University of Alberta

Quantification of deep drainage flux and drainage water quality characterization below the root zone of a short rotation coppice of willow and poplar receiving municipal treated wastewater irrigation in the lower foothills natural subregion of Alberta

By

Amy Elizabeth Gainer

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of

Master of Science
In
Soil Science

Department of Renewable Resources

ABSTRACT

Short rotation coppice is a type of bio-energy that involves the management of woody species to be harvested for energy purposes. Short rotation coppice in combination with municipal treated wastewater irrigation offers various benefits, mainly a low cost form of both nutrients and irrigation water. However, wastewater contains plant essential nutrients that can impact groundwater if the systems are mismanaged, i.e. nitrate. To prevent soil salinization from occurring in the root zone, a leaching fraction (LF) is applied. Leaching fraction is the fraction of surface infiltrated water that drains past the root zone. The research objectives involve quantifying and qualifying drainage in a fine textured soil below a SRC with wastewater irrigation system. Drainage was estimated under the wastewater irrigated soil and non-irrigated soil (control) using two methods, the soil water balance and a model based on the chloride mass balance by Rose et al (1979). The drainage in 2010 and 2011 using the water balance was -9.2 and 28 cm, respectively, and the model results were -18.5 and 11.7 cm, respectively. Drainage quality was monitored over 2010 and 2011 for nitrate-N, orthophosphate-P and other solutes using soil solution samplers and a transport model by Schoups and Hopmans (2002). Solute loading rates to groundwater were greater under the wastewater irrigated soil than the control. Nitrate-N in the soil solution at 150 cm below ground surface for both monitoring years never exceeded either the potable water guidelines (10 mg L^{-1}), however, these years had relatively low irrigation amounts and the Schoups and Hopman (2002) model predicted soil solution nitrate-N concentration in exceedance of potable water guidelines at LF's above about 0.55. Based on the findings from this research, it is recommended to use a LF between 0.2 and 0.5 to protect groundwater users, prevent soil salinization and utilize the large supply of municipal treated wastewater available.

Acknowledgements

I would like to thank Miles for all his time and patience with me and Gary for his support and insight. Thank you to Heather Shurgot, Sunny Song, Andre Chistensen, Gro LIlbaek, Kaitlin Zinz, Dick Puurveen and Cory Kartz for field and lab help. Thank you to Richard and Martin at CFS and all the Renewable Resource staff for their assistance. I would like to thank all my friends and family, especially Matt and my Mom, for their support. I would also like to thank Suzanne Abele who gave me my first taste of research and life as a graduate student.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

1. GENERAL INTRODUCTION

1.1 BACKGROUND

Currently, mitigating global warming is a top environmental concern for many countries around the world. One resolution for this issue is adopting energy forms with minimal greenhouse gas emissions. Biomass energy, or bio-energy, is a form of renewable energy that consumes plant material as a fuel (IEA 2005). Common bio-energy crops include sugar cane and maize for ethanol and oilseed rape for biodiesel (OECD 2004). A bio-energy system that is gaining popularity in Europe is short rotation coppice (SRC) with wastewater irrigation (Dimitrou et al 2009). SRC plantations are the management of fast growing woody species like willow in short intervals, of typically 3 to 5 years, for energy purposes (Labrecque and Teodorescu 2003; Bonaria et al 2004; Heller et al 2004). Plant growth is accelerated by applying water and nutrients in the form of readily available and relatively inexpensive secondary treated wastewater. Currently, there is between 1,500 and 15,000 ha of land being cultivated with woody species in Sweden, Poland, Germany, Italy and other European countries. SRC plantations have been identified as one of the most energy efficient carbon conversion technology to reduce greenhouse gas emissions (Styles and Jones 2007).

Short rotation coppice plantations typically use perennial, non-edible plant species like willow (*Salix* sp.) and poplar (*Populus* sp.). The common clones and hybrids in Sweden are *Salix viminalis, S. dasyclados* and *S. schwerinii* (Dimitrou and Arronson 2004). These species are desirable due to their high water requirements and the ability to take up nutrients with minimum leaching losses (Dimitrou and Aronsson 2004; Aronsson 2000). SRC crops are typically harvested in the winter or early spring and in short intervals varying from 2 to 15 years (Dimitrou et al 2009). The above ground biomass is typically chipped on site and stored or burned for heat or power (Dimitrou and Arronson 2004). After harvest, these plants have the ability to resprout and subsequent planting is not required. The estimated economical lifespan of a willow SRC systems is 20 to 25 years (Dimitrou and Arronson 2004; Heller et al 2004; Dimitrou et al 2009).

SRC systems irrigated with wastewater irrigation provide a complement to conventional wastewater treatment. Rather than discharging nutrient rich wastewater to water bodies it is applied to land where plants aid in filtering of water by taking up many nutrients. The secondary treated wastewater typically contains nutrients such as nitrogen and phosphorus which the SRC plants will utilize for growth and development. Nitrogen and phosphorus rich wastewaters discharged to surface waters can lead to a reduction in dissolved oxygen (eutrophication). By applying the wastewater to land rather than surface water bodies, water quality is hopefully improved.

SRC plantations with wastewater irrigation systems offer various social and economic benefits (IEA 2005). Studies have found that the cost efficiency of SRC crops with wastewater irrigation to be higher than conventional and, sometimes, unconventional tertiary wastewater treatments (Borjesson and Berndes 2006). In general, bio-energy crops produce considerabley less greenhouse gas emissions. Studies have estimated that bioenergy has 10 to 20 times lower net carbon emissions than fossil fuel based generation (Mann and Spath 2000; Matthews and Mortimer 2000). There is also potential for the trade of carbon credits in areas where SRC bioenergy plantations are established. Social benefits from bio-energy crops include improved access to basic services like electric lighting, creation of jobs and reduced oil dependency. An additional benefit from SRC crops is that it may boost farm incomes and the rural economy (Askew and Holmes 2001).

There are environmental and human health concerns with wastewater irrigation systems. Wastewater contains various salts that can accumulate in the root zone due to evapotranspiration effects. This creates osmotic pressures on the plant roots, decreased plant water uptake and decreased crop yields. Depending on the type of salts present, soil quality may be reduced. An abundance of sodium causes soil particle dispersion, degrading soil structure (Hamilton et al 2007). Heavy metals in wastewater can cause environmental problems. Heavy metals are typically cations and are strongly sorbed to the organic matter or soil surfaces, eventually accumulating in the soil over time (Hamilton et al 2007). Some metals, like boron, that remain mobile, threaten groundwater resources. Pathogens in drinking water are a rising public concern. The possible pathogens present in wastewater, depending on its origin, and may include bacteria such as *Salmonella* sp., enteropathogenic *Escherichia coli*, *Cryptosporidium parvum* or *Giardia intestinalis* (Hamilton et al 2007). Wastewater contains various plant essential nutrients like nitrogen, phosphorus and potassium (Dimitriou and Aronsson 2004). If these nutrients are applied in excess of the plants uptake capacity, leaching of mobile forms may occur. Over fertilization contributes to various plant problems including reduced fruit size, delayed maturation and the plant's ability to resist disease (Hamilton et al 2007). Research on these potential hazards is important for human and environmental health.

1. 2 SOIL AND WATER QUALITY

1.2.1 Nitrogen

Nitrogen is an essential plant macronutrient and commonly a growth-limiting nutrient for woody species (van Meigroet et al 1994). There are various nitrogen sources, one of which is via wet or dry deposition. Deposition occurs when burning of fossil fuels release NO_x gases to the atmosphere and HNO_3 forms (Galloway and Dillon 1983). Dry deposition typically occurs in the form of nitrogen dioxide (NO_2) or nitric acid (HNO_3) (Fowler et al 1999). Wet deposition of nitrogen occurs as rain or snow, typically in the form of nitrate (Fowler et al 1999). Atmospheric deposition on to the prairies is estimated to be about 13.9 x10^3 tonnes per year or 0.8 kg ha^{-1} yr^{-1} of total nitrogen (Chambers et al 2001).

Plants take up nitrogen in two main forms: ammonium and nitrate. Ammonium is preferred by plants because it requires less energy to assimilate (Crouzet et al 1999). Typically wastewater contains all forms of nitrogen except in the gaseous forms. These forms can change depending on the conditions they are exposed to. The major processes of the nitrogen cycle related to wastewater irrigation are mineralization, immobilization, nitrification and denitrication. Nitrogen mineralization involves the breakdown of organic nitrogen forms to ammonium by micro organisms (Brady and Weil 2002). The ammonium produced by mineralization is subject to immobilization, plant uptake, clay adsorption, volatilization or nitrification. Nitrogen immobilization occurs when there is an imbalance between the ratio of available carbon to nitrogen in the soil and simple inorganic compounds like ammonium or nitrate are taken up (immobilized) by microbes to maintain their metabolic processes. Plant uptake of nitrogen is a large sink. For example, hyrbid poplar on average removes between 241 and 420 kg ha^{-1} yr^{-1} of N (Heilman and Norby 1998). Clay adsoprtion involves bonding between the negatively charged clay surface and positive ammonium ion. Ammonium (NH_4^+) and ammonia gas (NH_3) are both produced by the breakdown of organic residues (Brady and Weil 2002). The relationship between the two is given by:

$$NH_3 + H_2O \leftrightarrow NH_4^+ + OH^- \quad [1]$$

Ammonia volatilization typically occurs at high pH levels, when soil is dry and when ammonium levels are high (Brady and Weil 2002). Volatilization is the conversion of ammonium to ammonia gas which is considered lost from the soil system. Nitrification is the oxidation of ammonium ions in soil and is a two part process (Brady and Weil 2002). The first part is the conversion of ammonium to nitrite by the bacteria *Nitrosomonas*. The nitrite formed is immediately utilized by the bacteria *Nitrobacter* and nitrate is product. Similar to ammonium, nitrate can be taken up by plants and immobilized by microorganisms (Brady and Weil 2002) . Denitrification is a process resulting in gaseous nitrogen forms, lost to the atmosphere. During anaerobic conditions, nitrate is typically converted to either nitric oxide gas (NO), nitrous oxide gas (N_2O) or dinitrogen gas (N_2) depending on the stage of the process and type of bacteria. Heilm and Norby (1998) found that denitrification can account for as much as 70 % of nitrate losses under irrigated soils. The nitrogen cycle is affected by various soil conditions like pH, organic carbon content, temperature, soil water content, texture and soil biogeochemical characteristics (Almasri and Kaluarachchi 2004, Brady and Weil 2002).

1.2.1.1 Nitrate

Managing nitrate movement under wastewater irrigation systems is important for efficient system design with minimal negative human and environmental health impacts. Nitrate is a soluble form of nitrogen that is readily taken up by plants. It is produced by either nitrification, the conversion of ammonium to nitrate, or direct fertilizer additions (Tisdale et al 2005). As mentioned earlier, ammonium sources are atmospheric nitrogen fixation, nitrogen deposition, mineralization of organic nitrogen forms or direct fertilizer additions.

Nitrate is an environment risk due to its behaviour in soils. The net charge of the nitrate ion is negative so it is not readily attracted to clay surfaces and remains mobile in the soil solution (Tisdale et al 2005). Nitrate leaching can occur if the amount of nitrate and water additions exceeds removal (Tisdale et al 2005). Previous studies show that when elevated nitrogen losses occur, it is mainly in the nitrate form rather than ammonium (Dimitrou and Aronsson 2004). Leaching of nitrate from the soil can lead to contamination of groundwater and, subsequently, a hazard for potable water users. Consumption of nitrate contaminated drinking water can cause *methemoglobinemia* in infants and stomach cancer in adults (Addiscott et al 1991; Lee et al 1991; Wolfe and Patz 2002). The Health Canada Drinking Water standard for nitrate-nitrogen is 10 mg L^{-1} and Alberta Environment Tier 1 Groundwater Remediation level is 13 mg L^{-1} (Health Canada 2010; Alberta Environment 2010).

Nitrate leaching to groundwater systems can have various detrimental impacts on the physical environment. Nitrate and ammonium contaminated groundwater discharged to surface water bodies can cause eutrophication, a widespread surface water quality problem near livestock feedlots. Symptoms of eutrophication are accelerated plant growth, changes in plant species composition and decreased dissolved oxygen concentrations (Zhann 1995). Recent studies also indicate that nitrate in surface water bodies may affect metabolic processes in fishes and reptiles (Edwards 2005; Guillette and Edwards 2005). Many aspects of the soil system affect the fate of nitrate in soil.

Nitrate movement is affected by numerous factors in the soil system including plants. Vegetation characteristics like age and transpiration rates affect the degree of nitrate leaching. Previous studies show that leaching from SRC plantations are low except during the establishment year when the plants are young (Aronsson 2000). Heilman and Norby (1998) found that on *Plantus occidentalis* plantations, nitrate concentrations at 60 cm were strongly influenced by the age of the trees when fertilized. Also from this study, they found as trees aged nutrient uptake increased and there were fewer nitrates in soil water. Dimitrou and Aronsson (2004) found that nitrate loading rates below the root zone as measured by lysimeters were negatively correlated to transpiration rate. Nitrate leaching was highest when transpiration was lowest.

Application rates affect the amount of nitrate retained in the soil system. Mantovi et al (2006) found that when nitrogen inputs from a manure slurry were substantially higher than plant uptake, nitrate accumulated in the surface layer of soil. Continuous irrigation with wastewater has different nitrate leaching dynamics than alternating wastewater irrigation with freshwater. Sakadevan et al (2000) established that continuous wastewater irrigation has greater amounts nitrate leaching than wastewater applied alternately with freshwater.

Soil texture properties affect nitrate movement and potential for leaching. Typically, coarse textured soils have the largest nitrate leaching potential (Bergstrom and Johansson 1991). Studies have found that finer textured soils can also have high leaching amounts. A study by Dimitrou and Aronsson (2004) found that a breakthrough of nitrate was recorded earlier in a clay soil than sand, attributed to preferential flow of water in cracks and macropores in the clay. Barton et al (2005) also had greater nitrate leaching in a poorly drained clay dominated soil due to macropore preferential flow. Macropores are the relatively larger soil pores created by roots or earthworms. Macropores can have the capacity to transmit large amounts of water, especially when near field saturation (Seyfriend and Rao, 1987; Magesan et al 1995; Magesan et al 1999). In the same study by Barton et al (2005), significant leaching losses also occurred from a well drained sandy soil.

1.2.1.2 Ammonium

As mentioned earier, ammonium is a plant essential nutrient and the preferred form of nitrogen by plants. Similar to nitrate, ammonium in surface and groundwater reserves can be detrimental to human and environmental health. Ammonium is typically bound to soil particles but under certain conditions it it can be mobile. A study by Wachendorf et al (2008) found significant concentrations of ammonium in seepage water form soil exposed to cattle urine. Various conditions affect the potential for ammonium leaching such as input concentrations and preferential flow.

Ammonium movement past the root zone is affected by input amounts and the organic carbon presence in soil. Mian et al (2009) found that a large proportion of ammonium was adsorbed by the soil but a significant amount remained in soil solution and was considered mobile. The amount of immobile ammonium may be related to the amount of organic matter present in soil. Burge and Broadbent (1961) found a relation between the percentage of carbon in soil and the amount of ammonium fixed. Ammonium fixation to organic matter has been attributed to the various constituents associated with humic and fulvic acids such as keytones, aldehydes and sugars (Stevenson 1994). Ammonium leaching may be a result of accumulated ammonium from organic matter mineralization. Livesley et al (2007) found larger than expected ammonium concentrations in soil water at depths of 30 and 100 cm indicating leaching due to direct ammonium inputs and indirect mineralization of organic matter.

Ammonium leaching to groundwater may occur due to preferential flow through macropores. Both Livesley et al (2007) and Singleton et al (2001) found high ammonium concentration in the subsurface which they attributed to near saturation macropore flow. Leaching losses from application of cow urine to soil increased when soil reached field capacity as macropore flow was initiated (Silva et al 2000). Ammonium prone to leaching is also affected by the soil residence time of applied ammonium in soil. With longer residence times, the ammonium is more exposed to transformations and immobilizations and concentration in soil water can be reduced (Livesley et al 2007).

Wastewater contains various types of cations that are attracted to the soil particle surfaces. Long term application of water containing cations may lead to ammonium accumulation in soil solution due to lack of cation exchange sites. Divalent cations like calcium and magnesium have a greater charge per surface area and more strongly attached to exchange sites than ammonium. Results from a study by Phillips (2002) suggest that long term application of piggery wastewater may encourage leaching of nitrogen and phosphorus in soils with limited capacity to retain these nutrients. Mian et al (2009) found that texture and the amount of ammonium leaching had a greater correlation than the cation exchange capacity of the soil and ammonium leaching. The reasoning for this is that texture reflects drainage conditions which likely result in denitrification (Mian et al 2009). Mian et al (2009) also found no significant correlation between soil mean initial ammonium N concentrations and C:N soil mass ratio. Other studies found leaching depends on the time for transformation and immobilization to occur (Livesley et al 2007).

1.2.2 Phosphorus

Like nitrogen, phosphorus (P) is a plant essential nutrient and a common fertilizer constituent. Phosphorus is a naturally occurring element in nature and is an essential component for many physiological processes in both plants and humans (University of Minnesota 2011). Phosphorus has no known direct toxic effects to humans or animals but can have damaging impacts on the environmental (Pierzynski et al 2005). Phosphorus in surface water bodies stimulates biological activity causing eutrophication, similar to nitrogen. Phosphorus is typically found in secondary treated wastewater used with SRC projects. Research on the fate and transport of this nutrient is vital for protection of aquatic water quality.

Phosphorus in the soil or water will typically occur as phosphate. Orthophosphate is a simple form of phosphate (University of Minnesota 2011). In water, orthophosphate occurs as $H_2PO_4^-$ in acidic conditions or HPO_4^{2-} in alkaline conditions. There are three main pools of phosphorus in the soil: 1) solution, 2) active and 3) fixed (University of Minnesota 2011). Solution phosphorus is a very small fraction of the total phosphorus pool. It is typically in orthophosphate form, the form preferred by plants, but organic phosphorus forms can also be present. Solution phosphorus is important because plants utilize this pool (University of Minnesota 2011). The active phosphorus pool is the phosphorus in the solid phase that can be readily released to soil solution. Phosphate in the active pool is released to replenish the phosphate removed by plants in the soil solution phase (University of Minnesota 2011). The active phosphorus pool is the main source of available phosphorus for crops and the ability of the active pool to replenish the soil solution varies with soil (University of Minnesota 2011). Forms of phosphorus in the active pool include inorganic phosphate adsorbed to soil particles, phosphate bound to calcium or aluminum and organic phosphorus than can be readily mineralized. Soil solution phosphorus is positively related to the active phosphorus pool (University of Minnesota 2011). The fixed phosphorus pool contains very insoluble inorganic phosphate compounds and organic phosphorus compounds that are crystalline in structure and not mineralized by soil microorganisms. This pool has little impact on the soluble phosphate but there is slow conversion between the fixed and active phosphorus pools. The relationship between all soil phosphorus pools is important for assessing phosphorus movement in soil.

Phosphorus in soil goes through various physical, chemical and microbiological processes, controlling its availability for leaching. Initially, phosphate additions to soil are soluble and available but reactions start occurring that limit its availability. These reactions depend on various soil conditions such as pH, moisture content, temperature and minerals present (University of Minnesota 2011). Once phosphate is added to the soil, it typically dissolves if adequate moisture is present. Once in the soil solution phosphate is available to plants and movement of phosphate is slow but may be increased by water flowing through soil (University of Minnesota 2011). The majority of the phosphate in solution will react with the minerals in the soil. Phosphate ions in solution typically move to the active pool by adsorbing to soil particles or forming compounds with elements like calcium, magnesium, aluminum or iron (University of Minnesota 2011). Phosphorus fertilization is considered inefficient because soil solution additions over time become fixed, and relatively unavailable to plants. The majority of the phosphorus applied to the soil will not be utilized by the crop but continued application typically increases the fertility of the soil (University of Minnesota 2011). Typically fine texture soils hold more than coarse (University of Minnesota 2011). Environmental pollution from phosphate is typically from sediment with phosphate attached. Phosphate is found in wastewater and although the soil retains much of the phosphate applied, there is the possibility of soluble phosphate leaching under irrigation systems (Olson et al 2010).

The risk of phosphorus leaching is considered lower when calcareous sub soils are present (Olsen et al 2010). Lutwick and Graveland (1978) estimated that the phosphate adsorption capacity of silty clay and silty clay loam soils from southern Alberta ranged from 236 to 950 mg P kg^{-1} and estimated that sewage effluent could be applied at 40 kg P $ha^{-1}yr^{-1}$ for 120 to 153 years before phosphate saturation. But some studies have found phosphate leaching can occur in calcareous soils. Eghabll et al (1996) found phosphate from long term manure applications leached through calcareous layers to a depth of 1.8 m in a very fine sandy loam soil. Olsen et al (2010) suggest phosphate leaching in calcareous soils occurs and more so in a coarse texture soil than medium textured with manure application.

There are various factors that affect phosphate leaching. Leaching is typically greatest when storm events follow recent fertilizer applications (Barton et al 2005). The presence of macro pores and preferential flow pathways contribute to phosphate leaching. Heathwaite and Dils (1999) found a peak P concentration of 1 mg P L^{-1} in subsurface tiles which was attributed to macro pore flow. The degree of phosphate saturation is an important factor for phosphate leaching. Soils with a history of receiving phosphate additions above crop requirements can become saturated with phosphate. Butler and Coale (2005) found the degree of phosphate saturation in the top 60 cm of the soil to be 25% after 4 years of dairy manure application. Phosphate in the subsoil may lead to higher leaching losses overtime as the subsoil becomes a source of phosphate through desorption into soil water (Gachter et al 2004).

1.2.3 Dissolved Organic Carbon

Irrigation with wastewater contributes organic carbon to the soil, part of which is dissolved organic carbon (DOC). Leaching of DOC is typically minimal compared to total soil organic carbon content and other carbon loss pathways like mineralization (Brye et al 2002). Findings from Siebe and Fischer (1996) show increased DOC in upper soil layers due to irrigation with sewage effluent. There is water quality concerns associated with potential leaching of DOC to groundwater from wastewater irrigated soils. Unsafe products such as trihalomethanes can be formed when chlorine in drinking water reacts with DOC (Xie 2004). The pesticide, atrazine was mobilized in a study due to binding with DOC from effluent irrigation (Graber et al 1995). Drinking water characteristics such as color, taste and odor can be altered negatively by DOC (Ruark and Linquist 2010). In subsurface soils denitrification is limited due to lack of carbon source but when DOC leaches to the subsurface soils, denitrification is enhanced (Sotomayor and Rice, 1996; Yeomans et al 1992). Studies have found that DOC levels in the soil are related to the survival of pathogenic bacteria (Jamieson et al 2002).

Research has shown that DOC can enhance transport of metals. Siebe and Fischer (1996) found enhanced metal mobility and transport due to irrigation with wastewater. Lensing et al (1995) found increased metal mobility due to DOC-cadmium complexes in soil. Fine et al (1999) found zinc and total organic carbon concentration were linearly correlated in a study with wastewater irrigation and biosolids. Studies have found that DOC also contributed to the transport of copper and mercury in the soil (Tetzlaffe et al 2007; Schuster et al 2008).

1.2.4 Salinity

Irrigation water typically contains soluble salts which may lead to salinization of the root zone over time if irrigation is not managed properly. The major salts of concerned include cations: Ca, Mg and Na and anions Cl, SO_4 and HCO_3

(Hillel 2005). Salinization occurs if irrigation water applied is equal to or less than the evapotranspirational demand of the plants, then salts from irrigation water will accumulate as plants successively take up water and not the salts.

Accumulation of salts in surface soils can have various detrimental affects on the environment. Soil structure degradation can occur under saline conditions. Unsuitable top soil is considered to be >8 dS m^{-1} and poor is 4 to 8 dS m^{-1} (Alberta Environment 2009). Fireman and Bodman (1939) noted that when soil solution salinity increases the tendency for swelling, aggregate failure and dispersion increases. Various studies show soil hydraulic conductivity decreases after application of treated municipal wastewater due to dispersion of soil particles caused by accumulation of sodium and other salts (Thomas et al,1966; Clanton and Slack, 1987; Cook et al 1994; Balks et al 1998; Goncalves et al 200; Magesan et al 1999; Halliwell et al 2001). The infiltration rate is also affected by soil salinity. Formation of a seal or crust at the soil surface occurs due to reduced aggregate stability and clay dispersion, reducing the infiltration rate of the soil and water available to plants (Grattan and Oster 2003). Another surface condition that affected by soil salinity ishard setting which occurs when soils are near saturation and results in a large, compact and hard surface upon drying (Mullins et al 1990).

To prevent salt accumulation, water in excess of the evapotranspiration needs of the crops is applied to leach the soluble salts below the root zone. This additional irrigation water is commonly called the leaching fraction, or leaching requirement. In agriculture, leaching involves the dissolution and transportation of soluble salts by the downward movement of water (U.S. Salinity Laboratory Staff 1954). The U.S. Salinity Laboratory Staff (1954) first defined the leaching fraction as the fraction of infiltrated water that moves through root zone to prevent soil salinity from exceeding a level that would reduce crop yield. The leaching fraction depends on the composition of the irrigation water, plant water needs and salt tolerance of crop (Hillel 2005).

1.3 OBJECTIVES

The overall objectives of this research are to determine the impacts of repeated applications of wastewater over the life cycle of a SRC plantation on the fate and transport of plant essential nutrients that can possibly contaminate water reserves. Specific objectives include:

I. Quantify the drainage under SRC with wastewater irrigation root zone using the soil water balance and chloride mass balance methods.
II. Characterize drainage water quality of the SRC plantation at the study site using an in-situ field-based sampling method and a transport model.
III. Estimate wastewater treatment, or removal of solutes, by the SRC system.

IV. Assess the soil salinity with varying irrigation management scenarios using a solute transport model.

V. Make recommendations to mitigate environmental impacts and promote sustainability of SRC plantations with wastewater irrigation systems.

1.4 REFERENCES

Addiscot, T. M., A. P. Whitmore and D.S. Powlson. 1991. Farming, fertilizers and the nitrate problem. CAB International. Wallingford, United Kingdom.

Alberta Environment. 2010. Alberta Tier 1 Soil and Groundwater Remediation Guidelines. Government of Alberta. Edmonton, AB.

Almasri, M.N. and J.J. Kaluarachchi. 2004. Implications of On Ground nitrogen loading and soil transformations on groundwater quality management. Journal of American Water Resources Association. February. 165-186.

Aronsson, P. 2000. Nitrogen retention in vegetation filters in short rotation willow coppice. Doctoral thesis, Silverstia 161, Swedish University of Agriculture Sciences. Department of Short Rotation Forestry. 14-32.

Askew, M.F. and C.A. Holmes. 2001. The potential for biomass and energy crops in agriculture in Europe, in land use, policy and rural economy terms. Biomass and Energy Crops II, Aspects of Applied Biology. 65: 365 – 374.

Balks, M.R., Mclay, C.D.A., Harfoot,C.G., 1997. Determination of the progressing in soil microbial response and changes in soil permeability, following application of meat processing effluent to soil. Applied Soil Ecology. 6:109-116.

Barton,L., L.A. Schipper, G.F. Barkle, M. McLeod, T.W. Speir, M.D. Taylor, A.C. McGill, A.P van Scaik, N.B. Fitzgerald and S.P. Pandey. 2005. Land application of domestic effluent onto four soil types plant uptake and nutrient leaching. Journal of Environmental Quality. 34:635-643.

Bergstrom, L. and R. Johansson. 1991, Leaching of nitrate form monolith lysimeters of different types of agricultural soils. Journal of Environmental Quality. 20:801-807.

Bonaria, E., G. Picchi, A. Fraga, M. Ginanni, W. Guidi and E. Piccioni. Comparison of the three coppice intervals on a nine years poplar biomass production. Proc. Of IPC 22nd Session: The Contributions of Poplars and Willows to Sustainable Forestry and Rural Development. November 28 to December 2 2004. Santiago, Chile.

Borjesson, P. and G. Berndes. 2006. The prospect for willow plantations for wastewater treatment in Sweden. Biomass Bioenergy. 30:428-438.

Brady, N.C. and R. R. Weil. 2004. Elements of the nature and properties of soils, Second Edition. Pearson, Prentice Hall. Upper Saddle River, New Jersey.

Brye, K.R. and J.M. Norman, L.G. Bundy and S.T. Gower. 2002. Carbon.budgets for a prairie and agroecosystems: effects of land use and inter annual variability. Ecological Applications. 12:962-979.

Burge W.D. and F.E. Broadbent. 1961. Fixation of ammonia by organic soils. Soil Science Society of America. 25:199-204.

Butler, J.S. and F.J. Coale.2005. Phosphate leaching in manure amended Atlantic Coastal Plain soils. Journal of Environmental Quality. 34: 370-381.

Chambers, P.A., M. Guy, E.S. Roberts, M.N. Charlton, R.A. Kent, C. Gagnon, G.Grove and N. Foster. 2001. Nutrients and their impacts on the Canadian environment. Agriculture and Agri Food Canada, Environment Canada, Fishers and Oceans Canada, Health Canada and Natural Resources Canada. 241 pp

Clanton, C.J. and Slack. D.C. 1987. Hydraulic properties of soil as affected by surface application of wastewater. Transactions of American Society of Agricultural Engineering. 30: 683-687.

Crouzet, P., J. Leonard, S. Nixon, Y. Rees, W. Parr, L. Laffon, J. Bogestrand, P. Kristensen, C.Lallana, G. Izzo, T. Bokn, J. Bak and T.J. Lack. 1999. Nutrents in European ecosystems.. Thyseen, N. (ed). Report no. Environmentall Assessment Report no.4 European Environment Agency. Copenhagen, Denmark.

Dimitrou, I. and P. Aronsson. 2004. Nitrogen leaching from short-rotation willow coppice after intensive irrigation with wastewater. Biomass and Bionenery. 26. 433 - 441.

Dimitrou, I., C. Baum, A. Baum, G. Busch, U. Schulz, J. Kohn, N. Lamersdorf, P. Leinweber, P. Aronsson, M. Weigh, G. Berndes and A. Bolte. 2009. The impact of Short Rotation Coppice (SRC) cultivation on the environment. 2009. Agriculture and Forestry Research. 59: 159-162.

Edwards, T.M., 2005. Environmental Influences on Mosquitofish Reproduction. Ph.D. Dissertation. University of Florida, Gainesville, Florida. 187 pp.

Eghabll, B., G.D. Binford and D.D. Baltensperfer. 1996. Phosphorus movement and adsorption in a soil receiving long term manure and fertilizer application. Journal of Environmental Quality. 25: 1339-1343.

Fine, P., A. Hasss, A. Berezking and N. Atzmon. 1999. Fate of organic matter, copper and zinc in effluent irrigated or fertigated sludge amended sand: A lysimeter study. *In* Y. Chen et al (ed.) Modern agriculture and environmental quality. Inter. Conf. of the Ministry of Science, Israel. April 1999. Jerusalem, Israel. Fireman, M. and G.B. Bodman. 1939. The effect of saline irrigation water upon the permeability and base status of soils. Soil Science Society of America Proc. 4:71-77.

Fowler, D., R.I. Smith, U.M. Skiba, M. Coyle, C. Flechard, C.E.R. Pitcairn and M.A. Sutton. 1999. The atmospheric nitrogen cycle and the role of anthropogenic activity. In: Managing Risks of Nitrates to Humans and the Environment. Wilson, W.S., A.S.Ball and R.H. Hinton (editors). The Royal Society of Chemistry. Cambridge, UK. Pp. 121-138.

Gachter,R., S.M. Steingruber, M. Reinhardt and B. Wehrli. 2004. Nutrient transfer from soil to surface waters: difference between nitrate and phosphate. Aquatic Science. 66: 117-122.

Galloway, J.N. and P.J. Dillon. 1983. Effects of acid deposition: The importance of nitrogen. Report no. SNV OM 1636. In: Ecological Effects of Acid Deposition:Report on Background Papers-1982 Stockholm Conference on the Acidification of the Environment. Expert Meeting 1. National Swedish Environment Protection Board. Pp. 145-160.

Goncalves, R.A.B, M.V. Folegatti, T.V. Gloaguen, P.L. Libardi, C.R. Montest, Y. Lucas, C.T.S. Dias and A. J. Melfi. 2007. Hydraulic conductivity of a soil irrigated with treated sewage effluent. Geoderma. 139:241-248.

Graber, E.R., E. Yagil, Z. Gersil, F. Berkovitch, and P. Fine. 1999. Copper binding ability of dissolved organic matter derived from anaerobically digested biosolids. Journal of Environmental Quality. 22:731-737.

Grattan, S.R. and J.D. Oster. 2003. Use and Reuse of Saline-Sodic Waters for Irrigation of crops. Journal of Crop Production. 7(1):131-162.

Guillette, Jr., L.J. and T.M. Edwards, 2005. Is Nitrate an Ecologically Relevant Endocrine Disruptor in Vertebrates- Integrative and Comparative Biology. 45:19-27.

Halliwell,D.J., K.M. Barlow, D.M. Nash. 2001. A review of the effects of wastewater sodium on soil physical properties and their implications for irrigation systems. Australian Journal Soil Research. 39: 1259-1267.

