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GRADE / DEGREE

Department of Earth Sciences
FACULTÉ, ÉCOLE, DÉPARTEMENT / FACULTY, SCHOOL, DEPARTMENT

Evaluation of Agriculture Beneficial Management Practices:
[2]Cattle Access Restriction to Surface Waters
[3]Subsurface Tile Drainage Management

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EVALUATION OF AGRICULTURAL BENEFICIAL MANAGEMENT PRACTICES:

^{2}CATTLE ACCESS RESTRICTION TO SURFACE WATERS

^{3}SUBSURFACE TILE DRAINAGE MANAGEMENT

Emilia Craiovan

Thesis submitted to the
Faculty of Graduate & Postdoctoral Studies
University of Ottawa
in partial fulfillment of the requirements for the
M.Sc. degree in the Earth Sciences

Ottawa-Carleton Geoscience Centre
and
University of Ottawa
Ottawa, Canada

Thèse soumise à
Faculté des études supérieures et postdoctorales
Université d'Ottawa
en vue de l'obtention de la maîtrise ès sciences en
Science de la Terre

Centre Géoscientifique d'Ottawa-Carleton
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ABSTRACT

The aim of this study was to investigate two potential beneficial management practices that will reduce the impact of fecal and fertilizers contamination on the environment. The study was conducted in Eastern Ontario on typical small scale agricultural practices.

The first part evaluates the impact of a conventional pasture system and an excluding pasture system on surface water quality. The three year study compared water quality endpoints such as: indicator bacteria, pathogens, parasites and nutrients between treatments. Microbial source tracking indicated that livestock was the main source of fecal contamination in the stream. Greater bacteria and nutrient loads were observed in the unrestricted pasture system than the excluding pasture. Moreover, parasite and indicator bacteria concentrations increased after cattle introduction in both systems.

The second part compares nitrogen mass balance between managed and conventional subsurface tile drainage. Over two years, nitrogen loads in groundwater, in tile flow, in plants, in soil and denitrification were compared between treatments. Nitrogen was mostly removed from managed tile drainage fields through plant uptake. Compared to nitrogen plant uptake, denitrification, the second greatest nitrogen removing process was 10 times smaller. The denitrification was greater in the unmanaged tile drainage fields. Generally, managed subsurface tile drainage reduces nitrogen mass loads to surface waters and increases nitrogen uptake by plant, which resulted in greater yields.

RÉSUMÉ

L'objectif de cette étude était d'investiguer deux pratiques de gestions bénéfiques pouvant potentiellement réduire l'impact des fertilisants et des matières fécales contaminant l'environnement. L'étude a été conduite sur des terres agricoles typiques de l'Est Ontarien.

La première partie de cette thèse évalue l'impact du pâturage conventionnel et du pâturage exclusif sur les eaux de surface. Pendant trois ans, des analyses ont été conduites afin d'évaluer l'impact du pâturage exclusif sur la qualité du cours d'eau. Les traitements ont été comparés sur le plan des bactéries indicatrices, des parasites, des pathogènes et des nutriments. Le dépistage des sources de pollution microbienne a indiqué que le bétail été la source principale de contamination fécale dans les cours d'eau. L'apport de bactéries et de nutriments était plus grand dans la partie du pâturage conventionnel que dans celle du pâturage exclusif. Par contre, la quantité de parasites et de bactéries a augmenté dans les deux pâturages après l'introduction du bétail.

La deuxième partie de l'étude compare le bilan d'azote entre le drainage agricole conventionnel et le drainage sous gestion. Pendant deux ans, le bilan de masse d'azote dans l'eau souterraine, dans les plantes, dans le sol et dans la denitrification a été comparé entre les traitements. L'azote a été assimilé en grande partie par les plantes. La denitrification, moins important que la consommation par les plantes, était le second processus de perte d'azote et cela dans le système conventionnel. Généralement, le système de drainage sous gestion a réduit la quantité d'azote entrant dans les cours d'eau et a accrue la rétention d'azote dans les plantes, résultant en une augmentation du rendement.

ACKNOWLEDGMENT

First of all, I would like to thank David Lapen and Mark Sunohara for all their financial and equipment support. I also would like to thank them and Michel J.L. Robin for believing and giving me the opportunity of working on this project as well as for their patience and encouragement. I also want to thank; Natalie Gottschall, Patricia Russell, Graham Wilkes and Claudia Fall for their great help with the data analysis, lab work and their friendship; Mark Edwards for the technical support; Dawn Paszkowski, Nick Lalumiere, Colleen Mortimer, Rob Lindsay, Karina Lamay, Victoria Mungall, Luke Andersson, Harun Cicek, Mira Krotneva, Jessica Weirdsma and Marielle Fortin-Mccuaig for their help with field work. Finally, but not least, I would like to thank my parents and my sister for their support and encouragement.

The funding for this project was provided by Agriculture and Agri-Food Canada, Ontario Ministry of Agriculture and Food, Ontario Ministry of the Environment, and Health Canada.

STATEMENT OF ORIGINAL CONTRIBUTION

This thesis was prepared and written by Emilia Craiovan. Emilia Craiovan and David Lapen's crew from Agriculture and Agri-Food Canada gathered all data essential to this study. Emilia Craiovan installed field instruments, collected water and soil samples and analysed these samples for: nitrate, ammonium and indicator bacteria. Due to the number of samples, limit resources and knowledge some of the analyses were done at specialised laboratories as described in the methods section of this thesis. A list of people and their affiliation information is listed in Appendix D. David Lapen supervised and provided guarantors for the project. Mark Sunohara was the project manager and responsible for all field operations. Emilia Craiovan is the primary author on both coauthored manuscripts presented in this thesis. The coauthors provided funding for the project and/or technical support and/or guidance.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

1.1 Problem

Canada, with the largest freshwater reservoirs, had 3100 boiling water advisory days across its municipalities in 1998, an increase of 24% since 1992 (Statistics Canada, 2003). Tragedies such as Walkerton, ON, and North Battleford, SK, have raised concerns about water quality in and around agricultural areas (O'Connor, 2002). Research studies have indicated that suspended sediments, phosphorus, nitrogen and fecal contamination are the main pollutants of surface and ground water in agricultural land-use (Kuhnle et al., 2000; Vanni et al., 2001; Rock and Mayer, 2006; Royer et al., 2006; Tarkalson et al., 2006). Beneficial management practices such as: buffer strips implementation around watercourses, off-stream cattle watering, and nutrients, soil and water management would reduce the negative impact of agriculture on the environment. These practices appeal less to farmers when they have to give up parts of their land and invest their revenue in order to protect the environment. This study has looked at two agricultural beneficial management practices to possibly reduce land loss and cost of their implementation while protecting the environment.

First, the unprotected streams and riparian areas in pasture systems can be physically impacted by cattle traffic as well as polluted directly or indirectly by microorganisms and nutrients from fecal contamination (Hooda et al., 2000; Sovell et al., 2000; Parkin et al., 2003; Agouridis et al., 2004; Parkin, 2004). Studies have shown that restricting or excluding cattle access to surface waters will improve surface water quality in pasture areas. Fisher and Endale (1999) found that ponds situated between the pasture area and surface water bodies will reduce *Escherichia coli* and enterococcus bacteria runoff. Providing an off-stream water source and a non-riparian shade have shown to improve surface water quality (Byers et al., 2005). Mayer et al. (2007) noted that a 50 meter buffer will be preferable to remove nitrate runoff. Line's 2003 study suggested that riparian buffer needs to be greater then 10 meters in order to reduce fecal coliforms and nutrient loads to adjacent surface waters. These suggestions are too demanding on farmers as they will result in great land loss and fencing costs; therefore, for small scale operations such as those in eastern Ontario, a smaller riparian buffer would be more appealing and would probably permit to reduce the contamination. This study hopes to demonstrate that a smaller riparian buffer (3 meters) would reduce fecal and nutrient loads to streams.

Indicator bacteria such as *Escherichia coli* and fecal coliforms, found in animal feces, are used as freshwater quality indicators due to their relatively simple analysis and their relation to other pathogens; i.e. high concentrations indicate the potential presence of pathogens, for example *Salmonella* and *Shigella* (USEPA, 2003 and Health and Welfare Canada, 1992). The pathogenic bacteria cause illness to their host. Usually pathogens are specific and do not survive for a long time outside their host. They are greatly found in contaminated soils, therefore infections from food and water ingestion can occur. The infection depends upon the ability of the host to defend itself (its immunity) and prevent injury (Health Canada, 2008). The most common known pathogens are *E. coli*, *Listeria*, *Salmonella*, C*ryptosporidium*, *Giardia* and *enterococcus*. Each one of them can cause serious illness and in some cases death, i.e. Walkerton, Ontario and recent *Listeria* poisoning events. In Agricultural areas the contamination by these pathogens can be important and this is the reason why this thesis is studying the impact of pasture on surface water quality.

Second, the subsurface tile drainage management on the nitrogen mass balance is another aspect that needs to be examined in order to reduce pollution due to agricultural practices. Sadly, most of the fertilizers, pesticides and manure applied on agricultural fields can end up in surface waters due to leaching and tile flow. Subsurface tile drainage limits the residence time of water in the soil and therefore contributes to the contamination of adjacent surface waters with nutrients and pesticides (Fausey et al., 1995; David et al., 1997; Kladivko et al., 2001; Tomer et al., 2003; Heppell and Chapman, 2006). Studies have shown that managed tile drainage reduces nitrogen tile flow (Drury et al., 1996; Ng et al., 2002; Wesström and Messing, 2007). Most of the studies conducted on subsurface tile drainage management looked at a specific component of the system such as; yields, nutrient loads lost through tile flow, nitrogen lost through denitrification, and nutrient conservation in soil and groundwater. This study has looked at the whole system regarding nitrogen mass balance, which will help indicate where everything goes and if this practice is sustainable.

Nitrogen is essential for life on Earth. The atmosphere is composed of approximately 78% of elemental gaseous nitrogen, which needs to be transformed into forms usable by living organisms. There are four processes in the nitrogen cycle: nitrogen fixation, decay, nitrification and denitrification (Brady, N.C., 1990). Nitrogen fixation is mostly accomplished by micro organisms present in soil (n*itrosomonas* and *nitrobacter*) or in symbiosis (*Rhizobia*) with plants. The elemental nitrogen is transformed into ammonia, which is immediately incorporated into proteins and organic nitrogen. The autotrophic n*itrosomonas* and *nitrobacter* bacteria transform some of the ammonia into nitrite and nitrate, respectively, and therefore make it available for plant uptake. In managed tile drainage these forms of nitrogen can be easily taken by plant roots when water is present. On the other hand, in the same wet conditions denitrification can be very important as the soil conditions turn to be anaerobic and that anaerobic bacteria can use nitrate as an alternative to oxygen for there electron acceptor in their respiration. In this case it is believed that managed tile drainage risks of increasing greenhouse gases releases from agricultural fields. Year round tile drainage management will permit to keep most if not all the fertilizer and organic matter on the fields as leaching through tiles is reduced or not existent (Drury et al., 1996; Ng et al., 2002; Wesström and Messing, 2007).

1.2 Study Purpose, Objectives and Thesis Organisation

The purpose of this study is to expand our knowledge regarding surface and ground water contamination in agricultural settings in order to reduce the impact on the environment.

The primary objectives of this thesis are to:

i. *Quantify water quality improvements associated with excluding pasturing cattle from streams*

ii. *Characterize nitrogen mass balance in managed subsurface tile drainage*

This thesis is written as two review-ready articles that are linked by a general introduction that gives the rationale for the study, an overview of the methodology, and the link between the two parts; and by a concluding chapter that overarches the two parts.

1.3 Conceptual Framework and Overview of Approach

This thesis is based on two studies. The first consists of evaluating the impact of restricting cattle access to surface waters on fecal contamination and nutrient loads to surface waters. The second part concerns the nitrogen mass balance on managed subsurface tile drained agricultural fields.

