DENITRIFICATION AND NITROUS OXIDE DYNAMICS IN THE SOIL PROFILE UNDER TWO CORN PRODUCTION SYSTEMS

By

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NITROUS OXIDE DYNAMICS IN THE SOIL PROFILE UNDER TWO FARMING SYSTEMS

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ABSTRACT

Ph. D. **Abdirashid A. Elmi** **Natural Resource Sciences**

Concerns for environmental quality stimulate the development of various management strategies that mitigate nutrient losses to the environment.

Field experiments were conducted at St. Emmanuel, Quebec, from 1998 to 2000 to investigate the combined effects of water table management and N fertilizer application rates on corn yield, concentrations of NO_3^--N in the soil profile and tile subsurface drainage water, denitrification and N_2O production rates, and N_2O:N_2O+N_2 production ratios in the soil profile. There were two water table treatments: free drainage (FD) with open drains at a 1.0 m depth from the soil surface and subirrigation (SI) with a water table depth of 0.6 m below the soil surface, and two N fertilization rates: 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}) arranged in a split-plot design. Compared to FD, subirrigation reduced NO_3^--N concentration in the soil by up to 50% and in drainage water by 55 to 73%. Water table had little effect on corn yield during the study period. Greater denitrification rates under SI were not accompanied with greater N_2O emissions as ratios of N_2O:N_2O+N_2 were lower under SI than in FD plots. Denitrification rate, N_2O emissions, and their ratios were unaffected by N rate.

A second field experiment was initiated from 1999 to 2000 to assess impacts of tillage systems on NO_3^- -N, denitrification, N_2O, and ratios of denitrification end-products (N_2O:N_2O+N_2). The experiment was conducted on long-term momocropped corn experimental plots under conventional tillage (CT), reduced tillage (RT), and no-till (NT), located at the Macdonald Research Farm, McGill University. Soil NO_3^--N concentrations tended to be lower under RT than under NT or CT. Denitrification and N_2O were similar among tillage systems.

Approximately 50% of soil denitrification activity was measured within the 0.15-0.45 m soil layer. Consequently, we propose that sampling the 0-0.15 m soil layer alone, as is usually done, may not give an accurate picture of soil denitrification activity. Dissolved organic carbon concentrations remained high in all soil depths sampled, but was not affected by water table, N rate or tillage system.

RÉSUMÉ

Ph. D. Abdirashid A. Elmi Natural Resource Sciences

La question environnementale a stimulé le développement de stratégies visant à réduire la pollution d'origine agricole.

Des essais en champs ont été menés à St-Emmanuel, Québec, de 1998 à 2000 pour vérifier l'effet de pratiques de gestion de la nappe phréatique (GNP) et de la fertilisation azotée (N) sur le nitrate dans le profile du sol et l'eau de drainage, sur le rendement du maïs, ainsi que sur la dénitrification et la production absolue et relative du N_2O. Deux traitements de GNP ont été utilisés: le drainage libre (DL) avec drains à 1 m et l'irrigation souterraine (IS) avec maintient de la nappe à 0.6 m, ainsi que deux niveaux de fertilisation: 120 kg N ha^{-1} et 200 kg N ha^{-1}.

Les concentrations en NO_3^--N du sol sous IS était de 55 à 73 % plus basses que sous DL. La GNP eu peu d'effet sur le rendement. Ni la concentration en NO_3^--N du sol et de l'eau de drainage, ni le rendement du maïs n'ont été influencés par la fertilisation. La dénitrification plus importante sous IS que sous DL n'était pas nécessairement accompagnée par une production de N_2O plus importante car la proportion du N_2O dans le gaz de dénitrification y était plus basse. Ni la dénitrification, ni la production absolue et relative du N_2O n'ont été influencés par la fertilisation.

Un deuxième essai conduit de 1999 à 2000 sur des parcelles longue-durée en monoculture de maïs grain de la station de recherche agronomique de l'Université McGill, a défini l'effet du travail du sol réduit (TR), du semis direct ou du travail conventionnel sur le niveau de NO_3^--N du sol, la dénitrification, et la production absolue et relative du N_2O. Le NO_3^--N étaient moins abondant sous TR. La dénitrification et la production de N_2O n'étaient pas influencées par le mode cultural.

L'abondance du N_2O (50%) dans la couche 0.15-0.45 m du sol suggèrent que l'échantillonage de la couche 0-0.15 m qui se pratique couramment, ne décrit pas adéquatement le processus de dénitrification des sols. Le carbone organique dissoult étaient

abondant dans tout le profile et et sans égard à la GNP, la fertilisation azotée ou la pratique culturale.

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First I wish to express my sincere thanks to Dr. Chandra Madramootoo and Dr. Chantal Hamel, my thesis supervisors, for patiently and skillfully guiding me through my research and thesis preparation. They listened and corrected me while I spoke incoherently. Many thanks. I am extremely grateful to those who took time out of their tight schedules and reviewed part or all of the manuscript; most especially Drs. Joann Whalen, Jim Fyles, Brian Driscoll, and Georges Dodds. I owe a special debt to Dr. Joann Whalen. She read and marked up the original version of the complete manuscript. Her pointed criticism, attention to detail, and sound advice gave my editing a sense of direction and helped carve the manuscript into shape. I am also indebted to Drs. Angus MacKenzie, Guy Mehuys, and Suzelle Barrington who served as a member of my comprehensive examination committee. I heartily acknowledge Dr. Jim Fyles's helping hand, particularly at the beginning of my study. His office was the first place where I used to run, often panicking, whenever I detected a serious problem with the GC machine for N_2O analysis, and he always had an open-door policy and smile for me.

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Many thanks go to my fellow students and staff whose kindness and hospitality made my graduate studies at McGill pleasant. In addition to science, we learn and respect differences in culture, religious and spiritual beliefs, philosophy, and lifestyle. I suspect I've learned at least as much from my colleagues as they have from me.

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My long time friends deserve recognition here for their friendship and encouragement: Dr. Abdirahman Beileh, my former teacher Dr. Hussein Warsame, Bashir Good, Mohamed O Musa, (Ph. D. Candidate, University of British Columbia), Mohamoud H Egeh (Ph. D. Candidate, Iowa State University), Musa Farah, Said A. Ahmed, and many others whose names not being listed here will not diminish their role in completing this work.

Finally, I should like to acknowledge the debt I owe to my family in supporting me during the extended period over which this work was done and in forgiving me for spending the time involved that would otherwise have been devoted to them. I owe an unpayably large debt to my wife, Fahima Tani, also a graduate student at Macdonald College of McGill University. With her wise judgements, she freed me from the thickets of responsibility, even at the temporary cost of marital harmony.

DEDICATION

This work is dedicated to my daughter and son, Ifrah and Osman, in the hope that it will encourage them in all that they do.

For inspiration, I wish to dedicate this work to Gen. Yusuf Tallan Ali (1935-2000) who planted a culture of hope and optimism in my heart.

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MANUSCRIPT AND AUTHORSHIP

This thesis has been written in the form of manuscripts submitted or to be submitted to scientific journals. This format has been approved as outlined in the "Guidelines Concerning Thesis Preparation" by the Faculty of Graduate Studies and Research, "must be cited in full in the introductory section of any thesis to which it applies:

"Candidates have the option of including, as part of their thesis copies of the text of paper(s) submitted or to be submitted for publication, or the clearly-duplicated text of a published paper(s). These text must be bound as an integral part of the thesis".

"**If this option is chosen, connecting texts that provide logical bridges between the different papers are mandatory.** The thesis must be written in such a way that it is more than a mere collection of manuscripts; in other words, results of a series of papers must be integrated.

"The thesis must still conform to all other requirements of the "Guidelines for Thesis Preparation". **The thesis must include, as separate chapters or sections:** (1) a table of contents, (2) a general abstract, in English and French, an introduction which clearly states the rationale and objectives of the study, a comprehensive review of the literature, and final conclusion and/or summary, and a thorough bibliography or reference list.

"Additional material must be provided where appropriate and in sufficient detail (e. g., in appendices) to allow a clear and precise judgement to be made of the importance and originality of the research reported in the thesis.

"In the case of manuscripts co-authored by the candidate and others, **the candidate is required to make an explicit statement in the thesis to who contributed to such work and to what extent.** Supervisors must attest to the accuracy of such statement at the doctoral oral defense. Since the task of the examiners is made more difficult in these cases, it is in the candidate's interest to make perfectly clear the responsibilities of all authors of the co-authored papers. **Under no circumstances can a co-author of any component of such a thesis serve as an examiner for that thesis".**

This is presented as a series of papers prepared for publication; **all of them co-authored by myself as a lead author and my supervisors; Dr. Chandra Madramootoo,**

Professor within the Department of Agricultural and Biosystems Engineering, and Dr. Chantal Hamel, Professor within the Department of Natural Resource Sciences. The research was planned and conducted under the responsibility of the student, with technical assistance as noted in the acknowledgments. The manuscripts were written by the student. My co-authors provided supervisory guidance and funds from the onset of this study, arranged technical assistance during field and laboratory operations, offered many valuable suggestions and reviewed all of the manuscripts.