Ham, J.M. and T.M. DeSutter. 1999. Seepage losses and nitrogen export from swine waste lagoons: a water balance study. Journal of Environmental Quailty. 28: 1090-1099.

Hamilton, A.J., F. Stagnittie, X. Xiong, S.L. Kreidl, K.K. Benke and P. Maher. 2007. Wastewater irrigation: the state of play. Vadose Zone Journal. 6:823-840.

Heatlh Canada. 2010. Health Canada Guidelines for Canadian Drinking Water Quality. Prepared by the Federal-Provincial-Territorial Committee on Drinking Water of the Federal-Provincial-Territorial Committee on Health and the Environment. Government of Canada. Ottawa, Canada.

Heathwaite, A.L. and R.M. Dils. Characterising phosphorus loss in surface and subsurface hydrological pathways. The Science of the Total Environment. 251/252: 523-238.

Heller, M.C., G.A. Keoleian, M.K. Mann and T.A. Volk. 2004. Life cycle energy and environmental benefits of generating electricity from willow biomass. Renewable Energy. 29 (7) 1023-1042.

Heilman, P. and R.J. Norby. 1998. Nutrient Cycling and fertility management in temperate short rotation forest systems. Biomass and Bioenergy. 14:361-370.

Hillel, D. 2005. Salinity management. Encyclopaedia of Soil Science, vol 3. Elsevier/Academic Press, 435-442.

IEA (International Energy Agenxy). 2005. Benefits of Bioenergy. International Energy Agency, http:// www.ieabioenergy.com /library/179_BenefitsofBioenergy.pdf. Accessed Nov 7, 2011.

Jamieson, R.C., R.J. Gordon, K.E. Sharples, G.W. Stratton and A. Madani. 2002. Movement and persistence of fecal bacteria in agricultural soils and subsurface drainage water: a review. Canadian Biosystems Engineering. 55:1.1-1.9.

Labrecque, M. and T.Teodorescue. 2003. High biomass yields achieved by Salix clones in SRC following two 3 years coppice rotations on abandoned farmland in southern Quebec, Canada. Biomass Bioenergy. 15:71-7.

Lee,Y.W., M.F. Dahab and I. Bogardi. 1991. Nitrate risk management under uncertainty. Journal of Water Resources Planning and Management. 118(2): 151-165.

Lensing,H.J., J. Von Sachs, A.Oberle and B. Herrling. 1995. Modeling of complexation of heavy metals by dissolved organic substances and its effects on adsorption and transport in aquifers. 9165-172. *In* K.Kovar and J. Krasny (ed.) Groundwater Quality: Remediation and Protection. Proceedings of the International

Conference. Prague. May 15-18, 1995. IAHS Publications no. 225. Institute of Hydrology IAHS Press. Wallingford, UK.

Livesley, S.J., M.A. Adams and P.F. Grierson. 2007. Soil water nitrate and ammonium dynamics under a sewage effluent irrigate Eucalypt plantation. Journal of Environmental Quality. 36: 1883-1894.

Lutwick, G.W. and D.N. Graveland. 1978. A study of phosphorus adsorption capacities of several southern Alberta soils. Technical Report. Alberta Environment Technical Development Branch. Edmonton, AB.

Magesan, G.N., I. Vogeler, D.R. Scotter, B.E. Clothier and R.W. Tillman. 1995. Solute movement through two unsaturated soils. Australian Journal of Soil Research. 33:585- 596.

Magesan,G.N., Williamson, J.C., Sparling, G.P., Schopper, L.A., Lloyd-Jones, A.R. 1999. Hydraulic conductivity in soils irrigated with wastewaters of differing strengths: field and laboratory studies. Australian Journal of Soil Research. 37: 391-402.

Mann M.K. and P.L. Spath. 2000. Life Cycle Assessment of a Direct-Fired Biomass Power Generation System. National Renewable Energy Laboratory, Golden, CO, USA.

Mantovi, P., L. Fumagalli, G.P. Beretta and M. Guermandi. 2006. Nitrate leaching through the unsaturated zone following pig slurry applications. Journal of Hydrology. 316: 195-212.

Matthews, R.W. and N.D. Mortimer. 2000. Estimation of carbon dioxide and energy budgets of wood-fired electricity generation systems. ETSU Report B/U1/00601/05/REP. ETSU: Harwell.

Mian, I.A., M. Riaz and M.S. Cresser. 2009. The importance of ammonium mobility in nitrogen impacted unfertilized grasslands: A critical reassessment. Environmental Pollution. 157:1287-1293.

Mullins, C.E., D.A. MacLeod, K.H. Northcote, J.M. Tisdall and I.M. Young. 1990. Hardsetting soils: behaviour, occurrence and management. Advanced Soil Science. 11:38-108.

OECD. 2004. Proceedings of Biomass and Agriculture-Sustainability Markets and Policies. ISBN 92-64-10555-7. OECD, Paris.

Olson, B.M., E.Bremer, R.H. McKenzie and D.R. Bennett. 2010. Phosphorus accumulation and leaching in two irrigated soils with incremental rates of cattle manure. Canadian Journal of Soil Science. 90:355-362.

Phillips, I. R. 2002. Phosphorus sorption and nitrogen transformations in two soils treated with piggery wastewater. Australian Journal of Soil Research. 40: 335-349.

Pierzynski, G.M., J.T. Sums adn G.F. Vance.2005. Soils and Environmental Quality, Third Edition. CRC Press Taylor and Francis Group. Boca Raton, FL.

Ruark,D.M. and B.A. Linquist. 2010. Seasonal losses of dissolved organic carbon and total dissolved solids form rice production systems in northern California. Journal of Environmental Quality. 39:304-313.

Sakadevan, K., B.L. Mheshwari and H.J. Bavor. 2000. Availability of nitrogen and phosphorus under recycled water irrigation. Australian Journal of Soil Reserach. 38. 653-664.

Schuster, P.F., J.N., Shanely, M.Marcin-Dispasquale, M.M. Reddy, G.R. Aiken, D.A. Roth. H.E. Taylor, D.P. Krabbenhoft and H.F. Dewilde. 2008. Mercury and organic carbon dynamics during runoff episodes from a Northeastern USA watershed. Water Air Soil Pollution. 187:89-108.

Seyfried, M.S. and P.S.C. Rao. 1987. Solute transport in undisturbed columns of an aggregated tropical soil: Preferential flow effects. Soil Science Society of America Journal. 51: 1434- 1444.

Siebe, C. and W.P. Fischer. 1996.Effect of long term irrigation with untreated sewage effluents on soil properties and heavy metal absorption of Leptosols and Vertisols in Central Mexico. Z. Pflanzenernaehr. Bodenkd. 159:657-364.

Silva, R.G., K.C. Cameron, H.J. Di, N.P. Smith and G.D. Buchan. 2000. Effect of macropore flow on the transport of surface applied cow urine through a soil profile. Australian Journal of Soil Research. 38: 13-23.

Singleton, P.L., C.D.A. McLay and G.F. Barkle. 2001. Nitrogen leaching soil lysimeters irrigated with dairy shed effluent and having managed drainage. Australian Journal of Soil Research. 39: 385-396.

Sotomayor, D. and C.W. Rice. 1996. Denitrifitcation in soil profiles beneath grassland and cultivated soils. Soil Science Society of American Journal.60:1822-1828.

Stevenson, F.J. 1994.Humus chemistry: Genesis, composition, reactions. 2nd edition. John Wiley and Sons Inc. New York, USA.

Styles, D. and M. Jones. 2007. Energy crops in Ireland: quantifying the potential life cycle greenhouse has reductions of energy crop electricity. Biomass Bioenergy. 31 (11-12): 759-772.

Tetzlaff, D., I.A. Malcolm and C. Soulsby. 2007. Influence of forestry, environmental change and climatic variability on the hydrology, hydrochemistry and residence times of upland catchments. Journal of Hydrology. 3674: 93-111.

Thomas , R. E., Scwatz, W.A., Benedixen, T.W. 1966. Soil chemical changes and infiltration rate reduction under sewage spreading. Soil Science Society of American Proceedings 30. 641 -646.

Tisdale, S.L., W.L. Nelson, J.D. Beaton and J.L. Havlin. 2005. Soil Fertility and Fertilizers. Edition 7. MacMillan Publishing Company.

U.S. Salinity Laboratory Staff. 1954. Diagnosis and Improvement of Saline and Alkali Soils. U.S. Department of Agriculture handbook 60. U.S. Government Printing Office. Washington, Dc. 160 pp.

University of Minesota (Extension). 2011. Nature of Phoshporus in Soils. http://www.extension .umn.edu/distribution/ crop systems/DC6795.html. Accessed October 3, 2011.

van Meigroet, H., R.J. Norby and T.J. Tschaplinksi. 1994. Nitrogen fertilization strategies in a short rotation sycamore plantation. Forest Ecology and Management. 64: 13-24.

Wachendorf, C, C. Lamp, F. Taube and K. Dittert. 2008. Nitrous oxide emissions and dynamics of soil nitrogen under ^{15}N-labelled cow urine and dung patches on a sandy grassland soil. Journal of Plant Nutrition and Soil Science. 210/211: 171-180.

Wolfe, A.H. and J.A. Patz. 2002. Reactive nitrogen and human health: Acute and Long term implications. Ambio. 31(2):120-125.

Xie, Y. 2004. Disinfection by products in drinking water: Formation, analysis and control. Lewis Publications, Boca Raton, FL, USA.

Yeomans, J.C., J.M. Bremner and G.W. McCarry. 1992. Denitrification capacity and denitrification potential of subsurface soils. Comm. Soil Science Plant Anal. 23:919-927.

Zhann, L.P. 1995. Our sea out future major findings of the state of the marine environment report for Australia. Department of Environment Sport and Territories, Canberra.

2. DRAINAGE

2.1 INTRODUCTION

Mitigating the causes of global warming is a top environmental concern for many countries. Energy forms that minimize greenhouse gas emissions are of great interest at present time. Biomass energy, or bio-energy, is an emerging form of renewable energy that utilizes plant material as fuel. Common bio-energy crops include sugar cane and maize for ethanol and oilseeds (e.g., rapeseed or canola) for biodiesel (OECD 2004). A bio-energy system that is becoming increasingly popular in Europe is short rotation coppice (SRC) with wastewater irrigation. SRC plantations are the management of fast growing woody species in short intervals for energy purposes. SRC plantations are considered cellulose and a solid energy crop (Sims et al 2006). Cellulose crops use hemicellulose in the plant material to produce ethanol. Solid energy crops use the whole plant, rather than the only the hemicellulose, to produce heat or electricity through combustion or conversion to biofuels like methanol or ethanol (Sims et al 2006).

For adequate plant growth, water and nutrients are needed. A readily available and cost-effective source of both plant essential nutrients and water is municipal wastewater that has undergone primary or secondary treatment. This wastewater still contains plant nutrients such as nitrogen and phosphorus. An additional benefit of using wastewater to irrigate SRC plantations is that the plant will likely remove nutrients as the wastewater percolates through the root zone, resulting in lower nutrient loading rates to rivers and streams compared to direct disposal of the wastewater to surface water bodies. However, there are still environmental concerns associated with SRC crops grown under wastewater irrigation.

Drainage is the movement of water below the root zone. Adequate drainage in irrigated systems is important for maintaining root zone aeration and preventing secondary salinization of the root zone (Hillel 1998). When applying irrigation water containing salts, the salts may accumulate in the root zone over time, causing secondary salinization. This occurs because the plants take up the water but not most of the salts. Depending on the irrigation management, salts may accumulate or leach from the root zone. Soil salinity is detrimental to the environment because it reduces soil quality and decreases plant growth (Hillel 1998). Salinity directly affects plants by altering soil water osmotic potential, making it harder for roots to extract soil water and creating toxic conditions due to ions like boron, chloride and sodium (Hillel 1998). When the main salt accumulating in the soil is sodium, soil quality is impacted through degraded soil structure, clay dispersion, reducing porosity and decreased connectivity of soil pores (Hillel 1998).

To prevent secondary soil salinization, irrigation water in excess of the plant requirements is applied to leach the salts below the root zone. The fraction of water that moves below the root zone is called the leaching fraction or leaching requirement (Hillel 1998). The average annual salt flux below the root zone depends on the leaching fraction. In systems with no artificial subsoil drainage, the leaching fraction needs to be high enough to leach salts but not cause the water table to rise too close to the soil surface or cause excess groundwater contamination.

An ion found in secondary treated municipal wastewater that is a common groundwater contaminant under agricultural scenarios is nitrate. Nitrate is a soluble salt and is also a plant available form of nitrogen. If nitrate is not removed from wastewater by plants or through denitrification, it is subject to leaching. Excess nitrate in groundwater can be detrimental to human and environmental health. The limit for nitrate in drinking water is 10 mg L^{-1} and consumption of nitrate contaminated water may lead to *methemoglobinemia* in newborn infants (Health Canada 2010; USEPA 2011). Nitrate in surface water promotes algal blooms termed eutrophication, reducing the dissolved oxygen and water quality (Brady 1990). Another plant essential nutrient found in wastewater with the potential to impact surface water quality is phosphorus. Plant available phosphorus in soil is typically in a form of phosphate (HPO_4^{2-} and $H_2PO_4^-$) (Brady and Weil 2004). Although phosphate has a negative charge it is relatively immobile in soil and the majority of phosphorus is not in the soil solution. In alkaline soils, available phosphate is predominantly bound to calcium compounds and in more acidic soils it is bound to iron and aluminum compounds (Brady and Weil 2004). Typically, soluble phosphate additions are quickly transformed into insoluble states. However, some studies have found phosphate leaching (Eghabll et al 1996; Olsen et al 2010). Phosphate leaching is a water quality concern in prairie watersheds (i.e. Lake Winnipeg). Like nitrate, phosphorus is a plant essential nutrient that can lead to potentially toxic algal blooms in surface water bodies.

An approach to assess the potential environmental impacts of SRC plantations irrigated with wastewater is to quantify drainage and its quality. There are various methods used for estimating drainage. The research presented in this chapter will employ an indirect physical based method, soil water balance, and an environmental and applied tracer method, the chloride mass balance to estimate drainage (Allison et al 1994).

2.1.1 Soil Water Balance

The soil water balance is a simple, indirect method of estimating drainage. The method involves monitoring the hydrologic components of the water cycle. Hillel (1998) describes the water balance as any change in the water storage of the soil during a specified time period as equal to the difference between water inputs and water outputs. The main water inputs are precipitation (P, mm), as rain or snow (snow water equivalent, SWE, mm), and irrigation (I, mm). Water losses include evapotranspiration (ET, mm), drainage (D, mm) and runoff (R, mm). In the absence of significant runoff, the soil water balance in its simplest form can be expressed as (Hillel 1998):

$$\Delta S = P + I - ET - D \qquad [2.1]$$

Where, ΔS is change in soil storage over a time period (mm), Δt (days, year, etc). Positive ΔS values represent an increase in soil water storage in the root zone and could be a result of precipitation or groundwater discharge. A negative ΔS represents a loss of soil water from the root zone as a result of drainage or ET.

Evapotranspiration is the combination of soil evaporation and crop transpiration (Raes 2009). Evapotranspiration can be estimated by determining potential evapotranspiration (PET), which then can be multiplied by a crop coefficient to determine crop evapotranspiration. Potential evapotranspiration can be estimated in numerous ways including evaporation pans, atmometers or using meteorological measurements like temperature and relative humidity (Maulé et al 2006). The most widely accepted model using meteorological data is Penman-Monteith but this requires a large number of parameters that are difficult and expensive to measure. Maulé et al (2006) developed an empirical regression equations using commonly measured meteorological data (i.e., min., max, and average daily air temperature) to estimate potential evapotranspiration on the Canadian prairies based on a Penman-Monteith model. These regression equations are convenient because they allow estimates of potential evapotranspiration for dry continental weather using limited data that is relatively easy to measure, but is more uncertain than direct estimates. Using a crop coefficient for the woody species, the actual crop evapotranspiration can be estimated.

2.1.2 Natural Tracers

Natural tracers can be used to estimate deep drainage in arid and semi arid regions (Allison et al 1994). Typically, a natural tracer is a naturally-occurring element deposited via rain or snow and once in the soil not altered or taken up by plants. Common natural tracers include: ^3H and Cl (Allison et al 1994). The chloride mass balance technique estimates drainage in the unsaturated zone using soil and atmospheric chloride concentrations (Allison et al 1994). The method assumes that over the long-term the mass flux of chloride at the soil surface is equal to the mass flux of chloride below the root zone (Allison and Hughes 1978). The chloride mass balance in the vadose zone can be simply expressed as:

$$\overline{P}\,\overline{C_p} = \overline{D}\overline{C_s} \qquad [2.2]$$

Where

$\overline{P}$ =mean annual infiltrating water (mm yr^{-1})

$\overline{D}$=mean annual drainage (mm yr^{-1})

$\overline{C_p}$=mean Cl^- concentration in infiltrating water (mg mm^{-3})

$\overline{C_s}$ =mean Cl^- concentration in soil water (mg mm^{-3})

Assumptions associated with Equation [2.2] are: 1) surface chloride flux is relatively constant with time and there are no chloride sources or sinks in the soil; and 2) flow is quasi steady state, one dimensional, downward piston flow (Allison and Hughes 1978). The Allison and Hughes (1978) model has some uncertainties. The model assumes that the concentration of chloride in the precipitation stays relatively constant which may be difficult to verify. Assuming one dimensional piston flow does not account for the processes of diffusion or dispersion which may be occurring but the occurrence of preferential flow may be more of an issue. Studies have found that diffusion could affect chloride concentrations below the root zone (Tyler et al 2006; Scanlon et al 1999).

Based on the principles of the chloride mass balance, Rose et al (1979) proposed a transient model to predict the short-term concentrations of chloride in the soil, following a change in the soil water balance. The Rose et al (1979) model was used because it was developed for irrigation systems in semi-arid regions. The Rose et al (1979) model uses measurements of soil chloride concentrations at two times in a soil profile with known input chloride concentrations to estimate drainage (Rose et al 1979). This approach assumes chloride is water soluble and there is complete mixing of soil and water and one dimensional downward flow (Silburn et al 2009). Applying the mass conservation principles, for a time period, the root zone chloride mass balance may be described by:

$$q_0 c = D\left(s_z - \frac{ds_z}{2}\right) = z\theta_{FC}\left(\frac{d\bar{s}(t)}{\delta t}\right) \qquad [2.3]$$

Where,

q_0=annual infiltrating water (mm yr^{-1})

D=leaching, or drainage, flux density at depth z (bottom of the root zone; mm yr^{-1})

c =chloride concentration in infiltrating water (mg mm^{-3})

z = root zone depth (mm)

$s_z(t)$=chloride concentration in soil at depth z (mg mm^{-3})

$\bar{s}(t)$=spatial mean solute concentrations in soil water in the top z cm of soil at time t (mg mm^{-3})

t=time measured from an initial time when $\bar{s}(t = 0) = \bar{s}_0$ (yr)

θ_{FC}= moisture content in root zone at field capacity (mm^3 mm^{-3})

Since drainage typically occurs at higher soil water contents, it is assumed that the volumetric water content is at field capacity, $\bar{\theta}_{FC}$ (Rose et al 1979). Another reason to use field capacity is that the irrigation system at the study site is presently set to maintain the root zone at field capacity. Expressed in differential form, Equation [2.3] becomes:

$$z\theta_{FC}\left(\frac{d\bar{s}}{dt}\right) = q_0 c - Ds_z \qquad [2.4]$$

The solution to Eq. [4] (substituting $\lambda = \frac{s_z}{\bar{s}}$) is:

$$\bar{s}(t) = \bar{s}_0 + (\frac{q_0 c}{D\lambda} - \bar{s}_0)(1 - e^{-\frac{D\lambda t}{\theta_{FC} z}}) \qquad [2.5]$$

It is assumed that λ is constant. The Equation[2.5] predicts average soil chloride concentrations in the root zone over time. Drainage can be determined by optimizing the *D* parameter to fit the soil concentration data obtained from field measurements.

2.2 OBJECTIVES

In 2006, the Canadian Forest Service (CFS) of Natural Resources Canada established a SRC plantation with wastewater irrigation project at Whitecourt, Alberta. The purpose of the project was to test and demonstrate the ability of a SRC willow crop to treat wastewater and as an alternative energy source. The challenge with SRC plantations with wastewater irrigation systems is to apply enough wastewater to leach salts from the root zone but minimize groundwater contamination and rising of the groundwater table. The objectives of this research are as follows:

I. Quantify the drainage below the root zone of the SRC plantation with wastewater irrigation using the soil water balance and the chloride mass balance model of Rose et al (1979).
II. Compare and contrast the drainage estimate methods used to quantify drainage below the root zone of SRC crops with wastewater irrigation systems.
III. Make recommendations to improve the efficiency and management of irrigation on SRC plantations with wastewater systems while limiting any adverse environmental impacts.

2.3 MATERIALS AND METHODS

2.3.1 Site Description

The study site was field monitored over the 2010 and 2011 growing season approximately from May to October. The study site is located in the north east section of the town of Whitecourt, Alberta, adjacent to the sewage treatment plant. Whitecourt is approximately 180 km northwest of Edmonton, Alberta, on Highway 43 (Figure 2.1). The study site is located at 01-01-60-012-W5 and has coordinates of approximately latitude 54 09 08 dms and longitude 115 39

04 dms. The study area is located in the Western Alberta Uplands, Clear Hills Uplands eco region of Alberta and the Lower Foothills natural sub-region (Agroclimatic Atlas of Alberta 2009; Natural Regions Committee 2006).

The climate of the Whitecourt area is characterized as humid continental with warm summers and long, cold winters (Longley 1968). The mean daily temperature at Whitecourt is 2.6 ^{o}C with extremes ranging from -41^{o}C in January to 33.5 ^{o}C in August (Figure 2.2). Average annual precipitation, rainfall and snow water equivalent, respectively, are as follows: 577 mm, 440.3 mm and 134.2 mm (Environment Canada 2010) (Figure 2.1). The majority of rainfall is between May and September, the greatest amount falling in July. Snowfall is primarily between November and March, with the greatest average amount occurring in January. The average amount of frost free days is about 65 (Wynnyk et al 1969). Annual average wind speed is 7.4 km h^{-1} and most frequently in the west direction.

The bedrock geology of the greater area consists of two continental shale/sand geologic formations, the Paskapoo Formation of Paleocene-Late Cretaceous age and the Wapiti Formation of Late Cretaceous age (Tokarksy 1977). The study site is situated on the Wapiti Formation from the Upper Cretaceous. The Upper Wapiti is characterized by grey, feldspathic, clayey sandstone; grey bentonititc mudstone and carbonaceous shale; concretionary ironstone beds; scattered coal and bentonite beds of variable thickness; and minor limestone beds (AGS 2009). The upper most bedrock unit is consolidated gray clay, about 11 m from surface. The study site is located in the Athabasca river flood plain, with an elevation of about 685 masl (Figure 2.3). Topography is gently rolling and aspect is north facing with a slope class between 2 and 5 %.

The soil in the study site area is classified as a Gleyed Dark Gray Chernozem of the High Prairie series (Wynnyk et al 1969). The soils are imperfectly drained and developed on recent alluvial material deposited by the nearby Athabasca River (Table 2.2; Wynnyk et al 1969). The measured solumn depth at the site varies with slope position from about 50 cm on slope shoulders to 1 m on slope toes. Below the solumn there is a thin layer of sand and a thick gravel layer (estimated at about 7 m thick). The measured groundwater table at the site ranges from 1.5 to 3.8 m below the ground surface (Figure 2.4). Piezometers on the site indicate that groundwater direction is mostly south in the spring and typically east in the summer and fall (Figure 2.5).

Prior to planting of SRC crop, the study site was an agricultural field planted with canola for at least 5 years. In 2006, five willow and two poplar clones were planted. The five willow clones were Charlie (*Salix miybeaba*), SX64 (*S. udensis*), SX61 (*S. udensis)*, SV01 (*S. dasyclados*) and Psuedo (*S. alba*) (Zsuffa 1995). The two poplar species were Brooks 1 and Green Giant, both *Populus deltoids* x *P. petrowkyana* clones.

The natural vegetation in the area is typical of lower elevation species in the lower foothills natural sub region of Alberta. There are mainly deciduous stands composed of: *Populus tremuloides* Michx., *Populus balsamifera* L. and *Betula papyrifera* Marsh (Natural Regions Committee 2006). Understory vegetation typically consists of *Vacciniumvitis-ideae* L., *Alnus viridian* D.C, *Epilobium angustifolium* L. and *Calamagrostis Canadensis* Michx (Natural Regions Committee 2006).

2.3.2 Experimental Design

The site consists of four zones, two receiving wastewater through irrigation and two receiving no irrigation (Figure 2.6). Within each zone, woody species are planted in paired rows with a subsurface irrigation line in the center of the row. The dimensions are roughly 60 cm between plants within paired rows and 2.2 m between rows (Figure 2.7). There are three adjacent rows of the same plant clone with 21 rows in each zone. In total there are about 6000 plants per zone.

The site is irrigated with secondary treated wastewater using a subsurface irrigation system. The irrigation line is buried about 25 cm below ground and runs north-south on the site. The length of the irrigation pipe is 80 m and the inner diameter is 1.4 cm. Outlets are situated every 60.96 cm and have a square shape with dimensions of 0.13 by 0.13 cm.

The woody species were planted in the spring of 2006 and the first irrigation year began that summer. The first harvest was in the winter of 2009 after three growing seasons. Irrigation is activated based on soil moisture potential sensors located in each irrigated zone. There are two Watermark® sensors, each in the same location in a paired row of cuttings. The sensors are located 20 cm below the soil surface and determine the average soil moisture potential in the rooting zone of the cuttings. Irrigation is designed to maintain the root zone at roughly field capacity, -0.5 bars. Irrigation rates in the two irrigated plots within the irrigated zone are independent of each other.

2.3.3 Soil Property Characterization

2.3.3.1 Chemical and Physical

In May and July of 2010, soil samples were taken from the irrigated and non-irrigated plots for soil property characterization. In May, undisturbed soil samples were taken using a Geoprobe drilling rig to depths between 50 and 100 cm, with a diameter of about 5 cm, encased in thin PVC tubing. The samples collected in May were used for neutron probe calibration and soil property characterization. They were split into 20 cm increments and weighed in the laboratory for moisture content. The samples collected in July were sampled using a hand auger with a diameter of about 3 cm to a depth between 50 cm and 100 cm. The July samples were used for characterization and were split up by horizon. Samples for characterization were analyzed for the following: electrical conductivity (EC), pH, cation exchange capacity (CEC), sodium adsorption ratio (SAR), nitrate-N ($N\text{-}NO_3^-$), ammonium-N ($N\text{-}NH_4^+$), major cations including calcium (Ca^{2+}), magnesium(Mg^{2+}), potassium(K^+) and sodium (Na^+), organic nitrogen (ON), organic carbon(OC), chloride (Cl^-) and percent organic matter (OM %). A selected number of historical samples from 2007-2009 were submitted for EC and chloride analysis. Electrical conductivity and soluble salts were analyzed with a saturated paste method based on the method presented by Miller and Curtin (2007). Total nitrogen was determined by Leco Combustion, as outlined in and Total Carbon, Organic Carbon and OM analysis was based on McKeague (1978). Particle size distribution for each sample was determined using a hydrometer, as outlined by Kaddah (1974) and Angers and Larney (2007). Readings were taken after 40 sec and 7 hours. Calculations were corrected for hydrometer and the dispersing agent. Percent sand, silt and clay were calculated and a texture class was assigned using the texture classification triangle (Toogood 1958).

2.3.3.2 Moisture Retention Curve

The moisture retention curve is the relationship between soil volumetric water content ($L^3\ L^{-3}$) and soil matric potential (L). It was derived using a combination of laboratory measurements (suction table) and field measurements (the Instantaneous Profile Method (IPM)). Data from both methods were combined to form segments of the moisture retention curve for the study site. Pressure plates were not used to determine the drier ends (-1 to -50 m) of the moisture retention curve as these conditions are typically not found in field conditions.

The suction table was used for determining near saturated (>-0.91 m) water head points of the moisture retention curve (Romano et al 2003; Reynolds and Topp 2007). Undisturbed soil core samples were collected in 2010 from IPM locations around the study site using a Geoprobe drilling rig. Cheese cloth was secured to the bottom of undisturbed soil cores of varying lengths with a diameter of about 5 cm. The cores were saturated from the bottom up by gradually adding temperature equilibrated de-aired tap water to the cores over 3 to 4 days, depending on core length. The cores were then weighed and the saturated mass of each core was recorded. The bottom of the core was placed on the saturated suction table and hydraulic contact was established using a silica sand slurry (Reynolds and Topp 2007). The cores were allowed to equilibrate and covered with plastic to prevent evaporation. After a 48 hour equilibration period, the difference between the water level in the burrette and the top of the table was recorded as the matric potential and the core weight recorded. This process continued with lower placements of the burrette each time. A pressure plate was not used to determine the dry end of the curve because it was assumed that the soil on the study site would always remain in the wet end of the moisture retention curve, similar to field conditions. The oven dry weight of the cores was determined at the end of the experiment. Using the oven dry weight, the gravimetric water content, θ_g (M M^{-1}), and volumetric water content, θ_v(L^3 L^{-3}), of each core was determined (Reynolds and Topp (2007).

The IPM is an in-situ, field-based technique for determining the moisture content-matric potential relationship and estimate hydraulic conductivity (Reynolds 2007a). Four locations around the treatments plots were selected for the IPM hydraulic property measurement (Figure 2.6). The locations were selected to cover the spatial variability of soil hydraulic properties. In May 2010, tensiometers were installed at depths of 10, 30, 50 and 90 cm, coinciding with the varying soil horizons, to measure the soil potential. Boreholes were created manually and backfilled with a silica flour slurry to create contact between ceramic cup and soil and prevent preferential flow down borehole. Tensiometers were filled with de-aired water. Two neutron probe access tubes and TDR steel rods were installed in close proximity to the tensiometers for moisture content readings. Access tube boreholes were created using a Geoprobe drilling rig and maintained by thin walled aluminum tubes to a depth of about 1 m. Vegetation was cleared and the soil surface covered with wood chips and a tarp to prevent any evaporation or transpiration. An area about 3 m by 3 m was enclosed with landscape edging to allow ponding of water. “Tap” water was applied uniformly using watering canisters and ceased at near saturated field. At varying time intervals after flooding stopped, paired measurements of moisture content and soil potential at each depth was monitored. Soil total potential was sampled from the tensiometers using a calibrated pressure transducer (i.e. tensimeter, Soil Measurements Systems, Arizone). The tensimeter readings were corrected so all total head readings were reflective of similar reference elevation. The matric potential was determined using the total head potential and gravimetric potential.

Moisture content was measured using time domain reflectometry (TDR) and a neutron probe (CPN 503DR Hydroprobe). Stainless steel TIG welding rods (1.66 mm diameter, 15 cm length) were placed in the A horizon at a depth of about 9.5 cm for determining soil moisture content. The steel rods were between 2 and 3 cm apart. The apparent dielectric permittivity was sampled manually and estimated from TDR waveforms created by Tektronix 1502 B or 1502 C cable tester (Tektronix, Beaverton, OR) (Dyck and Kachanoski 2009).

A neutron probe was used because it is rapid, non-destructive and allows monitoring of the same locations and depths at a point (Hillel 1998). During access tube installation, soil cores were sampled and returned to lab for bulk density and water content measurements. Upon access tube installation, neutron probe readings were obtained every 20 cm to the bottom of the access tube for the calibration. A count ratio was determined as the count at a specific depth over the standard count. A linear relationship between the count and the measured soil water content from the soil cores was established. The relationship can be expressed as:

$$\theta = 0.1459 \frac{depth\ count}{standard\ count} \qquad r=0.91\ \ (\alpha=0.05) \qquad [2.6]$$

2.3.3.3 Saturated Hydraulic Conductivity

The soil on site was of finer texture so saturated hydraulic conductivity was measured using the falling head method developed by Reynolds (2007b). Undisturbed soil core samples were collected in 2010 from IPM locations around the study site using a Geoprobe drilling rig. The cores were split up into lengths between 5 and 15 cm. The undisturbed soil cores were prepared for the experiment by covering the bottom with cheese cloth. Soil cores were saturated using the same methods as mentioned for the hanging water column experiment, gradually adding temperature equilibrated, de-aired tap water to the cores over 3 to 4 days. A plumbing couple link with similar diameter to the core was attached to the top of the core and to a water reservoir with similar diameter. The inner diameter of the core was about 4.95 cm and the reservoir was 5.095 cm. The soil cores were fixed vertically to a wall mount using clamps with the soil core positioned on the bottom and the water reservoir on top. A basin was placed below the soil core to catch the infiltrating water. Water was placed in the reservoir and the height of the ponded water was taken at regular time intervals. Saturated hydraulic conductivity was calculated using Reynolds (2007b) falling head method. Measurements of head levels were performed between 10 and 12 times and averaged for each depth.

