# An Assessment of a Wetland-Reservoir Wastewater Treatment and Reuse System Receiving Agricultural Drainage Water in Nova Scotia

by

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The undersigned hereby certify that they have read and recommend to the Faculty of Graduate Studies for acceptance a thesis entitled "An Assessment of a Wetland-Reservoir Wastewater Treatment and Reuse System Receiving Agricultural Drainage Water in Nova Scotia" by Michael J. Haverstock in partial fulfillment of the requirements for the degree of Master of Science.

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# ABSTRACT

A wastewater treatment and reuse system consisting of a tile drainage system, a constructed treatment wetland (CTW), a reservoir, and an irrigation system was established. The system supplied 780 mm of irrigation water for the 1.8 ha of drained land for the 2008 growing season. A hydraulic tracer study conducted in the CTW supported the use of a length to width ratio of 10:1. During 2008, annual nitrate-nitrogen ($NO_3^-$-N) and *Escherichia coli* (*E. coli*) mass reductions were 67.6 and 63.3%, respectively. Elevated *E. coli* levels were observed in the reservoir during the warm season. Therefore, water may not be safe for irrigating crops consumed raw. The mean first-order areal uptake rate constants generated for $NO_3^-$-N and *E. coli* were 8.0 and 6.4 m $y^{-1}$, respectively, and are recommended for similar CTWs. A wetland area to drainage area ratio of 4.5% is recommended to achieve ≈ 70 % mass reduction of $NO_3^-$-N and *E. coli*.

# LIST OF ABBREVIATIONS USED

| | |
|---|---|
| A | Wetland surface area |
| $A_D$ | Drainage surface area |
| $A_W$ | Wetland surface area |
| APHA | American Public Health Association |
| BEEC | Bio-Environmental Engineering Centre |
| $Br^-$ | Bromide |
| C* | Background concentration |
| C(t) | Exit tracer concentration |
| $C_{in}$ | Concentration at the constructed treatment wetland inlet |
| $C_{out}$ | Concentration at the constructed treatment wetland outlet |
| $C_{out\_corr}$ | Corrected concentration at the constructed treatment wetland outlet |
| CDN | Canadian Dollar |
| CTW | Constructed Treatment Wetland |
| dV / dt | Change in volume ($m^3$ $month^{-1}$) |
| E. coli | Escherichia coli |
| ET | Evapotranspiration |
| f(t) | Residence time distribution function |
| FC | Fecal coliform |
| k | First-order areal uptake rate constant |
| m | Total tracer mass collected |
| MR | Mass Reduction |
| $M_{Recov}$ | Total tracer mass recovered |
| N | Nitrogen |
| $NO_2$ | Nitrite |
| $NO_3^-$ | Nitrate |
| $NO_3^-$-N | Nitrate – Nitrogen |
| P | Phosphorus |
| PRECIP | Precipitation rate |
| Q | Annual inflow |
| $Q_i$ | Wetland inflow rate |
| $\overline{Q_o}$ | Mean daily wetland outflow rate |
| $Q_o$ | Wetland outflow rate |
| RT | Residence Time |
| RTD | Residence Time Distribution |
| SRP | Soluble Reactive Phosphorus |
| t | Time |
| $t_a$ | Actual measured residence time |
| $t_n$ | Nominal residence time |
| TN | Total Nitrogen |
| TP | Total Phosphorus |
| USD | United States Dollar |
| V | Wetland volume |
| $V_{et}$ | Volume of evapotranspiration over constructed treatment wetland area |

| | |
|---|---|
| $V_{in}$ | Volume of inflow to the constructed treatment wetland |
| $V_{in}$ | Volume of outflow from the constructed treatment wetland |
| $V_{precip}$ | Volume of precipitation over constructed treatment wetland area |
| W1 | Wetland 1 |
| W2 | Wetland 2 |
| WRIS | Wetland-Reservoir Irrigation System |
| $\eta$ | Porosity |
| $\sigma^2$ | Residence time distribution variance |

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# CHAPTER 1 - INTRODUCTION

Artificial subsurface drainage is used extensively throughout Nova Scotia to remove excess water from soil (Gartley et al., 1986). Removing excess water permits earlier field trafficability and enhances growing conditions, which ultimately results in a more productive crop system. However, agricultural non-point source pollution, including subsurface drainage water, is a major source of surface and groundwater degradation (USEPA, 2007), as it can contain nutrients, pathogens, pesticides, and sediment. The export of these pollutants can have major ecological, health, and socio-economic effects. Waterborne illnesses (Hunter, 1997), eutrophication (Bricker et al., 2007; Harper, 1992), and toxicity to biota (CCME, 2003) are among the most notable.

Another water management issue affecting Nova Scotia is water availability during the growing season. Nova Scotia has an abundance of groundwater, lakes, and rivers and receives up to 1,600 mm of annual precipitation (Davis and Browne, 1996; Environment Canada, 2007). However, it has endured droughts in recent years and dry periods are projected to become more frequent and severe (Vasseur and Catto, 2008).

As Canada's largest water consumer and a significant polluter (Coote and Gregorich, 2000; Environment Canada, 2004b), the agricultural industry has a responsibility to use water efficiently and mitigate water impacts. Wastewater treatment and reuse systems have the potential to address both pollution from agricultural drainage, and water supply issues. A wetland-reservoir irrigation system (WRIS) is an integrated wastewater treatment and reuse system that captures surface runoff and/or tile drainage discharge, uses a constructed treatment wetland (CTW) to improve water quality, and stores the treated water in a reservoir (Allred et al., 2003; Tan et al., 2007). This water can be utilized for irrigation, upon which the cycle of drainage, capture, and treatment may continue. For these systems to be effective and efficient they must be assessed in and adapted to local environmental conditions and needs, such as cold climates. Specifically, a better understanding of system hydraulics and water quality is required.

## 1.1 OBJECTIVES

The goal of the present study is to assess and adapt a wetland-reservoir wastewater treatment and reuse system receiving agricultural drainage water. The specific objectives include:

1. Design and construct a wastewater treatment and reuse system consisting of a tile drainage system, a CTW, and an irrigation reservoir;
2. Assess system overall hydraulics;
3. Assess CTW performance by determining annual mass reductions of nitrate-nitrogen ($NO_3^-$-N), total phosphorus (TP), soluble reactive phosphorus (SRP), and *Escherichia coli* (*E. coli*);
4. Determine first-order areal uptake rate constants (*k*s) for $NO_3^-$-N, TP, SRP, and *E. coli*; and
5. Assess the reservoir water quality for $NO_3^-$-N, TP, SRP, and *E. coli*.

# CHAPTER 2 - LITERATURE REVIEW

## 2.1 ARTIFICIAL SUBSURFACE DRAINAGE

In Nova Scotia, artificial subsurface drainage, often referred to as tile drainage, consists of perforated plastic pipes that are typically installed at depths of up to 1.0 m and spaced 9 to18 m apart, depending on topography and soil characteristics (Gartley et al., 1986). Tile drains remove excess water from the soil profile caused by precipitation or a shallow water table. The benefit of removing this water is to support increased crop yields and quality. Removing excess water increases productivity by permitting earlier field trafficability (Kornecki and Fouss, 2001), facilitating an earlier increase in soil temperatures for earlier germination (Gartley et al., 1986), and creating an aerobic environment that promotes root development (Gartley et al., 1986) and is required by beneficial aerobic microorganisms (Higa and Parr, 1994). Tile drainage also enhances infiltration and thereby reduces surface erosion and runoff, which helps to maintain soil health (Skaggs et al., 1982).

Tile drainage of agricultural land is used extensively throughout Nova Scotia with ≈ 18,000 km being installed between 1940 and 1998 (Cochrane, 2008) through provincial Land Improvement Programs (NSDAM, 1991). In most cases, no provisions were made for limiting the direct discharge of drainage effluent into surface water systems.

## 2.2 DRAINAGE WATER QUALITY

Agricultural subsurface drainage water may contain a variety of pollutants, the most prevalent being nitrogen (N), phosphorus (P), pathogens, salts, sediment, and pesticides (Kladivko et al., 2001). Studies examining tile drainage water quality in Nova Scotia have reported $NO_3^-$-N, TP, SRP concentrations as high as 36 mg $L^{-1}$ (MacDonald, 2001), 5 mg $L^{-1}$ (Kinley et al., 2007), and 1 mg $L^{-1}$ (Kinley et al., 2007; Lamond, 2005), respectively. *E. coli* levels as high as 34,000 CFU 100 $mL^{-1}$ (Thiagarajan et al., 2007) have also been reported. Currently, no guidelines exist for drainage water quality in Nova Scotia; however, effluent concentrations often exceed water quality guidelines for other purposes, which are presented in Table 2.1. Fecal coliform (FC) and *E. coli* are the most commonly used indicator bacteria to indirectly monitor pathogens (Edberg et al., 2000). When comparing FC levels to *E. coli* levels it should be noted that *E. coli* cannot be present in higher levels because it is a single species within the FC group.

**Table 2.1** Selected water quality guidelines for nitrate-nitrogen ($NO_3^-$-N), total phosphorus (TP), and *Escherichia coli* (*E. coli*).

| Guideline | $NO_3^-$- N (mg $L^{-1}$) | TP (mg $L^{-1}$) | *E. coli* (CFU 100 $mL^{-1}$) |
|---|---|---|---|
| Drinking Water | 10[A] | NA | 0[A] |
| Irrigation Water | 30[B] | NA | 100[C,†] |
| Livestock Water | 100 [C,‡] | NA | NA |
| Protection of Aquatic Life | 3[D] | 0.03[E]; Site specific[F,G] | NA |
| Recreation and Aesthetics | NA | 0.03[E] | 2000[H,†] |
| Wastewater Discharge | Site specific[I] | Site specific[I] | 200[I] |

[A] FPTCDW (2008); [B] Ayers and Westcot (1994); [C] CCME (2005); [D] CCME (2003); [E] (Environment Canada, 2004a); [F] CCME (2004); [G] CCME (2007); [H] CCME (1998); [I] CBCL (2006)
[†] Reported as fecal coliform
[‡] Reported as nitrate + nitrite

A number of factors work, often in combination, to affect subsurface drainage water quality. They must be considered when developing measures to mitigate water pollution from drainage water. Both soil moisture content and precipitation govern tile drainage flow rates and can also have a significant effect on water quality. Leaching and macropore preferential flow are the two main mechanisms that transport pollutants, particularly soluble pollutants such as nitrate ($NO_3^-$) and dissolved P (Beven and Germann, 1982). Less soluble pollutants, such as particulate P and pathogens, can enter drainage lines attached to soil sediment (Chapman et al. 2001; Reddy et al., 1981), although they are more commonly associated with surface runoff. Higher concentrations of $NO_3^-$-N (Bakhsh et al., 2005; Raisin et al., 1997; Randall et al., 2003), TP and SRP (Fink and Mitsch, 2004; Kinley et al., 2007; Raisin et al. 1997) and *E. coli* and FC

(Coulter, 2005) are observed during high flow events. Managing these periods of high pollutant loading is a major challenge for drainage water treatment.

The other major factor that affects subsurface drainage water quality is the application of fertilizers, particularly if the timing coincides with high drainage flow events. High fertilizer application rates have been shown to cause greater $NO_3^-$-N losses than lower rates (Bakhsh et al., 2005; Jaynes et al., 2001). Increased concentrations of SRP have been observed 3 weeks to one year after application (McDowell and Sharpley, 2001). Elevated levels of FC have been observed as soon as 20 minutes (Dean and Foran, 1992; Jamieson et al., 2002) and as late as several months (Coulter, 2005; Jamieson et al., 2002) following manure applications. Tillage practices (Drury et al., 1993; Thiagarajan et al., 2007), tile drain depth (Astakie et al., 2001; Skaggs and Cheschier, 2003), crop type (Randall et al., 1997) and soil conditions (Sharpley et al., 2003) are other factors which affect drainage water quality.

## 2.3 Water Pollution from Agricultural Drainage

Discharging drainage water into the environment can have significant ecological, health, and socio-economic effects. Effects are most likely observed in intensively farmed areas where the effects are cumulative (Spaling and Smit, 1995).

### 2.3.1 Ecological Effects

Nutrient enrichment of surface water can lead to eutrophication; a rapid population growth of phytoplankton and their subsequent decomposition (Bricker et al., 2007; Harper, 1992). Eutrophication causes numerous changes to biological communities and ecological processes by reducing sunlight transmission, depleting dissolved oxygen levels, and creating a toxic environment for aquatic life (CCME, 2003). Nitrogen and P are usually the growth-limiting nutrients in saltwater and freshwater systems, respectively (CCME, 2007; Goldman et al., 1990). Nutrient export from agriculture, including drainage water, can be a contributing factor to eutrophication (Chambers et al., 2002; Coote and Gregorich, 2000; Sharpley et al., 2003). The most cited example that implicates agriculture is hypoxia in the Gulf of Mexico (Burkhart and James, 1999;

Magner et al., 2004; Rabalais et al., 2001). Strain and Yeats (1999) compiled an eutrophication index for 34 inlets in Nova Scotia. These inlets showed a wide range in eutrophication. However, the authors did not find any correlation between eutrophication index and agricultural activity. Srivastava et al. (1995) investigated 14 lakes in Nova Scotia of varying trophic status and found a strong correlation between TP and total nitrogen (TN) concentrations and differences in aquatic vegetation. Direct toxicity to biota, such as livestock, is also a concern.

### 2.3.2 Public Health Effects

Perhaps the most significant effect is the public health risk created if drainage water pollutes drinking, recreational or irrigation water sources. Many waterborne illnesses are caused by enteric organisms, such as bacteria, protozoa, and viruses, which often originate from the land application of manure (Hunter, 1997; Roosen et al., 2000). A recent high profile case of waterborne illness that implicated agriculture was the *E. coli* O157:H7 spinach outbreak originating in California. Three people died and 203 became ill after consuming spinach that was irrigated from wells near surface water that had been contaminated by livestock manure (CalFERT, 2007).

Consuming water with high concentrations of $NO_3^-$ can cause methemoglobinemia (Camp, 2007; Knobeloch et al., 2000). Methemoglobinemia occurs when the body converts $NO_3^-$ to nitrite ($NO_2$), thereby decreasing the oxygen-carrying capacity of blood, causing shortness of breath, a blue discoloration of the skin, and potentially death (Camp, 2007; Knobeloch et al., 2000).

These public health risks are relevant to Nova Scotia. According to Statistics Canada (2006) irrigation is used on 7% (255) of farms in Nova Scotia, accounting for 0.8% (3217 ha) of total farmland. Irrigation is projected to increase (Environment Canada, 2004b) as farmers attempt to compensate for more frequent and severe periods of water deficit due to climate change (Vasseur and Catto, 2008) and increased water demands (NSDEL, 2002). Primarily fruit and vegetables crops are irrigated in Nova Scotia (Statistics Canada, 2006), which are often consumed raw and thereby are more susceptible to the transmission of waterborne illness through irrigation water. The risk of waterborne illness transmitted through drinking water is a concern in Nova Scotia because over 40% of the population, mostly in rural areas, use private wells (NSDEL, 2002). Furthermore, not all municipal water supplies have source protection programs (NSDEL, 2002), which could address the risk of contamination from agricultural non-point source pollution.

### 2.3.3 Socio-Economic Effects

In a few cases, ecological and public health effects can transform into broader socio-economic effects. For example, the spinach industry was impacted by recalls and consumer apprehension following the relatively small CalFERT (2007) case. Commercial fisheries have disappeared in the region of the Gulf of Mexico where hypoxia has been observed. Recreation and tourism can also be affected when fishing, swimming, and scenic areas are polluted (Bricker et al., 2007).

## 2.4 Drainage Water Management Options

There are numerous management strategies and alternative practices that can be used to mitigate agricultural impacts on drainage water quality. Dinnes et al. (2002) reviewed management strategies for reducing N leaching from tile drained fields. Improved timing of N application at appropriate rates, optimizing N application technology, reducing tillage, diversifying crop rotations, using cover crops, and using nitrification inhibitors all showed potential for reducing leaching losses.

Oquist et al. (2007) compared N and P leaching from tile drained fields under alternative and conventional practices. Alternative practices significantly lowered losses, partly attributed to reduced infiltration. Alternative practices investigated included organic management practices, crop species biodiversity, and practices that reduced inputs of synthetic fertilizers and pesticides.

Controlled drainage is the practice of using tile drainage during wet periods to remove excess soil water and restricting tile drainage flow during dry periods to maintain a sufficient water table (Belcher and D'Itri, 1995). A review of controlled drainage research found that N and P losses could be reduced by 30 to 50% by using controlled drainage instead of conventional drainage (Evans et al., 1995).

## 2.5 Constructed Treatment Wetlands

Constructed treatment wetlands are one of the more promising end-of pipe wastewater treatment technologies because they can be less expensive, less energy intensive, and more easily operated and maintained compared to conventional water treatment systems (i.e. chlorination or ultra-violet systems). They have been used to treat a variety of agricultural and industrial wastewaters, including drainage water. Constructed treatment wetlands are engineered aquatic systems comprised of soil, vegetation, and water environments, facilitating biogeochemical cycling. Biogeochemical cycling is the transport and transformation of chemicals by interrelated chemical, biological, and physical processes, and ultimately results in improved water quality (DeBusk, 1999; Kadlec and Knight, 1996; Mitsch and Gosselink, 2007). Free water surface flow wetlands are the most common type of CTW. They are comprised of shallow vegetated zones and deep unvegetated zones, which provide the aerobic and anaerobic environments required for nitrification and denitrification, respectively (Patrick and Reddy, 1976). Additional benefits of CTWs are that they improve aesthetics and increase biodiversity by creating new wildlife habitat in the farm landscape (Feierabend, 1989). Increased biodiversity can provide benefits such as increased crop production through pollination and natural pest control (Gurr et al., 2003). The new wildlife habitat can also be utilized for hunting and fishing (Kovacic et al. 2000).

### 2.5.1 Constructed Treatment Wetlands Receiving Drainage Water

Drainage water has a relatively low biological oxygen demand compared to other wastewaters. However, the dynamic pollutant loading of drainage water presents a treatment challenge. Hydraulics, specifically residence time (RT), is one of most significant factors that affect treatment (Kadlec, 1994; Kadlec and Knight, 1996; Mitsch and Gosselink, 2007). Residence time is the length of time that an individual parcel of water resides in the wetland. During high flow events RT may be too short to provide the desired treatment. For example, Raisin et al. (1997) reported as much as 55% of N reduction in surface runoff during low flow events but less than 5% reduction during high flow events, and attribute the difference to RT. Designing the CTW to provide the desired treatment during high flow events is important because a significant percentage of the annual pollutant load occurs during these events. For example, Reinhardt et al. (2005) reported that 43% of annual SRP load in subsurface drainage occurred during only a few high flow events.

Braskerud et al. (2005) and Carleton et al. (2001) summarized studies on wetlands receiving dynamic pollutant loading. The wetlands examined included constructed, restored, or natural wetlands receiving field runoff, urban stormwater, or diverted river water. Table 2.2 summarizes selected case studies on wetlands receiving agricultural subsurface drainage water. The studies summarized by Braskerud et al. (2005), Carleton et al. (2001), and Table 2.2 show a wide range of N reduction and P retention. No information on pathogen reduction was presented in any of the studies. The range in treatment between studies may be attributed to differences in design specifications, meteorological conditions, and flow. It also suggests that design methods and specifications need to be refined so that the desired treatment can consistently be achieved. Kovacic et al. (2000; 2006) recommended developing a framework for selecting where to implement CTWs receiving drainage water and utilizing the potential for additional treatment when CTWs are used in combination with other technologies, such as in-situ bioreactors (Dinnes et al., 2002; Jaynes et al., 2008), riparian buffers (Dinnes et al., 2002), in-stream reservoirs (Gannon et al., 2005), and wet detention ponds (Mallin et al., 2002; Murphy et al. 2010) to increase treatment. Another benefit of CTWs receiving drainage water is capability to reduce flooding (Knight, 1992).

**Table 2.2** Summary of selected case studies on wetlands receiving agricultural subsurface drainage water, including nitrate-nitrogen ($NO_3^-$-N) and total phosphorus (TP) treatment.

| **Location**† | **Wetland Area (ha)** | **Wetland Area to Drainage Area (%)** | **$NO_3^-$-N Mass Reduction (%)** | **TP Mass Retention (%)** | **Annual Precipitation (mm)** |
|---|---|---|---|---|---|
| Kleine Aa River, Switzerland[A] | 0.24 | 1 | NA | 23 | 1250 |
| Lake Bloomington, Illinois (2)[B] | 0.16 - 0.40 | 3 – 4 | 31 – 42 | 53 | 957 – 1038 |
| Embarras River, Illinois (3)[C] | 0.30 - 0.80 | 3 – 6 | 34 – 45 | -53 | 790 – 991 |
| Kent Island, Maryland[D] | 1.3 | 9 | 52 | 27 | 1090 – 1150 |
| Indian Lake, Ohio[E] | 1.2 | 7 | 40 | 59 | 407 – 841 |
| Kiwitahi, New Zealand[F] | 0.03 | 1 | 33 | -76 | 854 – 1004 |
| Gundowring, Australia[G] | 0.045 | <1 | 11‡ | 17 | 569 – 1030 |

[A]Reinhardt et al. (2002); [B]Kovacic et al. (2006); [C]Kovacic et al. (2000); [D]Jordan et al. (2003); [E]Fink and Mitsch (2004); [F]Tanner et al. (2005); [G]Raisin et al. (1997)
† Number of wetlands shown in brackets
‡ Reported as total nitrogen

### 2.5.2 Climate Considerations

Cold climate conditions, such as those experienced in Nova Scotia, can have significant effects on the performance of CTWs receiving drainage water and must be considered in their design and operation. A seasonal slow-down in treatment can be expected, as temperature affects several biogeochemical processes (Kadlec and Reddy, 2001; Wood et al., 1999). Sub-zero temperatures can also cause operational difficulties if flow is obstructed by ice. The amount and distribution of precipitation govern flow rates (Section 2.5.1) and affect drainage water quality (Section 2.2), and thus treatment.

The climate of Nova Scotia is classified as a cold, continental, fully humid, warm summer climate (McKnight and Hess, 2000). Daily mean temperatures reach as low as -7 ºC in Jan and as high as 19 ºC in Jul (Davis and Browne, 1996; Environment Canada, 2007). Nova Scotia receives a large range in annual precipitation, depending on the region, with less than 1000 mm to more than 1600 mm (Davis and Browne, 1996; Environment Canada, 2007). High flow events are most likely to occur during late fall and early winter when precipitation is greatest (Environment Canada, 2007). Annually, there are 8 d of precipitation > 25 mm (Environment Canada, 2008). There is a high ratio of mean annual runoff to mean annual precipitation (Mitsch and Gosselink, 2007), which suggests that CTWs receiving drainage water have higher flow rates than CTWs in most regions for a given amount of precipitation. High flow rates may also be observed from spring snowmelt events.

A few CTWs have been investigated in Nova Scotia (Rochon et al., 1999; Smith et al., 2006; Wood et al., 2008). These studies indicate potential for year-round operation of CTWs. To date however, no CTWs receiving drainage water have been investigated in the province. Studies conducted in cold climates such as Switzerland, Illinois, and Ohio provide some information on the viability of CTWs receiving drainage water in cold climates (Table 2.2). However, Nova Scotia is colder, receives more precipitation, has a different distribution of precipitation, and has a greater ratio of mean annual runoff to mean annual precipitation than these climates. The effect these differences have on hydraulics and water quality need to be assessed to adapt CTWs receiving drainage water to Nova Scotia.

## 2.6 WATER AVAILABILITY IN NOVA SCOTIA

The agricultural industry is Canada's largest consumer of water, accounting for approximately 70% of water withdrawals. The availability of good quality water is a primary concern of maintaining a productive and sustainable industry (Coote and Gregorich, 2000; Environment Canada, 2004b). Two of the biggest threats to water availability are climate change and pollution (De Kimpe, 2002).

Nova Scotia has an abundance of groundwater, lakes, and rivers from which to draw water and the amount of precipitation it receives should be more than sufficient to meet crop water demands, and compares favourably to the Canadian prairies, where annual precipitation can be as low as 300 mm. However, Nova Scotia has endured droughts in recent years because there has been a deficit of precipitation during the growing season and a surplus during the non-growing season (Environment Canada, 2007). Crop insurance reports document these events: "Record temperatures and extended periods of drought reduced crop yield for most commodities." (CNSCI, 2002). Periods of water deficit in Atlantic Canada are projected to become more frequent and severe due to climate change (Vasseur and Catto, 2008) and increased water demands from population and industrial growth (NSDEL, 2002). To compensate for these deficits and to ensure high yields more farmers will implement irrigation (Environment Canada, 2004b).

