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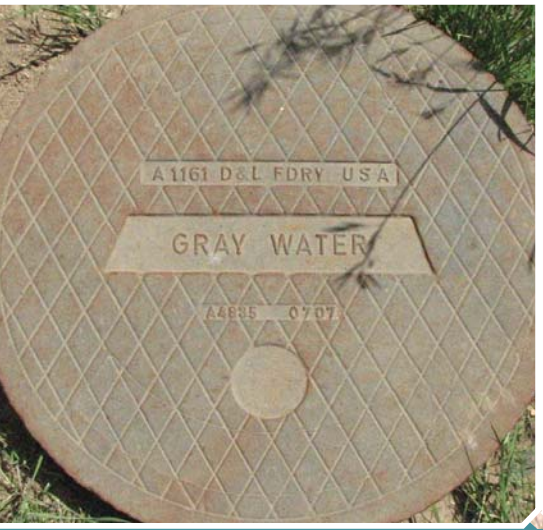
Graywater Treatment Using Constructed Wetlands



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Graywater Treatment Using Constructed Wetlands

by

Adam Jokerst[1], Meg Hollowed[1], Sybil Sharvelle[1], Larry Roesner (retired)[1], A. Charles Rowney[2]

[1]The Urban Water Center
Department of Civil and Environmental Engineering
Colorado State University
Fort Collins, CO 80521-1372

[2]ACR, LLC
Longwood, Florida, 32750

Contract No. 8C-R056NTSX

Project Officer
Thomas P. O'Connor

Technical Advisor
Richard Field (retired)

Urban Watershed Management Branch
Water Supply & Water Resources Division
National Risk Management Research Laboratory

NATIONAL RISK MANAGEMENT RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
CINCINNATI, OH 45268

Notice

Abstract

Mounting pressure to conserve water resources has prompted the notion that the separation of graywater from sewerage through the use of dual-plumbed systems may enable graywater to be reused at the household level for such non-potable demands as landscape irrigation or toilet flushing. Although graywater reuse holds great promise as a means to reduce potable water demands, water quality and health concerns arising from the level of contamination typically found in graywater have impeded the practice from gaining wide-scale application in the United States. Simple treatment schemes, such as those using constructed wetlands, could reduce graywater pollution and allow for expanded reuse applications. This report documents the results of a study examining the treatment efficiency of a pilot-scale constructed wetland system for graywater over a one-year period, and explores the potential for onsite reuse of the treated effluent for outdoor irrigation. The wide range of measurements taken, including those of anionic surfactants and of indicator microorganisms, provides a comprehensive assessment of the treatment efficiency and outflow water quality characteristics of such systems.

Results of water quality measurements indicated that the constructed wetland substantially reduced organics, solids, nutrients, pathogens, and surfactants throughout the one-year sampling period. In particular, removal rates of biochemical oxygen demand and total suspended solids averaged 91% and 77% respectively, while removal of anionic surfactants averaged 94% and never dropped below seasonal mean of 88% throughout the year. The wetland reduced pathogenic indicator microorganism concentrations by approximately two orders of magnitude on average, producing effluent concentrations below primary contact standards for all seasons except winter. A comparison of the wetland effluent quality with state reclaimed water quality regulations indicated that effluent would typically meet reclaimed water quality standards for restricted irrigation reuse. However, treatment efficiencies decreased precipitously during winter months, producing an effluent likely unsuitable for unrestricted reuse.

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Acronyms and Abbreviations

ABS	Alkyl benzene sulfonate
ACB	Atmospheric Chemistry Building
APHA	American Public Health Association
ANOVA	Analysis of Variance
BDL	Below detection limit
BMP	Best Management Practice
BOD	Biochemical oxygen demand
CoAgMet	Colorado Agricultural Meteorological Network
CDPHE	Colorado Department of Public Health and Environment
Cl^-	Chloride
CSU	Colorado State University
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DTN	Dissolved total nitrogen
E. coli	*Escherichia coli*
EPA	Environmental Protection Agency
ES	Enzyme substrate
FCES	Florida Cooperative Extension Service
FWS	Free water surface wetland
HDPE	High density polyethylene
HRT	Hydraulic retention time
IBR	Increasing block rate
I/I	Inflow/infiltration
LAS	Linear alkyl benzene sulfonate
MBAS	Methylene blue active substances
MBR	Membrane bioreactor
MF	Membrane filtration
NH_3	Ammonia
NO_3^-	Nitrate
ORD	Office of Research and Development
O&M	Operation and maintenance
PO_4^{3-}	Phosphate
PVC	Polyvinyl chloride
QAPP	Quality assurance project plan
QA/QC	Quality assurance/quality control
SC	Specific conductivity
SF	Subsurface flow wetland
SM	Standard Methods
SO_4^{2-}	Sulfate
Temp	Temperature

TDS	Total dissolved solid
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorus
TS	Total solids
TSS	Total suspended solid
TVS	Total volatile solid
U.S.	United States
VSS	Volatile suspended solid
WWF	Wet-weather Flows
WWTP	Wastewater treatment plant
s	standard of deviation

Acknowledgements

An undertaking of this type requires the dedication and cooperation of a team. The technical direction and coordination for this project was provided by the technical project team of the Urban Watershed Management Branch (UWMB), under the direction of Mr. Thomas P. O'Connor, the Project Officer and Mr. Richard Field, the Technical Advisor (retired). Ms. Carolyn Esposito of UWMB reviewed the quality assurance project plan and the report as well. Mr. Anthony Tafuri of UWMB, Mr. Robert Goo of EPA's Office of Water, and Dr. Francis Digiano of the University of North Carolina provided technical reviews. Yongdeok Cho contributed to early versions of the design of the constructed wetlands. Finally, the contributions of the many authors and professionals who were contacted or cited in this work are acknowledged, as it is their efforts that underlay the discussion and advances contained in the report.

Executive Summary

Mounting pressure to conserve water resources has prompted the notion that the separation of graywater from sewer effluents through the use of dual-plumbed systems may enable graywater to be reused at the household level for such non-potable demands as landscape irrigation or toilet flushing. Although graywater reuse holds great promise as a means to reduce potable water demands, water quality and health concerns arising from the level of contamination typically found in graywater have impeded the practice from gaining wide-scale application in the United States. Simple treatment schemes, such as those using constructed wetlands, could reduce graywater pollution and allow for more reuse applications.

This report documents the results of a study examining the treatment potential of constructed wetlands on graywater. The specific objectives of this research were to demonstrate the treatment efficiency of wetlands on graywater, determine whether wetland treatment produced water suitable for reuse, and assess the primary contact risks of the graywater in the wetland and treated effluent. A constructed graywater wetland scheme featuring a free water surface (FWS) bed followed in series by a subsurface flow (SF) bed was built and investigated on the campus of Colorado State University (CSU). Built at the pilot-scale, the wetland was evaluated using graywater, i.e., hand wash and shower water only, over a one year period. Water quality samples were taken from three locations in the wetland system: influent, effluent of the FWS, and effluent from the SF. Samples were collected and analyzed approximately every three weeks following the procedure given in the quality assurance project plan (QAPP) approved by the EPA prior to sampling. Samples were analyzed for a wide range of water quality parameters including organics, solids, nutrients, indicator microorganisms, and surfactants. Additionally, detailed assessments of flow and hydraulic retention times (HRT) were conducted to evaluate the hydraulics of the wetland. The wide range of measurements taken provides a comprehensive assessment of the treatment of graywater via wetland systems.

Results of water quality measurements showed that the wetland substantially reduced organics, solids, nutrients, pathogens, and surfactants throughout the sampling period. A majority of the removal for all constituents was found to occur in the FWS; however, the effluent quality of the FWS suffered from considerable algae growth in the open water which maintained elevated levels of TSS and turbidity. Further conditioning in the SF removed most of these solids. Flow monitoring through the wetland indicated significant flow losses during warm months resulting from evapotranspiration losses, which decreased the amount of effluent available for reuse. Throughout the experiment, no odor issues were noticed and no detrimental effects on plant growth from using graywater were observed.

By way of this research, constructed wetlands were shown to be a viable means to condition graywater for reuse. In particular, removal rates of biochemical oxygen demand and total suspended solids averaged 91% and 77% respectively, while removal of anionic surfactants averaged 94% and did not drop below seasonal mean of 88% throughout the year. The wetland reduced pathogenic indicator microorganism concentrations by approximately two orders of magnitude on average, producing effluent concentrations below primary contact standards for all seasons except winter. A comparison of the wetland effluent quality with Colorado state reclaimed water quality regulations showed that effluent would typically meet reclaimed water quality standards for restricted irrigation reuse although effluent would not meet standards of same states that had stricter microbial regulations. In addition, given the limited assessment performed during this study, as compared to EPA regulations governing recreational swimming waters, the graywater wetland effluent contained adequately low number of select bacteria but was not considered to have an average of non-detect as required; as such greywater would potentially exceed national recreational contract standards.

In particular, winter removal efficiencies were lowest of any season, producing effluent with *Escherichia coli*, biochemical oxygen demand, and turbidity levels above EPA recommended reclaimed water standards. The wetland effluent would not meet unrestricted irrigation standards consistently throughout the year. If unrestricted landscape irrigation is desired, the wetland effluent would require additional treatment including disinfection prior to reuse during the winter. However, if the wetland's effluent is intended for irrigation purposes, typically irrigation would not be carried out during the winter season and the potential for primary contact would be minimized.

Whether or not the above reclaimed water regulations truly represent the acceptable risk to humans and the environment from graywater reuse remains a matter of further investigation. Additional microbial analyses could better reveal the true pathogenic risks. It is believed that the scale and duration of this evaluation, along with the wide range of measurements taken, could promote additional research and application of wetland treatment of graywater.

Chapter 1 Introduction

This report documents the results of a study examining the efficiency of a pilot-scale constructed wetland system for treating graywater over a one year period, and explores the potential for onsite reuse of the treated effluent for outdoor irrigation. The wide range of measurements taken, including those of anionic surfactants and pathogenic indicator microorganisms, provides a comprehensive assessment of the treatment efficiency and outflow water quality characteristics of such systems.

Overview

Mounting pressure to conserve water resources has prompted the notion that the separation of graywater (all wastewater not including toilet and kitchen sources) from sewer effluents through the use of dual-plumbed systems may enable it to be reused at the household level for such non-potable demands as landscape irrigation or toilet flushing. Estimates of graywater generation rates vary, with several studies reporting as much as 50-80% of residential wastewater being graywater (Gross et al., 2007a). Past studies have shown significant reductions in water demands, wastewater generation rates, and associated costs resulting from graywater reuse. Such potential water and cost savings have prompted several European and Asian countries to adapt graywater reuse on a wide scale basis. However, in the United States (U.S.), water quality and health concerns arising from the level of contamination typically found in graywater have impeded graywater reuse from gaining wide-scale application. In particular, levels of solids and organics (e.g. Eriksson et al., 2002) as well as pathogens (Rose et al., 1991) in graywater have been shown to regularly exceed reuse and primary contact regulations.

Simple treatment schemes, such as those using constructed wetlands, could reduce graywater pollution and allow for more reuse applications. The efficacy and the design of wetlands for municipal wastewater streams have been documented by EPA (EPA 2000a and 2000b). Wetland systems can take on various designs, with two of the most common being the free water surface (FWS) and subsurface flow (SF) types. FWS wetlands feature open water similar to natural wetlands, while SF wetlands contain a matrix of sand or gravel through which water flow and in which vegetation is rooted (EPA, 2000a). The use of constructed wetlands for treating graywater has had few applications (and none in the U.S.). Past studies involving treatment of graywater using various wetland configurations have shown promising pollution removal efficiencies (Dallas and Ho, 2005; Winward et al, 2008b; Gross et al., 2007b); however, these wetland treatment systems were small, highly controlled, and operated over short periods of time. Specifically, the study by Winward et al. (2008a) featured horizontal wetlands supplied with low-load graywater. These wetlands were quite small and operated in a greenhouse, but with only a 2.1 day hydraulic retention time (HRT) achieved mean biochemical oxygen demand (BOD_5) and total suspended solids (TSS) removal rates of 90% and 63% respectively. Other published graywater wetland treatment schemes have been designed on a similarly small scale as above, and information is lacking on the long term performance of such systems at the pilot scale. In particular, no studies have examined the treatment effects of combining FWS and SF systems.

This report details a novel constructed graywater wetland treatment scheme involving both a FWS and SF in tandem to treat graywater from a building. Built at the pilot-scale, the wetland was evaluated using low-load graywater from a Colorado State University (CSU) building with a design graywater flow of 330 L/d. Water quality samples were taken from three locations in the wetland approximately every three weeks and were analyzed for temperature, pH, turbidity, dissolved oxygen (DO), specific conductivity (SC), total organic carbon (TOC), BOD_5, total nitrogen (TN), dissolved total nitrogen (DTN), ammonia (NH_3), nitrate (NO_3^-), total phosphorus (TP), phosphate (PO_4^{3-}), chloride (Cl^-), sulfate (SO_4^{2-}), *Escherichia coli* (*E. coli*), TSS, total dissolved solids (TDS), total volatile solids (TVS), volatile suspended solids (VSS), and anionic surfactants. Additionally, detailed assessments of flow and HRT were conducted to evaluate the hydraulics of the wetland. The above water quality parameters were chosen as they represent a detailed assessment of the overall effluent water quality, specifically for irrigation reuse applications. In its current configuration, due to local regulations, the graywater wetland effluent is not reused; however, there is potential to use the effluent in the future for landscape irrigation on the grounds of CSU. The treatment system here could also be modified or added to if other end uses of the graywater, e.g. toilet flushing, were desired.

Objectives

The objectives of this research were to demonstrate the treatment potential of wetlands on graywater, determine whether wetland treatment produced water suitable for reuse, particularly as irrigation water, and assess the primary contact risks of the graywater in the wetland and of the treated effluent.

Chapter 2 Conclusions and Recommendations

Wetland Performance

Results of water quality measurements showed that the wetland substantially reduced organics, solids, nutrients, pathogens, and surfactants throughout the sampling period. In particular, removal rates of BOD_5 and TSS averaged 91% and 77%, respectively, while removal of anionic surfactants averaged approximately 88% or more throughout the year. Removal rates of nutrients, i.e., TN and TP, were 84% and 77%, respectively.

The majority of the removal for all constituents was found to occur in the FWS, however, the effluent quality of the FWS suffered from considerable algae growth in the open water of this system. These algae contributed elevated levels of TSS and turbidity. Further treatment in the SF removed most of these solids in the spring months. Through the summer months TSS and turbidity levels were lower in the FWS effluent but these increased in the SF effluent, likely a result of biofilm formation on gravel matrix in the SF. Increased vegetation density in the FWS during the first summer may have reduced the high algae growth, due to shading of open water. Future spring seasons may not have as high TSS effluent conditions, as this was the first season monitored and the system was still being established.

This research demonstrated that constructed wetlands may be a viable means to condition graywater for reuse. A comparison of the wetland effluent quality with state reclaimed water quality regulations showed that effluent would typically meet reclaimed water quality standards for restricted irrigation reuse. Also, in comparison to EPA regulations governing primary contact, i.e., recreational swimming waters, the graywater wetland effluent contained sufficiently low values of select bacteria for all seasons except winter. The wetland's treated effluent was intended for irrigation purposes, which would not be carried out during the winter, reducing the potential for primary contact. Unfortunately, during the summer months when irrigation water would be most beneficial, little effluent was observed beyond the FWS cell. In this particular study, this is attributed to the following three factors, the first two of which are attributable to the arid climate: high evapotranspiration rates, limited direct rainfall and decreased production of graywater.

The wetland effluent did not meet unrestricted standards consistently throughout the year. In particular, winter removal efficiencies were lowest of any season, producing effluent with *E. coli*, BOD_5, and turbidity levels above both EPA recommended reclaimed water guidelines and primary contact regulations. If unrestricted urban landscape irrigation is desired, the effluent would require additional treatment including disinfection prior to reuse in winter months. If the effluent were to be collected indoor use, i.e. toilet flushing, additional treatment including disinfection would also be necessary. Whether or not the current reclaimed water guidelines truly represent the acceptable risk to humans and the environment from graywater reuse remains a matter of further investigation. The main pathway to infection from any wastewater remains "fecal-oral" transmission (EPA, 2004) though skin contact and inhalation are other vectors for disease transmission. As graywater systems separate out black water, the primary source of fecal material is not present in this treated wastewater. Reducing risk of disease transmission by graywater systems may be as simples as prohibiting spray irrigation system by using drip systems only, as aerosols from spray irrigation may

contain pathogens (viruses and bacteria – EPA, 2004), requiring safety procedure for those handling systems, i.e., wearing gloves and washing hands after coming in contact with the gray water system components, or incorporating disinfection treatment, as necessary. Many such safety procedures have been incorporated into a Water Environment Research Foundation (WERF) Final Report (Bergdolt et al., 2011) entitled "Guidance Manual of Graywater from Blackwater for Graywater Reuse.

Further microbial analyses could potentially better reveal the true pathogenic risks. It is believed that the scale and duration of this evaluation, along with the wide range of measurements taken, could promote further research and/or application of wetland treatment of graywater.

Public Acceptance

It is surmised that public acceptance of wetland treatment of graywater hinges primarily on public safety and aesthetics. Public safety risks, primarily through contact with graywater, are inherent unless access to wetland water is restricted. As in the case of SF wetlands, public access is easily restricted without any perimeter barriers which could be considered more appealing in urban areas. Aesthetically, wetland treatment is considered highly pleasing as a BMP for WWF, and wetland treatment of graywater should be no different. Throughout the experiment, no odor issues were noticed and no detrimental effects on plant growth from using graywater were observed. In fact, both the cattail and bulrush appeared to grow more quickly after irrigated with graywater shortly after vegetation establishment. Only after winter ice melted was the FWS considered mildly unattractive when algal growth in the FWS was prolific and formed thick bloom mats. However, algal blooms lasted only a week, quickly dissipating with warming temperatures.

Several animal and bird species frequented the wetland, which added to the overall aesthetic appeal. Such multiple benefits, i.e., water treatment, animal habitat, and green space, should increase the public acceptance of graywater wetlands. Additionally, such treatment is typically seen as more sustainable than traditional wastewater management methods, which may also increase public approval. As with any open treatment system, it should be noted that animals and birds are potentially additional vectors for nutrient and pathogenic indicators.

Recommendations for Future Study

Data in this report represents one year of data collection from the pilot-constructed wetland facility. Long-term monitoring past the one year mark would further demonstrate the wetland treatment potential, especially since the sampling results presented were within the first year of wetland establishment. Other future research should include:

1. Further seasonal data collection to verify effects observed in this study.
2. Studying graywater wetlands in other climatic conditions.
3. Further monitoring of the salinity content of the effluent (as it was shown to increase through the wetland and salinity content is a concern for landscape irrigation waters).
4. Investigating the metals and trace contaminant content (particularly pharmaceuticals and personal care products) of the wetlands' plants, soils, and treated effluent.
5. Determining the optimal loading rates of the wetland to achieve specific end purposes, i.e., potentially decreasing winter hydraulic loading rates to meet unrestricted contact for irrigation waters and increasing summer hydraulic loadings to produce sufficient effluent for irrigation.
6. Examining in detail the economics and public acceptance of using wetlands for graywater treatment as opposed to other treatment mechanisms.
7. Addressing the legal issues of reusing the wetland effluent both on local and state levels.
8. Investigating supplementary disinfection and/or secondary treatment, particularly during winter months or if the wetland effluent must be held to reclaimed water standards.
9. Investigating the full suite of pathogen indicator organisms and tracking specific viral pathogens to have a greater sense of the true risk of the discharge from graywater treatment systems.
10. If the purpose is to develop a source of unrestricted irrigation water for summer use, in arid areas, design modifications to the current system might be needed. This could entail adjusting sizing or flow through of the wetland system or seasonally adjusting sources of graywater need sources.

Chapter 3 Background

Graywater Definition, Quantity, and Reuse Savings

A general definition of graywater (also spelled greywater) is all wastewater not including contributions from toilets; however, in the U.S., this definition also excludes kitchen sources. Further restrictions on the definition of graywater can be made by more closely specifying the source of graywater; for example, Friedler (2004) and Winward et al. (2008a) define low-load graywater as coming from showers, baths, and sink faucet while excluding kitchen and laundry sources.

Estimates of graywater generation rates vary, with several studies reporting as much as 50-80% of residential wastewater being graywater (Gross et al., 2007a). An American Water Works Association sponsored study of 1,200 households across 14 North American cities found that on average over 50% of all residential indoor water demands generate graywater (Mayer et al., 1999). This same study showed that on average over 50% of all residential potable water is used for outdoor irrigation, the largest single residential demand. Considering this large demand and that the source for outdoor irrigation water need not be of the same quality as indoor potable water, reusing graywater for irrigation offers a substantial water savings. In addition to landscape irrigation, recycled graywater is a potential source for other demands not requiring potable source water, such as toilet flushing.

The degree to which graywater reuse can reduce household potable water demand is variable depending on geographical location, landscaping preferences, and personal water use habits. Karpiscak et al. (2001) and the City of Los Angeles (1992) among others have estimated that reusing graywater for outdoor irrigation and toilet flushing could reduce household water demand by 20-50%. In addition to reducing total water demand, recycling graywater reduces total wastewater generation rates, and is viewed as a much more efficient reuse alternative as compared with, for example, reusing wastewater effluent. This is because graywater is produced and used directly at the residence and requires very little infrastructure modifications or delivery costs.

Graywater Quality

Such potential water and cost savings have prompted several European and Asian countries to adapt graywater reuse on a wide-scale basis, and have provoked increasing interest in the practice in the U.S. However, many U.S. states do not advocate the practice due to environmental and public health concerns arising from the potential chemical and microbial content of graywater.

Physical and Chemical Properties of Graywater

The physical and chemical properties of graywater are highly variable depending on the source, and are influenced by many factors including the number of household occupants, types of cleaners and personal care products used, grooming and hygiene habits, and sink waste disposal practices (Eriksson et al., 2002). Concentration ranges for common water quality constituents compiled from three studies are listed in Table 3-1. The values presented for Eriksson et al. (2002) are for low-load graywater only derived from bathroom sinks, showers, and baths. The values

from Rose et al. (1991) are in reference to graywater not including kitchen sources composited in a storage tank. The values from Gross et al. (2007b) refer to graywater mixed artificially to replicate graywater from mixed sources.

Table 3-1. Graywater Quality Characteristics from Literature

Pollutant (mg/L)[1]	Source of Graywater		
	Bathroom[2]	Composite[3]	Composite[4]
pH	6.4 – 8.1	6.54	6.3 - 7.0
EC, uS cm^{-1}	82 - 250	–	1,000 - 1,300
BOD_5	76 - 200	–	280 - 688
TOC	30 - 104	–	–
TN	5 - 17	1.7	25 - 45
NH_3	< 0.1 - 15	0.74	0.1 - 0.5
NO_3^-	0.3 – 6.3	0.98	0 - 5.8
TP	0.1 - 2	–	17.2 - 27
PO_4^{3-}	0.9 - 49	9.3	–
SO_4^{2-}	–	22.9	–
Cl^-	9 - 18	9.0	–
TSS	54 - 200	–	85 – 285
TDS	137 - 1260	–	–
Turbidity, NTU	28 - 240	76.3	–
Anionic Surfactants	–	–	4.7 - 15.6

[1] All units in mg/L unless otherwise noted.
[2] Eriksson et al. (2002); values of TP and PO_4^{3-} taken from different studies of bathroom water.
[3] Rose et al. (1991).
[4] Gross et al. (2007b).

Microbial Properties of Graywater

The quantities of microbial indicators in graywater are highly variable, but generally have been shown to regularly exceed state water reuse or primary contact regulations (Rose et al., 1991; Cassanova et al., 2001). Commonly measured bacteria for indicating fecal contamination include total coliforms, fecal coliforms, *E. coli*, and enterococci. Of these, fecal coliforms and *E. coli* are the principle indicators of pathogenic enteric bacteria, while enterococci are often used to indicate fecal virus contamination. Traditionally, fecal coliform levels are used to determine the degree of fecal contamination; however, many states, including Colorado, have recently moved to adopt *E. coli* levels as the primary indicator (CDPHE, 2007). *E. coli* are thought to be entirely of fecal origin, as opposed to coliform bacteria which are known to grow in soil (Tchobanoglous and Burton, 1991), and therefore the substrate of the graywater system.

