2017

The Application of Floating Treatment Wetlands for Stabilization Pond Enhancement in Southern Alberta

Banmann, Christopher

UNIVERSITY OF CALGARY

The Application of Floating Treatment Wetlands for Stabilization Pond Enhancement in Southern Alberta

by

Christopher Liam Banmann

A THESIS

SUBMITTED TO THE FACULTY OF GRADUATE STUDIES

IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE

DEGREE OF MASTERS OF SCIENCE

GRADUATE PROGRAM IN CIVIL ENGINEERING

CALGARY, ALBERTA

JANUARY, 2017

Abstract

Floating Treatment Wetlands (FTWs) are a means of stabilization pond enhancement by providing a floating platform for vegetation that sits atop a water body. To determine the impacts of FTWs, a pilot-scale stabilization pond system was constructed at the Carseland Sewage Lagoons in Southern Alberta, Canada. Treatment cells with FTWs (22% surface coverage) and control cells were exposed to low and high nutrient wastewater from May to October, in 2015 and 2016. Results indicate FTW cells do not outperform control cells (with the exception of ammonium) when treating low nutrient influent. However FTW cells significantly improved the removal efficiency of several pollutants including reactive and total phosphorus, ammonia, nitrate, total nitrogen and biochemical oxygen demand when treating high nutrient influent. The FTW also significantly reduced temperature, electrical conductivity, dissolved oxygen, and pH. Therefore FTWs are an ideal, passive treatment enhancement for stabilization ponds, particularly when treating high nutrient wastewater.

Acknowledgements

Many thanks to my academic supervisors, Dr. Angus Chu and Dr. Jianxun He, and my industrial supervisor Bernie Amell (from Source2Source Inc.), for their support, guidance, and encouragement during this masters thesis. Also, thanks to Daniel Larson and Mirsad Berbic for providing technical support in the laboratory that was indispensible to my analysis.

Specific individuals who provided additional materials, labour, and guidance for this project were: Sadegh Hosseini (University of Calgary), Cecilia Chung, Wendell Koning and Rob Wolfe (Alberta Environment and Parks), Dave Churchill and his staff (Wheatland County), Erik Vandist (Vita Water Technologies Ltd.), Ray Shaw (Knutson and Shaw Growers), and Anton Skorobogatov (Source2Source Inc.). Financial support for this project was provided by Environment Canada (through the Lake Winnipeg Basin Stewardship Fund), Source2Source, and the Natural Sciences and Engineering Research Council of Canada (NSERC).

Finally, thanks to my family and friends for their support through the previous two years. Particularly my girlfriend Caitlyn, who endured (among other things) a car littered with mud and field equipment, camping coolers converted to sewage sample carriers, and long hours listening to complaints about malfunctioning pumps.

Table of Contents

List of Tables

List of Figures

List of Abbreviations

BOD – Biochemical Oxygen Demand

COD – Chemical Oxygen Demand

CFU – Colony Forming Units

DO – Dissolved Oxygen

HSSF – Horizontal Sub-Surface Flow

HRT – Hydraulic Retention Time

F-C – Facultative treatment Control cell

F-FTW – Facultative treatment Floating Treatment Wetland cell

FTW – Floating Treatment Wetland

FWS – Free-Water Surface

RP – Reactive Phosphorus

S1-C – Stage 1 Control (Two-Stage Treatment)

S1-FTW – Stage 1 Floating Treatment Wetland cell (Two-Stage Treatment)

S2-C – Stage 2 Control (Two-Stage Treatment)

S2-FTW – Stage 2 Floating Treatment Wetland cell (Two-Stage Treatment)

S-C – Storage treatment Control cell

S-FTW – Storage treatment Floating Treatment Wetland cell

SP – Stabilization Pond

TP – Total Phosphorus

TN – Total Nitrogen

TSS – Total Suspended Solids

VF – Vertical Flow

Chapter 1: INTRODUCTION

1.1 Background

Approximately 40% of the world's population lacks access to basic sanitation, of which 75% reside in rural regions. Lack of sanitation increases risk of diseases such as cholera, trachoma, and hookworm, and increases stress on the ecosystem (WHO, 2002). Nutrient contamination from untreated or insufficiently treated sewage, is a major contributor to eutrophication, which threatens the health of freshwater ecosystems in both developed and developing nations (Anderson et al, 2002). Nutrient and contaminant loading can originate from a variety of municipal, industrial and agricultural sources, and the need for more effective wastewater treatment is increasing as the population grows.

Though the general world population continues to urbanize, a significant minority continue to live in rural and remote regions. Wastewater treatment is a major capital cost, as centralized wastewater treatment systems are often expensive to construct and require skilled personnel to operate. In developing nations centralized treatment facilities often fail due to lack of maintenance or improper operation (Boller, 1997).

In small and isolated settlements with low population densities, decentralized treatment systems are superior to centralized treatment systems. Comparatively, decentralized treatment systems that are located close to the wastewater source are more cost effective since little infrastructure is devoted to wastewater collection. Oftentimes, wastewater can be treated, disposed or reused on site, which consequently reduces environmental impact. When designing decentralized treatment systems, it is crucial to

understand the operational environment, and the receiving environment. The treatment efficiency of individual systems will vary considerably with climate, population, local regulations, and ecological sensitivity (Boller, 1997).

Two treatment technologies well suited to decentralized treatment are waste stabilization ponds (SPs) (also referred to as sewage lagoons) and treatment wetlands. SPs have long been applied to wastewater treatment since they are relatively simple to construct and operate. They primarily rely on biological treatment provided by phytoplankton, bacteria and aquatic macrophytes, combined with a long hydraulic retention time (HRT) (greater than 10-days in most systems). SPs can be enhanced with aeration (creating an aerated lagoon), however a majority of systems are passive in order to reduce operational costs (Stanley & Smith, 1992).

Treatment wetlands have been commonly applied to wastewater treatment in North America ever since the 1970's. In addition to physical processes such as precipitation and sedimentation, macrophytes and their associated microorganisms are responsible for contaminant removal (Fisher & Acreman, 2004; Vymazal, 2007). They can be operated as either free water surface (FWS) wetlands in which emergent macrophytes are grown in shallow water (typical of natural wetlands), or horizontal sub-surface flow (HSSF) and vertical flow (VF) wetlands in which macrophytes are grown in a porous medium and wastewater is directed through the root system (Kadlec & Wallace, 2008).

When treating similar volumes of water, treatment wetlands are generally considered to be superior to SPs in terms of treatment efficiency (Baldizon et al, 2002; Garcia et al, 2008), although in some situations SPs perform better due to increased aeration (Liu et al, 2014; Munoz et al, 2016). A major advantage of SPs, and likely the primary reason for

their popularity compared to treatment wetlands, is they are less land intensive. Emergent macrophytes require shallow water, (less than 0.5m depth), compared with 1-3m in SPs, and so consequently a treatment wetland treating the same volume of wastewater as a SP requires between 40-400% more area, depending on influent water quality and effluent water quality regulations (Mara, 2006).

Floating Treatment Wetlands (FTWs) may be an ideal means of combining the biological treatment processes of FWS, HSSF and VF wetlands, with the HRT and depth of a SP. FTWs use a floating platform that remains on the surface of water bodies, and supports emergent macrophytes. Similar to a SP, a basin containing a FTW would provide sections of open water to allow efficient oxygen transfer from the atmosphere, and shallow water supports the growth of phytoplankton and aquatic macrophytes. FTWs provide additional treatment by diverting contaminants into macrophyte biomass, and providing a large surface area of ideal microbial habitat along the macrophyte roots that hang suspended in the water column.

Studies using FTWs have investigated stormwater, sewage, and industrial effluent treatment (Borne et al, 2013; Li et al, 2012; Van de Moortel et al, 2010), in addition to eutrophic lake and stream remediation (De Stefani et al, 2011; Song et al, 2009). FTWs appear to have a significant impact on nutrient reduction, such as nitrogen, phosphorus, and carbon, in addition to suspended solids and metals (Headley & Tanner, 2012). In sewage applications, FTWs increase the concentration of ammonifying, nitrifying and denitrifying bacteria (Zhou et al, 2012). Furthermore, naturally occurring floating wetlands have been observed to reduce fecal coliform bacteria (Kansiime & van Bruggen, 2001).

While the treatment potential of FTWs has been investigated in previous studies, most of the research methodology has been focused on smaller scale mesocosms (<10m^3 volume) with a HRT less than 20days. The only pilot-scale sewage treatment study (Wu et al, 2006) added aeration to the system, and had 100% coverage by FTWs. This thesis investigates the application of FTWs as retrofits to an existing SP, using a pilot-scale treatment system with a depth equal to that of a SP. Using only 20% coverage and passive aeration only, this study determined if FTWs are an acceptable and effective means of enhancing SP treatment under real world operating conditions.

1.2 Research Objectives and Hypotheses

FTWs are a promising passive enhancement for SPs in many regions. However, the application of FTWs in Alberta is reliant on performance in Alberta's climate as FTW treatment efficiency is likely site and climate-specific. It was hypothesized that, in Alberta's climate, the FTW would increase treatment efficiency relative to the control and would have a greater impact commensurate with incoming contaminant concentration. It was also hypothesized, that macrophytes would provide a minor sink for phosphorus within the treatment system.

Therefore, the purpose of this thesis was to measure the impact of FTWs and determine if they are a suitable additional to SPs in Alberta's climate. The primary objective was to measure the removal efficiency of various pollutants by FTWs relative to a control. The role of macrophytes in removing phosphorus was also assessed. The experimental cells were exposed to different levels of nutrient loading to determine whether the FTW was capable of enhancing the treatment of various pollutants, and more specifically, determined where a FTW would best be located in a full-scale system based

on nutrient concentration (i.e. the primary cell, the facultative lagoon, or the storage lagoon). The impact of FTWs on basic physicochemical parameters was weekly and continuously monitored to determine the expected impact on a full-scale lagoon.

The secondary objective was to observe the growth and winter survival of macrophytes to ensure that the species selected were suitable for the use in a cold-continental climate. During the experimental period, the macrophytes and the FTW platforms experienced sub-zero conditions and complete ice-coverage in winter.

The results obtained from the pilot-scale facility in this thesis could be translated into a full-scale system, since the experimental cells were designed to mimic the conditions of a full-scale lagoon.

1.3 Brief Methodology

Three treatments were completed over a period of two years, with each treatment designed to mimic a different component of a full-scale SP system. Two sea-containers were installed at the Carseland SPs in Southern Alberta. The sea-containers were divided each into two cells, with one designated the FTW cell (designed to mimic a SP with 20% coverage by a FTW) and the other the control (designed to mimic an unenhanced SP). In 2015, one sea-container was designated as the storage treatment, and was filled with low nutrient wastewater form the storage lagoon; it was operated as a static system (no additional wastewater added during the study period). The second sea-container was designated as the facultative treatment, and was filled with partially treated wastewater from the facultative lagoon; it was operated as a continuously fed system with a 50day HRT. In 2016, both sea-containers were merged to create a two-stage system designated

the two-stage treatment, which was continuously fed with high-nutrient wastewater from the effluent of the primary cells (located onsite as part of the Carseland SPs).

Treatments in both years were operated from June to October/November and water samples were collected twice per week. Wastewater samples were analyzed for NH_4^+, NO_3^-, total nitrogen (TN), reactive phosphorus (RP), total phosphorus (TP), total suspended solids (TSS), turbidity, biochemical oxygen demand (BOD) and fecal coliform bacteria (2016 only). In addition, basic physicochemical parameters (temperature, electrical conductivity (EC), dissolved oxygen (DO), and pH) were measured in situ, twice per week during both field seasons, and continuously for a period of three weeks in late-summer 2016.

Macrophyte and FTW survival was visually monitored for the winter of 2015/2016. At the end of the 2016 season, above-mat biomass was sampled and dry-biomass was measured. Phosphorus concentration in the macrophyte shoots was analyzed, to determine the percentage of the total phosphorus load that was diverted into the biomass.

1.4 Overview of Chapters

This thesis consists of six chapters, including this introductory chapter. Chapter 2 is a review of the literature on SPs, treatment wetlands, and research to date on FTWs applied to wastewater treatment. Chapter 3 describes the methods, materials, and experimental design used to conduct the study. Results and observations obtained from two field seasons are presented is Chapter 4, and the discussion of the results is given in Chapter 5. Finally, Chapter 6 presents the conclusions and future recommendations.

Chapter 2: LITERATURE REVIEW

2.1 Stabilization Ponds

SPs (also referred to as sewage lagoons) are a common treatment system used in both centralized and decentralized treatment systems worldwide. They have been widely adopted since they are cost effective, simple to operate and maintain, and have a high buffering capacity that enables them to deal with fluctuating influent concentration. SPs are operationally diverse and can be aerobic, anaerobic, or facultative. Their hydraulic retention times (HRTs) can range from 10 to greater than 100 days. The dominant treatment processes within lagoons include 1) sedimentation of organic and inorganic solids, 2) anaerobic digestion of settled organic solids, 3) aerobic stabilization of organic waste, 4) nutrient removal, and 5) natural disinfection (Stanley & Smith, 1992).

Simple SPs systems may contain only one large pond in which all treatment occurs, but most systems contain multiple cells. Modern SPs can be divided into three cells, each with a different role in the treatment process. The primary, or settling cell, is where sedimentation occurs and usually has a HRT of 1-10 days. The secondary, or stabilization cell, is primarily designed to remove dissolved nutrients and reduce BOD. This stage dominates the treatment process, and can have a HRT of several months. The third cell is a polishing lagoon and is occasionally required to further reduce dissolved nutrients and to meet release guidelines. Oftentimes, the third stage is not a lagoon, and in many decentralized systems a treatment wetland or a sand filter is used for final polishing. In cold climates like those seen in Alberta, Canada, the third cell functions as a storage lagoon in which effluent can be held until release once or twice a year, and thus allows

the system to compensate for the reduced treatment that occurs during the winter months (Price et al, 1995).

SPs that are operated without any mechanical treatment are commonly operated as facultative lagoons. In facultative lagoons the depth is less than 1.5m, which allows an aerobic upper layer (created through oxygen transfer from the atmosphere) to lay overtop a lower anaerobic layer. This allows a multitude of necessary chemical and biological process to occur simultaneously. Aerated lagoons are used to increase oxygen concentration, and allow rapid degradation of organics and increased rate of aerobic reactions.

While SPs can be easily improved by adding mechanical treatment, the operational costs of such treatment systems can nullify the passive nature of SPs, which is what makes them appealing to operators in the first place. Any enhancement to a SP would ideally require little to no power and maintenance, and rely primarily on biological treatment

2.2 Treatment Wetlands

Treatment wetlands have been widely used since the 1970's to treat municipal and industrial wastewater and stormwater (Kadlec & Wallace, 2008). Many studies have demonstrated that treatment wetlands are effective at reducing contaminant concentrations particularly nitrogen, phosphorus, suspended solids, and BOD (Fisher & Acreman, 2004; Vymazal, 2007). They have also been widely used in wastewater treatment systems (Martinez-Guerra et al, 2015). The design of treatment wetlands varies significantly depending on the application. Most are designed to take advantage of naturally occurring characteristics of a wetland ecosystem, while also considering

hydrologic parameters such as HRT and hydraulic loading rate. Conventional treatment wetlands can be divided into three categories: Free Water Surface (FWS), Horizontal Subsurface Flow (HSSF), and Vertical Flow (VF) (Kadlec & Wallace, 2009).

2.2.1 Free Water Surface

FWS wetlands have an open water surface, and are most similar in appearance to naturally occurring wetlands, making them aesthetically pleasing. A FWS wetland has one or more large basins, ranging from 10cm to 1m in depth. Usually designers avoid depths greater than 1m since it negates much of the benefits of wetland processes, and is unable to support emergent wetland vegetation. Treatment processes within FWS wetlands include sedimentation, filtration (through macrophyte stems and roots), oxidation/reduction (in the water column and the soil), adsorption, and precipitation. All these processes occur in SPs, but the wetland ecosystem enhances or concentrates these processes (Gottschall et al, 2007). They are cost effective when compared with other treatment technologies, and have low operating costs (U.S. EPA, 2000).

FWS wetlands experience a noticeable decrease in treatment efficiency in the winter months. They are prone to nutrient release in the fall season when macrophytes senesce and biomass degrades within the water (Kroger et al, 2007; Werker et al, 2002)

2.2.2 Horizontal Sub-Surface Flow and Vertical Flow

HSSF wetlands are often designed to treat primary effluent. Water flows through a bed of gravel, in which the macrophyte roots are embedded. This means HSSF wetlands are particularly effective at removing particulate matter, and the roots assist in removing dissolved nutrients. VF wetlands are designed for pulse loading applications, where water is filtered through a layer of sand and gravel, interspersed with roots from emergent

macrophytes. Water is drawn downwards, and exits near the bottom of the wetland. The system can only handle continuous flow if applied at a low hydraulic loading rate, so as not to completely flood the wetland or create anaerobic conditions. Oftentimes a layer of water is maintained near the bottom of the wetland, to create anaerobic conditions that allow for crucial anaerobic process such as denitrification (Lemon, Bis, Rozema, & Smith, 1996). Both HSSF and VF wetlands are better suited to cold climates since the gravel layer is insulated, resulting in better efficiency in the winter (Werker et al, 2002).

2.3 Floating Treatment Wetlands

Floating Treatment Wetlands (FTWs) are at their most basic, a means of supporting macrophytes on the surface of a water body. As yet, there is no consensus on the correct name for the system, as in the literature they have also been referred to as Artificial Macrophyte Filters, Rafted Reed Beds, Floating Wetland Systems, Vegetated Rafts/Mats, Floating Islands, Artificial Floating Wetlands, and Floating Emergent Macrophyte Treatment Wetlands (FEMTW) (Headley & Tanner, 2012). FEMTW might be the most correct variant, since the term macrophyte includes fully aquatic vegetation, or naturally floating species like duckweed (subfamily *Lemnoideae*) and water hyacinth (*Eichornia crassipes*) that need to be distinguished from emergent vegetation in terms of treatment applications. However, the term FTW will be used for the remainder of this thesis, to better encompass the wide variety of FTW systems that exist in the literature.

FWS wetlands are the most common system of treatment wetland, however a limitation of macrophytes rooted in soil is that they cannot be inundated with depths greater then 30-50cm. This can stress or kill the macrophytes if flood conditions are allowed to persist (Greenway et al, 2007). This limitation requires FWS wetlands to be

large enough to attenuate and absorb peak flows, which can be realized by the construction of bypass structures that reduce treatment.

FTWs remain on the surface of water body and thus are independent of water depth (Figure 2.1). This feature makes them appealing for stormwater and sewage treatment systems with highly variable loading and depth. Macrophyte species are grown hydroponically, which allows roots to extend into the water column. If the depth of the water body is substantial, there is no physical contact between the roots and the soil. Theoretically, these features make FTWs better suited to removing dissolved contaminants than conventional treatment wetlands, as they are completely reliant upon the water column for nutrition and not the soil (Headley & Tanner, 2012).

Classifying a FTW within the existing wetland nomenclature (Kadlec & Wallace, 2008) can be difficult, as they contain several elements from different systems. FTWs are placed in water bodies that are deep enough to be considered ponds or lagoons (>1.5m), and may contain large sections of open water. However, ponds rely on phytoplankton and submerged macrophytes for treatment, rather than emergent or terrestrial macrophytes. FWS wetlands are reliant on emergent macrophytes, and are designed to accommodate surface flow, which is similar to FTWs. However, FWS wetlands are shallow (<0.6m) and flow passes the shoots, but not the roots that are embedded in soil (Headley & Tanner, 2012).

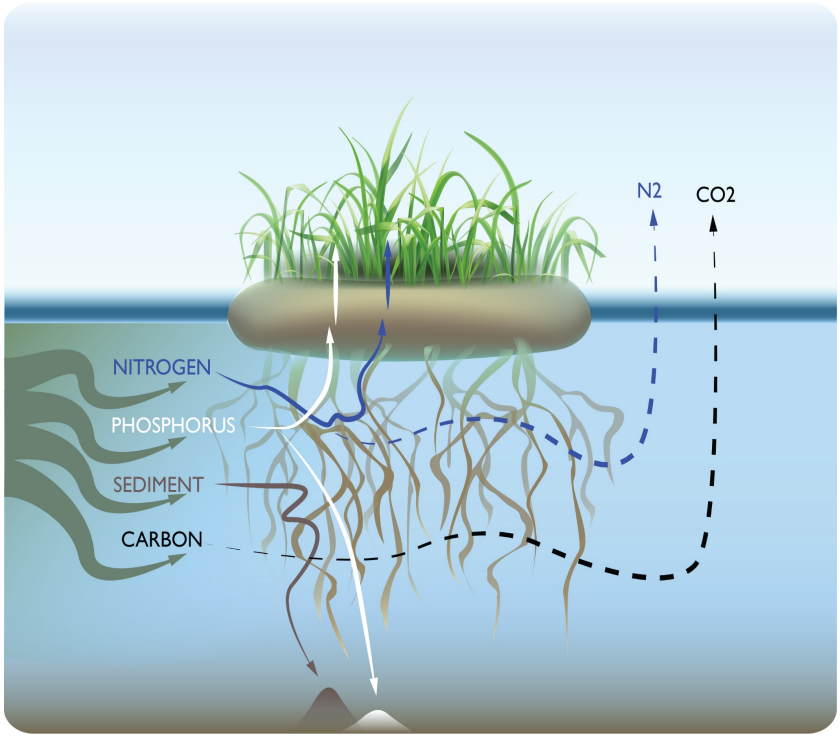

Figure 2.1 FTW diagram showing the root systems hanging beneath the floating platform. The various removal pathways for nitrogen, phosphorus, sediment, and carbon represent removal by macrophyte uptake and sedimentation (solid lines) or conversion to a gaseous form (dotted line). (Banmann et al, 2016)

Directing wastewater past the rhizosphere of the root system is a commonality between FTWs and HSSF/VF wetlands. The difference is, HSSF and VF wetlands force water through a substrate embedded with macrophyte roots and thus hydraulic short-circuiting may occur as macrophytes mature and root density increases. Over time, sediment and organic build-up may serve to block access to the rhizosphere (Breen & Chick, 1995). FTW root systems could also hinder hydraulic flow, but a lack of tight pore spaces and minimal sediment/organic deposition on roots could avoid significant short-circuiting.

The number of studies on FTWs has been steadily increasing since the early 2000's, as there is interest in using treatment wetlands that can bypass some of the limitations of

existing conventional treatment wetlands. Since the 1980's, FTWs have been investigated for applications in water treatment, bank stabilization, habitat creation, and beautification (Hoeger, 1988). In regards to water quality, the studies conducted to date for specific applications have been:

- **Stormwater** (Borne, 2014; Borne et al, 2014a; Borne et al, 2014b; Borne et al, 2013; Chang et al, 2012; Chua et al, 2012; Kerr-Upal et al, 2000; Ladislas et al, 2013; Nichols et al, 2016; Revitt et al, 1997; Tanner & Headley, 2011; Wang et al, 2014; Winston et al, 2013)
- **Sewage** (Ash & Truong, 2004; Ayaz & Saygin, 1996; Kyambadde et al, 2005; Saeed et al, 2014; Todd et al, 2003; Van de Moortel et al, 2010; Wu et al, 2006)
- **Swine effluent** (Hubbard et al, 2004; Xian et al, 2010)
- **Oil Refinery Wastewater** (Li et al, 2012)
- **Eutrophic lake water/reservoir remediation** (Garbett, 2005; Hu et al, 2010; Li et al, 2007; Li et al, 2009; Song et al, 2009; Wu et al, 2006)
- **Polluted stream remediation** (De Stefani et al, 2011; Sun et al, 2009; Zhou et al, 2012; Zhu et al, 2011)

In addition, there are many bench-scale/mesocosm studies investigating macrophyte growth, nutrient uptake, and nutrient removal from simulated wastewater (Bartucca et al, 2016; Billore et al, 2009; Chang et al, 2012; Faulwetter et al, 2011; Tanner & Headley, 2011; Wang & Sample, 2014; Wang et al, 2012; Wen & Recknagel, 2002; White & Cousins, 2013; Yang et al, 2008; Zhou & Wang, 2010).

Of those studies investigating the impact of FTWs on municipal sewage treatment, only one is considered pilot-scale (>10m^3 volume) (Wu et al, 2006). While previous

studies support the hypothesis that FTWs significant reduce nutrient concentrations in sewage effluent, none have explicitly focused on the application of FTWs for SP enhancement. The depth of the treatment cells in this study was specifically chosen so results could be scaled up to a larger system, should the FTW prove promising. No previous study has documented the use of FTWs in Canada over consecutive summers and observed successful overwintering in exposed conditions.

2.3.1 FTW Construction

Naturally occurring floating wetlands are composed of macrophytes growing on a buoyant mat of roots and organic matter. There are two broad categories of natural floating wetlands: deep-mats composed from plant roots dominated by larger plant taxa and large deposits of organic matter, and shallow-mats with little to no organic matter dominated by small plant taxa. They can range in size from <0.1hectares to >500 hectares (Mallison et al, 2001). Macrophytes with fibrous root systems are best suited to floating wetlands, as they form a dense mat that creates an ideal environment to trap decaying organics and associated gases (Sasser et al, 1996).

Constructed FTWs are designed to emulate these natural floating wetlands, but also to avoid disadvantages such as inconsistent buoyancy and slow establishment. Buoyancy in naturally occurring floating wetlands can be highly seasonal (most buoyant in the summer months when metabolic gas production is highest), and a floating wetland may fully submerge in fall and winter, only to re-emerge in the spring. Deep mat wetlands are more stable but may take decades to establish, making them impractical for treatment applications (Sasser et al, 1996). To avoid this, most FTW applications involve a constructed method of buoyancy to support the macrophytes.

Amongst the FTW studies to date, a variety of different support systems have been developed. Lines can be suspended 10cm above a treatment pond to support planting pots that allow macrophytes to grow on the water surface (Kyambadde et al, 2005), or a buoyant growing medium can be used that sits on the surface of water body (Hu et al, 2010). The latter is preferred since it allows the FTW to remain on the surface regardless of changes in water depth. Simple, cost effective systems may be used in a lab setting, but field application require more robust systems that can resist wave/wind action and resident waterfowl/mammals.

Rigid FTWs are built from a buoyant frame that encompasses a fibrous planting medium. The system used in this thesis study were BioHaven® floating islands (Floating-Island-International, 2016), which are comprised of tangled polyester fibre injected with patches of buoyant polyurethane foam. The foam maintains structural integrity, while the polyester fibre creates a growing medium for the macrophytes and provides additional surface area for bacterial growth. The same system has been used in previous studies investigating FTW performance in stormwater ponds, though never in a wastewater setting (Borne, 2014; Borne et al, 2014b; Borne et al, 2013; Nichols et al, 2016).

After construction a FTW must be placed where it will have the greatest impact on wastewater treatment. An advantage of FTWs is they are easily relocated, especially if built from independent platforms (Headley & Tanner, 2012). To effectively treat large water bodies with a small area of FTWs, an impermeable barrier or baffle system can be used to redirect flow underneath the wetland and through the root system (Nichols et al, 2016).

2.3.2 The Role of Macrophytes

In most FTWs, emergent wetland macrophytes are preferable to terrestrial macrophytes. Although, almost any terrestrial macrophyte species can be used in a FTW as long as the root-crown is kept above the water and dissolved oxygen concentration remains high, many terrestrial species cannot tolerate wastewater conditions. This could be caused by an intolerance of high contaminant concentrations (such as ammonia or metals), hostile physicochemical conditions, or vulnerability to parasitic/microbial attack. Species not adapted to wetland conditions should only be used in applications with relatively low contaminant loads and high dissolved oxygen, such as lake or stream remediation (Bartucca et al, 2016; De Stefani et al, 2011). Submerged vegetation often does not perform well in wastewater, since it requires dissolved oxygen that is often limited in wastewater with high BOD.

There are unique adaptations present in emergent macrophyte species that make them ideally suited to a FTW. Most emergent species contain tissue in the shoots (and in some species in the roots) called the aerenchyma (Figure 2.2), with the primary purpose of transporting oxygen from the shoots to the roots (Schussler & Longstreth, 1996). Wetland soils are characteristically hypoxic, thus emergent macrophytes need to be able to meet the oxygen demands of the roots and rhizosphere. The presence of the aerenchyma makes macrophytes ideal for treating wastewater streams where dissolved oxygen is often very low, and removes the need for additional aeration. Transport of oxygen to the rhizosphere also prevents the production of toxic compounds that are associated with anaerobic respiration, such as hydrogen sulphide (H_2S) (Neori et al,

2000). The aerenchyma is extremely porous and filled with air, which can add additional buoyancy to a FTW (Schussler & Longstreth, 1996).

Figure 2.2 Cross-sections of dried *Scirpus microcarpus* shoots clearly showing the porous aerenchyma tissue in the centre of each stem.

The literature to date contains a wide variety of macrophyte species used in FTWs. This is due to geographical differences, as most FTW studies select native emergent vegetation that can tolerate the climactic conditions of the study area. In many treatment wetlands the percentage of inorganic nutrients removed by the macrophytes (as opposed to bacteria and abiotic processes) is relatively small (Krzciuk & Galuszka, 2015). For example, only 5% of the phosphorus load in municipal wastewater is removed and incorporated into macrophyte biomass (Kim & Geary, 2001). Similarly for nitrogen, macrophytes only remove 5-10% of the nitrogen load. Considering this, nutrient uptake by macrophytes may not always be an important design factor, and in Europe,

macrophyte harvesting in treatment wetlands is typically not even recommended (Stottmeister et al, 2003).

2.3.3 The Role of Microbial Communities and the Rhizosphere

Microorganisms are responsible for the majority of contaminant removal within treatment wetlands, and in general, approximately 80% of nutrients are removed by microbial processes in treatment wetlands (Tao et al, 2007). Macrophytes provide an ideal habitat for bacteria by providing high surface area along the shoots (in a FWS system) and the roots (in a HSSF, VF, and FTW system) (Kadlec & Wallace, 2008).