### 2.7 Wastewater Treatment and Reuse Systems Receiving Agricultural Drainage Water

Environment Canada (2004b) and the United States Climate Change Science Program (Baron et al., 2008) recommend offsetting decreased water availability by increasing agricultural wastewater reuse. Drainage water has good potential to be reused because of its abundance and relatively low biological oxygen demand compared to other wastewaters, such as liquid manure or milkhouse washwater. The general approach to reusing wastewater is to combine various water management strategies and systems into a larger integrated system. Studies that examine wastewater reuse for agriculture include an investigation into challenges relating to the reuse of saline drainage water for irrigation in California (Oster and Grattan, 2002), a simulation model of a system that captures drainage water using a series of reservoirs and reuses it for irrigation, fish harvesting, and salt harvesting in India (Singh and Kumar, 1998), and an economic analysis of a system comprised of improved irrigation practices, irrigation of salt-tolerant plants with drainage water, and on-farm disposal of drainage water using a solar evaporator (Wichelns, 2005).

Wastewater treatment and reuse systems are primarily used in arid regions, and are not widely used in Canada. Approximately 65 projects use treated municipal effluent as a source of water for irrigating agricultural crops, trees, or golf courses in the Prairie Provinces (Coote and Gregorich, 2000). However, the implementation of wastewater treatment and reuse systems in Canada is projected to grow because of increasing threats to water availability from pollution, climate change, and increased water demands (Exall et al., 2006).

An innovative type of wastewater treatment and reuse system receiving agricultural drainage water is a WRIS. Wetland-reservoir irrigation systems capture surface runoff and/or tile drainage water, use a CTW to improve water quality, and store the treated water in a reservoir. The water can then be used for irrigation, upon which the cycle of drainage, capture, and treatment may continue. Wetland-reservoir irrigation systems are attractive because they have the dual benefit of mitigating water pollution from drainage water and offsetting decreased water availability.

Limited studies have investigated WRISs. Allred et al. (2003) established wetland-reservoir subirrigation systems at three sites in Ohio. Water quality was only monitored during selected storm events. One site reported mean mass reductions of total filterable solids, $NO_3^-$-N, and organic carbon of 74, 51, and 62%, respectively (Baker et al., 2004). Further sampling and analysis was recommended to explain discrepancies of ammonia and P reductions between sites. Corn and soybean yields, respectively, increased 35 and 38% during drier growing seasons, 14 and 10% during near average to wetter growing seasons, and 20 and 17% overall compared to non-irrigated crops (Allred et al., 2003). Tan et al. (2007) investigated a system comprised of a tile drainage/subsurface irrigation system and a merged wetland-reservoir in Ontario. Reductions of $NO_3^-$-N, dissolved inorganic P, dissolved organic P, and total dissolved P of 41, 18, 47, and 36%, respectively, were reported. Corn and soybean yield increased 91 and 41%, respectively, during dry growing seasons compared to non-irrigated crops.

Wetland-reservoir irrigation systems have not been implemented in Nova Scotia. For WRISs to be effective and efficient their design, construction, and operation need to be assessed in and adapted to local environmental conditions. This includes assessing CTW hydraulics and water quality, reservoir water quality and availability, and system costs. Design manuals exist for the individual components of WRISs but little information is available on integrating them into a unified system.

# CHAPTER 3 - DESIGN, CONSTRUCTION, AND OPERATION OF A WETLAND–RESERVOIR WASTEWATER TREATMENT AND REUSE SYSTEM RECEIVING AGRICULTURAL DRAINAGE WATER IN NOVA SCOTIA

## 3.1 INTRODUCTION

A variety of environmental factors influence tile drainage water quality and flow, treatment in CTWs, and general construction considerations. Therefore, local environmental conditions must be considered when establishing a WRIS. In Nova Scotia, the effects of cold conditions and high precipitation on water availability and CTW performance are a concern. It is hypothesized that a relatively large CTW will be required, as a seasonal slow-down in treatment is expected. It is also hypothesized that a relatively large reservoir will be required, as a relatively greater amount of precipitation falls in Nova Scotia.

The documentation of the design, construction, and operation of a WRIS in Nova Scotia will serve as a case study that can be consulted when implementing future WRISs in the region. The entire process will also provide practical experience to engineers and contractors involved in the project. The site will serve as a demonstration site where farmers, industry, and government can observe the technology in practice.

The objective discussed in this chapter is to design, construct, and operate a wastewater treatment and reuse system consisting of a tile drainage system, a CTW, and an irrigation reservoir. Design, construction, and operational challenges are identified and recommendations are made to prevent or address the challenges in future systems. Once a WRIS is constructed in Nova Scotia it can be assessed so that the process of adapting it to local environmental conditions may continue. Recommendations for adapting the design and operation of WRISs to the climate of Nova Scotia are made in Chapter 4 and 5, based on hydraulic and water quality assessments, respectively.

## 3.2 SITE DESCRIPTION

Wetland-reservoir irrigation systems may be most effective at mitigating water pollution from agricultural drainage when implemented as part of an integrated watershed approach that uses various best management practices to cumulatively improve water quality (Environment Canada, 2004b). More specifically, CTWs (and therefore WRISs) may be most effective when located at the heads of catchment areas because of their cumulative effect of storing and retarding drainage water, which may increase RT and thereby treatment in downstream natural wetlands (Raisin et al., 1997). Other factors that may govern the most suitable location for WRISs include the sensitivity of receiving water bodies to water pollution, drainage water quality, existing system components, and the drought frequency and severity.

The study site is located at the Bio-Environmental Engineering Centre (BEEC) in Bible Hill, Nova Scotia, Canada (N 45E 23' and W 63E 15') as shown in Fig 3.1. Bible Hill has a daily mean temperature of -7 ºC in Jan and 19 ºC in Jul, and receives 1170 mm of precipitation annually, with peak amounts during the fall (Environment Canada, 2008). The site (Fig 3.2) consists of a 5.0 ha agricultural field, an existing tile drainage system, and a pasture and forested gully directly south of the field. The predominant soils in the field are of the Pugwash (Gleyed and Orthic Melanic Brunisol) and Debert (Gleyed Melanic Brunisol) texture classes (Webb and Langille, 1996). Pugwash soils are moderately to well drained and Debert soils are imperfectly drained. Additional information on these soils can be found in CASCC (1998), Webb et al. (1991), and Webb and Langille (1996). The field has less than a 3% slope.

The tile drainage system that supplies the WRIS is underneath 1.8 ha of the field. It is comprised of 100 mm diameter tile lines installed at a depth of 80 cm and spaced every 12 m (Fig 3.2) (Lamond, 2005; Thiagarajan, 2005). Seven lines converge at monitoring Hut I where water samples and flow measurements were collected.

Cultural practices have been consistent since 2002. Both conventional and no tillage practices, covering 43 and 57% of the drainage area, respectively, were used.

Conventional tillage consisted of annual fall tillage to a depth of 25-30 cm using a moldboard plow followed by secondary tillage using disk harrow and spring cultivation to 10 cm using a disk harrow. The no till plots consisted of direct seeding using a Tye™ no tillage seeder. Liquid dairy manure was applied annually in the spring by a vacuum tanker. Manure was immediately incorporated into the conventional tillage plots and left on the surface of the no till plots. A history of manure applications are presented in Table 3.1. A three-year rotation of barley-soybean-spring wheat was maintained (Table 3.1).

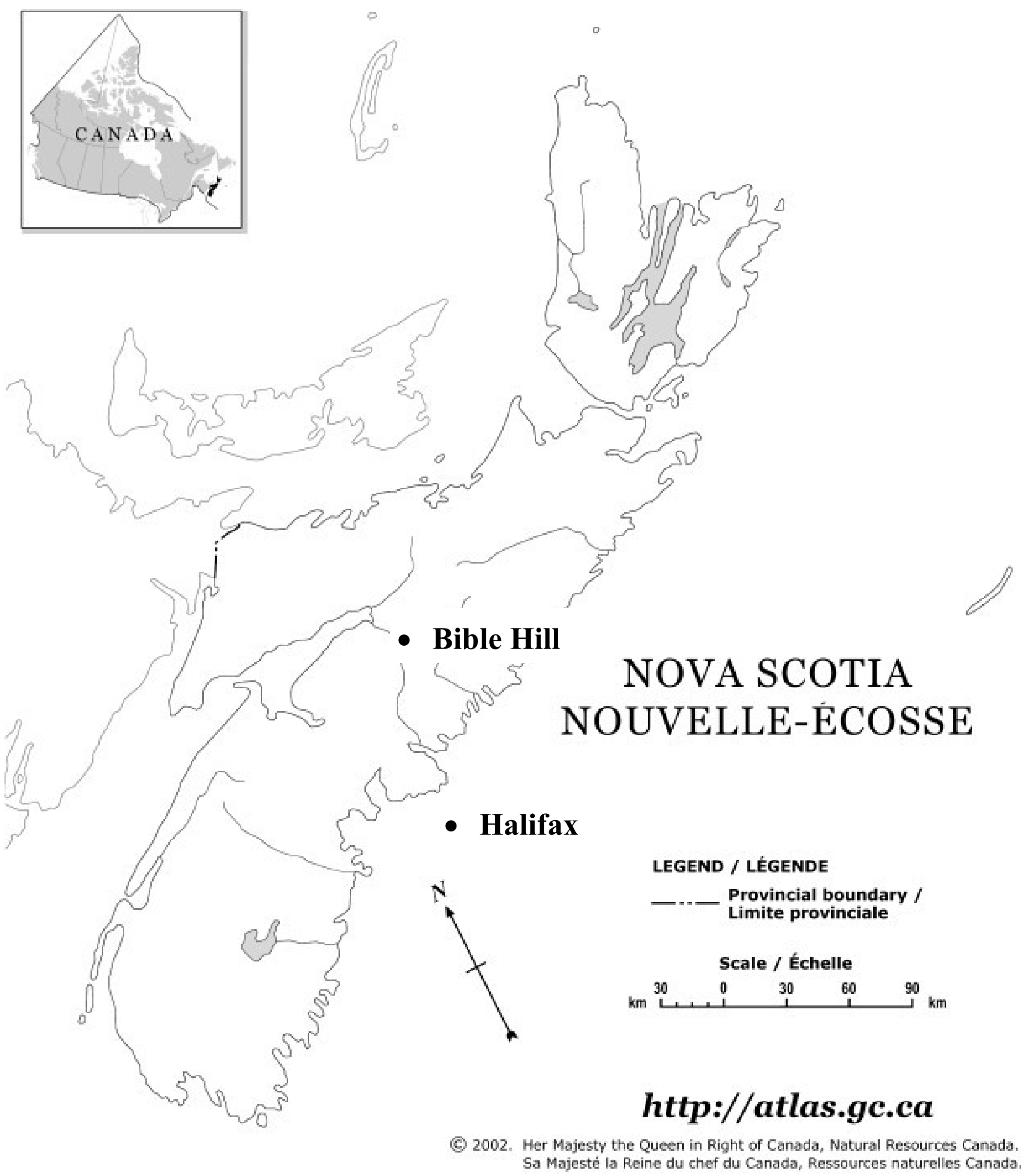


**Figure 3.1** Map of Nova Scotia, Canada.

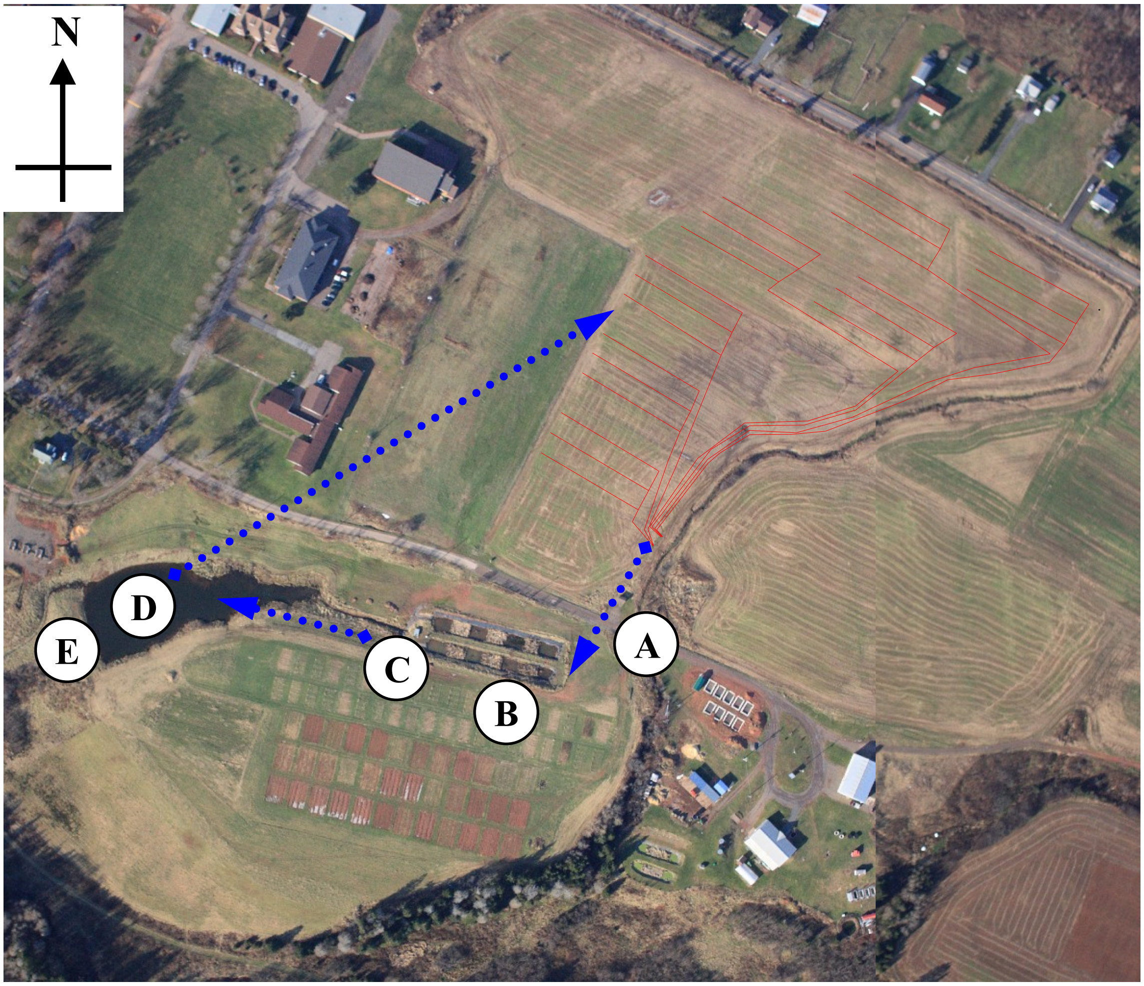


**Figure 3.2** Plan view of the wetland – reservoir irrigation system (1:3500). Labelled are Hut I (A), constructed treatment wetlands (B), Hut II (C), reservoir (D), and dam (E). Tile drainage lines are shown in red. Blue arrows illustrate the cycle of flow between components.

**Table 3.1** Crop and manure application details for the tile drained field from 2001 to 2008.

| Year | Crop | Manure Type | Manure Rate (t ha$^{-1}$) |
|---|---|---|---|
| 2008 | Barley | Liquid dairy | 70 |
| 2007 | Soybeans | Liquid dairy | 20 |
| 2006 | Spring wheat | Liquid dairy | 85 |
| 2005 | Barley | Liquid dairy | 65 |
| 2004 | Soybeans | Liquid dairy | 25 |
| 2003 | Spring wheat | Liquid dairy | 25 |
| 2002 | Barley | Liquid dairy | 40 |
| 2001 | Soybeans | Solid manure | 23 |

## 3.3 Constructed Treatment Wetland Design and Construction

The function of the CTW within the WRIS is to improve drainage water quality so that it can be subsequently used for irrigation or safely discharged into the environment. Constructed treatment wetlands also improve aesthetics, increase biodiversity, and reduce flooding.

### 3.3.1 Site Investigation

The location of CTW is a pasture directly south of the tile drainage system (Fig 3.2). Limited land was available, however this location provided some important benefits; it was in close proximity to the tile drainage system, situated between the tile drainage system and the reservoir, had adequate grade to achieve gravitational flow, and the land was not in agricultural production. Gravitational flow was desired so that the cost and operation and maintenance requirements of pumps, as used by Allred et al. (2003), would not be required to transport water between the tile drainage outlet, CTW, and reservoir.

The area was surveyed to help determine the allowable size, shape, and position of the CTW, and to help estimate work quantities. A mapping review of the site revealed a small catchment area and the possibility of diverting surface runoff away from the CTW to facilitate easier monitoring of water quality and inflow rates.

Test pits and auger holes revealed a shallow water table, between 0.5 m and 1.5 m deep depending on ground elevation. Once excavated to grade, the CTW floor would be less than 0.5 m above the water table, which is considered unsuitable for sewage lagoons (Webb et al., 1991). The shallow water table presented a concern because of the potential for movement between groundwater and the wastewater residing in the CTW. In this case, wastewater could pollute groundwater and challenges to construction, treatment and monitoring may occur.

Another important aspect of the site investigation was the collection of historical drainage flow and water quality data, which were used to size the CTW. Using data from the site allows the design to be adapted to the local environmental conditions, yielding a more efficient design than if standard values were used because there can be significant variability between sites (Section 2.2).

### 3.3.2 Design

A free water surface wetland was selected because they have been successfully implemented in Nova Scotia (Rochon et al., 1999; Smith et al., 2006; Wood et al., 2008) and are less expensive than subsurface flow wetlands (Kadlec and Knight, 1996).

Kadlec and Knight (1996) described steady state, first-order plug flow models, such as the k-C* model (eqn 1), which have been shown to adequately describe treatment in CTWs receiving dynamic pollutant loading, including drainage water (Carleton et al., 2001; Wong and Geiger, 1997) as:

$$A = -\frac{Q}{k} * \ln\left(\frac{C_{out} - C^*}{C_{in} - C^*}\right) \qquad [1]$$

Where:

$A$ = Wetland surface area ($m^2$)

$Q$ = Annual inflow ($m^3\ y^{-1}$)

$k$ = First-order areal uptake rate constant ($m\ y^{-1}$)

$C_{out}$ = Concentration at the constructed treatment wetland outlet ($mg\ L^{-1}$ or $CFU\ 100\ mL^{-1}$)

$C_{in}$ = Concentration at the constructed treatment wetland inlet ($mg\ L^{-1}$ or $CFU\ 100\ mL^{-1}$)

$C^*$ = Background concentration ($mg\ L^{-1}$ or $CFU\ 100\ mL^{-1}$)

It is important to consider however, this is only the case for long-term treatments. Short-term treatments, such as during high flow events, may not accurately follow this model (Carleton et al., 2001; Wong and Geiger, 1997).

The proper sizing of a CTW is essential to minimizing capital costs and to ensure that agricultural land is not unnecessarily taken out of production. Wong and Geiger (1997) adapted the k-C* model to dynamic flow rates by incorporating hydrologic effectiveness curves. The percentage of runoff that resides in the CTW for the target RT can be specified using these curves. For a CTW as part of WRIS, hydrologic effectiveness curves need to be adapted to include tile drainage water. None were available for the present site, however, historical flow and water quality data from the tile drainage system were available, enabling the design to be localized. Historical drainage flow and water quality data are typically absent, therefore standard values, which may result in a less efficient design, or the use of a pilot scale system, which may not be cost effective, are used. Kadlec and Knight (1996) recommend applying the k-C* model seasonally for CTWs treating nutrients to account for dynamic concentrations at the CTW inlet ($C_{in}$s). This also applies to the annual inflow ($Q$) for CTWs receiving drainage water, because of dynamic flow rates.

The present study applied the k-C* model using tile drainage flow and water quality data from 2005. These data allowed the CTW to initially be sized to treat the peak loads. The coinciding flow-weighted average $C_{in}$ (for $NO_3^-$-N, TP, and *E. coli*) and $Q$ (extrapolated to annual flow) from the entire field that produced the peak load were used. Only tile drainage water is treated by the CTW as surface runoff around the CTW was diverted away using berms and ditches. Seepage and groundwater intrusion were assumed to be negligible because a synthetic liner was used. The outflow concentrations ($C_{out}$) were based on the drinking water quality guideline for $NO_3^-$-N (10 mg $L^{-1}$), a 70% reduction for TP, and the irrigation water quality guideline for *E. coli* (100 CFU 100 $mL^{-1}$). The background concentrations ($C*$) were specified by Kadlec and Knight (1996); 0 mg $L^{-1}$ for $NO_3^-$-N, 0.01 mg $L^{-1}$ for TP, and 45 CFU 100 $mL^{-1}$ for *E. coli* (specified for FC). Carleton et al. (2001) showed that first-order areal uptake rate constants ($k$s) for CTWs receiving dynamic pollutant loading were similar to those reported in CTWs receiving steady state loading. First-order areal uptake rate constants have been shown to be dependant on temperature and should be adjusted to site specific temperatures (Kadlec and Knight, 1996; Wood et al., 1999). The $k$s used were the mean monthly $k$s adjusted for dilution or concentration effects generated by Jamieson et al. (2007) from a CTW receiving livestock wastewater in Nova Scotia. Jamieson et al. (2007) reported lower *k*s than Kadlec and Knight (1996), resulting in larger wetland areas (*A*s). This method yielded a maximum *A* of 24,370 $m^2$, which was considered too large to be economically feasible, did not fit within the available land, and raised a concern of the CTW drying up during low flow periods.

To generate a feasible *A* the k-C* model was applied in the same manner except the monthly mean *Q* (extrapolated to annual flow) during the month when the peak load occurred and the maximum, non-adjusted *k*s generated by Jamieson et al. (2007) were used (Table 3.2). Less treatment during periods of high loading was therefore expected. This method yielded a maximum *A* of 1,027 $m^2$, which was used as the design *A*. In general, a minimum RT of 6 d is recommended (NRCS, 2002). The mean nominal residence time ($t_n$) of the CTW is 9 d, based on the annual tile drainage flow volume from 2005. Monthly $t_n$s ranged from 4 to 95 d, based on monthly flow volumes from 2005.

**Table 3.2** k-C* model variables for nitrate-nitrogen ($NO_3^-$-N), total phosphorus (TP), and *Escherichia coli* (*E. coli*) during the month that yielded the largest constructed treatment wetland area.

| Variable | October 2005 $NO_3^-$-N | March 2005 TP | October 2005 *E. coli* |
|---|---|---|---|
| $Q$ ($m^3$ $y^{-1}$) | 16,990 | 6,240 | 16,990 |
| $k$ (m $y^{-1}$) | 12.0 | 38.4 | 54.5 |
| $C_{in}$ (mg $L^{-1}$) | 12.8 | 0.25 | 1503† |
| $C_{out}$ (mg $L^{-1}$) | 9.0 | 0.06 | 100† |
| $C^*$ (mg $L^{-1}$) | 0.0 | 0.01 | 45† |
| $A$ ($m^2$) | 499 | 247 | 1027 |

† Expressed as CFU 100 $mL^{-1}$

An alternate method of sizing a CTW receiving drainage water is to use a specific ratio of the wetland surface area ($A_W$) to the drainage surface area ($A_D$) (Carleton et al., 2001). Dinnes et al. (2002) concluded that sizing the CTW using an $A_W$ to $A_D$ ratio may not provide the desired RT, and thereby treatment, during high flow events. Rather, the $A_W$ to $A_D$ ratio was used in this design to verify the validity of the results from the k-C* model. Kovacic et al. (2000) recommend an $A_W$ to $A_D$ ratio of 4 to 7% for optimum $NO_3^-$-N reduction. Reinhardt et al. (2005) recommend an $A_W$ to $A_D$ ratio of 4% for 50% SRP retention. The $A_W$ to $A_D$ ratio of this design is 5.7%, indicating that the results of the k-C* model are valid. Another alternate sizing method is to use the drainage volume resulting from a specific storm event as the CTW volume. Allred et al. (2003) used the runoff and subsurface drainage volume from a storm with a 1 in 2 year return period and 24 h duration. These methods may be an over simplification and are not firmly established because of limited data.

The configuration of the CTW consists of two, identical, side-by-side, and independent CTWs, Wetland 1 (W1) and Wetland 2 (W2) (Fig 3.3). Wetland 1 and W2 are each 512 $m^2$ (half the design *A*) and received the tile drainage outflow split equally between them. This configuration permits data replication or comparison and facilitates repairs or modifications to W1 or W2. Specifications were based on several design manuals (NRCS, 2002; Schueler, 1992; USEPA, 1988) and are listed in Table 3.3.