The first study consisted of determining the fecal contamination sources and comparing bacteria and nutrient loads between protected and unprotected streams in pasture areas. The protected part of the stream, where a 3 meter riparian buffer was fenced around the stream over a 400 meters stretch, was adjacent and upstream from the unprotected area (conventional pasture). *Escherichia coli,* enterococcus, total coliforms, *Cryptosporidium, Giardia, Clostridium Perfringens,* nitrate-nitrogen (NO_3-N), ammonium-nitrogen (NH_4-N) and ortho-phosphate concentrations were measured on samples collected in the stream from June to November in 2006 and 2007. Stream velocity and depth were also collected during this period and were used to calculate the loads. Samples were collected from four sites; two in each treatment situated at the beginning and at the end of each treatment.

The second study consisted of measuring the effect of managed subsurface tile drainage on nitrogen mass balance. Nitrogen loads were determined in tile water, groundwater, greenhouse gases released, plant uptake, soil and soil water. Measurements were made from June to November in 2006 and 2007 on privately owned managed tile drained agricultural fields situated near St-Albert, Eastern Ontario (45°16'48"N, 75°9'36"W).

As mentioned earlier, the final chapter of the thesis will summarise the findings and tie the two studies together.

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CHAPTER 2

QUANTIFYING THE ENVIRONMENTAL EFFECTS OF RESTRICTING CATTLE ACCESS TO SURFACE WATERS WITHIN A WATERSHED

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2.1 Introduction

Unprotected streams and riparian areas in pasture systems can be physically impacted by cattle traffic as well as polluted directly or indirectly by microorganisms and nutrients from fecal material (Hooda et al., 2000; Sovell et al., 2000; Parkyn et al., 2003; Agouridis et al., 2004; Parkyn, 2004). Line et al. (2000) found that fecal material released by cattle directly in streams increased phosphorus availability. Lardner et al. (2005) found that cattle defecation in streams increased pathogen numbers, which can pose human health risks as well as decrease cattle production. Wood and Armitage (1997) found that direct cattle access to surface water bodies can also destroy streambed morphology, thereby disrupting ecological function and habitats for aquatic biota.

Pathogens and nutrients are the greatest concerns when it comes to water quality protection (Health and Welfare Canada, 1992); thus the Canadian Water Quality Guidelines (CWQG) include concentration limits for these critical parameters to protect human and aquatic life. For instance, the concentrations of *Escherichia coli* and fecal coliforms, in recreational freshwaters, should not exceed 200 colonies per 100 ml and in drinking water they should not be present at all (EPA, 2008; Health and Welfare Canada, 1992). *Escherichia coli* and fecal coliforms, found in animal feces, are used as freshwater quality indicators due to their relatively simple analysis and their relation to other pathogens; high concentrations indicate the potential presence of other pathogens, for example *Salmonella* and *Shigella* (USEPA, 2003; Health and Welfare Canada, 1992). *Cryptosporidium* spp., *Giardia* and *Listeria* could also be present during this high concentrations and their presence can be used to determine the contamination source (Ruecker et al., 2007; Lyautey et al., 2007). Although, there is no numerical limit for nutrients in recreational water, in drinking water, the concentration of nitrate-nitrogen should not exceed $10 \mathrm{mgL}^{-1}$ and phosphorus-32 should not exceed $50\ \mathrm{BqL}^{-1}$ (Health and Welfare Canada, 1992). Ammonium-ammonia (NH_4-NH_3) has no limits as it has not been proven to cause any human health problems (Health and Welfare Canada, 1992). No pathogen limits have been set for aquatic biota protection, but the concentration of nitrate (NO_3) cannot exceed 13mgL^{-1}, nitrite (NO_2) cannot exceed 60µgL^{-1}, ammonia-nitrogen (NH_3-N) cannot exceed 15.2µgL^{-1}, while phosphorus concentrations can vary from <4 to >100µgL^{-1} depending on the natural phosphorus levels in the freshwater system (Environment Canada, 2007). Because remediation is always more costly than prevention, water should be protected at the source.

Nevertheless, regulations on cattle access to surface water bodies are limited and not consistent among jurisdictions. For instance, in New Brunswick (NB), Canada, a 75 meter setback from the banks of watercourses has been implemented in watersheds from where surface water is taken for human consumption; although, ditches and ephemeral streams are not protected under these regulations (NB Watershed Protection, 1990). In Ontario, it is recommended to maintain a minimum 3 meter riparian buffer around the streams to improve water quality (OMAFRA, 2006). In Alberta, it is suggested to protect the riparian zone by rotating grazing zones and installing off stream watering systems (Zylstra, 2005). Because the cost of installing a fence and the loss of land are too important to the producer, the riparian zone protection is then less attractive. This is why the government of Manitoba encourages establishing a riparian buffer zone greater than 10 meters by covering 50% of the work costs for up to 20000 dollars (Manitoba Agriculture, Food and Rural Initiative, 2007). Similar costsharing is offered to farmers from the province of Ontario who want to protect water sources (Ontario Farm Environmental Coalition, 2008).

However, while it seems prudent to keep cattle away from surface water courses for purposes of improving water quality land loss and fencing costs can be too costly for small livestock operations. Some studies such as Mayer et al. (2007) noted that a 50 meter buffer will be preferable to remove nitrate runoff. Line (2003) noticed a reduction as high as 65.9% and 57.0% in fecal coliforms and enterococcus, respectively, after fencing a 335 meter long and 10 to 16 meter wide riparian corridor along a small North Carolina stream. Although, the above studies indicate reductions of contamination in surface water when large riparian zones are present, the question remains as to how beneficial smaller (<5m) riparian zones are?

For small pasturing operations that are common in eastern Canada, the purpose of this study is to quantify water quality improvement associated with excluding pasturing cattle from streams. The specific objectives of this study were to: i) determine the source of *E. coli* and *Cryptosporidium* in areas where the stream is protected from cattle, compared to areas that are unprotected; and ii) compare enteric bacteria pathogens and nutrient loads between the protected and unprotected stream sections.

2.2 Materials and Methods

2.2.1 Study Site Description

The study area is located near St.-Albert, Ontario, in the South Nation watershed, within the micro-watershed of Philippe Blanchard Municipal Drain (45°16'48", -75°9'36"). The drain is an ephemeral stream used to collect waters from agricultural fields where liquid dairy manure is usually spread in early May and mid-November. The two adjacent and privately owned pastures were situated downstream from these agricultural fields.

The restricted cattle access area (RCA) is located upstream from the unrestricted cattle access area (URCA) (Figure 2.1) and this represents the above-and-below (USDA, 1994) watershed statistical design. Cattle density was maintained at 1head/acre in both pastures; the URCA area is 9.86 ha and RCA is 5.15 ha. The average slope of the stream, between sampling sites 18 and 15 (Figure 2.1) is 0.1%. The average slope perpendicular to the stream is 11.7% on the north side and 9.4% on the south side in both treatments.

2.2.1.1 Restricted Cattle Access (RCA)

In 2006, a three meter wide riparian buffer was fenced for a 400 meter stretch along both sides of Blanchard stream, from site 18 to site 22 (Figure 2.1-yellow). Typically, seven adult beef cows and four calves, mostly Hereford breed, were introduced in June and taken out in November for each year of the study. To provide cattle with access on both sides of the stream, a bridge was constructed. Water was provided by two nose pumps acquiring water from the stream and two stock tanks connected to a well. Hay feeders were installed to draw cattle on to the south side of the stream to ensure complete grazing coverage as it happens in free pasture areas. Tree-shaded areas are present near site 22 on the north side and near site 18 on the south side of the stream (Figure 2.1).

On the south side, approximately 100 meters upstream from site 22, two family houses with septic tanks and 3 hogs (in 2006) were present. A hobby farm growing: sheep, horses, hogs and goats, was located approximately 200 meters upstream from site 18 on the south side of the road.

2.2.1.2 Unrestricted Cattle Access (URCA)

The unrestricted pasture is grazed by young dairy cows, Holstein breed heifers, and as in the controlled pasture, cattle are introduced in June and taken out in November. Hay feeders and water troughs are located on the south side of the stream near site 23. In the spring 2007, a manure pile was present uphill, on the south side, at approximately 50 meters from site 23. Shaded areas are present near site 22, past site 23 and beyond site 15

(Figure 2.1). Fields in crop rotation (corn, soybean, wheat) are present on both sides of the pasture field. The stream bed has been completely flattened downstream from site 22 to site 23, where cattle have spent time in the shade. Cattle also spent time near site 15, where the vegetation had been flattened on the stream banks and the stream bed was stepped on.

2.2.2 Stream Water Sampling

Portable water samplers (ISCO model 6712) were installed to collect stream water samples at monitoring sites: 18, 22 and 15. To capture rain events concentrations, the samplers were programmed to start collecting 1 litre of water in sterilized ISCO bottles when a prescribed amount of precipitation was reached. This was used as a surrogate for a change in stream flow, assuming the amount of rainfall would cause stream flow to increase. The amount was set between 5 and 15 mm, depending on the season and the water level in the stream. Rain gauges (ISCO 674) were used to measure the amount of precipitation. Manual samples (1 litre Grab Samples) were collected twice a week from April to December from all monitoring sites. During this time, water grab samples were also taken on a bi-weekly basis for the Health Canada-Agriculture and Agri-Food Canada Microbial Source Tracking (MST) project.

2.2.3 Laboratory Analysis

Unless mentioned otherwise, all samples were analyzed in the Agriculture and Agri-Food Canada laboratories, Ottawa, Ontario, Canada. Water samples collected from all monitored sites were analyzed for *Escherichia coli*, enterococcus and total coliforms using the Colilert (*Escherichia coli*) and Entrolert (Idexx Laboratories, Westbrook, MA, USA) enzyme substrate method, within 24 hours after collection time. The remaining water samples were split and frozen until analyzed for ammonium (NH_4), nitrate (NO_3), ortho-phosphate and total phosphorus analysis. A TrAAcs 800 autoanalyzer (Bran and Luebbe Analyzing Technologies, Inc., Elmsford, NY, USA) was used to analyse water samples for nitrate and ammonium. Ortho-phosphate was analyzed with a Lachat QuikChem FIA+8000 series autoanalyzer (Zellweger Analytics, Inc., Lincolnshire, IL, USA). Total phosphorus was analyzed with the Smart Spectro spectrophotometer from LaMotte Company (Chestertown, MD, USA). Only samples taken from site 15 and 18 were analyzed for *Cryptosporidium* and *Giardia* at the Alberta Provincial Laboratory for Public Health (Microbiology) in Calgary, Alberta, Canada as described in Ruecker et al., 2007. Analyses for *Listeria* spp. (pathogens) were conducted at the Agriculture and Agri-Food Canada laboratories in London, Ontario, Canada. Fecal coliforms and *Clostridium Perfringens* analyses were done at Agriculture and Agri-Food, London, Ontario, Canada. *Escherichia coli*, *Cryptosporidium* and *Giardia* sourcing was determined by genotyping in the Alberta Provincial Laboratory for Public Health, Calgary, Alberta, Canada. The complete method used for genotyping is described in Ruecker et al. (2007) although five replicates of nested PCR-restriction fragment length polymorphism (RFLP) analysis on the basis of the small-subunit (18S) rRNA gene was done.

2.2.4 Data Processing

Flow rate was determined using a 4150 Area Velocity Flow Logger (ISCO, Lincoln, NE, USA) installed in the stream at site 18 only, where velocity and depth were measured. The sensor was located in a round shape culvert which made it easy to determine the hydraulic radius at a given stream level needed to calculate discharge rate.

Velocity and level data points were imported into ISCO Flowlink 4.14 for Windows (ISCO, Lincoln, NE, USA) where stream discharge (m^3s^{-1}) was calculated on a 15-minute interval from April to December for each year. The 15-minute discharge rates and the contaminant concentrations (mgL^{-1}) were then used to calculate the instantaneous contaminant loads. To calculate stream discharge and contaminant loads for monitoring sites where discharge was not measured (sites 22, 23, 15); we applied a proportional amount of discharge based on the contributing areas for each site relative to that of site 18. We determined that base flow (low flow) was smaller than $0.024m^3s^{-1}$.

