THESIS

FREE WATER SURFACE AND HORIZONTAL SUBSURFACE FLOW CONSTRUCTED WETLANDS: A COMPARISON OF PERFORMANCE IN TREATING DOMESTIC GRAYWATER

Submitted by

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ABSTRACT

FREE WATER SURFACE AND HORIZONTAL SUBSURFACE FLOW CONSTRUCTED WETLANDS: A COMPARISON OF PERFORMANCE IN TREATING DOMESTIC GRAYWATER

Communities throughout the United States and abroad are seeking innovative approaches to sustaining their freshwater resources. Graywater reuse for non-potable demands is gaining popularity because it allows for the reuse of minimally contaminated wash water, generated and treated on site. Graywater is defined as any wastewater generated at the home or office including wastewater from the laundry, shower, and bathroom sinks but excluding water from the toilets, kitchen sinks, and dishwasher. When compared to other wastewater generated in the home, graywater is contaminated with lower concentrations of organics, solids, nutrients, and pathogens. These characteristics make the water suitable for reuse with negligible treatment when compared to other domestic wastewater sources. Graywater reuse for non-potable demands reduces the demand for treated water and preserves source waters. One method of treating graywater at a community scale for irrigation reuse is constructed wetlands. Despite widespread interest in this innovative approach, limited guidance is available on the design and operation of constructed wetlands specific to graywater treatment. The foremost objective of this research was to compare the performance of a free water surface constructed wetland (FWS) to a horizontal subsurface constructed wetland (SF) for graywater treatment and to assess their ability to meet water quality standards for surface discharge and reuse. This was done by comparison of percent (%) mass removal rates and requisite surface areas (SA) based on determined removal rates (*k*). Aerial loading rates were compared to EPA suggested aerial loading rates in an attempt to provide recommendations for target effluent concentrations. Determining contaminant removal rates is important for creating wetland design standards for graywater treatment and reuse. Contaminant removal rates were evaluated over the summer

and fall of 2010 and 2011 for a SF wetland. These removal rates were compared to the removal rates evaluated over a two year period (2008-2010) for a FWS wetland. Another objective was to determine the % mass removal of three common anionic surfactants in constructed wetlands (both FWS and SF) and finally, the possibility of incorporating constructed wetlands into greenhouse community garden centers as an option to reduce the losses resulting from evapotranspiration (ET) in arid climates was explored briefly.

The results indicate that SF wetlands provide relatively stable and more efficient treatment year round when compared to FWS wetlands. In particular, the SF wetland showed statistically significant higher mass removal of both biological oxygen demand (BOD_5) and total nitrogen (TN) than the FWS wetland during winter months ($P=0.1$ and 0.005; $\alpha=0.1$). When all the seasons were compared for each wetland individually there was a statistically significant degree of removal for BOD_5 and TN between the seasons in the FWS wetland ($P=0.09$ and 0.04; $\alpha=0.1$) while there was none in the SF wetland ($P=1.0$ and 0.9; $\alpha=0.1$). These results are consistent with other findings in the literature. When mass removals were compared to HLRs, the trends support the ability of SF wetlands to function across a wide range of HLRs and climatic conditions, whereas FWS wetlands are less capable of performing well under less than ideal conditions. Results of the k-C^* and SA analyses, though limited in their completeness, suggest once again that SF wetlands are capable of increased rates of removal not only during the warm summer months but also during the winter and transition months. Specifically, nitrification and denitrification processes may be contributing to TN removal in the SF wetland, particularly during senescent periods. Surfactant removal was also consistent with findings in the literature, with 50% removal of LAS and greater than 70% removal of AES/AS, suggesting that LAS is more persistent.

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DEDICATION

In Loving Memory

of

Natasha Gutierrez

May 22, 1987 – October 8, 2005

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CHAPTER 1: INTRODUCTION

Water shortages have served to motivate individuals and municipalities to begin reusing wastewater. While wastewater reuse in the form of reclaimed wastewater is emerging as an essential instrument for effective management of water, graywater reuse may provide even more benefit. Graywater is minimally contaminated when compared with domestic wastewater and thus requires less treatment to be reused for non-potable demands such as irrigation. In addition, graywater reuse promotes preservation of freshwater sources, potentially reduces pollutants in the environment, and reduces total costs in treating water and wastewater (Jefferson *et al*, 2001).

Although extensive treatment is not required, it is important to treat graywater to meet the standards for limited contact use. One method of treatment is through application of constructed wetlands. Constructed wetlands have low capital cost and limited maintenance after implementation and have been utilized for wastewater treatment for decades. There have been a variety of studies conducted on the performance of constructed wetlands and hydroponic systems as treatment options for graywater (Dallas and Ho, 2005; Frazer-Williams *et al*, 2008; Garland *et al*, 2000; Garland *et al*, 2004; Gross *et al*, 2007a; Gross *et al*, 2007b; Jokerst *et al*, 2011; Kadewa *et al*, 2010; Masi *et al*, 2010). Jokerst *et al* (2011) examined treatment of graywater through a constructed free water surface (FWS) and horizontal subsurface flow (SF) wetland series, as is common in wastewater treatment. Similar to previous studies, the results indicated that both types of constructed wetlands are capable of significant removal of key water quality constituents but unlike previous studies Jokerst *et al* (2011) concluded that using both a FWS and SF wetland, in series, may be excessive in the case of graywater treatment. One goal of the constructed wetland studies has been to better understand the ability of constructed wetlands to treat graywater to sufficient quality for discharge to surface waters or for irrigation and toilet reuse. Depending on the end application, the water quality parameters of importance vary. In the case of reuse for irrigation some nitrogen and phosphorus is beneficial while pathogens must be closely monitored. In the case of surface discharge there are nitrogen and phosphorus concerns related to receiving water health (i.e., algal blooms, nitrogen toxicity). In addition, regulations regarding nitrogen and phosphorus in discharge waters will become more stringent in the next ten years.

Table 1.1: Comparison of important water quality parameters

Application	BOD_5 (avg 30 day, mg L^{-1})	TSS (avg 30 day, mg L^{-1})	pH	*E.coli* (CPN/100mL)	Ammonia/TIN (mg N/L)	TP (mg P/L)
Reuse	30	30	6-9	25-500/100mL	n/a	n/a
Discharge	10	10	6-9	n/a	10 (6*)	n/a (0.1*)

*Regulations predicted for future discharge; Adapted from Sharvelle *et al* (2012) and CDPHE website

Despite repeated indication of constructed wetland's feasibility as a treatment option, there is minimal design guidance related specifically to graywater treatment. Paulo *et al* (2009) used literature values for domestic wastewater to design a graywater system and concluded that these values designed a system that treated graywater sufficiently but was prone to peak flow deterioration. By determining design parameters meant specifically for graywater, system weaknesses can be addressed to produce a reliable system. Bergdolt *et al* (2012) examined further the treatment of a FWS wetland, providing *k* rates and design recommendations. This work demonstrated that contaminant removal rates derived for domestic wastewater do not apply to treatment of graywater. Some preliminary results were also discussed of the SF wetland but insufficient data was available for a full analysis. Analysis of the systems on their own provided sufficient evidence to determine that use of a FWS or SF wetland alone would provide adequate treatment while also being more economically feasible.

Here, the SF wetland is examined further to provide a comparison of the FWS treatment efficiency to the SF treatment efficiency. Similar comparisons are limited and thus far have only been provided for municipal and industrial wastewater (Hijosa-Valsero *et al*, 2010; Jia *et al*, 2011; Kadlec and Knight, 1996). Results from these studies indicate that FWS wetlands may be more prone to seasonal variation (Figure 1.1). In addition, SF wetlands may be able to adjust to larger variations in influent loading, which is often an issue with graywater, making SF wetlands a more consistently effective option (Kadlec and Knight, 1996).

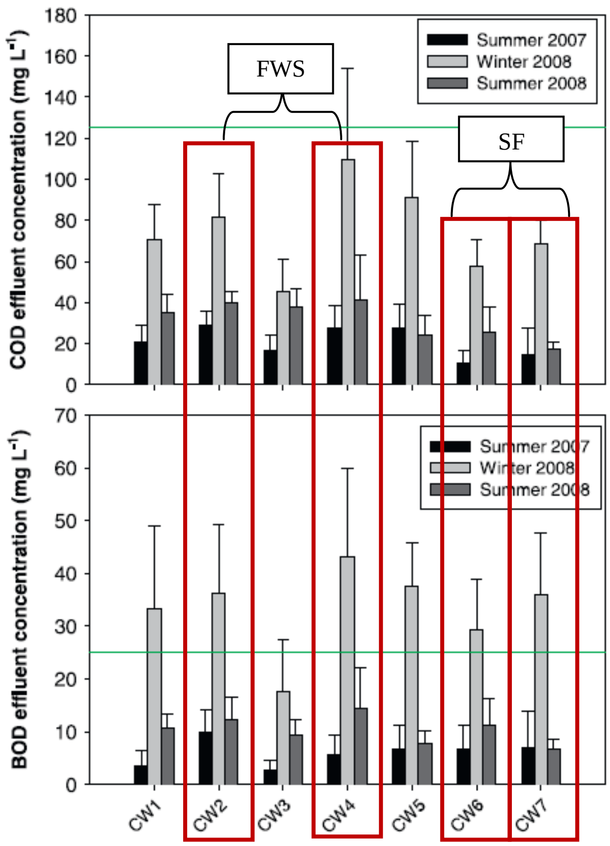

Figure 1.1: BOD_5 and COD effluent concentrations for various wetland configurations (Hijosa-Valsero *et al*, 2010)

In addition to traditional nutrients, graywater contains high levels of surfactants, which have been found to accumulate in river sediments (Barber *et al*, 1995). The fate of surfactants in constructed wetlands is largely unknown, especially when the media is soilless, as in the case of a rock-media SF wetland. The information herein, in addition to Jokerst *et al* (2011) and Bergdolt *et al* (2012), is important for future design of cost effective and efficient treatment systems meant to treat graywater for secondary use.

Hypothesis and Objectives

The goal of the constructed wetland system studied was to achieve water quality suitable for discharge to surface water or for unrestricted urban reuse. It is hypothesized that the performance of the FWS and SF wetlands will have seasonal variation but the SF wetland will more adequately meet treatment requirements for water reuse and surface discharge despite seasonal variations. The overarching objective of this research was to compare the performance of a FWS to SF for graywater treatment in a semi-arid/arid climate to provide the most efficient option for graywater treatment in an area of the US where water conservation is critical for long-term sustainability. This was done by comparison of mass removal rates and requisite surface areas required based on determined removal rates (k) for BOD_5 and TN. Once again it is important to note that removal of TN is important for surface discharge to receiving waters because of its adverse environmental health effects but when the effluent is used for irrigation, the TN may be beneficial. Aerial loading rates were compared to EPA suggested aerial loading rates in an attempt to provide recommendations for target effluent concentrations. Guidance on design of SF wetlands for treatment of graywater is provided through determination of k values and aerial loading rates. Another objective was to determine the fate of three common anionic surfactants in constructed wetlands (both FWS and SF). Finally, the last objective was to explore the possibility of incorporating constructed wetlands into greenhouse community garden centers to reduce the losses resulting from evapotranspiration (ET) in arid climates.

CHAPTER 2: BACKGROUND ON PERTINENT TOPICS

Water Conservation

Water is our most valuable resource; the quantity of freshwater available for our use is finite and must be protected. While global demand for freshwater will only increase as the global population continues to grow, the quality of freshwater in our streams and lakes is declining due to an increased input of pollution from point sources, including wastewater treatment plants, and non-point source pollution, such as agricultural runoff. To ensure sufficient freshwater to continue to meet our potable demand, there is a need for implementation of greater water conservation efforts.

EPA's *WaterSense: Be the Change* program (2012) provides many suggestions by which individuals may lower their water consumption. A few of the major contributors include installing low flow water fixtures, ensuring that your dishwasher and washing machine are full (or set at an appropriate water level) before running, turning off the sink while brushing your teeth and practicing sustainable landscape design by planting native vegetation and xeriscaping. Reusing reclaimed or graywater for non-potable applications such as irrigation, car washing, laundry, and toilet flushing also assists in water conservation efforts.

Water Use

In the United States, potable water is commonly used for non-potable demands such as irrigation, toilet flushing, and other uses that do not require the use of drinking quality water. In a study conducted by the American Water Works Association Research Foundation (AWWARF) the average US resident uses approximately 169.3 gallons of fresh water per capita per day (Mayer and DeOrea, 1999). Indoor water use accounted for approximately 69.3 gpcd (42% of total water use) while outdoor use averaged 100.8 gpcd (58% of total water use).

On average, graywater is generated at a rate of 31.4 gpcd (45% of total indoor water use). Potential potable water savings from graywater reuse is estimated to be 21% for landscape irrigation and 31% if irrigation reuse is combined with a toilet reuse system (ChristovaBoal *et al*, 1996).

Graywater

Graywater is wastewater from a home or office, including wastewater from laundry, shower, and bathroom sinks. In the US, graywater does not include kitchen sinks and dishwashers due to an increased potential for pathogens and a high organic content which leads to oxygen depletion and increased microbial activity in the stored graywater (Roesner *et al*, 2006). In addition, utility sinks are often excluded because they can be contaminated with oils, paints and grease, which may be toxic to plants and have the ability to clog graywater filters. When compared to domestic wastewater, graywater is minimally contaminated, with lower concentrations of organics, solids, nutrients, and pathogens (Table 2.1), which renders the water suitable for reuse with little treatment compared to what is required for treatment of domestic wastewater (Mayer and DeOrea, 1999; Tchobanoglous *et al*, 2003).

Table 2.1: Composition of Graywater to Domestic Wastewater

	Graywater Range[1]	**Domestic Wastewater Range[2]**
Chemical Oxygen Demand (COD; mg L^{-1})	77-240	250-1000
Biochemical Oxygen Demand (BOD_5; mg L^{-1})	26-130	110-400
Total Suspended Solids (TSS; mg L^{-1})	7-207	100-350
Total Nitrogen (TN; mg L^{-1})	0.36-0.64	20-85
Total Phosphorus (TP; mg L^{-1})	0.28-0.779	4-15
Total Coliform (CFU/100mL)	6.0×10^3-3.2×10^5	10^6-10^9
E. Coli (CFU/100mL)	<100-2800	

[1](Mayer and DeOrea, 1999) [2] (Tchobanoglous *et al*, 2003)

Despite being minimally contaminated, graywater is still wastewater and may contain harsh chemicals including oils, paints, solvents, and heavy metals such as zinc and copper (ChristovaBoal *et al*, 1996) or boron that can be harmful to plant health even at low concentrations (Madungwe and Sakuringwa, 2007). In addition, a number of studies have identified unsafe levels of indicator organisms in graywater (ChristovaBoal *et al*, 1996; Novotny, 1990; Rose *et al*, 1991). Because indicator organism concentrations in excess of standard concentrations permitted for recycled drinking, bathing, and surface irrigation water (Sheikh, 2010) are possible, graywater should be treated to levels that are safe for human contact when exposure is likely (i.e. toilet reuse). Current regulations applicable to water reuse in Colorado are summarized in Table 2.2. Regulations are provided by the Environmental Protection Agency (EPA) and the Colorado Department of Public Health and Safety (CDPHE).

Table 2.2: Current Regulations applicable to Colorado for water reuse

Regulatory Agency	BOD_5 (avg 30 day, mg L^{-1})	TSS (avg 30 day, mg L^{-1})	pH	E.coli (CPN/100mL)	
				Unrestricted Reuse	Restricted Use
EPA 40 CFR	30	30	6-9		
Colorado				25/100	500/100

Adapted from Sharvelle *et al*, 2012

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 disposal practices (Eriksson *et al*, 2002). Education and the implementation of safe handling and best management practices are necessary to safely and effectively reuse graywater.

Water reuse and other water conservation efforts are necessary to sustain human health, adequate agricultural production, and continued availability of water for recreational and industrial uses. Onsite graywater treatment and storage allows the home/business owner to benefit directly from water reuse and allows the participation of individuals interested in reuse without incurring large infrastructure costs to municipalities. As urban populations expand and existing infrastructure reaches capacity, separating graywater from blackwater may also extend the life of existing facilities.

Graywater reuse has already been implemented in other countries, including Australia, Asia, Europe, and parts of the Middle East. In most cases, graywater reuse has been implemented out of necessity, in response to drought or limitations brought on by high population density (Pinto *et al*, 2010; Al-Jayyousi, 2003). In the United States, graywater reuse is rapidly gaining popularity and a portion of the states have regulations concerning graywater use, however, these regulations vary significantly. In addition, there are also states that have not directly addressed the issue yet or are treating graywater as reclaimed water or septic effluent.

Sharvelle *et al* (2012) recently completed a review of the treatment, public health and regulatory issues associated with graywater reuse in an attempt to assist those in the regulatory community as they develop regulations for safe graywater reuse. Table 2.3 summarizes the current state of graywater reuse regulation in the U.S.

Table 2.3: Current State of Graywater Reuse Regulation in U.S (as of July 2012)

States that lack a graywater regulation or do not allow graywater use			**States that allow graywater use**		
States that allow wastewater reclamation	**States that fail to define graywater**	**States that treat graywater as septic**	**States that permit graywater using a tiered* approach**	**States that regulate graywater use w/o a tiered* approach**	**States that only allow residential irrigation**
Alabama	Illinois	Connecticut	Arizona	Florida	Hawaii
Alaska	Kansas	Kentucky	California	Georgia	Idaho
Arkansas	North Dakota	Maryland	New Mexico	Montana	Maine
Colorado	Ohio	Michigan	Oregon	Massachusetts	Nevada
Delaware	South Carolina	Minnesota	Washington	North Carolina	
Indiana	Tennessee	Nebraska		South Dakota	
Iowa		New Hampshire		Texas	
Louisiana		New Jersey		Utah	
Mississippi		New York		Virginia	
Missouri		West Virginia		Wisconsin	
Oklahoma				Wyoming	
Pennsylvania					
Rhode Island					
Vermont					

Adapted from Sharvelle *et al*, 2012
*Tiers based on volume of graywater being used

Graywater Reuse

Water shortages, either due to population increase or environmental conditions, have served as motivation for water reuse, however, water reuse should be more than a response to a problem—it should be implemented in recognition of water as a valuable resource and as an effort to protect our resources for future use.

Graywater reuse is emerging as an essential instrument for effective management of water because it promotes preservation of freshwater sources, potentially reduces pollutants in the environment, and reduces total costs in treating water and wastewater (Jefferson *et al*, 2001). The additional nutrients found in graywater may also reduce the need for artificial fertilizers (Hassan and Ali, 2002). Within a domestic residence, graywater reuse represents the largest potential source for reuse (Al-Jayyousi, 2003). However, improper reuse methods could potentially result in the spread of illness and/or unintended effects on the environment. Therefore, when reusing graywater, standards and best management practices must be implemented to safely reuse graywater and to avoid contaminating water with unsafe constituents (Bergdolt *et al*, 2011).

Graywater Reuse for Irrigation

Irrigation demand is typically the largest household water demand, estimated to be about 100 gallons per capita per day or approximately 60% of a typical home's overall water use (Mayer, 1999), depending on climate, region, irrigation area, and season (typically during the spring/summer months). On average, a person will generate enough graywater to meet only a portion (30 gpcd) of their irrigation needs unless irrigated areas are xeriscaped. Generally speaking, xeriscaped and landscaped areas require less graywater than turf lawns.