Concentrations of fecal indicators in stored graywater can be higher than freshly generated graywater, indicating that microbial growth occurs in storage tanks (Dixon et al., 1999). Rose et al. (1991) indicated that shower and bath wastewater are likely vectors of fecal contamination. A comparison of microbial levels in graywater of various sources from several studies is presented in Table 3-2.

Table 3-2. Microbial Quality of Graywater

Graywater Sources	Fecal coliforms (cells/100 mL)	*E. coli* (cells/100 mL)	Enterococci (cells/100 mL)	Reference
Composite	5.6×10^5	–	–	Casanova et al. (2001)
Composite	$1.8 \times 10^4 – 7.9 \times 10^6$	–	–	Rose et al. (1991)[1]
Composite	_	$10^0 – 2.4 \times 10^8$	$10^1 – 2.7 \times 10^5$	Eriksson et al. (2002)
Bath & shower	$10^0 – 3.3 \times 10^3$	$<10^2 – 2.8 \times 10^3$	_	Eriksson et al. (2002)
Bath & shower[1]	6×10^2	–	–	Surendran et al. (1998)
Composite	–	$<10^2 – 6.3 \times 10^3$	$<10^2 – 5.0 \times 10^3$	Winward et al. (2008b)

[1] From university residence halls

Surfactants

In addition to the aforementioned graywater quality constituents, recent interest has been paid to the effects of personal care products on the soils and plants receiving graywater irrigation. Surfactants represent one common class of graywater contamination arising from personal care products. These chemicals are found in most soaps and detergents, and are used to lower the surface tension of a liquid such to allow emulsification of hydrophobic compounds. Surfactants are classified by their ionic state: nonionic, cationic, anionic, and amphoteric. Many of the surfactants in common soaps and body cleansers fall in the anionic category, and include: sodium dodecyl sulfate, ammonium lauryl sulfate, linear alkyl benzene sulfonate (LAS).

While many of the above anionic surfactants are not directly toxic through contact or consumption, they may pose threats to human health via environmental accumulation or through the toxicity and xenobiotic characteristics of their degradation byproducts (Shcherbakova, 1999). Anionic surfactants have been shown to exist in graywater at elevated levels, with Gross et al. (2007b) reporting concentrations in the range of 4.7 – 15.6 mg/L.

Applicable Water Quality Standards

Although few water quality regulations exist specifically for onsite graywater reuse, urban water reuse standards – developed primarily for reclaimed wastewater effluent – provide a useful comparison by which to evaluate graywater quality. Several states have developed reuse regulations and/or guidelines specific to the end use of the reused water (Table 3-3). The EPA (2004) gives the following suggested guidelines for reclaimed water for such types of reuse as urban (e.g., all landscape irrigation, vehicle washing, toilet flushing, and other uses with unrestricted exposure) and recreational impoundments, i.e., for incidental and full body contact: pH = 6-9, $BOD_5 \leq 10$ mg/L, turbidity ≤ 2 NTU, and no detectable fecal coliforms per 100 mL of sample. EPA reuse guidelines given for other applications are less stringent, such as those for restricted or agricultural irrigation, industrial reuse, and landscape impoundments: pH = 6-9, $BOD_5 \leq 30$ mg/L, TSS ≤ 30 mg/L, and fecal coliforms ≤ 200 MPN/100 mL. However, there are stricter guidelines for indirect potable reuse and food crops not subject to commercial processing.

Table 3-3. State Reclaimed Water Quality Standards for Unrestricted Urban Reuse (EPA, 2004)

State	Arizona	California	Florida	Hawaii	Nevada	Texas	Washington
Treatment	Secondary treatment, filtration, & disinfection	Oxidized, coagulated, filtered, and disinfected	Secondary treatment, filtration, and high-level disinfection	Oxidized, filtered, and disinfected	Secondary treatment and disinfection	Not Specified	Oxidized, coagulated, filtered, and disinfected
Pollutant	**Water Quality Standard**						
BOD_5 (mg/L)	NS	NS	20 mg/L	NS	30 mg/L	5 mg/L[1]	30 mg/L
TSS (mg/L)	NS	NS	5.0 mg/L	NS	NS	NS	30 mg/L
Turbidity (NTU)	2 (mean)	2 (mean)	NS	2 (max)	NS	3	2 (mean)
	5 (max)	5 (max)					5 (max)
Total Coliform	NS	2.2/100 mL (mean)	NS	NS	NS	NS	NS
		23/100 mL (max in 30 days)					
Fecal Coliform	Non-detect (mean)	_	75% of samples below detection	2.2/100 mL (mean)	2.2/100 mL (mean)	20/100 mL (mean)	2.2/100 mL (mean)
	23/100 mL (max)	_	25/100 mL (max)	23/100 mL (max)	23/100 mL (max)	75/100 mL (max)	23/100 mL (max)

[1] This increases to 10 mg/L for landscape impoundments.
[2] NS – No Standard.

EPA primary contact regulations for fecal coliforms and *E. coli* are provided in Table 3-4. Reuse regulations specific to Colorado are based primarily on turbidity and *E. coli* numbers (CDPHE, 2008). Colorado specifies that *E. coli* concentrations be undetectable in over 75% of sample for unrestricted landscape irrigation and meet primary contact regulations (Table 3-4) for restricted reuse. The regulations limit turbidity to less than 3 NTU on a monthly average for unrestricted irrigation. No turbidity standards are given for restricted reuse; instead a TSS limit of 30 mg/L is applied.

Table 3-4. Pathogen Indicator Limits for Primary Contact in Freshwater (EPA, 1986)

Pathogen Indicator	30-day Geometric Mean, cells/100 mL	Single Sample Maximum 75th percentile, cells/100 mL[1]
Enterococci	200	375
E. coli	126	235
Fecal Coliform[2]	33	62

[1] For infrequent bathing.
[2] Extrapolated values based on equations presented in EPA (1986).

Aerosols from spray irrigation may contain viruses and bacteria which can come into contact with humans (EPA, 2004 and Bergdolt et al., 2011). The strict microbial standards presented in Table 3-4 for landscape irrigation would necessitate disinfection of recycled graywater, however if non-contact subsurface (drip) irrigation is performed, potential human contact would eliminated and these microbial standards could be viewed as overly cautious. Previous studies have suggested microbial regulations developed for primary contact recreational waters are applicable to recycled graywater since activities associated with primary contact recreational water and with graywater reuse both involve body exposure and accidental ingestion (Dixon et al., 1999).

Salinity, typically measured as SC, guidelines are not included in the above reuse guidelines. However, salinity is an important parameter to consider when using recycled water for irrigation since high salinity can reduce osmotic potential in plant cells, subsequently causing plant mortality. The Florida Cooperative Extension Service (FCES, 2004) recommended SC levels for irrigation water below 750 uS/cm.

Constructed Wetlands

Elevated pathogen indicator numbers in graywater make treatment desirable, if not necessary, for graywater reuse to gain wide-scale public acceptance and application. Many treatment systems have been proposed to condition graywater. For instance, Friedler et al. (2006) evaluated three technologies commonly used in municipal wastewater treatment on graywater: sand-bed filtration, membrane bioreactors (MBR), and biological contactors. These mechanical/biological treatment units showed effective graywater treatment; however, their application at a household level would require considerable time and large expenditures. Sand filtration units for instance require frequent back flushing, and MBRs have extensive operation and maintenance requirements. An alternative to mechanical treatment of graywater may be the use of constructed wetlands.

Constructed wetlands are treatment systems that replicate natural wetlands to improve water quality through physical, chemical, and biological treatment mechanisms (Kadlec and Knight, 1996), and are commonly employed to treat municipal wastewater (Tchobanoglous et al., 2003; EPA 2000a; and, Hammer, 1989). Not only can constructed wetlands provide important water quality improvements with low energy and maintenance requirements, wetlands can also offer pleasing aesthetics, ecological benefits, and wildlife habitat (EPA, 2000a).

Common Constructed Wetland Designs

Constructed wetlands can have a variety of designs, with two of the most common being the free water surface (FWS) and the subsurface flow (SF) types (Figure 3-1). FWS closely resemble natural wetlands in both appearance and function, and feature elements typical of a natural wetland such as emergent vegetation and open-water areas (EPA, 2000a). SF wetlands contain a bed of gravel or sand in which aquatic plants are rooted. Water passes through the matrix and root zone, but is not exposed to the surface; therefore water is not visible or accessible by humans or wildlife. Because of the large surface contact of the growth matrix, SF wetlands feature higher microbial reaction rates for most contaminants than FWS wetlands, which reduce the required footprints of SF wetland (EPA, 2000c). However, if loaded with high-solid content influent or if too fine a matrix is employed, SF wetlands may clog.

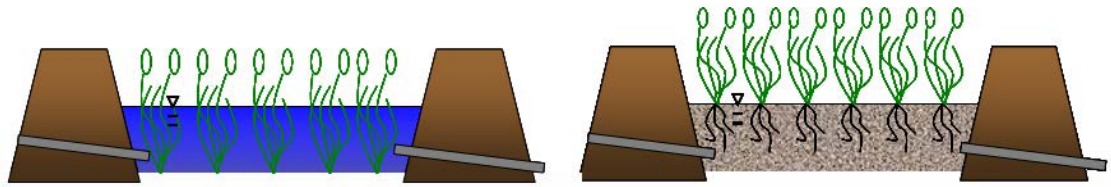


Figure 3-1. Typical free water surface (left) and subsurface flow (right) wetland configurations.

Wetland Treatment Mechanisms

Wetlands reduce pollutants via sedimentation, biodegradation, filtration, plant uptake and storage, and volatilization (EPA, 2000a). The primary variables influencing the degree to which the above wetland mechanisms treat process water are HRT and water temperature (Kadlec and Knight, 1996; Hammer, 1989). Increasing HRT results in a longer contact time and more efficient contaminant treatment, while lower water temperatures reduce reaction kinetics and subsequently treatment rates. Hence, it is generally expected that treatment efficiency is the lowest during winter months when temperatures are the coldest.

Microbial treatment in wetlands results primarily by settling, predation, solar irradiation, and temperature inactivation (Kadlec and Knight, 1996). Close correlations of TSS removal with pathogen (and indicator) removal have been reported (EPA, 2000a), with HRT being the primary factor influencing pathogen treatment efficiency. While

treatment rates of fecal pathogenic indicators in constructed wetlands vary widely by design, operation, and location, observed reductions of one to three orders of magnitude are common in wetlands treating municipal sewage.

Wetland Hydrology

Because the HRT of a wetland system is the preeminent criteria to determine treatment efficiency, a sound understanding of wetland hydrology is necessary for adequate design and operation of such systems. Wetland hydrology characterizes the pattern of the inflows and outflows through the wetland system. Inflows can include direct influent inputs, precipitation, runoff intrusions, and groundwater contributions. Outflows include effluent flows, evapotranspiration (ET), and losses to groundwater. In natural wetland systems, the balance of these inflows and outflows continually adjust the storage volume in the wetland. Constructed wetlands, however, are typically designed to maintain a fixed water depth and volume and to limit groundwater fluxes and runoff inflows. In this way, the HRT can be managed to achieve the desired level of treatment.

HRT is defined as the ratio of total void volume (or water volume) to inflow rate. Void volume is the part of the total volume available through which water may flow, and is simply the total wetland volume less any volume occupied by plants, water regulation structures, matrix (substrate) and piping. Void volume may be computed by multiplying the total volume by the porosity of the wetland. Wetland porosity varies by matrix, vegetation, and age, with more established wetlands having a lower porosity than recently established wetlands. FWS wetlands generally have greater porosities than SF wetlands due to the sand or gravel matrix in SF types. The EPA (2000a) reports FWS porosities between 0.65 and 0.95, with a value of 0.75 given for medium dense, fully established wetlands. Porosity values for SF wetlands mainly depend on the mean size of the gravel or sand growth matrix. Typical SF matrix porosity values range from 0.26 for course sand to 0.45 for course rock (EPA, 2000c).

Chapter 4 Materials and Methods

A pilot-scale constructed wetland system was built during the summer of 2007 at CSU. The system was built to receive and treat graywater from a campus building comprised of office and laboratory space. Upon completion of construction, the wetland was vegetated with wetland vegetation typical of the region. Supplemental graywater was provided from a residential dormitory and after reaching hydraulic equilibrium, a one-year study of the treatment effectiveness and effluent quality of the wetland was initiated to determine the suitability of the wetland effluent for reuse purposes. This chapter provides a description of the graywater wetland system, and describes the methods and materials used to monitor the physical, chemical, and microbial characteristics of the system.

Wetland Configuration

The graywater wetland was constructed at the base of a steep grade below the Atmospheric Chemistry Building (ACB) on the CSU Foothills Campus (Figure 4-1). Because of state and local permitting requirements, effluent from the graywater wetland was also routed back to the municipal sewer after passing through the wetland system.

In its current configuration, effluent from the wetland was not reused due to state regulations; however, there is future potential to use the effluent for landscape irrigation on the grounds of CSU. CSU currently uses non-potabale water for irrigation, and could supplement their irrigation water with the wetland effluent either directly or indirectly. If done directly, the effluent would be released to the surface where it would likely flow to the Colorado State Forest Service grounds (Figure 4-1) where it could be used for tree irrigation.

The graywater wetland consisted of two beds in series: a FWS bed followed by a SF bed (Figure 4-2). This arrangement was designed to promote settling of course materials in the FWS before reaching the SF, preventing clogging of the SF gravel matrix. Each wetland bed was constructed with distribution and collection headers to facilitate flow dispersion, denoted “e” in Figure 4-2, impermeable bottom liners to prevent groundwater flux, denoted “b”, and perimeter berms to prevent runoff intrusion and rip-rapped side slopes, denoted “c”. Overflow inlets, “f”, were provided in the event of the header blockage. Manholes were constructed at each inflow and outflow location, and contained piping to regulate water depth in the beds and flow meters, denoted “g”. Manholes were denoted numerically, with Manhole #1 being upstream of the wetland system, Manhole #2 located downstream of the FWS wetland and Manhole #3 located downstream of the SW wetland (Figure 4-2). The respective substrate for each wetland is denoted “d” and “h”, while “a” is a compositing tank.

A 300 L high density polyethylene (HDPE) storage tank, located upstream of the FWS and denoted by “a” in Figure 4-2, composited the inflow for sampling. The tank was sized to collect approximately a one day composite sample. Due to footprint and plumbing limitations, the final volume of the composite tank was 300 L.

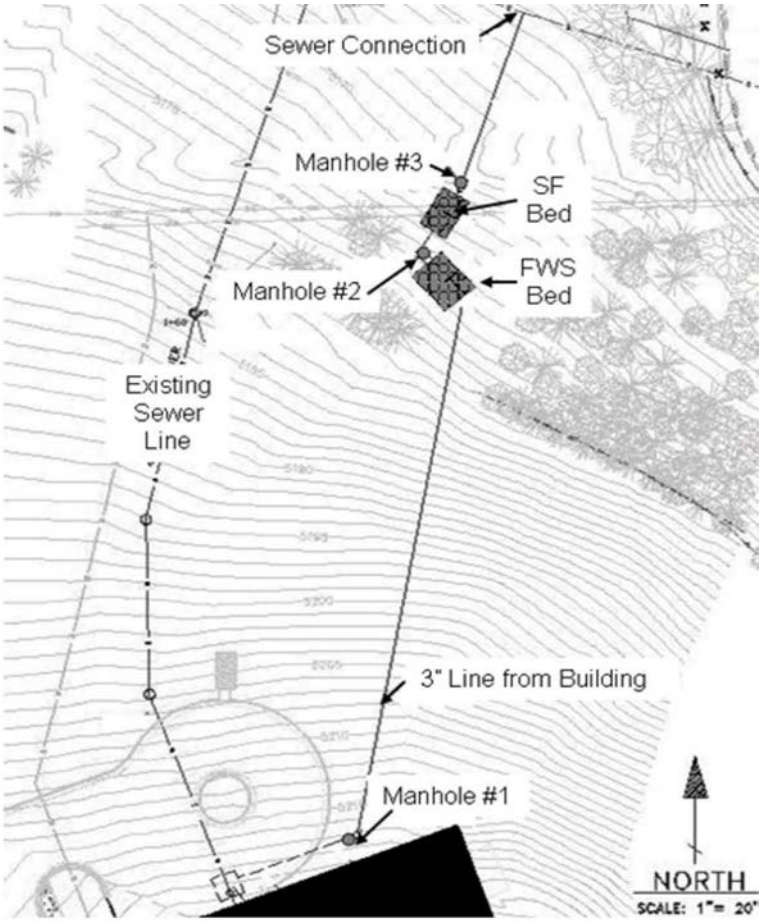


Figure 4-1. Graywater wetland location and layout. (Aerial photo © DigitalGlobe, 2009)

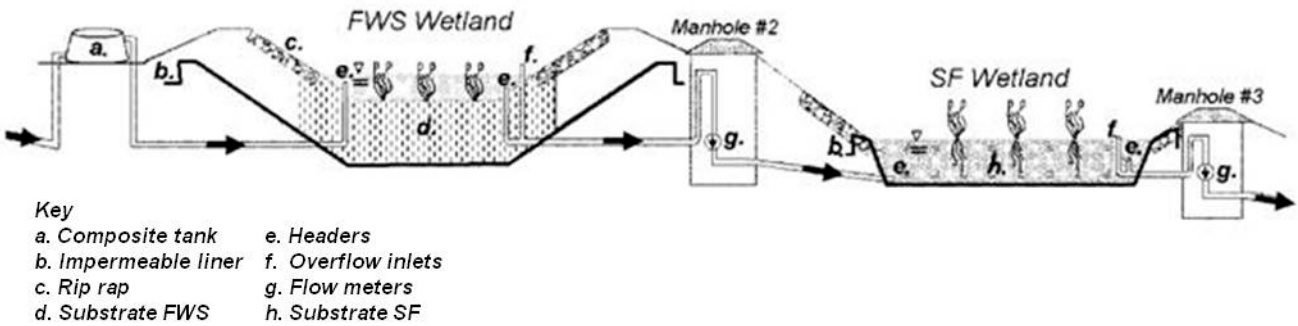


Figure 4-2. Graywater wetland configuration.

The size of the wetland was based on expected flow and pollutant loading rates. Graywater inflow rate was estimated assuming a building occupancy of 38 people and an average graywater generation rate of 5.7 L/cap/d giving a design graywater flow rate of 216 L/d. The design for the wetland anticipated removing over 99% of BOD_5 and TSS, even during winter months. Based on these criteria, an estimated flow (void) volume of 4.6 m^3 was specified resulting in a HRT of approximately 21 days.

Free Water Surface Design and Operation

The FWS bed measured 2.7 m by 3.7 m and maintained a water depth of approximately 0.3 m (Figure 4-3 and Figure 4-4). It was given a 2:1 (rise-to-run), rip-rapped side slope, and was planted with common cattails (*Typha latifolia*). The base of the FWS bed was filled with 0.9 m of amended soil, denoted letter "d" in Figure 4-2, consisting of approximately 50% native soil and 50% peat. The FWS volume was approximated at 2010 L assuming an overall vegetation porosity (η) of 0.8. The EPA (2000a) reports a porosity value of 0.75 for medium dense, fully established wetlands. Because it was uncertain how quickly the wetlands in this location would become fully established and porosity was important in computing effluent volumes and hydraulic retention times, the porosity of the FWS bed was computed by estimating that 80% of the bed was covered in medium-dense, established vegetation with a mean porosity of 0.75 (EPA, 2000a) and 20% of the FWS was open water ($\eta = 1.0$), leading to an overall weighted porosity of 0.8. The HRT of the FWS was nominally 9.3 d based on 216 L/d.

Figure 4-3. Freshwater surface wetland spring 2009.

Figure 4-4. Freshwater surface wetland fall 2009.

The cattails in the FWS bed were established in September 2007 at a density of one plant per 0.3 m^2. The cattails were irrigated with municipal water for the first 45 days after planting and afterwards with graywater. Cattails grew vigorously the first two months of establishment, growing approximately 0.35 m in the first 43 days after planting. After winter dormancy, the cattails resumed active growth in May, 2008, reaching a final mean height of 1.4 m by the fall of 2008. Plant heights were determined by measuring and averaging the height of 12 plants selected at random. The cattails also spread via rhizome migration to cover nearly 80% of the bottom of the FWS bed by fall 2008. By August 2009 the cattails were approximately 2 m tall and covered well over 80% of the bottom of the FWS bed.

Subsurface Flow Wetland Design and Operation

The SF bed was nominally 3.7 m by 4.6 m, and given a water depth of 0.6 m this provided a volume of 10.2 m^3. The SF was built with 2:1 rip-rapped side slopes (Figure 4-5 and Figure 4-6 and denoted by "h" in Figure 4-2). The SF was filled with cleaned, rounded native stone with an approximate mean diameter of 15 mm, and planted with hardstem bulrushes (*Scirpus acutus*). Porosity was physically measured by excavating the gravel and root matrix and placing in a bucket. The gravel and root matrix was allowed to drain and dry. Porosity was estimated by filling the bucket with a measured amount of water and then dividing by the total bucket volume (11.4 L). The procedure was completed for three locations in the wetland selected at random (Table 4-1). SF void volume was approximately 2610 L considering an average porosity of 0.29 for the gravel and established root matrix. The HRT of the SF was nominally 12.1 d based on 216 L/d.

Figure 4-5. Subsurface wetland spring 2009.

Figure 4-6. Subsurface wetland fall 2009.

Table 4-1. Porosity Determination of the Subsurface Flow Wetland

Locations	Total Volume (L)	Void Volume (L)	Porosity
1	11.4	4.3	0.38
2	11.4	2.5	0.22
3	11.4	3.0	0.26
Mean		**3.3**	**0.29**

Bulrushes in the SF bed were first established in September 2007 at a density of one plant per 0.3 m^2. Because they were planted late in the growing season and experienced an unusually cold winter in 2007/2008, the bulrushes suffered 100% mortality and were replanted in May, 2008. The bulrushes were watered with tap water for approximately 120 days after planting in 2008. The great length of time during which the SF was watered with tap water was due to a lack of graywater from the ACB as described under the heading "Graywater Sources". After supplemental graywater was acquired from a CSU dormitory, the bulrushes were switched to graywater irrigation. Because they were rooted in cleaned gravel void of any nutrients and irrigated with tap water, supplemental nutrients were supplied to the bulrush for approximately 90 days following planting in May, 2008. Soluble garden fertilizer was provided at a rate of 0.25 kg/wk in the following ratio: 24% N; 8% P; and 16% K. After this establishment period, nutrient application ceased 30 days prior to the introduction of graywater to allow the system to equilibrate. Very little mortality was observed over the winter of 2008/2009, and by spring 2009, the bulrushes had spread to occupy approximately 50% of the SF area.

Graywater Sources

Low-load graywater was provided to the wetland primarily from two sources: lavatory sink water from the dual-plumbed ACB, and lavatory sink and shower water from a residential dormitory.

Atmospheric Chemistry Building Graywater

The ACB was built with separate wastewater plumbing to separate blackwater (i.e., toilet water), lab water and graywater from lavatory sinks. Blackwater and lab water were routed directly to the municipal sewer, while ACB graywater was collected from sinks in four lavatories, which were plumbed directly to the wetland. Initial flow measurements during summer, 2008 showed an average graywater flow rate of 59 L/d being produced in the ACB, far less than the peak 216 L/d that was anticipated. Graywater flow rates were lower than expected because the building was not fully occupied. The estimated occupancy throughout the experiment was 20 persons as compared to the expected 38. Graywater production in the ACB occurred almost entirely during weekdays, with nearly no flow observed during weekends. Such low flow rates were less than the daily seepage and ET rates observed in the FWS, which prohibited any flow from reaching the SF wetland and necessitated that supplemental graywater be provided.