2.3.4 Soil Water Balance Data

2.3.4.1 Soil Water Storage

Soil moisture was monitored to determine the change in soil water storage in the root zone and below for the soil water balance in 2010. Eight soil moisture monitoring locations were positioned on the study site, four in the control and four in the wastewater irrigated treatment. At each monitoring location, soil moisture was determined beside the irrigation line source to varying depths. Shallow soil moisture was determined with TDR and deeper with a neutron probe, similar to the methods discussed in Section 3.3.2. Soil moisture was sampled weekly from May to October, 2010 and a standard count was taken every sampling day. The calibration relationship (Equation [2.6]) was used to determine volumetric moisture content. Soil storage was determined for each sampling location by summing the moisture contents over all depths and multiplying by depth. Change in storage was determined by determining difference between two time periods.

2.3.4.2 Evapotranspiration

A meteorological (MET) station was established on the research site in 2006 and there is a Government of Canada weather office in the area. From the MET station on site, daily precipitation and temperature was available for 2010. Precipitation was collected using in a tipping bucket. Meteorological data for 2007, 2008, 2009 and 2011 was obtained from the Government of Canada weather office in Whitecourt.

A temperature-based model from Maulé et al (2006) was used to determine potential evapotranspiration (ET_o) using the MET data available. ET_o is the evapotranspiration (mm day^{-1}) from an extensive surface of green grass of uniform height, actively growing, well watered and completely shading the ground (Irmak and Haman 2003). The Maulé et al 2006 model has the form:

$$ET_o = 0.134(T_{max} - T_{min}) + 0.0109T_{avg} + 0.708\,\Delta R_a - 0.669 \qquad \text{SE} = 0.90 \text{ mm day}^{-1} \qquad [2.7]$$

Where

T_{max}=maximum daily temperature (˚C)

T_{min}=minimum daily temperature (˚C)

T_{avg}=average daily temperature (˚C)

R_a=extraterrestrial radiation (MJ m^{-2} d^{-1}) from Smithsonian Institute (1951) tables

Δ = slope of the saturation vapour pressure temperature curve at temperature, T_{avg}, (kPa ˚C^{-1}); given by

$$\Delta = \frac{2504 \cdot e^{\frac{17.27\,T_{avg}}{T_{avg}+237.3}}}{(T_{avg}+237.3)^2} \qquad [2.8]$$

SE=standard error of estimate by model

The crop evapotranspiration (ET_c) can be estimated by multiplying ET_o by a crop coefficient (k_c).

2.3.5 Calculations

2.3.5.1 Soil Water Balance Calculations

The soil water balance was calculated on a weekly basis for 2010 because daily irrigation data was available for this year. For 2006 through 2009 and 2011, daily irrigation data was not available so the water balance was calculated on a growing season basis assuming that ΔS = 0. Note for 2010, the dates used for calculations were April 27 to October 28 while for the remaining years a full calendar years was used.

Irrigation was determined by dividing the irrigation volume for a specific day or year by the total area of the irrigated zone (7040 m^2). It was assumed that the line source supplied the entire area with irrigation water evenly because the study site has a high clay content which draws water laterally away from the irrigation line sources. Also, moisture content readings taken between irrigation rows were similar to measurements beside the line. Most fine textured soils such as these have a Gardner-alpha (Gardner 1958) value between 0.1 and 0.3 cm^{-1} (Reynolds 2008c). The Gardner-alpha value estimated for this soil is 0.27 cm^{-1}. Based on analytical solutions to the 2D Richard's equation under line source boundary conditions (Zhang et al 2000), there is likely significant overlap of the wetted area surrounding each irrigation line.

To account for snow water contributions to the soil water balance, Environment Canada estimates of snow water equivalent (SWE) were summed up over the winter months. The Environment Canada data estimates SWE based on the snow that has fallen, but does not account for losses due to sublimation and translocation. Based on estimates from Pomeroy (1995), we assumed that 50% of the incident SWE that fell over the winter months was lost to sublimation and translocation. Thus, the P in Eq. [2.1] is the sum of growing season precipitation and 50% of the winter SWE.

The evapotranspiration model of Maulé et al (2006) is valid only for periods when plants are actively transpiring. This model was used to estimate daily evapotranspiration during the growing seasons (April 30 to October 1) of the years in question (2006 – 2011). Crop coefficient values used for $\mathrm{ET_c}$ estimates ranged from 0.75 to 1.2, depending on age and time since coppice (Table 2.5). A $\mathrm{k_c}$ of 0.75 was used for 2009 and 2006 because this was the first growing season year and the first year after coppice. A higher $\mathrm{k_c}$ of 1.2 was used for the oldest plant. A typical $\mathrm{k_c}$ range for willow is between 0.6 to 1.7 and for poplar between 0.4 and 0.8 (Pistocchi et al 2009; Persson 1995). Numerous studies have found that the $\mathrm{k_c}$ for willow and poplar increase with age, mainly due to variations in leaf area index (LAI) (Persson 1995; Afas et al 2005; Fang et al 1999; Pontailler et al 1999). In a study by Fang et al (1999) the LAI of three poplar clones increased gradually over 6 years of growth. The $\mathrm{k_c}$ in the growing season after coppice has been found to be greater than during the establishment year. Pontailler et al (1999) found that certain coppiced poplar clones were able to reach similar LAI values of a mature forest stand.

Drainage was estimated using the soil water balance equation (Eq. [2.1]). Weekly drainage values were summed to yield total growing season drainage for a particular soil monitoring location. For estimating the drainage for past irrigation years (2006-2011) using the soil water balance, ΔS was assumed to be zero over the growing season and the growing season totals for remaining parameters were used. A positive drainage indicated a downward flux and a negative drainage indicates an upward flux.

2.3.5.2 Chloride Mass Balance Calculations

Due to the short time period of irrigation in 2010, the transient Rose et al (1979) model (Eq. [2.5]) was used as another method to estimate drainage. The Rose et al (1979) model was used to estimate drainage on an annual basis and over the 5 year period since irrigation commenced. For annual drainage (D_a), the initial average profile soil water chloride concentration was assumed to be equal to the value measured from the previous year and all other parameters (q_0, c and λ) also varied from year to year. Drainage (D in Eq. [5]) was then estimated by matching the $s(t=1)$ predicted by Eq. [2.5] with the measured value of $s(t=1)$ for next year.

The irrigation system was installed during the growing season of 2006. Irrigation application in 2006 was much smaller compared to other years. For Rose et al (1979) model calculations, the first full year of irrigation, 2007, was considered to be year 1 (i.e. t=0 is the start of the 2007 growing season). Therefore, the initial root zone soil chloride concentration, s_0 (Eq. [5]) was based on samples taken from the control (un irrigated) plots in 2007.

The 5-year average, drainage was estimated using the initial profile soil water concentration and 5-year average q_0, c and λ values. Drainage (D in Eq. [2.5]) was then estimated by matching the $s(t=5)$ predicted by Eq. [2.5] with the measured value of $s(t=5)$ for 2011.

Parameters for the Rose et al (1979) model were determined for 2007 to 2010 from soil measurements and metereological data measured on-site and from nearby Environment Canada MET stations. The irrigation system was installed during the growing season of 2006 but this year had very low irrigation amounts. For Rose et al (1979) model calculations, the first full year of irrigation, 2007, was considered to be year 1 (i.e. t=0 is the start of the 2007 growing season). Therefore, the initial root zone soil chloride concentration, s_0 (Eq. [2.5]), was based on samples taken from the control (un irrigated) plots in 2007. Soil samples from the irrigated soils provided values of s_z and s_m and control plot soil for samples values of s_0. The soil samples collected from 2007 to 2011 were analyzed for soil chloride using the saturated paste method and solution chloride form wastewater was determined using ion chromatography (Dionex 600) (Miller and Curtin 2007; Tabatabai and Basta 1991). The concentration of chloride in the infiltrating water (C_i) was calculated as the weighted mean of chloride concentrations in the irrigation, rain and snow using the following equation:

$$C_i = \frac{C_I*I+C_r*R+C_{snow}*(0.5*SWE)}{I+R+(0.5*SWE)} \qquad [2.9]$$

Where,

C_I is concentration of chloride in irrigation water (mg mm^{-3})

I is amount of irrigation water (mm)

C_r is concentration of chloride in rain water (mg mm^{-3})

R is amount of Rain water (mm)

C_{snow} is concentration of chloride in snow water (mg mm^{-3})

SWE is the snow water equivalent (mm)

Infiltrating water is the sum of irrigation, precipitation and infiltrating snow water. Infiltrating snow water was estimated as 0.5* snow water equivalent (SWE) to account for water losses from ablation, melt or evaporation, and sublimation, direct transformation of snow to water vapour (Pomeroy et al 1999). Wastewater samples were collected biweekly and analyzed for chloride using ion chromatography. Rain water chloride was estimated to be 0.041 mg L^{-1} based on a study by Hayashi et al (1998). Snow water chloride was estimated to be 8.86 x 10^{-5} mg L^{-1} from findings by Lilbaek and Pomeroy (2007). Irrigation was determined by dividing the irrigation volume for a specific day or year by the total area of the irrigated zone (3520 m^2). Infiltrating water volumes for each year were the sum of precipitation, SWE and irrigation. The rooting depth of a mature willow typically reaches 90 to 130 cm with up to 90 % of roots found in top 10 to 20 cm of the soil (Persson and Jansson 1989; Volk et al 2001). Rooting depth of willow and poplar has been linked to water stress (Rytter and Hansson 1996). Based on the age of the trees at the site and the lack of water stress for the plants receiving wastewater a rooting depth of 50 cm was chosen.

2.4. RESULTS AND DISCUSSION

2.4.1 Results

Various soil chemistry parameters were assessed for the control and wastewater irrigated zones in 2010 (Table 2.1). In general, the wastewater irrigated zones had higher major cations, TOC and OM in the topsoil while the control had higher amounts of these in the subsoil. All depths were classified as either silty clay loam or silty clay. The depths of 10 to 30 cm and 50 to 80 cm had the highest clay content, 42 and 40 %, respectively (Table 2.2).

Hydraulic properties were characterized for the study site. The moisture retention curve properties differ with depth (Figure 2.8). The slope of the moisture retention curve is greatest for depths at 10 cm and 80 cm below ground surface. Saturated hydraulic conductivity varied with depth (Table 2.3). The highest saturated conductivity was between the 0 to 10 cm and 70 to 80 cm depths below ground surface, with values of 8.5 x10^{-5} and 9 x10^{-5} m s^{-1}, respectively, typical of a silt (Hillel 1998). The lowest saturated hydraulic conductivity was evident at the 20 to 30 cm and 30 to 40 cm depths with values of 3.3 x10^{-6} and 1.3 x10^{-5} m s^{-1}, accordingly, typical of a clay (Hillel 1998). The highest variability in saturated hydraulic conductivity measurements was at the surface soils, from 0 to 10 cm, with a coefficient of variance value of 6.5 $x10^{-5}$ m^2 s^{-3}. The lowest saturated hydraulic conductivity variance was 1.4 $x10^{-5}$ m^2 s^{-3} for depths 90 to 100 cm (Table 2.3).

Components of the 2010 and 2011 soil water balance method were monitored weekly over the growing season. Please note that there was no weekly irrigation data available for 2011 so the water balance uses annual amounts only. In 2010 and 2011, respectively, irrigation commenced July 14 and June 6 and was off September 28 and August 16. The total annual precipitation for 2010 was lower than 2011 with 50.2 and 70.5 cm, respectively, compared to the long term normal of 57.7 cm. Note 2011 has a higher than normal precipitation amount. Irrigation for 2010 was lower than 2011 with 10.4 cm as opposed to 11.1 cm (Table 2.4). Total crop evapotranspiration was 57.31 cm in 2010 and 53.6 cm in 2011 with the highest levels found in June for 2010 and September for 2011. The greatest precipitation events in 2010 were on May 20 and July 22 with about 40.7 and 39.2 mm, respectively, and in 2011 were August 7 and 8 with 48.4 and 63.6 mm, respectively (Figure 2.9). The wastewater irrigation zone had higher average soil water storage than the control zone in 2010 while the control had slightly higher storage than the wastewater irrigated zone in 2011. This may be due to the above normal precipitation that occurred in 2011. The average change in soil storage for both irrigation zones over the growing season under the control was -0.03 cm and wastewater was -0.09 cm in 2010 and in 2011 the average change in storage in the control and wastewater irrigated soil were 0.12 and 0.06 cm, respectively. In 2010 prior to commencement of irrigation, the sum of drainage was -19.87 cm in the wastewater irrigated plot but only 10.72 cm for the remaining weeks that the irrigation system was turned on. The control drainage values ranged from -5.12 to 7.38 cm over the growing season in 2010. The sum of the drainage in 2010, under the wastewater irrigated zones for the entire growing season was estimated to be -9.2 cm and about -27 cm under the control plots.

Figure 2.10 summarizes the cumulative components of the weekly soil water balance over the 2010 growing season. As expected, the slope of the cumulative precipitation curve is always greater than or equal to zero for both years because precipitation is always zero or greater. The slope of the cumulative ET_c curve is always positive indicating there is always evaporative demand for water during the growing season. Irrigation was not applied to the WW plots until July in 2010 resulting in the slope of cumulative irrigation being zero, and then greater than or equal to zero between July and September and then zero once irrigation was discontinued for the winter. Cumulative drainage for both irrigated and control treatments was negative in 2010 indicating a net water deficit over the growing season but there were positive drainage events indicating periods where the cumulative drainage curve shows a positive slope. Large positive drainage values occurred after large precipitation events in 2010. At the onset of irrigation in both years, the cumulative drainage values gradually increase. The influence of added irrigation on weekly drainage is especially apparent between July and August.

The soil water balance was used to estimate historical drainage amounts based on growing season precipitation, irrigation, estimated evapotranspiration and assuming ΔS = 0 (Table 2.5). Irrigation quantities and WW drainage amount vary from year to year (Figure 2.11). The lowest amount of irrigation applied was in the first start-up year, 2006, with 4.19 cm and the highest was in 2008 with 92.5 cm. The higher drainage years were in 2007 and 2008 with 79.2 and 68.0 cm, respectively, and the lower in 2006, 2009 and 2011 with -3 cm, 39.7 cm and 28 cm, respectively. Drainage under the control only occurred in 2011 with 16.9 cm due to higher precipitation than normal. In the remaining years, drainage under the control was -7.2, -4.7, -24.5 and -6.8 cm for 2006, 2007, 2008 and 2009, respectively. Drainage for the 2010 growing season was also estimated using total growing season precipitation, ET and irrigation and assuming ΔS = 0. The resulting 2010 drainage was estimated to be 3.3 cm, higher than the sum of the growing season drainage total for drainage estimate of -9.2 cm. The results differ due to a longer time period used for the annual calculations, rather than the growing season. The annual method used a whole calendar year while the growing season method used dates from April 5 to October 10. Drainage under the control assuming ΔS = 0 was -7.1 cm in 2010.

The Rose et al (1979) model provided a 5 year average and annual drainage estimates following a change in the soil water balance resulting from the implementation of irrigation. In 2010 and 2011 annual drainage estimates were -18.5 and 8 cm and the 5 year average was 19.4 cm (Table 2.5). In 2009, 2008 and 2007 the Rose et al (1979) model annual drainage estimates were 67.2, 21.8 and 34.1 cm yr^{-1}, respectively.

The Rose et al (1979) model was also used to predict the 5 year average and annual root zone solute concentrations from irrigation (Figure 2.12). All the years except 2010 predicted a long term trajectory to a new steady state chloride concentration. The predicted time the new steady state conditions are reached with 2007 being the earliest after 2.9 years of irrigation followed by 2009 with 4.2 years, 2008 with 5.1 years, the 5 year average with 7.3 years, 2011 with 14 years and 2010 with 10+ years (Table 2.6). All years except 2009 showed increased soil chloride concentrations since irrigation was initiated at the site.

2.4.2 Discussion

Drainage quantities were estimated using two methods: the soil water balance and the Rose et al (1979) model. For the Rose et al (1979) model, drainage was estimated using annual changes from year to year and a 5 year average drainage. The estimated drainage results from the Rose et al (1979) model vary from the soil water balance method due to various uncertainties with each method. Both models shared uncertainties. One of the soil water balance uncertainties is in the accuracy of the infiltrating water components and evapotranspiration. The amount of infiltrating snowmelt was estimated as 0.5*SWE due to losses from snowmelt runoff and sublimation. A study by Pomeroy et al (1999) in the Yukon estimated 38 to 45% of water lost to sublimation. Another source of uncertainty with both methods is that they assume application of irrigation water is uniform. Ayars et al (1999) found that the uniformity coefficient of a subsurface drip irrigation system ranged from 76% to 96%. The outlet points along the irrigation line were spaced every 60.96 cm, further reducing the uniformity of applied irrigation water. Another source of uncertainty is that interception was not accounted for. Interception on a SRC plantation in Britain reported 21 % of rainfall was intercepted while a similar study in Italy reported 26 % interception (Hall and Allen 1997; Tarsia 1980). However, the irrigation on site is injected in the subsurface so interception only applies to rainfall.

2.4.2.1 Water Balance Method

The main drawback with using the soil water balance method for estimating drainage is determining the crop evapotranspiration, ET_c. There is uncertainty in both the model by Maulé et al (2006) and in the crop coefficient, k_c, values used to estimate ET_c. Given that there is minimal meteorological data available, a temperature based model created by Maulé et al (2006) was used for estimating ET_c. The standard error for the Maulé model was 0.9 mm d^{-1}. Over a growing season of about 100 days this amounts to about 90 mm of uncertainty in ET_c. There are also uncertainties with the k_c value used to estimate ET_c. The k_c values used in calculations changed to reflect plant age but the plants on site also varied in terms of species type and health (Pistocchi et al 2009; Persson 1995).

Using the soil water balance method, drainage was calculated on a weekly basis for 2010 and on a growing season basis for 2006 to 2011. In general, the drainage estimates from the soil water balance are sensitive to the amount of infiltrating water. Years with high infiltrating water, like 2007 and 2008, also had higher drainage values. The weekly estimate using growing season times and field storage calculations for 2010 differed from the annual 2010 estimate. This is most likely due to difference in dates used for calculations. The drainage determined annual using yearly totals has a full calendar year of data while the weekly soil drainage estimates was just over the growing season. When using a full calendar year in calculations, irrigation and evapotranspiration values are similar to the weekly, however precipitation is larger, resulting in a greater drainage estimate. Also this may be due to timing of first soil moisture content monitoring when determining the weekly drainage estimate. The first soil moisture reading taken for the weekly estimate was after the ground was free of snow. During snowmelt there may have been drainage occurring that the weekly estimate was not able to monitor but the annual drainage estimate captured.

2.4.2.2 Rose et al (1979) Model

The Rose et al (1979) model has uncertainties due to the various model assumptions. For the 5 year average drainage the irrigation parameter was an average of the previous year's amounts while the annual drainage estimate for the Rose et al (1979) model and the soil water balance used the actual amount of irrigation applied. The actual annual irrigation amounts varied greatly from 2006 to 2010. From this, the Rose et al (1979) model 5 year average drainage estimates was an average over the years, and doesn't give any indication of year-to-year variability. Another uncertainty with the Rose et al (1979) model may be from spatial variability in soil sampling. The soil samples collected for the model were taken randomly between irrigation lines, a space of about 160 cm. In general, chloride concentrations decrease as you move away from the irrigation lines. Samples not taken close to the irrigation line may result in an overestimate of drainage.

The Rose et al (1979) model had consistently lower drainage estimates than the WB except in 2009. The coefficient of correlation between the two methods for annual drainage estimates is 0.57 (P=0.31). This may be due to uncertainties in evapotranspiration estimates in the soil water balance calculations. In the fall of 2008, the trees were harvested resulting in much smaller trees and reduced leaf area in the stand for the 2009 growing season. The crop coefficient used in the calculations for this year may not be an accurate representation. This lower actual ET in 2009 would likely result in higher drainage and flushing of chloride from the root zone which is consistent with the observed lower chloride concentration for this year.

As mentioned earlier, the estimation of evapotranspiration is one of the main drawbacks with the soil water balance. A benefit with the chloride mass balance method is that evapotranspiration does not need to be calculated as the soil chloride concentrations in the root zone reflect this. For this main reason, the chloride mass balance estimates of drainage are considered to the most representative of the study site.

The Rose et al (1979) model may be used to extrapolate soil chloride levels beyond the year in question. The long-term trajectory of the soil chloride levels predicted by the Rose et al (1979) model may be used to make inference about current irrigation management (Rose et al 1979). For example, the 1-year simulations for 2007-2008, 2008-2009 and 2009-2010 all predict a new steady state soil root zone chloride concentration at times greater than four years after the commencement of irrigation. For the 2009-2010 growing season, however, a breakdown in irrigation equipment resulted in much less irrigation being applied than the previous year. The long-term trajectory for this simulation is a constantly increasing root zone chloride concentration for an indefinite period, indicating the potential for salt loading in the root zone if wastewater irrigation was continued at this rate. The long-term trajectory of the five year simulation, however, shows that the soil chloride levels again approach a new steady state at times greater than four years after the beginning of irrigation. In other words, despite the annual variability in irrigation application rates, the Rose et al (1979) model predicts that the long-term risks of salt loading in the root zone are low. Long-term monitoring of soil salt levels is still advised to verify this prediction. A more in-depth assessment of salt loading in the root zone as a function of irrigation amounts will be explored in the following chapter.

The water balance and Rose et al (1979) model methods gave a range of drainage estimates. Both methods yielded similar results, values on the same order of magnitude. The most limiting aspect of the WB method is estimating ET while the Rose et al (1979) model is limited by soil chloride spatial variability. A benefit from the Rose et al (1979) model is it can be a management tool for predict future drainage rates.

2.5 CONCLUSIONS AND RECOMMEDATIONS

The general objectives of this research were to quantify and qualify the drainage under SRC with wastewater irrigation systems. More specifically the objectives of this chapter are to quantify drainage, compare two varying methods used for estimating drainage and make recommendations based on findings to improve the sustainability of these projects.

Two methods were used to estimate drainage, the soil water balance and the chloride mass balance. The soil water balance involved monitoring the various aspects of the soil water balance. Throughout the growing season of 2010 and 2011, precipitation, irrigation, evapotranspiration and soil storage were monitored and drainage estimated. Estimates of drainage for the previous irrigation years were made using historical meteorological data and assuming change in storage, ΔS, was zero. The chloride mass balance is based on the mass conservation of a non-transformed, non-absorbed solute like chloride (Rose et al 1979). A simple model proposed by Rose et al (1979) was used to estimate annual and 5 year average drainage. Soil samples and wastewater samples were analyzed for chloride concentrations for the Rose et al (1979) model.

A range of drainage estimates for 2010 and previous irrigation years were determined using the water balance and chloride mass balance Rose et al (1979) model. Years with large irrigation events typically have higher drainage below the root zone. The Rose et al (1979) model predicted that soil salinity will be a problem for irrigation years similar to 2010 and larger irrigation volumes and leaching fractions need to be maintained to prevent salt accumulation in the root zone. It is recommended to monitor soil salinity throughout the life of SRC with wastewater irrigation sites. Both methods indicated significant drainage in the system recieving wastewater irrigation. Due to the drainage occurring on site, it is recommended that fertilizer additions be carefully managed to prevent contaminant transport to the groundwater. Additional information is needed on the quality of the drainage water and salinity of the soil to make further recommendations. By contributing to and advancing the knowledge about SRC with wastewater irrigation systems the longevity of these projects is expanded.

2.7 TABLES

Table 2.1. Soil chemical properties at various depths for control and wastewater zones at the SRC plantation with wastewater irrigation study site in Whitecourt, AB. Note samples were collected prior to start of irrigation in 2010.

								Major Cations (mg kg^{-1})						
Zone	Depth (cm)	pH	EC (dS m^{-1})	SAR	TOC (%)	OM (%)	Total Nitrogen (%)	Calcium	Magnesium	Sodium	Potassium	Nitrate-N	Ammonium-N	Sulfate
Wastewater Irrigated	0-10	6.1	0.8	0.2	4.8	9.5	0.4	131.4	21.7	9.0	12.7	6.5	3.5	28.0
	20-45	6.5	0.4	0.2	2.1	4.1	0.2	43.3	7.5	5.4	2.5	3.6	2.1	10.2
	45-60	7.0	0.3	0.2	1.3	2.5	0.1	31.1	6.3	4.0	1.7	2.9	1.8	9.6
	60-100	6.9	0.3	0.2	1.2	2.5	0.1	34.3	6.9	4.8	1.8	3	1.9	9.5
Control	0-10	6.2	0.6	0.3	4.5	8.9	0.4	92.3	16.1	10.6	13.0	3.27	3.7	20.8
	20-45	6.8	0.3	0.5	1.6	3.3	0.1	34.0	6.0	8.5	1.4	2.69	2.2	10.9
	45-60	7.4	0.3	0.6	1.1	2.1	0.1	22.5	5.0	9.5	1.0	1.93	1.8	10.5
	60-100	7.0	0.4	0.3	1.2	2.4	0.1	43.7	8.6	8.6	1.8	1.44	1.8	12.4

Table 2.2. Particle size classes by hydrometer and texture class for the control soil from 2010 at the SRC plantation with wastewater irrigation project in Whitecourt, AB.

	Particle Size Class			
Depth Range (cm)	% Sand	% Silt	% Clay	Texture Class
0-10	15.3	48.9	35.8	Silty Clay Loam
10-30	7.5	50.3	42.2	Silty Clay
30-50	4.6	56.5	39.0	Silty Clay Loam
50-80	11.0	48.7	40.3	Silty Clay
80+	7.1	56.5	36.4	Silty Clay Loam

Table 2.3. Saturated hydraulic conductivity by falling head method for varying depths for the control soil at the SRC plantation with wastewater irrigation site in Whitecourt, AB, in 2010. Note standard deviation and coefficient of variation were determined assuming normal distribution of data even though some data was not normally distributed. The assumption was applied to get a general idea of data spread.

Depth (cm)	Mean (cm s^{-1})	Standard Deviation (cm s^{-1})	Coefficient of Variation (%)
0-10	0.010	0.025	252.0
10-20	0.018	0.018	104.6
20-30	0.004	0.034	786.7
30-40	0.002	0.003	102.6
40-50	0.008	0.010	131.8
50-60	0.010	0.018	181.1
60-70	0.006	0.007	130.4
70-80	0.010	0.007	66.0
80-90	0.005	0.003	55.5
90-100	0.007	0.002	29.2

Table 2.4. Soil water balance (WB) and Rose et al (1979) model parameters and drainage for the SRC plantation with wastewater irrigation project at Whitecourt, AB.

Year	Annual Precipitation (cm)	Actual Irrigation (cm)	Reference ET (ET_o) (cm)	k_c[a]	Crop ET (cm) [ET_o*k_c]	WB Irrigation Drainage (cm)	WB Control Drainage (cm)	Rose et al (1979) model Annual Drainage (cm)
2006	33.3	4.2	53.9	0.75	40.5	-3	-7.2	na
2007	49.4	83.8	49.2	1.1	54.1	79.2	-4.7	34.1
2008	38.3	92.5	52.3	1.2	62.8	68.0	-24.5	21.8
2009	32.0	46.4	51.7	0.75	38.8	39.7	-6.7	67.2
2010*	32.2	10.4	57.3	1.0	57.3	-9.17	-27	
2010	50.2	10.4	57.3	1.0	57.3	3.3	-7.1	-18.5
2011	70.5	11.1	44.7	1.2	53.6	28	16.9	11.7
5 Year Average								19.4

*drainage estimated using weekly measurements of storage; dates April 5 to as of Oct 10, 2010
[a] crop coefficient
Note water balance (WB) drainage estimate were calculated using full calendar year used and assuming ΔS=0 except 2010*

Table 2.5. Parameters for the Rose et al (1979) model for SRC plantation with wastewater irrigation project at Whitecourt, AB.

Parameter	Units	2007	2008	2009	2010	2011	5 Year Average
$\bar{s}_0$	mg mm^{-3}	8.69E-05	1.517E-04	2.326E-04	5.286E-05	1.81E-04	8.69E-05
s_z	mg mm^{-3}	1.64E-04	2.65E-04	5.57E-05	1.67E-04	1.69E-04	1.69E-04
$\bar{s}$	mg mm^{-3}	1.52E-04	2.33E-04	5.29E-05	1.81E-04	1.45E-04	1.45E-04
λ		1.0787	1.14	1.05	0.92	1.16	1.07
z	mm	500	500	500	500	500	500
θ_{FC}	mg mm^{-3}	0.4	0.4	0.4	0.4	0.4	0.4
q_0	mm yr^{-1}	1332.3	1307.2	784.4	566	683.74	924.47
c	mg mm^{-3}	4.53E-05	5.06E-05	4.27E-05	1.28E-05	1.14E-05	3.26E-05
t	years	1	1	1	1	1	5

Table 2.6. Average time in years to reach steady state and steady state concentrations using the Rose et al (1979) model for the SRC plantation with wastewater irrigation project at Whitecourt, AB.

Year	Average (years)	Concentration (mg mm^{-3})
2007	2.90	1.64E-04
2008	5.10	6.52E-04
2009	4.20	4.75E-05
2010	15+	n/a
2011	14.00	8.42E-05
5 Year Avg	7.3	1.45E-04

2.8 FIGURES

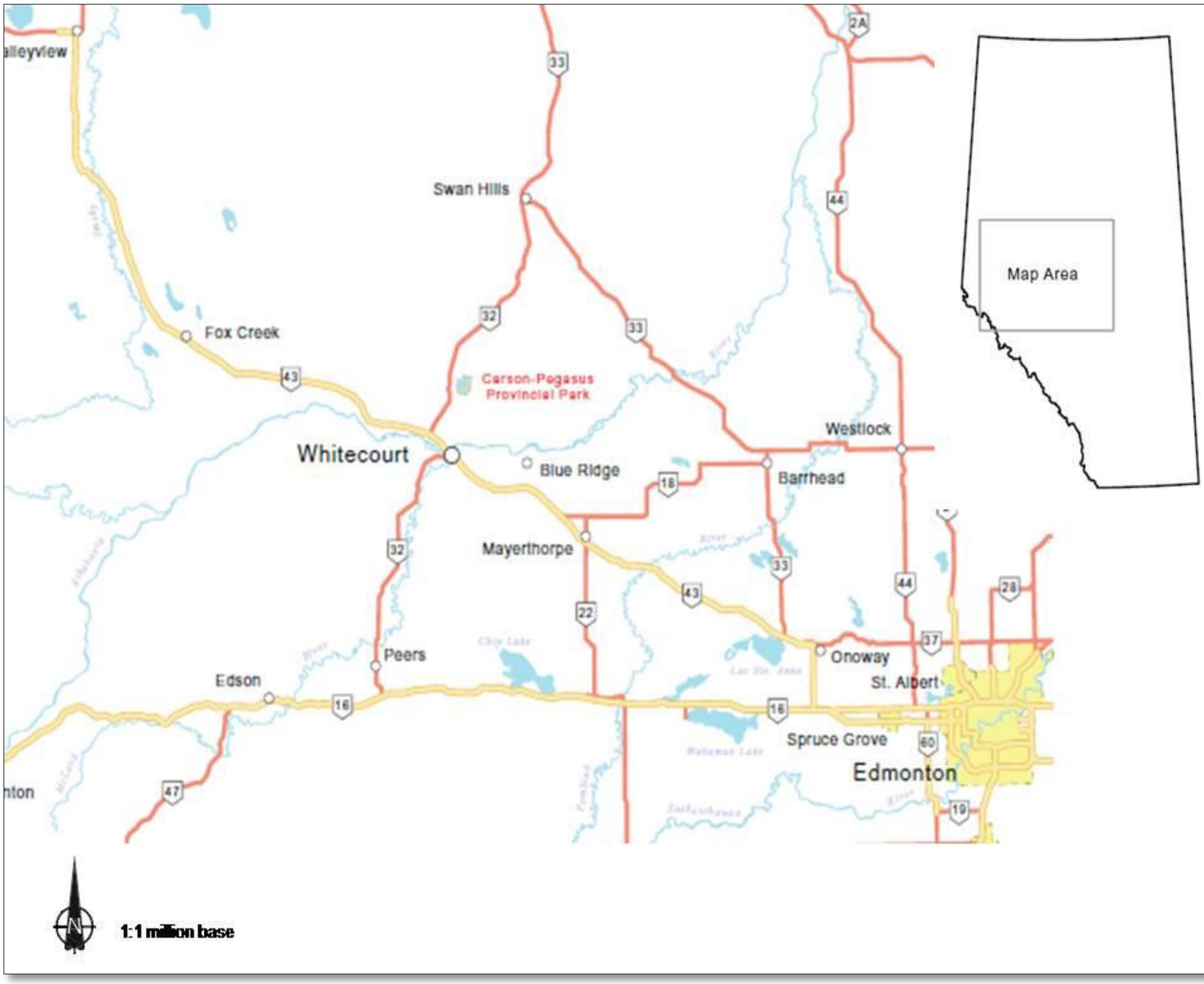

Figure 2.1. Map of study site at Whitecourt, Alberta, Canada, in proximity to Edmonton.