2.2.5 Load Calculation

Using the concentrations and instantaneous flow from each monitored site, we calculated instantaneous 15-minute loads of *Escherichia coli*, enterococcus, total coliforms, fecal coliforms and *Clostridium Perfringens* using the following equation:

$$\text{instantaneous load (cfu)} = \frac{cfu}{100mL}\cdot\frac{10^6 mL}{m^3}\cdot\frac{m^3}{\text{sec}}\cdot 15\,\text{min}\cdot\frac{60\,\text{sec}}{\text{min}}$$

where (cfu/100mL) is the concentration of either *Escherichia coli*, enterococcus, total coliforms, fecal coliforms or *Clostridium Perfringens* in the water samples and (m^3/sec) is the measurement of the 15 minutes instant flow rate in the stream. *Cryptosporidium* and *Giardia* loads were calculated as followed:

$$\text{instantaneous load (oocysts or cysts)} = \frac{oocysts\ or\ cyst}{100L}\cdot\frac{10^3 L}{m^3}\cdot\frac{m^3}{\text{sec}}\cdot 15\,\text{min}\cdot\frac{60\,\text{sec}}{\text{min}}$$

where (oocyst or cysts/100L) is the concentration of *Giardia* (cysts) and *Cryptosporidium* (oocysts) in the water samples and the (m^3/sec) is defined above. Nutrient loads (ammonium, ortho-phosphate and total phosphorus) were calculated as followed:

$$\text{instantaneous (mg)} = \frac{mg}{L} \cdot \frac{10^3 L}{m^3} \cdot \frac{m^3}{\sec} \cdot 15\min \cdot \frac{60\sec}{\min}$$

where (mg/L) is the concentration of ammonium, ortho-phosphate and total phosphorus in the water samples. These calculations are needed in order to determine if there is any significant difference between treatments.

2.2.6 Statistical Analysis

Statistical analysis was conducted to determine if there was any significant difference in loads between treatments. Studies, similar to ours, such as Byers et al. (2005), Lewis et al. (2005) and Line (2003) have shown that data collected in similar circumstances are not normally distributed and consequently nonparametric statistical analyses should be applied. According to the National Handbook of Water Quality Monitoring (USDA, 1994) an above-and-below watershed statistical design is analysed as a t-test of the differences between paired observations at the above and below stations. In our case, due to the nature of this study and data distribution, Mann-Whitney statistics were used under the Kruskal-Wallis nonparametric test in SYSTAT 10 (SPSS Inc., 2000).

2.3 Results and Discussion

2.3.1 General Observations

During the study period, June to November, the total amount of precipitation recorded in our study area was 559.1 mm in 2005, 523.9 mm in 2006 and 263.8 mm in 2007. The average temperature during these years and period was 16°C. During 2007, the number of collected samples was much smaller due to the dry conditions.

2.3.2 Pathogens Sourcing and Behaviour

2.3.2.1 *Cryptosporidium* and *Giardia*

Samples taken from sites 15 (URCA) and 18 (RCA), during the 2006 MST sampling period, were analyzed for *Cryptosporidium* and *Giardia*. The results show that in the majority of time, after livestock introduction, the concentrations of both parasites increased at site 15 (in the URCA) (Figure 2.2). Statistical analyses show no significant difference in loads between treatments (15-URCA vs 18-RCA) although the rank sum indicates that the URCA has greater loads of *Cryptosporidium* and *Giardia*. The stream level and flow have been extremely low during most of the summer at both sites, which corresponded to high concentrations. Water samples taken from low stream levels might have been contaminated with sediments, which could increase the oocysts recovery resulting in high concentrations (Feng et al., 2003). Nevertheless, the few samples analysed for *Cryptosporidium* and *Giardia* sourcing in late fall, at site 15, indicate that adult cattle is the main source. At the same time, adult cattle and wildlife are the main source of *Cryptosporidium* and *Giardia* at site 18. These results have certainly been affected by the fall manure application occurring on the upstream agricultural fields. Ruecker et al. (2007) concluded that the majority of the *cryptosporidia* in the South Nation watershed appears to be derived from mature cattle based on field observations and on the high presence of *C. andersoni* in the water samples. Although, *Cryptosporidium* and *Giardia* are not the best indicators of surface water quality, we still have noticed a slight improvement in the RCA, the rank sum indicated the URCA area to have greater load inputs.

2.3.2.2 *Escherichia Coli*

The *Escherichia Coli* sourcing, also conducted during the MST sampling period at site 15, supports the previous findings of the URCA, since during all seasons, livestock was the main source of bacteria (Figure 2.3). Human, avian, and other wildlife sources have percentages of occurrence less than 12%. Although, no significant difference was observed between treatments, the rank sum indicates that *Escherichia coli* loads are greater in the URCA area. Also, 59% of the observed *Escherichia coli* concentrations were greater than the recreational water limit (200 colonies per 100mL) after cattle introduction in the URCA area, i.e. a 35% increase compared to 28% in the RCA after cattle introduction. The high concentrations in the RCA area (Figure 2.4) could have resulted from the samples being taken from shaded areas, where bacteria are known to persist due to moisture and shade protection (Jamieson et al., 2002). Then again, when the before and after cattle introduction loads were compared at each site, the URCA sites were significantly different (p=0.012) with greater loads after introduction. A significant difference in loads (p=0.000) was also observed during stream low flow ($<0.024 m^3 s^{-1}$) in the URCA area. At greater flow, mostly occurring after a rain event, the *Escherichia coli* loads did not differ between treatments.

2.3.3 Other Bacteria Behaviour

Total coliforms, fecal coliforms, enterococcus and *Clostridium perfringens* showed similar results, i.e. no significant difference between treatments, but the rank sum indicated greater loads in the URCA area. More precisely, fecal coliforms, *Clostridium perfringens* and enterococcus loads were not significantly different between treatments before or after cattle introduction in the pasture areas, however, rank sum indicated greater loads in the URCA area. No similar studies have used and reported any impacts of these pathogens in surface water quality monitoring.

2.3.4 Pathogens

Over the two years of our study, *Listeria spp* was present at all sites over 88 percent of the times that analyses were done (Figure 2.5). *Listeria monocytogens* and *Campylobacter* had a low presence at site 15, but consistent through the other sites. *Salmonella* was rarely present at all sites, less than 7 percent of the time analyzed. *Escherichia coli* O157:H7 was not found at either of the sites.

Overall, before cattle introduction, *Listeria* and *Listeria monocytogenes* were the two most present pathogens at all sites. The percentage of presence at site 15 (25%) was three times lower then that at site 18 (75%). After cattle introduction, *Listeria* and *Listeria monocytogens* presence has decreased at all sites, especially at site 22 where the percentage of occurrence dropped from 50 to 0 percent. The presence of *Campylobacter* and *Salmonella* increased at all sites. The occurrence of pathogens presence at site 18 was always higher than the others sites, possibly due to surface runoff from the hobby farm upstream.

The data was not only impacted by the upstream hobby farm activities but it had also been greatly impacted by the important quantity of manure spread on the upstream fields, i.e. the manure application method employed does not permit to have a good control on the direction of application therefore an important amount was noticed to end in the stream. It was difficult to get a good back ground as the ephemeral stream starts where these activities take place.

2.3.5 Nutrients

Nutrient loads did not indicate any significant difference in loads between treatments. The nitrate drinking water guideline was not reached, the maximum concentration observed was 6.7 and 6.5 mgL^{-1} in the URCA and RCA area, respectively. Generally, no significant difference between treatments for nutrients was observed, but the rank sum showed greater loads of NH_4^+ in the unrestricted area before cattle introduction and after cattle introduction at low flow. After cattle introduction the rank sum indicated greater loads in the URCA area of ortho-phosphate and total phosphorus, which is not surprising as previous studies (Line et al., 2000) have indicated important contamination by this element. Ortho-phosphate and total phosphorus loads were also greater in the URCA area during high and low flow, this supports Line (2003) and Mayer et al. (2007) studies on riparian buffer protection impact on surface water quality.

2.4 General Discussions

Cattle intensity could had slightly varied between pastures as the cattle in the unrestricted area were observed to spent more time in the front pasture near site 22 (Figure 2.1). Theoretically this will suggest that the concentrations of contaminants flowing from the unrestricted area will be high, although this was not the case in our study. Studies such as Campbell and Allen-Diaz (1997) concluded that grazing intensity had no effect on surface water quality.

The slope intensity (~10%) in the restricted area might have impacted the results, as the small buffer strip could not reduce the momentum and magnitude of surface runoff as explained by Parkyn (2004). The slope intensity was similar between treatments although during base flow bacteria loads were greater in the unrestricted area therefore protecting the stream will reduce contamination.

2.5 Conclusions

All in situ bacterial non-point source contamination is difficult to monitor. As mentioned by Jamieson et al. (2002), many factors can affect bacterial behaviour: temperature, pH, moisture, soil type, slope, nutrients and microorganism competition. Other methods less expensive and less demanding could also be used, such as the following studies have suggested. Fisher and Endale (1999) found that *Escherichia coli* and enterococcus bacteria numbers decreased in surface waters when a pond is present in the pasture area between the surface water body and the grazing area. Byers et al. (2005) determined that providing a non-riparian shade and alternative off stream water source would keep livestock away from the stream; thus protecting surface water quality.

Upstream activities, in this type of study design, can also affect the results. Livestock is the main source of *Escherichia coli* and *Cryptosporidium* in our study area, with greater loads where cattle are allowed to move freely in the stream. Generally, no significant difference was observed between treatments, but the rank sums have shown that bacteria and nutrient loads were greater in the unprotected part of the stream. The fencing system seems to work for *Escherichia coli* load reduction at base flow, but not at high flow, i.e. after a rain event. A longer term study and within site comparison (comparing the before vs after cattle introduction impact at the same site and then comparing ameliorations between treatments) could better indicate treatment ameliorations in this small scale study type.