Roots provide an ideal environment for microbial growth (Figure 2.3). Macrophytes provide enormous surface area through the proliferation of fine structures and microscopic root hairs, up to 4.6–9.3 m^2 (of root structure) per m^2 of FTW. However, surface area is not the only requirement for bacteria growth, as floating platforms with added artificial roots (composed of polyester yarn) were not found to increase treatment compared with live roots, particularly in regard to phosphorus (Tanner & Headley, 2011). This indicates the release of bioactive compounds from live roots, or the manipulation of physicochemical conditions to create a more favourable environment for microbial establishment.

All macrophyte species retain a zone surrounding the root system affected by bioactive chemicals released from the roots, referred to as the rhizosphere. Due to the limitation of gas transfer into water, combined with a high BOD, wetland soils are often anoxic. In many cases the bacterial community is completely reliant on oxygen supplied from the roots of wetland macrophytes through the aerenchyma tissue (Neori et al, 2000). In anaerobic conditions, this can create an oasis of oxygen in an otherwise oxygen-

depleted environment, allowing aerobic reactions to take place. However, there is little evidence that wetland macrophyte species can significantly increase dissolved oxygen concentration in the surrounding water body, since most oxygen is consumed before exiting the rhizosphere (Bezbaruah & Zhang, 2005).

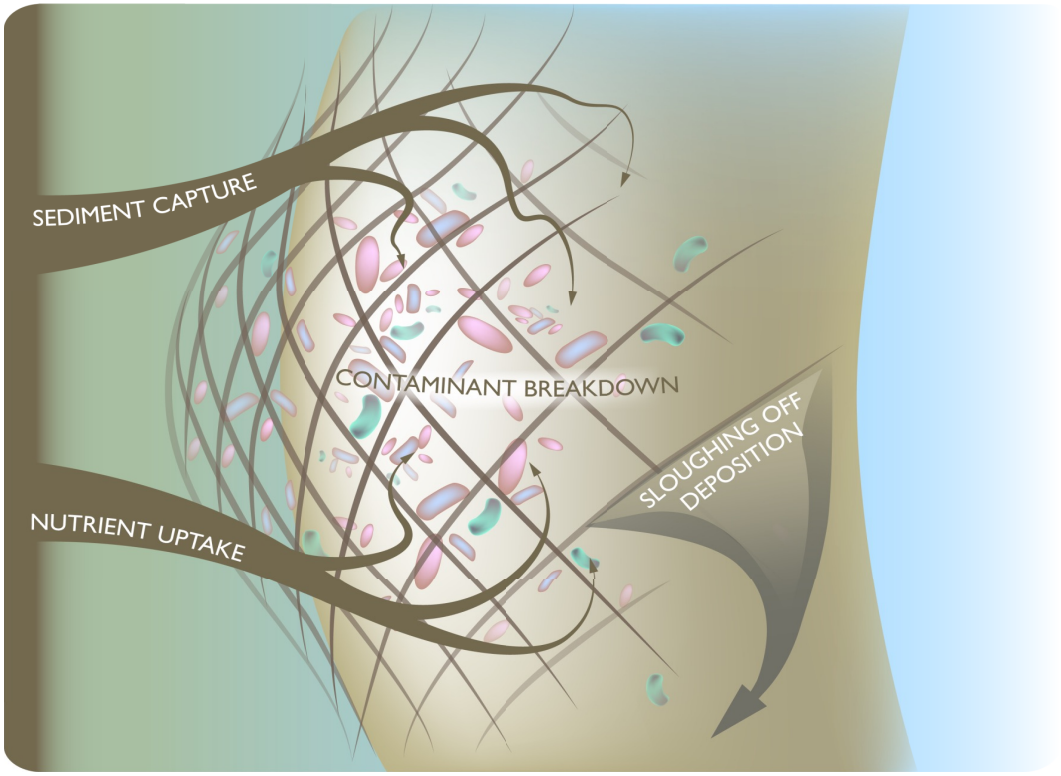

Figure 2.3 Diagram of bacterial growth within the rhizosphere. Sediment and nutrients are captured within the biofilm, and eventual sloughing off of bacterial cells will deposit nutrients into the benthic sediment. (Banmann et al, 2016)

Macrophytes also release polysaccharides into the rhizosphere that provide a carbon source for host bacteria. This carbon source can be extremely important in carbon-depleted conditions (such as treating stormwater or low-nutrient streams), and can allow a dense bacterial colony to function where it otherwise would be carbon-limited (Truu et

al, 2009). The formation and eventual degradation of microbial biofilms is an extremely important nutrient removal process in FTWs and wastewater treatment in general. In a treatment wetland/SP setting, biofilms will eventually slough off the root surface, and settle at the bottom of the water body and ideally sequester any nutrients contained within the bacteria into the sludge (Andersson et al, 2008). This treatment process provides much higher levels of nutrient removal than macrophyte uptake, especially when removing key nutrients like phosphorus that have no significant gaseous form (Kadlec & Wallace, 2008; Tanner & Headley, 2011).

2.3.4 Physical and Physicochemical Effects on the Water Column

The addition of a FTW has numerous physical impacts on the water body as well as the chemistry of the water column, even at a relatively small coverage of 1% or 5% of the surface area of a water body. The most obvious effect is the shading provided by the FTW itself and the consequent impact on water temperature. The FTW platform in most applications is opaque and the mature macrophytes absorb or reflect most incoming radiation before it enters the water. This reduction in radiation reduces water temperature and the effect is greatest at high ambient temperatures or during midday. A seasonal study on FTW treating wastewater by Van de Moortel et al. (2010) found that temperature in the FTW cell was restrained when air temperature rose above 15°C. In addition, several studies (particular those performed in warm climates) have found a reduction in water temperature by FTWs (Wang et al, 2012; White & Cousins, 2013; Zhou & Wang, 2010)

Various studies have also observed that FTWs neutralize pH in the water column beneath the roots or within the surrounding water body (Borne, 2014; Borne et al, 2014b;

Van de Moortel et al, 2010; White & Cousins, 2013; Xian et al, 2010; Zhou & Wang, 2010; Zhou et al, 2012). White & Cousins (2013) found that pH was reduced from 8.6 to 6.2 in a trough system with 100% FTW coverage and a 2-day HRT. The causes of the reduction in pH are the results of either organic acid released from the roots (Mucha et al, 2005; Neori et al, 2000), chemical reactions that occur in the enhanced treatment zone beneath the FTW (Headley & Tanner, 2012), or the inhibition of algae and submerged vegetation that can increase pH through carbonate consumption during photosynthesis (Xian et al, 2010). The impacts of low/neutral pH on the treatment performance of FTWs are numerous, and will be discussed in the sections below in regard to specific nutrients and contaminants.

Even though emergent macrophytes release oxygen through the root tissue (Bezbaruah & Zhang, 2005), the majority of studies observed that FTWs lower dissolved oxygen levels in the surrounding water (Van de Moortel et al, 2010; White & Cousins, 2013; Yang et al, 2008; Zhou et al, 2012). The release of organic acids, polysaccharides and other carbonaceous compounds increases BOD in the rhizosphere and surrounding water, which consequently increases oxygen consumption (Zhou et al, 2012). In addition, the presence of a FTW reduces the surface area for oxygen transfer from air to water, which is the primary pathway by which oxygen enters water in non-aerated sewage treatment systems and treatment wetlands (Kadlec & Wallace, 2008). Oxygen consumption may reduce aerobic reactions in the water body as a whole, but oxygen release from the roots creates a higher redox potential and allows aerobic reactions to continue even if anaerobic conditions dominate in the rest of the pond. Yang et al. (2008) concluded that even with dissolved oxygen concentrations below 0.7mg L^{-1}, nitrification

(an aerobic reaction) and ammonia reduction was still progressing in a system with 100% FTW coverage, which could indicate oxygen release from the rhizosphere.

2.3.5 Treatment: Suspended Solids

The removal of suspended solids is an important function of natural and treatment wetlands. In FWS wetlands, low water velocity combined with the presence of macrophyte shoots and litter provides an ideal environment for the settling and interception of solids. Many pollutants, such as metals and organic compounds, are associated with suspended solids as they adsorb strongly to particulates. In conventional wetlands, suspended solids can be problematic as they contribute to decreased function over time within the wetlands. In FWS wetlands, the settling of solids is ascribed to a gradual increase in the elevation of the bottom of the wetland. This can gradually decrease HRT as FWS wetlands are easily affected by even small changes in depth (Kadlec, 2009).

Compared to conventional treatment wetlands, FTWs seem ideally suited to solids removal. Settling is driven by the combination of reduced flow velocity (as a result of the FTW basin) and the physical filtering and entrapment of particles within the hanging root-biofilm network. FTWs prevent particle resuspension by suppressing wind action on the surface (Headley & Tanner, 2012). Additionally the FTW acts as a barrier to light and prevents the growth of algae that acts as an endogenous source of suspended solids within wetlands. Karnchanawong & Sanjitt (1995) found that facultative lagoon mesocosms (no FTW or surface area coverage) consistently caused an increase in TSS concentrations in primary-treated sewage due to algal growth; Meanwhile parallel FTW mesocosms consistently lowered TSS.

The root network is effective at removing particulates that are not typically removed in ponds, such as fine suspended clay particles with slow settling velocities. Tanner and Headley (2011) found as much as 2-3 times greater removal of fine clay by FTWs versus open water controls.

2.3.6 Treatment: Carbon

Wetlands are a significant source of carbon, due to the proliferation of macrophytes, bacteria, fungi, and animal life which all release carbon when they decay. In addition, treatment wetlands often receive large inputs of carbon, particularly when treating municipal sewage. BOD is the most common measure of carbonaceous compounds within sewage. It is an indirect measure to determine the oxygen consumption by bacteria within a water sample over a given time period (typically five days). Oxygen consumption is proportional to the concentration of biochemically degradable carbon sources. Nitrogenous compounds can also be oxidized (such as ammonium), so in some cases carbonaceous biochemical oxygen demand (CBOD) is used to denote when a BOD test using a nitrogen inhibitor has been performed (Sawyer et al, 1994). Chemical Oxygen Demand (COD) is a more inclusive measure of carbon and organic matter. It uses a powerful oxidant (typically potassium dichromate) that attacks a larger group of carbonaceous compounds than BOD, though it does not oxidize nitrogenous compounds (Kadlec & Wallace, 2008). In wastewater, the typical ratio of BOD_5 to COD is 0.4-0.8 (Stanley & Smith, 1992).

Though treatment wetlands produce significant amounts of biologically available carbon, they are efficient consumers of carbon and rapidly convert organic carbon into CO_2. Similar to conventional treatment wetlands, FTWs are likely to produce irreducible

background levels of organic carbon and BOD (Kadlec & Wallace, 2008). BOD removal efficiency is generally high even in systems with low HRT (White & Cousins, 2013) or in stream remediation (Billore et al, 2009; De Stefani et al, 2011). The additional surface area provided by the roots of FTWs, and the consequent increase in microbial activity that it provides, enhances COD and BOD reduction. Van de Moortel et al. (2010) observed a 53% COD reduction in FTW cells treating municipal wastewater, compared to 33% in the control for the same volume.

2.3.7 Treatment: Nitrogen

Nitrogen is often the primary contaminant of concern within wastewater, due to the impacts of ammonia toxicity and nitrate contamination in surface/ground waters. Fortunately, the existence of an atmospheric sink (meaning nitrogen can be volatilized or converted into nitrogen gas) makes complete nitrogen removal feasible within wastewater treatment systems (Sartoris et al, 2000). The majority of nitrogen within wastewater is derived from proteins and therefore is organic in origin. Many studies investigating FTWs have reported nitrogen removal as total nitrogen (TN, a combination of the various forms of nitrogen) or TKN (organic nitrogen and ammonia). Nitrogen is degraded to inorganic forms through three steps: ammonification, nitrification, and denitrification (Sawyer et al, 1994). FTWs appear to increase the concentration of ammonifying, nitrifying and denitrifying bacteria (Zhou et al, 2012).

Ammonification is the degradation of organic nitrogen (typically in the form of proteins) into ammonia. Ammonifying bacteria or fungi convert protein to ammonium (NH_4^+). In municipal sewage, most of the organic nitrogen is already converted to ammonia, and ammonifying bacteria play a limited role in sewage treatment. While

macrophytes use ammonium as a nitrogen source, it can be toxic if significant quantities are volatilized into ammonia ($NH_{3(g)}$) (Van Oostrom, 1995). Volatilization of ammonium to ammonia is greatly increased when the pH exceeds 8.5 (Sawyer et al, 1994).

Nitrification is the oxidation of ammonium to nitrate (NO_3^-). It is actually multiple reactions, in which ammonium is firstly oxidized to nitrite (NO_2^-) and then to NO_3^-.

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2O + 2H^+$$

The first step (ammonium to nitrite) is completed by bacterial species within the *Nitrosomonas* and *Nitrosococcus* genera. The second step (nitrite to nitrate) is completed by bacterial species within the *Nitrobacter* and *Nitrospira* genus (Wagner et al, 1996). The reaction is aerobic and requires 4.6mg L^{-1} of oxygen to nitrify 1mg L^{-1} of NH_4^+ (Sawyer et al, 1994). Because of the oxygen requirements, this step is often limited in SP systems without sufficient aeration, since dissolved oxygen concentrations within wastewater can be less than 1mg L^{-1} (Stanley & Smith, 1992). Nitrate is also useable as a nitrogen source for macrophytes (though less preferred than ammonium), and unlike ammonium will not volatilize, and is highly soluble (Van Oostrom, 1995). This process is extremely important in treating municipal sewage, since most nitrogen in wastewater can be found in the form of ammonia.

Denitrification is the process of nitrate reduction to nitrogen gas. This is an important step in wastewater treatment as nitrogen gas is inert, non-polluting, and allows a pathway for nitrogen removal from the system.

$$2NO_3^- + 10e^- + 12H^+ \rightarrow N_2 + 6H_2O$$

Heterotrophic bacteria from a variety of genera complete this process (Wagner et al, 2002). The process is negatively correlated with DO concentration, as nitrate in this reaction is used as an electron acceptor but it is inferior to oxygen. Therefore, this reaction only occurs in anoxic conditions. It requires a carbon source to oxidize, and denitrification will halt if carbon limited (Sawyer et al, 1994). Providing an anoxic zone with an ample carbon supply is critical in sewage treatment if denitrification is the goal.

Among the various forms of nitrogen, ammonium and nitrate are of specific interest since they are important for understanding nitrification and denitrification progresses within wetlands. For example, Ayaz & Saygin (1996) observed that ammonia decreased at a rate of 1.2-2.6g day^{-1} while nitrate increased at a similar rate in a FTW with 100% coverage by a floating mat treating secondary effluent from a wastewater treatment facility, which strongly indicates the occurrence of nitrification. Similar to SPs, it is challenge to create a system that can both nitrify and denitrify nitrogen, as a mixture of aerobic and anaerobic zones in a system is required to support both nitrification and denitrification. FWS wetlands are such systems (Kadlec & Wallace, 2008) and denitrification occurs in wetland soils, as it is a highly anoxic environment. This environment may be absent in a FTW, but anoxic zones can be created below the mat (Li et al, 2009).

Ammonia removal efficiency in FTW systems (as in SPs) is high, often upwards of 70% for both conventional treatment wetlands and FTWs (Fisher & Acreman, 2004; Headley & Tanner, 2012; Martinez-Guerra et al, 2015; Sartoris et al, 2000). Aeration can

be used to increase the rate of nitrification. Wang et al. (2012) increased ammonia removal in a FTW system from 35% to 71.3% with aeration and with a gas/water ratio of 10. In addition, temperature can also affect ammonia removal. At low temperatures nitrifying bacteria are less active and at high temperatures ammonia concentration can increase due to the release of ammonia from benthic and suspended sediment. Van de Moortel et al. (2010) found that the optimal temperature for ammonium removal is between 5-15°C. However, Wang et al. (2012) found that ammonium removal efficiency increased from 71.3% to 87.3% when temperatures exceeded 13°C.

In low-carbon systems (such as stormwater ponds and tertiary ponds in SPs) macrophyte roots appear to be far more effective for the growth of denitrifying bacteria than inert media such as plastic or gravel. Faulwetter et al. (2011) used carpet fibre and plastic media to remove nitrogen in an aerated system and found that ammonia and COD was reduced by 100%, but denitrification (and the growth of denitrifying bacteria) did not occur until molasses was added to the system as a carbon source. Macrophytes may transfer 30-70% of the carbon they produce to their root systems (Neori et al, 2000), and thus provide a consistent carbon source for denitrification. Borne et al. (2013) observed that a stormwater pond with a FTW, under which DO was low (mean of 0.9mg L^{-1}), reduced all forms of nitrogen except ammonia and increased organic matter because of compounds released from the roots. This created an ideal environment for denitrification and likely contributed to significantly reduced nitrate levels.

In controlled laboratory conditions, nitrogen uptake by some macrophytes in FTW can be up to 36mg m^{-2} day^{-1} (Chang et al, 2012). However, this can vary considerably among species, as Chua et al. (2012) recorded uptake rates of 16.2, 1.74, and 2.82 mg m^{-2}

day^{-1} for *Typha angustifolia*, *Chrysopohon zizanioides*, and *Polygonum barbatum* respectively. Sorption media is often added to FTW systems to increase treatment, but while effective for phosphorus, it was found to have little impact on nitrogen removal (Chang et al, 2012).

Sedimentation can be a significant removal mechanism for nitrogen. Borne et al. (2013) and Wu et al. (2006) applied a mass balance to determine the nitrogen remaining in a FTW treating stormwater, and found that a majority of the nitrogen ended up within the sediment rather than stored within the tissue of the macrophytes. Fu & LI (2011) found that 62.5% of the nitrogen was removed through the sedimentation of particulate nitrogen in a FTW with 100% coverage.

2.3.8 Treatment: Phosphorus

Phosphorus is a critical nutrient for biological growth, and is a limiting nutrient in many terrestrial and freshwater environments. In macrophytes and algae, the optimal ratio (termed the Redfield ratio) of carbon to nitrogen to phosphorus is 106:16:1 (Klausmeier et al, 2004). However in wastewater this ratio rarely occurs and phosphorus is far less limiting, with nitrogen to phosphorus ratios of 5:1, 3:1, or even 1:1 (Kadlec & Wallace, 2008). This can make acceptable phosphorus removal difficult, and minor inputs from wastewater into natural ecosystems can have negative impacts. Phosphorus release from municipal and agricultural wastewater is one of the primary contributors to eutrophication and harmful algae blooms (Pick & Lean, 1987).

Unlike nitrogen removal, there is no denitrification equivalent for phosphorus removal, meaning there is no significant gaseous form that phosphorus may be converted to that easily removes phosphorus from the system. Because of this limitation,

phosphorus entering a treatment wetland either remains in the water, or is diverted to macrophytes, microbiota, litter, or soil. Diversion into soil or sediment is the preferred method of sequestering phosphorus, as it is a long-term solution and the most stable (Kadlec & Wallace, 2008). Macrophyte biomass on FTW provides a sink for phosphorus accumulation, but macrophyte mortality and senescence in colder climates may produce a phosphorus source at certain times of the year (Werker et al, 2002).

Phosphorus inputs in treatment wetlands can exist in inorganic or organic forms (Figure 2.4). The generic term for inorganic phosphate ions is orthophosphate, denoted by PO_4-P. This is the most dominant form of dissolved phosphorus in the typical pH ranges found in treatment wetlands ($4<pH<9$). Soluble reactive phosphorus (SRP) includes orthophosphate and dissolved organic phosphorus, which is composed of a variety of organics that are all readily hydrolyzed by soil enzymes. Phosphorus bound to suspended particles is referred to as particulate phosphorus (PP). Total Phosphorus (TP) is a common parameter in wastewater quality and it encompasses orthophosphate, SRP, PP, and phosphorus that has been adsorbed onto organic biomass and metals such as Fe, Ca, Al, and Mg (Sawyer et al, 1994). Creating an environment that encourages the sequestration of phosphorus into macrophyte biomass or non-soluble compounds is an important criterion in treatment wetland design (Kadlec & Wallace, 2008). In general, abiotic phosphorus retention by phosphorus soils is regulated by physicochemical properties such as pH, redox potential, Fe, Al, and Ca content of soils, organic matter content, phosphorus loading, and ambient phosphorus content of soils (Lindstrom & White, 2011)

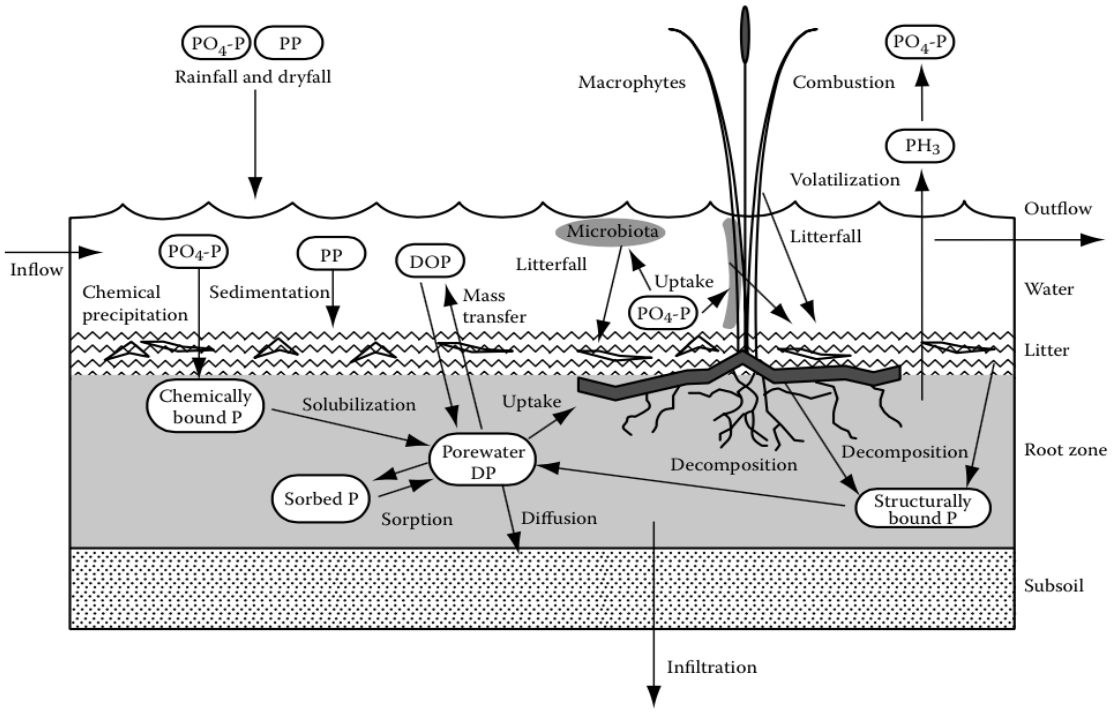

Figure 2.4 Phosphorus storages and transfers in the wetland environment. PO_4–P = orthophosphate; PP = particulate phosphorus; DP = dissolved phosphorus; PH_3 = phosphine. (Kadlec & Wallace, 2008)

Much of the phosphorus removal in treatment wetlands and SPs can be explained through the interception of particulate phosphorus. Borne (2014) found a 27% reduction in TP in the FTW pond relative to a control. However, since it was a stormwater pond, SRP levels were extremely low and there was no notable decrease in concentration in either pond. Therefore the increased treatment of the FTW was explained by the interception of PP within the suspended solids. Solids were significantly reduced as stormwater flowed through the root system of the FTW, and it was found most of the PP was diverted into the sludge. It was hypothesized by the authors that a more neutral pH and higher organic content beneath the FTW promoted phosphorus sorption onto particulate matter. The neutral pH and low redox conditions also did not induce phosphorus release. Van de Moortel et al. (2010) found that the temperature reducing

effect of the FTW (particularly when ambient temperature exceeded 15°C) was beneficial for phosphorus storage as high temperatures can increase phosphorus release from the sludge.

In general, phosphorus removal is positively correlated with higher HRT and FTW coverage, and negatively correlated with incoming concentration (Headley & Tanner, 2012). White & Cousins (2013), using a system with a 2-day HRT, found two differing efficiencies based on incoming concentration, a 75% and 45% reduction at 0.02mg L^{-1} and 0.12mg L^{-1} TP respectively. For municipal wastewater, Wu et al. (2006) saw a removal efficiency of 72.6% for TP , though aeration was present.

Due to the difficulty in removing phosphorus from a treatment system, macrophyte harvesting has been investigated as a removal method. In general, about 0.15-1.05% of macrophyte biomass is phosphorus, through this varies considerably between species. Regular harvesting of macrophytes in a FWS wetland (above ground biomass only) was found to remove no greater than 5% of the total phosphorus present in the system (Kim & Geary, 2001), indicating that macrophytes store a relatively small amount of the phosphorus in a treatment wetland. This finding has been supported in FTW research as well (Borne, 2014; Chang et al, 2012; Li et al, 2012; Wu et al, 2006). Opposed to these results (Zhou & Wang, 2010) found that *Oenanthe javanica* (Chinese celery) caused a 76% reduction in TP during the growth phase (63 days), but caused a 100% increase during the decay phase when the macrophytes senesced in the fall. Nutrient release in the fall is a common problem in conventional treatment wetlands in cold climates (Werker et al, 2002), and should be monitored in FTW systems as well. Most wetland species accumulate more phosphorus in the roots than the shoots, suggesting total macrophyte

harvesting is required in order to remove significant amounts of phosphorus (Ladislas et al, 2013; Wang et al, 2014; White & Cousins, 2013).

2.3.9 Treatment: Pathogens

Pathogens are a serious concern in wastewater treatment, present in a diversity of genera that include viruses, bacteria, fungi, protozoans, and helminths. Wastewater treatment facilities often include specific treatment process to deal with pathogens, such as chlorination, ozonation, and ultraviolet radiation. Wetlands have the potential to remove pathogens through natural die-off rates (especially with high HRTs) and hostile environmental conditions (Kadlec & Wallace, 2008).

No published studies have investigated pathogen removal with FTWs, though conventional treatment wetlands and natural floating wetlands have been investigated. Quinonez-Diaz et al. (2001) observed that even with HRTs of 1-2 days, a FWS treatment wetland showed a 99% reduction in pathogens when wastewater flowed through a treatment cell planted with bulrush (*Scirpus validus*). Compared to a control cell only containing sand, the wetland cell was capable of preventing virus penetration into the sand layer. Kansiime & van Bruggen (2001) observed that in a large, naturally occurring floating wetland that received regular wastewater inputs, there was a 1-2log reduction in fecal coliform bacteria. The authors concluded that the main removal mechanisms were entrapment within the macrophyte roots, sedimentation, and natural die off as wastewater flowed through the floating wetlands. It is likely that the protozoa and metazoa associated with FTW serve a predatory role with pathogens like fecal coliform bacteria and cyanobacteria (Song et al, 2009).

Chapter 3: MATERIALS AND METHODS

3.1 Study Site

The Carseland SPs (Figure 3.1) are located near the town of Carseland, in Wheatland County, Southern Alberta, Canada (latitude 50.847120, longitude -113.462309). The ponds treat the municipal wastewater of the hamlet of Carseland, and the nearby golf course/community of Speargrass. Total population is less than 1000 persons. The system functions similar to other SPs found in Alberta.

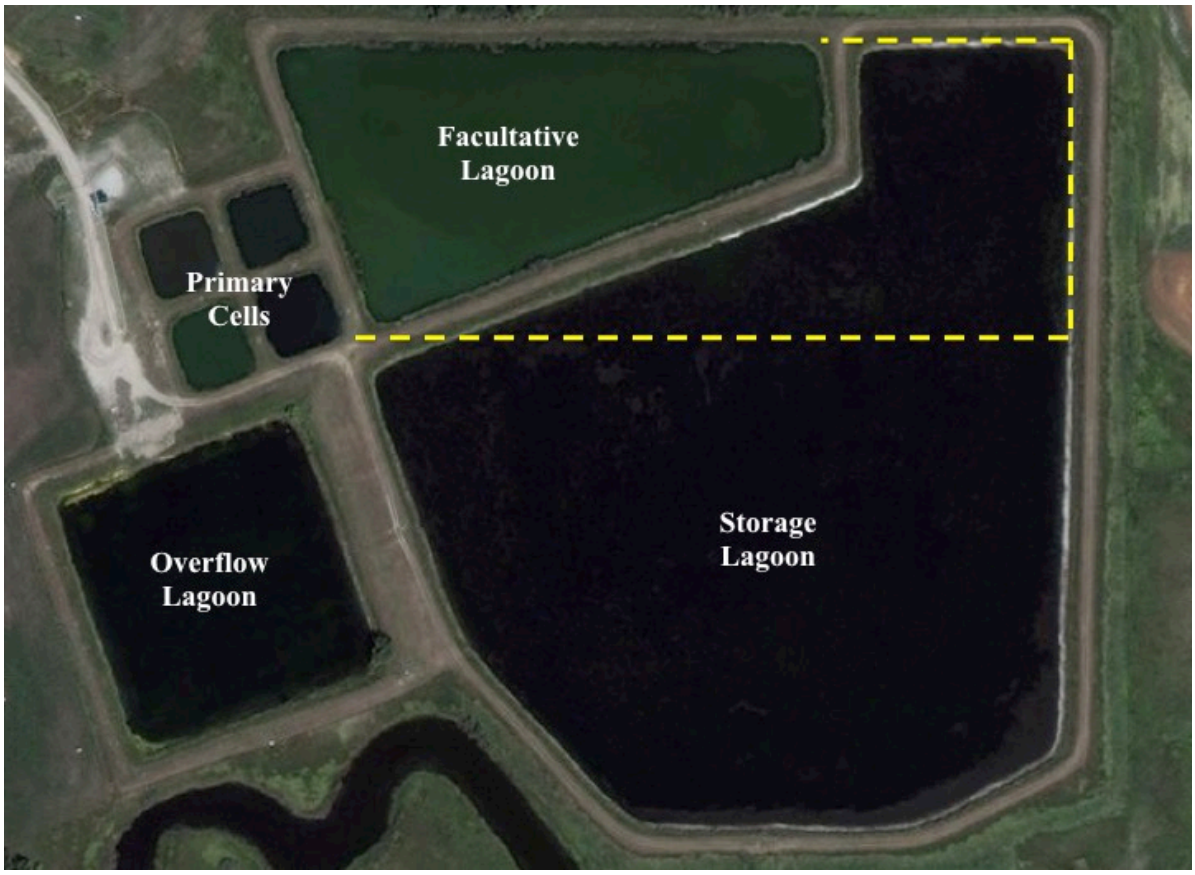

Figure 3.1 Satellite image of the Carseland stabilization ponds, highlighting the primary cells, facultative lagoon, storage lagoon, and overflow lagoon. The facultative lagoon was expanded in 2013 at the expense of the storage lagoon, and the current border is denoted by the dotted line.