Dormitory Graywater and Transfer System

Supplemental graywater was provided from a residential dormitory at CSU. Dormitory graywater was collected from sinks from three communal lavatories serving approximately 72 residents and from showers serving approximately 24 residents. Graywater was collected in the basement of the dormitory in a 1,100 L HDPE tank (Figure 4-7), where it was stored until transfer. Transfer and hauling to the wetland site was conducted every three days via a 1,800 L stainless steel, trailer-mounted tank (Figure 4-8). At the wetland site, the hauled graywater was dispensed using a variable speed, peristaltic metering pump (Masterflex VA10, Vernon Hills, Illinois) into Manhole #1 where it was mixed with the ACB graywater before gravity flowing to the wetland system. Dormitory graywater was hauled to the wetland twice a week during the experiment, restricting the average residence time of the water in the trailer-mounted tank to under approximately 3.5 days.

The metering pump was housed in an insulated pump box (Figure 4-9). In the pump box, approximately 3.5 m of 6.5 mm diameter tubing was coiled around a 75 W heat source (light bulb) to prevent freezing during transfer between the pump box and the manhole in the winter. The pump box heat source was wired through a household thermostat (set at 27 ˚C) to prevent overheating and conserve electricity. Other freeze prevention measures included insulating all exposed piping between the tank and manhole and installing a 500 W submersible heater in the trailer-mounted tank. The submersible heater was also wired through a thermostat which activated the heater only when the water temperature dropped below 1.6 ˚C and deactivated the heater when the water reached 10 ˚C. During non-winter months the tank heater, along with the pump box heat source and insulation, were removed.

Figure 4-7. Graywater storage tank, pump, and piping system used to collect graywater from the dormitory.

With the addition of supplemental dormitory graywater, average inflow rates were set to approximate the low-load graywater flow estimated for a single-family household. Although the wetland was not explicitly designed for residential use, there exists great potential to transfer the current design to residential applications. Thus, since hauling water from the dormitory allowed for a specific inflow rate to be maintained, that flow rate was set approximately to the expected flow from a single-family household. The desired influent rate was computed using the per capita water uses presented in Mayer et al. (1999), which established an average indoor water demand of 262 L/d/person. Without considering the demands of toilets, dishwashers, and cloth washers, indoor demand was reduced to 132 L/d/person. This portion was considered the quantity of indoor demand which generates low-load graywater. Multiplying the low-load graywater generation rate by an average population per household of 2.6 (U.S. Census, 2000) gave an influent rate of 343 L/d.

Figure 4-8. Trailer-mounted tank used to haul dormitory graywater to the wetland system.

Figure 4-9. Metering pump in insulated box used to transfer dormitory graywater from trailer-mounted tank to wetland system.

Influent Composition

The graywater inflow to the wetland varied by season due in part to differing compositing methods. During winter months (December through February), only dormitory graywater was supplied to wetland, as the 300 L composite tank and ACB source water was taken off line to prevent it from freezing, and inflow samples were instead drawn directly from the heated, trailer-mounted tank. It should also be noted that during January, 2009, graywater was provided from a single-family, residential household for approximately 12 days when dormitory graywater became

unavailable during the CSU winter break. In all other seasons, the 300 L composite tank was utilized, and both dormitory and ACB graywater were mixed and provided to the wetland.

During the summer of 2009, a large quantity of rainfall was followed by a long period of high temperatures and increased evapotranspiration. The result was an initial period of high flows, followed by nearly no flow the remainder of the summer to the second cell despite supplementing dormitory and the ACB influent with residential water in an attempt to overcome high ET rates. Higher graywater flow rates from the ACB were observed during the spring season than initially observed in the fall season. Thus, less dormitory graywater was used during the spring season than the fall season to achieve the same composite flow rate.

Flow Monitoring

Flow meters were installed in each manhole, and provided measurements of the inflow and outflow rates of each wetland. The inflow rate, measured in Manhole #1 (Figure 4-1), was taken downstream of the point where graywater from the ACB and dormitory was mixed. Flow rates throughout the system were highly variable, and were observed to range from less than 4 L/d when the ACB was unoccupied and no dormitory graywater was being dosed, to over 1,000 L/hr during periods of intense precipitation. To accommodate a wide range of flows, as well as high solid contents, a dosing flow meter system was devised.

Flow Meter Design

In each manhole, flow collected in a 5.7 L plastic dosing vessel, denoted by "A" in Figure 4-10. When full, a multi-level float switch activated a submersible pump, "B" and "C", respectively, which pumped the flow out of the dosing vessel and through a 20-mm diameter turbine flow meter (Great Plain Industries TM100-N, Wichita, Kansas), denoted "D" in Figure 4-10. A course, 1-mm inline strainer, marked "E" in Figure 4-10, was inserted upstream of each turbine meter to prevent turbine damage. A second float switch, denoted "F", was installed on the dosing vessel, which activated a counter (Red Lion Controls CUB-1, York, Pennsylvania) each time the vessel filled and emptied. This counter provided measurement redundancy since the approximate volume pumped per dose remained fairly constant throughout the experiment.

The turbine meter was orientated vertically to prevent water from standing in the turbine casing as stagnant water promoted biofilm growth on the turbines, which in turn affected meter accuracy. In fact, all plumbing in the manholes was done in a manner to minimize any standing water throughout the system. Straight pipe runs of 200 mm (10 times the inside diameter of the turbine meter) were installed both up- and down-stream of each turbine meter per manufacturer's instructions to measure flow more precisely.

Flow metering systems were self-contained, and powered by 12 V batteries. Since they were located in damp manholes, all electronics were sealed and isolated. An overflow in each dosing vessel was provided in the event a battery died or a pipe obstruction occurred. Vents and check valves were also provided in the metering system to prevent siphoning and backflow, respectfully. Batteries required replacement and recharging approximately every 5,500 to 7,500 L of transferred water or 14 days, whichever came first. Although larger batteries would have allowed for fewer recharges, size and weight limitations were imposed on the batteries, since they were frequently lowered into relatively deep manholes.

Flow Rate Determination

The turbine flow meters gave cumulative or totalized volume readings, which were recorded approximately three times a week throughout the experiment. Cumulative volumes were divided by the length of time between readings to yield averaged flow rates as given in Equation 4-1.

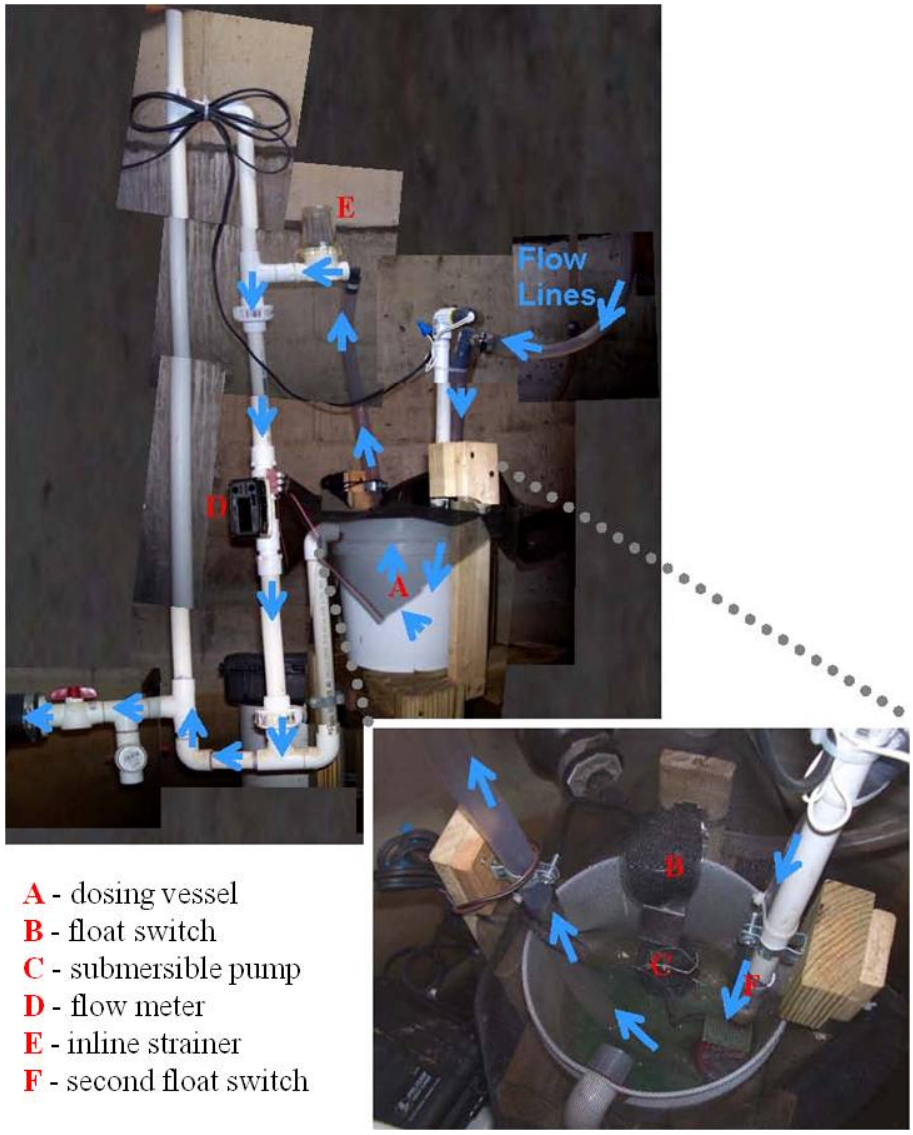


Figure 4-10. Flow meter design.

$$\overline{Q}_j = \frac{V_j - V_{j-1}}{t_j - t_{j-1}}$$ Equation 4-1

Where: t_j = the time that the flow reading was taken (hr)
t_{j-1} = the time that the previous flow measurement was made (hr)
V_j = the cumulative volume read from the flow meter at t_j (L)
V_{j-1} = the cumulative volume read from the flow meter at t_{j-1} (L)
= the mean flow rate between t_{j-1} and t_j (L/hr)

The cumulative volumes were also divided by the number of counts (one count equaled one dose of the vessel) to yield an average volume/dose. This value was observed to remain fairly constant, and was therefore used to alert any potential malfunction of the flow metering system. Malfunctions typically arose from biofilm growth on the turbine meter impellors, which required that the meters be cleaned and recalibrated approximately every 8 weeks.

Water Quality Analysis

Water quality sampling was conducted to evaluate the treatment efficiency and effluent concentration of the wetland for common organic, nutrient, solid, pathogen, and surfactant constituents. The treatment effectiveness and effluent quality of each wetland bed and the system as a whole was determined through the implementation of a systematic sampling plan. This sampling plan was developed within a quality assurance project plan (QAPP) prepared for and approved by the EPA. For sake of review, a generalized overview of the sampling plan is provided in this section, along with sampling location and frequency and data analysis methods.

Sampling Locations

Water quality measurements were made at three locations (Figure 4-11):

1. The influent, or inflow to the FWS system
2. The FWS effluent, or between the FWS and SF
3. The SF effluent, or SF system outflow

These locations were chosen because they represented the inflows and outflows of each wetland bed, thereby allowing evaluation of each bed independently and the system as a whole. During spring and fall months, influent grab samples were taken from the 300 L composite tank and represented approximately a daily composite. During winter months, influent samples were drawn directly from the 1,800 L trailer-mounted tank and represented approximately a 72 hour composite. FWS and SF effluent samples were drawn from the dosing vessels of the flow meter in manhole #2 and #3 respectfully. Effluent samples from the FWS and SF were not composited under the assumption that appropriate mixing occurred in each bed prior to outflow.

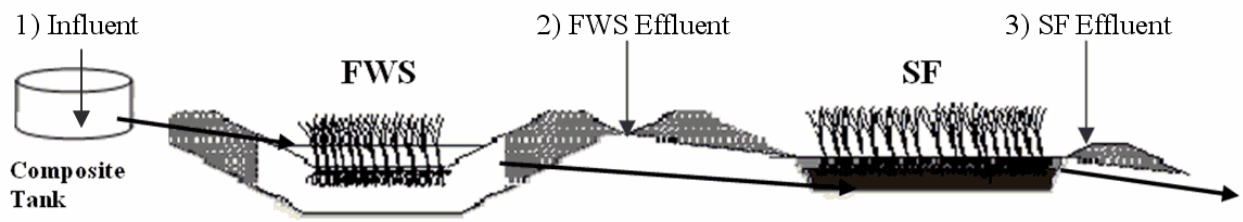


Figure 4-11. Water quality sampling locations.

Analyzed Parameters

Samples were analyzed for temperature, pH, turbidity, DO, SC, TOC, BOD_5, TN, NH_3, NO_3^-, TP, PO_4^{3-}, Cl^-, SO_4^{2-}, *E. coli*, TSS, TDS, TVS, VSS, and anionic surfactants. These parameters were chosen as they represent a detailed assessment of the overall effluent water quality, specifically for irrigation reuse applications. Also, the wide range of constituents measured gave a good assessment of the treatment efficiency of the graywater wetland on organics, nutrients, solids, and pathogens.

Physio-chemical Parameters: Turbidity, pH, DO, and SC were measured since all are common metrics of water quality and these measurements relatively inexpensive and readily available. Both pH and turbidity are standards given in the urban reuse regulations (EPA, 2004) and SC provides a good indication of water salinity, which is valuable to know for irrigation purposes given that some plants have low salt tolerances.

Organics: Both BOD_5 and TOC are a measure of the organic loading in wastewater, and are used to indicate the overall water quality biotic suitability of environmental systems. Specifically, BOD_5 is often cited as a water quality standard in regulations.

Nutrients: Both TN and TP were chosen for measurement since they provide cumulative macronutrient contents. In addition, NH_3 and NO_3^- were chosen since they typically represent the bulk of soluble nitrogen in environmental waters. Initially, NO_2^- was also measured, but was discontinued due to concentrations consistently below detection limits.

Solids: Solids were measured as they can negatively affect water quality and reuse applications. Initially, only TSS was analyzed since it is this category of solids that typically clogs irrigation systems. Later, TDS analysis was added to verify SC values. Algal growth can be assessed by VSS which is a simple addition to the TSS and TDS analyses.

Surfactants and Anions: Surfactants were measured out of concern of bioaccumulation and toxicity of degradation byproducts and since little prior research has examined their degradation in graywater through constructed wetlands. Specifically, anionic surfactants were examined as they are the most commonly found surfactants in body soaps and other personal care products. The levels of Cl^-, a conservative constituent, were useful in tracing the flow additions and losses.

Pathogen Indicators: *E. coli* was chosen as the routinely measured fecal indicator bacteria since it is a very specific indicator of pathogenic bacteria, and since the Colorado Department of Public Health and Environment (CDPHE) is moving towards using *E. coli* as its sole pathogen indicator (CDPHE, 2007). Although other routine microbial testing was desired, such tests were cost prohibitive.

Sampling Frequency

Sampling frequency was determined by analyzing the costs of various sample packages at various sampling frequencies over a one year. Note that while the aforementioned measurements were justified from a water quality perspective, budget constraints yielded the ultimate decision as to which constituents were measured. Based on the desire to analyze a wide range of parameters over an extended period of seasonal variation and the available budget, a sampling frequency of every three weeks was selected.

Analysis Methods

As described in the QAPP, Standard Methods (SM) (APHA et al., 1998) were used for all water quality analyses except anionic surfactants. Temperature and DO were analyzed in the field using a membrane electrode (Yellow Springs Instruments DO200, Yellow Springs, Ohio). Both pH and NH_3 were analyzed using an ion selective electrode (Thermo Scientific Orion 250A, Waltham, Massachusetts). Turbidity was measured with a nephelometric turbidimeter (Hach 2100N, Loveland, Colorado). Both TOC and TN were analyzed via combustion of acidified samples (Shimadzu TMN1, Columbia, Maryland). An ion chromatograph (Metrohm Peak 861, Herisau, Switzerland) was used to measure NO_3^-, PO_4^{3-}, Cl^-, and SO_4^{2-}. *E.Coli* were quantified via enzyme substrate (Idexx Laboratories Colilert/Quanti-Tray/2000, Westbrook, Maine). Anionic surfactants can be measured using a methylene blue active substances (MBAS) method (SM 5540C). This method, considered to be a broad estimation of all anionic surfactants, has been compared to other methods, such as the crystal violet method available through Hach, with the intention of finding a simpler method of determining anionic surfactants. Although all methods have shown some susceptibility to interference, other methods, including crystal violet, have been validated (Cross, 1998). Thus, anionic surfactants were analyzed by extraction of surfactant/crystal violet dye ion-pair complexes into benzene following the procedure of Hedrick and Berger (1966) and using a commercial surfactant testing kit (Hach Kit #2446800, Loveland, Colorado). The crystal violet method available from Hach measures surfactants that include an aromatic ring in their chemical structure, i.e., LAS and alkyl benzene sulfonate (ABS). Through electrophilic aromatic substitution the cationic, crystal violet dye bonds to the aromatic ring of LAS and ABS. When benzene is added to the solution a similar reaction dissolves the surfactant/cationic dye product in the benzene, turning the benzene solution a violet-blue color. MBAS uses methylene blue dye and chloroform to extract the surfactants and thus detection of a greater number of anionic surfactant species is possible.

All other measurements, along with sample collection, preparation, and storage were conducted following standard analytical methods (APHA et al., 1998). Quality assurance samples (e.g., blanks, duplicate analyses, and standards) were analyzed throughout the experiment at a rate of 10%. Multiple replications (generally three) were used for every

analysis when possible, and highest dilutions were always reported. In the case of bacterial samples, at least three replications were made at every dilution.

Chapter 5 Wetland Performance

Graywater influent characteristics, treatment efficiencies, and effluent quality all varied by season. Results of wetland monitoring were evaluated seasonally as distinct sample populations, however, due to limited seasonal data collection, statistical analysis by ANOVA was only performed on the complete data set (Table 5-5). The potential for reuse was evaluated by comparing the seasonal effluent water quality to applicable regulations (Table 3-3 and Table 3-4). The treatment efficiency of each wetland bed, and the system as a whole, were evaluated in terms of pollutant mass removal.

Water quality sampling and flow measurement commenced in September, 2008 and continued until August, 2009. At the onset of sampling, sampling events were conducted on a weekly basis for four weeks, after which samples were collected every three weeks on average. Initially, sampling was conducted weekly to establish equilibrium conditions following wetland startup. Wetland startup was an approximate three week period in September, 2008, immediately following the point when dormitory graywater became available. Prior to this time, graywater from the ACB was supplied to the FWS, but due to low flow rates, tap water supplemented the greywater feed to the SF. To flush the tap water and allow the wetland to come to equilibrium with dormitory graywater, sampling of the FWS and SF effluents were postponed 7 and 15 days after dormitory graywater was first supplied to approximate HRTs of the beds respectively.

Treatment Efficiency Determination

Simple comparison of the influent and effluent water qualities may not adequately evaluate the treatment performance of the wetland system because the effects of flow additions and losses on the effluent concentrations are not considered. However, by computing the mass loading at each point in the wetland, the effects of flow additions and subtractions are abated. In other words, comparing pollutant mass flux through the system instead of pollutant concentration changes resolved the dilution effects of precipitation and the concentration effects of ET.

Treatment efficiency, or mass removal, was computed by the average difference in mass loading between the influent and effluent of each wetland bed. Mass removals were considered seasonally, and were computed using the average mass loading over a season for the influent and effluent of each bed (Equation 5-1). Mass loading equals the product of flow rate and concentration, and was averaged seasonally using the measured concentration and the average flow rate over one HRT preceding the sampling event for all sampling events within a season (Equation 5-2). Influent loading rates considered the HRT of the FWS. FWS and SF loading rates considered the HRT of the SF. Flow rates were averaged over one HRT since that is the length of time over which flow rates affect the HRT, and therefore the wetland treatment efficiency. Because flow rates determine HRT, the calculation of an average flow rate over one HRT was performed iteratively.

$$TE = \frac{\overline{\dot{m}}_{Influent} - \overline{\dot{m}}_{Effluent}}{\overline{\dot{m}}_{Influent}} x100\%$$

Equation 5-1

$$\overline{\dot{m}} = \frac{\sum_{i=1}^{n}\left(C_i * \overline{q_i}\right)}{n}$$

Equation 5-2

Where:

TE	= treatment efficiency of a given system or bed for a particular season
$\overline{\dot{m}}$	= mean mass loading rate over a given season for a particular influent or effluent (mg/season)
n	= number of sampling events in a given season
C_i	= concentration of an influent or effluent for sampling event *i* (mg/L)
	= mean daily flow rate (L/d)

Hydrologic Performance

Flow Characteristics

Flows to and through the wetland varied by month and by season (Table 5-1 and Figure 5-1), and were influenced by the source and availability of graywater, losses from groundwater seepage and ET, and precipitation additions. Runoff intrusions were prevented via the protective berm surrounding each bed. Inflow rates fluctuated in response to seasonal variations in the graywater quantity generated from the office/laboratory building. In addition, during winter months, access to dormitory graywater was restricted, and the inflow rate was reduced accordingly. Seasons were defined as initial measurements in September through November for fall, December through February for winter, March through May for spring and June through last measurements in August for summer.

Table 5-1 Seasonal Mean Flow Rates and Standard Deviations

Season	Mean Flow ± s (L/d) (coefficent of variation)		
	Influent (1)[1]	FWS Effluent (2)	SF Effluent (3)
Fall	334 ± 33 (0.10)	285 ± 86 (0.30)	264 ± 86 (0.32)
Winter	247 ± 106 (0.43)	214 ± 105 (0.49)	206 ± 111 (0.54)
Spring	303 ± 119 (0.39)	305 ± 220 (0.72)	320 ± 275 (0.86)
Summer	276 ± 152 (0.55)	273 ± 249 (0.91)	111 ± 183 (1.65)

[1] Number refers to location in Figure 4-11.

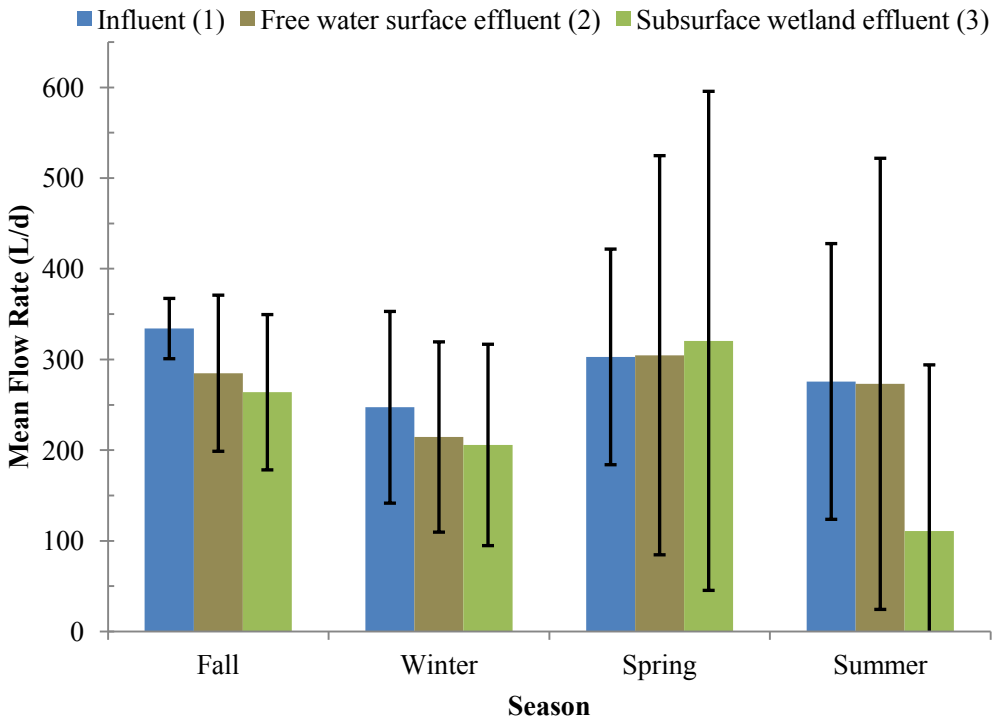


Figure 5-1. Seasonal averaged flow rates in the wetland system.