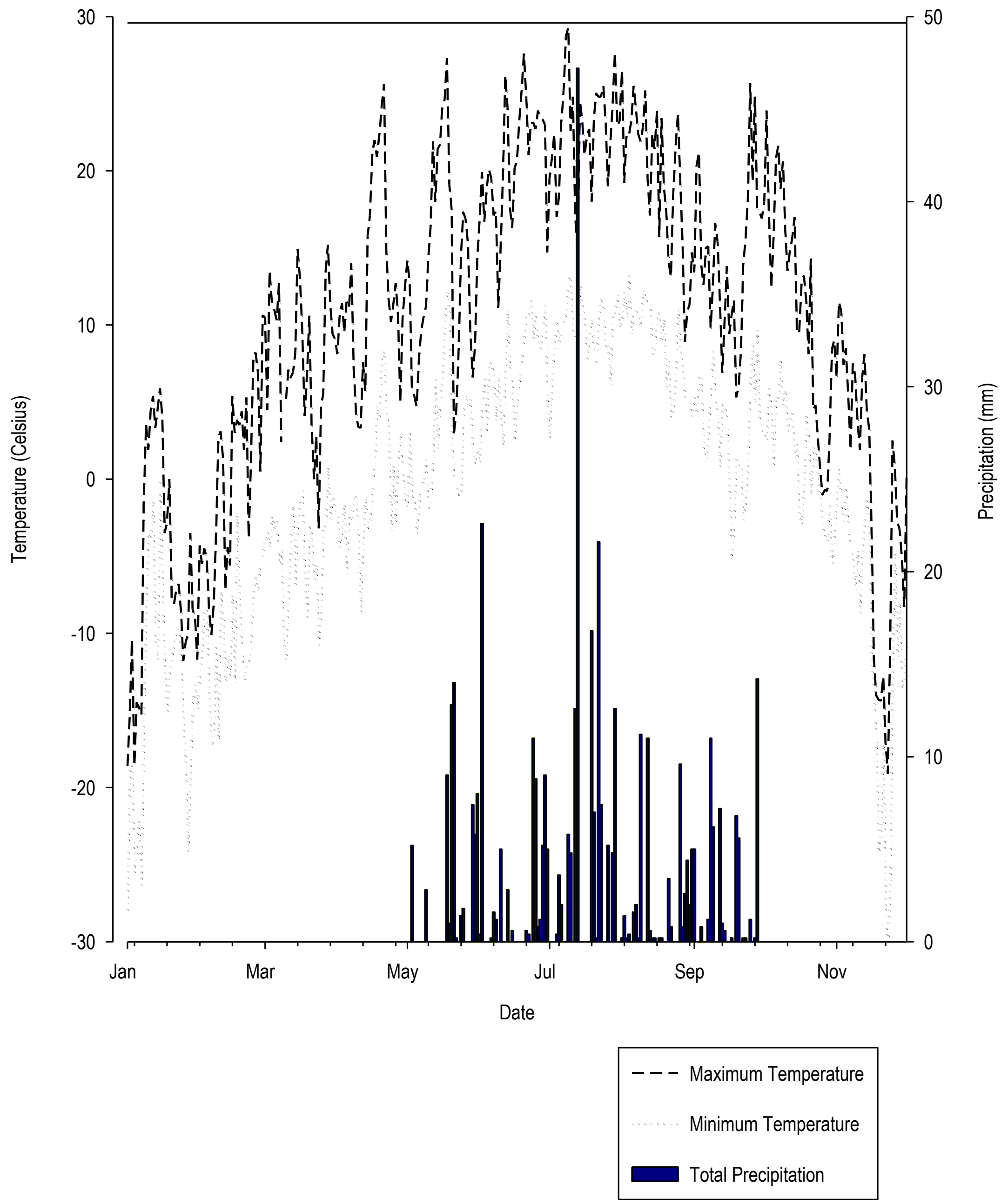

Figure 2.2. Long term climate normal temperature and precipitation weather data for Whitecourt, Alberta, based on data from 1970-2010.

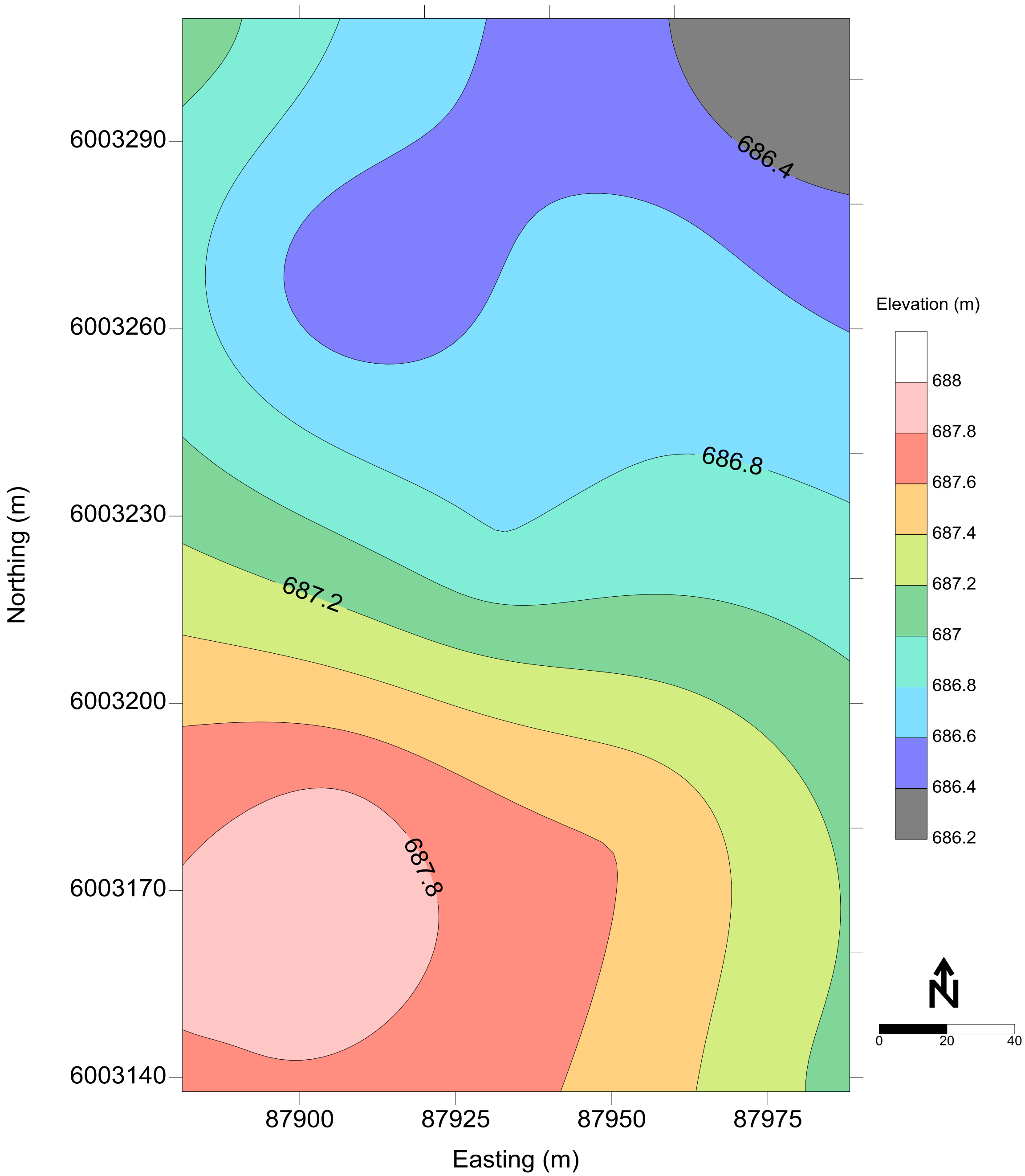

Figure 2.3. Topographic map of SRC plantation with wastewater irrigation project at Whitecourt, Alberta.

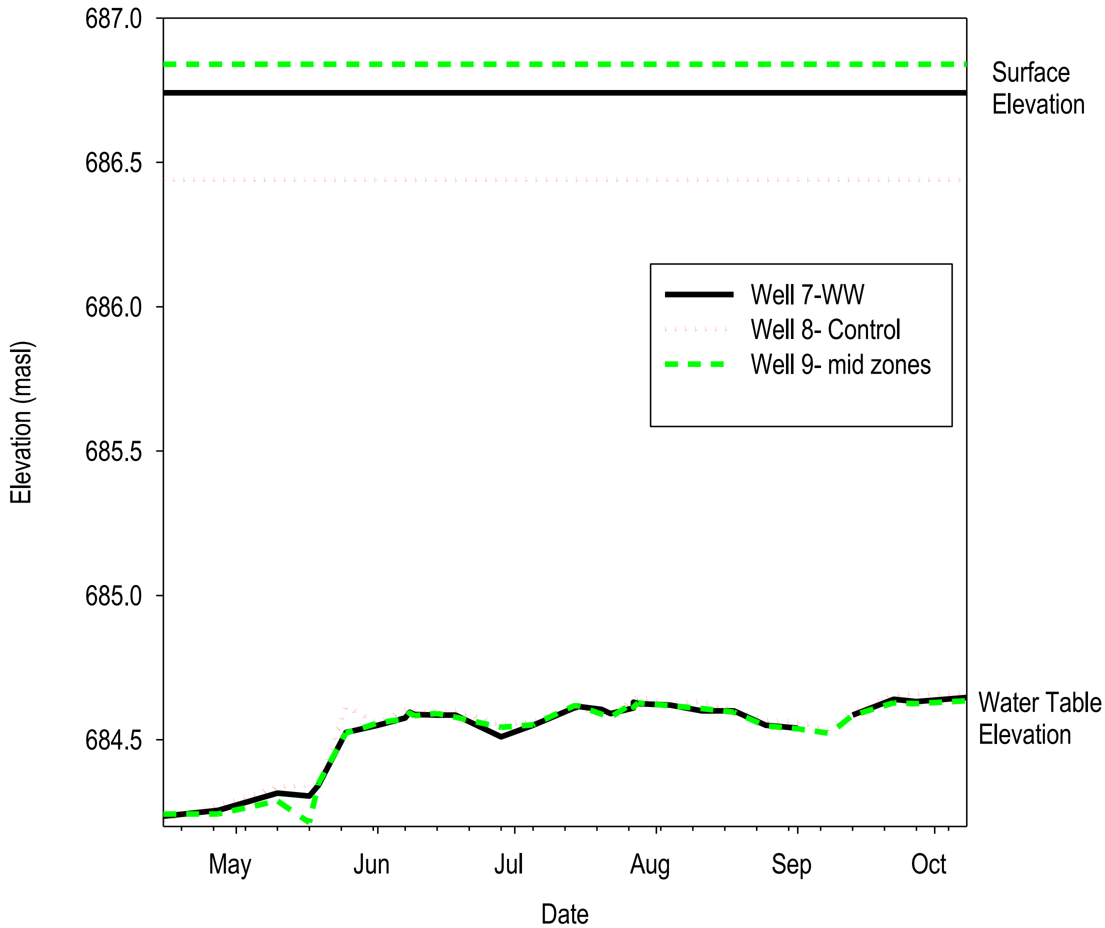

Figure 2.4. Groundwater table variations at the SRC plantation with wastewater irrigation site in Whitecourt, Alberta. Note only showing 3 of the 9 wells situate on the study site.

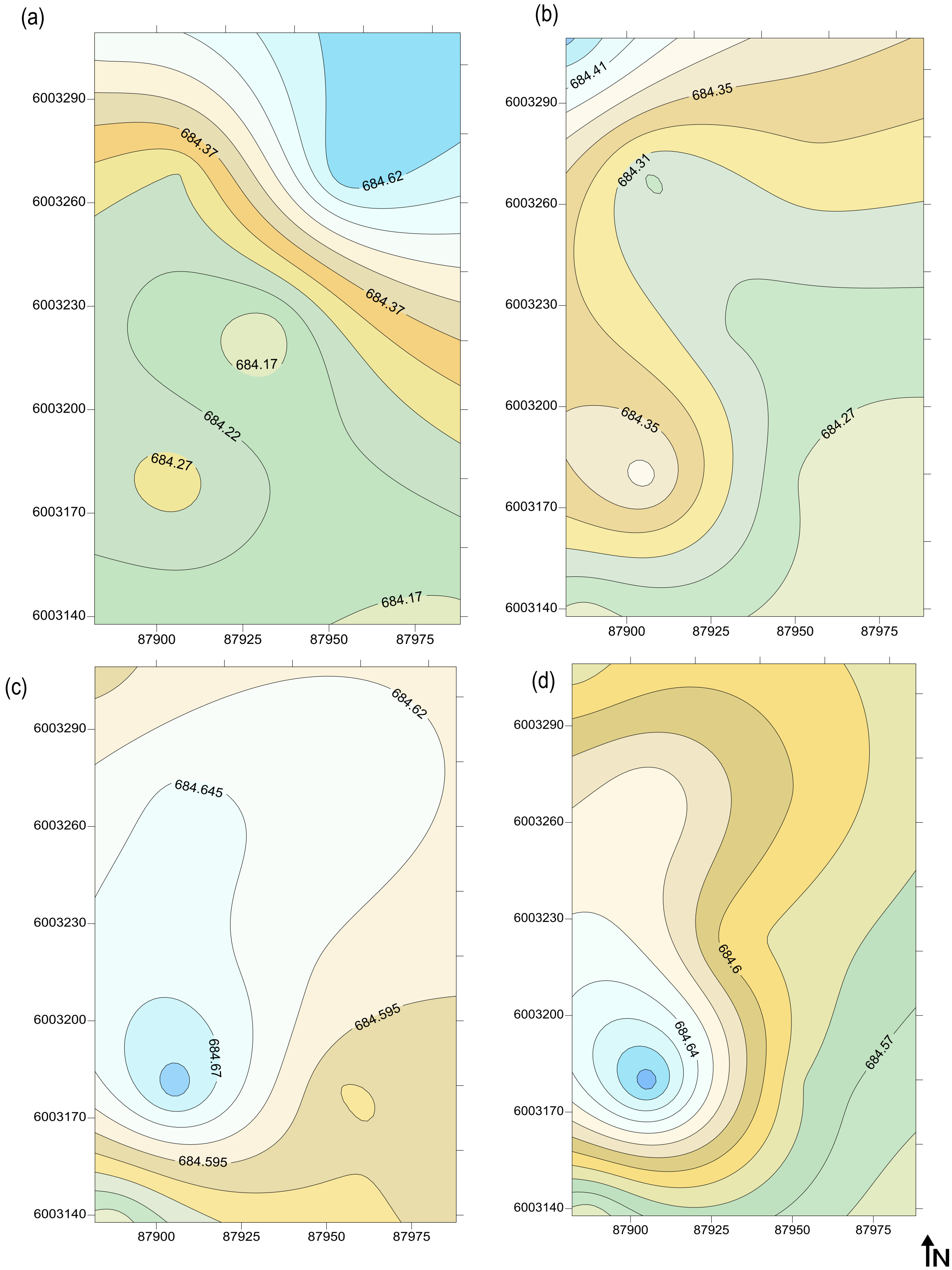

Figure 2.5. Ground water total head readings from (a) May, (b) July, (c) August and (d) September in 2010 at the study site showing groundwater direction.

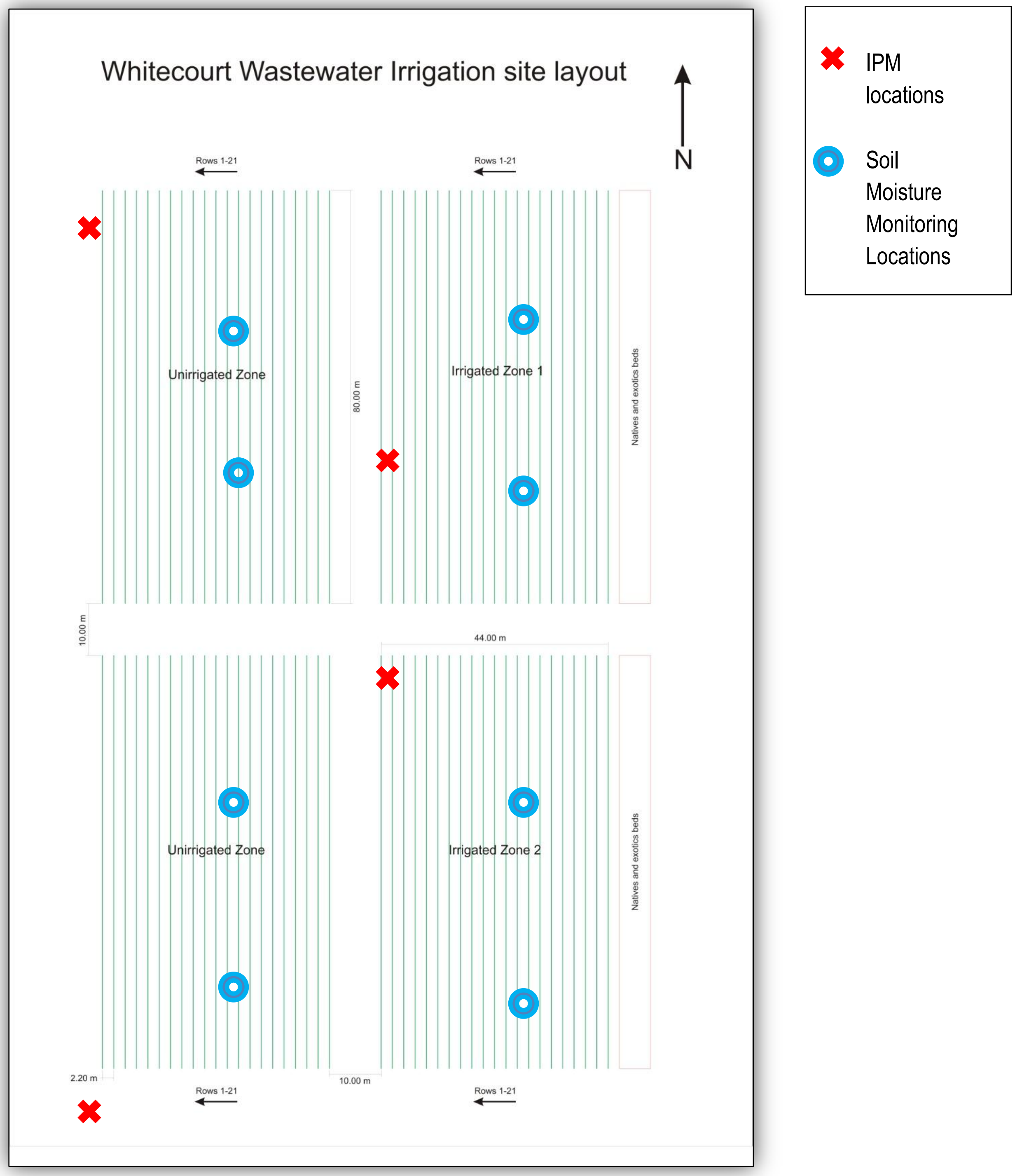

Figure 2.6. Experimental design of SRC plantation with wastewater irrigation project at Whitecourt, AB, displaying IPM (Instantaneous Profile Method) and soil monitoring locations.

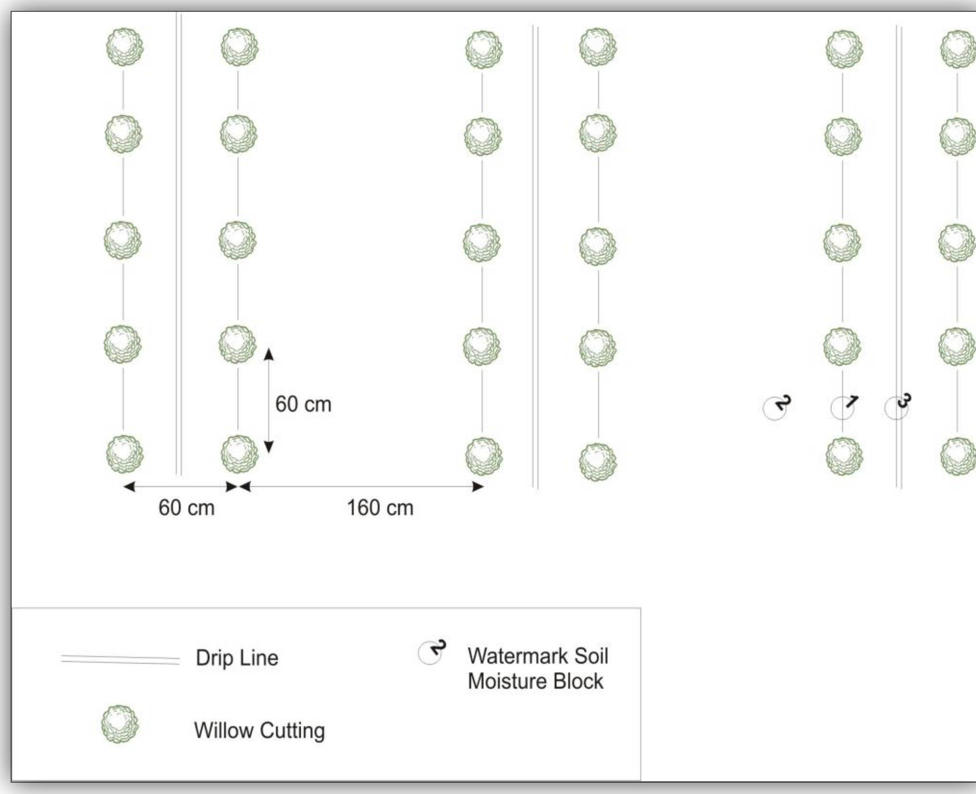

Figure 2.7. Planting arrangement for woody species at the SRC plantation with wastewater irrigation project in Whitecourt, Alberta.

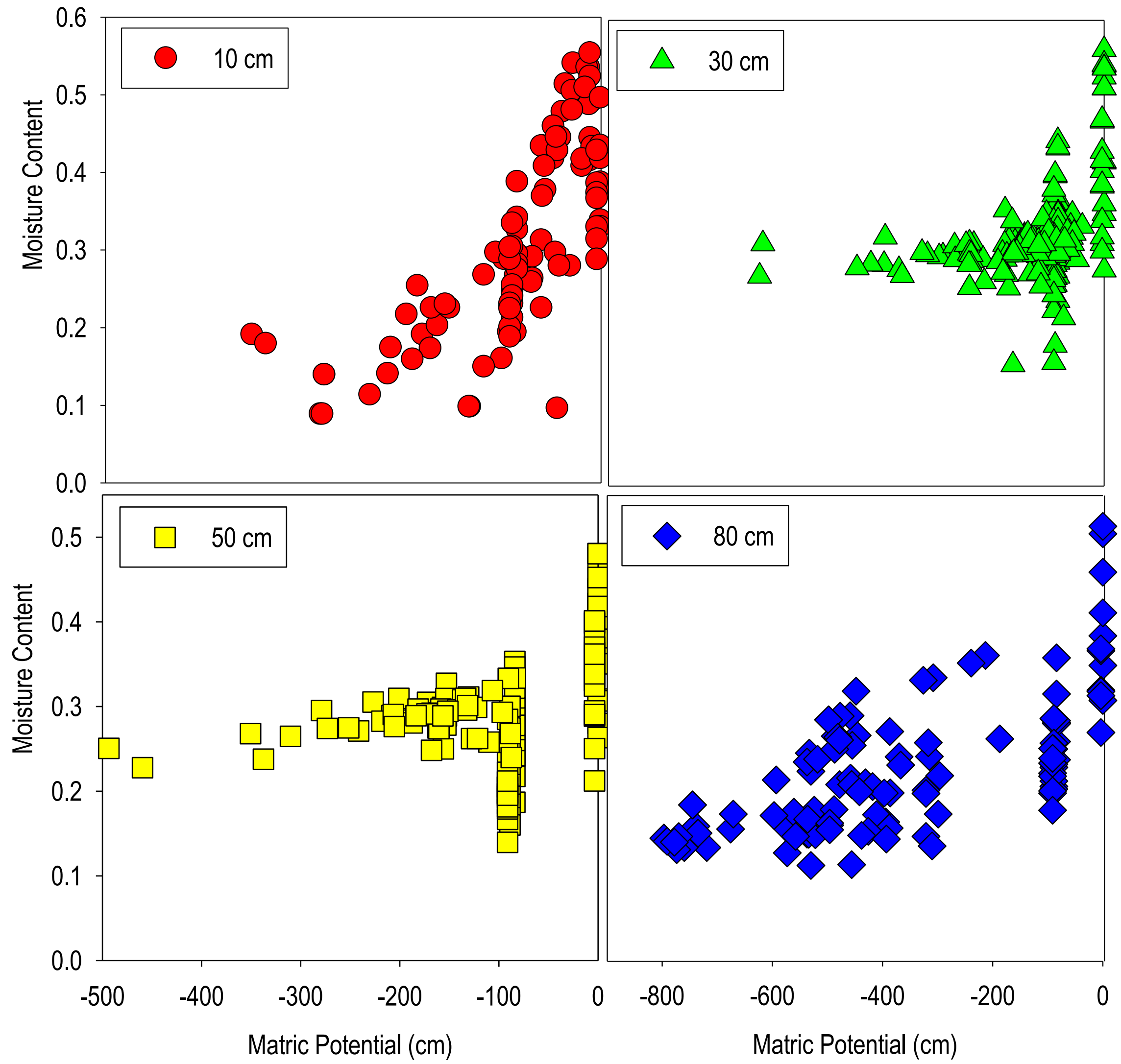

Figure 2.8. Moisture retention curve for varying depths at the SRC plantation with wastewater irrigation site in Whitecourt, AB.

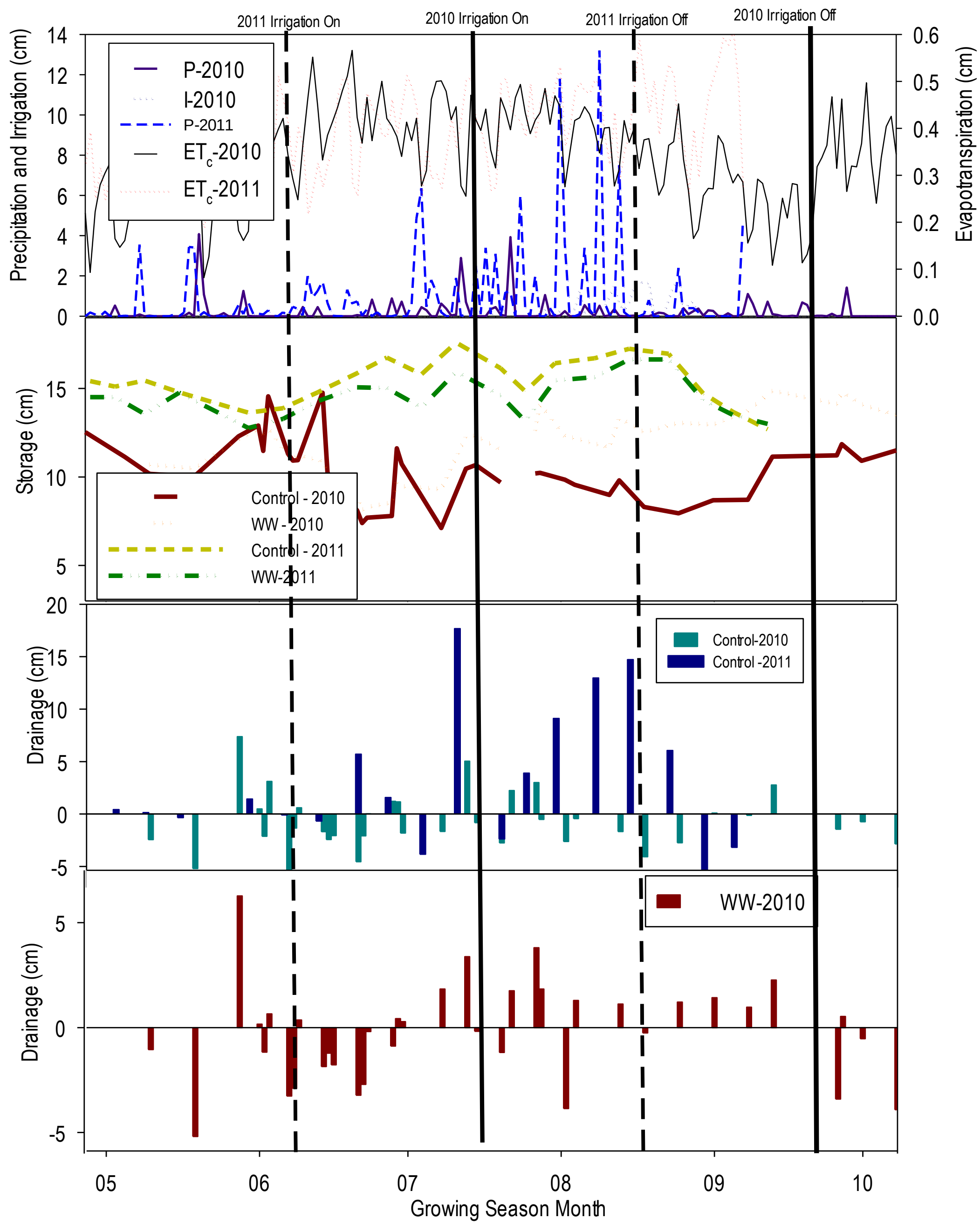

Figure 2.9. Soil water balance displaying precipitation, irrigation, evapotranspiration, storage within 50 cm and drainage under the wastewater irrigated (WW) and control zones for the SRC plantation with wastewater irrigation project in Whitecourt, AB, from April 27 to October 10.

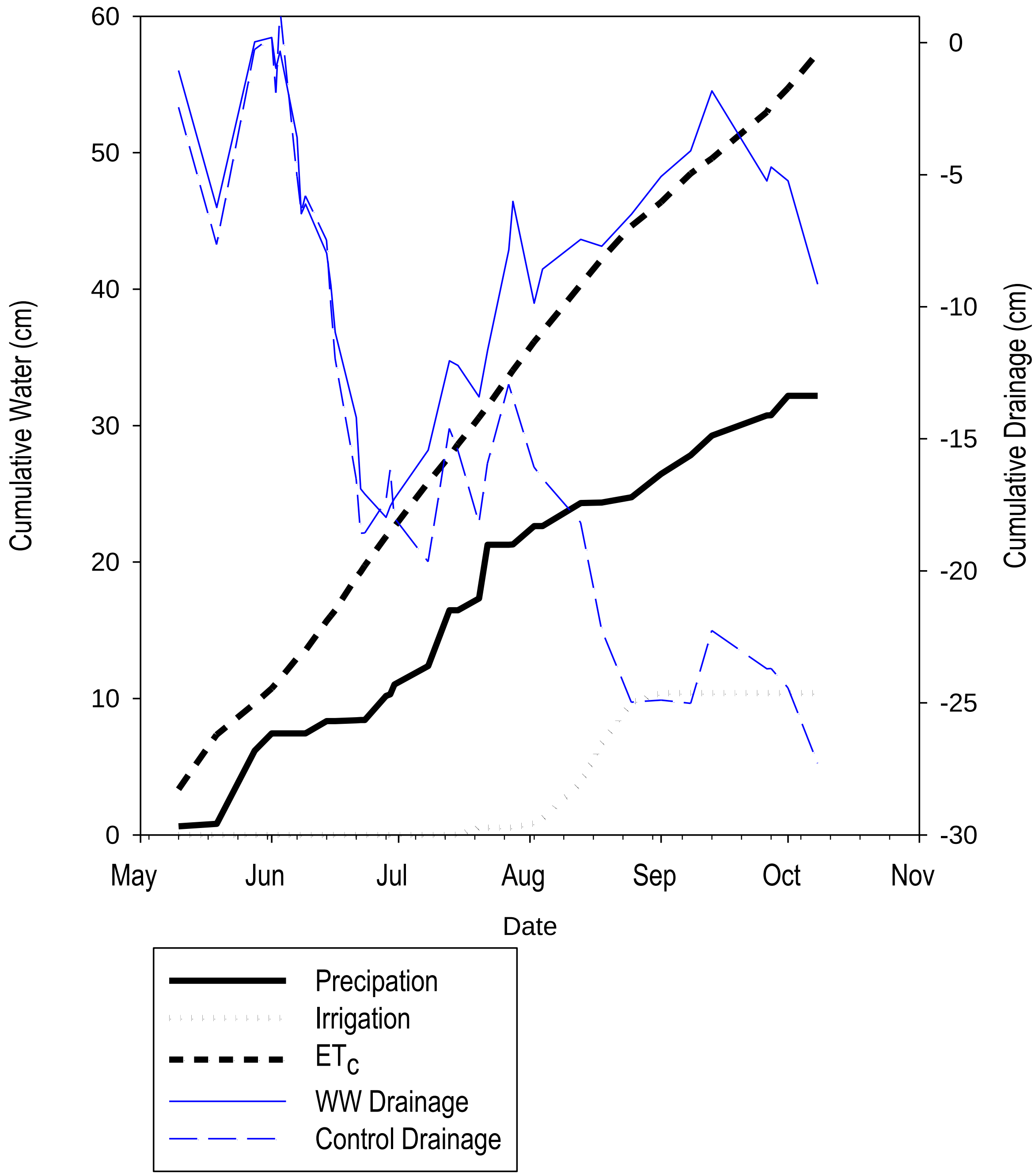

Figure 2.10. Cumulative precipitation, irrigation, evapotranspiration (ET_c) and wastewater (WW) irrigated soil and control soil drainage over the 2010 growing season at the SRC plantation with wastewater irrigation project at Whitecourt, AB.