The City of Los Angeles, California conducted a study in 1992 about the soil characteristics of homes that used graywater irrigation. The results in that study revealed an increase in sodium levels and sodium adsorption ratio (SAR) in the area of graywater irrigation but the plant health appeared to be unaffected. What continues to be the concern are the long term effects of graywater reuse on soil, plants, and groundwater. Currently Water Environment Research Fund (WERF) and the Soap and Detergent Association (SDA) are funding research to determine the potential threats of graywater reuse to human health, and potential long term effects of graywater on plant health, soil chemistry, and microbiology. The first phase of this research, a literature review, has been published and concludes that there are knowledge gaps regarding the long term effects of graywater reuse and additional research is required (Roesner *et al*, 2006). The second phase of this research evaluates the long term (5+ years) effects of graywater irrigation on the changes in soil chemistry, soil microbiology, indicator organism, and impacts on residential landscape plants (Sharvelle *et al*, 2010). Further research on surfactants indicates that there are surfactants found in both freshwater irrigated soils and graywater irrigated soils, however, in graywater irrigated soils concentrations are higher and can be found at greater soil depths (Negahban-Azar *et al*, 2012). Negahban-Azar *et al* (2012) also found that surfactants did not harm plant growth; in fact, in most cases the plants irrigated with graywater grew better than those receiving potable water. In a greenhouse study utilizing soil columns, Negahban-Azar *et al* (2013) found that between 92 and 96 percent of surfactant parent compounds in the synthetic graywater were biodegradable.

Wetlands

One approach to treating graywater is through wetlands. Treated water can be discharged to surface waters or reused for irrigation or toilet flushing. Both natural and constructed wetlands can be utilized for wastewater treatment. Natural wetlands are one of the most biologically productive ecosystems on the planet and they can effectively treat most constituents found in municipal and agricultural wastewater (Kadlec and Wallace, 2009). Natural wetlands were used for treatment of wastewater long before research on their treatment process began (Kadlec and Knight, 1996). The first round of research, conducted from the 1950s through the 1970s, concluded that widespread applicability of constructed wetlands as a treatment option for municipal wastewater may be feasible. Constructed wetlands have been successful in treating a variety of wastewater sources including agricultural, industrial and mining runoff, municipal effluent, landfill leachate and urban storm water (Kadlec and Wallace, 2009; Kouki *et al*, 2009). Constructed wetlands are designed to mimic the natural wetland treatment process through a combination of biological, physical and chemical processes including sedimentation, precipitation, adsorption to soil particles, assimilation by plant tissue, and microbial transformations and interactions in a system isolated from surface and ground water through impervious liners (Babatunde *et al*, 2011; US EPA, 2000). Wetland technology offers several advantages over traditional treatment options including (Davis, 1995):

- Low capital cost
- Only periodic on-site operation and maintenance (reduced time and cost requirements)
- Adjust to fluctuations in flow
- Scalable (single family home to community)
- Serve as habitat for plant and animal species
- Can be integrated into sustainable landscape design

The most common types of constructed wetlands that are used to treat wastewater are FWS wetlands and subsurface flow wetlands. Subsurface flow wetlands can further be classified as horizontal (SF) and vertical flow (VF) wetlands. Design of constructed wetlands includes depth of water column, type of soil substrate and type of vegetation. Each of these criteria can assist in achieving a system with varying portions of aerobic and anaerobic treatment. FWS wetlands resemble natural wetlands because they consist of aquatic plants which root in a soil layer at the bottom of the wetlands. Water is treated as it flows through the soil, roots and stems of the plants (US EPA, 2000). SF wetlands (also called vegetated submerged beds (VSB)) do not have exposed standing water, but rather wastewater flows horizontally beneath the surface of a bed of media (typically crushed rock, small stone, or sand), which are planted with aquatic plants. Depending on the root mass density, SF wetlands may be aerobic or anaerobic. If the density of roots in the substrate is limited, SF wetlands can be considered anaerobic without artificial aeration except immediately around the root zone where some aerobic microbiological activity would occur. If, however, the plants are well established with dense root systems the system may function aerobically with intermittent pockets of anaerobic zones. VF wetlands distribute wastewater across the surface of a sand or gravel bed and treatment occurs as the wastewater percolates through the plant root zone (Kadlec and Wallace, 2009). VF wetlands generally have a gradient from aerobic at the surface and surrounding the root zones to anaerobic deeper within the media, similar to FWS wetlands. Treatment occurs through filtration, plant uptake and aerobic and anaerobic microbiological pathways (Kadlec and Knight, 1996). Pathogens are reduced through exposure to sunlight (FWS), predation and competition for resources (Cronk and Fennessy, 2001).

Regardless of the type of wetland, they have been proven to treat graywater effectively. Gross *et al* (2007a) determined that nearly 100% of TSS and COD were removed at the effluent in addition to a 3-4 order of magnitude reduction in fecal coliforms. Gross *et al* (2007b) also found that a significant portion of nitrogen species were removed in a recycled vertical flow wetland. Kadewa *et al* (2010) found that greater than 96% of organics were removed in a vertical flow planted wetland and that the plants assisted in surfactant degradation. Masi *et al* (2010) concluded that 98% of the influent COD was removed in multiple configurations and that nitrification within the wetland removed 92-99% of influent ammonia. Jokerst *et al* (2009) reported 85% removal of *E. coli* from the FWS and an additional 9% removal from the SF. In most sampling events, the *E. coli* concentration was less than 100 CPN cells/ 100mL. The EPA has created the treatment wetland database (TWDB), which contains information about various constructed wetlands including the system descriptions, locations, size, and the constituents that were monitored during operation. Although the TWDB provides information on existing wetlands it does not provide sufficient information to assist in constructed wetland design.

Kadlec and Knight (1996) developed sizing criteria for the design of a constructed wetland using a first order removal rate constant (*k*) with background concentration (*C**). The *k*-*C** model assumes ideal plug flow conditions, which evenly distributes flow through the wetlands. This model uses removal rates (*k*), design influent and effluent concentrations of a particular constituent in addition to an estimated background concentration to determine the required surface area necessary to provide the desired treatment. The *k* -*C** model has been used successfully to design wetlands but the k rates are dependent on a number of variables including influent concentration, temperature and the type of wetland. It is important to use the appropriate *k*-*C** values for each type of wastewater to determine accurate treatment effects of a particular constructed wetland (Knight *et al*, 2000). Researchers have also developed more complex models (TIS and PFD) to try to compensate for the non-ideal hydraulics characteristic of constructed wetlands (Babatunde *et al*, 2011), however, Rousseau *et al* (2004) reviewed wetland design approaches, concluding that the *k*-*C** method remains the best tool for evaluating treatment of wastewater through constructed wetlands. Equation 2.1 uses *k* and *C** in addition to design influent and effluent concentrations to determine the design surface area (SA) of the wetland.

$$\mathrm{SA} = \left(\frac{Q}{k}\right) ln\left[\frac{(C_{in}-C^{*})}{(C_{eff,adj}-C^{*})}\right] \qquad \text{(Equation 2.1)}$$

Where: k = first order removal rate constant (m day^{-1})

Q = flow rate (m^3 day^{-1})

$C_{eff,\,adj}$ = effluent constituent concentration (mg L^{-1})

C^* = background constituent concentration (mg L^{-1})

The EPA has indicated that typical background concentrations for BOD_5 and TN are 1-10 mg L^{-1} and 1-3 mg L^{-1}, respectively, for both FWS and SF constructed wetlands (Table 2.4).

Table 2.4: Typical Background Nutrient Concentrations

Nutrient	Conc (mg L^{-1})
BOD_5	1-10
TN	1-3

EPA 832-F-00-024 (HLR = 0.4-4 in d^{-1}); EPA 832-F-00-023 (HLR = 3-12 in d^{-1})

In 2000, the EPA developed their own criteria for sizing both FWS and SF wetlands that treat municipal wastewaters using average loading rates (US EPA, 2000). While the values are similar, SF influent values and aerial loading rates are consistently higher, suggesting the SF wetlands may have a greater capacity for treatment of wastewater. FWS influent values range from 5-100 mg L^{-1} BOD_5 and 2-20 mg L^{-1}TN while SF influent values range from 30-175 mg L^{-1} BOD_5 and 20-40 mg L^{-1}TN. FWS loading rates range from 1-10 g m^{-2}d^{-1} BOD_5 and 0.2-1 g m^{-2} d^{-1} TN while SF loading rates range from 6.7-15.7 g m^{-2}d^{-1} BOD_5 and 0.3-1.2 g m^{-2}d^{-1} TN (Tables 2.5 and 2.6).

Table 2.5: Typical FWS Loading Rates and Effluent Concentrations (HLR = 0.01- 0.1 m d^{-1})

	Influent (mg L^{-1})		Target effluent (mg L^{-1})		Aerial Loading Rate (g $m^{-2}d^{-1}$)	
BOD_5	5-60 ^	5-100 *	5-20 ^	5-30 *	1-5 ^	1-10 *
TN	2-20 ^	2-20 *	1-10 ^	1-10 *	0.2-1 ^	0.2-1 *

^EPA 832-S-99-002; *EPA 832-F-00-024

Table 2.6: Typical SF Loading Rates and Effluent Concentrations

	Influent (mg L^{-1})	Target effluent (mg L^{-1})	Aerial Loading Rate (g $m^{-2}d^{-1}$)
BOD_5	30-175	10-30	6.7-15.7
TN	2-40	1-10	0.3-1.2

EPA 832-F-00-023 (HLR = 0.08 – 0.3 m d^{-1})

It should be noted that these values are based on municipal wastewater, not graywater and should therefore be used strictly as a reference and indication of treatment performance. The EPA loading rates can be universally applied to all areas of the country and across different climates and temperatures. These loading rates contain several safety factors within the model to allow for safe effluent conditions in various climates despite the influent concentrations.

The differences in suggested influent concentrations and aerial loading rates raise a question concerning the benefit of one type of wetland over the other. There have been a limited number of studies on the comparison of FWS and SF performance. Hijosa-Valsero *et al* (2010) and Jia *et al* (2011) used municipal and industrial wastewater, respectively. Their results suggest that FWS constructed wetlands have more seasonal variation than SF constructed wetlands. Neither study commented on the comparison of overall performance.

Kadlec and Knight (1996) provided estimated annual average treatment performance capabilities which suggest that SF constructed wetlands may perform better overall, reporting FWS constructed wetlands were capable of 67% BOD_5 removal and 69% TN removal while SF constructed wetlands were capable of 67-80% BOD_5 removal and 76% TN removal but again, this is a comparison of secondary treatment of municipal wastewater (Table 2.7).

Table 2.7: Comparison of Average Annual Treatment Performance Capabilities

	BOD_5 (% removal)	TN (% removal)
Free Water Surface (FWS)	67	69
Subsurface Flow (SF)	67-80	76

Kadlec and Knight, 1996

In addition to the flow pattern of the wetland, plant species may also play a role in treatment efficiency. Two common types of vegetation used in treatment wetlands are *Typha spp*. (cattail) and *Scirpus spp*. (bullrush). Both are emergent aquatic plants that provide rapid nutrient uptake during the growing season and provide high nutrient storage as a result of their high quantity of above-ground biomass (Cronk and Fennessy, 2001). The type of vegetation plays a role in the level of background nutrient concentrations, particularly during periods of the year when the plants are senescent. Cronk and Fennessy (2001) suggest that the rapid release of nutrients back to the water following plant death can range from 10 to 100% of the nutrients originally taken up by the plant.

Burke (2011) assessed the carbon, nitrogen and phosphorus storage of *Scirpus acutus* and *Typha latifolia* with the intention of recommending one as a more effective choice for wastewater treatment based on the capacity to store nitrogen and carbon and the ability to retain nutrients during senescent periods. Burke (2011) found that *Scirpus acutus* was a more suitable species for wastewater treatment as it demonstrated a greater capacity for storage while contributing less decomposition by-products from above-ground tissues than *Typha latifolia*. *Scirpus acutus* stored more biomass and nutrients in below-surface tissues and decayed 30% slower over one year than *Typha latifolia*. *Scirpus acutus*' ability to produce less above-ground biomass could result in reduced volume of ET during a hot, dry summer while its slow decay rate would allow for a less demanding maintenance/harvest schedule.

Surfactants

One common component of graywater is surfactants. Surfactants are synthetic organic compounds which exhibit tensioactive properties, making them key ingredients of household and industrial detergents, personal care products and pesticides, as they lower the surface tension of a liquid to allow emulsification of hydrophobic compounds (Lara-Martin, 2008). 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) and alkyl ethoxysulfates (AES).

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 *et al*, 1999). According to Huang *et al* (2004), LASs are the most widely used anionic surfactants, accounting for approximately 28% of the total surfactant production in the US, Western Europe and Japan. Alkyl ethoxysulfates (AES), the second most common anionic surfactant, are also produced in great quantities (Lara-Martin, 2008). Barber *et al* (1995) identified LAS in Mississippi River bottom sediments (0.1-20 mg kg^{-1}) as well as in 21% of water samples (0.1-28 μg L^{-1}). Sanderson *et al* (2006) found lower concentrations of AES in river sediments in the United States. Lara-Martin *et al* (2007) found that when water was spiked with commercial LAS, a major portion was found to be attached to sediments (99.2% ± 0.5%). Sorption to sediments increased as alkyl chain length increased. In the spiked trials, the degradation of LAS proceeded quickly until the concentrations reached what was seen in natural sediments and waters. Regardless of their presence in sediments and water samples, LAS and AES surfactants have been shown to degrade under both aerobic and anaerobic conditions (Huang *et al*, 2004; Lara-Martin *et al*, 2007; Fountoulakis *et al*, 2009).

Fountoulakis *et al* (2009) conducted a pilot scale study with a 42 m^2 FWS wetland (including an anoxic portion), a 45 m^2 SF wetland and a 5 m^2 gravel filter. It was reported that the FWS, SF and gravel filter cells generated on average 30%, 55.5% and 40.9% removal of LAS, respectively. Inaba (1992) reported that in one surface flow wetland system removal of LAS was more than 90% in the summer and over 40% in the winter. These results reinforce the accepted behavior that LAS is more degradable under aerobic conditions and show that wetlands may be a viable treatment alternative. Ying (2006) also found that LAS may be more persistent under anaerobic conditions and that, in general, anionic surfactants had lower sorption rates than nonionic and cationic surfactants. After investigating AES parent compound degradation, it was reported that AES had a shorter degradation half-time than LAS under similar test conditions (Pojana *et al*, 2004). The researchers also reported that AES concentrations in the river Po were less than 0.5 μg L^{-1} in all samples, whereas LAS concentrations ranged from 0.5-1.7 μg L^{-1}. Based on review of biodegradation of LAS and AES, LAS may be more persistent in the wetland treatment system to be studied than AES.

Anionic surfactants are the most common surfactants found in personal care products in the US today. While they are known to be readily biodegradable, it is unknown how elevated concentrations in graywater influence rate of biodegradation and/or accumulation in the soil and plant matter in which graywater is applied. In this study, the ability of the wetlands to remove anionic surfactants from graywater was investigated. Graywater samples were taken at the inflow of the FWS bed, after the FWS bed and after the SF bed; however, the accumulation of surfactants remaining in the wetlands was not analyzed.

Need for Community Integration

This thesis encompasses a seemingly wide spectrum of topics; however, the world in which we live is complex and an interdisciplinary approach to problem solving is inevitable. At this point in history, there is a great disconnect between similar disciplines which has become a hindrance to efficient management of our resources. We can no longer look at the world as disconnected systems, instead there is a great need for interdisciplinary collaboration and an alliance of intellect between overlapping disciplines. Throughout the completion of my studies I encountered a variety of disciplines, including civil and environmental engineering, ecological engineering, applied ecology, microbiology, biotechnology, restoration, landscape architecture, environmental psychology and environmental economics. While each of these is important to the discussion at hand, ecological engineering is perhaps the newest and most relevant.

H.T. Odum first defined ecological engineering in the 1960s. Ecological engineering, by definition, utilizes biological species and communities within natural ecosystems to treat wastewater. This form of treatment is becoming increasingly popular due to its economic benefits. These systems, often referred to as “green machines” or “living machines”, do not depend on significant construction or operation costs and they can be used to generate profitable resources in addition to treating waste (Mitsch, 1999; Todd, 1999). In addition, they are both self-organizing and self-sustaining after their initial start-up. In theory, constructed wetlands would provide this type of platform to both large and small scale graywater treatment. However, there are certain aspects to an open constructed wetland system that make utilizing this technology, especially in arid climates, difficult to justify. The plants used in constructed wetlands respire large volumes of water which makes utilizing them in water-limited regions seem counterproductive. FWS wetlands, depending on how large the open water surface is, may be prone to evaporation, may be a breeding ground for mosquitoes and increases the risk of direct human contact. Liners are used to minimize exchange with the surrounding environment through the bottom but SF wetlands provide a barrier topside which may assist in reducing evaporation from the wetland during hot summers. Proper sizing would also make wetlands more efficient and less prone to mosquito issues but there will always be ET from the plants. Evaluation of ET rates associated with different wetland plant species and careful selection of plants may help in reducing ET losses. Adding a greenhouse-like structure that returns ET to the system would also minimize losses, however, this addition would add capital and maintenance costs if it were not otherwise being used. Community gardens provide the opportunity to balance the cost of such additions with the long-term benefit of food production, community growth and education.

Community gardens have long been used in urban centers around the world to facilitate a number of social, economic and political goals. Inclusion of graywater constructed wetlands into community garden centers or community supported agriculture (CSA) gardens would accomplish a number of goals and perhaps make the use of constructed wetlands more feasible for graywater treatment by providing a link between government utilities and public use areas. The quality of water necessary for irrigation is very similar to the quality of water discharged from constructed wetlands. While the levels of nutrients are ideal for irrigation purposes, there is potential for high levels of pathogens as well. The history of community gardens in the United States can be traced as far back as the 1890s. Much of its history can be linked inextricably with political and economic instability, with sharp increases in popularity exhibited during the 1890s and 1930s depression eras and during and immediately following the World Wars (Hou, 2009). We are, in fact, in the middle of the most recent movement as evidenced by the drastic increase in the number of urban gardening groups in US cities. In 1996 it was estimated that there were 6,000 active community gardens in the United States (Lee, 2012). Today, the American Community Garden Association estimates that there are more than 18,000 community gardens in the US and Canada (ACGA, 2012), most of them present in the continental US (Figure 2.1).