CHAPTER 1

1 GENERAL INTRODUCTION

1.1 Statement of the Problem

Sustainable farming systems have attracted much attention in recent years. As non-point source pollution and soil erosion due to conventional farming practices continue to raise environmental concerns (Bouwer, 2000; Hussain et al., 1999), the need to identify best management practices that will ensure the continued productivity of agricultural lands while minimizing adverse environmental impacts is being recognized. Natural drainage is not sufficient in humid regions, especially in areas with fine-textured soils. Wet soil conditions decrease root respiration in plants, which in turn impedes growth and reduces yields (Evans et al., 1996). Subsurface drainage enhances the productivity of poorly drained lands by lowering the water table and improves root aeration and, hence crop growth. However, this technology has been found to play a major role in the transport of plant nutrients and other agrochemicals from drained lands and then discharged into adjacent surface waters (Randall and Mulla, 2001; Patni et al., 1998; Milburn et al., 1997).

In recent years, water quality monitoring has demonstrated that nitrate-N (NO_3^--N) is the most ubiquitous pollutant in surface and subsurface waters, and that the level of contamination is increasing (Spalding and Exner, 1993; Smith et al., 1987). Nitrate-N is highly water-soluble and therefore migrates easily to the subsurface drainage systems and then discharged into adjacent surface waters. This could worsen water quality problems by causing eutrophication and by contaminating groundwater. In addition, NO_3^--N contamination of groundwater is of serious concern to society because of its potential health hazards to humans and animals (Gelberg et al., 1999; Prasad and Power, 1995; Comly, 1945).

Water table management (WTM) has been identified as a best management practice that reduces NO_3^--N pollution of surface and ground waters while sustaining or increasing crop yield (Cooper et al., 1999; Evans et al., 1996; Kalita and Kanwar, 1993; Madramootoo et al., 1993). Water table management consists of two main alternatives: controlled drainage (CD) and subirrigation (SI). Under CD, water is prevented from exiting the soil profile at the drain outlet by means of plugging or raising the drainage outlet. Subirrigation is achieved by installing a control mechanism at the drain outlet and supplying water to the drainage system during drought periods in order to maintain an elevated water table depth in the field. Besides creating a favorable crop growing environment, WTM technology reduces NO_3^--N pollution problems by restricting the volume of drainage and by creating anaerobic conditions which enhance denitrification. In Ontario, Drury et al. (1997) reported that a CD-SI system reduced NO_3^--N concentration in tile drainage water by 43% compared to free drainage (FD) systems. A possible environmental consequence of CD-SI systems, however, may be increased nitrous oxide (N_2O) emission to the atmosphere through denitrification.

Tillage is another management practice that can have a major impact on nutrient dynamics in the soil profile. Tillage practices directly affect the soil-water properties of the surface soil and, thus, water movement characteristics. Continuous use of conventional tillage (CT), for example, can accelerate the decomposition of soil organic matter and lead to the deterioration of the soil structure (Hussain et al., 1999; Martel and MacKenzie, 1980). As a consequence, the risk of erosion and nutrient loss associated with surface runoff increases, which may stimulate the eutrophication of receiving water bodies. Furthermore, soil erosion is a major form of environmental degradation, reducing the fitness of soil and its productivity that may ultimately threaten the long-term sustainability of food production systems. Conservation tillage practices, including no-tillage (NT) and reduced tillage (RT), have been widely advocated as alternatives to CT practices because they improve soil and water quality, and reduce labor and energy costs (Hussain et al., 1999; Uri, 1999). In addition, more crop residue is retained on the surface under NT and RT than CT, protecting soil against raindrop impacts, reducing runoff and erosion, improving surface water quality, and enhancing water infiltration to benefit crops during dry or low rainfall periods (Ogden et al., 1999).

One disadvantage of NT and/or RT systems is that there is less mixing of applied fertilizers and crop residue into the soil surface, reducing the amount of fertilizer reaching the soil. This situation may lead to the application of greater rates of fertilizers, N in particular (Fawcett, 1987). Concern has been raised about the impact of increased fertilizer and herbicide use associated with conservation tillage practices, especially NT, on surface and ground water quality (Elliot and Efeltha, 1999; Edward et al., 1993). Formation of macropores and reduced runoff under NT or RT may enhance NO_3^--N leaching to groundwater. Increase in N_2O production under NT due to increased soil moisture content and organic C is an additional problem in some soils (Fan et al., 1997; Doran, 1980).

Nitrous oxide produced during denitrification is of serious concern as it contributes to greenhouse effects (Smith, 1990) and participates in the depletion of the ozone layer (Mooney et al., 1987). This concern has emphasized the need for information about the proportion of denitrification gases entering the atmosphere. To properly assess the ecological impact of the denitrification and subsequent release of N_2O, research leading to understanding of the proportion of N_2O:N_2O+N_2 gases entering the atmosphere is essential. Little information is available at present on the N_2O:N_2O+N_2 ratio under field conditions. The few studies reported so far were performed on soil columns under controlled conditions (e. g., Jacinthe et al., 2000; Drury et al., 1997; Yeomans et al., 1992).

Sustainable agricultural management should involve measures to minimize NO_3^--N leaching and N_2O emissions without concomitant reduction in yield. The primary aim of the research presented in this thesis was to generate information leading to achieve sustainable crop production and environmental goals.

1.2 Description of the Study

Field experiments were conducted on an experimental setup at St. Emmanuel, about 30 km southwest of Macdonald Campus of McGill University, from 1998 to 2000 to study the effects of water table and N management practices on the dynamics of NO_3^--N, denitrification, N_2O, and ratios of denitrification end-products the soil profile and grain corn yield. A second field experiment was initiated from 1999 to 2000 on a long-term experimental plots located at the at the Macdonald Research Farm, McGill University, to assess impacts of tillage systems on NO_3^--N, denitrification, N_2O, and ratios of denitrification end-products. In the first phase of this thesis, I examined total denitrification rates and N_2O fluxes at the soil surface (0-0.15 m), NO_3^--N concentrations dynamics in the soil profile (0-0.75 m), and NO_3^--N concentrations in the drain discharge during 3 cropping seasons (St. Emmanuel site). I expected that denitrification rates from the 0-0.15 m depth would vary among the agroecosystems investigated, based on information available in the scientific literature. I was not able to find reports on the quantities of N_2O+N_2 and N_2O produced in the subsoil (<0.15 m) and their ratios. Therefore, I focused the second phase of my research on N transformations at the 0-0.45 m depth of the soil profile. I was particularly interested in estimating the $N_2O:N_2O+N_2$ ratio during denitrification at different depths (0-0.15 m; 0.15-0.30 m; 0.30-0.45 m) in the soil profile at both sites.

1.3 Objectives

The specific objectives of my study were to:

1) Assess the effectiveness of controlled drainage/subirrigation (SI-CD) in reducing NO_3^--N concentrations in the soil profile and subsurface tile drainage water.

2) Define impacts of water table and different N fertilizer rates (120 kg N ha^{-1} and 200 kg N ha^{-1}) on denitrification and N_2O emission rates .

3) Evaluate combined effects of water table and different N fertilizer rates (120 kg N ha^{-1} and 200 kg N ha^{-1}) on grain corn yields.

4) Assess effects of long term tillage practices on NO_3^--N dynamics, denitrification and N_2O emissions in both surface and subsurface soils.

5) Estimate the ratio of $N_2O:N_2O+N_2$ during the growing seasons.

1.4 Hypotheses

This study tests the following four main hypotheses.

1) Controlled drainage-subirrigation (St. Emmanuel site) is expected to produce greater denitrification rates than free drainage because of higher soil moisture content.

2) No-till and RT systems (Macdonald Research Farm) produce greater denitrification rates than CT due to the higher water retention associated with the crop residue on the soil surface.

3) The proportion of N_2O emitted in denitrification gases ($N_2O:N_2O+N_2$) is lower under SI than under FD, and under NT and RT than under CT, because N_2 is expected to be the main end-product of denitrification in these treatments.

4) It is hypothesized that NO_3^--N concentrations in the soil profile is lower under SI compared to FD as a result of increased denitrification and/or crop uptake, but higher in NT and RT plots than CT plots.

1.5 Method of Thesis Presentation

Chapters 1 and 2 present the general introduction and literature review. The results of these field research experiments are reported in Chapters 3, 4, 5, and 6 in the form of four papers with connecting text. All the literature cited in the thesis are listed in the reference section at the end of this thesis. The format has been changed to be consistent within this thesis. Tables and figures are presented at the end of each chapter.