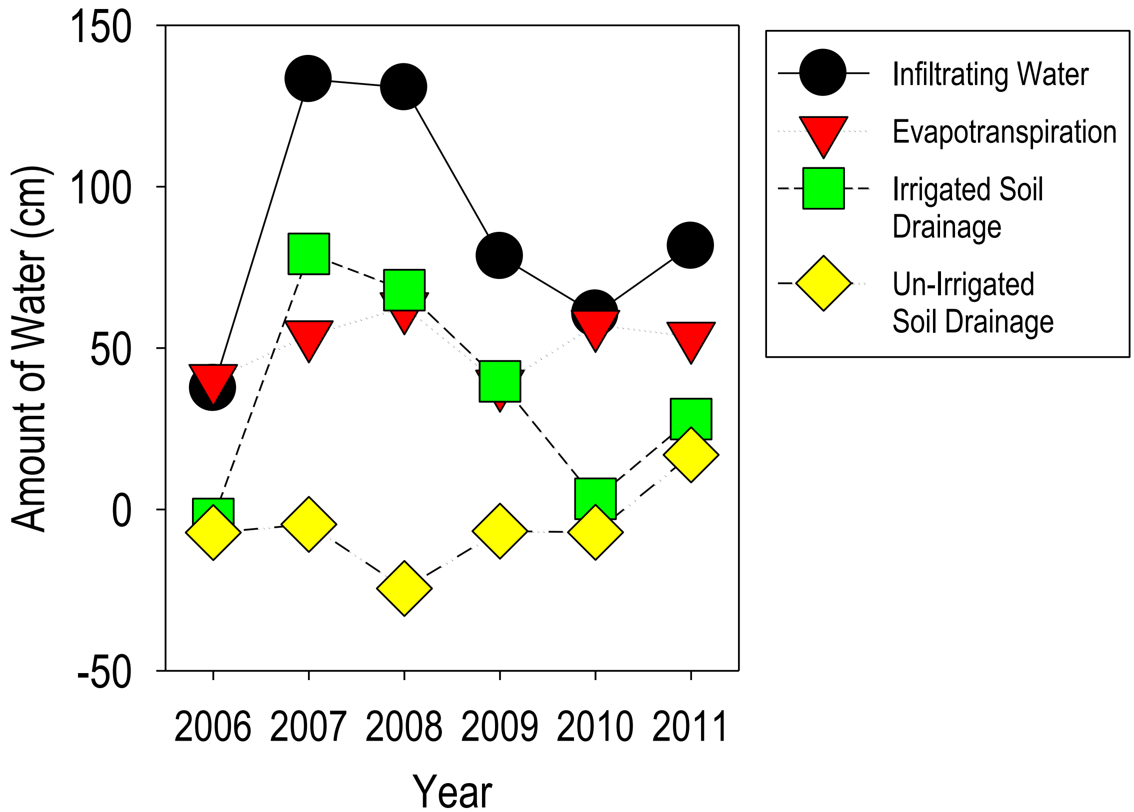

Figure 2.11. Historical soil water balance parameters for 2006 to 2011 at the SRC with wastewater irrigation project in Whitecourt, AB, using yearly data and assuming $\Delta S = 0$.

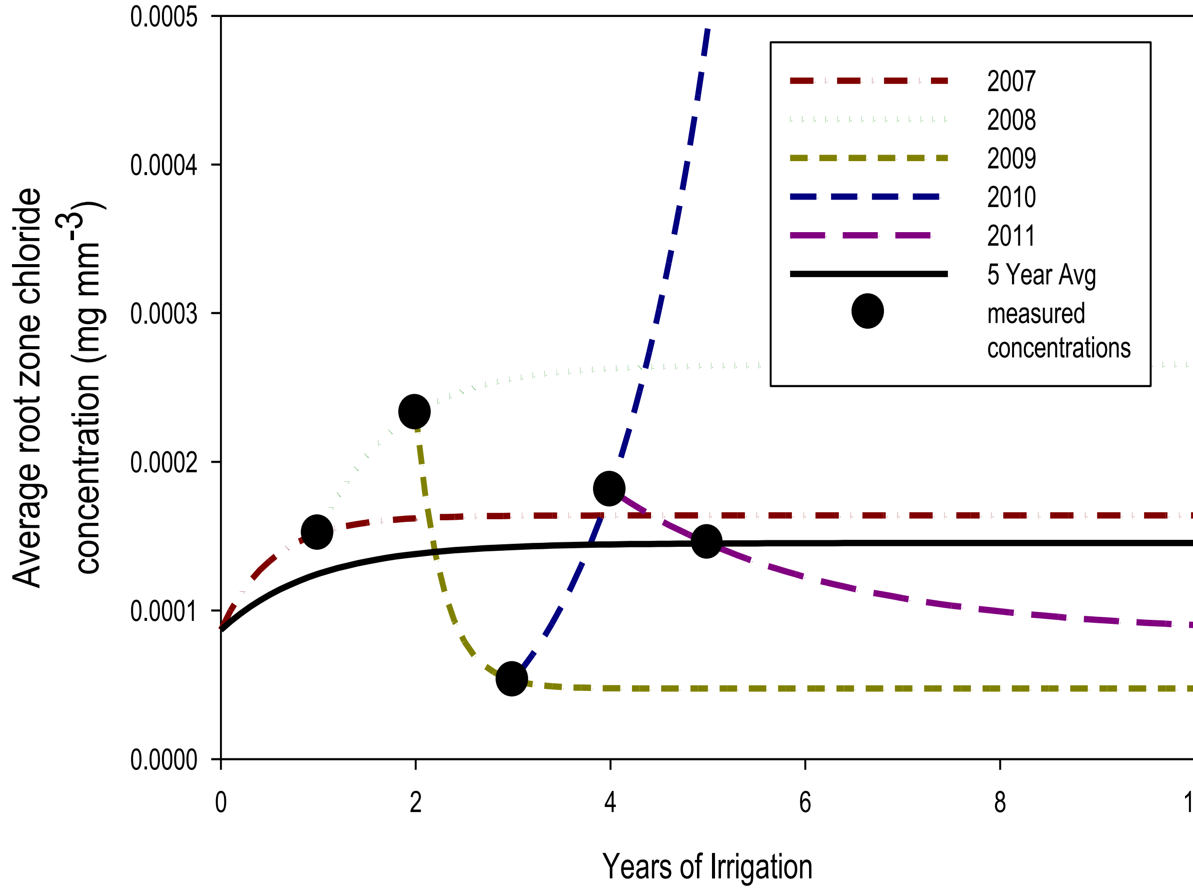

Figure 2.12. Average root zone chloride concentration for 2007 to 2011 and the 5 year average over irrigation years as predicted from the Rose et al (1979) model for steady sate conditions at the study site at Whitecourt, AB.

2.9 REFERENCES

Afas, N.A., A. Pellis, Ü. Niinemets and R. Ceulemans. 2005. Growth and production of a short rotation coppice culture of poplar. II. Clonal and year to year differences in leaf and petiole characteristics and stand leaf area index. Biomass and bioenergy. 28: 536-547.

Agroclimatic Atlas of Alberta. 2009. Climate of Alberta. Agriculture and Rural Development, Government of Alberta. http://www1.agric.gov.ab.ca/$department/deptdocs.n sf/all/sag6299. Accessed January 24, 2011.

Alberta Geological Survey (AGS). 2009. Geological Map of Alberta. http://www.ags.go v.ab.ca/defa ult.html. Accessed February 2, 2011.

Allison, G.B., G.W. Gee and S.W. Tyler. 1994. Vadose zone techniques for estimated groundwater recharge in air and semi arid regions. Soil Science Society of America Journal. 68:6-14.

Allison, G.B. and M.W. Hughes. 1978. The use of environmental chloride and tritium to estimate local recharge to an unconfined aquifer. Australian Journal of Soil Research. 16:181-195.

Angers, D.A. and F.J. Larney. 2007. Ch 55 Soil Physical Analyses in Soil Sampling and Methods of Analysis, Second Edition. Canadian Society of Soil Science. CRC Press. Ottawa, Canada.

Ayars, J.E., C.J. Phene, R.B. Hutmacher, K.R. Davis, R.A. Schoneman, S.S Vail and R.M. Mead. 1999. Subsurface drip irrigation of row crops: a review of 15 years of research at the water management research laboratory. Agricultural Water Management. 42: 1-27.

Brady, N.C. 1990. The nature and properties of soil, 10th edition. Maxwell MacMillan, New York.

Brady, N.C. and R. R. Weil. 2004. Elements of the nature and properties of soils, Second Edition. Pearson, Prentice Hall. Upper Saddle River, New Jersey.

Dyck, M.F. and R.G. Kachanoski. 2009. Measurement of Transient Soil Water Flux Across a Soil Horizon Interface. Soil Science Society of American Journal. 73,5:1604-1613

Environment Canada. 2010. Canadian Climate Data and Information Archive. http://www.clima te.weatheroffice.ec.gc.ca. Accessed November 1, 2010.

Fang, S., X. Xu, S. Lu and L. Tang. 1999. Growth dynamics and biomass production in short rotation poplar plantations:6 year results for three clones at four spacings. Biomass and bioenergy. 17:415-425.

Gardner, W. R. 1958. Some steady state solutions of the unsaturated moisture flow equation with applications to evaporation from a water table. Soil Science Society of America Journal. 85:228-232.

Hall, R.L. and S.J. Allen. 1997. Water use of poplar clones as short rotation coppice at two sites in the United Kingdom. Aspects of Applied Biology. 49: 163 -172.

Hayashi, M., G. van der Kamp and D.L. Rudolph. 1998. Water and solute transfer between a prairie wetland and adjacent uplands, 2. Chloride cycle. Journal of Hydrology. 207:56-67.

Heatlh Canada. 2010. Health Canada Guidelines for Canadian Drinking Water Quality. Prepared by the Federal-Provincial-Territorial Committee on Drinking Water of the Federal-Provincial-Territorial Committee on Health and the Environment. Government of Canada. Ottawa, Canada.

Hillel, D. 1998. Environmental Soil Physics. Academic Press. San Diego, California, USA.

Irmak, S. and D.Z. Haman. 2003. Evapotranspiration: Potential or Reference?. Agricultural and Biological Engineering Department, Florida Cooperative Extension Service, Institute of Food and Agricultural Sciences, University of Florida. ABE 3. 4 pp.

Lilbaek , G and J.W. 2007. Modelling enhanced infiltration of snowmelt ions into frozen soil. Hydrological Processes. 21: 2641–2649.

Longley, R.W. 1968. Climatic maps for Alberta. University of Alberta, Edmonton, Alberta. 8 pp.

Maulé,C., W.Helgason, S. McGinn and H. Cutforth. 2006. Estimation g of standardized reference evapotranspiration on the Canadian Prairies using simple models with limited weather data. Canadian Biosystem Engineering. 48:1.1-1.11.

McKeague,J.A (Editor). 1978. Manual on Soil Sampling and Methods of Analyses. Canadian Society of Soil Science. Ottawa, Canada

Miller, J.J. and D. Curtin. 2007. Ch 15 Electrical Conductivity and Soluble Ions in Soil Sampling and Methods of Analysis, Second Edition. Canadian Society of Soil Science. CRC Press. Ottawa, Canada.

Natural Regions Committee. 2006. Natural Regions and Sub-regions of Alberta. Compiled by Downing, D.J. and W.W. Pettapiece. Government of Alberta. Pub. No T/852. 264 pp.

OECD. 2004. Proceedings of Biomass and Agriculture-Sustainability Markets and Policies. ISBN 92-64-10555-7. OECD, Paris.

Persson, G. and P.E. Jansson. 1989. Simulated water balance of a willow stand on clay soil. In: Perttu K.I., Kowalik, P.J.; editors. Modeling of energy forestry: growth, water relations and economics. Wageningen: Pudoc. 147-162.

Persson, G. 1995. Willow stand evapotranspiration simulated for Swedish soil. Agricultural Water Management.28:271-293.

Pistocchi, C., W. Guidi, E.Piccioni and E. Bonari. 2009. Water requirements of poplar and willow vegetation filters grown in lysimeter under Mediterranean conditions: Results of the second rotation. Desalination. 246:137-146.

Pomeroy, J. W., Hedstrom, N. R. & Parviainen, J. (1999) The snow mass balance of Wolf Creek, Yukon: effects of snow sublimation and redistribution. In: Wolf Creek Research Basin - Hydrology, Ecology, Environment (Proc. Workshop held in Whitehorse, Yukon, Canada, 5-7 March, 1998). 15-29. NHRI Publication 37-121/1999E. Saskatoon, SK, Canada.

Pontailer, J.Y., R. Ceulemans and J. Guittet. 1999. Biomass yield of poplar after five 2 year coppice rotations. Insitute of Chartered Forester. 72: 157-163.

Raes, D. 2009. The ETo Calculator, reference manual version 3.1. Food and Agriculture Organization of the United Nations land and Water Division. 38 pp.

Reyolds, W.D. 2007a. Chapter 83: Unsaturated Hydraulic Properties Instantaneous Profile. In: Carter, M.R. and E.G. Gregorich Second Edition Soil Sampling and Methods of Analysis. Canadian Society of Soil Science. CRC Press, Boca Raton.

Reynolds, W.D. 2007b. Chapter 75: Saturated Hydraulic Properties: Laboratory Methods. In: Carter, M.R. and E.G. Gregorich Second Edition Soil Sampling and Methods of Analysis. Canadian Society of Soil Science. CRC Press, Boca Raton.

Reynolds, W.D. 2007c. Chapter 76: Saturated Hydraulic Properties: Well Permeameter. In: Carter, M.R. and E.G. Gregorich Second Edition Soil Sampling and Methods of Analysis. Canadian Society of Soil Science. CRC Press, Boca Raton.

Reynolds, W.D. and G. Clarke Topp. 2007. Chapter 72 Soil water desorption and imbibition's: Tension and pressure techniques. In: Carter, M.R. and E.G. Gregorich Second Edition Soil Sampling and Methods of Analysis. Canadian Society of Soil Science. CRC Press, Boca Raton.

Romano, N., J.W. Hopmans and J.H. Dane. 2003. Suction Table, Chapter 3 Water Retention and Storage. Encyclopedia of Water Science. Taylor and Francis. New York. 692-698.

Rose, C.W., P.W.A. ayananda, D.R. Nielsen and J.M. Biggar.1979. Long Term Solute Dynamics and Hydrology in Irrigated Slowly Permeable Soils. Irrigation Science. 1: 77-87.

Rytter, R.M. and Hansson, A.C. 1996. Seasonal amount, growth and depth distribution of fine roots in an irrigated and fertilized *Salix viminalis* L. plantation. Biomass and Bioenergy 11:129-137.

Scanlon, B. R., Langford, R. P., and R. S. Goldsmith. 1999. Relationship Between Geomorphic Settings and Unsaturated Flow. Water Resources Research 35: 983– 999.

Silburn, D.M., B.A. Cowie and C.M. Thornton. 2009. The Brigalow Catchment Study revisited: Effects of land development on deep drainage determined from non-steady chloride profiles. Journal of Hydrology. 373:487-498.

Sims, R.E.H., A. Hastings, B. Schlamadinger, G. Taylors and P. Smith. 2006. Energy Crops: current status and future prospects. Global Change Biology. 12: 2054-2076.

Tabatabai, W.M.A. and N. T. Basta.1991. Ion Chromatography. Soil Analysis. Modern Instrumental Techniques. Edited by K.A. Smith.Second Edition.

Tarsia, N. 1980. Measurement of rainfall inputs to poplar plantations. Pubblicazioni del Centro di Sperimentazione Agricolae Forestale. 13,3:239-249.

Tokarksy, O.1977. Hydrogeology of the Whitecourt area, Alberta. Alberta Research Council. Report 76-3. 13 pp.

Toogood, J.A. 1958. A Simplified Textural Classification diagram. Canadian Journal of Soil Science. 35:54-55.

Tyler, S. W., Chapman, J. B., Conrad, S. H., Hammermeister, D. P., Blout, D. O.,Miller, J. J., Sully, M. J., and J. M. Ginanni. 1996. Soil-water Flux in the Southern Great Basin, United States: Temporal and Spatial Variations over the last 120,000 years. Water Resources Research 32: 1481 – 1499.

United States Environmental Protection Agency (USEPA). 2011. Drinking Water Contaminants. http://water.epa.gov/drink/contaminants/index.cfm. Accessed July 20, 2011.

Volk, T.A., L.P. Abrahamson and E.H. White. 2001. Root dynamics in willow biomass crops. Biomass Power for Rural Development Technical Report. State University at New York. 30 pp.

Wynnyk, A., J.D. Lindsay and Wm. Odyksky. 1969. Soil Survey of the Whitecourt and Barrhead Area. Soils Division, Research Council of Alberta. Alberta Soil Survey Report No. 27. 69 pp.

Zhang, Z.F., R. G. Kachanoski, G.W. Parkin and B. Si. 2000. Measuring hydraulic properties using a line source :I Analytical Expressions. Soil Science Society of America Journal. 64 : 1554-1562.

Zsuffa, L. 1995. Characterization of poplar and willow clones and cultivars. Biomass and Bioenergy. 9:53-68.

3. ROOT ZONE SOIL SOLUTION QUALITY

3.1 INTRODUCTION

Climate change is a global concern and many countries are seeking renewable forms of energy to mitigate carbon emissions to the atmosphere. Short rotation coppice (SRC) systems are a form of renewable energy utilizing plant biomass to produce heat or electricity. SRC systems have been identified as one of the most energy efficient carbon conversion technologies to reduce greenhouse gas emissions (Styles and Jones 2007). SRC plantations consist of fast growing woody species that can regenerate after harvest, or coppice, in short time intervals for energy purposes (Labrecque and Teodorescu 2003, Bonaria et al 2004). Willow and poplar are the most common species used as they have the ability to take up large quantities of water as well as some contaminants like heavy metals. Using municipal, treated wastewater (referred to as wastewater hereafter) to irrigate SRC plantations offers various benefits including a low cost source of many plant essential nutrients and an abundant supply of irrigation water. Municipalities that adopt these systems gain various benefits, mainly, a low greenhouse gas emitting source of heat and electricity.

Recent studies have found that SRC systems are beneficial to mitigating carbon emissions to the atmosphere. A study by Baum et al (2009) found that SRC systems can be a carbon sink when planted in certain areas. The system serves as a carbon sink mainly due to annual leaf litter stored, minimum tillage and other management techniques (Dimitriou et al 2009). Dimitriou et al (2009) found increased carbon sequestration when SRC is planted on formerly cultivated soils.

Drainage is an important soil hydrologic process for plant and soil health. Drainage occurs when water inputs, like precipitation and irrigation, are larger than evapotranspiration (Hillel 1998). Drainage is particularly important for aerating roots and preventing salinization of the root zone when using irrigation water containing salts (Hillel 1998). Secondary soil salinization is a common irrigation problem that occurs when using irrigation water with salts like calcium, sulfate and sodium (Hillel 2005). If the amount of irrigation and precipitation applied equals or is less than the evapotranspirational demand of the plants, then salts from irrigation water will accumulate as plants successively take up water and not the salts (Hillel 2005). To mitigate this process, excess irrigation water is applied according to a desired leaching fraction (LF). The leaching fraction is defined as the ratio of drainage to precipitation and irrigation and was created to manage soil salinity in irrigated soils (U.S. Salinity Laboratory Staff 1954). LFs are an important aspect of irrigation management but when using wastewater as an irrigation source there is potential groundwater pollution due to solutes other than salts found in wastewater.

Wastewater typically contains plant essential nutrients such as nitrogen and phosphorus. The nitrate ion is mobile in aqueous solution (not readily sorbed to negatively-charged clay particles) and susceptible to leaching but also may be transformed by microbial processes. Nitrate leaching to groundwater is common in agricultural settings and contaminated groundwater has detrimental human and environmental health effects. In humans, severe nitrate contaminated drinking water can lead to *methemoglobinemia* in newborn infants (Addiscott et al 1991; Lee et al 1991; Wolfe and Patz 2002). In surface water, nitrate may impact water quality through eutrophication, characterized as accelerated plant growth causing decreased dissolved oxygen levels.

Phosphorus is another plant essential nutrient found in wastewater. The plant-available form of phosphorus is represented by phosphate and orthophosphate. In soil and water, phosphate typically occurs as orthophosphate. Orthophosphate forms depend strongly on pH, forming $H_2PO_4^-$ in acidic conditions and HPO_4^{2-} in alkaline conditions. Orthophosphate concentrations in soil solution depend on the sorbed concentrations, which is affected by soil texture. Although orthophosphate leaching is uncommon, there have been findings in which this has occurred. Eghabll et al (1996) found orthophosphate leaching from long term manure applications and Olsen et al (2010) suggest orthophosphate leaching in calcareous soils occurs especially when the soil is coarse-textured. Like nitrate, orthophosphate in surface water bodies contributes to eutrophication and reduces surface water quality.

3.2 OBJECTIVES

The Canadian Forest Service (CFS) of Natural Resources Canada established a SRC plantation with wastewater irrigation project at Whitecourt, Alberta, in 2005. The goal of the project was to demonstrate the ability of SRC willow biomass systems to treat municipal wastewater and their feasibility as an alternative energy source in Alberta. The purpose of this research is to assess the water usages, and nutrient loading rates to groundwater of this system to develop water management strategies that minimize environmental risk and maximize water use efficiency. The objectives of this research are as follows:

I. Using an in-situ field-based sampling method and a transport model to assess nitrogen and phosphorus leaching to groundwater under SRC crops with wastewater irrigation

II. Using a solute transport model, assess irrigation management and water useage to minimize soil salinization.

III. Estimate treatment of municipal treated wastewater by SRC plantations

IV. Make recommendations to mitigate environmental impacts and promote sustainability of SRC plantations with wastewater irrigation systems

3.3 MATERIALS AND METHODS

3.3.1 Site Description

The study was conducted over the 2010 and 2011 growing seasons (May to October). The study site was located in the north east section of the town of Whitecourt, Alberta, adjacent to the town wastewater treatment plant. Whitecourt is approximately 180 km northwest of Edmonton, Alberta, on Highway 43 (Figure 3.1). The study site has coordinates of approximately latitude 54 09 07.90930 dms and longitude 115 39 03.86188 dms. The study area is located in the Western Alberta Uplands, Clear Hills Uplands eco region of Alberta and the Lower Foothills natural sub-region (Agroclimatic Atlas of Alberta 2009; Natural Regions Committee 2006).

The climate of the study area is characterized as humid continental with warm summers and long, cold winters (Longley 1968). The annual mean daily temperature at Whitecourt is 2.6 ^{o}C and can range from -41^{o}C in January to 33.5 ^{o}C in August. Average annual precipitation, rainfall and snowfall depth, respectively, are as follows: 577 mm, 440.3 mm and 178.1 cm (Environment Canada 2010). Snowfall occurs primarily between November and March, with the average annual wind speed is 7.4 km h^{-1} and most frequently in the west direction.

The bedrock geology of the greater area consists of two continental shale and sand geologic formations, the Paskapoo Formation of Paleocene-Late Cretaceous age and the Wapiti Formation of Late Cretaceous age (Tokarksy 1977). The study site is situated on the Wapiti Formation from the Upper Cretaceous. The Upper Wapiti is characterized by grey, feldspathic, clayey sandstone;

grey bentonititc mudstone and carbonaceous shale; concretionary ironstone beds; scattered coal and bentonite beds of variable thickness; and minor limestone beds (AGS 2009). The upper most bedrock unit is consolidated gray clay at a depth of about 11 m from surface. The study site is located on the Athabasca river flood plain, with an elevation of about 685 masl. Topography is gently rolling and the dominant aspect is north facing with a slope class between 2 and 5 %.

The soil in the study site area is classified as a Gleyed Dark Gray Chernozem profile of the High Prairie series (Wynnyk et al 1969). The soils are imperfectly drained and developed on recent alluvial material deposited by the nearby Athabasca River (Wynnyk et al 1969). The observed soil depth at the research site varies with slope position from about 50 cm on slope shoulders to 1 m on slope toes. Below the solumn, there is a thin layer of sand and a thick gravel layer (estimated to be about 7 m thick). The observed groundwater table depth ranges from 1.5 to 3.8 m below the ground surface, depending on study site position and time of year. Piezometers on the site indicate that groundwater direction is mostly south in the spring and south east in the summer and fall.

Prior to planting of SRC woody species, the study site was an agricultural field planted with canola for about 5 years. In 2006, five willow and two poplar clones were planted as cuttings. The five willow clones were Charlie (*Salix miybeaba*), SX64 (*S. udensis*), SX61 (*S. udensis)*, SV01 (*S. dasyclados*) and Psuedo (*S. alba*) (Zsuffa 1995). The two poplar species were Brooks 1 and Green Giant, both *Populus deltoides* x *P. petrowkyana* clones.

The natural vegetation in the area is typical of lower elevation species in the lower foothills natural sub region of Alberta. There are mainly deciduous stands composed of *Populus tremuloides* Michx., *P. balsamifera* L. and *Betula papyrifera* Marsh (Natural Regions Committee 2006). Understory vegetation typically consists of *Vaccinium vitis-ideae* L., *Alnus viridian* D.C, *Epilobium angustifolium* L. and *Calamagrostis canadensis* Michx (Natural Regions Committee 2006).

3.3.2.1 Experimental Design

The site consists of four zones, two receiving wastewater through subsurface irrigation and two receiving no irrigation (control) (Figure 3.2). Within each zone, woody species are planted in paired rows and the irrigated zones have the irrigation line buried in the middle of the row. The dimensions are roughly 60 cm between plants within paired rows and 2.2 m between rows. There are three adjacent rows of the same plant clone with 21 rows in each zone. In total there are about 6000 plants per zone.

The site is irrigated with secondary treated municipal wastewater using a subsurface irrigation system. The irrigation line is buried about 25 cm below ground and runs north-south on the site. The length of the irrigation pipe is 80 m and the inner diameter is 1.4 cm. Outlets are situated every 60.96 cm and have a square shape with dimensions of 0.13 by 0.13 cm.

The woody species were planted in the spring of 2006 and the first irrigation year began that summer. The first harvest was in the winter of 2009 after three growing seasons. Irrigation is activated based on soil moisture potential sensors located in each irrigated zone. There are two sensors, each in the same location in a paired row of cuttings. The sensors are located 20 cm below the soil surface and determine the average soil moisture potential in the rooting zone. The irrigation system is designed to maintain the root zone at roughly field capacity, -0.5 bars. Irrigation rates in the two irrigated plots within the irrigated zone are independent of each other.

3.3.2.2 Soil Solution Samplers

Soil solution samplers, Model 1900, from Soil Moisture Corp. (Santa Barbara, CA) were installed below the root zone of the study site to sample the soil solution. The soil solution samplers consisted of a 4.8 cm outside diameter PVC tubes, a porous ceramic cup with a 2 bar (200kPa) air entry value and a Santoprene stopper. Neoprene tubing with a clamp was attached to the stopper to apply suction. Nests of 2 to 4 samplers with varying depths were positioned around the study site (Figure 3.2). Soil solution sampler depths were typically 50 cm, 90 cm and 150 cm. Samplers were installed by creating a borehole of the same diameter as the samplers and backfilling with silica flour slurry to create contact between soil and ceramic cup and prevent preferential flow. Sample collection in 2010 started in June and ended in October and in 2011 the sampling period was from April to September.

A suction was applied to the samplers using a vacuum pump designed by Soil Moisture Corp. (Santa Barbara, CA). Vacuums of between 300 kPa and 500 kPa were applied for varying time periods. In 2010, the vacuum was typically applied for up to 4 days and in 2011 up to 6 days prior to sampling. Samples were collected using a vacuum pump and an extraction kit consisting of a 1

L flask, tubing and stopper. Samples were transferred into bottles and put on ice while outdoors and frozen before transferred to an analytical laboratory.

The soil solution was analysed for the following: nitrate-N (NO_3 -N), nitrite-N (NO_2-N), ammonium-N (NH_4-N), ammonia-N (NH_3 -N), total Kjeldahl nitrogen (TKN), total nitrogen (TN), dissolved organic carbon (DOC) and orthophospahtes (PO_4^{3-} -P). Total nitrogen was determined by the sum of TKN and nitrate-N. Organic nitrogen was estimated by subtracting N-NH_4 and N-NO_3 from total nitrogen. Nitrate-N and nitrite-N were determined by Automated Colorimetry according to Smart Chem (EPA 1993). Ammonium-N was determined using SmartChem (APHA 2005). DOC was determined as Non Purgeable Organic Carbon (NPOC) with a Shimadzu carbon and nitrogen analyzer (McGill et al 1986). In 2010, total nitrogen analysis was performed using the Dumas method (AOAC 2000). In 2011, a different lab performed soil water analysis and TKN was determined rather than total nitrogen. TKN was determined using the International Organization of Standards Water Quality – determination of nitrogen (ISO11905-2). Orthophosphate-P was analyzed using Automated Ascorbic Acid Reduction Method, 4500-P F and only in 2011 (APHA 2005).

There are limitations with using soil solution samplers that should be noted. Soil solutions cannot be gathered when the soil suction is lower than the negative pressure in the soil solution sampler cup, which often occurs during drought and high root uptake activity (Normand et al 1997). During times of rainfall or high irrigation, the soil solution is more representative of water in the soil macroprores than in the soil aggregates (Normand et al 1997; Moutonnet et al 1993). Ions, like orthophosphate, may be excluded from or adsorbed to the ceramics (Cheverry 1983). This is not the case with nitrate due to its mobility and lack of adsorption capabilities (Litaor 1988).

3.3.2.3 Wastewater Characterization

In 2010 and 2011, wastewater samples from the treatment plant were collected about every two to four weeks. Samples were analyzed for the same ions as the soil solution sampler and chloride using methods previously mentioned. Chloride was determined using an Ion Chromatograph (Dionex 600) (Tabatabai and Basta 1991). Historical information on wastewater solutes was also available from previous sampling years.

3.3.3 Schoups and Hopmans (2002) Model

Wastewater used in SRC projects contains nutrients that are essential for plant growth but may contaminate groundwater under high drainage conditions. The movement of potential contaminants under these systems is important for the sustainability of these projects. Solute transport models can be used to estimate solute loading rates to the groundwater under various hypothetical management scenarios. They can be used to simulate long-term relationships between land use decisions and contamination levels in the environment that may take a long time to measure in the field (Schoups and Hopmans 2002).

Schoups and Hopmans (2002) created a simple, analytical model for solute transport considering root water and solute uptake. The model allows for calculating one-dimensional solute transport in the soil in the presence of a vertically distributed root system that extracts water and solutes. This model was chosen because it took into account root water uptake and the spatial variability in root distribution. The model is based on the analytical solution of the convective transport equation. The model requires surface boundary conditions (precipitation, irrigation), wastewater solute concentrations and average soil solute concentrations as input variables. The model is represented conceptually in Figure 3.3 and its mathematical derivation is presented below.

The continuity equation for one dimensional vertical water flow through the root zone is as follows:

$$\frac{\delta \theta}{\delta t} = \frac{\delta q}{\delta z} + r_w \qquad [3.1]$$

Where

$\theta(z,t)$ is volumetric water content (L^3 L^{-3})

$q(z,t)$ is soil water flux (L T^{-1}, positive downwards)

$r_w(z,t)$ is a sink term for root water uptake and evaporation (L^3 $L^{-3}T^{-1}$)

t is time (T)

z is vertical dimension (L, positive downwards)

The continuity equation for one dimensional transport of a solute undergoing root uptake and equilibrium sorption and neglecting dispersion and diffusion is:

$$\frac{\delta}{\delta t}(\theta c + p_b c_a) + \frac{\delta qc}{\delta z} + r_s = 0 \qquad [3.2]$$

Where

c is dissolved solute concentration (M L^{-3}); mass per unit volume of soil solution

c_a is adsorbed solute concentration (M M^{-1}), mass of sorbent per mass of dry soil

p_b is dry bulk density (M L^{-3})

r_s is root solute uptake [M L^{-3}T^{-1}]

Combining Equation [3.1] and [3.2] leads to the following:

$$\frac{\partial c}{\partial t} + v_R \frac{\partial c}{\partial z} = \frac{c\, r_w - r_s}{R\theta} \qquad [3.3]$$

Where

$V_R(z,t) = \frac{v}{R} = \frac{q}{R\theta}$ is the retarded solute velocity (L T^{-1})

$v(z,t)$ is the pore water velocity (L T^{-1})

$R = 1 + \frac{p_b k_d}{\theta}$ is the retardation coefficient

K_d is the distribution coefficient for linear sorption

For constant solute concentrations in the infiltrating water and constant surface flux surface boundary conditions, the steady state solution to Eq. [3.3] for a semi-infinite domain is:

$$c(z,t) = \left[\frac{\hat{q}(z_s)}{\hat{q}(z)}\right]^{1-a} c_i(z_s) H[\tau(z) - T_0(t)] + \left[\frac{\hat{q}(z_0)}{\hat{q}(z)}\right]^{1-a} c_0(t_s) H[T_0(t) - \tau(z)] \qquad [3.4]$$

Where, H is Heaviside function ($H[x]=1$ for x>0 and $H[x]=0$ otherwise), $c_i(z)$ is the initial solute concentration as a function of depth, $c_0(t)$ is the solute concentration in the irrigation (infiltrating) water as a function of time, and a is the root solute uptake parameter ($r_s = acr_w$).