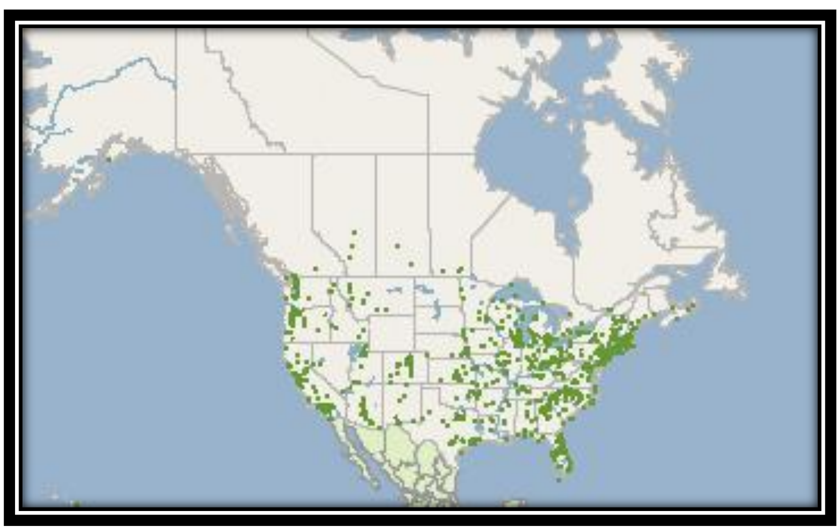

Figure 2.1: Estimated distribution of community gardens in the US and Canada (ACGA, 2012)

Community gardens may be merely open space in an urban setting or it can serve as a setting for urban agriculture. When community gardens are used to supply food products, they are called community supported agriculture (CSA) or agriculture supported community gardens (Smith, 2010). CSA gardens involve a subscription one pays to support a larger urban garden, similar to a food coop. Agriculture supported community gardens consist of many smaller plots, each tended by a member in the community.

There are many benefits which spring from community gardens. In addition to the obvious benefit of local food production, community gardens provide a link to natural systems which has been lost to modern urbanization (Smith, 2010; Irvine, 1999). They also provide a place for community activism, for communal work that acts as an opening for true and meaningful engagement, and a place in which members can learn about a wide variety of topics, including the tangible knowledge that our body is an instrument of productivity in its own right—we don't need highly technical instruments to function all the time (Smith, 2010). They provide a place for recycling and composting, improve air quality and can assist in stormwater management. In some cases, community gardens have been incorporated into juvenile court and after-school programs, housing projects and women's shelters and have been used to supplement resources in food bank programs and soup kitchens for the homeless (Hou, 2009; Smith, 2010; Irvine, 1999).

Unfortunately, there are many hurdles to starting and maintaining successful gardens. There are many sources that give examples of successful community gardens in the US but because the success is largely dependent on the community involved, each case must inevitably find its own path. In addition, community planning has failed, in general, to acknowledge the link between sustainable land use, ecological restoration and community gardening (Irvine, 1999). In the course of city development and planning, community gardens have traditionally been characterized as transient amenities which are easily replaced by more valuable land uses if necessary. For long-term community gardens to be successful, the physical space and the gardeners must be supported by organization, outreach and a clear sense of the site's future intended use (Lawson, 2004). Community gardens have long since been recognized as useful public investments but the current approach to public support of community open space lacks the substance and organization required to maintain a space conceptualized as both public resource and private investment. Collaborative partnerships are required to deal with these diverging issues to establish a balance between the participatory, evolving nature of user-initiated spaces with the long-term vision and structure of planning (Lawson, 2004).

Saxgren (1997) contends that a large part of the problem of our water consumption is the amount of water we turn into wastewater. He suggests that one way to force more responsibility on the population as a whole is to shorten the distance between our generation of wastewater and the process of treating it to reusable standards. By placing it within sight, it becomes a tangible problem rather than something we are able to take for granted. For many, wastewater is an inherently repulsive topic but our understanding of the treatment of wastewater has developed through the years into an intricate and fascinating process. In the setting of a community garden, the topic of wastewater and the process of cleaning our wastewater could be tackled in such a way to facilitate education and community growth. Because wetlands are largely self-sustaining, beyond the initial education and long term oversight, the community members could be responsible for maintaining the system on their own. Long term oversight by a government program would be necessary to ensure continued safety and could also act as a continual presence in neighborhoods where there is a high turn around. Through education in community gardens, the feasibility of graywater reuse can be emphasized while ensuring that best management practices are followed to protect against harmful effects of graywater reuse. In addition it provides a link between long-term government participation and community involvement.

Summary

Water conservation is the practice of reducing the demand of water in order to conserve our most valuable natural resource. The United States uses approximately 169.3 gpcd, of which 31.4 gpcd is considered graywater. Graywater is wastewater from the laundry, shower and bathroom sinks. This type of wastewater is minimally contaminated compared to domestic wastewater and can therefore be reused with limited treatment. Twenty states have regulations directly allowing for graywater reuse (July 2012) with various restrictions and recommendations for treatment. One method of treatment is constructed wetlands. Constructed treatment wetlands have been used for the treatment of wastewater for decades. They are particularly attractive because they offer an option with a low capital cost and low maintenance. They have been proven to effectively treat wastewater across a wide variety of flows and loading rates. Design of constructed wetlands for graywater treatment using the k-C* method has been proven effective. However, by determining rate coefficients specific to graywater rather than relying on values determined for other types of wastewater, treating graywater with constructed wetlands can be made even more effective and efficient. Surfactants, a main component in detergents and soaps, have been found in the natural environment despite claims that they are readily biodegradable. The fate of surfactants in treatment constructed wetlands is relatively unknown but it is likely that the biological activity present in wetland systems could potentially facilitate degradation. Community gardens provide many benefits including local food production but lack the necessary connection between public community and government acknowledgement that would ensure long-term continuity. Incorporating graywater treatment wetlands into community gardens could provide the necessary link to make graywater reuse and community gardens a reality.

CHAPTER 3: COMPARISON OF FREE WATER SURFACE AND SUBSURFACE WETLANDS FOR TREATMENT OF GRAYWATER

Introduction

Communities throughout the United States and abroad are seeking innovative approaches to sustaining their freshwater resources. Graywater reuse for non-potable demands is gaining popularity because it allows for the reuse of minimally contaminated wash water, generated and treated on site. Graywater is defined as any wastewater generated at the home or office including wastewater from the laundry, shower, and bathroom sinks but excluding water from the toilets, kitchen sinks, and dishwasher. When compared to other wastewater generated in the home graywater is minimally contaminated with lower concentrations of organics, solids, nutrients, and pathogens. These characteristics make the water suitable for reuse with minimal treatment when compared to other domestic wastewater sources. One method of treating graywater at a community scale for irrigation reuse is constructed wetlands. Constructed wetlands can offer a scalable, economically sound, low tech and easily maintained method of treating graywater for large scale irrigation reuse. While constructed wetlands are an appropriate technology for graywater treatment there is little research providing the removal rates used in the design of constructed wetlands for graywater reuse. The main objective of this research was to compare the performance of a FWS and a SF wetland for graywater treatment. This was accomplished by comparing the mass removal rates and requisite surface areas required for treatment based on determined *k*. Aerial loading rates were compared to EPA suggested aerial loading rates in an attempt to provide recommendations for target effluent concentrations. In addition, the removal of two common anionic surfactants from the water column in constructed wetlands (both FWS and SF) was evaluated. The results not only provide important information for the proper sizing of constructed wetlands but may also provide information on what type of wetland is more appropriate for graywater treatment.

Materials and Methods

Wetland Configuration

Details of the designs and operations of previous years are described in Cho (2007), Jokerst *et al* (2011) and Bergdolt *et al* (2012). Briefly, the experiments were conducted in Fort Collins, Colorado on the Foothills Campus of Colorado State University (CSU). A pilot scale constructed wetland system that consisted of a FWS and a SF wetland was constructed in the summer of 2007, outdoors, in a semi-arid climate and with no protection from temperature, rainfall, or evapotranspiration. The FWS was running as often as possible starting in October 2008 through November 2010. The SF was running independently from the FWS from June 2010 through May 2012. During the entire operation period of the wetlands (3.5 years), the system was also shut down occasionally for maintenance or due to insufficient graywater. The wetlands were constructed with distribution and collection headers to prevent short circuiting and to evenly distribute the flow (Figure 3.1) and impermeable ethylene propylene diene monomer rubber liners to prevent seepage or groundwater flux.

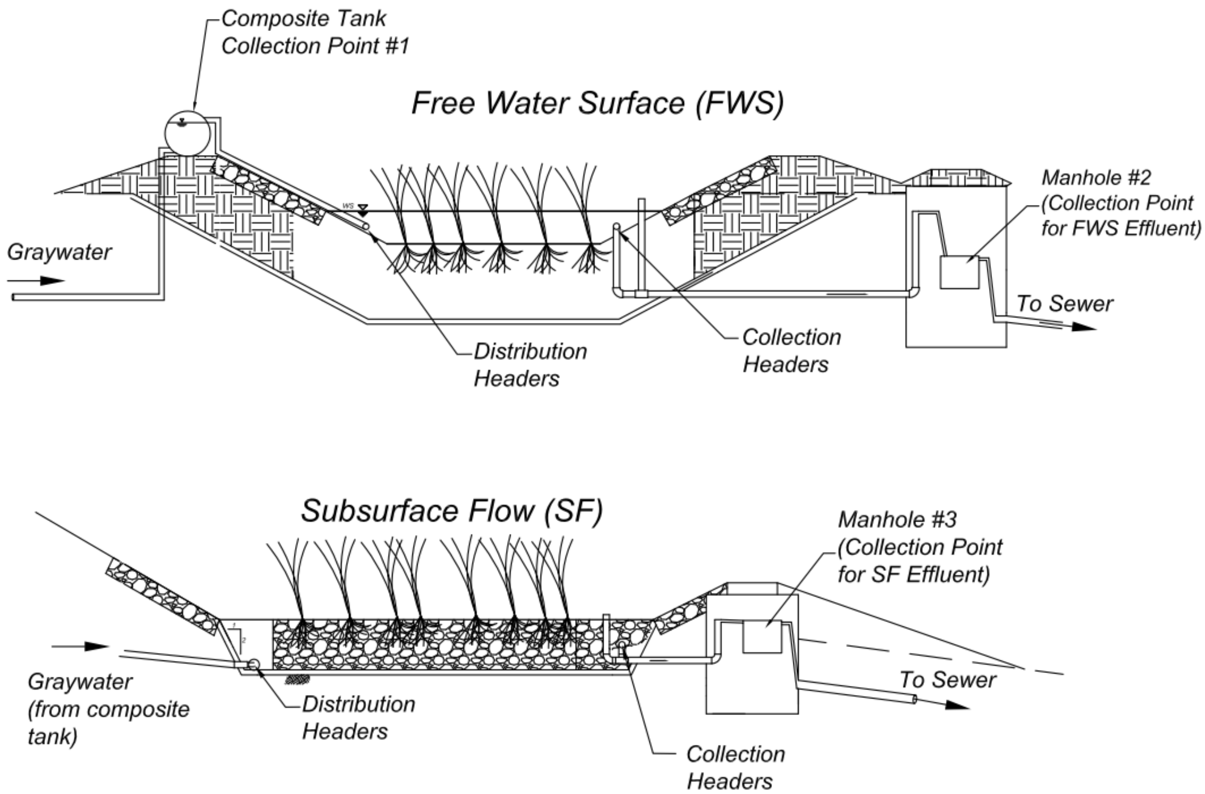

Figure 3.1: Wetland Configuration

A berm was constructed around both wetlands to minimize surface runoff into the wetlands. Initially, the wetlands cells were configured in series such that raw graywater entered the FWS wetland and the effluent from the FWS flowed into the SF wetland (Cho, 2007; Jokerst *et al*, 2011). The wetlands were separated to operate in parallel during the summer of 2010 to evaluate the performance of each wetland system separately. While this study focuses on the treatment capacity of the SF wetland, Bergdolt *et al* (2012) discusses the treatment efficiency of the FWS wetland at length. Only information on the FWS wetland that assists in comparing the SF wetland to the FWS wetland will be presented below.

The FWS wetland (Figure 3.2), planted with cattails (*Typha latifolia*), measured 9’ by 13’ (2.7m x 4.0 m) corresponding to a surface area of 120 ft^2 (11.2 m^2). The depth was assumed to be constant at 14” (0.36m) and the wetland was constructed with a 1:2 rip rapped side slope. The volume of the FWS was approximately 530 gallons (2.0 m^3), assuming an overall porosity of 0.8.

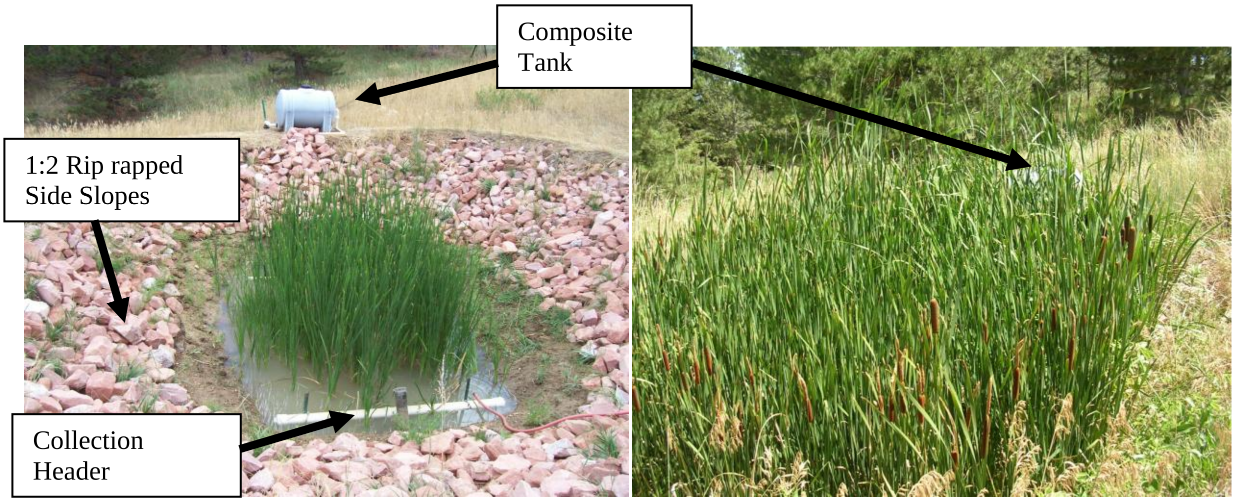

Figure 3.2: FWS in 2008 (left) and FWS in 2010 (right)

The SF wetland (Figure 3.3) was planted with hardstem bulrush (*Scirpus acutus*), measured 11’ by 17’ (3.3m x 5.2m) with a corresponding surface area of 185 ft^2 (17.2 m^2). The depth of the gravel matrix was 19” (0.5m) and included a 1:1 rip rapped side slope. The SF wetland was filled with clean rounded native gravel with an average diameter of ½” (15mm). The SF wetland maintained a volume of 590 gallons (2.2 m^3) considering an average porosity of 0.3.

Figure 3.3: SF in 2008(left) and SF in 2010 (right)

Water samples were collected at two locations after the fall of 2011 to determine the quality of effluent and treatment provided by the SF wetland. The first sampling location was at the composite tank, before the graywater entered SF wetland (Figure 3.1). During the winter this tank was heated to prevent freezing. The second sample was collected from a 1 gallon bucket located in a manhole at the downhill end of the SF wetland (Figure 3.1) that held approximately 1 gallon of effluent before triggering a float switch to empty the bucket. This float switch was used to track the volume of effluent. Samples were collected in 1 liter amber glass bottles and analysis was started immediately upon return to the laboratory.

Graywater Sources

At various times during the life of this pilot project graywater originated from three sources: directly from four lavatory sinks in the Atmospheric Chemistry Building (ACB) and from two different residential dormitories located on CSU's main campus. Briefly, the ACB was located south of the wetland and equipped with a dual plumbed system such that graywater could flow directly to the wetlands. Graywater production from the ACB fluctuated between 5 to 40 gallons per week depending on occupancy. Due to the low graywater flow rates, additional sources were needed to conduct the experiment. Graywater from Edwards Hall was collected using a retrofitted dual plumbing system installed during the summer of 2008. The dual plumbing system collected graywater from the sinks and showers of 34 residences and conveyed the graywater to a 300 gallon storage tank located in the basement of the residence hall until the graywater could be transported to the wetlands. During the summer of 2010 and 2011 graywater was also collected from an additional residence hall. The dual plumbing system at Aspen Hall collected graywater from fourteen showers and sinks used by twenty seven students and conveyed the graywater to two 300 gallon storage tanks located in the basement. Average influent concentrations for BOD_5, TN, NH_4^+ and TSS were 98 mg L^{-1}, 12.5 mg N L^{-1}, 8.8 mg NH_4^+-N L^{-1} and 35 mg L^{-1}, respectively (Table 3.1).

Table 3.1: Average Influent Concentrations of Study Graywater

Constituent	Influent Concentration (average)
BOD_5	11-233 (98) mg BOD L^{-1}
TN	3.3-35 (12.5) mg N L^{-1}
NH_4^+	1.7-30 (8.8) mg NH_4^+-N L^{-1}
TSS	7-190 (35) mg TSS L^{-1}

Graywater was transported from the residence halls to the wetlands using a 500 gallon metal tank affixed to a trailer. Previous methods of pumping water from the trailer to the wetland are discussed in Jokerst *et al* (2011) and Bergdolt *et al* (2012). Briefly, a variable speed peristaltic pump (Masterflex, Vernon Hills, Illinois) was used to move graywater from the trailer to a one gallon bucket within the first manhole, where it was mixed with graywater from the ACB. Beginning in the fall of 2011, graywater from the ACB was turned off and only graywater from Edwards Hall was used. In preparation for a self-sustaining system, the first manhole was altered so that water was pumped directly from the trailer to the composite tank and into the SF wetland, first using a small utility pump and finally a bilge pump (Rule 24, Gloucester, Massachusetts).

Definition of Seasons

Mass removal rates of graywater constituents were determined to be significantly different for each season (Jokerst *et al*, 2011). It was therefore important to separate data analysis based on seasons. Fort Collins, CO is located east of the Rocky Mountains in a semi-arid region where large weather fluctuations can occur throughout the year. Seasons were determined using both effluent water temperatures measured at the wetland and average daily temperatures recorded at a National Climatic Data Center (NCDC) station (Table 3.2 and 3.3). Seasonal temperatures can vary substantially in the Fort Collins, CO area so the seasons were not defined by a calendar year but were instead based on a range of temperatures and observation of the wetland vegetation's growth stages. The summer season was defined when the average daily temperatures were consistently above 15°C and the wetland vegetation had established growth. The spring and fall seasons were defined when the average daily temperatures ranged between 7-15°C and the plants had begun to visibly grow or the leaves of the plants started to die (showing yellow and brown color), respectively. The winter season was defined when the average daily temperature was lower than 7.5°C (Figure 3.4).