Water losses due to ET and seepage were large, especially ET during summer months, and were the cause of losses through the system and the decreasing flow rates out of the SF cell. Seepage occurred up and over the impermeable wetland liners in both wetland cells, which resulted from an inadequate elevation difference between the water levels and the top of the liners (this problem was discovered after the vegetation in the wetland beds was established, which precluded altering the liner elevation). During the spring, frequent precipitation events caused FWS and SF wetland effluent flow rates to be 1% and 6% greater than the average influent rate respectively. In the summer, several large precipitation events and varying inflow rates caused the average FWS effluent flow rate to be almost as high the average influent flow rate while very little SF effluent was observed due to high ET and evaporation rates. A record of precipitation is included in Figure-1 in Appendix A. Rainfall data were collected from a Colorado Agricultural Meteorological Network (CoAgMet) Class A weather station located approximately 650 m from the wetland.

Summer conditions were much more variable than the other seasons due to weather conditions and insufficient graywater coming from the dormitory and ACB. As mentioned previously, after a period of high precipitation and cool temperatures, the weather changed dramatically bringing high temperatures and nearly no precipitation. During this time there was also little water coming from the dormitory or the ACB, so graywater (including shower/bath water, laundry water, and hand wash water) from a residential source in Fort Collins, CO was used. Despite this extra water, very little effluent was collected from the end of the system. To fill the collection bucket, a section of tubing was lowered. Once enough water was available for collection, the tubing was returned to its original position. The flow meters and counters showed that very little water flowed out of the second cell of the wetland during the summer months. Except during spring months, inflow rates were larger than FWS or SF effluent rates. As shown in Figure 5-1, large deviations between the desired and actual inflow rates occurred during winter months when dormitory graywater collection was restricted.

Hydraulic Retention Time

The HRT was calculated by dividing the mean seasonal volume by the mean seasonal flowrate. The HRT in each wetland bed varied with seasonal flow conditions (Table 5-2), and were also affected by bed volume fluctuations in the winter resulting from ice coverage. Ice thickness was measured approximately twice weekly throughout the winter, and was considered in volume computations. During the 2008/2009 winter, a maximum ice thickness of 170 mm was observed on the FWS, while the gravel matrix in the SF provided insulation such that the maximum ice thickness was only 38 mm (Table 1 Appendix A). While the winter ice reduced the volume of each cell, reduced flow rates during winter months resulted in the higher observed HRTs than spring and fall. HRTs were lowest during the spring, largely because of frequent precipitation and were highest during the summer because of infrequent precipitation and reduced influent. Precipitation was treated simply as an additional inflow, which reduced overall HRT.

Table 5-2. Seasonal Mean Hydraulic Retention Times and Standard Deviations

Season	Mean Hydraulic Retention Time (d)		
	FWS Effluent (2)[1]	SF Effluent (3)	Total
Fall	7.0	9.9	16.9
Winter	6.4	12.4	18.8
Spring	6.6	8.1	14.7
Summer	7.3	23.5	30.8

[1] Number refers to location in Figure 4-11.

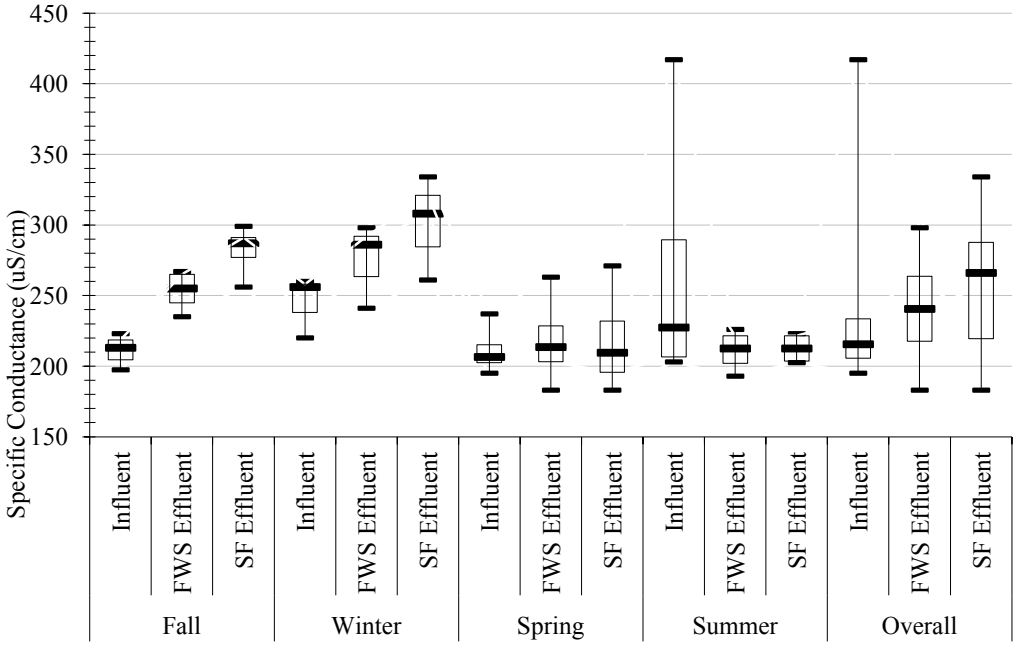
The HRTs observed in the wetland were six to ten times greater than those published for other constructed wetlands used to treat graywater. For example, the wetlands presented by Winward et al. (2008b) and Gross et al. (2007b) both featured HRTs that were maintained nearly constant at approximately two days. Near constant HRTs are typical of the treatment wetlands presented in the literature, but were not the case with the graywater wetland, which experienced large seasonal fluctuations in HRT. Of note is that the wetland studied here was designed for contaminant removal during the cold winter months observed in Fort Collins, CO. The HRTs observed in the wetland are longer than the HRTs of constructed wetlands used in wastewater treatment which typically range from one to six days (EPA, 1993).

Water Quality Results

The subsequent section provides water quality results and statistics from 18 sampling events conducted between September, 2008 and August, 2009. Unabridged water quality data and statistical analysis are presented in Appendix A. The seasonal and annual results for water quality parameters are presented using boxplots, created using Microsoft Excel. Boxplots allow condensed data statistics to be displayed, and show the following statistics of a population: median, range, and lower (1st) and upper (3rd) quartiles. Significant differences in water quality between influent and effluent concentrations were examined using one-way ANOVA at $P < 0.05$. Statistics on significance are provided only for the complete data set. Due to the low number of seasonal data points (which is denoted by "n" for number of date points), statistics are not provided seasonally.

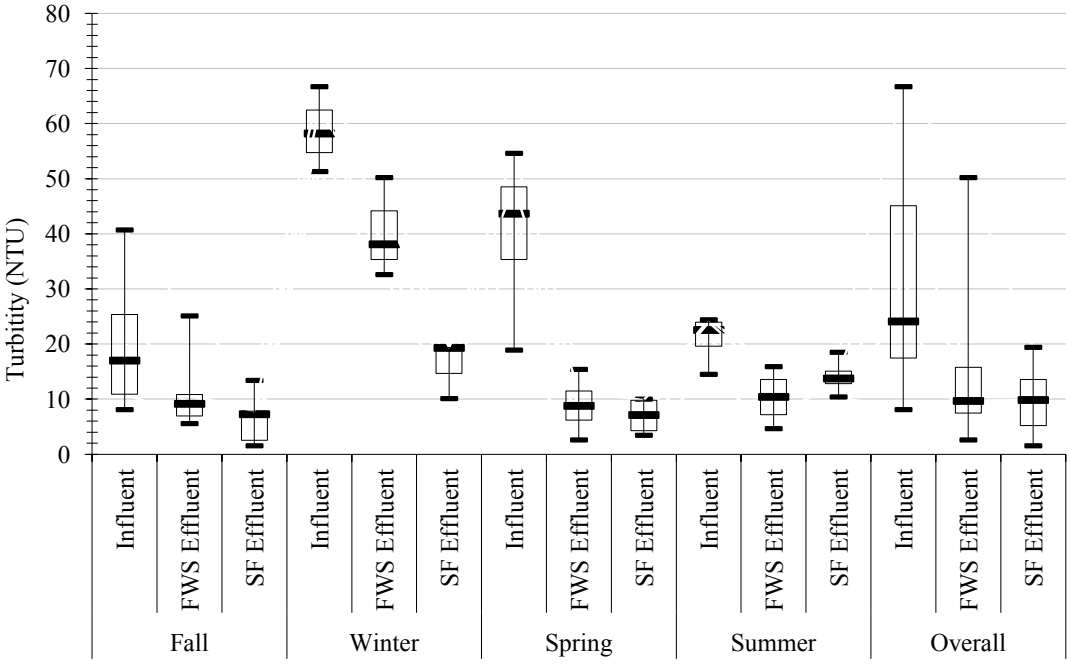
Physio-chemical Parameters

The physio-chemical properties of the raw graywater influent and the effluent of each wetland bed varied seasonally. SC and turbidity results are segregated seasonally and are presented in Figure 5-2 and Figure 5-3, respectively. Seasonal averages, standard deviations (s), and measurement ranges of temperature, DO, and pH are subsequently included in Table 5-3.



n = 7 (fall), 3 (winter) and 4 (spring and summer)

Figure 5-2 Plotted box plots for specific conductivity.



n = 7 (fall), 3 (winter) and 4 (spring and summer)

Figure 5-3 Plotted box plots for turbidity.

Table 5-3. Seasonal Mean Physio-chemical Parameters Measured in the Greywater Wetland.

Influent					
Analysis	**Parameter**	**Fall**	**Winter**	**Spring**	**Summer**
Temp (^oC)	Mean ± s	12.6 ± 3.7	16.0 ± 5.4	7.8 ± 6.4	18.9 ± 1.4
	Range	5.9 - 17.6	11.3 - 21.9	0.4 - 16	17.5-20.7
DO (mg/L)	Mean ± s	0.11 ± 0.05	0.08 ± 0.03	0.3 ± 0.21	0.3 ± 0.23
	Range	0.02 - 0.2	0.05 - 0.1	0.14 - 0.62	0.08-0.53
pH	Range	6.0 - 6.6	6.6 - 6.8	6.2 - 6.4	6.2-6.8
Free Water System Effluent					
Analysis	**Parameter**	**Fall**	**Winter**	**Spring**	**Summer**
Temp (^oC)	Mean ± s	11.3 ± 3.1	4.4 ± 3.0	7.7 ± 2.8	16.8 ± 0.55
	Range	7.8 - 15.1	2.5 - 7.8	4.7 - 11.5	16.1 - 17.3
DO, (mg/L)	Mean ± s	3.2 ± 1.6	1.2 ± 0.06	4.7 ± 2.3	1.1 ± 0.4
	Range	1.3 - 5.5	1.1 - 1.2	1.7-6.7	0.71 - 1.64
pH	Range	6.6 - 6.9	6.4 - 6.7	6.5 - 6.8	6.5-7.1
Submerged Flow Effluent					
Analysis	**Parameter**	**Fall**	**Winter**	**Spring**	**Summer**
Temp (^oC)	Mean ± s	12.4 ± 3.2	5.2 ± 4.5	9.0 ± 3.9	18.9 ± 1.2
	Range	8.6 - 16.3	1.9 – 10.3	4.5 - 14	17.8 - 20.6
DO, (mg/L)	Mean ± s	2.4 ± 0.6	2.2 ± 0.2	2.5 ± 0.5	1.7 ± 0.3
	Range	1.4 - 3.0	2.1 - 2.4	1.8 - 2.8	1.3 - 2.1
pH	Range	6.3 - 6.8	6.5 - 6.7	6.1 - 6.6	6.3 - 7.3

Note: n = 7 (fall), 3 (winter) and 4 (spring and summer)

Average SC values increased through the wetland for all seasons ('through the wetland' implies from the influent to the FWS effluent and from the FWS effluent to the SF effluent), indicating that dissolved solids and/or salts were likely accumulating in each wetland bed. Accumulation of dissolved ions can be common in wetland systems and results primarily from evapotranspiration. Conversely, precipitation contains relatively little dissolved solids, and would be expected to lower the SC in the wetland beds. This is evident by the relatively lower increase in the SC through the wetland system in the spring season which was the rainiest season observed.

Regardless of accumulation, measurements indicated that the wetland effluent SC was less than 350 uS/cm for any season, rendering the effluent water as "good" (Table 5-4) for irrigation based on SC levels recommended by FCES (2004). Furthermore, SC readings of the raw graywater influent were generally less than either bed's effluent, indicating that salinity levels in the low-load graywater were also suitable for irrigation. Influent SC values were in line with those given in Eriksson et al. (2002).

Table 5-4. Recommended Specific Conductivity Levels (FCES, 2004)

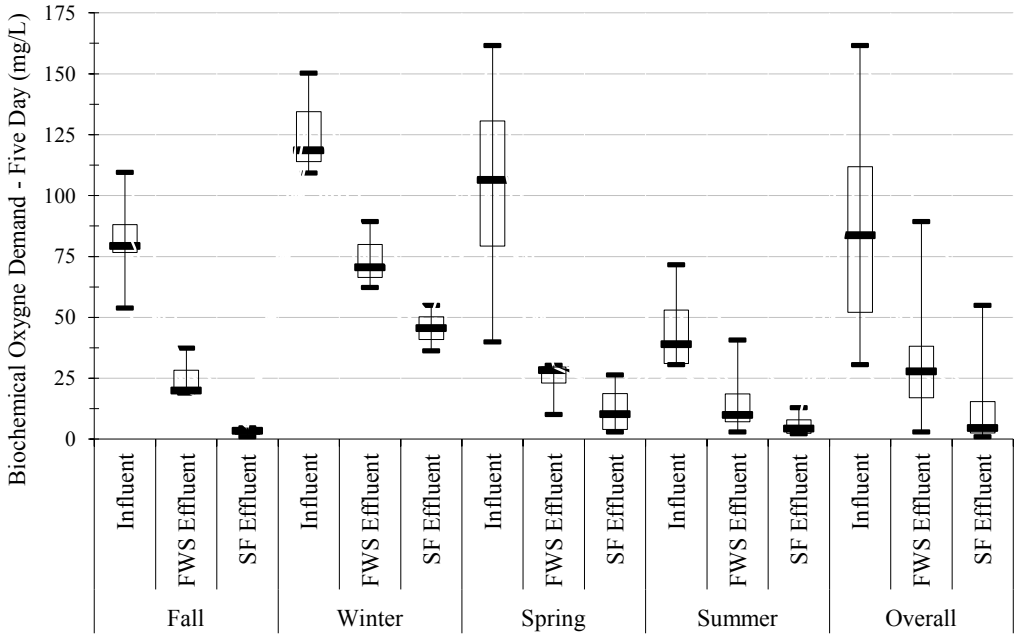
Class of Water	Specific Conductivity, µS/cm
Excellent	< 250
Good	250-750
Permissible	750-2000
Doubtful	2000-3000
Unsuitable	> 3000

Turbidity measurements decreased through the wetland for all seasons except for summer. Percent reduction (by measurement, not mass) between the influent and SF effluent ranged from 69% in the fall to 83% in the spring. The mean turbidity of the system effluent was approximately 6 NTU during the fall and spring, and nearly 17 NTU during the winter. Summer effluent results from the FWS cells were comparable to results from the fall (~10 NTU) but SF effluent results were about twice that of the fall results. Much of the solids during this time were thought to be from biofilm disturbed by the change in collection methods and algae that was present in the FWS earlier in the season moving through the system. VSS concentrations were slightly higher in the SF effluent during the summer (nearly 7 mg/L in the summer compared to about 4 in the winter and spring) but as a percentage of TSS, VSS were actually lower (50%, 63%, 60%, 43% for summer). The solids present in SF effluent in the summer months were largely inorganic in nature and it is difficult at this time to determine the source of inorganic solids from the SF cell. This will be examined with future sampling efforts at the constructed wetland. Unrestricted reuse guidelines for several states (e.g., Arizona, California, and Washington; see Table 3-3) give maximum and average turbidity values of 5 NTU and 2 NTU respectively. The EPA (2004) also gives turbidity recommendations of 2 NTU for reclaimed water used for irrigation. Under these strict regulations, the wetland effluent would be out of compliance for use as unrestricted irrigation water.

Reuse regulations also stipulate an acceptable pH range between 6 and 9 (EPA, 2004). The effluent pH ranged between 6.14 and 7.26 throughout the year, which is acceptable for reuse. Despite this range, the pH remained fairly constant (around 6.5) for all measurement locations through all seasons. The DO of the influent was quite low, indicating anaerobic conditions of the stored graywater. DO generally increased through the wetland system, with the highest DO measured during spring and fall seasons in the FWS effluent. This was likely due to the open water nature of the FWS, which probably promoted more oxygen transfer than the closed surface nature of the SF. DO at the FWS effluent was very low during winter months, explained by the thick ice cap on the FWS which prevented atmospheric contact. DO was also low in the FWS effluent during summer months. It is possible that extensive growth of plants in the FWS resulted in decreased oxygen transfer into the cell during this time period. Temperature through the wetland generally remained constant in the fall, decreased in the winter, and increased in the spring and summer. The high influent temperature in the winter is partially attributed to the submersible heater in the trailer- mounted tank.

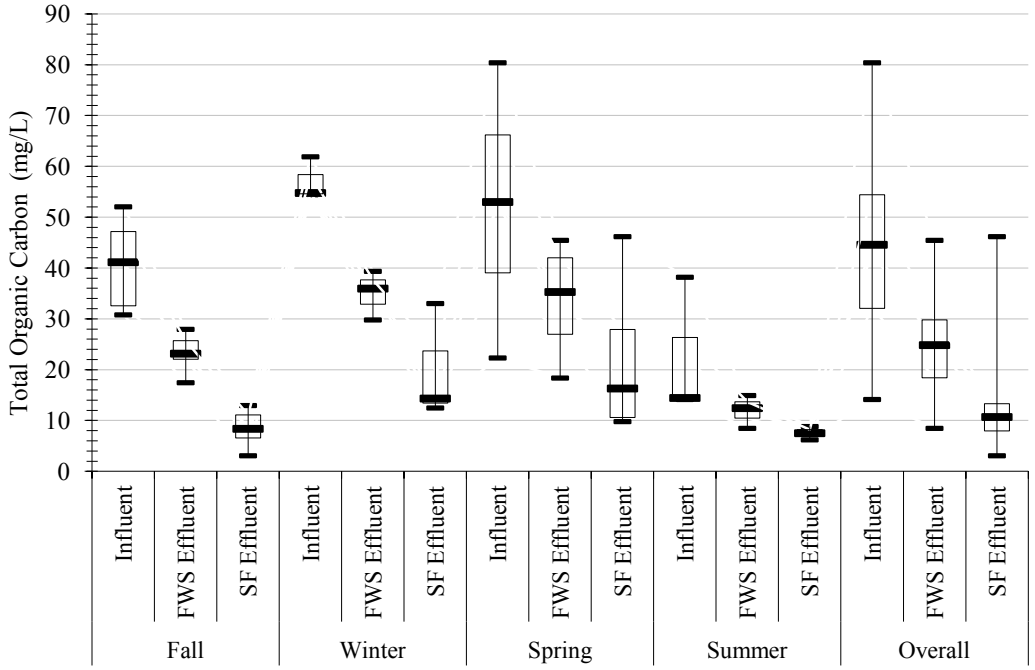
Organics

Organics, measured by BOD_5 and TOC decreased substantially though the wetland for all seasons – see Figure 5-4 and Figure 5-5, respectively.



n = 5 (fall influent and FWS effluent), 4 (fall SF effluent and summer all sites), and 3 (winter and spring all sites)

Figure 5-4. Plotted box plots for biochemical oxygen demand.



n = 7 (fall), 3 (winter, spring and summer influent and FWS effluent) and 2 (summer SF effluent)

Figure 5-5. Plotted box plots for total organic carbon.

Influent concentrations of BOD_5 and TOC were common to those given in literature (Erikkson et al., 2002 and Cassanova et al., 2001). Effluent concentrations of BOD_5 were lowest in the fall (3.1 mg/L) and highest in the winter (45.6 mg/L). Only during the fall and summer were effluent BOD_5 levels below the state mandated (from Table 3-3) and EPA (2004) recommended level of 10 mg/L for urban reuse of reclaimed water. The treatment efficiency (percent removal by mass) of BOD_5 through the wetland was greater than 89% throughout the year, with removal rates of 96%, 92% and 94% observed during fall, spring and summer seasons respectively (Figure 5-6). Lower removal efficiencies during winter months, 80%, were primarily attributed to lower temperatures.

Similarly, average effluent TOC levels were lowest in the summer (7.53 mg/L), but were greatest in the spring (22.17 mg/L). TOC removal rates were less than BOD_5, but always remained greater than 50% (Figure 5-6). This indicates that removal of readily biodegradable organic material, as measured by BOD_5, is effective through the wetland. However, there are likely recalcitrant organics (e.g., degradation byproducts, soluble microbial products, or plant exudates) that are not as readily removed through the wetland system. During spring months, TOC removal was particularly low in the SF. This low removal efficiency in the spring is partially explained by the increased effluent flow rates resulting from precipitation, which increased the computed mass loading at each effluent location. As evident in Figure 5-6, a majority of BOD_5 and TOC was removed in the FWS – although the HRT of the FWS was less than the HRT of the SF, the FWS was first in series and was likely the location where the bulk treatment of readily biodegradable organics occurred.

The dissolved portion of total organic carbon – dissolved organic carbon (DOC) was also measured periodically. DOC decreased through the wetland system similarly to TOC, with DOC concentrations being approximately 50% to 75% of TOC levels. Removal of DOC was less than TOC for most seasons except summer.

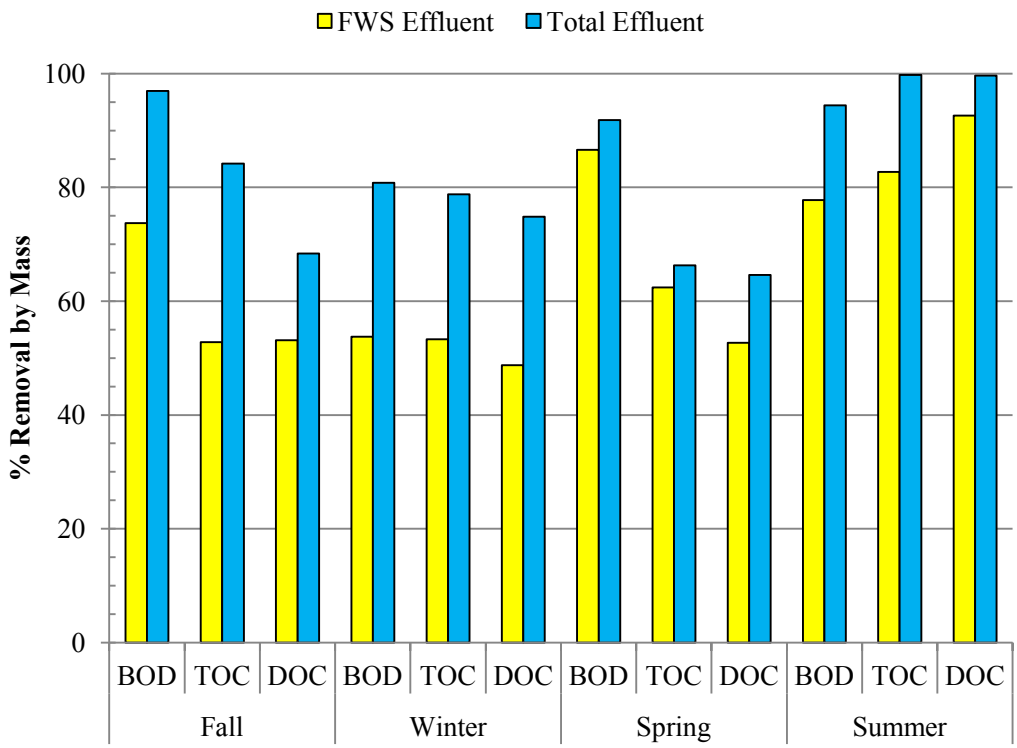
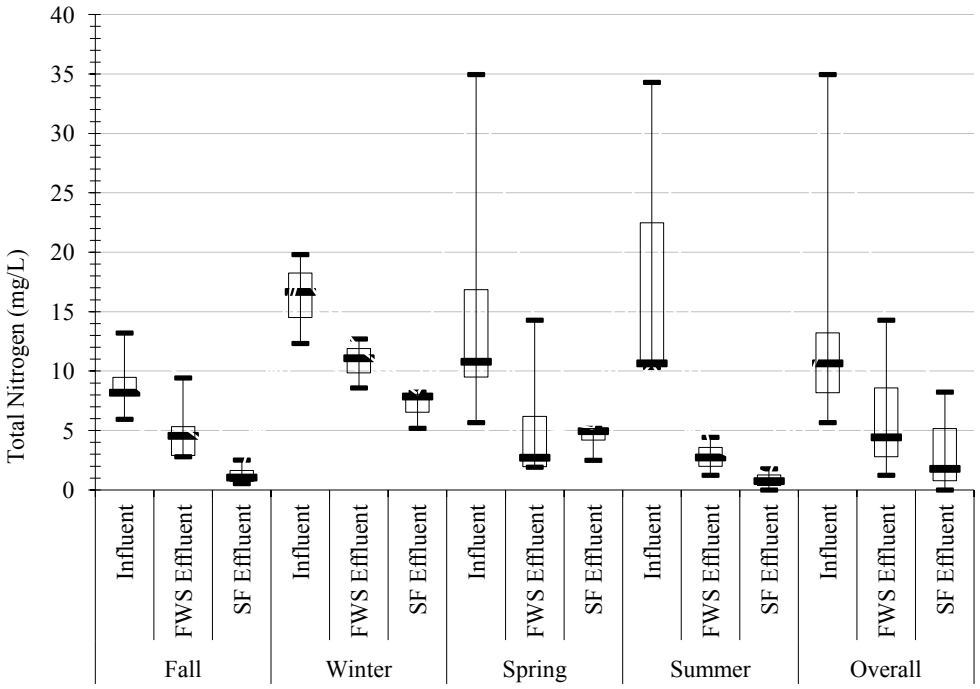


Figure 5-6. Percent organic removal efficiencies.