For the case of an exponential root distribution function and uniform average soil water content:

$\hat{q}(z)$ is the normalized soil water flux:

$$\hat{q}(z) = 1 - p \cdot exp\left(-\frac{z}{\delta}\right) \qquad [3.5]$$

where p is the precipitation efficiency ($1\text{-}p$ is the leaching fraction), and δ is the root distribution function parameter (set at 150 mm which resulted in 95% of the root mass in the first 50 cm of the root zone in line with observations by (Volk 2001)

$\tau(z)$ is the travel time of a solute parcel moving from z_0 to z :

$$\tau(z) = \frac{R\bar{\theta}\delta}{\bar{q}_0} ln\left[\frac{p + (1-p)\cdot exp(z/\delta)}{p + (1-p)\cdot exp(z_0/\delta)}\right] \qquad p \neq 1$$

$$\tau(z) = \frac{R\bar{\theta}\delta}{\bar{q}_0}[exp(z/\delta) - exp(z_0/\delta)] \qquad p = 1 \qquad [3.6]$$

Where $\overline{q_0}$ is the long-term average surface infiltration flux (i.e., precipitation plus irrigation, L T^{-1}), and $\bar{\theta}$ is the average water content (estimated to be 0.26 cm^3 cm^{-3} from field measurements) in the soil profile

$T_0(t)$ is the rescaled time that accounts for deviations of the infiltration rate from its annual average value (T):

$$T_0(t) = t - t_0 \qquad [3.7]$$

Where t_0 is the initial time when a solute parcel enters the soil.

z_s is the depth from which the solute particle that is currently at depth z originated (L):

$$\begin{aligned} z_s &= \delta \cdot ln\left[\frac{(p+(1-p)\cdot exp(z/\delta))\cdot exp[(p-1)T_0(t)\overline{q_0}/R\bar{\theta}\delta]-p}{1-p}\right] && p \neq 1 \\ z_s &= \delta \cdot ln\left[exp(z/\delta) - \frac{T_0(t)\overline{q_0}}{R\bar{\theta}\delta}\right] && p = 1 \end{aligned} \qquad [3.8]$$

t_s is time at which the solute particle that currently is at depth z was introduced at the boundary (T)

$$t_s = t - \tau(z) \qquad [3.9]$$

More details about the solution can be found in the article by Schoups and Hopmans (2002). The solution was implemented using Mathcad software assuming a constant initial solute concentration with depth and a temporally constant concentration in the irrigation water.

Two applicable quantities can be derived from the model:

$$C_{wm}(t) = \int_0^\infty c(z,t)r(z)dz \qquad [3.10]$$

$$S_{cum}(z,t) = \int_{t_0}^{t} s(z,\omega)d(\omega) = \int_{t_0}^{t} q(z,\omega)c(z,\omega)d\omega \qquad [3.11]$$

Where,
C_{wm} is the solute concentration in the root zone weighted by the root distribution function (M L^{-3})
S_{cum} is the cumulative solute flux at depth z (M L^{-2})

The solute flux, S_{cum}, can be used to estimate loading rates to the groundwater, under varying irrigation conditions (drainage solute flux density in Fig. 3).

To run the Schoups and Hopmans (2002) model, long term average infiltrating flux was determined for various leaching fractions , $LF=p-1$ (Table 3.1). Even though the actual irrigation system was a subsurface system, for modeling purposes, it was assumed that irrigation was introduced at the soil surface. As mentioned in the results and discussion section, the purpose of the modeling exercise was to gain understanding about the dynamics of the system rather than accurately simulate it. The long-term average infiltrating flux is the sum of growing season precipitation, snow water infiltration and irrigation. This was determined by using long term historical records for precipitation (1969 to 2012). Infiltration of snow water was assumed to be 0.5*snow water equivalent of snowfall (as estimated by the Environment Canada data) to account for losses via sublimation and redistribution during the winter and evaporation and runoff during spring snow melt. The value of 0.5 was shown to be fairly accurate by comparing meteorological records of total snowfall with measured snow water equivalent in the snowpack sampled in late winter of 2010 (Section 2).

The amount of irrigation added to growing season precipitation and infiltrated snowmelt was determined for each leaching fraction according to:

$$LF = \frac{P+0.5*SWE+I-ET}{P+0.5*SWE+I} = \frac{D}{P+0.5*SWE+I} \qquad [3.12]$$

Where,
P is growing season precipitation [L]

SWE is snow water equivalent of incident snowfall [L]

I is irrigation [L]

D is drainage [L]

LF is the leaching fraction

Solving this equation for I yields:

$$I = \frac{(LF-1)*(P+0.5*SWE)+ET}{(1-LF)} \qquad [3.13]$$

Evapotranspiration was estimated using a temperature-based model from Maulé et al (2006) using meteorological data from a nearby Environment Canada weather station. Crop evapotranspiration was estimated using the reference ET and a crop coefficient value of 1.2, typical for a mature willow (Pistocchi et al 2009; Persson 1995).

The concentration of solutes in the irrigation was assumed to be constant regardless of the amount of irrigation. The concentration in the infiltrating water, however, was different for each leaching fraction because the amount of irrigation increased with increasing leaching fraction.

Using soil water concentrations predicted with the model, the following were calculated: 1) root zone solute concentration weighted by the root distribution function, C_{wm} (Eq. [10]); 2) drainage (L T^{-1}); and 3) groundwater loading rates (M $L^{-2}T^{-1}$) for chloride, nitrate, orthophosphate and total dissolved solutes (TDS) and/or electrical conductivity (EC). For modeling EC, the EC of the wastewater was converted to TDS using the following relationship from Chang et al (1983) :

$$TDS = 765.1\ EC^{1.087} \qquad [3.14]$$

Where
TDS has units of mg L^{-1}
EC has mS cm^{-1}

Various parameters were estimated for the model (Table 3.2). Initial solute concentrations in the soil and wastewater concentrations were based on measured values in the field from control soils. For nitrate-N, the root solute uptake parameter, a, was estimated to be 1.27 based on findings by Shi and Zuo (2007). For orthophosphate, an a of 1.38 was used as estimated by Zhu et al 2010. For nitrate and orthophosphate higher a values were also used to get an idea of the model sensitivity to a. A distribution coefficient, k_d, of 7.369 and 3.38, corresponding to a R value of 43.5 and 20.5, were used for orthophosphate to establish a range of values (Jiao et al 2007). Initial soil solute concentrations were estimated using soil solution sampler data from the control soil in 2010 and 2011. An exponential root distribution was assumed and the δ value used was 150 (Figure 3.4). The root-weighted root zone solute concentration for varying LF values was determined for steady state (SS) times. Drainage at 100 cm was determined by the product of surface infiltrating water and normalized water flux at that depth. Solute flux was determined to estimate the loading rates to the groundwater. Loading rates were determined by the product of drainage and solute concentration at 100 cm.

3.3.4 Wastewater Treatment Efficiency

The amount of solute removed by the plants and soil system, the treatment efficiency (TE), was determined for the soil solution sampler (field) data and results from the Schoups and Hopmans (2002) model for determining root zone solute concentrations. Treatment efficiency was calculated as follows:

$$TE = \frac{LR_{ww} - LR_{dw}}{LR_{ww}} \times 100 \qquad [3.15]$$

Where,

TE = treatment efficiency (%)

LR_{ww} = wastewater loading rate [M L^{-2}]

LR_{dw}= drainage loading rate [M L^{-2}]

The wastewater loading rates were calculated assuming that the wastewater was discharged directly to the river (which would occur in the absence of the SRC system) instead of being applied as irrigation. The groundwater loading rates were estimated from either (i) output from the Schoups and Hopmans (2002) model or (ii) from direct measurement of soil solution concentrations from the in situ soil solution samplers.

The TE for the Schoups and Hopmans (2002) model was determined using Equation [3.15]. The wastewater loading rates (LR_{ww}) were determined as the product of the irrigation water applied for a particular LF and the average wastewater solute concentrations [Equation 3.13](Table 3.1). The amount of irrigation water applied was the difference between average total surface flux (q_0) for a particular LF and the long term average rain and 0.5*SWE (472 mm) (Table 3.1). The drainage water loading rates (LR_{dw}) was calculated as the product of the solute concentration at 100 cm and the water flux at 100 cm, both output from Schoups and Hopmans (2002) model. The water flux at 100 cm was determined by multiplying the surface infiltrating flux by the normalized soil water flux at 100 cm (Equation [3.5]).

The components for deriving the soil solution sampler TE were slightly different than the Schoups and Hopmans (2002) model. The wastewater loading rates, LR_{ww}, were determined by multiplying the actual irrigation water volume applied by the average concentration of various solutes in the wastewater (Chapter 2, Table 2.4). The drainage water loading rates, LR_{dw}, were determined as the product of growing season drainage and average growing season solute concentration. The TE for the soil solution samplers were determined using Eq. [3.15].

3.4 RESULTS & DISCUSSION

3.4.1 Schoups and Hopmans (2002) Model Results

The limitations of the Schoups and Hopmans (2002) model should be noted. The model was used to gain an understanding of the solute transport through and below the root zone and to provide comparison to the soil solution sampler data. The goal with the model was not for accuracy but the relationship between irrigation management and solute leaching.
The Schoups and Hopmans (2002) model was ran til steady state conditions were reached at 100 cm for comparing results. All solute concentrations at 100 cm reached steady state conditions at varying times (Table 3.5). For all solutes, the lower leaching fractions took longer to reach steady state conditions because of lower drainage. Orthophosphate-P took the longest to reach steady state because of its high retardation factor and chloride was the quickest. Surface infiltrating flux and drainage increased with LF (Table 3.1).

The steady state root distribution function weighted soil solution solute concentrations (C_{wm}; Equation [3.10]) in the root zone were determined for various solutes. For nitrate-N and orthophosphate-P, C_{wm} was greater with a lower a value. The steady state root distribution function weighted soil solution solute concentrations, C_{wm}, increased with LF for all solutes (Figure 3.5). The steady state root distribution function weighted soil solution solute concentrations, C_{wm}, of orthophosphate-P was the same for both R values of 20.5 and 43.5, because the R values only affect the amount of time required to reach steady state.

Loading rates were determined for various solutes (Figure 3.7). For all solutes, loading rates increased with LF. Lower a values had higher loading rates for nitrate-N and orthophosphate-P as would be expected with lower solute uptake. Loading rates were the same for both R values for orthophosphate-P.

Steady state soil solution concentration profiles were determined for all solutes (Figure 3.8, 3.9). The concentration profiles in Figure 3.8 and 3.9 show abrupt changes in concentrations for the various solutes at depths greater than 100 cm which represent the separation between initial solute concentrations and new solute concentrations under the new irrigation regime. The concentration profiles show the influence of the exponential root distribution function. The high surface flux is exponentially damped by the root extraction of water at each depth. As a solute parcel moves down through the root zone, it is concentrated as the roots extract more and more water with depth. Nitrate and orthophosphate soil solution concentrations will be higher after irrigation for most LF. However, chloride and EC also show an increase in concentrations from irrigation but can be reduced by increasing the LF.

3.4.2 Schoups and Hopmans (2002) Discussion

Surface infiltration flux and drainage increased with LF because irrigation was increased to achieve the higher LF under constant long-term average precipitation and evapotranspiration. Because higher LFs were associated with increased surface fluxes, the time to reach steady state conditions decreased with higher LFs. In this system, the majority of the solutes are contained in the irrigation water. Therefore, the steady state root zone distribution function weighted solute concentrations for all solutes increased with LF. At greater LFs there is more wastewater applied and additional solutes being added to the soil.

Steady state soil chloride concentrations and EC at 100 cm decreased with increasing LF while nitrate and orthophosphate increased (Figure 3.6, 3.7), showing the influence of simulated nutrient (nitrate and orthophosphate) uptake by plants. Due to its mobility and lack of plant uptake, more chloride is leached below the root zone with higher LF. The model assumed that EC experienced no plant uptake or retardation, behaving similarly to chloride. Even though many solutes that make up the total dissolved solids in the soil solution are cations and may be subject to adsorption, at steady state, it is likely that charges in the soil solution are balanced (solution cations are in equilibrium with sorbed cations) and using EC as a conservative tracer is likely justified. Nitrate is mobile like chloride but is taken up by plants. Nitrate soil solution concentrations at 100 cm increased with LF because larger LFs are adding more nitrogen to the soil than initially present and plants not taking up nitrogen at same rate as it is supplied. Orthophosphate is also taken up by plants but is highly non-mobile and readily binds to soil particle surfaces and forms sparingly soluble complexes with calcium. The soil solution orthophosphate increases with LF because the rate of orthophosphate additions is greater than removal, like nitrate. Aqueous orthophosphate-P concentrations are lower than the other solutes because, once added, it is immediately bound to the soil. The model assumes that partitioning is an instantaneous linear equilibrium relationship between solution orthophosphate and bound soil particle orthophosphate. Loading rates for all solutes increase with LF. With increasing LF, more solutes are being added and, coupled with higher drainage, results in higher loading rates.

The steady state concentration profiles determined using the Schoups and Hopmans (2002) model give additional insight into how the solutes are behaving in the soil and can be used to get a general idea of the long term impacts of the irrigation with the wastewater with respect to irrigation management. Chloride and EC increase with depth for most LF which indicates that solutes are concentrating as they move through the root zone due to plant water uptake (Figure 3.8). Figure 3.9, nitrate-N and orthophosphate-P concentration profiles tend to be higher at shallower depths and this is more pronounced with higher LFs. This indicates a concentrating effect from roots which is more pronounced at lower LFs. The wastewater is contributing nitrate and orthophosphate to the soil, which is beneficial for plants. If accumulation of solutes like chloride in the root zone is undesirable, then irrigation can be adjusted to leach more solutes below the root zone. This practice, however, increases the additions of nitrate and orthophosphate to the soil system that may leach to groundwater.

Secondary salinization of the root zone is a threat to long term sustainability and productivity of SRC plantations irrigated with wastewater. The simulations show that irrigation increases the EC in the soil solution of the root zone over initial soil conditions (Figure 3.6 and 3.7). Actual soil EC measurements taken from various depths in the root zone have also shown an increase in soil EC with irrigation years (Table 3.3). However, only very low LFs (0.1) result in EC soil solution concentrations in the root zone close to the “fair” (2 to 4 dS/m) category under Alberta Tier 1 Salt Remediation Guidelines for topsoil and “good” for subsoil (Alberta Environment 2010). The topsoil guidelines will be applied as they are more conservative.

According to these simulations, lower LFs are result in increased removal of nitrate and orthophosphate from the wastewater by plant uptake. The simulations indicate that the longer residence time of the solutes in the root zone under lower irrigation rates increases nutrient uptake, or treatment efficiency. The greater uptake by the trees has benefits for plantation productivity and decreased nutrient loading to groundwater. The simulations suggest that managing irrigation such that LF are around 0.1 results in greater nutrient removal from the wastewater without serious threat of detrimental salinization of the root zone. However, long-term monitoring of the salinity levels in the root zone is still recommended. Also to note, nitrate-N soil solution concentrations did not exceed Health Canada potable water guidelines of 10 mg L^{-1} until a LF of about 0.5 (Health Canada 2010).

The Schoups and Hopmans (2002) model provided benefits and insight into the soil system but there are various limitations with the model. The model is most accurate when using conservative solutes. Plant uptake of nutrients is simulated as a root sink but the model not does account for other nitrogen and phosphorus cycling processes like mineralization, immobilization, denitrification, fixation, etc. Also, the model assumed that the distribution coefficient followed linear sorption for orthophosphate. Another limitation with the Schoups and Hopmans (2002) model is that it assumes steady state conditions and a homogeneous soil profile. Due to the steady state assumptions, it is not capable of simulating year to year variability. Furthermore, from field observations, the soil profile is not vertically homogenous but has a clay layer at 30 cm and gravel lense at about 1 m. Another assumption with the model is that water is applied at the surface but at the study site the irrigation water is released in the sub surface at about 20 cm.

3.4.3 Field Measurements of Soil Water Quality Results

In 2010 and 2011, soil water at various depths below the root zone of the control and wastewater irrigated areas were monitored for various solutes. All soil solution samplers had concentrations less than that of the wastewater (Table 3.4). For all solutes the variance was greater in 2011 than 2010. This may be due to less samples collected in 2011 than 2010.

In 2010 and especially in 2011, TN soil concentrations were typically higher in the wastewater irrigated soil (Table 3.4). In the control and wastewater irrigated soil, the highest TN concentration in 2010 was at 150 cm (2.08 mg L^{-1}) and in 2011 varied between 50 cm and 90 cm (3.85 and 3.29 mg L^{-1}), both in the wastewater irrigated soil. In the wastewater irrigated soil, the 2011 samples had higher concentrations than the 2010 samples but there was more variability in the former. TN loading rates for 2011 were higher in the wastewater irrigated soil than the control (Table 3.5).

In 2010 and 2011, nitrate-N soil concentrations were higher in the wastewater irrigated soil than the control (Table 3.4). In the control, the nitrate concentrations were similar between years and depths. In the wastewater irrigated soil, the highest concentrations were found at 150 cm in 2010 (1.53 mg L^{-1}) and 90 cm in 2011 (2.63 mg L^{-1}). Nitrate-N concentrations did not vary greatly over the growing season except at 90 cm in 2011. Nitrate-N loading rates for 2011 were greater in the wastewater irrigated soil than the control (Table 3.5).

Ammonium-N concentrations in the control were similar for 2010 and 2011 (Table 3.4). In the wastewater irrigated soil, the ammonium-N concentrations and variance were greater in 2011 than 2010. The highest ammonium-N concentrations were typically at 50 cm (1.31 mg L^{-1}) for the wastewater irrigated soil. The ammonium-N concentration may be higher at 50 cm than deeper depths because it is not especially mobile like nitrate. In 2010, the wastewater irrigated soil concentrations were similar to the control concentrations, with typical concentrations of 0.04 mg L^{-1} at 50 cm. It should be noted that the majority of ammonium samples collected in 2011 were below detection limit, reducing the accuracy. Concentrations of ammonium-N in the control were similar in 2010 and 2011. Ammonium-N loading rates for 2011 were greater in the wastewater irrigated soil than the control (Table 3.5).

In 2011, orthophosphate-P concentrations were similar between treatments (Table 3.4). The highest orthophosphate-P concentrations were at 50 cm (0.35 mgL^{-1}) in the wastewater irrigated soil but had high variance. Loading rates for orthophosphate-P were greater in the wastewater irrigated soil than the control for 2011 (Table 3.5).

DOC concentrations were greater in 2011 than in 2010 (Table 3.4). In 2010, DOC concentrations in the control and wastewater irrigated soil were similar and concentrations were similar between depths. In 2011, the DOC concentrations were higher in the control than the wastewater irrigated soil and the shallow depths had greatest concentrations. The greatest concentrations were in the control at 50 cm. The higher variance for 2011 is also due to the variability in concentrations over the growing season (Figure 3.10). DOC loading rates for 2011 were greater in the wastewater irrigated soil than the control (Table 3.5).

3.4.4 Field Measurements of Soil Water Quality Discussion

Soil solution concentrations found in 2010 and 2011 were similar to a few other related studies involving wastewater irrigation. The concentrations of TN in the soil below the wastewater irrigated zone were similar to a study by Hasselgren (1997). Nitrate concentrations at 150 cm were similar to a study by Sopper (1970) where hardwoods were irrigated with municipal treated wastewater. Nitrate-N concentrations at all depths and for 2010 and 2011 did not exceed Health Canada potable water guidelines of 10 mg L^{-1} (Health Canada 2010) or Alberta Environment Tier 1 Groundwater Remediation Guidelines of 13 mg L^{-1} (Alberta Environment 2010).

There is high variance for many of the solutes sampled in the soil solution samplers. Variability of solute concentrations within treatments may be due to various factors. Within the wastewater irrigated zones, variability may be due to position of samplers in reference to the irrigation line outlet points. For example, the soil solution sampler at location T6 was situated near an outlet point along the irrigation line, as you could hear the water exiting during irrigation. Samples collected at T6 typically had higher concentrations than the other samplers in the treatment. Variability between samplers may also be due to plant health. Plants near soil solution samplers T1 and T2 in the control had minimal foliage and growth for most of 2010 compared to the other locations in the control, which may have resulted in elevated concentrations in the control because of reduced plant uptake of nutrients. Dimitriou and Aronsson (2010) noted that more established root systems, found in healthier plants, have greater nutrient uptake and lower nutrient concentrations in soil solution. Within the control zone, the variability in soil concentrations may also be due to moisture content. Locations T1 and T2 have higher moisture contents than T3 and T4, mostly due to topographic position. As mentioned earlier, variability within both treatments may be due to temporal variation in concentrations. For example, DOC and ammonium varied over the growing season, increasing the variance (Table 3.4). Solute concentration in the wastewater also had temporal variability in concentrations (Table 3.6).

Concentrations of various solutes tend to be higher in shallower depths, except for nitrate-N at 90 cm in the wastewater irrigated soil. In the wastewater irrigated soil, the irrigation water is released at about 25 cm, a source of solutes at a shallow depth. Nitrate's mobility may be responsible for the elevated concentrations at 90 cm. In the control, the higher concentrations at lower depths maybe attributed to increased organic matter additions at shallower depths.

In general, the nitrogen species had higher concentrations in the wastewater irrigated soil than the control whereas DOC are greater in the control. This is especially pronounced in 2011 data while the 2010 data shows similar concentrations between treatments. Livesley et al (2007) also found higher nitrate and ammonium concentrations in the root zone when wastewater was applied. The leaching of nitrate during irrigation period may be due to increased moisture content. Studies by Silva et al (2000) have suggested that leaching losses are due to the dominant macropore flow that occurs at higher moisture contents. The DOC concentrations were lower in the wastewater irrigated soil but the overall loading rates were higher in the wastewater irrigated soil. The orthophosphate concentrations had high variance and the wastewater irrigated soil having slightly higher concentrations at lower depths.

In 2010, ammonium-N concentrations decreased over the growing season while in 2011 they increased. This variation may be due to limited data collected in 2011. Most of the 2011 ammonium data was below the detection limit, so a more precise trend may not be apparent. Also the high variance may be due to a few high concentrations measured toward the end of the growing season but no overall pattern was observed. When considering just the 2010 data an explanation for the decrease in ammonium-N levels may be due to nitrification. Nitrification is the conversion from ammonium to nitrate by bacteria. Nitrification is temperature sensitive, hence, increasing in the summer months (Brady and Weil 2004). In a study by Livesley et at (2007) where a eucalyptus plantation was irrigated with sewage effluent, ammonium-N concentrations increased in the spring and then decreased for the remaining growing season for both the effluent irrigated and just regular water irrigated soils due to nitrification. Nitrification of ammonium may explain the increase in nitrate concentrations later in the growing season of both 2010 and 2011. In a study by Hermann et al (2005), they also found a decrease in ammonium concentrations due to leaching below the root zone of a forested soil in the summer months.

DOC in soil water of both the control and wastewater irrigated soil decreased over the growing season for both years. Martin-Olmedo and Reeves (1999) found a decrease in DOC after manure application. Decrease in DOC in the control with time may be due to lack of organic matter contributing to DOC. Moore et al (2008) found a decrease in DOC in mineral soils due to lack of organic matter additions. Peichl et al (2007) had decreasing DOC levels in the soil solution of an Ah soil horizon from a 15 year old white pine (*Pinus strobus* L.) stand from June to September. DOC concentrations remain low in fall, even though there are organic matter contributions, due to cold temperatures limiting microorganisms. Various studies show that temperature is a major control on DOC, with higher temperatures associated with greater DOC release (Peichl et al 2007; Dalva and Moore 1991; Guggenberger et al 1998; Kalbitz et al 2000). Soil solution DOC decreases over the growing season may also be due to reduced precipitation. A study by Peichl et al (2007) found that DOC concentrations in forest floor soil solutions had a negative expontential relationship with soil moisture. Decreases in DOC in soil solution may also be attributed to adsorption processes (Dosskey and Bertsch 1997; Kalbitz et al 2000; Michalzik et al 2001). DOC concentrations also varied from year to year. The soil solution DOC concentrations were greater for both treatments in 2011 which is mostly likely due to increased age of trees and biomass contributing to soil.

3.4.5 Wastewater Quality and Treatment Efficiency Results

Wastewater samples were monitored over the 2010 growing season and historical data was available for DOC, nitrate-N, orthophosphate-P, ammonium-N, chloride and TN (Table 3.6). In 2010, wastewater concentrations ranged from 9.1 to 42.5 mg L^{-1} for DOC, 20.9 to 29.3 mg L^{-1} for TN , 9.4 to 23.5 mg L^{-1} for nitrate-N, 0.1 (detection limit) to 1.43 mg L^{-1} for ammonium-N and 0.05 to 4.63 mg L^{-1} for organic nitrogen. In 2011, wastewater concentrations ranges were 11.3 to 14.1 mg L^{-1} for DOC, 15.2 to 21.7 mg L^{-1} for TN, 13.4 to 20 mg L^{-1} for nitrate-N, below detection limit (0.05 mg L^{-1}) for ammonium-N and 0.37 to 1.44 mg L^{-1} for orthophosphate-P. In 2010 and 2011, DOC concentrations in the wastewater decreased over the growing season while the remaining solutes were variable. DOC concentrations in the wastewater were greater in 2010 than 2011. The remaining solutes were similar between years.

To get an estimate of irrigation water solute removal by plants and soil system, treatment efficiencies were determined for both model and soil solution sampler methods of estimating leaching to groundwater. From the Schoups and Hopmans (2002) model findings, the highest TE was consistently greater for orthophosphate and nitrate (Figure 3.11). Also, with orthophosphate-P and nitrate-N, higher TEs are associated with lower LF. The TE for chloride and TDS were typically negative or near zero because these do not get taken up my plabts. The year with the highest Schoups and Hopmans (2002) TE was 2006.

TE was calculated using the soil solution sampler data as well (Figure 3.11). The highest TE was for nitrate-N in 2011. The control had higher TEs than the wastewater irrigation soil for nitrate-N, TN and orthophosphate-P. This is similar to the Schoups and Hopmans (2002) TE results in that lower LF have higher TE.

3.4.5 Wastewater Quality and Treatment Efficiency Discussion

The soil solution sampler TE's were positive for both years. Similarly to the Schoups and Hopmans (2002) results, the soil solution sampler TE were higher under lower LF environments. A lower LF increases the solute residence time in the soil and therefore increases the possibility of plant uptake. A study by Livesley et al (2007) found with longer residence times ammonium was more exposed to transformations and immobilizations resulting in reduced soil solution concentration of the nutrient. It should be noted that there are limitations in the derivation of the TE. For example, the growing season average solute concentration was used so the TE for each solute is generalized.

Findings from a 15 year study in Sweden also have high nitrate and ammonium removal rates for landfill leachate and municipal treated wastewater applied to SRC systems. Hasselgren (1998) found reductions of 85 to 95% and 95 to 96% for total N and total P, respectively, from municipal treated wastewater irrigation. The study also found reductions in ammonia from 96.8 to 99.9% and total nitrogen from 43.4 to 93.3 % in landfill leachate applied to SRC plantations. Dimitrou and Aronson (2010) found similar results with efficiency for landfill leachate with willow and poplar. Retention of total N ranged from 82 to 94 %, for total P was between 56 and 80% and TOC was between 72 and 86% (Dimitrou and Aronson 2010).

3.5. CONCLUSIONS AND RECOMMENDATIONS

Renewable forms of energy are important in mitigating global warming. SRC crops with wastewater irrigation are a form of renewable energy with low greenhouse gas emissions that utilizes plant material for energy. SRC plantations involve the management of woody species like willow harvested in short time intervals for energy purposes. By irrigating with municipal treated wastewater the plant and water needs of the plants are addressed in a relatively low cost approach. There are groundwater and drinking water contamination concerns when using wastewater as an irrigation source. This research focused on the quality of nutrient leaching using a field based method and a transport model.

From this research, it was found that land application of municipal treated wastewater is effective at removing various solutes including nitrate and orthophosphate. The loading rates of nutrients to groundwater under SRC with wastewater irrigation crops are less than the loading rates of solutes discharged directly to the river, which is the common practice. The field collected nitrate-N soil solution concentrations above the groundwater table (at 150 cm) in 2010 and 2011 never exceeded the Alberta Tier 1 Guidelines for Groundwater Remediation limit of 13 mg L^{-1} or the potable water limit for nitrate-N of 10 mg L^{-1} (Figure 3.12) (Alberta Environment 2010; Health Canada 2010). These years were lower than usual LF and the previous irrigation years may have higher concentrations due to higher LF. However, from the Schoups and Hopmans model predictions, LF > 0.6 results in solution nitrate-N concentrations in exceedance of potable water guidelines (Figure 3.8). As there is not field collected soil solution samples from years with high LFs (>0.25) and the Schoups model predicts a LF>0.6 resulting in soil solution nitrate-N above potable water guidelines, to protect drinking water quality it is recommended to not exceed a LF of about 0.5. The historical TE from Schoups and Hopmans model were relatively high for nitrate and orthophosphate for the study site (Figure 3.11).

Over the last six years the nutrient application rates at the Whitecourt study site varied and, on average, were 2.21 kg P ha^{-1} yr^{-1} and 94.49 kg N ha^{-1} yr^{-1}(Table 3.7). A study by Brown and Van den Driesche (2005) found that applications of 100 to 200 kg P ha^{-1} yr^{-1} and 500 kg N ha^{-1} yr^{-1} gave substantial growth for poplar hybrids. With findings from Dimitriou and Aronsson (2010) work in Sweden, they recommend applying about 126 kg N ha^{-1} yr^{-1}. From their study in Finland, Hytönen and Sarrsalmi (2009) found increases in willow growth when applying 100 kg N ha^{-1} yr^{-1} and 40 kg P ha^{-1} yr^{-1}, especially in the first ten years of a SRC plantation. Ledin (1986) recommended an application of 60 kg N ha^{-1} yr^{-1} in the establishment year followed by 80 to 120 kg N ha^{-1} yr^{-1} and 30 kg P ha^{-1} yr^{-1} in subsequent years, from his study in Sweden. In Sweden the current production recommendations are 45, 100 to 150 and 90 to 120 kg N ha^{-1} yr^{-1} for the second, third and fourth years, respectively, of a four year rotation coppice crop (Danfors et al 1998). A LF of at most 0.8 will result in a TN application rate close to these recommendations. However total phosphorus will be lower than recommendations and fertilizer may be required. A site specific phosphorus test would be beneficial. Also, variable application rates for each growing season would be beneficial as plant size varies greatly on SRC plantations, especially post-coppice.

For SRC plantations with wastewater irrigation systems similar to the research site, root zone salinization could become an issue. From the Schoups and Hopmans (2002) model, the EC of the soil solution varied with LF but mostly remained “fair” quality (Alberta Environment 2010). As mentioned earlier, the greatest treatment of nitrate and orthophosphate are at lower LF but this may cause soil salinization. Long term monitoring of root zone salinity is recommended. Actual LFs on the study site have varied greatly over the operating years (Table 3.7). This is mainly due to the variation in tree biomass due to age. Based on the findings from this research, the SRC with wastewater irrigation systems similar to the research site at Whitecourt, Alberta, present a low threat to soil salinization and groundwater contamination by nitrate or orthophosphate when following recommendations made from authors.

As mentioned earlier, the nitrate soil solution concentrations deep in the soil for 2010 and 2011 are below the Health Canada Drinking Water Quality Guidelines and Alberta Environment Guidelines and, thus, groundwater contamination is a low threat. However, in the past there have been years with high LFs (0.55) but field soil solution samples are not available to determine if concentrations are above guidelines. From the Schoups and Hopmans (2002) model, soil solution nitrate-N concentration exceeds potable water limits at LF of between 0.5 and 0.6. Also, a too small LF may have higher TE but the amount of nitrogen being added via wastewater irrigation may be not meeting plant requirements and root zone salinization will most likely occur. Using the field data as well as the model, it is recommended to use a LF in the range of 0.2 to 0.5. By applying this range of L, plants are receiving nutrients and water, groundwater contamination is minimized and municipal wastewater is utilized.

3.6 TABLES

Table 3.1. Average total surface flux (q_0), based on historical climate data from 1970 to 2010 and hypothetical irrigation amounts to determine leaching fractions for Schoups and Hopmans (2002) model. Solute concentrations in infiltrating water (c_0) for varying leaching fraction (LF) values for the study site at Whitecourt, Alberta.

					c_0 (mg L^{-3})			
LF	p	q_0 (mm yr^{-1})	Irrigation (mm)	Drainage (mm)	Cl^-	NO_3^--N	PO_4^{3-}-P	TDS
0.1	0.9	636.2	164.2	64.4	0.023	0.005	1.35E-04	0.155
0.25	0.75	760.2	288.2	190.8	0.033	0.007	1.98E-04	0.227
0.4	0.6	950.3	478.3	380.8	0.044	0.010	2.63E-04	0.302
0.5	0.5	1140.0	668.0	570.7	0.051	0.011	3.06E-04	0.351
0.6	0.4	1425.0	953.0	855.9	0.058	0.013	3.49E-04	0.401
0.75	0.25	2281.0	1809.0	1711.5	0.069	0.016	4.14E-04	0.475
0.9	0.1	5702.0	5230.0	5131.8	0.080	0.018	4.79E-04	0.549

*determined as difference between q_0 and long term average rain and 0.5*SWE (472 mm)

Table 3.2. Parameters for the Schoups and Hopmans (2002) model based on data from the study site in Whitecourt, Alberta and scientific literature.