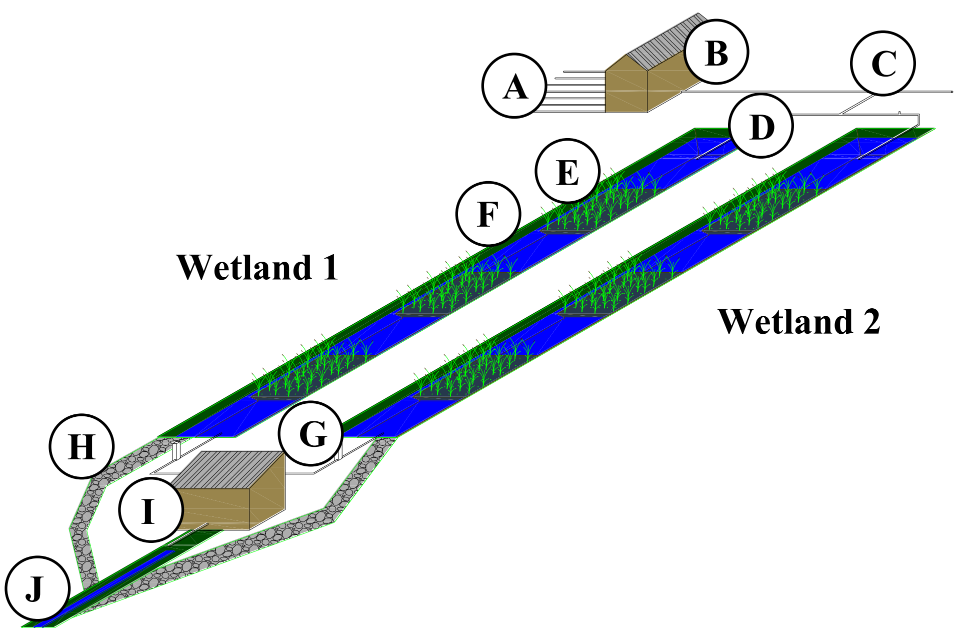


**Figure 3.3** Constructed treatment wetland diagram. Labelled are the seven tile drain outlets (A), Hut I (B), diversion control structure (C), Wetland 1 sampling port (D), shallow zone (E), deep zone (F), wetland 2 outlet control structure (G), Wetland 1 emergency spillway (H), Hut II (I), and surface channel (J).

The shape of the CTW is important because it can affect plug-flow hydraulics, and thereby RT and treatment (Kadlec and Knight, 1996). A high length to width ratio is recommended to encourage plug-flow hydraulics, while balancing construction costs (USEPA, 1988). Steiner and Freeman (1989) recommend a length to width ratio of 10:1, which was used for W1 and W2. A shallow marsh type of design, with alternating shallow and deep zones, was selected to ensure wastewater resides in both aerobic and anaerobic environments during high flow periods (Schueler, 1992). The first zone is a 1 m deep forebay (35% of total CTW volume) that allows sediment accumulation (Schueler, 1992) and may increase RT. There is 0.6 m of freeboard that can increase water holding and delay overtopping if outflow is restricted. The water levels in W1 and W2 can be set by adjusting plates in an in-line water level control structure (Agri-Drain, Adair, IA) (Fig 3.4) at their outlets.

**Table 3.3** Individual constructed treatment wetland specifications.

| | |
|---|---|
| Area ($m^2$) | 512 |
| Wetland area to drained area ratio (%) | 5.7 |
| Volume at standard operating depth ($m^3$) | 185 |
| Length to width ratio | 10:1 |
| Length (m) | 72 |
| Width (m) | 7 |
| Forebay standard operating water depth (m) | 1 |
| Deep zone standard operating water depth (m) | 0.60 |
| Shallow zone standard operating water depth (m) | 0.15 |
| Soil depth (m) | 0.45 |
| Freeboard (m) | 0.60 |
| Number of deep zones | 3 + forebay |
| Number of shallow zones | 3 |
| Length of zones (m) | 10 |
| Percent of wetland area occupied by shallow zones (%) | 43 |
| Bank slopes | 2:1 |
| Primary vegetation | Cattail (*typha spp.*) |
| Inlet | 150 mm PVC |
| Outlet | 150 mm in-line control structure |
| Emergency outlet | Rock spillway |
| Liner | 12 mil woven polyethylene |

**Figure 3.4** In-line control structure installation at the outlet of Wetland 2.

To address the concerns presented by a shallow water table a 12 mil woven polyethylene liner was installed. The CTW floors are sloped 0.2% and two 50 mm perforated pipes, wrapped in filter fabric, were laid in the last two zones of W1 and W2 to drain any water trapped beneath the liner. The disadvantages of using a liner are cost and not benefitting from potential treatment provided by seepage (Larson et al., 2000).

Pipe sizes were based on peak tile drainage flow rates recorded between 2003 and 2005. Tile drainage water is transported to the CTW by a 200 mm underground pipe. A 200 mm in-line control structure between Hut I and the CTW allows water to be diverted away from the CTW and into a ditch. A vertical plastic fin glued inside a T splits flow equally between W1 and W2. Knife valves after the T allow flow to be shut-off to W1 or W2, thereby facilitating repairs or modification. The primary outlet for each CTW is a 150 mm in-line control structures. An emergency spillway at the each outlet is used to safely discharge excess water if the control structure cannot. Water flows out of the in-line control structures and into Hut II, where water samples and flow measurements were collected. Water is then directed to the reservoir by a 1 m wide, 1 m deep, 30 m long surface channel, which may provide additional treatment.

### 3.3.3 Construction

Construction began in Nov 2006 and took ≈ 14 d. The timing of construction was a challenge because it occurred too late for vegetation establishment, leaving the site susceptible to erosion. Erosion control measures were used, namely a temporary sedimentation pond, silt screens, straw bales, and straw mulch (NSDE, 1988). Significant damage from erosion still occurred over the winter, therefore it is recommended that construction occur in late spring or early summer to permit vegetation establishment. Engineering drawings, staking, and on-site problem-solving were important methods of communicating design specifications to the contractor.

The site was cleared of vegetation and the CTW was excavated to grade. The liner was manually installed but utilizing machinery to position the liner is recommended. Standing water and lack of liner slack posed a challenge to inlet and outlet installation that could have been avoided if the inlets and outlet were installed before the liner. Inlet and outlet inverts were set 15 cm above the CTW floor to prevent them from being blocked by sediment accumulation. Sediment accumulation was observed in the forebays after one year of operation, therefore an inlet riser pipes is recommended. Shallow zones were backfilled with a soil containing a high organic matter content because increased substrate organic matter has been shown to increase $NO_3^-$-N reduction (Burchell et al., 2007). Fig 3.5 shows the CTW after the shallow zones were backfilled in Nov 2006, and after one year of operation in Sept 2008.

Cattail (*typha spp.*) shoots were transplanted at a spacing of 1 $m^{-2}$ (Kadlec and Knight, 1996) into the shallow zones from a natural wetland in May 2007 (Fig 3.6A) and grew to cover the entire shallow zone by Aug 2007 (Fig 3.6B). Cattails were selected because they are indigenous, hardy, readily transplanted, and have been used successfully in other CTWs in Nova Scotia (Rochon et al., 1999; Smith et al., 2006; Wood et al., 2008). Other species were established naturally, however cattails were dominant. Luckeydoo et al. (2002) monitored natural succession on a similar system and found that vegetation was established quickly but the number of wetland species was low. Areas susceptible to erosion, such as berms and ditches, were seeded with a mixture of clover, red fescue and timothy. Remaining areas were covered with vegetation through natural succession.


**Figure 3.5** (A) Constructed treatment wetlands during construction in November 2006 and (B) in September 2008.


**Figure 3.6** (A) Newly transplanted cattails in May 2007 and (B) Cattail growth through August 2007.

### 3.4 Reservoir and Dam Design and Construction

The function of the reservoir as part of the WRIS is to store the water treated by the CTW until it is used for irrigation. Reservoirs have also been shown to provide additional treatment (Gannon et al., 2005; Mallin et al., 2002; Murphy et al., 2010), which is important if the CTW does not provide the desired treatment.

### 3.4.1 Site Investigation

The location of the reservoir was a forested gully south of the tile drainage system and downstream of the CTW (Fig 3.2), thereby allowing gravitational flow. An on-stream reservoir, which is formed by constructing a dam across the downstream end of a gully, was selected rather than a dugout reservoir primarily because of land availability and cost (NRCS, 1997a). The proximity of the gully to the field is close enough to allow a feasible length of irrigation pipe.

A good understanding of site geology is an essential part of designing and constructing safe and effective dams and reservoirs (Kutzner, 1997; NRCS, 1997a; Schwab et al., 1993). A field investigation consisting of test pits, permeability tests, and a review of soil maps was conducted to assess seepage potential of the reservoir, the suitability of site soil for use as dam fill, and to help estimate work quantities for the construction tender (Webster, 2006). Test pits dug along gully slopes revealed a shallow water table at most locations, which may act as a recharge source. In-situe and Guelph permeability (Soil Moisture Corp., Santa Barbara, CA) tests indicated that the site is characterized as slowly permeable, which raised a concern about water retention by the reservoir (Webster, 2006). Pollution of groundwater by reservoir water was also a concern if the CTW did not provide the desired treatment. Webb et al. (1991) classify the soil as the Woodville (Gleyed Humo-Ferric Podzols) texture class. The soil profile was fairly uniform but some layers of fractured siltstone and clay loam were observed. The fractured siltstone was a concern because it may impede compaction when used as dam fill, however, the compactor was able to crush this material. The clay loam material was a source of impervious material that was used for constructing the dam core and key. A dam foundation material that is impervious, has a low compressibility, and is capable of bearing the heavy dam load is required for dam stability and to avoid seepage losses underneath the dam (Kutzner, 1997, NCRS, 1997a). Test pits dug at the dam foundation site revealed a sandstone layer below the topsoil, capable of bearing the dam load (Webster, 2006).

A laboratory investigation consisting of a particle size analysis, Atterberg limit test, direct shear test, and a standard Proctor density test was conducted to assess dam stability. Soil with a high clay content is recommended for reservoir sites and to be used as dam fill because it is relatively impervious (Cummings, 1999; NRCS, 1997a). Soil samples from test pits were mostly classified as silt loam (USDA classification system) or low plasticity silt (ML) (USCS and AASHO classification systems), and therefore somewhat lacking in clay content. The Atterberg limit test indicated that the soil becomes plastic and liquid at relatively low moisture contents, and therefore careful attention must be paid to keep soil dry during construction. A standard proctor compaction test yielded an optimum dry density of 1.85 g $cm^{-3}$ at an optimum moisture content of 15.5%, which was used to verify proper compaction during construction. These tests indicate that the site material is not ideal as a reservoir site or as dam fill but may be acceptable if proper design specifications and construction practices are followed.

### 3.4.2 Reservoir Design

The main challenge encountered during reservoir design was to determine the volume of water available in Nova Scotia's climate and to balance the reservoir's capacity with limited land availability and cost.

Reservoir volume should be based on water demand, application efficiency, losses, and inputs (NRCS, 1997a). Schwab et al. (1993) recommend allowing as high as 60% of the total volume for seepage and evaporation losses, and non-usable storage. Allred et al. (2003) based the volume of the reservoirs in their WRISs on the subirrigation requirements of crops in 8 out of 10 y using DRAINMOD (Skaggs, 1978), however, this method yielded reservoirs that were not economically feasible. Surface runoff is typically the primary input to reservoirs. However, it was not a major input to this reservoir because most surface runoff was diverted from the WRIS by berms and ditches. For reservoirs that are part of WRISs, the major input is CTW outflow, which was estimated from tile drainage outflow. A mean annual outflow of 6,000 $m^3$ was recorded from the tile drainage system between 2003 and 2005, when the mean annual precipitation was 1,157 mm. Annual precipitation was estimated to supply ≈ 3,500 $m^3$, based on the 0.3 ha allocated to the reservoir. A monthly water budget projected a maximum potential reservoir volume of 8,500 $m^3$. Water availability could be augmented by incorporating surface runoff, although it may present treatment challenges. A contour survey of the gully was conducted and reservoir volumes were calculated for various dam locations. A capacity of 5,000 $m^3$ was calculated when the dam was situated at the downstream limit of the gully. The reservoir has a surface area of 0.3 ha and a maximum depth of 4.5 m, deep enough to provide fish habitat and limit aquatic plant growth (NRCS, 1997a). This volume is less than the potential volume of 8,500 $m^3$ because it was limited by land availability, topography, and cost, as experience by Allred et al. (2003).

### 3.4.3 Dam Design

The main challenge encountered during the dam design was a dam stability concern created by the use of less than ideal fill material. Design features that addressed this challenge include an impervious upstream layer, a pea stone drain, and toe drains. Dam specifications are listed in Table 3.4 and were based on earthen dam design books (Kutzner, 1997; Schwab et al., 1993).

The impervious upstream layer prevents seepage into the dam It consists of bentonite, a highly expandable clay, incorporated into the upstream dam surface and covered with soil (NRCS, 1997a). Bentonite was applied at a rate of 12 kg $m^{-2}$, as determined from a pilot scale study.

The pea stone and toe drains remove water that has seeped into the dam, thereby preventing flow paths and soil saturation that threaten stability. The pea stone drain is a 0.6 m by 0.6 m trench inside the dam and runs across the dam width. The trench contains a 15 cm perforated pipe, covered by washed pea stone, all of which is wrapped in a geotextile filter. The two toe drains are 15 cm perforated pipes, buried 3 m in from the downstream toe. Three, galvanized steel drain outlets discharge water into an outlet ditch at the center of the downstream toe. Drain outlet inverts should be set to account for settling and sedimentation, as observed during the present study.

The spillway removes excess water from the reservoir, thereby preventing dam overtopping and failure. Flow rates used to size the spillway were based on a storm with a 1 in 50 y return period and 24 h duration, which is a more conservative approach than that recommendation by the NRCS (1997a). Surface runoff from the entire catchment area was included as part of the peak flow in case the CTW berms failed or ditches overtopped, as observed during a major storm event in Sept 2008. The spillway control section is 4 m wide and 6 m long. It is lined with a geotextile and covered by rock < 0.2 m wide.

**Table 3.4** Specifications for the dam that formed the on-stream reservoir component of the Wetland-Reservoir Irrigation System.

| | |
|---|---|
| Top width (m) | 5 |
| Freeboard (m) | 1.2 |
| Upstream slope | 3:1 |
| Downstream slope | 3:1 |
| Keyway depth (m) | 0.6 |
| Maximum lift thickness (m) | 0.2 |
| Compaction | 100% Standard Proctor |
| Optimum dry density (kg $m^{-3}$) | 1850 |
| Optimum water content (%) | 15.5 |
| Seepage controls | Bentonite upstream layer, pea stone drain, toe drains |
| Spillway | < 0.2 m rock, 4 m wide control section |

### 3.4.4 Construction

The reservoir site was cleared of trees in Nov 2006 but dam construction was postponed until Jun 2007 when weather and soil conditions improved, so that proper compaction could be achieved. Erosion control practices were used, namely a temporary sedimentation pond, silt screens, straw bales, and straw mulch (NSDE, 1988), but the best practice would have been for site preparation and dam construction timing to coincide. Dam construction took ≈ 10 d. Engineering drawings, staking, and on-site problem-solving were important methods of communicating design specifications to the contractor.

The main concern during construction was dam stability. In addition to the timing of construction, soil water content was maintained near the optimal 15.5% by diverting runoff using berms, ditches, and pumps.  The gully topsoil was excavated at the dam location until the sandstone foundation was exposed.  A bull-dozer moved soil from the adjacent borrow pit to the dam, reserving soil with a high clay content for the key and core.  A 10 tonne vibrating smooth drum compactor (Ingersoll Rand, Piscataway, NJ) was used to compact 15 cm lifts of soil (Fig 3.7A).  A neutron probe operated by a trained professional was used to monitor compaction.

The pea stone drain and toe drains were installed once dam height had reached the top of the pea stone drain.  The impermeable upstream layer of bentonite was incorporated once the dam was fully constructed.  The surface layer was loosened and a 1 tonne sac of bentonite, suspended from an excavator, was spread evenly over a marked area (Fig 3.7B).  The bentonite layer was covered with a 0.6 m layer of soil to protect it from punctures and drying.

The dam was hydro seeded and covered with straw mulch so that vegetation would quickly establish and prevent erosion.  The remaining bare soil was naturally colonized.  A riparian area comprised of various species of trees and shrubs was planted along one edge of the reservoir to trap sediment from surface runoff, utilize nutrients, stabilize banks, improve aesthetics and increase biodiversity.  Fig 3.8 shows the completed reservoir and spillway.


**Figure 3.7**  (A) Dam construction and (B) bentonite application.


**Figure 3.8** (A) Reservoir and (B) spillway.

## 3.5 IRRIGATION SYSTEM DESIGN

The function of an irrigation system as part of a WRIS is to use the water stored in the reservoir to irrigate crops, thereby increasing crop yield and quality. This increased productivity is the primary incentive for farmers to construct a WRIS. Wetland-reservoir irrigation systems may also provide the opportunity for farmers to grow new crop types that would not have been viable under previous conditions. Systems are designed so that water quality should be suitable for most crops when used directly. Other reuse systems require blending or cycling with freshwater (Shannon et al., 1997; Tanji and Kielen, 2002), an option for WRISs if the system does not perform as expected. Pollutant, particularly salt, accumulation in the surface layer of soil from the repeated reuse of water (De Villers, 2000) may be a concern for WRISs. This is site specific, depending on initial concentrations and soil type (De Villers, 2000). Residual P in WRIS irrigation water is not likely economically valuable to farmers because approximately only 2% of P originally applied to the field is lost via surface runoff and subsurface drainage (Sharpley et al., 2003).

Existing WRISs have used controlled drainage/subirrigation systems (Allred et al., 2003; Tan et al., 2007), which utilize the tile drainage lines to distribute water to crops (Belcher and D'Itri, 1995). The benefits of using controlled drainage/subirrigation are that irrigation infrastructure may already be in place and that nutrient losses via subsurface drainage are reduced (Evans et al., 1995; Lalonde et al., 1996; Wesstrom et al., 2001).

Existing tile drainage systems are not likely optimally designed for this dual purpose and certain field characteristics, such as the undulating topography often found in Nova Scotia, may prohibit their use (Belcher and D'Itri, 1995). In Nova Scotia, sprinkler irrigation is the most common type of irrigation (Environment Canada, 2004b), therefore an intermittent move sprinkler irrigation system was selected for this WRIS. A controlled drainage/subirrigation system would have interfered with other research conducted on the field. Design manuals (BCMAFF, 2001; NRCS, 1997b; Schwab et al., 1993) and consultations with irrigation experts guided the design process. The irrigation system specifications are listed in Table 3.5. The number and frequency of irrigations will depend on meteorological parameters, soil type, and crop. The reservoir capacity of 5000 $m^3$ is equivalent to 277 mm of water for irrigation over the 1.8 ha of drained land. This is may not be enough water to meet crop water requirements throughout numerous years of drought as recommended by the Prairie Water News (2002). However, additional water may be available through unaccounted recharge, increasing capacity, or capturing surface runoff. It should be noted that it is not necessary to irrigate the same field that supplies the wastewater to the WRIS. Irrigating a field at a lower grade than the reservoir may reduce pump requirements and cost.

Installation challenges included the pump's proximity to a power supply and selecting the appropriate type of intake system. The pump was situated as close as possible to a power source but still required power cable and a meter, which had a considerable cost. An alternative would be to use a power take-off or gasoline engine but the operational requirement of refuelling was not desired. The intake system, such as a floating intake or a wet well (Prairie Water News, 2002), is most easily installed during reservoir construction.

**Table 3.5** Specifications for the irrigation system used as a component of the Wetland-Reservoir Irrigation System.

| | |
|---|---|
| Type | Intermittent-move sprinkler |
| Irrigated area (ha) | 5.0 |
| Readily available moisture (mm) | 66 |
| Peak rate of water use (mm $d^{-1}$) | 6.0 |
| Precipitation rate (mm $h^{-1}$) | 7.6 |
| System capacity ($m^3$ $h^{-1}$) | 46 |
| Total design head (m) | 78 |
| Maximum Irrigation frequency (d) | 11 |
| Time per set (h) | 9 |
| Moves ($d^{-1}$) | 1 |
| Lateral spacing (m) | 18 |
| Sprinkler spacing (m) | 18 |
| Sprinkler | 25 mm brass impact (4.5 bars @ nozzle, 19.1 m radius, 2.59 $m^3$ $h^{-1}$) |
| Pump | Submersible (20 HP, 45 $m^3$ $h^{-1}$) |
| Motor | 3 phase, 60 Hz, 20 HP, 575 V, 21.5 A |
| Pipe | 9 m length, 10 mm diameter aluminum |

## 3.6 APPROVALS

An approval to construct the dam was required by the Nova Scotia Department of Environment and Labour through the Nova Scotia *Environment Act* (Province of Nova Scotia, 1994). Supporting documentation for the approval included project and site details, design drawings with a professional engineer's seal, and a dam safety plan, which was drafted based on the Canadian Dam Association's Dam Safety Guidelines (2007). This project also triggered a Canadian Environmental Assessment Screening. There was no response from the public participation component of the environmental assessment and only one mitigation measure was required; to use an impermeable liner for the CTW to minimize movement of wastewater to groundwater. Nearby residents had experienced elevated levels of $NO_3^-$ in their well water (NSDEL, 2001) but this mitigation measure was unexpected because the WRIS is expected to cause less groundwater pollution than the original situation of free drainage.

### 3.7 OPERATION AND MAINTENANCE

This WRIS was designed to have minimal operation and maintenance activities so that it would be more attractive to farmers. Operation and maintenance activities for WRISs are not well documented (Allred et al., 2003; Tan et al., 2007); however, selected activities (Table 3.6) can be found in design manuals for individual system components.

The common problem of siltation in dams (De Villers, 2000) is not expected to be a concern for many years because sediment is intercepted by the CTW. However, dredging may be required to remove sediment from the CTW as sediment accumulates. Dredging is more likely to be required in CTWs that receive surface runoff, which typically has higher concentrations of sediment. The use of a CTW to intercept sediment supports using a separate CTW and reservoir, rather than the combined wetland-reservoir used by Tan et al. (2007).

Vegetation litter that was transported during fall high flow events, frequently obstructed CTW outlet control structures. This could be addressed by using intake screens or larger control structures. Alvarez and Becares (2008) investigated harvesting vegetation to prevent the release of nutrients, which could also help with obstructions. Channelling, a distinct path of disturbed vegetation and eroded soil, was observed in the CTW shallow zones after very high flow events. Therefore, a rock buffer perpendicular to flow at the end of the forebay (Schueler, 1992) is recommended to disperse flow energy. During very high flow events the outlet control structures were too small to meet discharge requirements. This was likely due to surface runoff breaching berms, which should be included as a component of peak flow when sizing control structures.

The WRIS operated through winter without many challenges. An ice layer was observed inside the control structures but they did not completely freeze, likely due to a constant outflow. Had they frozen, they could leak and eventually contribute to the failure of the downstream berm. The tops of the control structures were exposed, therefore it may be worthwhile to insulate the top to prevent freezing. There was a concern that portions of the CTW inlet piping that could not be buried below the frost line would freeze, however, near constant inflow prevented freezing. The importance of the emergency spillways was observed when precipitation accumulated on top of the CTW ice layer during a storm event, unable to be discharged by the control structures. Without the emergency spillway the CTW would have flooded and potentially overtopped the berms. The shallow water table persisted, and in some areas the liner was lifted by groundwater pressure, despite the sloped CTW floor and drainage pipes beneath the liner. Additional efforts to address the shallow water table included installing a tile drain around the perimeter of the CTW and raising the water level to increase downward pressure. These efforts apparently reduced movement between wastewater and groundwater; however, it is more appropriate to avoid shallow water tables during site selection or to consider the pollutant loading and water volume contributions from groundwater during the design process. Monitor the WRIS beyond one year is recommended to determine additional operation and maintenance requirements.

