

OPERATION AND OPTIMIZATION OF A TWO-STAGE, VERTICAL FLOW
CONSTRUCTED WETLAND SYSTEM AT
BRIDGER BOWL SKI AREA

by

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ABSTRACT

Treatment wetlands are an internationally accepted technology for treatment of domestic wastewater and modern designs have become the preferred option for small communities in several European countries for their ability to produce a high quality effluent. To evaluate performance of modern treatment wetland designs with respect to carbon and nitrogen removal in Montana and other challenging contexts, a two-stage, vertical flow system with recycle capabilities has been constructed and tested at a ski area near Bozeman, Montana. Site climatic and operational conditions provide a “worst-case” scenario to test the efficacy of treatment wetlands in Montana. Intensive sampling of influent and effluent concentrations of chemical oxygen demand (COD) and nitrogen containing compounds after the second season of plant growth is used to optimize and correlate performance as influenced by loading rate, dose volume, and recycle ratio. COD removal was greater than 90% and increased linearly with loading rate even when loading rate exceeded European design guidelines by nearly a factor of 10, with effluent concentrations approximately $100 \text{ mg}\cdot\text{L}^{-1}$. The system was also able to nitrify and denitrify. With the use of water recycling, effluent could be optimized for complete removal of ammonium and total nitrogen removal around 50%, even though influent concentrations were approximately 4 times greater than typical domestic wastewater. Mass removal rates were as high as $20 \text{ g}\cdot\text{N}\cdot\text{m}^{-2}\cdot\text{day}^{-1}$, higher than expected based on European guidelines. These results indicate that treatment wetlands are capable of high nitrogen and organic carbon removal, even when applied at a high concentrations, low temperatures, and variable flow situations.

INTRODUCTION

Natural wetlands have been utilized to treat wastewater effluents since the early days of municipal sewage collection and treatment (Kadlec & Wallace, 2009). More recently, wetlands have been constructed for the explicit purpose of treating not only domestic wastewater, but a variety of other wastewaters as well. Animal waste, mine drainage, industrial waste, and stormwater have also been treated in constructed wetland systems (CW) also known as treatment wetland systems (TW), but the majority of CW systems in the United States and abroad are utilized to provide secondary treatment of on-site and small-scale municipal wastewater effluents (Kadlec & Wallace, 2009).

Research and implementation of CW began in Europe in the late 1950's and 1960's, where researchers experimented with various construction techniques and loading schemes to treat animal waste. Use of CW slowly gained popularity in both Europe and North America through the 1970's and 1980's, and began to be utilized widely in the 1990's (Kadlec & Wallace, 2009). Designs have evolved as implementation and research have advanced. Current, state-of-the-art designs have become the preferred treatment option for small communities in several European countries (Molle et al., 2005).

Currently, the field distinguishes three main types of constructed wetlands; free water surface wetlands are characterized by water levels at a higher elevation than the substrate, which allows water to be exposed directly to the atmosphere. Horizontal subsurface flow wetlands feature water that flows below the substrate, horizontally across the wetland cell. Finally, vertical flow wetlands are characterized by water flow along the vertical plane, either top to bottom (downflow) or vice versa (upflow). Each of these

types of systems is planted with hydrophilic plants of one or multiple species, usually after establishment in a nursery, but sometimes as seed.

The nature of each type of treatment wetland system enhances certain characteristics, which can be leveraged to optimize treatment performance. Free water surface wetlands limit oxygen transfer to the substrate while maintaining a shallow aerobic layer on the surface, making them ideal for mine waste, landfill leachate, and industrial waters with contaminants that are best degraded anaerobically. These wetlands also maintain consistent water levels, even during unpredictable pulse loading, and therefore are commonly used for stormwater treatment as well. Horizontal subsurface flow wetlands are characterized by similar oxygen transfer conditions as free water surface wetlands, but are advantageous for applications where water must not be exposed on the surface. This flow design minimizes potential for human exposure to toxic metals and pathogens, and limits mosquito habitat as well. Finally, vertical flow wetlands can be utilized to optimize oxygen transfer processes for either highly aerobic or highly anaerobic conditions. To promote oxygen transfer, pulse loaded, downflow vertical flow wetlands provide highly aerobic conditions. In contrast, continuously loaded upflow vertical flow wetlands minimize oxygen transfer quite well. By selecting the wetland design best suited to achieve the conditions desired, high quality water treatment can be realized.

The current state-of-the-art for CW treatment of domestic wastewater has been developed in several European countries and features a two-stage, downward vertical flow treatment wetland system (Molle et al., 2005; Arias & Brix, 2005). Despite wide

acceptance in Europe, this technology has not been widely adopted in the USA and there are no such municipal systems in Montana, even though most municipalities produce a wastewater that could be efficiently treated with CW. To assess the applicability of CW in Montana, a two-stage downward vertical flow system was constructed in 2012 at Bridger Bowl, a non-profit ski area near Bozeman, Montana, USA. The construction and concurrent research has been funded by the Montana Department of Environmental Quality (MDEQ) to test the efficacy of treatment wetlands in Montana's climate, and, if suitable, to provide a framework for developing regulatory guidelines for treatment wetlands in Montana.

Bridger Bowl provides a unique study site because it features many challenges to effective wastewater treatment, thus it inherently provides a conservative estimate for treatment wetland efficacy in Montana. The ski area operates from early December until early April each year, therefore generating almost all wastewater during the coldest months of the year. Furthermore, because Bridger Bowl has a high altitude (1966 m, 6450 ft.), northerly latitude ($45^{\circ}49'01''$), and continental climate, snow cover lingers well into spring and returns early in the fall. The growing season is relatively short, typically lasting only from early June to late September, which further challenges plant establishment and growth. Additionally, the waste stream is generated primarily from toilet and commercial kitchen waste and Bridger Bowl's management employs water conservation measures, so it is considerably more concentrated than typical domestic wastewater. Chemical oxygen demand in the septic water which feeds the wetland system exceeds $800 \text{ mg}\cdot\text{L}^{-1}$, and ammonium levels exceed $150 \text{ mg}\cdot\text{L}^{-1}$ for most of ski season.

Depending on the number of skiers, flow varies widely from 6,000 to 40,000 L·d⁻¹ (~1,500 to 10,000 gal·d⁻¹). During the summer months, when only a few, year-round employees are utilizing the system, average flow rates are approximately 1000 L·d⁻¹ (250 gpd). These factors make Bridger Bowl a challenging location to treat wastewater, but provide a good opportunity to test treatment wetland systems under challenging conditions.

The objectives of this study are threefold. The first is to evaluate the applicability of wetland systems in Montana's climate. There have been concerns that operation of treatment wetlands may be challenging or impossible due to problems posed by harsh climates, despite many similar systems operating in cold climates in other states and countries. This study seeks to confirm that treatment wetlands can not only function in sub-freezing temperatures, but also provide high levels of treatment for typical secondary treatment parameters. Additionally, the ability of wetland plants to establish and grow well given the environmental challenges of the site has been questioned, so plant growth and performance have been evaluated. Finally, this study investigates how operational parameters can be varied to optimize the system for total nitrogen removal. Regulatory agencies in the United States are increasingly looking at nitrogen removal as a promising method of preventing and reversing eutrophication of waters receiving waste effluents. Both at Bridger Bowl and across Montana, treatment of nitrogen in other types of wastewater systems has proved challenging, yet nitrogen regulations are becoming increasingly stringent.

This thesis will consist first of a literature review that examines research conducted on treatment wetlands with a focus on nitrogen removal for systems treating domestic waste and investigates design parameters and guidelines that have been established elsewhere for downflow vertical flow treatment wetland systems. Next, the experimental methods utilized in this study will be discussed, followed by the results and discussion of these experiments. Finally, a conclusion coalesces the presented tests into a unifying message.

LITERATURE REVIEW

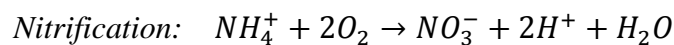
Constructed wetlands have been utilized to treat domestic wastewater for decades, but achieving high rates of nitrogen removal has not been a great concern until recently. In recent years, however, considerable research has been conducted in an effort to determine which design parameters and operational conditions can enhance nitrogen removal. This has included manipulating loading schemes, utilizing different types of plants, building with unconventional substrate, and manipulating carbon to nitrogen ratios. Improvements in nitrogen removal have been observed in many systems, but considerable opportunity for improvement of treatment performance and prediction capabilities remains.

Nitrogen exists in wastewater in numerous forms, but is overwhelmingly found in typical raw domestic wastewater as urea or part of organic molecules. In most systems, both of these compounds are easily and rapidly degraded to ammonium in the early stages of treatment. There are numerous biotic and abiotic pathways that are then capable of transforming ammonium to nitrogen forms that are released as gas to the atmosphere. In treatment wetlands systems, classical microbial nitrification and denitrification is typically assumed to be the primary nitrogen removal mechanism (Arias, et al., 2005; Brix et al., 2003; Langergraber et al., 2009), but other microbial pathways, including anaerobic ammonium oxidation (“ANAMMOX”), completely autotrophic nitrite removal over nitrate (“CANON”), and partial nitrification-denitrification have been identified in constructed wetland systems (Faulwetter et al., 2009; Saeed & Sun, 2012). Additionally,

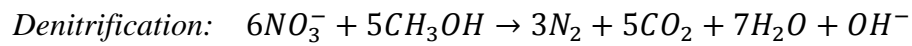
non-microbial mechanisms such as physical adsorption, plant uptake, and ammonium volatilization have been observed (Davis, 2013; Saeed & Sun, 2012).

Constructed wetlands are dynamic, living systems that rely on complex interactions between biotic and abiotic mechanisms to provide treatment. It can be assumed that each of the aforementioned removal mechanisms will be taking place to some degree somewhere in the system at any given time. Researchers have thus attempted to improve performance by optimizing the system to enhance the removal pathways that are assumed to be most dominant (Kadlec & Wallace, 2009). The relatively rapid kinetics of traditional nitrification and denitrification implies that these pathways are likely to provide the majority of nitrogen removal (Faulwetter et al., 2009). The long-term nature of treatment operations implies adsorption and plant uptake are incapable of providing significant nitrogen removal due to the fact that they do not provide a mechanism for nitrogen to escape the system, but rather store it away from water for some period of time. In taking all of this into consideration, it is reasonable and prudent to assume that optimization of classical microbial nitrification and denitrification pathways is the best method to improve treatment performance for total nitrogen (TN) removal (Fuchs et al., 2012).

In any wastewater treatment system, managing oxygen and carbon supplies is the greatest challenge to achieving high total nitrogen removal rates via the nitrification and denitrification pathways. The autotrophic nitrification process oxidizes ammonium to nitrate, and requires aerobic conditions, as shown below (Saeed & Sun, 2012):



Following nitrification, nitrate can be biologically reduced to dinitrogen gas by the heterotrophic denitrification process, which requires anoxic conditions and the presence of organic carbon (Saeed & Sun, 2012):



The challenge arises because the incoming wastewater is high in both organic carbon and ammonium. Aerobic heterotrophic organisms generally outcompete the autotrophic nitrifiers for available oxygen, thus the organic carbon is mostly consumed prior to the nitrification step, leaving too little organic carbon and/or too much oxygen for the heterotrophic denitrification process. As a result, numerous methods of carbon and oxygen management have been employed to ensure these reactants are available at the right stage so that microbial degradation of nitrogen compounds may occur.

The importance of carbon and oxygen in coupled nitrification and denitrification processes was underscored in a comparative study of constructed wetlands by Fan et al. (2013b). Two sets of bench scale systems were treated with four different synthetic wastewaters with C:N ratios of 2.5, 5, 10, and 20; one set was aerated intermittently, the other was not. Average total nitrogen removal levels were approximately 60% for reactant limited systems (unaerated and/or low carbon), but systems with both aeration and high carbon supply averaged 90% total nitrogen removal. Recognition of the importance of carbon and oxygen supply has led to the implementation of two fundamental strategies for operation to achieve total nitrogen reduction; the recycling of nitrified effluent and the use of pulse loading schemes.

It should be noted that considerable differences in regulatory requirements occur in different countries. For example, the United States regulates both ammonium and nitrate and hence total nitrogen removal (Federal Water Pollution Control Act, 2002), whereas Denmark, for instance, sets lower removal requirements for total nitrogen, and does not regulate nitrate at all (Brix & Arias, 2005). Varying jurisdictional requirements for treatment performance have created many different, localized strategies for optimization of different treatment processes. In the US, removal of nitrate to meet regulatory requirements has proved challenging, so designs in this context should focus on creating conditions to foster both nitrification and denitrification.

In pursuit of these various treatment requirements, several countries have created guidelines to aid in the design of treatment wetlands for domestic wastewater treatment. Most guidelines prescribe a wetland surface area based on the population equivalent (PE) of the community or facility served, which is an assumed average nutrient and hydraulic load introduced to the treatment system by one user in a day. Values for PE vary by country, but are generally 60-120 g·d⁻¹ biological oxygen demand (BOD), 10-15 g·d⁻¹ TN, and a flow of 150 L·d⁻¹ (Brix & Arias, 2005; Brix & Johansen, 2004; Molle et al., 2005; ONORM B 2505, 2005). Minimum surface area requirements for vertical flow systems treating domestic waste vary between 2 m² PE⁻¹ in France (Molle et al., 2005) and 4 m² PE⁻¹ in Austria (ONORM B 2505, 2005) but have generally trended downward as more research has been done and better optimization procedures have been developed. Additionally, relatively large variability exists in media depth requirements; the Danish guideline is 1 m of media depth (Brix & Johansen, 2004), but the French guideline is only

0.3 m (Molle et al., 2005). Media size is similar across all design guidelines, and ranges between 0.25 and 8 mm, (medium sand to medium gravel, respectively). Uniformity of the media is important to avoid clogging, so some guidelines prescribe a minimum uniformity coefficient as well (Brix & Johansen, 2004). The differences between current guidelines for a prescribed area and depth are substantial; resulting in very different material and areal requirements. These components represent two major project costs, and hence critically affect acceptability of the technology. France seems to have adopted treatment wetlands for municipal systems most aggressively, likely due in part to the smaller suggested size. Furthermore, the French systems typically use the wetlands for both primary and secondary treatment, a design that further reduces total system infrastructure cost, whereas all other countries continue to require separate primary treatment prior to the wetland.

The French systems are also the only systems that are prescribed in the guideline as a two-stage process (i.e. two vertical flow wetland cells in series), and these types of systems have been fittingly referred to as ‘French style (CW) systems’ in literature (Molle et al., 2005). Studies show that two-stage systems are more efficient on an aerial basis than their single stage counterparts (Langergraber et. al., 2009), so this may explain why the French guidelines require less area and media than others. In light of this research, more studies have examined various combinations of systems with multiple stages in hopes of further improving treatment performance and/or reducing the footprint requirement (Molle et al., 2008). These are typically called hybrid systems when they combine two wetlands of different flow types (i.e. horizontal and vertical, or free water

surface and vertical), but the term multi-stage is also used, and encompasses systems with more than two stages, as well as systems with two stages of the same flow type. A recent review of such systems found that hybrid and multi-stage systems exhibited greater TN removal capacity when compared to single stage systems (Vymazal, 2013). Mean TN removal rates for the multi-stage systems surveyed varied between 2 and 5 $\text{g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ depending in part on the design, compared to only 1.13 and 1.85 $\text{g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ for single-stage horizontal and vertical flow systems, respectively. One recent study observed 80% TN removal in a three-stage system without recycle in a cold climate, with effluent ammonium values below 5 $\text{mg-N}\cdot\text{L}^{-1}$, even during winter (Vymazal, 2015). The potential for hybrid and multi-staged systems to remove total nitrogen is high, but is not the only method for enhancing nitrogen removal.

In hopes of maximizing the potential of single stage constructed wetlands, a few recent studies have created saturated zones within the bottom portion of vertical downflow wetland cells. This creates an anoxic, saturated zone at the bottom of the cell, while preserving a free-flowing, aerobic zone in the upper portion of the cell. Thus, the different aerobic/anoxic conditions of a hybrid or multi-stage system are contained in a single stage. This is a novel approach, and thus data are sparse, but initial results indicate that the saturation of a relatively small portion of the cell can improve denitrification processes without greatly impacting other processes (Arias, personal communication; Langergraber et al., 2011; Panuvatvanich et al., 2009; Silveria et al., 2015). Impoundment does, however, prevent utilization of drain pipes for aeration, a common strategy to promote oxygen transfer throughout the full depth of the wetland. This has the potential

to reduce oxygen transfer and inhibit oxygenation in the upper reaches of the cell to some extent (Arias, personal communication). In order to mitigate this, it is possible to include a passive aeration pipe network in the middle of the cell (Silveria et al., 2015). Tracer tests conducted on partially saturated wetland cells indicate that global mixing within the cell increases when the cell is impounded, but local mixing decreases (Giraldi et al., 2008). Thus, flow under saturated conditions diverges from ideal plug flow condition, which could reduce treatment performance in some respects. These negative impacts may be acceptable in certain circumstances, as impoundment has been shown to improve removal of nitrogen (Silveria et al., 2015).

Several recent studies have examined the effect of recirculating nitrified effluent from constructed wetland systems back to the influent of the wetlands. This causes mixing of nitrate-rich recirculation water with the organic carbon of the untreated influent, creating conditions in which denitrification can place. The data indicates substantial gains in total nitrogen removal can be made by recirculation of water. Although performance of different systems varied, total nitrogen removals in the 60-80% range are common (Arias et al., 2005; Laber et al., 1997; Tuncsiper, 2009). One study, utilizing a three-stage system that was operated between 50% and 100% recycle (Recycle/Septic) achieved overall ammonium and nitrate reductions of 91% and 89% respectively (Tuncsiper, 2009). Laber et al. (1997) experimented with a single-stage CW operated with 0% and 80% recycle and saw total nitrogen removal more than double and inorganic nitrogen removal more than quadruple with recirculation. In another study, total nitrogen removal was increased from only 1% without recycle to nearly 67% with

200% recycle (Arias et al., 2005). This study also compared recycle rates of 100% and 300%. Gains in total nitrogen removal were observed when recycle rate was increased from 100% to 200%; however, further increasing the recycle rate to 300% did not achieve significantly different treatment levels.

Recycle schemes have increased total nitrogen removal over treatment designs without recycle in all cases (Arias et al., 2005), thus it is prudent to incorporate the ability to recycle water in any constructed wetland system where total nitrogen removal is the goal. It should be noted that while recycle increases total nitrogen removal by stimulating denitrification, it may decrease performance for BOD, chemical oxygen demand (COD), and nitrification. Recycling water necessarily increases the total hydraulic loading on the system compared to a single pass, decreasing the residence time for removal of other parameters and creating a dilution effect which likely reduces the rates of other pollutants' removal. These interactions make optimization of recycle rate quite complex and dependent on both operational and design factors such as hydraulic and nutrient loading rate, form of nitrogen applied, wetland design, the ratio of various pollutants to be removed etc., and also on the regulatory environment dictating which parameter is most critical. Furthermore, the optimal recycle rate is likely system dependent so testing and calibration provides the best means of achieving maximum removal efficiency. In lieu of using such an intensive evaluation process, existing literature shows that recycle rates of 100-200% are appropriate (Arias et al., 2005).

Pulse loading of constructed wetland beds has been established as an effective, relatively low cost method of increasing oxygen supply in vertical downflow treatment

wetlands (Brix et al., 2003; Kadlec & Wallace, 2009; Langergraber et al., 2007; Tanner et al., 1999). Oxygen diffusion and entrapment in the media is facilitated by alternating wet and dry periods several times a day. When water is applied to the system, it moves through the bed unimpeded and drains freely, maximizing oxygen exposure within the bed. Nitrification takes place readily in aerobic conditions, and therefore proceeds well in pulse-loaded systems.

Some studies have further facilitated oxygen transfer by adding aeration systems. Continuous aeration has the ability to ensure nearly complete (99%) nitrification. This inhibits denitrification, however, so total nitrogen removal has been observed to be higher in systems that are intermittently aerated (Fan et al., 2013a). Total nitrogen removal in that study was maximized in an intermittently-aerated, frequently-dosed system at 82%. While this is a high level of treatment, similar levels have been achieved using less energy intensive methods (Tuncsiper, 2009) and without a mechanical component with high operating costs.

Constructed wetlands are typically operated outdoor year-round and therefore can be subjected to harsh environmental conditions. In continental climates, cold winter temperatures are a concern. As temperature drops, microbial processes have been observed to slow or stop all together, and wetlands plants undergo senescence (Faulwetter et al., 2009; Kadlec & Wallace, 2009). This may lead to an inference that constructed wetland systems in cold climates would not function well; however, many systems provide effective treatment in such conditions (Chouinard et al., 2015; Speer et al., 2012; Werker et al., 2002). It is typical to note seasonal changes in treatment levels,

due to not only water temperature, but also due to plant seasonal cycles and other environmental conditions (Kadlec & Wallace, 2009).

Conflicting opinions exist as to the importance of water temperature, and what minimum operating temperatures might be with respect to nitrogen removal processes, but it is well known that organic carbon removal in wetlands is not inhibited by cold temperatures, and may in fact be enhanced due to increasing availability of oxygen at colder temperatures as a result of higher solubility and/or transfer (Kadlec & Reddy, 2001; Stein et al., 2007; Stein & Hook, 2005; Werker et al., 2002). Some researchers have noted declining nitrogen treatment levels below water temperatures of 12°C, whereas others have achieved satisfactory levels of treatment at temperatures as low as 2°C (Brix et al., 2003; Langergraber et al., 2007). Freezing of entire wetlands systems is possible under especially extreme conditions, and is challenging to reverse, given the high specific heat of water (Kadlec, personal communication). However, a study of 12 systems in mountainous regions in France found the systems consistently performed well; freezing was never a problem and nitrification dipped only slightly during the winter and only for loads greater than 10 g N·m⁻²·d⁻¹, which is more than double the recommended design load (Molle et al., 2005; Prost-Boucle et al., 2015). Duration and intensity of the cold period also seem to effect treatment. Performance during moderately cold temperatures for long durations appears to be sustainable, but sudden, extremely cold events cause depressed treatment, and require some recovery time when warming occurs (Langergraber et al., 2009). It has been shown that two-stage systems have more inherent ability to buffer against the effects of temperature, providing increased stability and faster

recovery at cold temperatures (Langergraber et al., 2008). However, variability in treatment level has been observed to increase as temperature drops (Langergraber et al., 2007). There is more work to be done investigating the changes in treatment as a result of cold temperature, but it has been shown that CW systems are effective in cold temperature water treatment scenarios.