CHAPTER 2

2 LITERATURE REVIEW

2.1 Importance of Nitrogen

Nitrogen is a key element in plant nutrition. Food grain production over the past few decades has kept pace with population growth in a large part because of increased use of fertilizers, and particularly of N fertilizer (Constant and Sheldrick, 1992). Most forecasts indicate that global population will increase, and crop yields will have to increase correspondingly to meet food demand. It is expected that more N mineral and animal manure fertilizers will be applied to agricultural lands to achieve these future yield goals. In the province of Quebec, for example, N fertilizer use increased nearly three-fold between 1976 and 1996 (Statistics Canada, 1996).

High yielding crops, like corn (*Zea mays L.)*, require large amounts of N fertilizer to ensure optimum yield. Liang et al. (1992) reported a maximum grain corn yield of 15.2 Mg ha^{-1} resulting from an optimal combination of hybrid, high population density, high N fertilizer rate and irrigation. In an attempt to maximize yields, excessive amounts of N fertilizer are often applied in corn producing farms. Tan et al. (1996) pointed out that high fertilizer rates used with sound field management practices and favorable climatic conditions can increase crop yield. When, however, high N fertilizer rates are applied and management practices are poor, NO_3^--N accumulation in the soil profile and subsequent leaching to groundwater may result (Tan et al., 1996; Angle et al., 1993; Roth and Fox, 1990). Clearly, it is important to select N application rates that are realistic for management schemes under which the crop is to be grown.

The objective of agriculture has always been to feed populations. While this objective is still vital and necessary, it alone is not sufficient, and a number of other objectives, including minimizing adverse environmental impacts, must now be met.

2.2 Nitrate Pollution Problems

Concerns for environmental impacts associated with NO_3^--N began to emerge in the 1940s (Comly, 1945), but grew rapidly in the 1980s and 1990s. Movement of NO_3^--N below the root zone of crops has the potential to increase the NO_3^--N content of groundwater. Elevated levels of NO_3^--N have been found in wells in many areas of USA and Canada, giving rise to health and environmental concerns (Gelberg et al., 1999; Patni et al., 1998). High concentrations of NO_3^--N in drinking water have been linked to condition known as methemoglobinemia (also known as blue baby syndrome) that occurs when nitrite (NO_2^-) forms in the newborn infant's intestine (Comly, 1945). This condition has been fatal in many cases (Johnson et al., 1987). It can be concluded from the above publications that concerns over NO_3^--N increases are legitimate due to the potential ill effect of NO_3^--N on health of humans, particularly young children. As a safeguard against methemoglobinemia, many countries, including Canada, have a drinking water standard limit of 10 mg NO_3^--N (N in NO_3^- form) L^{-1} or 45 mg NO_3^- L^{-1} (Health Canada, 1996) for municipal water supplies.

In addition to health concerns, increased levels of NO_3^--N in lakes and rivers may cause excessive growth of aquatic plants. Along with phosphorus, N is often a limiting element in aquatic ecosystems. Its introduction into such ecosystem may significantly contribute to eutrophication of surface waters by stimulating algae growth (Spalding and Exner, 1993). Eutrophication is the biological response of water to over-enrichment by plant nutrients. Dissolved oxygen levels drop drastically when algae die and start to decompose, which can result in fish kills and loss of biodiversity in aquatic ecosystems (Carpenter et al., 1998). Certain algae species produce toxins which can be dangerous to health, making waters unfit for human consumption (Pitois et al., 2001). Eutrophication, thus, not only can impair the aesthetic qualities of the water, but also prohibits the use of water for drinking supply, fisheries, and recreation.

There is also an economic dimension to the NO_3^--N pollution problem. If NO_3^--N in a public water supply reaches or exceeds the safety standard limit, expensive water treatment measures may become inevitable.

Growing public awareness has led to the perception that NO_3^--N contamination of surface and groundwater is linked to the large increase in N fertilizer used in agricultural ecosystems to enhance food production. This perception has been supported by research findings that established a close relationship between N fertilizer use and NO_3^--N concentrations in water that exceed the drinking water limit in many agricultural regions of North America (Patni et al., 1998; Madramootoo et al., 1992; Hubbard and Sheridan, 1989; Smith et al., 1987). David et al. (1997) indicated that NO_3^--N contaminated drainage water from subsurface drainage systems in agricultural regions is a primary source of NO_3^--N loadings to surface waters. As a result, public concern over the impact of agricultural activities on water quality may escalate, and farming systems should be improved to prevent further degradation of water quality. Furthermore, NO_3^--N not used by crops is subject to denitrification, which may result in N_2O emission from soils that can impact on the ozone layer and contribute to the greenhouse effect (Smith, 1990; Mooney et al., 1987).

2.3 Denitrification as a Natural Bioremediation for Nitrate Pollution

The major losses of N from soils are due to crop removal and leaching (Roth and Fox, 1990). When, however, soils become saturated with water, O_2 is restricted or excluded and anaerobic reactions take place. Some facultative anaerobic bacteria have the ability to use nitrite (NO_2^-) and nitrate (NO_3^-) as electron acceptors instead of O_2 (Aulakh et al., 1992). They obtain energy from the oxidation of organic compounds with the accompanying release of N_2 and N_2O. This process is called biological denitrification. Denitrification is an anaerobic process that converts the nitrogen oxides NO_3^- and NO_2^- to the nitrogen gases N_2O and N_2 in the following sequence:

$$NO_3^- \rightarrow NO_2 \rightarrow NO \rightarrow N_2O \rightarrow N_2 \qquad (1)$$

$$NO_3^- \rightarrow NO_2 \rightarrow NO \rightarrow N_2O \qquad (2)$$

Where high NO_3^--N levels in soil-water pose a pollution hazard, reduction of NO_3^--N to N_2 (eq. 1) is a desirable process by which to lower NO_3^--N levels. Nitrous oxide emissions from agricultural soils and the relative proportion of the gases (N_2O:N_2O+N_2 ratio) evolved are influenced by a variety of environmental conditions (temperature, rainfall), management practices (tillage, fertilization, irrigation), soil properties (pH, texture, bulk density, organic matter), and the size of denitrifying microbial communities in the soil (Burton et al., 1997; MacKenzie et al., 1997; Granli and Bøckman, 1994). Depending on prevailing environmental conditions NO_3^--N may not be completely reduced to N_2 and intermediates of NO and N_2O

may escape into the environment (eq. 2). Under this situation, denitrification can be a significant source of N_2O. This topic is given further consideration in section 2.10.

2.4 Ecology of Denitrifying Community

The size of denitrifying community in the soil is largely determined by the amount of available C substrate. There are large populations of denitrifying organisms in arable soils, especially in the vicinity of plant roots (Richards and Webster, 1999). Exudates produced from actively functioning plant roots are believed to support the growth of denitrifying bacteria in the rhizosphere by providing them with C as source of energy (Klemedtsson et al., 1987; Shahrawat and Keeney, 1986). The important relationship between soil organic C and denitrification was established decades ago (Bremner and Shaw, 1958). Despite the passage of nearly five decades, the interaction between plant roots and denitrification are not yet well defined. Rudaz et al. (1999) reported higher denitrification rates at the end of a growing season. They postulated that this could be related to the presence of higher amount of available C derived from dead roots at the end of the growing season, which would stimulate the growth of denitrifying populations and the synthesis of denitrifying enzymes. Other authors (e.g., Mahmood et al., 1997) reported that plant growth increased denitrification by reducing O_2 availability. However, Haider and Heinemeyer (1987) have shown a negative effect of growing plant roots on denitrification rates. This may be due to NO_3^--N depletion by root uptake of N and reduction in soil moisture content by transpiration (Bakken, 1988). These conflicting results may be attributed to the strong sensitivity of denitrification to environmental conditions including carbon and nitrate substrate supply and oxygen concentration in the vicinity of the denitrifiers. More research is needed in this area.

2.5 Measurement of Denitrification

To obtain reliable estimates of denitrification in soil, precise and sensitive measurements of the total N_2O and N_2 evolution from the soil are necessary. The acetylene inhibition method (AIM) (Yoshinari et al., 1977) has been widely used during the last two decades to determine denitrification rates from soil in field and laboratory studies. Since the development of AIM, denitrification studies in agricultural systems have increased considerably. This technique has been reviewed by many authors, including Malone et al., (1998), Bollman and Conrad (1997), and Aulakh et al. (1991). Acetylene (C_2H_2) inhibits the enzymatic reduction of N_2O to N_2. Acetylene concentrations of 1-10% in air (vol/vol) are usually sufficient for inhibition to occur, causing only N_2O to be evolved (Granli and Bøckman, 1994). Total denitrification rate ($N_2O + N_2$) and N_2O can then be measured as the amount of N_2O gas produced in soil treated with and without C_2H_2, respectively. Measurement of N_2O evolution rates using gas chromatography (GC) is an economical and practical technique (Aulakh et al., 1991).