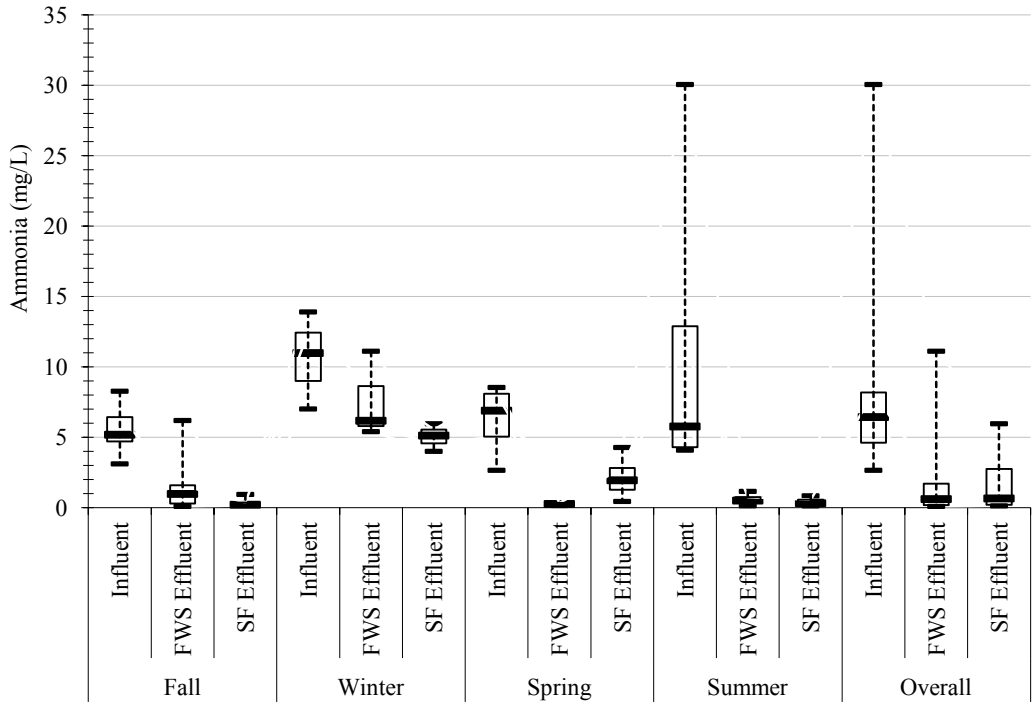
Nutrients

Nitrogen was evaluated via TN, NH_3, and NO_3^-. Results of TN and NH_3 monitoring are given in Figure 5-7 and Figure 5-8, respectively. Observed concentrations of NO_3^- were below or near the detection limit for all measurement locations and for all seasons except for one measurement during the summer at the influent and at the SF effluent during the fall and summer, when the average NO_3^- was 0.24 mg/L. The detection limit for NO_3^-, as well as PO_4^{3-}, Cl^- and SO_4^{2-}, was 0.2 mg/L. This was the lowest discernible concentration obtainable from the ion chromatograph, which was used to measure anions.



n = 7 (fall) and 3 (winter, spring and fall influent and FWS effluent) and 2 (fall SF effluent)

Figure 5-7. Plotted box plots for total nitrogen.



n = 7 (fall influent), 6 (fall FWS effluent), 5 (fall SF effluent), 4 (summer all sites) and 3 (winter,

Figure 5-8. Plotted box plots for ammonia.

NH_3 represented 40% to 65 % of the TN measured in the influent throughout all seasons, with both parameters within published ranges for raw graywater (Eriksson et al., 2002). At the effluent of the FWS and SF, the ratio of NH_3:TN was variable, but generally lower than the influent, indicating biotic uptake of soluble ammonia. Average concentrations of TN and NH_3 in the wetland effluent were highest in the winter (7.1 mg/L and 5.0 mg/L respectively) and lowest in the summer (0.84 mg/L and 0.4 mg/L). These effluent nitrogen levels, especially for NH_3, are quite high with regard to toxicity levels in freshwater. General EPA freshwater guidelines for NH_3 depend on temperature, pH, and the type biotic organisms present (EPA, 1986). This reference gives applicable total ammonia limits between 1.5 mg/L (at 20°C, pH = 6.5, salmonids present) to 2.4 mg/L (at 5°C, pH = 6.5, salmonids absent). Except during fall and summer months, the effluent levels of NH_3 could be toxic to freshwater biota, which should be considered if the effluent were to be stored in or directly released to an environmental water body. However, if the effluent is applied directly for irrigation, high NH_3 concentrations should be of little concern. In fact, high levels of NH_3 and TN may in fact be beneficial to the landscaping receiving the effluent for irrigation. Limits for nitrate depend on end use, Drinking water supplies have NO_3^- limits of 10 mg/L, while toxicity levels for salmonid species have been shown to be between 0.6 mg/L and 1.6 mg/L, and toxicity levels for warm water, i.e., not salmonid species, are greater than 10 mg/L. As discussed, the graywater wetland had effluent concentrations of NO_3^- less than 0.24 mg/L for all seasons, keeping the effluent well within NO_3^- regulations for all end uses.

Concentrations of TN and NH_3 were reduced through the wetland for all seasons, except through the SF during the spring (Figure 5-9). During the spring, the NH_3 concentration increased through the SF. This concentration increase was consistently observed for all sampling periods during the spring, and may have resulted from increased algae uptake of ammonia in the FWS and subsequent nitrogen release from decomposing algae or detritus in the SF.

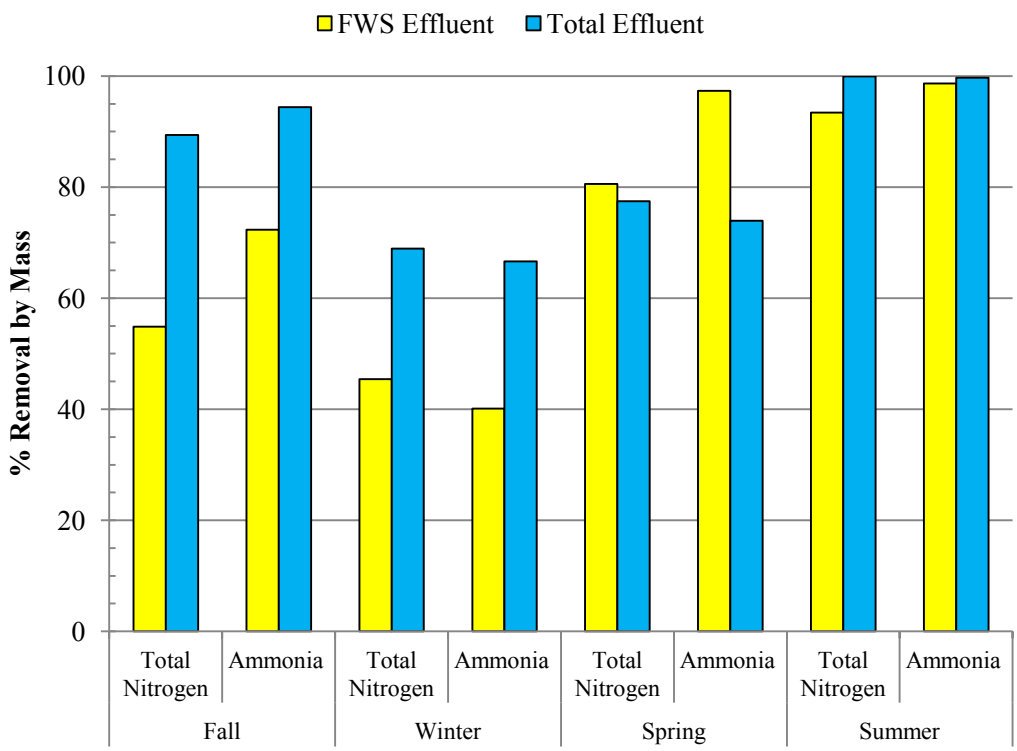
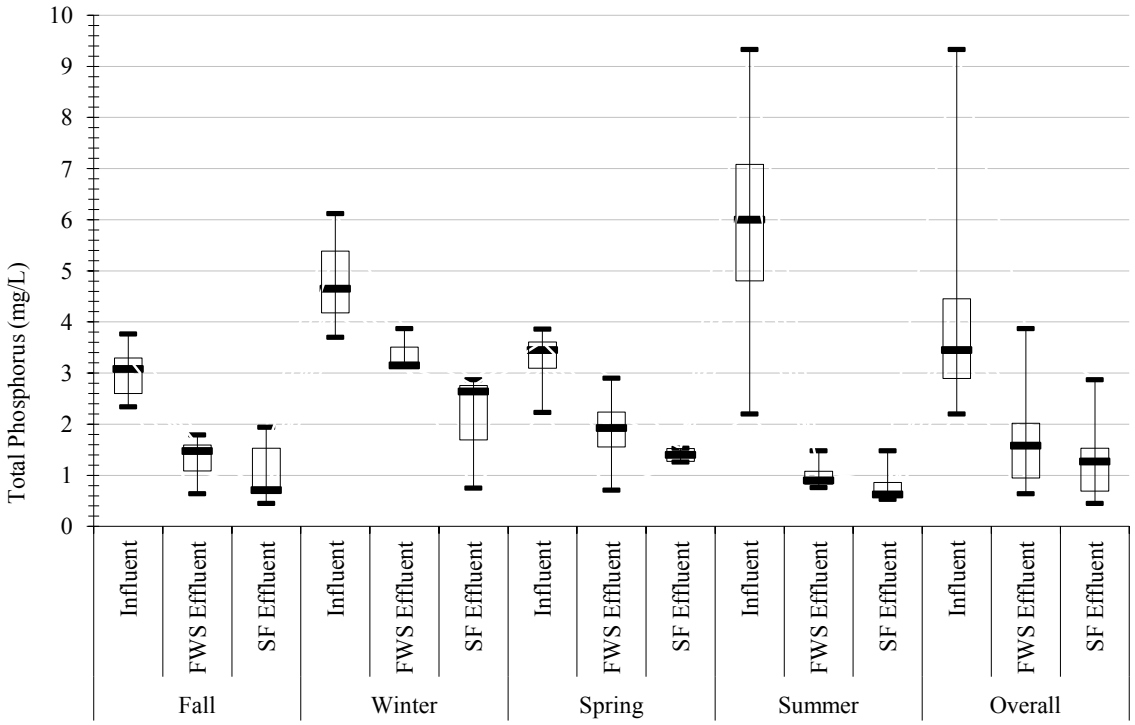


Figure 5-9. Percent nitrogen removal efficiencies.

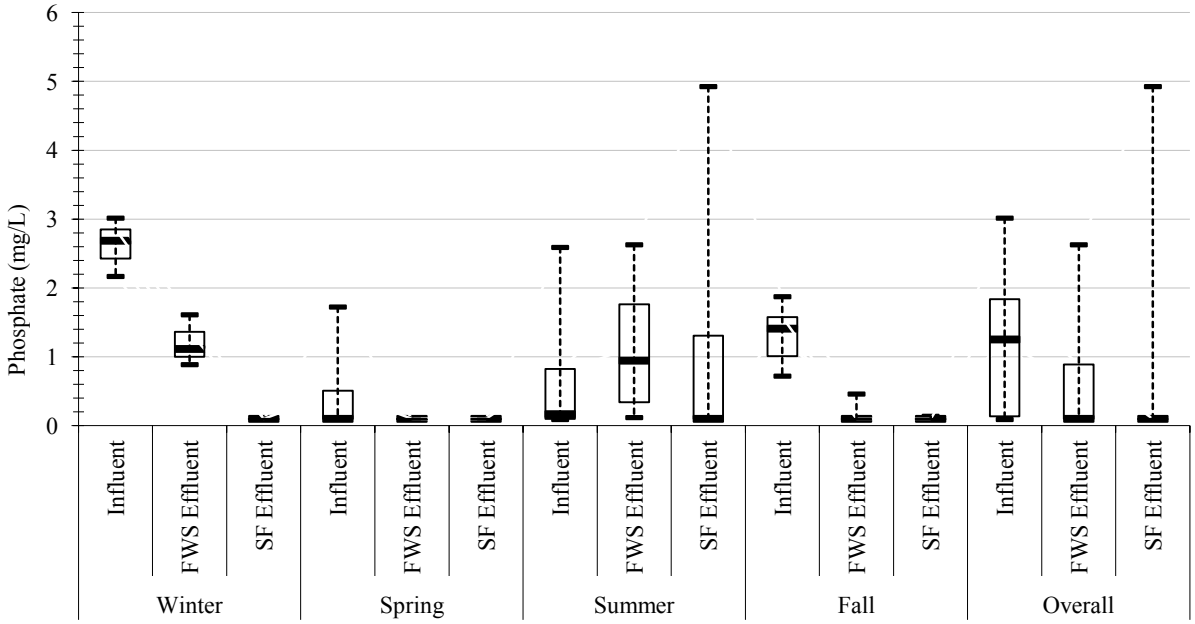
As noted NO_3^- were low, even for influent values and were lower than literature values (Table 3-1). The concentrations of NH_3 influent values were a mean of 5 mg/L or greater for all seasons and theses values were higher than the characteristic literature values (Table 3-1) which were all < 1mg/L. The low NO_3^- and high NH_3 values are potentially an indication of biological activity and nitrogen conversion. Rose et al. (1991) noted the increase in NH_3 and PO_4^{3-} may be due to nutrients available for microorganisms. This is discussed further under the sub heading ***Surfactants.***

Phosphorus was measured by TP and PO_4^{3-} (Figure 5-10 and Figure 5-11). PO_4^{3-} concentrations were always below detection (0.2 mg/L) at the SF effluent for sampling events except for one summer event. During the spring, all PO_4^{3-} measurements at all sites were below detection. Along with SF effluent samples, several PO_4^{3-} measurements in FWS effluent were also below detection. For the sake of comparing concentrations above and below the detection limit, PO_4^{3-} concentrations below the detection limit were assumed to be half the detection limit (or 0.1 mg/L) when computing seasonal concentration averages and percent removals (Figure 5-12) during fall and winter seasons.



n = 7 (fall influent), 6 (fall FWS effluent), 5 (fall SF effluent), 4 (summer all sites) and 3 (winter and spring all sites)

Figure 5-10. Plotted box plots for total phosphorus.



n = 7 (fall influent), 6 (fall FWS effluent), 5 (fall SF effluent), 4 (summer all sites) and 3 (winter and spring all sites)

Figure 5-11. Plotted box plots for phosphate.

TP and PO_4^{3-} concentrations in the influent were reasonable compared to literature values (Table 2.1). Approximately 45 – 55% of influent total phosphorus was in the dissolved phosphate form. This percentage dropped to 10 – 30% in the FWS effluent and to less than 9% in the SF effluent, showing that phosphate was quickly metabolized in the wetland. Treatment efficiencies of phosphate were very high (> 90%), but less so for total phosphorus, especially during colder, winter months (Figure 5-12). As mentioned, in the spring, influent and effluent phosphorous concentrations were below detection limits so no loadings were calculated.

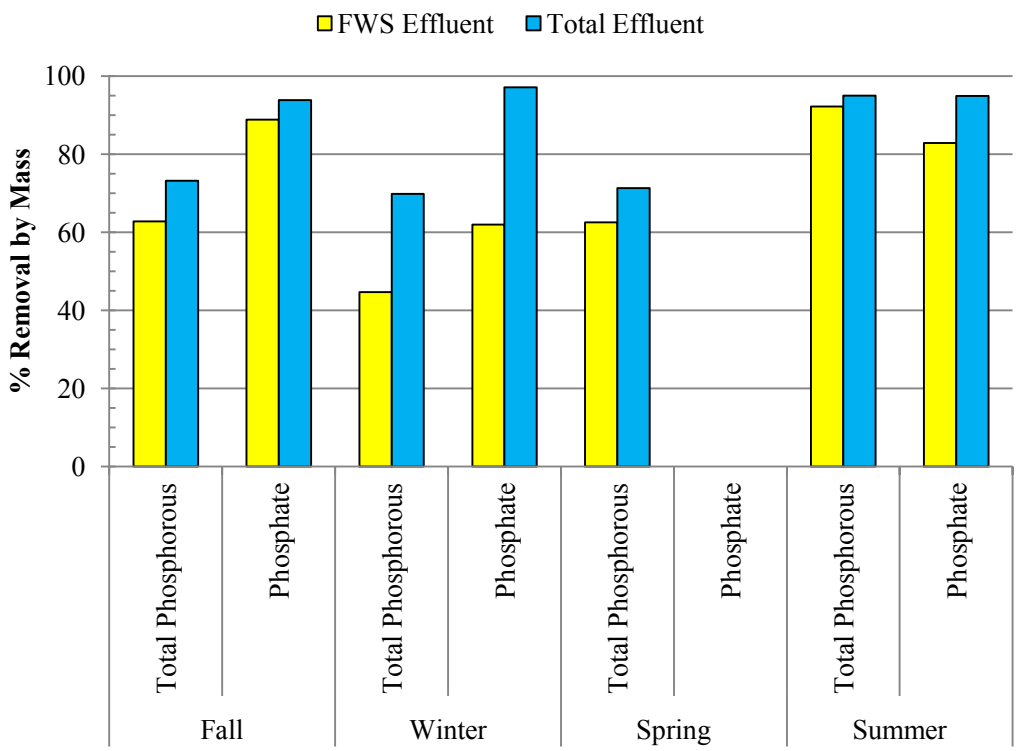
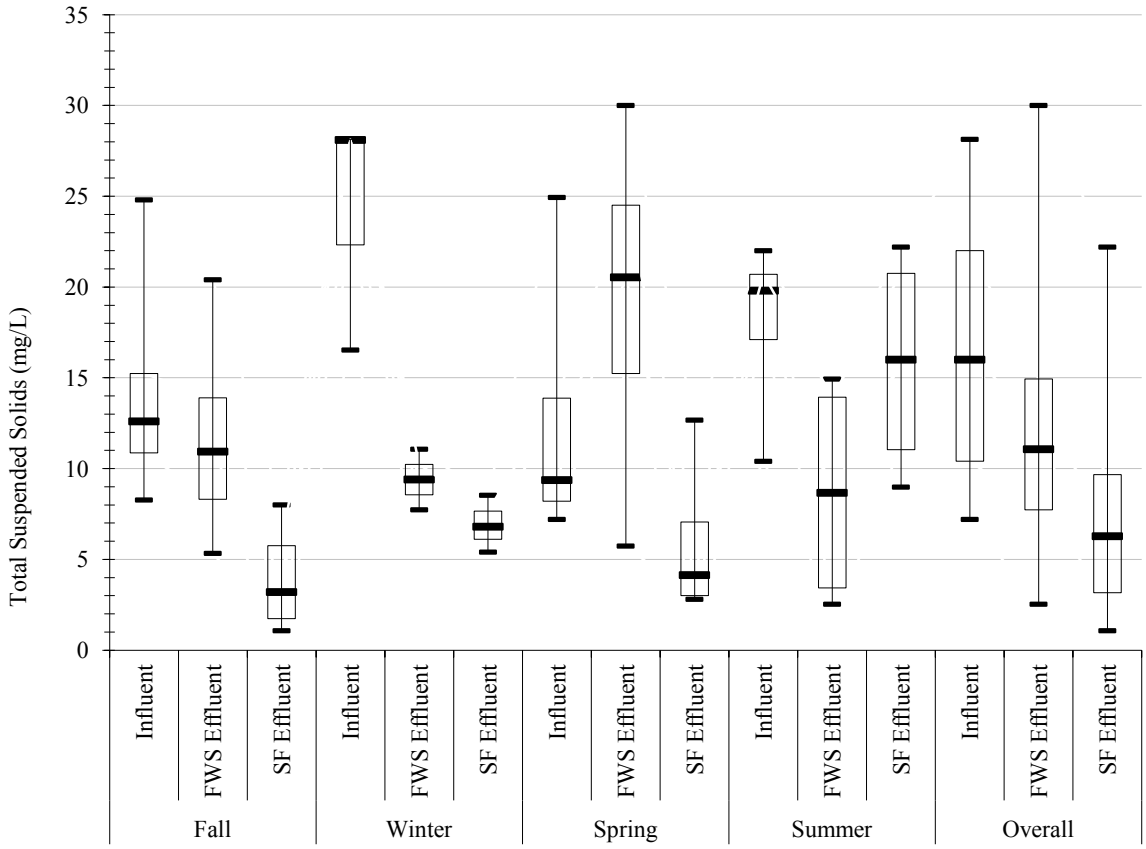


Figure 5-12. Percent phosphorus removal efficiencies.