There are four small anaerobic cells termed the 'primary cells' where solids settle. The primary cells have a HRT of 14-days, a depth of 2.95m (for each cell), and a total volume of 1549m^3. The facultative lagoon is 1.5m deep and has a HRT of 134.5days and a volume of 73,411 m^3. The storage lagoon has a maximum operating depth of 3m and a maximum volume of 310,000m^3. Finally the overflow lagoon holds wastewater when the storage lagoon exceeds capacity; it remained dry for the duration of the study. Similar to 80% of the SPs operating in Alberta, the Carseland SPs operate using controlled discharge, meaning effluent is only discharged once per year in late summer/early fall (August/September) after a long period of ice-free treatment (Price et al, 1995). The storage lagoon is then slowly filled over the course of a year. SPs in cold climates typically experience a drop in performance during the winter months and periods of sustained ice-cover (Price et al, 1995). The effluent of the lagoons is released into the nearby Bow River after infiltrating through soil rather than by surface flow. The ponds were designed to have a HRT of at least 60 days, to meet the standards set forth by Alberta Environment and Parks. The primary cells must have a minimum 2-day HRT for each cell, and the storage lagoon must have a 12 month storage capacity (Reid, 2012).

Two sea-containers (40ft standard dry cargo container) were selected to act as pilot scale SPs for the purpose of the study (Figure 3.2). The interior dimensions of the containers were 13.5m long, 2.3m wide, and 2.7m tall. Staff from Wheatland County modified the containers by removing the top panels and welding the exterior doors shut to create open topped vessels. A spray-on waterproof liner was used to coat the interior of the containers (Ecodur 201 Protective Coating, non-toxic, solvent free) to prevent leaking. A barrier (sheet metal, 5mm) was welded into the centre of the containers to

divide them into two treatment cells, each of which was 6.75m long and 2.3m wide. The containers were placed onsite on April 22, 2015 approximately 100m west of the primary cells with an east-west orientation. Approximately 1.2m of soil were added around the perimeter of the containers to act as thermal insulation.

Figure 3.2 Sea-containers installed on site. Soil was piled to a depth of 1.2m around the containers to provide thermal insulation. Photo taken April 22, 2015.

3.2 Floating Treatment Wetlands Construction and Planting

On April 17, 2015 the FTWs were prepared in a greenhouse in Vulcan, Alberta (60km south of Carseland). The floating mats were BioHaven^{TM} floating islands (Floating-Island-International, 2016), which are composed of a polyethylene plastic matrix and injected with polyurethane marine foam. The matrix creates a support system for macrophyte and biofilm growth, while the marine foam provides buoyancy. Two sets

of mats (each 20cm thick) were prepared (one for each sea-container) (Figure 3.3). Each set of mats has a size of 3.7m^2 (2.44m x 1.52m) in area, which equals 22% surface area coverage for the treatment cells within the sea containers. The selection of 22% coverage was based upon a study by Winston et al. (2013), which concluded that this degree of coverage is needed to provide adequate treatment without significantly hindering oxygen transfer from atmosphere. Also, since the purpose of this study was to ascertain whether the use of FTWs would enhance the treatment efficiency of the SPs, 22% coverage was chosen as it is reasonable for full-scale applications within SPs. Although greater than 20% coverage is physically feasible, it may not be financially feasible.

Figure 3.3 FTW construction in April 2015. Photo on the left shows the bare polyethylene FTW matrix with recently added rockwool within the planting holes and on the surface. Photo on the right is a completed FTW with macrophytes plugs (*Carex a.* along the outside *Scirpus m.* in the centre) and coir covering to protect from U.V. radiation in the field.

Each mat was planted with 70 macrophyte plugs ranging from 20-30cm in length. Planting holes within the BioHaven^{TM} matrix were 15cm deep. The macrophytes selected for each mat were evenly divided between Water Sedge (*Carex aquatilus* Wahlenb 1803)

and Small Flowering Bulrush (*Scirpus microcarpus* J. Presl and C. Presl 1828). These plant species were selected because they are perennial, emergent macrophytes native to southern Alberta and therefore can be expected to tolerate anoxic conditions and very cold temperatures (<-20ºC). The same macrophyte species have been used previously in FTW projects for stormwater ponds in Southern Alberta, and macrophytes from both the *Carex* and *Scirpus* genus have been used in past FTW studies (Ladislas et al, 2013; Tanner & Headley, 2011; Van de Moortel et al, 2010; Wang & Sample, 2014), though never the exact same species as was used in this study. The macrophytes were transferred into the planting holes with packed rockwool, which allows water to wick to the roots until the macrophytes established, but avoids adding nutrients which can occur with other growing mediums like peat. Each mat was covered in a coir-fibre blanket (2cm thick) in order to protect the polyethylene fibers within the mat from ultraviolet degradation (Figure 3.3). Two completed FTWs were placed in a 15cm deep pool in a greenhouse for two months (April 17-June 16, 2015) to allow the macrophytes to establish. The macrophytes were lightly fertilized during their establishment in the greenhouse. The FTWs were transferred to the sea-containers in Carseland on June 16, 2015 (Figure 3.4). Each FTW was placed in the middle of the western cell of each sea-container and as little space as possible was left on the outside edges for water to bypass the perimeter of the wetland.

Figure 3.4 Completed FTW installed in the south sea-container on June 16, 2015. Visible in the background is the control cell.

3.3 Experimental Design and Monitoring

A total of three different treatments were run between 2015 and 2016, the storage and facultative treatment in 2015 and the two-stage treatment in 2016. The purpose of these treatments was to determine where the FTW would have the greatest impact if scaled up and used in a SP. For 2015, the storage treatment investigated the impact of a FTW on a relatively static, low nutrient system (as seen in many storage lagoons), and the facultative treatment investigated the impact of a FTW on regularly fed, partially treated wastewater (as seen near the outlet of a facultative lagoon, or near the input of a storage lagoon). For 2016, the two-stage treatment investigated the impact of a FTW on high-nutrient wastewater that had only undergone primary treatment. The operation and design of each treatment is discussed in greater detail below.

3.3.1 Experimental Design in 2015 Season

In the 2015 field season (from June – November), the two sea-containers were divided into two separate treatments (Figure 3.5). The northern sea-container was designated as the storage treatment and divided into two cells, the FTW cell (S-FTW) and the control cell without a FTW (S-C). On June 5, 2015 each cell was filled with wastewater (transferred via water truck) withdrawn from the western side of the Carseland storage lagoon. Cells were filled up to a depth of 2.2m in the cells, which gave a volume of 35.6m^3 per cell. There were no further wastewater additions during the 2015 field season, as the purpose of this treatment was to determine how the FTW performed as a final treatment step for low-nutrient wastewater with little input.

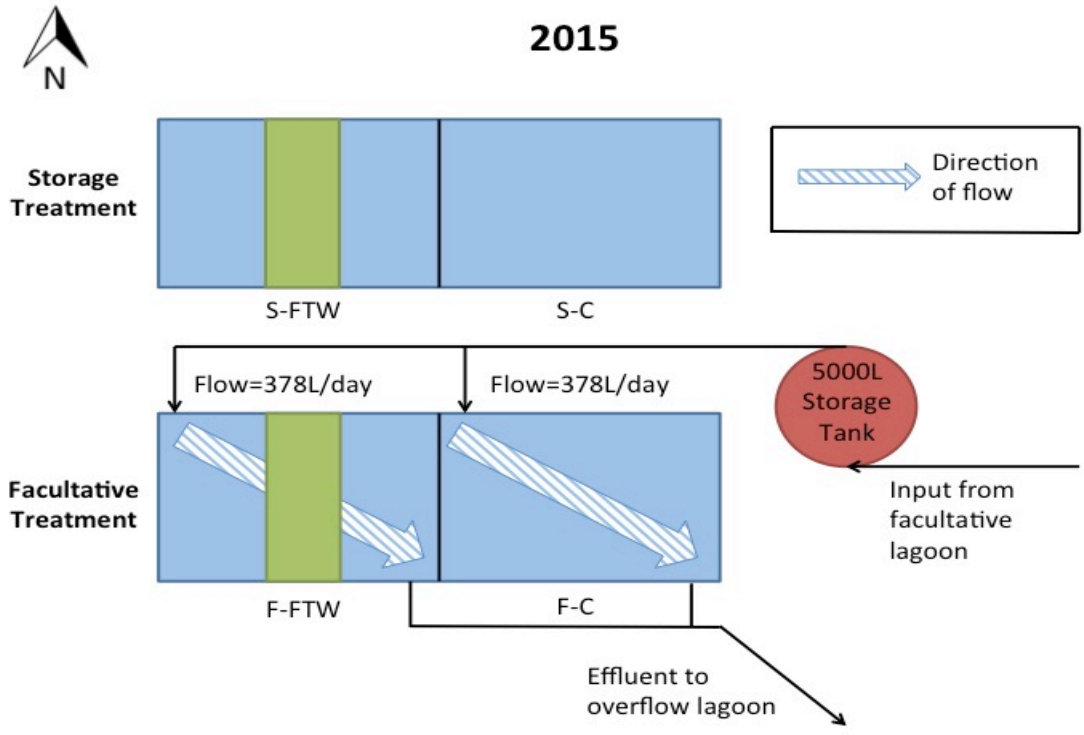

Figure 3.5 Experimental design for 2015, showing both the storage treatment and facultative treatment

The southern sea-container was designated the facultative treatment and also divided into two cells, the FTW cell (F-FTW) and control cell without a FTW (F-C) (Figure 3.5). In each cell, 1m of clay was laid at the bottom (to mimic the bottom of the facultative lagoon). Both cells were filled with wastewater withdrawn from the western side of the Carseland facultative lagoon. Cells were filled up to a depth of 1.2m, which gave a volume of $19m^3$ per cell. Wastewater to be fed into the system was withdrawn from the western end of the facultative lagoon, nearest to the input from the primary cells. Wastewater was pumped twice per week from the facultative lagoon to a 5000L storage tank located adjacent to the F-FTW and F-C cells. Two submerged 1/3 horsepower electric pumps on timers, placed in the storage tank, were used to feed the wastewater into each cell at a rate of 21L $minute^{-1}$ for three minutes every four hours. Thus, wastewater was added at a rate of 378L day^{-1}, which gave a HRT of 50.3 days for each cell. This HRT was chosen since it is slightly lower than the minimum 60-day HRT required for facultative lagoons in Alberta (Reid, 2012). The selection of a slightly lower HRT is necessary as the pilot scale system is expected to be more efficient than the full-scale lagoons considering scaling effects (which is beyond the scope of this thesis). Wastewater entered from the NW corner of the cells, and an outlet was placed at the SE corner, where effluent was drained to the nearby overflow lagoon. Pumps were operated from June 16–Nov 2, until temperatures below 0°C which made operating the pumps impossible.

3.3.2 Experimental Design in 2016 Season

In the 2016 field season, the experimental design was altered to feed the cells with high-nutrient wastewater (from the primary cells) and changed into a two-stage system

(Figure 3.6). First, an additional 5,000L storage tank was placed at the site and connected to the existing storage tank, to double the holding capacity for wastewater. The input to the storage tanks was withdrawn from the outlet of the primary cells (rather than from the facultative lagoons).

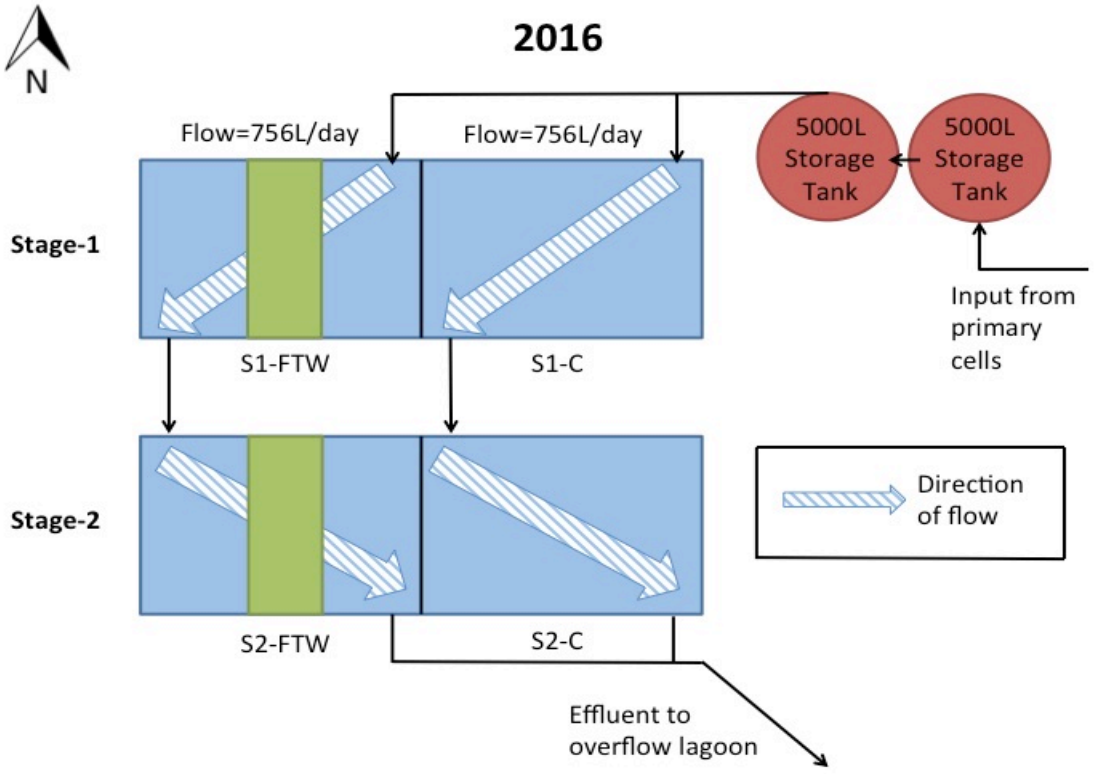

Figure 3.6 Experimental design for 2016, showing the two-stage treatment

At the beginning of the field season, all cells were completely drained and 1m of clay was added to the bottom of the both cells in the northern sea-container (formerly the storage treatment). The cells were refilled on May 17, 2016 with wastewater taken from the outlet of the primary clarifiers. Input from the storage tanks was diverted to the NE corner of northern sea-container, now called stage-1. In the SW corner, a 19litre bucket was submerged into each cell, and inside a 1/3 horsepower pump with a float switch was

added. The pump was connected to the input of the southern sea-container (now called stage-2), where wastewater was once again able to flow from the NW to the SE corner. Wastewater was fed into both the southern and northern sea-containers at a rate of 756L day^{-1} per cell, which gave a 25.15day HRT for each cell, in both stage-1 and stage-2. Thus, the HRT for the whole system is 50.3days, same as the facultative treatment from 2015. Pumps were operated from June 7-Oct 10, until temperatures below 0°C made operating the pumps impossible.

3.4 Basic Physicochemical Water Quality Measurement

A multiparameter meter (YSI ProPlus) was used to measure in situ physicochemical water quality parameters, namely water temperature, EC (2016 only), pH, and DO. Measurements were taken twice per week, between 9:30-10:00am in all cells. The measurements were taken at two depths: 0.15m and 1m for the facultative treatment and two-stage treatment, and 0.15m and 2m for the storage treatment. Measurements at different depths were to monitor if stratification was occurring. In both the control and FTW cells for all treatments, measurements were taken at two opposite corners (NW and SE), at 30cm diagonal from the edge. To capture the overall condition of the cells, values from opposite corners (but not separate depths) were averaged for analysis.

In the 2016 season, two sondes were installed (YSI EXO2 Multiparameter Sondes) in late summer to gather continuous data (15-minute intervals) for temperature, EC, pH and DO. For stage-1, the sondes were placed on the western side of both the S1-C and S1-FTW cells, 50cm from edge and with the sensors located at a 30cm depth. Stage-1 data was gathered from August 18-25, and September 5-8. For stage-2 the sondes were placed on the eastern side of both the S2-C and S2-FTW cells 50cm from edge and with

the sensors located at a 30cm depth. Stage-2 data was collected from August 25 to September 5. Air temperature data was obtained from Environment Canada (http://weather.gc.ca/city/pages/ab-6_metric_e.html), from a weather station recording hourly temperature data located in Strathmore, AB (26.7km from the study site).

3.5 Water Sampling

In the 2015 field season, water samples were gathered at 10:00am twice per week, in 1L plastic bottles (rinsed with deionized water). For the facultative treatment, water samples were collected from the drainage outlets of the F-FTW and F-C cells. In addition, a sample was taken from the surface of the storage tank shortly after filling with wastewater from the facultative lagoon, to determine influent water quality. For the storage treatment, water samples were taken from all four corners (250 ml per corner) of the S-FTW and S-C cells at 15cm below the surface and immediately compiled at the site to form a 1L sample for each cell.

In the 2016 field season, 1L water samples were taken twice per week from all cells, S1-FTW and S1-C in the stage-1 and S2-FTW and S2-C in the stage-1. The water samples were collected from the 19L effluent buckets for stage-1, and from the drainage outlets for stage-2. In addition, a sample was taken from the storage-tank, immediately after refilling with wastewater from the primary clarifiers.

3.6 Water Quality Parameter Analysis

After collection, samples were immediately placed into a 10ºC cooler. Samples were transported to the University of Calgary and analyzed. For nutrients the parameters measured were reactive phosphorus, total phosphorus, nitrate, ammonia, and total nitrogen. Tests were performed using Hach reagents, a DR 2800 spectrophotometer

(Hach Instruments), and a DRB 200 digester (Hach Instruments). The methods used were:

- Reactive Phosphorus (Hach method 8048)
- Total phosphorus (Hach method 8180)
- Nitrate (Hach method TNT 835)
- Ammonium (Hach method 10023)
- Total Nitrogen (Hach method 10072)

All methods are outlined in the Hach Water Analysis Handbook, 8th edition (Hach, 2013). Organic nitrogen (ON) was calculated by subtracting nitrate and ammonium concentration from total nitrogen concentration. RP was subtracted from TP, and then divided by the total to determine the percentage of TP that was not reactive.

TSS measurements were performed according to Standard Methods 21st Edition (Eaton et al, 2005c). Turbidity measurements were performed using a LaMotte 2020 turbidity meter. Tests were performed twice per week for the first two months (June and July), and once per week for the remainder of the season due to very low values for both parameters.

5 day BOD was measured and calculated according to standard methods (Eaton et al, 2005a), twice per week. From August 4 to October 10, 2016 fecal coliform analysis was performed on all water samples, twice per week. Analysis was performed according to standard methods 21st Edition (Eaton et al, 2005b).

To determine removal efficiency in the facultative and two-stage treatment at the time of sampling, the following formula was used:

$$Eff(\%) = 100 \times \frac{C_i - C_e}{C_i}$$

Where *Eff* is the removal efficiency, C_i is the concentration of the influent, and C_e is the concentration in the effluent leaving the treatment cell. Efficiency was based off concentration (and not loading) since effluent limits in Alberta are based on concentration.

3.7 Macrophyte and Water Level Monitoring

In the storage treatment, the water level would continuously drop due to evapotranspiration, and as such the water depth was recorded once per week in the field season. Photographs were taken every week to qualitatively compare the growth of macrophytes in the both treatments. In addition, all cells were visually monitored for submerged vegetation and algal growth. When water sampling ceased in the fall, all cells remained filled so as to observe the effects of winter on the FTWs. The FTWs were inspected once per month from November to April to ensure that the platforms and the sea-containers were not damaged by ice formation. In the spring, growth and emergence of the macrophytes was visually monitored weekly to determine winter survival.

On October 17, 2016, shoots from six macrophytes (three of both *Carex a.* and *Scirpus m.*) from each FTW were harvested (cut at the base, where the shoot emerged from the plastic matrix). The shoots were dried in an 80°C oven for 48hrs and then weighed. Weight was averaged for each species, on each FTW. Dried tissue was then finely ground, and 1g of tissue was thoroughly mixed with 1L of water for 3 minutes. The solution was then digested and total phosphorus concentration was recorded using Hach Method 8048 (Hach, 2013). Phosphorus concentration in macrophyte tissue was averaged per species.

3.8 Statistical Analysis

All analysis was performed using Excel 2011 (Microsoft Inc. USA), using the Real Statistics add-in. Average contaminant concentration and removal efficiency was calculated for each parameter, along with standard deviation. Average contaminant concentration (in all treatments) and average removal efficiency (facultative and two-stage) were first analyzed with the Shapiro-Wilk test for normality. All data sets were found to be non-parametric, and therefore a non-parametric test for significant difference in means had to be selected. The Mann-Whitney U-Test does not assume normality, and was used to determine if physicochemical parameters in the FTW cells were significantly higher, or lower, than the paired control.

Chapter 4: RESULTS

4.1 Macrophyte Monitoring/Sampling

In the 2015 season, the macrophytes appeared to grow larger in the storage treatment than in the facultative treatment (both species). By August, the macrophytes in the facultative FTW developed a yellow-tinge in the shoots (Figure 4.1). Since there were no additions of wastewater in the storage treatment, over the course of the season depth in both cells was reduced by 31.5cm and 33cm in the S-C and S-FTW cells respectively. Both cells in the facultative treatment developed significant coverage by floating algae and were colonized by submerged vegetation; this did not occur in the storage treatment. Both FTWs remained in the sea containers over the winter, and a thick layer of ice (>10cm) developed.

By April 2016 all macrophytes successfully returned, except for one individual macrophyte (*Carex aquatilius*) in the facultative FTW. The deceased macrophyte was replaced with a *Carex a.* specimen found onsite in the nearby overflow lagoon. For the 2016 season all macrophytes appeared healthy and grew significantly. Harvesting at the end of the 2016 season (October) measured an average height of 75cm and 115cm for *Carex a.* and *Scirpus m.* respectively in stage-1, and 58cm and 75cm in stage-2. Average dry-biomass per macrophyte was 57g and 54g for *Carex a.* and *Scirpus m*. respectively in stage-1, and 58g and 63g in stage-2. Estimated dry biomass for each FTW was therefore 3,891g and 4,239g for the stage-1 and stage-2 FTW respectively. Phosphorus content within the shoots was 2.6 and 5.3mg (g of dry biomass)$^{-1}$ for *Carex a.* and *Scirpus m*. respectively. Therefore, the estimated mass of phosphorus within the dry biomass was

15.4 and 17.2g for the stage-1 and stage-2 FTW respectively. Colonizing species were observed establishing themselves on the FTW throughout the summer. Both cells in stage-2 (formerly the facultative treatment) developed significant coverage by floating algae and were colonized by submerged vegetation.

September 8, 2015

September 8, 2016

Figure 4.1 Macrophyte growth on the FTW at the end of the first season (2015) and the second season (2016)

4.2 Basic Physicochemical Properties

4.2.1 Temporal Variation 2015

In the S-FTW cell, pH and temperature was lower, (-0.16 and -0.2°C respectively), and DO was higher (+0.3mg L^{-1}) relative to the control (S-C) (Table 4-1). The storage treatment cells (S-C and S-FTW) showed no significant difference for temperature and dissolved oxygen (DO) ($p>0.05$), but pH in the S-FTW cell was significantly lower than the control ($p<0.05$). Temperature fluctuated between 15 and 22°C in both cells for the first three months of data collection (June-August 2015), then decreased in conjunction with ambient air temperature (Figure 4.2a). In both cells pH gradually increased, while pH in the S-FTW cell remained lower than the S-C cell until September 29 when it became approximately equal in both cells (Figure 4.2b). DO increased continuously in both cells until July 21, 2015, where it remained between 8-10mg L^{-1} for the remainder of the season. The exception was between September 25 and Oct 2, when DO within the S-FTW cell was supersaturated (>100%) and exceeded 10 mg L^{-1} (Figure 4.2c). The storage treatment was the only treatment to experience thermal stratification, however it was not consistent. Thermal stratification (greater than 1°C difference at the 0.15m and 2m sampling depths) only occurred on ten sampling dates, and always occurred simultaneously in both the S-C and S-FTW cells. The dates were June 30, July 3, 10, 21, 24, August 4, 11, 25, and September 11. Based on observations in the field these stratification events were associated with warm ambient temperature and low wind velocity.

In the facultative treatment, no significant difference was detected for temperature, DO, and pH between the F-C cell and the F-FTW cell. In the F-FTW cell, temperature

and pH was slightly lower than those in the F-C cell at an average of -0.1°C and -0.12, respectively; whereas DO was higher than the F-C cell with an average of +0.8mg L^{-1} (Table 4-1). Trends for temperature in both the F-C and F-FTW cells were similar to the storage treatment (Figure 4.2a). In both cells pH continuously increased and reached a maximum on September 29, 2015 (10 and 9.9 in the F-C and F-FTW cells respectively) (Figure 4.2b). DO was consistently higher than 10mg L^{-1} from July 21 till the end of the season (Figure 4.2c). Maximum saturation occurred on October 2, 2015 with 129% and 192% saturation in the F-C and F-FTW cells respectively.

Table 4-1 Average basic physicochemical values (± standard deviation) for 2015 in the storage treatment, facultative treatment, and the influent

	Storage Treatment		Facultative Treatment		
	S-C	S-FTW	F-C	F-FTW	Influent
Temperature (°C)	13.8±5.03	13.6±4.98	13.6±5.27	13.51±5.28	-
DO (mg L^{-1})	7.46±1.85	7.77±2.44	12.2±2.45	13.01±3.17	-
DO (% Saturation)	73.1±26.9	75.3±30.3	119.4±40.1	126.9±46.6	-
pH	8.3±0.27	8.14±0.24	9.51±0.55	9.39±0.59	8.70±0.49

4.2.2 Temporal Variation 2016

In stage-1 there was no significant difference in temperature, EC, or pH, but the DO concentration was significantly lower in the S1-FTW cell than in the S1-C cell ($p<0.05$). On average, temperature, DO and EC were all lower in the S1-FTW cell (-0.4°C, -0.9mg L^{-1}, and -18µs cm^{-1} respectively) relative to the control (Table 4-2), whereas the pH was equal with a mean of 7.9 in both cells. Temperature in both cells fluctuated between 15 and 20°C during June-August 2016, and then declined continuously in September (Figure 4.3a). EC in both cells gradually increased until reaching a maximum on September 1 (1874 and 1845µs cm^{-1} in the S1-C and S1-FTW cells respectively) (Figure 4.3b). DO dropped rapidly in the first month, and remained below 2mg L^{-1} for June-August (Figure 4.4a). In both stage-1 cells pH fluctuated between 7.6 and 8.2 for most of the season, though the S1-C cell had higher variability than the S1-FTW cell (standard deviation 0.24 versus 0.08) (Figure 4.4b).