2.5.1 Soil Core Method

Several methods have been used for direct in-field measurement of denitrification. Soil core incubation (Ryden et al., 1987) and closed chamber (Rolston et al., 1978 and Ryden et al., 1979) with or without C_2H_2 incubation are the most popular methods to study denitrification. Soil core method with the C_2H_2 inhibition technique offers a particularly versatile approach to the direct field measurement of denitrification N loss. The method is sufficiently simple that it can be used for routine monitoring of denitrification. These features make the technique much more generally applicable than other methods. However, drawbacks with the soil core method include: 1) damage to the experimental plots through sample removal, 2) spatial variability due to limited sample area, and 3) soil sampling process may disturb soil, altering bulk density, aeration, and C_2H_2 diffusion into the soil and N_2O diffusion out of the soil into the headspace.

2.5.2 Limitations of Acetylene Inhibition Method (AIM)

While the C_2H_2 inhibition method is technically simple to operate, inexpensive and able to measure cumulative denitrification rates over 24 h, there are indications that the AIM may underestimate denitrification. Measurement of denitrification loss in soil cores assumes that the depth of cores (mostly between 0-0.15-m) represent the zone where denitrification occurs. Hence, N losses from deeper depths are not accounted for. In my study, I attempted to overcome this limitation by measuring denitrification in lower depths (0-0.45 m) of the soil profile. Furthermore, the effectiveness of C_2H_2 varies with soil type. Achieving a uniform distribution of C_2H_2 throughout the pore space is necessary to effectively inhibit N_2O and C_2H_2 diffusion takes longer in soils with high clay or organic matter (OM) content than other soils (Ryden et al., 1987). Though denitrification is often considered to be the major source of N_2O emissions from soil, such emissions can be produced during nitrification (Bremner and Blackmer, 1978). If nitrification is an important mechanism for N_2O production, the AIM may underestimate the total N_2O emissions from soil since N_2O from nitrification cannot be accounted for because C_2H_2 also inhibits nitrification. Despite these limitations, the C_2H_2 inhibition technique continues to have several useful applications, the most obvious of which is its ability to determine the natural denitrification rates of soil samples.

2.6 Water Table Management Strategy

Water table management practices can be grouped into three types: 1) free drainage, 2) controlled drainage, and 3) controlled drainage-subirrigation. Free drainage (FD) alone lowers the water table during wet periods. Controlled drainage (CD) is achieved by placing a control structure, such as flashboard riser in the outlet ditch or subsurface drain outlet, to control the rate of subsurface drainage. Controlled drainage-subirrigation (CD-SI) is similar to the CD system, except that supplemental water is pumped into the system during the growing season to maintain the water table at a desired level. Drainage is provided during wet periods by allowing excess water to flow over the control structure, which may be adjusted in elevation depending upon the rainfall. As a result of intensive rains during growing season, the shallow water tables that result from CD-SI can leave fields vulnerable to flooding. To prevent this, weather forecasts may be used to evaluate the best time for lowering the water table. Knowing when to reverse from drainage mode to the subirrigation mode requires experience as well as monitoring soil moisture content. Maintaining a depth of water table low enough to prevent aeration problems and high enough to permit capillary movement into the rooting zone for plant uptake poses a challenge.

2.6.1 Environmental Implications

Protecting water quality is the key starting point for ensuring sustainable freshwater supply for drinking and food production. Water table management practices have been shown to be an effective technology for reducing NO_3^--N leaching to groundwater and surface waters via subsurface drainage as N loss through denitrification is enhanced under shallow water tables (Jacinthe et al., 1999; Wright et al., 1992). By keeping the water table at a shallow depth, soil remains saturated and O_2 diffusion in soil pores is restricted, creating reducing conditions that promote NO_3^--N loss by denitrification. Jacinthe et al. (1999) studied the effects of various water table depths on NO_3^--N concentrations and concluded that NO_3^--N concentrations were reduced when water tables were maintained at shallow depths. They reported a 40% reduction in NO_3^--N using WTM technology. Similarly, Kalita and Kanwar (1993) demonstrated that raising the water table depth from 0.9 m to 0.3 m below the soil surface resulted in a proportional decrease in NO_3^--N concentrations in the soil with the lowest average concentrations of NO_3^--N over the season in groundwater being recorded under a 0.3 m water table depth. Furthermore, Gilliam and Skaggs (1986) estimated about a 32% decrease in NO_3^--N leaching due to controlled drainage. It can be concluded from these findings that where high NO_3^--N concentrations in the soil-water system pose health and environmental hazards, one way to reduce NO_3^--N pollution may be to enhance denitrification.

It must be emphasized, however, that there are environmental consequences to increasing denitrification if it increases N_2O emissions to the atmosphere. Nitrous oxide is a powerful atmospheric trace gas, with an atmospheric lifetime of nearly 120 yr and a global warming potential 320 times that of CO_2 (IPCC, 1995). Furthermore, N_2O participates in depletion of the ozone layer (Mooney et al., 1987). The depletion of the ozone increases the strength of harmful ultraviolet at the earth surface and may affect many parts of the ecosystem. Fortunately, N_2O is not the end product of denitrification as it may further be reduced to N_2, which has no known deleterious impact on the atmosphere. Knowledge of N_2O and N_2 fluxes and their ratios under natural conditions is needed to devise management practices that reduce NO_3^--N pollution without concomitantly increasing pollutant N_2O in the atmosphere. If a major portion of the gases produced is N_2O, then application of WTM techniques could increase atmospheric N_2O loading, and an attempt to solve the NO_3^--N pollution problem may, instead, lead to another pathway of pollution. If N_2, rather than N_2O, is the predominant gas, then WTM techniques would represent environmentally sound ways of removing NO_3^--N from the soil. Jacinthe et al. (2000) estimated a relatively high mole fraction of N_2O (i.e., N_2O:N_2 ratio) associated with WTM and concluded that the impact of this practice on the global N_2O budget will depend on the total area where WTM practices would be applicable. Other workers have reported that N_2 is the dominant end product of denitrification (Drury et al., 1997; Maag and Vinther, 1996). All these studies were conducted on soil columns under a controlled environment where conditions can be manipulated. There is insufficient information on N_2O:N_2O+N_2 ratio fluxes under field conditions and factors that control the N_2O to N_2O+N_2 ratio. Therefore, the ecological significance of gaseous denitrification products with WTM under field conditions is still largely a matter of speculation. Sustainable agricultural management practices should involve measures to minimize NO_3^--N and N_2O without a concomitant reduction in yield.

2.6.2 Agronomic Aspects

Acceptance of new technologies by farmers largely depends on their impact on crop yields. With the establishment of a reasonably shallow water table depth (WTD), water availability to crop plants is increased, which can lead to higher crop yields. Drury et al. (1997) and Skaggs et al. (1999) suggested that raising the water table generally increased evapotranspiration and, hence, yield. Optimum WTD is a function of crop species and soil types. Broughton and Madramootoo (1995) indicated that WTDs ranging from 0.6 m to 0.8 m produced maximum soybean yield. Kalita and Kanwar (1993) showed that when corn was grown on loam and silty loam soils, the highest yield was obtained at a WTD of 0.6 to 0.9 m and the lowest yield with a WTD of 0.2 to 0.3 m. Similarly, Doty (1980) found that the best water table depth for corn in sands or sandy loams was 0.76-0.89 m. Tan et al. (1996) have reported their highest corn yield to occur with a WTD of 0.6 m with corn yield being reduced by 15% at a WTD of 0.3 m relative to a 0.6 m WTD depth. Wesseling (1974) found that too shallow a WTD reduces oxygen supply to roots, reduces nutrient uptake and crop growth, and restricts rooting volume.

These observations suggest that precise management of the water table is required with SI, particularly during rainy periods/seasons. For controlled drainage/subirrigation systems to be successful, water table depth must be high enough to permit capillary rise into the root zone and low enough to ensure adequate soil aeration.

Subirrigation is expected to be more beneficial than conventional drainage during drier cropping seasons as it supplements rainfall to meet crop evapotranspiration demand. Cooper et al. (1999) recorded a significant yield increase from a SI treatment in 1991, a very dry year, whereas the wetter growing conditions of 1992 resulted in conventional drainage yields not differing from those obtained under SI. These results suggest that in wet growing years there is no significant yield advantage for SI systems.

2.7 Tillage Systems

2.7.1 Conservation Tillage and Agricultural Sustainability

Concerns over declining soil productivity due to intensive tillage and erosion has brought about much interest in conservation tillage farming systems. Conservation tillage has evolved from practices that range from reducing the number of trips over the field to raising crops without primary or secondary tillage. Currently, NT and various forms of RT practices are highly promoted as one of the key means through which soil quality and soil organic matter (SOM) can be maintained (Karlen and Cambardella, 1996). Recently, the concept of soil quality has gained prominence with respect to the sustainability of agriculture. Soil quality is increasingly conceptualized as the major linkage between agricultural conservation management strategies and achievement of the major goals of sustainable agriculture (Parr et al., 1992). A growing body of evidence suggests that a greater number of earthworms and microorganisms are found under no-till than conventionally tilled agroecosystems (Hubbard et al., 1999). Management practices that promote earthworm populations and their activities may provide enormous benefits in terms of soil quality and sustainable productivity. According to Buckerfield et al. (1997), abundance of earthworms generally indicates healthy and productive soils. Conventional moldboard plow tillage destroys earthworm burrows and may kill earthworms in the process. Other benefits of conservation tillage practices include reduced labor and energy costs.

