharge of Pb and Cu at the outlet of their study wetland during baseflow conditions and Birch *et al.* (2004) found their study wetland to be a source of Fe and Mn during all storm events sampled and a source of TSS during extreme storm events; however, positive removal rates of nutrients, metals and bacteria are noted in the majority of studies. Of importance is the fact that the reviewed literature pertains to studies on a short-term basis (<5yrs) and little is known about the function of treatment wetlands as contaminant sinks over the long term.

2.4 Research Needs and Gaps

In general, there is a noted absence in research pertaining to the use of both natural and constructed FWS wetlands in the treatment of stormwater. The majority of completed research on treatment wetlands to date has focused on the treatment of municipal and industrial wastewater using both VF and HSSF wetlands. Of the research that exists on stormwater wetlands, the reduction of nutrient loading has been the primary focus. Very few wetland studies focused on multiple parameters, highlighting only the ability of a wetland to remove a specific contaminant as opposed to a suite of contaminants normally found in stormwater. A large portion of relevant FWS wetland research has been performed in Australia, and while comprehensive, differences in climate may affect wetland performance. In particular, cold climate stormwater wetlands studies are limited.

The research which has been conducted on stormwater treatment wetlands has provided evidence that both constructed and natural wetlands can be used to improve stormwater quality. However, differences in sampling duration and frequency, technique and selection of parameters make it difficult to compare current research, and make conclusions on wetland performance and design criteria challenging. Expected contaminants in stormwater vary greatly based on land use and season. The variability of contaminant inflow into a proposed treatment wetland is noted by researchers as one of the primary obstacles in designing high-functioning treatment systems. This highlights the need for detailed site-specific studies to characterize stormwater quality, climate, and hydrology in order to effectively design wetland treatment systems.

3.0 METHODOLOGY

3.1 Site Description and Research Strategy

Kuhn Marsh is a small urban wetland located within the Halifax Regional Municipality, in Dartmouth, Nova Scotia, located at 44°41’12” N and 63°31’41” W (Fig 3-1). The marsh encompasses an area of approximately 2 ha and is situated in the headwaters of a 7 km^2 sub-watershed draining to the inlet of Morris Lake. The primary inlet to Kuhn Marsh is a stormwater outlet pipe currently servicing the Westphal neighborhood, with urban development commencing meters from the marsh inlet. The outlet of the marsh is located several hundred meters southeast of the inlet. The outlet drains through a culvert under Main Street, and continues for several kilometers before draining into Morris Lake. Kuhn Marsh is located in an urban environment which has a direct impact on the environmental quality of the marsh. Large waste items, such as automobiles and housing debris, remain in the marsh from a housing settlement demolished in the late 1970’s and this situation is compounded by poor solid waste management from the current surrounding apartment complexes. Both Kuhn Marsh and Morris Lake are used by the public as natural hiking and wildlife observation areas, and Morris Lake also has great community value as a recreational water body. The optimization of Kuhn Marsh as a stormwater treatment wetland and the proper management of urban waste in the surrounding area could show positive impacts on water quality both in the marsh, and in downstream water bodies. As Kuhn Marsh is a natural wetland, there is inherently a large amount of uncertainty concerning hydrologic function and treatment capacity. The following section details research strategies which were developed to characterize the current function of the natural wetland.

3.2 Monitoring Methods

Monitoring within the marsh began in May 2011 and continued through October 2012. Methodology for monitoring was organized into three subsections: 1) Physical characterization, 2) Hydrologic characterization, and 3) Water quality monitoring.

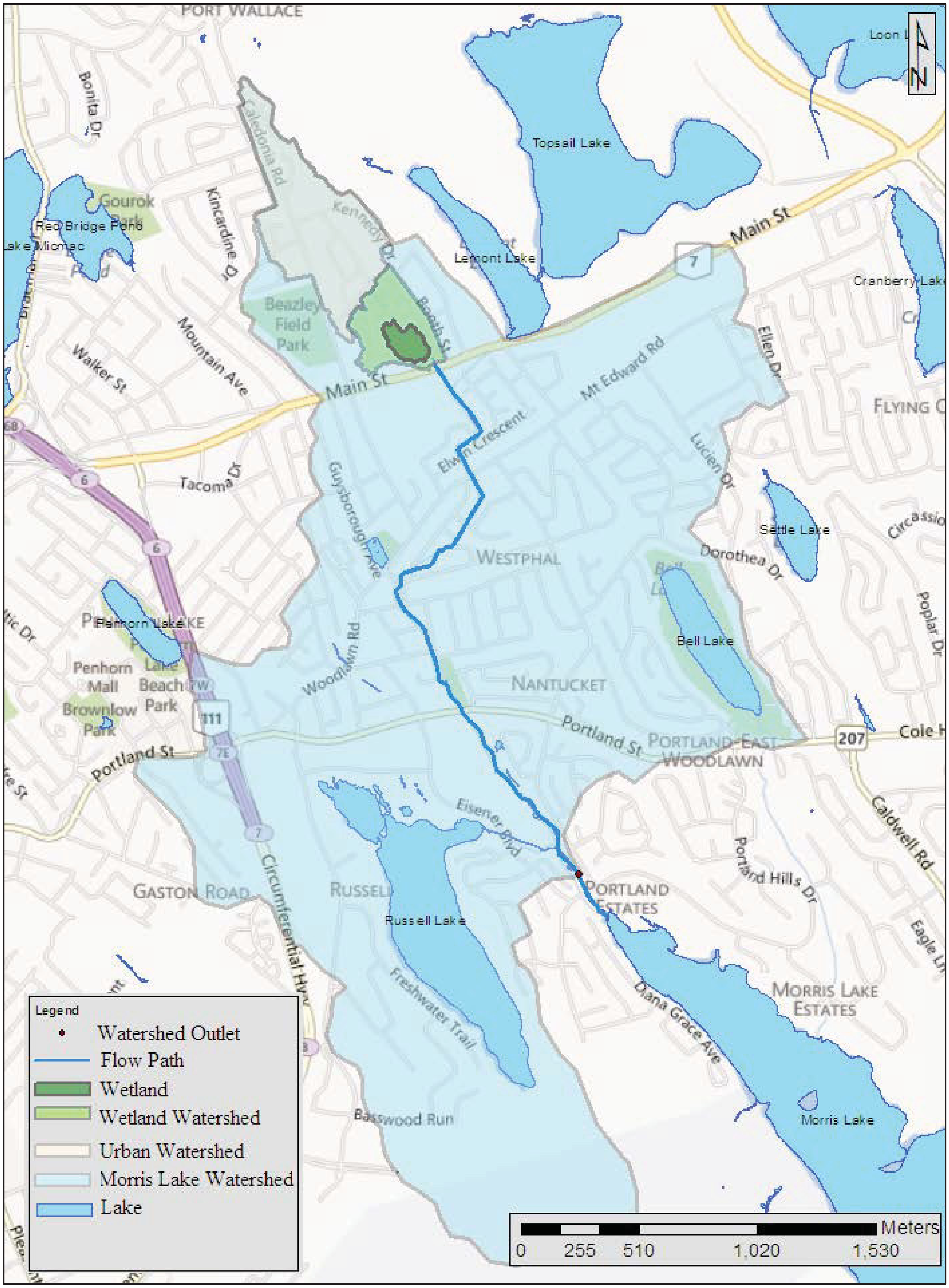

Figure 3-1 Kuhn Marsh location within Morris Lake watershed

3.2.1 Physical Characterization

Physical characterization involved three main areas of study; the urban watershed draining to the stormwater outfall discharging into Kuhn Marsh, the wetland watershed draining directly into the wetland, and the wetland area itself (Fig 3-2). Four linear transects (Fig 3-3) were laid perpendicular to direction of flow across representative cross-sections within the wetland drainage area. Transect locations were mapped out using ArcGIS™ and found and staked in-field using a TopCon Real-Time Kinematic Global Position System (RTK-GPS) (Waterloo, Ontario). Transects were used to provide locations for soil and vegetation analysis as well as groundwater monitoring. The entire study site (watershed and wetland) is underlain by the Halifax bedrock formation (ЄOMh) falling under the Meguma Group (ЄOM). The Halifax formation is characterized by slope-outer shelf slate, siltstone, minor sandstone, and Fe-Mn nodules (Nova Scotia Department of Natural Resources, 2000). According to the 1963 Soil Survey of Canada, the two primary soil series in the area of Kuhn Marsh are peat and the Bridgewater (Bw) soils series. The Bw soil series is categorized by a surface soil layer and subsoil of brown shaly loam over yellowish-brown shaly loam, with a parent material of olive shaly loam till derived from Precambrian slates, with the topography defined as gently undulating to gently rolling with good drainage capacity (Soil Survey of Halifax County Nova Scotia, 1963).

3.2.1.1 Urban Watershed Characteristics

The urban watershed (Fig 3-2) encompasses an area of approximately 28.5 ha. Of this 28.5 ha area, approximately 7.7 ha is considered impermeable, which accounts for 27% of the total urban watershed area. Of the impermeable areas in the urban watershed, 10% is attributed to large buildings, 30% to homes, 42% percent to roadways and 18% to paved areas such as parking lots. Delineation of the urban watershed area was completed using remote sensing Light Detection and Ranging (LiDAR) data for the area. A digital elevation model (DEM) of 2m resolution was created from the LiDAR and manipulated with the ArcHydro function of ArcGIS v10 (ESRI, Redlands, CA, USA). A hand-delineation of the watershed on a 1:2400

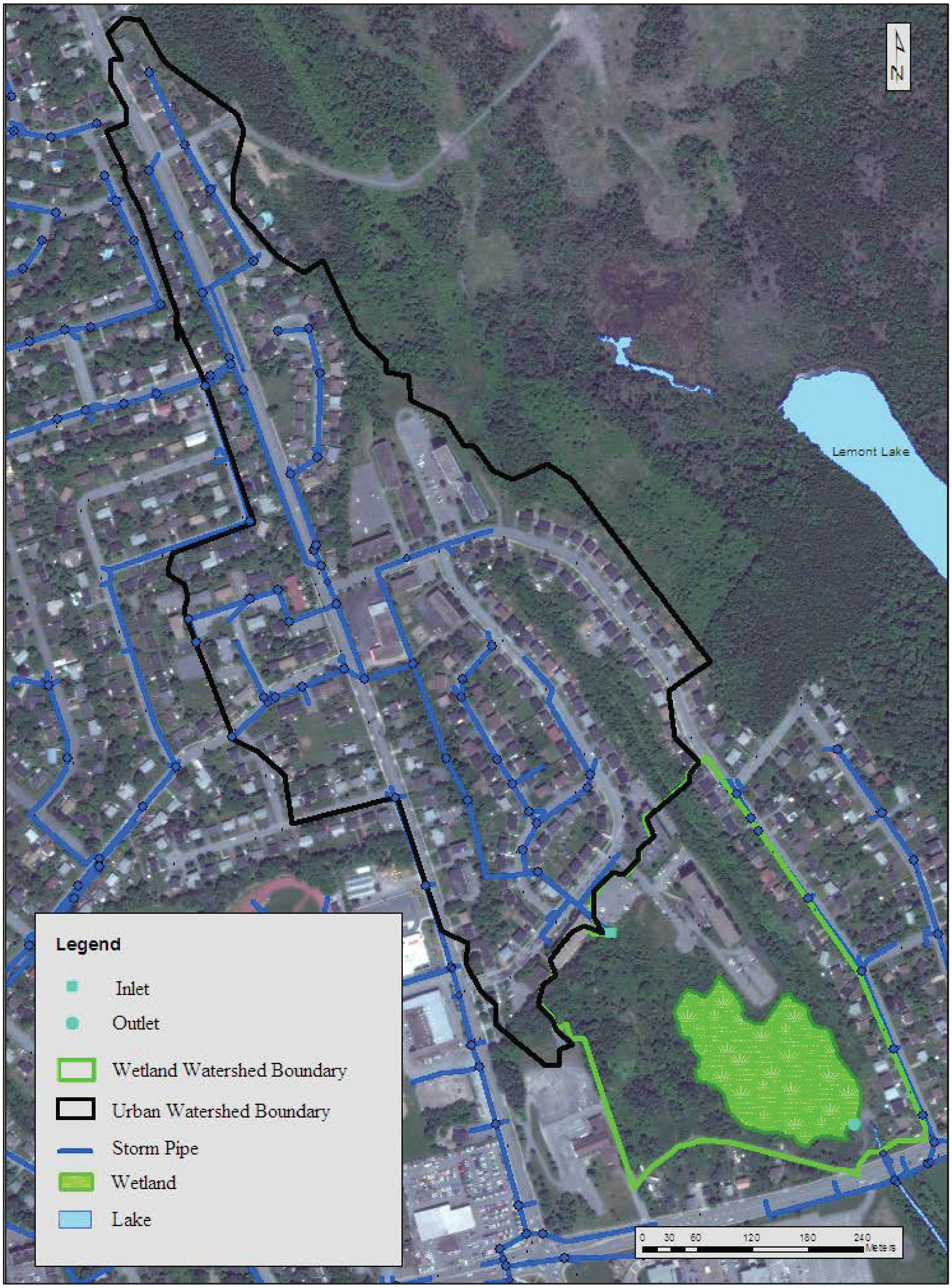

Figure 3-2 Watershed boundary detail with stormwater infrastructure

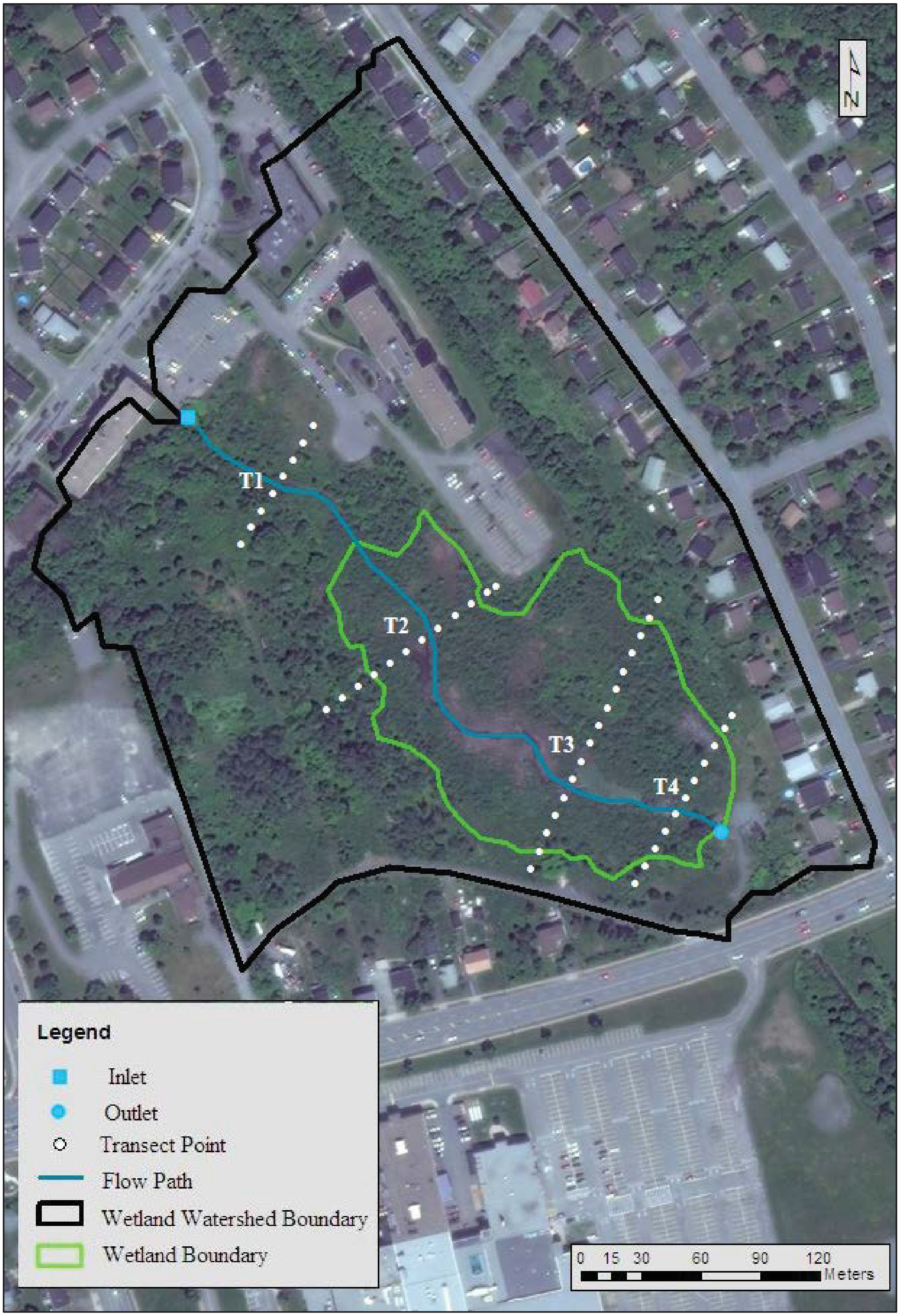

Figure 3-3 Kuhn Marsh wetland area with transect points

scale topographic map was used to verify the ArcHydro delineation. The urban watershed has both sanitary and storm sewer systems. As the storm sewer system defines the drainage into Kuhn Marsh from the urban watershed, 'ground-truthing' was used to verify actual drainage area from the ArcHydro delineation. All LiDAR and Geographical Information System (GIS) data for the Halifax area was obtained through the GISciences Centre at Dalhousie.

3.2.1.2 Wetland Watershed Characteristics

The wetland watershed (Figure 3-2) encompasses an area of approximately 10.84 ha. The wetland watershed is almost entirely vegetated with approximately 1 ha considered impermeable, which accounts for 9.2 % of the wetland watershed. The impermeable areas are attributed to two large apartment complexes and the resulting paved areas constructed within the wetland watershed. A settlement of approximately a dozen homes had existed within the wetland watershed and was demolished in the mid-1970s. The demolition of this settlement has left large waste items within the wetland watershed, such as remnants of automobiles and house foundations. The topography of the wetland watershed is basin-like with steep slopes draining into the wetland on both east and west borders. The wetland watershed was delineated in ArcHydro using LiDAR data for the area.

3.2.1.2.1 Wetland Watershed Soils Characterization

Soils characterization was completed through the excavation of shallow test-pits at various locations along four transects within the wetland watershed (Fig 3-3). The Canadian System of Soil Classification (1998) was used to aid in the identification of soil type and structure of the identifiable soil horizons in each test-pit. Soil samples were taken from each test-pit for grain-size analysis. Sieve analysis was completed on the samples to determine grain size distributions for particles greater than 0.075 mm in size, while the hydrometer testing was done to determine the distribution of fine particles less than 0.075 mm in size. Sieve analysis was completed in accordance with *Soil Sampling and Methods of Analysis* (Canadian Society of Soil Science, 2008). Hydrometer testing was completed using a laboratory method based upon American Society for Testing and Materials (ASTM, 2007) standard D422 and the American Association of State Highway and Transportation Officials (AASHTO, 2012) standard T88. Using grain size distribution data obtained from sieve and hydrometer analyses, soil texture was classified based on percent sand, silt and clay using a standard soil texture diagram as given by Mays (2012).

3.2.1.2.2 Wetland Watershed Vegetation Analysis

Vegetation analysis within the wetland watershed was completed using field data and information from satellite images of the area. Dominant vegetation types (grasses, shrubs, trees, etc.) were identified at each transect point using a radial study area of 10-20 metres. A mixed forest area consisting of trees and medium-sized shrubs interspersed with grasses and a meadow area consisting of small shrubs and grasses are the primary vegetation areas within the wetland drainage watershed. For each vegetation area (mixed forest, meadow and wetland) the dominant vegetation species were also identified and their wetland status determined using the Nova Scotia Wetland Indicator Plant List (Atlantic Canada Conservation Data Centre, 2011).

3.2.1.3 Wetland Delineation

Wetland delineation was performed using the guidance of the US Army Corps of Engineers Wetland Delineation Manual (US Army Corps, 1987). Initial wetland area was determined by assessing the presence of wetland (hydrophytic) vegetation. Approximate wetland boundaries were defined between areas of wetland and upland vegetation and analysis of soil was used to determine final wetland boundaries. The presence of gleying or organic accumulation was used as an indication of hydric soil conditions. Soils with brightly coloured mottles were used as indications of fluctuating water levels along the boundary of the wetland. The final wetland boundary was flagged with survey tape and the RTK-GPS was used to collect GPS coordinates along the boundary for mapping purposes.

3.2.2 Hydrologic Characterization

Hydrologic characterization of the wetland involved the study of both surface water and groundwater within the wetland area and surrounding watersheds. Hydrologic monitoring and data collection methods are outlined in the following sections.

3.2.2.1 Surface Water Characterization

3.2.2.1.1 Meteorological data

Meteorological data was taken from several different locations for use in characterizing the hydrology of Kuhn Marsh. Barometric pressure data was taken from the Environment Canada climate station at Bedford Range, located approximately 12 km north-east of the marsh. Average monthly climate data for the Halifax area was taken from the Environment Canada climate station at the Halifax Stanfield International Airport, located approximately 23 km north of the marsh (Table 3-1). All Environment Canada climate data was accessed through the National Climate Data and Information Archive (NCDIA, 2013). For the purpose of hydrologic calculations within the Kuhn Marsh watershed, rainfall data was taken from two Hobo® weather stations (Onset®, Southern MA, USA) located in Dartmouth; one on Anderson Street located 1.5 km south of the marsh, and one at Lake Lemont located less than 1 km east of the marsh.

Table 3-1 Halifax International Airport Climate Averages from 1971-2000

1971-2000 Averages	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Temperature (°C)	-6	-5.6	-1.4	4	9.8	15	18.6	18	14.1	8.3	3.1	-2.8
Rainfall (mm)	101	69	96.4	96.1	106	98	102	93	104	126	133	115
Snowfall (cm)	54.6	50.1	41.1	20.9	3.3	0	0	0	0	2.3	14	43.9
Precipitation (mm)	149	114	135	118	110	98	102	93	104	129	146	155

3.2.2.1.2 Surface Water Velocity and Discharge

Stream gauging was carried out at both the inlet and outlet of Kuhn Marsh during baseflow and storm flow sampling to monitor surface water flow through the system. The Velocity-Area method (Herschy, 2009) was used to calculate flow across the stream section. Velocity and depth measurements were taken using either a USGS Model 6205 Pygmy current meter or a FlowTracker Acoustic Doppler Velocimeter (SonTek/YSI, San Diego, CA, USA). All velocity measurements were taken at a location equivalent to 60% of the total depth from the water surface.

3.2.2.1.3 Continuous Water Level Monitoring

To enable the determination of surface water flow on a continuous basis, both the inlet and outlet of the marsh were instrumented with Levelogger® (Solinst Canada Ltd, Georgetown, Ontario) pressure transducers programmed to continuously log water level on 15 minute intervals. Pressure transducers were installed in May 2011 and logged data continuously until November 2012. The transducers were not vented to the surface and therefore water level was corrected for differences in barometric pressure using data from the Bedford Range climate station. Verification of water level readings was done in-field ten times by manually measuring the water level above the sensors at a known point in time and comparing against transducer readings.

3.2.2.1.4 Stage discharge

Stage discharge relationships were constructed for both the inlet and the outlet of the marsh using flow calculated from in-field velocity-area measurements in combination with corresponding water level readings from the pressure transducers. See appendix for curves and fitted relationships.

3.2.2.1.5 Tracer Studies

Three separate tracer studies were completed in Kuhn Marsh during Fall 2012. Two tracers were completed during high flow events, on September 20th and 22nd respectively, and one during a low flow event on October 18th. Rhodamine WT dye (20% by weight) was used for all three tracer studies. Rhodamine is a conservative, fluorescent dye typically used to track water movement through a surface flow wetland. Dosages of 250, 100 and 50 mL were used for the first, second and third tracers. An in-situ Sonde® (YSI 6920, Yellow Springs, OH, USA) fitted with an optical Rhodamine sensor (YSI 6130) was used to log Rhodamine concentration readings on a 1 minute interval for the duration of the studies.

3.2.2.2 Groundwater Characterization

3.2.2.2.1 Horizontal groundwater movement

Groundwater monitoring infrastructure was installed in Kuhn Marsh on transects perpendicular to the main inlet flow path using. In total, twelve monitoring wells were installed. The wells are located on either side of the flow path, in upland and wetland areas when topography permitted (See Fig 3-4). In preparation for well installation, holes were hand-augured using a 2-inch diameter Dutch auger with a fixed handle length of 1m. Well holes were augured either to refusal or a depth of 1 m. Wells were screened up to near-surface using 1-inch diameter polyvinyl chloride (PVC) well screen and finished with solid PVC piping to an average distance of 0.5 meters above-ground. Filter sock was used on all wells to prevent sediment intrusion through the well casing. Upland wells were backfilled with silica sand whereas wetland wells

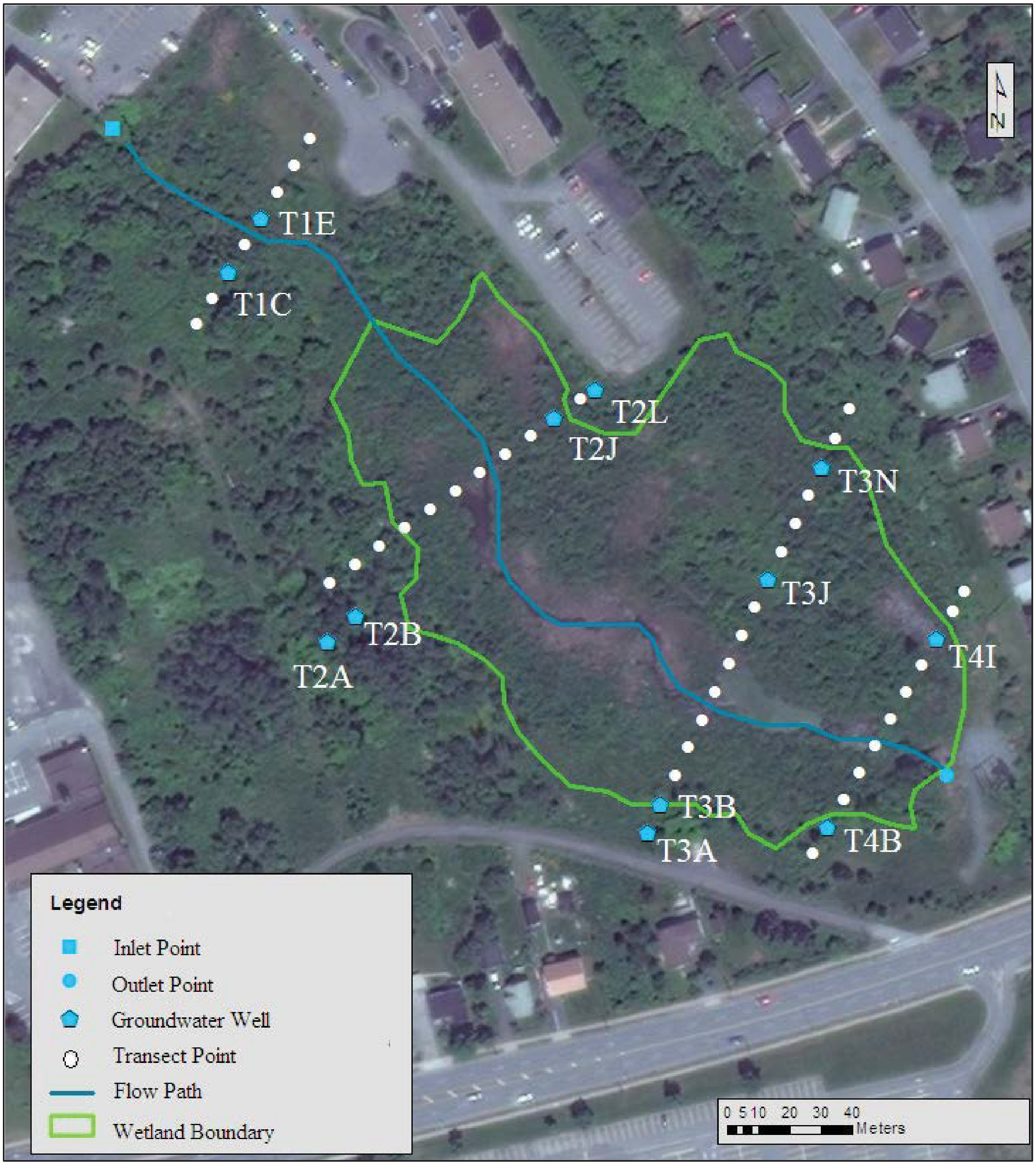

Figure 3-4 Groundwater well location

were backfilled with parent material. All wells were fitted with a bentonite-clay surface plug. Water levels were measured on ten occasions from June through September 2012 using a dipper-T water level reader (Heron Instruments, Dundas, Ontario). In addition to water level measurements, distance from the well cap to the ground surface and total well depths were also measured to check for indications of sedimentation and vertical well movement. A standard rod-and-level survey was conducted to determine ground surface elevation at each well location within Kuhn Marsh.

3.2.2.2.2 Rate of Well Recharge

Well bail-down tests were performed on all groundwater wells in Kuhn Marsh on September 12th, 2012 to determine the rate of recharge of each well. A dipper-T water level reader was used to measure the initial water level within each well before beginning the bail-down test. Using a hand pump, wells were pumped dry and the water level reader was immediately re-inserted into the well casing. Water level measurements were taken at increasing time intervals until the pre-test water level was achieved. In cases where time to full recharge was exceeding one hour, measurements were taken until the water level reached within 5 cm of the original measurement. Bail-down tests were performed with guidance from *Applied Hydrogeology* (Fetter, 2001).

3.2.3 Water Quality Monitoring and Analysis

Surface water samples were taken beginning in May 2011 and continuing through October 2012, with the exception of the winter months between November 2011 and February 2012. Groundwater quality monitoring was conducted between July and September 2012. All laboratory analysis was completed at Dalhousie University.

3.2.3.1 Surface Water Quality Monitoring

With the exception of direct surface runoff, the stormwater outlet pipe located at the headwater of the marsh is the sole inlet to Kuhn Marsh. All surface flow is discharged from the marsh by way of a control structure located at the outlet. Discrete and continuous surface water monitoring was performed at the inlet and outlet of the wetland during both baseflow and stormflow conditions.

Figure 3-5 Kuhn Marsh Inlet during a) baseflow and b) stormflow

Figure 3-6 Kuhn Marsh Outlet during a) baseflow and b) stormflow

3.2.3.1.1 Discrete

Discrete surface water sampling consisted of grab sampling and in-situ water quality measurements at specific points in time at both the inlet and outlet of Kuhn Marsh. In-situ water quality measurements were taken using a handheld Sonde (YSI 600R), which measures temperature, DO, conductivity and pH. During seventeen months of study, twenty discrete baseflow sample points and eleven storm events were captured at both inlet and outlet. When possible, baseflow sampling was conducted between each storm event. Between two and seven discrete sample points were collected during each storm event, depending on storm duration and time of storm onset. Discrete water quality monitoring was not performed at night due to safety considerations.