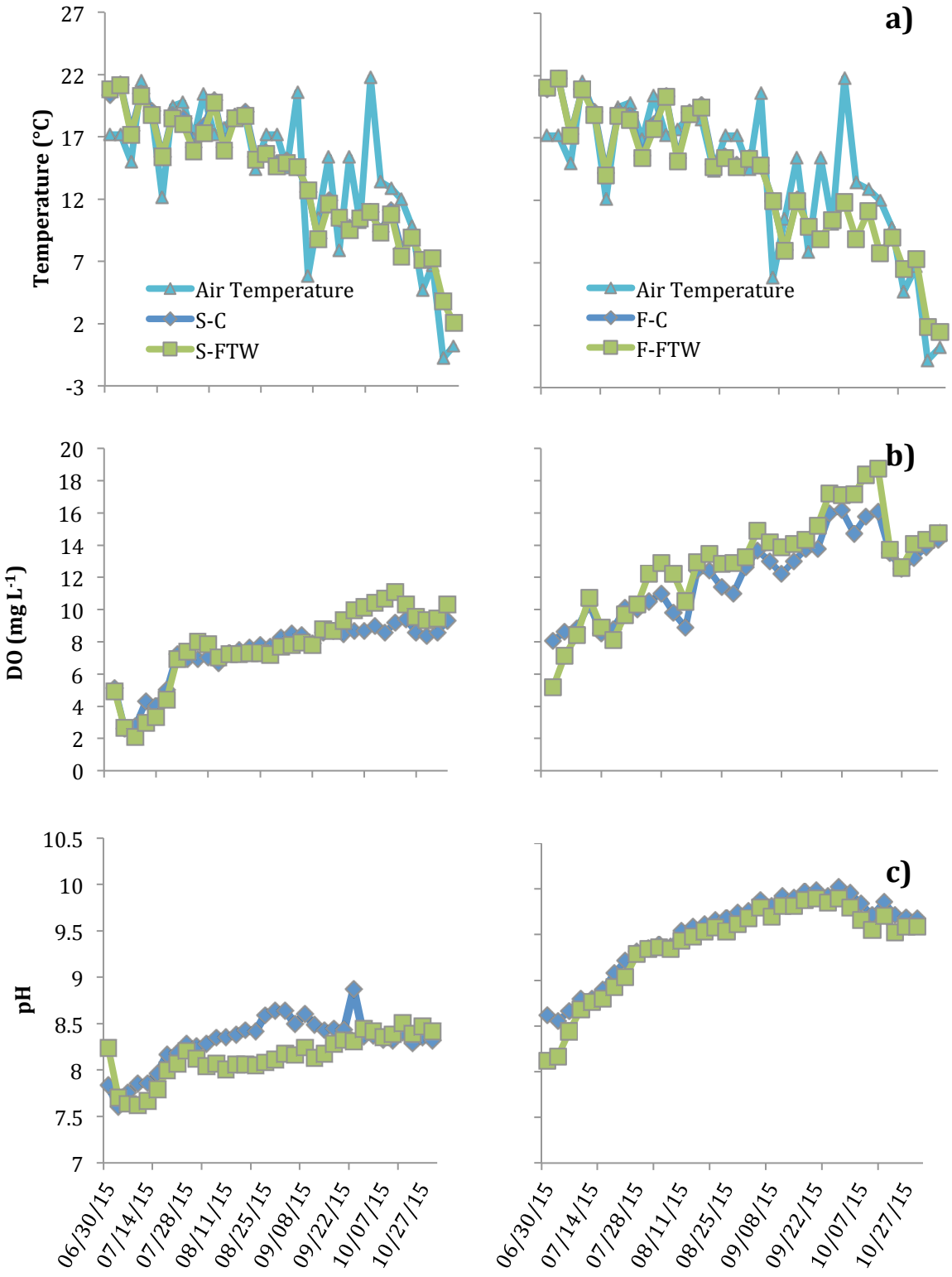

Figure 4.2 Temporal variation of temperature (a), DO (b) and pH (c) between the storage treatment (left column) and the facultative treatment (right column)

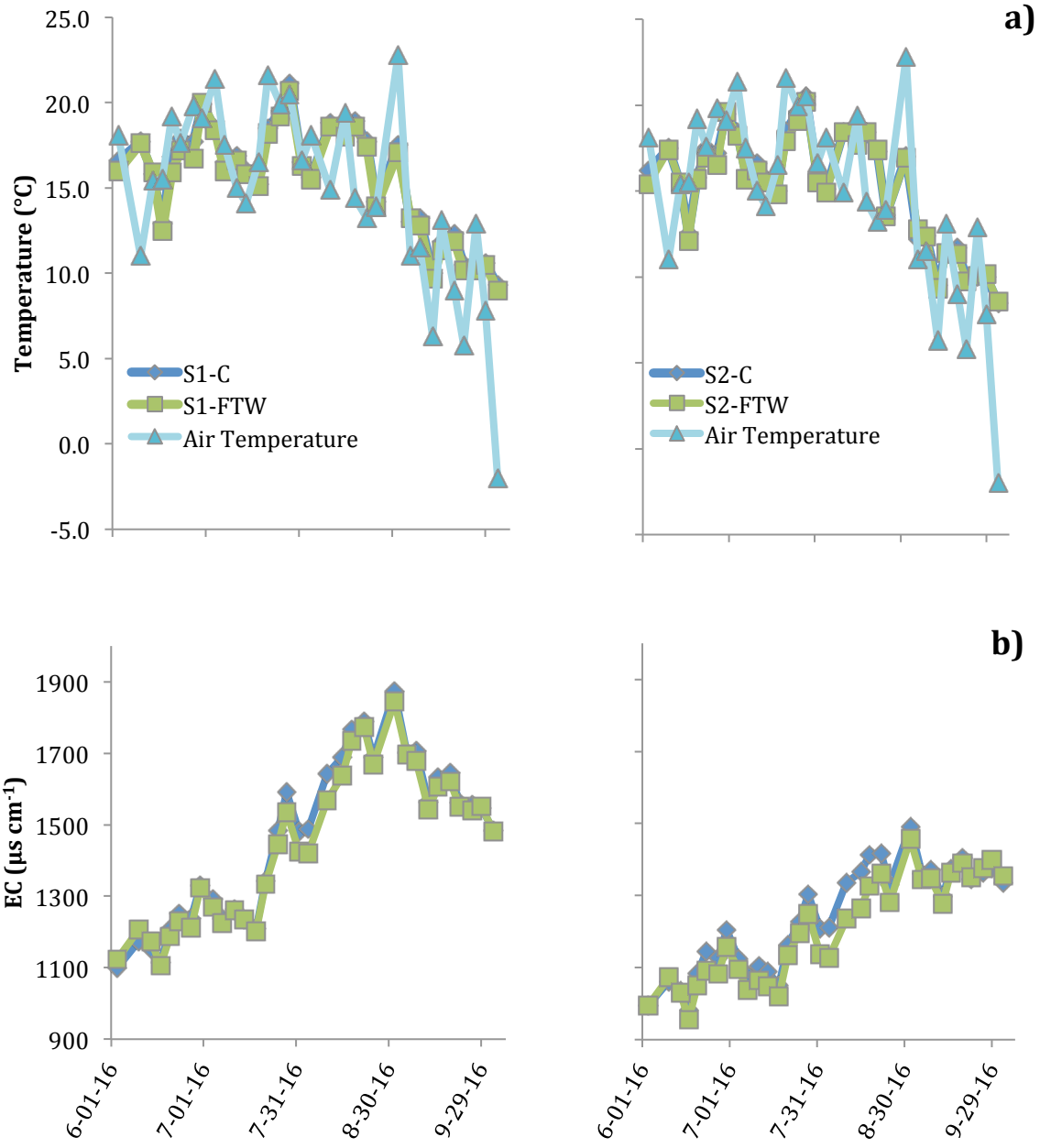

Figure 4.3 Temporal variation of temperature (a), and EC (b) between stage-1 (left column) and stage-2 (right column) in the 2016 two-stage treatment

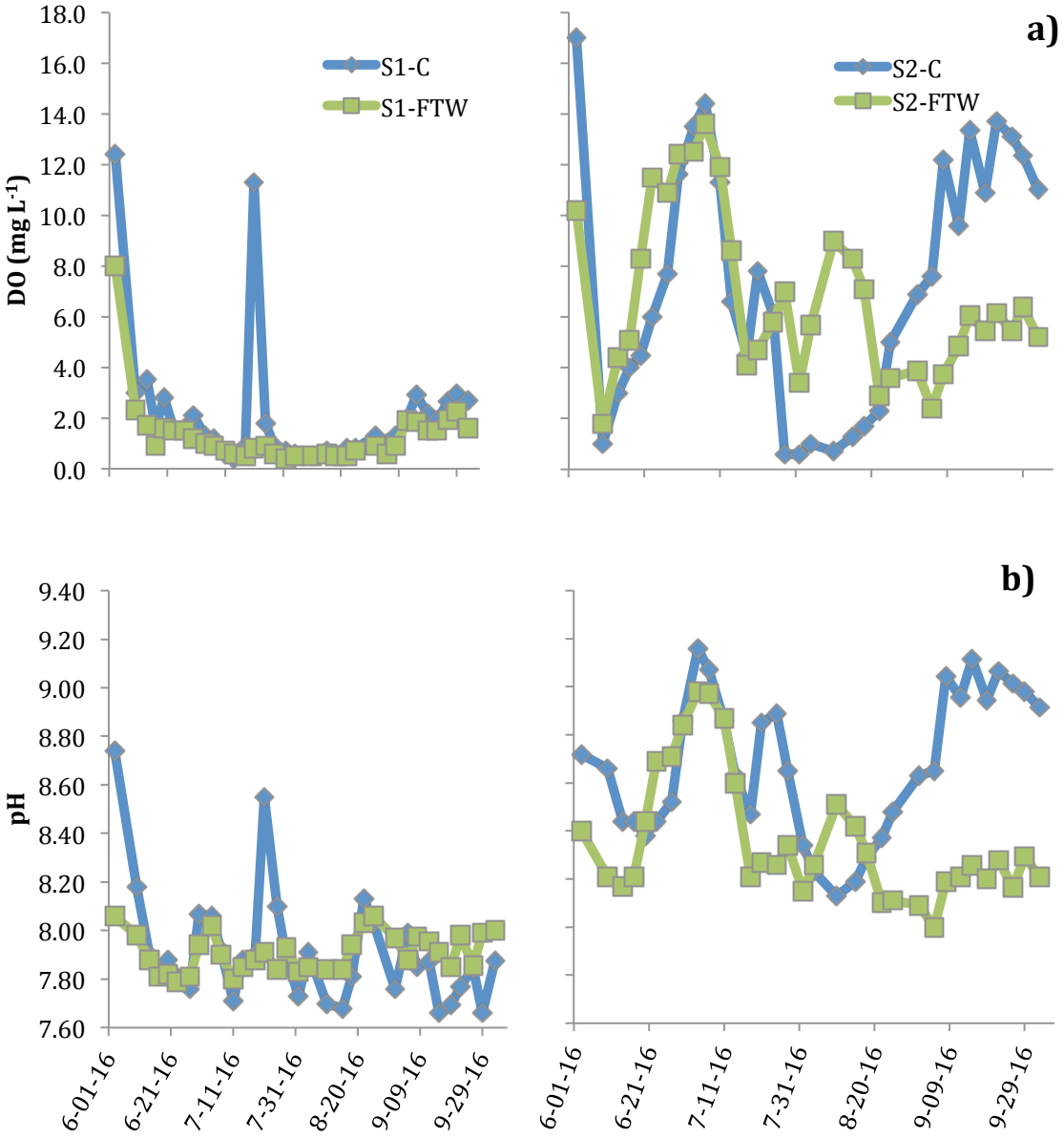

Figure 4.4 Temporal variation for DO (a), and pH (b) between stage-1 (left column) and stage-2 (right column) in the 2016 two-stage treatment

In stage-2, there was no significant difference in temperature, DO, and EC, but the pH was significantly lower in the S2-FTW cell than in the S2-C cell ($p<0.001$). On average temperature, DO, EC, and pH were all lower in the S2-FTW cell (-0.2°C, -0.7mg L^{-1}, -33µs cm^{-1}, and -0.3 respectively) (Table 4-2). Temperature in both cells followed a similar trend to the stage-1 cells (Figure 4.3a). EC increased continuously in both cells over the season, reaching a maximum on Sept 1, 2016 (1491 and 1455µs cm^{-1} in the S2-C and S2-FTW cells respectively) and then remained more or less constant until the end of the field season (Figure 4.3b). DO concentration fluctuated considerably in both cells, at times low (<2mg L^{-1}) at other times high (>10mg L^{-1}) (Figure 4.4a). From Sept 8, 2016, onwards DO was consistently supersaturated in the S2-C cell, while in the S2-FTW cell it remained around 50% saturation. Overall, pH followed a similar trend as DO, and was highly variable throughout the season (Figure 4.4b).

Table 4-2 Average physicochemical values (± standard deviation) for 2016 for the two-stage treatment and the influent

	Stage-1		Stage-2		
	S1-C	S1-FTW	S2-C	S2-FTW	Influent
Temperature (°C)	15.7±3.22	15.3±3.2	15.2±3.3	15±3.24	-
DO (mg L^{-1})	2.2±2.6	1.3±1.3	7.4±4.9	6.7±3.2	0.5±0.5
DO (% Saturation)	24.9±31	14.6±15	80.8±53.2	76.1±39	-
EC (µs cm^{-1})	1455±223	1437±212	1236±144	1203±145	1534±361
pH	7.9±0.24	7.9±0.08	8.7±0.3	8.4±0.26	7.9±0.21

4.2.3 Continuous Monitoring 2016

For stage 1, the sondes were installed from August 18-25, and September 5-8. Over 10-days of continuous monitoring, the S1-FTW cell had significantly lower DO, EC and pH (-0.5mg L^{-1}, -14µs cm^{-1}, and -0.04 respectively ($p<0.001$)), and a lower (but not significantly so) temperature (-0.2°C ($p>0.05$)) (Table 4-3). Minimum daily temperature in both cells occurred between 5-7am, and maximum temperature between 4-6pm (Figure 4.5a). Temperatures typically fluctuated 2-3 degrees over a 24hr period, but the largest variation during the monitoring period was 9.4 degrees (Table 4-3). EC followed a similar trend to temperature, reaching a maximum in late afternoon, and a minimum in the morning (7-10am) (Figure 4.5b). EC often fluctuated between 100-200µs cm^{-1} daily. DO was consistently lower in the S1-FTW cell, especially during the day. During the afternoon (12-6pm) DO saturation was often 15% higher in the S1-C cell compared to the S1-FTW cell, but <5% higher during the early morning (2-8am) (Figure 4.5c). For pH, there was little fluctuation (standard deviation±0.04 in both cells), the daily max occurred between 2-4pm, and the minimum occurred between 4-6am (Figure 4.5d).

For stage-2, the sondes were installed from August 25-September 5. Over the 12 days of continuous monitoring, the S1-FTW cell had significantly lower temperature, DO, EC, and pH (-0.3°C, -3.36mg L^{-1}, -54µs cm^{-1}, -0.45pH respectively ($P<0.001$)) (Table 4-3). For temperature, the greatest difference between the S2-C and S2-FTW cells was during the late-afternoon/evening, occasionally exceeding 1.5°C difference (Figure 4.6a). DO and pH reached daily maximum values late in the day, between 8-12pm (Figure 4.6c and d). In the S2-FTW cell pH was less variable than the S2-C cell (standard deviation 0.03 versus 0.14).

Table 4-3 Average physicochemical parameters (± standard deviation) and minimum/maximum values from continuous monitoring within the 2016 two-stage treatment between August 18 and September 8, 2016

	Stage-1			
	S1-C		S1-FTW	
	Average (±SD)	Min-Max	Average (±SD)	Min-Max
Temperature (°C)	16.1±2.5	12.3–21.7	15.9±2.3	12.2–21
DO (mg L^{-1})	1.26±0.7	0.11–3.4	0.76±0.5	0–2.2
DO %	13±7	1.1–33.7	7.6±5	0–22.6
EC (µs cm^{-1})	1741±60	1632–1935	1727±58	1630–1918
pH	7.69±0.04	7.58–7.8	7.65±0.04	7.58–7.74
	Stage-2			
	S2-C		S2-FTW	
	Average (±SD)	Min-Max	Average (±SD)	Min-Max
Temperature (°C)	15.7±1.7	12.1–19.4	15.4±1.4	12.8–18.6
DO (mg L^{-1})	5.72±1.7	2.2–12.2	2.36±1.1	0.5–5.2
DO %	58±18	23.6–133	24±11	4.9–56
EC (µs cm^{-1})	1437±62	1327–1575	1383±56	1281–1503
pH	8.04±0.14	7.78–8.43	7.59±0.03	7.53–7.68

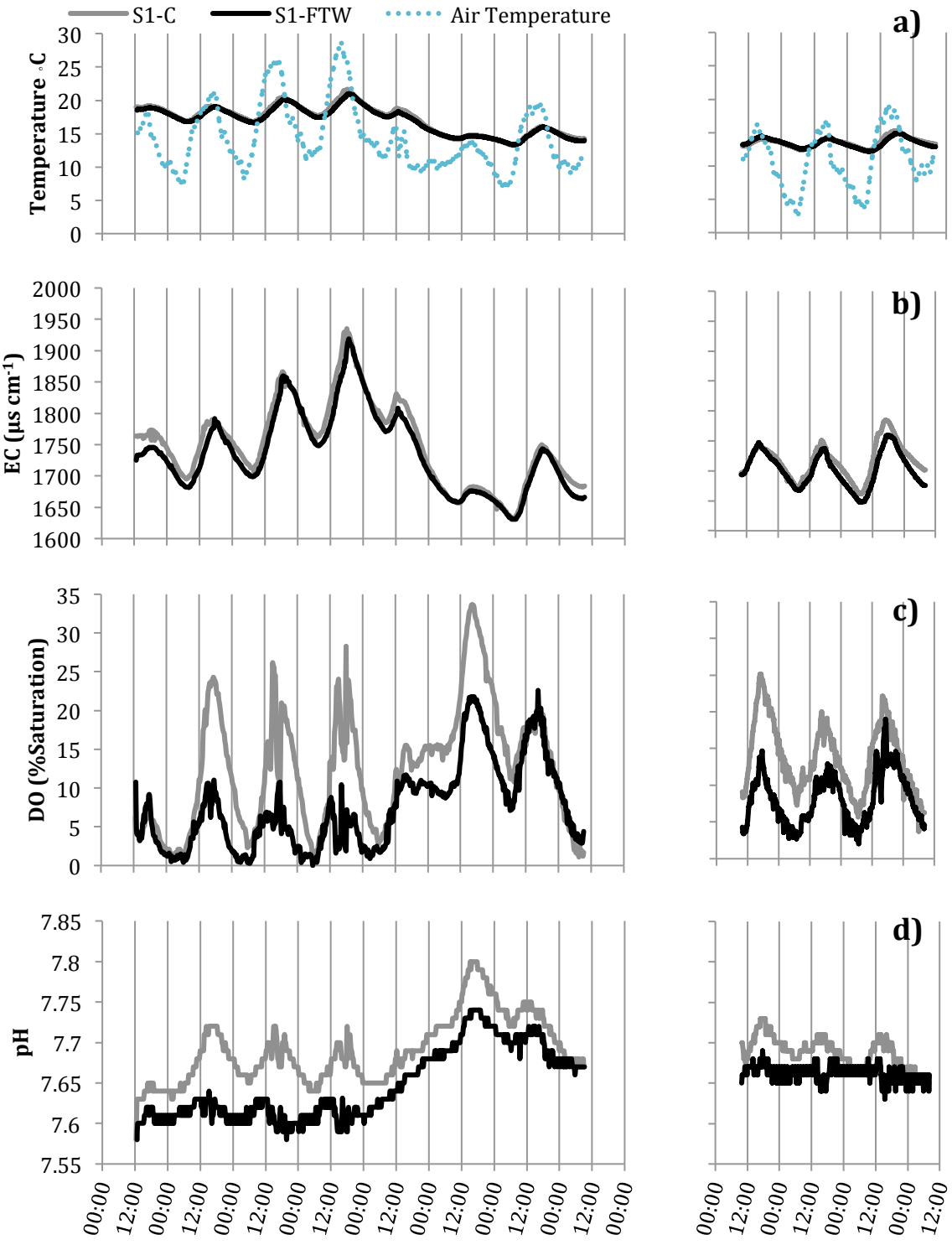

Figure 4.5 Diurnal variation of temperature (a), EC (b), DO saturation % (c) and pH (d) within stage-1. Data graphed on the left column was gathered from Aug. 18 to 25, and the right column from Sept. 5-8.

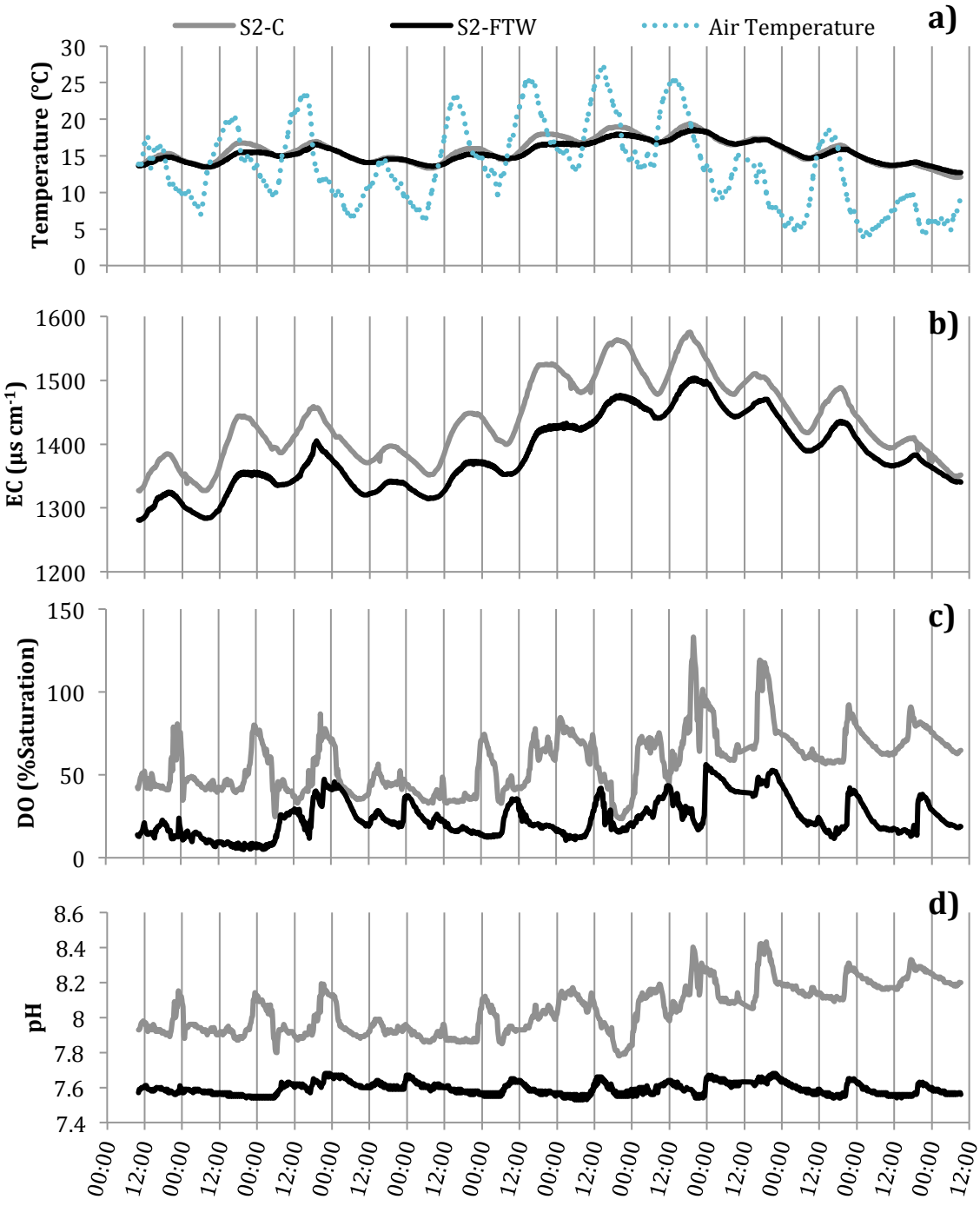

Figure 4.6 Diurnal variation of temperature (a), EC (b), DO saturation % (c), and pH (d) within stage-2. Data gathered from Aug. 25-Sept. 5 2016

4.3 Nutrient Properties in 2015

4.3.1 Storage Treatment

The wastewater used to initially fill the S-FTW and S-C cells was low in nutrients (0.2mg L^{-1} RP, 0.42 mg L^{-1} TP, 0.679 mg L^{-1} $N0_3^-$–N, 0.8 mg L^{-1} NH_4^+–N, and 3 mg L^{-1} TN). During the field season, there was no significant difference between the S-FTW and S-C cells for the average concentration of RP, TP, NO_3^-, and TN ($P>0.05$), but NH_4^+ concentration was significantly lower in the S-FTW cell than in the S-C cell ($P<0.001$).

NH_4^+ was consistently lower in the S-FTW cell from June 30, 2015 until the end of the field season. Concentration increased in the S-FTW and S-C cells until reaching a maximum on July 7, 2015 (1.27 and 0.65mg L^{-1} for the S-C and S-FTW cell respectively). Afterwards concentration decreased in both cells, and by July 21 2015 concentration was very low in the S-FTW cell (0.03mg L^{-1}), and remained less than 0.05mg L^{-1} for the remainder of the season, with the exception of Aug. 17 when it reached 0.06mg L^{-1}. Concentration in the S-C cell decreased at a slower rate, and never decreased below 0.06mg L^{-1} (Figure 4.7a). Note that there was no influent fed into the storage treatment cells after they were filled in the beginning of the season.

NO_3^--N concentration quickly diverged between the two cells, and rapidly increased in the S-FTW cell from 0.41 to 1.14 mg L^{-1} between June 23 and July 21, 2015. Afterwards, NO_3^- concentration sharply decreased in the S-FTW cell, until Sept 8 2015 when it fell below the detection limit for the test (0.23mg L^{-1} NO_3^-–N). In the S-C cell, concentration increased until Aug. 11 (reaching a maximum of 0.76mg L^{-1}) and steadily decreased until Sept. 15, where it maintained a concentration between 0.39 and 0.49mg L^{-1} (Figure 4.7b). When comparing values for the entire season there was no significant

difference between the S-C and S-FTW cell, but analysis comparing only the values from August 7 2015 onwards indicates concentration within the S-FTW cell was significantly lower ($P<0.001$). The maximum NO_3^- concentration in the S-FTW cell coincided with the onset of when NH_4^+ concentrations decreased below 0.05mg L^{-1} (July 21, 2015).

There was little difference in TN concentration between the S-C and S-FTW cells. The initial concentration in both cells was approximately 3mg L^{-1}, which increased to 5mg L^-1 in mid-July, before steadily decreasing to approximately 1mg L^{-1} by Oct. 27 (Figure 4.8). Average organic nitrogen was 1.7±1.2, and 1.6±1mg L^{-1} N for the S-C and S-FTW cells respectively, and there was no significant difference between the cells ($p>0.05$).

Average RP concentration was 0.23±0.09 and 0.24±0.11mg L^{-1} in the S-C and S-FTW cells, respectively. Initially concentration increased in the S-FTW cell relative to the control, and remained higher until July 17 2015 (Figure 4.7c). For the remainder of the season, concentration in both cells was similar (between 0.1 and 0.2 mg L^{-1}), except for isolated peaks in concentration in each cell (Aug. 25 and Sept. 4 for the S-C cell, Sept. 22 and Oct. 16 for S-FTW cell). TP followed a similar trend to RP (Figure 4.7d), and the average concentration for the S-C and S-FTW cell was 0.58±0.22mg L^{-1} and 0.64±0.21mg L^{-1} respectively. Initial TP concentration was greater in the S-FTW cell until July 17 2015, after which values in both cells remained quite similar. On average, TP was 57±19% and 62±21% higher then RP in the S-C and S-FTW cells respectively. There was no significant difference between the FTW and control cell ($p>0.05$).

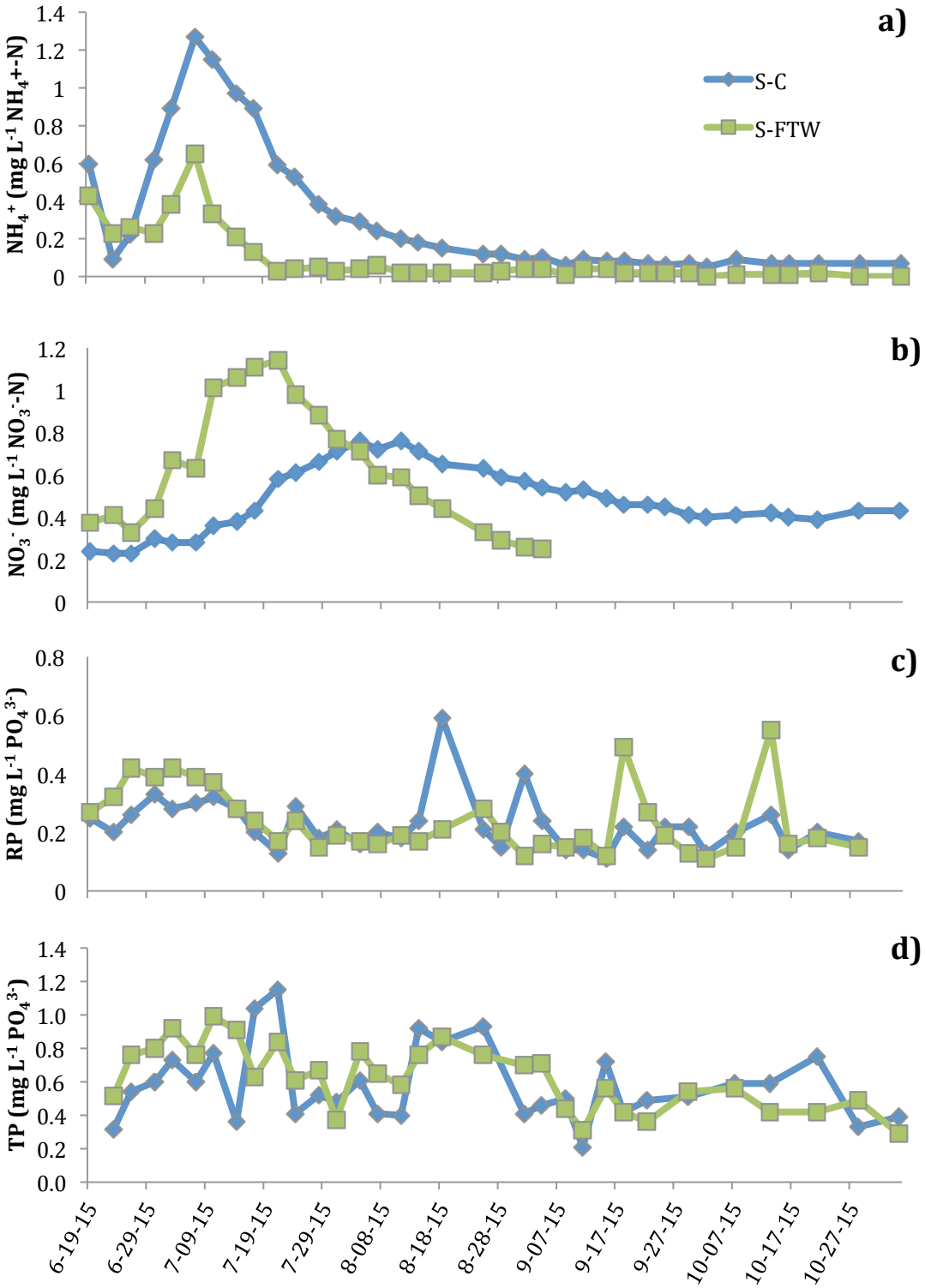

Figure 4.7 Temporal variation of nutrient concentration for Ammonia (a), nitrate (b), RP (c), and TP (d) for the 2015 storage treatment.