The effect that plants have on treatment in CW systems has been debated for more than a decade, and many questions have yet to be resolved. Numerous studies have confirmed that systems with plants provide higher levels of treatment than similar systems without plants (Canga et al., 2011; Fan, et al., 2013a; Kadlec & Wallace, 2009; Taylor et al., 2011; Tuncsiper, 2009). Researchers have proposed that plants aid in transfer of both oxygen and carbon into the substrate of CW, but these theories have proved challenging to confirm (Faulwetter et al., 2009), although elevated oxidation-reduction potentials and dissolved oxygen concentrations are widely observed (Fan, et al., 2013a; Tuncsiper, 2009; Wu et al., 2013), and seem to implicitly confirm these hypotheses. The mechanisms and rates of oxygen transfer, however, remain unclear and challenging to measure; reported values for plant oxygen transfer rates in the literature vary by as much as three orders of magnitude (Kadlec & Wallace, 2009).

Despite widespread evidence on the importance of plants, the influence of plant species selection has been largely ignored. European researchers have generally planted CW with *Phragmites australis* (Arias et al., 2005; Brix et al., 2003; Langergraber et al., 2010; Tuncsiper, 2009) and have overlooked other plants to the point where some use the terms “constructed wetland” and “reed bed” interchangeably, despite the fact that there

are many other wetland plants that can be, and are, used in other locales. Virtually all studies with a plant species component in the experimental design confirm that plant presence (versus absence) improves performance for virtually all contaminants but conclusions from field studies on specific species selection are inconclusive as to which are best (Brisson & Chazarenc, 2008). Some bench studies indicate that species selection affects treatment performance with especially pronounced differences in treatment occurring during the colder months in temperate climates (Allen et al., 2002; Taylor et al., 2011). Unfortunately, many bench and pilot scale studies fail to account for seasonal effects at all, with studies being conducted at room or summer temperatures over relatively short time intervals (Fan et al., 2013a; Fan et al., 2013b; Fuchs et al., 2012).

Constructed wetlands researchers have also observed that systems require a significant “start-up” period before a steady-state treatment level is reached. During this time, plants become established and mature, and sorption sites are occupied by contaminants. This process can take months or years (Kadlec & Wallace, 2009). Despite this, numerous studies fail to account for an adequate start-up period (Fan, et al., 2013a; Fan, et al., 2013b; Tuncsiper, 2009; Wu et al., 2013). Data from such studies, therefore, fail to take into account how systems may perform over a longer, more realistic lifespan.

In summary, nitrogen treatment in wetlands has proved challenging, but recent research has made headway toward improved techniques for removal of nitrogen-containing compounds. Downflow vertical systems utilizing a pulse loading scheme have largely replaced horizontal flow systems because of their ability to increase oxygen transfer and ensure conversion of ammonium to nitrate. In the same vein, forced aeration

has been used to facilitate oxygen transfer in some studies but at a higher operational cost. These operations have been shown to increase dissolved oxygen levels and reduce ammonium levels. Recycling nitrified effluents back to the system inflow has proved particularly effective at improving total nitrogen removal. This strategy combines carbon-rich influent waters with nitrate laden recycle waters to create conditions for denitrification to take place. While effective at providing total nitrogen treatment, recycling water, especially at high rates, may decrease performance for other parameters.

There are also several areas where researchers have yet to come to definitive conclusions. The importance and effect of temperature is one example. It is known that microbial and plant metabolic pathways slow or stop at low temperatures, yet much data indicate that it is possible to continue treatment at near-freezing temperatures. The reasons why effective treatment continues in cold weather treatment remain unclear, but gaining greater understanding in this area would allow researchers and operators to achieve more effective treatment with greater consistency. Additionally, the importance of plant species on treatment performance remains unsettled. Some bodies of data indicate that plant species affects treatment levels significantly during the winter, but many researchers continue to operate under the assumption that plant species selection does not significantly affect performance. More research is needed in this area before consensus can be reached.

METHODS

System Design and Construction

The Bridger Bowl treatment wetlands are a two-stage system with two cells in parallel at each stage for a total of four cells (Figure 1). All design parameters, including the physical dimensions, were drawn from the latest published scientific literature, consultations with international collaborators, and nearly a decade of record of Bridger Bowl's wastewater characteristics. Each of these four cells is square, with a surface area of 23.8 m² (256 ft²), and has vertical sidewalls approximately 1.1 m deep.

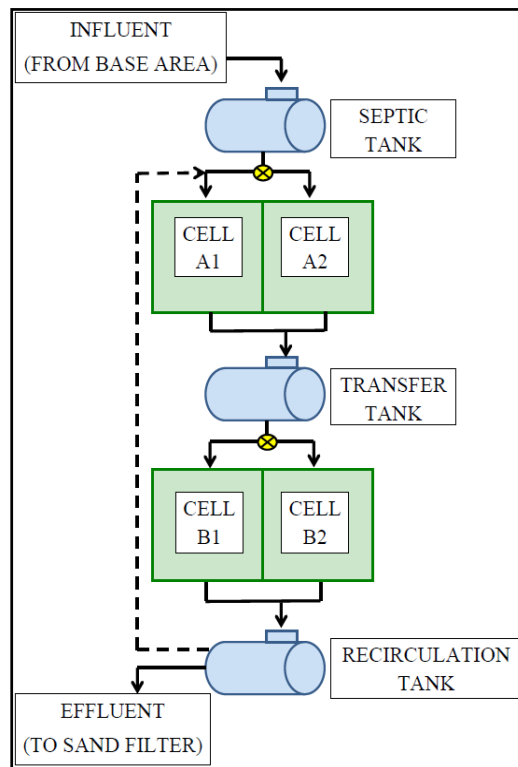


Figure 1: Bridger Bowl treatment wetland flow schematic.

The dimensions were selected to be representative of a small to medium sized full-scale system, but the system does not treat all of the wastewater stream at Bridger Bowl. A design flow rate of $3800 \text{ L}\cdot\text{d}^{-1}$ ($1000 \text{ gal}\cdot\text{d}^{-1}$) split evenly between the parallel trains of the system results in loading rates of $38\text{g COD m}^{-2}\cdot\text{d}^{-1}$ and $7\text{g TN m}^{-2}\cdot\text{d}^{-1}$ for the system as a whole, because the seasonal average concentrations exiting Bridger Bowl's septic tanks are approximately $1,000 \text{ mg}\cdot\text{L}^{-1}$ COD and $180 \text{ mg}\cdot\text{L}^{-1}$ total nitrogen. These loading rates are on the higher end of European guidelines developed for less extreme environmental conditions. The loads can be increased or decreased as part of the experimental protocols, because Bridger Bowl has an independently permitted and redundant wastewater system, thus the experimental wetland system does not need to treat all the influent water, nor does it need to meet any specific discharge requirements. Duplicate cells in parallel allow the system to be utilized for testing and comparison of different loading scenarios under identical environmental conditions and influent concentrations.

The cells are lined with a geotextile fabric and contain (top to bottom); a gravel protective layer, distribution pipes, a treatment layer, drain rock, and drain pipes (Figure 2). The treatment layer of the first cells in series ("A" cells) feature a coarse media (gravel, $d_{50}\approx 5 \text{ mm}$), while the treatment layer in second series ("B" cells) is a medium sand ($d_{50}\approx 0.6 \text{ mm}$). This is the only physical difference between the two stages of cells. The treatment zones of both cells lie above two, 10 cm layers of coarse drainage rock ($d_{50}=5\text{-}10 \text{ cm}$) and pea gravel ($d_{50}= 1.0 \text{ cm}$) and beneath a gravel protective layer ($d_{50}\approx 5 \text{ mm}$) that provides thermal insulation and a barrier to direct human contact with

wastewater. The network of distribution pipes on top of the treatment layer consists of a 2 inch main line in the center of the cell, from which 1 ¼ inch lines branch off laterally at 50 cm intervals. These lateral distribution lines have 8 mm orifices drilled every 33 cm along the length of the pipe to discharge water. These orifices were covered with protective caps to prevent clogging, and the distribution laterals have cleanouts on each end. Drain pipes lay on the bottom of the cell in a similar pattern, but with laterals placed at 163 cm intervals and constructed from 4 inch standard perforated drain pipe. All cells were fitted with an Agri-drain© weir water level control mechanism to allow manual changes to water surface height within the wetland cells.

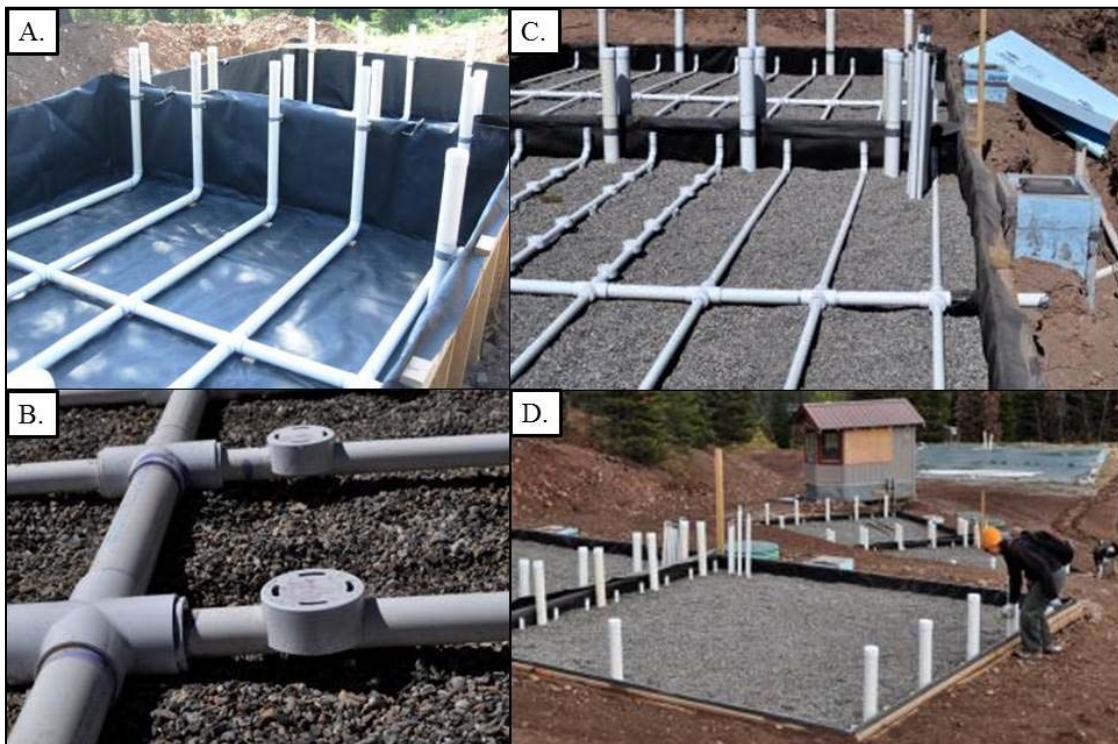


Figure 2: Components of the system. Geotextile liner and drain pipe network (A), distribution main line and laterals with orifice caps (B), two cells with treatment media and distribution networks (C), and final grade with distribution network covered by a protective layer (D).

Several tanks, totaling 87,000 L (23,000 gal), receive all the raw wastewater from the base area buildings, which includes ticket concessions, restaurants, bars, ski shops, maintenance facilities, and administrative offices. After clarification in these tanks, this septic water is pumped to a 3,800 L (1,000 gal) pump tank at the wetlands site. This pump applies the clarified septic water to the first stage (A cells) of the system. A 3,800 L transfer tank mixes the effluent from both A cells and is the tank from which the second stage (B cells) are loaded. A 5,700 L (1,500 gal) recirculation tank receives the effluent from the B cells and is used to recycle the partially treated wastewater back directly to the A cells (Figure 1). All 3 tanks have overflow pipes that drain to the existing permitted system, a recirculating sand filter, allowing any excess water to leave the wetlands treatment system from any tank.

V-notch weirs were built from 10 inch pipe to provide flow measurements at the effluent of each wetland cell (Figure 3). These weirs sit inside the tanks and capture all the effluent from each cell before it flows into the tank. Infrared sensors were calibrated and installed to provide continuous monitoring of water surface changes in each weir as water flows out of the cells.

When the media was being placed in the cells during construction, an in-cell sampling system was installed to sample water from depths of 7, 12, and 20 cm below the top of the treatment media. The system, modeled after another previous developed (Jamie Nivala, personal communication), consisted of water collection “trays” made from a 61 cm section of 6 inch polyvinyl chloride (PVC) pipe capped on the ends with the middle 46 cm cut as a half-pipe. A slotted, 1.5 inch PVC pipe was inserted along the bottom of



Figure 3: Cell outflow weirs before installation, with infrared sensor mounts attached, but without sensors installed.

the tray and each tray collects water at the specified depths (Figure 4). The smaller pipes were connected to a vertical pipe extending above the gravel media via a tee, which provides a small reservoir from which to sample collected water. The diverter also has a cleanout pipe to allow for clearing of roots that may grow into the smaller slotted pipe in the future. The water collected in the tray, after filling the sampling reservoir, flows out an overflow pipe back into the wetland.



Figure 4: In-cell sample collection tray shown during installation.

Wetland construction was finished and the system was hydraulically operational just prior to the 2012-2013 season, but it remained unplanted and installation of many of the experimental components of the system, such as flow measuring devices, a water quality automatic sampling unit, and automated control systems were not yet completed. Though unintended, the grab sample water quality data collected during this time by another student (Davis, 2013) provides a baseline period to compare performance of the system with and without plants.

Following the spring thaw in June 2013, the system was planted with approximately 4,000 individual, bare-root plants on approximately 15 cm centers. *Carex utriculata* and *Schoenoplectus acutus*, two plants native to the region, were selected based on their superior performance for both COD reduction and nitrogen removal in previous studies at MSU (Taylor et al., 2011). The plants were grown and delivered as bare-root stock by North Fork Natives, a wetlands nursery in Idaho that specializes in growing wetlands plants native to the western United States. Each of the four cells was planted in quadrants with a monoculture of each species in a checkerboard pattern within each cell, so both cells in each stage (i.e. A1 and A2) are identical, and each contains two quadrants of each species. The collection pipes in each cell collect water from all four quadrants, thus effluent from the cells represents an average performance of both plant species. However, quadrants were arranged so that the in-cell sampling ports were under one monoculture of each species in both stages of the system, which allows the assessment of individual species effects.

Immediately after planting, water levels were raised to within 5 cm of the surface to provide plants with ample water for establishment. During this time, leaks were discovered at the junction between the cell liners and inlet and outlet pipes. This had been unnoticed the previous year because the system remained unsaturated and water losses were negligible, but it resulted in extensive water losses and lower than desired water levels when the cells were fully saturated. In the days immediately following planting, the area around the inlet and outlet piping was excavated so leaks could be identified and repaired. After repair, gravel was replaced, plants were replanted along the original grid at the original density, and water levels in the cells were raised again. Over the course of the summer, water levels were incrementally dropped in 12 or 18 cm increments to encourage deep root growth.

The design and installation of the supervisory control and data acquisition (SCADA) system and flow measuring devices was also completed during the summer and fall of 2013. The SCADA system uses Labview© software and provides operational control over all pumps, valves, and the automatic sampler. Additionally, the program monitors and stores data collected from pumps and weir sensors.

Plant coverage surveys were conducted to evaluate plant growth and establishment after the first growing season in the fall of 2013, and repeated annually late in the growing season thereafter. The total number of distinct stems in each cell was counted for both species in 2013 and for *Schoenoplectus* in 2014 and 2015. However, due to the high density of *Carex* in 2014 and 2015 counting all of the stems for the entire system was impractical, so stems were counted in smaller subsections of each quadrant.

These subsections were chosen semi-randomly by tossing a hoop into each quadrant. All plant counts were conducted by multiple individuals, and results averaged together for each quadrant. Plant density was then calculated for both counting methods by dividing the number of stems counted by the applicable area surveyed.

In the summer of 2015, after two growing seasons in which *Schoenolpectus* grew poorly, project collaborators collectively decided to plant *Carex* in the quadrants that were originally planted with *Schoenolpectus*. Approximately 1,750 additional *Carex* were planted between the living *Schoenolpectus*, plants on approximately the same 15 cm grid used for initial planting. Following planting, the water levels in the cells were raised to less than 5 cm below the wetland surface for 12 weeks.

Operation

The wetland system began operation on January 14th 2013 and was operated throughout the remainder 2012-2013 ski season as a gravel filter (unplanted). Three different operational schemes with different hydraulic loads, dose volumes, and recycle ratios were tested on each of the cells (Table 1). These operational schemes are labeled in three parts; the first is a two digit year that indicates the year the season started, the second is a letter that corresponds to the cell tested (A or B cell), and the final is a numerical value that increases for each scheme tested in a given season. Note that odd numbers have been made to correspond to odd cells A1 or B1 and even numbers to even cells A2 or B2. Data for the 2012-13 season was collected by and reported in a MS Professional Paper of a graduate student previously working on the project (Davis, 2013).

The system was operated after one season of plant growth during the following 2013-14 season. Four different operational schemes were tested on each cell. At the beginning of the season, the system was hydraulically operated in a manner similar to the prior season. As the season progressed, dose volumes were decreased and recycle was eliminated (Table 1). Only minimal grab sample data was collected during this season, as most effort was placed on perfecting the automated controls for the operation and the automated data acquisition system. Most operational challenges were overcome by the end of the 2013-2014 season, therefore extensive sampling utilizing the composite autosampler was conducted during the 2014-15 ski season in which eleven different operational schemes were tested on the A cells and ten different operational schemes on the B cells to evaluate the effect of dose size, recycle ratio, and daily hydraulic load. Note names 14-B-9 and 14-B-11 were not utilized in order to preserve parity between scheme number and cell label.

Water Quality

An automated sampling system was constructed to allow sampling in the remote environment. Samples were drawn via a diaphragm pump from sample lines plumbed to each of the three water distribution tanks and the effluent weirs of each cell. The collection lines were purged before and drained after each sample to ensure accuracy of the sample. Samples drawn from the cells using the autosampler were always timed so that the samples were taken from the effluent at a time point near the centroid of the

Table 1: Hydraulic loading rate (HLR), dosing frequency and depth for each wetland cell over the entire experimental period. Each cell and time period is given a unique identifier called a scheme that is used for future reference.

Scheme	Cell	Season	Start Date	Total Number of Doses per Day	Target Dose Load (cm)	Target Recycle Ratio (R:S)	Target Septic HLR (cm·d ⁻¹)	Target Recirculation HLR (cm·d ⁻¹)	Target Total HLR (cm·d ⁻¹)
12-A-1	A1	2012-13	1/14/2013	6	5.3	1:1	15.9	15.9	31.8
12-A-2	A2	2012-13	1/14/2013	6	5.3	1:1	15.9	15.9	31.8
12-A-3	A1	2012-13	1/21/2013	6	4.0	2:1	8.0	15.9	23.9
12-A-4	A2	2012-13	1/21/2013	6	4.0	2:1	8.0	15.9	23.9
12-A-5	A1	2012-13	2/10/2013	6	5.3	2:1	10.6	21.2	31.8
12-A-6	A2	2012-13	2/10/2013	3	5.3	2:1	5.3	10.6	15.9
12-B-1	B1	2012-13	1/14/2013	6	5.3	-	-	-	31.8
12-B-2	B2	2012-13	1/14/2013	6	5.3	-	-	-	31.8
12-B-3	B1	2012-13	1/21/2013	6	4.0	-	-	-	23.9
12-B-4	B2	2012-13	1/21/2013	6	4.0	-	-	-	23.9
12-B-5	B1	2012-13	2/10/2013	6	5.3	-	-	-	31.8
12-B-6	B2	2012-13	2/10/2013	3	5.3	-	-	-	15.9

Table 1 (Continued)

Scheme	Cell	Season	Start Date	Total Number of Doses per Day	Target Dose Load (cm)	Target Recycle Ratio (R:S)	Target Septic HLR (cm·d ⁻¹)	Target Recirculation HLR (cm·d ⁻¹)	Target Total HLR (cm·d ⁻¹)
13-A-1	A1	2013-14	11/15/2013	6	5.3	2:1	10.6	21.2	31.8
13-A-2	A2	2013-14	11/15/2013	6	1.9	2:1	5.3	10.6	11.2
13-A-3	A1	2013-14	2/21/2014	12	0.7	2:1	2.7	5.3	8.0
13-A-4	A2	2013-14	2/21/2014	12	0.7	2:1	2.7	5.3	8.0
13-A-5	A1	2013-14	3/26/2014	24	0.7	2:1	5.3	10.6	16.0
13-A-6	A2	2013-14	3/26/2014	24	0.7	2:1	5.3	10.6	16.0
13-A-7	A1	2013-14	4/7/2015	8	0.7	0:1	5.3	0.0	5.3
13-A-8	A2	2013-14	4/7/2015	8	0.7	0:1	5.3	0.0	5.3
13-B-1	B1	2013-14	11/15/2013	2	15.9	-	-	-	31.8
13-B-2	B2	2013-14	11/15/2013	2	8.0	-	-	-	15.9
13-B-3	B1	2013-14	2/21/2014	0	0.0	-	-	-	0.0
13-B-4	B2	2013-14	2/21/2014	24	0.7	-	-	-	16.0
13-B-5	B1	2013-14	3/26/2014	8	2.0	-	-	-	15.9
13-B-6	B2	2013-14	3/26/2014	8	2.0	-	-	-	15.9
13-B-7	B1	2013-14	4/7/2015	8	0.7	-	-	-	5.3
13-B-8	B2	2013-14	4/7/2015	8	0.7	-	-	-	5.3

Table 1 (Continued)

Scheme	Cell	Season	Start Date	Total Number of Doses per Day	Target Dose Load (cm)	Target Recycle Ratio (R:S)	Target Septic HLR (cm·d ⁻¹)	Target Recirculation HLR (cm·d ⁻¹)	Target Total HLR (cm·d ⁻¹)
14-A-1	A1	2014-15	12/10/2014	8	1.3	0:1	10.1	0.0	10.1
14-A-2	A2	2014-15	12/10/2014	16	0.6	0:1	10.1	0.0	10.1
14-A-3	A1	2014-15	1/22/2015	32	0.6	0:1	20.2	0.0	20.2
14-A-4	A2	2014-15	1/22/2015	32	0.6	1:1	10.1	10.1	20.2
14-A-5	A1	2014-15	2/19/2015	64	0.6	1:1	20.2	20.2	40.4
14-A-6	A2	2014-15	2/19/2015	64	0.6	1:1	20.2	20.2	40.4
14-A-7	A1	2014-15	3/11/2015	96	0.4	2:1	12.8	25.6	38.4
14-A-8	A2	2014-15	3/11/2015	96	0.4	1:2	25.6	12.8	38.4
14-A-9	A1	2014-15	3/15/2015	0	0	0:0	0	0	0
14-A-10	A2	2014-15	3/15/2015	192	0.4	1:1	38.4	38.4	76.7
14-A-11	A1	2014-15	3/25/2015	48	0.6	3:1	7.2	21.5	28.7
14-A-12	A2	2014-15	3/25/2015	48	0.6	3:1	7.2	21.5	28.7
14-B-1	B1	2014-15	12/10/2014	8	1.3	-	-	-	10.1
14-B-2	B2	2014-15	12/10/2014	2	5.4	-	-	-	10.8
14-B-3	B1	2014-15	1/22/2015	4	5.4	-	-	-	21.7
14-B-4	B2	2014-15	1/22/2015	3	3.4	-	-	-	10.2
14-B-5	B1	2014-15	2/19/2015	3	6.8	-	-	-	20.3
14-B-6	B2	2014-15	2/19/2015	4	5.4	-	-	-	21.7
14-B-7	B1	2014-15	3/18/2015	12	2.4	-	-	-	28.8
14-B-8	B2	2014-15	3/11/2015	4	5.8	-	-	-	23.2
14-B-10	B2	2014-15	3/18/2015	4	6.2	-	-	-	24.9
14-B-12	B2	2014-15	3/25/2015	12	2.4	-	-	-	28.8

hydrograph from each pulse dose, during two pulses each day, 12 hours apart. These two samples were combined in the same container and thus represent a daily average for the day they were collected. Samples from the dosing tanks were also time-averaged in a similar manner.