3.2.3.1.2 Continuous

The use of two in-situ continuous-monitoring Sondes (YSI 6600) provided the ability to monitor several parameters within Kuhn Marsh over the span of several days. The two in-situ Sondes were both fitted with a DO probe (YSI 6150+), a turbidity probe (YSI 6136), a pH probe (YSI 6561), and a conductivity probe with temperature sensor (YSI 6560). The in-situ Sondes were tethered at both the inlet and outlet, ensuring proper submergence of sensor and probes, and measurements were taken on ten minute intervals. Throughout the course of the study, five separate continuous monitoring events were captured using in-situ continuous-monitoring. Event durations ranged between 24 and 48 hours. Calibration of the in-situ Sondes was completed in accordance with the YSI 6600 User Manual using laboratory grade purchased standards for all calibrations with the exception of DO.

3.2.3.2 Surface Water Quality Analysis

Surface water samples were analyzed in the laboratory at Dalhousie for the following parameters: TSS, turbidity, TOC, TN, TP, *E.coli*, and total metals. All sample bottles were transported to Dalhousie in a refrigerated cooler and were stored or preserved immediately upon return to the laboratory.

3.2.3.2.1 TSS and Turbidity

Samples for TSS analysis were taken using 1L plastic bottles and stored at 4°C pending analysis. TSS samples were analyzed in duplicate within 7 days of sampling, after procedure 2540 D in *Standard Methods for the Examination of Water and Wastewater* (Clesceri *et al.,* 1998). Samples of known volume were filtered through pre-dried and weighed Whatman™ 934-AH™ glass fibre filters (Maidstone, UK). Filters were then dried at 105°C overnight and placed in a desiccator to cool prior to final weighing. TSS concentrations are reported in mg/L. Turbidity was measured using a standard Hach 21400AN Turbidimeter (Loveland, CO, USA). An initial blank of deionized water was measured to account for discrepancies in the sample vessel and four subsequent turbidity readings were taken per sample and averaged. Turbidity is reported in NTU.

3.2.3.2.2 E.coli

Samples for bacterial analysis were taken using 500 mL plastic bottles which had been sterilized with 70% ethanol (EtOH) prior to sampling. Bacteria samples were stored at 4°C and analyzed in duplicate within 24 hours of sampling. Bacterial colonies were quantified using the membrane filtration method. Sample volumes ranging from 0.1-100 mL were filtered through a 0.45 μm sterile membrane filter and plated atop a sterile pad saturated with 2 mL of m-ColiBlue24® broth (EMD Millipore Corporation, Billerica, MA, USA). Sample volumes of less than 10 mL were brought up to a volume equivalent to 10 mL using a sterile 0.85% saline solution. Plates were incubated at 35±0.5 °C for 24 hours prior to counting. *E.coli* counts are reported in CFU/100 mL.

3.2.3.2.3 TOC and TN

Total N samples were taken in 250 mL bottles and frozen immediately upon return to the laboratory pending analysis. In preparation for analysis, TN samples were thawed in a water bath and decanted into smaller glass vessels and preserved to a pH below 2 using sulfuric acid (H_2SO_4). TOC samples were taken in 250 mL bottles and stored for a maximum of 24 hours at 4°C before being decanted into smaller glass vessels and brought to a pH below 2 for preservation using phosphoric acid (H_3PO_4). Acidified TN and TOC samples were stored at a temperature of 4°C until time of analysis. Both TN and TOC were analyzed using a Shimadzu ASI-V (Kyoto, Japan), fitted with a TOC-V analyzer for TOC and a TNM-1 analyzer for TN. Both TOC and TN are reported in mg/L.

3.2.3.2.4 TP

Total P samples were taken in 250 mL bottles and frozen immediately upon return to the laboratory pending analysis. Samples were thawed in a water bath prior to analysis. Due to samples having high turbidity and the possible adsorption of P to organic matter, TP samples were digested using potassium persulfate ($K_2S_2O_8$) prior to analysis. Analysis of TP was completed by LKB Biochrom Ultrospec 4051 spectrophotometer (Cambridge, UK) using the Ascorbic Acid Method 4500-P E. from Clesceri *et al.* (1998). TP concentrations are reported in mg/L.

3.2.3.2.5 Total Metals

Metals were sampled using 50 mL plastic vials and preserved using nitric acid (HNO_3) upon return to the laboratory. Due to turbidity readings greater than 1 NTU for most samples, an acid digestion step was performed on all samples prior to analysis using a ThermoScientific XSeries2 inductively coupled plasma mass spectrometry (ICP-MS) unit (Toronto, Ontario). Acid digestion was completed using trace metal grade HNO_3 according to USEPA Method 200.8 for the Determination of Trace Elements in Waters and Wastes by ICP-MS (USEPA, 1994). Samples exceeding the maximum detection limit of 500 ppb were diluted using deionized water and re-analyzed.

3.2.3.3 Groundwater Quality Monitoring

Groundwater quality monitoring was completed on all twelve wells located within Kuhn Marsh, with wells sampled on four occasions beginning in July through September 2012. Groundwater Monitoring consisted of a purge cycle and a sampling cycle with an average of two hours between cycles to allow for recharge. A hand pump and length of tubing was used to purge one full well volume from each well prior to sampling. Sterilization of well pumping equipment was done in-field using 70% EtOH to prevent cross-contamination between wells. Each well had separate, designate tubing and all tubing was rinsed and sterilized in-lab prior to each use in-field. Each well sample was decanted into a sterile 500 mL plastic bottle.

3.2.3.4 Groundwater Quality Analysis

Due to highly variable well volume yields, groundwater quality analysis was limited to the quantification of *E.coli* within the wells. Quantification of *E.coli* was done using the Colilert® Quanti-tray/2000™ (IDEXX Laboratories, Westbrook, ME, USA) method. Samples of 100 mL volume were dosed with Colilert® powder and incubated for 24 hours at 35±0.5 °C. Fluorescence under UV light indicated the presence of *E.coli*, and Most Probable Number (MPN) of bacteria per 100 mL sample was calculated using an MPN calculation table provided with the method. The method provides quantification of up to 2,419 MPN per 100 mL sample. Samples found to exceed an MPN of 2,419 after the initial analysis were diluted for subsequent analyses.

3.2.3.5 Microbial Source Tracking

Microbial source tracking was completed on samples from the inlet and outlet of the marsh as well as each groundwater well. Each groundwater and surface water sample retrieved was filtered through a 0.45 µm cellulose-nitrate membrane (Whatman Laboratory Division, Maidstone, UK), with filter volumes of approximately 500 mL for surface water samples and between 100 and 500 mL for groundwater samples, depending on well volume yield. Nucleic acids were extracted from each filter using a PowerWater deoxyribonucleic acid (DNA) extraction kit (MoBio Laboratories Inc., Carlsbad, CA, USA) according to manufacturer instruction, and the extracted DNA was stored at -20°C prior to analysis. The DNA extracts were probed for human-specific *Bacteroidales* 16S rRNA gene sequences using a quantitative polymerase chain reaction (qPCR) protocol and primers developed by Reischer et al. (2007). The qPCR analysis was performed using a CFX96 Touch system (Bio-Rad Laboratories Inc., Hercules, USA). Each reaction was run in 25 µL batches, containing 12.5 µL of 2x SsoFast Probes Supermix (Bio-Rad Laboratories Inc.), 200 nM of the BacH forward primer (5'-CTTGGCCAGCCTTCTGAAAG-3'), 200 nM of the BacH reverse primer (5'-CCCCATCGTCTACCGAAAATAC-3'), 100 nM each of the two dual-labeled fluorescent probes BacH_pC (FAM-5'-TCATGATCCCATCCTG-3'-NFQ–MGB) and BacH_pT (FAM-5'-TCATGATCCCATCCTG-3'-NFQ–MGB), 10 µg of bovine serum albumin, and 4 µL of sample DNA. Eight tenfold serial dilutions of human-specific *Bacteroidales* plasmid standards (10^0-10^7) were run in triplicate to generate the standard curve used for sample enumeration. Blank DNA extraction controls, no template controls, and negative DNA controls (DNA from *E. coli* ATCC 25922 culture) were included in the qPCR run.

3.3 Data Analysis

3.3.1 Surface Runoff

The volume of surface runoff generated during storm events within both the urban drainage watershed and the wetland drainage watershed was calculated using the NRCS (SCS) curve method (US Department of Agriculture Soil Conservation Service, 1972; Mays, 2012), where surface runoff is computed as:

$$R = \frac{(P - I_a)^2}{(P - I_a) + S} \qquad [3.1]$$

Where:

P_r = precipitation (mm)

I_a = initial abstraction (mm)

S = potential maximum retention (mm)

The initial abstraction (I_a) is taken as twenty percent of the potential maximum retention (S). The value of S is dependent upon a curve number (CN) and is calculated using the following equation:

$$S = \frac{25400 - 254CN}{CN} \qquad [3.2]$$

The CN is chosen based on hydrologic soil group and land use within the watershed of interest, and may be altered to account for either low or high antecedent moisture conditions (AMCs).

3.3.2 Horizontal Groundwater Movement

Horizontal groundwater movement within the wetland watershed was calculated between pairs of upland and wetland groundwater wells located on either side of the main flow path. The Bouwer and Rice well bail-down method (Bouwer and Rice, 1976; Fetter, 2001; Mays, 2012) was used to determine the hydraulic conductivity (K) of the soil surrounding the wells. The Bouwer and Rice method pertains to fully or partially penetrating wells in an unconfined aquifer, and K is given as:

$$K = \frac{r_c^2 \ln(R_e/R)}{2L_e} \frac{1}{t} ln\left(\frac{H_0}{H_t}\right) \qquad [3.3]$$

Where:

r_c = radius of the well casing (m)

R_e = effective radial distance over which head is dissipated (m)

R = radius of gravel pack surrounding well (m)

L_e = length of well screen (m)

t = time (days)

H_0 = well drawdown at time=0

H_t = well drawdown at time=t

Using the calculated K values, and water level measurements, the Dupuit equation (Fetter, 2001) was used to calculate flow per unit width in the upper saturated soil layer, as follows:

$$q = \frac{K}{2L}(h_1^2 - h_2^2) \quad [3.4]$$

Where:

K = hydraulic conductivity (m/day)

L = distance between well points (m)

h_1 = up-gradient well head (m)

h_2 = down-gradient well head (m)

3.3.3 Hydraulic Retention Time and Volumetric Efficiency

The mean tracer detention time, or hydraulic retention time (HRT), was calculated using a moment analysis, as outlined by Kadlec and Wallace (2009):

$$\tau = \frac{1}{M_i}\int_0^\infty tQ_{out}Cdt \quad [3.5]$$

Where:

M_i = mass of tracer into wetland (g)

t = time (hours)

Q_{out} = outlet flow (L/hour)

C = outlet tracer concentration (g/L)

Continuous monitoring data for both outlet flow and concentration were used in the calculation of the mean HRT. Integrals were evaluated using the five-point quadrature numerical integration formula (Fogler, 1992), as follows:

$$\int_{X0}^{Xn} f(x)dX = \frac{h}{3}(f_0 + 4f_1 + 2f_2 + 4f_3 + 2f_4 + \cdots + 4f_{n-1} + f_n) \quad [3.6]$$

Volumetric efficiency is a measure of the ineffective volume of a treatment wetland, and is given as the ratio of effective volume to actual wetland volume. Volumetric efficiency was calculated using the following formula, from Kadlec and Wallace (2009):

$$e_v = \frac{V_{active}}{V_{nominal}} = \frac{\tau * Q_{out}}{A * d} \quad [3.7]$$

Where:

τ = hydraulic retention time (hours)

Q_{out} = outlet flow (L/hour)

A = wetland area (m^2)

d = estimated wetland depth (m)

3.3.4 Contaminant Loading

Changes in contaminant loading between marsh inlet and outlet were categorized using both a contaminant reduction and mass removal approach. Concentration reduction was calculated between each grab sample event, during both baseflow and stormflow, using the following formula (Kadlec and Wallace, 2009):

$$\%[C]_{reduction} = 100x\left(\frac{C_i - C_o}{C_i}\right) \quad [3.8]$$

Where:

C_i = inlet concentration (mg/L; CFU/100 mL)

C_o = outlet concentration (mg/L; CFU/100 mL)

Concentration reduction values provide important information on the lowering of contaminant concentrations below specified guideline levels; however, if dilution is occurring within a wetland, concentration reduction may occur while large masses of contaminants are still being discharged from a wetland. To quantify the mass of contaminants leaving the wetland, the following equation was applied (Kadlec and Wallace, 2009):

$$\%M_{removal} = 100x\left(\frac{Q_iC_i - Q_oC_o}{Q_iC_i}\right) \quad [3.9]$$

Where:

C_i = inlet concentration (mg/L; CFU/100 mL)

C_o = outlet concentration (mg/L; CFU/100 mL)

Q_i = inlet flow (L/s)

Q_o = outlet flow (L/s)

Baseflow mass flux calculations were completed using average concentration and flow values computed for all baseflow grab samples. Linear interpolation was used to estimate stormflow concentration values between grab sample points. Linearly interpolated concentration values were then applied to continuous flow data to compute mass flux of contaminants during storm events.

3.3.5 Catchment Lag

For the purpose of this research, catchment lag was defined as the time between the center of mass effective rainfall and the resulting peak of the discharge hydrograph (Leopold, 1991). Catchment lag was calculated using the following equation expressed by Watt and Chow (1985):

$$t_L = 0.000326\left(\frac{L}{\sqrt{S}}\right)^{0.79} \quad [3.10]$$

Where:

L_b = basin length (m)

S_b =slope of basin (m/m)

A model was used to alter calculated catchment lag values based on actual discharge data. A unit hydrograph was created using a gamma function equation, as presented in Akan and Houghtalen (2003). The unit hydrograph was converted to a direct runoff hydrograph by multiplying the unit hydrograph by the effective rainfall.

$$Q_u = Q_{up}\left[\left(\frac{t}{t_p}\right)exp\left(1-\frac{t}{t_p}\right)^{n-1}\right] \quad [3.11]$$

Where:

Q_u = unit hydrograph flow (m^3/s)

Q_{up} = unit hydrograph peak flow (m^3/s)

t = time (s)

t_p = time to peak flow (s)

n = dimensionless shape parameter

3.3.6 Statistical Analysis

All 95% confidence intervals were computed using the one-sample t-test function of the Minitab v16 (Minitab Inc., State College, PA, USA) statistical software package. Regression was used to determine contaminant correlation and create stage-discharge rating curves. All regression and necessary data transformations were performed using Microsoft® Excel (Redmond, WA, USA).

4.0 RESULTS AND DISCUSSION

4.1 Physical Characterization

4.1.1 Watershed and Wetland Land Use

Land use areas for the urban and wetland drainage watersheds are presented in Table 4-1. The urban drainage watershed refers to the residential stormwater catchment area up gradient of Kuhn Marsh. The stormwater outfall located at the drainage point of the urban watershed is the primary inlet to Kuhn Marsh. The wetland drainage watershed refers to the area downstream of the marsh inlet point that generates storm runoff contributing to flow at the outlet of Kuhn Marsh. Impermeable surfaces make up approximately 27% of the urban watershed area, largely due to the high density of homes and roadways in the area. Two large apartment complexes with parking lots are located within the wetland drainage watershed and this accounts for the wetland drainage watershed having 9% impermeable surfaces. Including both the urban watershed and the wetland drainage watershed, the wetland-to-watershed ratio for Kuhn Marsh is 0.06. A wetland-to-watershed ratio range of 0.03-0.05 has been suggested by Fink and Mitsch (2004) to ensure a wetland has the capacity to effectively treat the stormwater runoff from a given watershed. Catchment lag is defined as the time between the center of mass of the effective rainfall and the peak of the rainfall hydrograph (Watt and Chow, 1985). The average catchment lag for the urban watershed was calculated as 12.4 minutes. Leopold (1991) suggests catchment lag values of between 9 and 15 minutes for a basin of similar size and percentage of impermeable area.

Table 4-1 Land use distribution and percent impermeable area within study watersheds

Urban Watershed	Area (ha)	Percent	Wetland Watershed	Area (ha)	Percent
Total Area	28.4	100.0	**Total Area**	10.9	100.0
Impermeable Area	7.7	27.2	Wetland Area	2.2	20.2
Parking Lots	1.4	4.8	Wetland Drainage Area	8.7	79.8
Roads	3.3	11.5	**Impermeable Area**	1.0	9.2
Large Buildings	0.8	2.7	Parking Lots	0.8	7.1
Homes	2.3	8.2	Large Buildings	0.2	2.1

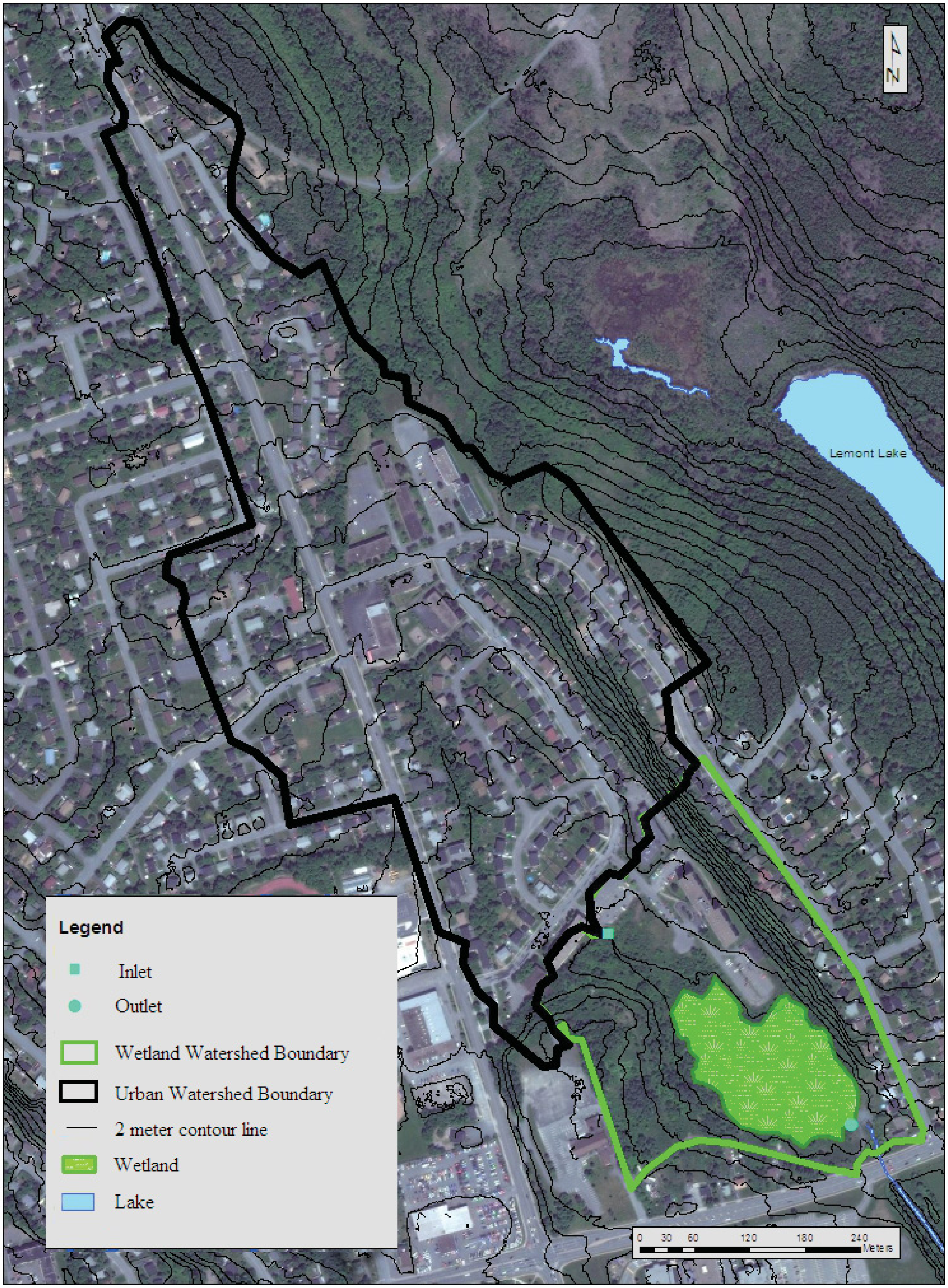

Figure 4-1 Kuhn Marsh watershed boundaries with local topographic contours

4.1.2 Soil Analysis

Soil samples were taken at various transect point locations within Kuhn Marsh (Section 3, Fig 3-3). The results from the sieve and hydrometer soil analysis are presented in Table 4-2. Sandy Loam is the primary soil class within the marsh. Sandy Loam consists primarily of sand (>45%) and silt (<50%) and a small fraction of clay (<20%), and is of moderate permeability (Mays, 2012).

Table 4-2 Soil characteristics for transect points within wetland drainage watershed

Well ID	Horizon	% Sand	% Silt	% Clay	Soil Class
T1C	A	68.8	30.4	0.8	Sandy Loam
	B	76.9	22.2	0.9	Loamy Sand
	C	63.4	35.8	0.9	Sandy Loam
T1E	A	73.7	26.2	0.2	Loamy Sand
T1G	A	41.8	56.1	2.0	Silt Loam
	B	45.3	53.0	1.7	Silt Loam
	C	55.1	44.4	0.5	Sandy Loam
T2B	A	72.3	26.0	1.7	Loamy Sand
	C	60.7	37.4	1.9	Sandy Loam
T2E	B	61.5	37.4	1.1	Sandy Loam
T2J	C	57.6	41.4	1.0	Sandy Loam
T3A	Fill	53.6	44.1	2.3	Sandy Loam
	C	53.9	45.5	0.6	Sandy Loam
T3B	C	53.5	45.5	1.0	Sandy Loam
T3J	Fill	70.1	28.9	1.1	Sandy Loam
T3N	C	66.4	32.5	1.1	Sandy Loam
T4B	A	63.3	35.8	0.9	Sandy Loam

4.1.3 Vegetation

A map of the spatial distribution of vegetation types within the wetland drainage watershed boundary is presented in Figure 4-1 and the dominant vegetation species are presented in Table 4-3. OBL plants almost always occur in wetlands. FACW plants have a 67-99% probability of occurring in wetlands. FAC plants are equally likely to occur in wetlands and uplands. FACU plants have a 67-99% chance of occurring in uplands and UPL plants almost always occur in uplands (Atlantic Canada Conservation Data Centre, 2011).

Table 4-3 Dominant plant species within wetland drainage watershed

Plant Species	**Common Name**	**Wetland Status**
Wetland		
Polygonum sagittatum	Arrowleaf Tearthumb	OBL
Myrica gale	Bog Myrtle	OBL
Typha latifolia	Cattail	OBL
Nuphar lutea	Pond Lily	OBL
Impatiens capensis	Spotted Touch-Me-Not	FACW
Alnus	Alder	FAC
Betula	Birch	FAC
Mixed Forest		
Acer saccharum	Sugar Maple	FACU
Sorbus americana	Mountain Ash	FACU
Syringa vulgaris	Lilac	UPL
Rubus idaeus	Red Raspberry	FAC
Meadow		
Phleum pratense	Timothy	FACU
Solidago canadensis	Goldenrod	FAC

Kuhn Marsh has all dominant wetland plants species within the OBL, FACW and FAC species groups, fulfilling the 1987 US Army Corps requirements that a wetland must have more than 50% of the dominant wetland species in the OBL, FACW or FAC category. As depicted in Figure 4-1, *Myrica gale* is the predominant wetland plant species. This medium-sized, shrubby bush has root nodules containing bacteria in the genus *Frankia*, which act to fix nitrogen from the atmosphere, converting N_2 into NH_4 (Baker and Parsons, 1997). Thus, *Myrica gale* has the ability to increase N concentrations in soil.

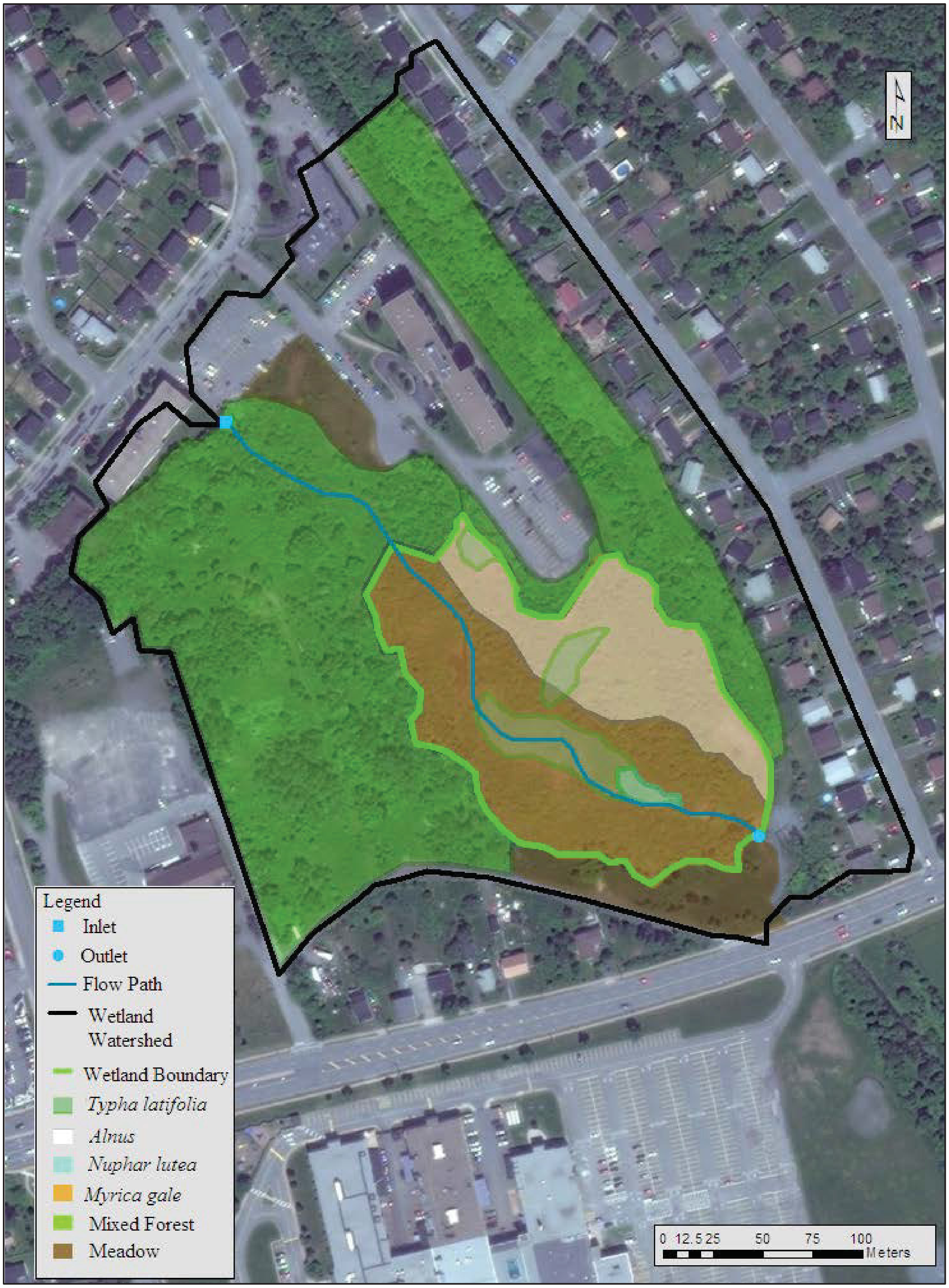

Figure 4-2 Vegetation map for wetland and wetland drainage watershed

4.2 Hydrologic Function

4.2.1 Hydraulic Retention Time

Tracer studies were performed during both stormflow and baseflow conditions to determine hydraulic retention times within the marsh. The tracer studies on September 20th, 2012 (Fig 4-2a) and September 22nd, 2012 (Fig 4-2b) were both done during stormflow conditions. A water volume of approximately 3.5 million litres passed through the wetland on September 20th, with a mean hydraulic retention time of 2 hours. On September 22nd, a water volume of approximately 1.5 million litres passed through the wetland, with a mean hydraulic retention time of 4 hours. A baseflow tracer study was conducted on October 18th, 2012 (Fig 4-2c). A baseflow mean hydraulic retention time of 11 hours was determined, with a water volume of approximately 0.6 million litres flowing through the wetland during the study. Both Schueler (1987) and Whipple and Randall (1983) recommend a minimum hydraulic retention time of 18 hours for adequate removal of contaminants through settling, which is higher than current stormflow retention times within Kuhn Marsh. Short-circuiting or channelization of flow occurs within a wetland when a primary flow path between inlet and outlet is created, thus reducing hydraulic retention times and volumetric efficiency. Volumetric efficiency is the ratio of actual wetland treatment volume, a function of actual hydraulic retention time, to the available wetland volume, a function of total wetland surface area and estimated water depth. With an assumed decrease in water depth over the wetland surface area over time, the volumetric efficiency for Kuhn Marsh was calculated as approximately 20% for each measured retention time. This means only 20% of the available wetland volume is currently being used for treatment purposes. In their study on two natural wetlands, Reinelt and Horner (1995) observed a correlation between removal of TP and hydraulic retention time. An increase in TP removal between wetlands, from 8 to 82%, corresponded with an increase in hydraulic retention time from 3.3 hours to 20 hours and a subsequent increase in volumetric efficiency from 22 to 38%.