Table 3.2: Seasonal Dates, Effluent Water Temperatures and Average Daily Air Temperatures –FWS analysis

Season	Season Dates	Mean Effluent Water Temp (°C)			Average Daily Air Temp (°C)		
Spring 2009	3/31/09-6/12/09	7.7	±	2.8	11.7	±	7.1
Summer 2009	6/13/09-9/20/09	16.7	±	0.8	19.9	±	5.5
Summer 2010	7/1/10-10/8/10	16.0	±	4.1	20.3	±	6.3
Fall 2008	9/24/08 -11/19/08	11.3	±	3.1	9.7	±	6.9
Fall 2009	9/21/09-10/27/09	8.2	±	0.9	7.9	±	7.1
Fall 2010	10/9/10-11/4/10	9.6	±	2.3	10.1	±	5.7
Winter 2008-2009	11/20/08-3/30/09	4.5	±	2.4	1.4	±	8.2
Winter 2009-2010	10/28/09-4/1/10	2.0	±	0.9	0.1	±	7.3

Table 3.3: Seasonal Dates, Effluent Water Temperatures, and Average Daily Air Temperatures—SF analysis

Season	**Season Dates**	**Mean Effluent Water Temp (°C)**			**Average Daily Air Temp (°C)**		
Spring 2010	4/2/10-6/30/10	19.4	±	1.1	13.7	±	7.5
Spring 2012	3/11/12-6/1/12	12.1	±	2.3	13.5	±	6.7
Summer 2010	7/1/10-10/8/10	17.6	±	1.4	20.3	±	6.3
Fall 2010	10/9/10-11/4/10	10.6	±	2.8	10.1	±	5.7
Winter 2011-2012	11/3/11-3/10/12	4.2	±	1.8	0.7	±	6.8

While there was a wide range of values collected for both the influent and the effluent concentrations (Table 3.4), there was no significant difference in the influent values based on season or wetland type (ANOVA P-value >0.05 in all cases).

Table 3.4: Comparison of Influent and Effluent Concentrations

Constituent	Influent Range (average)	FWS Effluent Range (average)	SF Effluent Range (average)
BOD_5 (mg BOD L^{-1})	11-233 (98.0)	3-161 (37.0)	1-67 (20.0)
TN (mg N L^{-1})	3.3-35 (12.5)	0.6-18.9 (7.1)	0.3-8 (2.7)
NH_4^+ (mg NH_4^+-N L^{-1})	1.7-30 (8.8)	0.1-13.6 (4.4)	0.04-7 (2.8)
TSS (mg TSS L^{-1})	7-190 (35.0)	1.9-35 (14.0)	0.7-38.8 (14.8)

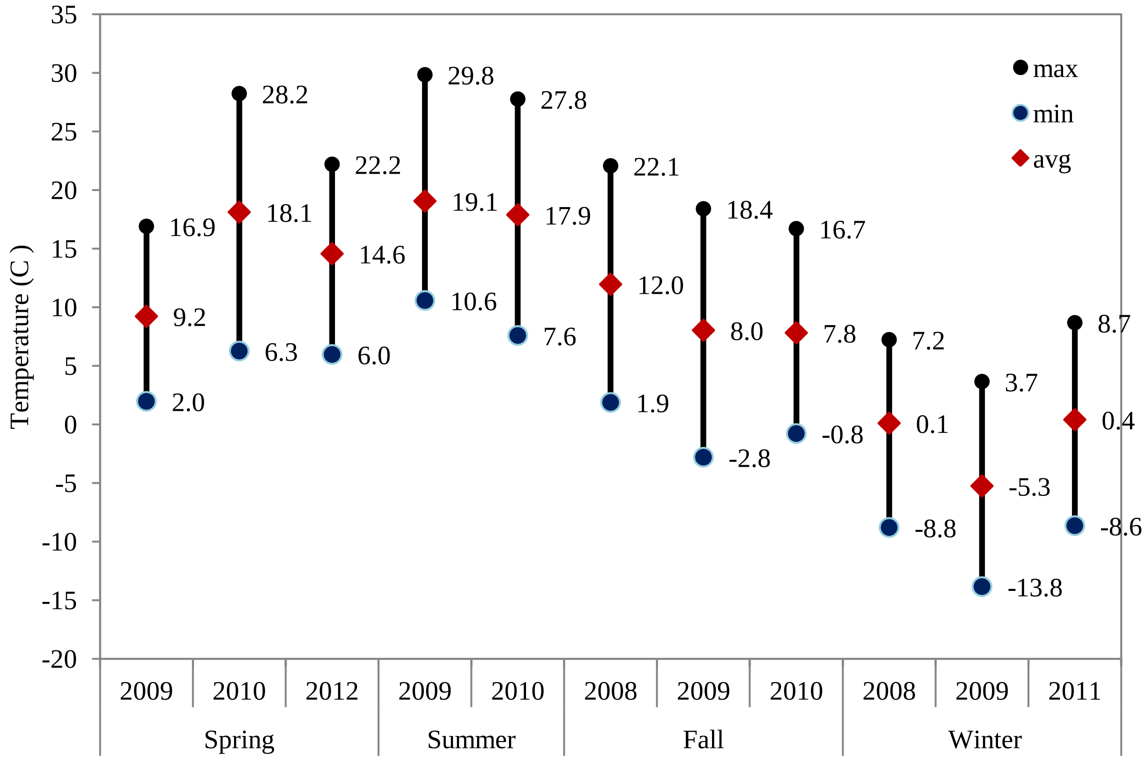

Figure 3.4: Range of Average Daily Air Temperatures over the experiment

Plants in both wetlands typically started to grow in mid-May (spring), reached full maturity in July and began to turn yellow in October. Although Fort Collins generally has well defined fall, winter and summer seasons, spring is not so well defined and is often too short to collect meaningful data.

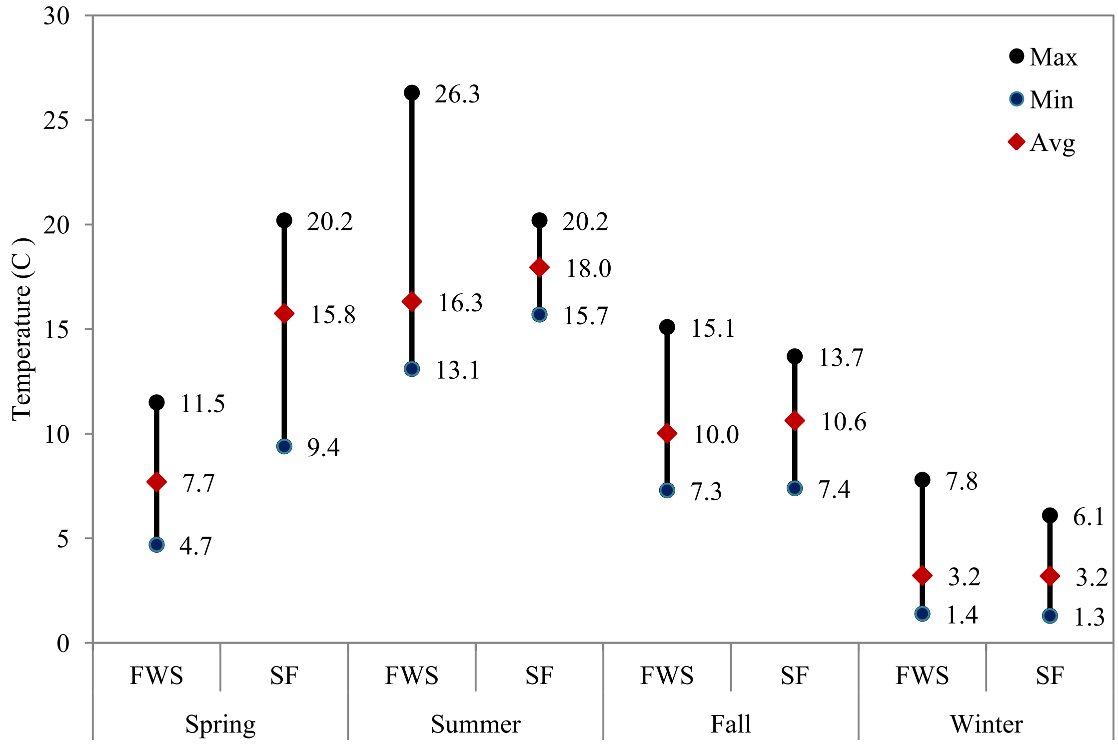

Figure 3.5: Seasonal Comparison of Effluent Water Temperatures

While previous spring seasons were short, the spring of 2012 started early and lasted longer than previous years. The winter season proceeding the spring of 2012 was also uncharacteristically warm and evidence of growth started in March, with full maturity reached by the beginning of June. Because of the warm winter days, it was also necessary to focus more on effluent water temperatures than average daily temperatures. Although it was warm during the day, freezing still occurred at night keeping the effluent water temperatures low until March (Figure 3.5). While there is a significant difference (P = 0.04) in the average effluent water temperatures for the wetlands in the spring, this significance is also present in the average air temperatures (P = 0.05) when the season is split into the years which were analyzed for the FWS and which were analyzed for the SF.

Flow Monitoring

Details of past flow monitoring methods are described in Jokerst *et al* (2011) and Bergdolt *et al* (2012). Briefly, graywater flowed into each manhole (Figure 3.1) and was collected in a 1 gallon bucket. When the bucket was full, a float switch activated a submersible pump (Rule 25D, Gloucester, Massachusetts) which pumped the flow through a turbine meter (Great Plain Industries TM100-N, Wichita, Kansas). Cumulative flow readings were monitored and recorded and divided by the length of time between readings to determine a flow rate.

During the period of time when graywater was entering both wetland cells simultaneously, flow into each wetland could not be measured directly. Therefore, it was estimated by calculating the portion of the total inflow that was leaving from the respective cell and multiplying that ratio by the total inflow (Equation 3.1).

$$Q_{i,out}\Big|_{i=1,2} = \left(\frac{Q_{m1,out} + Q_{m2,out}}{Q_{mi,out} * Q_{m,in}}\right)^{-1} \qquad \text{(Equation 3.1)}$$

Where $Q_{i,out}$ = Estimated Q_{out} for the wetland (1=FWS, 2=SF)

$Q_{m1,out}$ = Measured Q_{out} for FWS

$Q_{m2,out}$ = Measured Q_{out} for SF

$Q_{mi,out}$ = Measured Q_{out} for the wetland (1=FWS, 2=SF)

$Q_{m,in}$ = Measured total flow into the wetlands

During the fall and winter of 2011, several changes were made to the pumping system. First, the peristaltic pump was replaced by a small utility pump and irrigation valve (necessary to stop siphoning from below) temporarily until a more permanent solution could be installed. In January 2012 a submersible bilge pump (Rule 24, Gloucester, Massachusetts) was installed inside the metal tank (Figure 3.6).

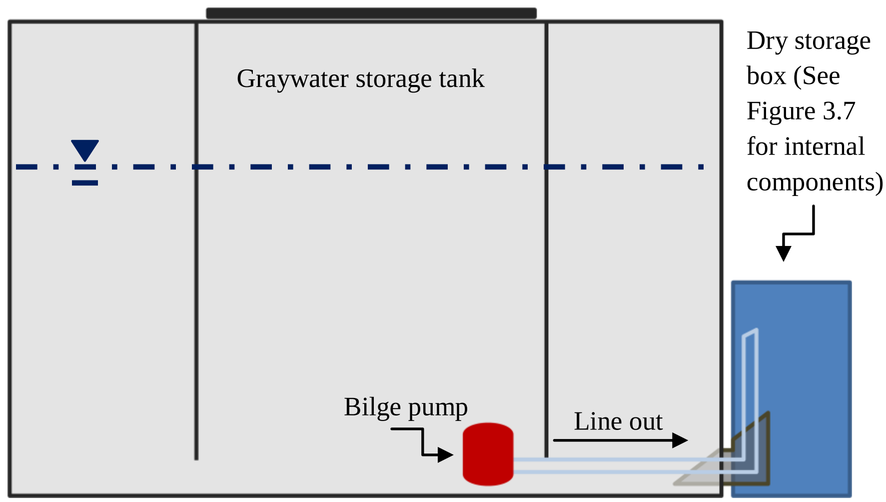

Figure 3.6: Schematic of graywater storage trailer and new system orientation

Graywater was pumped through an irrigation valve and a flowmeter before moving to the composite tank. The pump and irrigation valve were connected with a timer, which allowed for adjustment of flow for varying hydraulic loading rate (Figure 3.7).

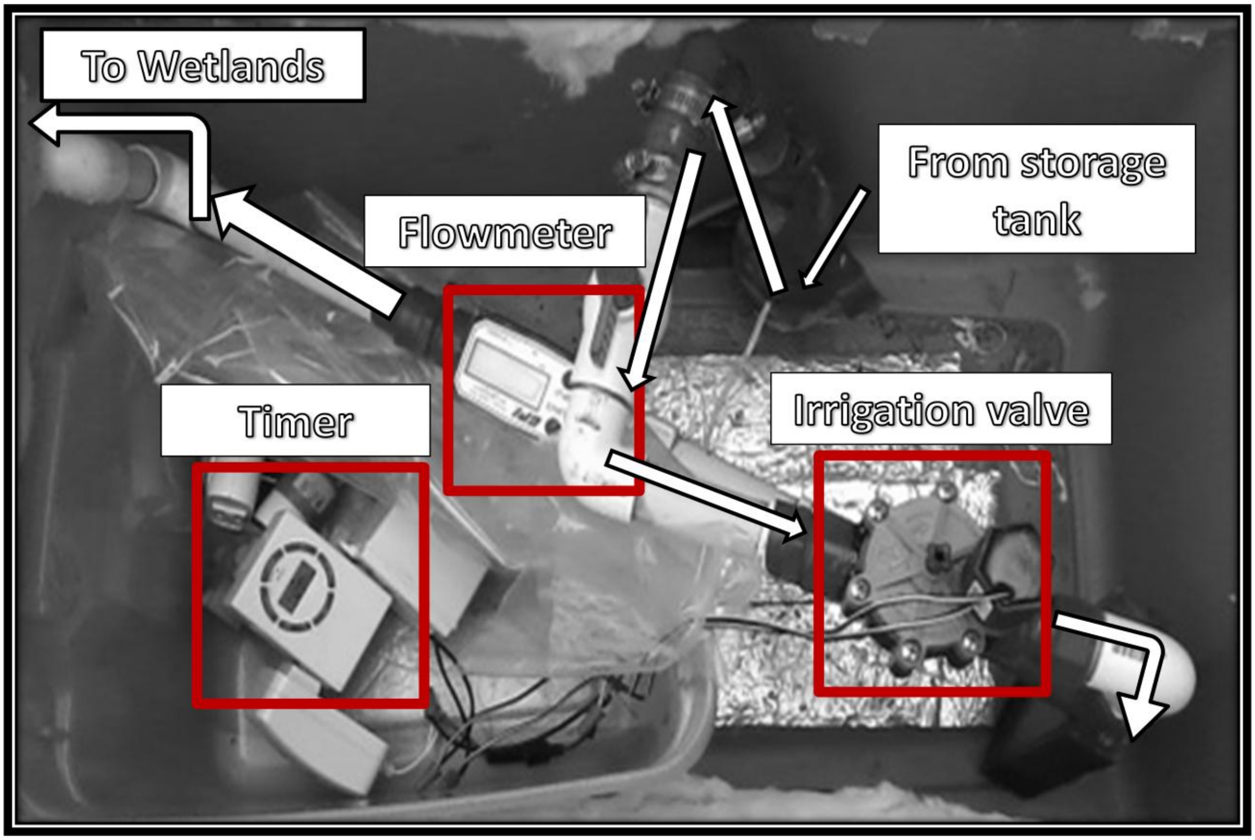

Figure 3.7: Flowmeter, irrigation valve and timer setup

Because of the syphone, the flowmeter was unreliable. In hindsight, it may have been better to put the irrigation valve on the opposite side of the flowmeter. Having the flowmeter on the upstream side of the irrigation valve may have prevented false readings brought on by air flow rather than water flow through the flow meter. In the interest of time, the system was left alone and the flow was estimated by measuring the change in the water level in the tank.

Experimental Plan

The wetland was initially designed for a baseline hydraulic loading rate (HLR) of 0.036 m d^{-1} (approximately 5 day hydraulic retention time (HRT)). This HLR was used from the fall of 2008 to the summer of 2009. Once the baseline was established, HLRs were intentionally varied within each season to obtain required data for parameter estimation (Table 3.5).

Table 3.5: Desired and observed HLRs and average HRTs for FWS and SF wetlands

HLRs for the FWS Wetland			
Dates	Desired HLR (m day^{-1})	HRT (days)	Observed HLR± SD (m day^{-1})
10/1/08 - 5/18/09	0.036	5	0.032 ± 0.005
5/19/09-8/20/09	0.026	7	0.027 ± 0.007
8/21/09-3/9/10	0.06	3	0.082 ± 0.039
3/10/10-5/13/10	0.026	7	0.027 ± 0.001
5/13/10- 8/22/10	-	-	-
8/23/10- 9/3/10	0.06	3	0.058 ± 0.005
9/4/10- 9/28/10	0.026	7	0.030 ± 0.012
9/29/10-10/13/10	0.045	4	0.046 ± 0.002
10/14/10-11/4/10	0.02	9	0022 ± 0.005
HLRs for the SF Wetland			
Dates	Desired HLR (m day^{-1})	HRT (days)	Observed HLR± SD (m day^{-1})
7/30/1/10- 8/5/10	0.022	6	0.02 ± 0.003
8/6/10-9/3/10	0.037	4	0.037 ± 0.013
9/4/10- 9/28/10	0.043	3	0.047 ± 0.010
9/29/10- 11/4/10	0.018	7	0.02 ± 0.002
11/5/10-10/13/11	-	-	-
10/14/11-11/15/11	0.015	8	0.033 ± 0.035
11/16/12-2/1/12	-	-	-
2/2/12- 3/9/12	0.008	13	0.01 ± 0.002
3/10/12- 5/1/12	0.015	8	0.016 ± 0.001

A range of HLRs were selected based on the amount of graywater that was available during each season and chosen to adequately estimate *k*-*C** parameters. This range was revised during the experiment depending on the results that were obtained from the baseline tests and the results obtained after each set of completed tests with a given HLR. A minimum of three sample points were desired at each HLR range. The desired HLR was approximated by varying the pump speed of the peristaltic pump or duration of time the pump was "on" as described above.

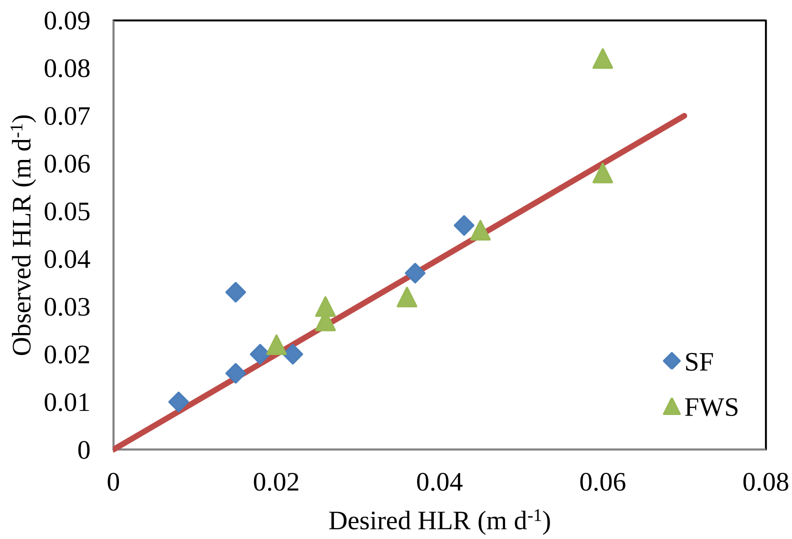

Figure 3.8: Comparison of Desired HLR values and Observed HLR values

Actual HLR was subjected to fluxuation due to climate conditions and the variability in the generation of graywater from both the dorm and building sources (Figure 3.8).