Additionally, grab samples were occasionally collected from the dosing tanks, effluent of the cells, and from sample ports within the cells. Samples from the tanks were obtained simply by submerging a bottle in the water, then transferring this sample to another container for transport. Effluent grab samples from the cells were taken as the effluent spilled through weirs by placing the collection vessel in the spillover water. Finally, samples from in-cell sampling ports were taken, after first clearing the sample port, using a syringe and tube to vacuum the water into the syringe, where it was then transferred to another container. Grab samples were compared, when possible, to samples collected by the autosampler to ensure consistency between the methods.

Water samples were either preserved at collection by reducing the pH of the sample to below 2 with sulfuric acid, or cooled in snow or ice prior to transport in coolers. Upon arrival at laboratory facilities, all samples were either transferred to refrigerators or freezers or immediately analyzed. Samples for COD, TKN, TSS and BOD were not filtered prior to testing; however, samples for ion chromatography (IC) analysis were filtered with 0.22 μm nylon membrane syringe filters prior to analysis. Deionized water was utilized in dilutions for BOD tests, and 18.2 M Ω reverse osmosis water was used for dilutions for TKN, COD, and IC tests.

Water quality parameters were analyzed using several methods. Chemical oxygen demand (COD) data was obtained using HACH Co. test kits (HACH Method 8000, EPA Standard Method 5220D). Ammonium, nitrate, and nitrite were analyzed using a Metrohm ion chromatograph (IC) with a Dionex AS-22 Fast 4x150mm column for anion analysis and either a Metrohm C4 2x150mm column or a Metrohm C6 4x250 mm column for cation analysis (EPA Method 300.1). Total nitrogen (TN) and total Kjeldhal nitrogen (TKN) were analyzed using HACH TNT800 kit (HACH Method 10242) which is a method with preliminary EPA approval for simultaneous analysis of both TN and TKN (Walker, 2013). Standard analysis methods were used to test biological oxygen demand (BOD) (EPA Method 405.1).

In many instances, water quality data was sorted by the scheme in which it was collected and subjected to statistical analysis. These analyses were completed primarily in Minitab 17® computing software, although some simple analyses were completed in Excel®. Comparisons between mean values and associated statements of significance were completed using the one-way analysis of variance (ANOVA) method with a two-sided, 95% confidence interval and Tukey's test.

Tracer Studies

Tracer studies were conducted in one cell of both stages (A and B) of the system during the summer of 2014 to gain insight into hydrologic and biological dynamics within the wetlands. Ammonium bromide was utilized as the tracer for its ability to act as both a conservative hydrologic and biologically available tracer. The bromide ion is well

accepted as a conservative tracer in wetlands systems, and was utilized for hydraulic modeling (Davis et al., 1985; Headley et al., 2002; Kadlec & Wallace, 2009).

For each cell, a mass of 1,000 g of ammonium bromide was measured and dissolved in approximately 2 L of deionized water. Application lines for each cell were tapped and connected to the ammonium bromide solution vessel with tubing and a control valve. This vessel was pressurized with compressed air, the main water pump was started, and the control valve opened, which quickly injected the high pressure ammonium bromide into the pumped water as it traveled towards the cell. The ammonium bromide solution was assumed to be well mixed in the pipe before the water and tracer arrived at the cell, which is approximately 100 linear feet of pipe for the A cells and 30 linear feet for the B cells. Dosing of ammonium bromide was terminated at least 60 seconds before the pumps turned off in an effort to ensure all the ammonium bromide was applied to the cell, but samples were taken in the dosing tanks following application to quantify any backflow of ammonium bromide.

Both cells were dosed with pulse loads, to simulate realistic wastewater application conditions. Cell B1 was dosed with 1900 L (8 cm) doses approximately every two hours for the 24 hour length of the experiment, whereas Cell A2 was dosed with 950 L (4 cm) doses every hour for 38 hours.

RESULTS

Cold Weather Viability

Winter weather conditions at the experiment site can be extremely harsh. Despite air temperatures occasionally as low as -20°C and sustained temperatures below 0°C , there was no freezing of wastewater within the wetlands or associated pipes. Data collected in prior studies at the site indicate septic tank influent wastewater temperatures remain between 0°C and 5°C throughout the winter months when the majority of wastewater must be treated (Davis, 2013). Investigation of the surface conditions of the wetlands during January of 2014 showed formation of an ice lens several centimeters above the surface, which appeared to be structurally supported by the plants (Figure 5).



Figure 5: Ice lens just above the wetland surface, January 2014.

This ice lens supported approximately 1 meter of snow at the time it was observed and shows that snow cover is providing thermal insulation. During the 2014-15 season,

air temperatures dropped below -20°C several nights (11/10-13, 11/15, 12/29-30, 2/22-23, and 3/3), including several nights before the start of ski season (11/10-13, and 11/15). At this time, a climate monitoring station near the project site recorded only 10 cm of snow cover, yet no problems with freezing were noted.

Plant Growth

The wetland cells were planted in June 2013. In each subsequent October, plant surveys show stark differences in plant density between the *Carex* and *Schoenoplectus* plants (Table 2). Survival and growth through the first two growing seasons were high for the *Carex*, but low for the *Schoenoplectus*. *Carex* plantings experienced little mortality following planting, and surviving plants recovered rapidly from any transplant shock. Stem densities only decreased from $16.5 \text{ stems}\cdot\text{m}^{-2}$ to $15 \text{ stems}\cdot\text{m}^{-2}$ during the first growing season, and were much higher than the original density by the end of the second growing season. In contrast, *Schoenoplectus* density by decreased more than 50% in the first season and was only $4.3 \text{ stems}\cdot\text{m}^{-2}$ after the second growing season. A visual representation of plant growth (Figure 6) in June 2013, soon after planting, and on the date of each plant count shows these large differences in plant density for each species at various stages of development year to year and quadrant to quadrant.

During the summer of 2015, the quadrants with *Schoenoplectus* were planted with *Carex* due the low density of *Schoenoplectus* measured at the end of the previous season. However, the surviving *Schoenoplectus* began to spread rhizomonously for the first time in the spring of 2015. By the middle of the growing season, they had achieved heights of

Table 2: Average plant densities for the entire wetlands system, counted annually at the end of each growing season

Survey Date	Density (Plant stems m ⁻²)	
	<i>Schoenoplectus</i>	<i>Carex</i>
Planting (6/19/2013)	16.5	16.5
First Growing Season (9/24/2013)	7.2	15.0
Second Growing Season (10/1/2014)	4.3	27.5
Third Growing Season (10/15/2015)*	20.0	41.1

*Fall 2015 density for *Carex* includes the addition of plants which were planted in the *Schoenoplectus* quadrants during the summer of 2015.

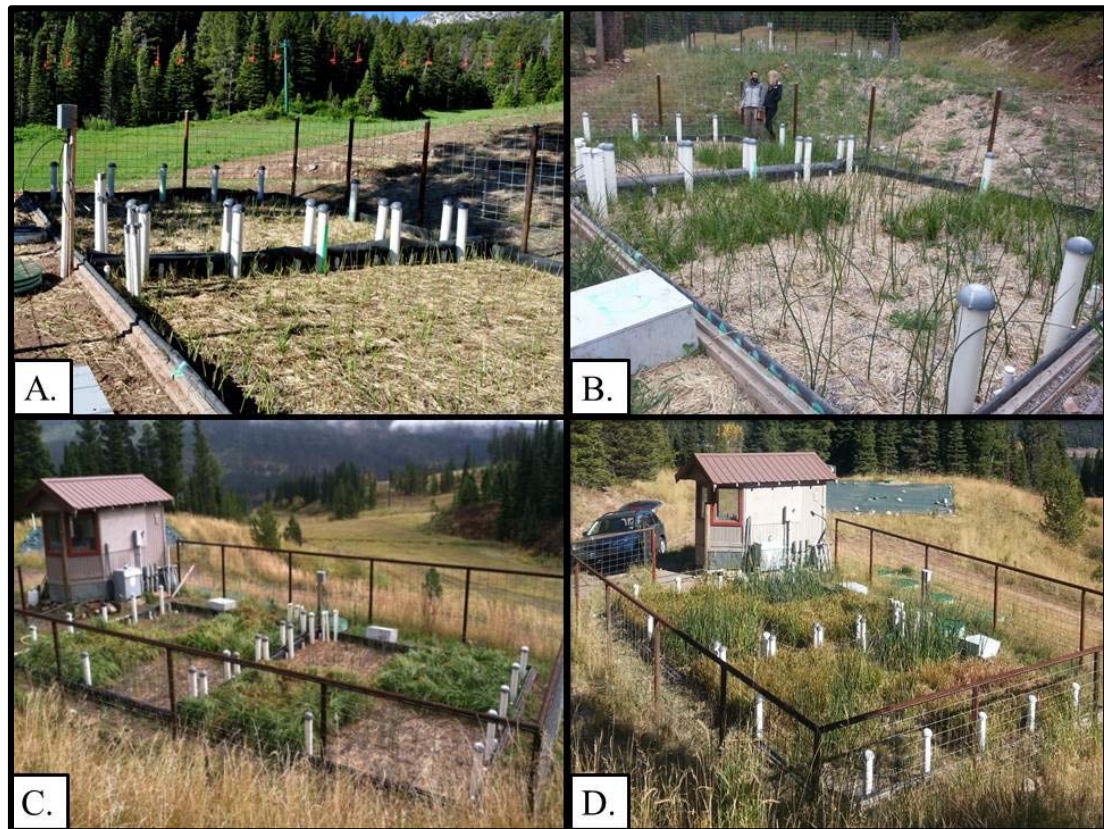


Figure 6: Plant growth in the B cells in June 2013 immediately after planting as bare-root stock (A), on September 24, 2013 after one growing season (B), on October 1, 2014, after two growing seasons (C), and on October 15, 2015 after additional *Carex* were planted and a third growing season (D).

growth greater than 1.75 m, a height typical of a mature plant. At the time of the plant survey, at the end of the growing season, the *Schoenoplectus* had rebounded to a stem density higher than that which was originally planted, although still only half of the *Carex* density at that time.

It is possible that the hydrologic start-up and operational conditions were not adequate for *Schoenoplectus* to grow well the first two years. *Schoenoplectus* require more inundated water conditions than *Carex*, so low water levels likely negatively affected *Schoenoplectus* growth more, as the roots may have been drier than ideal. This could have occurred as a result of unsaturated water conditions due to system leakage, which left all the plant's roots relatively dry immediately following planting, or it could have occurred as a result of lowering the water saturation zone too quickly over the course of the first summer. Additionally, herbivory was noted to be high during the summer of 2014, the second growing season, especially for the *Schoenoplectus*, which further challenged the plants (Figure 7). A combination of these factors likely explains the observed decrease in stem density through the first two seasons.

During the summer of 2015, water levels were maintained just below the gravel surface for the entire summer, *Carex* were planted in quadrants alongside surviving *Schoenoplectus*, and little herbivory was observed. It is unknown which, if any, of these factors explain the rapid increase in *Schoenoplectus* stem density observed in the third growing season. It is possible that the species simply takes longer to become established enough to spread and/or is more vulnerable than *Carex*. Although *Schoenoplectus* struggled to grow for the first two growing seasons, data collected up to this point in time

indicate that they can mature and spread in Bridger Bowl's climate. Continued monitoring of plant density is advisable in order to track long-term species trajectory and suitability.



Figure 7: Predation on *Schoenoplectus* during the summer of 2014. Many of the plants were eaten completely through (left) or partly damaged (right).

Water samples taken from the lowest depth in-cell sampling ports in the B cells during the 2014-15 season show a small, but statistically significant difference in ammonium reduction capacity between the quadrants where the *Carex* and *Schoenoplectus* were planted. Influent loading was identical on both cells, with $10 \text{ g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ of ammonium applied on each cell. Mean removal efficiency for the *Carex* quadrant was 97% compared to only 82% for *Schoenoplectus*. Water samples collected at the effluent of each cell, which mixes water from each plant species quadrant, did not show a significant difference in treatment efficiency, indicating the difference in treatment was

not due to a difference in cell performance. *Carex* density was more than six times higher than *Schoenoplectus* just prior to that winter season, so it is more likely that this is a result of the large differences in plant growth and associated treatment functions rather than any inherent difference between species. Treatment in the system will likely improve as plants become better established; given the large difference in performance observed, further plant establishment should result in substantial improvements in overall performance for the system in future seasons.

Tracer Study

Tracer studies were conducted on one of the A and B cells during the summer of 2014. For the A cell, dose loads of 4 cm were applied, but unfortunately approximately 90% of the conservative bromide tracer had exited the cell between the first sample, which was taken at the same time the tracer was being applied, and the second sample, which was taken at the peak of the first dose outflow hydrograph. Subsequent doses show rapid washout of the remaining 10% of the tracer (Figure 8), indicating that this dose load was significantly larger than the storage capacity of the media in the A cells. Three distinct doses can be easily identified as tight clusters of points in the figure at hours one, two, and three, which shows dilution of the remaining bromide held in the cell as each dose is applied. These points were collected over a short time interval as the water from each individual dose exited the cell to gain insight into how tracer behavior varies within a single dose, as well as over multiple doses. These clusters further show bromide concentrations increase with time within each dose. In contrast, flowrate at the outflow of

the cell follows a classical hydrograph pattern, with very little flow as the dose is applied, a rapid increase in flow as the water reaches the outflow, and a tapering tail as water traveling on slower flow paths exits. These contrasting patterns indicate that when the water is moving through the system rapidly, it does not have sufficient contact time to pick up high concentrations of bromide. In contrast, as flow decreases towards the end of the dose hydrograph, bromide concentrations rise. Thus, it seems that slower flow allows more contact between the bromide water applied in the first dose and the clean water applied subsequently. Clearly the dose volumes utilized for this tracer test were much larger than the retention volume of the cell. Future work done on the system should include a tracer test with a more appropriate dose volume so that more detailed residence volume analysis can be completed.

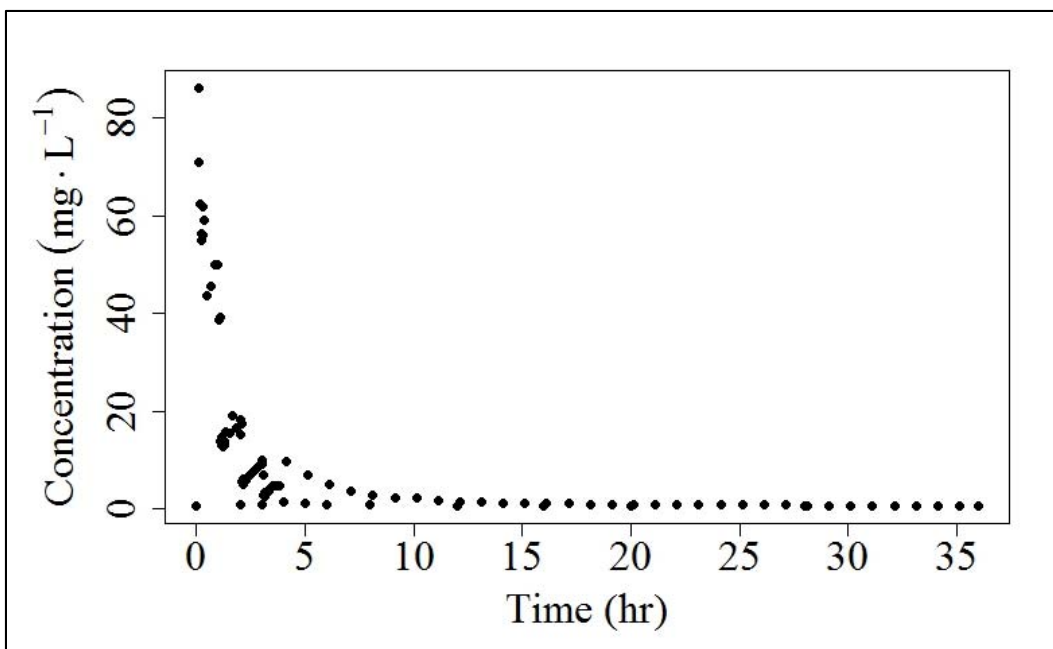


Figure 8: A cell bromide tracer samples show that no samples were collected on the rising edge or peak of the tracer curve, causing low tracer recovery and making retention analysis impossible. Three tightly grouped points are also visible on the descending slope of the curve, showing rapid washout of bromide with each dose.

Water Quality

2012-13 Season

Water quality results analyzed from grab samples taken during the first year of operation, when the system was run as a gravel filter (2012-13 season), were reported previously (Davis, 2013) but are reinterpreted here (Figure 9) for completeness of the data record, to compare between seasons, and to assess the influence of plants and start-up effects. Over the course of the season, influent COD concentration to the A cells (septic tank) averaged $832 \text{ mg}\cdot\text{L}^{-1}$ while the averaged effluent concentration from the A cells (and influent to the B cells) was $226 \text{ mg}\cdot\text{L}^{-1}$, which was further reduced to $135 \text{ mg}\cdot\text{L}^{-1}$ after passing through the B cells (Figure 9, A). No discernable time trend is evident in the data, and large variance in transfer tank concentrations is observed.

Total nitrogen ion (TNI) is defined as the sum of NH_4^+ , NO_2^- and NO_3^- ions. TNI concentration applied to the A cells from the septic tank averaged $147 \text{ mg N}\cdot\text{L}^{-1}$ and displayed little trend with time (Figure 9, B). The A cells initially removed 76% of incoming TNI, but removal decreased throughout the season and was below 5% by season's end. The B cells exhibited a similar temporal trend, with 23% TNI removal (transfer tank to recycle tank) initially, although removal quickly decreased and was below 10% within the first month of operation. Throughout the year, nearly all of the TNI in the system was ammonium; nitrate values never rose above $1 \text{ mg N}\cdot\text{L}^{-1}$ (Figure 9, C and D). While it is possible that simultaneous nitrification and denitrification was occurring, the temporal removal pattern strongly suggests that the observed reduction in

ammonium is due to sorption to the media which declined during this initial start-up phase as sorption sites were gradually saturated.

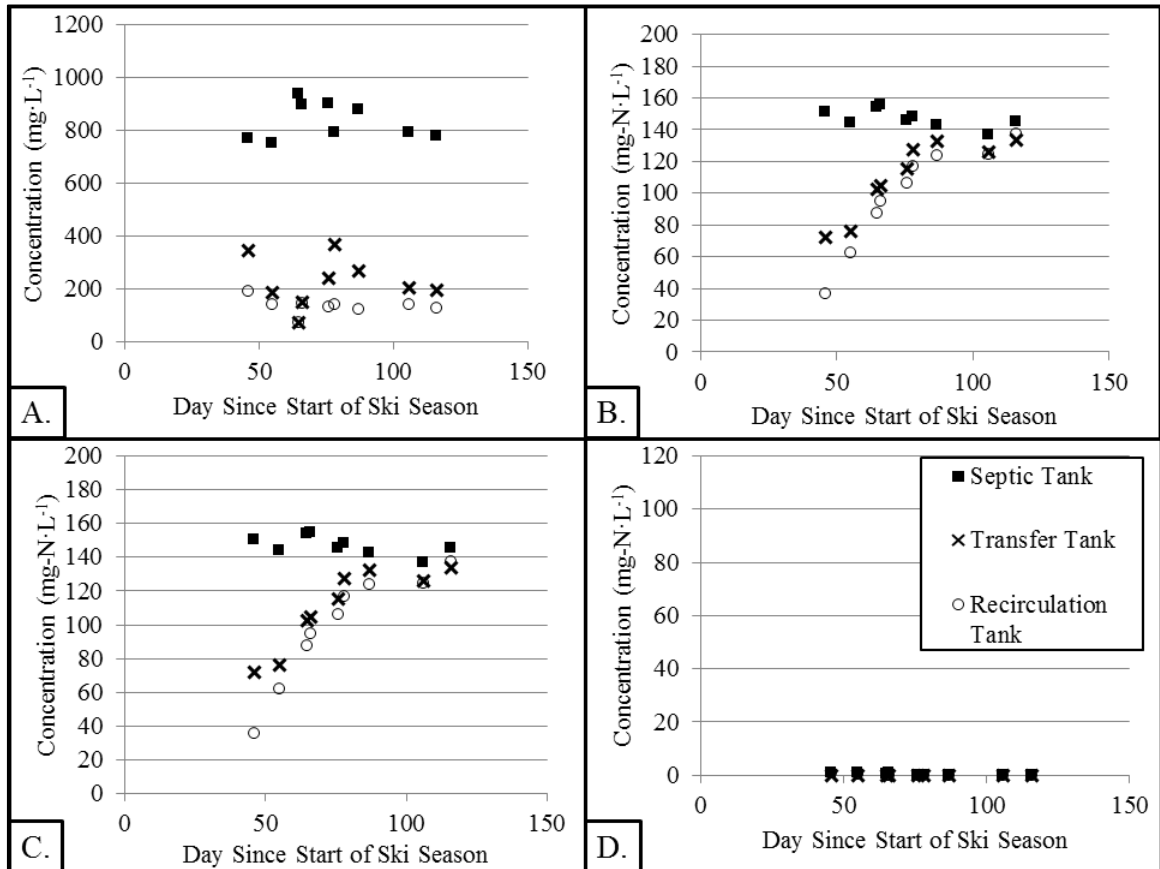


Figure 9: Measured COD (A), TNI (B), ammonium (C), and nitrate (D) concentrations in the tanks during the 2012-13 season, when the wetland system was operated without plants (Davis, 2013). Effluent from the A cells is mixed in the transfer tank and enters the B cells. The recirculation tank mixes effluent from the B cells. Operational parameters of individual cells varied with time (See Table 1).