With a few exceptions, practices that enhance one soil quality parameter often also reinforce other parameters. For example, reduction of soil erosion under conservation tillage systems usually maintains or improves soil productivity and surface water quality (Logan et al., 1987; Ogden et al., 1999; Uri, 1999).

2.7.2 Conservation Tillage Systems and Environmental Concerns

In recent years, popularity of conservation tillage including NT has grown steadily in Canada and elsewhere. In the province of Quebec alone, RT increased acreage by 24% (from 103,599.2 to 128,311 ha) and NT by 63% (from 21353.2 to 34889 ha) between 1991 and 1996, with the crop land area under CT declining by about 8.5% (Statistics Canada, 1996). This brings a sense of agricultural revolution.

Reduced or No-till systems that retain crop residues on the soil surface creates a greater reliance on fertilizers because surface crop residue intercepts applied fertilizers, thereby reducing the amount reaching the soil (Fawcett, 1987). Additionally, NT and/or RT may rely more heavily on herbicides to control weeds than CT. Therefore, the continuing increase in the area under various forms of conservation tillage systems, especially NT, raises potential concerns about the impact of increased fertilizer and pesticide use associated with conservation tillage systems on water quality (Edward et al., 1993; Elliot and Afeltha, 1999; Kanwar et al., 1997). Surface residues in NT and/or RT provide protection against surface sealing, promotes water infiltration and decreases surface runoff. Because abundance of continuous macopores under NT and RT increases infiltration, surface runoff is low, but the potential for leaching of NO_3^--N to groundwater is increased (Elliot and Afeltha, 1999; Kanwar et al., 1997; Varshney et al., 1993).

While the likelihood of NO_3^--N leaching is higher under conservation tillage compared to conventional tillage systems, some data are conflicting. Kanwar et al. (1997) have reported higher concentrations of NO_3^--N in tile drainage water under CT than under NT. In Ontario, Patni et al. (1998) reported significantly higher soil NO_3^--N concentrations under CT than NT in samples collected at 0.9 m depth. This finding can be attributed to the lower mineralization and higher denitrification rates in soils under NT (Rice and Smith, 1982; Aulakh et al., 1984). Soils under NT retain greater moisture than those under CT, and the higher moisture content of NT systems is thought to contribute to denitrification and thus reduced NO_3^--N movement down the profile of NT soil (Parkin et al.,1987; Rice and Smith, 1982; Doran, 1980). In contrast, Randall and Iragavarapu (1995) showed that soil NO_3^--N was not greatly affected by tillage in the first few years but the difference between CT and NT began to widen as time progressed. Kanwar et al. (1985) studied the effects of NT and CT, and single versus split N applications on NO_3^--N leaching with subsurface drainage of continuous corn. Significant reduction of NO_3^--N in subsurface water with NT relative to CT was observed, but only after the third year of the experiment. This is an important suggestion that short term results may be misleading, and caution must be exercised if they were to be used to develop public policy. It is possible that after NO_3^--N migrates into NT soil, it may be absorbed and denitrified more quickly than in conventionally tilled soils. This is because NT soils contain more organic material in the surface layer and support a larger and more active microbial population (Aulakh et al., 1984; Doran, 1980). Other workers did not find consistent differences in NO_3^--N leaching from NT and CT systems with the obvious conclusion that the choice of tillage method will have a minor impact on groundwater quality.

Hence, the effects of tillage practices on NO_3^--N concentrations in the subsurface soil-water system have not always been consistent among different investigations. Long-term experiments such as our site which was established a decade ago are perhaps the only way to determine if new agricultural system will sustain agricultural and environmental quality.

2.7.3 Nitrous Oxide Emissions and Tillage Practices

The increase in N_2O production under NT and/or RT during denitrification processes is an additional concern. Doran (1980) indicated that with the build up of organic matter on the soil surface of NT and/or RT fields there was an increase in the number of microorganisms capable of reducing NO_3^-, suggesting the potential for greater N_2O emissions. Mummey et al.

(1998) found that the NT scenario had on average greater denitrification fluxes than the CT scenario. This finding is consistent with other studies suggesting that NT or RT systems generally have higher denitrification rates than CT (Burton et al., 1997; Mackenzie et al., 1997; Linn and Doran, 1984). Mackenzie et al. (1998) and Aulakh et al. (1992) reported increased denitrification under NT compared with CT. They felt that this effect was due to denser soil and higher moisture contents with NT.

2.8 Tillage Effects on Crop Yield

Although agronomic impacts of tillage systems was not part of my study, I felt it was important to address this issue in this literature review chapter by summarizing information available in the literature.

The adoption of new production technology by farmers always involves a degree of risk and uncertainty concerning its impact on output. The decision to change production technology is based on many factors. Farmer's risk perceptions, level of managerial expertise, and local climate condition have all been identified as important factors affecting farmers choice among cropping systems (Uri, 1999). Tillage effects on crop yields depend on cropping systems including amount and characteristics of crop residue, soil type, and local climatic conditions. There are numerous reports comparing the effects of NT practices with RT and CT systems for corn production, most of them concluding that NT or RT can improve crop yields. This may be attributed in part to the higher moisture content often found in NT soils that can benefit crops during low rainfall periods. However, in cool and wet climate regions, soil warming in Spring and seedling emergence may be delayed in systems that leave high levels of residue (Burgess et al., 1996). Epplin et al. (1994) reported that the lowest wheat grain yield was obtained in fields under NT and that yield was inversely related to the amount of crop residue cover on the field prior to planting.

Yield benefits associated with the continuous use of conservation tillage take a relatively long time to materialize. Ismail et al. (1994) suggested that a minimum of 10 years are required to fully realize any potential yield effects associated with NT and RT practices. Griffith et al. (1988) found that continuous corn yield was reduced in NT treatments during the first 3 years of a 7- year study on imperfectly drained soils. During the final 4 year of the study the NT corn yields were either equal to, or better than, yields from corn in plowed plots. Kapusta et al. (1996) found that continuous NT on a poorly drained soil did not reduce corn yield. A 15-year continuous corn study in Ontario showed that NT systems significantly depressed yields at a nonirrigated tile-drained site (Vyn and Rainbault, 1993). In a 25-year tillage experiment in Ohio, moldboard plow corn initially outyielded NT treatment on a poorly drained fine textured soils, but similar yields were recorded during the latter years (Dick et al., 1991). On a well-drained soil, however, the same authors noted that NT proved to be consistently superior to moldboard plowing with yield differences increasing as time progressed. Hussain et al. (1999) suggested that increased profits associated with NT and RT are attributed to yield increase, reduced production costs such as fuel and labor .

This summary of available data suggests that no clear consensus emerges concerning the effects of conservation tillage on crop yields relative to conventional tillage. Development of production recommendations from short-term tillage studies is hampered by year-to-year variation in results which may mask the real impact of the tillage system being evaluated. To accurately evaluate the benefits and disadvantages of RT and NT systems, it is important to examine long-term impacts of these systems.

2.9 Denitrification and Nitrous Oxide in Subsurface Soil

Denitrification and nitrous oxide emissions in subsurface soils have been a topic of increasing concern for years, yet microbial N transformations in subsurface soil are poorly documented. Although it is generally true that biological denitrification flux is greatest at the soil surface, some researchers in laboratory-based studies have reported increased denitrification with depth when soluble carbon (C) is not limiting (Jarvis and Hatch, 1994; Ryan et al., 1998; Weier et al., 1993). Dissolved organic C (DOC) is generally considered to provide a rough measure of the amount of C available to microorganisms in the soil. Lind and Eiland (1989) observed appreciable amounts of N_2O below the root zone (4 m), even without the addition of glucose, although levels markedly increased when C was added. Downward movement of soluble C (i. e., DOC) is, therefore, a critical factor controlling subsoil denitrification.

It is plausible to assume that management practices that increase DOC in the subsurface soil may lead to increased denitrification. Conditions under which DOC may be exported from top soil to subsoil are not fully understood. Myers and McGarity (1971) observed that soluble C, added as glucose, cowpea straw, and wheat straw leached from the zone of application. These authors concluded that high denitrification activities measured in some B horizons of Australian Solonetz soils were due to soluble C leached from A horizons. In contrast, McCarty and Bremner (1993) found that the increase in soluble C concentrations of surface soil following addition of plant residues was short lived. They concluded that this C was so rapidly decomposed in surface soil that it could not leach and, hence, did not promote denitrification in subsoil environment. Further, Richards and Webster (1999) reported that, in the inorganic N-treated soils, denitrification activity in the subsoils amounted only about 10% of the surface soil.