Water Quality Analysis Methods

All measurements, sample collections, preparations, storage methods and water quality analyses were conducted following standard analytical methods (APHA, 1998). Water quality analysis included pH, Dissolved Oxygen (DO), conductivity, turbidity, Biological Oxygen Demand (BOD_5), Total Organic Carbon (TOC), Total Nitrogen (TN), Total Solids (TS), Total Suspended Solids (TSS), Total Volatile Solids (TVS) and three anionic surfactants (Linear Alkylsulfonate (LAS), Alkylethoxysulfate (AES) and Alkylsulfate (AS)). Quality assurance samples (blanks, duplicate analyses, and standards) were analyzed throughout the experiment. Multiple replications (generally 3 subsamples from the same collection bottle) were used for every analysis when possible, and highest dilutions were always reported. Temperature and DO were analyzed in the field using a membrane electrode (Yellow Springs Instruments DO200, Yellow Springs, Ohio). Turbidity was measured with a nephelometric turbidimeter (Hach 2100N, Loveland, Colorado). TOC and TN were analyzed via combustion of acidified samples (Shimadzu TOC-V CSH/CSN, Columbia, Maryland).

Surfactant analysis required additional preparation. All glassware used in sampling and testing was rinsed with hot tap water, DI water and methanol before being rinsed once more in DI water. Collection and storage vessels were also placed in an oven (at least 105C) for a minimum of 12 hours. Methods for detecting AES, AS, and LAS were developed using an LC/MS (Negahban-Azar *et al*, 2012). Solid phase extraction was completed using Oasis® Extraction Cartridges (Waters, Milford, Massachusetts), loading each cartridge with one milliliter of methanol, one milliliter of DI water, three milliliters of sample and one milliliter 5% methanol in water (v/v). Finally, one milliliter of methanol was drawn through each cartridge and collected in clean vials and transferred to 2mL autosampler vials to be stored in the freezer until analysis. LAS, AES, and AS concentrations were quantified using an Agilent 1200 High-throughput HPLC system in conjunction with an Agilent 6220 Accurate Mass Time of Flight mass spectrometer (Agilent Technologies, Santa Clara, California). Chromatographic separation was performed using a XTerra® MS C18 column (2.5 μm, 50 × 2.10 mm; Waters, Milford, Massachusetts). The data was stored and analyzed using MassHunter Workstation software (Agilent Technologies, Santa Clara, California). Sample volumes of 20 μL were introduced to the instrument with an auto-sampler. Both right and left column temperatures were maintained at 40^{o}C. A gradient method was used with a mixture of water with 10 mM ammonium acetate (A) and acetonitrile with 10 mM ammonium acetate (B). The gradient method included an initial mix of 62% water and 38% acetonitrile. This was followed by a linear increase to 65% acetonitrile within 25 minutes, a linear increase to 80% acetonitrile for 10 minutes and a post run at initial conditions for 10 minutes. A flow rate of 0.32 mL/min was used. A DAD-UV (λ = 254 nm) detector was used for the determination of LAS and AES/AS (Table 3.6). To determine C10-13 LAS, an electrospray ionization (negative ion mode) mass spectrometer was used.

Table 3.6: Method of surfactant detection

Compound	*Column*	*Solvents*	*Gradients Method*
LAS **AS** **AES**	XTerra® MS C18 column (2.5 µm, 50×2.10 mm), Waters Corp (Milford, MA)	1) water + 20 mM ammonium acetate 2) acetonitrile + 20 mM ammonium acetate	1) 60% water and 40% acetonitrile 2) Linear increase to 65% acetonitrile within 20 min 3) Linear increase to 80% acetonitrile within 10 min 4) Constant 80% acetonitrile for 5 min 5) Linear decrease to initial condition within 1 min 6) Postrun at initial condition for 10 min

The LAS ions were monitored as C10 = m/z 297, C11 = m/z 311, C12 = m/z 325, C13 = m/z 339. The AS was monitored as C12 = m/z 265 and AES homologues were monitored as C12-EO1 =m/z 309, C12-EO2 = m/z 353, C12-EO3 = m/z 397 (Table 3.7).

Table 3.7: Surfactant Homologues

Compound	*Carbon Chain*	*m/z*	*Ionization Mode*
LAS	C_{10}-C_{13}	297; 311; 325; 339	
AS	C_{12}	265	negative
AES	C_{12} EO_{1-3}	309; 353; 397	

Negahban-Azar *et al* (2012) performed recovery tests prior to experimentation and determined that recovery rates for SPE were above 92%.

Performance evaluation

HLR indicates the rate at which wastewater contaminants enter the wetland. HLR is the volumetric flow rate (Q) divided by the aerial surface area (SA; Equation 3.2; EPA, 2000).

$$\mathrm{HLR} = \frac{\mathrm{Q}\,[\mathrm{m^3 d^{-1}}]}{\mathrm{SA}\,[\mathrm{m^2}]} \qquad \text{(Equation 3.2)}$$

Constructed wetlands can be evaluated using the hydraulic retention time (HRT), which measures the amount of time that the constituents spend in the wetland, typically in days. Literature values for HRT can vary depending on the type of influent and desired treatment, but most of the results show a treatment range of 5-25 days, depending on influent contaminant concentration. HRT is the total volume of the wetland (V) divided by the flow rate (Q; Equation 3.3).

$$\mathrm{HRT} = \frac{\mathrm{V}\,[\mathrm{m^3}]}{\mathrm{Q}\,[\mathrm{m^3\ d^{-1}}]} \qquad \text{(Equation 3.3)}$$

Determination of *k* values for the wetlands required taking samples under several different HLRs during each season. The *k* of constituents within the water directly correlates with the HLR (Equation 3.2)

Percent mass removals were calculated for each of the four seasons. To better demonstrate the effect of the HLR, mass removals were further divided into two HLR ranges. The removal rates of the SF wetland reported are the removal rates collected after the wetlands were separated and untreated graywater was supplied to the cell. Those determined prior to the summer of 2010 had undergone prior treatment in the FWS (Jokerst *et al*, 2011).

Removal rates (*k*) were determined through an iterative process, minimizing the residual sum of square error (RSSE) between the observed effluent concentration and the predicted effluent concentration, utilizing Equation 3.4 to determine the predicted effluent concentration.

$$C_{eff,predicted} = C^{*} + (C_{in} - C^{*})e^{-k/\mathrm{HLR}} \quad \text{(Equation 3.4)}$$

Where: $C_{eff,\,predicted}$ = Effluent constituent concentration (mg L^{-1})

C^* = Background constituent concentration (mg L^{-1})

C_{in} = Influent constituent concentrations (mg L^{-1})

The RSSE was then divided by the number of data points in each sample set (n) to determine a goodness of fit. The solver function embedded in Microsoft Excel was applied to minimize the calculated value for RSSE (based on k and C^* estimates) and provide best parameter estimates. Minimum and maximum background constraints were set for C^*. Upper and lower bounds for C^* were established based on literature values. The upper bound for C^* for BOD_5 was 10 mg L^{-1} (EPA, 2000; Kadlec and Knight, 1996), and for TN and ammonia the upper bound was 3 mg L^{-1} (Kadlec and Knight 1996).

To account for precipitation and ET the observed concentration was adjusted as shown in Equation 3.5:

$$C_{eff,adj} = \left(\frac{Q_{out}}{Q_{in}}\right) C_{eff} \quad \text{(Equation 3.5)}$$

Where: $C_{eff,\,adj}$ = Flow adjusted effluent constituent concentration (mg L^{-1})

Q_{out} = Flow out of the wetland (m^3 day^{-1})

Q_{in} = Flow into the wetland (m^3 day^{-1})

C_{eff} = Measured effluent constituent concentration (mg L^{-1})

More details on this method can be found in Bergdolt *et al* (2011). Once the estimation of k and C^* are complete, they can be applied to design criteria for sizing constructed wetlands. The surface area (SA) of a wetland can be determined using Equation 3.6.

$$\text{SA} = \left(\frac{Q}{k}\right) ln \left[\frac{(C_{in} - C^*)}{(C_{eff,adj} - C^*)}\right] \qquad \text{(Equation 3.6)}$$

Where: Q = flow rate (m^3 day^{-1})

k = the first order removal rate constant (m day^{-1})

C^* = the background constituent concentration (mg L^{-1})

$C_{eff, adj}$ = the effluent constituent concentration (mg L^{-1})

To determine comparable estimates of surface area the following assumptions of parameter values were applied to Equation 3.6. The flow rate (Q) was assumed to be 0.4 m^3 d^{-1}; C_{in} was estimated by averaging the influent concentration of all BOD_5 and TN data and was assumed to be 87 and 12 mg L^{-1} for BOD_5 and TN, respectively. C_{eff} was estimated based on literature values and water quality goals. When C^* was equal to the estimated C_{eff} values, C_{eff} was raised by ten percent so that a SA could be determined. Future state discharge standards allow for a running average of 5.7 mg TIN L^{-1} over the most recent 12 months and a 30 day average of 30 mg L^{-1} BOD_5; however, a well-functioning wetland is capable of treating graywater to a higher standard so *Ceff* was taken to be 20 and 4 mg L^{-1} for BOD_5 and TN, respectively.

In addition to mass removal and removal rates, aerial loading rates (ALR) were determined for the FWS and SF. Aerial loading rates can be estimated by utilizing the following equation:

$$\text{ALR} = \frac{C_i * q_i}{1000 * \text{SA}} \qquad \text{(Equation 3.8)}$$

These ALR values were compared to those suggested by the EPA for domestic wastewater and suggestions for graywater were attempted.

Statistical Analysis

Microsoft Excel was also used for all statistical analyses. When evaluating the effluent characteristics the % mass removals of the constituents were analyzed using ANOVA: Single-Factor analysis (α=0.1). When evaluating the influent for seasonal variability, the constituent concentration was analyzed using ANOVA: Single-Factor analysis (α =0.5). A linear regression analysis was used when comparing BOD_5 and TN HLR and % mass removals. XLSTAT (Addinsoft, New York, NY) software was applied for this analysis. The null hypothesis of a linear regression is that the slope of the line (*m*) created based on the data points is equal to zero. When the slope of the regression is not zero the null hypothesis can be rejected. The severity of the slope along with the degree to which the linear regression fits the data points (R^2) results in statistical significance. Trends were determined to be statistically significant if the null hypothesis that the slope (*m*) is zero was rejected at the 90% confidence interval.

Results and Discussion

Mass Removal Rate

Mass removal rates were analyzed based on both season and HLR. Because HLR does impact mass removal rate (Bergdolt *et al,* 2012), analysis of mass removal was separated based on two ranges of HLR, 0.009-0.02 m d^{-1} and 0.02-0.04 m d^{-1}. Initially, smaller ranges were selected in an attempt to determine how a wide range of HLRs affected the performance of the wetlands, however, missing data made comparison between wetlands impossible so the HLRs ranges were condensed until a good comparison could be made while still providing some division of HLRs. These HLR values correspond to the HRTs for the FWS and SF wetland reported in Table 3.8.

Table 3.8: HLR values and corresponding HRTs for the FWS and SF wetlands

HLR (m d^{-1})	HRT (days)	
	FWS	*SF*
0.009	23	14
0.02	10	7
0.04	5	4

The SF provided consistently higher average mass removal (%). The boxplots below were created such that bottom of the boxplot represents the lower 25% of the data (1^{st} quartile), the line inside the boxplot represents the median and the upper edge of the boxplot represents the third quartile data. The low and high error bars extend to the minimum and maximum values, respectively. The mean value is represented as the black diamond data points within the boxplots.

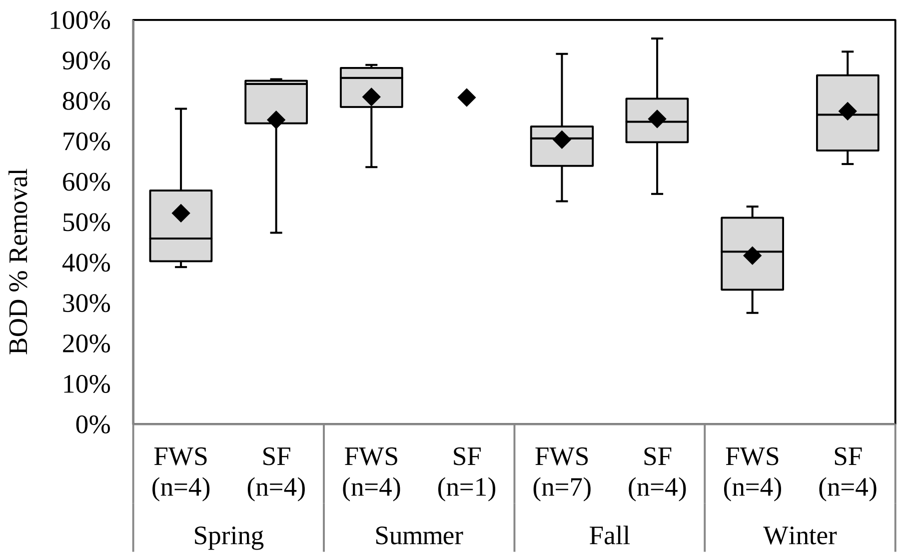

Figure 3.9: BOD_5 Removal Comparison at HLR = 0.009-0.02 m d-1

The seasonal trends are more visible in the lower HLR range (Figure 3.9) compared to the higher HLR range (Figure 3.10). Over the whole range of HLR values (high and low HLR combined), the SF wetland performed significantly better compared to the FWS wetland (P=0.04). In addition, for the lower range of HLR (Figure 3.9) there is a statistically significant difference in the performance during the spring and winter seasons (P = 0.03 and 0.1, respectively; Note: one outlier was removed from each of the wetlands during the spring season) with a higher mass removal being observed for the SF wetland. More data is required at the higher HLR range to make conclusions about the performance of both wetlands (Figure 3.10).

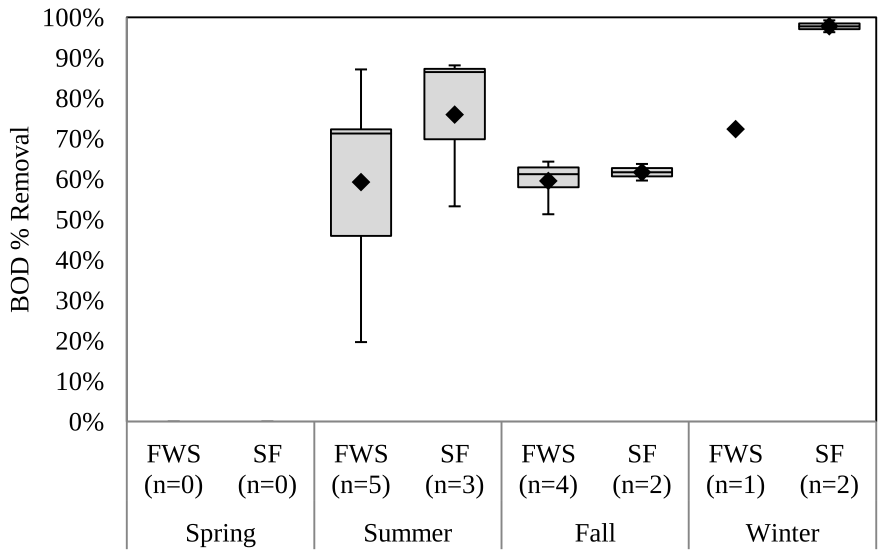

Figure 3.10: BOD_5 Removal Comparison at HLR = 0.02-0.04 m d^{-1}

Similar to BOD_5 results, the SF was more successful in removing TN; however, some seasonal variation is visible to some extent in both wetlands (Figure 3.11). This trend suggests that plant uptake plays a significant role in removal in both wetlands during the growing seasons. While microbial activity is contributing to a certain extent in both wetlands as well, the ability of the SF wetland to maintain high removal rates during the wetland plants senescent periods indicates that microbial activity is greater in the SF wetland.

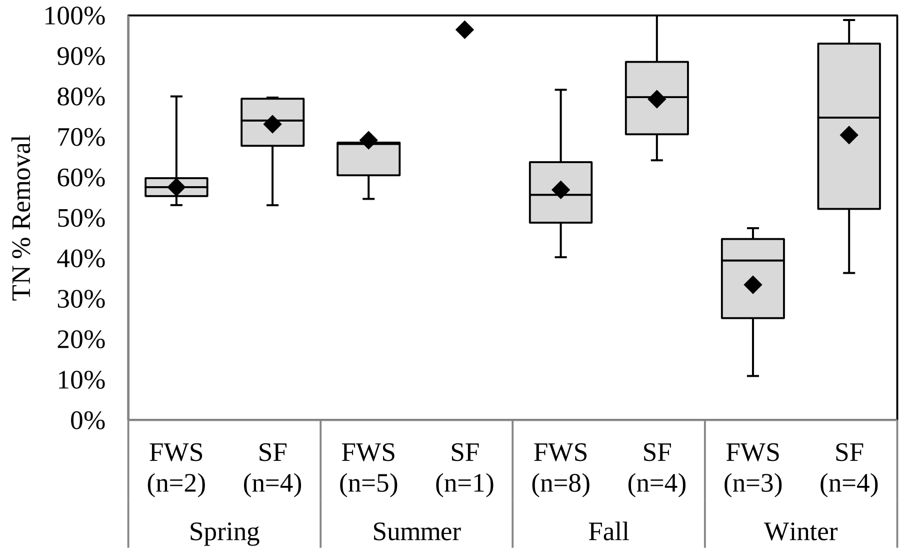

Figure 3.11: TN Removal Comparison at HLR = 0.009-0.02 m d^{-1}

Statistical analysis of the mass removal (%) for BOD_5 and TN indicated there was a significant difference between the seasons in the FWS (P = 0.09 and 0.04, respectively) but no significant difference between the seasons in the SF (P = 1.00 and 0.88, respectively; Table 3.9).

Table 3.9: Statistical analysis comparing seasonal % removal by mass in the FWS and SF wetlands (Low HLR only)

	P-values from ANOVA analysis (α=0.1)	
	FWS	SF
BOD_5	0.09	1.00
TN	0.04	0.88

Comparison of between % removal by mass for the SF and FWS wetlands in each season indicates that there is a significant difference in the spring and winter (Table 3.10) for BOD_5. The summer and fall indicated no significant difference between the FWS and SF. Unlike BOD_5, TN results indicated that there was a significant difference between the two wetlands for every season except spring (Table 3.10). Combined with the results summarized in Table 3.9, these results suggest that the FWS wetland was more prone to seasonal variability than the SF wetland;

however, it should be noted that the differences in the spring and winter may be due, in part, to the extreme variations in these seasons over different years.