Since 2001, Bridger Bowl has used a recirculating sand filter (RSF) for their primary wastewater treatment system, which provides a useful reference to compare the performance of the unplanted wetland. In the five seasons of data collected by prior students (Allen & Davis, 2013), mean total nitrogen removal was only 39% with large sample to sample variability and no discernable trends (Figure 10). This consistently poor

performance is further evidence that sorption and not microbial activity was responsible for the initially high nitrogen removal that was observed in the wetland during its first season of operation. Additionally, the RSF data record suggests that biofilm establishment was not a factor limiting TNI removal in the wetland when run as an unplanted filter during the first season of operation.

2013-14 Season

The limited 2013-14 season water quality data, taken as grab samples at two periods of time after one season of plant establishment and one previous year of exposure to wastewater, show improved performance relative to the prior season without plants. (Table 3). Concentration removal of COD was similar, but unlike the previous season, nitrate was generated within the system in relatively high concentrations; 74% and 50% of total nitrogen in the recirculation tank was as nitrate in January and April, respectively.

Additionally, there was removal of total nitrogen, measured as either TNI or TKN. At the end of the season, with no recycle, TKN concentration removal in the A cells was 52%. The B cells further reduced TKN by another 41% at that time, for a total removal of 71% across the whole system (Table 3). These results indicate that nitrogen removal mechanisms other than sorption, likely nitrification and denitrification, were active in the wetland system during the second year of operation and after one year of plant establishment.

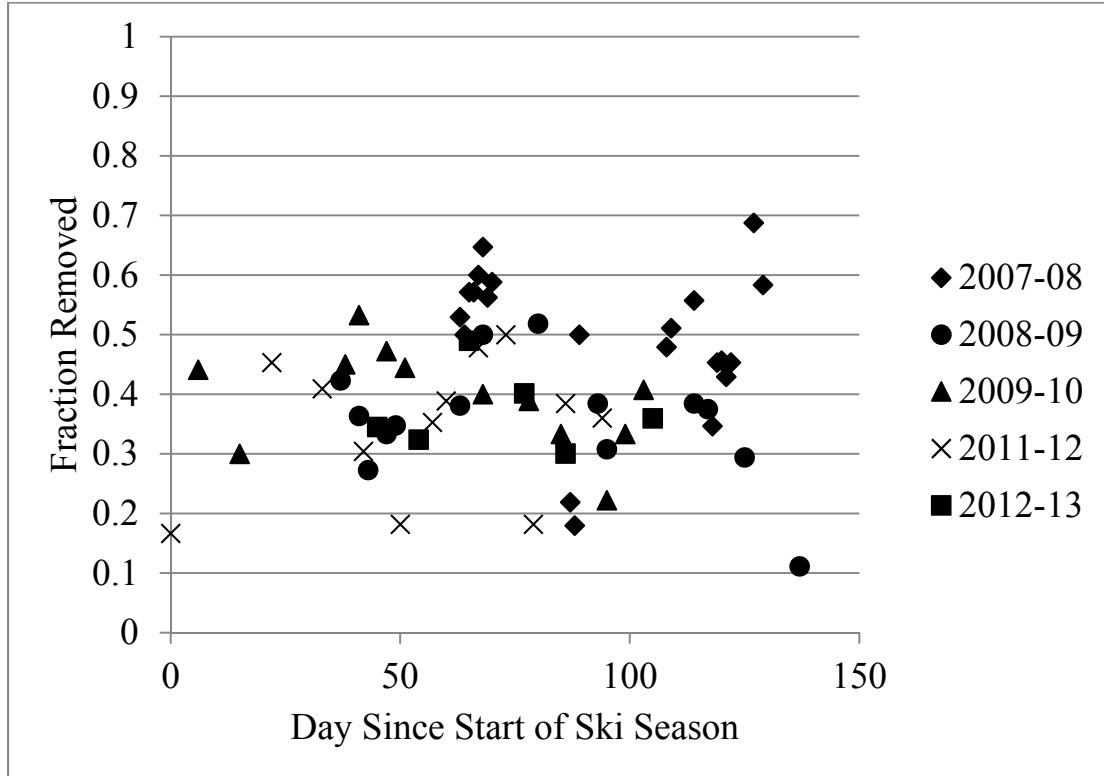


Figure 10: Total nitrogen removal by the recirculating sand filter (2007-08 to 2011-12 seasons) and by the newly constructed wetland and the sand filter combined (2012-13 season), calculated using base area septic tank samples for influent values and drainfield dosing tank samples as effluent values.

Table 3: Measured COD, TNI and TKN concentrations in the wetland tanks during the 2013-2014 ski season

	COD (mg L ⁻¹)		TNI (mg-N L ⁻¹)	TKN (mg-N L ⁻¹)	Nitrate (mg-N L ⁻¹)	
Sample Date	1/17/14	4/7/14	1/17/14	4/7/14	1/17/14	4/7/14
Septic Tank	487	691	136	135	0.5	5.5
Transfer Tank	210	161	116.7	65.4	11.4	5.4
Recirculation Tank	52	120	53.5	38.5	39.4	19.0

2014-2015 Season

During the 2014-15 season, water quality samples were collected using the autosampler on several days within each scheme from individual cells and each of the three pump tanks. Average values at each loading scheme for calculated hydraulic load,

dose volume, as well as mass loading and mass removal for COD, TN, NH_4^+ and NO_3^- are shown in Table 4. The hydraulic load is calculated by dividing the volume of water applied to each cell from the respective source(s) by the cell surface area, while dose volume is the daily load divided by the number of daily doses. The calculated mass loadings represent the total load, or the sum of the daily mass supplied from the septic and recycle tanks ($Q_s \cdot C_s + Q_r \cdot C_r$) divided by the individual cell surface area. Both the target septic tank hydraulic load and target recycle ratio varied throughout the season and often varied between parallel cells at any given time (Table 1). The actual volume applied from the recycle and septic tank to each cell deviated somewhat from the target value because of operational limitations. The reported hydraulic and mass loads are calculated from calibration curves relating the flowrate to pump timing; thus the hydraulic load and the mass loading of any given parameter is typically unique for each scheme. The mass removal is calculated as the difference between the daily mass in and the daily mass out, expressed on a per unit area basis.

Hydraulic loads to the A cells varied between $5.6 \text{ cm} \cdot \text{d}^{-1}$ and $68.5 \text{ cm} \cdot \text{d}^{-1}$, and between 9.5 and $29.8 \text{ cm} \cdot \text{d}^{-1}$ on the B cells. This hydraulic load was applied in dose sizes that were as small as 0.4 cm and as large as 15.8 cm . Mean COD loads to the A cells varied depending on loading scheme between 61 and $368 \text{ g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$, but were only 13 to $51 \text{ g} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$ in the B cells. Mean ammonium loading on the A cells was as low as $8 \text{ g} \cdot \text{N} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$ and as high as $52 \text{ g} \cdot \text{N} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$. The maximum daily load on the A cells was nearly ten times higher than published guidelines. For the B cells, ammonium loading was much less variable, ranging from 9 to $20 \text{ g} \cdot \text{N} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$. The A cells were loaded

Table 4: Hydraulic loading, mass loading, and mass removal conditions during the 2014-15 season, organized by loading scheme. All nitrogen parameters reported as mass N. Values shown are a mean \pm one standard deviation.

Scheme	Dose Load (cm)	Actual Septic HLR (cm·d ⁻¹)	Actual Recirc HLR (cm·d ⁻¹)	Total HLR (cm·d ⁻¹)	Total COD Load (g·m ⁻² ·d ⁻¹)	Total NH ₄ ⁺ Load (g·m ⁻² ·d ⁻¹)	Total NO ₃ ⁻ Load (g·m ⁻² ·d ⁻¹)	Total TNI Load (g·m ⁻² ·d ⁻¹)	COD Removal (g·m ⁻² ·d ⁻¹)	NH ₄ ⁺ Removal (g·m ⁻² ·d ⁻¹)	NO ₃ ⁻ Removal (g·m ⁻² ·d ⁻¹)	TNI Removal (g·m ⁻² ·d ⁻¹)
14-A-1	1.3	5.4	0.2	5.6	- -	8 \pm 2	0 \pm 0	9 \pm 2	- -	1 \pm 1	0 \pm 0	1 \pm 1
14-A-2	0.6	8.3	0.2	8.5	- -	13 \pm 3	1 \pm 0	13 \pm 4	- -	2 \pm 1	0 \pm 0	3 \pm 1
14-A-3	0.6	19.2	2.0	21.2	176 \pm 50	31 \pm 10	3 \pm 1	33 \pm 11	152 \pm 36	6 \pm 5	0 \pm 1	7 \pm 5
14-A-4	0.6	7.3	8.9	16.2	78 \pm 18	13 \pm 3	5 \pm 1	18 \pm 3	61 \pm 19	-1 \pm 3	3 \pm 1	3 \pm 3
14-A-5	0.6	17.4	17.1	34.5	168 \pm 42	31 \pm 7	13 \pm 2	44 \pm 9	116 \pm 28	2 \pm 4	8 \pm 1	10 \pm 5
14-A-6	0.6	16.8	14.7	31.4	184 \pm 28	30 \pm 5	11 \pm 3	41 \pm 7	114 \pm 17	2 \pm 3	7 \pm 1	9 \pm 3
14-A-7	0.4	12.0	22.5	34.5	91 \pm 24	16 \pm 4	12 \pm 1	29 \pm 3	81 \pm 26	-6 \pm 5	7 \pm 0	1 \pm 4
14-A-8	0.4	23.6	11.5	35.0	249 \pm 22	30 \pm 9	7 \pm 0	37 \pm 9	185 \pm 25	8 \pm 6	3 \pm 0	11 \pm 5
14-A-9	0	0	0	0	0	0	0	0	-	-	-	-
14-A-10	0.4	35.4	33.1	68.5	368 \pm 44	52 \pm 7	17 \pm 3	71 \pm 6	265 \pm 72	11 \pm 9	10 \pm 2	20 \pm 8
14-A-11	0.6	7.5	21.2	28.7	61 \pm 9	10 \pm 2	12 \pm 1	23 \pm 1	40 \pm 15	0 \pm 2	7 \pm 1	8 \pm 2
14-A-12	0.6	7.5	21.1	28.5	84 \pm 9	10 \pm 2	12 \pm 1	23 \pm 1	42 \pm 15	0 \pm 2	6 \pm 2	6 \pm 2
14-B-1	1.3	-	-	9.5	- -	12 \pm 1	1 \pm 0	12 \pm 1	- -	7 \pm 1	-5 \pm 0	2 \pm 1
14-B-2	5.4	-	-	9.9	- -	12 \pm 1	1 \pm 0	13 \pm 1	- -	9 \pm 1	-6 \pm 0	2 \pm 1
14-B-3	5.4	-	-	10.8	13 \pm 6	9 \pm 2	1 \pm 0	10 \pm 2	12 \pm 2	7 \pm 2	-6 \pm 0	1 \pm 2
14-B-4	3.4	-	-	16.1	51 -	20 \pm 5	2 \pm 1	22 \pm 6	34 -	16 \pm 4	-11 \pm 2	5 \pm 3
14-B-5	6.8	-	-	20.5	39 \pm 0	17 \pm 2	3 \pm 1	20 \pm 2	12 \pm 4	14 \pm 1	-8 \pm 1	5 \pm 1
14-B-6	5.4	-	-	17.1	33 \pm 5	15 \pm 1	3 \pm 1	17 \pm 1	13 \pm 1	13 \pm 1	-10 \pm 0	3 \pm 1
14-B-7	2.4	-	-	27.8	40 \pm 6	11 \pm 2	5 \pm 2	16 \pm 1	12 \pm 4	10 \pm 2	-9 \pm 1	1 \pm 3
14-B-8	5.8	-	-	21.5	40 -	14 \pm 1	3 \pm 0	17 \pm 1	16 -	14 \pm 1	-9 \pm 0	5 \pm 1
14-B-10	6.2	-	-	23.0	49 -	14 \pm 3	2 \pm 1	17 \pm 2	22 -	14 \pm 3	-10 \pm 0	4 \pm 2
14-B-12	2.4	-	-	29.8	37 \pm 5	10 \pm 2	6 \pm 2	16 \pm 1	12 \pm 2	10 \pm 2	-11 \pm 1	-1 \pm 2

with between 0.4 and 13 $\text{g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ of nitrate and the B cells received 0.7 to 5 $\text{g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ of nitrate, depending on the loading scheme.

Mean removal of COD was high in the A cells, with up to 265 $\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ removed when the system was receiving the highest septic loads tested. In the B cells, COD removal was lower by nearly an order of magnitude. Mean ammonium removal was generally low in the A cells, and sometimes even negative, but operational schemes with no recycle did remove ammonium and in all cases standard deviations were relatively large. In contrast, ammonium removal in the B cells proceeded more rapidly and with considerably more consistency; removal was as high as 15 $\text{g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ and consistently above 10 $\text{g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ in the latter part of the season. Net nitrate removal was moderate to high in the A cells under loading schemes with recycle, but was low without recycle because very little nitrate was applied. The highest mean nitrate removal was 10.3 $\text{g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ when the system was receiving the highest septic tank hydraulic load, and hence the greatest amount of influent COD. The B cells always produced more nitrate than they removed, so net nitrate removal is negative for all schemes tested. TNI was removed at rates of between 1.2 and 19.7 $\text{g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ in the A cells, and between -0.8 and 4.9 $\text{g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ in the B cells.

Figures 11 through 21 compare mass loading, mass removal, and measured concentrations at various points with the wetland system for COD, TNI, NH_4^+ , and NO_3^- , respectively. For all parameters, the septic tank concentration represents the influent to the system, but the concentration applied to the A cells is shown as a flow-weighted average of the septic and recycle tank water, which varies depending on the recycle ratio.

The recycle ratio was varied throughout the season and often differed between the A cells at a given time (Table 1). Water from both A cells was mixed in the transfer tank, so both B cells received the same influent concentration at any given time, but were often operated under different loading schemes. Measured concentration results are shown for each individual sample, but mass loading and removal figures show the average of samples collected within each loading scheme (Table 4), with error bars to show the standard deviation. Furthermore, mass loading data has been fitted with a linear regression line, which shows average performance for all data collected during the season. These regression lines were forced through (0,0) and R^2 values are shown. Data has been grouped by chemical compound, with organics first, followed by ammonium, then nitrate, and finally total nitrogen ions.

Organics Removal. Averaged over the entire season and all operational schemes, the first stage (A cells) of the wetland system reduced COD concentrations from 1003 $\text{mg}\cdot\text{L}^{-1}$ in the septic tank to 165 $\text{mg}\cdot\text{L}^{-1}$ in the transfer tank for an 84% reduction (Figure 11). Taking into account the dilution of COD due to mixing of recycled water, the first stage removed only 70% as the flow-weighted influent concentration was only 543 $\text{mg}\cdot\text{L}^{-1}$. The second stage (B cells) further reduced COD concentration to 91 $\text{mg}\cdot\text{L}^{-1}$ in the recycle tank, thus the wetland system as a whole reduced COD concentration by 91%.

Septic tank COD concentration was fairly constant with time, but influent to the A cells was higher without recycle and lower at high recycle ratios. Effluent concentrations reflect this influent variation but also demonstrate a performance influence due to recycle ratio; greater differences in COD concentrations across the cell are apparent when recycle

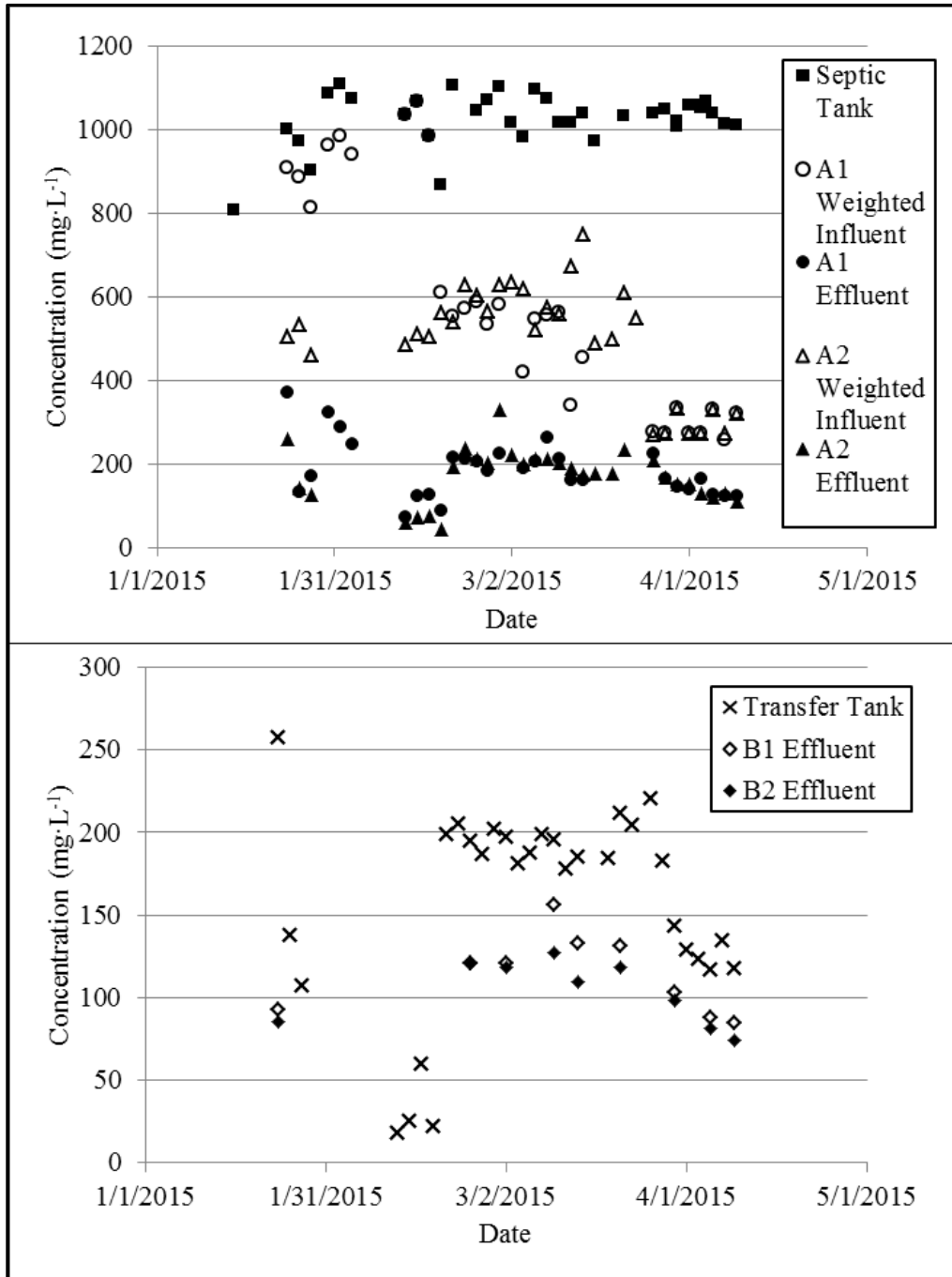


Figure 11: Measured COD concentrations during the 2014-15 season for the A cells (top) and the B cells (bottom). Influent to the A cells is a flow-weighted average of the septic tank and recirculation tank which depends on the recycle ratio. Effluent from the A cells is mixed in the transfer tank and enters the B cells. The recirculation tank mixes effluent from the B cells, but data are not shown. Operational parameters of individual cells varied with time (See Table 1).

ratios are low, likely due to the recycle water containing COD that is relatively recalcitrant.

This is evident in the COD and BOD₅ concentrations measured in the three tanks during the 14-A-11 and 14-A-12 schemes (Table 5). The ratio of BOD₅ to COD is 68% in the septic tank which reduced to just 19% in the transfer tank and 11% in the recycle tank. Thus, recycling water returns COD to the head of the system which is not easily biodegraded, albeit at a relatively low concentration.

Table 5: Mean BOD and COD concentrations \pm one standard deviation in the tanks near the end of the 2014-2015 season

	COD (mg·L ⁻¹)	BOD ₅ (mg·L ⁻¹)	BOD ₅ /COD Ratio
Septic Tank	1062 \pm 5	724 \pm 137	0.68
Transfer Tank	128 \pm 12	23 \pm 4	0.19
Recirculation Tank	93 \pm 8	10 \pm 1	0.11

During the 2014-2015 season, A cell dose volumes ranged from 0.4 to 1.3 cm. Dose volumes smaller than 1 cm generally reduced A cell effluent COD concentrations to near or below 250 mg·L⁻¹ regardless of the septic water HLR, recycle ratio or any other parameter (Figure 11). The COD effluent from the A cells exceeded 300 mg·L⁻¹ in only three of 132 samples (all at an HLR of 1.3 cm) and the transfer tank COD did not exceed 225 mg·L⁻¹ in any samples taken after the dose volume was reduced to below 1 cm.