**Table 3.6** Wetland-reservoir irrigation system operation and maintenance requirements.

| System Component | Operation and Maintenance Activities |
|---|---|
| Infrastructure | Ensure tile drains, ditches, spillways and control structures are not obstructed |
| Constructed treatment wetland | Manage ice level during winter months; Manage nuisance wildlife; Dredging; Replace shallow zones |
| Reservoir | Follow dam emergency preparedness plan and dam safety plan |
| Irrigation system | Move lateral lines to irrigate different field sections; Standard equipment maintenance; Remove pump during winter months |

## 3.8 COST

A major obstacle of implementing WRISs is cost. Factors such as existing components, irrigation requirements, drainage area, water quality, topography, and site geology all influence the system costs. Cost is not linear with drainage area, as larger systems may be more economically feasible because the proportion of land removed from production would be less and the cost of constructing larger reservoirs is non-linear (Richards et al., 1999).

The total cost of the present study was \$105,800 (\$59,000 $ha^{-1}$) and a summary of the costs is listed in Table 3.7. Costs are reported in CDN and a conversion factor, based on the average exchange rate from Jan 2010 through Sep 2010, of \$1.035 CDN = \$1 USD is used when comparing costs. Table 3.7 does not include the costs of supervising construction, developing safety and operational plans, preparing documents for construction approvals, operation, and maintenance, which are estimated to be \$3,000. Constructed treatment wetland operation and maintenance costs have been reported to be \$1,035 $ha^{-1}$ (Kadlec and Knight, 1996). If the tile drainage system needs to be installed or expanded an additional \$4,000 $ha^{-1}$ would be required. Three similar systems in Ohio receiving drainage from 10.9, 16.2, and 18.3 ha cost \$48,645 (\$4,463 $ha^{-1}$), \$62,100 (\$3,833 $ha^{-1}$), and \$89,010 (\$4,864 $ha^{-1}$), respectively (Allred et al., 2003; Richards et al., 1999). The difference in cost between the present WRIS and the systems in Ohio may attributed to scale, existing reservoirs at two sites in Ohio, a larger $A_W$ to $A_D$ ratio at this WRIS, and site specific differences. Over half (\$55,200) of the total cost of the present WRIS was incurred during dam construction. Dam construction cost per unit of volume was more than 3 times that of straight excavation. The volume of compacted fill material used to construct the dam was ≈ 3500 $m^3$ and created a reservoir with a capacity of only 5000 $m^3$. This suggests that a dug-out type of reservoir would have been more economical at this site.

The benefits of irrigation, namely increased crop productivity and the ability to grow higher value crop types that would not have been viable under previous conditions, will help offset the capital cost. Adapting WRIS to local environmental conditions will maximize economic efficiency; accurately sized reservoirs will capture all available water to maximize irrigation benefits and accurately sized CTWs will provide the desired treatment without unnecessary construction and land use expenses. A full economic analysis of the present WRIS could not be performed as yield data was not collected. Yield increases would depend on crop type and growing season weather. A better understanding of predicted drought frequency and severity would be beneficial when selecting a site that will receive the greatest irrigation benefit from a WRIS. Richards et al. (1999) concluded that the WRIS they studied was not profitable; however, their economic analysis only considered the benefit of increased yields and not the additional benefits of WRISs, such as pollution mitigation, water conservation, increased biodiversity, and flood prevention. These additional benefits and their value need to be expressed to support the adoption of this type of system. Wichelns (2005) investigated the economic feasibility of an integrated on-farm drainage management system in California and concluded that economic incentives or drainage water disposal regulations may be required to encourage farmers to bear the cost of the system. It is recommended to investigate alternative applications of WRISs, such as crop processing, aquaculture, and golf industry applications, to determine if they are more economically viable.

**Table 3.7** Wetland-reservoir irrigation system costs.

| Material | Cost ($CDN) |
|---|---|
| *Irrigation system* | |
| Pump | $4,800 |
| Pipes | $9,000 |
| Sprinklers | $2,400 |
| Fittings | $2,400 |
| *Constructed treatment wetland* | |
| Excavation | $10,000 |
| Synthetic liner | $5,700 |
| Control structures | $2,000 |
| Pipes | $2,500 |
| Valves | $600 |
| Spillway construction | $400 |
| Spillway rock | $400 |
| *Reservoir* | |
| Soil tests | $1,500 |
| Engineering fees | $4,700 |
| Grubbing and stump disposal | $10,400 |
| Dam construction | $40,000 |
| Compaction monitoring | $600 |
| Bentonite | $3,000 |
| Spillway geotextile | $800 |
| Spillway rock | $2,400 |
| Site seeding | $2,200 |
| Total | $105,800 ($59,000 $ha^{-1}$ of drained land) |

## 3.9 SUMMARY

The present study serves as a case study for the design, construction, and early operation of a wastewater treatment and reuse system consisting of a tile drainage system, a CTW, and an irrigation reservoir. The design of the individual system components were based on standard design manuals and adapted to the challenges presented by the environmental conditions that exist in Nova Scotia. Construction and operation challenges were identified to assist with future WRIS's. Historical tile drainage flow and water quality data were important for adapting the CTW design to the local environmental conditions and estimating available water.

The CTW was sized using the k-C* model, applied monthly to each pollutant using $Q$, the monthly peak $C_{in}$, and $k$ determined from a cold climate CTW study. An $A_W$ to $A_D$ ratio within the recommended range supported this method of applying the k-C* model. The soil found on site was not ideal for use as dam fill, therefore conservative compaction specifications, an impervious upstream dam face, and a pea stone drain inside the dam were used to ensure dam stability. The reservoir volume was limited by land availability, topography, and cost, and may not provide enough water to meet crop water requirements throughout numerous years of drought. An intermittent move sprinkler irrigation system was selected for this WRIS because of its widespread use in Nova Scotia and because a controlled drainage/subirrigation system would have interfered with other research conducted on the field. An approval to construct the dam and an environmental assessment were required. In addition to the design methods and specifications used in the present WRIS, design recommendations are to:

- Utilize a rock buffer perpendicular to flow at the start of the CTW to disperse flow energy, and thereby prevent channelling
- Utilize riser pipes at the CTW inlet to prevent the inlet from being buried beneath sediment accumulation
- Include surface runoff as a component of peak flow when sizing control structure so that the CTW does not flood if berms and ditches are breached or overtopped

- Consider pollutant loading and water volume contributions from groundwater when sizing the CTW and reservoir
- Consider irrigating a field at a lower elevation than the reservoir to reduce pump requirements and cost

Limited land availability and grade, as may often be the case, forced the CTW to be constructed where there was a shallow water table. A tile drain around the perimeter of the CTW and raising the water level to increase downward pressure apparently reduced movement between wastewater and groundwater. Construction recommendations are to:

- Time construction in the late spring orearly summer to permit the establishment of a vegetation cover that will mitigate erosion
- Use machinery to position liner
- Install CTW inlets and outlets before liner to avoid installation challenges
- Install the irrigation system intake during reservoir construction

The WRIS operated through winter without many challenges. Additional operational requirements identified through the present study are to:

- Utilize intake screens or larger control structures to prevent vegetation litter from obstructing CTW outlet control structures
- Insulate tops of control structures to prevent freezing
- Monitor WRIS beyond one year to determine additional operation and maintenance requirements

The cost of the present WRIS was greater than similar systems. Over half of the total cost was incurred during dam construction, suggesting that an on-stream reservoir may not have been the most economical type of reservoir. Recommendations for decreasing WRIS costs and encouraging their adoption are to:

- Utilize existing system components

- Construct on a large scale, as cost is non-linear with drainage area
- Adapt WRIS to local environmental conditions so that they are more effective and efficient
- Gain a better understanding of predicted drought frequency and severity to select a site that will receive the greatest irrigation benefit
- Monitor yield to permit a full economic analysis
- Express the additional benefits of WRISs and their value to encourage their implementation
- Consider economic incentives or drainage water disposal regulations to encourage WRIS implementation
- Investigate alternative applications of WRISs, such as crop processing, aquaculture, and golf industry applications, to determine if they are more economically viable

# Chapter 4 - Hydraulic Assessment of a Wetland–Reservoir Wastewater Treatment and Reuse System Receiving Agricultural Drainage Water in Nova Scotia

## 4.1 Introduction

A hydraulic assessment of a WRIS can provide information that can be used to further the process of adapting it to local environmental conditions, as hydraulics affect water availability and CTW performance.

In Nova Scotia, the potential exists to have more water available for irrigation than WRISs in Ohio (Allred et al., 2003) and Ontario (Tan et al., 2007), as it receives greater precipitation. Water budgets can be used to determine when and how much water is available, which will help size reservoirs to maximize the amount of water available for irrigation. Greater amounts of water available may also mean higher flow rates. Monitoring flow rates can provide data that can be used to properly size CTWs and WRIS infrastructure.

Constructed treatment wetland performance is affected by hydraulics because several biogeochemical processes, such as sedimentation and biochemical transformations, depend on how water moves through the CTW. Actual treatment may differ from design treatment if water is not active or does not follow plug flow hydraulics. In this case, parcels of water may reside in the CTW for varying amounts of time and therefore undergo varying amounts of treatment. Residence time distribution (RTD) can help characterize hydraulics and has been shown to have a strong correlation to treatment (Dierberg et al., 2005; Reinhardt et al., 2005). It is particularly important to understand CTW hydraulics during high flow events because they create periods of high pollutant loading and may release stored pollutants (Raisin et al., 1997), thereby making treatment critical. The actual measured residence time ($t_a$) in the CTW was expected to be similar to $t_n$ because a high length to width ratio (10:1) was used to promote plug-flow hydraulics.

The objective discussed in this chapter is to assess hydraulics of a WRIS constructed at the Bio-Environmental Engineering Center in Truro, Nova Scotia, Canada (Chapter 3). This is achieved by means of a water budget and RTD assessment. Recommendations for adapting the design and operation of future systems in Nova Scotia are discussed.

## 4.2 METHODOLOGY

The WRIS is composed of a tile drainage system beneath 1.8 ha, two identical and independent CTWs with a total surface area of 1025 $m^2$, a reservoir with a capacity of 5000 $m^3$, and a sprinkler irrigation system. A detailed description of the WRIS is provided in Chapter 3. Wetland-reservoir irrigation system hydraulics were assessed using a case study approach rather than utilizing simulation models, such as that used by Arheimer and Wittgren, (1994) or DRAINMOD (Skaggs, 1978). This approach is more site specific and allows the CTW design model to be assessed. The present hydraulic assessment focuses on CTW hydraulics. The WRIS was monitored from Nov 1, 2007 through Dec 31, 2009.

### 4.2.1 Meteorological Measurements

Hourly meteorological measurements were collected at a weather station adjacent to the CTW. Air temperature and relative humidity were monitored by a Campbell Scientific model CS500 air temperature and relative humidity probe (Campbell Scientific Corp., Edmonton, AB). Solar irradiance was monitored by a LI-COR model LI200S silicon pyranometer (Campbell Scientific Corp., Edmonton, AB). Precipitation was monitored by a Young 52202 0.1 mm heated tipping bucket rain gauge (R. M. Young Company, Traverse City, MI). Environment Canada (2008) precipitation data were also used to supplement when the rain gauge data were not available. All meteorological data were recorded by a Campbell Scientific CR 10X datalogger (Campbell Scientific Corp., Edmonton, AB).

### 4.2.2 Flow Monitoring

Flow rates were continuously monitored by tipping buckets at the tile drainage system outlet (Hut I), and the W1 and W2 outlets (Hut II). Flow data were recorded hourly by a

Campbell Scientific CR 10X datalogger (Campbell Scientific Corp., Edmonton, AB). A Palmer-Bowlus flume (Walkowiak, 2006) at the W1 and W2 outlets was used to measure flow rates that exceeded the tipping bucket capacity (0.5 tip $s^{-1}$). Flume water levels were manually recorded. An ISCO 730 bubbler flow module (Teledyne Isco, Inc., Lincoln, NE) and a WL-16 submersible pressure transducer and USB datalogger combination (Global Water, Gold River, CA) were tested to automatically record flume water levels but did not provide precise measurements. Utility lines were the power source for Hut I and it was heated by an electric base board heater to prevent tipping buckets from freezing. Hut II was situated at a more remote location so it used a KC130TM 130 W solar panel (Kyocera Solar Inc., Scottsdale, AZ) as a power source and a DV-210-75G 10,000 BTU $h^{-1}$ propane direct vent wall furnace (Empire Comfort Systems Inc., Belleville, IL) for heat.

### 4.2.3 Residence Time Monitoring

A common method of assessing RT is to conduct a hydraulic tracer study to determine a RTD curve, which shows a distribution of times that parcels of water reside in the CTW. A hydraulic tracer study can be conducted in a CTW by injecting a known amount of tracer into the inlet and monitoring tracer concentration at the outlet, along with inflow and outflow rates (Kadlec and Knight, 1996). Bromide ($Br^-$), an ion tracer, in the form of potassium bromide was selected for the present study because it is widely used, nonreactive, found at low background concentrations, has a low toxicity, and easy to analyze (Whitmer et al., 2000). Potassium bromide has been used in similar tracer studies in Nova Scotia (Jamieson et al., 2001; Miles, 2008; Smith et al., 2005). Rhodamine B, a dye tracer, was used simultaneously for a comparison but was too reactive and not detected at the CTW outlet. Rhodamine Wetland Tracer should have been used instead (Dierberg et al., 2005; Keefe et al., 2004, Lin et al., 2003).

Two tracer studies (A and B) were initiated in W2 on Sept 1 and 24, 2008, respectively. The months of Sept was selected because, historically, it is a period with significant rainfall (109.1 mm is the expected normal for Truro, Nova Scotia) occurs (Environment Canada, 2008). During this period, all inflow was directed to W2 to eliminate any unequal flow splitting. Bowman (1984) recommended that the mass of tracer be at least 20 x its background concentration multiplied by the wetland volume. For the present studies the mass of tracer was determined from 25 x the background concentrations, which were the detectable limits (0.5 mg $L^{-1}$ for $Br^-$, 0.01 µg $L^{-1}$ for Rhodamine B). Water levels in the deep and shallow zones during the studies were 0.65 m and 0.35 m, respectively. Water samples were collected from the outlet of W2 every 12 h by an Isco 6712 portable samplers (Teledyne Isco Inc., Lincoln, NE). Samples were analyzed for $Br^-$ by means of ion chromatography according to American Public Health Association (APHA) Method 4110 C: Single-Column Ion Chromatography with Electronic Suppression of Eluent Conductivity and Conductimetric Detection (Clesceri et al., 2005). Samples were syringe filtered with 0.45 µm nitrocellulose membrane filters (Millipore Corp., Billerica, MA) and analyzed using a Waters Ion Chromatography System (Waters Canada Ltd., Mississauga, ON). A total of 48 water samples were analyzed for $Br^-$ in tracer study B.

Nominal residence time was calculated using eqn 3. A porosity ($\eta$) of 0.95, as recommended by Kadlec and Knight (1996), and a mean daily wetland outflow rate ($\overline{Q_o}$) during the tracer study, to account for dynamic flow rates, were used.

$$t_n = \frac{V\eta}{\overline{Q_o}} \qquad [2]$$

Where:

$t_n$ = nominal residence time (d)

$V$ = wetland volume ($m^3$)

$\eta$ = porosity, and

$\overline{Q_o}$ = mean daily wetland outflow rate ($m^3$ $d^{-1}$)

Actual measured residence time during the tracer study was determined from the first moment (eqn 3) of the residence time distribution function (*f(t)*) (eqn 4).

$$t_a = \int_0^\infty tf(t)dt \qquad [3]$$

Where:

$t_a$ = actual measured residence time (d)

$t$ = time (d), and

$f(t)$ = residence time distribution function ($d^{-1}$)

$$f(t) = \frac{Q_o(t)C(t)}{m} \qquad [4]$$

Where:

$f(t)$ = residence time distribution function ($d^{-1}$)

$C(t)$ = exit tracer concentration (mg $m^{-3}$)

$Q_o$ = wetland outflow rate ($m^3$ $d^{-1}$)

$m$ = total tracer mass collected (mg), and

$t$ = time (d)

Mass recovery of the tracer was verified from the zeroth moment (eqn 5) of the $f(t)$ (Kadlec and Knight, 1996).

$$M_{\text{Recov}} = \int_0^\infty Q_o(t)C(t)dt \qquad [5]$$

Where:

$M_{Recov}$ = total tracer mass recovered (mg)

$Q_o$ = wetland outflow rate ($m^3$ $d^{-1}$)

$C(t)$ = exit tracer concentration (mg $m^{-3}$), and

$t$ = time (d)

The degree of mixing in a CTW can be a sign of the variation in treatment of parcels of water. The degree of mixing can be characterized by the RTD variance ($\sigma^2$), that is, the spread of the tracer response curve around the mean of the distribution, $t_a$ (Kadlec, 1994). Residence time distribution variance was determined from the second moment of the *f(t)* (eqn 6).

$$\sigma^2 = \int_0^\infty (t - t_a)^2 f(t) dt \qquad [6]$$

Where:

$\sigma^2$ = residence time distribution variance ($d^2$)

$t$ = time (d)

$t_a$ = actual measured residence time (d), and

$f(t)$ = residence time distribution function ($d^{-1}$)

## 4.3 Results and Discussion

### 4.3.1 Meteorological Parameters

The amount and distribution of precipitation are important factors to consider during a hydraulic assessment because they govern flow rates and water availability. A summary of the 1971-2000 climate normals (Environment Canada, 2008) and monthly meteorological means and totals during the entire monitoring period are presented in Table 4.1. Annual precipitation was greater than normal and amounts reported by similar studies (Table 2.1), indicating WRISs in Nova Scotia may capture more water but have greater treatment challenges than WRISs in other regions.

### 4.3.2 Flow Rates

Flow rates from the tile drainage outlet are shown in Fig 4.1. They illustrate the dynamic nature of flow entering the CTW. The mean tile drainage outflow rate in 2008 was 0.72 $m^3 h^{-1}$. A larger value for $Q$, 1.9 $m^3 h^{-1}$, was used in the k-C* model was (Section 3.3.2) to account for dynamic pollutant loading. As much as 24% of the annual tile drainage outflow volume was contributed by flow rates that exceeded $Q$, and therefore this volume may not have a long enough RT to undergo the desired treatment, depending on the distribution of flow rates and inlet concentrations. When considering overall treatment, this potential treatment shortfall may be offset by additional treatment from longer RT during low flow periods and the increased RT due to the forebay, which is not considered by the k-C* model. Treatment must be examined before drawing firm conclusions but aside from using a larger CTW, design and management options, such adding additional stop plates to the CTW outlet control structure before high flow events to increase CTW volume, combining the CTW and reservoir into a single system, using a head pond and a lower rate pump to the CTW, or circulating CTW outflow through the CTW again, may help extend RT during high flow events and provide the desired treatment to all flow rates. Diverting high flows away from the WRIS, to prevent high pollutant loading to the reservoir, would significantly decrease water availability and is not recommended.

Peak flow rates of 6.18 $m^3\ h^{-1}$ (during a 30 mm storm event) and 11.40 $m^3\ h^{-1}$ (during a 46 mm storm event) were recorded from the tile drainage outlet and the combined W1 and W2 outlets, respectively. The difference may be attributed to potential gains to W1 and W2 from precipitation, surface runoff, and groundwater intrusion, and flow monitoring instrumentation errors. No backup of flow was expected or observed in the 200 mm pipe connecting the tile drainage system to the CTW because it has a larger capacity than that of the seven tile drainage outlets combined. During peak flow events backup of flow was observed in the 150 mm W1 and W2 outlet control structures. Only tile drainage outflow and precipitation were considered when sizing the control structures. Therefore, all potential gains, including those from groundwater intrusion and surface runoff that breaches berms, should be considered when sizing the CTW outlet. If the contributions from some potential gains are unknown, an emergency spillway is an important design feature, as observed in the present study. Low tile drainage outflow during Jun and Jul indicates a potential to treat alternative farm wastewater during this period, and is the best time to perform maintenance.

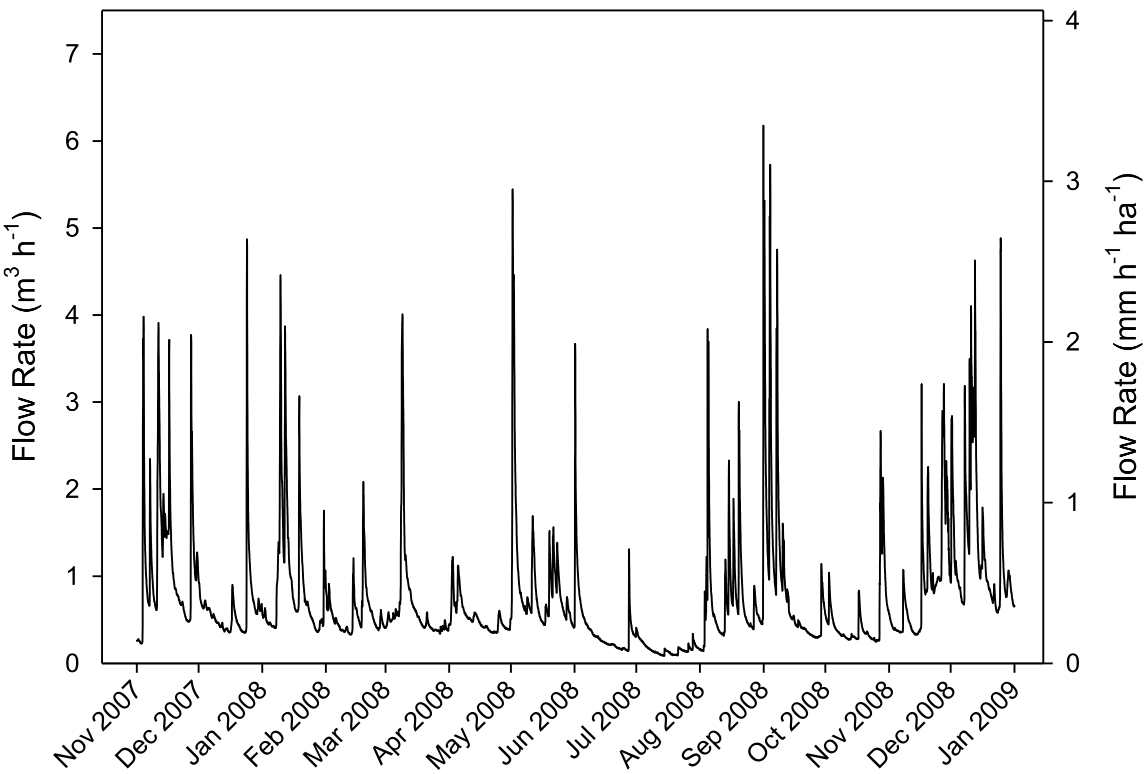


**Figure 4.1** Flow rates ($m^3\ h^{-1}$ and standardized as a depth over the constructed treatment wetland surface area per drained field surface area ($mm\ h^{-1}\ ha^{-1}$)) at the tile drainage outlet.

### 4.3.3 Constructed Treatment Wetland Water Budget

Annual (Table 4.2) and monthly (Figs 4.2 and 4.3) water budgets for W1 and W2 were calculated using eqn 7, which was adapted from USEPA (1988). Surface runoff is assumed to be negligible because it was diverted from the CTW using berms and ditches, although some was observed breaching berms during large storm events. Seepage is assumed to be negligible because an impermeable liner was used.

$$dV/dt = Q_i - Q_o + PRECIP * A - ET * A \qquad [7]$$

Where:

$dV/dt$ = change in volume ($m^3\ month^{-1}$)

$Q_i$ = wetland inflow rate ($m^3\ month^{-1}$)

$Q_o$ = wetland outflow rate ($m^3\ month^{-1}$)

$PRECIP$ = precipitation rate (m month$^{-1}$)

$ET$ = evapotranspiration rate (m month$^{-1}$)

$A$ = wetland surface area (m$^2$)

Evapotranspiration (ET) was initially calculated using the Penman-Monteith equation (Allen et al., 1998), however, this equation yielded abnormally low ET rates. Therefore, lake evaporation normals from Truro, Nova Scotia (Environment Canada, 2008) were used as ET. Evapotranspiration was also approximated by assuming all solar radiation was utilized as latent energy for the vaporization of water. This approximation supported using normals rather than the Penman-Monteith equation. The average ET during the warm months (May - Oct) was 2.8 mm d$^{-1}$. Brassard (2001) reported an average ET of 3.9 mm d$^{-1}$ from an agricultural constructed wetland in Nova Scotia. Evapotranspiration removed 524 m$^3$ (511 mm) from W1 and W2 combined in 2008 (Table 4.2), which is 6.7% of the contribution from tile drainage outflow and precipitation combined. Similar studies in warmer climates have reported ET as a significant loss in CTW water budgets (Kovacic et al., 2000; 2006; Kivaisi, 2001).