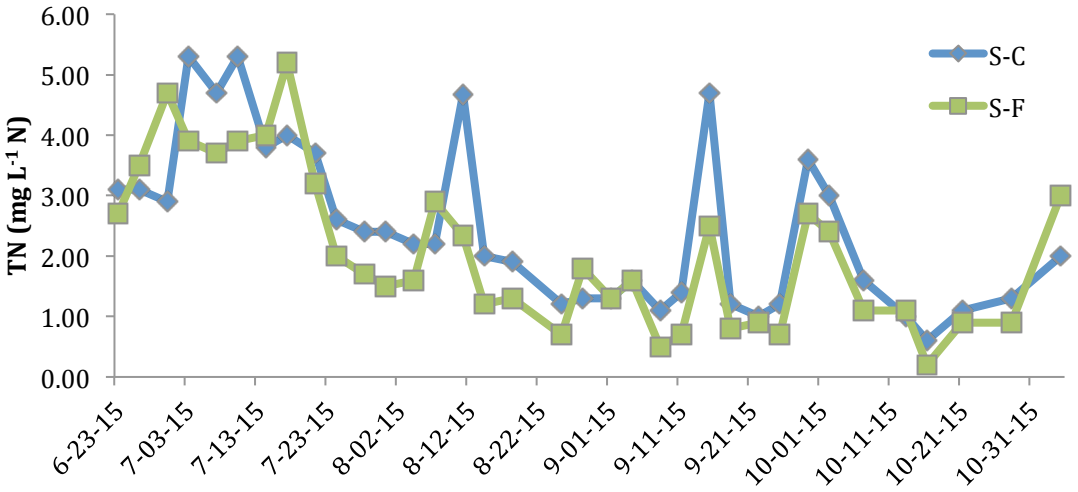

Figure 4.8 Temporal variation of total nitrogen (TN) in the 2015 storage treatment

4.3.2 Facultative Treatment

There was no significant difference in nutrient removal efficiency in the facultative treatment between the F-C and F-FTW cells (NH_4^+, NO_3^-, RP, TP, and TN) ($p>0.05$) (Table 4-4). Influent NH_4^+ varied during the 2015 season. There was an initial sharp decrease from 1.32mg L^{-1} on June 19 to 0.02mg L^{-1} NH_4^+–N on July 21, making the influent concentration at times lower then the effluent from the treatment cells. From July 24, onwards, the concentration of the influent was higher than that exiting the treatment cells, reaching a maximum of 1.72mg L^{-1} on Aug. 25. NH_4^+ concentration decreased rapidly in both the F-C and F-FTW cells, and from July 17 onwards the concentration never increased above 0.1mg L^{-1} (Figure 4.9a).

Influent NO_3^- generally increased over the season, from approximately 0.3 to 0.8mg L^{-1} as NO_3^-–N. Concentration in the F-C cell increased to a maximum 0.74mg L^{-1} on June 30, then rapidly decreased, falling below the detection limit (0.23mg L^{-1} NO_3^-–N) by July 17. In the F-FTW cell, concentration followed a similar trend, except it reached a higher

maximum of 1.16mg L^{-1} on June 30. It then decreased below the detection limit by July 17 (Figure 4.9b). TN concentration was highly variable, with no clear trend for the influent TN concentration. Concentration in both the F-C and F-FTW cells displayed a general decrease from 3-4mg L^{-1}, down to 0.5mg L^{-1} by the end of the season (Figure 4.10). Average organic nitrogen was 1.6±0.84, 1.6±1, and 1.8±0.94 mg L^{-1} N for the F-C, F-FTW, and influent respectively. There was no significant difference between the F-C and F-FTW cells ($p>0.05$)

Average influent RP concentration was 2.6±0.78mg L^{-1}. RP concentration in the influent was initially around 3mg L^{-1} but fluctuated between a minimum of 0.96mg L^{-1} (Oct. 21) and a maximum of 3.82mg L^{-1} (Aug. 4). RP within the cell decreased from approximately 1 to 0.2mg L^{-1} over the season (Figure 4.9c). TP concentration in the influent followed a similar trend to RP removal. Concentration of the influent increased from approximately 3 to 5mg L^{-1} by mid-August, before decreasing to 1.15mg by Oct 28, 2015. TP concentration in both F-C and F-FTW cells decreased from approximately 1.2 to 0.3mg L^{-1} (Figure 4.9d). On average, TP was 42±17%, 39±15%, and 18±10% higher RP in the F-C, F-FTW, and influent respectively. There was no significant difference between the FTW and control ($p>0.05$)

Table 4-4 Average nutrient removal efficiency (%) (± standard deviation) in the 2015 facultative treatment, and *p*-values from the Mann-Whitney U-Test

	F-C (%)	F-FTW (%)	*p*-value
RP	76±8.6	75±7.7	0.392
TP	66±9.4	65±10.6	0.955
NH_4^+	51±86	71±47	0.465
NO_3^-	63±62	44±102	0.767
TN	33±46	34±41	0.906

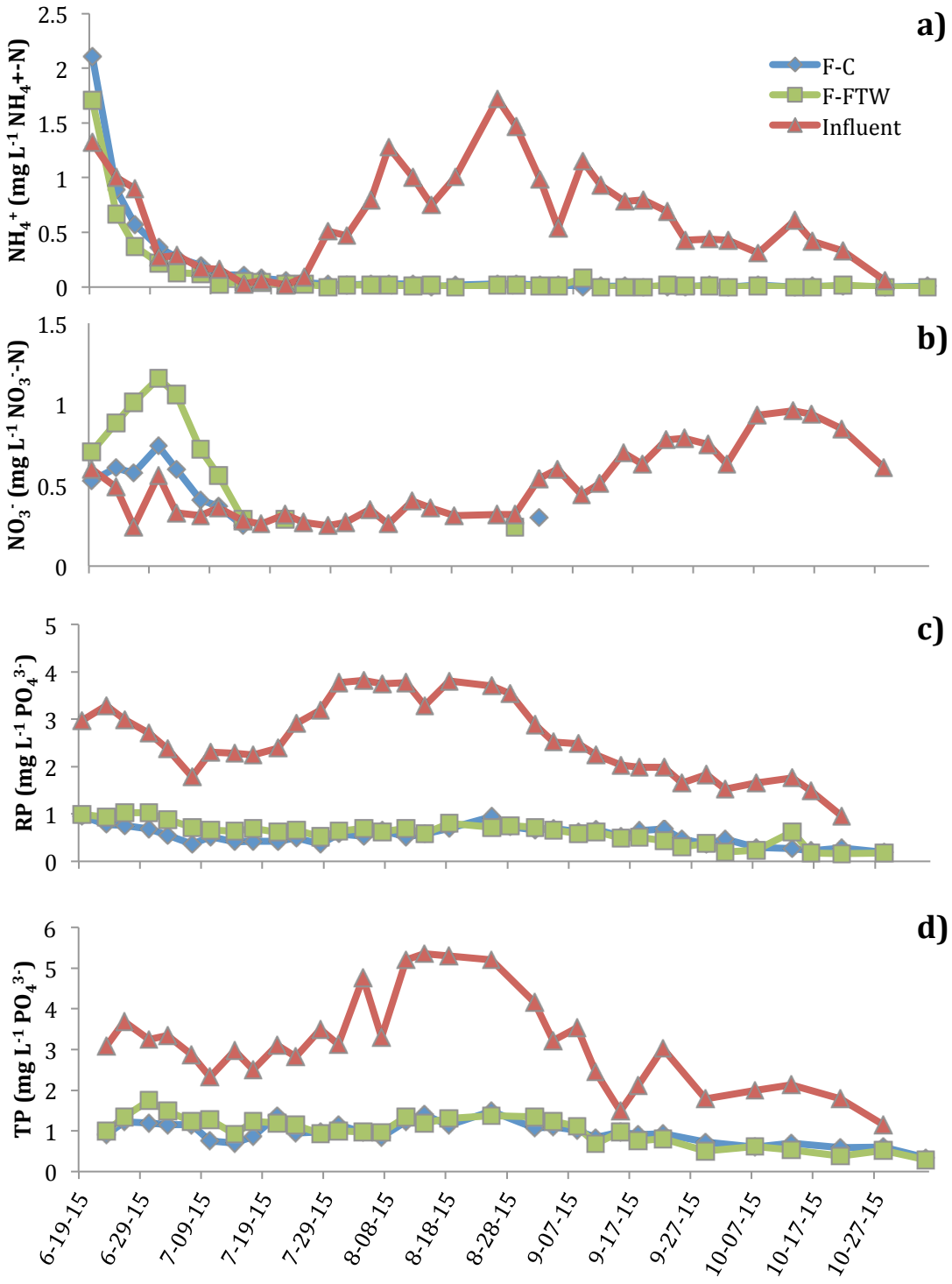

Figure 4.9 Temporal variation of nutrient concentration for Ammonia, nitrate, reactive phosphorus (RP), and total phosphorus (TP) for the 2015 facultative treatment

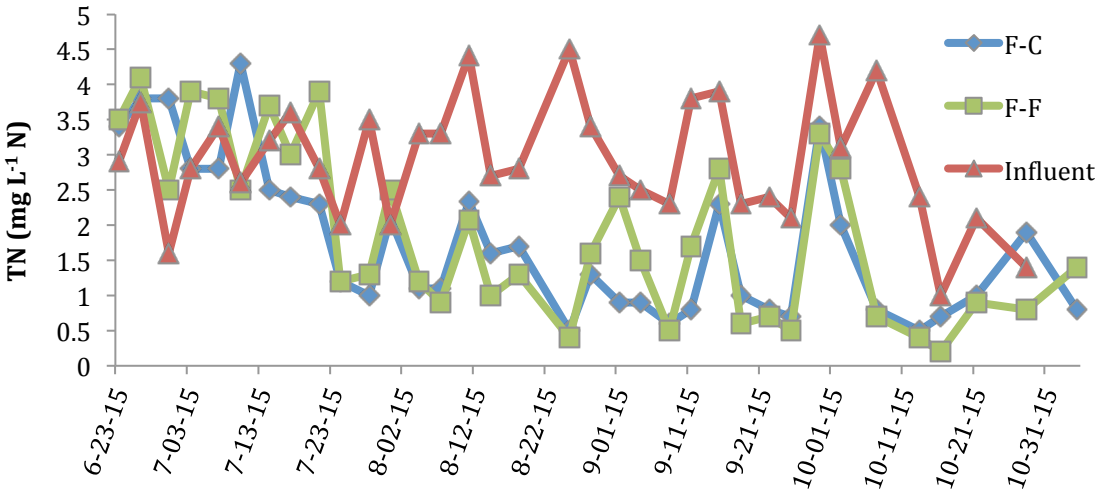

Figure 4.10 Temporal variation of total nitrogen (TN) in the 2015 facultative treatment

4.4 Nutrient Removal 2016

When determining the nutrient removal efficiency for the 2016 season, nutrient concentration in the stage-1 and stage-2 effluent was compared to influent nutrient concentration from the primary treatment cells on site. As such, the removal efficiency is always higher in stage-2, since it includes treatment in stage-1. In stage-1, efficiency in the S1-FTW cell was significantly higher than the control for NH_4^+ and TN ($p<0.001$ for both). There was no significant difference for RP, TP, and NO_3^-. In stage-2, the S2-FTW had significantly greater efficiency for all nutrient parameters, RP, TP, NH_4^+, NO_3^-, and TN (Table 4-5).

Table 4-5 Average nutrient removal efficiency (%) (± standard deviation) in the 2016 two-stage treatment, and *p*-values from the Mann-Whitney U-Test (p<0.05 denotes significance)

	Stage 1			Stage 2		
	S1-C	S1-FTW	*p*-value	S2-C	S2-FTW	*p*-value
RP	46±33	52±22	0.447	58±28	78±13	<0.001
TP	50±24	57±17	0.152	59±25	76±11	0.003
NH_4^+	51±18	72±14	<0.001	86±14	94±14	<0.001
NO_3^-	−34±76	−56±121	0.995	−111±157	−6±120	<0.001
TN	48±19	67±15	<0.001	76±13	85±8	0.001

As shown in Figure 4.11a, NH_4^+ was consistently lower in the S1-FTW cell (average concentration, 2.63mg L^{-1}) when compared to the S1-C cell (average concentration 4.88mg L^{-1}). In the S2-FTW cell, NH_4^+ concentration remained below 0.3mg L^{-1} from June 20 to the end of the season. NH_4^+ in the S2-C cell was consistently higher until September 1, when it also dropped below 0.3mg L^{-1} and remained low for the remainder of the season. Average incoming NH_4^+ concentration from the primary clarifiers was 10.81±3.6mg $^{L-1}$ NH_4^+-N and fluctuated between a maximum of 17.3mg L^{-1} on June 30, and a minimum of 4.6mg L^{-1} on August 4 (Figure 4.11a).

In general, NO_3^- concentration in all cells in both stage-1 and stage-2 treatment was higher than that of influent, except in the S2-FTW cell after July 31. The average NO_3^- concentration in the influent was low, only 0.34±0.12 mg L^{-1}. NO_3^- concentration in the S1-C and S1-FTW cell was highly variable, fluctuating between 1.0 and 0.15mg L^{-1} in the S1-C cell, and 1.16 and 0.10mg L^{-1} in the S1-FTW cell. Concentration in the S2-FTW cell reached a maximum of 1.16mg L^{-1} on June 23, before decreasing below 0.45 for the remainder of the season. The S2-FTW cell produced significantly less NO_3^- (+6.06% increase) than the S2-C cell (+112% increase) on average (Figure 4.11b). TN

concentration was consistently lower in the S1-FTW and S2-FTW cells when compared with their respective controls. Average influent TN concentration was 15.05±4.6mg L^{-1}, and fluctuated between a maximum of 22.8mg L^{-1} on July 7, and a minimum of 7.3 on Sept 16 (Figure 4.13). Average organic N content (TN minus NO_3^-–N and NH_4^+-N) was 1.9±1, 1.8±1.3, 1.8±1.3, 1.7±1.1, and 3.6±1.5mg L^{-1} N for S1-C, S1-FTW, S2-C, S2-FTW, and the influent respectively. There was no significant difference in organic nitrogen concentration between either the S1-C and S1-FTW cells, and the S2-C and S2-FTW cells ($p>0.05$).

RP concentration in the S1-FTW cell was lower than the S1-C cell from June 16 to August 22, after which the outgoing RP concentration was often higher in the S1-FTW cell. RP in the S2-FTW was consistently lower than the S2-C, except after Sept 22, 2016 when concentration in both cells was very low (<0.5mg L^{-1}). Average incoming RP concentration in the influent was 4.71±1.67mg L^{-1} as PO_4^{3-}, though it varied considerably during the season, reaching a maximum of 7.89mg L^{-1} on June 30, 2016, and a minimum of 2.01mg L^{-1} on Aug 4 2016 (Figure 4.12a). TP concentration followed a trend similar to RP. Average incoming TP concentration was 6.91±1.91mg L^{-1} PO_4, and reached a maximum of 9.96mg L^{-1} on June 27, 2016, and a minimum of 3.87mg L^{-1} on Aug 4, 2016 (Figure 4.12b). For stage 1, TP was on average 25±11% and 22±13% higher than RP for the S1-C and S1-FTW cells respectively, and there was no significant difference ($p>0.05$). For stage 2, TP was on average 30±12% and 41±12% higher than RP for the S2-C and S2-FTW cells respectively. This percentage was found to be significantly higher in the S2-FTW cell than the S-C cell ($p<0.001$). In the influent TP was on average 31±9.3% higher than RP.

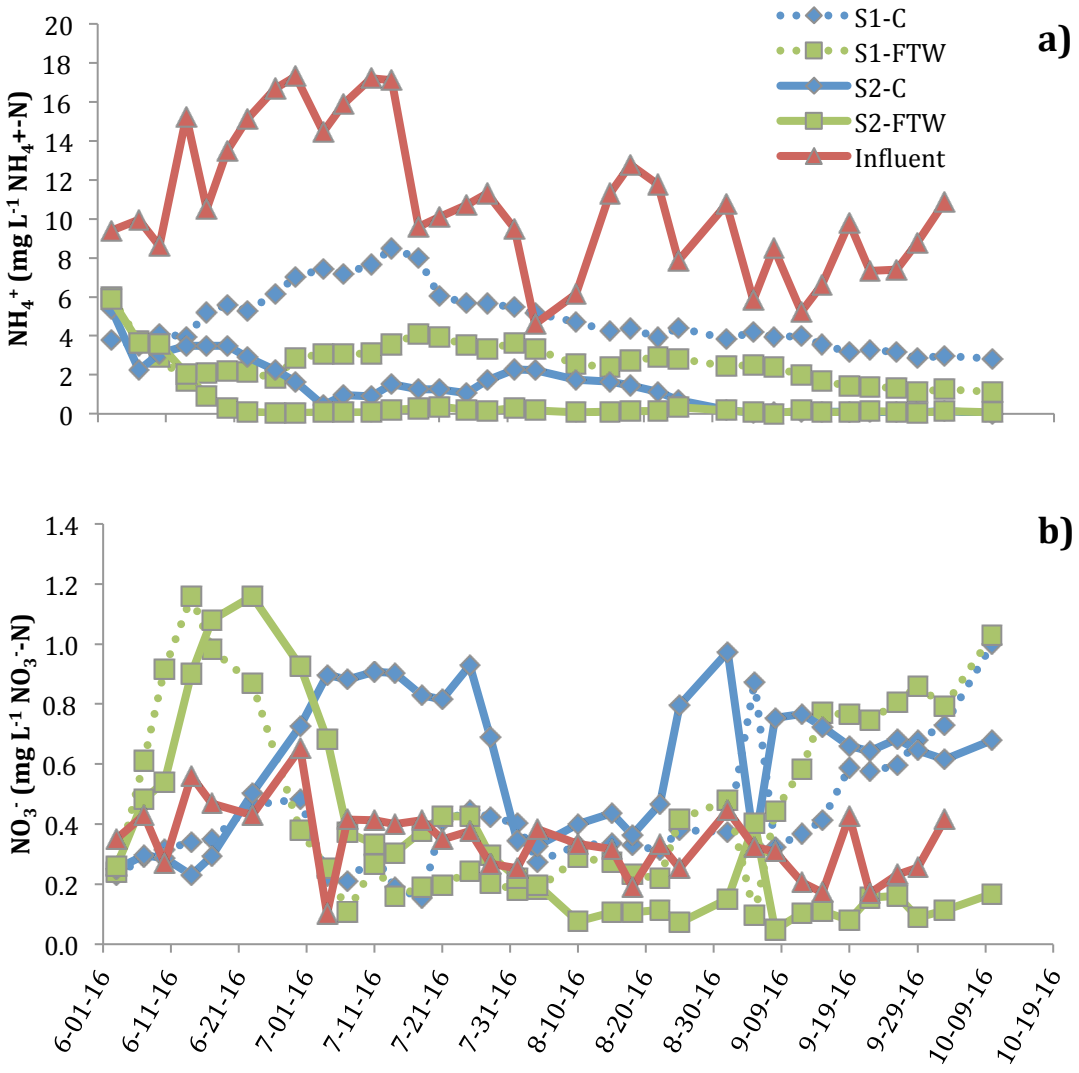

Figure 4.11 Temporal variation of nutrient concentration for ammonia (a) and nitrate (b) for the 2016 two-stage treatment

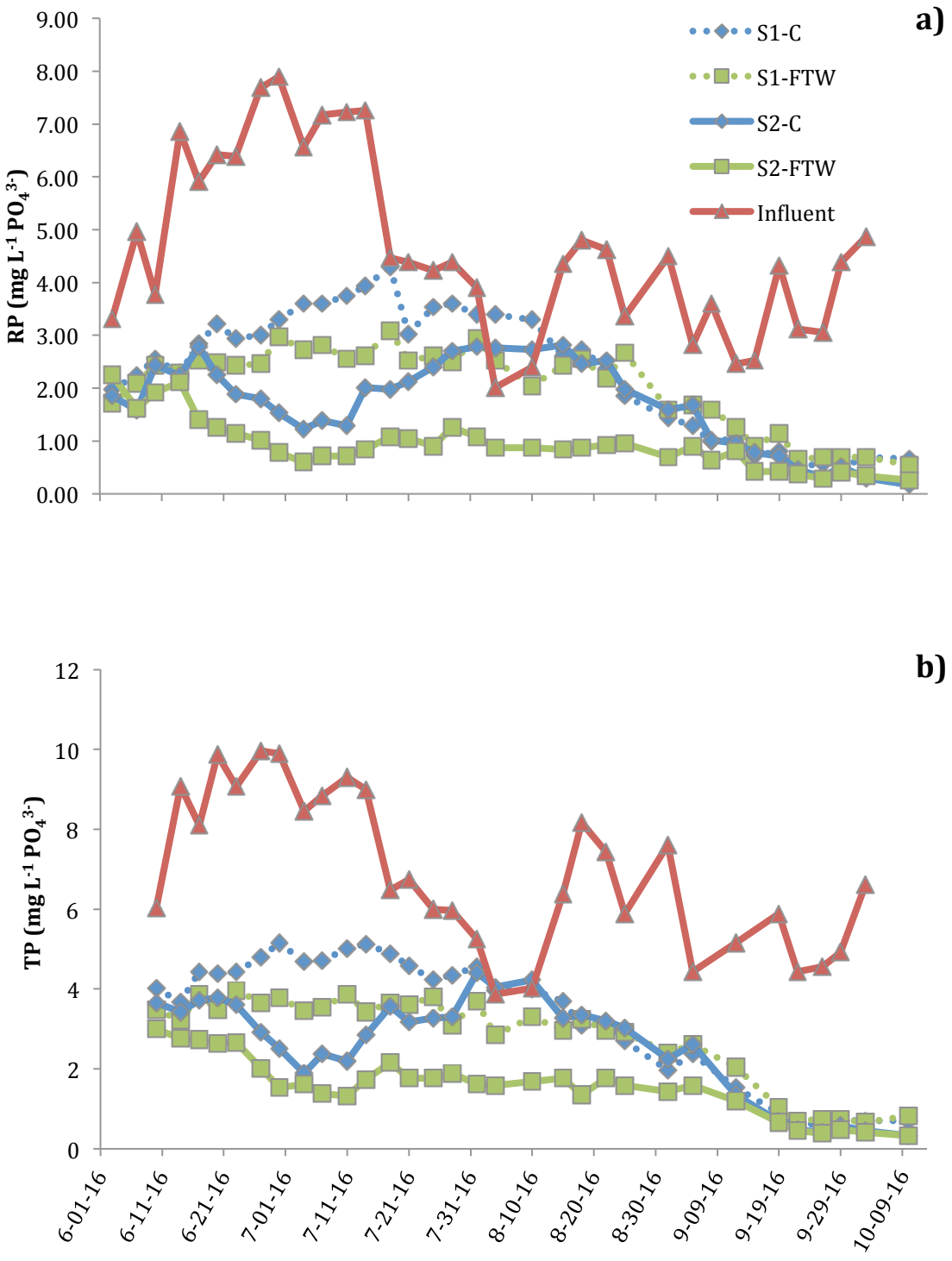

Figure 4.12 Temporal variation of nutrient concentration for reactive phosphorus (RP) (a) and total phosphorus (TP) (b) for the 2016 two-stage treatment

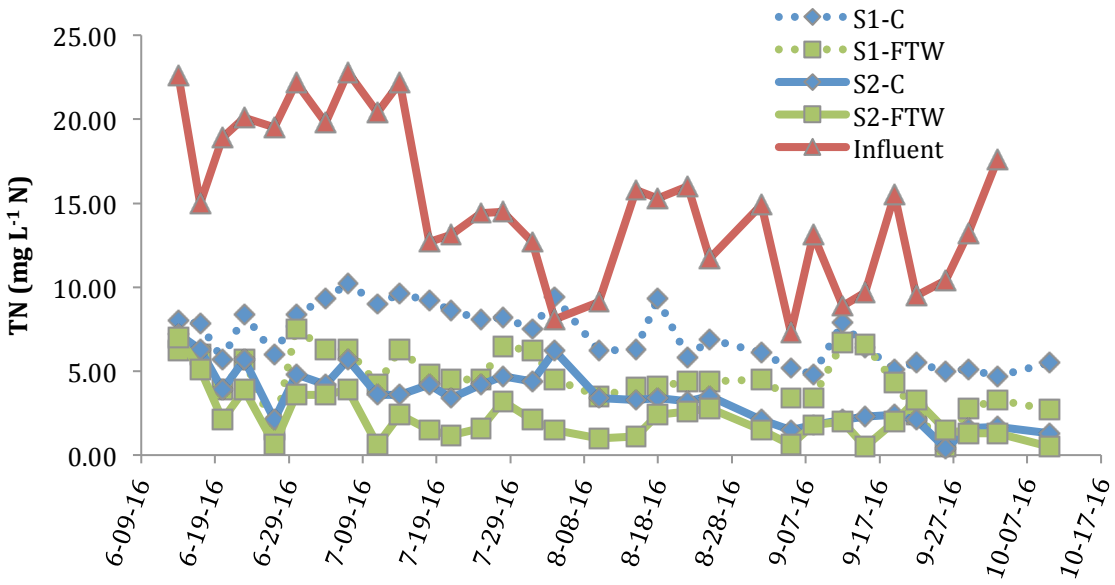

Figure 4.13 Temporal variation of total nitrogen (TN) in the 2016 two-stage treatment

4.5 BOD

For the 2015 season, there was no significant difference between the FTW and control cells for either the storage or the facultative treatment. For the storage treatment, average BOD_5 was 3.13±2.83 and 2.91±2.63mg L^{-1} for the S-C and S-FTW cells respectively. Initial BOD dropped rapidly from >12 to <4mg L^{-1} in both cells by July 7. It remained steady for the remainder of the season (Figure 4.14a). For the facultative treatment average incoming BOD_5 was 12.3±8.65mg L^{-1}, fluctuating between of maximum of 43.5 on Sept. 1, and a minimum of 2.3 on July 10. Average BOD_5 in the effluent was 3.32±1.65 and 3.66±3.46mg L^{-1} for the F-C and F-FTW cells respectively. Effluent from both cells remained below 4mg L^{-1} from June 30 until the end of the season (Figure 4.14b).

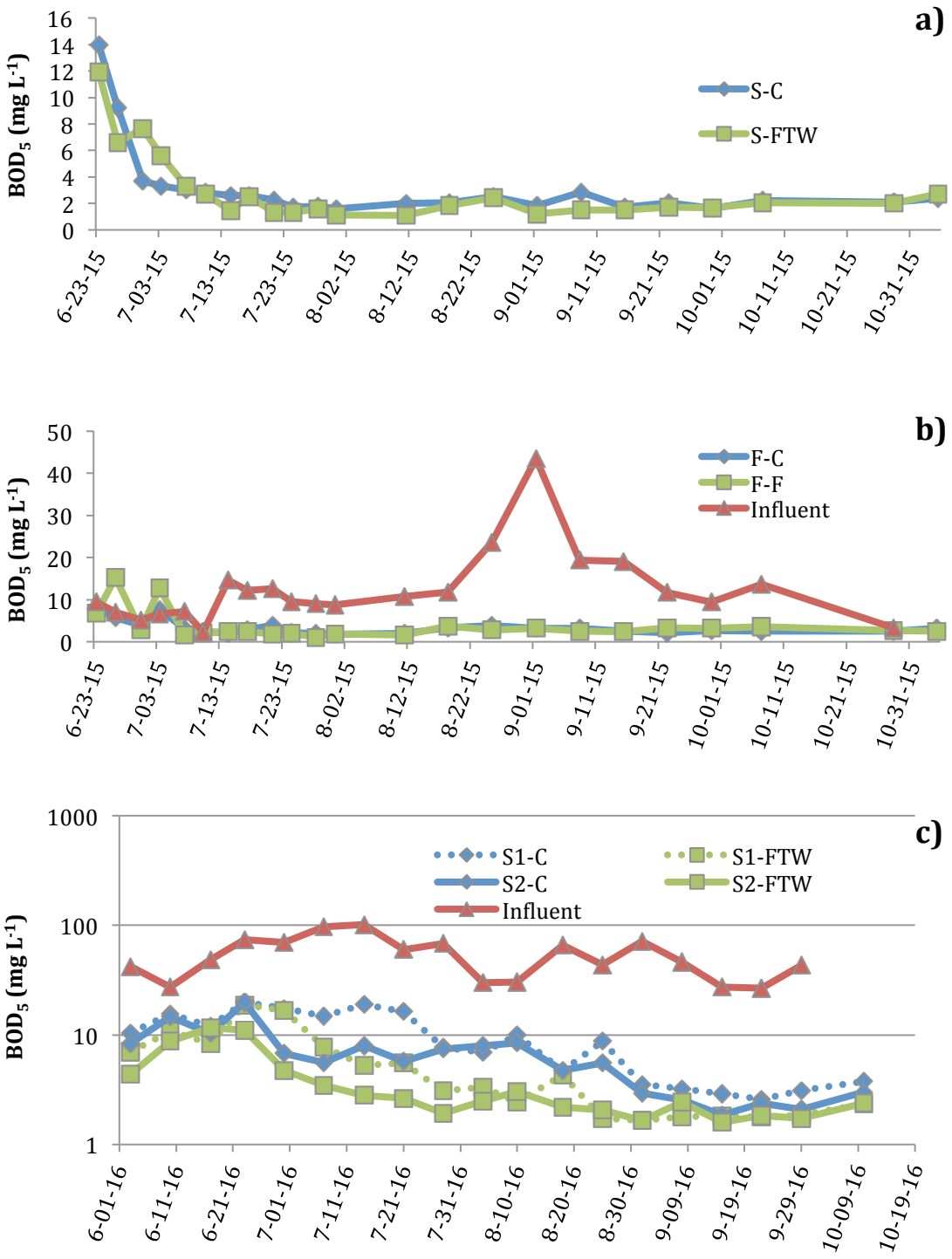

Figure 4.14 Temporal variation in 5-day biochemical oxygen demand (BOD_5) concentration in the storage treatment (a), the facultative treatment (b), and the two-stage treatment (c). Bottom figure graphed on a logarithmic y-axis.

For the 2016 season, BOD_5 removal efficiency was significantly higher in both the S1-FTW and S2-FTW cells relative to the control ($p<0.05$). For stage-1, removal was 80%±13 and 88%±9.7 for the S1-C and S1-FTW cells respectively. For stage-2, removal was 85%±13 and 92%±7.9 for S2-C and S2-FTW cells respectively. Average influent BOD_5 was 54±23mg L^{-1}, and fluctuated between a maximum of 101.7 on July 16 and a minimum of 26.7 on Sept. 22 (Figure 4.14c).