The mass removal of COD in the A cells significantly increased with an increase in mass loading. Removal for all schemes tested was very consistent and averaged 72% ($R^2=0.96$). This trend held at even at the highest loading (Figure 12), indicating that the A cells were not overloaded with organic carbon during the entire test period. There is no

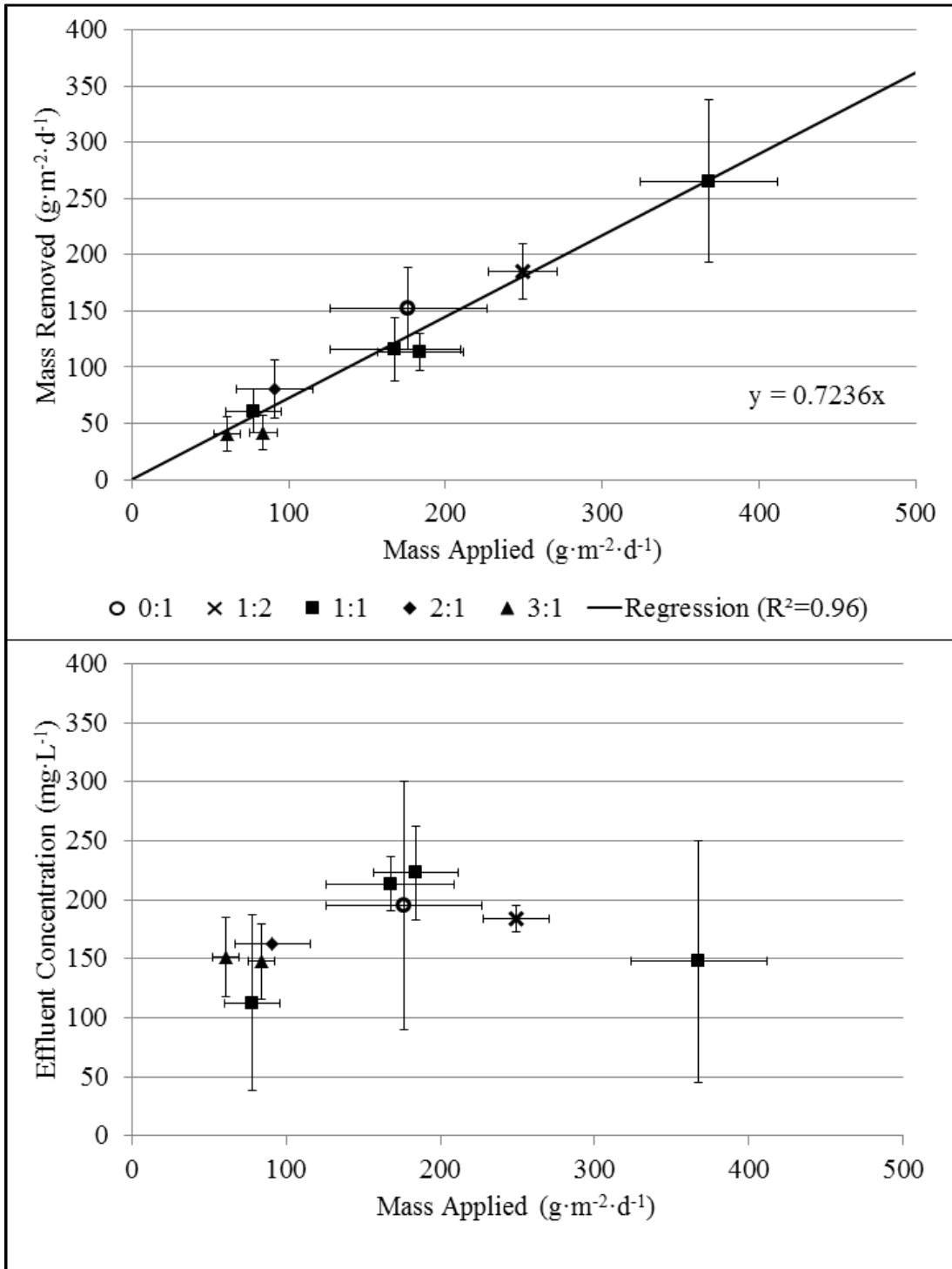


Figure 12: COD mass removal (top) and COD effluent concentration (bottom) in the A cells as a function of COD total mass load for each loading scheme tested during the 2014-15 season. Total mass load includes the mass from septic and recycle water. Symbols reflect mean values at different recycle ratios, and error bars show one standard deviation.

correlation between A cell COD effluent concentration and mass loading, again suggesting the system was not overloaded with respect to COD. Removal efficiency of COD was primarily a factor of the recycle ratio. The highest mass removal fraction of 86% was observed with no recycle, due to the highly biodegradable COD component in the septic water (Tables 1 and 5). In contrast, higher recycle ratios return less biodegradable COD to the A cells hence mass removal decreases for the same total mass loading.

Mass removal of COD in the B cells is comparatively low and more variable; on average, only 43% ($R^2=0.41$) of mass influent COD was removed because very little biodegradable COD remained after passing through the A cells (Figure 13). COD removal increases slightly as loading increases, especially at the higher loads, but effluent concentration does not follow any apparent trends with respect to loading, removal, or number of doses per day. This probably indicates that the B cells were removing all available biodegradable carbon regardless of operational conditions tested.

Ammonium Removal. Optimization of nitrogen removal processes was a major focus of the 2014-15 season. Septic tank ammonium concentrations were just over 180 mg-N·L⁻¹ at the beginning of the season, but decreased slightly over the course of the season, and averaged 150 mg-N·L⁻¹ (Figure 14, top). There was an average 39 mg-N·L⁻¹ or 25% reduction in the ammonium concentration without recycle (schemes 14-A-1, 14-A-2, and 14-A-3), but very little ammonium was removed in the A cells in schemes tested thereafter, and effluent concentrations are close to flow-weighted influent concentrations. As expected, the B cells were very effective at reducing ammonium

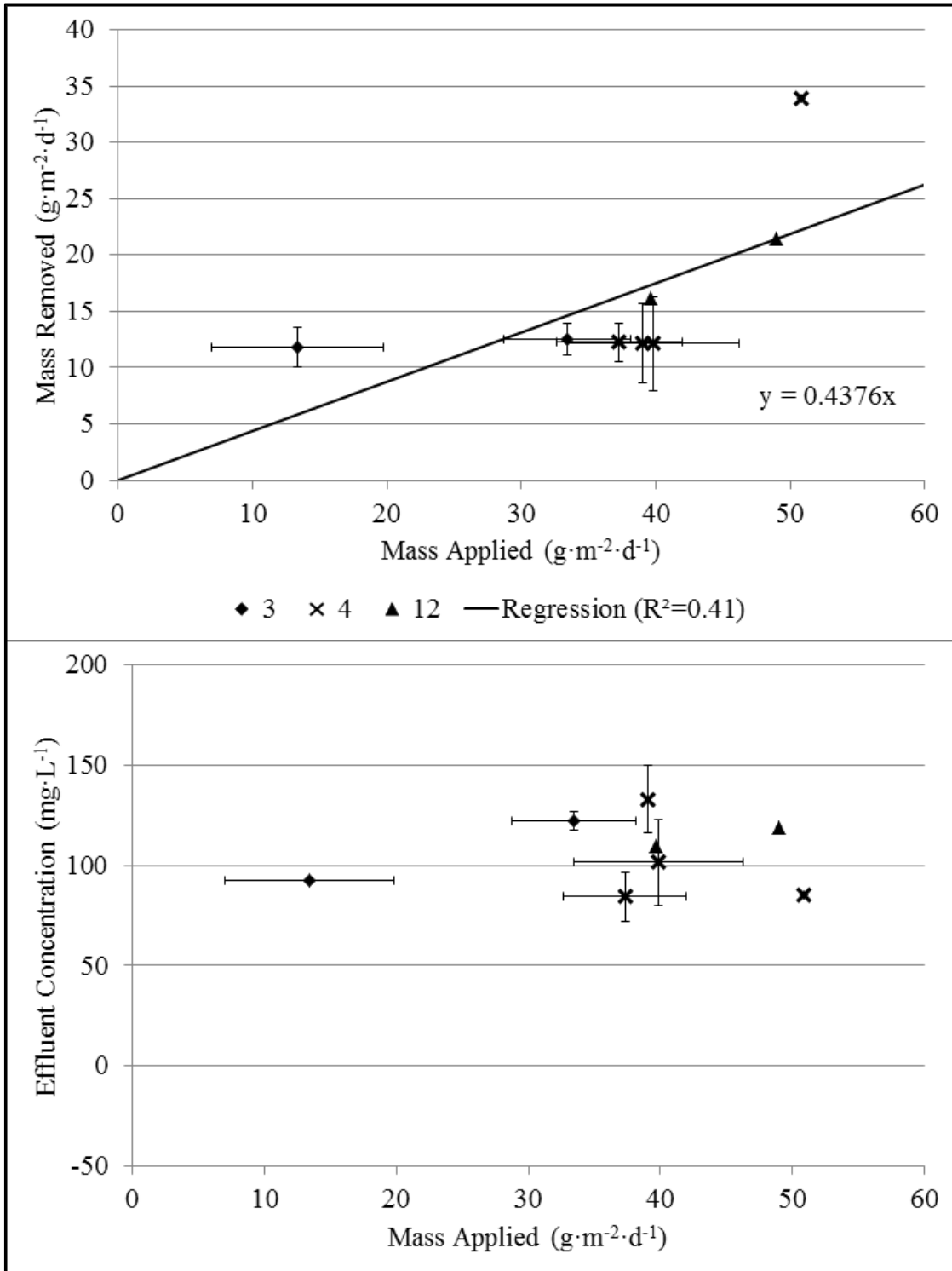


Figure 13: COD mass removal (top) and COD effluent concentration (bottom) in the B cells as a function of COD total mass load for loading schemes tested during the 2014-15 season. Symbols reflect mean values at different dose frequencies (doses per day) and error bars show one standard deviation, where applicable.

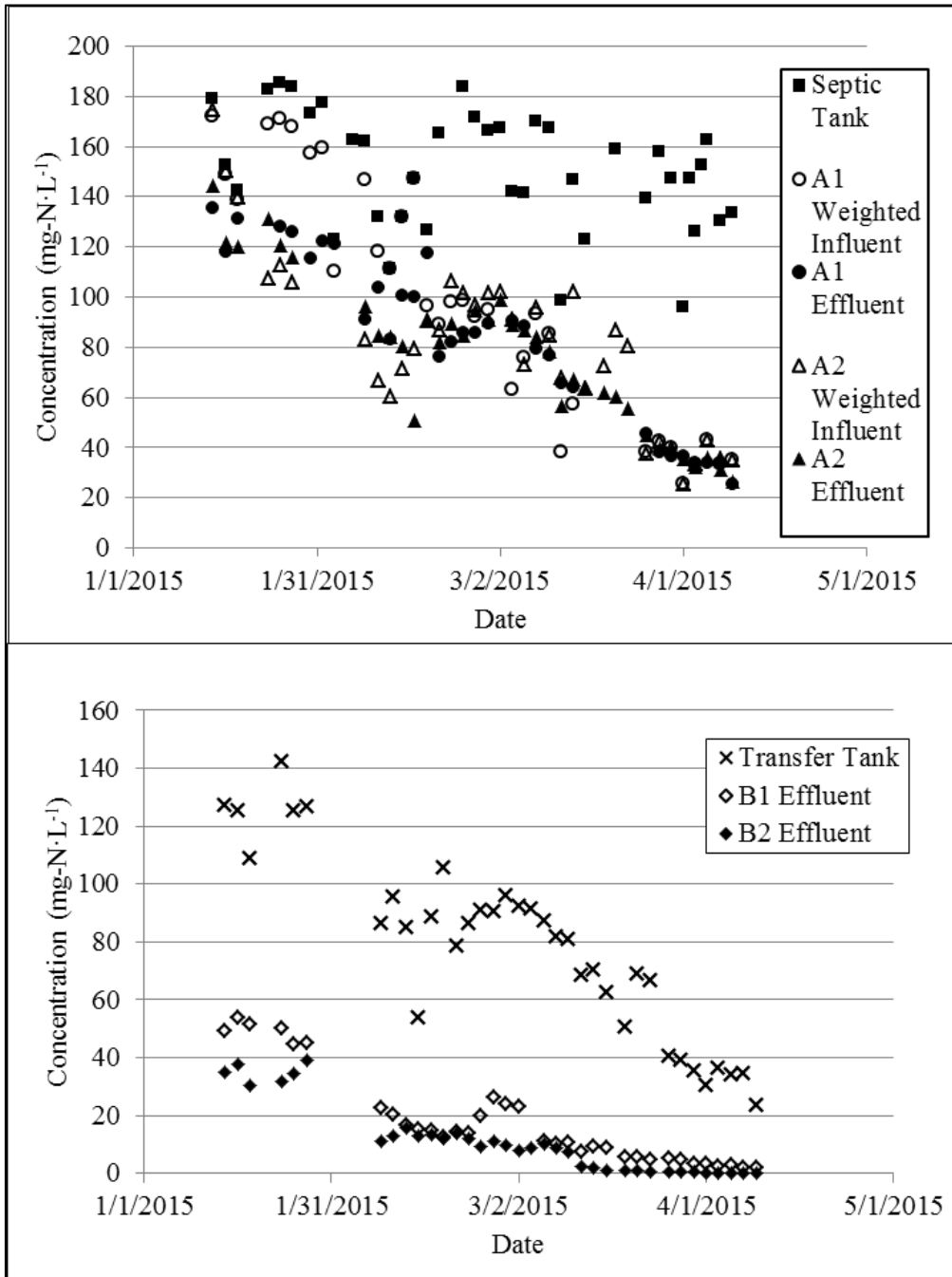


Figure 14: Measured NH_4^+ concentrations during the 2014-15 season for the A cells (top) and the B cells (bottom). Influent to the A cells is a flow-weighted average of the septic tank and recycle tank which depends on the recycle ratio. Effluent from the A cells is mixed in the transfer tank and enters the B cells. The recirculation tank mixes effluent relative to load was low and high effluent ammonium concentrations were generally observed without recycle. from the B cells. Operational parameters of individual cells varied with time (See Table 1).

concentrations (Figure 14, bottom). Average influent ammonium concentration in the transfer tank was $76 \text{ mg-N}\cdot\text{L}^{-1}$ and average removal efficiency in the B cells was 82% for the season, or $62 \text{ mg-N}\cdot\text{L}^{-1}$. Cell B2 consistently had effluent ammonium concentrations that were lower than cell B1 when compared on the same sample date, regardless of hydraulic scheme. Treatment also improved for both cells through the course of the season, and several samples had ammonium concentrations below detection limits of $0.25 \text{ mg-N}\cdot\text{L}^{-1}$ at the end of the season.

Ammonium loading in the A cells varied widely, between 8 and $55 \text{ g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$, as did mass removal -6 to $+8 \text{ g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ (Figure 15). Overall, little ammonium was removed in the A cells, with an average mass removal of only 11% of the mass applied but the correlation is poor ($R^2= 0.56$). Oxygen available for nitrification may be limited under schemes without recycle as it was consumed in the removal of the high COD loads, but even in schemes without recycle a small mass removal of ammonium was observed (Table 4). This may reflect the system's large capacity to remove COD, which would have left some oxygen available for nitrification. Nevertheless, ammonium mass removal

In schemes with 3:1 recycle, septic tank water was diluted with recycle water which was low in ammonium, thus ammonium concentrations in the effluent were the lowest observed all season, despite low mass removal. Under intermediate recycle ratios, ammonium removal is more variable, and even negative (i.e. net increase in ammonium concentration) when applied at lower loads. This may be due to large measurement variance, or may indicate that hydrolysis of organic nitrogen or dissimilatory nitrate reduction was occurring. Effluent ammonium concentrations for these intermediate

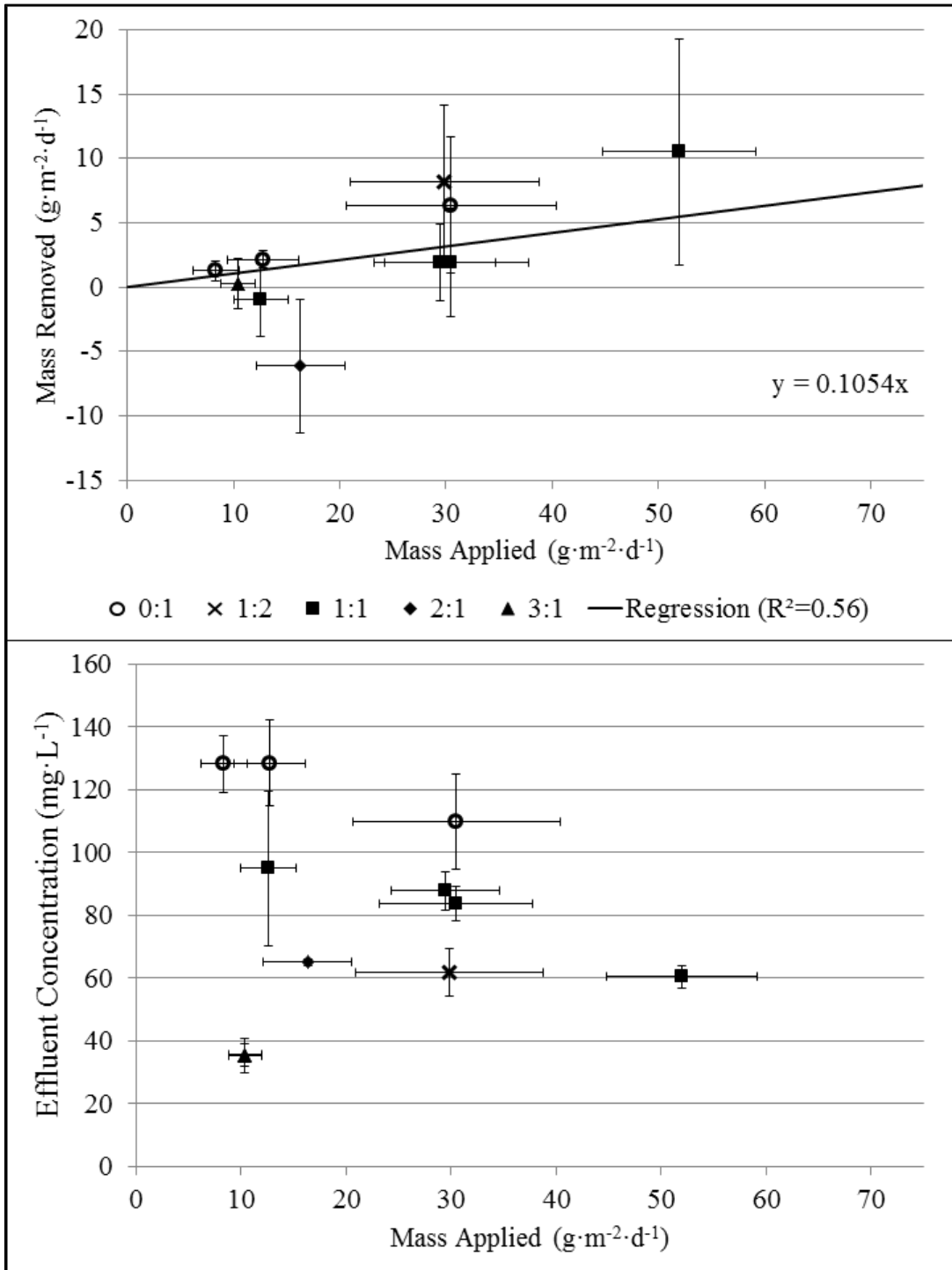


Figure 15: Ammonium mass removal (top) and ammonium effluent concentration (bottom) in the A cells as a function of ammonium total mass load for each of the loading scheme tested during the 2014-15 season. Total mass load includes the mass from septic and recycle water. Symbols reflect mean values at different recycle ratios and error bars show one standard deviation.

recycle schemes are generally in the 60-100 mg-N·L⁻¹ range and are fairly well bounded between the high concentrations with no recycle and the low concentrations with 3:1 recycle.

The B cells exhibited the ability to nitrify well during the 2014-15 season, with a mean removal of 84% ($R^2=0.74$) and effluent ammonium concentrations below detection limits of 0.25 mg-N·L⁻¹ for several samples at the end of the season (Figure 16). As previously shown, the effluent concentration from the A cells and hence influent to the B cells decreased throughout the season, but the daily hydraulic load was gradually increased from 9.5 cm to over 29.8 cm, so ammonium mass loading was relatively constant and much lower than the loads applied to the A cells, ranging between 9 and 20 g-N·m⁻²·d⁻¹. Mean ammonium removal showed a strong correlation with load and varied between 6 and 16 g-N·m⁻²·d⁻¹.