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2.6 Figures

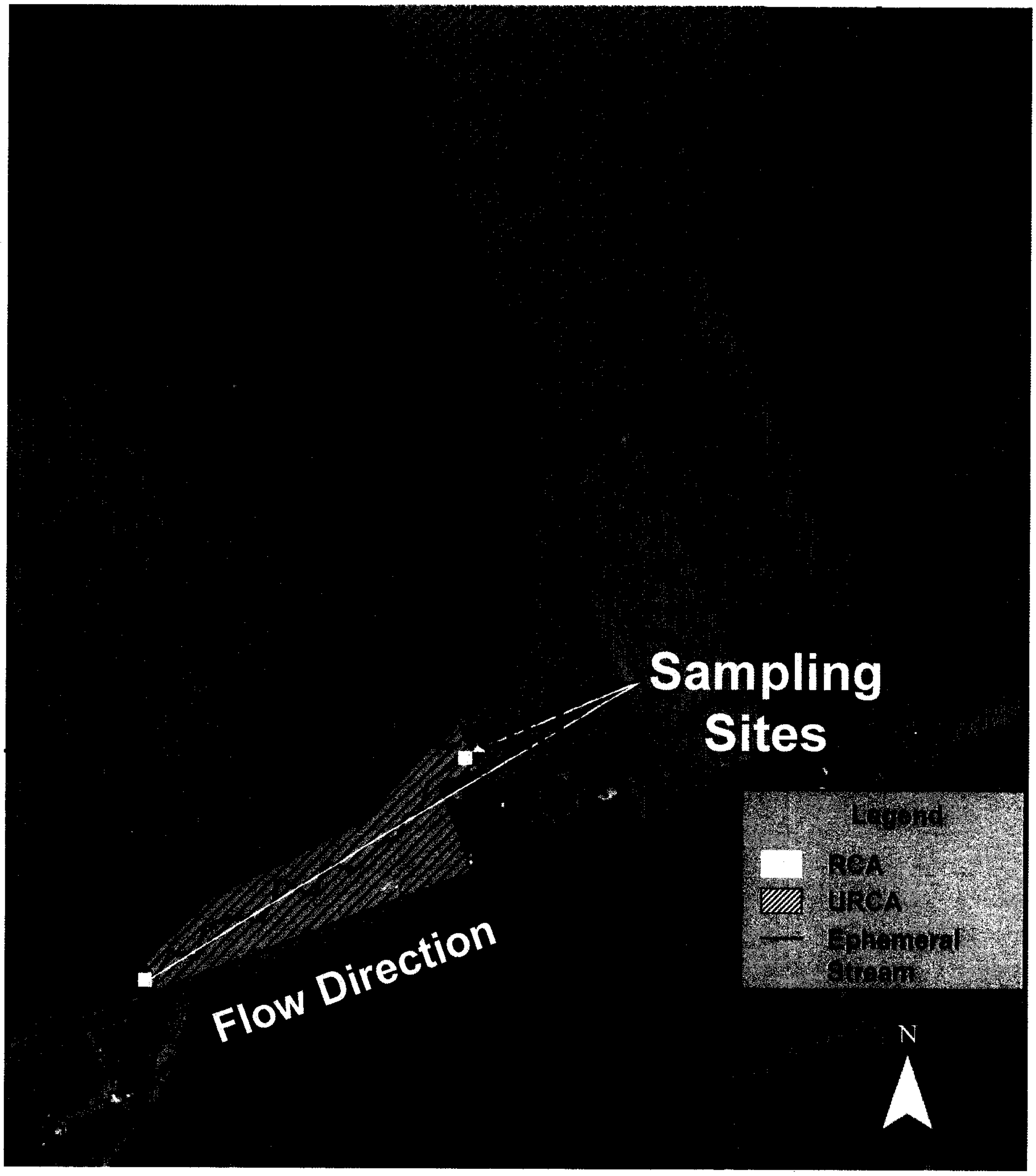

Figure 2.1 Map of pasture

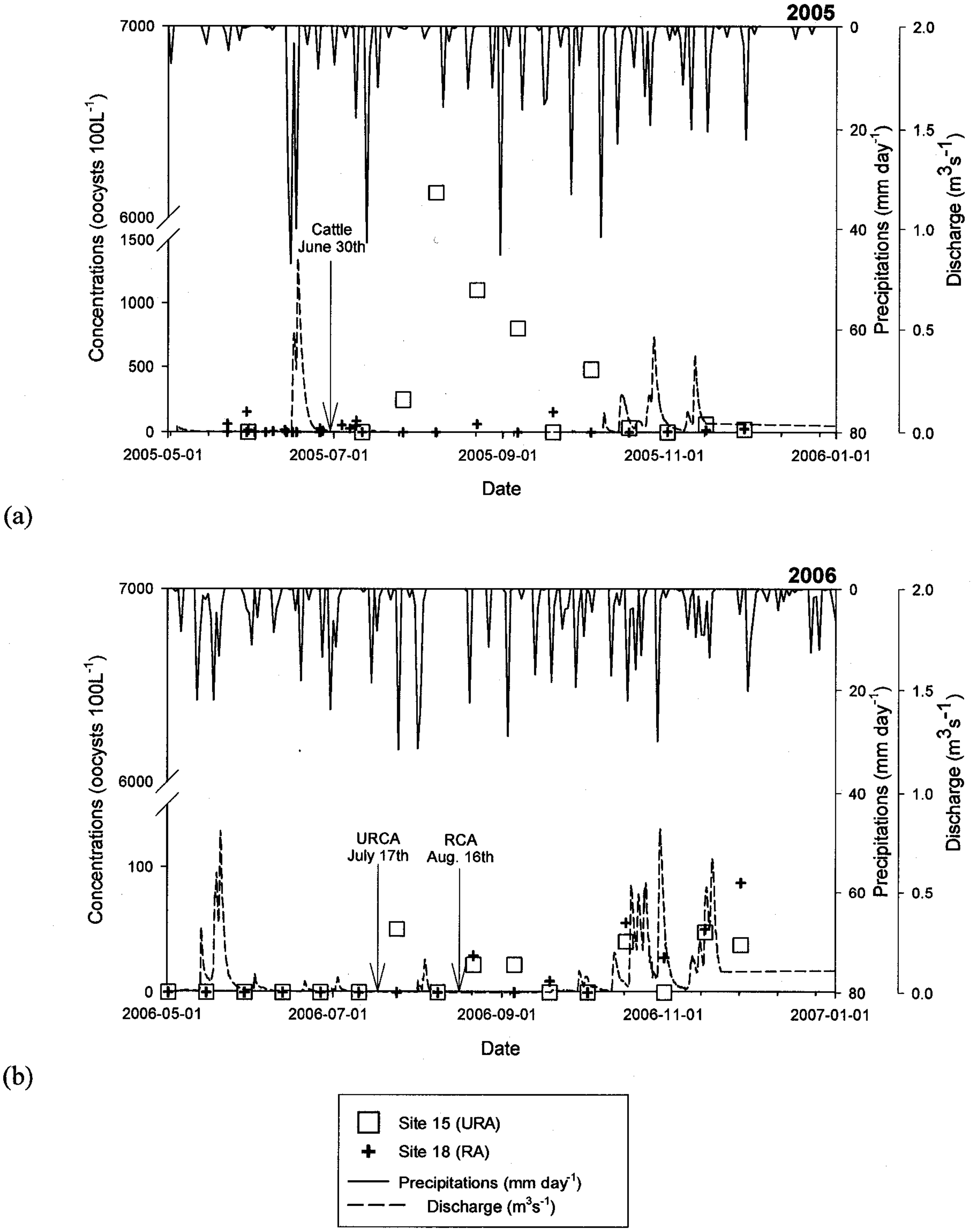

Figure 2.2 Measure (a) 2005 and (b) 2006 *Cryptosporidium* concentrations (oocyst 100L^{-1}). Precipitations (mm day^{-1}) and discharge (m^{3}s^{-1}) of the study area are also present. Arrows indicate when cattle have been introduced in the respective pasture.

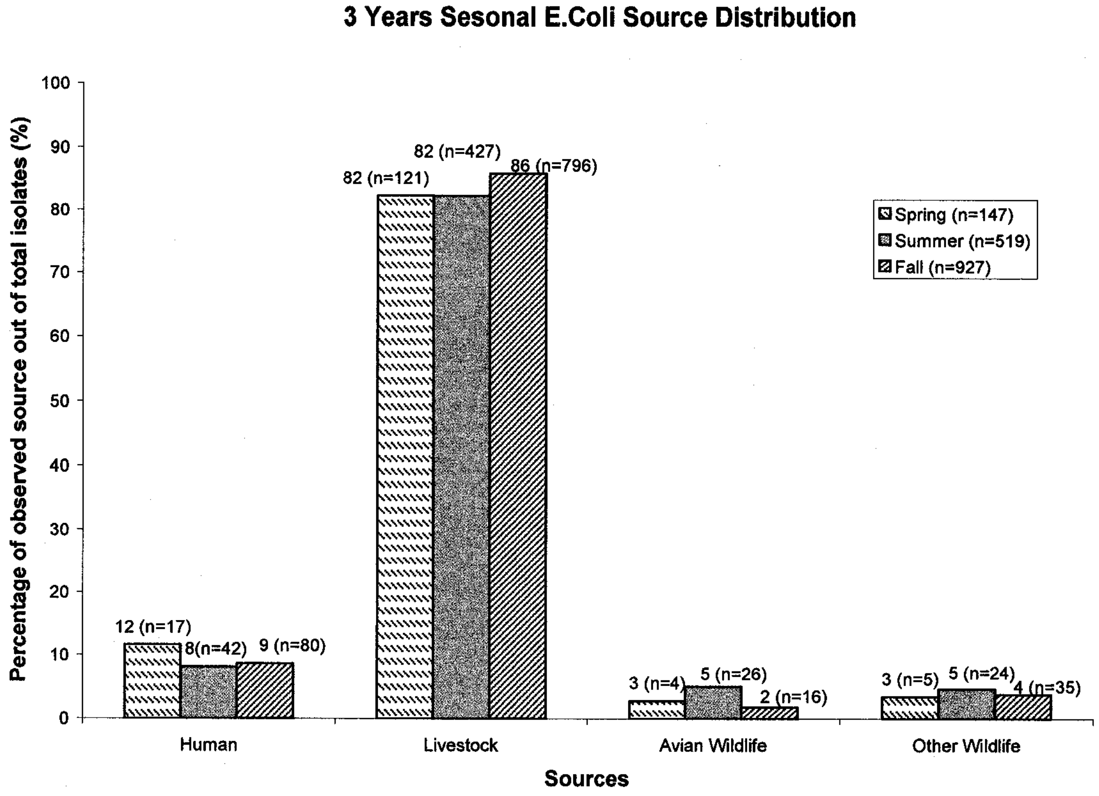

Figure 2.3 Seasonal variation of *Escherichia coli* source represented as percentage of observed source out of total isolates (%).

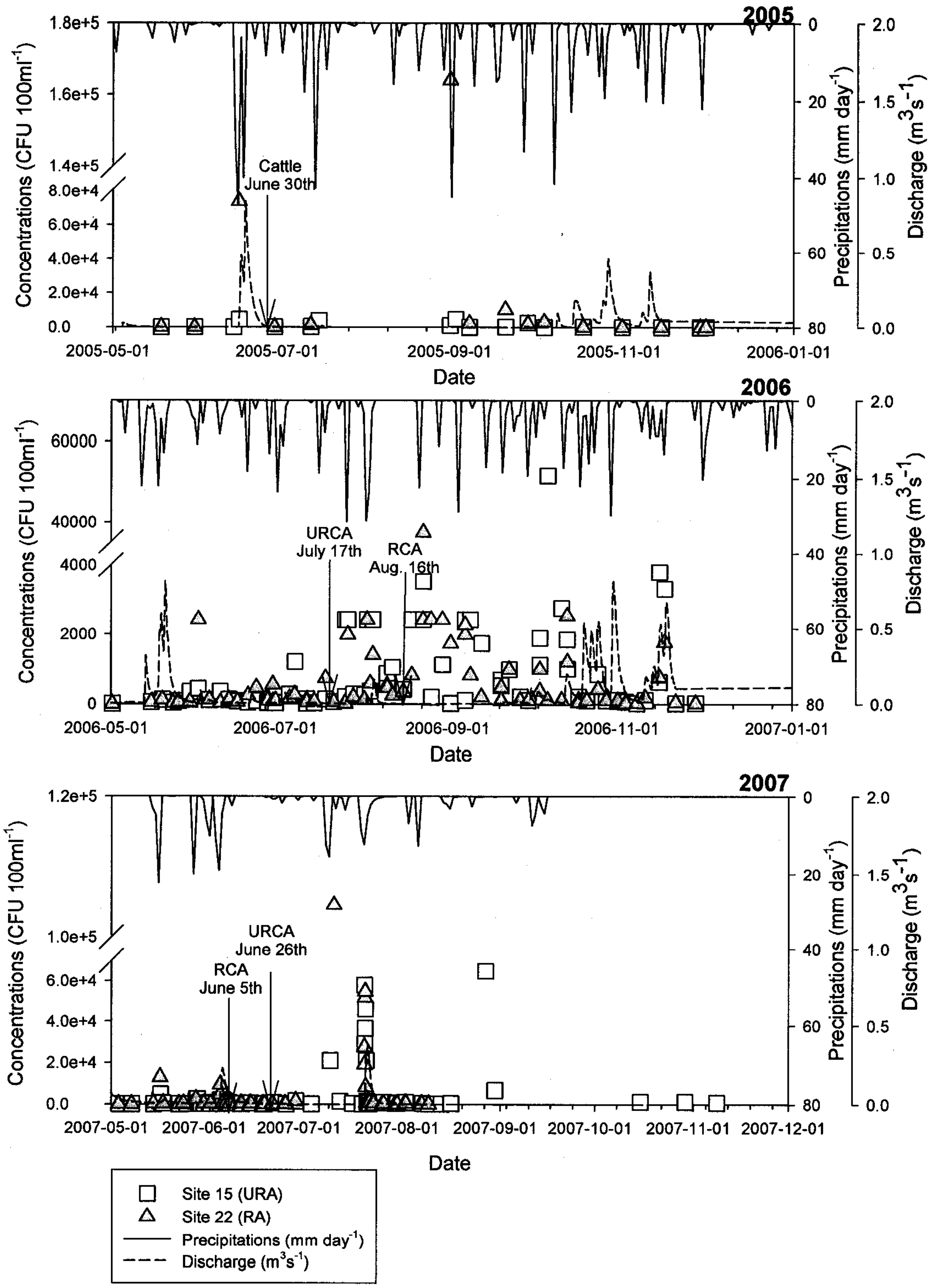

Figure 2.4 Measured 2005, 2006 and 2007 *Escherichia coli* concentrations (CFU 100ml^{-1}). Precipitations (mm day^{-1}) and discharge (m^{3}s^{-1}) in the study area are also present. Arrows indicate when cattle have been introduced in the respective pasture.