4.6 TSS and Turbidity

For the 2015 season, there was no significant difference in TSS removal in either the storage or the facultative treatment. In both treatments, TSS decreased rapidly and remained below 5mg L^{-1} for the remainder of the season (Figure 4.15 a and b). Average TSS in the storage treatment was 4.63±8.17 and 4.37±7.6mg L^{-1} for the S-C and S-FTW cell respectively. Average TSS in the facultative treatment was 2.34±0.7 and 1.63±0.46mg L^{-1} for the F-C and F-FTW cells respectively. Average incoming TSS was 6.63±2.79mg L^{-1}. For the 2016 season (Figure 4.15c) there was no significant difference between the FTW and control cells in both stages. Average incoming TSS was 47±24mg L^{-1}. For stage-1, TSS removal was 88±7.4 and 92±5.6mg L^{-1} for the S1-C and S1-FTW cell respectively. For stage-2, TSS removal was 88±11 and 91±5.6mg L^{-1} the S2-C and S2-FTW cells respectively. Average TSS concentration in the effluent of all cells was ≤5mg L^{-1}.

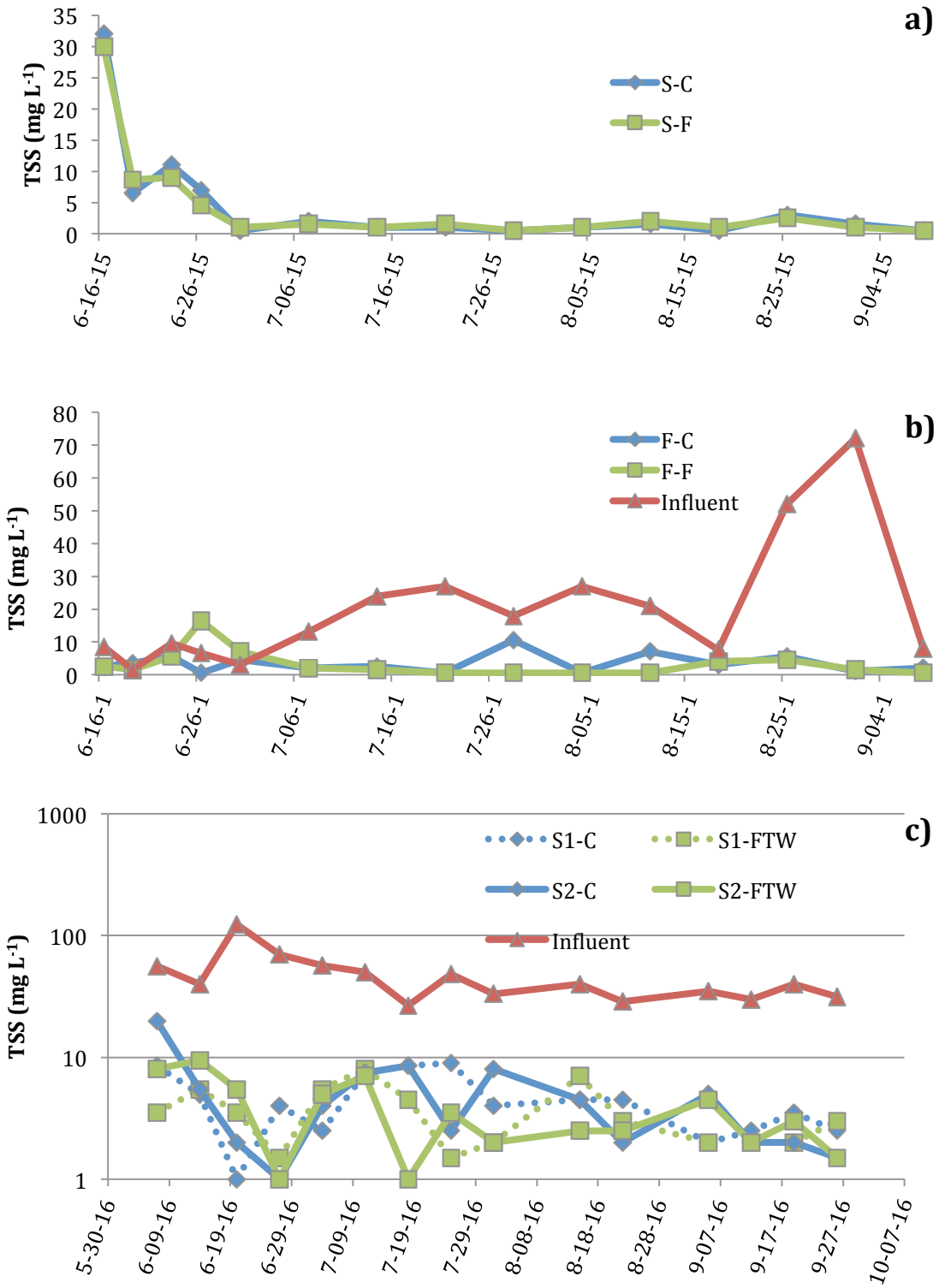

Figure 4.15 Temporal variation in total suspended solids (TSS) concentration in the storage treatment (a), the facultative treatment (b), and the two-stage treatment (c). Bottom figure graphed on a logarithmic y-axis.

For the 2015 season there was no significant difference in turbidity for either the storage or the facultative treatment. Similar to TSS, in both treatments turbidity dropped rapidly and remained <5 NTU for the remainder of the season. Average turbidity in the storage treatment was 4.04±1.9 and 3.68±3.8NTU for the S-C and S-FTW cells respectively. Average turbidity in the facultative treatment was 2.34±0.7, 1.63±0.46 and 6.63±2.79 NTU for the F-C, F-FTW, and influent respectively. For the 2016 season, the effluent leaving the S1-FTW cell was significantly lower than the S1-C cell ($p<0.05$). Average removal efficiency was 93%±3.1 and 96%±1.8 for the S1-C and S1-FTW cell respectively. In stage-2 there was no significant difference as average removal was 97%±1.4 and 98%±2.1 for the S2-C and S2-FTW cells respectively. Average incoming turbidity was 81±35 NTU.

4.7 Fecal Coliform

The two-stage treatment saw a 3-log reduction of fecal coliform bacteria in S1-C and S1-FTW, and a 4-log reduction in S2-C and S2-FTW. Since average incoming fecal coliform bacteria was 20,000±22,000cfu $100ml^{-1}$, there was an almost complete elimination of bacteria within both stage-2 cells. Influent concentration varied between a maximum and minimum of 80,000 and 1,900cfu $100ml^{-1}$ on Aug. 10 and Sept. 1 respectively. The average fecal coliform concentration in the effluent of the S1-C and S1-FTW cells was 20±40 and 12±26cfu $100ml^{-1}$ respectively, but there was no significant difference ($p>0.05$). Average concentration in the effluent of the S2-C and S2-FTW cells was 0±0.3 and 3±9mg $100ml^{-1}$ respectively, with no significant difference ($p>0.05$). There were 16 samples taken from Aug. 10 to Oct. 10, and of this 14 samples in the S2-C

cell revealed no fecal coliforms compared with 11 samples within the S2-FTW cell. Two unusually high counts were found in S2-FTW cell on Sept. 26 and 29, where the values were 32 and 17cfu 100ml^{-1} respectively (Figure 4.16).

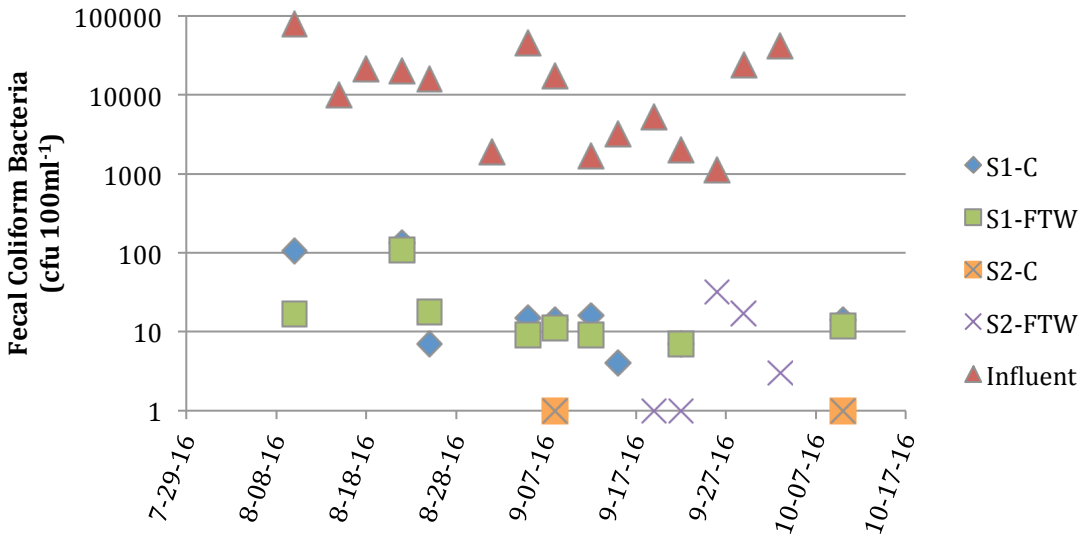

Figure 4.16 Temporal variation of Fecal Coliform Bacteria in the 2016 two-stage treatment, with a logarithmic y-axis. Values of zero are not shown.

Chapter 5: DISCUSSION

5.1 Factors Affecting Macrophyte Performance and Aquatic Vegetation Growth

Since macrophytes are arguably the most prominent and critical component within a FTW, ensuring the successful establishment and continued survival of macrophytes in the cold, semi-arid continental climate of Southern Alberta (*BS*k under the Koppen climate classification) was vital to the study. In 2015, all macrophytes successfully established over a two-month period in the greenhouse, and survived the transfer to the pilot-scale treatment system in the field. Interestingly there were notable visual differences in growth between S-FTW and F-FTW cells. By September 2015, the macrophytes grown in the S-FTW cell appeared larger and greener than those grown in the F-FTW cell. Macrophytes in the F-FTW cell had yellow/pale-green leaves, and appeared noticeably less healthy than those in the S-FTW cell.

Yellow leaves are indicative of chlorosis, which is a condition where macrophytes produce insufficient chlorophyll, and can be caused by a variety of factors. The most likely factors in this experiment were nitrogen limitation (Ramirez-Santiago et al, 2012) and high pH (Loeppert & Hallmark, 1985). Nitrogen limitation affects the production of proteins vital to the functioning of macrophytes, while high pH reduces the bioavailability of metals, especially Fe, which interferes with the production of chlorophyll. From late-July onwards in the 2015 field season, NO_3^- concentration was consistently below the detection limit, and NH_4^+ concentration was between 0-0.03mg L^{-1} in both the F-FTW and the S-FTW cells. From Aug. 18, onwards the pH was above 9.5 in the F-FTW cells, which would make Fe increasingly less soluble (Sawyer et al, 1994).

Considering that the S-FTW cell had very low nitrogen levels but pH level in the S-FTW cell was not as high as that in the F-FTW cell, high pH could be the primary cause of macrophyte stress observed in the F-FTW.

Despite the stress occurring in the late season in the F-FTW, both FTWs had high overwintering success. Only one *Carex a*. specimen failed to return in spring in the F-FTW cell. This observation indicates that both *Carex a*. and *Scirpus m*. are suitable for year-round use in FTWs in the Southern Alberta climate. Winter temperatures dropped below -30°C and ice thickness within the cells was above 15cm in January. Buoyancy in the platforms themselves did not appear to be impacted by the freeze-thaw cycle. If used in a full-scale system, these results show that little maintenance would be needed to ensure the macrophytes remain healthy, and the platform remains functional.

For the 2016 season there was little difference between the growth of the macrophytes in the S1-FTW and S2-FTW cells. Dry biomass in each FTW was similar (~3.9kg and ~4.2kg for the S1-FTW and S2-FTW dry biomass respectively), despite an observed difference in height. Macrophytes at the end of 2016 field season were taller in the S1-FTW cell. It should be note that the S-FTW from 2015 was deployed into the S1-FTW cell, and the F-FTW cell from 2015 was deployed into the S2-FTW cells. Since the S-FTW visibly outperformed the F-FTW in 2015, a fair qualitative comparison of macrophytes growth between S1-FTW and S2-FTW cannot be made, since the S1-FTW may simply have begun the 2016 season with more established/mature macrophytes. There was no visible indication of chlorosis of macrophytes in the 2016 season. Furthermore, the macrophytes were considerably larger in the 2016 season than those in the 2015 season. These observations demonstrate the quick maturity of macrophytes due

to an already established a root/shoot system in the 2015 season, and higher nutrient concentrations in the 2016 season. A cross-section of macrophytes harvested from the S1-FTW cell in October 2016 revealed a dense root network deeply embedded in the plastic matrix (Figure 5.1). Unfortunately, accurate determination of macrophyte root biomass was not possible, since it was difficult to safely extract the roots from the FTW.

Figure 5.1 Results from FTW matrix analysis. 1) Roots from Carex a. extending through the plastic matrix, 2) Roots of Scirpus m. embedded within the matrix, and 3) cross sections of the a Scirpus m. shoot, displaying the porous aerenchyma tissue

Carex a. had a phosphorus concentration of 2.6mg g^{-1} of dry biomass, which is similar to results from previous studies investigating emergent macrophytes grown in high-nutrient wastewater (Gottschall et al, 2007; Hubbard et al, 2004). Typically, phosphorus content within the shoots of most species ranges from 1-4mg g^{-1} (Kadlec & Wallace, 2008). However, *Scirpus m.* had relatively high phosphorus content at 5.3mg g^{-1}

and the reason for this disparity between the two macrophyte species is uncertain. A limitation of the macrophyte sampling in 2016 was it occurred in October, by which time the ambient temperature had already fallen below 0°C on several occasions (Environment Canada). Emergent macrophytes in cold climates become dormant in response to cold temperatures (Werker et al, 2002) and redirect phosphorus from the shoots into the roots (Zhou & Wang, 2010). As such, the phosphorus content that was measured in this study, is likely lower than if the shoots had been sampled in mid-summer. It is also likely that different species redirect phosphorus to the roots at differing rates, or at different times of the season. Therefore *Carex a.* may have already redirected more phosphorus to the roots or entered dormancy earlier relative to *Scirpus m.*

TP removed in the two-stage FTW system was calculated to be 500g (0.76 removal efficiency multiplied by the total TP load to the FTW cells of 658g (Table 5-1)). Since 32.6g of phosphorus was estimated to be located in the macrophyte shoot biomass, about 6.5% of the phosphorus load was redirected from the wastewater into the above-mat biomass. This value is located in the range of 5-10% of the TP load that is seen in other treatment wetland studies (Chang et al, 2012; Kadlec & Wallace, 2008; Kim & Geary, 2001; Li et al, 2012), including in a pilot-scale wastewater treatment FTW with aeration (Wu et al, 2006).

The macrophytes did not appear to be stressed by the high concentration of ammonia (influent concentration >15mg L^{-1} NH_4^+-N at times), or the low dissolved oxygen (<0.5mg L^{-1} O_2 at times in stage-1). During summer 2016, other macrophyte species, many of which were terrestrial but not emergent macrophytes, established on the FTW platforms in both the S1-FTW and the S2-FTW cells. Previous FTW studies have used

shading in FTW and control cells to prevent the growth of algae or submerged macrophytes which may enhance treatment in addition to the FTW (Tanner & Headley, 2011; Wu et al, 2006). However in this study, both floating algae and submerged vegetation which established in the 2015 F-FTW cell, and 2016 S2-FTW cell were purposely not removed in order to better mimic the conditions of a stabilization pond (Figure 5.2). During the study period there was significant growth of algae and aquatic macrophytes in the nearby facultative lagoon. Similar methodology was used by Chang et al. (2012) in which algae was allowed to proliferate to better mimic stormwater pond conditions. In this study, non-FTW vegetation would certainly have provided additional treatment in both the control and FTW cells, and the control cell appeared to have more submerged vegetation. This can be ascribed to the absence of shading created by a FTW.

The floating algae may have both improved and hindered treatment within the cells. The algae would have assisted with nutrient uptake, while conversely the thick mat likely inhibited atmospheric oxygen transfer that could have impacted critical chemical reactions like nitrification. In the storage treatment cells in 2015, and the stage-1 cells in 2016 there was no algae and/or macrophyte growth aside from periphyton growth on the walls of the sea-container and the FTW platform. The lack of growth in the storage treatment could be explained by an absence of additional wastewater (and therefore nutrients) during the study period, which likely limited algae introduction. The water was also >2m deep and lacked a clay layer at the base, which prevented the establishment of submerged vegetation. However in the stage-1 cell, low dissolved oxygen (less than 1mg L^{-1}) for most of the growing season most likely prevented the establishment of submerged vegetation and suspended algae.

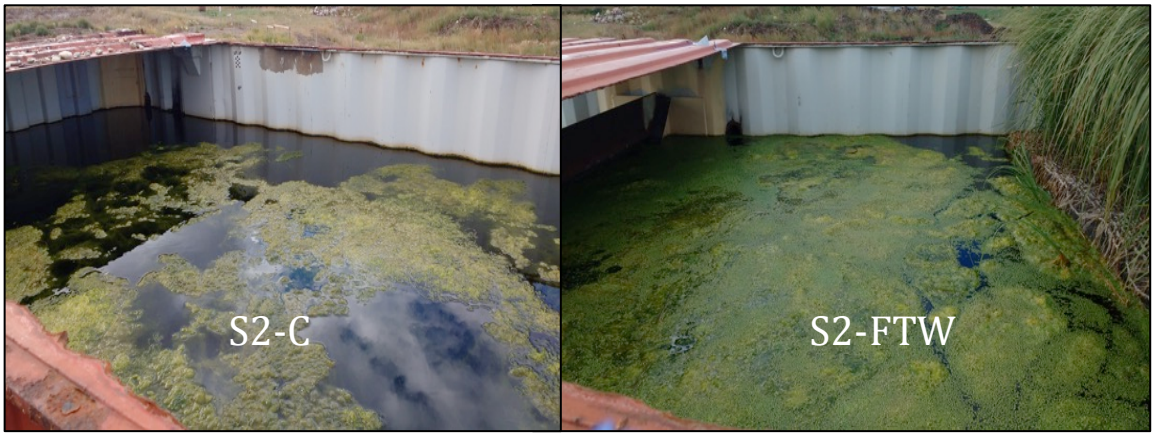

Figure 5.2 Floating algae present in the S2-C and S2-FTW cells. Notice the S2-FTW had considerably more coverage, and included a large covering of duckweed (which was not present in S2-C). Photo taken on Sept. 12, 2016.

5.2 Impact of FTW on Basic Physicochemical Properties

5.2.1 Variation in Field Seasons

FTWs appeared to alter physicochemical properties in the treatment cells, and the effect varied temporally across the two field seasons. When comparing the mean value for the entire season, there appeared to be no significant difference in most parameters in both 2015 and 2016 seasons when comparing FTW cells with control cells. However there also existed obvious differences between the cells at some time intervals for specific parameters. For both years, the FTW cell generally had slightly lower temp, pH and EC (in 2016). Similar results have been reported in several previous studies (Borne, 2014; Van de Moortel et al, 2010; Wang et al, 2012; White & Cousins, 2013; Yang et al, 2008; Zhou & Wang, 2010). Interestingly, in 2015 the FTW cells had higher average DO

concentration than the control cells, while the opposite results were obtained in 2016. The results from 2016 for DO are more similar to those seen in previous studies.

The difference in DO is interesting because it varied between the years, and between stage-1 and stage-2. In all cells, the FTW cells can be expected to inhibit oxygen transfer from the atmosphere, and thus reduce oxygen regeneration (Borne et al, 2014b). While wetland macrophytes are known to leach oxygen into the rhizosphere, previous research has shown that the rate of transfer is heavily dependant on oxygen demand within the water, and oxygen is often consumed before exiting the rhizosphere (Bezbaruah & Zhang, 2005); Thus macrophytes should not be expected to significantly contribute directly to DO concentration in the water. For the 2016 season, incoming DO in the effluent was low, on average 0.5±0.5mg L^{-1}, since it was coming from the primary cells, which had a high BOD (>50mg L^{-1}) and no artificial aeration. The DO was significantly lower in the S1-FTW cell than the control, for the entire season. To explain, the most important parameter may be BOD, which is a measure of the oxygen demand within the water.

The average influent BOD entering the stage-1 sea-container was 54mg L^{-1}, and was reduced by on average 80%±13 and 89%±10 for the S1-C and S1-FTW cell respectively. While this means that BOD, and therefore the oxygen demand, within the effluent of the FTW cell was lower, it requires that more oxygen be consumed as the wastewater is treated (Zhou et al, 2012). This, combined with the inhibition of O_2 transfer from the atmosphere by the FTW, most likely contributed to a reduced DO in the S1-FTW cell.

Water entering stage-2 already had a relatively low BOD, 9.6±6 and 5.6±5mg L^{-1} BOD_5 entering the S2-C and S2-FTW cell respectively. Figure 4.4b highlights that DO

was not consistently lower in either the S2-C and S2-FTW cell. In fact, from July 28 to August 22 the DO was consistently higher in the FTW cell, but from August 25 onwards the DO was consistently lower. BOD within the S2-FTW cell only dropped from 5.6 to 3.8mg L^{-1} upon reaching the outlet, a difference of only 1.8mg L^{-1}. In the S2-C cell, it was reduced from 9.6 to 6.73mg L-1, a difference of 2.9mg L^{-1}. This difference in oxygen consumption could have contributed to the periodically higher DO within stage-2, and could explain the higher average DO in the storage and facultative treatments in 2015.

From late summer onwards, both the S2-C and S2-FTW cells had high coverage by floating algae (Figure 5.2), but the S2-FTW cell appeared to have higher concentrations, in addition to supporting a large mat of common duckweed (*Lemna turionifera*), which was not present at all in the S2-C cell. This factor, combined with the FTW platform itself, would have created a significant barrier to O_2 transfer, and is the most likely cause of the late-season lower DO. There were consistent supersaturated (>100% O_2 saturation) DO conditions in both cells of the facultative treatment in 2015, and periodically in the S2-C and S2-FTW cells in 2016. Supersaturation is created by the presence of submerged vegetation and floating algae in the cells. Unlike emergent macrophytes, submerged vegetation releases O_2 directly into the water when undergoing photosynthesis, and is the primary cause of supersaturation (Krause-Jensen & Sand-Jensen, 1998). In late summer, and during warm days, it was common for cells with a low BOD (<10mg L^{-1}) to experience supersaturated conditions.

In general, pH was lower or slightly lower in the FTW cells than that in the control cells. In addition, a significant difference in pH between the FTW and control cells was detected in the 2015 storage treatment and stage-2 of the 2016 two-stage treatment. This

is consistent with previous research that demonstrated that FTWs tend to lower pH in the water column beneath the root system (Borne, 2014; Borne et al, 2014a; Van de Moortel et al, 2010; White & Cousins, 2013) due to the release of organic acids from the macrophyte rhizosphere (Neori et al, 2000). In the 2015 season, the average pH of the influent was significantly lower than the effluent of the F-C and F-FTW cells (8.7±0.5, 9.5±0.5, and 9.4±0.6 for the influent, F-C and F-FTW cells, respectively). Similarly, in the 2016 two-stage treatment, the influent pH was significantly lower then the S2-C and S2-FTW cells (7.9±0.2, 8.7±0.3, and 8.4±0.3 for the influent, S2-C and S2-FTW cells respectively).

Algae and submerged vegetation can increase pH through carbonate ion (CO_3^{2-}) consumption during photosynthesis (Krause-Jensen & Sand-Jensen, 1998). Figure 5.3, demonstrates a high correlation between DO and pH in the S2-C and S2-FTW cells (R^2 equals 0.72 and 0.9 respectively) as an example. Supersaturated conditions (likely created by submerged and floating vegetation) in the late summer in F-C and F-FTW cells in 2015 and the S2-C and S2-FTW cells in 2016, likely contributed to the high pH (>9). In the S1-C and S1-FTW cells the correlation between DO and pH is very weak (R^2<0.1 for both cells), indicating that when DO is lower and supersaturation does not occur, there is a reduced relationship between DO and pH. The lack of submerged macrophytes and floating algae in stage-1 means there was little photosynthesis (relative to stage-2) occurring within the water.

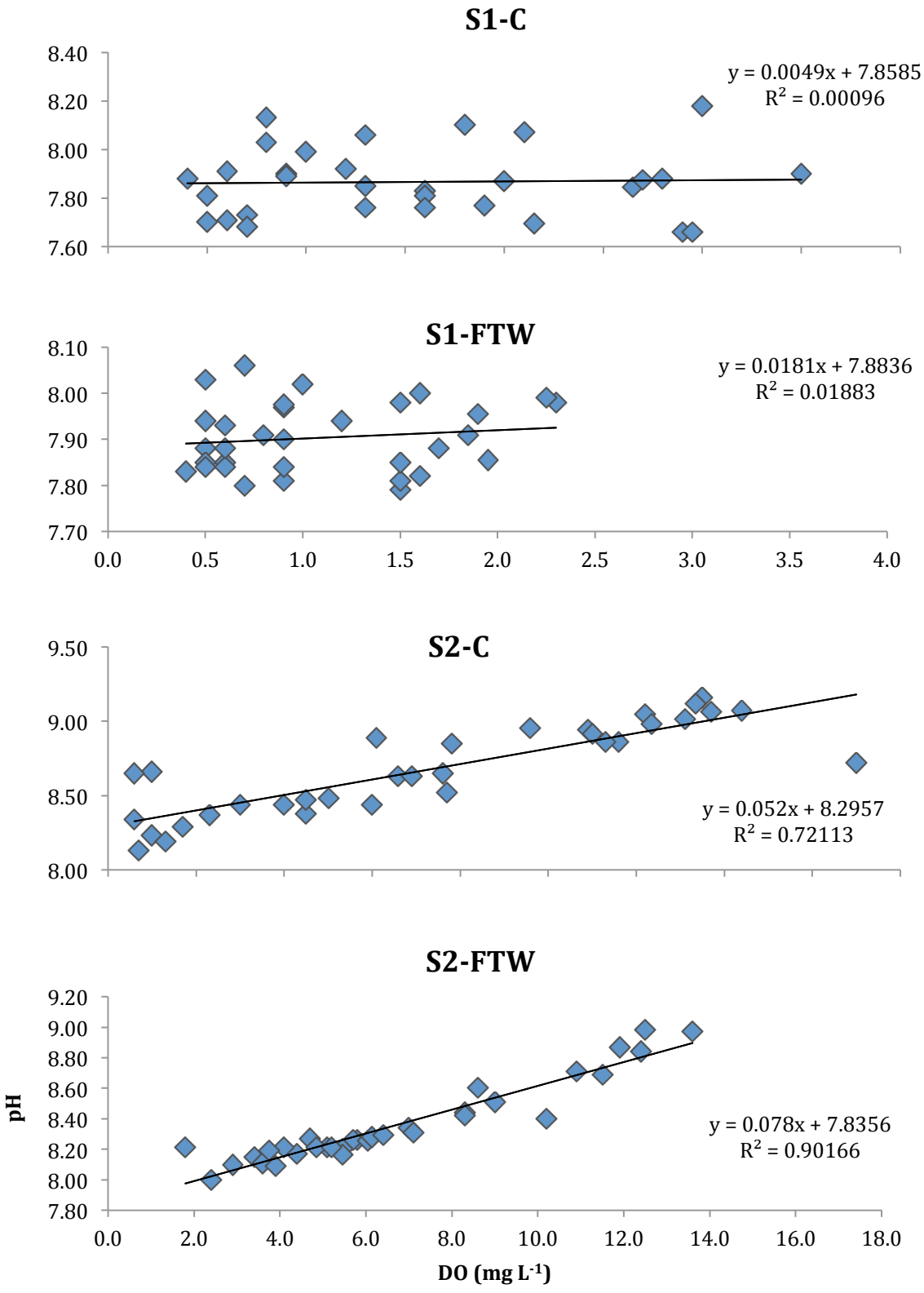

Figure 5.3 Scatter plot showing the relationship between dissolved oxygen (DO) and pH in the two-stage treatment cells. Trend line equation and R^2 values are included

EC is an indirect measurement of total dissolved solids (TDS) and is directly related to the concentration of ions in water. EC was only measured in the two-stage treatment in 2016. EC in the S1-FTW and S2-FTW cells was slightly lower (though not significant) than in the S1-C and S2-C cells. Low EC in FTW cells could be ascribed to the observed reduction of ions including NO_3^- (except in June, 2016), NH_4^+, and PO_3^{3-} during the field season. Furthermore, macrophytes or the microbial community may have removed other unmeasured ions. EC of the influent was higher than that in all FTW and control cells in the two-stage treatment indicating an overall reduction by the treatment system.

5.2.2 Factors Effecting Diurnal Variation

The continuous water quality monitoring by Sonde in late summer of 2016 revealed a prominent diurnal variation in all monitored parameters including temperature, DO, EC, and pH in all FTW and control cells. In the 2015 and 2016 seasons, the measurement of these parameters was performed at 9:30am, twice per week. The bi-weekly field monitoring of course could not reflect and capture the variation of these parameters on a diurnal scale, as the daily fluctuation leads to different readings at different times of the day. In addition, the greatest difference between the control and FTW cells often occurred in the late afternoon or evening. It was expected that water temperature fluctuated with air temperature in both the control and FTW cells, and the temperature fluctuation was attenuated slightly in the FTW cells. The diurnal variation of DO and pH was likely driven by biological activities, specifically photosynthesis during the daytime and respiration during the nighttime (Kadlec & Wallace, 2008). The fluctuation of conductivity was largely driven by changes in temperature, since conductivity is

positively correlated with water temperature (Sawyer et al, 1994). These results highlight the need to monitor and investigate the behaviour of the FTW in a finer temporal resolution as temperature, DO, and pH could largely affect physicochemical and biological processes and in turn the treatment efficiency of a FTW.

The difference in average temperature between the S1-C and S1-FTW cells and between the S2-C and S2-FTW cells were -0.2°C and -0.3°C, respectively. However the difference between the FTW and control was much greater in the afternoon and evening in stage-2, while this was not observed in stage-1 cells. The presence of a FTW absorbs or reflects incoming solar radiation before it can enter the water. This effect would only increase as the vegetation matures, since macrophytes are particularly reflective to infrared radiation (Daughtry et al, 2000). Since the FTWs covered 22% of the cell, it can be assumed the FTW cells were receiving 22% less solar radiation then the respective controls. However, radiation is not the only way the control cells would warm, as there would also be convection and conduction from the atmosphere. Half the height of the sea-containers was buried in soil, but the other half was exposed to the open air.