Nitrate Removal. Nitrate concentrations were typically low in the septic tank, averaging only 5 mg-N·L⁻¹, but recycling of nitrate rich water to the A cells created an average flow-weighted influent nitrate concentration of 30 mg-N·L⁻¹ for the season, with one value nearly 50 mg-N·L⁻¹ (Figure 17). An average nitrate concentration reduction of 53% occurred in the A cells, and effluent nitrate concentrations were typically 5-15 mg-N·L⁻¹ prior to March 22 when recycle ratios were 2:1 or less, but increased to 15-30 mg-N·L⁻¹ when the recycle ratio was increased to 3:1. The decrease in ammonium observed in the B cells (Figure 14) was accompanied by a similar increase in nitrate (Figure 17). Average effluent nitrate concentration from the B cells was 61 mg-N·L⁻¹, and varied between 40-80 mg-N·L⁻¹. The influence of different hydraulic schemes is apparent, and

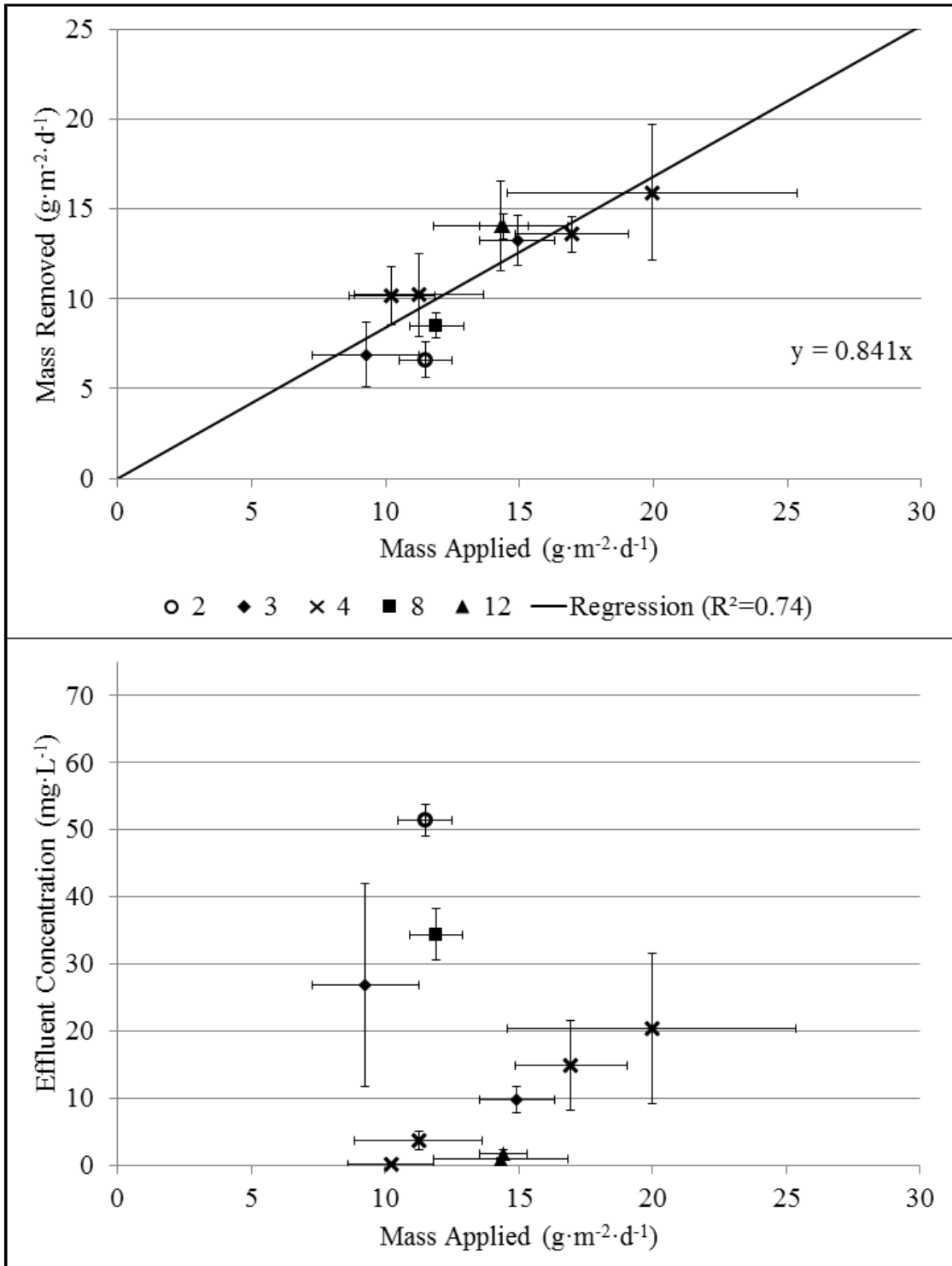


Figure 16: Ammonium mass removal (top) and ammonium effluent concentration (bottom) in the B cells as a function of ammonium total mass load for each loading scheme tested during the 2014-15 season. Symbols reflect mean values at different dose frequencies (doses per day) and error bars show one standard deviation.

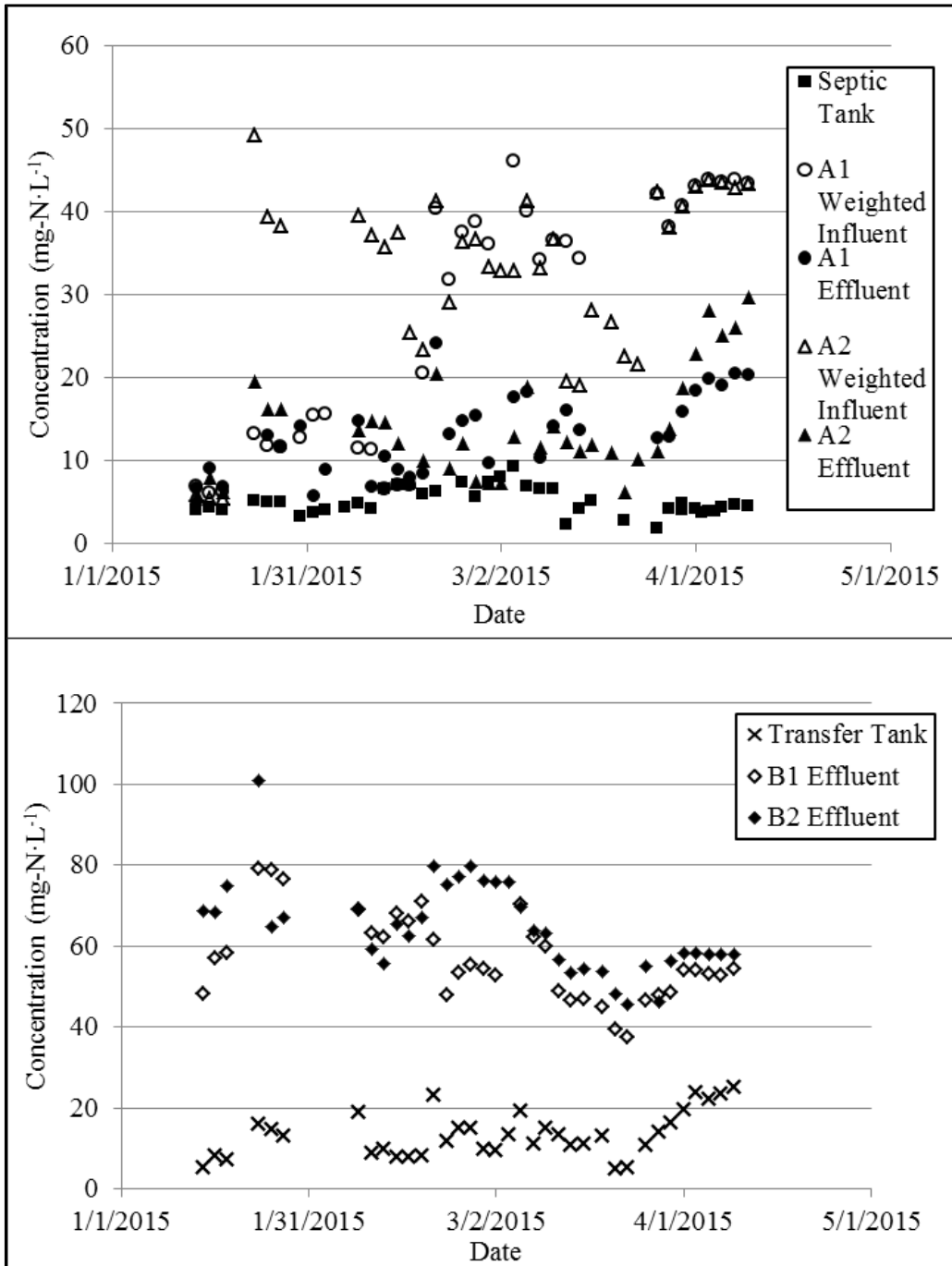


Figure 17: Measured NO_3^- concentrations during the 2014-15 season for the A cells (top) and the B cells (bottom). Influent to the A cells is a flow-weighted average of the septic tank and recycle tank which depends on the recycle ratio. Effluent from the A cells is mixed in the transfer tank and enters the B cells. The recirculation tank mixes effluent from the B cells. Operational parameters of individual cells varied with time (See Table 1).

the B2 cell consistently produced more nitrate than B1, even when dose volumes and hydraulic and TNI loads between B cells were similar. More importantly, the pattern of ammonium decrease relative to nitrate production due to these factors is remarkably similar, with a decrease in ammonium removal correlating to an increase in nitrate production.

Net nitrate mass removal in the A cells shows a very strong correlation with nitrate mass loading; an average of 55% ($R^2=0.97$) of applied nitrate was reduced (Figure 18). Schemes with no recycle showed little net change in nitrate mass, but were also lightly loaded. In contrast, schemes with 1:1 and 2:1 recycle ratio (R:S) consistently perform better than the season average. Although variance increases at high loading, a linear relationship between loading and removal continues up to the highest loading ($17 \text{ g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$) suggesting that nitrate removal was rate limited rather than capacity limited.

The B cells were designed and optimized primarily for nitrification and, as expected, they were net producers of nitrate throughout the 2014-15 season. Therefore a correlation between nitrate mass loading and either nitrate mass production or effluent nitrate concentration was not expected. Rather, nitrate production correlates with ammonium mass loading (63%, $R^2=0.27$) and ammonium mass removal (75%, $R^2=0.65$). Net nitrate mass production is close to the ammonium mass loading up to about $10 \text{ g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$, strongly suggesting that nitrification is the main process removing ammonium in the B cells at lower loading (Figure 19). Nitrate production peaks at about $12 \text{ g-N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ suggesting that this may be near the maximum possible nitrification in the B cells. Effluent nitrate concentrations were consistently between 60 and $80 \text{ mg-N}\cdot\text{L}^{-1}$.

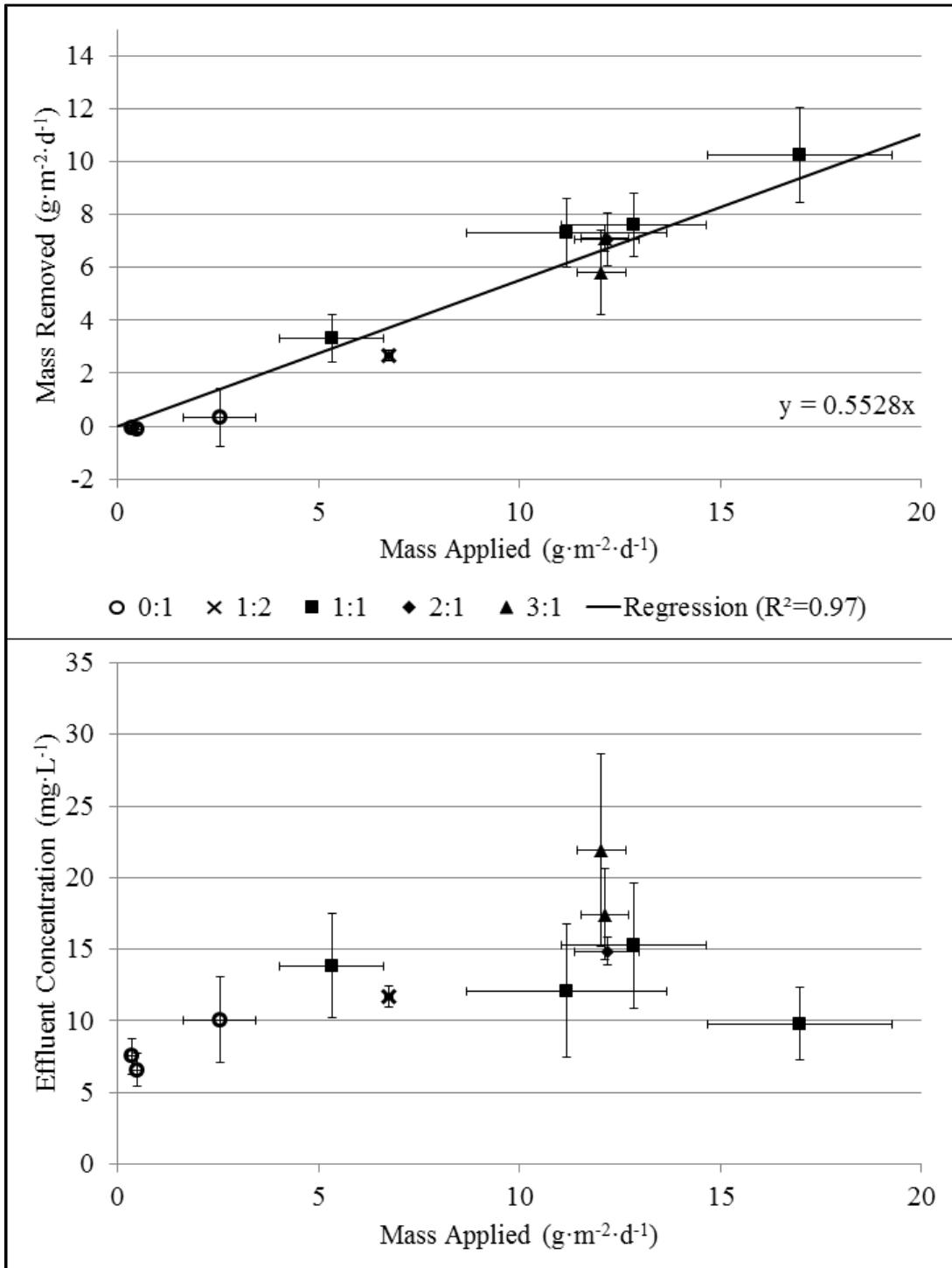


Figure 18: Nitrate mass removal (top) and nitrate effluent concentration (bottom) in the A cells as a function of nitrate total mass load for each loading scheme tested during the 2014-15 season. Total mass load includes the mass from septic and recycle water. Symbols are mean values for different recycle ratios, and error bars show one standard deviation.

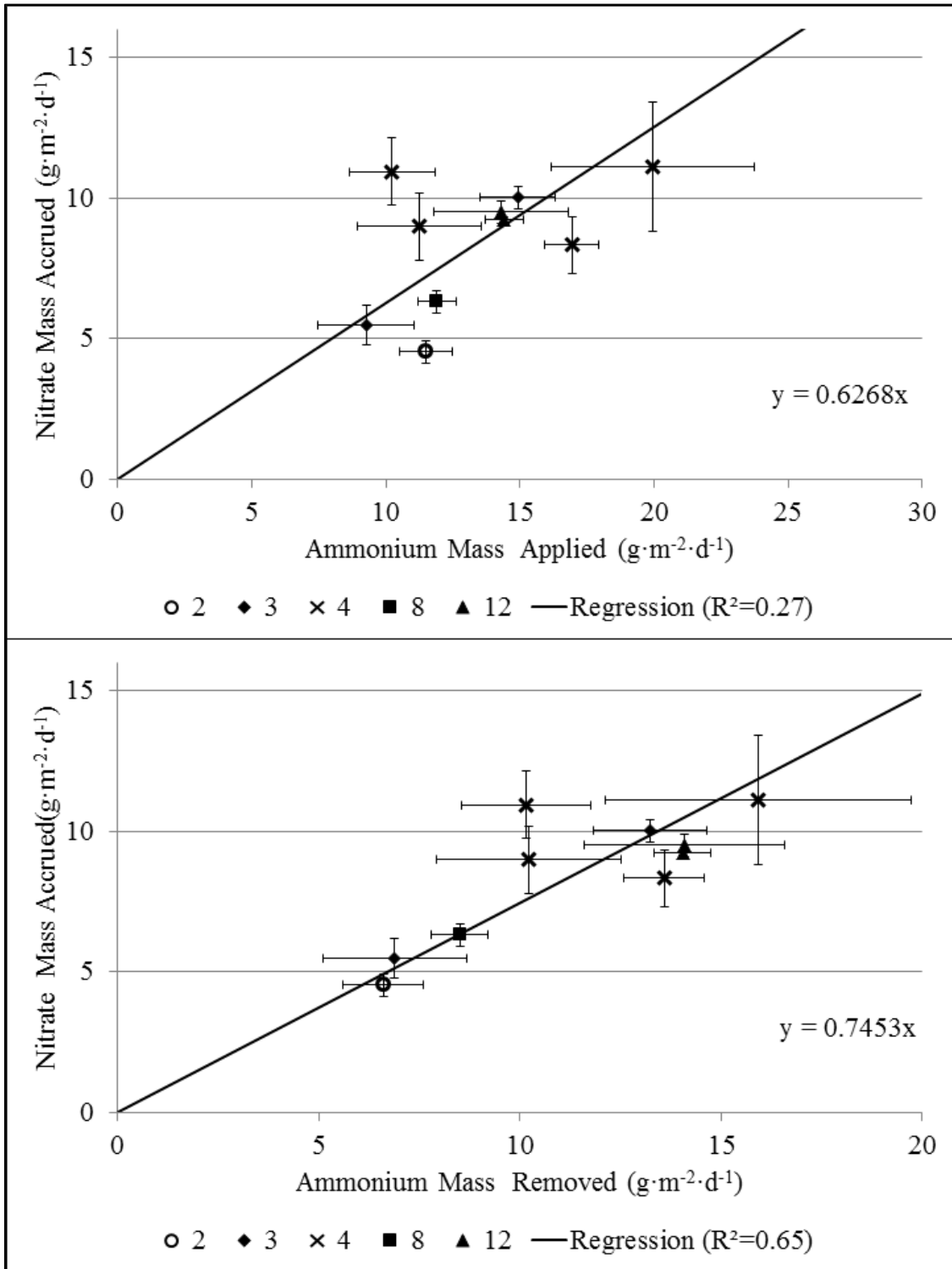


Figure 19: Nitrate mass accrual as a function of ammonium applied (top) and ammonium mass removed (bottom) in the B cells for each loading scheme tested during the 2014-15 season. Symbols are mean values at different dose frequencies (doses per day) and error bars show one standard deviation.

Total Nitrogen Ions Removal. Incoming septic tank TNI concentration was often over $180 \text{ mg-N}\cdot\text{L}^{-1}$ at the beginning of the season, but gradually decreased to approximately $140 \text{ mg-N}\cdot\text{L}^{-1}$ by the end, so the average for the season was $155 \text{ mg-N}\cdot\text{L}^{-1}$ (Figure 20). Flow-weighted influent TNI concentration also decreased as the season progressed, due to not only the septic tank concentrations decreasing, but also to the generally increasing recycle ratio as the season progressed (Table 1) and ranged from as high as $160 \text{ mg-N}\cdot\text{L}^{-1}$ at the beginning of the season, to as low as $45 \text{ mg-N}\cdot\text{L}^{-1}$ at the end.

The B cells reduced TNI concentrations by an additional $10\text{-}30 \text{ mg-N}\cdot\text{L}^{-1}$ for the first half of the season, but removal eventually fell to near zero in mid to late March (Figure 20). TNI concentrations exiting the B cells approached, but never dropped below $40 \text{ mg-N}\cdot\text{L}^{-1}$. Averaged across the entire season and all tested schemes, the A cells removed 42% or $66 \text{ mg-N}\cdot\text{L}^{-1}$ of TNI. Further reduction of $11 \text{ mg-N}\cdot\text{L}^{-1}$ in the B cells made average concentration-based removal across the entire system 49%.

Change in total nitrogen ions (TNI) is primarily the sum of changes in ammonium and nitrate because nitrite concentrations were generally small (less than $5 \text{ mg}\cdot\text{L}^{-1}$) with no discernable trend. TNI mass removal in the A cells during the 2014-15 season was strongly correlated with mass loading regardless of recycle ratio; removal trended near an average of 23% reduction ($R^2=0.84$) for all schemes tested (Figure 21). Because the A cells were generally not effective in removal of ammonium (Figure 15), but achieved high levels of nitrate removal (Figure 18), TNI removal in the A cells is expected to be better with recycling of nitrate rich water than without. This is readily apparent in the effluent concentration data (Figure 21, bottom), with higher TNI values without recycle

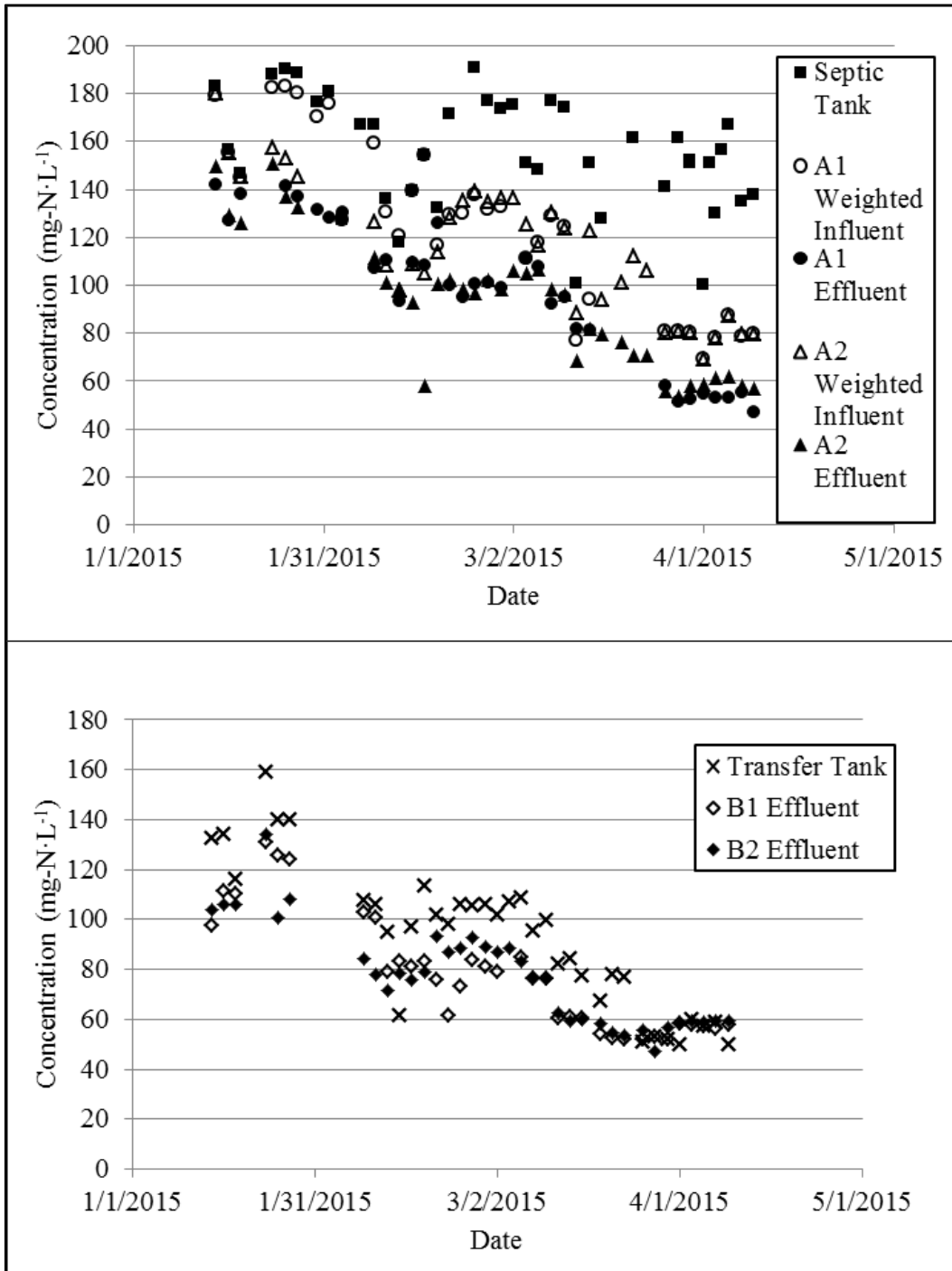


Figure 20: Measured TNi concentrations during the 2014-15 season for the A cells (top) and the B cells (bottom). Influent to the A cells is a flow-weighted average of the septic tank and recycle tank which depends on the recycle ratio. Effluent from the A cells is mixed in the transfer tank and enters the B cells. The recirculation tank mixes effluent from the B cells. Operational parameters of individual cells varied with time (See Table 1).