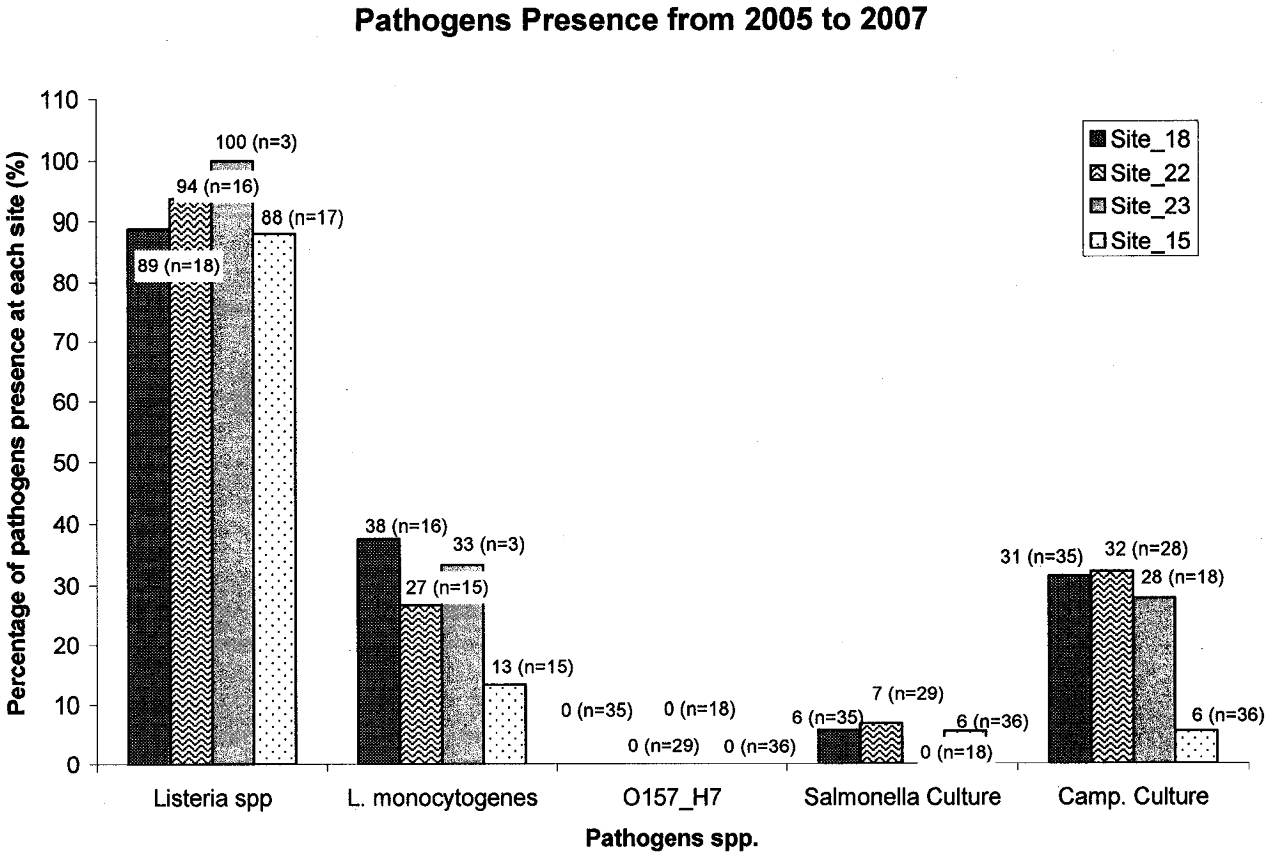

Figure 2.5 Pathogens presence (%) at each site (18-22RCA and 23-15URCA) over a three year period.

CHAPTER 3

IMPACT OF AGRICULTURAL SUBSURFACE TILE DRAINAGE MANAGEMENT ON NITROGEN MASS BALANCE

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3.1 Introduction

Subsurface tile drainage is a common practice used on agricultural fields in Eastern Ontario. It reduces soil erosion, assists in the reduction of phosphorus in streams, increases yields, and permits to start field work earlier in the spring (Planning to Drain Your Land, Land Improvement Contractors of Ontario, http://www.drainage.org/). On the other hand, subsurface tile drainage limits the residence time of water in the soil and therefore contributes to the contamination of adjacent surface waters with nutrients and pesticides (Fausey et al., 1995; David et al., 1997; Kladivko et al., 2001; Tomer et al., 2003; Heppell and Chapman, 2006).

Nitrogen mass losses through subsurface tile drainage are very considerable, both for the protection of the environment and for the producer's revenue (Randall and Mulla, 2001; Randall and Goss, 2001; Grigg et al., 2003; Burchell et al., 2003). Tan et al.

(2002) measured nutrient quantities draining out through tiles as high as 82 kg N ha^{-1} for fertilized continuous corn, 99.9 kg N ha^{-1} for fertilized rotation corn, and 69.8 kg N ha^{-1} for second year fertilized alfalfa, over a three year period in southern Ontario. During these three years, the flow weighted mean concentration exceeded the Canadian and European water guideline values of 10 mg N L^{-1} and 11.3 mg N L^{-1}, respectively. In humans, large amounts of nitrate (NO_3^-) (the most soluble and mobile form of nitrogen lost through subsurface tile drainage) or nitrite (NO_2^-) can result in methemoglobinemia, which first shows up as a blue discoloration of the skin, but can lead to asphyxia as the blood ability to carry oxygen is impaired (Ironside, 2001). The nitrate-nitrogen (NO_3-N) limit for aquatic life is set to 3 mg N L^{-1} (Canadian Council of Ministers of the Environment, 2007) as early life stages in aquatic organisms can be harmfully impacted by nitrate (NO_3^-) concentrations. It can reduce the oxygen-carrying capacity of the blood and can disrupt the ability to maintain proper balance of salts.

Managed (controlled) subsurface tile drainage was found to be a good alternative to direct tile outflow. Drury et al. (1996) reported a 25% decrease in mean nitrate concentration, and a 49% decrease in total annual nitrate load when subsurface tile drainage management was implemented on clay loam soil in southern Ontario. Ng et al. (2002) found that managed subsurface tile drainage not only reduced nitrate loss by 36%, but also increased corn yields by 64%. Similar results were found by Wesström and Messing (2007) where nitrogen drain outflow was reduced by 60 to 95% and yields were 2 to 18% larger on managed subsurface tile drainage fields.

Most of the research conducted on managed (controlled) subsurface tile drainage has been on the reduction of nitrogen outflow and yields increase, although soil mineral nitrogen, groundwater nitrogen loads and surface denitrification (nitrous oxide releases) are known to be affected. This study compared nitrogen mass balances between controlled (CTD / managed) and uncontrolled (UCTD / conventional) subsurface tile drainage in eastern Ontario.

Nitrogen mass balance research conducted on conventional subsurface tile drainage has indicated that total N outputs exceeded total N inputs (David et al., 1997; Jaynes et al., 2001; Webb et al., 2004; Libra et al., 2004; Jaynes and Karlen, 2005). Jaynes and Karlen (2005) noticed that nitrogen inputs exceeded nitrogen output in a corn-soybean rotation only when fertilizer application was greater then 280 kg N ha^{-1}. Nitrous oxide emissions from conventional subsurface tile drainage were determined to be affected by moisture, nitrate (NO_3) content and temperature in the top soil layer (Liang and Mackenzie, 1997; Hénault and Germon, 2000; Valé et al., 2007). Valé et al. (2007) measured values as high as 34 kg N ha^{-1} in the un-irrigated to 46 kg N ha^{-1} in the irrigated treatments on bare soil in France. Liang and MacKenzie (1997) measured values as high as 41 kg N ha^{-1} on Chicot sandy clay loam and as high as 53 kg N ha^{-1} on a Ste. Rosalie clay under corn (*Zea mays* L.) cultivation and conventional subsurface tile drainage.

Our ultimate goal is to determine how managed (controlled) subsurface tile drainage affects nitrogen mass balance. Is this practice sustainable or should it be improved? We compared nitrogen (N) inputs (fertilizer, wet-N deposition and N fixed) and outputs (tile outflow, groundwater, plant uptake, denitrification (N_2O) and change in soil mineral N). The following equation was used:

$$\boldsymbol{\Delta}\ \textbf{inputs - }\boldsymbol{\Delta}\ \textbf{outputs - }\boldsymbol{\Delta}\ \textbf{soil residual mineral N = residual}$$

in order to determine nitrogen mass balances.

3.2 Methods

3.2.1 Study Site

The sampling sites are located on privately owned tile-drained agricultural fields near St.-Albert, Ontario, in the South Nation watershed, more precisely in the micro-watershed of the Philippe Blanchard Municipal Drain (45°16'48"N, 75°9'36"W). The soil is classified as a Bainsville series, characterized by layered silt and fine sand overlying clay deposits, with poor natural drainage (Canadian and Ontario Department of Agriculture, 1962). The fields have a flat topography (slope less then 1%) with a shallow impermeable (clay) layer situated at a depth of approximately 1.2m. The region is characterized by dairy farm operations and cash-crop farms; typically corn, soybean, and forage rotation. Monitored fields were planted with corn during the study periods from June to November in 2006 and 2007; soybean was planted in one pair of fields in 2006.

3.2.2 Site Description

A total of eight fields of comparable size, soil type, and cropping management were paired in order to compare nitrogen mass balances between managed (CTD) and conventional (UCTD) tile drainage (Figure 3.1). Field surface areas varied from 2 to 7 hectares. Each field was tile drained by lateral subsurface tile drains (102 mm in diameter) spaced by approximately 15 m with an average tile slope of 0.1% (Planning to Drain Your Land, Land Improvement Contractors of Ontario, http://www.drainage.org/). These lateral tiles are connected to a main header tile and outlet which was retrofitted with an inline water level control structure (Agri Drain Corporation, Adair, USA). In the CTD fields, the control structures' stoplogs were set at a depth of 0.6 m from ground surface to maintain a potential water table at this level

(Figure 3.2). The control structures in the UCTD were left open to permit free flow as it would occur when these structures are not present (Figure 3.3).

In each field, two in-field sites equipped with suction lysimeters (30 and 60 cm deep), groundwater wells (2 m deep) and volumetric water content time domain reflectance profiling probes (90 and 120 cm deep), and GHG chambers were installed between lateral tiles at the beginning of the tile (shallow tile depth sites) and near the outlet (deep tile depth sites) to have a better estimation of the nitrogen occurrence in the fields.

Water samples collected from tile, groundwater and lysimeter were frozen and then analyzed the following winter for ammonium-nitrogen (NH_4-N) and nitrate-nitrogen (NO_3-N) with a TrAAcs 800 autoanalyzer (Bran and Luebbe Analyzing Technologies Inc., Elmsford, NY, USA) at Agriculture and Agri-Food Canada, Ottawa.

3.2.3 Tile Drainage

Tile flow was monitored continually in both controlled and uncontrolled tile outlets in order to quantify differences in the total amount of tile water discharge and nitrogen exported during the study period. Bubbler flow modules (ISCO model 730) attached to ISCO automatic samplers (ISCO model 6712) were used in 22° V-notch weir boxes installed at each outlet or directly in the water control structures to obtain continuous flow records of tile discharge. Portable automatic samplers were used to collect water samples at each tile outlet. Generally, composite water samples were collected twice a week during continuous tile flow, whereas more intensive sampling was done during rain event flow in order to capture hydrograph events and subsequent nutrient leaching. In 2006, automated samplers were triggered by actuators installed in the weir boxes from where water samples were taken. In 2007, the bubbler modules were set in the water control structures and permitted to program the sampler to start taking samples directly from the structure at a change of level. On the day of collection, the 1 L water samples were taken back to the lab in coolers with ice to maintain sample integrity. Turbidity, pH and electrical conductivity were measured on each water sample using a Global Water turbidity meter WQ770 (Global Water Instrumentation, INC., Gold River, CA, USA) and an YSI 556 MPS sonde (YSI Environmental, Yellow Springs, OH, USA). All water samples were frozen for subsequent analysis of NO_3-N and NH_4-N.