Table 3.10: Statistical Analysis comparing % removal by mass between FWS and SF wetland during each season

	P-values from ANOVA analysis (α=0.1)			
	Spring	Summer	Fall	Winter
BOD_5	0.03	0.88	0.64	0.1
TN	0.6	0.05	0.04	0.005

Over the whole range of HLR values (high and low HLR combined), the SF wetland performed significantly better compared to the FWS wetland at removing BOD_5 (P=8E-5). Despite the more pronounced seasonal variability seen in the TN data, the SF remained efficient at removing TN during the spring and winter months while the FWS decreased in performance in both HLR ranges (Figures 3.11 and 3.12). For the lower range of HLR (Figure 3.11) there is a statistically significant difference in the performance during the summer, fall and winter seasons (P = 0.05, 0.04 and 0.005, respectively) with a higher mass removal being observed for the SF wetland. With more data, there may also be a significant difference in the spring as well. More data is required at the higher HLR range to make conclusions about the performance of both wetlands (Figure 3.12); however the trends appear to apply to the higher HLR range as well. The greater removal of nitrogen during the growing seasons suggests that, in both cases, plant uptake contributes greatly to the removal of nitrogen from the system. The results also indicate that the SF wetland may provide some protection during the colder months allowing microbiological activity to continue year round.

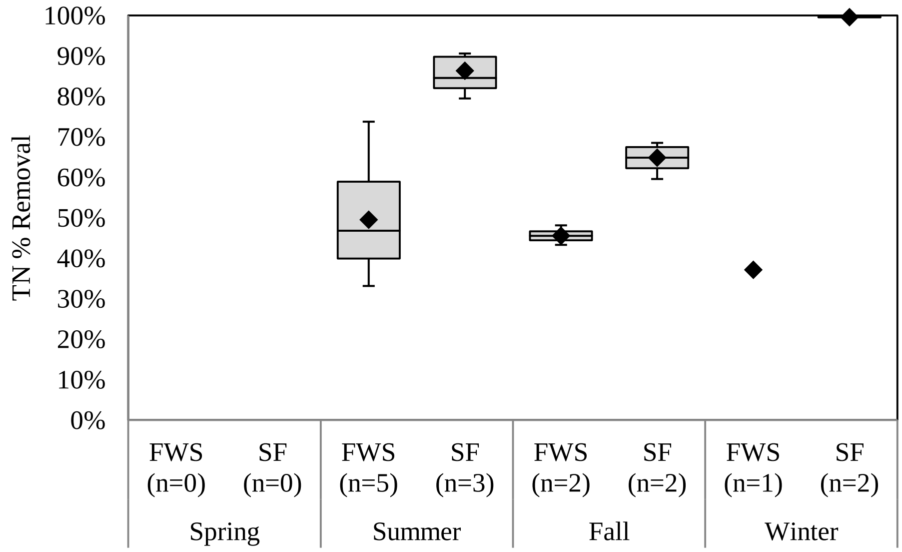

Figure 3.12: TN Removal Comparison at HLR = 0.02-0.04 m d^{-1}

The mass removal rates presented for BOD_5 and TN correspond to other results comparing FWS and SF treatment performance. As mentioned above, Kadlec and Knight (1996) reported that FWS wetlands removed 67% BOD5 and 69% TN while SF wetlands removed 67-80% BOD5 and 76% TN. Hijosa-Valsero *et al* (2010) reported that BOD_5 removals were similar; however, when comparing the figures, it appears that the FWS wetlands had slightly more variation both between summer and winter and between the two summers than was determined for the SF wetlands.

Mass removal of surfactants was analyzed for five sampling events in September 2010. Both wetlands showed comparable removal efficiencies for each of the two types of anionic surfactants, however, AES/AS were removed much more effectively than LAS (Figure 3.13).

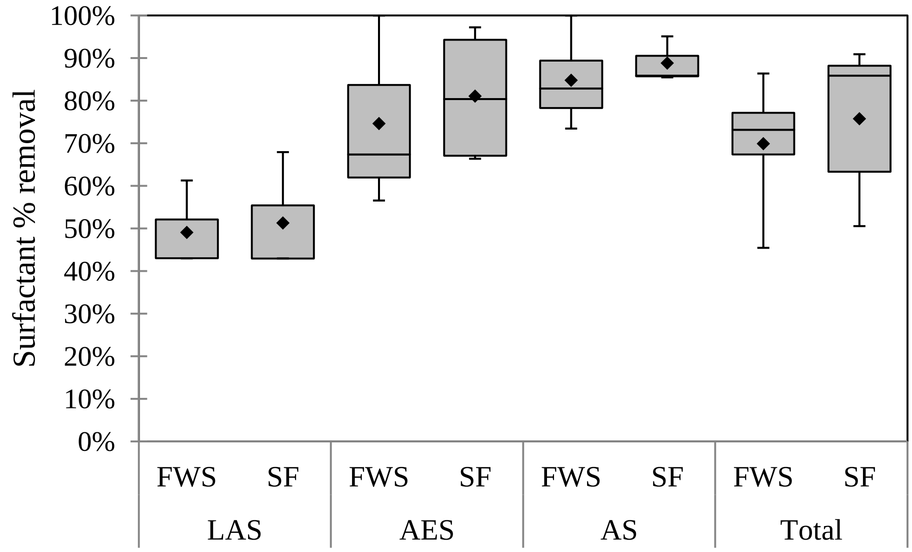

Figure 3.13: Removal of LAS, AES and AS from the FWS and SF wetlands

Part of the reason LAS removals were not as significant may have been the low influent concentrations (Table 3.11). While AES and AS had higher percent removal, the effluent concentration of all three surfactants averaged approximately 1 mol d^{-1}(Table 3.10). Another explanation for this difference is that LAS has been proven to be more persistent than AES under anaerobic conditions (Ying, 2006; Pojana *et al*, 2004) and in general, AES is found in lower concentrations in the natural environment, despite being more popular, when compared to LAS (Pojana *et al*, 2004).

Table 3.11: Average Influent and Effluent Surfactant Concentrations for the FWS and SF wetlands

	FWS (mol d^{-1})		SF(mol d^{-1})	
	Influent	Effluent	Influent	Effluent
AES	5.5	2.3	8.1	1.3
AS	4.5	0.2	4.1	1.0
LAS	1.3	1.1	1.7	1.4

HLR Relationship with % Mass Removal

A linear regression analysis indicated whether or not there was a correlation between HLR and the % mass removal. As mentioned before, trends were determined to be statistically significant if the null hypothesis that the slope (*m*) is zero was rejected at the 90% confidence interval. Figure 3.14 is an example of a plot used to summarize the data and determine the correlation of HLR to mass removal. In this case, the trend was significant. The remainder of the plots representing the SF can be found in Appendix B.

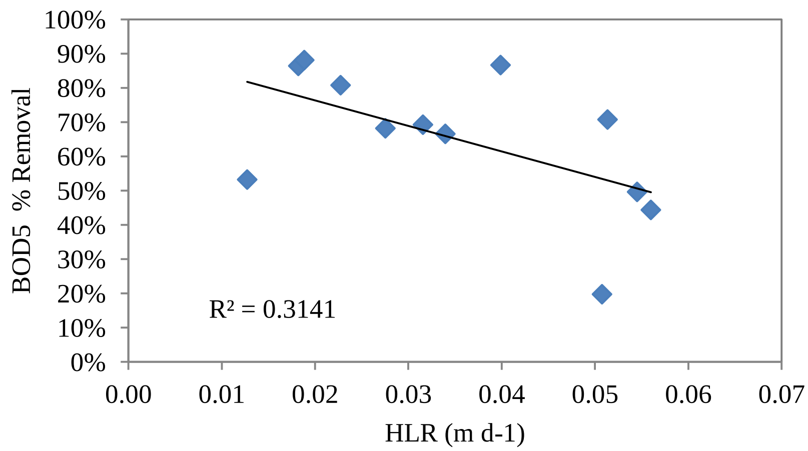

Figure 3.14: HLR vs. Mass Removal (%) for the SF BOD_5during the summer

The results of the regression analysis are summarized in Table 3.11 and indicate that during the summer and fall there is statistically significant correlation between HLR and % mass removal. The results given for the spring are inconclusive due to a lack of usable data points. These results were compared to the statistical analysis completed by Bergdolt *et al* (2012) for the FWS (Table 3.12).

Table 3.12: SF HLR vs. Mass Removal Correlation (* indicates statistical significance of the trend, P≤0.10)

		Spring	Summer	Fall	Winter
BOD_5	***m***	-5.9	-7.4*	-45*	2.2
	R^2	0.01	0.25	0.60	0.25
TN	***m***	17	-7.4*	-41*	4.0
	R^2	0.11	0.40	0.59	0.25

As Table 3.13 indicates, there was a significant correlation between HLR and % mass removal in the winter and fall BOD_5 for the FWS wetland and a significant correlation for summer, fall and winter TN for the FWS wetland. The correlation in the winter for the FWS was very strong while no statistically significant correlation was found in the winter for the SF (Table 3.12). Given these results, it can be concluded that not only does HLR have less of an impact on removal efficiency in the SF wetland but these results also support the conclusion that FWS are more sensitive to seasonal and temperature changes.

Table 3.13: FWS HLR vs. Mass Removal Correlation (* indicates statistical significance of the trend, P<0.10)

		Summer	Fall	Winter
BOD_5	***m***	-1.0	-5.1*	-6.8*
	R^2	0.02	0.51	0.33
TN	***m***	-4.0*	-12.7*	-7.5*
	R^2	0.36	0.52	0.33

*k-C** Parameter Estimation

In addition to mass removal, removal rates (*k*) associated with the SF wetland were determined for the summer and fall months and compared to the *k* rates reported in Bergdolt *et al* (2012). Summer *k* values for BOD_5 are not notably different between the wetlands. This is consistent with the ANOVA results which indicate that there is no significant difference between the FWS and SF wetland performance during the summer season. The summer *k* values for TN are also similar but through statistical analysis it was determined that there was a significant difference between wetland performance. The SF fall BOD_5 *k* value is notably larger than that of the FWS and the *C** value is also much higher. While *C** for the TN analysis remained the same, once again the *k* value for the SF wetland is significantly higher than that of the FWS wetland (Tables 3.14-3.15). Valid parameter estimates could not be determined for the spring and winter because there was a narrow range of HLRs represented in addition to a limited number of data points overall. In general, these results are consistent with mass removal results which indicated increased contaminant removal rates in the SF wetland.

Table 3.14: BOD5 k and C* Estimates for Graywater Processing (valid parameter estimates for spring and winter were not achieved)

		Summer	**Fall**
SF	***k*** **(m yr^{-1})**	14.7	29.5
	C* **(mg L^{-1})**	6.06	20.0
	RSSE/*n*	17	27
	n	8	7
FWS	***k*** **(m yr^{-1})**	15.9	15.2
	C* **(mg L^{-1})**	6.4	10.0
	RSSE/*n*	275	103
	n	13	9

Adapted from Bergdolt *et al* (2012)

Table 3.15: TN k and C* Estimates for Graywater Processing (valid parameter estimates for spring and winter were not achieved)

		Summer	**Fall**
SF	***k*** **(m yr^{-1})**	19.8	24.6
	C* **(mg L^{-1})**	1.0	3.0
	RSSE/*n*	1.63	0.15
	n	8	3
FWS	***k*** **(m yr^{-1})**	16.4	8.5
	C* **(mg L^{-1})**	1.0	3.0
	RSSE/*n*	1.2	3.0
	n	13	9

Adapted from Bergdolt *et al* (2012)

Given these results, the necessary surface area required to achieve a desired effluent concentration can be calculated (Table 3.16-3.17). There was no discernible difference between surface areas calculated for the summer (BOD_5 and TN) and fall BOD_5, particularly given the large uncertainty associated with *k-C** parameter estimation. The surface areas calculated for TN in the fall were notably different with the FWS area nearly three times the area of that calculated for the SF wetland (Table 3.17).

Table 3.16: Surface Area (m^2) required for treatment based on calculated k values for Summer

Summer	**Required Surface Area (m^2)**	
	BOD_5	**TN**
FWS	15.7	11.6
SF	16.8	9.6

Table 3.17: Surface Area (m2) required for treatment based on calculated k values for Fall

Fall	**Required Surface Area (m^2)**	
	BOD_5	**TN**
FWS	18.7	37.7
SF	20.8	13.1

Conventionally, FWS wetlands are expected to provide nitrification (aerobic conditions) and SF wetlands provide denitrification (anaerobic conditions). Because of this, SF wetlands require more surface area than FWS wetlands due to the nature of slow growing anaerobic microbes. Still, there are many variables that may influence the performance of both types of wetlands (i.e., climate, water depth, influent characteristics, etc.). It has also been suggested that the dosing regimen can have a significant effect on SF wetland performance (Jia, 2011). The SF wetland in this study was design to function as a horizontal continuous flow subsurface wetland. Due to availability of graywater during times of the year when the university was not in session and the changes made to the system during the fall of 2011, the system functioned as a dosed horizontal subsurface flow wetland for a time which may have facilitated its ability to provide both nitrification and denitrification. The reed bed of the SF wetland is also very dense and well established, leading to the theory that the plant's root mass is facilitating a beneficial network of aerobic and anaerobic zones and a stable environment for microbial function year-round. The results, while they do not entirely support convention, do support the claim that SF wetlands are more stable year-round.

ALR

ALR values were compared to effluent concentrations to try to determine some suggestions for design ALR values based on desired effluent concentration. Only in the summer were there sufficient data and a clear trend to provide good recommendations. Estimations were made for each case but only the recommendations that carry any confidence are highlighted by bold text in Table 3.18. The actual ranges of ALRs tested are given in parentheses in the table.

Table 3.18: Calculated ALRs for SF wetland based on $5 \le C_{out} \le 30$ mg BOD_5/L and $1 \le C_{out} \le 10$ mg TN/L (Measured range of ALR in parentheses)

	BOD_5 (g m^{-2} d^{-1})			TN (g m^{-2} d^{-1})		
	High		Low	High		Low
Summer	**3.02 (7.41)**	-	**0.50 (0.88)**	**1.4 (1.12)**	-	**0.14 (0.28)**
Fall	2.64 (3.43)	-	0.44 (1.46)	0.72 (0.29)	-	0.07 (0.07)
Winter	1.69 (1.81)	-	0.28 (0.65)	0.24 (0.05)	-	0.02 (0.04)
Spring	2.26 (1.41)	-	0.38 (0.47)	0.56 (0.26)	-	0.06 (0.06)

When compared to EPA's suggested loading rates, the results vary. TN rates are comparable to those of the SF wetland loading rates (0.3 – 1.2 g m^{-2} d^{-1}) but the BOD_5 loading rates are much lower than those suggested by the EPA (6.7 – 15.7 g m^{-2} d^{-1}) and do, in fact, fit comfortably within the range given for the FWS wetland (1 – 10 g m^{-2} d^{-1}). More data is required to provide useful recommendations. An example has been provided in Figure 3.15 and the remainder of the plots can be found in Appendix A. The plots do include the current recommended high and low effluent concentrations provided by the EPA for municipal wastewater effluent (designated as solid lines at 30mg/L BOD_5 and 10mg/L TN, 5mg/L BOD_5 and 1mg/L TN, respectively) in addition to an average background concentration (10mg/L BOD_5 and 3mg/L TN).

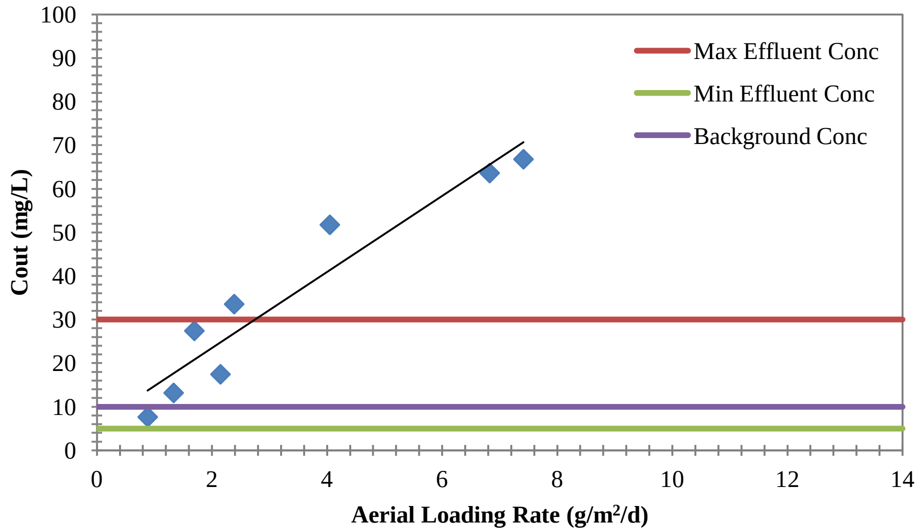

Figure 3.15: Summer SF BOD_5 Example plot of ALR vs. Effluent Concentration

Estimates were made by using the equation of the trend line. The low value corresponds to a C_{out} of 5 mg BOD_5 L^{-1} and 1 mg TN L^{-1}. Likewise, the high value corresponds to a C_{out} of 30 mg BOD_5 L^{-1} and 10 mg TN L^{-1}.

Conclusion

In conclusion, these results suggest that in the case for graywater treatment in arid climates, SF wetlands provide a better treatment option compared to FWS wetlands. Not only do the results indicate that SF wetlands provide more efficient treatment over the seasons but that treatment is relatively stable year round. In particular, the SF wetland showed higher mass removal of both BOD_5 and TN than the FWS wetland during winter months (P=0.1 and 0.005). When all the seasons were compared for each wetland individually there was a statistically significant degree of removal for BOD_5 and TN between the seasons in the FWS wetland (P=0.09 and 0.04) while there was none in the SF wetland (P=1.0 and 0.9). These results are consistent with other findings in the literature. When % mass removals were compared to HLRs, the trends support the ability of SF wetlands to function across a wide range of HLRs and climatic conditions, whereas FWS wetlands are less capable of performing well under less than ideal conditions. While results of the k-C* and SA analyses are limited by available data, the data that was usable suggests once again that SF wetlands are capable of increased rates of removal not only during the warm summer months but also during the cold and transition months. While the FWS wetland meets surface discharge and reuse standards for BOD_5 and TSS for parts of the year (during the summer and fall), the SF wetland meets the discharge requirements year round. More investigation into the efficacy of wetlands at removing harmful pathogens is necessary before reusing wetland effluent for edible crops; however, both wetlands effectively removed *E. coli* to Colorado's restricted use standards. The ability of both wetlands to perform well in regard to TN removal during the growing season indicates a strong reliance on the uptake of nitrogen by the plants. The SF wetlands ability to maintain a certain degree of removal during the senescent periods indicates that the SF wetland is also relying on nitrification and denitrification to a much greater extent than the FWS wetland. The dosing regimen, well established plant communities, low influent BOD concentrations compared to domestic wastewater (Table 2.1) and insulation from the weather provided by the rock media may all contribute to increased effectiveness of the SF wetland at removing TN. One of the main concerns of SF wetlands is clogging in the event of high solids loading. While graywater does have a tendency to vary in composition, high solids are rarely an issue ($TSS_{in,avg}$ = 35 mg/L). In light of the evidence presented herein, subsurface wetlands appear to be the better option for graywater treatment in arid climates, particularly under circumstances of cold winter temperatures. In addition, surfactant removal was comparable between the two wetlands and was consistent with findings in the literature, with 50% removal of LAS and greater than 70% removal of AES and AS in SF and FWS wetlands.