If all gains and losses are accounted for then the expected water balance would be 0 m$^3$. However, mean monthly imbalances of -578 m$^3$ (151 m$^3$ standard deviation) and 103 m$^3$ (118 m$^3$ standard deviation) were recorded in W1 (Fig 4.2) and W2 (Fig 4.3), respectively. Wetland 1 and W2 combined gained 6,360 m$^3$ from tile drainage outflow in 2008 (Table 4.2), as expected based on annual tile drainage outflow (Section 3.4.2), yet the CTW lost more than twice that amount, 12947 m$^3$ (Table 4.2). The water budget imbalances may be attributed to unequal flow splitting, monitoring errors, and unaccounted gains or losses. All inflow was directed to W2 after Aug 1, 2008 to eliminate any unequal flow splitting. A mean monthly imbalance of 176 m$^3$ was subsequently recorded in W2, therefore, unequal flow splitting was not a major source of the imbalances.

**Table 4.1** Summary of climate normals (1971-2000) and monthly meteorological parameters.

| | 2007 | | 2008 | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Nov** | **Dec** | **Jan** | **Feb** | **Mar** | **Apr** | **May** | **Jun** | **Jul** | **Aug** | **Sep** | **Oct** | **Nov** | **Dec** | **2008** |
| **Precipitation (mm)**† | 143 | 92 | 100 | 142 | 118 | 64 | 103 | 58 | 78 | 169 | 145 | 88 | 160 | 189 | 1414 |
| **Normal precipitation (mm)**† | 112 | 106 | 93 | 82 | 94 | 86 | 104 | 96 | 91 | 90 | 109 | 108 | 112 | 106 | 1170 |
| **Daily mean temperature ($^o$C)** | 3 | -5 | -4 | -4 | -3 | 5 | 9 | 15 | 20 | 18 | 14 | 8 | 4 | -1 | 7 |
| **Normal daily mean temperature ($^o$C)**† | 3 | -4 | -7 | -6 | -2 | 4 | 10 | 15 | 19 | 18 | 14 | 8 | 3 | -4 | 6 |
| **Relative humidity (%)** | 93 | 93 | 94 | 92 | 88 | 84 | 87 | 92 | 93 | 96 | 93 | 93 | 95 | 93 | 92 |
| **Solar radiation (MJ $m^{-2}$ $d^{-1}$)** | 5 | 4 | 6 | 8 | 12 | 16 | 16 | 18 | 19 | 14 | 13 | 9 | 5 | 4 | 12 |
| **Wind speed (m $s^{-1}$)** | 4 | 4 | 4 | 4 | 5 | 4 | 4 | 3 | 2 | 2 | 3 | 3 | 3 | 4 | 3 |

†(Environment Canada, 2008)

**Table 4.2** Summary of water budgets during 2008.

| | **Wetland 1** | **Wetland 2** | **Reservoir** |
|---|---|---|---|
| ***Gains*** | | | |
| **Inflow from tile drainage ($m^3$)**† | 1538 (3.00)‡ | 4822 (9.41)‡ | |
| **Inflow from constructed wetlands ($m^3$)**† | | | 12947 |
| **Precipitation ($m^3$)**† | 725 (1.41) | 725 (1.41) | 4242 |
| ***Losses*** | | | |
| **Outflow ($m^3$)**† | 9074 (17.70) | 3873 (7.56) | |
| **Evapotranspiration ($m^3$)**† | 262 (0.51) | 262 (0.51) | 1536 |
| **Change in volume ($m^3$)**† | -7073 (-13.80) | 1412 (2.76) | |

† Depth equivalent (m) over component surface area shown in brackets
‡ All tile drainage flow was directed to Wetland 2 at the start of August 2008

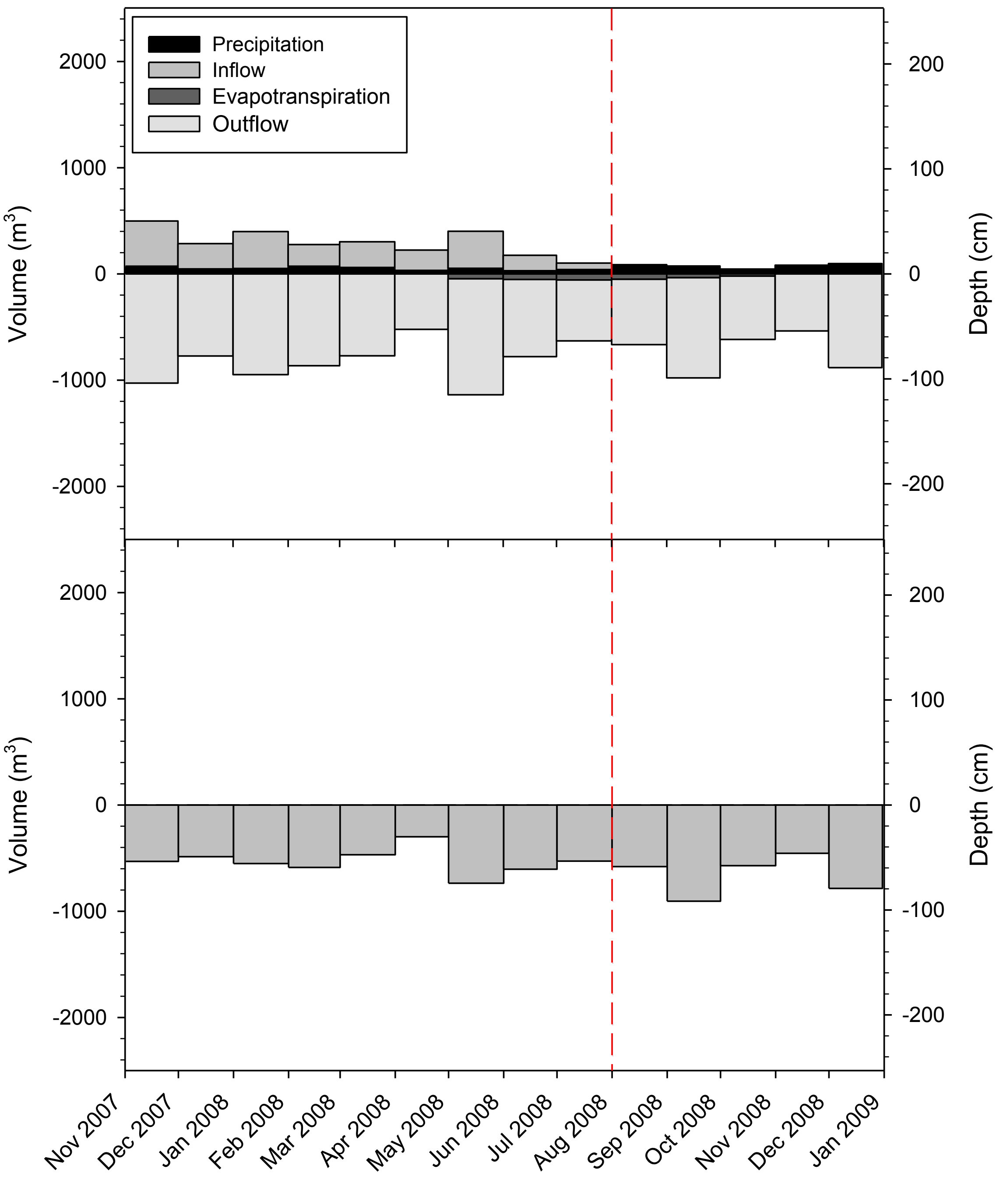


**Figure 4.2** Water budget and change in volume for Wetland 1. The date when all inflow was directed to Wetland 2 is shown by vertical dashed line.

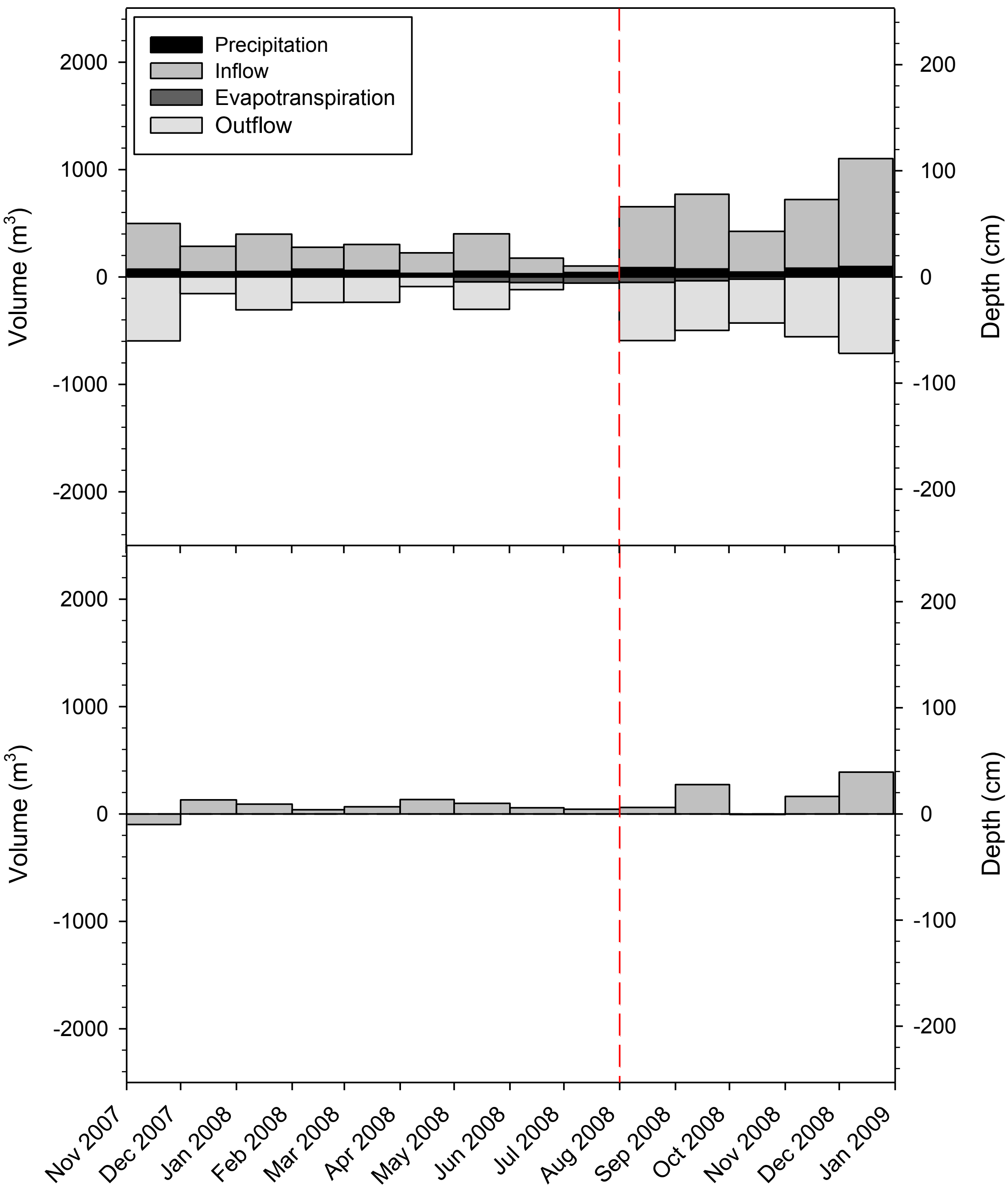


**Figure 4.3** Water budget and change in volume for Wetland 2. The date when all inflow was directed to Wetland 2 is shown by vertical dashed line.

Tile drainage outflow rarely exceeded tipping bucket capacity. Constructed treatment wetland outflow periodically exceeded tipping bucket capacity and flume readings were not very precise, therefore, improved flow monitoring of high flow events at the CTW outlet is recommended, possibly using an electromagnetic flow sensor. Continued outflow from W1 after all inflow was directed to W2 (Fig 4.2) indicates that the negative imbalance in W1 were primarily caused by unaccounted gains. Surface runoff was observed breaching and overtopping berms and ditches and entering the CTW during large storm events. A shallow water table was identified as concern during the site investigation and in some areas lifting of the liner by groundwater pressure was observed. Therefore the liner, sloped CTW floor, drainage pipes beneath the liner, tile drain around the CTW perimeter, and water table management may not have been completely effective at preventing groundwater intrusion. Less groundwater intrusion may have occurred in W2 because W2 had a positive water imbalance and W1 may have intercepted groundwater, based on a water table investigation and site topography. Reinhardt et al. (2005) also reported that groundwater was a significant component of their CTW water budget in a similar study. A site investigation should be conducted to identify the water table depth so that the CTW can be situated to avoid movement between groundwater and wastewater. Seventy-six percent of W1 annual gains and 25% of W2 annual losses in 2008 are unaccounted (Table 4.2). Therefore any conclusions drawn from the flow data should consider these discrepancies.

### 4.3.4 Water Availability

It was not possible to perform a complete reservoir water budget because reservoir outflow was not monitored, however, CTW outflow, precipitation, and ET data (Table 4.2) can provide some information on the availability of water for irrigation. To illustrate water availability the annual period of reservoir recharge is assumed to be a 12 month period starting Oct 1 (Fig 4.4), approximately when all reservoir water has been used for irrigation and the growing season is over. Between Oct 1, 2007 and Oct 1, 2008 the reservoir captured 15,600 $m^3$ (8,666 $m^3$ $ha^{-1}$ of drained land) from CTW outflow and precipitation combined (Fig 4.4). Evapotranspiration removed 1,536 $m^3$ (512 mm) from the reservoir during this period (Table 4.2), ≈ 9% of the contribution from CTW outflow and precipitation combined. The annual volume of available water of 14,064 $m^3$ (Fig 4.4) is equivalent to 780 mm of water for irrigation over the 1.8 ha of drained land, provided that the reservoir capacity was large enough to retain it all. This is considerably greater than the projected volume of 8,500 $m^3$ (4,722 $m^3$ $ha^{-1}$ of drained land). Gains from tile drainage and precipitation were close to expected. The additional captured water is attributed to unaccounted gains in the CTW and CTW outflow monitoring errors (Section 4.3.2), and unaccounted gains in the reservoir, such as groundwater, surface runoff, and outflow from the tile drain around the CTW perimeter. The WRIS was not intended to capture groundwater or surface runoff, however, the potential to augment water availability by incorporating them is shown. Reservoirs should be sized to capture all available water to maximize irrigation benefits, and thereby economic efficiency.

In a similar system in Ontario, Tan et al. (2007) reported annual tile drainage outflows of 2380 and 1670 $m^3$ $ha^{-1}$ under free and controlled drainage, respectively. Other studies on CTWs receiving solely tile drainage water have reported annual CTW outflow volumes ranging from 1500 to 2800 $m^3$ $ha^{-1}$ (Kovacic et al., 2000; 2006; Tanner et al., 2005). This indicates that WRISs in Nova Scotia may capture more water but also have greater treatment challenges than WRIS in other regions, even if the tile drainage outflow is considered the only gain. The differences in available water between the present study and others may be attributed to a greater amount of precipitation in Nova Scotia. This illustrates the importance of adapting WRIS designs to local environmental conditions. Accurately sized reservoirs will capture all available water to maximize irrigation benefits and accurately sized CTWs will provide the desired treatment without unnecessary construction and land use expenses.

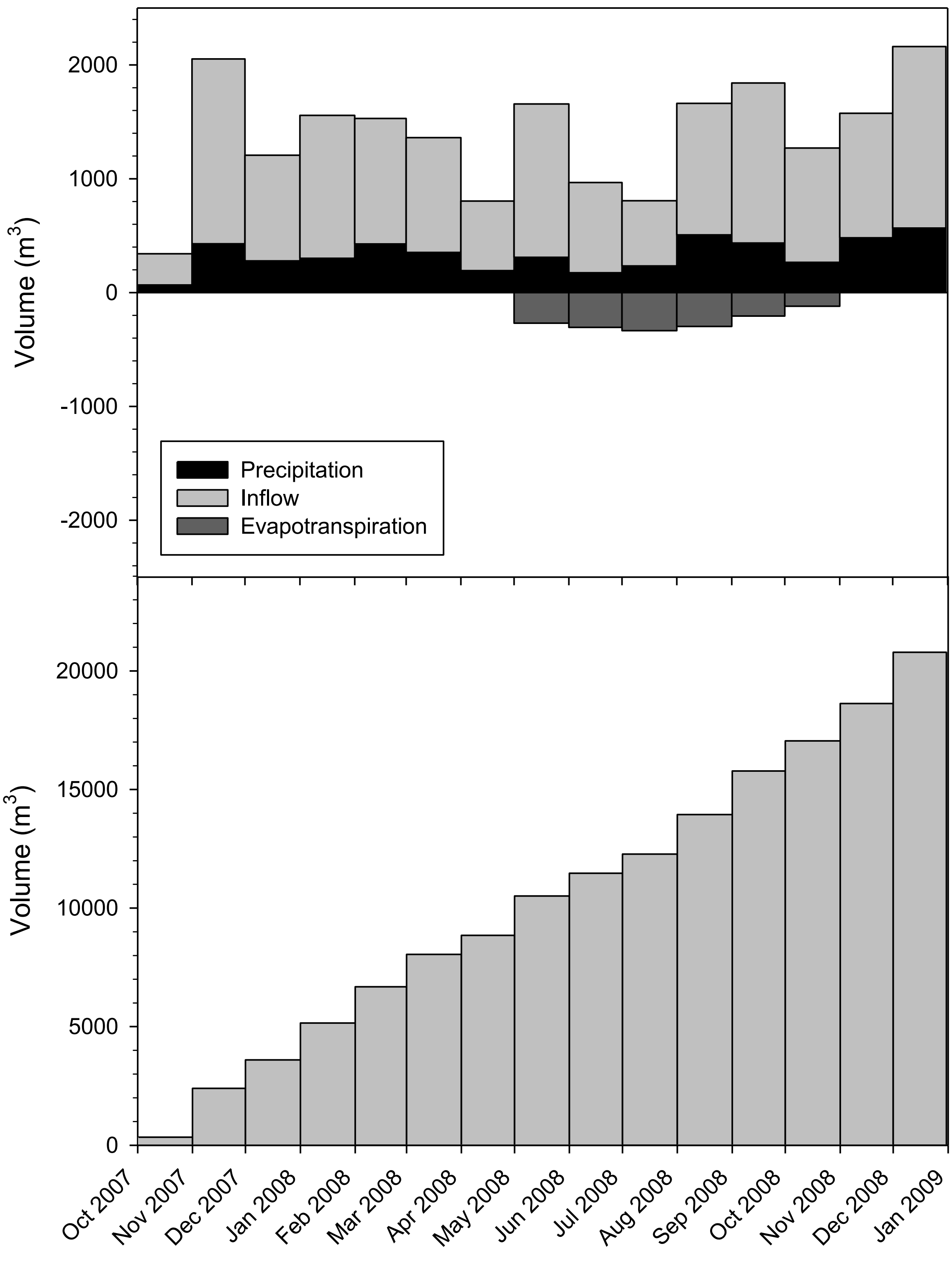


**Figure 4.4** Reservoir water budget (lacking outflow) and available water.

### 4.3.5 Residence Time

Hydraulic tracer study A was not completed because the CTW was flooded and damaged during a flash flood. The $t_n$ during hydraulic tracer study B was 14.5 d. The tracer response curve for tracer study B (Fig 4.5) illustrates that water did not follow plug-flow hydraulics and therefore varying treatment of parcels of water may be expected. The tracer response curve (Fig 4.5) was linearly extrapolated from the last known concentration because the collection of water samples was prematurely stopped before $Br^-$ concentrations returned to background levels because the tracer study duration had to be estimated due to delayed water sample analyses. The linear extrapolation is an approximation, however all tracer would likely have been discharged by the high flow event beginning on Day 33 and other approaches to extrapolating the curve are thought to have little affect on the centroid, that is $t_a$.

Water quality precision was verified during the tracer study by collecting and analyzing three replicate samples. The mean $Br^-$ concentration of the three replicates was 7.20 mg $L^{-1}$ and the standard deviation was 0.17 mg $L^{-1}$. The $t_a$ during tracer study B was 15.0 d. Theoretically the $t_a$ should be less than the $t_n$ but in tracer study B the $t_a$ was 103% of $t_n$. This is attributed to extrapolating the last section of the tracer response curve, missing flow data between Day 0 and 5, or monitoring or analytical errors. A $t_a$ that is similar to its associated $t_n$ suggests that there are few inactive zones and supports the use of a length to width ratio of 10:1 for the CTW shape. It also suggests that the overall treatment may be similar to that predicted by the k-C* model.

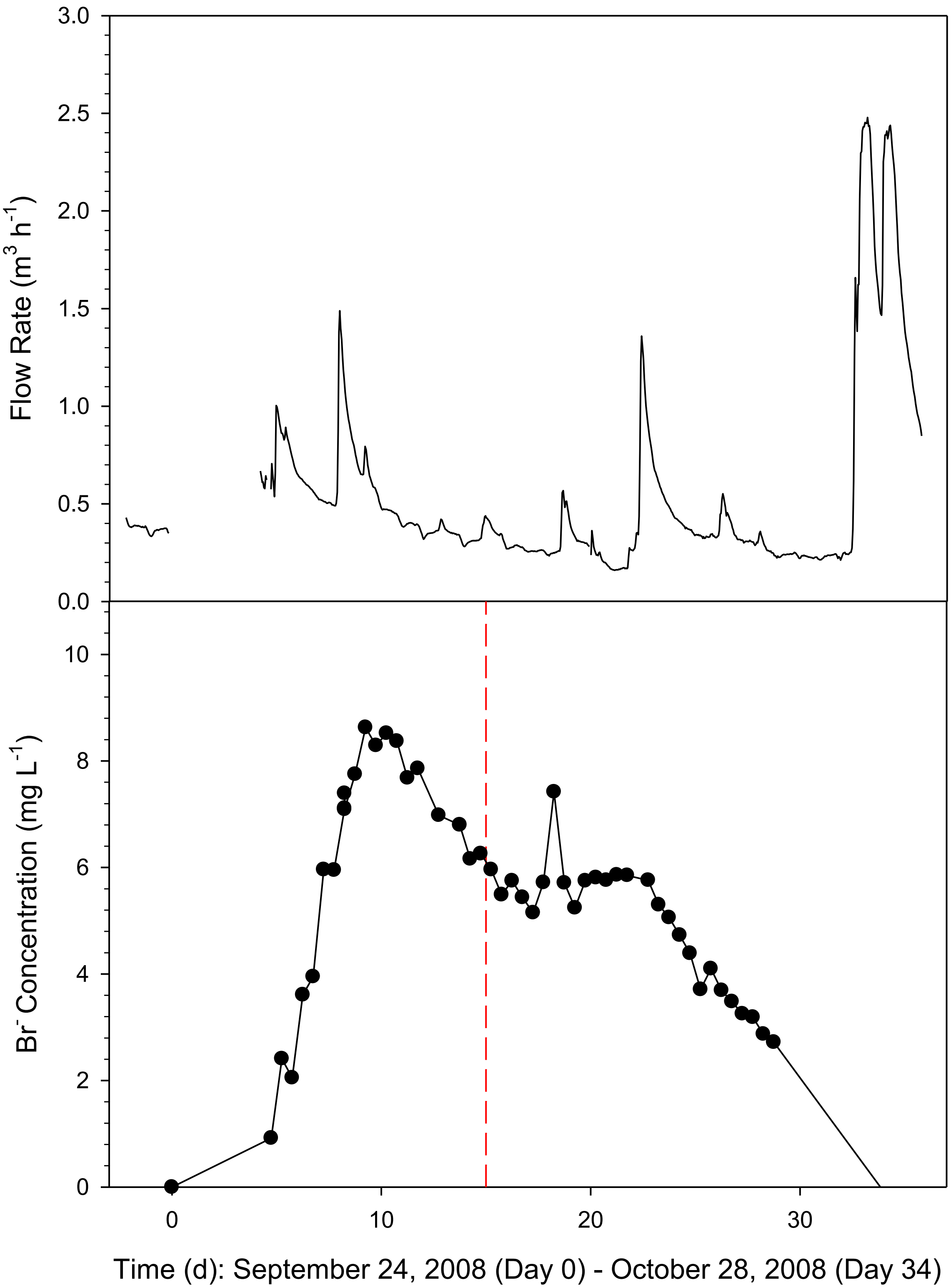


**Figure 4.5** Wetland 2 outlet flow rates ($m^3$ $h^{-1}$) and tracer response curve during tracer study B. The actual measured residence time is shown by the vertical dashed line.