3.2.4 Groundwater

At each infield site, a filtered perforated PVC pipe (50.8mm diameter) was installed between lateral tiles to a depth of 2m to measure free-standing water levels and to collect groundwater samples. Manual depth measurements and groundwater well purging were conducted 24 hours prior to sampling. For continuous monitoring of groundwater levels, each well was monitored with water level loggers (Global Water Instruments Inc., Gold River, CA, USA) which recorded levels every 15 minutes. One-litre water samples were taken back to the laboratory in coolers filled with ice where they were filtered using Whatman No. 42 filter paper and frozen for subsequent NO_3-N and NH_4-N analysis.

Bail tests at saturation were performed to determine the hydraulic conductivity at saturation (K_{sat}). K_{sat} was calculated with a slightly modified Hvorslev's solution (Annex B) (Hvorslev, 1951). Technically, the Hvorslev technique is valid in confined systems only, and another technique, such as the Bouwer & Rice technique, should be used. However, the difference in the resulting conductivity at this site is less than the variability caused by spatial heterogeneity. So, the simpler Hvorslev method was used. The Dupuit-Forchheimer equation (Annex B) was used to determine lateral groundwater flow (Hansen et al., 1980).

Horizontal flow and nutrient concentrations in groundwater were used to determine the total amount of nitrogen leaching from the fields in kg N ha^{-1} day^{-1}. The change in groundwater nitrogen content above the impermeable layer (1.2m below soil surface) was calculated as the difference between the amount of nitrogen present in the fall and spring based on the water level above the impermeable layer. Data was linearly interpolated between sampling times in order to have continuous data. Statistical t-test was conducted on measured groundwater concentrations to determine if there was any significant difference between treatments.

3.2.5 Lysimeters and volumetric soil moisture content

Ceramic cup suction lysimeters were installed at in-field sites and were used to collect weekly pore space water samples from 30 and 60cm depths. Samples were split for nutrients (NO_3-N and NH_4-N) and then frozen until analyzed.

Volumetric water content time domain reflectance profiling probes (E.S.I. Environmental Sensor Inc., Sidney, BC, Canada) were installed at all in-field sites. Soil moisture readings were taken manually twice a week from depths of 0-15, 15-30, 30-45, 45-60 and 60-90 cm (at shallow tile depth sites) and of 0-30, 30-60, 60-90, 90-120 and 120-150 cm (at deep tile depth sites).

3.2.6 Denitrification Estimates

Greenhouse gas releases were collected once a week from 8 sites in two pair of fields (2 sites per field). Clear acrylic plastic chambers (62cm in diameter) were installed between corn rows at a depth of 4 cm and left in situ for the study period as described by Rochette et al. 2004. Samples were collected from these chambers at 2 minute intervals for a total of 8 minutes in 30mL evacuated vials containing 0.2-0.3 g of $Mg(ClO_4)_2$ as desiccant. Nitrous oxide (N_2O) and carbon dioxide (CO_2) soil releases were analyzed at Agriculture and Agri-Food Canada (AAFC-AAC) in Ottawa. Total nitrogen released for the sampled day was calculated using the equation determined by Rochette et al. 2004 (Gregorich Lab, AAFC-AAC, Ottawa).

The NEMIS model (Hénault and Germon, 2000) was used to estimate the total gaseous nitrogen release over the study period through nitrous oxide (N_2O-N) (kg $ha^{-1}day^{-1}$). NEMIS uses a relationship between soil volumetric moisture content (F_W), ground temperature (F_T), soil nitrate concentration (F_N), and the potential denitrification rate of the soil (D_P) in order to best estimate denitrification (D_A) as:

$$\mathbf{D_A = D_P\ F_N\ F_W\ F_T}$$

Each factor in this equation has a specific relationship to denitrification and was validated with measured values. The most probable potential denitrification rate (D_P) was taken from Drury, et al. (1991) where the Tuscola loam soil type has similar characteristics as the Bainsville series. The D_P for this soil was 8.47 kg N $ha^{-1}day^{-1}$. NEMIS optimum volumetric moisture content for denitrification was determined to be around 30%. In our case, the volumetric water content optimum was dropped to 25% due to the low measured denitrification flux and the low volumetric moisture content of the soil. Soil total nitrogen concentrations were determined from volumetric moisture content, bulk density, and concentrations of NO_3-N and NH_4-N in lysimeter water samples taken from 30 centimeter depth. Soil temperature was measured during greenhouse gas collection every week. Volumetric moisture content, temperature, soil NO_3-N and NH_4-N concentrations at 30cm depth were linearly interpolated between sampling times in order to have continuous data.

3.2.7 Plant N Removal

Above-ground plant samples (without roots) were collected before harvest, dried at 85°C, then weighed, ground, and analyzed for total nitrogen and carbon content. Plant total nitrogen and carbon content was analyzed using a Carlo Erba Model NC2500-elemental analyzer N/C (Milan, Italy). Plant density per hectare was estimated by counting plants on three plots of 4 m^2 in soybean fields and on three plots of 9 m^2 in corn fields. Average plant weight, density and total nitrogen were used to estimate nitrogen uptake (kg N ha^{-1}) in plants.

3.2.8 Nitrogen Fixation

Nitrogen (N) fixation in soybean planted in 2006, was calculated using Barry et al. (1993) equation, a linear relationship between soybean grain yield (Mg ha^{-1}) and N fixed (kg N ha^{-1}) as:

$$\mathbf{N_{fixed} = 81.1 * yield - 98.5}$$

Soybean grain yields were measured by the farmer at harvest using a combine harvester and yield monitor.

3.2.9 Fertilizers, wet-Nitrogen deposition and soil samples

The quantity of fertilizers applied varied from 3.4 kg N ha^{-1} in soybean fields to 176.6 kg N ha^{-1} in corn fields. In the beginning of May, paired fields received the same amount and type of fertilizer prior to and at planting. Broadcast fertilizer applied prior to planting was applied as urea. Urea-ammonium-nitrate (UAN) fertilizer was applied at planting as a starter.

Wet-N deposition was calculated based on weekly mean concentrations of ammonium-N and nitrate-N measured in rain water at station NY22-Akwesasne Mohawk-Fort Convington by the National Atmospheric Deposition Program, USA (http://nadp.sws.uiuc.edu/). The concentrations were first calculated as total monthly nitrogen deposition (kg) and then converted to total amount in kilograms of nitrogen per hectare (kg N ha^{-1}) deposited over the study period (June to November of either 2006 or 2007) using total monthly precipitation data collected by a on-site HOBO weather station (Onset Computer Corporation, Bourne, MA, U.S.A.).

Integrated soil samples were taken in spring before planting and in fall after harvest from depth intervals of 0-15, 15-30 and 30-60 cm in infield sites (16 sites). Residual soil NO_3-N and NH_4-N concentrations were measured on these samples as described by Drury et al., (1991). Bulk density was measured on undisturbed core samples as described in the Methods of Soil Analysis (Klute and Page, 1986).

3.3 Results and Discussion

3.3.1 General Observations

The total annual precipitation registered on our study area was 795.9 mm in 2006 and 588.4 mm in 2007. Annual temperature varied from a minimum of -23.4°C to a maximum of 34.4°C in 2006 and from -30.2°C to 32.3°C in 2007 (Figure 3.4). During 2007, the number of water samples was much smaller due to dryer conditions.

In 2006, the subsurface tile drainage results from soybean paired fields indicated contrary results to what was expected, i.e. the volume of water drained from the controlled tile drainage site was greater than the comparative free flow tile drainage. In 2007, this same pair of fields was planted with two different species of corn, which compromised results.

3.3.2 Nitrogen Inputs

3.3.2.1 Fertilizers, Nitrogen Fixation and Wet-Nitrogen Deposition

Mineral fertilizer was the main source of nitrogen for the corn crop. Fertilizer applications were considered to be high (> 170 kg N ha^{-1}) in 2006 and medium in 2007 (137 kg N ha^{-1}) based on Jaynes et al. (2001). Based on the Barry et al. (1993) equation, soybean plants fixed 97 % of their nitrogen needs. The retention of nitrogen due to fixation process was 10 % greater in the controlled field; this is six times lower than what was found by Fisher et al. (1999) in subirrigated and controlled drainage fields. This difference could possibly be due to the fact that their value was directly measured on crop grain and ours was estimated from the measured yields.

Wet nitrogen deposition represented approximately 1.5 % of the total nitrogen applied on the fields. In 2006, the amount was about 2.62 kg N ha^{-1} and in 2007, a drier year, the amount was 1.61 kg N ha^{-1}. These estimated values are below 3.7 kg N ha^{-1} the minimal annual range of wet deposition occurring across the Corn and Soybean Belt, USA (Jaynes and Karlen, 2005). Although total nitrogen wet deposition over the entire year represented only 1.5% of the total nitrogen input, a better estimation will be done with nitrate-nitrogen and ammonium-nitrogen concentration measured in precipitation water occurring over the study fields.

3.3.3 Nitrogen Outputs

3.3.3.1 Tile Drainage

Nitrogen loss through tile was reduced from 25 to almost 100 % in the managed subsurface tile drainage. The 100% reduction is due to the fact that the water level in the managed tile drainage did not reached the level (60 cm below ground surface) set in the control structure. These percentages are the ratios of difference between paired uncontrolled and controlled field (ie: % = [(UCTD – CD)/UCTD]*100), calculated for each pair. The greater reductions were noticed during the drier year in 2007, when control structure overflow had not occurred in either one of the managed subsurface tile drains. These results concur with the conclusions drawn by other studies conducted on managed subsurface tile drainage, including: Drury et al. (1996), Fisher et al. (1999), Ng et al. (2002), and Wesström and Messing (2007).

Mean nitrate concentrations varied between 3.71 and 9.93 mg N L^{-1} in all treatments, which met the drinking water guidelines value (10 mg N L^{-1}). On the other hand, the limit concentration for freshwater protection (3 mg N L^{-1}) was not met in either treatment. Surprisingly, nitrogen mass lost through tile flow were much more conservative (between 2.2 and 12.4 kg N ha^{-1}) (table 3.1 and 3.2) than those measured by Ng et al. (2002) (between 36.8 and 57.9 kg N ha^{-1}), under similar rainfall and amount of fertilizer applied, although they also included the winter period.

3.3.3.2 Groundwater Leaching

Nitrogen loss through groundwater leaching represented less than 1 % of the total nitrogen removed from the fields and it was similar between paired fields (table 3.1 and 3.2). The concentrations mean over two years (3.59 to 11.24 mg N L^{-1}) exceeded the freshwater guideline in all treatments. Drinking water guideline was exceeded sometimes in the CTD and sometimes in the UCTD. The exceeding values occurred 26% of the time in samples collected in CTD and 22% of the times in samples collected in the UCTD. One of the possible reasons why our groundwater nitrogen contamination was low, is that shallow aquifers are mainly protected from NO_3 contamination due to plant uptake, denitrification, warm conditions and the carbon rich environments (only corn and soybean grain were harvested, the rest of the plant was left behind) (Spalding and Exner, 1993).

3.3.3.3 Denitrification

The denitrification process is responsible, at the most, for 9 % of total nitrogen loss (0.4 to 27.6 kg N ha^{-1}) in our fields. It was responsible at the most for 3.3% of total nitrogen loss in CTD compared to 7.7% in UCTD under corn. Measured daily N_2O-N releases varied from 0 to 0.119 kg N ha^{-1} in the CTD and from 0 to 0.045 kg N ha^{-1} in UCTD. Releases from soybean fields were smaller. No significant differences ($p>0.05$) were observed between treatments on measured values, although the mean was greater in the UCTD fields.

The predicted total amount of nitrogen released through denitrification over the study period varied in fields from nearly no release, in 2007, to 27.6 kg N ha^{-1} in 2006 (Table 3.2 and 3.1), this is almost 100 times greater than what was measured. Measured values were mainly taken at lower than optimum volumetric water content (30%) needed for the NEMIS model and this could explain the variation. We have measured field denitrification at volumetric water contents as low as 20%. This level of volumetric water content must have been enough to cause anaerobic conditions for the anaerobes to start using nitrate as an electron acceptor. No studies have been found comparing N_2O-N releases from controlled and uncontrolled subsurface tile drainage, although Liang and MacKenzie (1997) conducted a study on two Quebec soils (sandy clay loam and clay) and indicated similar magnitudes of denitrification rates (between 4 and 53 kg N ha^{-1}). The four studied fields are inconclusive regarding the impact of managed tile drainage on denitrification (50% occurrence).