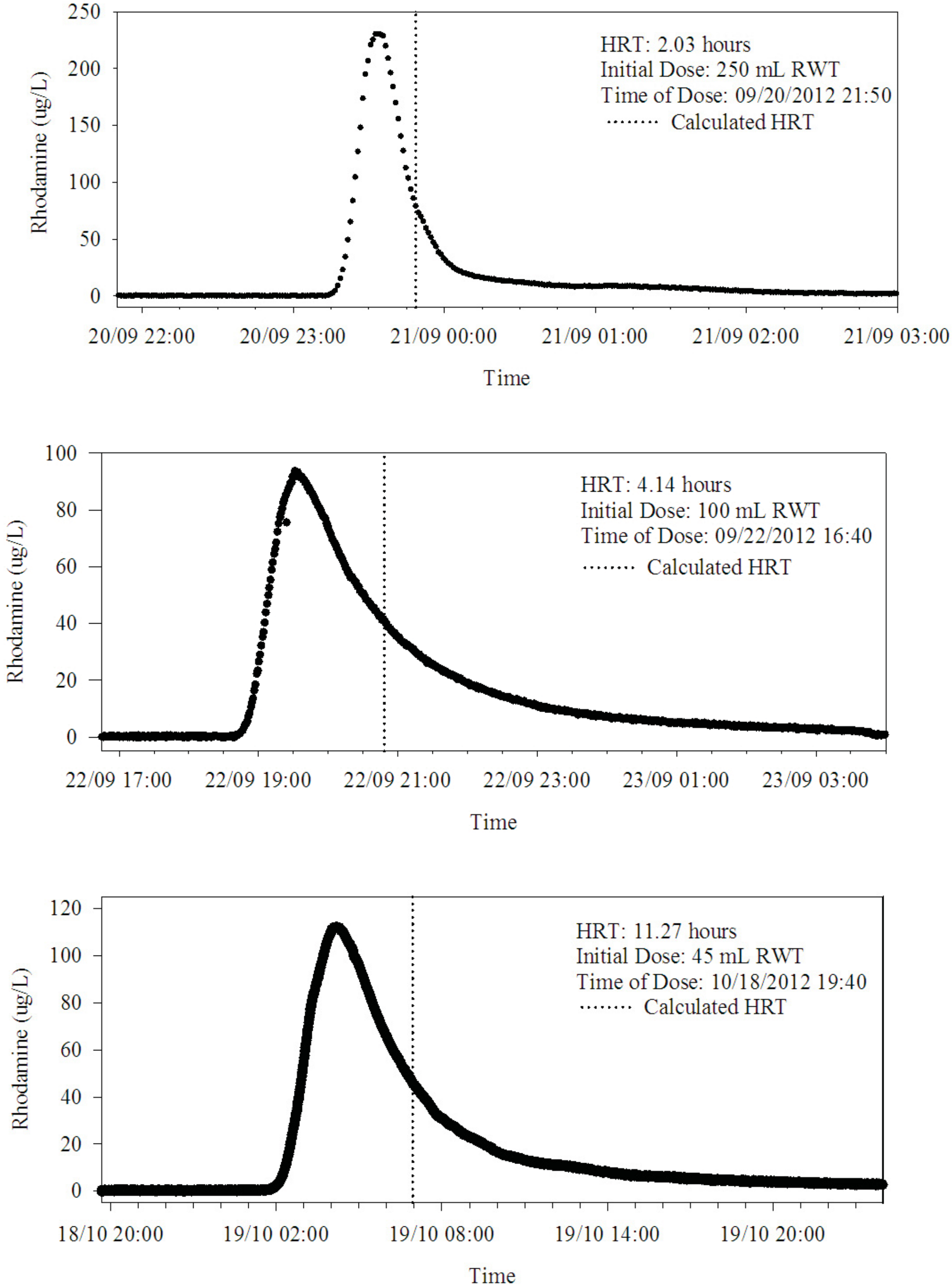

Figure 4-3 Kuhn Marsh tracer studies, completed on a) September 20th, 2012 b) September 22nd, 2012 and c) October 18th, 2012

4.2.2 Storm Event Hydrology

During the course of monitoring within Kuhn Marsh, surface water stage data at both inlet and outlet was successfully captured at a 15 minute interval during ten storm events. Stage-discharge relationships were developed for both inlet and outlet using measured flow values and these relationships were used in combination with the stage data to produce storm hydrographs for each event (see appendix). Total volume and peak flow values were calculated from the inflow-outflow hydrographs for each storm event and are presented in Table 4-4. Stage-discharge relationships for marsh inlet and outlet can be found in the appendix.

Table 4-4 Volume and peak flow data for storm events A through J.

Storm	Start Date	Volume In (ML)	Volume Out (ML)	Peak Flow In (L/s)	Peak Flow Out (L/s)
A	June 13 2011	11.1	24.5	130.8	409.8
B	July 30 2011	1.5	3.2	101.7	138.9
C	Aug 8 2011	5.4	17.2	131.8	622.7
D	Oct 1 2011	2.6	7.8	71.7	112.0
E	Oct 4 2011	5.4	19.8	91.9	532.0
F	April 23 2012	1.7	3.5	120.4	143.8
G	June 26 2012	1.2	5.8	85.2	263.6
H	July 24 2012	1.0	3.5	101.6	230.6
I	Sept 5 2012	1.6	7.5	137.2	462.4
J	Sept 10 2012	3.4	14.3	239.1	564.4

In comparing volumes of water moving through the wetland during events, the volume of water entering the wetland through the inlet pipe is on average 30% of the total volume discharging via the outlet control structure. As the inlet is the sole surface water inflow point, this is indicative of a large volume of water entering the wetland downstream of the inlet via alternate hydrologic mechanisms. Peak flows are higher at the outlet, which indicates that peak flow dampening does not occur within the wetland. Based on the fact that 70% of the surface outflow volume is entering the wetland area downstream of the inlet pipe, the lack of peak flow dampening may be attributed to this additional volume of water entering the wetland and discharging via the outlet control structure. The pattern of higher water volumes and peak flows at the outlet is visually apparent when looking at the storm hydrographs for Kuhn Marsh.

Hydrologic data for each storm event are presented in Table 4-5. The runoff coefficient for a watershed is calculated as the ratio of runoff volume to the total volume of rainfall over the watershed. Typical runoff coefficients are 0.25-0.40 for suburban areas (Mays, 2012; Akan and Houghtalen, 2003). Higher runoff coefficients are indicative of higher surface runoff rates. Abstractions such as evapotranspiration, interception, infiltration and depression storage are all mechanisms which reduce surface runoff volumes and may be factors in lowering runoff coefficients.

Table 4-5 Kuhn Marsh storm event data

Storm	**Start Date**	**Duration (hr)**	**Rainfall (mm)**	**5 day AMC (mm)**	**15 min Peak Intensity (mm/hr)**	**Urban Watershed Runoff Coefficient**
A	June 13 2011	78	119.4	0.6	9.6	0.33
B	July 30 2011	24	24.4	33.2	20.0	0.22
C	Aug 8 2011	48	71.2	19.4	15.2	0.27
D	Oct 1 2011	66	36.8	3.2	11.2	0.25
E	Oct 4 2011	42	74.0	59.0	8.8	0.26
F	April 23 2012	36	25.6	4.2	17.6	0.23
G	June 26 2012	36	45.3	26.2	22.4	0.09
H	July 24 2012	30	21.5	0.0	15.2	0.16
I	Sept 5 2012	60	56.0	0.0	26.4	0.10
J	Sept 10 2012	42	76.4	56.0	64.0	0.16

Rainfall intensities calculated using the intensity-duration-frequency (IDF) curve for the Halifax area show that return periods of 15 minute duration range from 2 years for intensities of 40 mm/hr to 25 years for intensities of 70 mm/hr (NCDIA, 2013). All storm events, with the exception of Storm J, were shown to have peak rainfall intensities of 15 minute duration below the 2 year return period. Monthly rainfall data for the Kuhn Marsh area (Table 4-6) show reduced rainfall amounts during the summer of 2012 as compared to 2011. In comparison with historic average monthly rainfall data, the 2011 sample season was characterized by above-average rainfall amounts, whereas rainfall amounts were below-average for the 2012 sample season. Below-average rainfall during the 2012 sample season may have been a contributing factor in lower runoff coefficients calculated for the 2012 season, as compared to 2011.

Table 4-6 Monthly precipitation amounts for Kuhn Marsh area

	Monthly Precipitation (mm)		
Month	**2011**	**2012**	**Average Rainfall (1971-2000)[1]**
April	124.2	91.4	96.1
May	124.2	101.8	106.0
June	162.6	72.3	98.0
July	107.6	47.7	102.0
August	143.2	62.8	93.0
September	51.8	316.2	104.0
October	310.8	46.2	126.0

[1]NCDIA, 2013

4.2.3 Horizontal Groundwater Movement

Horizontal groundwater fluxes were calculated between groundwater wells located on Transect 2 and 3 within the wetland drainage watershed. Well locations are identified in Section 3, Figure 3-4. The hydraulic conductivity values calculated for all wells are presented in Table 4-7. The average inferred hydraulic conductivity value was 0.06 m/day, which is one order of magnitude smaller than the suggested literature value of 0.5 m/day for soil that is predominantly sandy loam (Bouwer and Rice, 1976; Akan and Houghtalen, 2003). This discrepancy between measured and suggested hydraulic conductivity values may be explained by changes to the surrounding soil during well construction (*ie.* compaction) or potential fouling of the filter sock used on the wells to prevent fine sediment intrusion.

Table 4-7 Hydraulic conductivities for groundwater wells within Kuhn Marsh

Well ID	K Value (m/day)	Well ID	K Value (m/day)
T1C	0.009	T3A	0.067
T1E	0.034	T3B	0.045
T2A	0.056	T3J	0.165
T2B	--	T3N	0.008
T2J	0.019	T4B	0.072
T2L	0.074	T4I	0.073

Horizontal groundwater movement was calculated between sets of wells located on both sides of the main flow path through the wetland. Wells T2A, T2B, T3A and T3B are located on the west side of the main flow path. Wells T2L, T2J, T3N and T3J are located on the east side of the main flow path. Horizontal flow values for groundwater wells on Transects 2 and 3 are presented in Table 4-8. All computations were completed using measured hydraulic conductivity values. As all values are positive, this indicates horizontal movement into the wetland area.

Table 4-8 Horizontal groundwater movement into Kuhn Marsh

	T2A to T2B	**T2L to T2J**	**T3A to T3B**	**T3N to T3J**
Date	**q (m^2/day)**	**q (m^2/day)**	**q (m^2/day)**	**q (m^2/day)**
June 7 2012	0.00351	0.00007	0.00345	0.00091
June 14 2012	0.00350	0.00002	0.00332	0.00082
June 28 2012	0.00564	0.00002	0.00447	0.00097
July 11 2011	0.00275	0.00005	0.00188	0.00064
July 17 2012	0.00231	0.00018	0.00157	0.00043
July 25 2012	0.00413	0.00030	0.00424	0.00061
August 7 2012	0.00362	0.00011	0.00408	0.00058
September 6 2012	0.00470	0.00007	0.00425	0.00046
September 12 2012	0.00629	0.00006	0.00437	0.00093
September 25 2012	0.00593	0.00014	0.00403	0.00091

Based on a measured length of influence of 120 m for each set of wells, the volume of groundwater moving horizontally into the wetland averages 1 m^3/day. Although horizontal groundwater movement was being directed into the wetland during all sampling runs, the volume of groundwater moving horizontally into the wetland is evidently minimal and not considered to have an influence on the overall water balance.

4.2.4 Water Balance

A water balance on the wetland was completed for each measured storm event. Based on topography and on-site assessment, the only discharge point for the wetland was assumed to be the outlet control structure (Q_{out}). Surface flow to the wetland was attributed to three main sources: piped flow coming into the wetland from the urban drainage watershed (Q_{inlet}), surface runoff from the wetland drainage watershed ($Q_{surface}$), and direct precipitation on the wetland area (P). Discrepancies between surface flow to the wetland and discharge volumes calculated at the outlet control structure were attributed to groundwater discharge into the wetland (G_{in}). The water balance was calculated as follows:

$$\Delta S_{\Delta t} = P + G_{in} + Q_{in} - [ET + G_{out} + Q_{out}] \quad [4.1]$$

And,

$$Q_{in} = Q_{inlet} + Q_{surface} \quad [4.2]$$

As the change in storage over the duration of the storm event ($\Delta S_{\Delta t}$) is assumed to be zero, the inflow components of the water balance are made equivalent to the surface flow out of the wetland. Evapotranspiration (ET) rates are considered minimal during storm events and abstraction through evapotranspiration was not considered in the storm event water balance. Estimated groundwater discharge into the wetland refers to the vertical upwelling of groundwater into the wetland area and this, along with piped inflow from the urban drainage watershed, accounts for the majority of flow discharging from the wetland. As the majority of groundwater flow is presumed to exit via the outlet pipe, subsurface groundwater outflow (G_{out}) was considered minimal and was not included in the calculations. It is noted that during storm events, an average of 50% of the total volume of rainfall that fell on the entire study area was discharged via the outlet control structure. Depression storage and soil storage within the watershed may account for this volume of water retention. The water balance is presented in Figure 4-3.

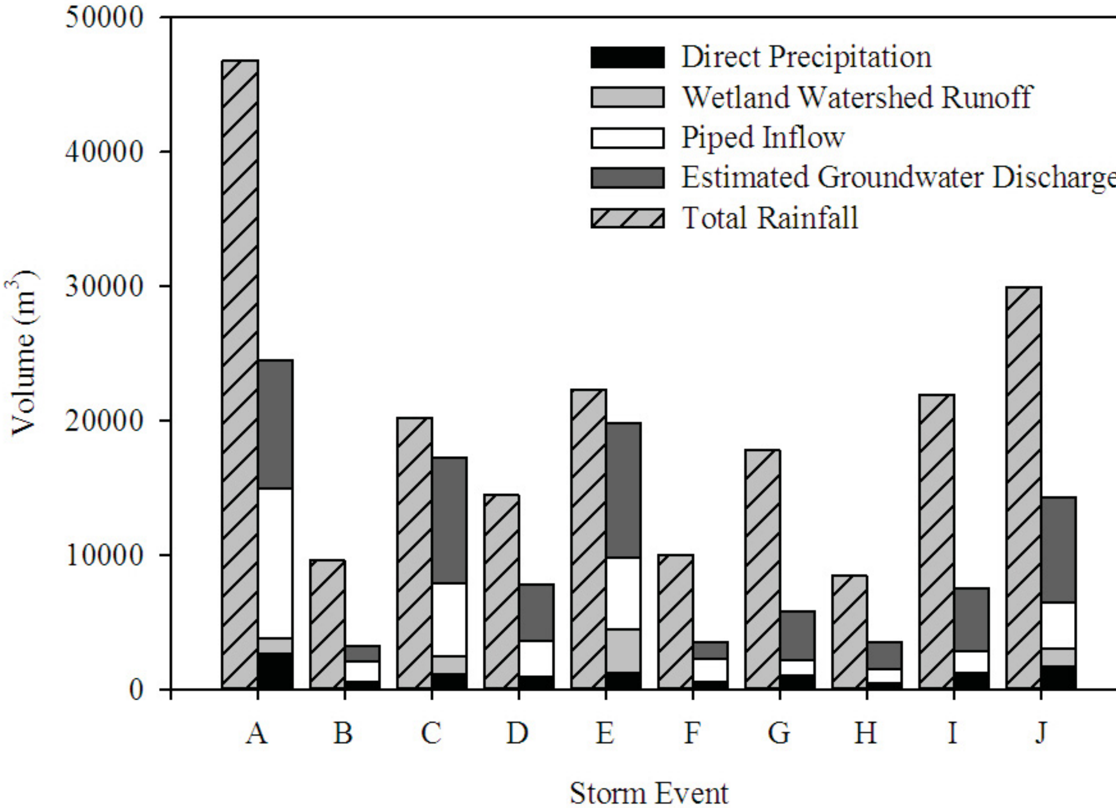

Figure 4-4 Inflow components of storm event water balance

Based on the large influx of groundwater, Kuhn Marsh may be defined as a discharge wetland. On average, 50% of the flow at the outlet of the marsh may be attributed to groundwater discharge. Although wetlands are typically presented as beneficial areas of groundwater recharge, several studies of natural wetlands and un-lined stormwater detention ponds have found groundwater discharge to be a large contributor to overall outflow volumes (Raisin *et al.*, 1999; Kantrowitz and Woodham, 1995; McCann and Olson, 1994). Constructed wetlands are typically designed with impermeable liner systems to eliminate groundwater influence. Groundwater discharge can contribute to the dilution of contaminants within a wetland or can act as an additional source of contaminants into a wetland, depending on groundwater quality.

4.3 Contaminant Removal Capacity

4.3.1 Event Mean Concentration

When considering contaminant loading from a watershed during a storm event, EMCs provide a quantification of the concentration of contaminants contained in a given volume of storm runoff during a rain event. EMCs calculated for the urban drainage watershed considered herein were based on contaminant concentrations and flow measured at the inlet of Kuhn Marsh during a storm event, and are presented in Table 4-9.

Table 4-9 EMCs for urban watershed runoff as compared to literature values

	TSS (mg/L)	**TN (µg/L)**	**TP (µg/L)**	**Pb (µg/L)**	**Cu (µg/L)**	**Fe (µg/L)**	**Al (µg/L)**	**Cd (µg/L)**	**Zn (µg/L)**
Median	12.5	4575.0	78.2	2.0	7.4	883.1	303.1	0.04	27.3
Mean	17.5	4722.3	75.4	2.4	6.8	1018.3	393.7	0.04	27.2
St.Dev.	12.8	877.0	18.3	1.2	1.6	347.7	160.4	0.02	6.0
Residential[1]	101	2600	383	144	33	--	--	--	135
Mixed[1]	67	1500	262	144	27	--	--	--	154
HDR[2]	47.7	2700	300	12	17.3	--	--	0.7	145.9
MDR[2]	30.5	1700	200	6.1	9.7	--	--	0.5	59.4

USEPA, 1983[1] CH2M HILL, 2000[2]

Literature values were taken from the USEPA PLOAD Manual (USEPA, 2001) and were chosen to highlight discrepancies when looking at literature values for EMCs for representative watersheds. Values presented by the National Urban Runoff Program (NURP) (USEPA, 1983) differ greatly from those suggested by the North Carolina Department of Environment (CH2M HILL, 2000) for high density residential (HDR) and medium density residential (MDR) areas. With the exception of TN, EMCs from the urban drainage watershed are lower than those values suggested by the NURP and are more aligned with the MDR values presented by the North Carolina Department of Environment; however, it is important to note that estimates of pollutant loading into Kuhn Marsh would be exaggerated if either set of EMC literature values were used in the calculations. If it is assumed that the mass of pollutants is the same for watersheds of similar land use, design EMCs should be chosen based on watersheds with similar land use as well as similar storm runoff volumes and catchment lag.

4.3.2 Guideline Exceedences

Guideline exceedences were evaluated using CCME Freshwater Aquatic Life (FAL) guidelines for TP, TN, and a selection of metals. There are no CCME FAL guidelines pertaining to bacteria so the CCME Recreational Water Guidelines were used to evaluate *E. coli* concentrations. All concentration plots can be found in the appendix. Trophic status characterization of inlet and outlet TP concentrations for 19 baseflow and 38 stormflow samples are presented in Table 4-10. During baseflow conditions, 95% of inlet TP sample concentrations were below the eutrophic zone, as compared to only 26% of outlet samples. During stormflow, 71% of inlet TP sample concentrations and 92% of outlet TP sample concentrations were within or exceeding eutrophic zone concentrations. While baseflow entering Kuhn Marsh may be below eutrophic zone guidelines, the majority of TP concentrations at the outlet control structure were found to be above eutrophic zone guidelines. This indicates a trend of increasing TP concentration between the inlet and outlet of Kuhn Marsh, with TP concentrations exceeding guidelines at the outlet control structure.

Table 4-10 Percent distribution of inlet and outlet TP during baseflow and stormflow conditions

TP	**Below Eutrophic Zone** (<35 μg/L)		**Eutrophic Zone** (35-100 μg/L)		**Hyper Eutrophic Zone** (>100 μg/L)	
	Baseflow	Stormflow	Baseflow	Stormflow	Baseflow	Stormflow
Inlet	95%	29%	0%	55%	5%	16%
Outlet	26%	8%	68%	50%	5%	42%

The CCME FAL guideline of 3 mg/L NO_3-N was used to evaluate guideline exceedences for TN within Kuhn Marsh. This guideline comparison is made under the assumption that NO_3 is the primary nitrogen species present in surface water samples (Voutsa *et al.*, 2001). Using this assumption, all inlet and outlet TN samples during both baseflow and stormflow exceed the guideline with one exception, an inlet stormflow sample falling just below 3 mg/L of NO_3-N.

Guideline exceedences for *E.coli* were evaluated against the Canadian Recreational Water Guideline of 200 CFU/100 mL. Evaluation was conducted on 17 baseflow samples and 37 stormflow samples. Percentages of inlet and outlet samples failing to meet this guideline are presented in Table 4-11. During baseflow, the majority of samples taken from the inlet and outlet were found to be below the guideline. During stormflow, the majority of samples taken from the inlet and the outlet were found to be above the guideline. In comparing inlet and outlet under both flow conditions, a slight increase in the number of samples exceeding the guideline at the marsh outlet is noted.

Table 4-11 Percentage of *E.coli* samples exceeding CCME recreational water quality guideline during baseflow and stormflow conditions

E.coli	**Above Guideline**	
	Baseflow	Stormflow
Inlet	24%	89%
Outlet	29%	92%

The CCME Water Quality Index (WQI) was used to quantify the number of metals exceeding specified CCME FAL guidelines for a suite of eleven common metals in 20 baseflow and 38 stormflow samples. The output of the WQI is presented in Table 4-12. All Al samples and the majority of Fe samples were found to exceed guidelines at the inlet and outlet during both flow conditions. During stormflow conditions, a large number of samples exceeded guidelines for Pb, Cd and Cu. Both As and Ag had minimal guideline exceedences, and all Ni, Se, and Ur samples were below guidelines at the inlet and outlet during both flow conditions. It is noted that for Zn, Cd and Cu, the percentage of samples exceeding guidelines were reduced between inlet and outlet during both baseflow and stormflow conditions. Based on the minimal number of guideline exceedences, As, Ag, Ni, Se and Ur were not considered to be metals of importance and were not used in evaluating concentration reduction and mass flux of contaminants.

4.3.3 Concentration Reduction

Concentration reduction within Kuhn Marsh was determined by comparing contaminant concentration values sampled at the outlet to those at the inlet. Although concentration reduction is used as a metric for determining wetland performance, dilution is a large contributing factor. Wetlands that are groundwater discharge areas, such as Kuhn Marsh, may see concentration reduction as a result of groundwater dilution. The lack of concentration reduction at the outlet of the wetland for certain parameters may be indicative of the presence of contaminants in groundwater entering the wetland or the discharge of contaminants previously stored in the wetland substrate. Contaminant concentration plots can be found in the appendix.

Table 4-12 Percentage of guideline exceedences of various metals within Kuhn Marsh during baseflow and stormflow conditions

	Inlet % Exceedence				Outlet % Exceedence		
	Stormflow	Baseflow	Total		Stormflow	Baseflow	Total
Pb	47%	10%	34%	**Pb**	45%	16%	35%
Ag	3%	0%	2%	**Ag**	0%	11%	4%
Al	100%	100%	100%	**Al**	100%	100%	100%
As	3%	0%	2%	**As**	0%	0%	0%
Cd	86%	55%	75%	**Cd**	63%	26%	51%
Cu	92%	45%	76%	**Cu**	84%	37%	68%
Fe	92%	100%	95%	**Fe**	97%	100%	98%
Ni	0%	0%	0%	**Ni**	0%	0%	0%
Se	0%	0%	0%	**Se**	0%	0%	0%
Zn	45%	10%	33%	**Zn**	0%	5%	5%
Ur	0%	0%	0%	**Ur**	0%	0%	0%

4.3.3.1 Storm Flow Conditions

Stormflow concentration reduction data is presented in Table 4-13. Concentration reduction is characterized as the percent reduction of contaminant concentration achieved at the outlet of Kuhn Marsh. Positive concentration reduction values indicate a lower concentration of contaminants at the outlet as compared to the inlet. A degree of positive concentration reduction at the outlet is achieved for all parameters at some point during stormflow conditions. Both TOC and Fe show the lowest number of reduction events with approximately 17% of the 38 individual storm samples showing a small reduction in concentration at the outlet. Both Cu and Zn show an average concentration reduction of 36% and 45% occurring at the outlet in virtually all samples, while Cd shows an average concentration reduction of 44% in three quarters of the samples.

Table 4-13 Stormflow concentration reduction values

Stormflow		**TOC**	**TSS**	***E.coli***	**TN**	**TP**
Reduction Events		6/38	16/37	17/36	19/38	19/38
% Reduction Events		16	43	47	50	50
Average % Reduction		+15	+53	+54	+17	+33
Average % Increase		-104	-142	-122	-20	-170
	Pb	**Cu**	**Fe**	**Al**	**Zn**	**Cd**
Reduction Events	17/38	36/38	7/38	24/38	38/38	24/32
% Reduction Events	45	95	18	63	100	75
Average % Reduction	+40	+36	+47	+50	+45	+44
Average % Increase	-87	-113	-184	-60	0	-146

4.3.3.2 Base Flow Conditions

Baseflow concentration reduction data is presented in Table 4-14. Minimal concentration reduction of TOC, TSS and TP occurs at the outlet, with all three parameters showing some degree of concentration reduction in only 6% of baseflow samples. Parameters showing some reduction in concentration for the majority of baseflow samples include TN, Cu, Zn and Cd.

Table 4-14 Baseflow concentration reduction values

		TOC	TSS	*E.coli*	TN	TP
Reduction Events		1/18	1/16	4/17	14/18	1/19
% Reduction Events		6	6	24	78	5
Average % Reduction		+31	+17	+59	+19	+25
Average % Increase		-216	-549	-513	-14	-475
	Pb	**Cu**	**Fe**	**Al**	**Zn**	**Cd**
Reduction Events	4/18	14/18	4/18	6/18	15/18	14/17
% Reduction Events	22	78	22	33	83	82
Average % Reduction	+17	+38	+40	+27	+42	+67
Average % Increase	-171	-41	-171	-172	-14	-391

4.3.3.3 Overall Concentration Reduction Function of Kuhn Marsh

Baseflow parameters showing an increase in percentage of concentration reduction events at the outlet as compared to stormflow are limited to Cd and TN. Both Zn and Cu show a slightly lower number of reduction events during baseflow. However, the number of reduction events during both flow regimes is considered high. All other parameters show a reduced percentage of concentration reduction events at the outlet during baseflow conditions as compared to stormflow. Lower instances concentration reduction events during baseflow may be indicative of the fact that groundwater intrusion into the wetland area during stormflow is contributing to dilution of contaminant concentrations between the inlet and outlet of Kuhn Marsh, resulting in an apparent increase in concentration reduction during stormflow conditions. High instances of concentration reduction reported for TN in baseflow and Cu, Zn and Cd in both stormflow and baseflow are most likely attributed to the effects of dilution.

Both Fe and TOC have minimal concentration reduction at the outlet during storm events. Under anaerobic conditions, Fe (III) in soil is reduced to the more soluble Fe (II) form and iron is able to move from the sediment to the water column. Discharge of soluble Fe is common in wetlands as saturated soil conditions provide ideal anaerobic conditions for release of Fe from sediments (Tammi, 2000). Wetlands are known sinks of C in the form of organic matter, which may be discharged in the form of dissolved or particulate C. Minimal concentration reduction of

Fe and TOC despite groundwater dilution may be attributed to the mass diffusion of both substances from the wetland during stormflow events.

4.3.4 Mass flux

Mass flux calculations within Kuhn Marsh were completed by comparing masses of contaminants at the outlet to those at the inlet using measured flow and concentration data. Mass flux data provides contaminant removal estimates that account for flow and are an important metric to determine quantity of contaminants moving through a wetland system. Where concentration reduction values provide valuable information from a guideline exceedence or dilution perspective, mass flux calculations provide valuable information concerning degree of contaminant removal within a wetland.

4.3.4.1 Storm Flow Conditions

Mass flux of contaminants between the inlet and outlet of Kuhn Marsh was calculated on a per storm basis using an average of all individual storm data points. Percent mass reduction data for each storm event is presented in Table 4-15. Positive values indicate mass removal of contaminants within the wetland area, whereas negative values indicate an increase in mass of contaminants at the wetland outlet. Mass reduction of TSS, Cu, Al, Zn and Cd was achieved at the outlet during a few individual storm events; however, the percentages of mass reduction are small during these events. The parameter with the highest mass reduction at the outlet is TSS, which has mass reductions of 17, 47 and 60% for three of nine storm events.

Mass flux and percent reduction values are presented on a parameter-basis in Table 4-16. These values are calculated as a sum of the average of all recorded storm events and are considered an estimate of the mass of contaminants discharging from the wetland during stormflow on an annual basis. Kuhn Marsh was found to be a source of all contaminants measured. Parameters having the highest percentages of mass flux out of the wetland include TOC, *E.coli*, and Fe followed by TN, TP and Cd. Parameters having lower percentages of mass flux out of the wetland include TSS, Al and Zn.

Table 4-15 Percent mass reduction of contaminants on a per storm basis within Kuhn Marsh

	TOC	**TSS**	***E.coli***	**TN**	**TP**	**Pb**	**Cu**	**Fe**	**Al**	**Zn**	**Cd**
Storm A June 13 2011	-168	-85	-63	-69	-47	-18	-39	-164	-29	-5	**+27**
Storm B July 31 2011	-192	**+59**	-4714	-119	-239	-47	**+10**	-586	**+4**	**+17**	-94
Storm C Aug 9 2011	-440	**+17**	-303	-190	-188	-19	-78	-158	-23	-113	-167
Storm D Oct 3 2011	-721	-53	-2	-263	-262	-115	-78	-688	-26	-81	-204
Storm E Oct 5 2011	-310	-273	-295	-212	-190	-226	-145	-442	-156	-72	-143
Storm G June 26 2012	-564	-354	-65	-406	-155	-356	-115	-934	-155	-160	-289
Storm H July 24 2012	-835	-540	-203	-323	-244	-318	-88	-1187	-142	-55	-239
Storm I Sep 5 2012	-557	-195	-345	-377	-374	-389	-221	-525	-184	-186	-414
Storm J Sep 10 2012	-472	**+47**	-642	-374	-160	-40	-122	-139	**+11**	-82	-665

Table 4-16 Annual stormflow mass flux and percent mass reduction of contaminants based on average stormflow values

		TOC (Kg)	TSS (Kg)	*E.coli* (10^9 CFU)	TN (Kg)	TP (Kg)
Inlet		114.8	476.3	17.9	185.5	2.7
Outlet		503.3	703.5	87.1	529.4	6.8
% Reduction		-338	-48	-386	-185	-153
	Pb (Kg)	**Cu (Kg)**	**Fe (Kg)**	**Al (Kg)**	**Zn (Kg)**	**Cd (g)**
Inlet	0.08	0.3	32.9	13.9	1.0	1.4
Outlet	0.15	0.5	147.7	20.4	1.6	4.3
% Reduction	-84	-82	-349	-47	-58	-202

4.3.4.2 Base Flow Conditions

Annual baseflow mass flux and percent mass reduction values are presented on a parameter-basis in Table 4-17. Baseflow averages for each parameter were used to calculate mass reduction of contaminants. Statistics pertaining to these averages can be found in the appendix. During baseflow, Kuhn Marsh is considered a source of contaminants for all parameters sampled. Both TSS and TP have the highest percentages of mass flux from the wetland during baseflow, followed by TOC and *E.coli*. Both Zn and Cd were found to have the lowest percentages of mass flux from the wetland during baseflow.

4.3.4.3 Overall Mass Reduction Function of Kuhn Marsh

Based on mass reduction data, Kuhn Marsh is found overall to be a source of all contaminants sampled during both stormflow and baseflow conditions. Larger masses of contaminants are being discharged from the wetland on an annual basis during baseflow and percent mass reduction between inlet and outlet is further decreased as compared to stormflow conditions. Specifically, percent mass reduction of sediment-associated parameters, such as TOC, TSS, and TP, is greatly reduced and Kuhn Marsh becomes a greater source of these parameters during baseflow. Exceptionally, percent mass reduction of Cd is improved during baseflow conditions and percent mass discharge of Fe and TN are relatively consistent for both flow conditions.