Bromide mass recovery from tracer study B was 72%, which is relatively high and indicates that the conclusions of drawn from tracer study B valid. Similar recoveries were reported by Jamieson et al. (2001) (75%), Miles (2008) (42 to 116%), and Smith et al. (2005) (72 to 81%). Unrecovered mass may be attributed to monitoring or analytical errors, or plant uptake (Whitmer et al., 2000).

A $\sigma^2$ of 49.8 $d^2$ was calculated from tracer study B. The $\sigma^2$ determined from the present study can be compared to the $\sigma^2$ in other CTWs receiving drainage water to assess how different designs affect the degree of mixing, however, little information on $\sigma^2$ in CTWs receiving drainage water has been presented in literature. While the similarity of the $t_a$ to $t_n$ during tracer study B suggests that overall treatment may be similar to that predicted by the k-C* model, the large $\sigma^2$ indicates that there may be a wide range of treatment of individual parcels of water.

A summary of tracer study B is presented in Table 4.3. Holland et al. (2004) conducted a RTD study in a stormwater treatment wetland and reported that flow rates did not have a significant effect on RTD characteristics while water level did. Additional tracer studies are recommended to help characterize RTD over a range of flow rates and water levels.

**Table 4.3** Summary of hydraulic tracer study B in Wetland 2.

| Date | Nominal Residence Time (d) | Actual Measured Residence Time (d) | Mass Recovery (%) | Variance ($d^2$)† |
|---|---|---|---|---|
| Sept 24 to Oct 28 2008 | 14.5 | 15.0 | 72.5 | 49.8 (0.22) |

†Dimensionless variance shown in brackets

## 4.4 SUMMARY

Wetland-reservoir irrigation system hydraulics were assessed to gain a better understanding of their affect on water availability and treatment, and thereby adapt the design and management of future systems to local environmental conditions. Annual precipitation was greater than normal and than amounts reported by similar studies, indicating that WRISs in Nova Scotia may capture more water but have greater treatment challenges than WRISs in other regions.

Flow rates were examined and a considerable portion the annual tile drainage outflow volume may not have sufficient RT to undergo the desired treatment. Aside from using a larger CTW, design and management options, such as adding additional stop plates to the CTW outlet control structure before high flow events to increase CTW volume, combining the CTW and reservoir into a single system, using a head pond and a lower rate pump to the CTW, or circulating CTW outflow through the CTW again, may help extend RT during high flow events and provide the desired treatment to all flow rates. All potential gains, including those from groundwater intrusion and surface runoff that breaches berms, should be considered when sizing the CTW outlet. Low tile drainage outflow during Jun and Jul indicates a potential to treat alternative farm wastewater during this period, and is the best time to perform maintenance.

A CTW water budget showed significant mean monthly imbalances in W1 and W2. These imbalances are primarily attributed to groundwater intrusion. Any conclusions drawn from the flow data should consider these discrepancies. A site investigation should be conducted to identify the water table depth so that the CTW can be situated to avoid movement between groundwater and wastewater. If an ideal location is unavailable, as was the case in this present study, measures such as a liner, a tile drain around the CTW perimeter, and raising the CTW water level may help reduce movement between groundwater and wastewater.

As predicted by the greater annual precipitation, the reservoir captured more water per area of drained land than other studies, even if tile drainage outflow is the only gain considered. This illustrates the importance of adapting WRIS designs to local environmental conditions to maximize water availability and economic efficiency. While the WRIS was not intended to capture groundwater or surface runoff, the potential to augment water availability by incorporating them is shown.

A bromide tracer study indicated that $t_a$ was similar to $t_n$. This suggests that there are few inactive zones in the CTW and supports the use of a length to width ratio of 10:1 for the CTW shape. This also suggests that overall treatment may be similar to that predicted by the k-C* model, however, a large $\sigma^2$ indicates that there may be a wide range of treatment of individual parcels of water. Additional tracer studies are recommended to characterize RTD over a range of flow rates and water levels.

# CHAPTER 5 - WATER QUALITY ASSESSMENT OF A WETLAND – RESERVOIR WASTEWATER TREATMENT AND REUSE SYSTEM RECEIVING AGRICULTURAL DRAINAGE WATER IN NOVA SCOTIA

## 5.1 INTRODUCTION

A water quality assessment of a WRIS can provide information that can be used to further the process of adapting it to local environmental conditions. The CTW of the WRIS that was constructed at the Bio-Environmental Engineering Center in Truro, Nova Scotia (Chapter 3) was designed using localized *k*s and historical flow and water quality data in the k-C* model (Section 3.3). Assessing CTW performance will help determine if these adaptations adequately account for the effects that temperature (Kadlec and Reddy, 2001; Wood et al., 1999) and tile drainage flow rates may have on treatment. Treatment during high flow events is expected to be critical, as high flow events create periods of high pollutant loading and may release stored pollutants in the CTW (Raisin et al., 1997).

Within a WRIS pathogens are a pollutant of concern, as they have the potential to cause waterborne illness via irrigation water. Pathogen treatment has not been reported in other WRIS studies (Allred et al., 2003; Tan et al. 2007), possibly because these studies irrigated corn and soy beans, which are not consumed raw and therefore have a small risk of transmitting waterborne illness. Studies on CTWs receiving tile drainage water often do not report pathogen treatment, as N and P are the main pollutants of concern. Phosphorus is also a pollutant of concern within a WRIS, as it is usually the growth-limiting nutrient if freshwater systems (CCME, 2007; Goldman et al., 1990). It can cause challenges by contributing to algae blooms that obstruct irrigation equipment or block solar disinfection. Nitrogen and P may be considered beneficial as fertilizer in irrigation water, although they may not be present in large enough quantities to provide an economic benefit. Assessing reservoir water quality will indicate if additional treatment (Gannon et al., 2005; Mallin et al., 2002; Murphy et al. 2010) or contamination occurs in the reservoir.

Wetland-reservoir irrigation systems can also be a source of environmental pollution, although likely no more than the preceding condition of discharging un-treated drainage water into the environment. Environmental pollution can occur via discharge from an open WRIS, such as the present WRIS because of an undersized reservoir (Section 4.3.4), via field runoff after irrigation, or via seepage. Pathogens may impair water for many uses, as they cause waterborne illness. Nitrogen may adversely affect aquatic life or contaminate drinking water supplies. Nitrogen and P may contribute to eutrophication of saltwater and freshwater system, respectively (CCME, 2007; Goldman et al., 1990).

The objectives discussed in this chapter are to:

(i) assess CTW performance by determining annual mass reductions of $NO_3^-$-N, TP, SRP, and *E. coli*;
(ii) determine *k*s for $NO_3^-$-N, TP, SRP, and *E. coli*; and
(iii) assess reservoir water quality for $NO_3^-$-N, TP, SRP, and *E. coli*.

These objectives are achieved by means of assessing of water quality and flow rates at the tile drainage system outlet, CTW outlet, and reservoir outlet. Recommendations for adapting the design and operation of future systems in Nova Scotia, are discussed.

## 5.2 METHODOLOGY

The WRIS design is described in Chapter 3. The WRIS was monitored from Nov 1, 2007 to Dec 31, 2009. Meteorological measurements and flow monitoring are described in Sections 4.2.1 and 4.2.2, respectively. Monitoring flow, water quality, and meteorological parameters beyond a single year is recommended to support k and $A_W$:$A_D$ recommendations and to gain a better understanding of loading and treatment variability as the WRIS ecologically matures

### 5.2.1 Water Quality Monitoring

Water quality was monitored from Nov 1, 2007 through Dec 31, 2008 at both the W1 and W2 inlets and outlets and at the reservoir outlet. From Nov 1, 2007 through Dec 14, 2007 composite water samples were collected weekly from each of the seven tile drainage lines that enter Hut I and were analyzed to determine the $C_{in}$s for $NO_3^-$-N, TP, and SRP. These composite samples were comprised of equal parts of six daily samples. The daily samples were collected automatically by ISCO 6700 portable samplers (Teledyne Isco Inc., Lincoln, NE). Flow weighted average pollutant concentrations from the entire tile drainage system were calculated using eqn 8. Tile drainage outlet water samples that were analyzed for *E. coli* during this period were collected manually from the seven tile drainage lines and a flow weighted average for entire tile drainage system was calculated as:

$$C_{in} = \frac{\sum_{i=1}^{7}(C_i * V_i)}{\sum_{i=1}^{7} V_i} \qquad [8]$$

Where:

$C_{in}$ = Flow weighted average concentration at tile drainage system outlet (concentration at the constructed treatment wetland inlet) (mg $L^{-1}$ or CFU 100 $mL^{-1}$)

$C_i$ = Concentration in composite sample from drain i (mg $L^{-1}$ or CFU 100 $mL^{-1}$) and

$V_i$ = Total flow volume from drain i over six day period (L)

From Dec 14, 2007 through Dec 31, 2008 both W1 and W2 inlet water samples were collected manually from sampling ports (Fig 3.4) twice per week and analyzed to determine all the $C_{in}$s. During high flow events the outlet water sampling frequency from W1 and W2 was increased to every 6 h. During the entire monitoring period W1 and W2 outlet water samples were collected automatically by ISCO 6712 portable samplers (Teledyne Isco Inc., Lincoln, NE) twice per week and analyzed to determine the $C_{out}$s of $NO_3^-$-N, TP, and SRP. Wetland 1 and W2 outlet water samples that were analyzed for *E. coli* were collected manually twice per week. During high flow events W1 and W2 outlet water sampling frequency was increased to up to every 6 h.

Reservoir water samples were collected manually from the reservoir spillway every two weeks. Water quality may vary throughout the reservoir. The reservoir spillway was used as the sampling point because it was easy to collect samples from a consistent point, it was the furthest point from the CTW outlet so any treatment or contamination would be evident, it was near the irrigation system site intake pipe so it provided a good indication of irrigation water quality, and it provided a direct measure of effluent water quality.

All water sample collection and storage procedures followed those of the Nova Scotia Agricultural College and Clesceri et al. (2005). Table 5.1 presents the number of samples analyzed for each pollutant in 2008 at each sampling location.

### 5.2.2 Analytical Methods

All water samples were analyzed at the Environmental Microbiology Research Laboratory at the Nova Scotia Agricultural College for $NO_3^-$-N, TP, SRP, and *E. coli*. Water samples were analyzed for $NO_3^-$-N and SRP by means of ion chromatography according to APHA Method 4110 C: Single-Column Ion Chromatography with Electronic Suppression of Eluent Conductivity and Conductimetric Detection (Clesceri et al., 2005). Samples were syringe filtered with 0.45 µm nitrocellulose membrane filters (Millipore Corp., Billerica, MA) and analyzed using a Waters Ion Chromatography System (Waters Canada Ltd., Mississauga, ON). Water samples were analyzed for TP by means of spectrophotometry according to the Ascorbic Acid Method outlined by Kovar (2003), which is based on APHA Method 4500-P E: Ascorbic Acid Method (Clesceri et al., 2005). Samples were first digested according to a procedure based on APHA Method 4500–P B4: Sulfuric Acid-Nitric Acid Digestion (Clesceri et al., 2005). Absorbance was then measured using a DR/2000 spectrophotometer (Hach Co., Loveland, CO). Water samples were analyzed for *E. coli* by means of membrane filtration using m-ColiBlue24® broth as a growth medium according to Hach Method 10029 (Hach Co., 1999). The detection limits of these analyses were 0.04 mg $L^{-1}$ for $NO_3^-$-N, 0.05 mg $L^{-1}$ for TP, 0.10 mg $L^{-1}$ for SRP, and 0 CFU 100 $mL^{-1}$ for *E. coli*. Field replicates collected at the same time and location were used to estimate sampling and laboratory analysis precision. Sources of error may include human error during sample collection, storage, and analysis, such as container contact with the reservoir spillway bed and not storing samples at appropriate temperatures, or instrumentation error, such as false calibration.

**Table 5.1** Number of water samples analyzed for nitrate-nitrogen ($NO_3^-$-N), total phosphorus (TP), soluble reactive phosphorus (SRP), and *Escherichia coli* (*E. coli*) in 2008 at each sampling location.

| | Number of Water Samples Analyzed | | | |
|---|---|---|---|---|
| **Sampling Location** | **$NO_3^-$-N** | **TP** | **SRP** | ***E. coli*** |
| Wetland 1 Inlet | 103 | 47 | 103 | 94 |
| Wetland 1 Outlet | 174 | 67 | 174 | 149 |
| Wetland 2 Inlet | 164 | 70 | 164 | 137 |
| Wetland 2 Outlet | 216 | 92 | 216 | 152 |
| Reservoir Spillway | 55 | 32 | 55 | 64 |

## 5.3 RESULTS AND DISCUSSION

### 5.3.1 Tile Drainage Water Quality

After Dec 14, 2007 the mean water quality at the W1 and W2 inlets was used as the tile drainage outlet water quality. Loads were calculated hourly and missing concentration points were assumed equal to the nearest known concentration.

#### 5.3.1.1 *Nitrate- nitrogen*

Nitrate-nitrogen concentrations and loads from the tile drainage system are shown in Fig 5.1. The mean $NO_3^-$-N concentration in tile drainage water in 2008 was 6.58 mg $L^{-1}$ (Table 5.2). Thiagarajan (2005) reported a mean concentration of 5.64 mg $L^{-1}$ from the same field as the present study in 2002-2004. Other studies on CTWs receiving tile drainage have reported $NO_3^-$-N concentrations between 0.79 and 20.30 mg $L^{-1}$ (Coote and Gregorich, 2000; Fink and Mitsch, 2004; Kovacic et al., 2000; 2006; Tanner et al., 2005), depending on the site, year, or cropping system.

The annual load was 24.4 kg $ha^{-1}$ (Table 5.2). Thiagarajan (2005) reported a mean annual load of 14.4 kg $ha^{-1}$ from the same field site from 2002-2004. Other reported annual loads range from 5.0 to 99.0 kg $ha^{-1}$ (Bakhsh et al., 2005; Fink and Mitsch, 2004; Jordan et al., 2003; Kovacic et al., 2000; 2006; Tanner et al., 2005).

Nitrate-nitrogen concentrations in tile drainage outflow exceeded the water quality guideline for the protection of aquatic life (3 mg $L^{-1}$) during all of 2008. Therefore some aquatic life, fish and frog eggs are the most sensitive to $NO_3^-$ concentrations (CCME, 2003), may be adversely affected in the CTW, reservoir, or in a surface waterbody if untreated effluent is discharged into it. Therefore, there may be a need to reduce $NO_3^-$-N concentrations in open WRISs, such as that of the present study that discharged reservoir into the environment because the reservoir was undersized (Section 4.3.4). Preventing water pollution from WRIS effluent is more rationale, in addition to maximizing water availability and economic efficiency (Section 4.3.4), for designing a closed system that captures and retains all gains.

Nitrate-nitrogen concentrations in tile drainage outflow exceeded the drinking water quality guideline (10 mg $L^{-1}$) during a total of 3.7 d in 2008, which accounted for 5.9% of the load in 2008. Therefore there was not a concern of contaminating drinking water supplies.

Nitrate-nitrogen concentrations in tile drainage outflow did not exceed the irrigation water quality guideline (30 mg $L^{-1}$) during 2008. Therefore, un-treated tile drainage water could be used directly for irrigation from an N perspective. Therefore CTWs as part of closed WRISs in Nova Scotia may only need to be sized to treat *E. coli*.

The highest $NO_3^-$-N concentrations were expected during the first storm event after manure application. However they were recorded during an 18.4 mm storm event immediately before manure application (Fig 5.1) when 5.5% of the load in 2008 occurred over 3 d. Lower flow rates (0-2 $m^3$ $h^{-1}$) accounted for 76% of the load in 2008 (Fig 5.2). This indicates that treatment during high flow periods may not be as critical as expected. Raisin et al. (1997) reported a similar distribution of 68% of annual TN loading during background flow and 32% during storm events. The distribution supports using the monthly mean *Q* (extrapolated to annual flow) during the month when the peak load occurred (1.9 $m^3h^{-1}$), as was used in the present study (Section 3.3.2).

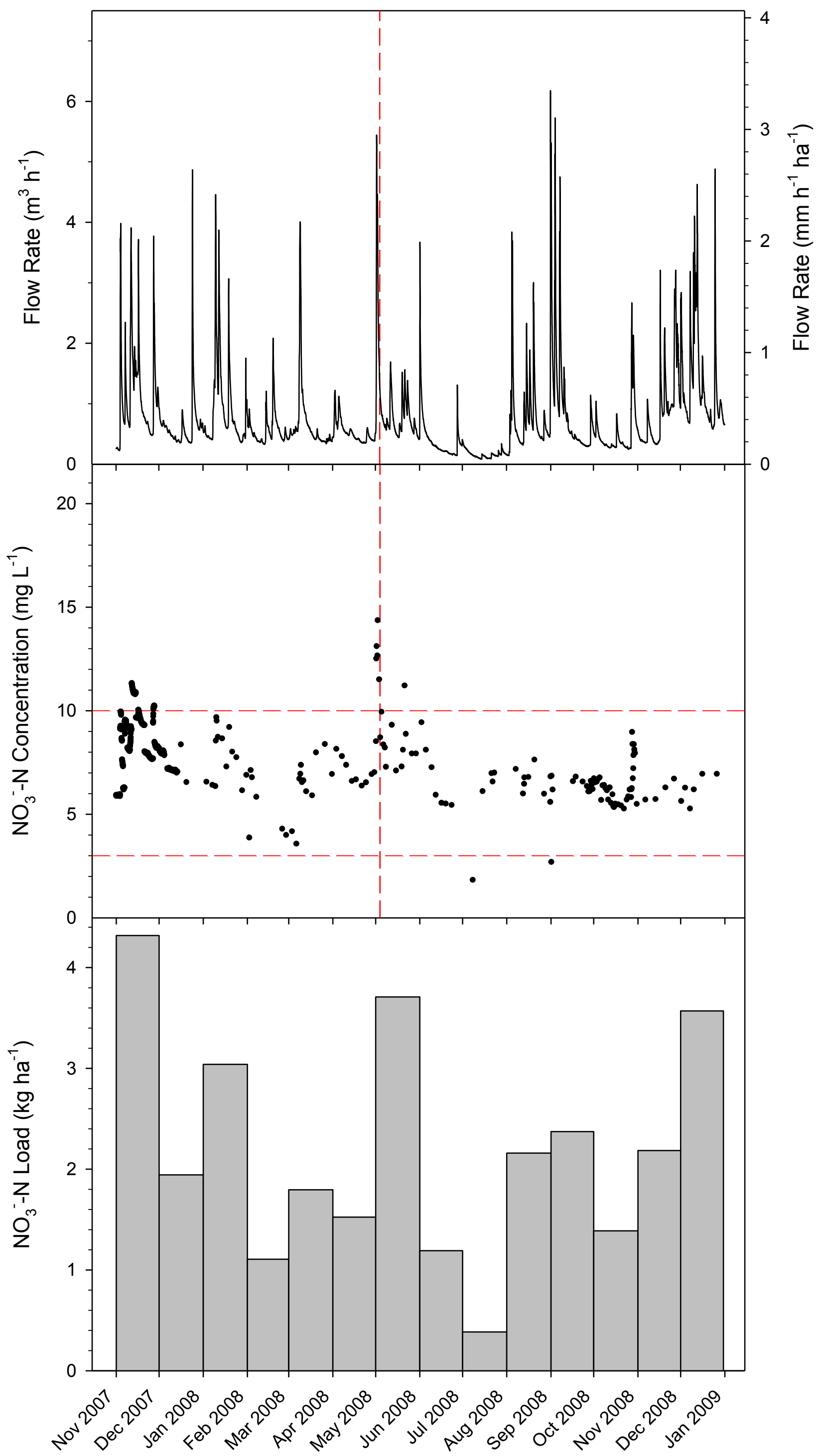


**Figure 5.1** Flow rates ($m^3\ h^{-1}$), flow rates standardized as a depth over the constructed treatment wetland surface area per drained field surface area ($mm\ h^{-1}\ ha^{-1}$), and nitrate-nitrogen ($NO_3^-$-N) concentrations ($mg\ L^{-1}$) and loads ($kg\ ha^{-1}$) from tile drainage outflow. The date of manure application is shown by the vertical dashed line. The protection of aquatic life and drinking water quality guidelines are shown by the horizontal dashed lines.

**Table 5.2** Mean, maximum, and minimum nitrate-nitrogen ($NO_3^-$-N) concentrations and *Escherichia coli (E. coli)* levels, and annual loads (kg $ha^{-1}$) from tile drainage outflow into the constructed treatment wetlands in 2008.

| | Concentration ($mg\ L^{-1}$) | | | Annual Load ($kg\ ha^{-1}$) |
|---|---|---|---|---|
| | **Mean** | **Max** | **Min** | |
| **$NO_3^-$-N** | 6.58 | 14.35 | 1.81 | 24.4 |
| ***E. coli*** | 112† | 3200† | 0† | 39.1 x $10^{8}$‡ |

† Reported as CFU 100 $mL^{-1}$
‡ Reported as CFU $ha^{-1}$

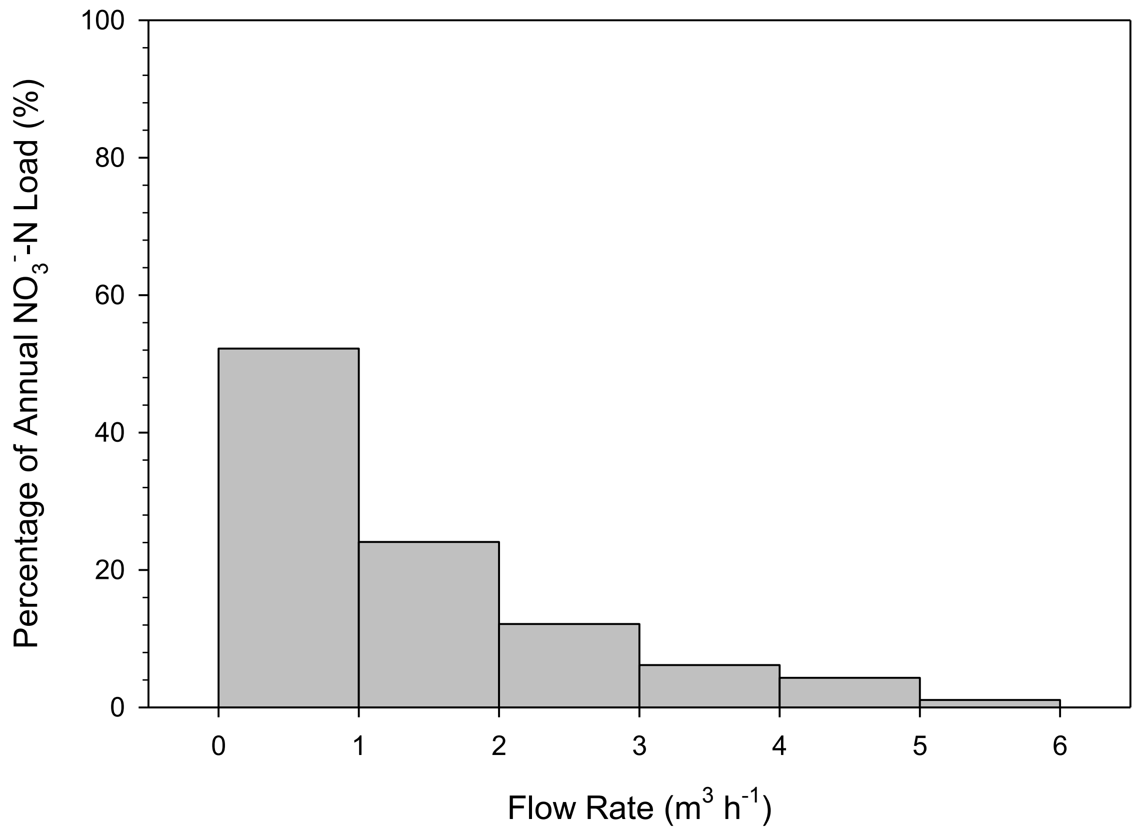


**Figure 5.2** Percentage (%) of the annual nitrate-nitrogen ($NO_3^-$-N) load in tile drainage outflow by flow rate ($m^3\ h^{-1}$) during 2008.