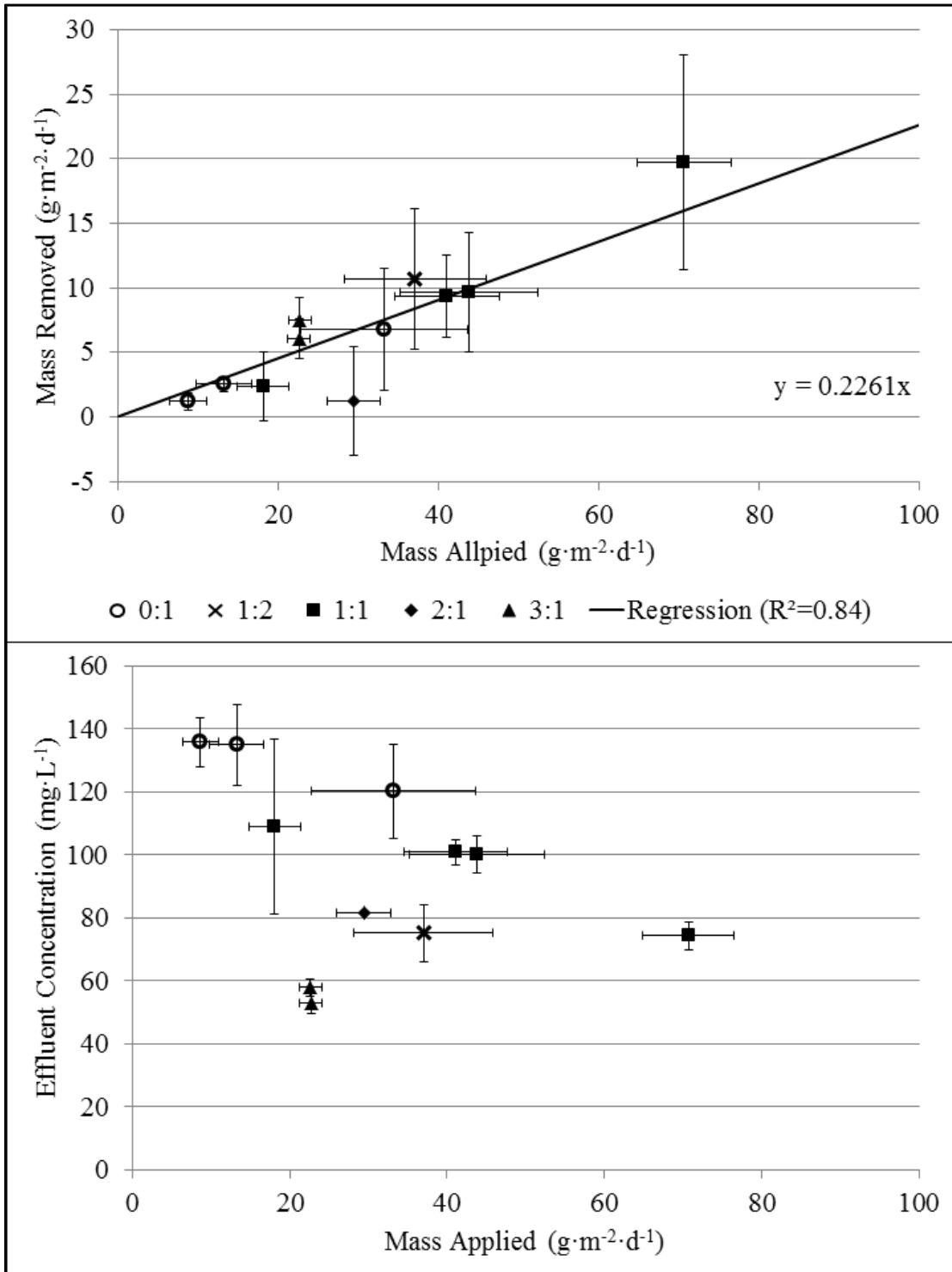


Figure 21: TNI mass removal (top) and effluent concentration (bottom) in the A cells as a function of total mass load for each loading scheme tested during the 2014-15 season. Total mass load includes the mass from septic and recycle water. Symbols are mean values at different recycle ratios, and error bars show one standard deviation.

and the lowest values in schemes with a 3:1 recycle ratio (14-A-11 and 14-A-12) but the effect of recycle is less obvious when TNI mass removal is compared to TNI mass applied (Figure 21, top). However, the recycle ratio influences the relationship between nitrate and ammonia removal in the A cells. Increasing the recycle ratio increased nitrate removal substantially but also tended to decrease the ammonium removal, thus masking the anticipated increase in TNI removal with higher recycle ratios (Figure 22).

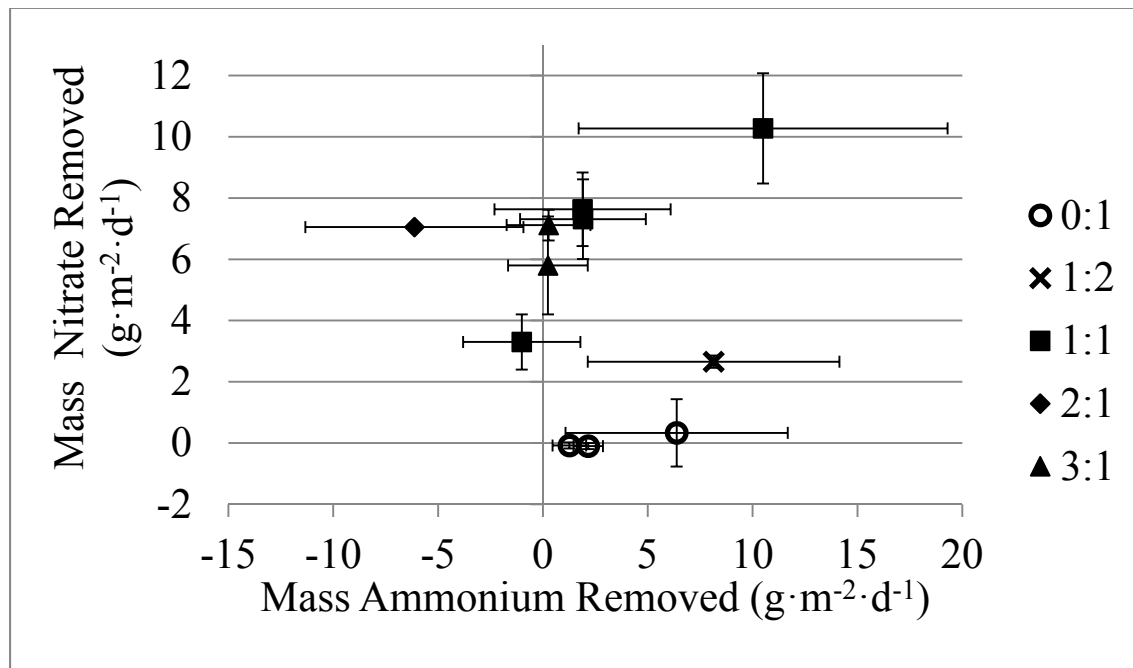


Figure 22: Nitrate and ammonium mass removal in the A cells for loading schemes tested during the 2014-15 season. Symbols represent mean values at different recycle ratios and error bars show one standard deviation

Total nitrogen mass reduction in the B cells was low during the 2014-15 season, never exceeding 50% and averaged only 17% ($R^2=0.28$) of the mass applied. Mean removal ranged from -0.8 to $5 \text{ g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ (Figure 23) with relatively large variance for individual measurements within a scheme. Low TNI removal is expected, as the B cells

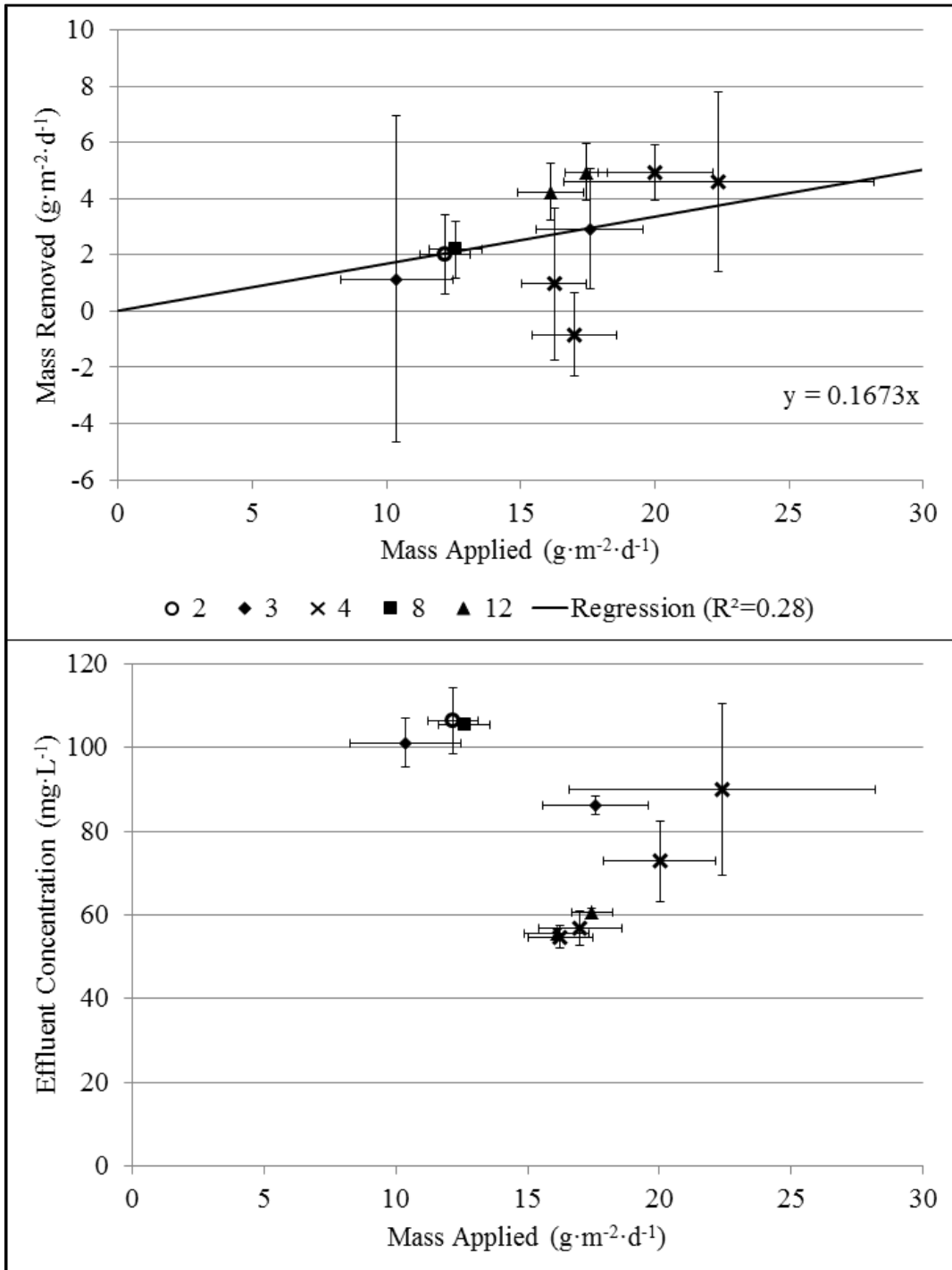


Figure 23: Mean TNI mass removal (top) and mean TNI effluent concentration (bottom) in the B cells as a function of mean TNI total mass load during the 2014-15 season. Symbols reflect different dose frequencies (doses per day) and error bars show one standard deviation.

primarily convert ammonium to nitrate, and hence do not remove TNI. Perhaps the more interesting observation is that they removed TNI at all. There appear to be mechanisms other than nitrification influencing TNI removal; higher loading did result in improved removal. Some denitrification could be occurring as there was certainly sufficient nitrate available and some reduction of COD in the B cells. Other possibilities include continued sorption of ammonium or other microbial removal processes such as ANAMMOX.

DISCUSSION

Retention Volume, Organic Loading and Clogging Problems

During all of the 2012-2013 season and the beginning of the 2013-2014 season, infrequent and large dose volumes ranging from 1.9 to 5.3 cm/dose were applied to the A cells to ensure even distribution of flow across the entire wetland surface. This resulted in generally high effluent COD concentrations ranging from 250 to 400 mg·L⁻¹. Subsequent tracer testing (see tracer section, prior) demonstrated that approximately 90% of the conservative tracer exited the A cells before the peak of the hydrograph was reached when a dose volume of 4 cm was applied, indicating that the hydraulic residence time of doses applied during the 2012-13 and early 2013-14 season may have been too short for significant treatment to occur. This explains the relatively high COD levels observed in the transfer tank, which was subsequently applied to the B cells. This high organic load is believed to have clogged cell B1, which was noted on 2/18/2014, when scheme 13-B-1 was in operation. Initial concerns were that the cell had frozen, but inspection of the media determined that this cell was biologically fouled. COD removal in the B1 cell was high at the time, despite receiving large doses of 15.9 cm. Tracer analysis for the B cells, which showed conservative tracer recovery but was completed by another student, indicates these cells have a hydraulic residence volume of approximately 3660 L. This volume corresponds to a load of 15.4 cm, which was 0.5 cm smaller than the doses used in scheme 13-B-1, but at least 7.4 cm larger than the dose of any other scheme.

Thus, large doses applied on the A cells caused organic carbon breakthrough and high transfer tank COD concentrations. This was applied to the B cells, where a large organic load was consumed microbially, leading to extensive biofilm growth and clogging of the pore spaces. It is unknown if clogging occurred during the previous 2012-13 season. The A cells were operated similarly in terms of daily hydraulic loading, recycle ratio, and dose volume, so it is likely that similar organic carbon breakthrough to the B cells occurred. The B cells received a similar daily hydraulic loading rate, but at much smaller dose volumes. COD concentrations in the tanks are similar in the two seasons, so it seems possible that one or both of the B cells could have clogged. However, it is possible that the small dose volumes on the B cells distributed the available COD through the cell differently and therefore the biofilm growth could have been different as well.

Following the discovery of clogging, the B1 cell was rested (not loaded) to allow shrinkage of the biofilm within the media, and periodic hydraulic testing was performed to assess the conductivity of the cell. This strategy proved effective for reversal of clogging and after 5 weeks, microbial fouling had been reduced enough to resume application of wastewater. During this time, dose volume to each of the A cells was reduced, which allowed for better removal of COD in the A cells and prevented breakthrough to the B cells. This proved to be an important strategy to reduce carbon loads to the B cells, and successfully prevented clogging problems for the remainder of the study.

These experiences show that media size plays a critical role in dictating the appropriate retention volume with important, but difficult to quantify, ramifications on system operation. Larger media has a lower hydraulic retention volume, thus dose volumes must be smaller to prevent rapid breakthrough of contaminants. In contrast, smaller media has a larger hydraulic retention volume, which allows larger doses to be applied. However, for a specified daily hydraulic load, dose load is not independent of the time between doses, which controls the time for necessary reactions to occur. Thus, media size, reaction kinetics, hydraulic load and dose load are intertwined. For a given media size, there is a maximum dose load to prevent contaminant breakthrough, and each contaminant has a minimum reaction time for a desired level of removal, which dictates the dose frequency. The daily hydraulic load is determined from the dose load and dose frequency; hence there is an optimal hydraulic load for a wetland cell that is dependent on this interaction of media size, the influent concentration of a limiting contaminant, and its removal kinetics. In concept, the smaller the media, the higher the optimal hydraulic load, however, smaller media has a much higher risk of clogging, which is primarily caused by organic overloading. Thus, it is logical to put a coarser media in first stage cells and finer media on second stage cells. Perhaps it will be possible, in the future, to specify an optimal media size for removal of a specific contaminant, but knowledge is currently insufficient, and it likely that media will continue to be specified generally and the optimal (or maximum) loading schemes determined by direct experimentation or estimates from the selected media size.

The media used in this system worked well, but may represent the upper and lower bounds of advisable media size, based on both experience in this study and comparison to design criteria in other countries. The media on the A cells required small doses to prevent breakthrough of organic carbon, to the point where smaller doses became impractical and inefficient. At the smallest dose volumes used to ensure high organic carbon removal, water was applied for only 50-75% of the time the pump was on (with the remainder of the time representing the time required to fill and pressurize the pipe network). Furthermore, these doses were too small to ensure broad distribution of water across the surface, thus creating inefficient use of an unknown fraction of the wetlands or dead zones which never received water. These factors combine to reduce treatment efficiency in the system, an effect that would be exacerbated if even larger media were used. No evidence from this experiment suggests that larger media would improve treatment either, so it is quite reasonable to set the upper bound for media in the first stage of a system similar to the one studied here at a d_{50} of about 5 mm.

Interestingly, the French guidelines prescribe a media d_{50} between 2 and 8 mm, which corresponds well to the suggestion drawn from this study (Molle, 2005). An important difference is that the French systems are used for both primary and secondary treatment, and therefore accumulate sludge on the surface of the wetlands. The sludge layer ensures even distribution of applied water across the surface and retards flow to the gravel surface. Dose volume, which was critical for the design used for this study, becomes less important when the sludge layer is serving as a hydraulic control; it slows flow to a trickle and hydraulic overloading with respect to dose volume becomes much

harder, if not impossible. Furthermore, the sludge ensures efficient use of the system by distributing water more evenly across the surface and limits the possibility of dead zones in the hydraulic path. However, other inefficiencies may be introduced by incorporating primary treatment; for example, French systems are operated with weekly rest periods to allow the sludge to dry and decompose which necessitates construction of redundant wetland cells to use during this period (Molle, 2005).

Despite the hydraulic inefficiencies of the A cells, it is important to realize that they performed quite well in regards to COD removal. COD was successfully removed at the highest mass loading of $368 \text{ g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$, which was approximately 10 times higher than the mass loading assumed in the design and 47% greater than the French guideline of $250 \text{ g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ (Molle, 2005). This loading was achieved by applying the entirety of Bridger Bowl's wastewater flow to a single A cell and was still below the maximum removal capacity of the system.

In summary, it is clear that hydraulic overloading of a system is possible both in terms of daily hydraulic load and in terms of dose volume, emphasizing the importance of media size selection when determining proper hydraulic loads in both design and operational guidelines of treatment wetland systems. But by changing the dosing scheme to utilize smaller doses, much higher daily hydraulic loads were possible compared to previous years.

The media size in the B cells proved problematic because the B1 cell clogged, but despite this, the size used is likely appropriate for use in the second stage of similar systems. This cell clogged only when the incoming COD load was quite high due to the

combined effects of high hydraulic load and high COD breakthrough from the A cells due to the large dose volumes. Cell B2, when operating at the same time and with the same COD concentration but lower hydraulic load (and hence lower COD load), did not clog. Additionally, the hydraulic capacity of the clogged B1 cell recovered with rest and with lower, more appropriate COD loads has not clogged again. Based on these results it appears that COD loading should not exceed $35 \text{ g}\cdot\text{m}^{-2}\cdot\text{dose}^{-1}$ for media with a d_{50} of 0.6 mm. It is not advisable, of course, to use such a small media in the first stage of a system, as this would be very likely to cause problems with COD concentrations close to more typical domestic waste levels (Tchobanoglous et al., 2003). High hydraulic retention is desirable in the second stage in order to allow sufficient time for nitrification to take place, therefore increasing the size of the media might reduce the nitrification efficiency of the system. Thus, the media selected seems to strike an appropriate balance between being too susceptible to clogging and reducing nitrification capacity.

Treatment of Nitrogen

High rates of ammonia and TN removal were achieved in this study as well as high COD removal. One scheme (14-A-10), at 1:1 recycle, achieved not only the highest nitrate removal rate but also the highest ammonium removal rate, with just over $10 \text{ g}\cdot\text{N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ of each removed. Additionally, this scheme resulted in the lowest effluent nitrate concentrations of schemes incorporating recycle, below $10 \text{ mg}\cdot\text{N}\cdot\text{L}^{-1}$. This loading is 5 times higher than the French guidelines with respect to ammonia, and because those systems don't typically recycle, the TN load was 7 times the French guideline (Molle,

2005). Thus, the system as operated under scheme 14-A-10 outperformed the industry standards by a considerable margin.

The strong correlation between nitrate applied and nitrate removed in the A cells indicates that nitrate removal in the system was likely substrate limited rather than capacity limited, not only during scheme 14-A-10, but throughout the season. If this limitation was a result of low nitrate concentrations in the cell, which ranged from 5 to 20 mg-N·L⁻¹ through the course of the season, then higher nitrate loading may be possible without negatively impacting removal. Alternatively, the limitation could be a result of low organic carbon availability, which was prioritized to prevent B cell clogging. Reducing the recycle ratio provides a higher organic carbon to nitrate ratio and if the system is carbon limited this should increase removal, however, the limited data collected at 1:2 (R:S) recycle show a lower removal efficiency, so carbon availability might not be the limiting factor. It is also possible that removal was limited by the hydraulic conditions; more contact time may increase removal rates and removal efficiencies if this were the case. However, the dose loads during the most of the 2014-15 season approach the practical minimum for the media size in the A cells, so residence time can only be increased by decreasing the daily hydraulic load and hence the dose frequency. Selection of a smaller media size would increase the hydraulic residence time and could provide better performance if dose retention time was limiting removal, but would make the system more susceptible to clogging, as already discussed.