Table 4-17 Annual baseflow mass flux and percent mass reduction of contaminants based on average baseflow values

		TOC (Kg)	**TSS (Kg)**	***E.coli*** **(10^9 CFU)**	**TN (Kg)**	**TP (Kg)**
Inlet		168.8	88.0	13.7	399.0	1.2
Outlet		1127.9	1830.5	92.9	1082.4	9.7
% Reduction		-568	-1980	-579	-171	-725
	Pb (Kg)	**Cu (Kg)**	**Fe (Kg)**	**Al (Kg)**	**Zn (Kg)**	**Cd (g)**
Inlet	0.06	0.2	118.0	10.9	1.1	2.4
Outlet	0.25	0.5	472.0	25.1	2.3	5.0
% Reduction	-310	-147	-300	-130	-112	-109

Several studies have reported treatment wetlands as sources of contaminants. Scholes *et al.* (1998) noted a discharge of Pb and Cu at the outlet of their study wetland during baseflow conditions and Birch *et al.* (2004) found the study wetland in question to be a source of Fe and Mn during all storm events sampled, and a source of TSS during extreme storm events. However, no study has reported a wetland to be a large source of all contaminants sampled. The consistent discharge of Fe and TN from the wetland considered herein may be due to the constant release of Fe from aerobic sediment and the fact that the majority of the wetland is covered by *Myrica gale* which has the ability to continually fix atmospheric N in soil. Factors influencing the poor function of Kuhn Marsh as a treatment wetland may include groundwater influence, chemical composition of wetland sediment and, perhaps most importantly, age of wetland. Groundwater influence is discussed further in Section 4.4.

Adsorption and immobilization of metals depends largely on the chemical composition of wetland sediments and factors such as pH may cause the release of metals into the water column. Elliott *et al.* (1986) ranked Pb, Cu, Zn and Cd based on their affinity for soil adsorption, from high affinity to low affinity. In mineral soils, metal adsorptivity is ranked Pb>Cu>Zn>Cd, whereas for organic soils, the ranking is Pb>Cu>Cd>Zn. This suggests that Pb and Cu are more likely to be associated with the release of sediment in a wetland, whereas Cd and Zn are more likely to be released in dissolved form. According to USEPA (1992), peak retention of cationic metals generally occurs at a pH>7, where peak retention of anionic metals generally occurs at a pH<7. Retention is largely based on the availability of, and competition for, charged adsorption sites. As reported by USEPA (1992), most studies concerning metal retention are performed in a controlled laboratory setting and do not take into account the complexity of natural interactions. The precipitation, oxidation and adsorption of metals are complex processes and largely depend on local conditions and inputs.

It is important to note that regardless of the mechanism of pollutant release, Kuhn Marsh is noted as a huge source of contamination. Based on the extent of mass discharge from Kuhn Marsh, age of the wetland may be the most important factor in poor wetland performance. After receiving stormwater discharge for decades, the wetland treatment capacity may now be reduced to the point where contaminants are being released from saturated sediments and is likely further compounded by the scouring of contaminated sediments during intense flow events. While minimal studies exist on the lifespan and function of stormwater treatment wetlands over the long-term, Fink and Mitsch (2004) noted a 30% drop in P retention within their newly constructed study wetland over a 2 year period and White *et al.* (2000) noted greatly reduced P adsorption capacity of wetland sediments in as little as five years after construction. Based on plant uptake and saturation, Weis and Weis (2004) suggest that marshes may become sources of metals over the long term when metals sequestered in plant matter are released back into the environment. Kraus (1987) also cautioned on the use of the word ‘sink’ to describe contaminant retention in wetlands and suggested ‘reservoir’ to be a more appropriate descriptor of actual wetland function.

4.3.5 Areal Contaminant Loading Rates

Areal loading rates of contaminants based on average storm event values are presented in Table 4-18. Values calculated for inlet and outlet are based on the respective contributing watershed areas. Inferred loading rates are greater at the outlet of Kuhn Marsh in all instances. Load increase values provide an estimate of the areal loading rate increase generated by the wetland area between the inlet and outlet of the marsh.

Table 4-18 Areal loading for average storm event

		TOC (Kg/ha)	**TSS (Kg/ha)**	**E.coli (10^9 CFU/ha)**	**TN (Kg/ha)**	**TP (Kg/ha)**
Inlet		4.0	16.8	0.6	6.5	0.09
Outlet		13.6	19.0	2.3	14.3	0.18
Load Increase		9.5	2.2	1.7	7.7	0.09
	Pb (Kg/ha)	**Cu (Kg/ha)**	**Fe (Kg/ha)**	**Al (Kg/ha)**	**Zn (Kg/ha)**	**Cd (g/ha)**
Inlet	0.0029	0.009	1.2	0.49	0.036	0.050
Outlet	0.0041	0.012	4.0	0.55	0.044	0.116
Load Increase	0.0012	0.003	2.8	0.06	0.008	0.066

Areal loading rates for Kuhn Marsh during baseflow conditions are presented in Table 4-19. All parameters show a load increase between inlet and outlet and, with the exception of Cd, load increases calculated during baseflow are greater than those determined for stormflow conditions. When comparing baseflow and stormflow areal loading rates for the inlet, increased loading rates for TSS, TP, Pb, Cu and Al occur during stormflow; whereas higher loading rates for TOC, TN, Fe *E.coli* and Cd occur during baseflow.

Table 4-19 Areal loading for average baseflow on a yearly basis

		TOC (Kg/ha)	TSS (Kg/ha)	*E.coli* (10^9 CFU/ha)	TN (Kg/ha)	TP (Kg/ha)
Inlet		5.6	2.9	0.5	13.3	0.04
Outlet		28.7	46.6	2.4	27.6	0.25
Load Increase		23.1	43.7	1.9	14.3	0.21
	Pb (Kg/ha)	**Cu (Kg/ha)**	**Fe (Kg/ha)**	**Al (Kg/ha)**	**Zn (Kg/ha)**	**Cd (g/ha)**
Inlet	0.0020	0.007	3.9	0.36	0.036	0.085
Outlet	0.0064	0.014	12.0	0.64	0.059	0.135
Load Increase	0.0043	0.006	8.1	0.28	0.023	0.050

Annual areal loading rates taken from literature values for various watersheds are presented in Table 4-20. Based on comparison with areal loading rates in the Kuhn Marsh watershed, all loading rates to and from the wetland are below literature values for both flow conditions.

4.3.6 Continuous Monitoring

Continuous monitoring of turbidity, conductivity, DO and temperature was completed in Kuhn Marsh for storm events starting on August 15th 2011, October 4th 2011 and April 23rd 2012. Monitoring was conducted on ten minute time intervals over a 2-3 day period and the results are presented with corresponding hydrographs in Figures 4-4, 5, 6. Temperature and DO levels remain relatively stable at the inlet of Kuhn Marsh while both fluctuate at the outlet on

Table 4-20 Annual areal pollutant loading rates from representative watersheds

Reference	Watershed	TN (Kg/ha)	TP (Kg/ha)	TSS (Kg/ha)	*E.coli* (10^9 CFU/ha)	Pb (Kg/ha)	Zn (Kg/ha)	Cu (g/ha)
Herrmann (2012)	Residential	332	31.5	3514	--	--	--	--
Sinclair *et al.* (2009)	Agricultural	--	--	--	16	--	--	--
Horner *et al.* (1994)	HDR	6.95	1.12	470.76	--	0.90	0.78	30
Horner *et al.* (1994)	MDR	4.37	0.56	212.96	--	0.22	0.22	160
Reinelt and Horner (1995)	Urban	--	0.63	107	42	--	0.43	--

what appears to be a diurnal basis. Slight increases in DO of approximately 1 mg/L at the inlet of the marsh during storm onset may be attributed to the increase in turbulence of flow discharging from the stormwater system at the inlet. Slight increases in temperature at the inlet during storm onset may be attributed to the inflow of warmer water from impermeable surfaces in the urban drainage watershed. Storm events on both April 23rd and August 15th show reduction in conductivity at the inlet upon onset of the events; however, the storm event monitored on October 4th shows a marked increase in conductivity at the onset of the event. Variability in stormwater quality may be an important factor in the behavior of conductivity at the inlet of the wetland. Build-up of ionic compounds on impermeable surfaces in the drainage watershed may cause high conductivities in the surface runoff during a storm event and may increase the conductivity at the inlet of the wetland. Alternatively, if stormwater entering the wetland has low conductivity the influx may dilute the concentration of ions at the inlet causing a drop in measured conductivity. Turbidity ranges fluctuate greatly during each monitoring event. The storm event on April 23rd saw peaks in turbidity of 1200 NTU at the inlet, whereas the events on August 15th and October 4th saw peaks at the inlet of 100 and 40 NTU, respectively. The marked increase in turbidity during the April 23rd event may be associated with spring runoff heavily laden with inorganic material, such as road sand, accumulated over the winter months. With the exception of the low-turbidity event on October 4th, turbidity is higher at the inlet during stormflow. This indicates a dampening of turbidity as flow moves from inlet to outlet and may be an indication that turbidity is not directly related to contaminant fluxes as the wetland was generally found to be a source of contaminants at the outlet. As continuous monitoring is an autonomous data collection method, potential errors may occur; as illustrated in the turbidity plot for the storm event on August 15th. Generally, turbidity responses at both inlet and outlet are highly correlated with flow. Both inlet and outlet monitoring on August 15th show large turbidity spikes that do not correspond with flow entering or exiting the wetland. This may be more indicative of interference in the vicinity of the probe than an indication of water quality and highlights the need for data examination to remove potential outliers.

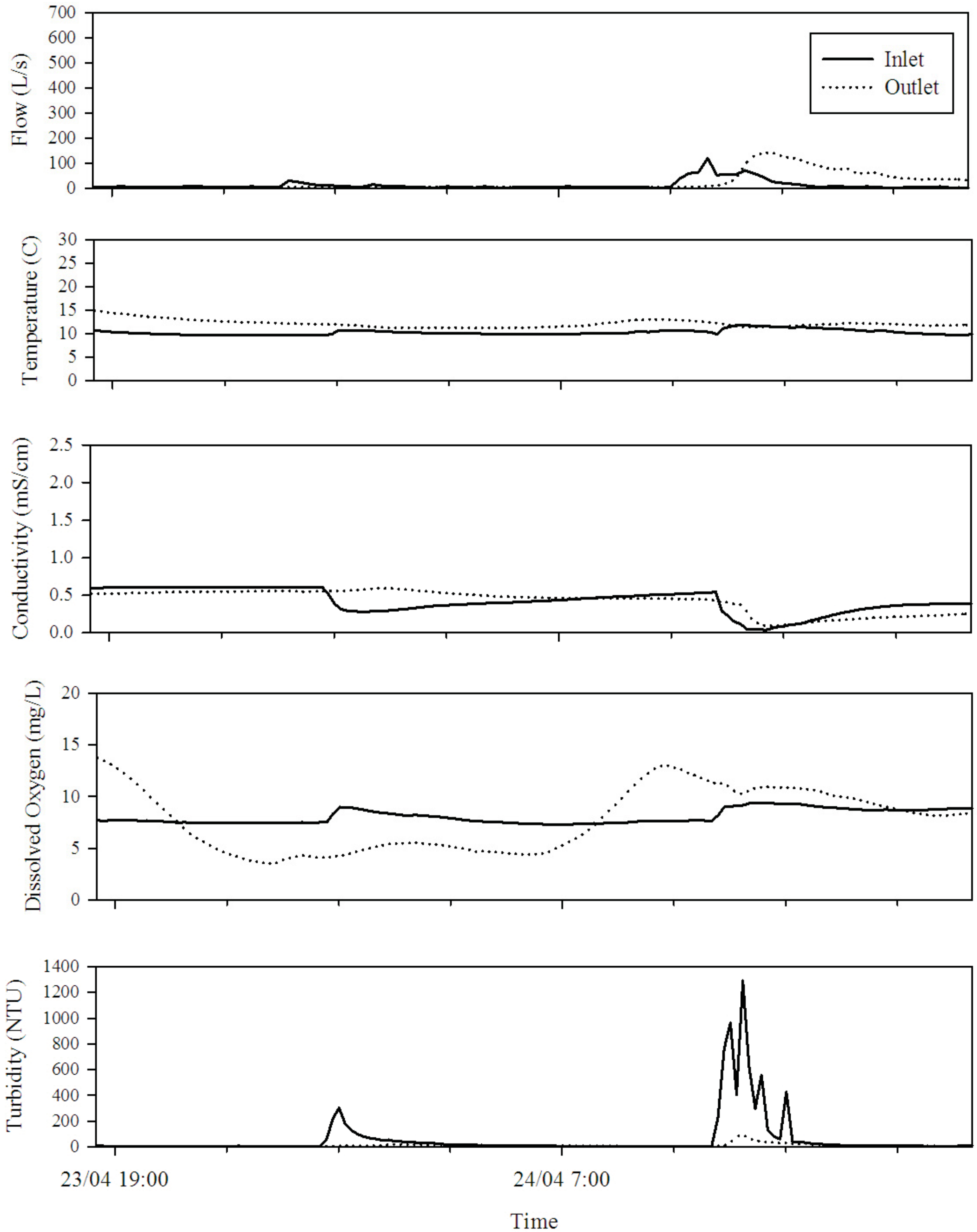

Figure 4-5 Continuous monitoring time series plots for April 23rd 2012

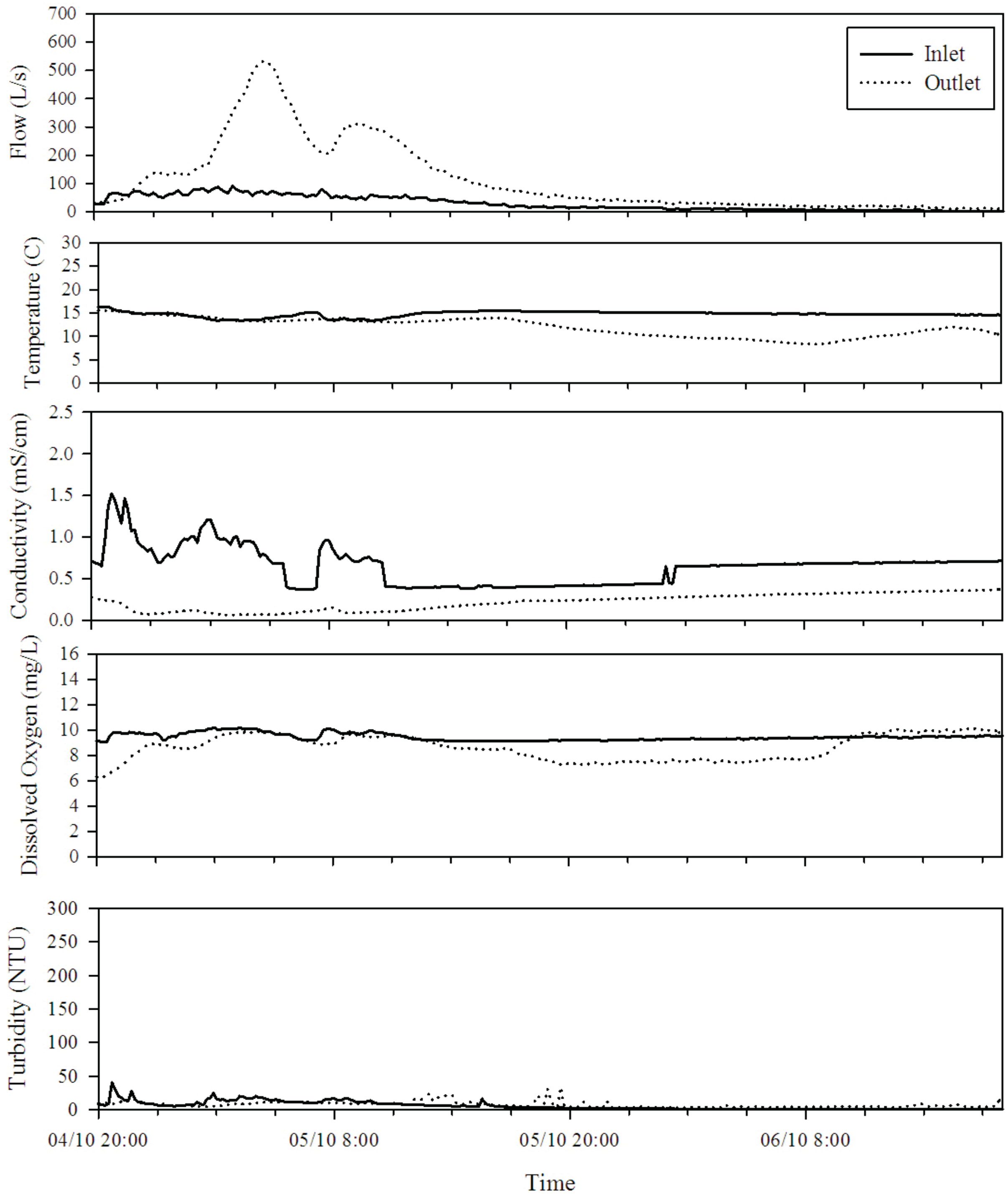

Figure 4-6 Continuous monitoring time series plots for October 4th 2011

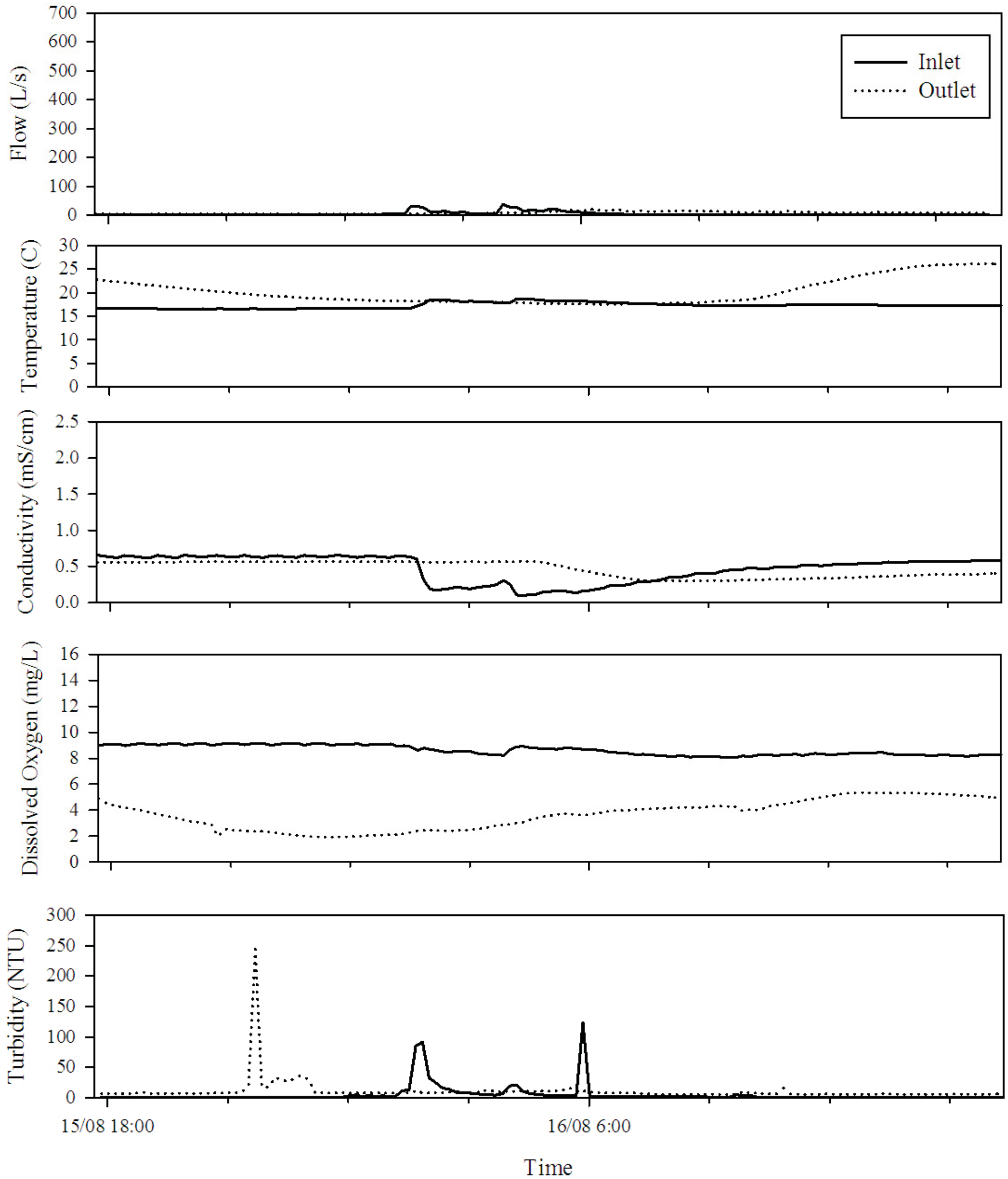

Figure 4-7 Continuous monitoring time series plots for August 15th 2011

4.4 Groundwater Interference

Based on information gained from both the water balance and storm hydrographs, it is apparent that groundwater has a large influence on the hydrology within Kuhn Marsh. With approximately 50% of flow discharging from the outlet control structure attributed to groundwater, there is potential for contaminant influx into the wetland via groundwater inflow. Constituents present in groundwater can vary depending on local geology and land use. According to NSE, common naturally occurring constituents within Nova Scotia groundwater include As, Fe, Mn, Ur, hardness and chlorides (Nova Scotia Environment, 2012). Improper chemical disposal, improper sewage management, accidental spills and agricultural areas can all contribute contaminants such as pesticides, nutrients, bacteria and hydrocarbons to local groundwater. The majority of groundwater contaminant studies in wetlands focus on nutrient import via groundwater inflow. Raisin *et al*. (1999) found their 2 ha study wetland to be largely fed by groundwater and sourced 50% of TN and TP exports from the wetland to groundwater. Reinelt and Horner (1995) also found groundwater to be a source of phosphorous transport into their study wetland and was the primary factor in considering the wetland a source of phosphorous export.

Shallow groundwater quality sampling was conducted on four separate occasions; all completed within several days of a storm event with the exception of the August sample run. Low well volume yields (<150 mL) from the majority of the wells limited the scope of water quality analysis. Based on increases in *E.coli* mass flux between inlet and outlet of the wetland and the presence of on-site sewage systems bordering the west side of the wetland watershed, groundwater bacterial analysis was made priority. Figure 4-7 represents the results of the groundwater *E.coli* sampling that took place in the marsh between July and September 2012.

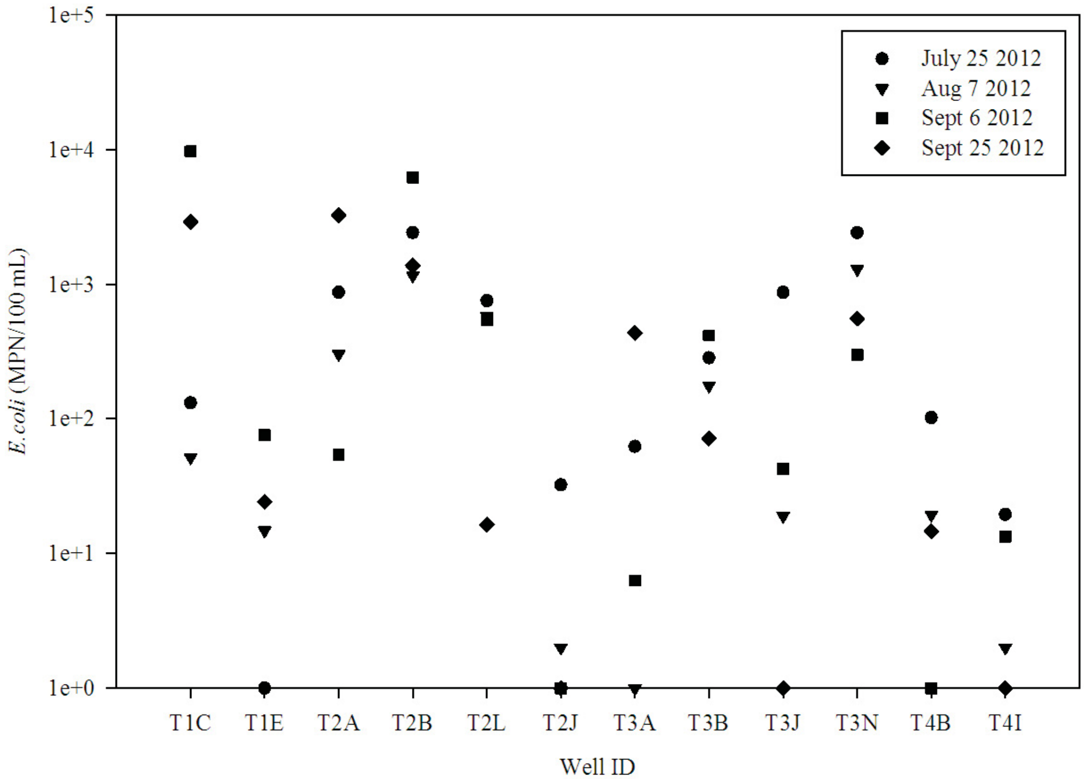

Figure 4-8 Enumeration of *E.coli* in groundwater wells

Microbial source tracking was performed on samples from each well, as well as samples from the inlet and outlet of the wetland, to determine if fecal waste of human origin was entering the wetland. DNA was extracted from *bacteroidales* in groundwater and surface water samples after the September 25th sample run. Although several groundwater wells show high *E.coli* levels, the outlet and all wells had negative indications of human-source bacterial contamination. The inlet sample had approximately 800 gene copy numbers (GCN) per 100 mL of sample. According to Sauer *et al.* (2011), <1000 GCN/100 mL is considered in the low range for human bacteroidales within stormwater samples. This confirms the presence of human fecal contamination at the inlet of Kuhn Marsh and may be indicative of sewer cross-connections in the stormwater system servicing the urban drainage watershed. While bacterial intrusion is not as common in deep groundwater wells, shallow wells that are under direct influence of surface water do have higher instances of bacterial contamination; however, several wells in Kuhn Marsh are showing *E.coli* levels in the range of 10^3-10^4 MPN/100mL, which indicates a high level of bacteria present in the wells. It is noted that negative indications of human source bacterial contamination in the groundwater wells may be due to the fact that travel time from the septic field could exceed the decay rate of the human *bacteroidales* markers. Tambalo *et al.* (2012) reported that a 2-log reduction in human *bacteroidales* markers could occur in as little as 1.5 days, with a 2-log reduction in *E.coli* occurring after 6 or more days. Alternatively, due to the lack of human *bacteroidales* markers, *E. coli* may have originated from a non-human fecal source, such as local wildlife or domestic animals, or may be indicative of a naturalized bacterial population within the wetland soil. According to Perchec-Merien and Lewis (2012), a naturalized *E.coli* population can evolve over time in the absence of a host and is no longer considered directly derived from fecal sources; meaning a naturalized strain of bacteria may not contain the same genetic markers as those directly associated with a host. The fact that well T3N is showing high bacteria counts despite being located on the east side of the wetland, which is not in the vicinity of the on-site sewage systems, may indicate that bacterial contamination in this well is of non-human origin which infiltrated into the shallow well from surface contamination or is from a naturalized bacteria population.

5.0 CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE WORK

5.1 Conclusions

The physical, hydrologic and water quality characterization of Kuhn Marsh took place over a seventeen month period from May 2011 to October 2012 in an attempt to assess the function of the natural, urban wetland in the treatment of stormwater. Based on the study, the following conclusions have been determined.

5.1.1 Hydrologic Function

1. Kuhn Marsh is largely influenced by groundwater discharge into the wetland. Based on the input water balance, groundwater accounts for approximately 50% of flow discharging through the outlet control structure during stormflow. Through the monitoring of groundwater wells in the wetland watershed, it was determined that horizontal groundwater movement into the wetland is minimal and discharge is most likely due to the vertical upwelling of groundwater.

2. As depicted in hydrographs plotted for the wetland, peak flow dampening and volume reduction does not occur during storm events. Larger peak flows and discharge volumes were reported at the outlet during all monitored storm events.

3. Hydraulic retention times within Kuhn Marsh are low, with calculated stormflow retention times of 2 and 4 hours during two separate events and a calculated retention time of 11 hours during baseflow conditions. Both stormflow and baseflow retention times are well below the minimum recommended retention time of 18 hours required for settling of suspended sediment.

4. Volumetric efficiency of Kuhn Marsh is low, with approximately 20% of the wetland area being used to treat stormwater. Low hydraulic retention times and poor volumetric efficiency are both indications that flow is short-circuiting through the wetland.

5.1.2 Treatment Capacity

1. The majority of stormflow samples analyzed exceeded Canadian guidelines at the outlet of Kuhn Marsh for all parameters sampled. Not only are treatment processes in Kuhn Marsh ineffective in reducing contaminant concentrations under guideline levels prior to discharge from

the wetland, in some cases, parameter concentration increased between inlet and outlet to the point of exceeding guideline levels at the outlet of the marsh.

2. Concentration reduction between inlet and outlet was reported for TN during baseflow conditions and Cd, Cu, Zn during both baseflow and stormflow conditions. Lower concentration reduction rates were found for all other parameters sampled, with TOC and Fe showing the lowest number of reduction events in both baseflow and stormflow. Positive concentration reduction was attributed to dilution based on the large quantity of groundwater discharge reportedly entering the wetland downstream of the inlet. Minimal concentration reduction despite the effects of groundwater dilution may be an indication of contaminant influx from groundwater entering the marsh, or contaminant discharge into the water column from wetland sediments.

3. Kuhn Marsh was found to be a source of all contaminants sampled during both baseflow and stormflow conditions, with mass flux calculations indicating the mass of contaminants exiting the wetland was greater than the mass of contaminants entering the wetland in all conditions. Potential contaminant influx from groundwater and the age of the wetland may be contributing factors in Kuhn Marsh being considered a source of contaminants. With the wetland having received stormwater inflow for decades, scouring of saturated wetland sediments or the diffusion of contaminants into the water column from saturated sediments may play a role in the mass discharge of contaminants from the wetland.

5.2 Recommendations for Future Work

1. Complete groundwater quality characterization should be executed in wetlands that are heavily influenced by groundwater discharge to properly assess contaminant flux and concentration reduction within the wetland. Depending on local groundwater quality, treatment wetlands receiving groundwater discharge may be considered sources of contaminants based on constituents entering the wetland via groundwater discharge that are not accounted for in contaminant reduction calculations.

2. Water quality parameters should be selected with consideration to the convenience of comparison between existing stormwater treatment wetland studies. With a limited body of research in existence on the use of wetlands in treating stormwater, comparison between studies is only possible when similar sets of parameters are used. A recommended minimum parameter suite should include TSS, TOC, TN, TP, *E.coli* and Fe, Cu, Cd, Zn and Pb.

3. Water quality monitoring should be executed on a finer scale when possible to increase the degree of precision when calculating contaminant reduction within the wetland. When feasible, the use of automatic sampling equipment is recommended.

4. Changes in the contaminant reduction capacity of stormwater treatment wetlands over the long term have not been well documented. Comprehensive studies on the fate and transport of stormwater contaminants in aging wetlands are necessary to fully assess the viability of using wetlands for long-term stormwater treatment.

5. Treatment optimization of Kuhn Marsh may include the dredging of contaminated sediments to recharge treatment capacity, construction of forebay to dissipate inflow velocities and aid in the settling of sediments, installation of baffle system to increase hydraulic retention time and volumetric efficiency, and the creation of a ponded area at the outlet to resettle any re-suspended sediments before discharge downstream.