One important point that must be communicated with regard to denitrification in subsoil is the hypothesis that denitrification activity in subsoil may be less environmentally damaging than denitrification in surface soils since the N_2O:N_2 ratio decreases with depth (Ryan et al., 1998; Jarvis and Hatch, 1994). While extensive research on denitrification has been done in southwestern Quebec (Elmi et al., 2000; MacKenzie et al., 1998; Fan et al., 1997; MacKenzie et al., 1997), information on the extent of these processes in the subsoil environment is severely lacking. One of the key objectives of my research was to investigate the evolution and proportion of N gases with depth, namely N_2O relative to N_2O+N_2.

2.10 Mole Fraction of N_2O (N_2O:N_2O+N_2 ratio)

Denitrification has been studied extensively by various methods. According to data compiled by Eichner (1990), losses varying from 0 to 70 % of the applied N have been reported. However, the N_2O:N_2 ratio can vary widely, and in some cases N_2O or N_2 can be the sole gaseous product of denitrification (Granli and Bøckmam, 1994). Data from grassland (Ryden, 1981) have indicated that approximately 25% of the denitrification losses may occur as N_2O. The functioning of denitrifying community may be a critical factor in regulating the emission of N_2O, since it is a net result of N_2O production and reduction to N_2. Smith and Arah (1990) indicated that the N_2O fraction may range from less than 10 to 100% of denitrification products in agricultural soils. This balance between production and reduction of N_2O is controlled chiefly by diffusion conditions and enzymatic activities of the denitrifying community (Richard and Webster, 1999).

The ratio of $N_2O:N_2O+N_2$ is also affected by soil moisture content and temperature. Weier et al. (1993) found that an increase in water filled pore space (WFPS) from 60% to 90% corresponded to a strong decrease in $N_2O:N_2O+N_2$ ratio. Maag and Vinther (1996) found by regression analysis that N_2 production was greater than N_2O production as temperature increased (2-25 °C). The $N_2O:N_2O+N_2$ ratio may also depend on the stage of plant development. Rudaz et al. (1999) reported $N_2O:N_2O+N_2$ ratio to be lower at the end of a growing season. They explained this could be related to the presence of higher amounts of available C derived from dead roots at the end of the growing season, which would influence the growth of denitrifying populations and synthesis of denitrifying enzymes favoring the reduction of N_2O to N_2. Other authors (e. g., Mahmood et al., 1997) showed that plant growth decreased the $N_2O:N_2O+N_2$ ratio by reducing oxygen availability. Maag and Vinther (1999) and Granli and Bøckman (1994) suggested that the $N_2O:N_2O+N_2$ ratio increases as pH decreases. Klemedsson et al. (1987) noted that this shift from N_2 to N_2O may have little effect on total N_2O+N_2 produced but that the last step of N_2O conversion to N_2 is suppressed by the low (acidic) pH. The obvious conclusion from these findings is that the mole fraction of N_2O should be taken into account when devising strategies to minimize N_2O emission to atmosphere. Until adequate information about the ratios of the products of denitrification is available, it may be difficult to prescribe overall solutions to mitigate environmental N pollution problems.

PREFACE TO CHAPTER 3

In view of growing awareness that water resources are vulnerable to NO_3^--N contamination, farm management practices that minimize water quality problems while sustaining food production must be researched and identified. Efficient management of our agricultural lands is the key in developing sustainable agricultural systems. While there is an increasing body of data on NO_3^--N losses from agroecosystems, there are inconsistences in the findings from different climatic regions as outlined in Chapter 2. At this project's inception, the need for more research was evident.

In chapter 3, the combined impact of water table and N fertilizer rate on N dynamics in the soil-water system and corn yield is discussed. This chapter is drawn from a manuscript prepared for publication by myself and co-authored by my supervisors, Dr. Chandra Madramootoo and Dr. Chantal Hamel, as outlined in the manuscript and authorship section. The format has been changed to be consistent within this thesis.

CHAPTER 3

Managing Water Table and Nitrogen Application Rate for Nitrate Pollution Control and Sustainable Crop Production in a Sandy Loam Soil in Southwestern Quebec, Canada

ABSTRACT

Nitrate-N (NO_3^--N) pollution of water resources is a widely recognized problem. Water and nitrogen (N) fertilizer are the two most important factors affecting crop production and NO_3^--N movement to surface and groundwater. Field trials were conducted from 1998 to 2000 growing seasons to investigate the combined impacts of water table management (WTM) and N fertilization rate on NO_3^--N concentration in the soil profile and drainage water, and on crop yield. There were two water table treatments: free drainage (FD) with open drains at a 1.0 m depth from the soil surface and subirrigation (SI) with a target water table depth of 0.6 m below the soil surface, and two N fertilizer rates: 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}) in a split-plot design. Compared to FD, SI reduced NO_3^--N concentration in the soil by up to 50% averaged over the two N rates. Concentrations of NO_3^--N in drainage water from SI were lower than those from FD by 55 to 73%. Corn yield was not significantly affected by water table treatments in 1999 and 2000, but 25% lower in 1998 under subirrigation because of the unusually heavy rainfall in June coupled with shallow water table, which resulted in waterlogging. These findings suggest that SI can be used as a means of reducing NO_3^--N pollution without compromising crop yields during normal seasons. Soil and drainage water NO_3^--N concentrations, and corn yield were unaffected by N rate treatments.

3.1 INTRODUCTION

Water and N fertilizer are the two inputs that have the greatest impact on agronomic, economic, and environmental outcomes in crop production systems (Letey et al., 1977). Food production has kept pace with population growth over the past decades largely because of the increased use of fertilizers as well as improved cultivars. Use of N fertilizer, for example, has increased 15-fold over the past 50 years worldwide, helping triple world grain production in support of doubled human populations (Constant and Sheldrick, 1992). Most forecasts suggest that global population will continue to increase and crop yields will have to increase correspondingly if future food demands are to be met. In order to achieve these future yield goals, an increase in N fertilizer and water use may be inevitable. Concomitantly, society is seeking ways to diminish the adverse impacts of agricultural activities on the environment.

The realization that our environment, water quality in particular, is something that needs to be protected is a concept which is spreading very quickly at all social levels. Heightened public awareness in recent years has led to the perception that NO_3^--N contamination of surface waters and shallow groundwater is closely linked to the extensive use of N fertilizer in agricultural crop production systems. This perception is supported by numerous research reports that established an obvious connection between N fertilizer use and NO_3^--N concentrations in water (Patni et al., 1998; Randall and Mulla, 2001; Smith at al., 1987). High levels of NO_3^--N in drinking water have long been regarded as dangerous for infants under 6 months of age. Methemoglobinemia (also known as Blue baby syndrome), for example, was first linked to NO_3^--N contamination of drinking water in Iowa in the 1940s (Comly, 1945). In addition, discharge of N into surface waters via subsurface drains can lead to excessive growth of algal blooms and eutrophication in aquatic ecosystems. Certain algae species produce toxins which can be dangerous to health, impairing the use of water resources for human consumption, irrigation, and recreational purposes (Pitois et al., 2001). David et al. (1997) indicated that NO_3^--N contaminated drainage water from subsurface drainage systems in agricultural regions is a primary source of NO_3^--N loadings to surface waters. Therefore, NO_3^--N contamination of water resources is increasingly becoming a critical public health issue. Pressure is growing to adopt farm management systems that ensure sustainable food production but also minimize negative environmental impacts.

Water table management (WTM) is a practice that has been shown to offer better water quality while enhancing or maintaining crop performance (Kalita and Kanwar, 1993; Drury et al., 1997; Cooper et al., 1999). Water table management consists of two main alternatives: controlled drainage (CD) and subirrigation (SI). Under CD, water is prevented from leaving the drain outlet by means of raising the drainage outlet. No supplemental water, other than rainfall, is added to the system. With this practice, water table drops below a designed level due to evapotranspiration and natural drainage. As the water table continues to drop, it may reach a level too low to support crop growth if rainfall or irrigation does not occur, resulting in crop stress and reduced yield. Subirrigation is similar to the CD system, except that supplemental water is pumped into the system to maintain the water table at a designed level during drought periods. Controlled drainage-subirrigation (CD-SI) reduces NO_3^--N pollution problems either by restricting the volume of drain discharge (Gilliam and Skaggs, 1986; Kliewer and Gilliam, 1995; Wright et al., 1992) and/or by creating anaerobic conditions which enhance denitrification (Elmi et al., 2000; Jacinthe et al., 2000).

There is a general lack of information regarding interactive effects of N and water table management on crop productivity and environmental quality. In this context, the effects of the interactions between N application rate and water table depth (WTD) on NO_3^--N accumulation in the soil profile, NO_3^--N losses to subsurface drainage, and yield need to be examined in order to develop environmentally and economically sustainable farming systems. Specific objectives of this study were to 1) investigate the combined impacts of water table depth and N fertilization rate on corn yield, 2) evaluate water table management effectiveness in lowering NO_3^--N in the soil profile, and 3) examine drainage water quality.