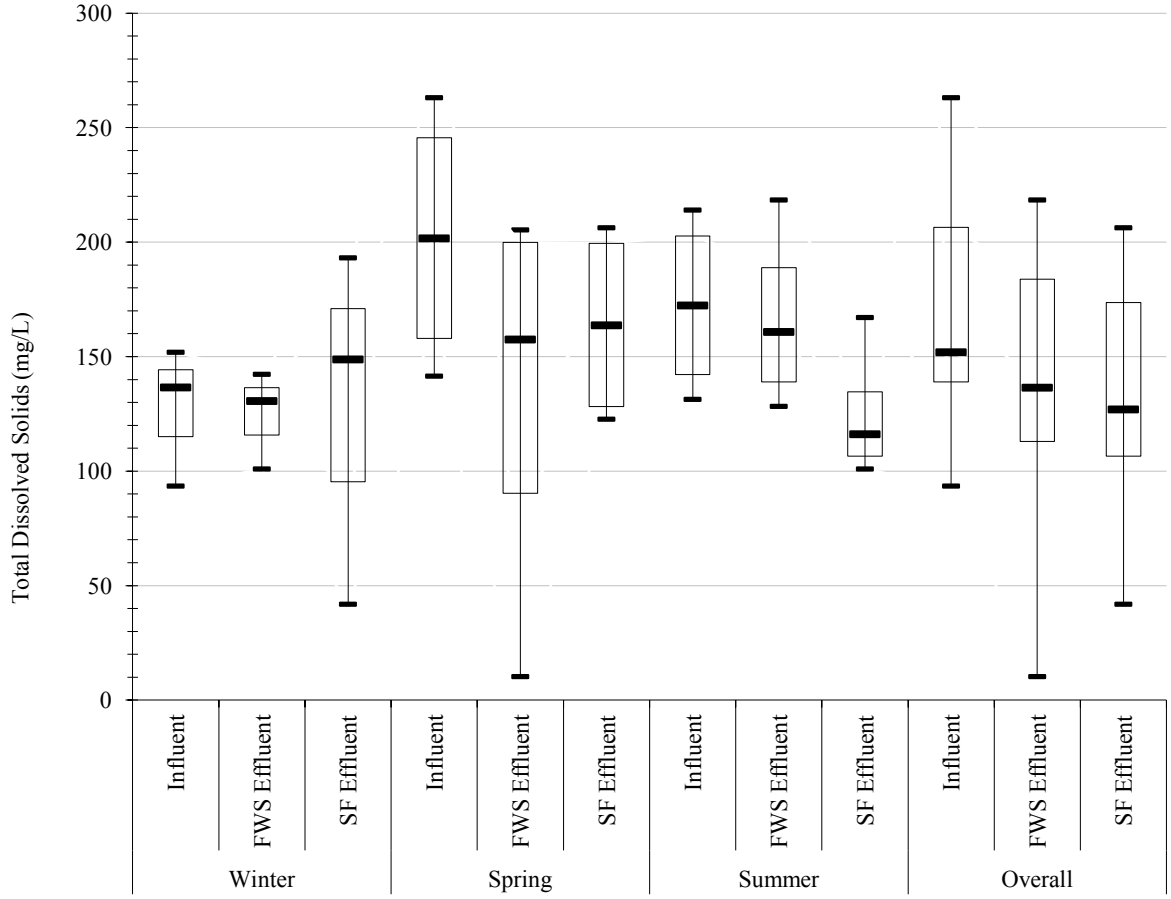
Solids

Initially, only TSS were analyzed. Starting in November, 2008 dissolved, volatile, and fixed solids were also measured. Seasonal TSS measurements are shown in Figure 5-13, and TDS measurements are shown in Figure 5-14. Volatile and fixed contributions to each seasonal suspended and dissolved solids category are shown in Figure 5-15.



n = 6 (fall influent and FWS effluent), 5 (fall SF effluent), 4 (summer all sites) and 3 (winter and spring all sites)

Figure 5-13. Plotted box plots for total suspended solids.



n = 3 (winter and spring) and 4 (summer)

Figure 5-14. Plotted box plots for total dissolved solids.

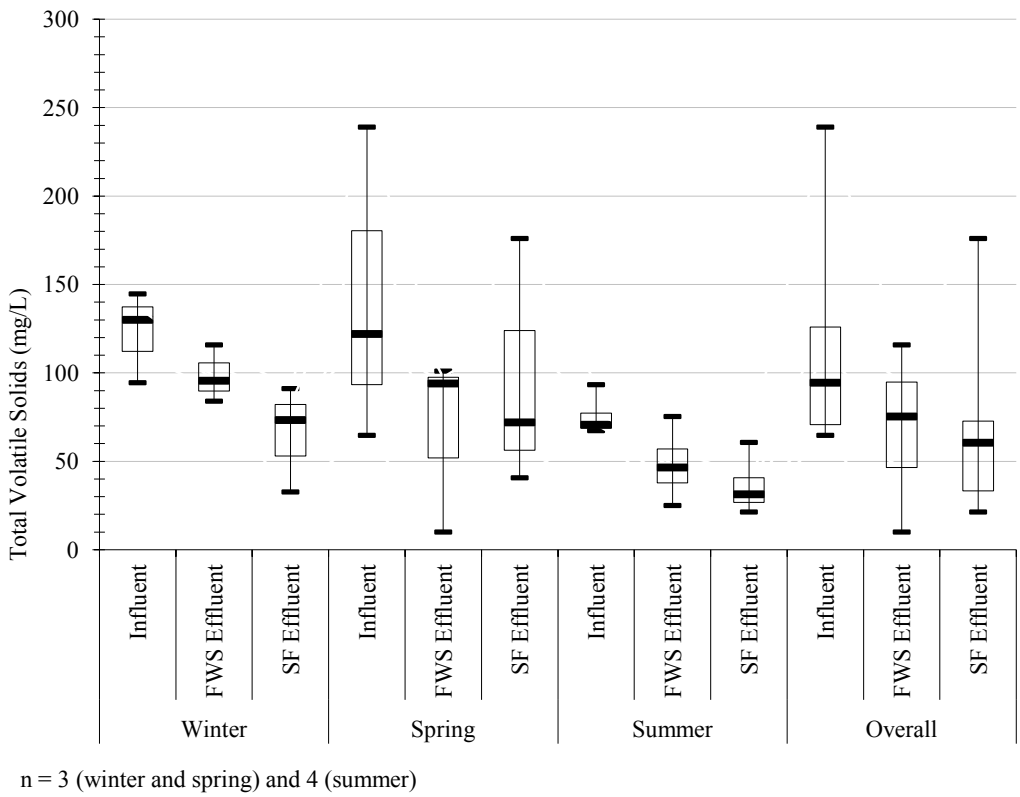


Figure 5-15. Plotted box plots for total volatile solids.

TSS concentrations were reduced substantially between the influent and SF effluent for all seasons other than summer, producing mean effluent concentrations of less than 4 mg/L for spring and fall, and less than 7 mg/L during winter months. Effluent TSS would meet even the strictest state or EPA reclaimed water regulation (5 mg/L) for irrigation during the fall and spring seasons. Even during the winter, effluent TSS was below EPA recommendations and most state regulations (except Florida) for irrigation water. However, effluent TSS was high in the summer months in comparison to other seasons even while TSS in the FWS cell was low (Figure 5-13).

The SF cell actually contributed solids during the summer. It is hypothesized that an extensive biofilm formed on the gravel present in the SF cell over summer months and the biofilm sloughed into the water, thus contributing to TSS. However, unlike the rest of the year when percent VSS in the SF effluent was between 50 and 60%, the percentage of VSS in the SF effluent during the summer was only 43%. At this time, it is difficult to determine the source of inorganic solids in the SF cell. Nonetheless, there was an overall decrease in TSS through the wetland during summer months despite the mean effluent concentration of nearly 16 mg/L. No substantial difference was observed in TSS concentrations between the influent and FWS effluent during spring or fall months. In fact, during these seasons FWS effluent TSS was often higher than the influent, which is attributed to the large amount of algae growing in the FWS. Algae in the FWS were especially plentiful immediately following ice-off in the spring. For approximately two weeks after the ice melted, thick algae blooms were observed in the FWS.

TDS contents fluctuated widely between every sampling event. Considering the observed s of the box plot (Figure 5-14), no significant difference was observed between the three measurement sites during any season. Surprisingly, TDS measurements did not correlate well with SC measurements as would be expected. The ratio of TDS:SC varied widely within sites and seasons, and ranged from 0.1 to 1.2 with a mean of 0.6. This mean TDS:SC ratio was within the common range of wastewater and those given by Eriksson et al. (2002) for graywater.

Dissolved solids were greater than TSS for all samples, and generally the volatile (organic) portion of the solids exceeded the fixed (inorganic) portion. It was also found that for fall and winter months, the volatile fraction decreased and the fixed fraction increased through wetland (this is especially clear during the winter months). Much larger fractions of volatile suspended solids were seen in the FWS effluent as compared to the SF effluent, which is again attributed to algae in the FWS.

Even though TSS influent concentrations were lower than the range of graywater values from the literature (Table 3-1), i.e. < 54 mg/L, mass removals were consistently high. Total mass removals of TSS were greater than 75%, with the highest removals realized during winter months (Figure 5-16). Even though TDS concentrations changed little between the influent and effluent, overall percent removals were positive, and resulted primarily from changes in mass loading due to differing flow rates (Figure 5-16).

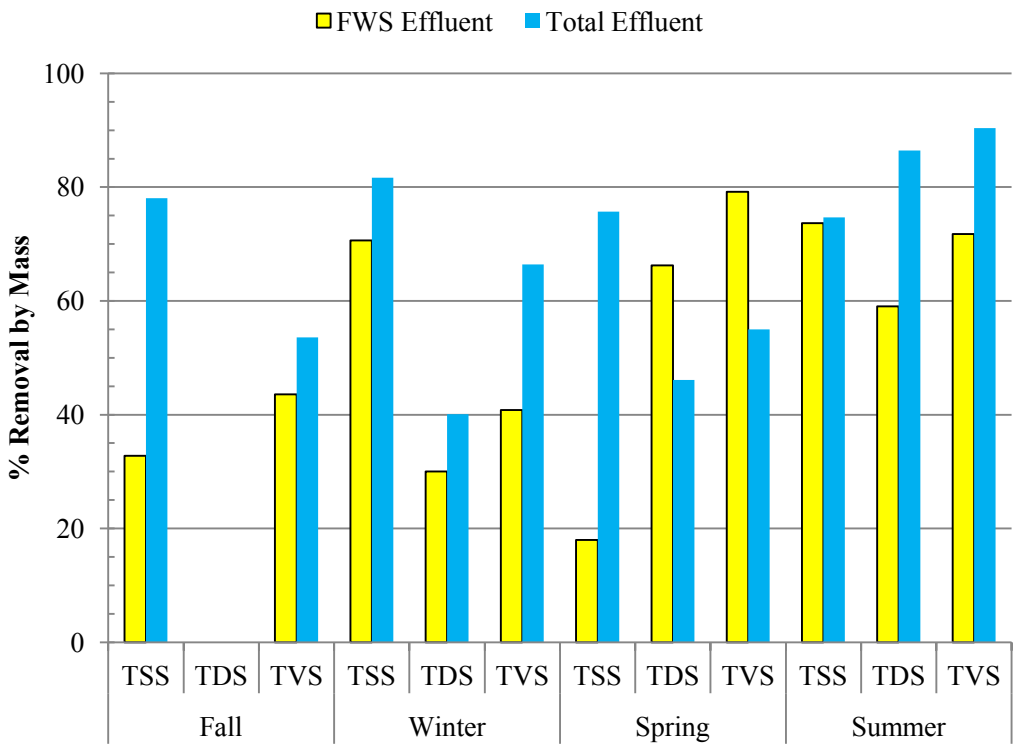


Figure 5-16. Percent suspended, dissolved, and volatile removal efficiencies.

Anions

Seasonal anion (Cl^- and SO_4^{2-}) concentrations are shown below in Table 5-5. Cl^- was evaluated mainly as a tracer through the wetland. Concentrations of Cl^- increased during winter, albeit slightly. During fall, Cl^- decreased slightly through the system. Spring Cl^- measurements were made with incorrect dilutions, and all results except one were over range (10 mg/L). Decreases in Cl^- likely occurred due to precipitation additions, while Cl^- increases were attributed to ET. Likewise, SO_4^{2-} generally decreased between the influent and the SF effluent, especially during the spring. However, SO_4^{2-} readings within seasons (especially the spring) generally had large variances resulting in very little measurement difference between any sites for any season. Concentrations of Cl^- in the inflow were within published ranges, while SO_4^{2-} concentrations were lower than those in the literature (Rose et al., 1991; Casanova et al., 2001).

Table 5-5 Chloride and Sulfate Measurements and Statistics

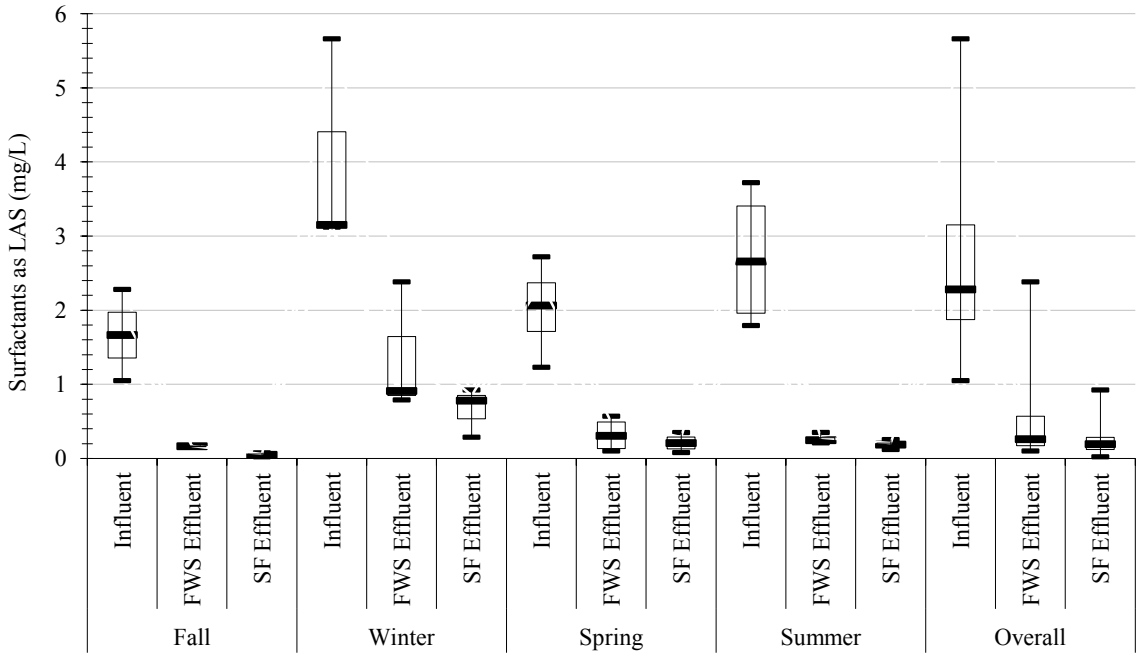
Analyte	Parameter	Concentrations (mg/ L) or Number (n) of Data Points			
		Fall	Winter	Spring	Summer
Influent					
Cl^-	Mean ± s	9.6 ± 0.8	10.2 ± 0.7	12.2 ± 4.4	8.4 ± 5.0
	Range	8.9 - 10.9	9.4 - 10.9	10.0 - 18.8	1.1 - 12.4
	n	7	3	4	4
SO_4^{2-}	Mean ± s	11.6 ± 3.4	7.4 ± 0.8	9.6 ± 4.8	10.9 ± 9.1
	Range	5.3 - 16.3	6.8 - 8.3	6.1 - 16.1	0.24 - 19.5
	n	7	3	4	4
FWS Effluent					
Cl^-	Mean ± s	9.6 ± 0.9	11.1 ± 0.7	11.4 ± 2.7	6.4 ± 4.8
	Range	8.6 - 11.0	10.7 - 11.9	10.0 - 15.4	0.51 - 10.54
	n	6	3	4	4
SO_4^{2-}	Mean ± s	12.6 ± 2.4	9.8 ± 0.7	4.81 ± 3.7	9.3 ± 5.3
	Range	10.6 - 16.8	9.2 - 10.5	<0.2 - 8.2	0.12 = 14.1
	n	6	3	4	4
SF Effluent					
Cl^-	Mean + s	8.6 ± 1.6	12.3 ± 1.8	6.9 ± 3.6	0.39 ±0.4
	Range	6.9 - 11.2	10.5 - 14.1	3.3 - >10	0.12 - 1.01
	n	5	3	4	4
SO_4^{2-}	Mean + s	8.9 ± 4.9	6.0 ± 3.5	1.8 ± 1.7	2.7 ± 4.0
	Range	3.2 - 15.6	3.9 - 10.1	0.77- 4.4	<0.2 - 8.5
	n	5	3	4	4

Surfactants

Anionic surfactants were reduced substantially through the wetland treatment system, with SF effluent concentrations being less than 1 mg/L for all seasons (Figure 5-17). Anionic surfactants include a wide class of molecular structures, each with different molecular weights. Therefore, surfactant concentrations given below are normalized to the molecular weight of linear alkyl benzene sulfonate (LAS).

Influent surfactant concentrations ranged from approximately 1 – 6 mg/L, which were less than the ranges given in literature (Gross et al., 2007b). Influent concentrations were thought to be affected by the storage time of graywater in the trailer-mounted tank, with lower concentrations resulting from increased storage time. To test this hypothesis, biodegradation in the trailer-mounted tank was tested during the fall season by collecting samples as the tank was filled and when the tank was nearly empty. Results of this one-time test showed a 49% reduction in surfactant concentration in the trailer-mounted tank over 48 hours of storage. Incidentally, TOC and TN had reductions of 19% and 6% respectively. Lower observed influent surfactant concentrations in comparison to published values likely was due to the storage time of the raw graywater.

Total removal efficiencies averaged approximately 90% or greater in all seasons (Figure 5-18). Removal efficiencies were greatest during the fall, spring, and summer seasons and lowest during winter months. As with other carbonaceous compounds, surfactant degradation heavily relies on temperature dependent biological kinetics, which was reduced during the winter season.



n = 2 (fall), 3 (winter and spring) and 4 (summer)

Figure 5-17. Plotted box plots for anionic surfactants as LAS.

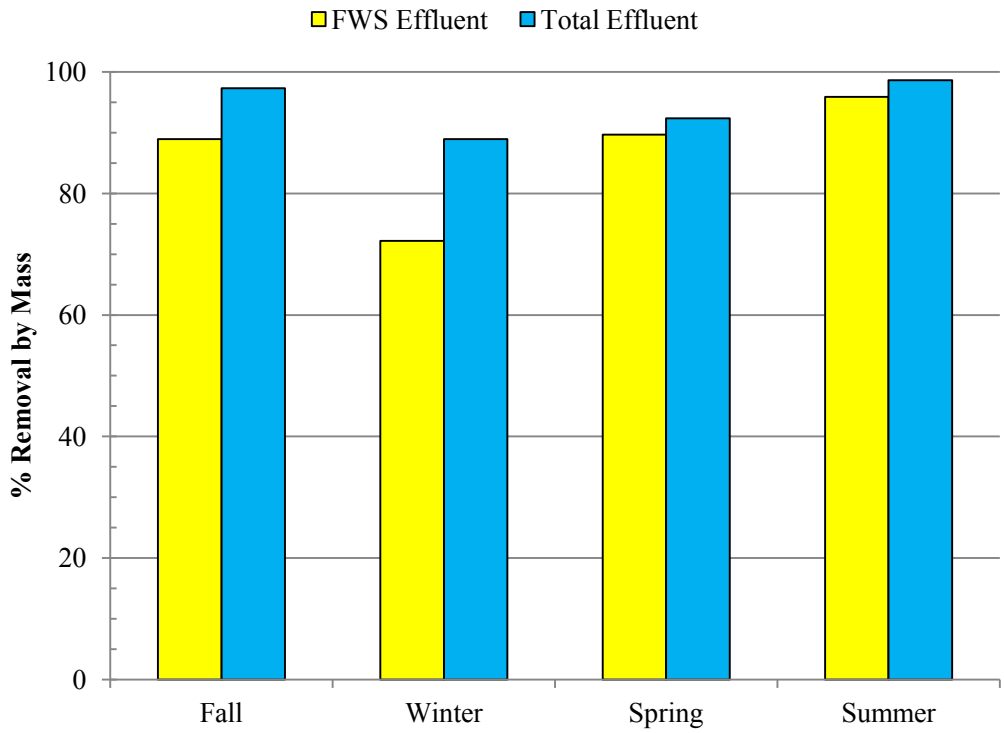


Figure 5-18. Percent anionic surfactant removal efficiencies.

Pathogen Indicators

Concentrations of *E. coli* are presented for every sampling event (Figure 5-19). Included in the plot are the single sample maximum and 30-d geometric mean regulations for primary contact recreational waters (EPA, 1986).

E. coli concentrations in the FWS and SF effluents never exceeded the primary contact maximum single-sample limit, except for two sampling periods in the winter. Note that primary contact regulations limit the 30-d sample mean, which by regulation requires at least five samples. Due to budget restrictions, less than five samples were collected monthly; therefore, as a conservative estimate, measured concentrations were also evaluated against the 30-d monthly geometric mean limit. Aside from winter measurements, only during March, 2009 were FWS effluent concentrations out of compliance with this limit. Non-winter SF effluents were always within geometric mean limits. EPA recommended reclaimed water regulations limit fecal coliform concentrations to non-detection on average. As such, the effluent from the graywater wetland would be out of compliance of these regulations nearly all year. Likewise, the effluent failed to meet the very strict microbial regulations developed by several states for reclaimed water.

Removal efficiencies of *E. coli* through the wetland were computed via concentration reductions (instead of using mass loading), and ranged from approximately 2.2 orders of magnitude in the fall to 1.0 order in the winter (Figure 5-20), with the FWS accounting for the majority of removal. Treatment efficiencies were evaluated by concentration change as is convention with microbials. Decreased removal rates in the winter likely resulted in the higher *E. coli* effluent concentrations observed during that time. Other wetland treatment studies have noted similar inverse relationships between effluent pathogen indicator concentrations and effluent temperatures (Winward et al., 2008b).

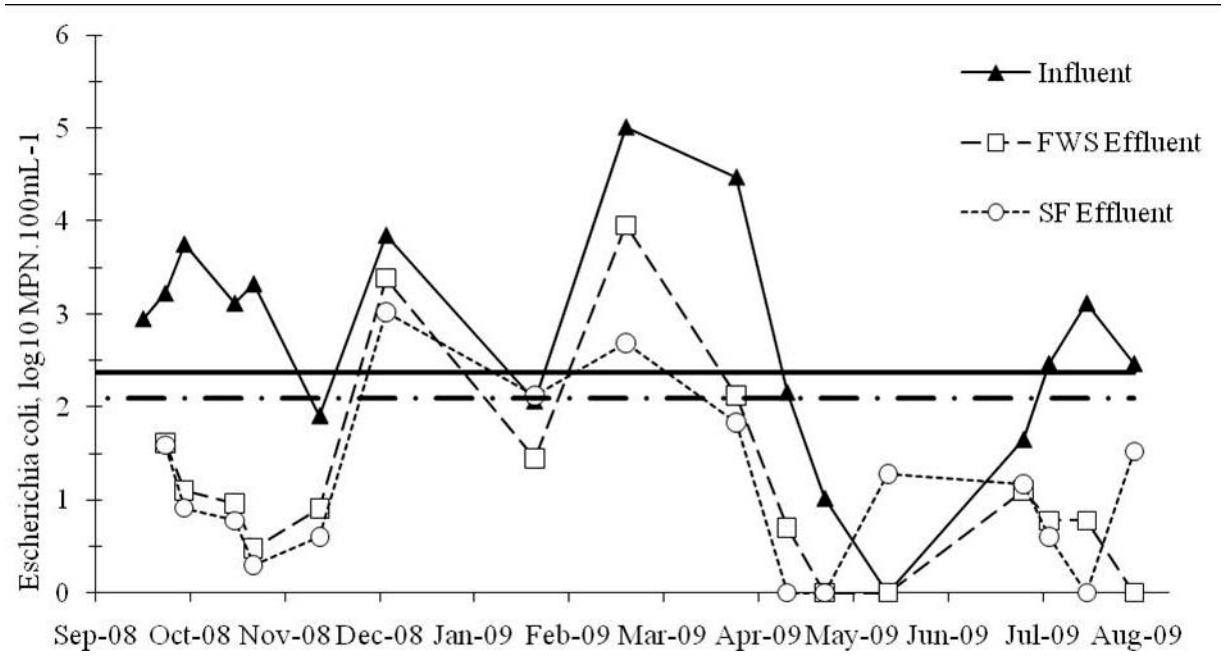


Figure 5-19. Single sample *Escherichia coli* log concentrations.