Due to these factors, 22% coverage by FTW alone may have a very minimal effect on temperature. However, the high perimeter to volume ratio in the sea-containers means it would experience greater edge effects relative to an SP. In a full scale application, the issue of radiation heating the edges of the system would be less prominent, since most SPs are surrounded by an earthen berm that would provide insulation (Stanley & Smith, 1992). In the S2-FTW cell, there was far more surface coverage than just the FTW, as it appeared to have more floating vegetation than the S2-C cell. This would have reflected more radiation, and is the likely explanation for the magnified effect of the FTW in stage-

2 versus stage-1. A reduction in water temperature was also found in other studies (Wang et al, 2012; White & Cousins, 2013; Zhou & Wang, 2010), though most of these studies have greater than 20% coverage, and as such the difference is expected to be larger. Van de Moortel et al. (2010) found in a seasonal study that temperature in the FTW cell was restrained when ambient temperatures rose above 15°C, which was not seen in the control. A similar effect was observed in this study. During the last three days of continuous monitoring, the ambient temperature dropped (either due to rain or consistent cloud cover), and this appeared to reduce the daily difference between the FTW and control cell. Temperature in the control and the FTW cells began to diverge around sunrise (between 6-7am during the study period), with the maximum difference occurring around 5-6pm. From then onwards, the temperatures converge and show very little difference by the middle of the night (12-2am).

This supports the assumption that the primary means by which the FTW (and associated floating vegetation) reduces temperature is by intercepting and reflecting solar radiation. After sunrise, temperature rises faster in the control as it absorbs more radiation, and by late afternoon this effect is at its greatest. However, during the night the FTW does not interfere with the conduction of heat into the atmosphere, and as such temperature in the control and FTW cells converged at this time.

DO is another crucial water quality parameter, as it affects other physicochemical parameters and is important to chemical reactions occurring in the treatment cells. The observations revealed that DO saturation in all cells fluctuated daily, reaching a peak in late day (between 2-3pm in stage 1, and between 7-10pm in stage-2), and reaching a minimum during the night, shortly before sunrise (between 6-7am). The later peak in

stage-2 is unusual, since most DO monitoring in ponds and rivers produces a DO maximum in the late afternoon. A possible explanation is that the sonde was located on the eastern side of the sea container, only 75cm from the eastern wall. Shading provided by the wall would have prevented direct sunlight from reaching that side of the container until later in the day, which may have caused DO to not increase until late day. Sunset during the study period was between 8:30-9:00pm. DO was almost certainly associated with primary productivity and photosynthesis in all treatment cells. Even though the stage-1 cells did not contain aquatic and floating macrophytes, the presence of periphyton on the container walls and suspended algae within the water would have undergone photosynthesis during the day.

In stage-1, the greatest difference between the FTW and control occurred in late afternoon, while during the night DO levels were quite similar, often reaching below 5% saturation in both cells. The explanation for lower DO in the FTW cell during the day is either a reduction in primary productivity in the S1-FTW cell (caused by shading from the FTW) or reduced oxygen transfer from the atmosphere.

Measured pH in both FTW and control cells also presented minor diurnal variation. The diurnal variation of pH can be largely explained by the diurnal variation of DO concentration, as both demonstrated similar variation patterns. The stage-1 cells showed a smaller fluctuation in pH relative to the stage-2 cells, with a standard deviation of 0.04 in both the S1-C and S1-FTW cells compared with 0.14 and 0.04 in the S2-C and S2-FTW cells respectively. Similar observation was obtained in DO concentration. These results suggest that the biological activities (photosynthesis and respiration) are more intensive in the stage-2 cells relative to the stage-1 cells.

In all cells, EC generally peaked between 4-6 pm daily, approximately when temperature peaked. EC was consistently lower in the FTW cells than in the control cells and was found to fluctuate with temperature. In stage-1 treatment, the difference between the S1-C and S1-FTW cells was negligible, whereas the different between stage-2 treatment cells was obvious. This difference seems to follow a similar pattern as temperature.

5.3 Wastewater Treatment Efficiency Under Low and High Loading Scenarios

When comparing the nutrient removal efficiency between the 2015 and 2016 season, there seems to be a clear difference between removal efficiency of FTW cells fed with high versus low-nutrient wastewater. Though the change in experimental design makes direct comparison difficult, the facultative treatment in 2015 can be generally compared with the two-stage treatment in 2016. The storage treatment in 2015 provides a good view into changing nutrient concentrations in a static treatment system without regular additions.

Though the facultative treatment and the two-stage treatment both had a 50-day HRT, the major differences in these treatments include: (a) one-cell treatment used in 2015 versus two-cell treatment in 2016, (b) a more mature FTW system in 2016 compared to 2015, and (c) higher nutrient input in 2016 (Table 5-1). These differences explain why the FTW cell outperformed the control cell in 2016 and not 2015. When examining nitrogen removal, the observations in 2015 made it difficult to discern if any additional treatment was introduced by FTW since low concentrations of both NH_4^+ and NO_3^- were measured in the effluent of both the F-C and F-FTW cells. For phosphorus,

there was a gradual decrease in both the control and FTW cell in 2015 (both treatments), but without any significant difference in removal efficiency.

Table 5-1 Influent properties for the facultative treatment (2015) and the two-stage treatment (2016). The total added column denotes total nutrient additions for the entire season

	Influent 2015 (Facultative Treatment)			
	Average Concentration (mg L^{-1})	Daily Loading (mg day^{-1})	Daily Loading (mg d^{-1} m^{-2})	Total Added (g)
NH_4^+–N	0.65	245.7	15.85	34
NO_3^-–N	0.52	196.56	12.68	27
TN	2.97	1122.66	72.43	155
RP	2.60	982.8	63.41	136
TP	3.16	1194.48	77.06	165
BOD	12.30	4649.4	299.96	642

	Influent 2016 (Two-Stage Treatment)			
	Average Concentration (mg L^{-1})	Daily Loading (mg day^{-1})	Daily Loading (mg d^{-1} m^{-2})	Total Added (g)
NH_4^+–N	10.8	8164.8	263.38	1029
NO_3^-–N	0.34	257.04	8.29	32
TN	15.1	11415.6	368.25	1438
RP	4.71	3560.76	114.86	449
TP	6.91	5223.96	168.51	658
BOD	54	40824	1316.90	5144

However in the two-stage treatment in 2016, a clear difference between FTW and control cells in terms of nutrient removal was observed. Sampling from the outlet of stage-1 and stage-2 revealed that treatment efficiency was not consistent, and physicochemical factors and nutrient loading rate likely play a large role in treatment.

The sections below discuss the relationship of nutrient removal with influent loading, physicochemical parameters, and retention time, in greater detail.

5.3.1 Nitrogen Removal

Previous research indicates that FTWs increase the concentration of ammonifying, nitrifying, and denitrifying bacteria (Zhou et al, 2012). The root network beneath the FTW is host to a diverse community of bacteria that facilitates bacterially driven reactions occurring in the nitrogen cycle (Davey & O'Toole, 2000). Results from this study support the conclusion that FTWs increase the rate of both nitrification and denitrification.

5.3.1.1 Ammonia Removal and Nitrification

The storage treatment provides a clear comparison between the FTW and control, as ammonia was practically removed from the water (concentrations reduced below 0.05mg L^{-1}) 41 days before concentrations fell below the same level in the control cell (Figure 5.4). Interestingly, although both cells had no additions, there was an initial increase in ammonia that peaked on July 7, 2015 in both cells, despite the absence of additional influent. The most likely explanation was the degradation of suspended algae in the sea-containers. Initially TSS was much higher in both treatment cells, and then rapidly decreased in June. Since effluent came from a stabilization pond with a high HRT, it was likely most of the solids were organic in origin. Since the degradation of organic proteins produces ammonia (Sawyer et al, 1994), this likely explains the initial increase in ammonia concentration. While macrophytes can use ammonia as a nitrogen source and appear to prefer it over nitrate (Van Oostrom, 1995), a large proportion of ammonia is removed by microbial community instead of by vegetation uptake (Tao et al, 2007). The

volatilization of ammonium ions (NH_4^+) to ammonia gas (NH_3) is greatly increased when pH approaches and exceeds 8.5, which provides an additional removal pathway in highly alkaline systems (Sawyer et al, 1994).

Both the 2015 facultative treatment and the 2016 two-stage treatment had regular additions of ammonia from the influent, but the removal efficiency in each treatment differed. In the facultative treatment the FTW cell did not perform significantly better, but both cells reduced incoming ammonia concentration, and by July 17, 2015 concentration in both cells (and also the influent) had fallen below 0.5mg L^{-1} NH_4^+-N. This nearly complete removal, combined with relatively low incoming ammonia (average 0.65mg L^{-1} NH_4^+-N in the influent) meant it was difficult to determine if the FTW was providing additional treatment. For the facultative treatment, the 50-day HRT proved more than adequate to remove ammonium in the control, just as well as in the FTW cell.

In the 2016 two-stage treatment, a much higher incoming concentration (average 10.8±3.6mg L^{-1} NH_4^+-N in the influent (Table 5-1) provided a clearer comparison between the FTW and control cell. In stage-1, after a 25-day HRT, effluent from the S1-FTW cell already had significantly lower ammonia than the control, indicating greater ammonia removal through biological uptake (macrophyte and/or bacteria), or increased nitrification. Nitrate concentration exiting the S1-FTW cell was higher than the S1-C, which suggests nitrification was occurring more rapidly in the FTW cell (Table 5-2). The low DO environment present in both cells (though lower in the FTW cell) was still sufficient for nitrification to occur.

Continuous monitoring revealed that concentrations of <0.5mg L^{-1} DO occurred during the night in both cells, but especially the FTW cell, which would have inhibited

nitrification. Macrophytes also release O_2 from their roots, which creates an aerobic environment where it otherwise would not exist (Bezbaruah & Zhang, 2005). Though aerobic conditions would most likely be constrained to the rhizosphere, it would provide treatment as wastewater flowed underneath the FTW and through the root system. Interestingly, in stage-2 the S2-FTW cell had consistently higher treatment than the control until September 1 2016 then concentration in both cells was equally low (below 0.2mg L^{-1} NH_4^+-N). Concentration in the S2-FTW cell had been at low concentrations since June 23, 2016, and there are a couple explanations for the sudden increase in treatment in the S2-C cell late in the season. First, incoming ammonia in the influent was higher on average for the months of June/July when compared with August/September (13.1 versus 8.5mg L^{-1} NH_4^+-N), which means the 50-day retention for the combined stage-1 and stage-2 control may have been adequate to almost completely remove ammonia in the late season. Another reason is pH was significantly higher in the S2-C cell, particularly in the late season. For the month of September, average pH in the S2-C cell was 8.93 and only 8.19 in the S2-FTW cell. Since ammonium volatilization increases significantly above pH of 8.5, the late season high pH in the S2-C cell likely contributed to ammonia removal.

5.3.1.2 Nitrate Removal and Denitrification

Unlike ammonia, incoming nitrate concentration was relatively low for all treatments. Thus it was more often the case that nitrate was produced in the treatment cells through nitrification rather than removed (Table 5-2). Since nitrification is the conversion of ammonium to nitrate, is it reasonable that the major source of nitrate in the cells was the aerobic conversion of ammonia by nitrifying bacteria. The storage

treatment, lacking regular additions of wastewater, provides a clear view into the relationship between ammonia and nitrate concentration (Figure 5.4). While ammonia concentration was consistently lower in the S-FTW cell, nitrate was consistently higher than in the control cell until August 4, 2015. Looking only at the S-FTW cell, ammonia reached its maximum on July 7, 2015, and then began to decrease while nitrate levels continued to increase. The maximum nitrate concentration was reached on July 21 2015, which coincides with the date in which ammonia concentration was reduced below 0.1mg L^{-1} NH_4^+-N in the FTW cell. After this point, with the ammonia supply practically depleted, nitrate concentration rapidly decreased and fell below the detection limit (0.23mg L^{-1} NO_3^--N). A similar process occurred in the S-C cell, but at a reduced rate. Nitrate concentration in the S-C cell reached a maximum on August 11, 2015 (21 days after the S-FTW cell), and gradually decreased.

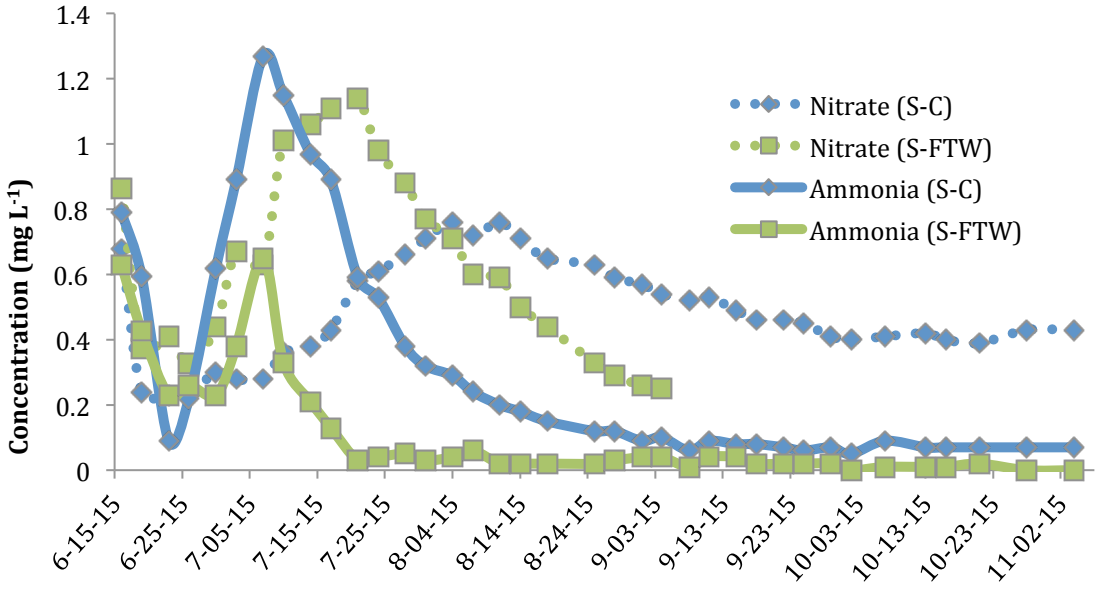

Figure 5.4 Temporal variation of concentration within the storage treatment for nitrate (as NO_3^-–N) and ammonia (as NH_4^+–N) revealing that a decrease in ammonia produces nitrate through nitrification.

The most likely removal mechanisms for nitrate include denitrification which eventually reduces NO_3^- into N_2 gas (Sawyer et al, 1994), and biological uptake since macrophytes and bacteria may also use nitrate as a nitrogen source (Van Oostrom, 1995). Bartucca et al. (2016) used a FTW system with aeration under high nitrate loading (>150mg L^{-1} NO_3^-) and found significant removal through the macrophytes, which grew proportional to nitrate concentration level.

Denitrification is an anoxic reaction, meaning it is inhibited by the presence of oxygen. The reaction also requires a carbon source, since NO_3^- is used to oxidize a carbonaceous compound that acts as an energy source for denitrifying bacteria. If O_2 is present in sufficient quantities, then it will be used preferentially since it is a stronger oxidizing agent (Sawyer et al, 1994). In both cells of the storage treatment, DO saturation in the water column reached a minimum of 24% early in the season, but for the remainder for the year remained between 80-100%. This relatively high level of DO means that denitrification was most likely inhibited within the water column of both the S-C and S-FTW cell. Thus the nitrate removal in the FTW cell (at least in the late season) was likely caused by biological uptake (due to the presence of macrophytes or dense microbial communities that were not present in the control), or an anoxic or anaerobic zone that may have occurred beneath the FTW.

Previous research has shown the zone beneath the FTW has reduced DO, likely due to the high BOD in the rhizosphere (Headley & Tanner, 2012). Continuous monitoring revealed that nightly DO concentration could fall below 5%, particularly in the stage-

FTW cell. This would have created a far more favourable environment for denitrification. Thus, the role of denitrification in removing NO_3^- might vary diurnally.

Table 5-2 Average NO_3^-–N production per cell in the two-stage treatment. For stage-1, NO_3^-–N concentration in effluent was subtracted from the influent, for stage-2 NO_3^-–N concentration in effluent was subtracted from stage-1 effluent. Negative value indicates NO_3^-–N reduction in the cell

	Stage-1		Stage 2	
	S1-C	S1-FTW	S2-C	S2-FTW
Average NO_3^-–N Production (mg L^{-1})	0.06	0.12	0.21	-0.13

Carbon released from the roots of emergent wetland macrophytes (Neori et al, 2000), would provide a carbon source for denitrification if the water column became carbon limited, which can occur in wastewater treatment systems (Faulwetter et al, 2009). The mat was composed of dense polyethylene fibres, and later in the season it was interspersed with macrophytes roots (Figure 5.2). It is likely this zone could have become anoxic, due to a combination of high BOD and the presence of a barrier to O_2 transfer from the atmosphere.

The F-C and F-FTW cells, had differing levels of nitrate early in the season, but for remainder of the season nitrate remained below the detection limit in both cells. Nitrate concentration in the influent was low (0.52±0.25mg L^{-1} NO_3^-–N) and initially the concentration in both F-C and F-FTW cells was higher than that in the influent. Similar to the storage treatment, nitrate was higher in the FTW (likely due to the greater nitrification rate of incoming ammonia), but then rapidly decreased in both cells, and remained below the detection limit for the remainder of the season, despite a gradual increase in incoming

nitrate concentration in the influent. Both cells were often supersaturated (>100% DO saturation) so denitrification was likely inhibited. It is probable that biological treatment (through the FTW, but also submerged and floating vegetation in both cells) was the primary removal mechanism for nitrate, and the long 50-day retention time provided adequate time for almost complete removal.

The 2016 two-stage treatment had a net production of nitrate, but with a large difference between the FTW and control cell (Table 5-2). Influent concentration was high in ammonia but low in nitrate, which is common for sewage which has only undergone primary treatment (Stanley & Smith, 1992). Considering the cells in stage-1, low DO in both cells created a more favourable environment for denitrification to occur, which would have been required to balance out the rapid nitrate production through the nitrification of ammonia that was occurring in both cells. Similar to the storage and facultative treatments, there was an initial nitrate concentration spike in the S1-FTW cell, likely caused by the sudden nitrification of incoming ammonia, which then rapidly decreased. The average nitrate concentration in the S1-FTW cell effluent was slightly higher than the control and is most likely caused by the higher rate of nitrification relative to the control. In stage-2 however, the nitrate in the effluent was consistently lower in the S2-FTW cell compared to the S2-C cell.

The relatively low BOD exiting both stage-2 cells (6.73±4.54 and 3.93±3.11 BOD_5 for the S2-C and S2-FTW cells respectively) could provide an explanation for the difference in nitrate removal between the S2-C and S2-FTW cells. Since BOD is an indirect measure of carbon, is could imply the water in both cells was relatively low in organic carbon. In which case the carbonaceous compounds leached through the roots of

the FTW would have allowed denitrification to occur at an increased rate, relative to the control, by providing a localized carbon source.

5.3.1.3 Total Nitrogen Removal

Total nitrogen includes most forms of inorganic and organic nitrogen. The FTWs outperformed the controls in terms of removal efficiency in both stages of the two-stage treatment. Much of the total nitrogen removal can be explained by the reduction of ammonia, which for most of the season, and in all treatments, was the most abundant inorganic form of nitrogen. Ammonia and nitrate are the most prominent and biologically available forms of inorganic nitrogen, and determining the portion of Total Nitrogen that is neither ammonia or nitrate (termed organic nitrogen), determines the percentage of nitrogen exiting the cell that is in a less available form, such as proteins or in complex inorganic compounds. Interestingly, the concentration of organic nitrogen exiting the FTW and control cell in both stages of the two-stage treatment did not differ significantly. This means that much of the increased nitrogen removal in the FTW cells can be explained through the uptake of inorganic ions, denitrification, and ammonia volatilization, instead of the digestion of organic nitrogen in stage-1 or stage-2.

5.3.2 Phosphorus Removal

Phosphorus is a limiting nutrient in most freshwater ecosystems in Canada and many parts of the world. Thus phosphorus is often targeted as a nutrient to control in order to prevent eutrophication in water bodies (Carey et al, 2013). Differing from nitrogen, phosphorus cannot be converted to a gaseous component, thus phosphorus removed from the water column is either diverted into biomass growth or into the sediment. Creating a more neutral pH and raising the organic content within a water body can increase

sorption of phosphorus onto particulate matter, thus both pH and organic content can impact phosphorus removal from the water column in FTW systems (Borne, 2014). Internal phosphorus loading can be a significant source into a SP ecosystem (Nurnberg et al, 2013), and factors which affect internal loading include aerobic or anaerobic conditions, pH, soluble Al^{3+} and Fe^{2+}, and organic detritus accumulation within the sediment (Lindstrom & White, 2011).

In 2015, there was no significant difference in treatment between the FTW and control cells for either the storage or the facultative treatment. Initially in the months of June and early July, RP and TP concentrations were higher in the FTW cells than those in the control cells in both treatments. Before placement in the field, both FTWs were grown in a greenhouse in which fertilizer was added to help the macrophytes establish. The macrophyte plugs were also initially grown in soil and though the roots were washed, some of the soil may have been transported into the FTW when they were transplanted. This could have introduced additional phosphorus into the treatment cells, and is likely the reason for the initial increase in phosphorus concentration in the FTW cells. However after the initial divergence early in the season, TP and RP remained similar between both FTW and control cells in both treatments. In the storage treatment, initial RP and TP concentration was quite low and throughout the season there was no significant reduction. It is likely that at such low concentrations neither the FTW nor the control was capable of removing significant amounts of phosphorus. Also internal phosphorus loading from the macrophyte roots in the FTW cell, or decaying zooplankton and other invertebrates in both cells, may have provided a phosphorus source that replaced any phosphorus losses created by biological uptake or sedimentation.

Differing from the storage treatment, the facultative treatment displayed high removal efficiency for RP and TP, in both cells (76±8.6% and 75±7.7% RP removal for the F-C and F-FTW cells respectively, and 66±9.4% and 65±11% TP removal for the F-C and F-FTW cells respectively). In addition, RP and TP concentration gradually decreased during the season in both cells. This was likely the result of lower RP and TP concentration in the influent. The influent was withdrawn from the edge of the Carseland facultative lagoon, where higher summer temperature and increased biological activity would have increased treatment during the summer months (July and August), which is common in SPs in northern climates (Price et al, 1995). Even though the influent for the facultative treatment was withdrawn from the western edge of the facultative treatment (closest to the outlet of the primary cells), mixing within the lagoon would have meant partially treated wastewater was entering the treatment cells. As such, the concentration of incoming phosphorus in the influent decreased during the 2015 season, in parallel with lowering concentrations in the facultative lagoon. Within all treatment cells, relatively colder temperatures in September and October likely caused a reduction in RP and TP in the effluent. Since much of the phosphorus was likely removed through sedimentation, colder temperature can prevent the release and resuspension of phosphorus, by inhibiting biological activity and chemical reactions within the water (Lindstrom & White, 2011; Soranno et al, 1997).

Previous studies have found a significant reduction in phosphorus concentration caused by a FTW (compared to a control), even at low incoming concentrations (Borne, 2014; De Stefani et al, 2011; White & Cousins, 2013). However the major difference between those studies and this experiment was HRT. Many mesocosm and stormwater

studies have a HRT between 1-20 days, while this study was 50-days. It seems that a relatively long HRT provides a high level of treatment in the controls, which is not necessarily improved by the addition of a FTW when treating low phosphorus wastewater.

In the 2016 season, the performance of FTWs relative to the control appeared to be different from FTWs in the 2015 season. Both the S1-FTW and S2-FTW displayed lower TP concentration compared to the S1-C and S2-C, respectively. The results in both the 2015 and 2016 season might suggest that the removal efficiency of the cells is not independent of influent concentration, and higher removal efficiency is obtained when given higher nutrient concentrations.

Interestingly, there is a noticeable difference in treatment between the stage-1 and stage-2 cells. In stage-1, RP and TP removal efficiency was higher in the S1-FTW cells then the S1-C, but was not statistically significant. Thus RP and TP concentrations in the effluent from the S1-FTW cell were lower than those in the effluent from the S1-C cell for most of the season, until late August when outgoing phosphorus in the FTW cell was actually higher than the control. However, in stage-2 the S2-FTW cell had consistently and significantly higher RP and TP removal efficiency until late September when both cells had similar concentrations. For RP, the removal efficiency of the S1-FTW cell was on average 6% higher than the S1-C cell, which is much lower than that of the S2-FTW cell, which had an average removal efficiency 20% higher than the S2-C cell.

There are a few factors that could improve the treatment of the S2-FTW cell. The low DO concentration (average 1.3±1.3mg L^{-1} DO) in the S1-FTW cells is more favourable to reduction in both the water column and in the bottom sediment. This would

facilitate the release of phosphorus adsorbed onto particulates. The majority of phosphorus removed by FTWs is likely caused by bacterial uptake along the roots, which then settles into the sediment (Borne, 2014; Headley & Tanner, 2012; Tanner & Headley, 2011). Thus when ambient environmental conditions are favourable to phosphorus release, the phosphorus removal efficiency of a FTW could be severely diminished. It should be noted that any phosphorus incorporated into the root or shoot tissue of the macrophytes (and not bacterial biomass) would not be as susceptible to release due to changing physicochemical conditions.

In stage-2, average DO in the S2-FTW cell was lower than the S2-C cell, but still greater than it was in the stage-1 cells. This likely would have inhibited phosphorus release, which consequently allowed more phosphorus to be stored in the sediment and enhanced the magnitude of treatment in both the S2-FTW and S2-C cells. Humic substances released from macrophyte roots are theorized to increase phosphorus removal from the water column by providing a reactive surface for phosphorus to adsorb (Borne, 2014). In stage-1, the S1-FTW cell was adding humic substances to wastewater that already had a high concentration of organic substances (implied by the high BOD of the influent). As such, the additional adsorbing capacity provided by the FTW may have been negligible in stage-1. However in stage-2, significantly higher humic content in the FTW-cell may have provided a removal pathway that was not present in the control, since the influent (water exiting the stage-1 cell) was already low in organic carbon, as implied by the reduced BOD.

In the effluent of the S2-C and S2-FTW cell, TP was on average 30% and 41% higher than RP respectively (Figure 5.5). The greater proportion of TP that was non-

reactive in the FTW cell indicated that much of the increased phosphorus removal in the cell was through the uptake of more biologically available forms of phosphorus like orthophosphate. Meanwhile the lower non-reactive TP% in stage-1 indicates that relatively more of the incoming phosphorus was in a biologically available form, possibly due to anaerobic digestion in the primary cells on site. The high non-reactive TP% in the storage treatment indicates that much of the reactive phosphorus had been removed by the system, likely a combination of initially low TP concentration, and a lack of any substantial phosphorus input.

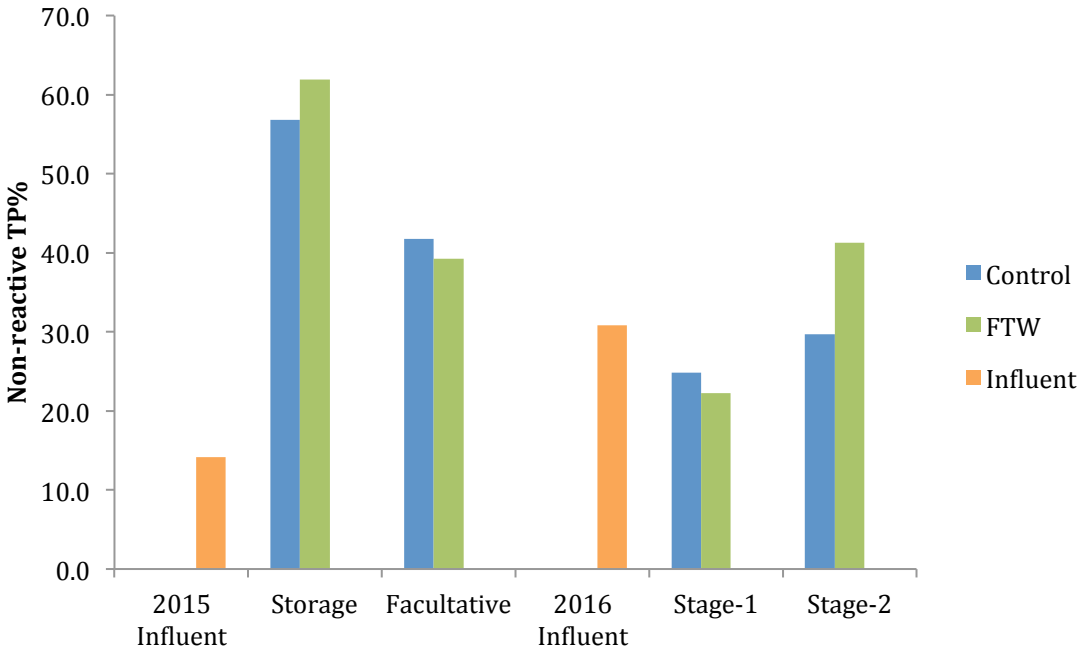

Figure 5.5 Comparison of the average non-reactive TP percentage in the control and FTW cell for the storage treatment, facultative treatment, both stages of the two-stage treatment, as well as the 2015/2016 influent used to feed the facultative and two-stage treatments

5.3.3 Suspended Solids, Turbidity, and Fecal Coliforms

A previous study by Borne et al. (2014b) displayed that FTWs can significantly reduce suspended solids (SS) in the water column. This reduction is caused by the physical entrapment of particulate matter within the roots and the eventual settling within the benthic sediment. If a majority of the sediment is organic, then a portion of the removal can be explained through the decomposition of organics by resident bacteria (Headley & Tanner, 2012).