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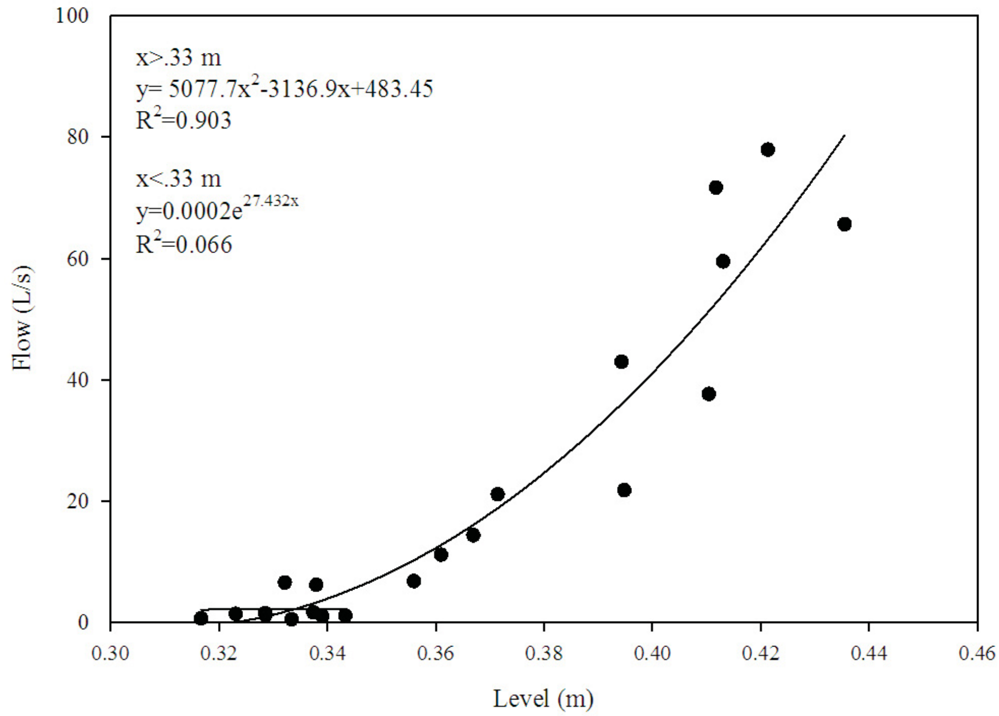
APPENDIX



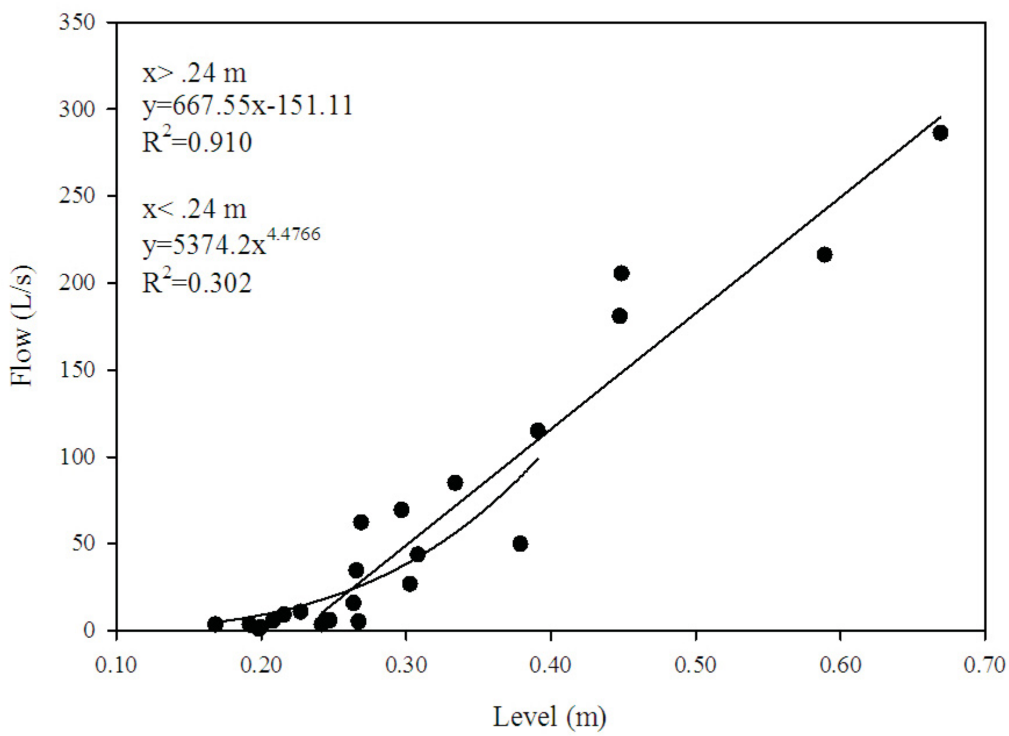


Figure A-1 Stage-discharge rating curves for a) inlet and b) outlet

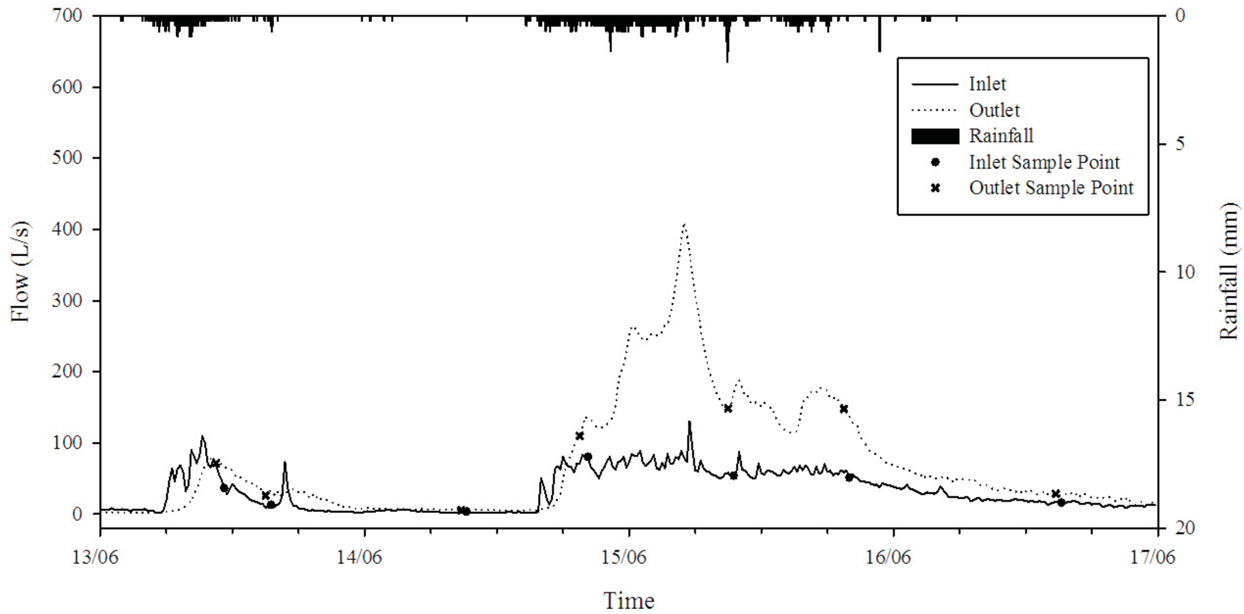


Figure A-2 Storm A Hydrograph: June 13-16 2011

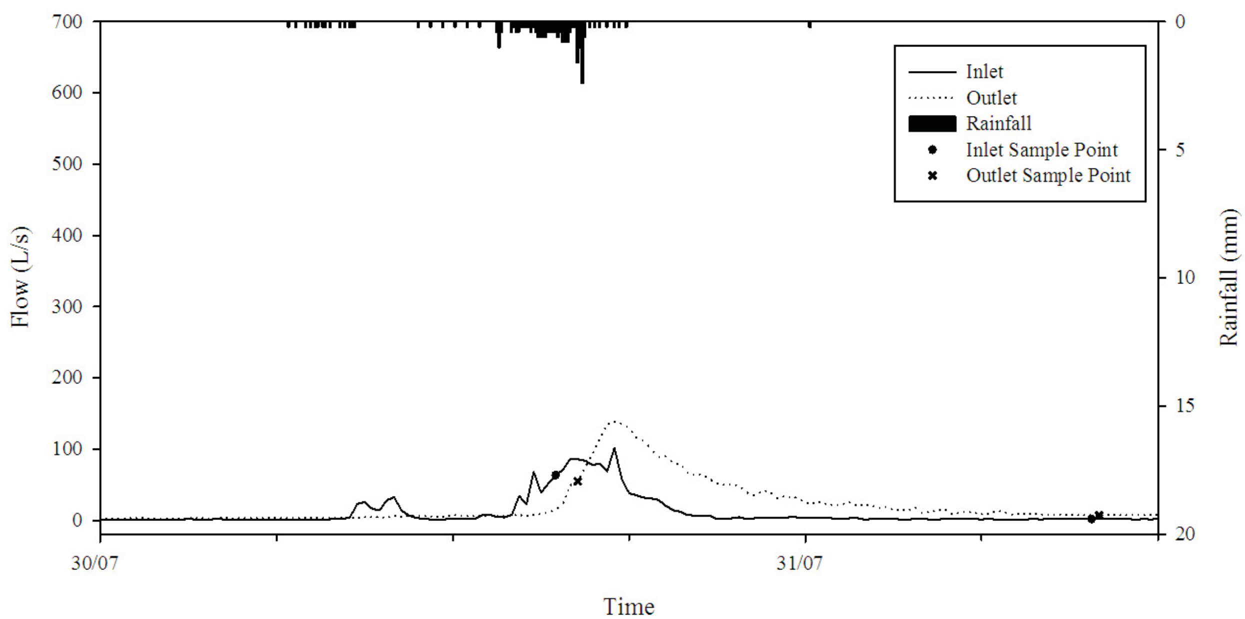


Figure A-3 Storm B Hydrograph: July 30-31 2011

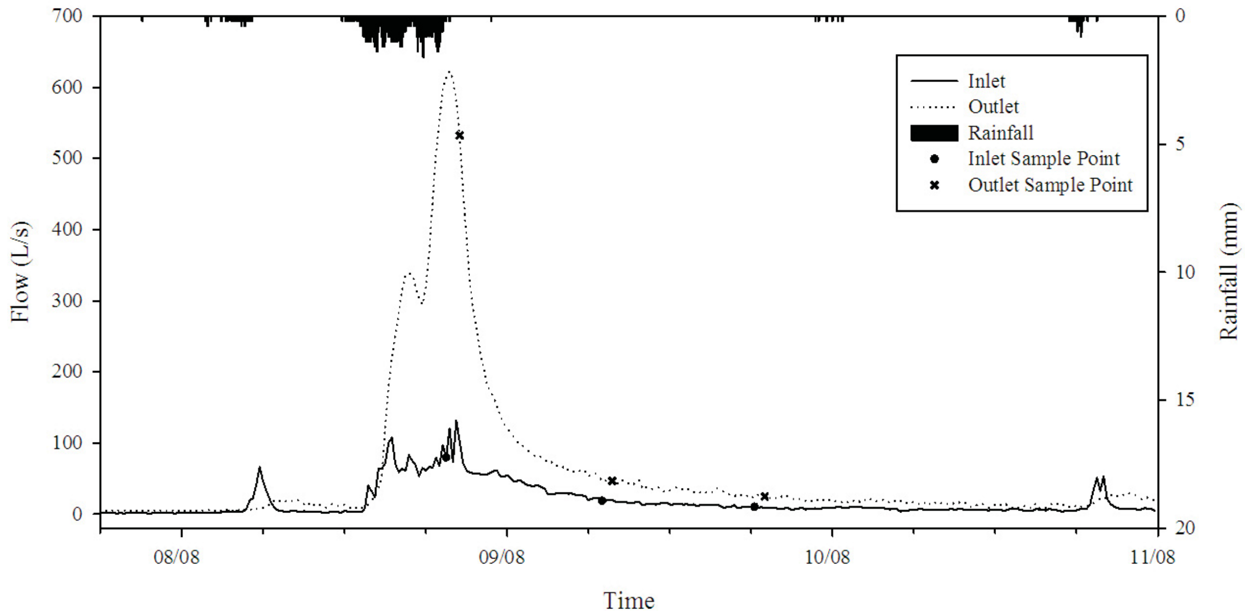


Figure A-4 Storm C Hydrograph: August 8-10 2011

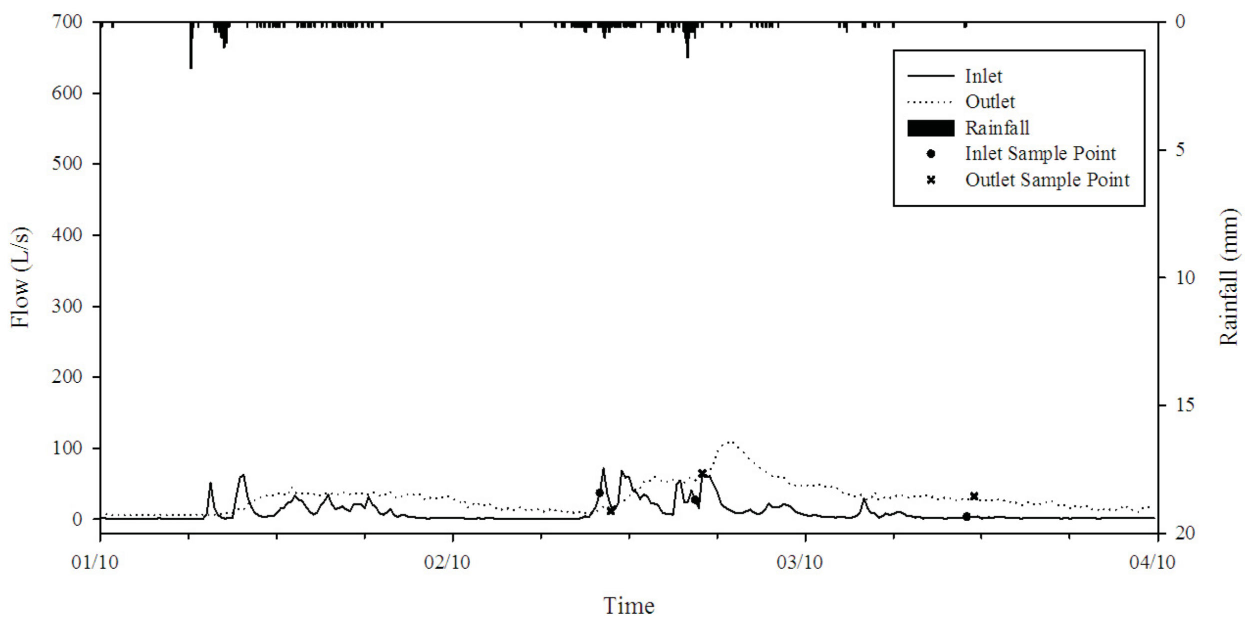


Figure A-5 Storm D Hydrograph: Oct 1-4 2011

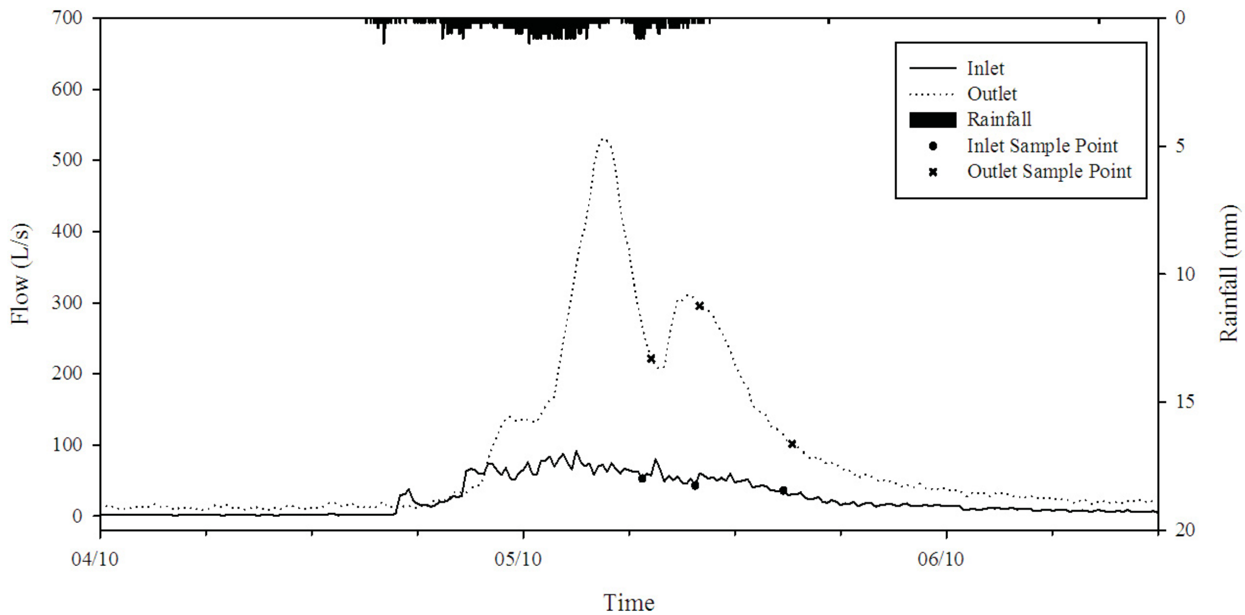


Figure A-6 Storm E Hydrograph: Oct 4-6 2011

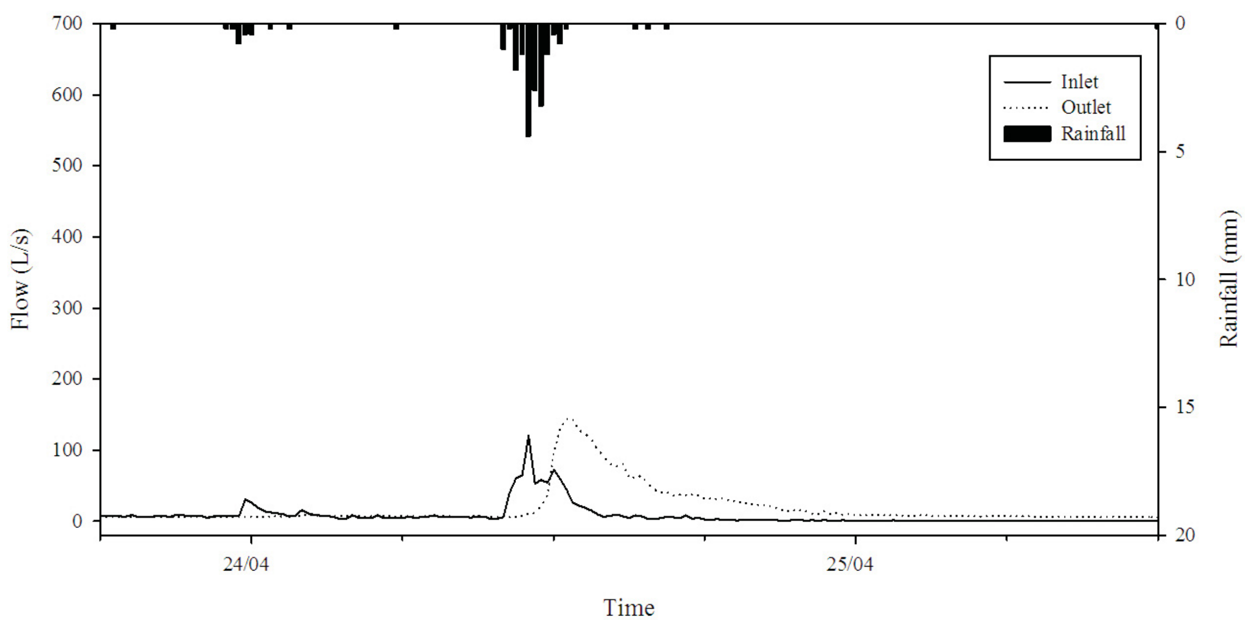


Figure A-7 Storm F Hydrograph: April 24-25 2012

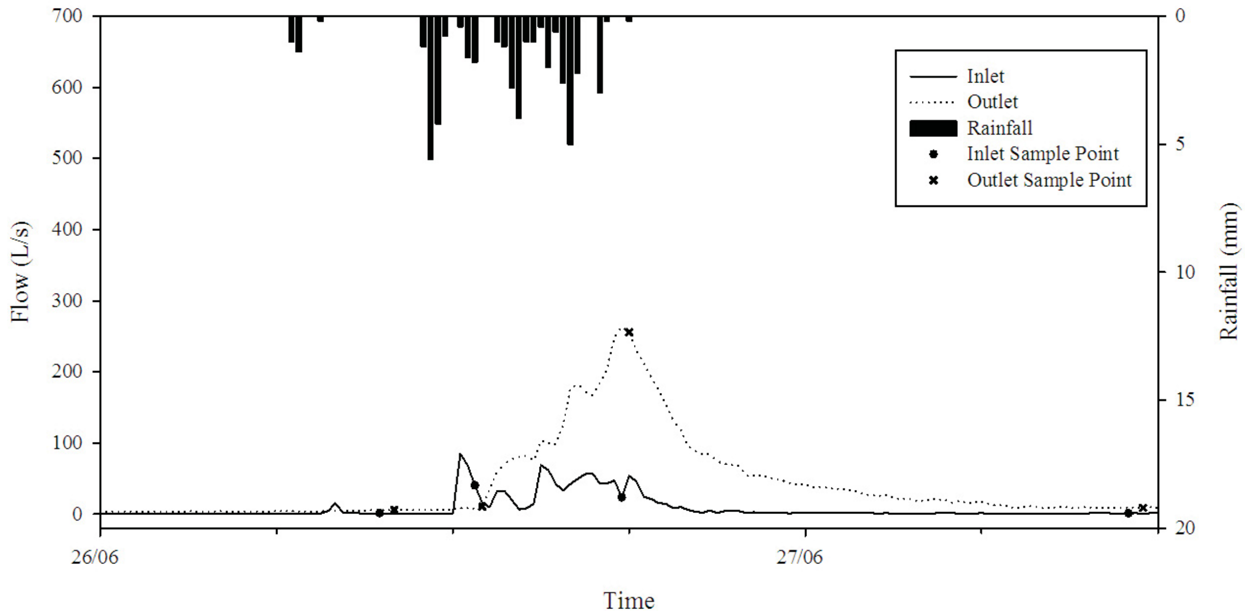


Figure A-8 Storm G Hydrograph: June 26-27 2012

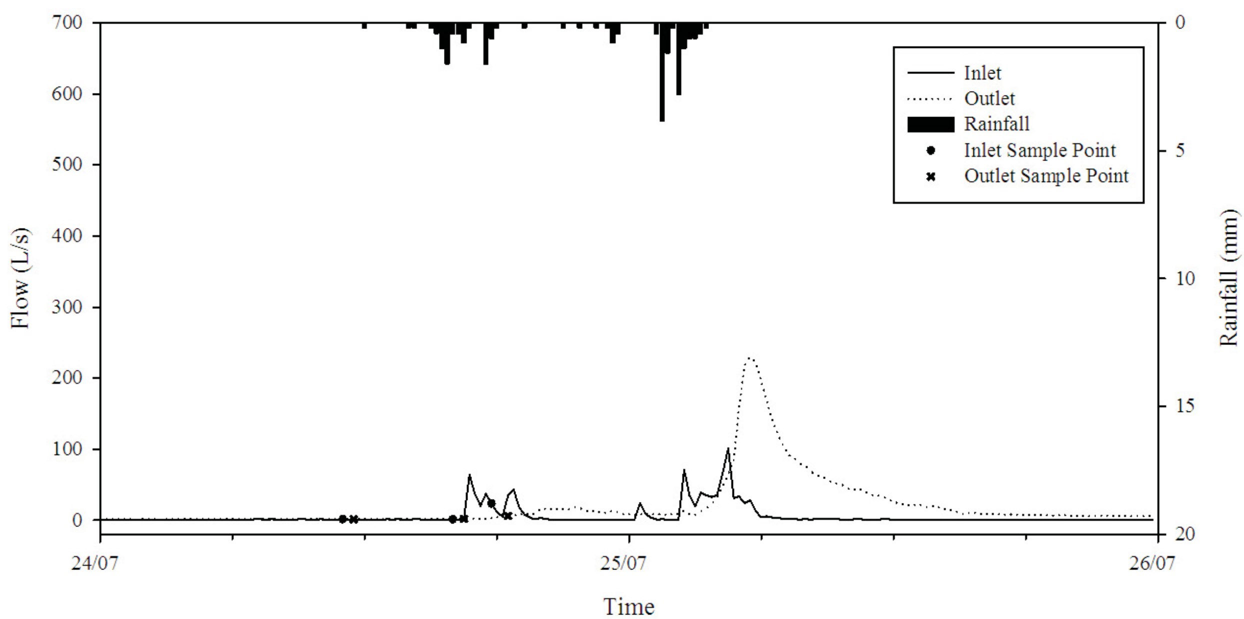


Figure A-9 Storm H Hydrograph: July 24-26 2012

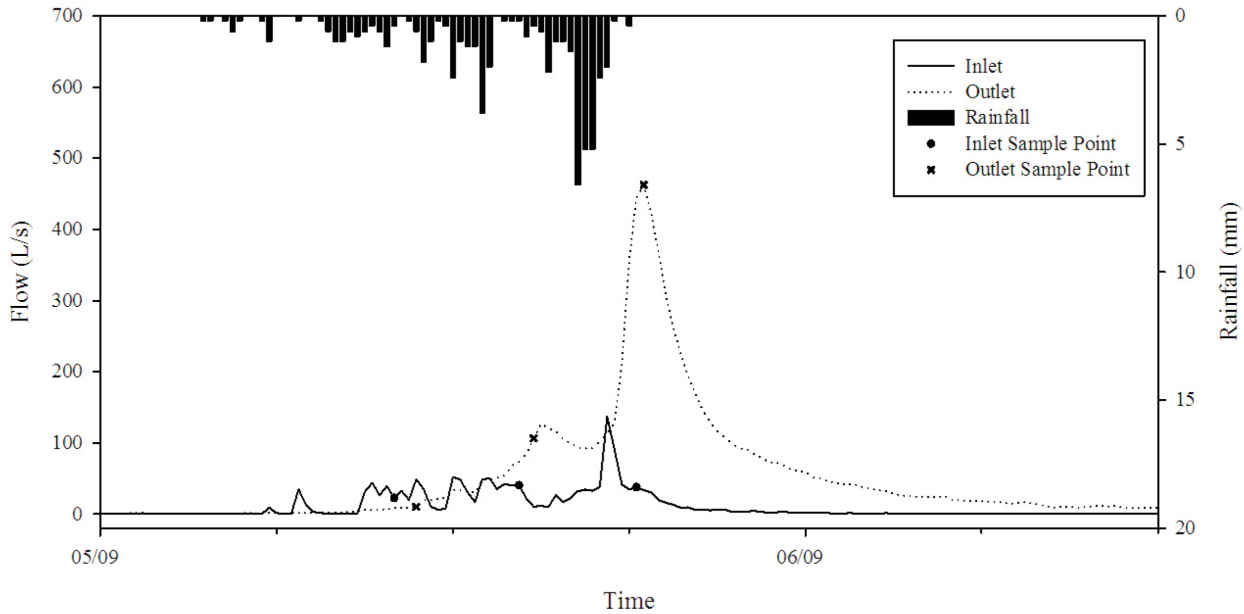


Figure A-10 Storm I Hydrograph: September 5-6 2012

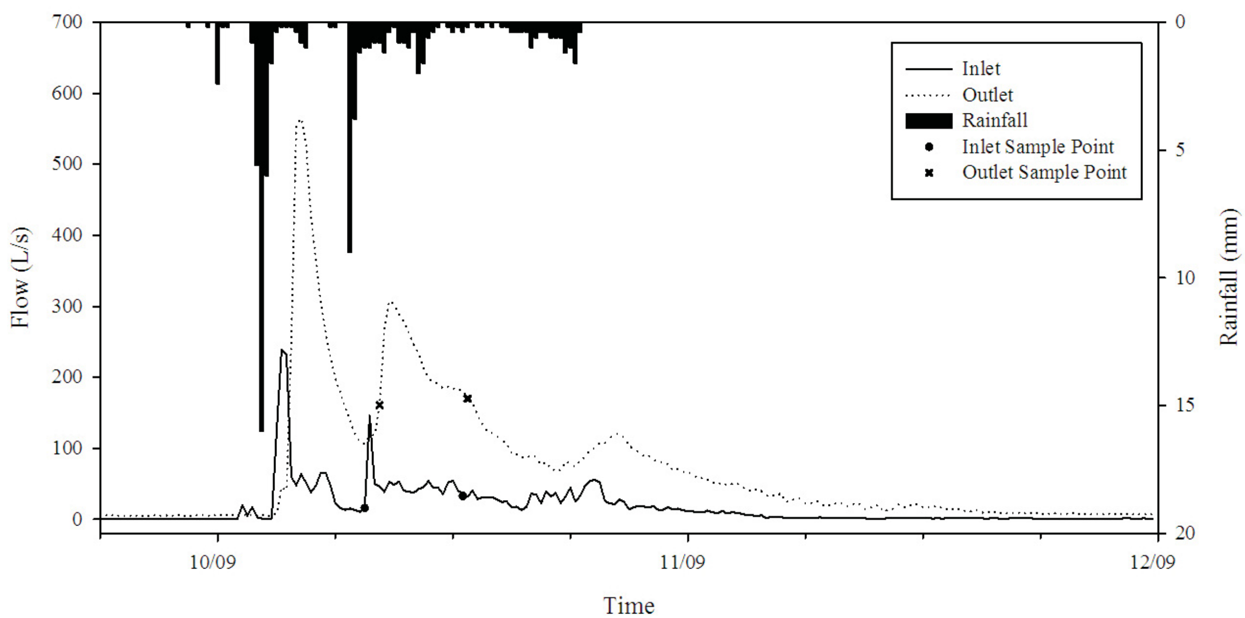


Figure A-11 Storm J Hydrograph: September 9-12 2012

Table A-1 Inlet baseflow parameter statistics

Inlet Baseflow					95% Confidence Interval	
Variable	**N**	**μ**	**σ**	**Maximum**	**Low**	**High**
Level (m)	14	0.32	0.01	0.34	0.32	0.33
Flow (L/s)	14	1.97	1.35	4.97	1.19	2.75
TOC (mg/L)	18	2.72	2.68	10.52	1.39	4.05
Turbidity (NTU)	15	3.54	2.91	9.99	1.92	5.15
TSS (mg/L)	16	1.42	1.22	4.38	0.77	2.07
***E.coli* (CFU)**	17	60.4	402	1350	13	427
TN (mg/L)	18	6.43	0.92	7.92	5.97	6.88
TP (mg/L)	19	0.019	0.035	0.163	0.002	0.036
DO (mg/L)	16	8.36	2.72	12.81	6.91	9.81
Conductivity (mS/cm)	16	0.73	0.42	1.77	0.51	0.95
pH	16	6.53	0.31	6.93	6.36	6.69
Hardness (mg/L)	20	110.6	29.0	203.6	97.0	124.2
Ag (ug/L)	8	0.04	0.02	0.08	0.02	0.06
Al (ug/L)	20	176	611	2764	-109	462
Cd (ug/L)	20	0.04	0.03	0.09	0.03	0.05
Cu (ug/L)	20	3.50	3.81	18.64	1.72	5.28
Ni (ug/L)	20	3.81	1.77	8.17	2.98	4.64
Pb (ug/L)	20	0.98	2.39	10.91	-0.14	2.10
As (ug/L)	20	0.53	0.34	1.32	0.37	0.69
Fe (ug/L)	20	1900	2344	10260	803	2996
Se (ug/L)	20	0.08	0.11	0.28	0.03	0.13
Zn (ug/L)	20	17.5	22.47	110.2	6.98	28.02
Ur (ug/L)	15	0.025	0.031	0.120	0.008	0.043