3.3.3.4 Plant Nitrogen Removal

Plant uptake represented the greatest amount of nitrogen removal. Between 88 and 99 % of total nitrogen loss was retained in plants and the retention was greater in the controlled drainage fields. In some cases, the retention was 21 % greater in the CTD fields and this happened during a relatively dry year (2007) (Table 3.2). Corn total plant-N removal varied between 185 and 314 kg N ha^{-1} in CTD and between 210 and 345 kg N ha^{-1} in UCTD. Soybean plant uptake was the greatest, although in UCTD, the results are contrary to what the literature has suggested (Jaynes et al., 2001).

3.3.3.5 Change in residual soil mineral N and groundwater N

Residual soil mineral nitrogen indicated that CTD had less or about the same amount of mineral nitrogen taken out from the soil during the wetter year. On the contrary, during a drier year, it was the UCTD that had a greater mass deficit in soil mineral N (table 3.2). Elmi et al. (2000) study concluded that if denitrification is low, then soil mineral N will build up, which is what happened in 50 % of our fields.

Residual groundwater nitrogen increased in CTD over the wetter year, this can be explained by enough water percolating from the surface, dissolving and carrying nutrients. These values were calculated over a depth of 1.2m (impermeable layer); therefore if the water table was lower than this depth (UCTD case) the result would be zero. In 2007, a small nitrogen deficit (-0.8 to -2.3 kg N ha^{-1}) was observed in all treatments.

3.3.3.6 Nitrogen Balance Residue

Nitrogen deficit was generally greater in CTD during the drier year, values vary from -8 to -144 kg N ha^{-1} (table 3.2). During a wetter year (2006), 50% of the CTD fields had also a high deficit (table 3.1). This means that managed subsurface tile drainage allows the release of nitrogen from some uncounted nitrogen sources. The unaccounted source, the nitrogen pool, was indicated to be the total soil organic nitrogen (or soil organic carbon) by Jaynes and Karlen (2005). They brought in the question of the long-term productivity of these soils if the pool of nitrogen is emptied each year. In this case, what happens if the crop is only harvested for the grain and the rest of the plant is left behind each year? Would there by more nitrogen available for the next season? Soil organic nitrogen or carbon should be accounted for any further nitrogen mass balance studies.

3.4 Conclusion

Generally, the study showed that managed subsurface tile drainage reduces the amount of nitrogen lost through tile, increases plant uptake, but it does not really load groundwater and does not seem to increase denitrification. Soil organic nitrogen should be accounted for in the mass balance as it seems to play a very important role, therefore soil organic nitrogen measurements should be made. Plant nitrogen uptake is the main nitrogen taker followed by denitrification. In order to better estimate denitrification, field measurement values should be taken at higher volumetric water content and more frequently. The controlled tile drainage system seems to function well, although long-term studies should be conducted to determine the viability of the soil and water quality protection.

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3.6 Figures

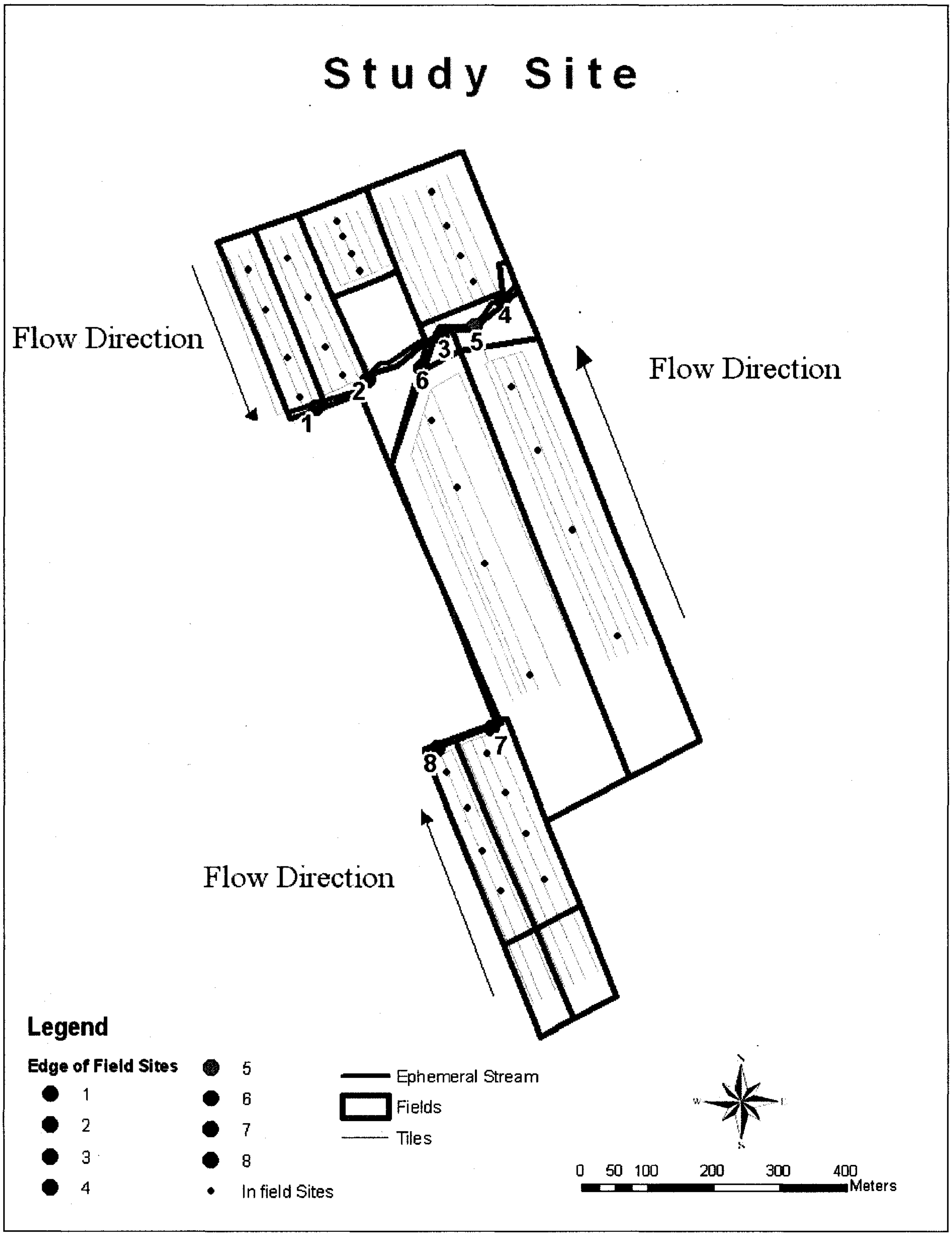

Figure 3.1 Study site

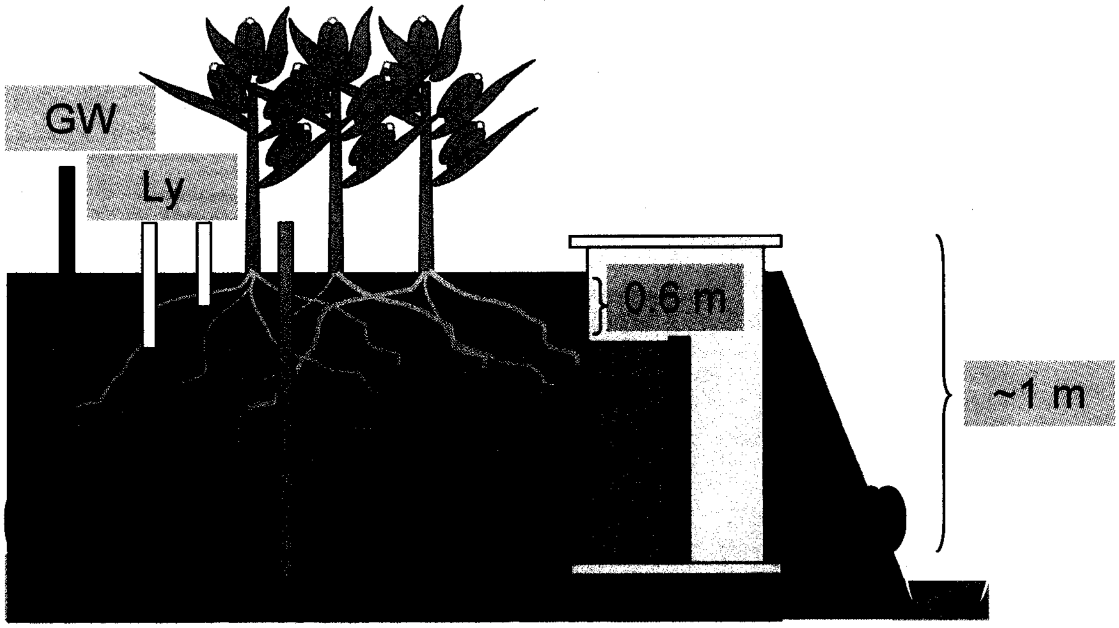

Figure 3.2 Managed subsurface tile drainage infield site setup. Tiles at approximately 1 meter depth and groundwater well (GW) at approximately 2 meter depth. Lysimeters (Ly) at 30 and 60 centimetres depth. Time domain reflectance profiling probes (TDR) and control structure where the water level was kept at 60 centimetres in the ground.

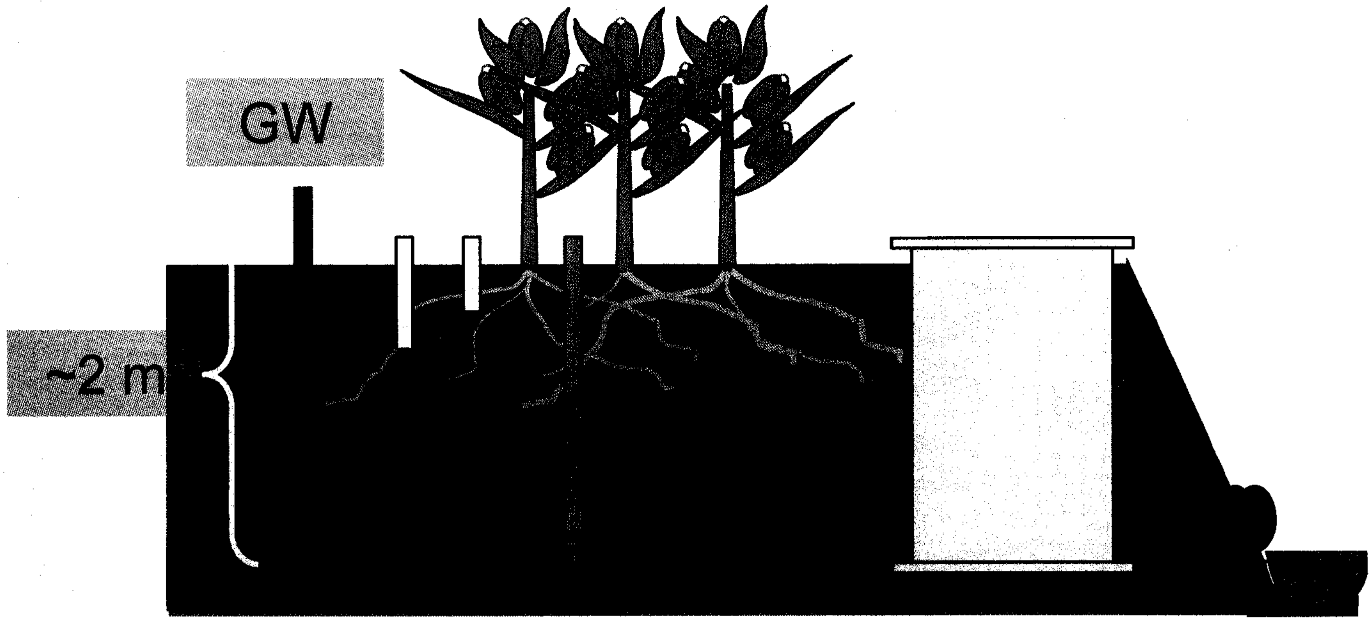

Figure 3.3 Conventional (free flow) tile drainage infield site setup. Same instruments as mentioned in Figure 3.2, although the control structure was not closed to maintain water in the ground.