In all treatments (both seasons), there was no significant difference is SS removal between FTW and control. However, removal efficiency was high in the facultative and two-stage treatment as all cells had ≤5mg L^{-1} TSS in the effluent on average. The storage treatment had initially high TSS in both cells (>30mg L^{-1} TSS) primarily caused by suspended algae. After the decomposition/settling of the algae TSS decreased below 5mg L^{-1} for the remainder of the season. In the facultative treatment incoming TSS was quite low and highly variable (19±18mg L^{-1} TSS), but was higher in the two-stage treatment when drawn from the more turbid primary treatment cells (47±24mg L^{-1} TSS). The likely cause of high SS removal in all cells was HRT. Even in the two-stage treatment, the 25-day HRT in stage-1 alone was higher than most primary settling systems used in wastewater treatment. This long HRT gave sufficient time for most solids to settle or decompose, thus providing an explanation why the enhanced impact of FTWs on SS removal that others studies have noted, was not seen in this study.

The results for turbidity generally mirrored those seen for SS. Understandably since turbidity is positively correlated with SS concentration. One interesting difference was the significant difference between the S1-FTW cell and the S1-C cell (96±1.8% turbidity

removal versus 93±3.2% removal in the S1-FTW cell and S1-C cell respectively). However this significant difference did not carry over to stage-2, where the removal efficiency was 97±1.4% and 98±2.1% for the S2-C and S2-FTW cells respectively. The FTW appeared to outperform the control in stage-1. An explanation for a significant difference in turbidity and not SS was the presence of *Daphnia* (water fleas) and other zooplankton, which may have increased the measured value for TSS, but would have had little impact on turbidity readings. All cells were host to occasional blooms of zooplankton throughout the study period, which likely played a major role in reducing turbidity and TSS.

Fecal coliform has been reduced in previous studies on naturally occurring floating wetlands (Kansiime & van Bruggen, 2001) but none have investigated constructed FTWs. In this study, Fecal Coliform bacteria were only measured in the 2016 two-stage treatment. Average incoming concentration was 19,000±22,000cfu $100ml^{-1}$, and varied considerably. Average removal efficiency in all cells (stage-1 and stage-2) was >99.99%. High HRTs have been associated with high fecal coliform reduction in SPs (George et al, 2002), and the treatment system in this study no different. Predatory microbes, zooplankton, and a hostile environment most likely combined to create a system that was highly effective at reducing fecal coliform bacteria. Garcia et al. (2008) observed that treatments wetlands (a FWS and HSSF system) outperformed a maturation pond in terms of bacteria reduction. This was not seen in this study, likely due to the long HRT, and the fact that HSSF and VF wetland appear to be the most effective form of bacterial treatment, since wastewater is forced through a porous medium interspersed with roots.

The S2-FTW cell had a period near the end of the season where Fecal Coliform levels were detectable (>1cfu $100ml^{-1}$), while at the same time similar results were not seen in the S2-C cell. During the same period of time, undetectable levels of fecal coliform bacteria were fed into the S2-FTW cell from S1-FTW, so it is unlikely the bacteria were originating from the S1-FTW effluent. During the study period, birds such as ravens, gulls, terns, and swallows were observed resting on the sea-containers around the FTWs. It is likely that excrement from these animals may have contributed to the low, (but detectable) levels of fecal coliform bacteria in the S2-FTW cell.

Chapter 6: CONCLUSIONS

6.1 Conclusions and Future Applications

The primary objective of this thesis was to determine the effectiveness of FTWs in a pilot-scale SP system (albeit, operating at water depth and HRT similar to those of a full-scale SP). Macrophyte growth and successful establishment of the FTWs, combined with high winter survival, indicates that FTWs using native vegetation are suitable for SPs within Southern Alberta. This also indicates that FTWs are suitable for many regions of Canada, as well as other cold-continental climates throughout the world.

Results indicate the effectiveness of the FTW is contingent upon nutrient loading and HRT within the treatment cells. For the storage and facultative treatments, fed by low nutrient influent in 2015, the FTWs did not significantly improve treatment (except for NH_4^+ in the storage cell), likely because the high HRT of the treatment cells was adequate to provide an already high level of contaminant removal without FTWs. However, in the two-stage treatment in 2016 fed by high nutrient influent, the FTW significantly increased the treatment efficiency of NH_4^+, NO_3^-, RP, TP, and BOD removal by 8%, 106%, 20%, 17%, and 7% respectively, relative to the control. In addition, all measured basic physicochemical parameters (temperature, EC, DO, pH) were reduced in the FTW cells during continuous monitoring. Phosphorus analysis of above-mat macrophyte biomass revealed that the FTW acts as a minor sink for phosphorus (6% of TP sequestered in above-mat biomass), and thus harvesting of the biomass could provide a method of removing nutrients from SPs.

Based on these findings, it is recommended that FTWs be applied to enhance treatment in high-nutrient SP systems. However, optimal placement would be directly downstream of the primary treatment cells, where FTWs could provide the greatest impact on nutrient removal. FTWs may be unlikely to improve treatment in storage lagoons, or secondary/tertiary facultative lagoons where the incoming wastewater is partially treated, and HRT is often quite high (e.g. >50days).

6.2 Recommendations for Future Research

While one objective of this thesis was to observe the winter survival of the macrophytes and ensure the structural integrity of the FTW was consistent, wastewater treatment during the winter was not monitored. Since treatment reduction in winter is a common issue in SPs in cold-climates, enhancing treatment with a passive system in the winter would be beneficial to isolated and remote communities that rely on decentralized treatment. Therefore the winter performance of a pilot or full-scale SP with a FTW should be monitored. In addition, a mass balance approach is recommended for evaluating the role of FTWs more accurately, considering the nutrient deposition in sediment and the rate of sludge accumulation. Sludge removal is a common practice in SPs, and is typically conducted every 5-15 years dependent on daily loading. The investigation of FTW's role in sludge deposition rate or alteration of sludge composition would benefit SP operators and regulators.

References

Anderson, D. M., Glibert, P. M. & Burkholder, J. M. (2002) Harmful algal blooms and eutrophication: Nutrient sources, composition, and consequences. *Estuaries*, 25(4b), 704-726.

Andersson, S., Rajarao, G. K., Land, C. J. & Dalhammar, G. (2008) Biofilm formation and interactions of bacterial strains found in wastewater treatment systems. *Fems Microbiology Letters*, 283(1), 83-90.

Ash, R. & Truong, P. (2004) The use of Vetiver grass wetlands for sewerage treatment in Australia, *Sewage Managment QEPA Conference.* Cairns, Australia.

Ayaz, S. C. & Saygin, O. (1996) Hydroponic tertiary treatment. *Water Research*, 30(5), 1295-1298.

Baldizon, M. E., Dolmus, R., Quintana, J., Navarro, Y. & Donze, M. (2002) Comparison of conventional and macrophyte-based systems for the treatment of domestic wastewater. *Water Science and Technology*, 45(1), 111-116.

Banmann, C. L., Chu, A., He, J., Amell, B. & Skorobogatov, A. (2016) Floating Treatment Wetlands for Wastewater Treatment in Carseland, Alberta, *The 2016 North American Lake Management Society Symposium.* Banff, Alberta, Canada.

Bartucca, M. L., Mimmo, T., Cesco, S. & Del Buono, D. (2016) Nitrate removal from polluted water by using a vegetated floating system. *Science of the Total Environment*, 542, 803-808.

Bezbaruah, A. N. & Zhang, T. C. (2005) Quantification of oxygen release by bulrush (Scirpus validus) roots in a constructed treatment wetland. *Biotechnology and Bioengineering*, 89(3), 308-318.

Billore, S. K., Prashant & Sharma, J. K. (2009) Treatment performance of artificial floating reed beds in an experimental mesocosm to improve the water quality of river Kshipra. *Water Science and Technology*, 60(11), 2851-2859.

Boller, M. (1997) Small wastewater treatment plants - A challenge to wastewater engineers. *Water Science and Technology*, 35(6), 1-12.

Borne, K. E. (2014) Floating treatment wetland influences on the fate and removal performance of phosphorus in stormwater retention ponds. *Ecological Engineering*, 69, 76-82.

Borne, K. E., Fassman, E. A. & Tanner, C. C. (2014a) Floating treatment wetland retrofit to improve stormwater pond performance for suspended solids, copper and zinc (vol 54, pg 173, 2013). *Ecological Engineering*, 63, 142-142.

Borne, K. E., Fassman-Beck, E. A. & Tanner, C. C. (2014b) Floating Treatment Wetland influences on the fate of metals in road runoff retention ponds. *Water Research*, 48, 430-442.

Borne, K. E., Tanner, C. C. & Fassman-Beck, E. A. (2013) Stormwater nitrogen removal performance of a floating treatment wetland. *Water Science and Technology*, 68(7), 1657-1664.

Breen, P. F. & Chick, A. J. (1995) Rootzone dynamics in constructed wetlands receiving wastewater: A comparison of vertical and horizontal flow systems. *Water Science and Technology*, 32(3), 281-290.

Carey, R. O., Hochmuth, G. J., Martinez, C. J., Boyer, T. H., Dukes, M. D., Toor, G. S. & Cisar, J. L. (2013) Evaluating nutrient impacts in urban watersheds: Challenges and research opportunities. *Environmental Pollution*, 173, 138-149.

Chang, N. B., Islam, M. K. & Wanielista, M. P. (2012) Floating wetland mesocosm assessment of nutrient removal to reduce ecotoxicity in stormwater ponds. *International Journal of Environmental Science and Technology*, 9(3), 453-462.

Chua, L. H. C., Tan, S. B. K., Sim, C. H. & Goyal, M. K. (2012) Treatment of baseflow from an urban catchment by a floating wetland system. *Ecological Engineering*, 49, 170-180.

Daughtry, C. S. T., Walthall, C. L., Kim, M. S., de Colstoun, E. B. & McMurtrey, J. E. (2000) Estimating corn leaf chlorophyll concentration from leaf and canopy reflectance. *Remote Sensing of Environment*, 74(2), 229-239.

Davey, M. E. & O'Toole, G. A. (2000) Microbial biofilms: from ecology to molecular genetics. *Microbiology and Molecular Biology Reviews*, 64(4), 847-+.

De Stefani, G., Tocchetto, D., Salvato, M. & Borin, M. (2011) Performance of a floating treatment wetland for in-stream water amelioration in NE Italy. *Hydrobiologia*, 674(1), 157-167.

Eaton, A. D., Clesceri, L. S., Rice, E. W. & Greenberg, A. E. (2005a) 5-Day BOD Test, in Franson, M. A. H. (ed), *Standard Methods for the Examination of Water and Wastewater*, 21st Edition edition. Washington D.C.: American Public Health Association, 5-2,5-3.

Eaton, A. D., Clesceri, L. S., Rice, E. W. & Greenberg, A. E. (2005b) Fecal Coliform Membrane Filter Procedure, in Franson, M. A. H. (ed), *Standard Methods for the Examination of Water and Wastewater*, 21st Edition edition. Washington D.C.: American Public Health Association, 9-66, 9-68.

Eaton, A. D., Clesceri, L. S., Rice, E. W. & Greenberg, A. E. (2005c) Total Suspended Solids Dried at 103-105ºC, in Franson, M. A. H. (ed), *Standard Methods for the Examination of Water and Wastewater Treatment*, 21st Edition edition. Washington D.C.: American Public Health Association, 2-58.

Faulwetter, J. L., Burr, M. D., Cunningham, A. B., Stewart, F. M., Camper, A. K. & Stein, O. R. (2011) Floating treatment wetlands for domestic wastewater treatment. *Water Science and Technology*, 64(10), 2089-2095.

Faulwetter, J. L., Gagnon, V., Sundberg, C., Chazarenc, F., Burr, M. D., Brisson, J., Camper, A. K. & Stein, O. R. (2009) Microbial processes influencing performance of treatment wetlands: A review. *Ecological Engineering*, 35(6), 987-1004.

Fisher, J. & Acreman, M. C. (2004) Wetland nutrient removal: a review of the evidence. *Hydrology and Earth System Sciences*, 8(4), 673-685.

Floating-Island-International (2016) http://www.floatingislandinternational.com/products/biohaven-technology/, 2016. Available online: [Accessed.

Fu, W. G. & Li, P. P. (2011) Characteristics of Phosphorus Adsorption of Aerated Concrete in Wastewater Treatment. *Environmental Biotechnology and Materials Engineering, Pts 1-3*, 183-185, 466-470.

Garbett, R. (2005) An investigation into the application of floating reed bed and barley straw techniques for the remediation of eutrophic waters. *Water and Environment Journal*, 19(3), 174-180.

Garcia, M., Soto, F., Gonzalez, J. M. & Becares, E. (2008) A comparison of bacterial removal efficiencies in constructed wetlands and algae-based systems. *Ecological Engineering*, 32(3), 238-243.

George, I., Crop, P. & Servais, P. (2002) Fecal coliform removal in wastewater treatment plants studied by plate counts and enzymatic methods. *Water Research*, 36(10), 2607-2617.

Gottschall, N., Boutin, C., Crolla, A., Kinsley, C. & Champagne, P. (2007) The role of plants in the removal of nutrients at a constructed wetland treating agricultural (dairy) wastewater, Ontario, Canada. *Ecological Engineering*, 29(2), 154-163.

Greenway, M., Jenkins, G. & Polson, C. (2007) Macrophyte zonation in stormwater wetlands: getting it right! A case study from subtropical Australia. *Water Science and Technology*, 56(3), 223-231.

Hach (2013) *Water Quality Analysis Guide*.Loveland, CO.

Headley, T. R. & Tanner, C. C. (2012) Constructed Wetlands With Floating Emergent Macrophytes: An Innovative Stormwater Treatment Technology. *Critical Reviews in Environmental Science and Technology*, 42(21), 2261-2310.

Hoeger, S. (1988) SCHWIMMKAMPEN - GERMANYS ARTIFICIAL FLOATING ISLANDS. *Journal of Soil and Water Conservation*, 43(4), 304-306.

Hu, G. J., Zhou, M., Hou, H. B., Zhu, X. & Zhang, W. H. (2010) An ecological floating-bed made from dredged lake sludge for purification of eutrophic water. *Ecological Engineering*, 36(10), 1448-1458.

Hubbard, R. K., Gascho, G. J. & Newton, G. L. (2004) Use of floating vegetation to remove nutrients from swine lagoon wastewater. *Transactions of the Asae*, 47(6), 1963-1972.

Kadlec, R. H. (2009) Comparison of free water and horizontal subsurface treatment wetlands. *Ecological Engineering*, 35(2), 159-174.

Kadlec, R. H. & Wallace, S. (2008) *Treatment Wetlands: Second Edition*. Boca Raton, Fl: CRC Press.

Kansiime, F. & van Bruggen, J. J. A. (2001) Distribution and retention of faecal coliforms in the Nakivubo wetland in Kampala, Uganda. *Water Science and Technology*, 44(11-12), 199-206.

Karnchanawong, S. & Sanjitt, J. (1995) Comparative study of domestic wastewater treatment efficiencies between facultative pond and water spinach pond. *Water Science and Technology*, 32(3), 263-270.

Kerr-Upal, M., Seasons, M. & Mulamoottil, G. (2000) Retrofitting a stormwater management facility with a wetland component. *Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering*, 35(8), 1289-1307.

Kim, S. Y. & Geary, P. M. (2001) The impact of biomass harvesting on phosphorus uptake by wetland plants. *Water Science and Technology*, 44(11-12), 61-67.

Klausmeier, C. A., Litchman, E., Daufresne, T. & Levin, S. A. (2004) Optimal nitrogen-to-phosphorus stoichiometry of phytoplankton. *Nature*, 429(6988), 171-174.

Krause-Jensen, D. & Sand-Jensen, K. (1998) Light attenuation and photosynthesis of aquatic plant communities. *Limnology and Oceanography*, 43(3), 396-407.

Kroger, R., Holland, M. M., Moore, M. T. & Cooper, C. M. (2007) Plant senescence: A mechanism for nutrient release in temperate agricultural wetlands. *Environmental Pollution*, 146(1), 114-119.

Krzciuk, K. & Galuszka, A. (2015) Prospecting for hyperaccumulators of trace elements: a review. *Critical Reviews in Biotechnology*, 35(4), 522-532.

Kyambadde, J., Kansiime, F. & Dalhammar, G. (2005) Nitrogen and phosphorus removal in substrate-free pilot constructed wetlands with horizontal surface flow in Uganda. *Water Air and Soil Pollution*, 165(1-4), 37-59.

Ladislas, S., Gerente, C., Chazarenc, F., Brisson, J. & Andres, Y. (2013) Performances of Two Macrophytes Species in Floating Treatment Wetlands for Cadmium, Nickel, and Zinc Removal from Urban Stormwater Runoff. *Water Air and Soil Pollution*, 224(2).

Li, H., Hao, H. L., Yang, X. E., Xiang, L. C., Zhao, F. L., Jiang, H. & He, Z. L. (2012) Purification of Refinery Wastewater by Different Perennial Grasses Growing in a Floating Bed. *Journal of Plant Nutrition*, 35(1), 93-110.

Li, M., Wu, Y. J., Yu, Z. L., Sheng, G. P. & Yu, H. Q. (2007) Nitrogen removal from eutrophic water by floating-bed-grown water spinach (Ipomoea aquatica Forsk.) with ion implantation. *Water Research*, 41(14), 3152-3158.

Li, X. N., Song, H. L., Lu, X. W., Xie, X. F. & Inamori, Y. (2009) Characteristics and mechanisms of the hydroponic bio-filter method for purification of eutrophic surface water. *Ecological Engineering*, 35(11), 1574-1583.

Lindstrom, S. M. & White, J. R. (2011) Reducing phosphorus flux from organic soils in surface flow treatment wetlands. *Chemosphere*, 85(4), 625-629.

Liu, G. J., Zheng, D., Deng, L. W., Wen, Q. & Liu, Y. (2014) Comparison of constructed wetland and stabilization pond for the treatment of digested effluent of swine wastewater. *Environmental Technology*, 35(21), 2660-2669.

Loeppert, R. H. & Hallmark, C. T. (1985) INDIGENOUS SOIL PROPERTIES INFLUENCING THE AVAILABILITY OF IRON IN CALCAREOUS SOILS. *Soil Science Society of America Journal*, 49(3), 597-603.

Mallison, C. T., Stocker, R. K. & Cichra, C. E. (2001) Physical and vegetative characteristics of floating islands. *Journal of Aquatic Plant Management*, 39, 107-111.

Mara, D. D. (2006) Constructed wetlands and waste stabilization ponds for small rural communities in the United Kingdom: A comparison of land area requirements, performance and costs. *Environmental Technology*, 27(7), 753-757.

Martinez-Guerra, E., Jiang, Y., Lee, G., Kokabian, B., Fast, S., Truax, D. D., Martin, J. L., Magbanua, B. S. & Gude, V. G. (2015) Wetlands for Wastewater Treatment. *Water Environment Research*, 87(10), 1095-1126.

Mucha, A. P., Almeida, C. M. R., Bordalo, A. A. & Vasconcelos, M. (2005) Exudation of organic acids by a marsh plant and implications on trace metal availability in the

rhizosphere of estuarine sediments. *Estuarine Coastal and Shelf Science*, 65(1-2), 191-198.

Munoz, M. A., Rosales, R. M., Gabarron, M., Faz, A. & Acosta, J. A. (2016) Effects of the Hydraulic Retention Time on Pig Slurry Purification by Constructed Wetlands and Stabilization Ponds. *Water Air and Soil Pollution*, 227(9), 13.

Neori, A., Reddy, K. R., Ciskova-Koncalova, H. & Agami, M. (2000) Bioactive chemicals and biological-biochemical activities and their functions in rhizospheres of wetland plants. *Botanical Review*, 66(3), 350-378.

Nichols, P., Lucke, T., Drapper, D. & Walker, C. (2016) Performance Evaluation of a Floating Treatment Wetland in an Urban Catchment. *Water*, 8(6).

Nurnberg, G. K., LaZerte, B. D., Loh, P. S. & Molot, L. A. (2013) Quantification of internal phosphorus load in large, partially polymictic and mesotrophic Lake Simcoe, Ontario. *Journal of Great Lakes Research*, 39(2), 271-279.

Pick, F. R. & Lean, D. R. S. (1987) The Role of Macronutrients (C, N, P) in Controlling Cyanobacterial Dominance in Temperate Lakes. *New Zealand Journal of Marine and Freshwater Research*, 21(3), 425-434.

Price, D. S., Smith, D. W. & Stanley, S. J. (1995) PERFORMANCE OF LAGOONS EXPERIENCING SEASONAL ICE COVER. *Water Environment Research*, 67(3), 318-326.

Quinonez-Diaz, M. D., Karpiscak, M. M., Ellman, E. D. & Gerba, C. P. (2001) Removal of pathogenic and indicator microorganisms by a constructed wetland receiving untreated domestic wastewater. *Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering*, 36(7), 1311-1320.

Ramirez-Santiago, P., Velasco-Velasco, V. A., Ruiz-Luna, J., Enriquez-del Valle, J. R., Campos-Angeles, G. V., Rodriguez-Ortiz, G. & Preciado-Rangel, P. (2012) Inducing Nutrient Deficiencies of Nitrogen, Phosphorus, Potassium, Sulfur and Iron in Agave potatorum Zucc. *Ii International Symposium on Soilless Culture and Hydroponics*, 947, 249-254.

Reid, D. (2012) Standards and Guidelines for Municipal Waterworks, Wastewater and Storm Drainage Systems Edmonton, Alberta: Alberta Environment and Sustainable Resource Development, Regional Integration Branch, Operations Branch.

Revitt, D. M., Shutes, R. B. E., Llewellyn, N. R. & Worrall, P. (1997) Experimental reedbed systems for the treatment of airport runoff. *Water Science and Technology*, 36(8-9), 385-390.

Saeed, T., Al-Muyeed, A., Afrin, R., Rahman, H. & Sun, G. Z. (2014) Pollutant removal from municipal wastewater employing baffled subsurface flow and integrated surface

flow-floating treatment wetlands. *Journal of Environmental Sciences-China*, 26(4), 726-736.

Sartoris, J. J., Thullen, J. S., Barber, L. B. & Salas, D. E. (2000) Investigation of nitrogen transformations in a southern California constructed wastewater treatment wetland. *Ecological Engineering*, 14(1-2), 49-65.

Sasser, C. E., Gosselink, J. G., Swenson, E. M., Swarzenski, C. M. & Leibowitz, N. C. (1996) Vegetation, substrate and hydrology in floating marshes in the Mississippi river delta plain wetlands, USA. *Vegetatio*, 122(2), 129-142.

Sawyer, C. N., McCarty, P. L. & Parkin, G. F. (1994) *Chemistry for Environmental Engineering 4th Edition*. New York, NY: McGraw-Hill Inc.

Schussler, E. E. & Longstreth, D. J. (1996) Aerenchyma development in wetland plants. *Plant Physiology*, 111(2), 255-255.

Song, H. L., Li, X. N., Lu, X. W. & Inamori, Y. (2009) Investigation of microcystin removal from eutrophic surface water by aquatic vegetable bed. *Ecological Engineering*, 35(11), 1589-1598.

Soranno, P. A., Carpenter, S. R. & Lathrop, R. C. (1997) Internal phosphorus loading in Lake Mendota: response to external loads and weather. *Canadian Journal of Fisheries and Aquatic Sciences*, 54(8), 1883-1893.

Stanley, S. J. & Smith, D. W. (1992) LAGOONS AND PONDS. *Water Environment Research*, 64(4), 367-371.

Stottmeister, U., Wiessner, A., Kuschk, P., Kappelmeyer, U., Kastner, M., Bederski, O., Muller, R. A. & Moormann, H. (2003) Effects of plants and microorganisms in constructed wetlands for wastewater treatment. *Biotechnology Advances*, 22(1-2), 93-117.

Sun, L. P., Liu, Y. & Jin, H. (2009) Nitrogen removal from polluted river by enhanced floating bed grown canna. *Ecological Engineering*, 35(1), 135-140.

Tanner, C. C. & Headley, T. R. (2011) Components of floating emergent macrophyte treatment wetlands influencing removal of stormwater pollutants. *Ecological Engineering*, 37(3), 474-486.

Tao, W. D., Hall, K. J. & Ramey, W. (2007) Effects of influent strength on microorganisms in surface flow mesocosm wetlands. *Water Research*, 41(19), 4557-4565.

Todd, J., Brown, E. J. G. & Wells, E. (2003) Ecological design applied. *Ecological Engineering*, 20(5), 421-440.

Truu, M., Juhanson, J. & Truu, J. (2009) Microbial biomass, activity and community composition in constructed wetlands. *Science of the Total Environment*, 407(13), 3958-3971.

Van de Moortel, A. M. K., Meers, E., De Pauw, N. & Tack, F. M. G. (2010) Effects of Vegetation, Season and Temperature on the Removal of Pollutants in Experimental Floating Treatment Wetlands. *Water Air and Soil Pollution*, 212(1-4), 281-297.

Van Oostrom, A. J. (1995) Nitrogen removal in constructed wetlands treating nitrified meat processing effluent. *Water Science and Technology*, 32(3), 137-147.

Vymazal, J. (2007) Removal of nutrients in various types of constructed wetlands. *Science of the Total Environment*, 380(1-3), 48-65.

Wagner, M., Loy, A., Nogueira, R., Purkhold, U., Lee, N. & Daims, H. (2002) Microbial community composition and function in wastewater treatment plants. *Antonie Van Leeuwenhoek International Journal of General and Molecular Microbiology*, 81(1-4), 665-680.

Wagner, M., Rath, G., Koops, H. P., Flood, J. & Amann, R. (1996) In situ analysis of nitrifying bacteria in sewage treatment plants. *Water Science and Technology*, 34(1-2), 237-244.

Wang, C. Y. & Sample, D. J. (2014) Assessment of the nutrient removal effectiveness of floating treatment wetlands applied to urban retention ponds. *Journal of Environmental Management*, 137, 23-35.

Wang, C. Y., Sample, D. J. & Bell, C. (2014) Vegetation effects on floating treatment wetland nutrient removal and harvesting strategies in urban stormwater ponds. *Science of the Total Environment*, 499, 384-393.

Wang, D. Q., Bai, S. Y., Wang, M. Y., Xie, Q. L., Zhu, Y. N. & Zhang, H. (2012) Effect of Artificial Aeration, Temperature, and Structure on Nutrient Removal in Constructed Floating Islands. *Water Environment Research*, 84(5), 405-410.

Wen, L. & Recknagel, F. (2002) In situ removal of dissolved phosphorus in irrigation drainage water by planted floats: preliminary results from growth chamber experiment. *Agriculture Ecosystems & Environment*, 90(1), 9-15.

Werker, A. G., Dougherty, J. M., McHenry, J. L. & Van Loon, W. A. (2002) Treatment variability for wetland wastewater treatment design in cold climates. *Ecological Engineering*, 19(1), 1-11.

White, S. A. & Cousins, M. M. (2013) Floating treatment wetland aided remediation of nitrogen and phosphorus from simulated stormwater runoff. *Ecological Engineering*, 61, 207-215.

WHO (2002) *WHO (World Health Organization) Environmental Health*(CEHA), E. M. R. C. f. E. H. A.

Winston, R. J., Hunt, W. F., Kennedy, S. G., Merriman, L. S., Chandler, J. & Brown, D. (2013) Evaluation of floating treatment wetlands as retrofits to existing stormwater retention ponds. *Ecological Engineering*, 54, 254-265.

Wu, Q. T., Gao, T., Zeng, S. C. & Chua, H. (2006) Plant-biofilm oxidation ditch for in situ treatment of polluted waters. *Ecological Engineering*, 28(2), 124-130.

Xian, Q. M., Hu, L. X., Chen, H. C., Chang, Z. Z. & Zou, H. X. (2010) Removal of nutrients and veterinary antibiotics from swine wastewater by a constructed macrophyte floating bed system. *Journal of Environmental Management*, 91(12), 2657-2661.

Yang, Z. F., Zheng, S. K., Chen, J. J. & Sun, M. (2008) Purification of nitrate-rich agricultural runoff by a hydroponic system. *Bioresource Technology*, 99(17), 8049-8053.

Zhou, X. H. & Wang, G. X. (2010) Nutrient concentration variations during Oenanthe javanica growth and decay in the ecological floating bed system. *Journal of Environmental Sciences-China*, 22(11), 1710-1717.

Zhou, X. H., Wang, G. X. & Yang, F. (2012) Nitrogen Removal from Eutrophic River Waters by Using Rumex Acetosa Cultivated in Ecological Floating Beds. *Fresenius Environmental Bulletin*, 21(7a), 1920-1928.

Zhu, L. D., Li, Z. H. & Ketola, T. (2011) Biomass accumulations and nutrient uptake of plants cultivated on artificial floating beds in China's rural area. *Ecological Engineering*, 37(10), 1460-1466.

Appendix

Quality Control and Quality Assurance

Quality control was closely monitored throughout the study period to ensure consistent and accurate water quality analysis. Analysis for TSS, BOD, and Fecal Coliforms followed the procedure outlined in Standard Methods for the Examination of Water and Wastewater (Eaton, 2005), and analysis for NH_4^+, NO_3^-, TN, RP, and TP followed their respective Hach methods as outlined in Chapter 3 (Materials and Methods) (Hach, 2013).

Before sampling began in June 2015, nutrient analysis was performed on standard solutions to ensure accuracy within 5% of the target concentration. Standard solutions used were 3mg L^{-1} PO_4^{3-} for RP and TP analysis, 10mg L^{-1} NH_4^+-N for NH_4^+ and TN analysis, and 1000mg L^{-1} NO_3^-–N (diluted to 10mg L^{-1}) for NO_3^- analysis. A standard solution of 20NTU was used to determine turbidity accuracy. During analysis, a sample from a standard solution was analyzed in parallel with wastewater samples. If the standard solution deviated greater than 10% from the expected value, then the analysis was repeated. For the first month of each field season (June 2015 and June 2016) every second wastewater sample was duplicated, to ensure the analysis method was consistent. If duplicated samples deviated greater than 10% from each other, then the test was repeated and the cause of inconsistency determined.

Once per month, a travel blank (1L of deionized water) carried in a plastic sampling bottle (identical to those used for wastewater sampling) was transported to the field site,

exposed to the open air (bottle placed on the ground near the sea-container, without any lid for 15 minutes), and returned to the laboratory for water quality analysis. The blank was never found to have a detectable concentration of nutrients, or any other contaminant analyzed during the study period.