Table A-2 Outlet baseflow parameter statistics

Outlet Baseflow					95% Confidence Interval	
Variable	**N**	**μ**	**σ**	**Maximum**	**Low**	**High**
Level (m)	14	0.21	0.02	0.25	0.20	0.23
Flow (L/s)	14	6.16	3.29	13.89	4.26	8.06
TOC (mg/L)	18	5.80	1.84	10.90	4.89	6.72
Turbidity (NTU)	16	11.96	17.62	76.97	2.57	21.35
TSS (mg/L)	17	9.42	16.14	67.38	1.12	17.72
***E.coli* (CFU)**	17	174	1257	5300	-168	1124
TN (mg/L)	18	5.57	0.81	6.75	5.17	5.98
TP (mg/L)	19	0.050	0.021	0.122	0.039	0.060
DO (mg/L)	16	6.49	2.81	11.23	4.99	7.99
Conductivity (mS/cm)	16	0.53	0.22	1.19	0.41	0.65
pH	16	6.63	0.26	7.14	6.49	6.77
Hardness (mg/L)	18	100.9	29.3	175.5	86.3	115.4
Ag (ug/L)	10	0.05	0.03	0.10	0.03	0.07
Al (ug/L)	18	129	306	1323	-23	282
Cd (ug/L)	18	0.03	0.03	0.12	0.01	0.04
Cu (ug/L)	18	2.76	2.87	9.92	1.34	4.19
Ni (ug/L)	18	2.87	1.23	6.35	2.26	3.47
Pb (ug/L)	18	1.29	1.74	7.52	0.42	2.15
As (ug/L)	18	0.60	0.18	0.96	0.51	0.69
Fe (ug/L)	18	2429	953	5035	1954	2903
Se (ug/L)	18	0.12	0.11	0.33	0.07	0.18
Zn (ug/L)	18	11.86	15.03	65.49	4.39	19.34
Ur (ug/L)	18	0.034	0.065	0.262	0.001	0.066

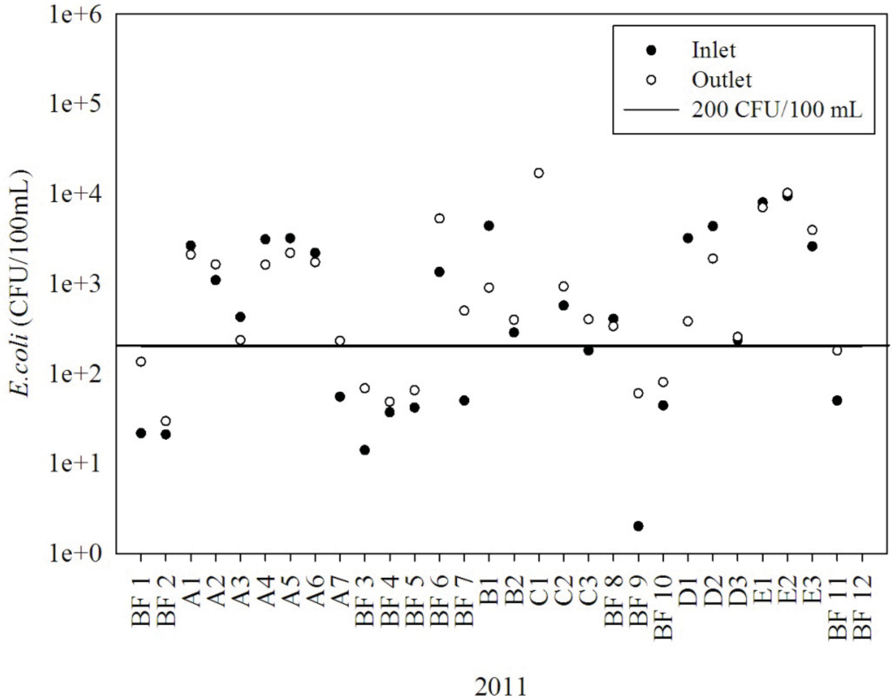
Table A-3 Stormflow concentration reduction percentages

% Reduction	TOC	TSS	*E.coli*	TN	TP	Pb	Cu	Fe	Al	Zn	Cd
Storm A June 13 2011	-51	-11	-29	+15	+9	-30	+36	-126	+17	+43	-169
A1	-3	-124	+21	+9	+26	-50	+40	-383	-34	+31	--
A2	-11	+41	-49	+6	+24	+28	+58	-77	+52	+51	--
A3	-164	--	+45	+15	-94	-151	+30	-95	-65	+2	--
A4	-65	-23	+48	+6	+46	+75	+67	+33	+67	+74	--
A5	-12	+32	+32	+27	+32	-8	+12	-152	+11	+48	--
A6	+7	+42	+21	+45	+56	+24	+38	-21	+56	+59	--
A7	-110	-31	-318	-0	-24	-126	+8	-188	+35	+33	-169
Storm B July 31 2011	-108	-88	+20	+8	-226	-131	+48	-208	-53	+51	+23
B1	-34	+83	+80	-17	-60	+23	+56	-302	+49	+59	-10
B2	-183	-259	-39	+33	-392	-285	+39	-114	-156	+43	+57
Storm C Aug 9 2011	-77	-9	-93	+9	-19	-27	+35	-126	+40	+44	+45
C1	-69	+84	--	-2	+32	+74	+54	+53	+70	+32	+14
C2	-39	-15	-63	+9	-17	+4	+32	-130	+62	+60	+68
C3	-122	-95	-122	+22	-72	-160	+20	-301	-14	+41	+53
Storm D Oct 3 2011	-196	-48	+45	-35	-50	-39	+41	-313	+14	+38	+13
D1	-279	+65	+88	-76	+3	+60	+70	-276	+83	+64	+30
D2	-141	+42	+56	-42	-16	-47	+23	-434	-4	+16	-39
D3	-168	-251	-9	+13	-136	-129	+30	-228	-36	+35	+48
Storm E Oct 5 2011	-6	+16	-16	+9	+29	+15	+31	-47	+30	+51	+38
E1	+8	+14	+12	+15	+31	+22	+40	-155	+36	+59	+33
E2	-7	+55	-9	-9	+37	+14	+17	+66	+19	+38	+39
E3	-20	-21	-52	+23	+21	+9	+34	-53	+36	+56	+43
Storm F Apr 23 2012	-70	+9	+85	+3	-133	+18	+58	-2	-1	+66	+70
F1	-239	-109	+82	+42	-480	-93	+10	-82	-165	+44	+78
F2	+16	+99	+97	-13	+52	+99	+98	+96	+99	+98	+97
F3	+14	+35	+75	-20	+28	+49	+67	-19	+62	+55	+34
Storm G June 26 2012	-19	-161	-44	-2	+21	-2	+56	-235	+39	+46	-29
G1	+51	-409	-75	-10	+57	+68	+86	-110	+82	+82	+56
G2	-109	-57	-138	+6	-60	-91	+19	-584	-27	+7	-193
G3	+1	-17	+80	-3	+68	+16	+63	-10	+62	+49	+49

Storm H July 24 2012	-238	-344	**+28**	-12	-467	-100	**+17**	-342	-56	**+29**	-130
H1	-224	-815	**+93**	**+6**	-1028	-164	**+1**	-132	-128	**+12**	-469
H2	-77	-143	-36	**+4**	-381	-120	**+2**	-344	-70	**+12**	**+69**
H3	-414	-73	--	-46	**+6**	-17	**+48**	-548	**+30**	**+63**	**+9**
Storm I Sep 5 2012	-98	**+11**	-34	-16	-18	-26	**+18**	-129	**+23**	**+33**	-40
I1	-160	-40	-102	-10	-51	-94	**+5**	-307	-14	**+31**	-84
I2	-107	**+19**	-21	-6	-17	**+7**	**+10**	-87	**+35**	**+25**	-54
I3	-26	**+54**	**+20**	-32	**+14**	**+9**	**+41**	**+8**	**+49**	**+45**	**+18**
Storm J Sep 10 2012	-94	**+60**	-117	-18	**+39**	**+50**	**+47**	**+40**	**+69**	**+58**	-96
J1	-200	**+95**	-241	-22	**+30**	**+79**	**+53**	**+73**	**+88**	**+63**	-98
J2	**+12**	**+24**	**+8**	-15	**+49**	**+20**	**+42**	**+7**	**+49**	**+54**	-94
Storm K Sept 20 2012	-42	-73	-195	**+4**	**+5**	-6	-24	-59	**+25**	**+46**	**+49**
K1	-69	-227	-557	**+5**	-31	-30	-282	-87	**+10**	**+37**	**+42**
K2	-34	-63	-29	-10	-9	-27	**+41**	-84	-4	**+20**	**+29**
K3	-5	**+62**	-9	-32	**+51**	**+75**	**+41**	**+39**	**+58**	**+56**	**+65**
K4	-34	-32	-36	**+11**	**+19**	-16	**+17**	-55	**+22**	**+55**	**+53**
K5	-86	-66	-400	**+21**	-16	-28	**+9**	-116	**+33**	**+44**	**+54**
K6	-22	-109	-137	**+28**	**+15**	-12	**+31**	-54	**+32**	**+66**	**+54**

Table A-4 Baseflow concentration reduction statistics

% Reduction	TOC	TSS	*E.coli*	TN	TP	Pb	Cu	Fe	Al	Zn	Cd
BF 1	-319	-55	-523	**+13**	-106	-42	**+64**	-60	-165	**+26**	-607
BF 2	-312	-362	-40	**+23**	-500	**+6**	**+14**	**+10**	-48	**+26**	**+81**
BF 3	-276	-106	-388	-16	-386	-94	**+11**	-91	-112	**+18**	**+52**
BF 4	-123	-138	-31	**+22**	-206	**+22**	**+58**	-64	**+55**	**+69**	**+100**
BF 5	**+31**	-186	-56	**+20**	-850	-255	-13	-137	-354	-9	-514
BF 6	-115	-229	-293	**+33**	-206	-78	**+42**	-254	-31	**+39**	--
BF 7	-306	-388	-900	**+17**	-667	--	--	--	--	--	--
BF 8	-115	-245	**+17**	**+34**	-200	-101	**+58**	-145	-77	**+52**	**+67**
BF 9	-270	-545	-2900	**+21**	-467	-52	**+47**	**+79**	**+10**	**+69**	**+78**
BF 10	-292	**+17**	-82	**+8**	-557	-175	**+47**	-76	-82	**+60**	**+76**
BF 11	-121	-715	-260	**+5**	-160	-467	**+19**	-372	-1	**+26**	**+56**
BF 12	-26	--	--	--	--	-476	-91	-420	-76	-25	**+14**
BF 13	-9	--	--	**+10**	**+25**	**+31**	**+47**	**+10**	**+52**	**+41**	-51
BF 14	-157	--	**+94**	-21	-200	-5	-38	**+60**	**+17**	**+42**	**+83**
BF 15	-326	-567	**+40**	-17	-83	-10	**+26**	-59	**+21**	**+32**	**+63**
BF 16	-301	-534	-831	-2	-723	**+10**	**+37**	-427	-76	**+47**	**+97**
BF 17	--	--	--	--	-449	-20	**+29**	-58	**+8**	**+28**	**+51**
BF 18	-328	-1025	-289	**+11**	-1475	-90	**+34**	-183	-210	**+56**	**+90**
BF 19	-278	-2724	**+82**	**+11**	-1200	-535	-21	-41	-834	-10	**+37**
BF 20	--	-423	-72	**+40**	-117	--	--	--	--	--	--



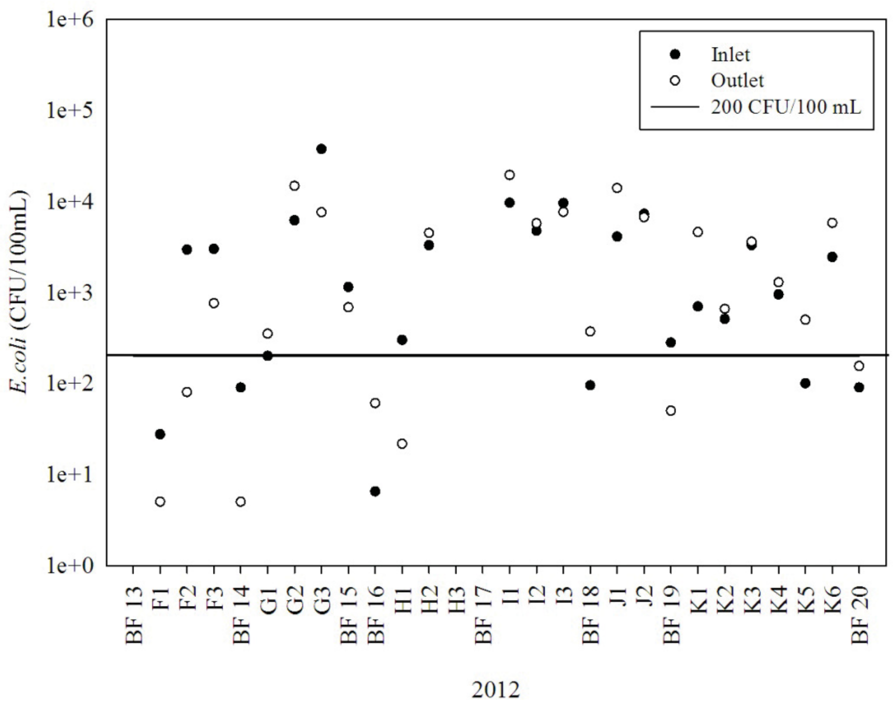
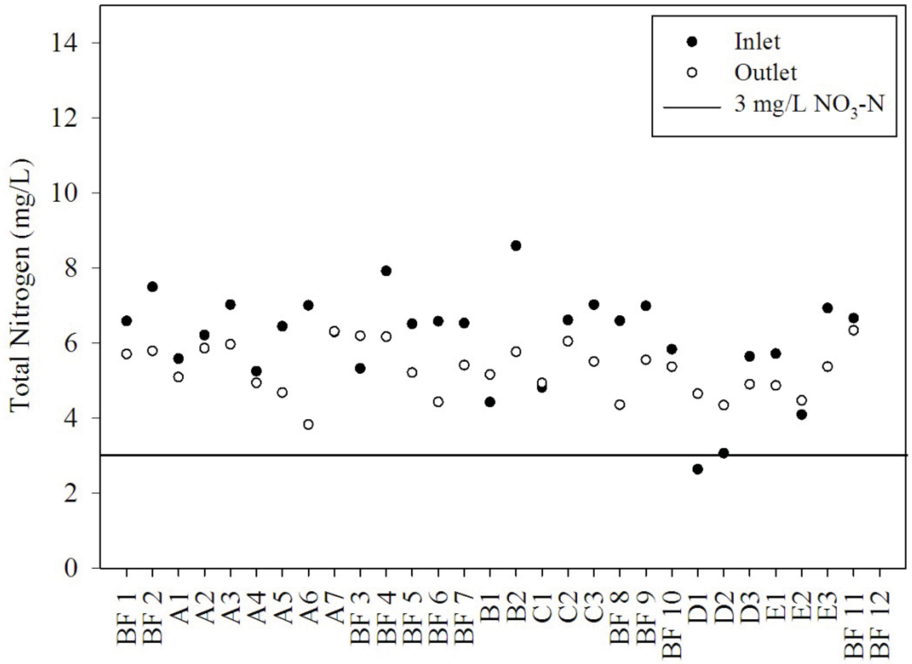
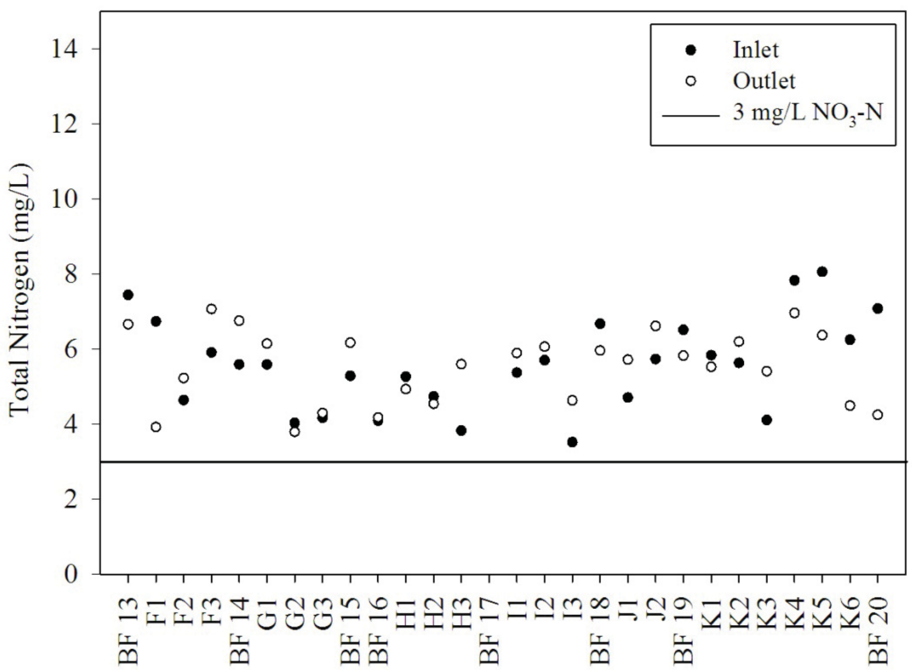


Figure A-12 *E.coli* grab sample concentration plots for baseflow and stormflow



2011



2012

Figure A-13 TN grab sample concentration plots for baseflow and stormflow

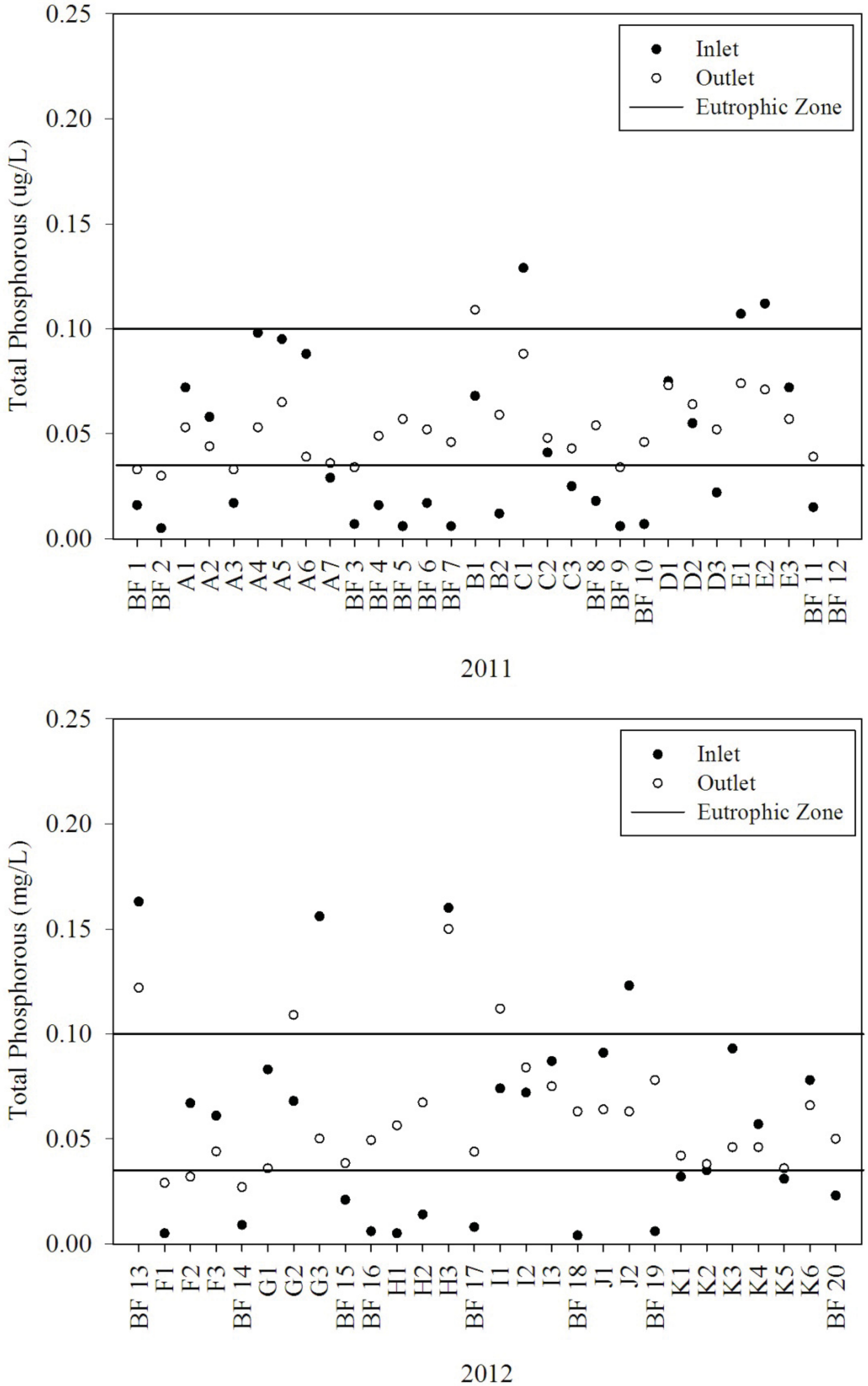
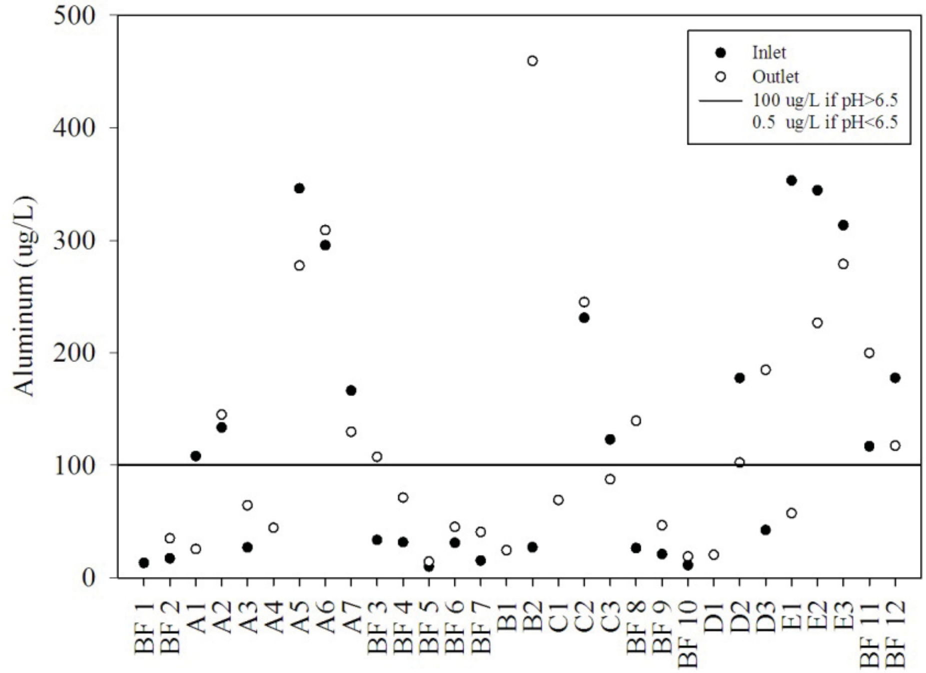
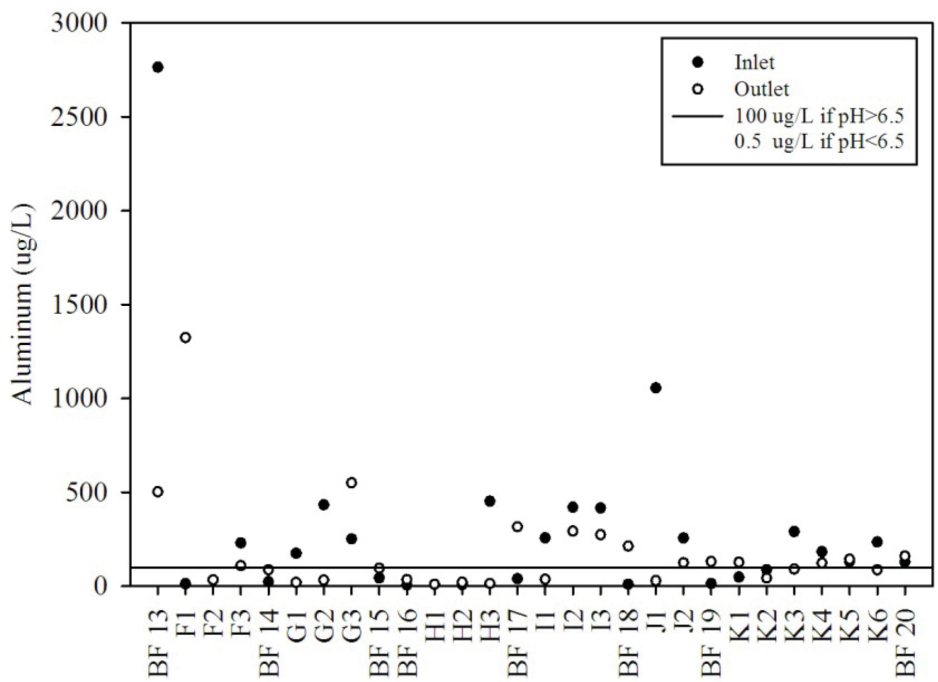