5.3.1.2 *Total Phosphorus and Soluble Reactive Phosphorus*

Total P and SRP concentrations were below detectable limits in 85 and 100%, respectively, of samples at the tile drainage outlet in 2008. Therefore a more accurate analysis method for TP and SRP is recommended. A maximum TP concentration of 0.35 mg $L^{-1}$ was measured in 2008. Concentrations were lower than expected because Thiagarajan (2005) reported mean TP and SRP concentrations of 0.33 mg $L^{-1}$ and 0.16 mg $L^{-1}$, respectively, from the same field as the present study in 2003-2004 and 2002-2004, respectively. Other studies on CTWs receiving tile drainage have reported TP concentrations between 0.01 and 1.3 mg $L^{-1}$ (Fink and Mitsch, 2004; Raisin et al., 1997; Reinhardt et al., 2002; Tanner et al., 2005). The TP and SRP concentrations in tile drainage water that are below detectable limits indicate that there is little concern of eutrophication in the CTW, reservoir, or freshwater environment surrounding the WRIS.

5.3.1.3 *Escherichia coli*

*Escherichia coli* concentrations in and loads from the tile drainage outflow are shown in Fig 5.3. The mean *E. coli* level in tile drainage water in 2008 was 112 CFU 100 $mL^{-1}$ (Table 5.2). Thiagarajan (2005) reported a mean level of 2891 CFU 100 $mL^{-1}$ from the same field as the present study in 2003-2004.

The annual load was 3.9 x $10^9$ CFU $ha^{-1}$ (Table 5.2). Thiagarajan et al. (2005) reported an annual load of 9.8 x $10^9$ CFU $ha^{-1}$ from the same field from 2002-2004, despite receiving a lower manure application rate (Table 3.1). This illustrates the potential year to year variability in tile drainage flow and water quality.

The *E. coli* levels in tile drainage outflow exceeded the drinking water quality guideline (0 CFU 100 $mL^{-1}$) during a total of 293.8 d in 2008. The *E. coli* levels in tile drainage outflow exceeded the irrigation water quality guideline (100 CFU 100 $mL^{-1}$) during a total of 78.8 d in 2008, which accounted for 88.0% of the load in 2008. Therefore using un-treated tile drainage water for irrigation purposes may pose a potential health risk, especially for crops consumed raw.

The highest *E. coli* levels were expected during the first storm event after manure application. However they were recorded in the fall (Fig 5.3). Lower flow rates (0-2 $m^3 h^{-1}$) accounted for 71% of the load in 2008 (Fig 5.4). This supports the finding from the distribution of $NO_3^-$-N load by flow rate that treatment during high flow periods may not be as critical as expected (Section 5.3.1.1). The distribution also supports using the monthly mean *Q* (extrapolated to annual flow) during the month when the peak load occurred (1.9 $m^3h^{-1}$), as was used in the present study (Section 3.3.2).

### 5.3.2 Constructed Treatment Wetland Water Quality

Constructed treatment wetland water quality was assessed to determine event, seasonal, and annual treatment. The treatment of individual parcels of water was not assessed because a tracer study conducted in W2 indicated that plug-flow hydraulics were not followed, that is, parcels of water reside in the CTW for varying times and therefore varying treatment of may be expected (Section 4.3.5). Event, seasonal, and annual treatment can be assessed because they consider the mean treatment of individual parcels of water. Mass reductions were used to quantify treatment using the following equation:

$$MR = 1 - \frac{\sum(C_{out} * V_{out})}{\sum(C_{in} * V_{in})} * 100\% \qquad [9]$$

Where:

$MR$ = Mass reduction (%)

$C_{out}$ = Concentration at the constructed treatment wetland outlet (mg $L^{-1}$)

$C_{in}$ = Concentration at the constructed treatment wetland inlet (mg $L^{-1}$)

$V_{out}$ = Volume of outflow from the constructed treatment wetland (L)

$V_{in}$ = Volume of inflow to the constructed treatment wetland (L)

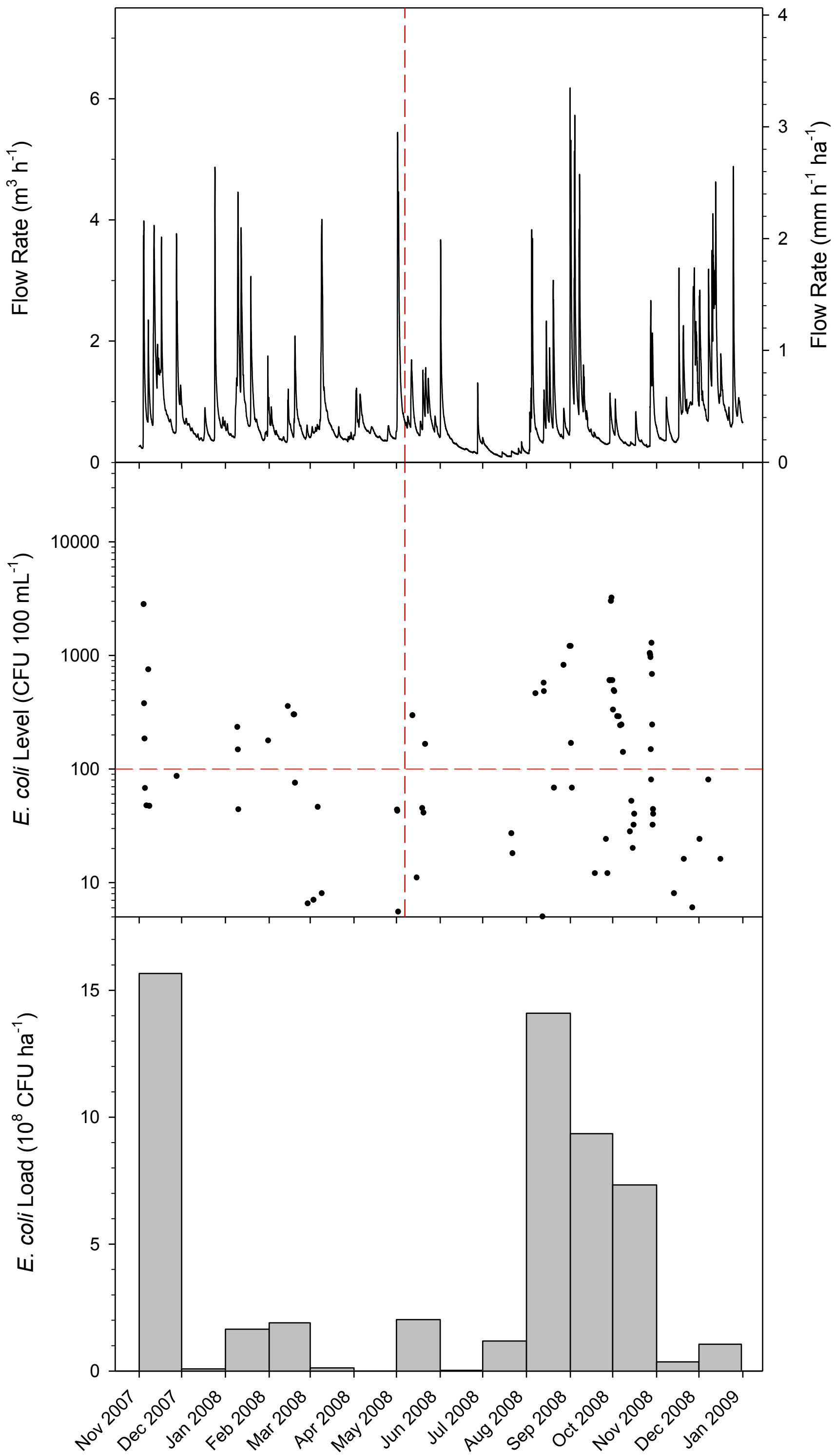


**Figure 5.3** Flow rates ($m^3\ h^{-1}$), flow rates standardized as a depth over the constructed treatment wetland surface area per drained field surface area ($mm\ h^{-1}\ ha^{-1}$), and *Escherichia coli* (*E. coli*) levels (CFU 100 $mL^{-1}$) and loads ($10^8$ CFU $ha^{-1}$) from tile drainage outflow. The date of manure application is shown by the vertical dashed line. The irrigation water quality guideline is shown by the horizontal dashed line.

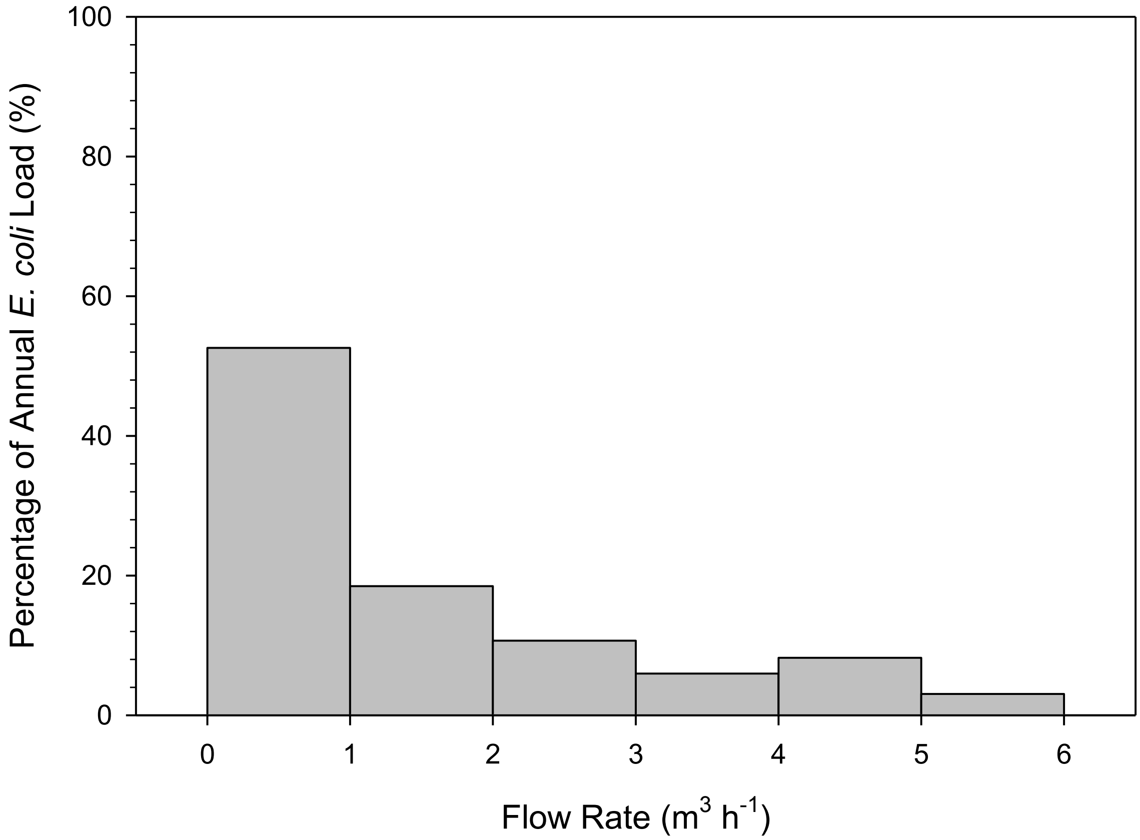

**Figure 5.4** Percentage (%) of the annual *Escherichia coli* (*E. coli*) load in tile drainage outflow by flow rate ($m^3$ $h^{-1}$) during 2008.

A seasonal slow-down in treatment is expected because temperature affects several biogeochemical processes and thus treatment (Kadlec and Reddy, 2001; Wood et al., 1999). The pollutant loading and loss effects from unaccounted gains and losses (Section 4.3.3) are integrated into the treatment assessment in W2. However, their effects were considered too great to accurately assess treatment in W1. Treatment was expected to be reduced by half after Aug 1, 2008 when all tile drainage outflow was directed to W2, effectively doubling *Q*. Loads were calculated on an hourly basis with missing concentration data assumed equal to the nearest known concentration.

5.3.2.1 *Nitrate-nitrogen*

Wetland 2 inlet and outlet flow rates and $NO_3^-$-N concentrations and loads in and out of W2 are shown in Fig 5.3. The mean, maximum, and minimum $NO_3^-$-N concentrations and annual loads in and out of W2 in 2008 are shown in Table 5.3. Nitrate-nitrogen mass reduction in W2 was 67.6% in 2008, which is greater than the mass reductions reported in WRISs (Baker et al., 2004; Tan et al., 2007) and in similar studies on CTWs receiving drainage water that had similar $A_W$ to $A_D$ ratios (Table 2.2). This may support the design features, such as a length to width ratio of 10:1 (Section 3.3.2) but is also may also be attributed to the specific loading regime at the site. These results support the use of CTWs as an effective technology for mitigating water pollution from agricultural drainage water.

Nitrate-nitrogen concentrations in W2 outflow exceeded the water quality guideline for the protection of aquatic life during a total of 86.1 d in 2008, which accounted for 65.6% of the load in 2008. While this was an improvement over the tile drainage water quality (5.3.1.1), there may still be a concern for the protection of aquatic life. A range of aquatic life was observed in the CTW, including insects, snails, and frogs. Nitrate-nitrogen concentrations in W2 outflow remained below drinking and irrigation water quality guidelines, as was expected based on the tile drainage water quality (Section 5.3.1.1).

5.3.2.2 *Total Phosphorus and Soluble Reactive Phosphorus*

As expected, based on the concentrations in tile drainage outflow (Section 5.3.1.2), TP and SRP concentrations were below detectable limits in all W2 outflow water samples. Therefore no conclusions regarding TP and SRP retention can be made and there is little concern of eutrophication in the CTW. Water in W2 did not appear turbid, as would be expected in a eutrophic environment. However, no tests were conducted to classify the trophic status of the CTW. Higher concentrations of TP may be observed in CTWs that capture surface runoff because P adheres to soil particles transported by surface runoff.

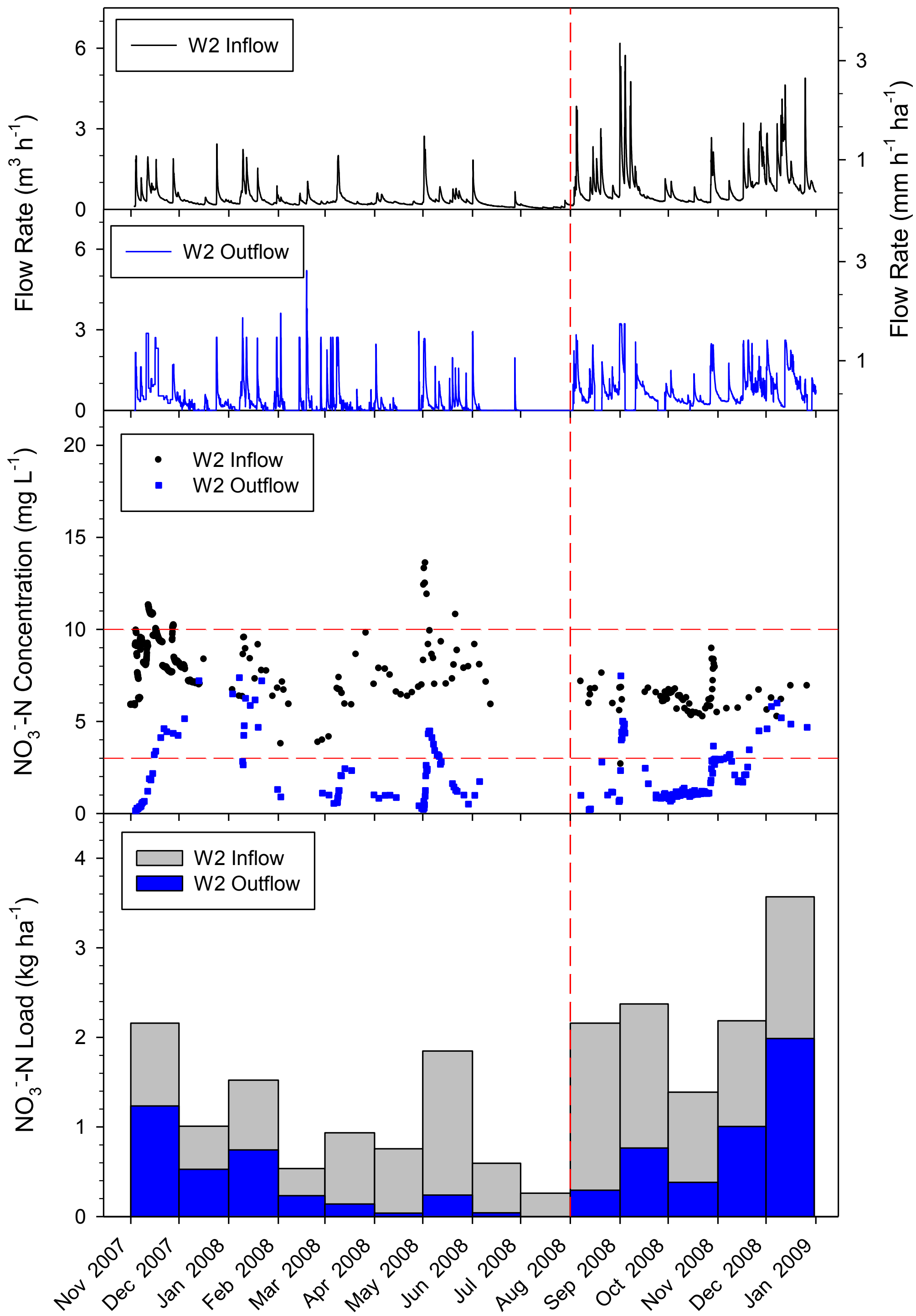


**Figure 5.5** Flow rates (m3 h-1), flow rates standardized as a depth over the constructed treatment wetland surface area per drained field surface area (mm h-1 ha-1), and nitrate-nitrogen (NO3--N) concentrations (mg L-1) and loads (kg ha-1) at the Wetland 2 (W2) inlet and outlet. The date when all inflow was directed to Wetland 2 is shown by vertical dashed line. The protection of aquatic life and drinking water quality guidelines are shown by the horizontal dashed lines.

**Table 5.3** Mean, maximum, and minimum nitrate-nitrogen ($NO_3^-$-N) concentrations (mg $L^{-1}$) and *Escherichia coli (E. coli)* levels (CFU 100 $mL^{-1}$), annual loads (kg $ha^{-1}$ $y^{-1}$ or CFU $ha^{-1}$ $y^{-1}$), and annual mass reductions (%) from Wetland 2 inflow and outflow in 2008.

| | Concentration (mg $L^{-1}$) Mean | Max | Min | Annual Load (kg $ha^{-1}$) | Annual Mass Reduction (%) |
|---|---|---|---|---|---|
| **$NO_3^-$-N In** | 6.7 | 13.6 | 2.7 | 18.1 | 67.6 |
| **$NO_3^-$-N Out** | 2.2 | 7.5 | 0.2 | 5.9 | |
| ***E. coli* In** | 122† | 3200† | 0† | 36.2 x $10^8$‡ | 63.3 |
| ***E. coli* Out** | 42† | 1160† | 0† | 13.3 x $10^8$‡ | |

† Reported as CFU 100 $mL^{-1}$
‡ Reported as CFU $ha^{-1}$

5.3.2.3 *Escherichia coli*

Wetland 2 inlet and outlet flow rates, and *E. coli* levels and loads are shown in Fig 5.4. The mean, maximum, and minimum *E. coli* levels and annual load in and out of W2 in 2008 are shown in Table 5.4. *Escherichia coli* reduction in W2 was 63.3% in 2008. This supports the use of CTWs as an effective technology for reducing pathogen in tile drainage water. However, *E. coli* levels must be examined to determine if water quality is improved enough to be used for irrigation.

*Escherichia coli* levels in W2 outflow exceeded the drinking water quality guideline during a total of 267.9 d in 2008. This is an improvement over tile drainage water quality (Section 5.3.1.3); however, there may still be a concern of contaminating drinking water supplies. It is not be feasible to use CTWs to reduce *E. coli* levels to meet drinking water guidelines because the *C** greater than the guideline.

*Escherichia coli* levels in W2 outflow exceeded the irrigation water quality guideline during a total of 28.8 d in 2008, which accounted for 63.1% of the load in 2008. This shows the potential of CTWs as an effective technology for improving tile drainage water so that it can be used for irrigation.

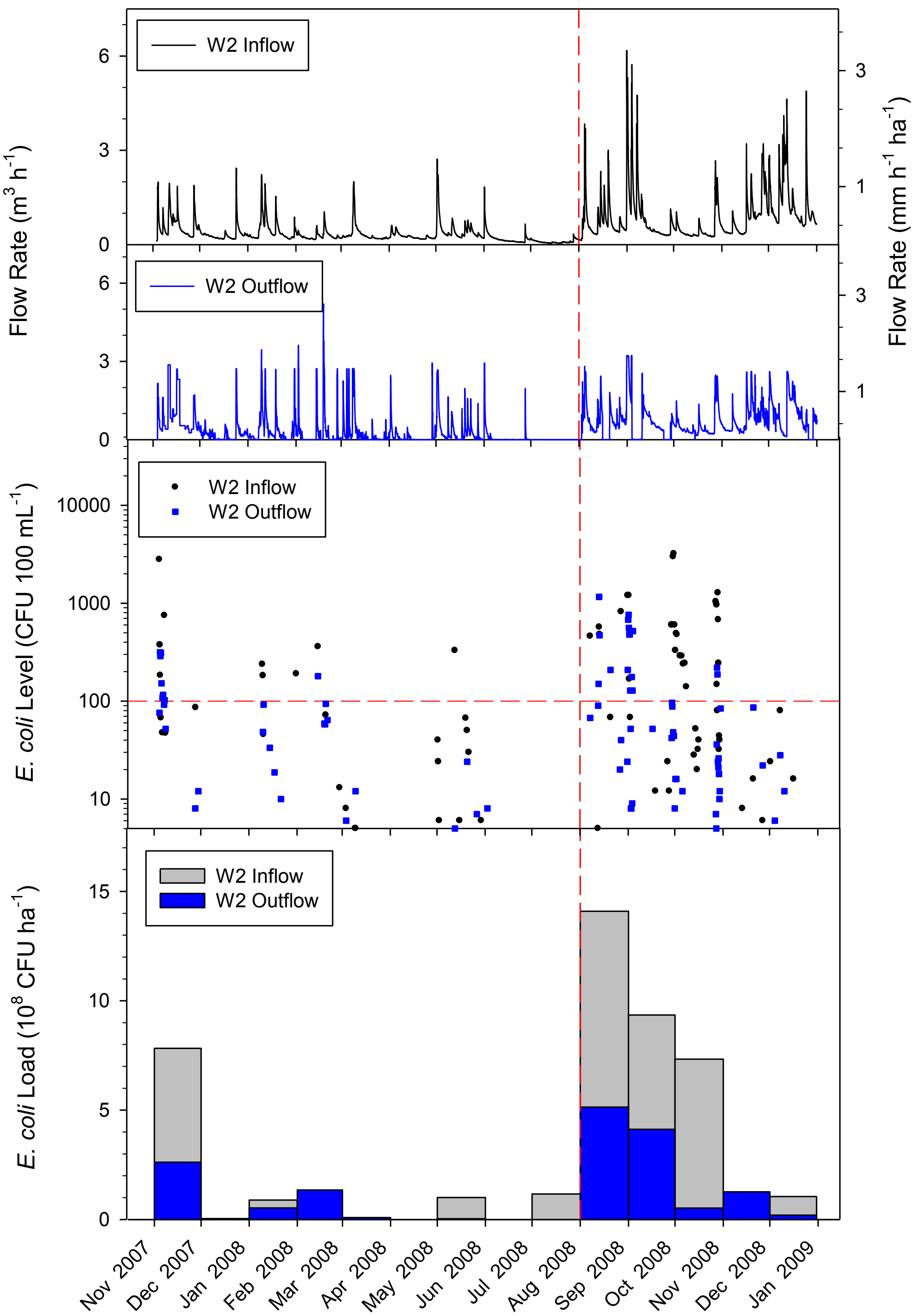


**Figure 5.6** Flow rates (m$^3$ h$^{-1}$), flow rates standardized as a depth over the constructed treatment wetland surface area per drained field surface area (mm h$^{-1}$ ha$^{-1}$), and *Escherichia coli* (*E. coli*) levels (CFU 100 mL$^{-1}$) and loads (10$^8$ CFU ha$^{-1}$) at the Wetland 2 (W2) inlet and outlet. The date when all inflow was directed to Wetland 2 is shown by vertical dashed line. The irrigation water quality guideline is shown by the horizontal dashed line.

### 5.3.3 Reservoir Water Quality

Reservoir water quality was assessed to determine if reservoir water can be safely used for irrigation or safely discharged into the environment. Assessing reservoir water quality is also part of determining if the surface channel or reservoir provide additional treatment. However, reservoir outflow rates must be monitored to properly assess treatment in the reservoir and they were not. Unaccounted gains or losses, as occurred in the CTW (Section 4.3.3), may also have occurred in the reservoir.