Beyond the quantitative benefits of SF wetlands, there are also a few qualitative benefits worth mentioning. Because the graywater is beneath a layer of rock media, there is less opportunity for mosquito habitation (reduced contribution to a modern vector of disease transmission such as West Nile) and reduced risk of direct contact with the graywater. The SF wetland also saw reduced ET rates when compared to the FWS which may be a result of the plant species, smaller surface area and rock-bed media. Protection from the rock media provides in the summer may also prevent the wetland from freezing in the winter.

CHAPTER 4: SUMMARY AND CONCLUSIONS

Graywater reuse for non-potable demands is gaining popularity because it allows for the reuse of minimally contaminated wash water, generated and treated onsite. The characteristics of graywater make the water suitable for reuse with a reduced degree of treatment when compared to other domestic wastewater sources. Graywater reuse for non-potable demands reduces the demand for treated water and preserves source waters. One method of treating graywater is through implementation of constructed wetlands. Constructed wetlands can offer a scalable, economically sound, low tech and easily maintained method of treating graywater for community scale irrigation reuse. While constructed wetlands are an appropriate technology for graywater treatment there is little research providing the removal rates used in the design of constructed wetlands for graywater reuse. In addition, despite widespread interest in this innovative approach, concerns about water loss due to evapotranspiration have arisen in connection to using constructed wetlands in the arid climates of the western United States.

The foremost objective of this research was to compare the performance of a FWS wetland to a SF wetland for graywater treatment and to evaluate each for their ability to provide treatment adequate for irrigation reuse in accordance with the EPA's surface discharge and Colorado's water reuse standards. This was done by comparison of mass removal rates and requisite surface areas required based on determined removal rates (k). Determining removal rates is important for creating wetland design standards for graywater treatment and reuse. Removal rates were evaluated over the summer and fall of 2010 and 2011 for a SF wetland. These removal rates were compared to the removal rates evaluated over a two year period (2008-2010) for a FWS wetland. The results indicate that SF wetlands provide relatively stable and more efficient treatment year round. In particular, the SF wetland showed higher mass removal of both BOD_5 and TN than the FWS wetland during winter months (P=0.1 and 0.005). When all the seasons were compared for each wetland individually there was a statistically significant degree of removal for BOD_5 and TN between the seasons in the FWS wetland (P=0.09 and 0.04) while there was none in the SF wetland (P=1.0 and 0.9). These results are consistent with other findings in the literature. When mass removals were compared to HLRs, the trends support the ability of SF wetlands to function across a wide range of HLRs and climatic conditions, whereas FWS wetlands are less capable of performing well under less than ideal conditions. Results of the k-C* and SA analyses, though limited in their completeness, suggest once again that SF wetlands are capable of increased rates of removal not only during the warm summer months but also during the winter and transition months. The SF wetlands ability to maintain a certain degree of nitrogen removal during the senescent periods indicates that the SF wetland is not only relying on plant uptake but also nitrification and denitrification to a much greater extent than the FWS wetland. The results not only provide important information for the proper sizing of constructed wetlands but also provide information on what type of wetland is more appropriate for graywater treatment. Subsurface constructed wetlands planted with bulrush provide a stable means of treating graywater for irrigation reuse in areas where there are large seasonal temperature variations and in arid climates.

Community gardens provide many benefits including local food production and community open space but lack the necessary connection between public community and government acknowledgement that would ensure long-term continuity. They provide a place for community activism and act as a platform for true and meaningful social engagement in which members can learn about a wide variety of topics, including the benefits of recycling and composting. Incorporating graywater treatment wetlands into community gardens could provide the necessary link to make graywater reuse and community gardens more universally practiced and supported. By combining community gardens with municipal wastewater treatment (treatment wetlands), the government becomes involved in the education necessary to make the process a safe community sustained entity. Treatment wetlands are adequately simple and can allow a person with minimal training to provide everyday maintenance. Through education in community gardens, the feasibility of graywater reuse can be emphasized while ensuring that best management practices are followed to protect against any harmful effects of graywater reuse (i.e, pathogens). Community gardens provide the opportunity to balance the cost of necessary system improvements with the long-term benefit of edible and non-edible crop production and community growth.

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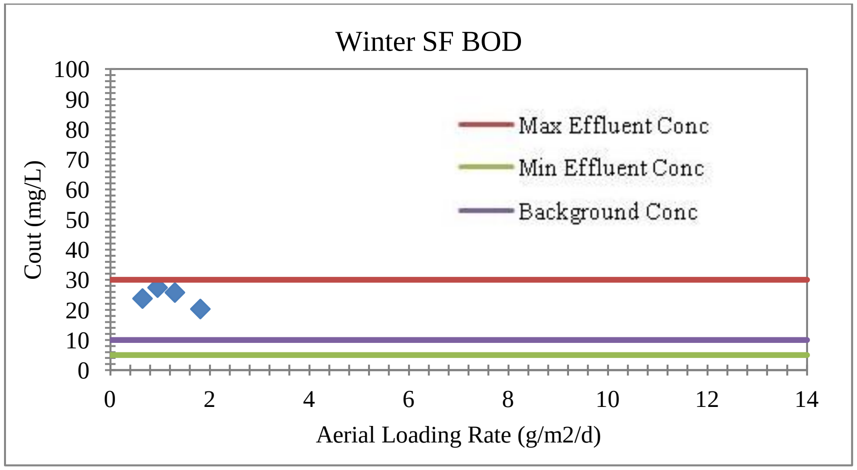
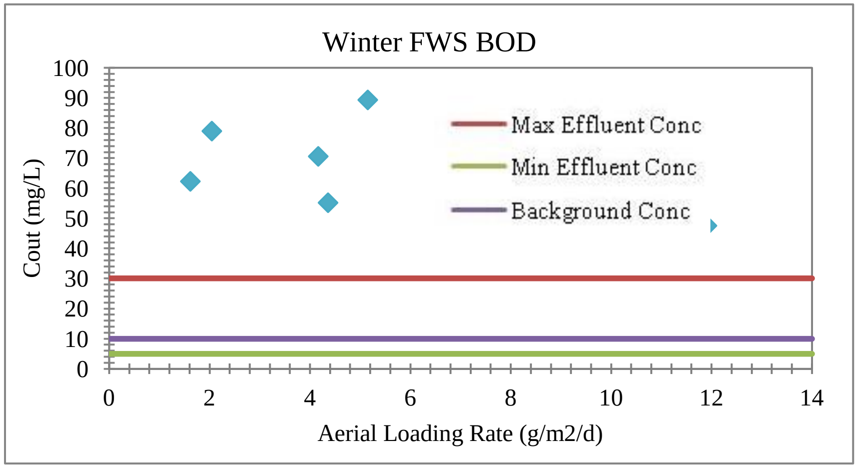
APPENDIX A: AERIAL LOADING RATES VS. C_{OUT}

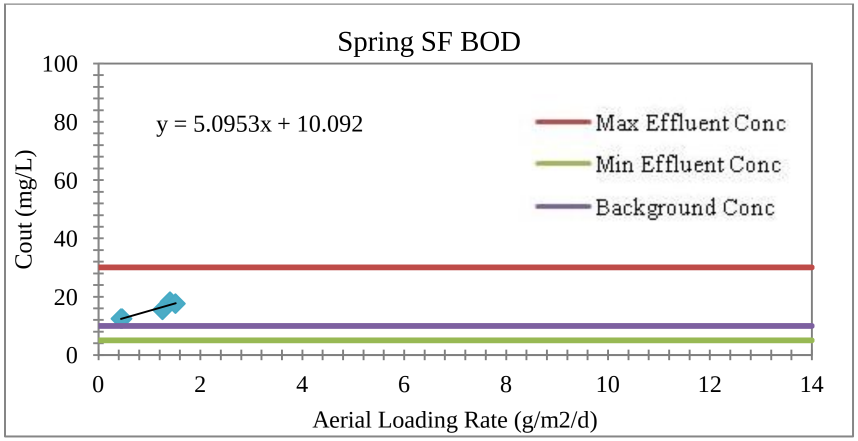
Red line indicates upper bound of EPA suggested effluent concentration (30mg/L BOD_5 and 10mg/L TN)

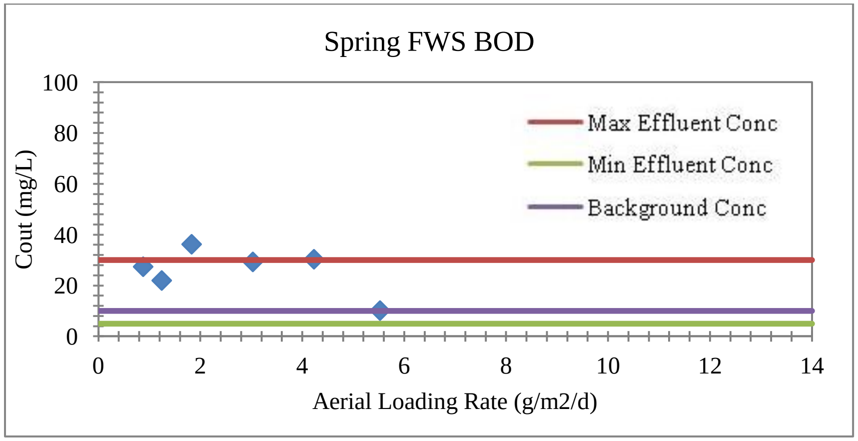
Green line indicates lower bound of EPA suggested effluent concentration (5mg/L BOD_5 and 1mg/L TN)

Purple line indicates estimated average background concentration (10mg/L BOD_5 and 3mg/L TN)





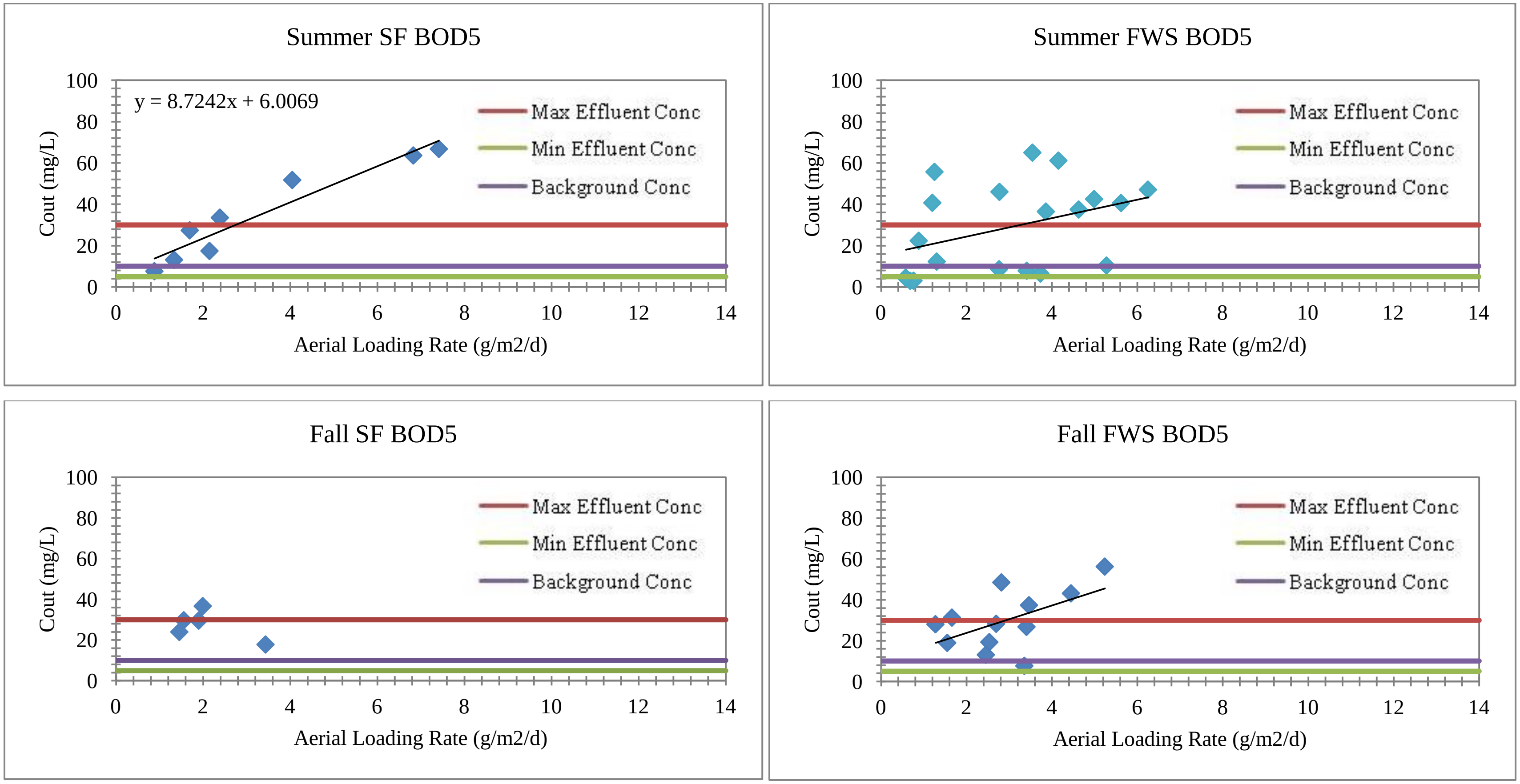




Red line indicates upper bound of EPA suggested effluent concentration (30mg/L BOD_5 and 10mg/L TN)

Green line indicates lower bound of EPA suggested effluent concentration (5mg/L BOD_5 and 1mg/L TN)

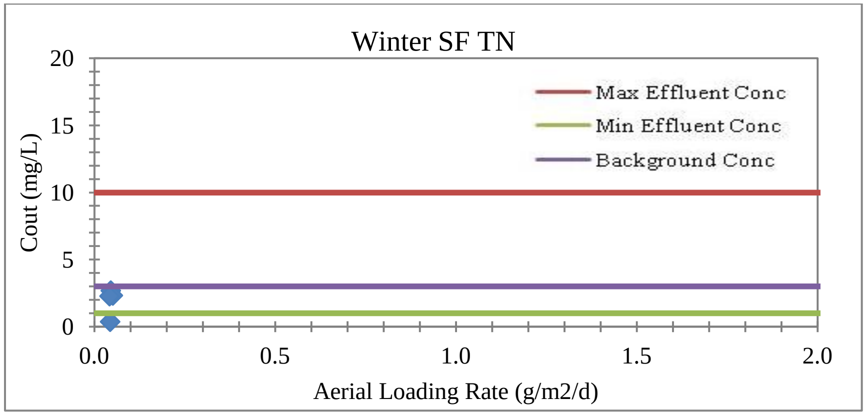
Purple line indicates estimated average background concentration (10mg/L BOD_5 and 3mg/L TN)

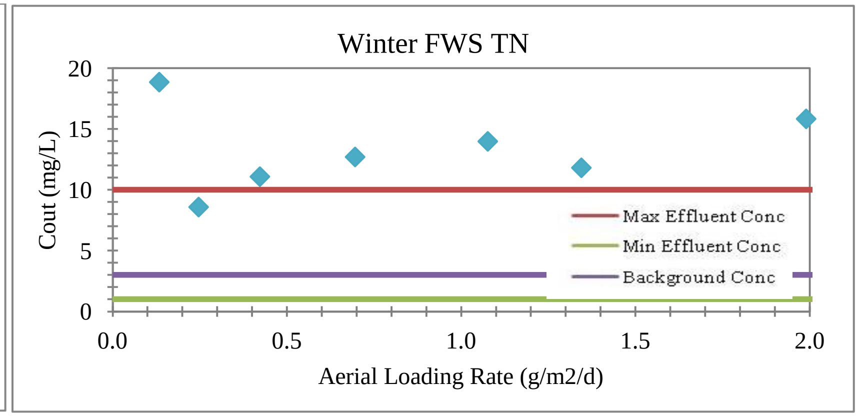


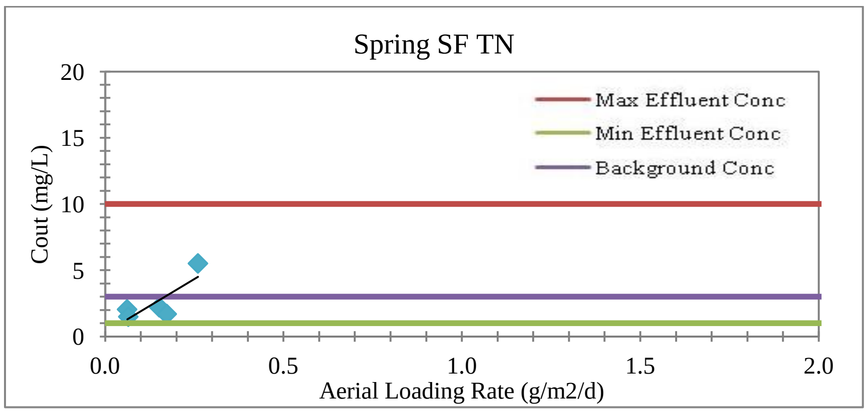
Red line indicates upper bound of EPA suggested effluent concentration (30mg/L BOD_5 and 10mg/L TN)

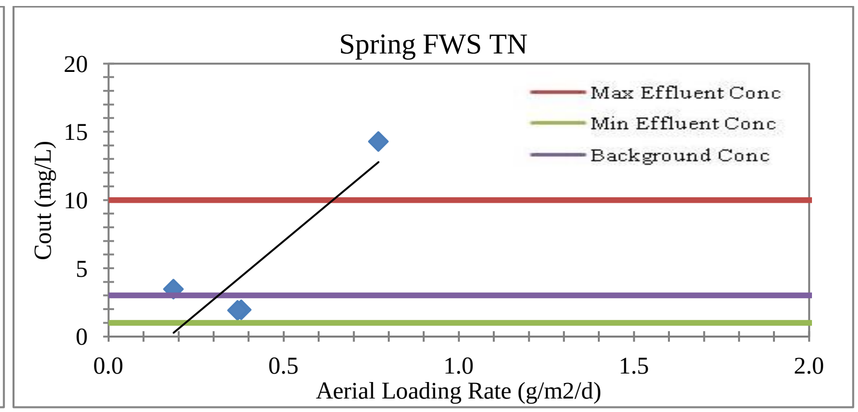
Green line indicates lower bound of EPA suggested effluent concentration (5mg/L BOD_5 and 1mg/L TN)

Purple line indicates estimated average background concentration (10mg/L BOD_5 and 3mg/L TN)