Parameter	Unit	Chloride	Nitrate-N	Phosphate-P	Total Dissolved Solids
C_i	mg L^{-1}	0.087	0.30	0.13	193.9
a		0	1.27	1.38	0
R		1	1	43.5 20.5	1

Table 3.3. Historical average root zone EC (mS cm^{-1}) and coefficient of variation (%) of the wastewater irrigated soil at various root zone depths at the study site in Whitecourt, Alberta. Note this data did not meet the assumptions of normality but normal distribution was assumed for this general analysis.

	Mean EC (mS cm^{-1}) (CV (%))		
	Depth (cm)		
Date	5	20	50
10/10/2006	0.24 (51)	0.17 (50)	0.16 (29)
10/1/2007	na	na	0.37 (51)
10/1/2009	0.31 (20)	0.22 (19)	0.20 (22)
5/21/2010	0.92 (25)	na	0.51 (23)
7/14/2011	0.31 (37)	0.24 (12)	0.26 (36)

CV=Coefficient of Variance (%)

Table 3.4. Arithmetic mean and coefficient of variation for soil solution solutes at various depths over the 2010 and 2011 growing season at the study site in Whitecourt, AB. Note this data did not meet the assumptions of normality but normal distribution was assumed for this general analysis.

		Arithmetic Mean (CV) (mg L^{-1})			
		2010	2010	2011	2011
Solute	Depth (cm)	Control	WW	Control	WW
TN	50	1.23 (66)	1.0 (54)	1.36 (6)	3.85 (317)
	90	1.2 (92)	1.61 (69)	1.67 (151)	3.29 (156)
	150	2.05 (44)	2.08 (56)	1.16 (91)	1.11 (89)
NO_3^--N	50	0.29 (171)	0.27 (15)	0.19 (329)	0.17 (368)
	90	0.25 (85)	1.00 (136)	0.32 (263)	2.63 (182)
	150	1.33 (92)	1.53 (93)	0.48 (210)	0.37 (241)
NH_4^+-N	50	0.05 (121)	0.04 (151)	0.03 (0.00)	1.31 (392)
	90	0.029 (66)	0.033 (124)	0.029 (54)	0.052 (185)
	150	0.095 (97)	0.06 (221)	0.03 (63)	0.03 (0.00)
DOC	50	29.97 (44)	28.62 (56)	367.8 (237)	121.15 (148)
	90	32.02 (4)	20.81 (65)	92.38 (171)	99.85 (249)
	150	18.66 (91)	15.37 (71)	206.68 (149)	302.98 (135)
PO_4^{3-}-P	50	na	na	0.13 (48)	0.35 (285)
	90	na	na	0.122 (56)	0.088 (63)
	150	na	na	0.1 (74)	0.12 (65)

na- data not available for this year
CV- coefficient of variance(%)

Table 3.5. Loading rates of various solutes to groundwater for 2010 and 2011 growing season at the study site in Whitecourt, Alberta, determined using soil solution sampler data at 150 cm.

	Groundwater Loading Rates (kg ha^{-1}) and Year		
	WW	C	WW
Solute	2010	2011	
TN	1.47	0.51	2.52
Nitrate-N	1.55	0.21	1.56
DOC	13.60	139.29	872.69
Ammonium-N	0.07	0.01	0.05
Orthophosphate-P	0.26	0.04	0.26
ON	1.43	0.28	1.43

Table 3.6. Wastewater solute concentrations for various solutes for the study site at Whitecourt, Alberta Note this data did not meet the assumptions of normality but normal distribution was assumed for this general analysis.

Solute	Temporal Range	N	Mean (mg L^{-1})	CV (%)
TN	2006-2011	23	18.9	25.8
NO_3^--N	2006-2011	32	18.3	18.1
NH_4^+-N	2006-2011	26	0.1	204.3
DOC	2010-2011	20	17.7	51.0
PO_4^{3-}-P	2006-2011	19	1.2	49.3
Cl^-	2006-2011	11	73.0	26.2

Table 3.7. Actual leaching fractions (LF) and solute application rates over the years of operation at the study site in Whitecourt, AB based on the soil water balance.

		Application Rate (kg ha^{-1} yr^{-1})	
Year	LF	TN	PO_4-P
2006	0.281	9.97	0.23
2007	0.594	199.53	4.62
2008	0.520	220.14	5.10
2009	0.506	110.50	2.56
2010	0.050	23.81	0.55
2011	0.225	26.43	0.61

3.7. FIGURES

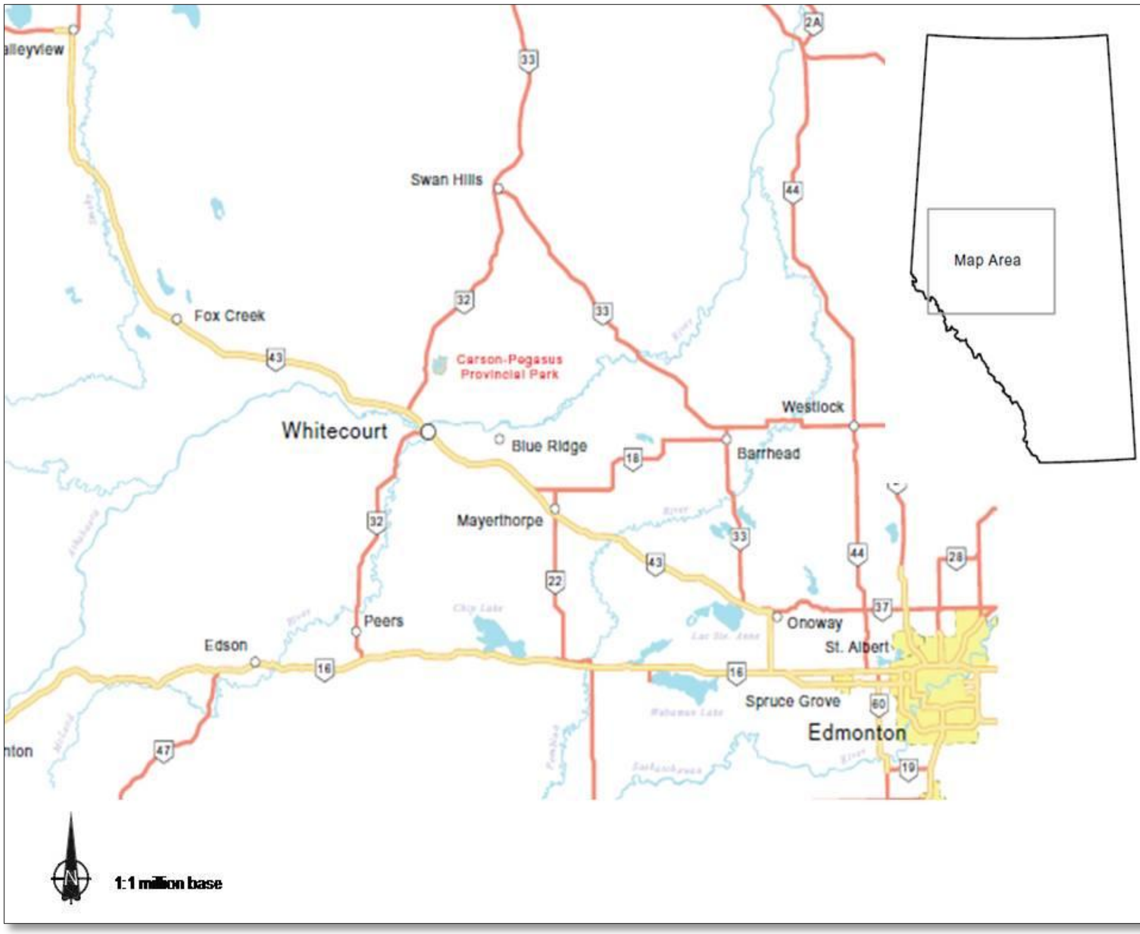

Figure 3.1. Map displaying the SRC with wastewater irrigation study site at Whitecourt, Alberta, Canada, in proximity to Edmonton

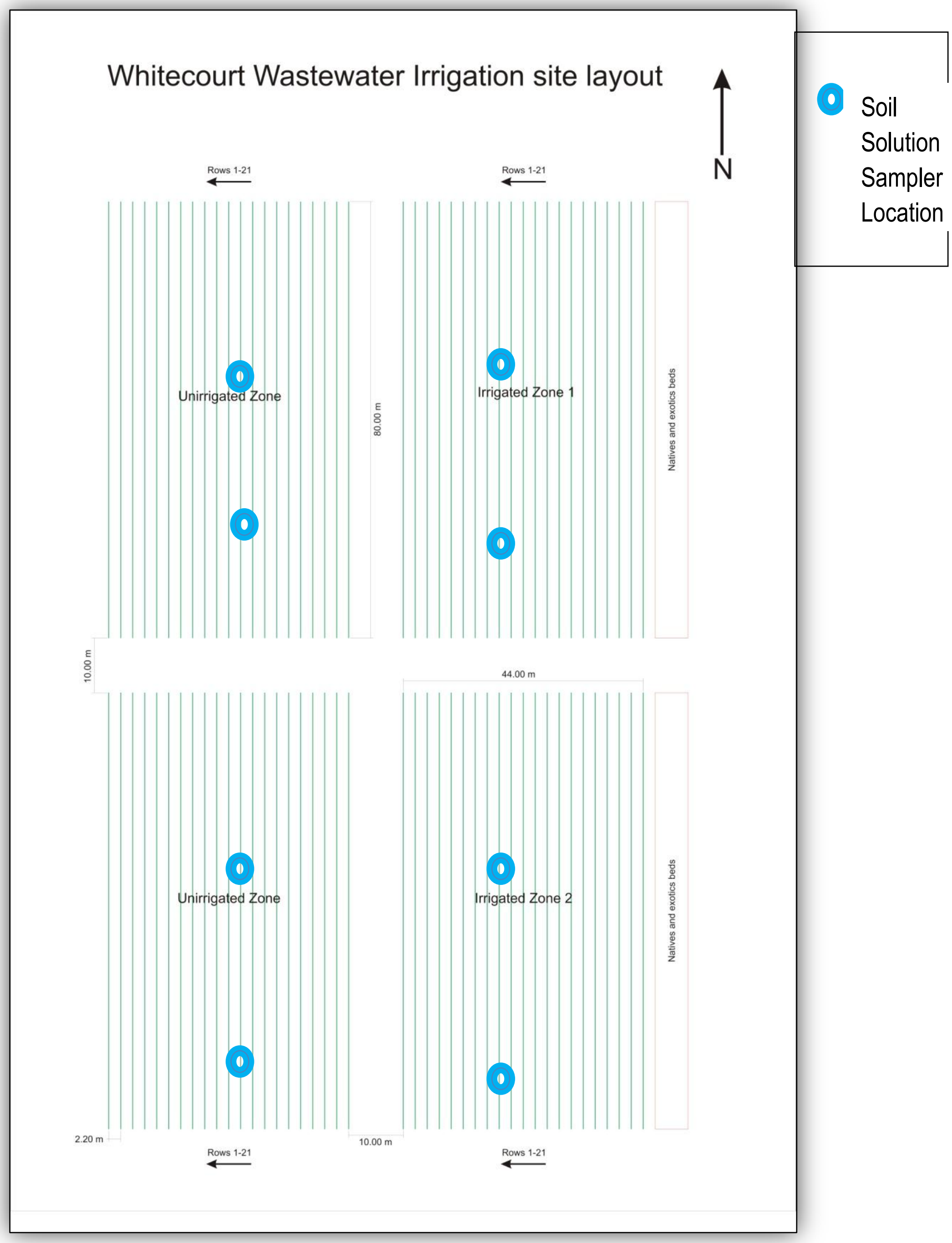

Figure 3.2. Experimental plot layout and soil solution sampler locations at the study site in Whitecourt, Alberta.

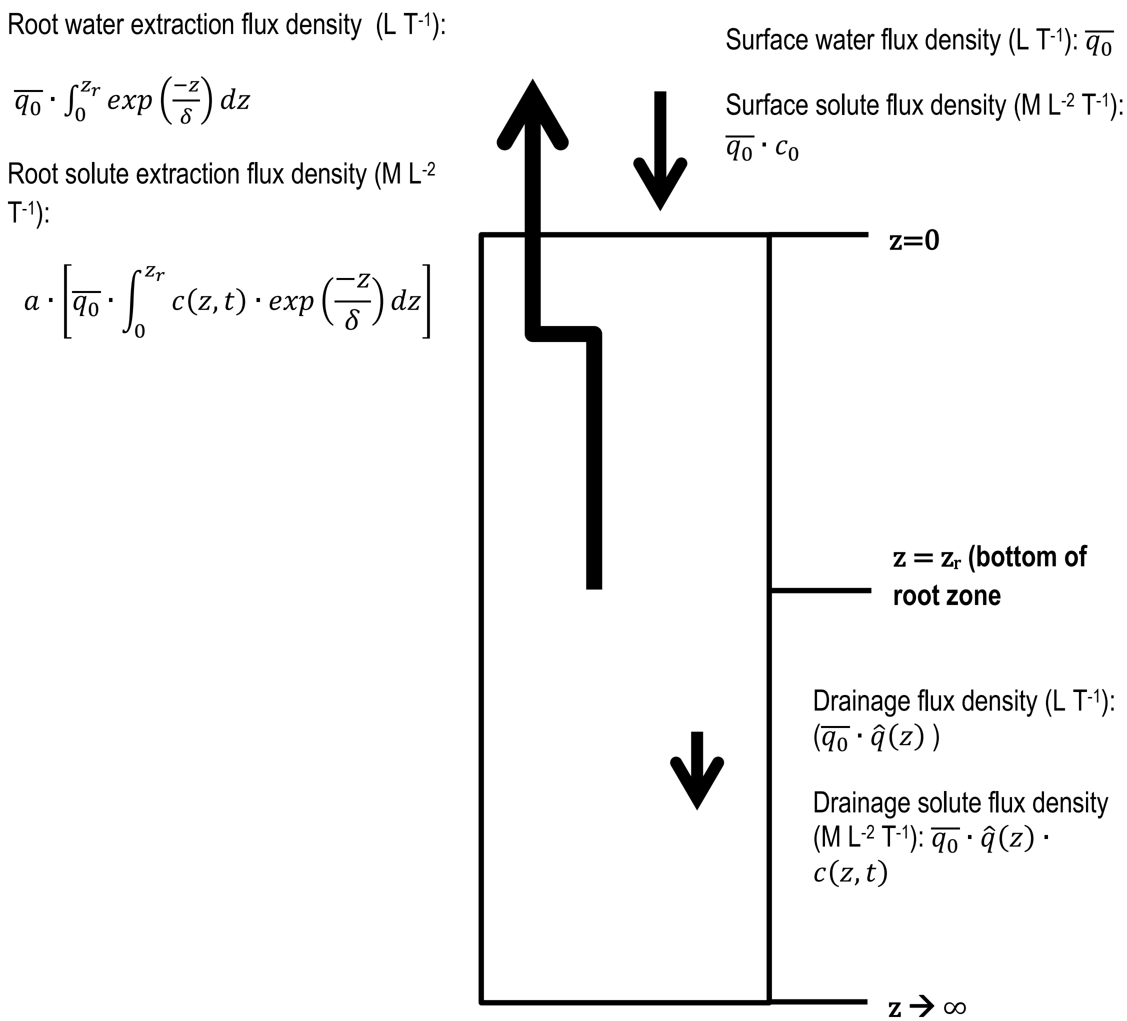

Figure 3.3. Basic schematic summarizing Schoups and Hopmans (2002) analytical solute transport model with root water and solute uptake.

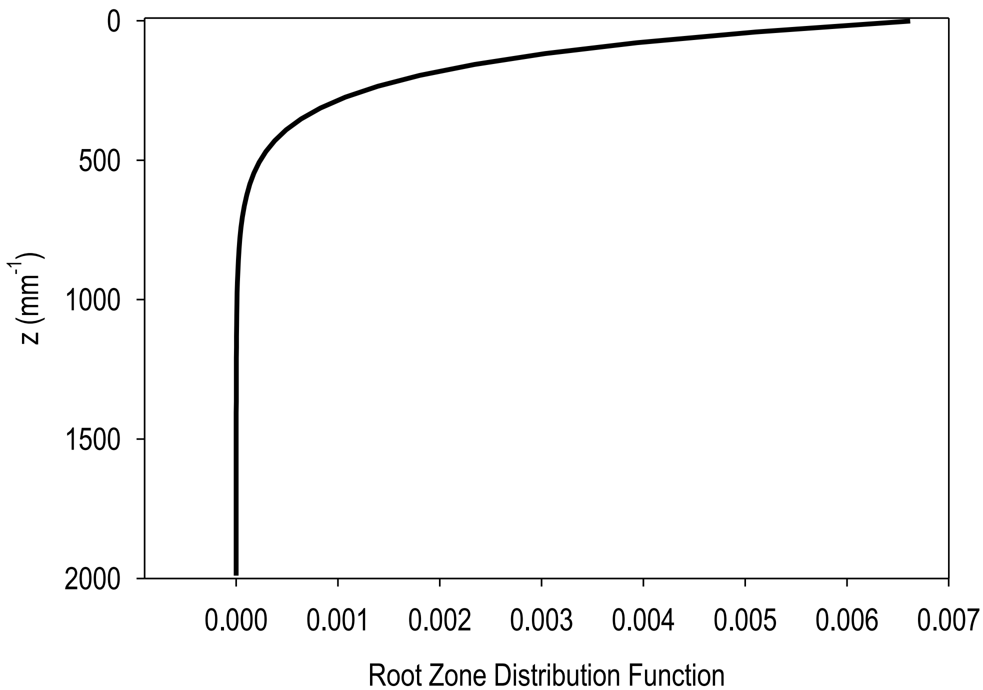

Figure 3.4. The root zone distribution function with depth used in the Schoups and Hopmans (2002) model for the study site at Whitecourt, AB.

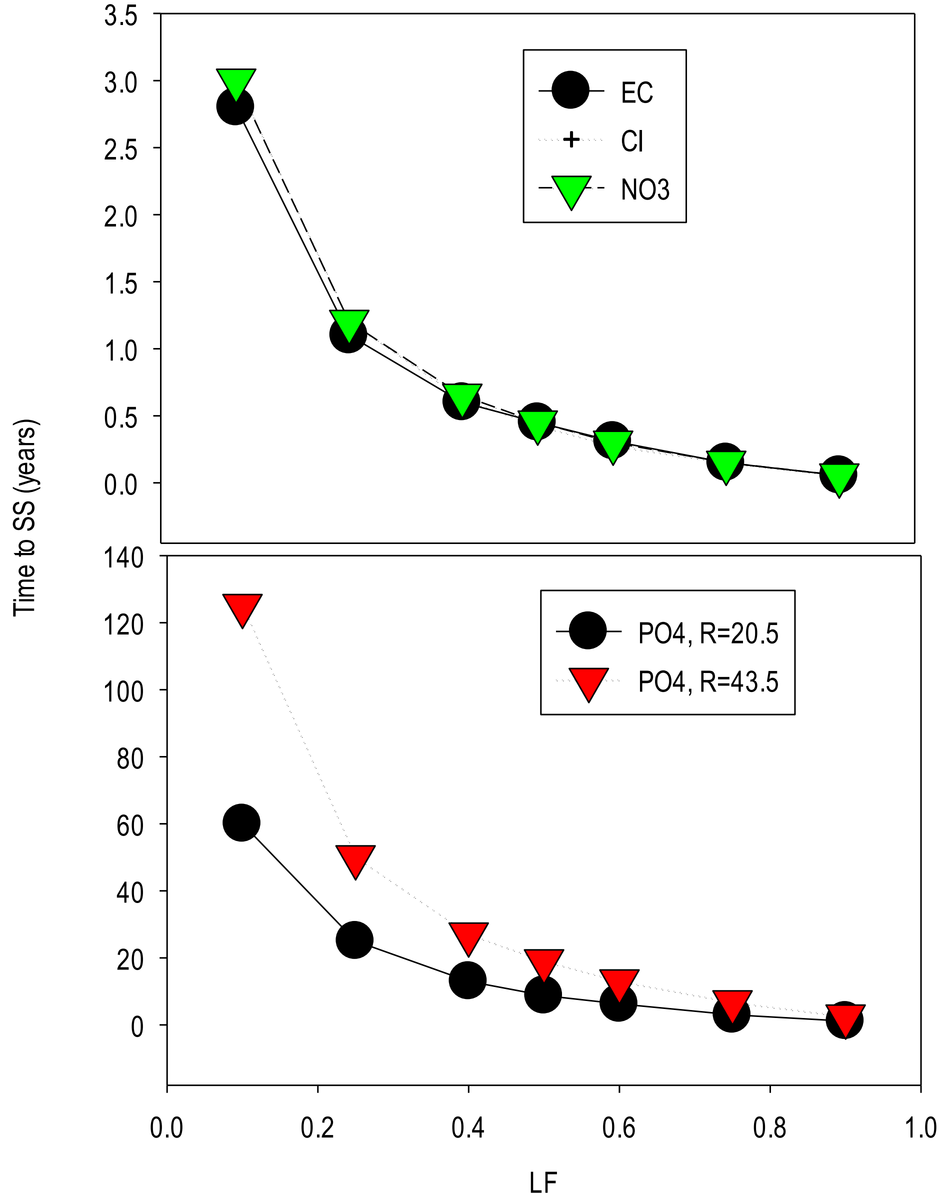

Figure 3.5. The time for steady state (SS) solute concentrations at 100 cm depth to be reached as a function of leaching fraction (LF) from the Schoups and Hopmans (2002) model for the study site at Whitecourt, AB.

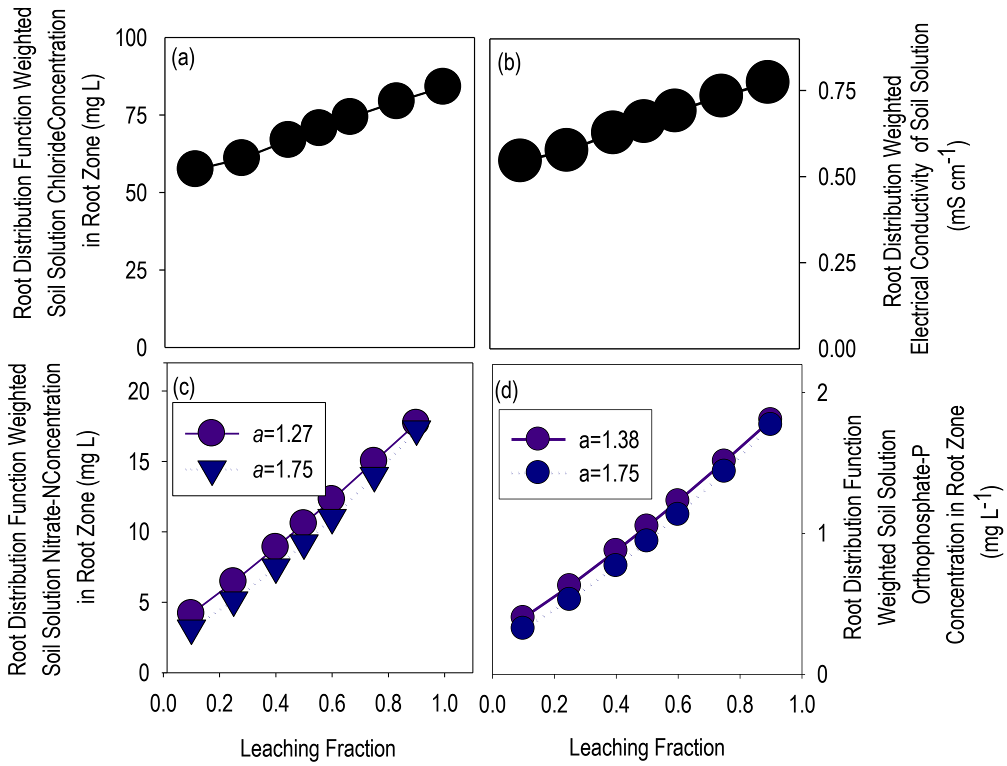

Figure 3.6. The steady state root distribution function weighted soil solution solute concentrations (C_{wm}; Eq. [10]) as predicted by the Schoups and Hopmans model for (a) chloride, (b) EC, (c) nitrate-N and (d) orthophosphate-P at study site for a root zone depth of 100 cm and various a values.

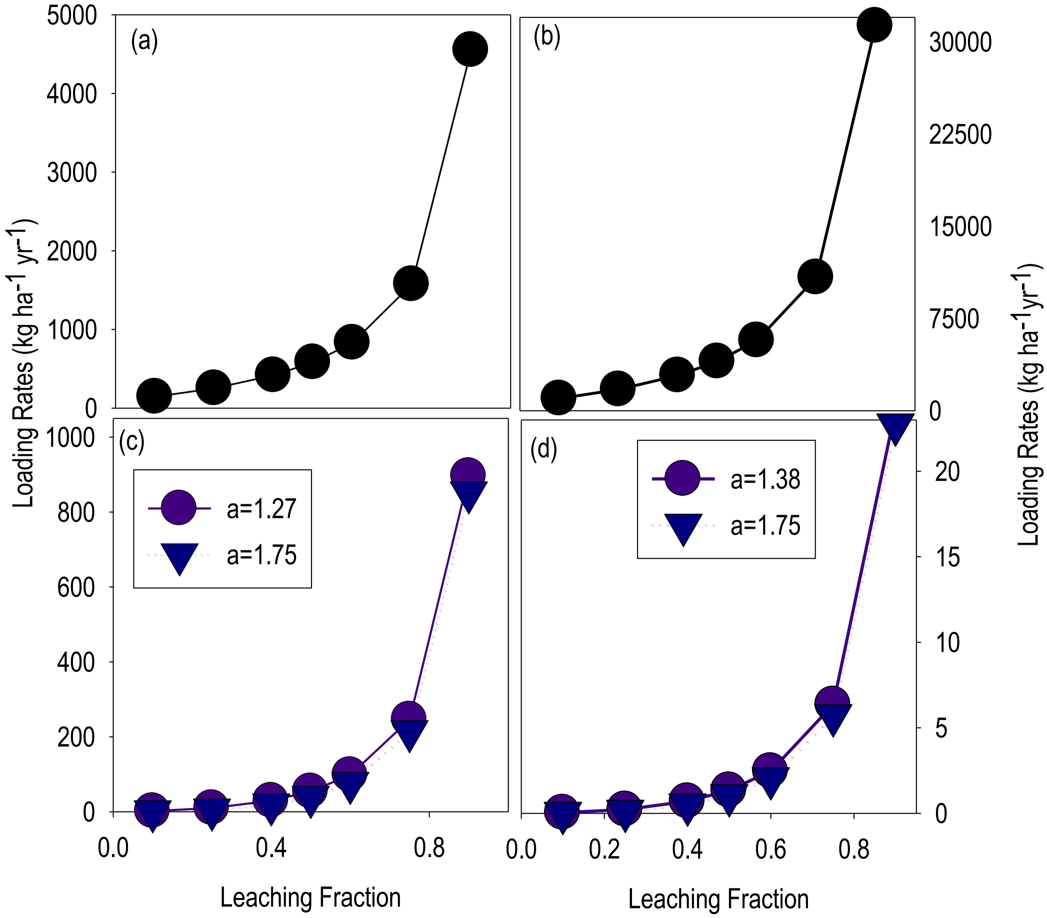

Figure 3.7. Steady state loading rates to groundwater calculated with results from the Schoups and Hopmans model steady state concentrations at 100 cm for (a) chloride, (b) TDS, (c) nitrate-N and (d) orthophosphate-P with various *a* values at the study site. Note the loading rates for orthophosphate-P were the same for both R values.

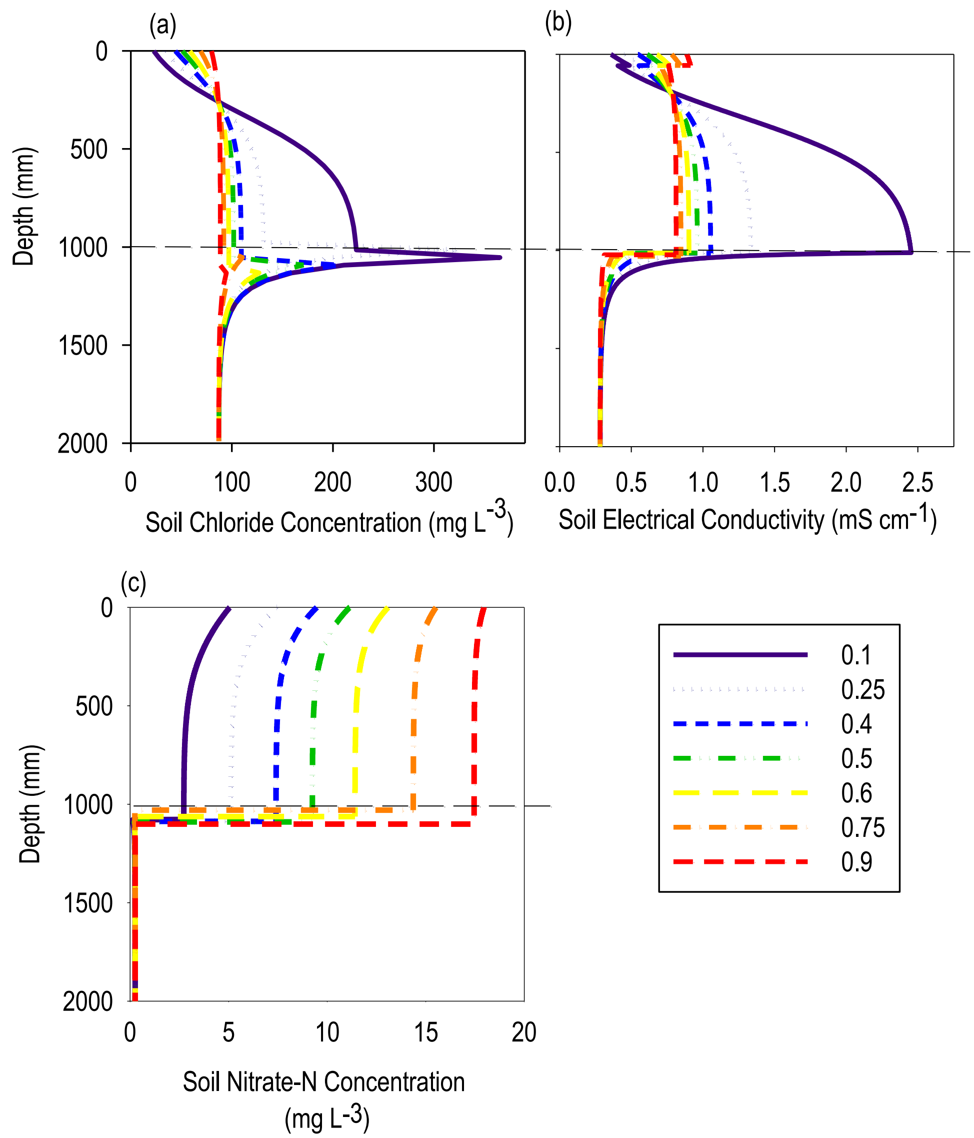

Figure 3.8. Soil solution (a) chloride concentrations, (b) EC profile and (c) nitrate-N concentrations for various leaching fractions determined by the Schoups and Hopmans (2002) models for the study site at Whitecourt, Alberta. The α used for nitrate was 1.27. The reference line indicates the depth to steady state, 1000 mm. Below this reference line is pre steady state conditions and the solute front.

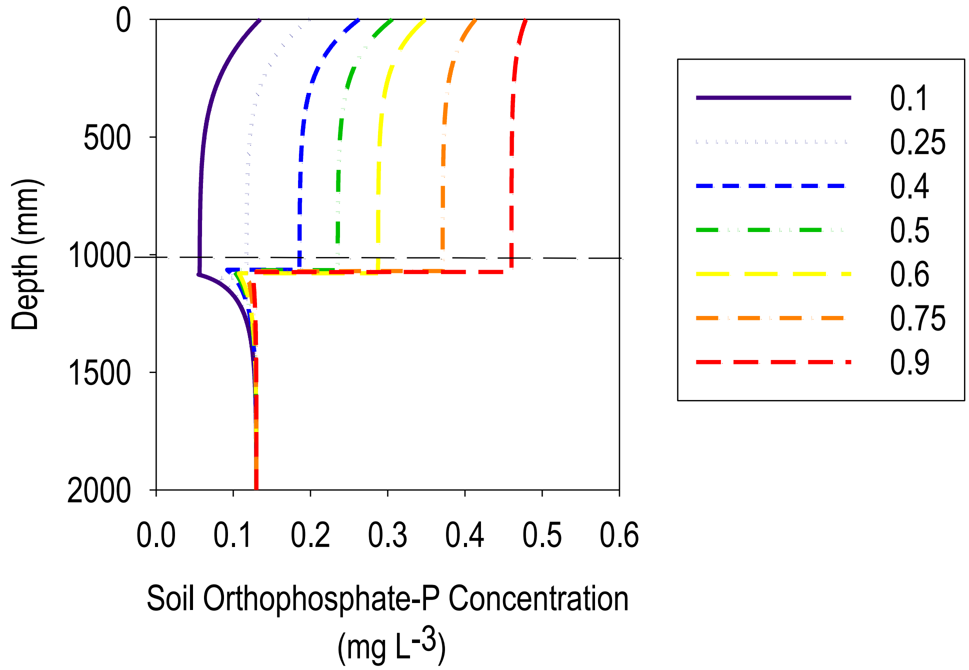

Figure 3.9. Soil solution orthophosphate-P concentration for various leaching fractions determined by the Schoups and Hopmans (2002) models for the study site at Whitecourt, Alberta, determined using Schoups model. The α used for orthophosphate was 1.38 and both R values resulted in the same concentration profiles. The reference line indicates the depth to steady state, 1000 mm. Below this reference line is pre steady state conditions and the solute front.