The high loading capacity of the A cells was leveraged under the schemes 14-A-11 and 14-A-12 which utilized relatively low septic loading but increased recycle ratio to

realize the lowest effluent TNI concentrations. Rates of removal, however, dropped below those observed during scheme 14-A-10 to 7.5 and 6.1 $\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ respectively. There were also higher concentrations of nitrate in the effluent during schemes 14-A-11 and 14-A-12. It is possible the hydraulic load exceeded the capacity of the system, but unlikely since hydraulic loading rate was higher under scheme 14-A-10 and high denitrification rates were observed at that time. Instead, these results suggest that the denitrification became carbon limited, as effluent nitrate concentrations exceeded 20 $\text{mg}\cdot\text{N}\cdot\text{L}^{-1}$ for the first time all season. The increase in recycle ratio was so large that the septic tank water no longer provided enough carbon for denitrification to reduce nitrate levels to single digit concentrations. Although nitrate removal was limited by carbon availability, high recycle *still* improved water quality compared to other schemes due to the large portion of the hydraulic load that was composed of low nitrogen concentration recycle water. This diluted the septic tank water to concentrations that were so low that modest nitrogen removal resulted in the lowest total nitrogen concentrations observed all season. It is likely, however, that higher rates of nitrogen removal and lower effluent concentrations would have been possible with slightly more available carbon.

To examine the effect of available organic carbon on nitrification in the A cells the ratio of organic carbon to nitrate was compared to nitrate removal (Figure 24). Both organic carbon and nitrate are required for denitrifying organisms to grow, so it is common for wastewater treatment operations to evaluate a carbon to nitrogen ratio (C/N). The amount of COD from the septic water is used in lieu of the applied COD because the recycled COD is not very biologically available and thus total COD loading rates do not

represent the portion of COD available for microbial uptake. Furthermore, the traditional C/N ratio is not used here due to the fact that low nitrogen values result in the ratio trending towards infinity; instead an N/C ratio is used, which forces small nitrogen values to zero. An N/C ratio that is too low would result in a nitrate limited reaction rate, and a system with a high N/C ratio would be organic carbon limited. The stoichiometry of denitrification requires a 0.41 mass ratio ($\text{g-N} \cdot \text{g-C}^{-1}$) between available nitrate and organic carbon for reduction of nitrate to occur, but some additional organic carbon is also required for growth of biomass as well. Therefore, the US Environmental Protection Agency reports nitrogen to COD ratios of 0.15 to 0.21 ($\text{g-N} \cdot \text{g-COD}^{-1}$) for commonly used carbon additives for denitrification such as methanol, ethanol, acetic acid, etc. (EPA 823-F-13-016, 2013).

Nitrate removal was greatest when the nitrate to septic COD ratio was near 0.05 which corresponds to a recycle ratio of approximately 1:1 (R:S) in this study, albeit with high removal variability (Figure 24). This value is much lower than the EPA's guidelines, but is likely reasonable since much of the organic carbon in the septic water is less biologically available than the EPA additives designed to promote denitrification. Higher recycle ratios resulted in slightly lower performance, but still showed high removal rates. In contrast, no recycle and 1:2 (R:S) recycle showed relatively low removal efficiency. This suggests that denitrification in the A cells is not carbon limited but might be limited by either too little nitrate or too little reaction time due to low retention volume of the media.

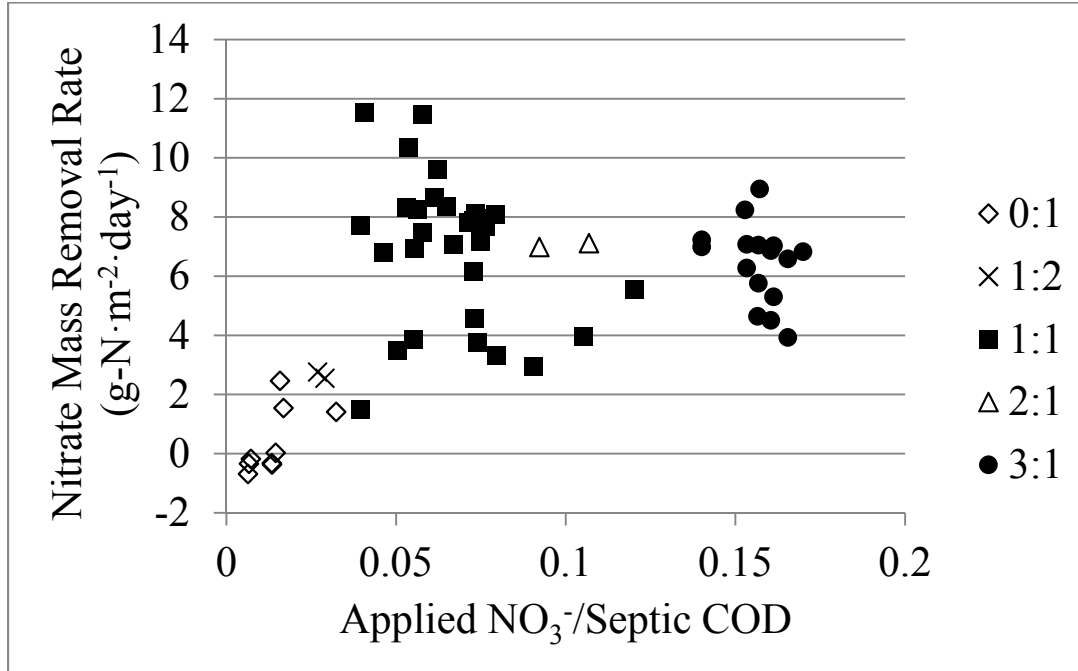


Figure 24: Measured mass of nitrate removed as function of the ratio of applied nitrate to applied septic COD in the A cells. Symbols represent individual values at different recycle ratios. Higher recycle ratios increase the available nitrate relative to available carbon. The highest mass removal of nitrate occurs at a ratio of approximately 0.05.

The effect of partial saturation in the A cells on denitrification was investigated during schemes 14-A-11 and 14-A-12. Water level was raised to 71 and 48 cm for each scheme respectively and recycle ratio was increased to 3:1 (R:S). All water quality parameters demonstrated a large temporal change within the scheme as the cells responded from a completely free-draining to partially submerged condition (Figure 25). Interestingly, net nitrate removal decreased while ammonium removal increased, but the net effect on TNI removal in the A cells is negligible. More anoxic conditions created by impoundment should have stimulated denitrification and limited nitrification. However, concurrent COD removal increased from only about 20% at the first sample to about 60% at the end of the two week sample period. This is consistent with the observed nitrogen

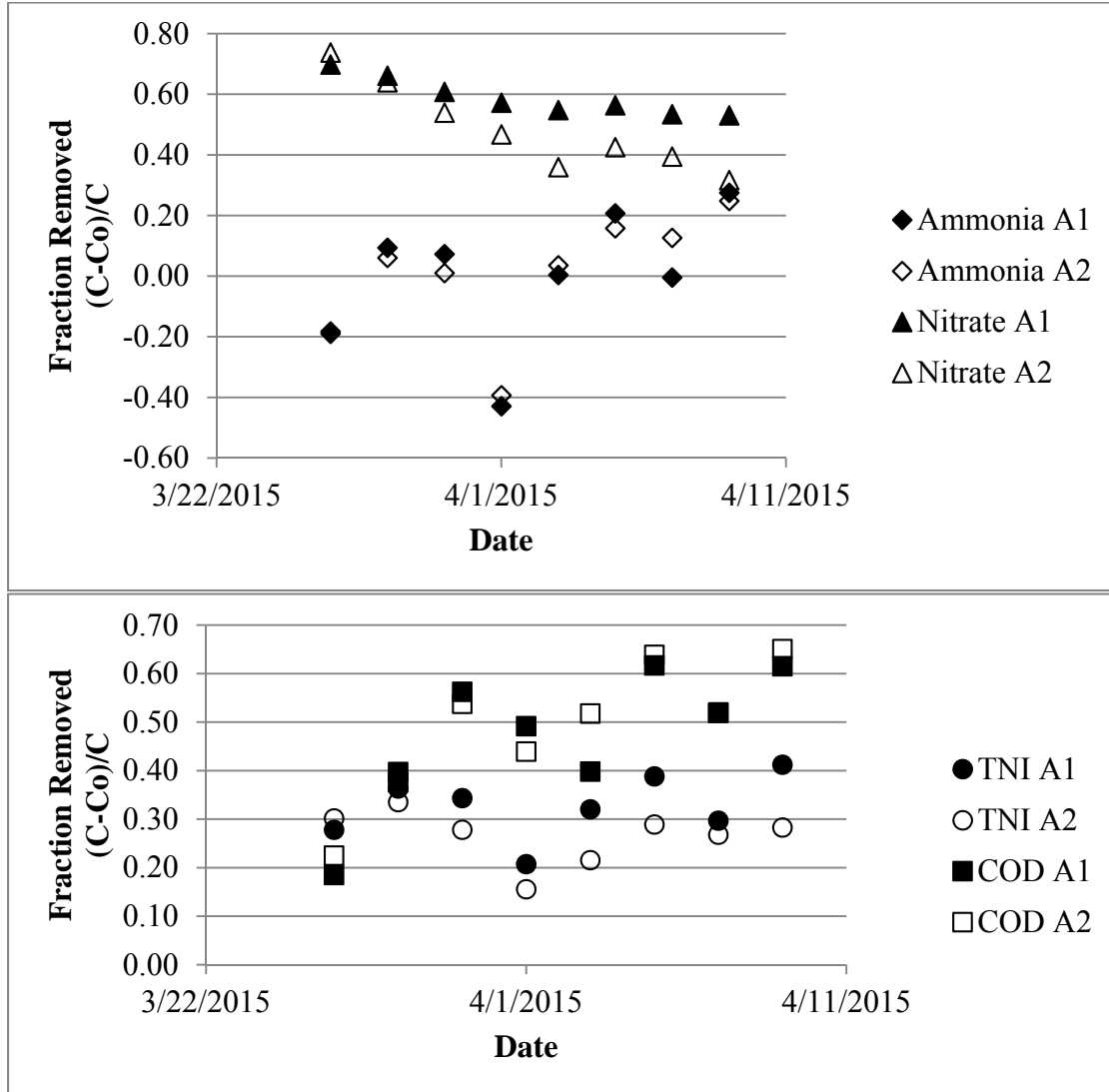


Figure 25: Removal of nitrate, ammonium, TNI and COD within loading schemes with impoundment (14-A-11 and 14-A-12), showing rapid degradation of nitrate removal and improvement in COD removal over the course of two weeks.

dynamics, as near complete removal of biologically available COD would tend to leave excess oxygen available for nitrification, but the cell as a whole behaved exactly opposite of what was anticipated. The rapid change in saturation clearly altered the microbial community structure more than anticipated and a more detailed investigation needs to be conducted to determine the influence of saturation on nitrogen dynamics.

The B cells were primarily utilized for nitrification of ammonia to nitrate, thus TNI removal was low throughout the season, but ammonium mass removal was as high as $16 \text{ g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$. Under this scheme (14-B-4) TNI loading was $20 \text{ g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$, a load that is 33% higher than the French system recommendation (Molle, 2005). Despite this, removal efficiency was not significantly different than the 73 French systems surveyed in Molle's study. Furthermore, it was shown that greater than 99% ammonium removal is possible (schemes 14-B-8, 14-B-10, and 14-B-12), highlighting the potential of the system.

During these schemes, ammonium application rates were at or below the French loading recommendation of $15 \text{ g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$. Interestingly, there was a strong seasonal effect to the ammonium removal efficiency, ranging from 50-60% at the beginning of the season and steadily increasing to 95-100% at the end (Figure 14). Furthermore, cell B2 begins with higher ammonium transformation efficiency and continues to outperform cell B1 for the entirety of the season. This suggests that slow, but steady, growth of nitrifiers throughout the season is the dominant driver of ammonium reduction in the B cells. This trend overwhelms any discernable effects due to operational changes made to the system and implies that the cells were overloaded at the beginning of the season and slightly underloaded at the end. This effect may have been intensified by pumping of the septic settling tanks at the base area, which is done annually during the summer months and results in 6-10 weeks of no new wastewater flow to the wetlands. In 2014, this was done at the beginning of August, and new flow to the wetlands did not resume until mid-October, meaning microbes were starved of nutrients until temperatures were low and growth rates slow. It appears that the growth of nitrifying bacteria early in the season

limits removal, so steps to maintain flow before the season starts could provide a means of improving system performance. Pumping and refilling of settling tanks is a relatively flexible event that can occur anytime during the summer months. Modifications to operation could eliminate the no-flow starvation period and provide some nitrogen to microbes throughout the summer, which would minimize microbial die-off.

Total System Performance

The wetlands were subjected to numerous different operational conditions during the 2014-15 season and a hydraulic balance between the A cells and B cells was not always prioritized (more water may have passed the A cells than B cells). However, a hydraulic balance was tested during the final weeks of the season to evaluate overall performance of the entire system when all cells were operated at a target total hydraulic load of $28.8 \text{ cm}\cdot\text{d}^{-1}$. During this time mean removal efficiency for total nitrogen ions was 58% with greater than 99% ammonium conversion (Table 6). At the loading rates utilized, this represents mean mass reduction of 5.0 and $3.1 \text{ g}\cdot\text{N}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ for ammonium and TNI respectively for the system as a whole. Removal of carbon was high, with a mean removal efficiency of 92% and a mass removal of $34.8 \text{ g}\cdot\text{COD}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$. This corresponded to only $10 \text{ mg}\cdot\text{L}^{-1}$ of BOD_5 remaining in the effluent.

Table 6: Removal rates and efficiencies evaluated across the entire system, with hydraulic balance between A and B cells during the final two weeks of the 2014-15 season (Schemes 14-A-11, 14-A-12, 14-B-7, and 14-B-12)

	COD	TNI	NH_4^+
System Removal Rate ($\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$)	34.8	3.1	5.0
System Removal Efficiency	92%	58%	99%

These results compare well with current leading research and highlight the potential of the system to provide high quality treatment at the site. Furthermore, other schemes tested demonstrated that individual components of the system could achieve even higher removal at higher loading rates, indicating that the data in Table 6 does not represent the best expected performance.

Stark contrasts in treatment performance are apparent when total nitrogen ion removal is compared between the first year of the study, when the the system was experiencing start-up effects and was unplanted, and the most recent year, when they system had been tested and optimized, and plants had matured (Figure 26). During the first season, adsorption of ammonium is witnessed, followed by the rapid decay of removal thereafter. In contrast, total nitrogen ion performance during the 2014-15 season increased linearly as the season progressed and as operational changes were made, even though the system was operated at the same or much higher hydraulic and mass loading rates. It is quite clear, therefore, that the addition of plants, a mature biofilm, and improved operating procedures each combine to make an enormous difference in total nitrogen removal.

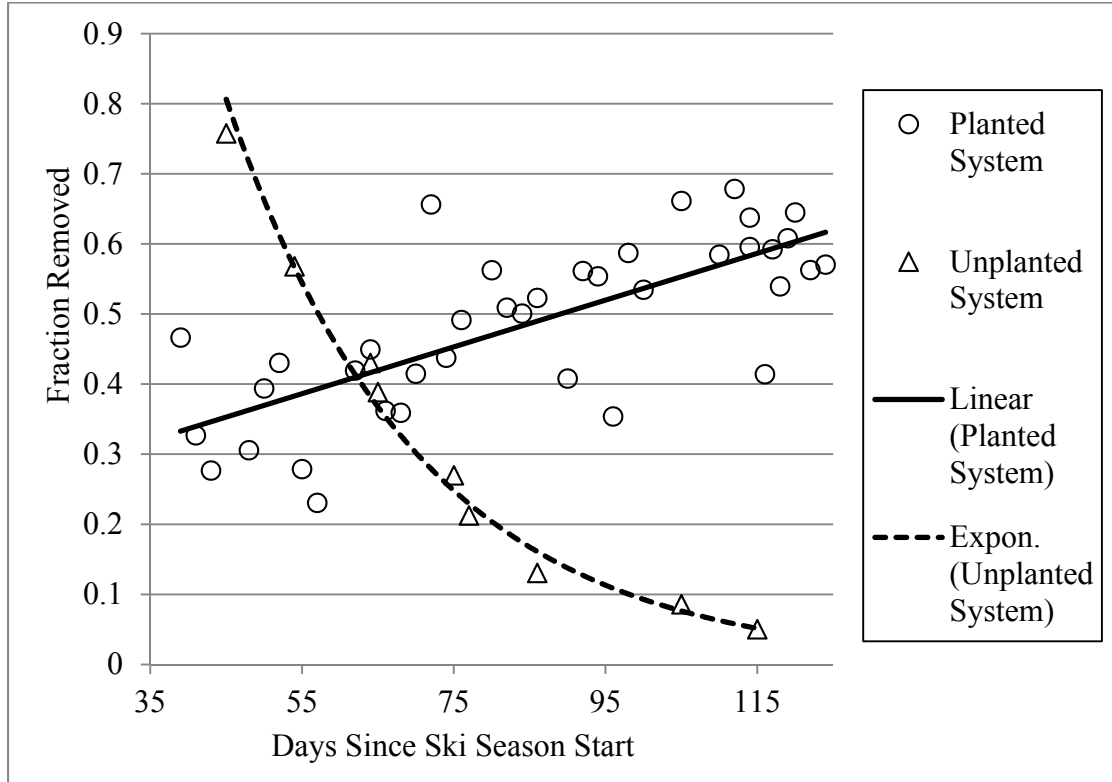


Figure 26: Total nitrogen ion removal during the 2012-13 season without plants and the 2014-2015 season with plants

CONCLUSION

Evaluation and optimization of the pilot scale treatment wetlands at Bridger Bowl provided a valuable opportunity to evaluate numerous aspects of treatment wetland construction and performance. This treatment wetland system has been shown to be robust when operated adeptly in this challenging wastewater treatment scenario where more conventional systems failed to achieve desired performance.

The study confirms that cold temperatures do not prevent treatment or inhibit usage of a wetland system and contributes to the body of evidence demonstrating that wetlands systems can and should be utilized in cold climates. Despite water temperatures that were only slightly above freezing and air temperatures that dropped below -20°C , no problems with freezing of the wetlands or associated distribution and collection pipes, or any other component of the system were encountered.

The ability of two plants to grow and thrive despite the short growing season was also evaluated. While differences in plant recovery following planting shock existed, both plants proved viable and adaptable to site conditions. Additionally, a vast improvement in treatment performance was observed following planting compared to the unplanted system. In fact, even when plants experienced slow growth and were not yet mature, they provided measurable improvements in treatment. Monitoring of plant growth and treatment performance should be continued in future years to evaluate long-term effects of plants in the specific climate of the project site.

Media size and resulting hydraulic conditions were shown to be critical in operation and optimization of the system. The larger media in the A cells created a small

retention volume which was easily exceeded, but minimized clogging potential and likely provided high rates of oxygen transfer throughout the cell. In the B cells, smaller media proved susceptible to clogging when overloaded with organic carbon. In the absence of carry-over organic carbon, however, this media increased retention time while still providing oxygen and thus was effective at creating conditions for nitrification to occur. The importance of evaluating hydraulic characteristics was highlighted, as lessons learnt about overloading provided a theoretical basis for effective decision making regarding methodologies to improve treatment. It is prudent to ensure dose volumes are smaller than hydraulic residence volume to allow sufficient time for treatment to occur. Furthermore, it was shown that by sufficiently decreasing dose volume compared to the hydraulic retention volume, higher daily hydraulic loading rates could be applied and thus higher mass loading rates realized. In light of this, it is advisable to conduct simple hydraulic tests on any wetland systems so that this information can be utilized to inform dosing regimens. Additionally, designers should be cognizant of the effect of media size on hydraulic characteristics when selecting media. Media utilized in the A cells probably represents a maximum feasible size ($d_{50} = 5\text{mm}$), as the appropriate hydraulic retention volume is so small that dose volumes are near the minimum that are hydraulically feasible to ensure even distribution of water across the cell.

Successful nitrification occurred within the system; complete ammonium transformation occurred at times. Although many other studies report limited nitrification at low temperatures, this study did not experience such problems. However, operational conditions were altered rapidly and the system probably never reached steady-state

performance for any one operational scheme. The possibility remains that growth of nitrifiers is slow and builds throughout the season, and a test should be conducted to evaluate if this is occurring and how it affects treatment performance. If possible, detailed water temperature data should be collected to see if correlations between temperature and nitrification can be found, as shown in other studies.

It is likely that the system's capability to nitrify changes seasonally based on loading, temperature, and other factors, so operational strategies may need to be modified to account for this if changes in performance over the course of a season are detected. Operational changes could be enacted to ensure higher nitrogen input loads to the wetlands during the summer, which would promote the growth of nitrifiers year-round. This might reduce the amplitude and intensity of the growth curve observed during the 2014-15 season. To do this, the primary settling septic tanks at the base area could be filled with spring water immediately after pumping, which occurs in the summer months. This would eliminate the several week period of no new septic flow to the wetland and at least maintain low concentrations of nitrogen to the system year round. Operating in this manner might sustain more nitrifying bacteria through the summer and result in higher rates of ammonium transformation at the beginning of the year.

Additionally, investigation of microbial communities at the site could yield interesting and important information about nitrification in cold climate systems. It is possible that the cold water source causes natural selection of organisms that are more cold tolerant than those found in other systems, which could dramatically change paradigms surrounding operation of cold weather treatment wetland systems.

Total nitrogen performance of the system showed promise, but there is likely room for improvement. Removal rates compare well with other systems in less extreme treatment environments, and the ability of the system to remove nitrogen at high rates has been shown. The primary barrier to further total nitrogen reduction is denitrification, which has not yet been fully optimized. Recycle of nitrified water improved denitrification, but it is possible that the process was still carbon limited. Impoundment of water within the A cells did not improve performance, possibly due to the carbon limitation, or the short duration of this test.

Future efforts should seek to better manage carbon and nitrate ratios in the system to improve removal of nitrate in the A cells without causing carbon inhibition of nitrification in the B cells. Creation of anoxic conditions in part of all the A cells seems particularly promising, as this should favor denitrification, while also removing a significant amount of carbon. Carefully balancing of recycled nitrate loads with carbon rich septic loads should result in near complete removal of both substrates within the A cells. Hydraulic loading rates under these conditions may need to be reevaluated to ensure sufficient contact time for denitrification and carbon removal to occur in anoxic conditions. Dosing strategies could also be tested to promote conditions in which both carbon and nitrate would be present in the same region at the same time, and thus available for denitrifying microbes. Utilization of the in-cell sampling ports could prove useful in investigating carbon and nitrate mixing and removal within the cell.

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