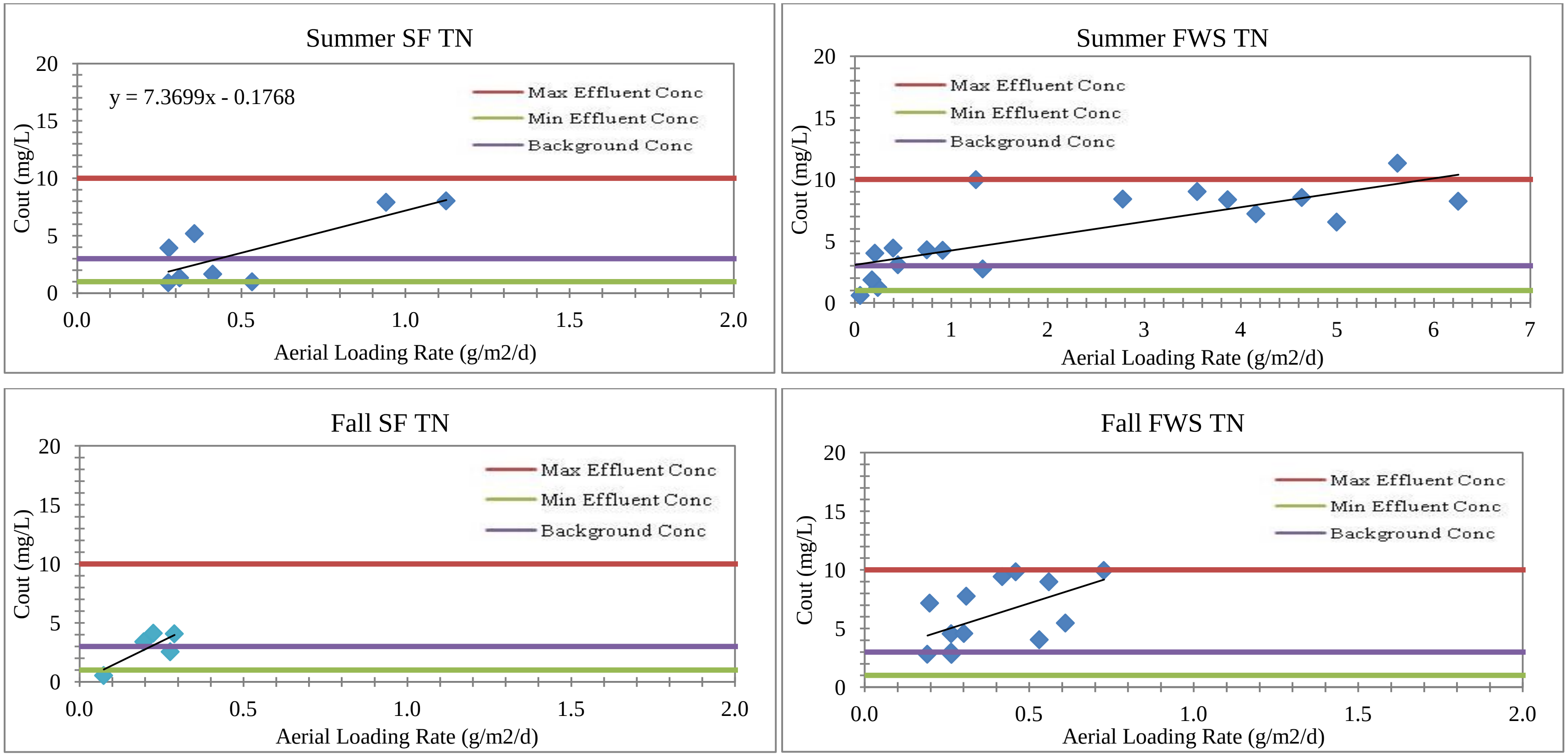




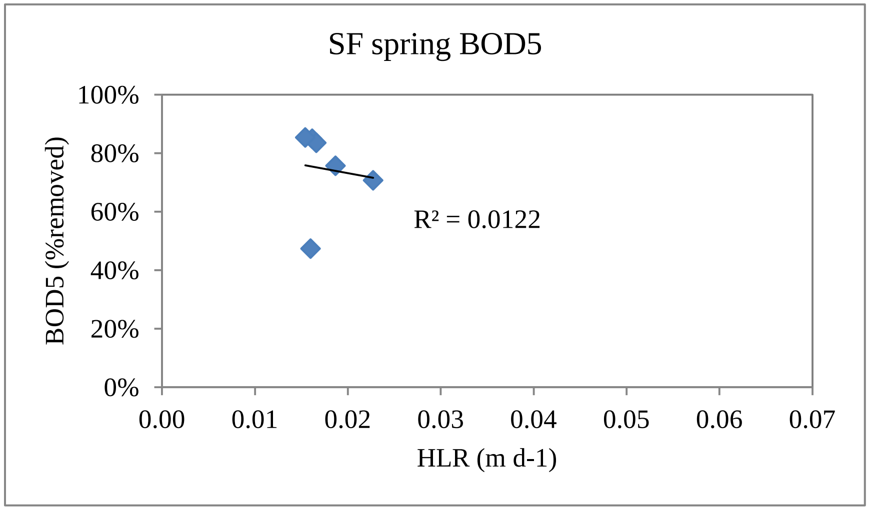
Red line indicates upper bound of EPA suggested effluent concentration (30mg/L BOD_5 and 10mg/L TN)

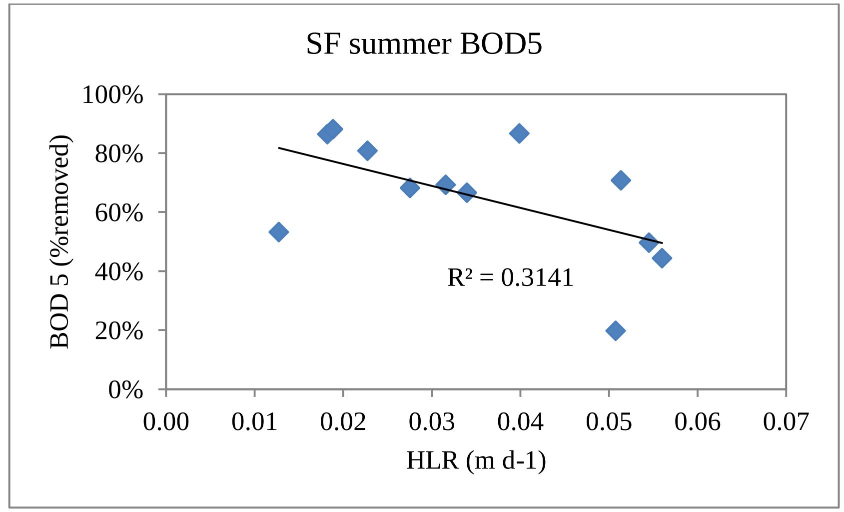
Green line indicates lower bound of EPA suggested effluent concentration (5mg/L BOD_5 and 1mg/L TN)

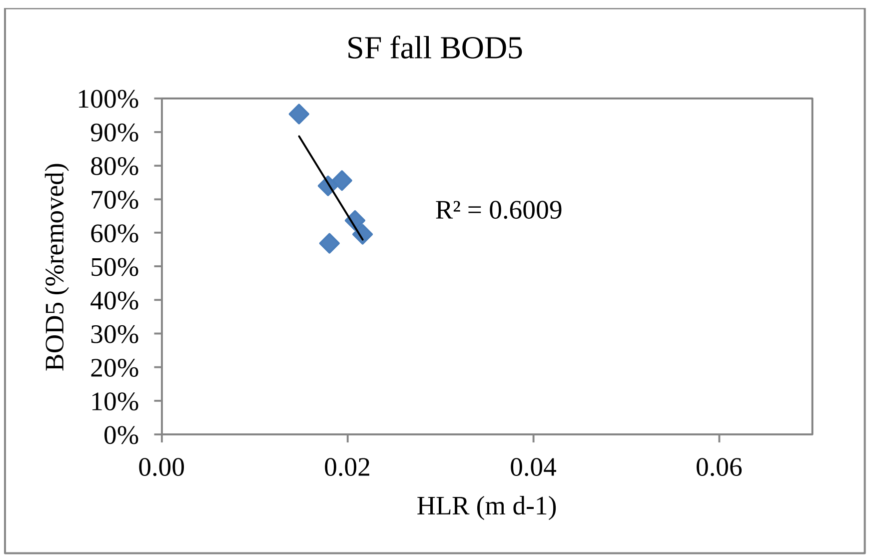
Purple line indicates estimated average background concentration (10mg/L BOD_5 and 3mg/L TN)

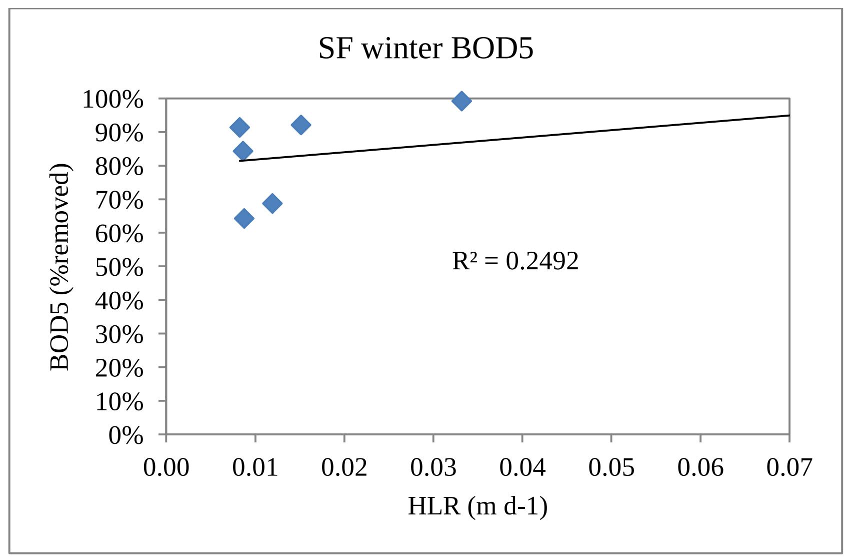


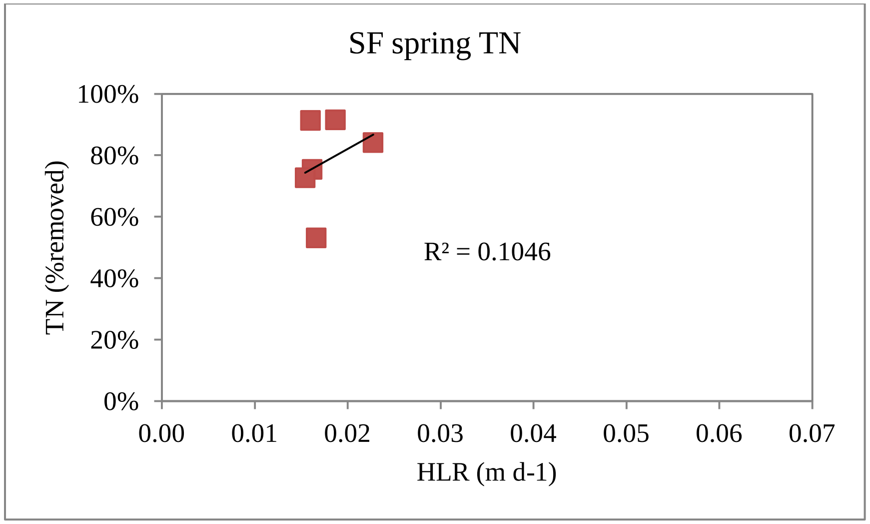
APPENDIX B: SF HLR VS. % MASS REMOVAL PLOTS

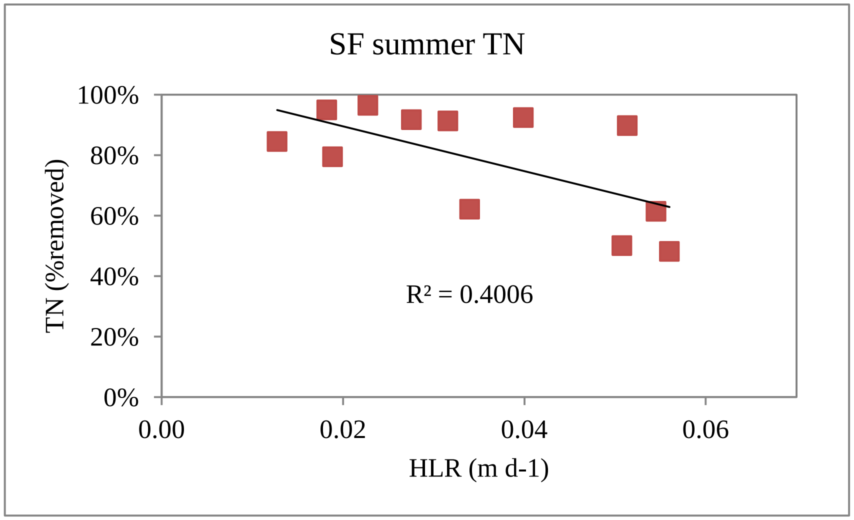


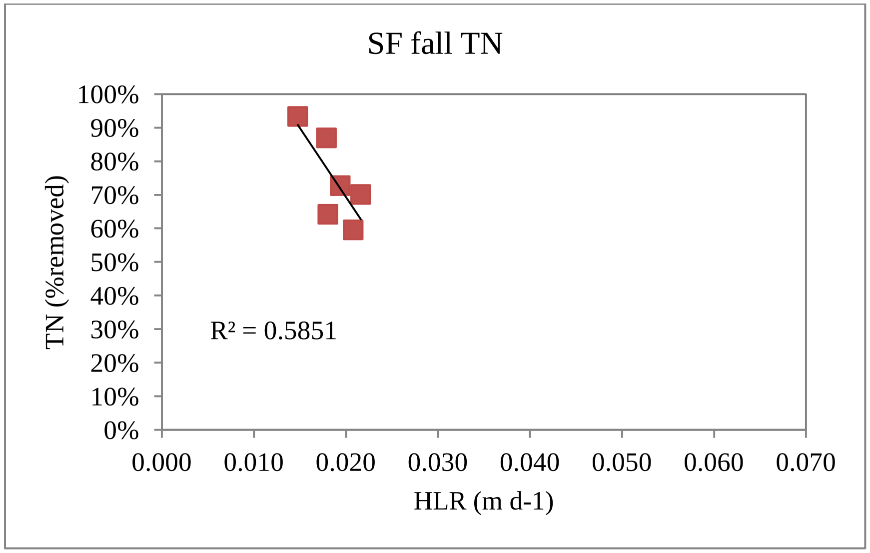


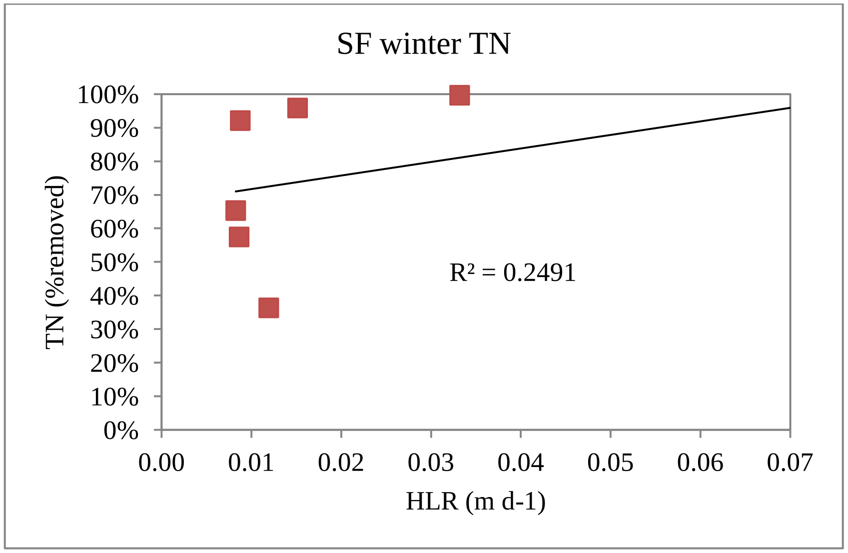





SF spring TN
TN (%removed)
HLR (m d-1)
R² = 0.1046
100%
80%
60%
40%
20%
0%
0.00
0.01
0.02
0.03
0.04
0.05
0.06
0.07


SF summer TN
TN (%removed)
HLR (m d-1)
R² = 0.4006
100%
80%
60%
40%
20%
0%
0.00
0.02
0.04
0.06


SF fall TN
TN (%removed)
HLR (m d-1)
R² = 0.5851
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
0.000
0.010
0.020
0.030
0.040
0.050
0.060
0.070


SF winter TN
TN (%removed)
HLR (m d-1)
R² = 0.2491
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
0.00
0.01
0.02
0.03
0.04
0.05
0.06
0.07

APPENDIX C: COMMUNITY GARDEN RESOURCES

American Community Garden Association website: http://www.communitygarden.org/

Campbell, C.S. and Ogden, M. (1999) *Constructed Wetlands in the Sustainable Landscape*, John Wiley & Sons, New York, NY

Ecological Engineering for Wastewater Treatment, 2nd Ed, edited by Carl Etnier and Bjorn Guterstam, CRC Press, Boca Raton, FL

Glover, Troy D.; Parry, Diana C.; Shinew, Kimberly J. (2005) *Building Relationships, Accessing Resources: Mobilizing Social Capital in Community Garden Contexts*, Journal of Leisure Research, Vol 37(4): 450-474.

Hou, Jeffrey (2009) *Greening cities, growing communities: learning from Seattle's urban community gardens*, University of Washington Press, Seattle, Washington.

Irvine, S., Johnson, L., and Peters, K. (1999) *Community gardens and sustainable land use planning: A case-study of the Alex Wilson community garden*, Local Environment, 4(1):33-46

Lawson, L. (2004) *The Planner in the Garden: A Historical View into the Relationship between Planning and Community Gardens*, Journal of Planning History, 3(2): 151-176

Lee, Vanessa (accessed 2012) *Community Gardens*, http://depts.washington.edu/open2100/pdf/2_OpenSpaceTypes/Open_Space_Types/cgarden_typology.pdf

Smith, Jeremy N (2010) *Growing a Garden City: How Framers, First Graders, Counselors, Troubled Teens, Foodies, a Homeless Shelter Chef, Single Mothers, and More are Transforming Themselves and Their Neighborhoods Through the Intersection of Local Agriculture and Community—and How You Can, too*/ Jeremy Smith, Forward by Bill McKibben, Photographs by Chad Harder and Sepp Jannotta.

Thompson, J. and Sorvig, K. (2008) *Sustainable landscape construction: a guide to green building outdoors; 2nd edition*, Island Press, Washington, DC.

UN Department of Economic and Social Development, Division for Sustainable Development (1992) Agenda 21: http://www.un.org/esa/dsd/agenda21/

APPENDIX D: NEAT PROGRAMS/COMPANIES

US Department of Energy: Solar Decathlon:

> *The U.S. Department of Energy Solar Decathlon challenges collegiate teams to design, build, and operate solar-powered houses that are cost-effective, energy-efficient, and attractive. The winner of the competition is the team that best blends affordability, consumer appeal, and design excellence with optimal energy production and maximum efficiency.*

Program Website: http://www.solardecathlon.gov/

Atlantis Corp:

> *Atlantis's objective is to create sustainable environmental solutions, turning major environmental problems into rejuvenated assets that enhance water quality and reduce or eliminate contaminated water discharge.*

Company Website: http://www.atlantiscorp.com.au/

OrganicaWater:

> *Organica Water's mission is to provide products and services which enable customers all over the world to build and operate space and energy efficient biological wastewater treatment plants that blend harmoniously into urban and residential population centers, ultimately lowering infrastructure costs and facilitating reuse and recycling of treated wastewater.*

Company Website: http://www.organicawater.com/about/history/

Ecological Engineering Group:

> *Ecological Engineering Group…[combines] conventional engineering, innovative ecological design, and permitting know-how. Our work ranges from residential homes to multi-building sites, many with environmentally sensitive or restricted conditions.*

Company Website: http://www.ecological-engineering.com/