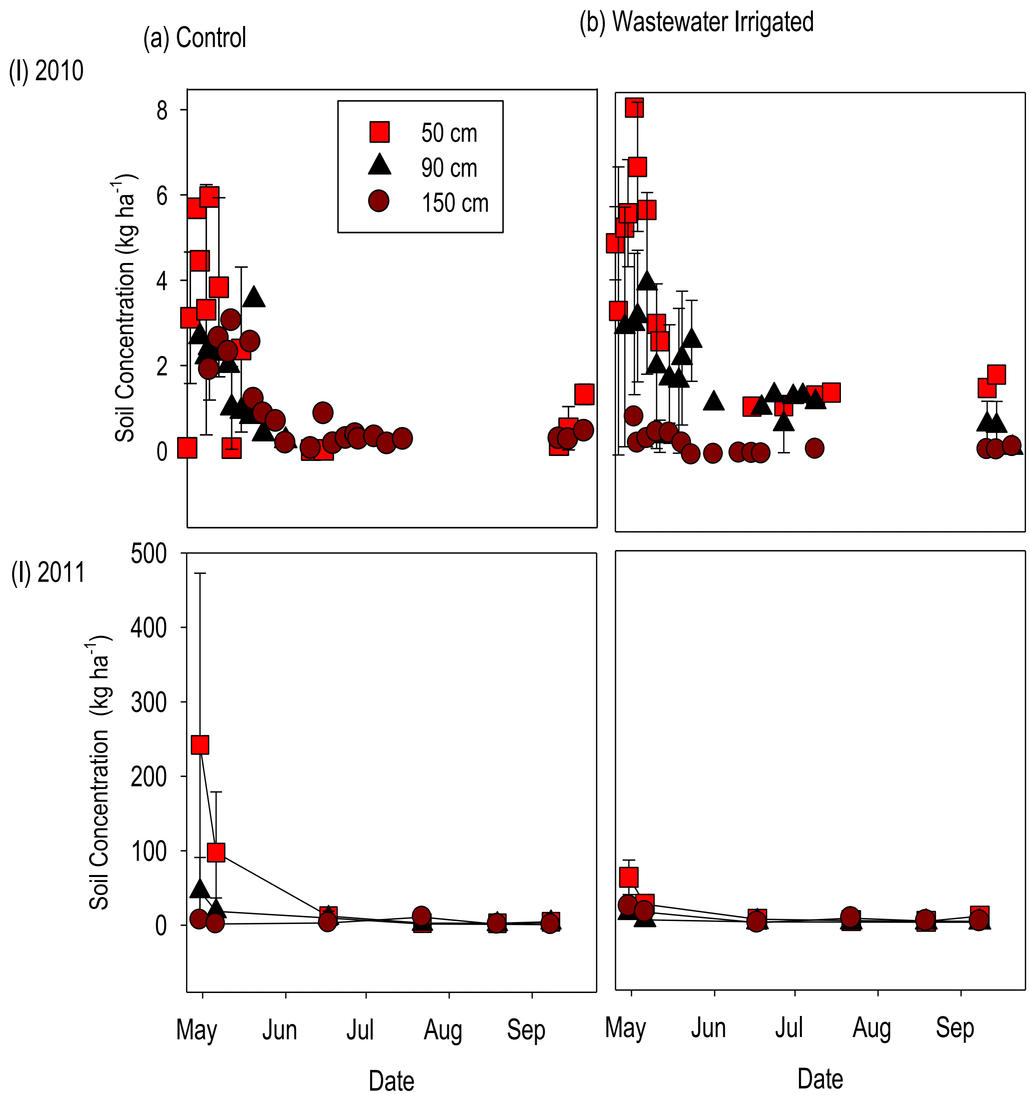

Figure 3.10. Soil concentration for DOC at various depths over the growing season in the (a) control and (b) wastewater irrigated soil for (I) 2010 and (II) 2011 at the study site in Whitecourt, Alberta. Error bars indicate standard error. Note change in y axis scale for (a) and (b) is different than (c) and (d).

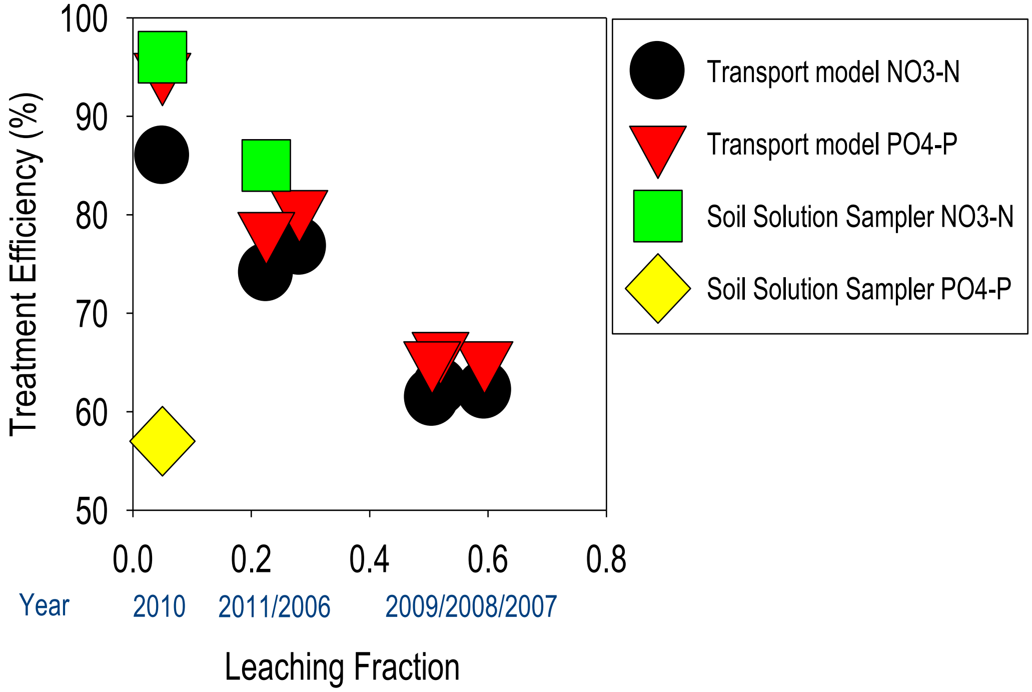

Figure 3.11. Estimated treatment efficiency (%) of wastewater by system based on Schoups and Hopmans (2002) and soil solution sampler gathered soil solution quality data at observed leaching fractions for nitrate-N (NO3-N) and orthophosphate-P (PO4-P) for the study site in Whitecourt, AB.

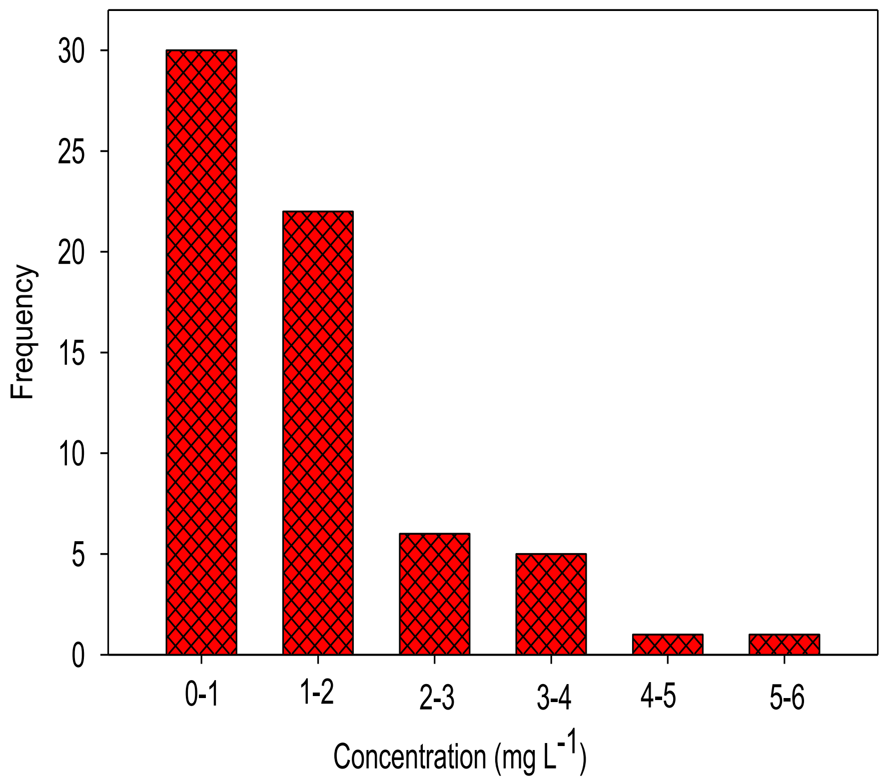

Figure 3.12. Histogram of nitrate-nitrogen concentrations in soil solution at 150 cm under the wastewater irrigated soil in 2010 and 2011 at the study site in Whitecourt, Alberta. Note that the soil solution nitrate-nitrogen concentrations are below the Alberta Tier 1 Groundwater Remediation Guideline for an Industrial Site with fine texture (13 mg L^{-1}) and potable water (10 mg L^{-1}) levels (Alberta Environment 2010).

3.8 REFERENCES

Addiscot, T. M., A. P. Whitmore and D.S. Powlson. 1991. Farming, fertilizers and the nitrate problem. CAB International. Wallingford, United Kingdom.

Agroclimate Atlas of Alberta. 2009. Climate of Alberta. Agriculture and Rural Development, Government of Alberta.http://wastewaterw1.agric.gov.ab.ca/$department/deptdocs.nsf/all/sag6299. Accessed January 24, 2011.

Alberta Environment. 2010. Alberta Tier 1 Soil and Groundwater Remediation Guidelines. Government of Alberta. Edmonton, AB.

AOAC. 2000. Method 972.43, Micro-chemical Determination of Carbon, Hydrogen, and Nitrogen, Automated Method. Official Methods of Analysis of AOAC International, 17th Edition. AOAC International, Arlington, VA.

American Public Health Association (APHA). 2005. Standard Methods for the Examination of Water and Wastewater, 21 st Edition. Edited by Eaton, A.D, L.S. Clesceri, E.W. Rice, A.E. Greenber and M.A.H. Franson (Editors).

Brady, N.C. 1990. The nature and properties of soil, 10th edition. Maxwell MacMillan, New York.

Brady, N.C. and R. R. Weil. 2004. Elements of the nature and properties of soils, Second Edition. Pearson, Prentice Hall. Upper Saddle River, New Jersey.

Brown, K.R. and R. Van den Driessche. 2005. Effects of nitrogen and phosphorus fertilization on the growth and nutrition of hybrid poplars on Vancouver Island. New Forests. 29 : 89 to 104.

Chang,C., T.G. Sommerfeldt, J.M. Carefoot and G.N. Schaalje. 1983. Relationship of electrical conductivity with total dissolved salts and cation concentrations of sulfate dominant soil extracts. Canadian Journal of Soil Science. 63 : 79-86.

Cheverry, C. 1983. L'extraction de al. solution du sol par le biasis des bougies poruses. Bull. Groupe Fr. Humidimetri Neutronique. 14:47-71.

Dalva, M. and T.R. Moore. 1991. Sources and sinks of dissolved organic carbon in a forested swamp catchment. Biogeochemistry. 15:1-19.

Danfors, B., S. Ledin and H. Rosenqvist.1998. Short rotation willow coppice grower's manual . Uppsala Sweden. Swedish Institute of Agricultural Engineering. 40 pp.

Dimitriou, I. and P. Aronsson. 2010. Landfill leachate treatment with willow and poplars- Efficiency and plant response. Waste Management. 30: 2137-2145.

Dimitriou, I., C. Baum, A. Baum, G. Busch, U. Schulz, J. Kohn, N. Lamersdorf, P. Leinweber, P. Aronsson, M. Weigh, G. Berndes and A. Bolte. 2009. The impact of Short Rotation Coppice (SRC) cultivation on the environment. 2009. Agriculture and Forestry Research. 59: 159-162.

Dosskey, M.G. and P.M. Bertsch. 1997. Transport of dissolved organic matter through a sandy forest soil. Soil Science Society of America Journal. 61:920-927.

Environment Canada. 2010. Canadian Climate Data and Information Archive. http://wastewaterw. climate. weatheroffice.ec.gc.ca. Accessed November 1, 2009.

Environmental Protection Agency (EPA). 1993. Determination of Nitrate-Nitrite Nitrogen by Automated Calorimetry, (SmartChem)- Method 353.2 Revision 2.0, in Methods for the Determination of Inorganic Substances in Environmental Samples, EPA/600/R-93/100.

Guggenberger, G., Kraiser, K. and W. Zech. 1998. Mobilization and immobilization of dissolved organic matter in forest soils. Z Pflanzenerahrung Und Bodenkunde. 161: 401-408.

Hasselgren,K. 1998. Use of municipal waste products in energy forestry: highlights from 15 years of experience. Biomass Bionenergy. 15:71 to 74.

Heatlh Canada. 2010. Health Canada Guidelines for Canadian Drinking Water Quality. Prepared

by the Federal-Provincial-Territorial Committee on Drinking Water of the Federal-Provincial-Territorial Committee on Health and the Environment. Government of Canada. Ottawa, Canada.
Hermann, M., J. Pust and R. Pott. 2005. Leachign of nitrate and ammonium in heathland and forest ecosystems in Northwest Germany under the influence of enhanved nitrogen deposition. Plant soil. 273:129-137.
Hillel, D. 2005. Salinity management. Encyclopedia of Soil Science, vol 3. Elsevier/Academic Press, 435-442.
Hytönen, J. and A. Sarrsalmi. 2009. Long term biomass production and nutrient uptake of birch, alder and willow plantations on cut away peatland. Biomass and Bionenergy. 22: 1197-1211.
Jiao, Y., J.K Whalen and W.H Hendershot. 2007. Orthophosphate sorption and release in a sandy loam soil as influenced by fertilizer sources. Soils Science Society of America Journal. 71:118-124.
Kalbitz,K., S. Solinger, J.H. Park, B. Michalzik and E. Matzner. 2000. Controls on the dynamics of dissolved organic matter in soils: a review. Soil Science. 165: 277-304.
Ledin, S. 1986. Management during the production period. In; Handbook for energy forestry. Uppsala, Sweden: Swedish University Agricultural Science. 19-20.
Lee,Y.W., M.F. Dahab and I. Bogardi. 1991. Nitrate risk management under uncertainty. Journal of Water Resources Planning and Management. 118(2): 151-165.
Litaor, M.I. 1988. Review of soil solution samplers. Water Resource Research. 24:727-733.
Livesley, S.J., M.A. Adams and P.F. Grierson. 2007. Soil nitrate and ammonium dynamics under a sewage effluent irrigated Eucalypt plantation. Journal of Environmental Quality. 36:1883-1894.
Longley, R.W. 1968. Climatic maps for Alberta. University of Alberta, Edmonton, Alberta. 8 pp.
Maulé,C., W.Helgason, S. McGinn and H. Cutforth. 2006. Estimation of standardized reference evapotranspiration on the Canadian Prairies using simple models with limited weather data. Canadian Biosystem Engineering. 48:1.1-1.11.
Moore, T.R., D. Pare and R. Boutin. 2008. Production of Dissolved organic carbon in Canadian Forest Soils. Ecosystems. 11:740-751.
Natural Regions Committee. 2006. Natrual Regions and Subregions of Alberta. Compiled by Downing, D.J. and W.W. Pettapiece. Government of Alberta. Pub. No T/852. 264 pp.
Normand, B.,S. Recous, G. Vacchaud, L. Kengni and B. Garino. 1997. Nitrogen-15 tracers combined with tension neutronic method to estimate to nitrogen balance of irrigated maize. Soil science society of America Journal. 61:1508-1518.
McGill, W.B., K.R. Cannon, J.A. Robertson and F.D. Cook.1986. Dynamics of soil microbial biomass and water-soluble organic C in Breton L after 50 years of cropping to two rotations, Can. J. Soil Scence. **66:** 1–19.
Michalzik, B., K. Kalbitz, J.H. Park, S. Solinger and E. Matzner. 2001. Fluxes and concentrations of dissolved organic carbon and nitrogen- a synthesis for temperature forests. Biogeochemistry. 52:173-205.
Moutonnet, P., J.F. Pagenel and J.C. Fardeau. 1993. Simulataneous field measurements of nitrate-nitrogen and matric pressure head. Soil Science Society of American Journal. 57: 1458-1462.
Olsen,S.R. and L.E. Sommers. 1982. Methods of Soil Analysis, Part 2- Chemical and Microbiological Properties,Second Edition. A,L, Page, R.H. Miller and D.R. Keeney, editors. American Society of Agronomy, Inc, SSSA,Inc.419-420.
Peichl, M., T.R. Moore, M.Altaf Arain, M. Dalva, D. Brodkey and J. McLaren. 2007. Concentrations and fluxes of dissolved organic carbon in an age sequence of white pine forests in southern Ontario, Canada. Biogeochemistry. 86:1-17.

Persson, G. 1995. Willow stand evapotranspiration simulated for Swedish soil. Agricultural Water Management. 28:271-293.

Pistocchi, C., W. Guidi, E.Piccioni and E. Bonari. 2009. Water requirements of poplar and willow vegetation filters grown in lysimeter under Mediterranean conditions: Results of the second rotation. Desalination. 246:137-146.

Schoups, G. and J.W. Hopmans. 2002. Model of vadose zone solute transport with root water and solute uptake. Symposium no. 54. 17 th WCSS, 14-21 August 2002. Paper 1094. 1-11.

Shi, J. and Q. Zuo. 2007. An inverse method to estimate the source sink term in the nitrate transport equation. Soil Science Society of America Journal. 71:26-34.

Silva, R.G., K.C. Cameron, H.J. Di, N.P. Smith and G.D. Buchan. 2000. Effect of macropore flow on the transport of surface applied cow urine through a soil profile. Australian Journal of Soil Research. 38: 13-23.

Sopper, W.E. 1970. Disposal of municipal wastewater through forest irrigation. Environmental Pollution. 1:263-284.

Tabatabai, W.M.A. and N. T. Basta.1991. Ion Chromatography. Soil Analysis. Modern Instrumental Techniques. Edited by K.A. Smith. Second Edition.

Tokarksy, O.1977. Hydrogeology of the Whitecourt area, Alberta. Alberta Research Council. Report 76-3.13 pp.

Volk,T.A., L.P. Abrahamson and E.H. White. 2001. Root dynamics in willow biomass crops. Biomass Power for Rural Development Technical Report. State University at New York. 30 pp.

Wolfe, A.H. and J.A. Patz. 2002. Reactive nitrogen and human health: Acute and Long term implications. Ambio. 31(2):120-125.

Wynnyk, A., J.D. Lindsay and Wm. Odyksky. 1969. Soil Survey of the Whitecourt and Barrhead Area. Soils Division, Research Council of Alberta. Alberta Soil Survey Report No. 27. 69 pp.

Zhu, X., Q. Zuo and J. Shi. 2010. Analyzing soil soluble phosphorus transport with root phosphorus uptake applying an inverse method. Agricultural Water Management. 97: 291-299.

4. CHAPTER 4: SYNTHESIS AND FUTURE RESEARCH

4.1 SUMMARY AND CONTRIBUTIONS

The general goal of the project was to determine the ability of a SRC crop to remove plant macronutrients, nitrogen and phosphorus, in municipal treated wastewater and the environmental implications of these systems on soil and groundwater. The specific objectives of this research were to quantify and qualify the drainage below the root zone of SRC plantations with wastewater irrigation and make recommendations to improve efficiency and management. Root zone salinity and leaching fraction (LF) requirements were also assessed. The major findings from this research were as follows:

I. Drainage varied greatly over the irrigation operating years of the study site mainly due technical problems limiting the amount of irrigation water applied. From running the Rose et al (1979) model, it was foreseen that irrigation years similar to 2010, with low LFs, would most likely have salt accumulation in the root zone.

II. Plant nutrient and solute of concern, nitrate, in the soil solution at 150 cm in 2010 and 2011 depths never exceeded the Alberta Tier 1 Groundwater Remediation Guideline for an Industrial Site with fine textured material (13 mg L^{-1}) and potable water (45 mg L^{-1}) guidelines (Alberta Environment 2010; Health Canada 2010). However, these years do not reflect high irrigation years so concentrations may have exceeded guidelines in the past. The Schoups and Hopmans (2002) model predicted soil solution nitrate-N concentration in exceedance of potable water guidelines at LFs above about 0.55.

III. There was not a great difference between control and wastewater irrigated soil solution concentration of orthophosphate and ammonium for both study years. However, solute loading rates were consistently higher in the wastewater irrigated soil than the control.

IV. Based on average application rates in the last five years, nitrogen and phosphorus additions to the SRC crop via wastewater irrigation are below recommended application rates by various authors (Dimitriou and Aronsson, 2010; Brown and Van den Driescche, 2005; Hytönen and Sarrsalmi, 2009; Ledin, 1986). Additional research and work into plant nutrient solute uptake and requirement is recommended.

V. From the Schoups and Hopmans (2002) model and historical soil samples, the electrical conductivity (EC) of the root zone is rising but not to detrimental values. From soil samples gathered in 2010, the root zone of the wastewater irrigated soil is classified as slightly saline, below 2.5 mS cm^{-1} (Brady 1990).

VI. The treatment efficiency (TE) is the amount of solute removed by the root zone as a fraction of the applied solute amount through wastewater. The TE was high (>50 %) for both the methods of determining soil solute concentrations for nitrate and orthophosphate. It was also observed that TE was higher at lower LFs, likely due to longer residence times when less water is applied.

4.2 RECOMMENDTIONS

Results from this study indicate that drainage is higher under irrigated soil than un-irrigated soil. It was also found that the SRC system was effective at removing many of the plant essential nutrients, like nitrate and phosphate, from the wastewater. However, the nutrients supplied by the wastewater may not be meeting the nutrients demands of the plants and additional fertilizer may be required. Drainage on site is variable, mainly depending on infiltrating water. If additional fertilizer is applied it recommended to use caution, with attention mainly on timing in relation to precipitation events.

It is recommended that the SRC with wastewater irrigation system at Whitecourt to slightly change its present regime of a growing season typically between May and August with the irrigation programmed to field capacity at 20 cm soil depth**.** For the majority of the irrigation years at the study site, except 2010 due to technical errors, drainage and LFs have been high (>0.5). However, field samples of the soil solution from 2006 to 2009 are not available to comment on the actual quality of the soil water. From the Schoups and Hopmans (2002) model, at higher LF (>0.6), the nitrate soil solution concentration exceeded the Alberta Tier 1 Guidelines for Groundwater Remediation and potable water guideline (Alberta Environment 2010; Health Canada 2010). It is recommended to use a LF between 0.25 and 0.55 to minimize groundwater contamination while supplying SRC plants with adequate macronutrients like nitrate and utilizing the large wastewater supply. A value less than 0.1 could possibly lead to salt accumulation and a change in the soil salinity category rating under the Alberta Tier 1 Salt Remediation Guidelines. It is recommended to change the irrigation system to be triggered by a soil moisture content slightly less than field capacity at 20 cm. Assuming that during years with no technical errors the LF would be higher (similar to that of 2007 to 2009) by slightly lowering the soil moisture content at which irrigation is triggered, the LF will be lowered. Also, care must be taken when irrigating during regeneration years due to the small plant size and low evapotranspiration. Long term soil salinity monitoring of the root zone and soil phosphorus testing is also recommended. Implementation of findings from this research into similar programs in Alberta and Canada will be important for the future of SRC with wastewater irrigation systems and global greenhouse gas reductions.

4.2.2 Future Research

Knowledge of the impacts of wastewater irrigation of SRC systems on the environment is crucial for creating an efficient and environmentally sound energy form. This research has contributed to the understanding of SRC plantations with wastewater irrigation systems in western Canada. The study addressed the potential for soil salinity and groundwater contamination from wastewater irrigation of SRC crops in Alberta but more information is needed on many other possible environmental threats. To build on this research and its findings more knowledge on SRC with wastewater systems in Canada is essential. Areas for future research may include the following:

I. fate and transport of other solutes found in wastewater like heavy metals, trace elements or plant micronutrients like sulphur or potassium;

II. monitoring of drainage and soil solution quality during the establishment year of a SRC with wastewater irrigation system;

III. plant response to varying nutrient applications through wastewater irrigation and assessment of plant nutrient requirements over the life cycle of a SRC;

IV. fate and transport of solutes found in wastewater on varying soil types and climates in Canada; and

V. detailed investigation of the potential for salinity and sodicity issues in the short and long term.

4.3 REFERENCES

Alberta Environment. 2010. Alberta Tier 1 Soil and Groundwater Remediation Guidelines. Government of Alberta. Edmonton, AB.

Brown, K.R. and R. Van den Driessche. 2005. Effects of nitrogen and phosphorus fertilization on the growth and nutrition of hybrid poplars on Vancouver Island. New Forests. 29 : 89 to 104.

Heatlh Canada. 2010. Health Canada Guidelines for Canadian Drinking Water Quality. Prepared by the Federal-Provincial-Territorial Committee on Drinking Water of the Federal-Provincial-Territorial Committee on Health and the Environment. Government of Canada. Ottawa, Canada.

Ledin, S. 1986. Management during the production period. In; Handbook for energy forestry. Uppsala, Sweden: Swedish University Agricultural Science. 19-20.

Rose, C.W., P.W.A. Dayananda, D.R. Nielsen and J.M. Biggar.1979. Long Term Solute Dynamics and Hydrology in Irrigated Slowly Permeable Soils. Irrigation Science. 1: 77-87.

Schoups, G. and J.W. Hopmans. 2002. Model of vadose zone solute transport with root water and solute uptake. Symposium no. 54. 17 th WCSS, 14-21 August 2002. Paper 1094. 1-11.

APPENDICES

APPENDIX A: Sample Calculations

A.1 Solute concentration in infiltrating water for Schoups model for chloride

C_{q0}= C_I*I + C_{Rain}*$Rain$ +C_{Snow}*$Snow$/ (I +$Rain$ +$Snow$)

Where
C_{q0} =solute concentration in infiltrating water [M L^{-3}]
C_I=solute concentration in irrigation water (wastewater) [M L^{-3}]
I=amount of irrigation water applied [L]
$Rain$=amount of rain fall [L]
$Snow$=amount of snowfall as 0.5* SWE (snow water equivalent) [L]

=((0.087 ug mm^{-3}*164 mm)+(4.1 x 10^{-5} ug mm^{-3} * 391 mm)+(8.86 x 10^{-7} ug mm^{-3}+79.97 mm))/(164.98 mm+79.95 mm+391.58mm)=0.02255 ug mm^{-3}

A.2 Converting mass concentration to solution concentration

Assume bulk density is 1.5 g cm^{-3} and moisture content is 0.22. Conversion form mg L-1 to ug mm^{-3}.

mg kg^{-1} soil *1.5 g cm^{-3} * (1 kg/1000 g)*(1 cm^3 soil/0.22 cm^3 water)*(1000 ug/mg) *(1 cm^3/1000mm^3) = ug mm^{-3}

A.3 Determining drainage from Schoups model.

D= q_0 * q_z

Where
D=Drainage [L]
q_0=infiltrating water [L^2 T^{-1}]
q_z= flux as a specific depth, *z* [L^2 T^{-1}]

A.4 Treatment efficiency of WW by SRC system

$$TE = \frac{LR_{ww} - LR_{dw}}{LR_{ww}} \times 100$$

Where,
TE= treatment efficiency (%)
LR_{ww} = wastewater loading rate [M L^{-2}]
LR_{dw}= drainage loading rate [M L^{-2}]

A.5 Determining groundwater loading rates for Schoups and Hopmans (2002) model

LR_{dw}= D *C_z

Where
LR_{dw}= = Loading rate [M L^{-2}]
D= Drainage at 1000 mm from model[L]
C_z= soil concentration at depth *z*, 1000 mm, from model [M L^{-3}]

A.6 Determining wastewater loading rates for Schoups and Hopmans (2002) model

LR_{ww} = I *C_I
where

$I =$ irrigation water applied for a specific leaching fraction
C_I =concentration of solute in irrigation water

A.7 Determining wastewater loading rates for soil solution samplers
LR_{ww} = $= \mathrm{I} * \mathrm{C_I}$

where
$I =$ irrigation water applied for a specific year [L]
C_I =concentration of solute in irrigation water {m L^{-3}]

A.8 Determining groundwater loading rates for soil solution samplers
LR_{ww} = $D * C_{ss}$

where
LR_{gw} = loading rate for soil solution sampler [M $L^{-2}T^{-1}$]
D = drainage below the root zone [L]
C_{ss} = soil solution concentration [M L^{-3}]

A.9 Soil mass concentration calculation
C_{sm} = $C_{ss} * \theta_v * L_{cup}$
where
C_{sm} = soil mass concentration [M L^{-2}]
C_{ss} = soil solution concentration [M L^{-3}]
θ_v = volumetric moisture content
L_{cup} = Length of ceramic cup used for sampling soil solution (5 cm for this study) [L]

A.10 Snow density determination

$$D_{snow} = \frac{M_{snow}}{V_{snow}}$$

Where
D_{snow} = density of snow [M L^{-3}]
M_{snow} = mass of snow [M]
V_{snow} = volume of snow [L^3] =$\pi r^2 h$
Where
h= depth of snow sample [L]
r = radios of snow sample

A.11 Snow water equivalent (SWE)

$$SWE = h\frac{D_{snow}}{D_{water}}$$

Where
D_{water}= density of water [M L^{-3}]
h= snow height [L]

APPENDIX B. Derivation of the Convective Transport Equation Without Dispersion or Diffusion from Schoups and Hopmans (2002)

Recall, the continuity equation for one dimensional vertical flow through the root zone,

$$\frac{\delta\theta}{\delta t} + \frac{\delta q}{\delta z} + r_w \qquad \text{[B1]}$$

Where
$\theta(z,t)$ is volumetric water content
$q(z,t)$ is soil water flux (L/T, positive downwards)
$r_w(z,t)$ is a sink term for root water uptake and evaporation (1/T)
t is time (T)
z is vertical component (L, positive downwards)

The continuity equation for one dimensional transport of a solute undergoing root uptake and equilibrium sorption and neglecting dispersion and diffusion is:

$$\frac{\delta}{\delta t}(\theta c + p_b c_a) + \frac{\delta qc}{\delta z} + r_s = 0 \qquad \text{[B2]}$$

Where
c is dissolved solute concentration (M L^{-3}); mass per unit volume of soil solution
c_a is adsorbed solute concentration (M M^{-1}), mass of sorbent per mass of dry soil
p_b is dry bulk density (M L^{-3})
r_s is root solute uptake [M $L^{-3}T^{-1}$]

To combine equation [B1] and [B2], the derivatives from equation [B2] are expanded:

$$c\frac{\partial\theta}{\partial t} + \theta\frac{\partial c}{\partial t} + p_b\frac{\partial c_a}{\partial t} + c_a\frac{\partial p_b}{\partial z} + c\frac{\partial q}{\partial z} + q\frac{\partial c}{\partial z} + r_s = 0 \qquad \text{[B3]}$$

Assuming p_b (bulk density) to be constant with time and inserting equation [B1] in [B3] will give the following:

$$-cr_w + \theta\frac{\partial c}{\partial t} + p_b\frac{\partial c_a}{\partial t} + q\frac{\partial c}{\partial z} + r_s = 0 \qquad \text{[B4]}$$

And letting,

$$c_a = F(c) \text{ and } \frac{\partial c_a}{\partial c} = F' \qquad \text{[B5]}$$

Where,
$F(c)$ is the sorption isotherm with slope F'

Combining [B4] and [B5] yields,

$$-cr_w + \theta\frac{\partial c}{\partial t} + p_b F'\frac{\partial c}{\partial t} + q\frac{\partial c}{\partial z} + r_s = 0 \qquad \text{[B6]}$$

Rearranging [B6] gives,

$$(\theta + p_b F')\frac{\partial c}{\partial t} + q\frac{\partial c}{\partial z} = cr_w - r_s \qquad \text{[B7]}$$

Defining the retardation factor (R) as

$$R = 1 + \frac{p_b F'}{\theta} \qquad \text{[B8]}$$

The convective transport equation is obtained.

$$\frac{\partial c}{\partial t} + v_R \frac{\partial c}{\partial z} = \frac{c r_w}{R\theta} \qquad \text{[B9]}$$

Where

$v_R(z,t) = \frac{v}{R} = \frac{q}{R\theta}$ is the retarded solute velocity (L T^{-1})

$v(z,t)$ = is the pore water velocity (L T^{-1})