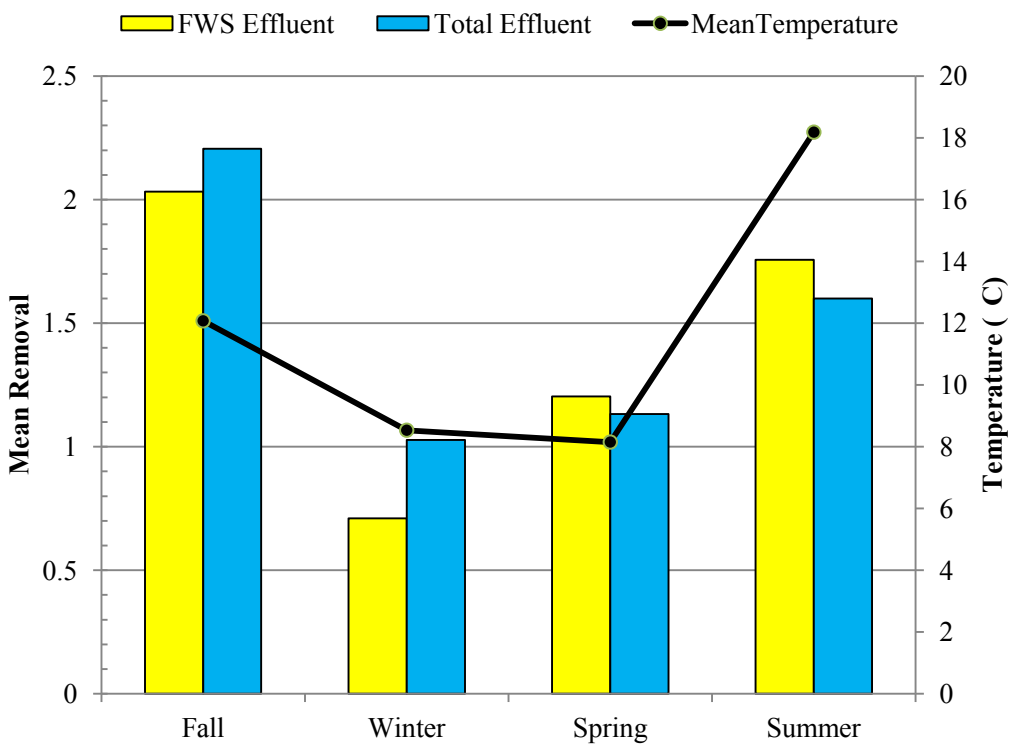


Figure 5-20. *Escherichia coli* mean removals by log concentrations.

Water Quality Statistical Analysis

Significant differences of the overall data (i.e., all seasons combined), summarized in Table 5-6, were computed using ANOVA single factor test (a=0.05) with each site tested against each the influent and each bed. Significance in Table 5-6 is shown through the use of letters, i.e., when two sites showed significant difference the letter is the same (e.g., A) and when there was no significant difference the letters are different (e.g., A, B, C). Results show that temperature, SC, TDS and *E. coli* had no significant change through the wetland. Of note is that *E. coli* concentrations decreased substantially through the wetland although the change was not statistically significant. BOD_5, TOC and TN were significantly reduced ($p < 0.05$) between each of the tested points while for other parameters (DO, turbidity, NH_3-N, TP, PO_4^{3-}, Cl^-, SO_4^{2-}, anionic surfactant, and TVS), a significant reduction was observed either between the influent and the FWS effluent or the influent and the SF effluent, but not between the FWS effluent and the SF effluent ($p < 0.05$). This analysis indicates overall performance of the wetland system.

Table 5-6. Water Quality Statistics for All Data Collected

Parameter	Location	Mean ± s	n	Significance	Range
Temp (^oC)	Influent	13.5 ±5.6	18	A	0.4 - 21.9
	FWS Effluent	11.9 ± 5.0	17	B	2.5 - 17.3
	SF Effluent	10.5 ± 5.6	17	C	1.9 - 20.6
DO (mg/L)	Influent	0.19 ± 0.17	17	A,B	0.0 - 0.6
	FWS Effluent	2.7 ± 2.0	17	A	0.7 - 6.7
	SF Effluent	2.2 ± 0.5	17	B	0.0 - 3.0
pH	Influent	NA[2]	18	NA	6.0 - 6.8
	FWS Effluent	NA	18	NA	6.4 - 7.1
	SF Effluent	NA	18	NA	6.1 - 7.3
SC (uS/cm)	Influent	229.7 ± 50.6	18	A	195.0 - 417.0
	FWS Effluent	239.9 ± 31.7	18	B	183.0 - 298.0
	SF Effluent	255.8 ± 44.1	18	C	183.0 - 334.0
Turbidity (NTU)	Influent	31.1 ± 18.2	18	A,B	8.1 - 66.7
	FWS Effluent	15.2 ±13.0	18	A	2.6 - 50.2
	SF Effluent	9.8 ± 5.7	18	B	1.5 - 19.4
BOD_5 (mg/L)	Influent	86.3 ± 40.3	16	A	30.6 - 161.6
	FWS Effluent	31.7 ± 24.0	16	A	3.0 - 89.3
	SF Effluent	12.7 ± 16.0	14	A	1.0 - 54.9
TOC (mg/L)	Influent	43.0 ± 17.6	17	A	14.2 - 80.4
	FWS Effluent	25.9 ± 10.3	17	A	8.5 - 45.5
	SF Effluent	14.0 ± 11.1	16	A	3.1 - 46.2
TN (mg/L)	Influent	13.5 ± 8.7	17	A	5.7 - 35.0
	FWS Effluent	5.6 ± 4.1	17	A	1.2 - 14.3

Parameter	Location	Mean ± s	n	Significance	Range
	SF Effluent	3.0 ± 2.6	17	A	0.0 - 8.2
NH_3 (mg/L)	Influent	7.9 ± 6.2	18	A,B	2.7 - 30.1
	FWS Effluent	2.1 ± 3.2	17	A	0.1 - 11.1
	SF Effluent	1.7 ± 2.0	16	B	0.1 - 6.0
TP (mg/L)	Influent	4.0 ± 1.8	18	A,B	0.4 - 9.3
	FWS Effluent	1.7 ± 1.0	17	A	2.5 - 3.9
	SF Effluent	1.3 ± 0.7	16	B	1.9 - 2.9
PO_4^{3-} (mg/L)	Influent	1.2 ± 1.0	18	A,B	0.1 - 3.0
	FWS Effluent	0.57 ± 0.74	17	A	0.1 - 2.6
	SF Effluent	0.4 ± 1.2	16	B	0.1 - 4.9
Cl^- (mg/L)	Influent	10.0 ± 3.2	18	A	1.1 - 18.8
	FWS Effluent	9.5 ± 3.1	17	B	0.5 - 15.4
	SF Effluent	6.8 ± 4.7	16	A	0.1 - 14.1
SO_4^{2-} (mg/L)	Influent	10.3 ± 5.0	18	A	0.2 - 19.5
	FWS Effluent	9.5 ± 4.6	17	B	0.1 - 16.8
	SF Effluent	5.0 ± 4.6	16	A	0.1 - 15.6
TSS (mg/L)	Influent	16.5 ± 7.2	17	A	7.2 - 28.1
	FWS Effluent	12.3 ± 7.3	17	B	2.5 - 30.0
	SF Effluent	8.0 ± 6.2	16	A	1.1 - 22.2
TDS (mg/L)	Influent	170.9 ± 51.5	11	A	93.5 - 263.1
	FWS Effluent	138.7 ± 57.8	12	B	10.3 - 218.4
	SF Effluent	136.6 ± 48.3	12	C	41.9 - 206.3
TVS (mg/L)	Influent	138.7 ± 50.9	11	A,B	64.7 - 239.0
	FWS Effluent	69.4 ± 33.6	11	A	10.0 - 115.8

Parameter	Location	Mean ± s	n	Significance	Range
	SF Effluent	62.8 ± 43.5	11	B	21.3 - 176.0
Anionic Surfactants as LAS (mg/L)	Influent	2.6 ± 1.2	13	A,B	1.1 - 5.7
	FWS Effluent	0.52 ± 0.62	13	A	0.1 - 2.4
	SF Effluent	0.28 ± 0.27	13	B	0.0 - 0.9
E. coli (MPN cells /100 mL)	Influent	$9.0 \times 10^3 \pm 2.5 \times 10^3$	17	A	$<10^0$ - 1.0×10^5
	FWS Effluent	$7.3 \times 10^2 \pm 23.0 \times 10^2$	16	B	$<10^0$ - 9.0×10^3
	SF Effluent	$1.2 \times 10^2 \pm 2{,}8 \times 10^2$	16	C	$<10^0$ - 1.1×10^3

[1] Same letters in "Significance" column for a pollutant denote significant differences at $P < 0.05$ for ANOVA, differing letters denote no significance.
[2] NA= not applicable.

Chapter 6 References

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Appendix A Data

Precipitation Record

The precipitation data as monitored by a Colorado Agricultural Meteorological Network (CoAgMet) weather station located approximately 650 m from the graywater wetland is presented in Figure-1.

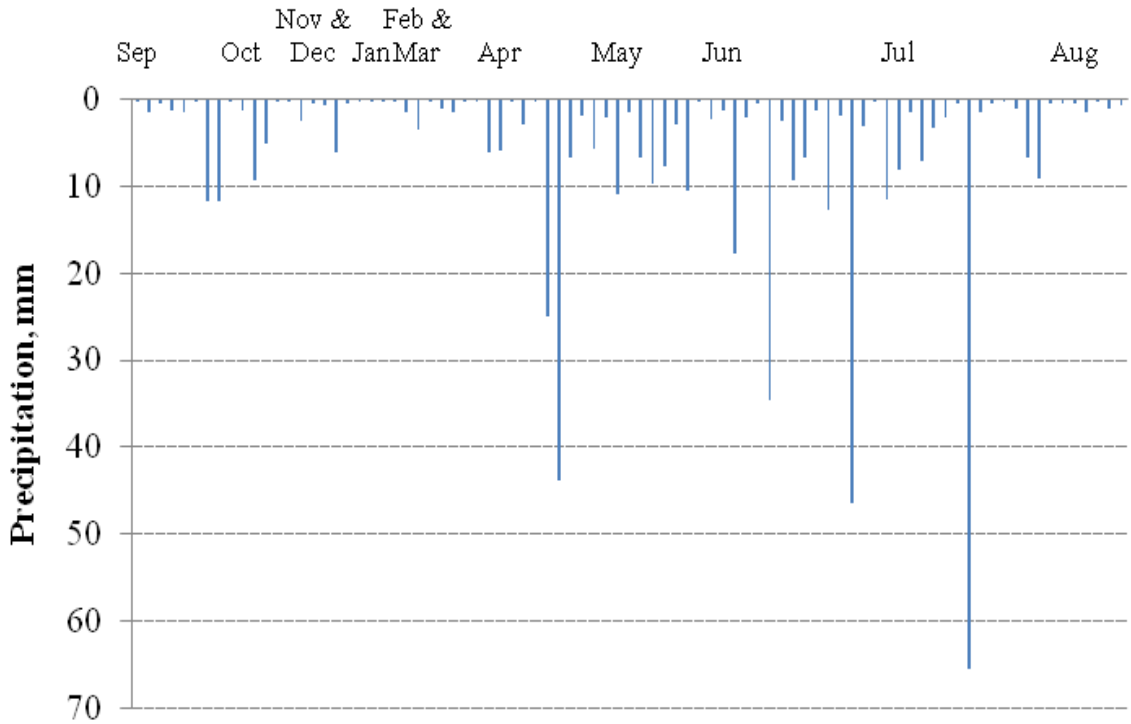

Figure-1. Precipitation record during the sampling period.

Flow Data

The flow data are presented in Table-1. The shaded rows in Table 1 represent sampling events.

Table-1. Flow Rates, Ice Thicknesses and Volumes

Date	Flow (L/d)			Ice Thickness (mm)		Volume (L)	
	Influent	FWS effluent	SW effluent	FWS	SF	FWS	SF
9/2/08	295	0	91	0	0	2010	2610
9/3/08	360	212	148	0	0	2010	2610
9/4/08	371	269	235	0	0	2010	2610
9/5/08	371	269	235	0	0	2010	2610
9/6/08	314	235	212	0	0	2010	2610
9/7/08	314	235	212	0	0	2010	2610
9/8/08	314	235	212	0	0	2010	2610
9/9/08	386	333	322	0	0	2010	2610
9/10/08	416	322	280	0	0	2010	2610

Date	Flow (L/d)			Ice Thickness (mm)		Volume (L)	
	Influent	FWS effluent	SW effluent	FWS	SF	FWS	SF
9/11/08	356	276	257	0	0	2010	2610
9/12/08	363	939	867	0	0	2010	2610
9/13/08	341	269	284	0	0	2010	2610
9/14/08	341	269	284	0	0	2010	2610
9/15/08	341	269	284	0	0	2010	2610
9/16/08	344	295	284	0	0	2010	2610
9/17/08	344	295	284	0	0	2010	2610
9/18/08	344	295	284	0	0	2010	2610
9/19/08	344	295	284	0	0	2010	2610
9/20/08	333	231	174	0	0	2010	2610
9/21/08	333	231	174	0	0	2010	2610
9/22/08	333	231	174	0	0	2010	2610
9/23/08	371	238	185	0	0	2010	2610
9/24/08	371	238	185	0	0	2010	2610
9/25/08	356	223	155	0	0	2010	2610
9/26/08	337	208	167	0	0	2010	2610
9/27/08	337	208	167	0	0	2010	2610
9/28/08	337	208	167	0	0	2010	2610
9/29/08	337	208	167	0	0	2010	2610
9/30/08	401	269	212	0	0	2010	2610
10/1/08	401	269	212	0	0	2010	2610
10/2/08	375	265	204	0	0	2010	2610
10/3/08	326	254	220	0	0	2010	2610
10/4/08	326	254	220	0	0	2010	2610
10/5/08	326	254	220	0	0	2010	2610
10/6/08	326	254	220	0	0	2010	2610
10/7/08	379	284	310	0	0	2010	2610
10/8/08	379	284	310	0	0	2010	2610
10/9/08	379	284	310	0	0	2010	2610
10/10/08	284	284	310	0	0	2010	2610
10/11/08	284	284	310	0	0	2010	2610
10/12/08	284	284	310	0	0	2010	2610
10/13/08	284	284	310	0	0	2010	2610
10/14/08	284	284	310	0	0	2010	2610
10/15/08	284	284	310	0	0	2010	2610
10/16/08	288	216	193	0	0	2010	2610
10/17/08	299	212	178	0	0	2010	2610
10/18/08	299	212	178	0	0	2010	2610
10/19/08	299	212	178	0	0	2010	2610
10/20/08	299	212	178	0	0	2010	2610
10/21/08	382	310	318	0	0	2010	2610
10/22/08	344	310	291	0	0	2010	2610
10/23/08	344	310	291	0	0	2010	2610
10/24/08	299	322	231	0	0	2010	2610
10/25/08	299	322	231	0	0	2010	2610
10/26/08	299	322	231	0	0	2010	2610

Date	Flow (L/d)			Ice Thickness (mm)		Volume (L)	
	Influent	FWS effluent	SW effluent	FWS	SF	FWS	SF
10/27/08	299	322	231	0	0	2010	2610
10/28/08	344	322	318	0	0	2010	2610
10/29/08	344	322	318	0	0	2010	2610
10/30/08	284	265	250	0	0	2010	2610
10/31/08	284	265	250	0	0	2010	2610
11/1/08	314	291	265	0	0	2010	2610
11/2/08	314	291	265	0	0	2010	2610
11/3/08	314	291	265	0	0	2010	2610
11/4/08	352	307	295	0	0	2010	2610
11/5/08	352	307	295	0	0	2010	2610
11/6/08	352	307	295	0	0	2010	2610
11/7/08	352	307	295	0	0	2010	2610
11/8/08	333	344	333	0	0	2010	2610
11/9/08	333	344	333	0	0	2010	2610
11/10/08	333	344	333	0	0	2010	2610
11/11/08	333	344	333	0	0	2010	2610
11/12/08	341	341	326	0	0	2010	2610
11/13/08	341	341	326	0	0	2010	2610
11/14/08	367	322	299	0	0	2010	2610
11/15/08	367	322	299	0	0	2010	2610
11/16/08	367	322	299	0	0	2010	2610
11/17/08	367	322	299	0	0	2010	2610
11/18/08	360	344	326	0	0	2010	2610
11/19/08	360	344	326	0	0	2010	2610
11/20/08	356	326	310	0	0	2010	2610
11/21/08	356	326	310	0	0	2010	2610
11/22/08	356	326	310	0	0	2010	2610
11/23/08	356	326	310	0	0	2010	2610
11/24/08	356	326	310	0	0	2010	2610
11/25/08	276	246	235	0	0	2010	2610
11/26/08	276	246	235	13	0	1896	2610
11/27/08	288	254	246	13	0	1896	2610
11/28/08	288	254	246	13	0	1896	2610
11/29/08	288	254	246	13	0	1896	2610
11/30/08	288	254	246	13	0	1896	2610
12/1/08	288	254	246	13	0	1896	2610
12/2/08	288	254	246	0	0	2010	2610
12/3/08	356	352	326	0	0	2010	2610
12/4/08	356	352	326	0	0	2010	2610
12/5/08	371	405	409	0	0	2010	2610
12/6/08	371	405	409	0	0	2010	2610
12/7/08	371	405	409	0	0	2010	2610
12/8/08	371	405	409	19	0	1843	2610
12/9/08	382	303	307	19	0	1843	2610
12/10/08	382	303	307	25	0	1790	2610
12/11/08	360	356	352	25	0	1790	2610

Date	Flow (L/d)			Ice Thickness (mm)		Volume (L)	
	Influent	FWS effluent	SW effluent	FWS	SF	FWS	SF
12/12/08	159	151	170	25	0	1790	2610
12/13/08	159	151	170	25	0	1790	2610
12/14/08	159	151	170	38	0	1684	2610
12/15/08	148	53	42	38	0	1684	2610
12/16/08	148	53	42	89	25	1294	2468
12/17/08	151	132	95	89	25	1294	2468
12/18/08	151	132	95	89	25	1294	2468
12/19/08	151	132	95	89	25	1294	2468
12/20/08	151	132	95	89	25	1294	2468
12/21/08	151	132	95	89	25	1294	2468
12/22/08	151	132	95	89	25	1294	2468
12/23/08	151	132	95	89	25	1294	2468
12/24/08	151	132	95	89	25	1294	2468
12/25/08	151	132	95	89	25	1294	2468
12/26/08	151	132	95	89	25	1294	2468
12/27/08	151	132	95	89	25	1294	2468
12/28/08	151	132	95	89	25	1294	2468
12/29/08	151	132	95	89	25	1294	2468
12/30/08	151	132	95	89	25	1294	2468
12/31/08	151	132	95	89	25	1294	2468
1/1/09	151	132	95	171	38	765	2396
1/2/09	151	117	121	171	38	765	2396
1/3/09	151	117	121	171	38	765	2396
1/4/09	151	117	121	171	38	765	2396
1/5/09	151	117	121	171	38	765	2396
1/6/09	151	117	121	171	38	765	2396
1/7/09	174	117	110	171	38	765	2396
1/8/09	167	117	110	152	13	874	2536
1/9/09	140	117	110	152	13	874	2536
1/10/09	140	117	110	152	13	874	2536
1/11/09	140	117	110	152	13	874	2536
1/12/09	140	117	110	152	13	874	2536
1/13/09	140	117	110	127	0	1033	2610
1/14/09	148	125	136	127	0	1033	2610
1/15/09	148	125	136	127	0	1033	2610
1/16/09	148	125	136	127	0	1033	2610
1/17/09	148	125	136	127	0	1033	2610
1/18/09	148	125	136	127	0	1033	2610
1/19/09	155	136	121	127	0	1033	2610
1/20/09	155	136	121	127	0	1033	2610
1/21/09	155	136	121	127	0	1033	2610
1/22/09	155	136	121	127	0	1033	2610
1/23/09	155	136	121	89	0	1294	2610
1/24/09	155	91	79	89	0	1294	2610
1/25/09	155	91	79	89	0	1294	2610
1/26/09	155	91	79	89	0	1294	2610

Date	Flow (L/d)			Ice Thickness (mm)		Volume (L)	
	Influent	FWS effluent	SW effluent	FWS	SF	FWS	SF
1/27/09	291	201	189	89	0	1294	2610
1/28/09	291	201	189	89	0	1294	2610
1/29/09	291	201	189	121	25	1075	2468
1/30/09	375	314	344	121	25	1075	2468
1/31/09	375	314	344	121	25	1075	2468
2/1/09	375	314	344	121	25	1075	2468
2/2/09	375	314	344	121	25	1075	2468
2/3/09	371	333	326	121	25	1075	2468
2/4/09	371	333	326	121	25	1075	2468
2/5/09	371	333	326	76	0	1389	2610
2/6/09	375	367	360	76	0	1389	2610
2/7/09	375	367	360	76	0	1389	2610
2/8/09	375	367	360	76	0	1389	2610
2/9/09	375	367	360	70	0	1438	2610
2/10/09	375	326	322	70	0	1438	2610
2/11/09	375	326	322	70	0	1438	2610
2/12/09	375	326	322	64	0	1484	2610
2/13/09	371	333	322	64	0	1484	2610
2/14/09	371	333	322	64	0	1484	2610
2/15/09	371	333	322	64	0	1484	2610
2/16/09	371	333	322	44	0	1635	2610
2/17/09	360	326	314	44	0	1635	2610
2/18/09	360	326	314	44	0	1635	2610
2/19/09	360	326	314	44	0	1635	2610
2/20/09	360	326	314	25	0	1790	2610
2/21/09	363	303	291	25	0	1790	2610
2/22/09	363	303	291	25	0	1790	2610
2/23/09	363	303	291	25	0	1790	2610
2/24/09	363	303	291	13	0	1896	2610
2/25/09	170	174	159	13	0	1896	2610
2/26/09	170	174	159	13	0	1896	2610
2/27/09	170	174	159	0	0	2010	2610
2/28/09	390	174	254	0	0	2010	2610
3/1/09	390	174	254	0	0	2010	2610
3/2/09	390	174	254	0	0	2010	2610
3/3/09	322	265	250	0	0	2010	2610
3/4/09	322	265	250	0	0	2010	2610
3/5/09	322	265	250	0	0	2010	2610
3/6/09	322	265	250	0	0	2010	2610
3/7/09	375	337	322	0	0	2010	2610
3/8/09	375	337	322	0	0	2010	2610
3/9/09	375	337	322	0	0	2010	2610
3/10/09	375	337	322	0	0	2010	2610
3/11/09	375	326	250	0	0	2010	2610
3/12/09	375	326	250	0	0	2010	2610
3/13/09	375	326	250	0	0	2010	2610

Date	Flow (L/d)			Ice Thickness (mm)		Volume (L)	
	Influent	FWS effluent	SW effluent	FWS	SF	FWS	SF
3/14/09	375	326	242	0	0	2010	2610
3/15/09	375	326	242	0	0	2010	2610
3/16/09	375	326	242	0	0	2010	2610
3/17/09	375	326	242	0	0	2010	2610
3/18/09	375	326	242	0	0	2010	2610
3/19/09	375	246	246	0	0	2010	2610
3/20/09	375	246	246	0	0	2010	2610
3/21/09	375	246	246	0	0	2010	2610
3/22/09	375	246	246	0	0	2010	2610
3/23/09	386	322	367	0	0	2010	2610
3/24/09	386	322	367	0	0	2010	2610
3/25/09	386	322	367	0	0	2010	2610
3/26/09	386	322	367	0	0	2010	2610
3/27/09	386	322	367	0	0	2010	2610
3/28/09	363	568	776	0	0	2010	2610
3/29/09	363	568	776	0	0	2010	2610
3/30/09	443	333	435	0	0	2010	2610
3/31/09	443	333	435	0	0	2010	2610
4/1/09	386	450	394	0	0	2010	2610
4/2/09	386	450	394	0	0	2010	2610
4/3/09	386	450	394	0	0	2010	2610
4/4/09	386	450	394	0	0	2010	2610
4/5/09	386	450	394	6	0	1953	2610
4/6/09	379	413	553	6	0	1953	2610
4/7/09	379	413	553	6	0	1953	2610
4/8/09	379	413	553	6	0	1953	2610
4/9/09	379	413	553	6	0	1953	2610
4/10/09	379	413	553	0	0	2010	2610
4/11/09	333	238	235	0	0	2010	2610
4/12/09	333	238	235	0	0	2010	2610
4/13/09	333	238	235	0	0	2010	2610
4/14/09	435	178	174	0	0	2010	2610
4/15/09	435	178	174	0	0	2010	2610
4/16/09	435	178	174	0	0	2010	2610
4/17/09	394	1075	1283	0	0	2010	2610
4/18/09	394	1075	1283	0	0	2010	2610
4/19/09	394	1075	1283	0	0	2010	2610
4/20/09	394	1075	1283	0	0	2010	2610
4/21/09	394	1075	1283	0	0	2010	2610
4/22/09	367	303	307	0	0	2010	2610
4/23/09	367	303	307	0	0	2010	2610
4/24/09	367	303	307	0	0	2010	2610
4/25/09	367	303	307	0	0	2010	2610
4/26/09	367	303	307	0	0	2010	2610
4/27/09	367	303	307	0	0	2010	2610
4/28/09	367	303	307	0	0	2010	2610