#### 5.3.3.1 *Nitrate-nitrogen*

Nitrate-nitrogen concentrations in the reservoir outflow exceeded the water quality guideline for the protection of aquatic life during a total of 24.8 d in 2008 (Fig 5.5). This is an improvement over CTW outlet water quality (Section 5.3.2.1) and may be attributed to additional treatment in the reservoir or dilution from precipitation or other unaccounted gains. The periods when the guideline was exceeded may not be a major concern because concentrations did not exceed the guideline by much, elevated concentrations occurred in the winter when irrigation does not occur, and a range of aquatic life was observed in the reservoir, including insects, snails, and frogs. The reservoir was likely not established long enough attain a fish population.

Nitrate-nitrogen concentrations in reservoir outflow remained below drinking and irrigation water quality guidelines, as was expected based on the CTW outflow water quality (Section 5.3.2.1). Therefore there was not a concern of contaminating drinking water supplies.

#### 5.3.3.2 *Total phosphorus and Soluble Reactive Phosphorus*

As expected, based on the concentrations in tile drainage outflow (Section 5.3.1.2) and W2 outflow (Section 5.3.2.2), TP and SRP concentrations were below detectable limits in all reservoir outflow water samples. Therefore there is little concern of eutrophication in the reservoir or the freshwater environment surrounding the WRIS. Reservoir water did not appear turbid, as would be expected in a eutrophic environment. However, no tests were conducted to classify the trophic status of the reservoir. This also indicates that TP and SRP loading from unaccounted gains, for example surface runoff, are not a concern.

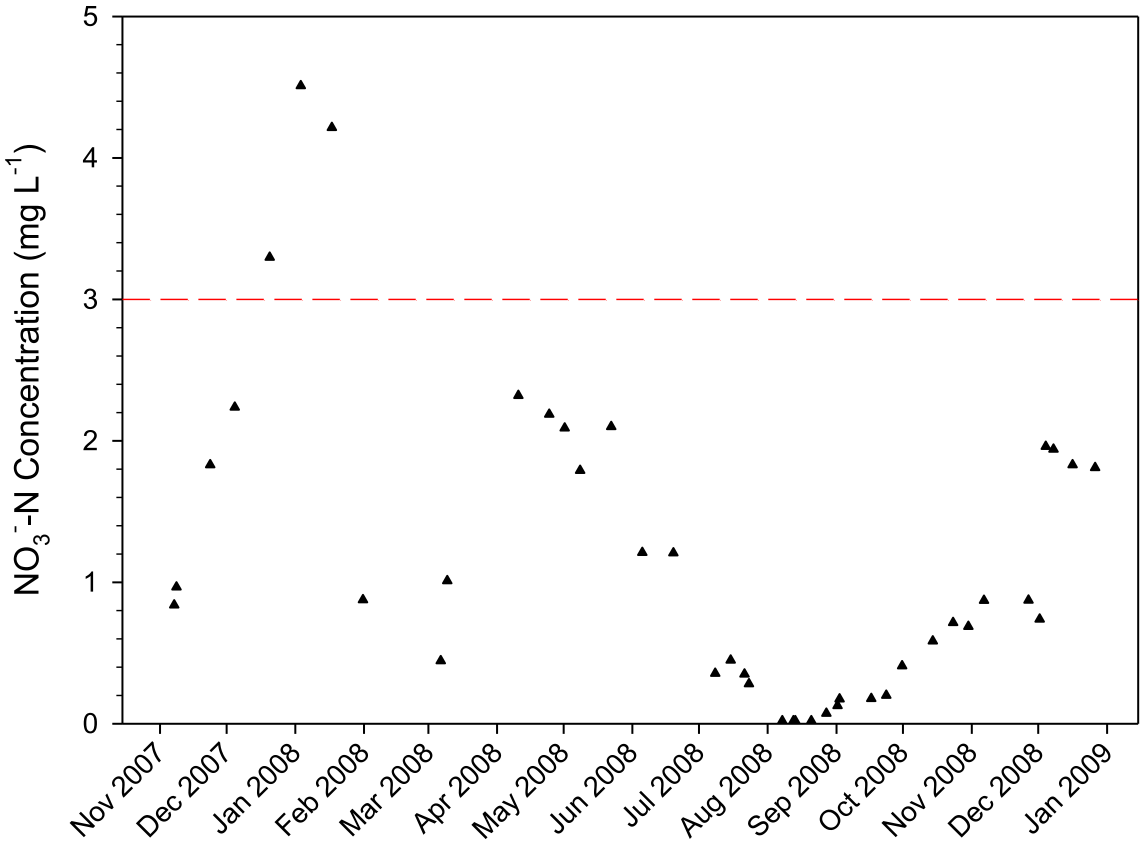


**Figure 5.7** Nitrate-nitrogen ($NO_3^-$-N) concentrations (mg $L^{-1}$) in reservoir outflow from Nov 2007 through Jan 2009. The protection of aquatic life water quality guideline is shown by the horizontal dashed line.

**Table 5.4** Mean, maximum, and minimum nitrate-nitrogen ($NO_3^-$-N) concentrations (mg $L^{-1}$) and *Escherichia coli (E. coli)* levels (CFU 100 $mL^{-1}$) in reservoir outflow in 2008.

| | Mean | Max | Min |
|---|---|---|---|
| **$NO_3^-$N (mg $L^{-1}$)** | 1.21 | 4.51 | 0.02 |
| ***E. coli* (CFU 100 $mL^{-1}$)** | 178$^{\dagger}$ | 2700$^{\dagger}$ | 0$^{\dagger}$ |

5.3.3.3 *Escherichia coli*

*Escherichia coli* levels in the reservoir outflow exceeded the irrigation water quality guideline during a total of 118.3 d in 2008 (Fig 5.6). This is an improvement over CTW

outlet water quality (Section 5.3.2.1) and may be attributed to additional treatment in the reservoir or dilution from precipitation or other unaccounted gains. Elevated levels were observed during the warm season and were typically greater than the preceding and coinciding levels in the W1 and W2 outflow. This increase is attributed to surface runoff or contamination from wildlife, as waterfowl were regularly observed at the reservoir during the warm season. This illustrates a challenge to any technology that does not treat irrigation water immediately before use. Therefore it may not be safe to use reservoir water for irrigation of crops consumed raw before immediately before harvest.
There may also be a risk of polluting drinking water supplies during this period.
Increases in *E. coli* levels were not observed after periods of high loading.

### 5.3.4 First-Order Areal Uptake Rate Constants

First-order areal uptake rate constants were assessed to determine if those used in the present study (Section 3.3.2) are applicable to CTWs receiving dynamic pollutant loading in Nova Scotia. Rate constants need to be assessed in local environmental conditions because temperature (Kadlec and Reddy, 2001; Wood et al., 1999) and vegetation (Bachand and Horne, 1998; Kadlec, 2008) may affect several biogeochemical processes and thus treatment. A seasonal slow-down in treatment is expected in colder climates (Kadlec and Reddy, 2001; Wood et al., 1999), such as that of Nova Scotia, therefore CTWs in Nova Scotia may need to be larger, and thus *k*s smaller, than those in warmer climates to provide the same treatment. The CTWs that were part of existing WRISs were not sized using the k-C* model (Allred et al., 2003; Tan et al., 2007). Carleton et al. (2001) demonstrated that *k*s for CTWs receiving dynamic pollutant loading were similar to *k*s for CTWs receiving typical wastewaters. Kadlec and Knight (1996) summarize the *k*s of 72 wetlands receiving a range of wastewaters at a range of locations (Table 5.5). However, the present study used *k*s generated from a CTW receiving livestock wastewater in Nova Scotia (Table 5.5) (Jamieson et al., 2007) because they may be specific to the environmental conditions of Nova Scotia. It was expected that the *k*s generated from the present study will be similar the mean monthly *k*s adjusted for dilution or concentration effects that were generated by Jamieson et al. (2007).

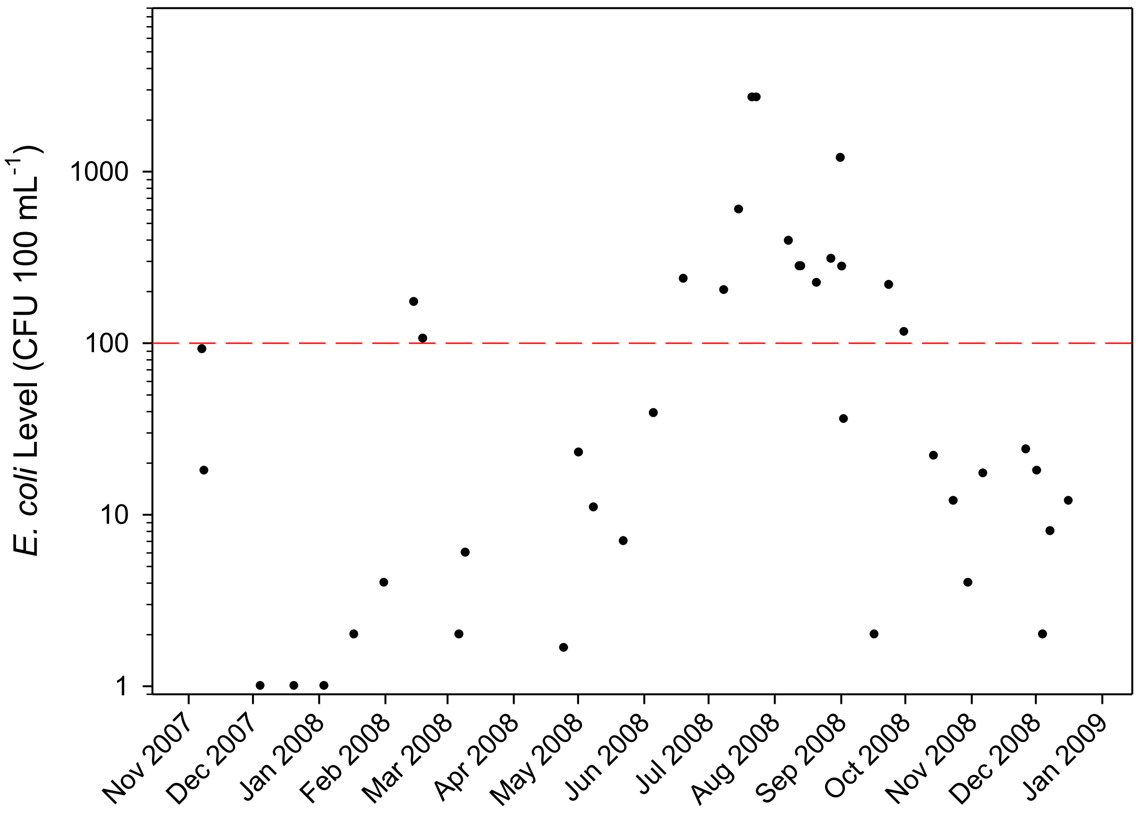


**Figure 5.8** *Escherichia coli* (*E. coli*) levels (CFU 100 mL-1) in reservoir outflow. The irrigation water quality guideline is shown by the horizontal dashed line.

First-order areal uptake rate constants are presented in Table 5.5 and were generated by rearranging eqn 1. Long-term $C_{in}$, $C_{out}$, and $Q$ data from W2 in 2008 and an assumption of plug-flow hydraulics were used. While a tracer study indicated that plug-flow hydraulics were not achieved (Section 4.3.5), it may be a reasonable assumption for the purpose of generating *k*s because the tracer study found that $t_a$ was similar to $t_n$, and therefore overall treatment may follow the k-C* model. Data from W1 were not used because the unaccounted gains in W1 (Section 4.3.3) had an unknown affect on water quality and flow data. The *k*s were adjusted to account for the dilution or concentration effects of hydrologic losses or gains by substituting $C_{out}$ with a corrected concentration at the CTW outlet ($C_{out\_cor}$) (eqn 10) using precipitation and evapotranspiration data (Section 4.3.3). The imbalances from the water budget (Section 4.3.3) were not used to adjust *k*s because they may include unaccounted gains or losses, such as surface runoff, that contribute or withdraw unknown pollutant loads.

The mean $k$s and $k_{adj}$s for both $NO_3^-$-N and *E. coli* were lower than the $k$s reported by Kadlec and Knight, as was expected because of a seasonal slow-down in treatment in cold climates, and may be more suitable for designing CTWs in Nova Scotia. They were closer to the $k$s and $k_{adj}$s generated by Jamieson et al. (2007), which may be even more suitable for designing CTWs in Nova Scotia because they were generated from 4 years of data as opposed to 1 year of data from the present study. The $k$s and $k_{adj}$s of the present study were lower than those generated by Jamieson et al. (2007). This is difference is attributed to the relatively large inflow volumes compared to hydrologic gains. Therefore it may be acceptable to use non-adjusted $k$s from literature when sizing CTWs receiving tile drainage water.

The $k$s generated from the present study account for the effects that dynamic pollutant loading and temperature have on treatment. They should be adjusted to temperature using the Arnhennius equation before they are applied to CTWs at other locations. The design of the present CTW used the maximum, non-adjusted $k$s generated by Jamieson et al. (2007) to yield a size that was economically feasible, fit within the allocated land, and did not dry up during low flow periods (Section 3.3.2). However the $k$s should not have been changed from the initial design method that used the mean $k_{adj}$s. The complete recommended application of the k-C* model is presented in Section 5.3.5.

$$Cout\_cor = Cout * \left(\frac{Vin + Vprecip - Vet}{Vin}\right) \qquad [10]$$

Where:

$C_{out}$ = Concentration at the constructed treatment wetland outlet (mg $L^{-1}$)

$C_{out\_cor}$ = Corrected concentration at the constructed treatment wetland outlet (mg $L^{-1}$)

$V_{in}$ = Volume of inflow to the constructed treatment wetland ($m^3$)

$V_{precip}$ = Volume of precipitation over constructed treatment wetland area ($m^3$)

$V_{et}$ = Volume of evapotranspiration over constructed treatment wetland area ($m^3$)

### 5.3.5 Application of the k-C* Model

The proper application of the k-C* model is key to designing efficient CTWs as part of a WRIS. The present water quality assessment provides data that can be used to adapt the CTW to the environmental conditions of Nova Scotia, rather than using data from literature on CTWs in other regions. Detailed historical drainage flow and water quality data is necessary to obtain accurate values for $Q$ and $C_{in}$.

A $k$ value generated from this study should be used. The distribution of annual load by flow rate supports using the maximum monthly mean $Q$ (extrapolated to annual flow) (Sections 5.3.1.1 and 5.3.1.3), however, this yields a non feasible $A$. Therefore, the annual $Q$ should be used, as the recommended k already accounts for the dynamic pollutant loading. In a closed WRIS, the less stringent irrigation water quality guidelines may be used as $C_{out}$s, rather than the drinking or protection of aquatic life water quality guidelines, because environmental pollution from the WRIS is less likely. This may permit a smaller $A$ and will maximize any crop benefit from nutrients in irrigation water. This is more rationale, in addition to maximizing water availability and economic efficiency (Section 4.3.4), and preventing water pollution from WRIS effluent (Section 5.3.3.1), for designing a closed system that captures and retains all gains. A range of $C^*$ for *E. coli* have been used in literature (Kadlec and Knight, 1996). The *E. coli* levels in CTW outflow (Fig 5.6) were frequently below the average value of 45 CFU 100 $mL^{-1}$ recommended by Kadlec and Knight (1996), which was used in the present study (Section 3.3.2). This indicates that a lower $C^*$ could be used, which would result in a smaller $A$. A lower $C^*$ could also be used because the $C^*$ recommended by Kadlec and Knight (1996) is for FC, which encompasses *E. coli*.

In most cases, historical drainage flow and water quality data will be absent, in which case values from the present study could be used. A much easier alternative design method is to use an $A_W$ to $A_D$ ratio, although it is less precise and scientifically sound. The present study recommends an $A_W$ to $A_D$ ratio of 4.5% to achieve an approximate reduction of $NO_3^-$-N and *E. coli* of 70%. This $A_W$ to $A_D$ ratio considers that all flow was directed to W2 after Aug 1, 2008. This recommendation is smaller than those from similar studies (Table 2.2) and is attributed to design features, such as a length to width ratio of 10:1 (Section 3.3.2) and the specific loading regime at the site.

**Table 5.5** Summary of first-order areal uptake rate constants (*k*s) for nitrate-nitrogen ($NO_3^-$-N) and *Escherichia coli* (*E. coli*) for the present study, calculated monthly, and selected studies.

| Reference | *k* Type | First-Order Areal Uptake Rate Constant ($m\ y^{-1}$) Mean | Maximum | Minimum |
|---|---|---|---|---|
| *$NO_3^-$-N* | | | | |
| Present Study | $k$ | 9.2 | 23.8 | 0.9 |
| Jamieson et al. (2007) | $k$ | 1.9† | 12.0† | -2.9† |
| Present Study | $k_{adj}$ | 8.0 | 22.0 | 1.6 |
| Jamieson et al. (2007) | $k_{adj}$ | -0.2† | 4.3† | -15.1† |
| Kadlec and Knight (1996) | $k$ | 30.0†‡ | 54.4†‡§ | 9.6†‡¶ |
| *E. coli* | | | | |
| Present Study | $k$ | 7.7 | 29.7 | -18.7 |
| Jamieson et al. (2007) | $k$ | 11.0† | 54.5† | 0.0† |
| Present Study | $k_{adj}$ | 6.4 | 28.8 | -20.5 |
| Jamieson et al. (2007) | $k_{adj}$ | 8.3† | 29.8† | 0.8† |
| Kadlec and Knight (1996) | $k$ | 83.0†# | 856.0†§# | 27.0†¶# |

† Reported as fecal coliform
‡ From 72 wetland studies
§ $90^{th}$ percentile of annual mean k
¶ $10^{th}$ percentile of annual mean k
# From 23 wetland studies

## 5.4 SUMMARY

A water quality assessment consisting of the collection of water samples and flow monitoring at the tile drainage system outlet and CTW outlet, and collection of water samples at the reservoir outlet was conducted to determine treatment in the CTW, assess *k*s, and determine if reservoir water can be used for irrigation or safely discharged into the environment.

A difference in tile drainage water quality from the present study, a previous study on the field, and similar studies in other regions illustrates the year to year and field to field variability of tile drainage water, and therefore the need to adapt CTW designs to local environmental conditions. Tile drainage water quality identified elevated $NO_3^-$-N concentrations as a potential concern for the protection of aquatic life and elevated *E. coli* levels as a potential concern for safe drinking and irrigation water. TP and SRP were typically below detection limits, therefore a more accurate analysis method for TP and SRP is recommended. In a closed WRIS the CTW may only need to be sized to reduce *E. coli*. The distribution of $NO_3^-$-N and *E. coli* loads by flow rates indicated that the treatment of high flow periods may not as critical as expected.

The CTW was shown to be effective at reducing both $NO_3^-$-N and *E. coli* loads. Water quality successively improved from the tile drainage outlet, to the W2 outlet, to the reservoir outlet, where there was no longer a concern for the protection of aquatic life from $NO_3^-$-N and much less for irrigation from *E. coli*. One exception was elevated *E. coli* levels at the reservoir outlet in the warm season, when the reservoir water could not safely be used for irrigating crops consumed raw. This is attributed to surface runoff or contamination from wildlife and illustrates the challenge to any technology that does not treat irrigation water immediately before use. Monitoring flow at the reservoir spillway is recommended to determine pollutant loads leaving the WRIS.

The *k*s generated from the present study were closer to those generated by from another study in Nova Scotia than to those commonly used in CTW design. First-order areal uptake rate constants that were adjusted to account for the dilution or concentration effects of hydrologic losses or gains were close to *k*s and indicate that it may be acceptable to use non-adjusted *k*s from literature when sizing CTWs receiving tile drainage water. Sizing the reservoir to retain all gains may allow less stringent $C_{out}$s to be used because environmental pollution would be less likely. This may permit a smaller $A$ and will maximize any crop benefit from nutrients in irrigation water. A lower $C^*$ for *E. coli* may be used based on low levels recorded at the CTW outlet. Monitoring flow, water quality, and meteorological parameters beyond a single year is recommended to support k and $A_W$:$A_D$ recommendations and to gain a better understanding of loading and treatment variability as the WRIS ecologically matures.

# CHAPTER 6 - CONCLUSION

Wetland-reservoir irrigation systems have shown potential to mitigate water pollution from agricultural drainage water and also provide an irrigation water supply. The present study adapted a WRIS to Nova Scotia conditions and assessed its function for future implementation in the region.

This WRIS design, construction, and operation serves as a case study that can be consulted when implementing future systems. The primary adaptation was to account for the relatively high amounts of precipitation in Nova Scotia so that the CTW would provide the desired treatment and so that the reservoir would retain the maximum amount of water. This was accomplished using historical drainage flow and water quality data. Construction and operational challenges were primarily due to a shallow water table, a condition that may often be encountered at other locations because the land available may be near waterbodies or in low lying areas. A CTW liner, a tile drain around the perimeter of the CTW, and raising the water level to increase downward pressure apparently reduced movement between wastewater and groundwater. However, it may be more appropriate to consider the pollutant loading and water volume contributions from groundwater during the design process. Surface runoff should be included as a component of peak flow when sizing control structures. A rock buffer, perpendicular to flow, at the end of the forebay should be used to disperse flow energy and prevent channelling. Cost may be prohibitive; therefore it is important to efficiently design the WRIS and express their additional benefits. The most efficient design may be a closed system that captures and retains all gains because it would maximize the volume of water available for irrigation and prevent water pollution from WRIS effluent, thus reducing $C_{out}$ requirements and CTW size, which furthermore would maximize any crop benefit from nutrients in irrigation water.

Wetland-reservoir irrigation system hydraulics were assessed to gain a better understanding of their affect on water availability and treatment. A CTW water budget showed significant mean monthly imbalances in W1 and W2. The imbalances are primarily attributed to groundwater intrusion, which supports considering it as a pollutant load and water volume source during the design process. The amount of water captured by the CTW shows that WRISs constructed in Nova Scotia may capture more water but have greater treatment challenges than WRISs in other regions. A tracer study showed that the $t_a$ was similar to the $t_n$, which suggests that there are few inactive zones and supports the use of a high length to width ratio for the CTW shape. It also suggests that overall treatment may be similar to that predicted by the k-C* model.

Water quality throughout the WRIS was assessed to determine treatment in the CTW, assess *k*s, and determine if reservoir water can be used for irrigation or safely discharged into the environment. The distribution of $NO_3^-$-N and *E. coli* loads by flow rates indicated that the treatment of high flow periods may not be as critical as expected. The CTW was shown to be effective at reducing both $NO_3^-$-N and *E. coli* loads. TP and SRP concentrations were typically below detectable limits, therefore their treatment could not be assessed. Water quality successively improved from the tile drainage outlet, to the W2 outlet, to the reservoir outlet. One exception was elevated *E. coli* levels at the reservoir outlet in the warm season, when the reservoir water may not be safe to irrigate crops consumed raw. This is attributed to surface runoff or contamination from wildlife and illustrates the challenge to any technology that does not treat irrigation water immediately before use. The water quality assessment provided information on the proper method of applying the k-C* model. The *k*s generated from the present study account for dynamic pollutant loading and temperature effects on treatment and should be used to design similar CTWs in the regions. The annual *Q* should be used rather than the maximum monthly mean *Q* (extrapolated to annual flow), as the recommended *k*s already account for the dynamic pollutant loading A lower *C** for *E. coli* may be used based on low levels recorded at the CTW outlet. In cases when historical drainage flow and water quality data is absent an $A_W$ to $A_D$ recommendation can be used to size the CTW. Design and management options, such as adding additional stop plates to the CTW outlet control structure before high flow events to increase CTW volume, combining the CTW and reservoir into a single system, using a head pond and a lower rate pump to the CTW, or circulating CTW outflow through the CTW again, may help extend RT during high flow events and provide the desired treatment to all flow rates.

Many of the conclusions from the present study are drawn from a single year of data. It is recommended that flow, water quality, and meteorological parameters be monitored beyond a single year to better understand system performance under extended conditions. Monitoring crop yields with and without irrigation will provide important information for the economic analysis. Research needs include a holistic assessment of the WRIS to further improve designs and understand benefits. This would include monitoring P retention in CTW soil to determine the P treatment lifespan of the CTW, monitoring air quality at the CTW to determine green house gas emissions, and conducting wildlife surveys to characterize the ecological benefit of WRISs.

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