Figure A-14 TP grab sample concentration plots for baseflow and stormflow



2011



2012

Figure A-15 Al grab sample concentration plots for baseflow and stormflow

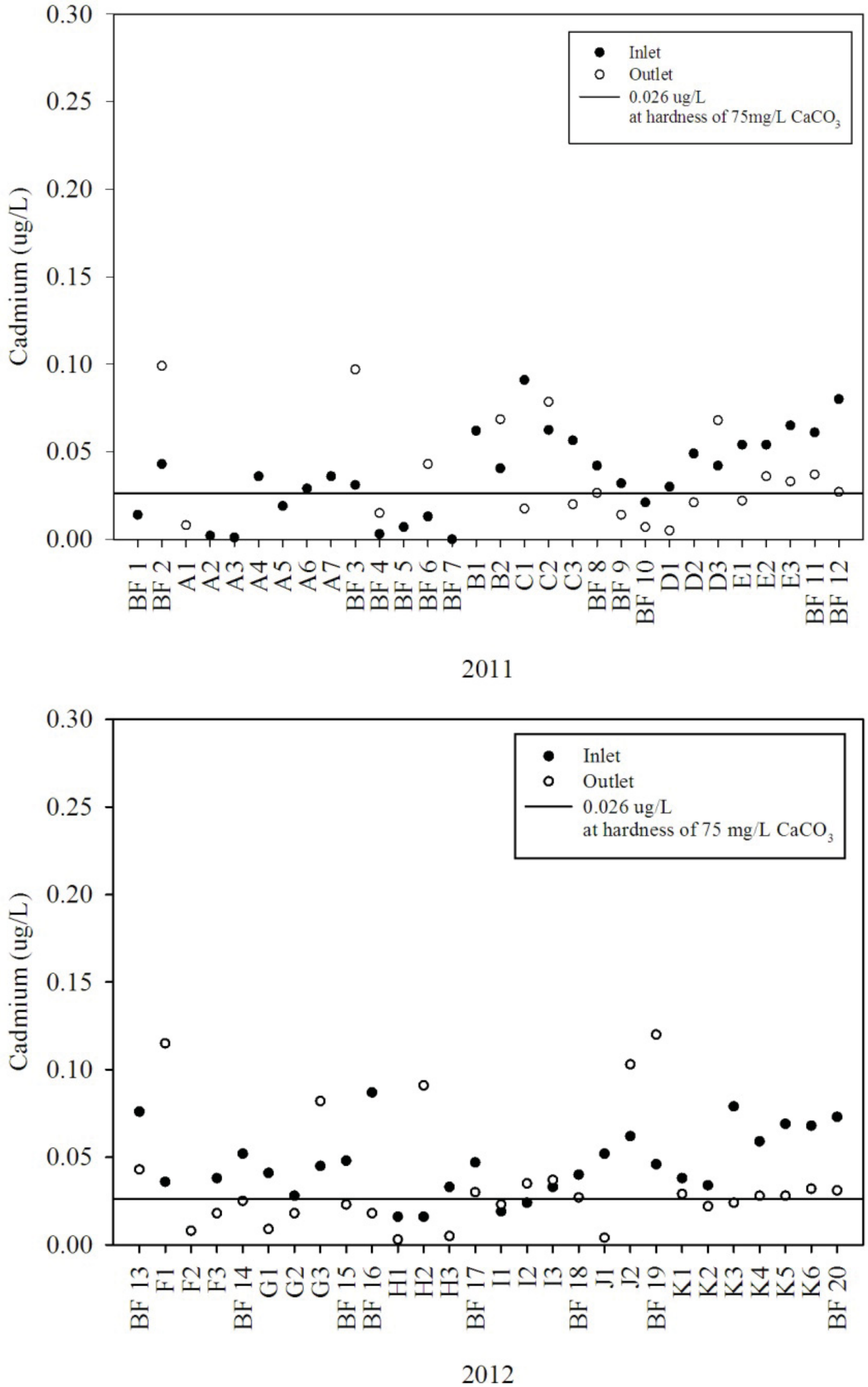


Figure A-16 Cd grab sample concentration plots for baseflow and stormflow

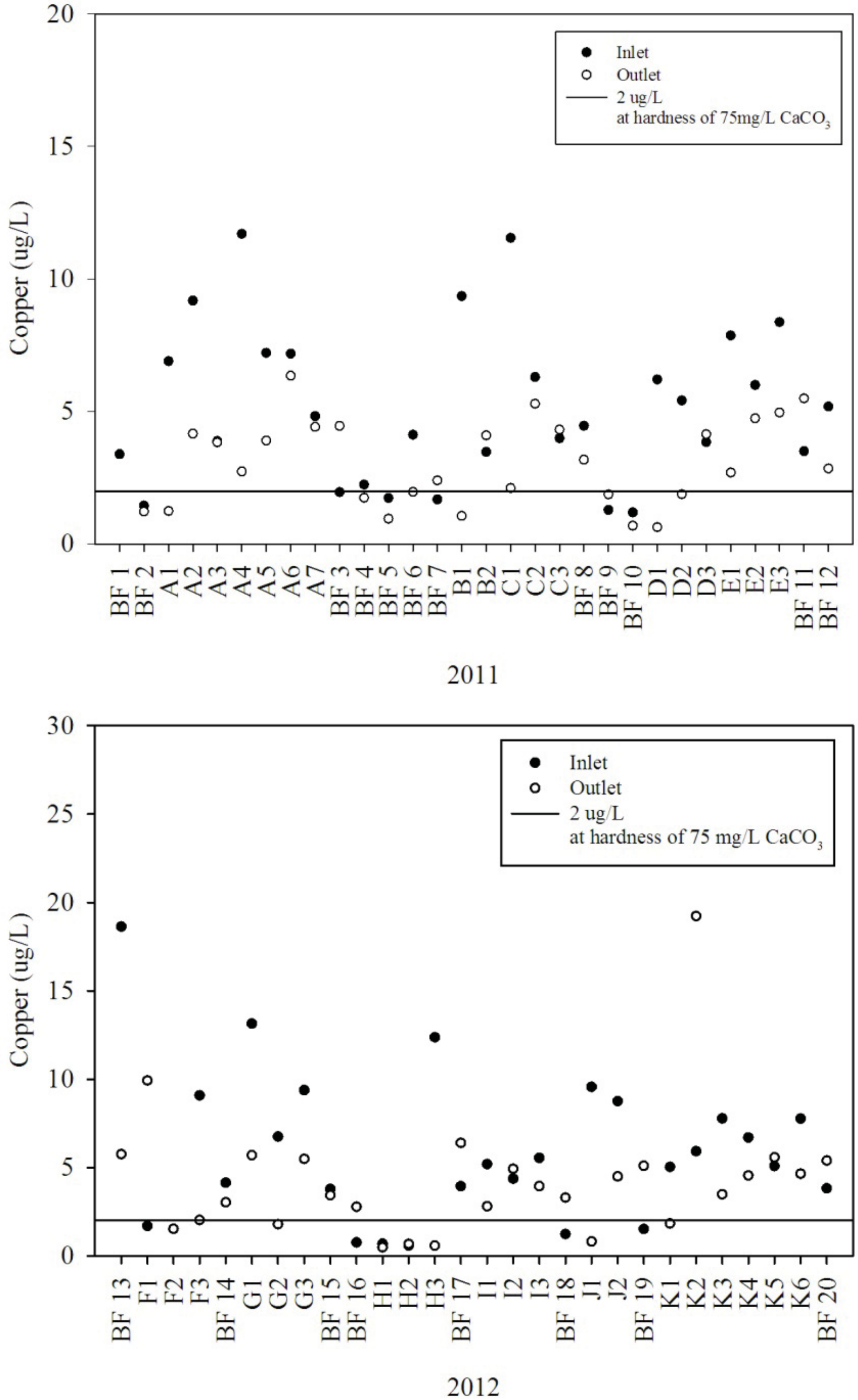


Figure A-17 Cu grab sample concentration plots for baseflow and stormflow

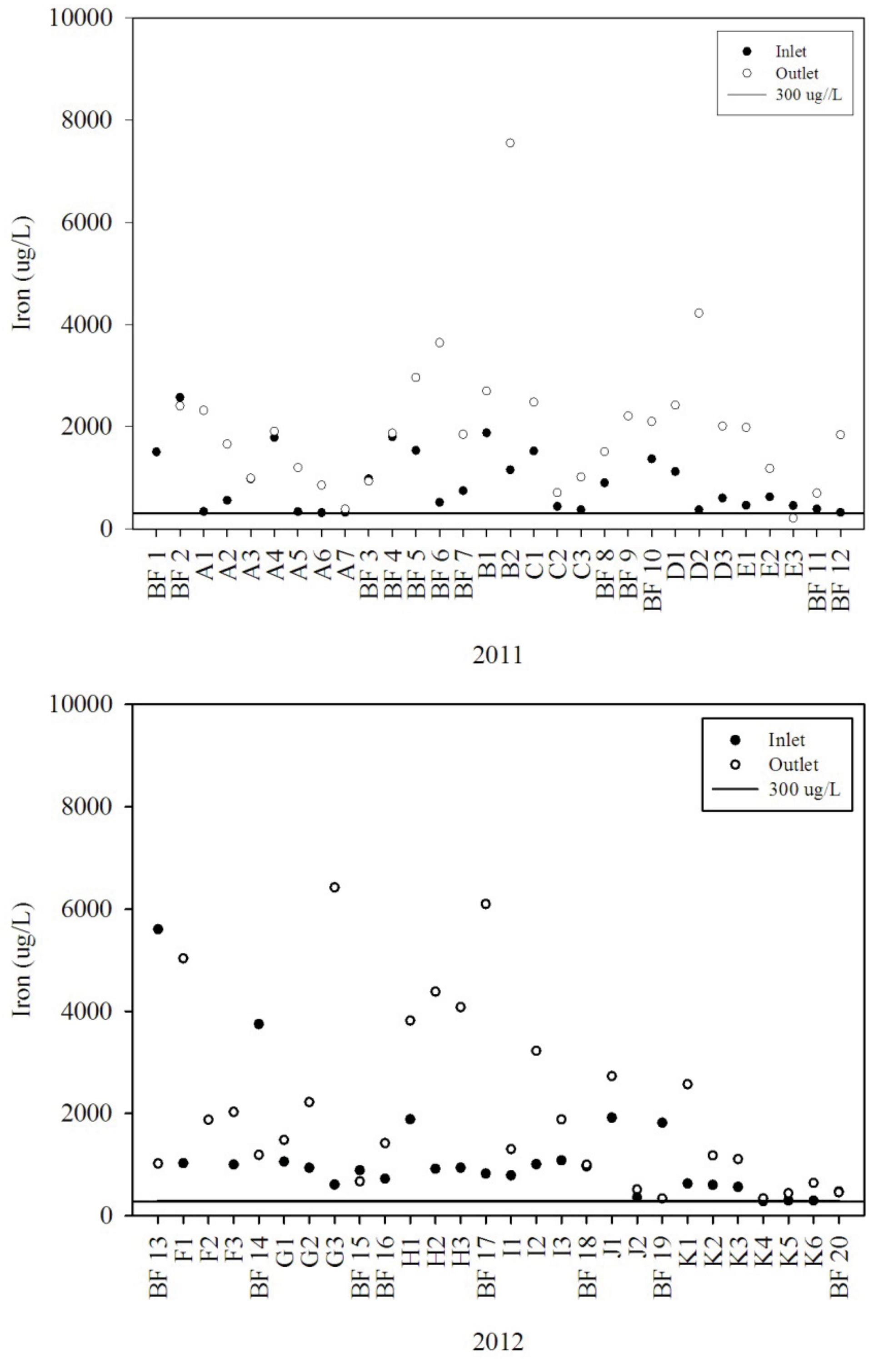
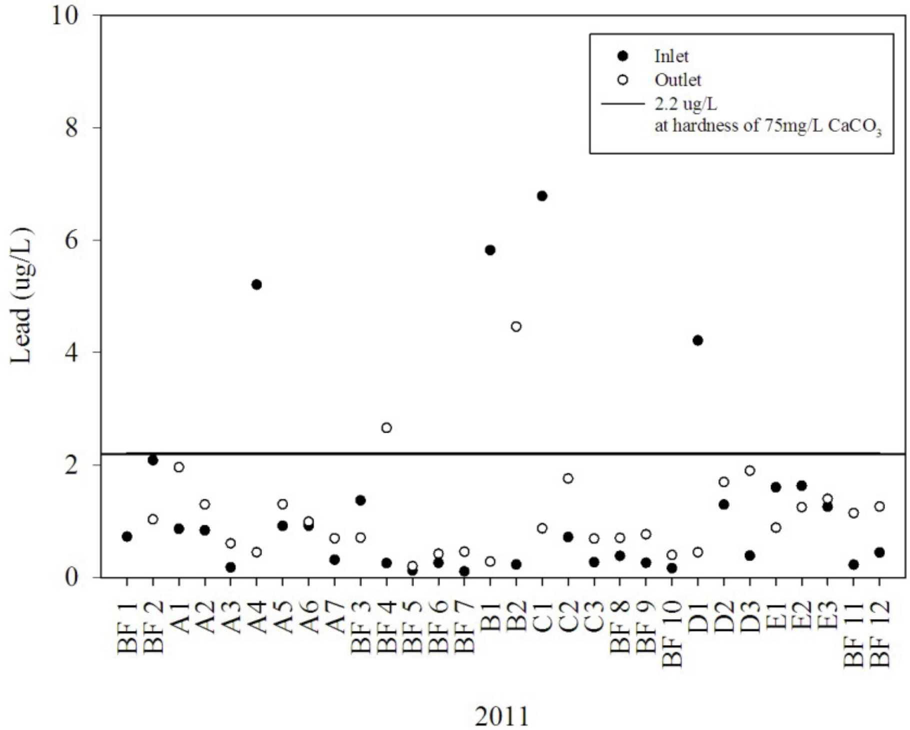


Figure A-18 Fe grab sample concentration plots for baseflow and stormflow



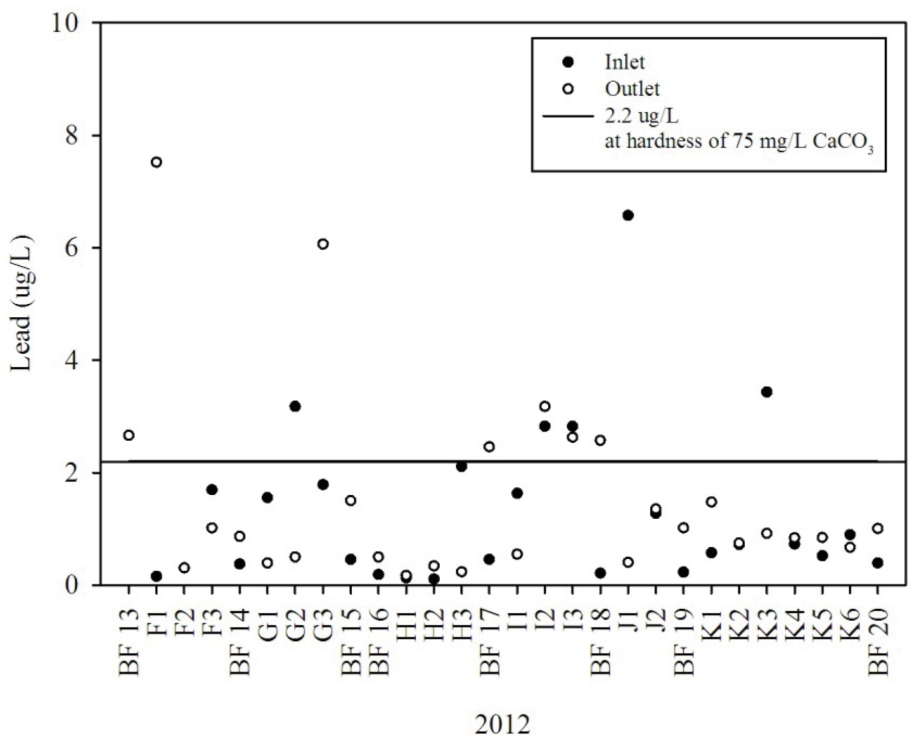
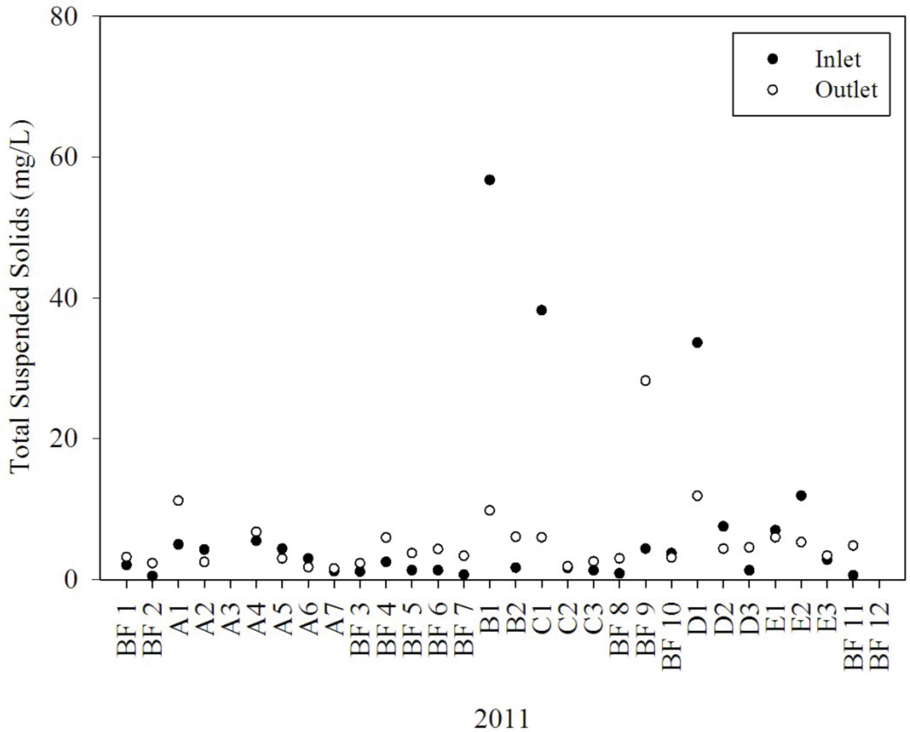


Figure A-19 Pb grab sample concentration plots for baseflow and stormflow



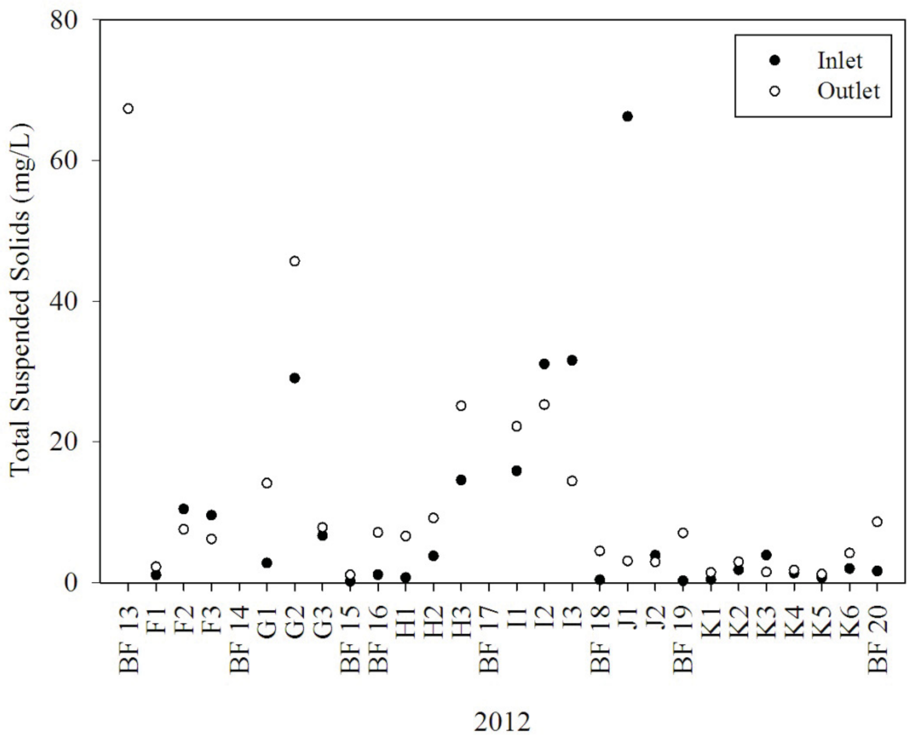


Figure A-20 TSS grab sample concentration plots for baseflow and stormflow

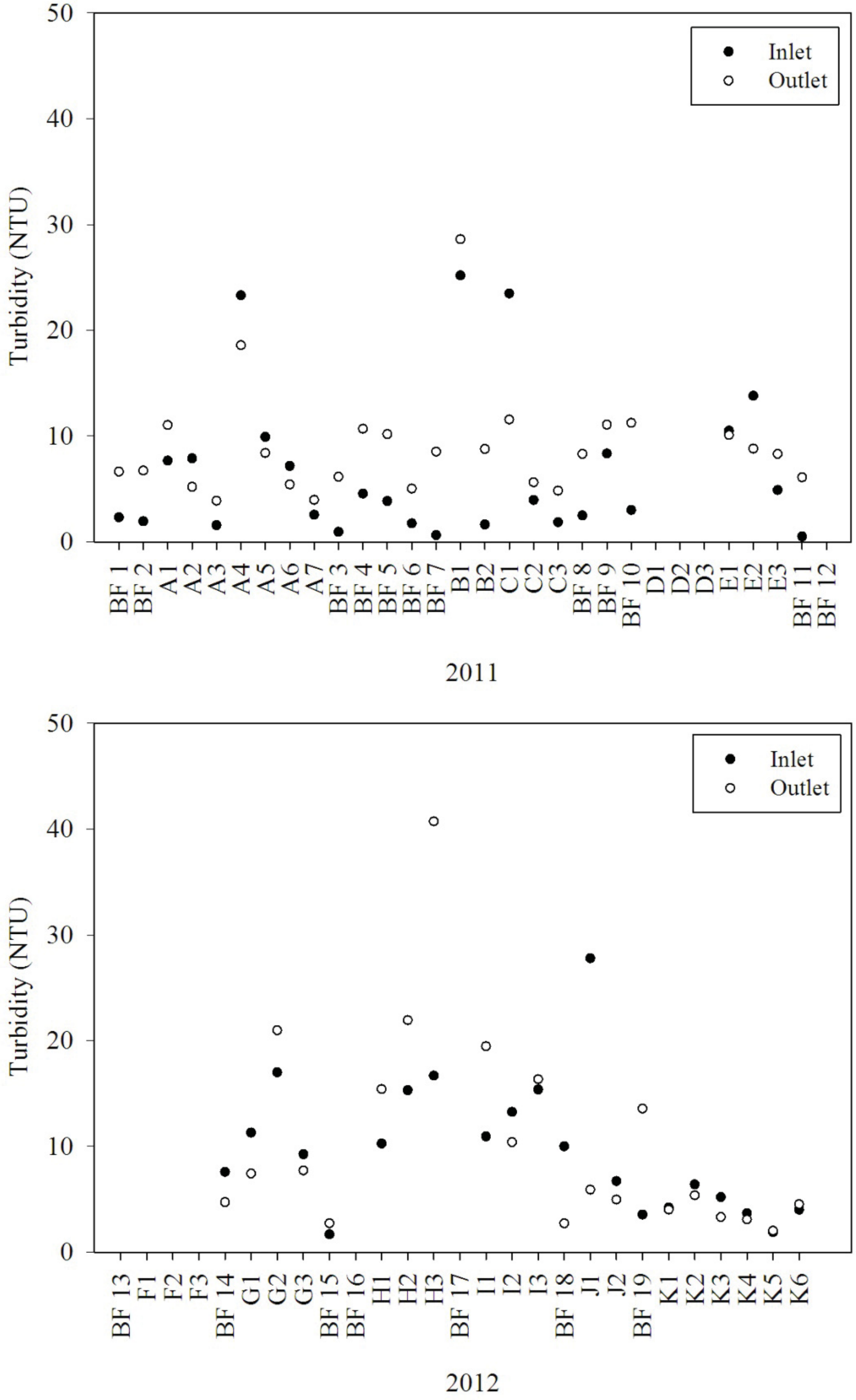


Figure A-21 Turbidity grab sample plots for baseflow and stormflow

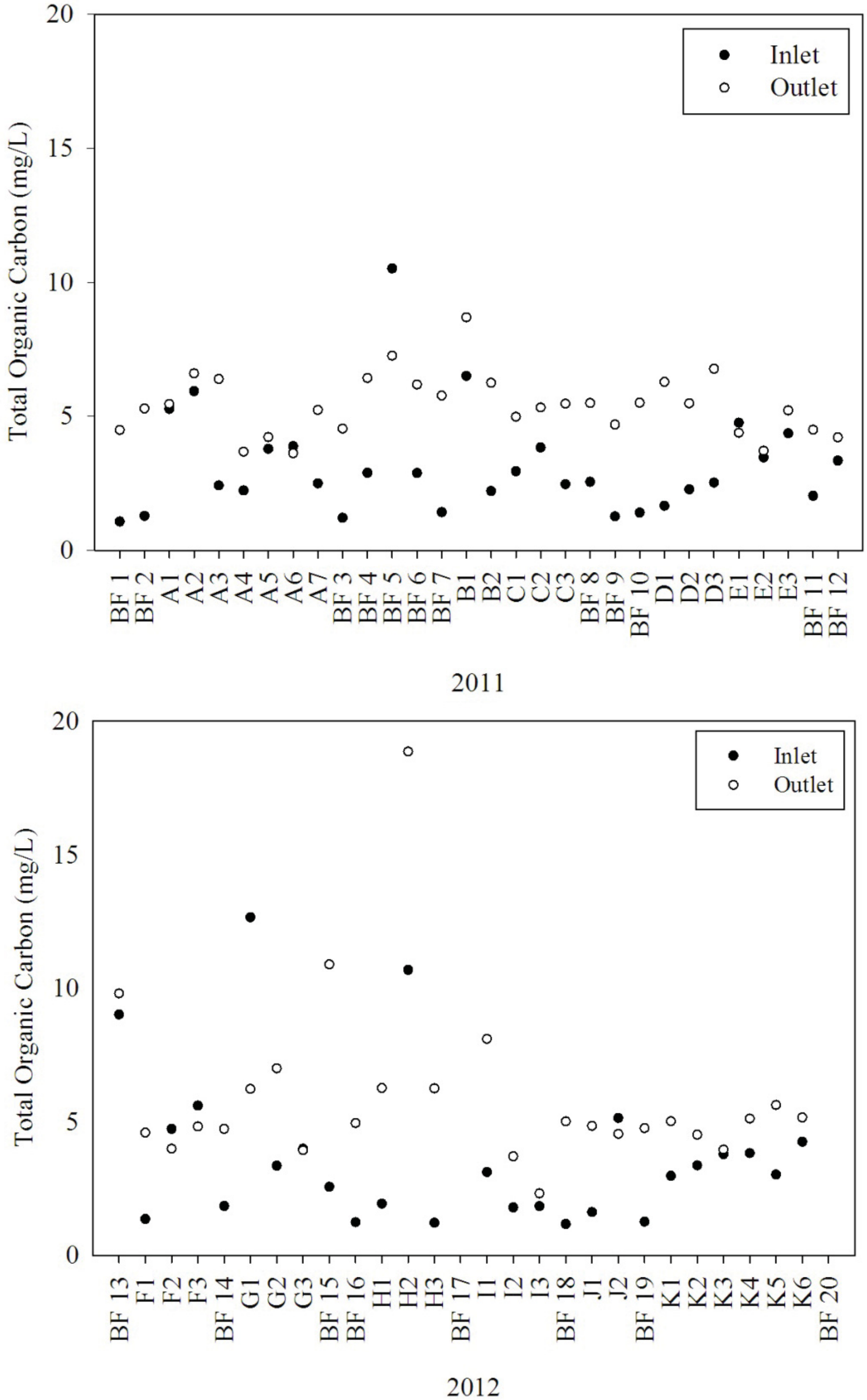
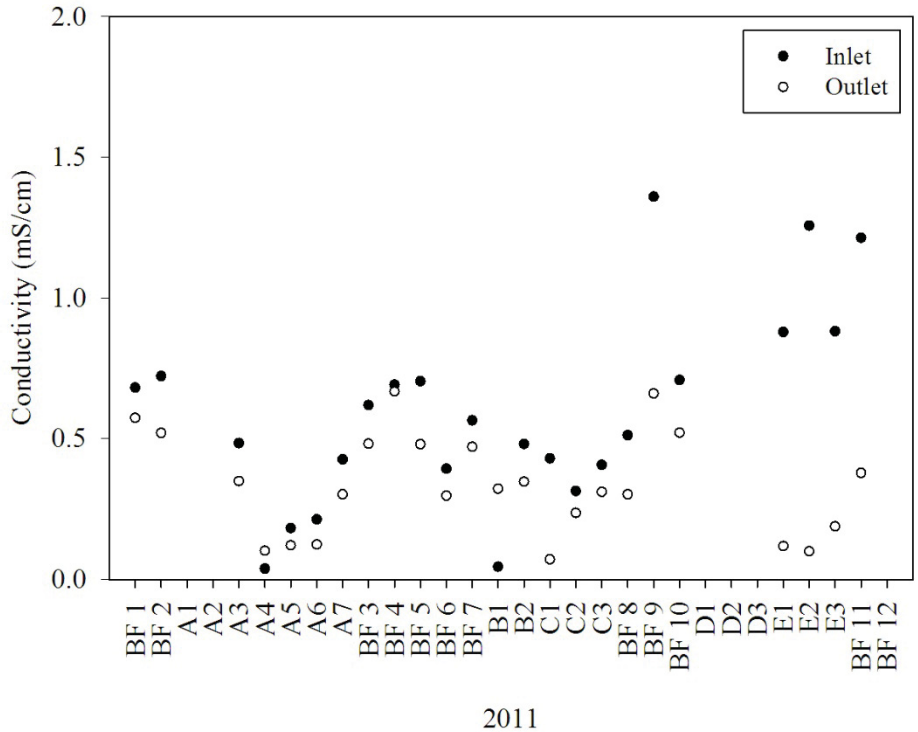


Figure A-22 TOC grab sample concentration plots for baseflow and stormflow



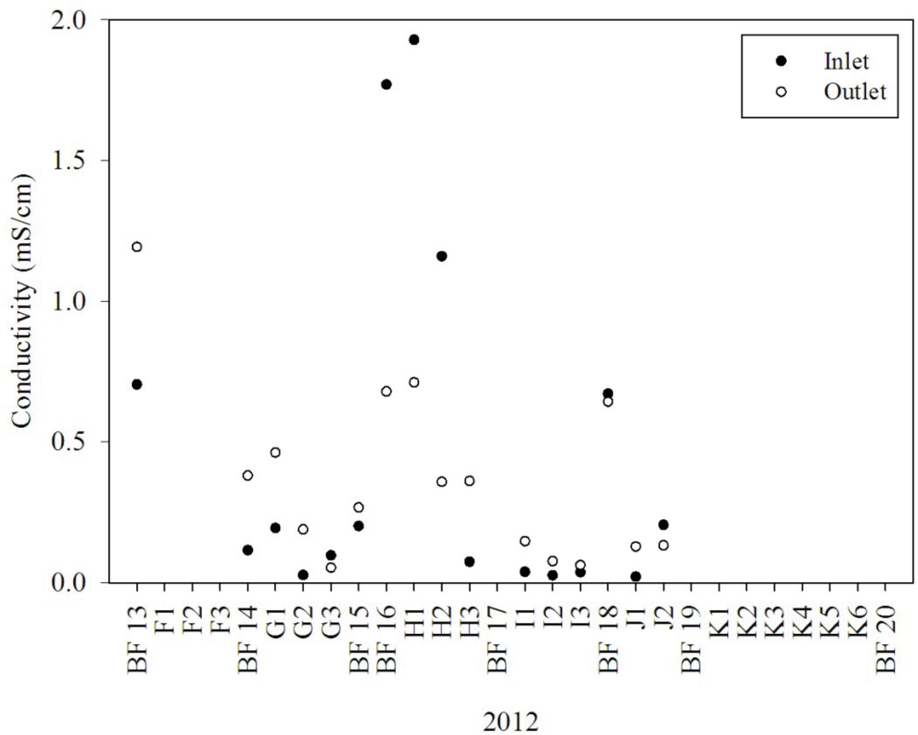


Figure A-23 Conductivity in-situ sample plots for baseflow and stormflow

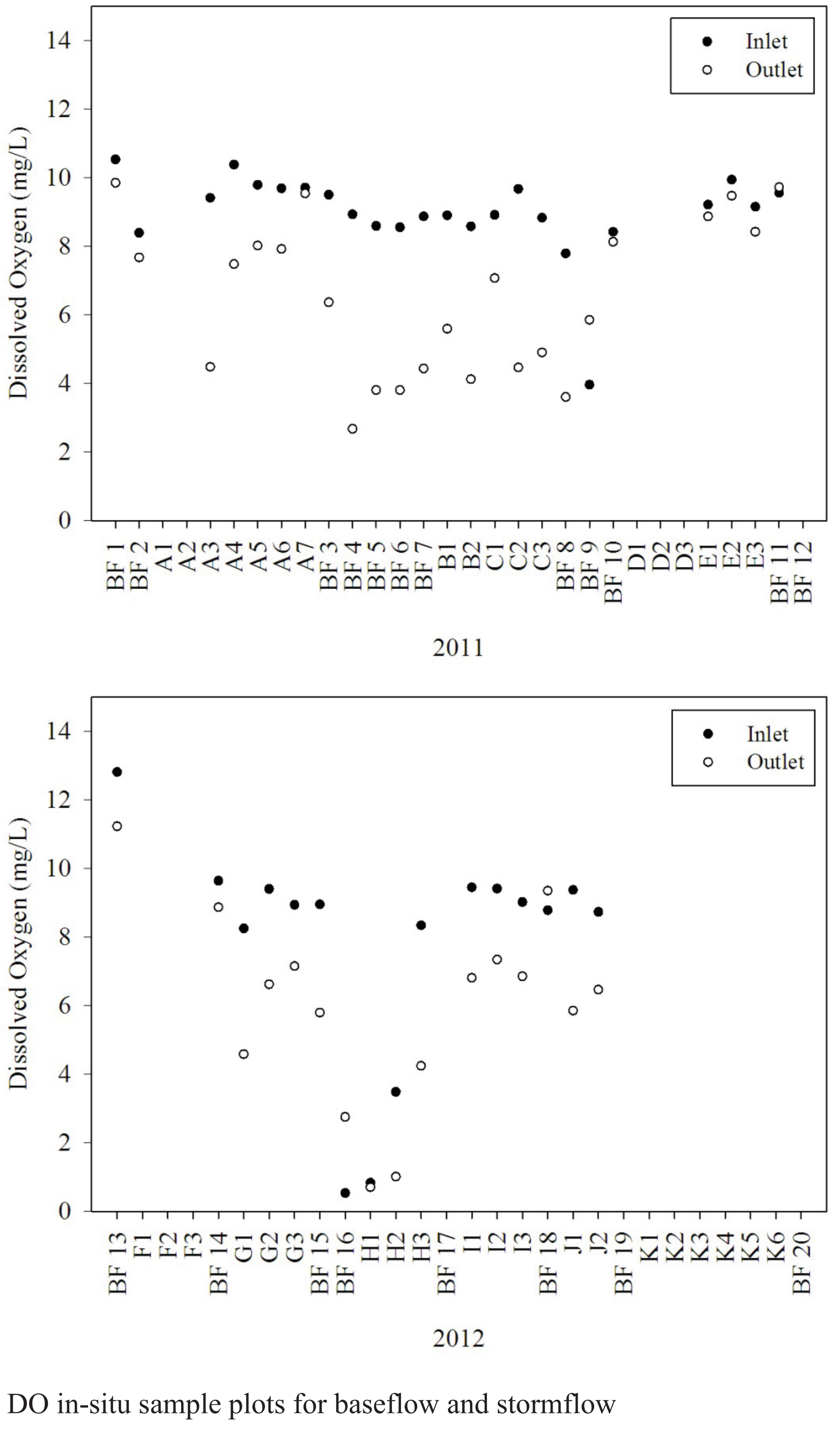
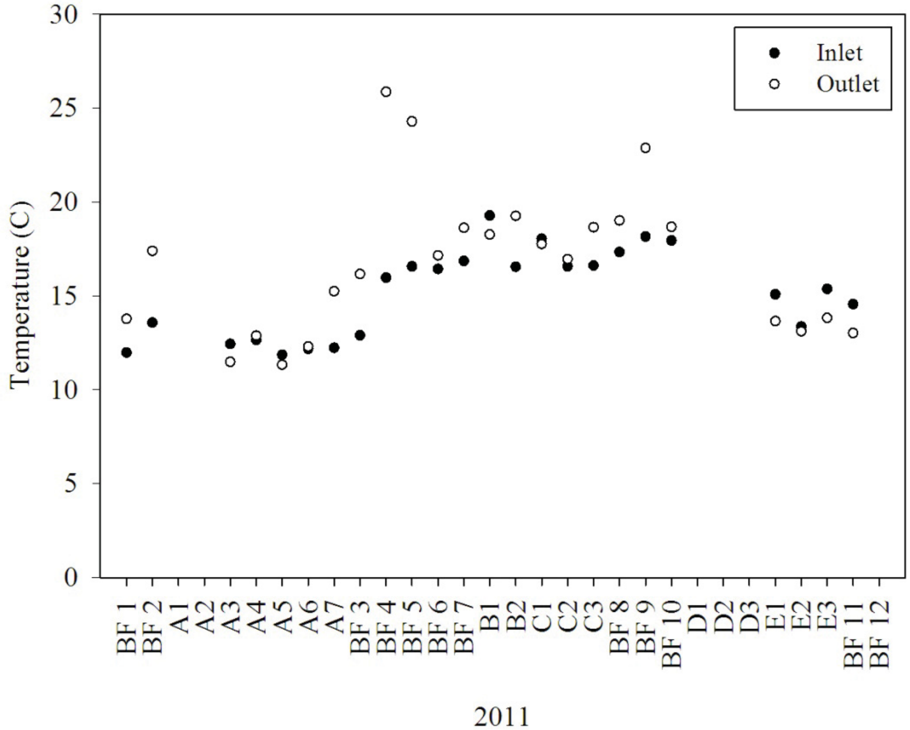