3.2 MATERIALS AND METHODS

3.2.1 Field History and Management

The research was conducted on a 4.2-ha privately-owned field located at St-Emmanuel near Côteau-du-Lac, Quebec (74° 11' 15" lat., 45 ° 21' 0" long.), about 30 km southwest of the Macdonald Campus of McGill University. Site design and instrumentation is detailed in Tait et al. (1995). The soil, a Soulanges fine sandy loam (fine silty, mixed, non-acid, frigid *Humaquept*, Gleysol, according to the FAO classification system), was of sedimentary origin. Surface topography was generally flat with an average slope of less than 0.5%. The fine sandy loam soil (0-0.25 m) was underlain by layers of sandy clay loam (0.25-0.55 m) and clay (0.55-1.0 m), and the clay layer impeded the natural drainage. Experimental plots were under a conventional tillage system (i. e., moldboard- plowed to 0.20 m in fall and disked in spring performed by the farm owner as part of his normal production practice), the common practice in the region. The site had been a pasture prior to 1992, when it was converted to corn production.

3.2.2 Experimental Design

Field layout and treatment arrangements are presented in Fig. 3.1. There were two water table management treatments: FD with open drains 1 m in depth from the soil surface and SI with a design water table 0.6 m below the soil surface, and two fertilizer rates: 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}). Nitrogen fertilizer was applied in a split dose: 23 kg N ha^{-1} banded as diammonium phosphate (18-46-0) at planting, and 97 kg N ha^{-1} or 177 kg N ha^{-1} broadcast as ammonium nitrate (34-0-0) one month after planting (8 June 1998, 10 June 1999, and 20 June 2000), resulting in rates of 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}), respectively. In addition, the farmer applied manure (cattle slurry) on the field in spring 1998 at a rate of approximately 20 Mg ha^{-1} (wet wt).

Treatments were laid out in a split-plot design with water table depth as main plot and N fertilization as subplot. The water table treatments were established in 30 m wide and 75m long plots, and each main plot was split into two 15 m x 75 m subplots. The water table treatments were replicated in 3 blocks, and fertilizer treatments were assigned randomly to the subplots. In the middle of each plot, 75 mm diameter subsurface drain pipes were installed, at 1.0 m depth, with a slope of 0.3%. Blocks were separated by a 30 m wide strip of undrained land (Fig. 3.1). To minimize seepage and chemical flow between plots, a 6-mil (0.6 mm) polyethylene sheeting was installed to a depth of 1.5 m between plots (Tait et al., 1995). In addition, adjacent to each WTM treatment were subirrigation treatment plots were buffer plots with the same water treatment (Fig. 3.1). All buffer plots received 120 kg N ha^{-1}.

The field was seeded on May 8 in 1998, May 4 in 1999, and May 23 in 2000 with grain corn (Pioneer hybrid 3905) at a planting density of 75,000 plants ha^{-1} with 0.75 m spacings. Potassium (muriate of potash, 0-0-60) was side-dressed at a rate of 90 kg K_2O ha^{-1} roughly one week before planting. Subirrigation treatment was imposed 2 weeks after planting and maintained until crop maturity in late September. Well water (Fig. 3.1) with no detectable nitrates was continuously pumped into the field to balance crop use and evaporative losses. The plots received no surface irrigation. Due to deep seepage it was difficult to maintain water tables at the desired depth. Following heavy rainfall events, pumping was stopped in SI plots and excess water drained until a 0.6 m water table depth was achieved in the field. To monitor water table fluctuations, three observation wells, a perforated 12-mm diameter polyethylene pipes wrapped with geotextile sleeves (Zodiac, London, ON), were installed diagonally in each treatment or buffer plot to a depth of about 1.4 m, and water table depth averaged for each plot. Water table depths were measured once or twice a week during the growing season by inserting a sonic water sensor mounted on a graduated rod. Drains were opened September 28 in 1998, September 17 in 1999, and September 25 in 2000 to facilitate trafficability for harvesting. Rainfall and air temperature data were obtained from an Environment Canada weather station situated 500 m from the experimental site.

3.2.3 Soil Sampling and Analysis

Soil samples (three samples per plot) for NO_3^--N analysis were taken prior to planting in the spring (April or early May), during summer (July), and shortly after harvest in fall (October) from 0-0.25 m, 0.25-0.50 m, and 0.50-0.75 m depth increments using hand-held auger sampling probe. Each sample was placed in a prelabeled soil sample bag and transported to the laboratory. All soil samples were promptly frozen (at -4 °C) after collection and kept for 1-3 weeks prior to analysis. Samples were then thoroughly mixed and moist subsamples of 10 g were shaken with 100 mL of 1 M KCl for 60 min. The soil suspensions were filtered through Whatman # 5 filter papers. Nitrate-N was quantified using Lachat flow injection autoanalyzer (Lachat Quickchem, Milwaukee, WI) according to Keeney and Nelson (1982). The detection limit was 0.05 mg L^{-1}.

3.2.4 Collection of Drainage Water Samples and Analysis

Tipping buckets, housed in the two buildings (Fig. 3.1), were located at the outlet of each subsurface drain to measure samples of drain discharge. Water sampling was done according to a flow weighted composite strategy, i. e., the frequency of water sampling was set according to accumulated volume of drain flow. Water samples were obtained from plastic containers (20 L) connected to each pipe using a water sampling valve located just before the tipping buckets, and brought to the laboratory twice a week or once in every two weeks, depending on tile flow rate, and frozen until analysis. Composite samples accumulated in the 20 L plastic containers, from which 20 mL sub-samples were taken and analyzed for NO_3^- -N in triplicates using a Lachat flow injection Autoanalyzer (Lachat Quickchem, Milwaukee, WI) according to Keeney and Nelson (1982).Nitrate -N mass loss in tiles was calculated by multiplying the computed volumes of subsurface outflow with measured concentrations of NO_3^- -N in the drain outflow.

Corn grain yield was determined by hand harvesting individual ears from a subplot consisting of a 5 m stretch of the three middle rows of each plot. Grain yield was reported on a dry-weight basis. The field was plowed in the first week of November, incorporating all corn stover (leaves plus stalks) into the soil.

3.2.5 Statistical Analysis

Analysis of variance (ANOVA) was performed on NO_3^--N concentrations in the soil profile, drain water and on corn grain yield. Data were analysed as a split plot with water table being the main plot and fertilizer N treatments as sub-plot. When a main plot effect was significant without any interaction, Fisher's F test statistic was used to determine statistical significance within each main plot treatments. Since the interaction between the water table and N fertilizer treatments was significant in 1999 and 2000 seasons with respect to NO_3^--N concentration in drain water, interactions rather than the main treatment effects are discussed. Unless otherwise noted, statistical significance is reported at the 5% probability level (alpha < 0.05). All statistical analyses were conducted using the General Linear Model (GLM) procedure of the Statistical Analysis System (SAS Institute, 1996).

3.3 RESULTS AND DISCUSSION

3.3.1 Seasonal Water Table, Drain Outflow, and Precipitations

Water table levels, and drainage outflows fluctuated throughout the growing seasons, responding primarily to rainfall events (Figs. 3.2-3.4). Total seasonal (May - October) rainfall in 1998 was 29% above normal. About 34% of the total seasonal rainfall in 1998 occurred in June. Globally, June 1998 was the wettest in 70 years and the second wettest on record (NOAA, 1998). July 1998 was also much wetter than the normal, accounting for about 20% of the seasonal total. Rainfall in these two months together accounted for nearly 60% of the seasonal (May-October) total. These two months of high rainfall (Fig. 3.2 c) resulted in periods of shallowest water table depths, especially in the SI treatment (Fig 3.2 a), and produced the greatest volume of drainage outflows, especially in FD plots, during the entire growing season (Fig. 3.2 b). In June and July 1998, drain outflow was significantly ($P \leq 0.01$) lower (48% and 15%, respectively) in SI than FD. This is likely to be due to WTD under FD remaining well above the drains (1. 0 m depth) from mid June to the end of July because of the excessive rainfall.

It is worth noting that the August 25 rainfall event, the second highest in 1998 (Fig. 3.2 c), raised the water table in the SI treatment to as shallow as 0.1 m below the soil surface (Fig. 3.2 a), but did not cause a significant increase in drain discharge (Fig 3.2 b). A plausible explanation for this observation was that a heavy rainfall event preceded by dry conditions may not cause significant water percolation to the subsurface drains. Another notable observation is the increase in the drain outflow from SI plots at the end of September. This increase coincided with the opening of drains (i. e., SI switched to FD mode), suggesting that water stored in the SI plots started to drain out in October. September and October rainfalls were 7% and 9% below normal, respectively, providing further evidence that the increased drainage outflow in the SI plots (Fig. 3.2 b) was from water stored in the soil profile due to subirrigation, rather than rainfall.