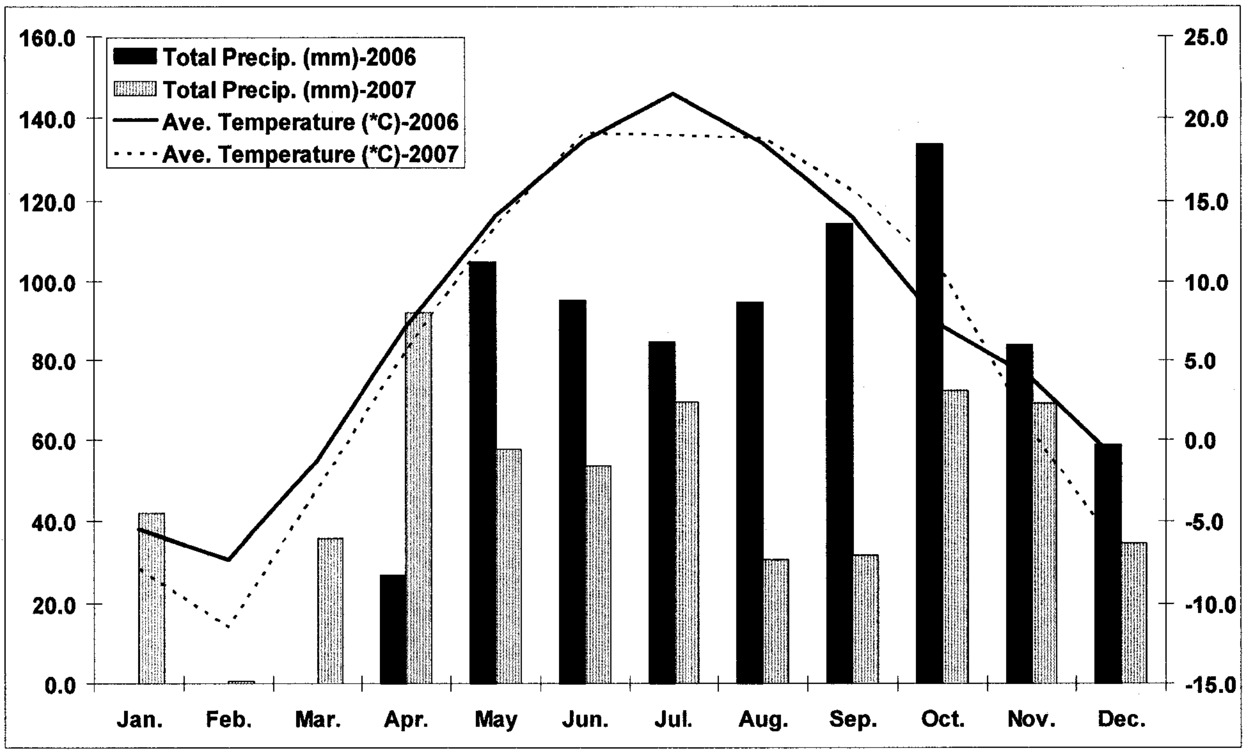

Figure 3.4 Monthly average temperature and total precipitations for 2006 and 2007.

3.7 Tables

Table 3.1 Partial, from 0 to 120 centimeters, nitrogen mass balance in kilogram nitrogen per hectare from June to November, 2006. Samples standard deviation is presented in brackets for concentrations (in parts per million).

Depth 0 to 120 cm	N inputs			N outputs						
	Fertilizer	Wet Deposition	N Fixed	Total Tile Drainage Loss	Total Groundwater Drainage Loss	Denitrification	Total Plant-N Removal	Change in Residual Soil Mineral N	Change in Residual Groundwater N	N Balance Residual
Fields	....	....	....	....	....	...kg N ha^{-1}...	....	....	....	....
1 (Cd-Corn)	171.3	2.6		2.2 (± 5.5)	1.1 (± 11.3)	13.3	314.9	-30.7	22.5	-149.4
2 (Ucd-Corn)	171.3	2.6		9.1 (± 14.9)	1.2 (± 14.1)	27.6	295.7	-65.6	0.0	-94.0
3 (Cd-Corn)	171.3	2.6		5.1 (± 7.4)	2.0 (± 3.4)	4.9	216.8	-39.8	19.1	-34.2
4 (Ucd-Corn)	171.3	2.6		6.8 (± 10.0)	2.5 (± 10.8)	12.7	210.9	8.9	4.5	-72.5
5 (Ucd-Soybean)	3.4	2.6	176.3	11.1 (± 1.5)	0.2 (± 4.5)	10.2	527.3	-12.8	0.0	-353.7
6 (Cd-Soybean)	3.4	2.6	195.6	12.4 (± 0.8)	1.7 (± 7.8)	12.8	433.9	-20.9	10.9	-249.1
7 (Ucd-Corn)	176.6	2.6		4.2 (± 4.2)	0.3 (± 3.2)		297.6	-47.9	3.8	-78.6
8 (Cd-Corn)	176.6	2.6		3.1 (± 0.9)	0.3 (± 4.0)		299.6	-49.8	17.8	-91.8

Table 3.2 Partial, from 0 to 120 centimeters, nitrogen mass balance in kilogram nitrogen per hectare from June to November, 2007. Samples standard deviation is presented in brackets for concentrations (in parts per million).

Depth 0 to 120 cm	N inputs		N outputs						
	Fertilizer	Wet Deposition	Total Tile Drainage Loss	Total Groundwater Drainage Loss	Denitrification	Total Plant-N Removal	Change in Residual Soil Mineral N	Change in Residual Groundwater N	N Balance Residual
Fields	....	....	....	....	...kg N ha^{-1}...	....	....	....	....
1 (Cd-Corn)	136.9	1.6	0.0 (± 1.6)	0.5 (± 37.5)	4.3	298.1	-27.3	-0.8	-132.5
2 (Ucd-Corn)	136.9	1.6	1.3 (± 4.1)	0.1 (± 4.7)	7.1	246.0	-46.5	-0.1	-62.7
3 (Cd-Corn)	136.9	1.6	0.0 (± 1.3)	0.3 (± 3.8)	11.8	185.0	-49.0	-1.0	3.0
4 (Ucd-Corn)	136.9	1.6	0.8 (± 1.4)	1.2 (± 17.1)	0.4	210.2	-90.2	-2.3	18.6
5 (Ucd-Corn)	136.9	1.6	1.7 (± 0.8)	0.1 (± 6.1)	5.6	345.8	-217.8	0.0	8.3
6 (Cd-Corn)	136.9	1.6	2.8 (± 0.9)	1.8 (± 50.6)	26.5	295.6	-121.5	0.0	-40.4
7 (Ucd-Corn)	136.9	1.6	1.3 (± 0.6)	0.3 (± 5.8)	0.0	256.6	-29.1	0.0	-90.8
8 (Cd-Corn)	136.9	1.6	0.3 (± 1.1)	0.3 (± 27.7)	0.0	285.6	-3.6	0.0	-144.4

3.8 Appendix

3.8.1 Appendix A

Hvorslev Equation

$$\frac{H - h(t)}{H - H_0} = \exp(-t/T_0) \quad \text{with} \quad T_0 = \frac{r^2 \ln(L/R)}{2 L K_{sat}}$$

to

$$K_{sat} = \frac{r^2 \ln(L/R)}{2 L K}$$

where **H** is the static hydraulic head (m), **H_o** is the hydraulic head when the bailer was removed (m), **h(t)** is the hydraulic head during the recovery (m), **t** is the time elapsed since the bailer is removed (sec), **T_o** is the basic time lag (sec), **K_{sat}** is the hydraulic conductivity at saturation (m sec^{-1}), **r** is the radius of the well casing (m), **R** is the radius of well screen including sand-pack (m), in our case this was equal to the radius of the well casing, and **L** is the length of the well screen (m) which, in our case, is the length of the well in the aquifer.

3.8.2 Appendix B

Dupuit-Forchheimer

$$\boxed{Q=\frac{1}{2}K_{sat}\frac{(h_1^2-h_2^2)}{L}}$$

where **Q** is the flow per unit area (m^3 ha^{-1}), **K$_{sat}$** is the hydraulic conductivity at saturation (m sec^{-1}), **h$_1$** is the head in the well (m), **h$_2$** is the head in the stream (m) and **L** is the distance between well and stream (m). Vertical movement was assumed to be negligible due to the shallow underlying clay layer (Delleur, 2007).

3.8.3 Appendix C

Nitrous Oxide (N_2O-N) calculations in non-steady-state (NSS) chambers

$$\boxed{N_2O - N\ Flux\ (\mu g\ m^{-2}\ hr^{-1}) = \frac{m\,P}{(R\,T)} \times \frac{60\,\text{min}}{hr} \times \frac{V}{A} \times \frac{28.02\ \mu g\ N}{\mu Mol}}$$

where **m** is the linear slope of ppmv·min^{-1} (or µL-N_2O·min^{-1}·L^{-1}), **P** is the mean atmospheric pressure during GHG sampling (atm), **R** is the ideal gas constant (0.082057 µL·atm·$(K·µmol)^{-1}$), **T** is the in-situ chamber air temperature converted into Kelvin; (273.14 + °C), **(60min / hr)** is the factor to convert from minutes to an hourly basis, **V** is the chamber headspace volume (L), **A** is the chamber surface area (m^2) and **(28.02 µg N / µMol)** is the factor to convert from N2O to N basis. Fluxes were converted to kilogram per hectare per day (kg·ha^{-1}·day^{-1}).

CHAPTER 4

GENERAL SUMMARY AND CONCLUSIONS

This thesis is divided into two review-ready articles (chapter 2 and 3). The objectives of this thesis are to (1) quantify water quality improvements associated with excluding pasturing cattle from streams (chapter 2) and (2) characterize nitrogen mass balance in managed subsurface tile drainage (chapter 3). The purpose of this fourth chapter is to summarize the information contained in the two previous chapters and to give general recommendations based on the study findings.

Based on *E. coli* and *Cryptosporidium* sourcing analysis, adult cattle were found to be the main source of fecal contamination in both the protected and unprotected part of the stream; however wildlife was also an extra fecal contamination source in the protected area. Generally, no significant difference was observed between treatments, yet the rank sums have shown that bacteria and nutrient loads are greater in the unprotected part of the stream. Upstream agricultural activities seem to have a great impact on our results. Spring and fall on-field manure applications could leach through tiles and end up in the stream, this is where the managed subsurface tile drainage could reduce the contamination. A longer-term study with subsurface tile drainage could result in a better protection as we would learn to better manage the tile flows. The riparian buffer and stream were observed to be a habitat for snapping turtles, frogs, muskrats and beavers, therefore, there is a need to conserve and protect this area. More studies should be conducted on the buffer strips regarding biodiversity to determine the benefits of protecting this area.

Generally, the study showed that managed subsurface tile drainage reduces the amount of nitrogen lost through the tiles, increases plant uptake, does not load groundwater to a large extent and does not seem to increase denitrification. During drier years nitrogen tile loads is not impacted although plant uptake and corn yields are greater in some of the cases. More measurements of mineral nitrogen and denitrification are needed to get a better estimation of nitrogen loss through nitrogen dioxide from the soil. Organic nitrogen in soil should be measured and accounted for in the mass balance as it seems to play a very important role. Plant nitrogen uptake is the main nitrogen taker followed by denitrification. In order to better estimate denitrification, field measurement values should be taken at higher volumetric water content and more frequently. The system seems to function well, although long-term studies should be conducted to determine the viability of the soil and water quality protection.

Even if the statistical analyses do not show any significant difference between any of the treatments these studies show these practices to be a step forward for the protection of the environment. In order to better manage the system more studies are required regarding wetter years, and the content of organic nitrogen and carbon in soil. With years the systems would be better understood which would, hopefully, show a better environment protection and greater yields, i.e. revenues for the farmers.

APPENDIX

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