Date	Flow (L/d)			Ice Thickness (mm)		Volume (L)	
	Influent	FWS effluent	SW effluent	FWS	SF	FWS	SF
4/29/09	352	291	269	0	0	2006	2610
4/30/09	352	291	269	0	0	2006	2610
5/1/09	352	291	269	0	0	2006	2610
5/2/09	26	110	167	0	0	2006	2610
5/3/09	26	110	167	0	0	2006	2610
5/4/09	26	110	167	0	0	2006	2610
5/5/09	284	197	159	0	0	2006	2610
5/6/09	284	197	159	0	0	2006	2610
5/7/09	284	197	159	0	0	2006	2610
5/8/09	284	197	159	0	0	2006	2610
5/9/09	227	246	257	0	0	2006	2610
5/10/09	227	246	257	0	0	2006	2610
5/11/09	227	246	257	0	0	2006	2610
5/12/09	227	246	257	0	0	2006	2610
5/13/09	227	246	257	0	0	2006	2610
5/14/09	201	117	68	0	0	2006	2610
5/15/09	201	117	68	0	0	2006	2610
5/16/09	201	117	68	0	0	2006	2610
5/17/09	201	117	68	0	0	2006	2610
5/18/09	201	117	68	0	0	2006	2610
5/19/09	132	19	0	0	0	2006	2610
5/20/09	106	4	0	0	0	2006	2610
5/21/09	106	4	0	0	0	2006	2610
5/22/09	114	246	269	0	0	2006	2610
5/23/09	114	246	269	0	0	2006	2610
5/24/09	114	246	269	0	0	2006	2610
5/25/09	114	246	269	0	0	2006	2610
5/26/09	114	246	269	0	0	2006	2610
5/27/09	53	30	11	0	0	2006	2610
5/28/09	53	30	11	0	0	2006	2610
5/29/09	53	30	11	0	0	2006	2610
5/30/09	53	30	11	0	0	2006	2610
5/31/09	53	30	11	0	0	2006	2610
6/1/09	117	34	79	0	0	2006	2610
6/2/09	117	34	79	0	0	2006	2610
6/3/09	117	34	79	0	0	2006	2610
6/4/09	140	587	632	0	0	2006	2610
6/5/09	140	587	632	0	0	2006	2610
6/6/09	140	587	632	0	0	2006	2610
6/7/09	140	587	632	0	0	2006	2610
6/8/09	140	587	632	0	0	2006	2610
6/9/09	140	587	632	0	0	2006	2610
6/10/09	189	87	79	0	0	2006	2610
6/11/09	189	87	79	0	0	2006	2610
6/12/09	189	87	79	0	0	2006	2610
6/13/09	189	87	79	0	0	2006	2610

Date	Flow (L/d)			Ice Thickness (mm)		Volume (L)	
	Influent	FWS effluent	SW effluent	FWS	SF	FWS	SF
6/14/09	189	87	79	0	0	2006	2610
6/15/09	189	87	79	0	0	2006	2610
6/16/09	102	344	132	0	0	2006	2610
6/17/09	102	344	132	0	0	2006	2610
6/18/09	484	238	132	0	0	2006	2610
6/19/09	484	238	132	0	0	2006	2610
6/20/09	484	238	132	0	0	2006	2610
6/21/09	484	238	132	0	0	2006	2610
6/22/09	484	238	132	0	0	2006	2610
6/23/09	534	942	132	0	0	2006	2610
6/24/09	534	942	132	0	0	2006	2610
6/25/09	534	942	132	0	0	2006	2610
6/26/09	534	942	132	0	0	2006	2610
6/27/09	360	310	273	0	0	2006	2610
6/28/09	360	310	273	0	0	2006	2610
6/29/09	360	310	273	0	0	2006	2610
6/30/09	360	310	273	0	0	2006	2610
7/1/09	522	280	4	0	0	2006	2610
7/2/09	522	280	4	0	0	2006	2610
7/3/09	522	280	4	0	0	2006	2610
7/4/09	148	363	4	0	0	2006	2610
7/5/09	148	363	4	0	0	2006	2610
7/6/09	148	363	4	0	0	2006	2610
7/7/09	231	121	8	0	0	2006	2610
7/8/09	231	121	8	0	0	2006	2610
7/9/09	231	121	8	0	0	2006	2610
7/10/09	231	121	8	0	0	2006	2610
7/11/09	117	4	8	0	0	2006	2610
7/12/09	117	4	8	0	0	2006	2610
7/13/09	117	4	8	0	0	2006	2610
7/14/09	269	4	4	0	0	2006	2610
7/15/09	269	4	4	0	0	2006	2610
7/16/09	522	0	4	0	0	2006	2610
7/17/09	522	0	4	0	0	2006	2610
7/18/09	522	0	4	0	0	2006	2610
7/19/09	431	0	4	0	0	2006	2610
7/20/09	431	0	4	0	0	2006	2610
7/21/09	155	409	4	0	0	2006	2610
7/22/09	155	409	4	0	0	2006	2610
7/23/09	155	409	4	0	0	2006	2610
7/24/09	155	409	4	0	0	2006	2610
7/25/09	155	409	4	0	0	2006	2610
7/26/09	155	409	4	0	0	2006	2610
7/27/09	155	409	4	0	0	2006	2610
7/28/09	155	409	4	0	0	2006	2610
7/29/09	235	250	4	0	0	2006	2610

Date	Flow (L/d)			Ice Thickness (mm)		Volume (L)	
	Influent	FWS effluent	SW effluent	FWS	SF	FWS	SF
7/30/09	235	250	4	0	0	2006	2610
7/31/09	235	250	4	0	0	2006	2610
8/1/09	276	57	4	0	0	2006	2610
8/2/09	276	57	4	0	0	2006	2610
8/3/09	276	57	4	0	0	2006	2610
8/4/09	367	91	4	0	0	2006	2610

Water Quality Data

The following tables contain the water quality data collected during the one-year monitoring period. The tables are broken down into different seasons, fall (Table-2), winter and spring (Table-3) and summer (Table--4).

Table-2. Fall Graywater Quality Data

Season		Fall	Fall	Fall	Fall	Fall	Fall	Fall
Date		9/19/2008	9/24/2008	10/1/2008	10/7/2008	10/23/2008	10/29/2008	11/19/2008
DO (mg/L)	Influent	0.2	0.1	0.1	0.1	0.2	0.1	0.0
	FWS Effluent		3.0	1.8	4.4	5.5	3.2	1.3
	SF Effluent		1.4	2.3	3.0	2.8	2.8	2.0
Temp (˚C)	Influent	17.6	13.7	13.1	10.8	5.9	12.0	14.8
	FWS Effluent		15.1	14.1	12.9	8.7	9.3	7.8
	SF Effluent		16.3	15.6	13.5	8.6	9.8	10.3
pH	Influent	6.5	6.1	6.0	6.2	6.2	6.2	6.6
	FWS Effluent	6.9	6.7	6.7	6.7	6.8	6.6	6.6
	SF Effluent	6.8	6.5	6.4	6.5	6.3	6.3	6.4
S C (uS/cm)	Influent	218.0	197.4	201.2	208.0	223.0	219.0	213.0
	FWS Effluent	235.0	240.0	249.7	266.0	264.0	255.0	267.0
	SF Effluent	256.0	283.0	294.2	299.0	287.0	288.0	271.0
BOD_5 (mg/L)	Influent	76.7	79.3			53.8	88.1	109.5
	FWS Effluent		19.3	20.0		18.9	28.3	37.4
	SF Effluent			4.5		2.4	1.0	4.7
TOC (mg/L)	Influent	46.5	33.1	47.9	30.8	32.1	41.2	52.1
	FWS Effluent	26.5	17.5	23.0	28.0	23.2	21.1	24.9
	SF Effluent	13.0	10.6	6.5	8.4	6.6	3.1	11.6
DOC (mg/L)	Influent	29.2	22.3					33.2
	FWS Effluent		13.5					18.1
	SF Effluent							9.8
TN (mg/L)	Influent	8.1	6.0	8.0	8.2	9.1	9.9	13.2

Season		Fall	Fall	Fall	Fall	Fall	Fall	Fall
Date		9/19/2008	9/24/2008	10/1/2008	10/7/2008	10/23/2008	10/29/2008	11/19/2008
	FWS Effluent	6.1	2.8	3.0	2.8	4.6	4.6	9.4
	SF Effluent	2.5	1.6	1.1	0.8	0.7	0.5	1.7
NH_3, mg/L	Influent	4.9	3.1	5.2	4.5	6.4	6.5	8.3
	FWS Effluent		0.7	0.1	0.2	1.3	1.7	6.2
	SF Effluent			0.4	0.2	0.2	0.2	0.9
NO_3^- (mg/L)	Influent	BDL[1]	BDL	BDL	BDL	BDL	BDL	BDL
	FWS Effluent		BDL	BDL	BDL	0.1	0.1	0.2
	SF Effluent			0.1	0.2	0.3	0.5	0.1
DTN (mg/L)	Influent	6.7	5.9					11.0
	FWS Effluent		2.3					7.9
	SF Effluent							1.5
TP (mg/L)	Influent	2.4	2.3	2.8	3.1	3.3	3.3	3.8
	FWS Effluent		0.6	1.0	1.6	1.6	1.4	1.8
	SF Effluent			1.9	1.5	0.7	0.5	0.7
PO_4^{3-} (mg/L)	Influent	0.9	0.7	1.1	1.4	1.7	1.4	1.9
	FWS Effluent	---	0.1	0.1	0.1	0.1	0.1	0.5
	SF Effluent	---	---	0.1	BDL	BDL	0.1	BDL
E. coli (MPN cells /100 mL)	Influent		884.0	1658.0	5610.0	1300.0	2103.0	80.0
	FWS Effluent			41.0	12.5	9.1	3.0	8.0
	SF Effluent			38.7	8.2	6.0	2.0	4.0
Anionic Surfactants as LAS (mg/L)	Influent						1.1	2.3
	FWS Effluent						0.2	0.2
	SF Effluent						0.0	0.1
Cl^- (mg/L)	Influent	10.9	8.9	9.7	8.9	10.1	8.9	9.5
	FWS Effluent		8.6	8.9	9.1	10.3	9.9	11.0
	SF Effluent			6.9	8.1	8.3	8.5	11.2
SO_4^{2-} (mg/L)	Influent	14.1	11.4	11.0	12.4	16.3	10.8	5.3
	FWS Effluent		16.8	11.8	11.0	14.2	11.4	10.6
	SF Effluent			3.2	6.2	15.6	12.0	7.6
Turbidity, (NTU)	Influent	28.3	10.6	8.1	11.2	17.0	22.4	40.7
	FWS Effluent	25.1	6.0	5.6	7.9	9.2	9.2	12.5
	SF Effluent	13.4	7.9	7.7	7.3	2.8	1.5	2.3
TS (mg/L)	Influent							161.5
	FWS Effluent							91.5
	SF Effluent							99.5
TS S (mg/L)	Influent	24.8	16.0	12.3	10.4	8.3	12.9	

Season		Fall	Fall	Fall	Fall	Fall	Fall	Fall
Date		9/19/2008	9/24/2008	10/1/2008	10/7/2008	10/23/2008	10/29/2008	11/19/2008
	FWS Effluent		5.3	13.2	20.4	14.1	8.7	8.2
	SF Effluent			5.8	8.0	1.7	1.1	3.2
TDSo (mg/L)	Influent							
	FWS Effluent							91.5
	SF Effluent							99.5
TVS (mg/L)	Influent							118.0
	FWS Effluent							69.5
	SF Effluent							60.5
VSS (mg/L)	Influent							
	FWS Effluent							6.2
	SF Effluent							2.0

[1] BDL - below detection limit.

Table-3. Winter and Spring Graywater Quality Data

Season		Winter	Winter	Winter	Spring	Spring	Spring	Spring
Date		12/10/2008	1/26/2009	2/24/2009	3/31/2009	4/16/2009	4/28/2009	5/18/2009
DO (mg/L)	Influent	0.1	0.1	0.1	0.3	0.1	0.2	0.6
	FWS Effluent	1.1	1.1	1.2	6.7	4.3	6.3	1.7
	SF Effluent	2.4	2.2	2.1	2.7	1.8	2.7	2.8
Tem (˚C)	Influent	14.8	11.3	21.9	0.4	8.1	6.5	16
	FWS Effluent	7.8	2.5	2.8	4.7	7.2	7.4	11.5
	SF Effluent	10.3	1.9	3.5	4.5	9.0	8.5	14
pH	Influent	6.7	6.8	6.6	6.4	6.3	6.2	6.2
	FWS	6.7	6.4	6.4	6.7	6.8	6.7	6.5
	SF Effluent	6.7	6.5	6.7	6.6	6.6	6.5	7.3
SC (uS/cm)	Influent	260.0	256.0	220.0	205.0	195.0	208.0	237.0
	FWS Effluent	298.0	286.0	241.0	210.0	217.0	183.0	263.0
	SF Effluent	308.0	334.0	261.0	219.0	271.0	183.0	200.0
BOD_5 (mg/L)	Influent	118.6	109.3	150.3	120.4	92.5	161.6	39.9
	FWS Effluent	70.6	62.3	89.3	30.3	29.3	10.1	27.4
	SF Effluent		54.9	36.3	16.2	26.4	4.3	3.0
TOC (mg/L)	Influent	54.8	61.9	54.4	80.4	44.6	61.4	22.3
	FWS Effluent	36.0	39.4	29.8	45.5	29.8	18.4	40.8
	SF Effluent	14.4	33.0	12.5	46.2	21.8	9.81	10.8
DOC (mg/L)	Influent	35.9	37.6	30.3	36.2	26.5	33.7	14.8
	FWS Effluent	24.5	31.4	17.8	24.5	20.2	17.8	26.9
	SF Effluent	12.8	26.5	5.71	20.1	19.1	8.3	8.4
TN (mg/L)	Influent	19.8	16.7	12.3	10.8	5.7	10.8	35
	FWS Effluent	12.7	8.6	11.1	2.0	3.5	1.9	14.3
	SF Effluent	5.2	8.2	7.9	4.8	5.2	2.5	5.2
NH_3	Influent	13.9	11.0	7.0	5.8	7.9	2.7	8.5

Season		Winter	Winter	Winter	Spring	Spring	Spring	Spring
Date		12/10/2008	1/26/2009	2/24/2009	3/31/2009	4/16/2009	4/28/2009	5/18/2009
(mg/L)	FWS Effluent	11.1	5.4	6.2	0.2	0.3	0.3	0.19
	SF Effluent	4.0	6.0	5.1	2.3	4.3	1.6	0.44
NO_3^- (mg/L)	Influent	BDL	BDL	BDL	BDL	BDL	BDL	BDL
	FWS Effluent	BDL	0.2	BDL	BDL	BDL	BDL	BDL
	SF Effluent	0.2	0.2	0.2	BDL	BDL	BDL	BDL
DTN (mg/L)	Influent	17.4	15.1	10.7	10.2	4.0	9.4	32.3
	FWS Effluent	11.2	7.8	9.7	3.8	1.6	1.8	7.0
	SF Effluent	5.2	7.9	7.3	5.1	4.8	2.3	4.0
TP (mg/L)	Influent	4.7	6.1	3.7	3.5	2.2	3.9	3.4
	FWS Effluent	3.1	3.9	3.2	2.9	1.8	0.7	2.0
	SF Effluent	0.8	2.9	2.6	1.3	1.5	1.3	1.5
PO_4^{3-} (mg/L)	Influent	3.0	2.7	2.2	BDL	BDL	BDL	BDL
	FWS Effluent	1.6	0.9	1.1	BDL	BDL	BDL	BDL
	SF Effluent	BDL	BDL	BDL	BDL	BDL	BDL	BDL
E. coli (MPN cells /100 mL)	Influent	7009.0	115.0	102035.0	29546.7	143.0	10.3	<1
	FWS Effluent	2420.0	28.0	8975.0	133.3	5.0	<1	<1
	SF Effluent	1054.0	132.0	488.4	67.7	1.0	<1	19
Anionic Surfactants as LAS, (mg/L)	Influent	3.2	3.1	5.7	1.9	2.7	2.3	1.2
	FWS Effluent	0.8	0.9	2.4	0.5	0.6	0.1	0.14
	SF Effluent	0.3	0.8	0.9	0.3	0.4	0.1	0.08
Cl^- (mg/L)	Influent	9.4	10.3	10.9	>10	>10	>10	>10
	FWS Effluent	11.9	10.7	10.7	>10	>10	>10	>10
	SF Effluent	14.1	12.2	10.5	>10	>10	3.3	>10
SO_4^{2-} (mg/L)	Influent	7.2	6.8	8.3	6.1	6.1	10.2	
	FWS Effluent	9.2	9.9	10.5	7.2	3.8	0.1	
	SF Effluent	4.1	10.1	3.9	4.4	0.8	0.9	
Turbidity (NTU)	Influent	58.2	51.3	66.7	46.5	40.8	54.6	18.9
	FWS Effluent	38.1	32.6	50.2	10.2	7.4	2.6	15.4
	SF Effluent	10.1	19.3	19.4	9.7	4.6	3.4	10.0
TS (mg/L)	Influent	180.0	110.0	164.7	172.0	250.0	288.0	148.7
	FWS Effluent	151.7	138.3	112.0	135.3	228.0	16.0	228.0
	SF Effluent	154.2	201.7	48.7	135.3	200.0	209.3	232.0
TSS (mg/L)	Influent	28.1	16.5	28.1	8.5	10.2	24.9	7.2
	FWS Effluent	9.4	7.7	11.1	18.4	22.7	5.7	30
	SF Effluent	5.4	8.5	6.8	5.2	2.8	3.1	12.7
TDS (mg/L)	Influent	151.9	93.5	136.5	163.5	239.8	263.1	141.5
	FWS Effluent	142.3	130.6	100.9	116.9	205.3	10.3	198.0
	SF Effluent	148.8	193.1	41.9	130.1	197.2	206.3	122.7
TVS (mg/L)	Influent	130.0	94.4	144.7		122.0	239.0	64.7
	FWS Effluent	115.8	95.6	84.0		101.0	10.0	94.0
	SF Effluent	73.3	91.1	32.7		72.0	176.0	40.7
VSS (mg/L)	Influent	24.4	13.3	21.7	7.5	8.4	23.1	6.7
	FWS Effluent	7.2	6.4	7.5	15.5	19.9	2.9	23.5
	SF Effluent	3.4	5.8	3.9	4.3	2.4	1.9	5.7

[1] BDL - below detection limit.

Table-4. Summer Graywater Quality Data

Season		Summer	Summer	Summer	Summer
Date		6/30/2009	7/8/2009	7/20/2009	8/4/2009
DO (mg/L)	Influent		0.08	0.30	0.53
	FWS Effluent	1.1	0.9	1.6	0.71
	SF Effluent	2.1	1.3	1.5	1.8
Temp (˚C)	Influent	20.7	17.5	19.1	18.2
	FWS Effluent	16.1	16.5	17.3	17.1
	SF Effluent	17.8	18.5	20.6	18.8
pH	Influent	6.2	6.7	6.8	6.8
	FWS	6.5	6.7	6.7	7.1
	SF Effluent	7.3	6.3	6.5	6.9
S C (uS/cm)	Influent	203.0	247.0	417.0	207.7
	FWS Effluent	226.0	205.0	220.0	192.9
	SF Effluent	223.0	204.0	221.0	202.5
BOD_5 (mg/L)	Influent	31.2	46.8	71.6	30.6
	FWS Effluent	40.7	11.3	8.6	3.0
	SF Effluent	12.9	2.2	6.3	2.5
TOC (mg/L)	Influent	14.5		38.2	14.2
	FWS Effluent	12.5		14.9	8.5
	SF Effluent			8.8	6.2
DOC (mg/L)	Influent	12.7		22.0	10.8
	FWS Effluent	1.7		10.1	8.8
	SF Effluent			5.6	5.7
TN, (mg/L)	Influent	10.3		34.3	10.7
	FWS Effluent	4.4		2.7	1.2
	SF Effluent			1.8	0.8
NH_3 mg/L	Influent	4.1	4.4	30.1	7.2
	FWS Effluent	0.61	0.13	1.2	0.40
	SF Effluent	0.16	0.13	0.47	0.84
NO_3^- (mg/L)	Influent	<0.2	<0.2	<0.2	<0.2
	FWS Effluent	<0.2	<0.2	<0.2	<0.2
	SF Effluent	<0.2	<0.2	1.3	<0.2
DTN (mg/L)	Influent	9.0		32.2	10.1
	FWS Effluent	2.6		2.4	1.2
	SF Effluent	0.76		1.4	0.7
TP (mg/L)	Influent	2.2	6.3	9.3	5.7
	FWS Effluent	1.5	0.95	0.85	0.76
	SF Effluent	1.5	0.60	0.65	0.53
(mg/L)	Influent	0.24	<0.2	2.6	<0.2
	FWS Effluent	<0.2	0.41	2.6	1.5
	SF Effluent	<0.2	4.9	<0.2	<0.2
E. coli (MPN cells/100 mL	Influent	44.3	290.9	1299.7	290.9
	FWS Effluent	12.7	6	6	<1
	SF Effluent	14.7	4	<1	33
Anionic Surfactants as LAS (mg/L)	Influent	2.0	3.3	3.7	1.8
	FWS Effluent	0.35	0.23	0.26	0.20
	SF Effluent	0.18	0.12	0.26	0.20

Season		Summer	Summer	Summer	Summer
Date		6/30/2009	7/8/2009	7/20/2009	8/4/2009
Cl^- (mg/L)	Influent	>10	9.9	>10	1.1
	FWS Effluent	4.5	0.51	>10	>10
	SF Effluent	0.20	<0.2	1.0	0.20
SO_4^{2-} mg/L	Influent	0.24	17.3	6.7	19.5
	FWS	<0.2	10.3	114.1	12.9
	SF Effluent	<0.2	2.1	8.5	<0.2
Turbidity(NTU)	Influent	14.5	21.3	23.8	24.4
	FWS Effluent	15.9	8.0	4.7	12.8
	SF Effluent	18.5	13.9	13.6	10.4
TS (mg/L)	Influent	209.3	236.0	166.0	150.7
	FWS Effluent	194.0	232.0	132.0	145.0
	SF Effluent	130.7	112.7	144.0	176.0
TSS (mg/L)	Influent	10.4	22.0	20.3	19.3
	FWS Effluent	14.9	13.6	3.7	2.5
	SF Effluent	22.2	11.7	20.3	9.0
TDS (mg/L)	Influent	198.9	214.0	145.7	131.3
	FWS Effluent	179.1	218.4	128.3	142.5
	SF Effluent	108.5	100.9	123.7	167.1
TVS (mg/L)	Influent	69.3	93.3	67.3	72.0
	FWS Effluent	51.0	75.3	25.0	42.0
	SF Effluent	34.0	28.7	21.3	60.7
VSS (mg/L)	Influent	9.1	19.1	18.4	16.3
	FWS Effluent	11.3	8.4	1.9	0.80
	SF Effluent	10.4	6.1	6.8	3.6