Figure A-24 DO in-situ sample plots for baseflow and stormflow



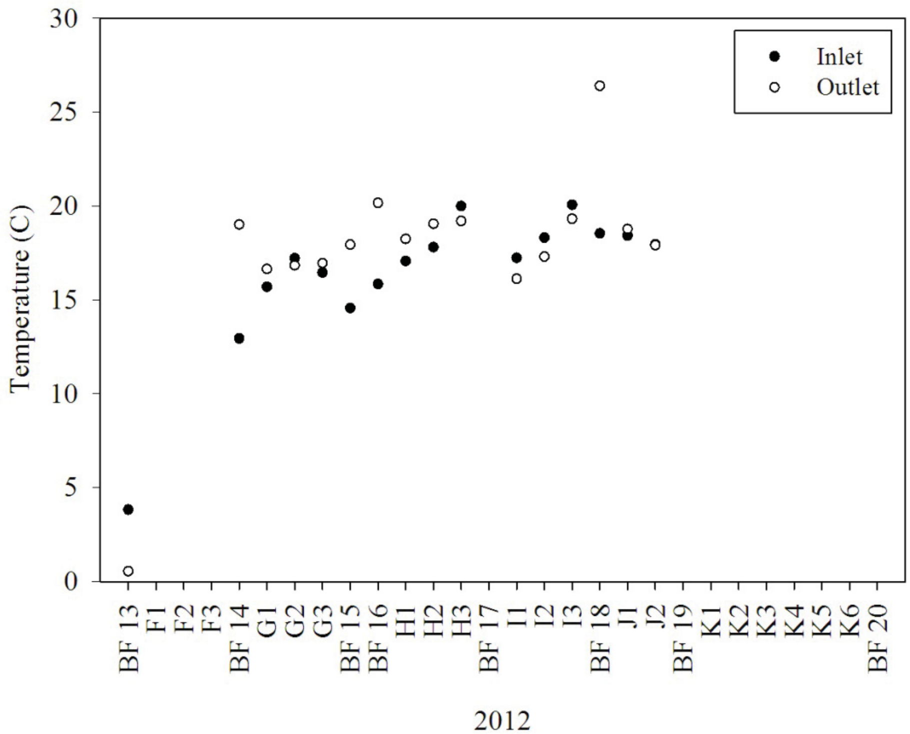
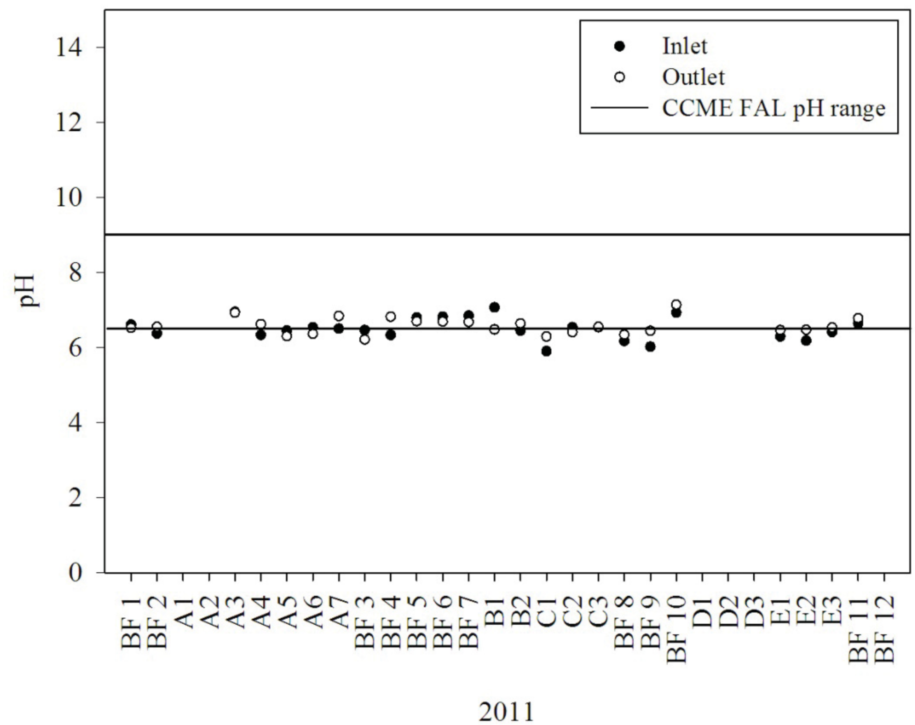


Figure A-25 Temperature in-situ sample plots for baseflow and stormflow



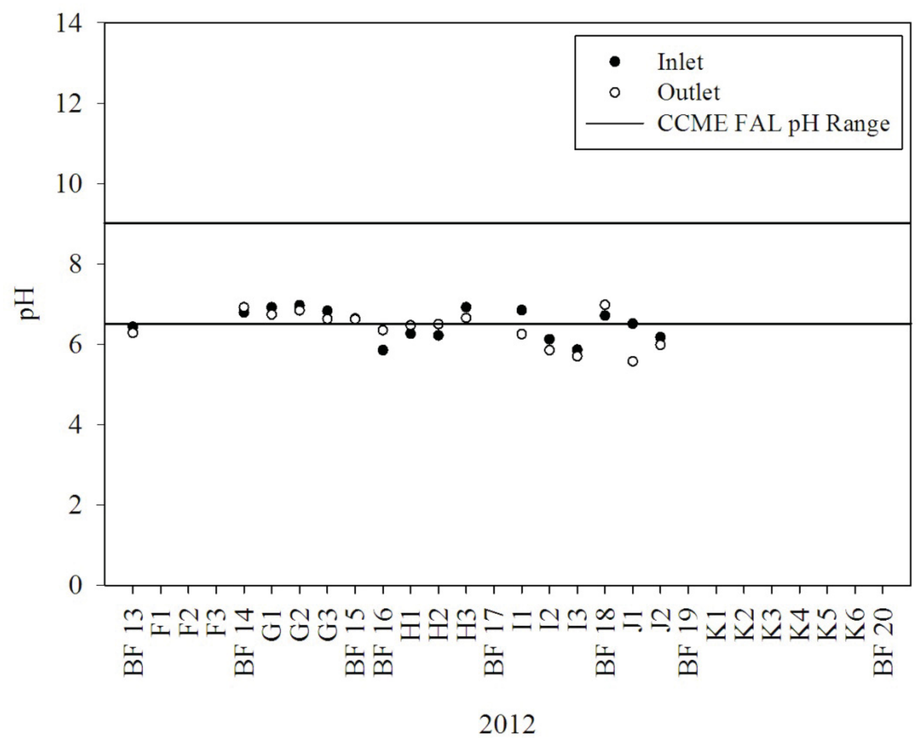
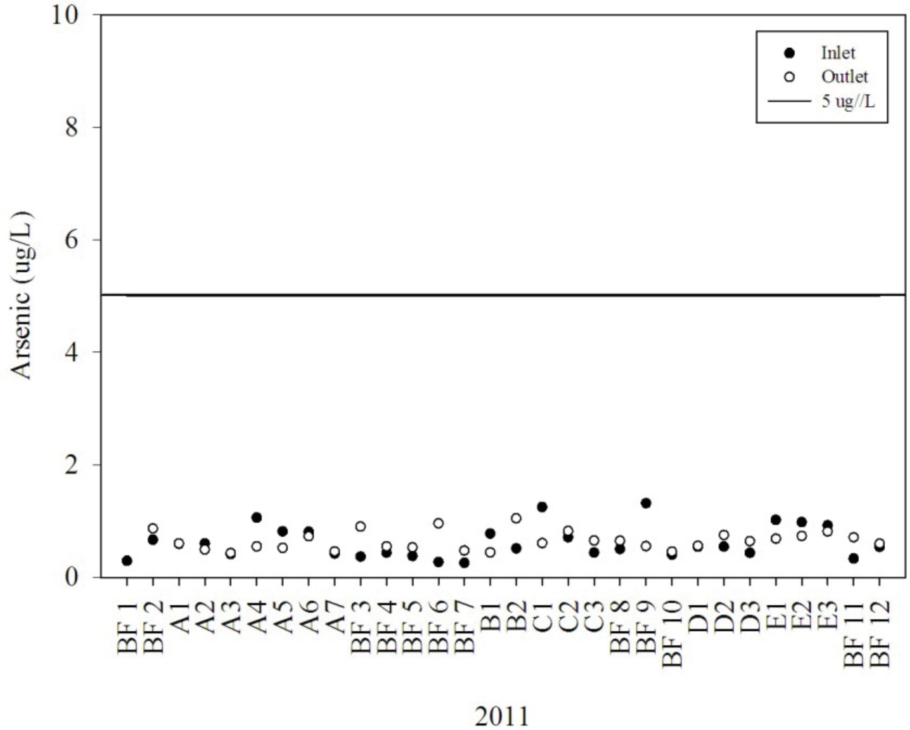


Figure A-26 pH in-situ sample plots for baseflow and stormflow



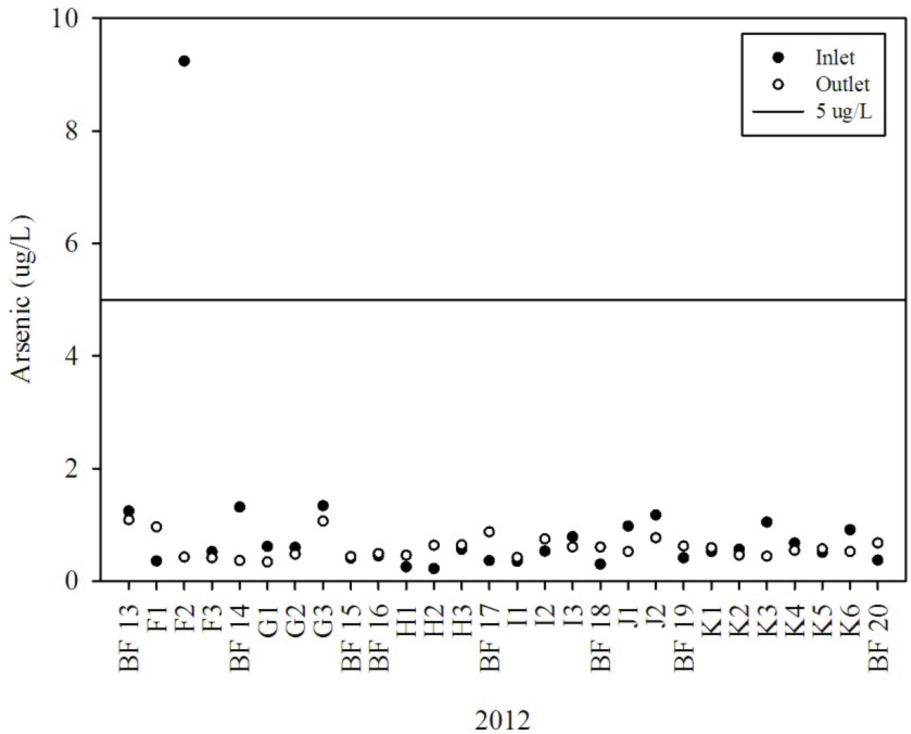
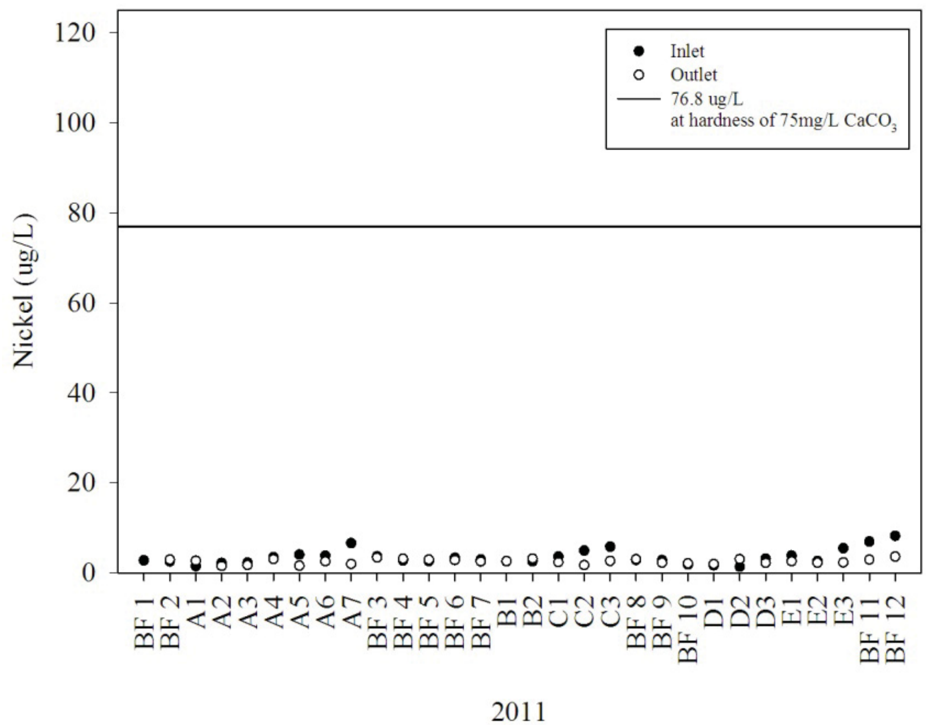


Figure A-27 As sample concentration plots for baseflow and stormflow



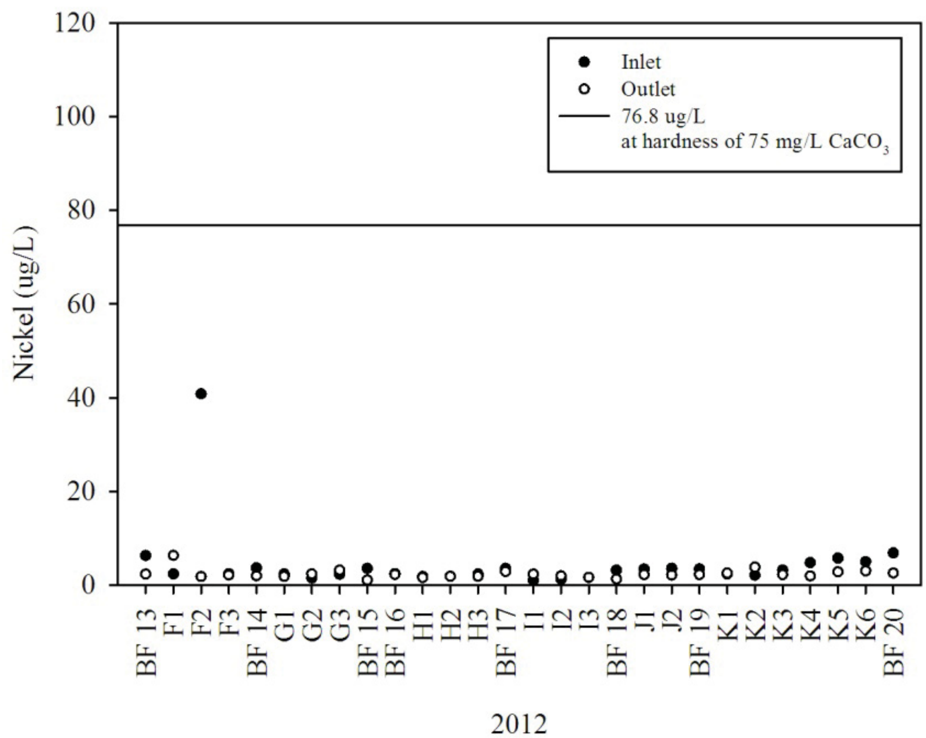
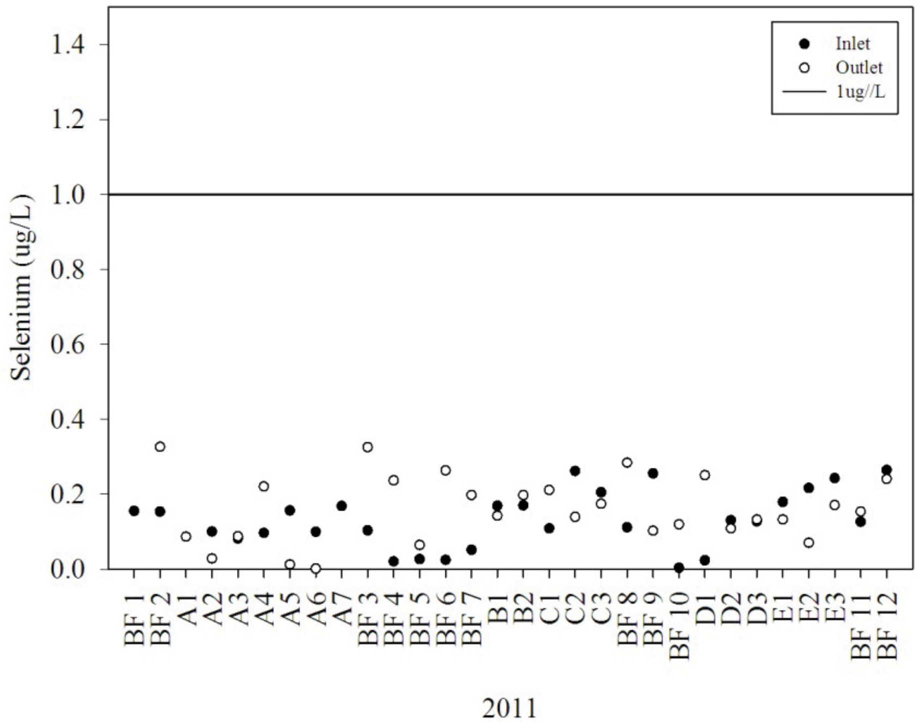


Figure A-28 Ni sample concentration plots for baseflow and stormflow



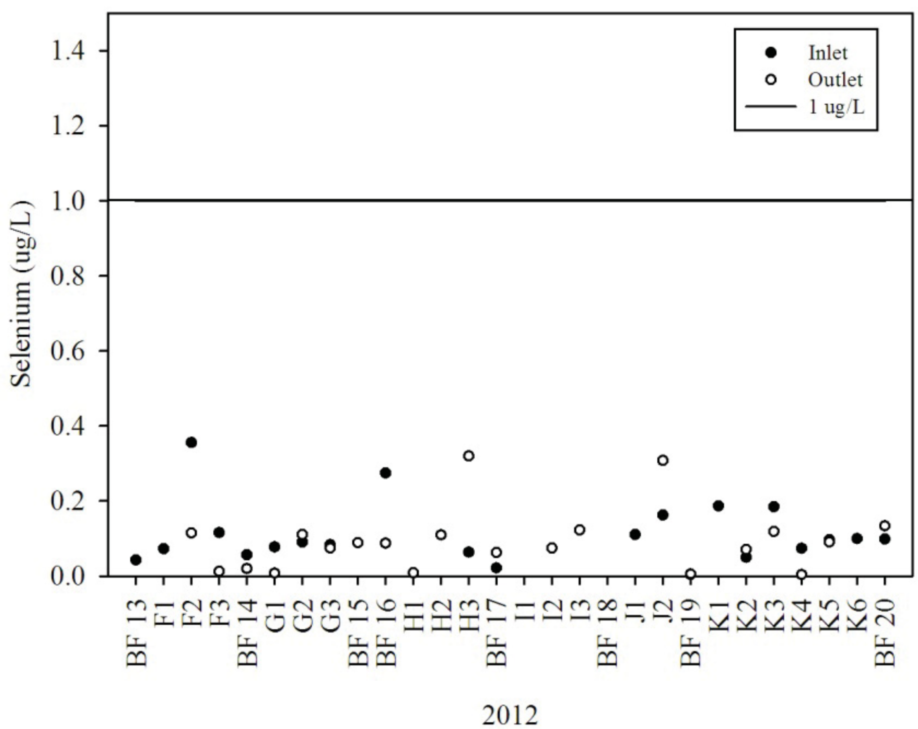
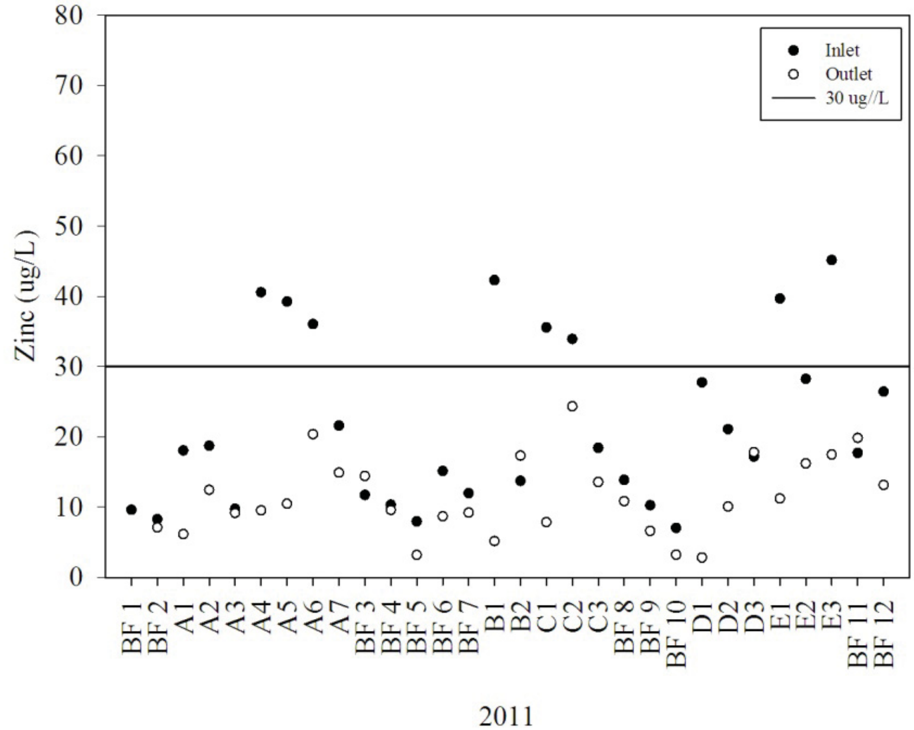


Figure A-29 Se sample concentration plots for baseflow and stormflow



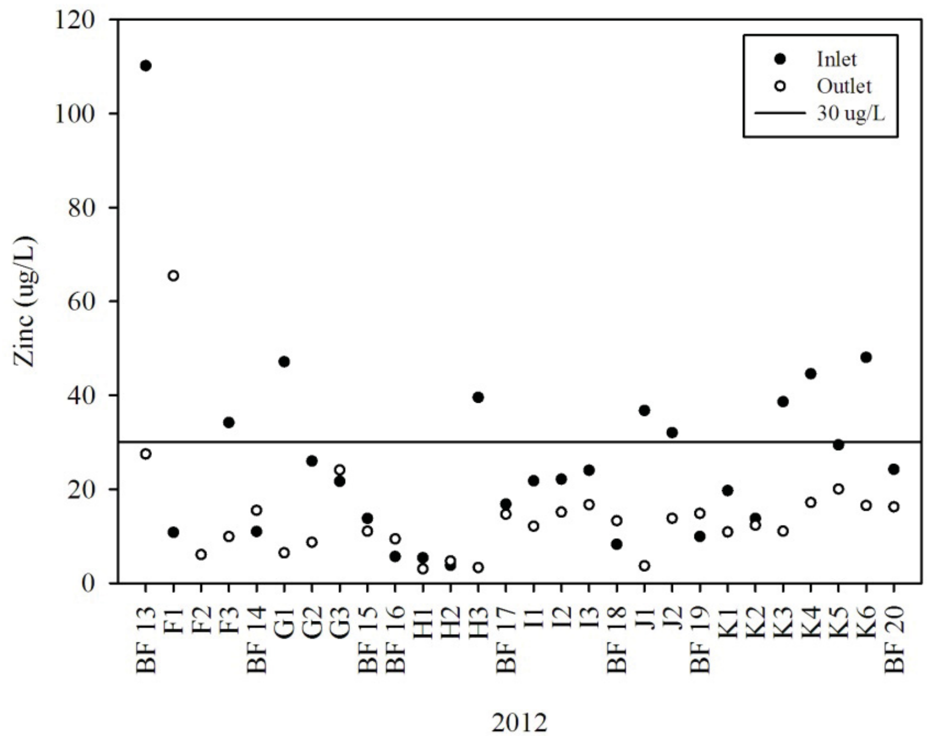


Figure A-30 Zn sample concentration plots for baseflow and stormflow

Table A-5 Sources, concerns and guidelines associated with common heavy metals

Metal	Anthropogenic Sources [1, 2]	Human Health Concerns [2,3,4]	Stormwater Concentration Ranges, from Literature [2]	FAL Guideline [3]	
				Short term	Long term
Aluminum	Flocculation, combustion of coal	Can cause skeletal problems if ingested in large quantities, may be associated with dementia and Alzheimer's disease	0.1-16 mg/L	---	5 µg/L if pH<6.5 100 µg/L if pH ≥ 6.5
Arsenic	Industrial emissions, fossil fuel combustion, smelting, herbicides, preservatives	Ingesting high concentrations can be rapidly fatal, lower concentrations can cause cancer, paralysis of hands and feet	0.001-0.21 mg/L	---	5 µg/L
Boron	Glass, biomass incineration, fossil fuel combustion, smelting	Can be lethal through ingestion of large doses	---	29,000 µg/L or 29 mg/L	1,500 µg/L or 1.5 mg/L
Beryllium	Combustion of fossil fuels	Inhalation causes pulmonary fibrosis, noted as probable carcinogen, causes skin reactions upon dermal contact	0.001-0.049 mg/L	---	---

Metal	Anthropogenic Sources [1, 2]	Human Health Concerns [2,3,4]	Stormwater Concentration Ranges, from Literature [2]	FAL Guideline [3]	
				Short term	Long term
Cadmium	Vehicular wear, corrosion of galvanized metals, fertilizers and pesticides, combustion of lubricants	Bioaccumulates in kidneys, can cause hypertension, calcium loss in bones, prostate cancer and if inhaled, emphysema and chronic bronchitis	0.00005-13.73 mg/L	---	Hardness dependant: 0.018 µg/L at 50mg/L 0.033 µg/L at 100 mg/L
Chromium, hexavalent (Cr(VI))	Corrosion of welded metal plating, vehicular wear, dyes, paints, ceramics, paper, pesticides and fertilizers	Carcinogenic upon inhalation, causes skin irritation upon dermal contact	0.001-2.3 mg/L	---	1 µg/L
Chromium, trivalent (Cr(III))		Considered an essential nutrient, required as a cofactor for insulin, however can be converted to hexavalent form within cells		---	8.9 µg/L
Copper	Vehicular wear, pesticides and fungicides, corrosion of building materials		0.00006-1.41 mg/L	---	Hardness dependant: 2.0 at 50mg/L 2.36 at 100 mg/L

Metal	Anthropogenic Sources [1, 2]	Human Health Concerns [2,3,4]	Stormwater Concentration Ranges, from Literature [2]	FAL Guideline [3]	
				Short term	Long term
Iron	Corrosion of vehicular bodies and other metal objects, coal combustion, landfill leachate		0.08-440 mg/L	---	300
Lead	Emissions from gasoline combustion, tire wear	Causes damage to kidneys and central nervous system, neuropathy, mental defects, may cause hyperactivity in children	0.00057-26 mg/L	---	Hardness dependant: 1.0 at 50mg/L 3.18 at 100 mg/L
Mercury	Cement manufacturing, waste incineration, smelting, electrical products, thermometers, coal and other fossil fuel combustion	Causes damage to kidneys and central nervous system, can increase sodium and potassium absorption in cells, can cause skin reactions	0.00005-0.067 mg/L	---	0.026
Molybdenum	Common alloy component, lubricants, printing inks, rubber, paint, fertilizer	Can be lethal through ingestion of large doses	---	---	73

Metal	Anthropogenic Sources [1, 2]	Human Health Concerns [2,3,4]	Stormwater Concentration Ranges, from Literature [2]	FAL Guideline [3]	
				Short term	Long term
Nickel	Corrosion of welded metal plating, vehicular wear, electroplating and alloy manufacturing	Inhalation of larger quantities considered carcinogenic	0.001-49 mg/L	---	Hardness dependant: 25 µg/L at 50mg/L 95.58 µg/L at 100 mg/L
Selenium	Effluents from lead and copper refineries, coal combustion		0.0005-0.077 mg/L	---	1 µg/L
Silver	Fungicides, medical and electrical waste, coal combustion, oil refining, seeding clouds to induce precipitation		0.0002-0.014 mg/L	---	0.1 µg/L
Thallium	Catalysts, dyes, imitation jewelry, pyrotechnics, electroplating, potash, smelting	Can be lethal through ingestion of doses from 6-40mg/kg. Can cause alopecia, neuropathy, neurological impairment.	0.001-0.014 mg/L	---	0.8 µg/L
Uranium	Uranium mill tailings and effluent, stack emissions	Radioactive substance.	---	33 µg/L	15 µg/L

Metal	Anthropogenic Sources [1, 2]	Human Health Concerns [2,3,4]	Stormwater Concentration Ranges, from Literature [2]	FAL Guideline [3]	
				Short term	Long term
Zinc	Tire wear, brake pads, corrosion of metal, combustion of lubricants	Causes anemia in mammals by interfering with utilization of copper and iron	0.0007-22 mg/L	---	30 µg/L

Vaccari *et al.*(2006)[1], Makepeace *et al.*(1995)[2], CCME (1999)[3],USEPA (2004)[4]