The 1999 cropping season was characterized by a dry summer (Fig. 3.3 c). While total seasonal (May-October) rainfall was 13% higher than the normal, May and August were 23% and 35% lower than the normal, respectively. The first week of July, just three weeks after the N application, received frequent precipitation resulting in a flush of drain outflow from SI plots (Fig. 3.3 b) and the shallowest water depth (Fig. 3.3 a) of the growing season. For much of 1999, water table in FD plots remained deeper than 1.0 m, whereas in SI plots, water table were on average 0.8 m below the soil surface. A single rainfall event on 16 September provided 29% of the total growing season rainfall (Fig. 3.3 c), and occurred just one day before SI system was switched into drainage mode, resulting in a significant increase in drain outflow from formerly subirrigated plots (Fig. 3.3 b) and a water table shallower than the design depth (0.6 m). Rainfall during the 1999 cropping season totaled 589.2 mm, with nearly half (276 mm) falling in September and October (Fig. 3.3 b).

During the 2000 growing season, total rainfall (Fig. 3.4 c) was about 12% higher than the normal. May was the wettest month, accounting for 25% of seasonal rainfall with October being the driest month, accounting for less than 5% of total rainfall. The subirrigation system was not functional in May, and there was a large drainage outflow both from conventionally drained and formerly subirrigated plots (Fig. 3.4 b). The shallowest WTD under SI (Fig. 3.4 a) corresponded to a rainfall event that delivered 39 mm on August 7 (Fig. 3.4 c) and resulted in a large drainage outflow on 9 August (Fig. 3.4 b). This suggest that drainage outflow can be greater in SI than FD when the SI system is opened to drain excess water.

3.3.2 Impacts of Water Table and N Fertilization Rate on Corn Yield

Corn yields for the three years are presented in Table 3.1. Corn yields in 1999 and 2000 were not significantly different in either treatment. In contrast, yields under SI were significantly lower than FD in 1998. Unusually heavy June rains (Fig. 3.2 c) coupled with the high water tables in the SI plots led to ponding of water on the field on some occasions, which resulted in poor crop growth and yield. When ponding occurred, the subirrigation system had to be manually shut off for about 24-36 hrs to allow drainage of excess water. Corn roots, particularly when they are young, are sensitive to even short periods of restricted aeration (Evans et al., 1996). Corn stalks were visibly shorter (approx. 0.5 m) under SI plots than FD plots, and yield was reduced by 25%. This observation suggests that precise management of the water table is required with SI, particularly during rainy periods.

For SI systems to be successful, water table depth must be high enough to permit capillary rise into the root zone and low enough to ensure adequate soil aeration. Skaggs et al. (1999) suggested that raising the water table generally increases evapotranspiration and, hence, yield. Tan et al. (1996) made similar observations and concluded that corn grain yields on a sandy loam soil were less with a water table depth of 0.8 m as compared to 0.6 m because stomatal conductance and transpiration rate were reduced by water stress. Doty (1980) found that the optimum water table depth for corn in sands or sandy loam to be 0.76-0.89 m.

Subirigation is expected to be more beneficial than conventional drainage during drier crop seasons as it supplements rainfall to meet crop evapotranspiration demand. Cooper et al. (1999) recorded a significant yield increase from a SI treatment in 1991, a very dry year, whereas wet growing conditions of 1992 resulted in conventional drainage yields not differing from those obtained under SI. These results suggest that in wet growing years there is no significant yield advantage for SI systems.

Corn yield was not responsive to N fertilization rate in all the three seasons covered in this study (Table 3.1). This is an indication that 120 kg N ha^{-1} were sufficient to optimize crop yield under conditions of no drought stress. Growing conditions (rainfall and temperature) were either above normal (1998 and 1999) or similar to normal (2000). From the same site, Elmi et al. (2002) reported a significantly higher yields under 200 kg N ha^{-1} than 120 kg N ha^{-1} treatment in 1997 during which rainfall and temperature were both below the normal. This may suggest that when growing conditions are not optimal, an increase in yield is possible with higher rate of N fertilizer. This situation may lead to a tendency for farmers to apply excessive amounts of these inputs to maximize productivity, but not without potential environmental cost as NO_3^--N in the soil-water system may be increased. It must be recognized that farmers are usually fearful that adoption of reduced fertilizer application rates will result in a decline in yield and ultimately in a decline in farm income.

Yields were lower in 2000 than 1998 and 1999 (Table 3.1). This is likely due to the late planting date of the field because of technical difficulty and weather conditions.

3.3.3 Water Table Effects on NO_3^--N Levels in the Soil Profile

Nitrate-N concentrations in the soil profile for the spring and fall are shown in Fig. 3.5. No significant difference between freely drained plots and plots formerly subjected to subirrigation was observed in spring 1998 at any depth (Fig. 5 a), whereas in the fall, the difference was significant at the deeper depths (Fig. 3.5 b). Similar trends were observed in 1999 and 2000 except that the difference between FD and SI was significant at the uppermost layer in the spring 1999 (Fig. 3.5 c), and at the intermediate depth in the spring of 2000 (Fig. 3.5 e). These findings suggest that the effects of water table treatments were more pronounced in the fall than spring. This may be because sampling in the spring was done long (7 months) after SI was switched into FD, and the carry-over effect of SI treatment was minimal. Whereas in the fall, samples were collected immediately after SI was put into FD mode and the SI effect was significant. Greater NO_3^--N levels at the uppermost soil layer (0-0.25 m) in the spring of 1998 (Fig. 3.5 a) and 1999 (Fig. 3.5 c) than 2000 (Fig. 3.5 e) may be due to the mineralization of manure applied by the farmer in 1998 spring. Liang et al. (1995) suggested that a major portion of the manure applied in the spring was mineralized during subsequent summer.

In the growing season (July sampling), NO_3^--N concentrations in the 0.25-0.50 m and 0.5-0.75 m soil layers were significantly lower in SI than FD in all the three growing seasons investigated (Fig. 3.6). In 1998, there was also significantly more NO_3^--N in the 0-0.25 m soil layer under FD than SI (Fig. 3.6 a). These significant effects of water table depth on NO_3^--N can be explained by the fact that SI was functional during summer months (May-September). The significant decrease in soil NO_3^--N associated with SI may be due to enhanced denitrification. Soil moisture content was significantly greater under SI than FD plots (See chapter 4, Fig. 4.4) during the 1998 to 2000 growing seasons, which might have created sufficiently anaerobic conditions to promote NO_3^--N loss through denitrification. Denitrification has long been recognized as an important mechanism for reducing NO_3^--N loading in the saturated zone of the soil profile (Yeomans et al., 1992).

In general, there was a trend for NO_3^--N concentrations in the soil to decrease with depth under both water table treatments. A notable exception was spring 2000 (Fig. 3.5 e).

An increase in NO_3^--N with soil depth would be expected if leaching had occurred. As pointed out earlier, heavy rains in the fall of 1999 producing the largest amount of drain discharge (Fig. 3.3 b) might have caused NO_3^--N leaching with percolation water. In coarse-textured soils such as the study site, NO_3^--N can be leached easily below the crop rooting zone and, consequently, discharged into water bodies during heavy rainfall events or excessive irrigation, particularly after harvest. This may explain the sharp increase of soil NO_3^--N concentrations with depth in the spring 2000 (Fig. 3.5 e), following the excessively wet fall of 1999. Hatfield et al. (1999) suggested that approximately 95% of NO_3^--N passing through the root zone is intercepted and moved eventually as discharge into surface waters. In spring, when evapotranspiration is low and precipitation and snow melt exceed the water holding capacity of the soil, residual NO_3^- -N can leach beyond the crop root zone and, consequently, contaminate surface water resources via subsurface drains or groundwater with percolating water. Patni et al. (1998) estimated that approximately 70% of NO_3^--N leaching occurs from fall to spring (October through April). Keeney and DeLuca (1993) found that NO_3^--N concentrations in the Des Moines river in Iowa, USA, were above 10 mg L^{-1} for about 14 days per year, mainly in the spring. Drury et al. (1996) reported that up to 88 to 95% of the NO_3^--N loss to subsurface drainage occurred during the noncrop period (November through April). All these observations indicate that NO_3^--N leaching in early spring appears to account for the majority of NO_3^--N loading losses to the subsurface drains which subsequently discharged into surface waters.

Averaged across all depths, seasonal reduction of total soil NO_3^--N under SI compared to FD ranged from 2% to 29% in the spring, 38% to 46% in the summer, and 36% to 50% in the fall. Nitrate-N reductions of 30% to 60%, resulting from controlled drainage/subirrigation have been reported in several studies. For example, Fogiel and Belcher (1991) found that controlled drainage-subirrigation reduced NO_3^--N loading through drainage by 25% to 59% over a two year period compared with conventional drainage. Gilliam and Skaggs (1986) predicted a 32% decrease in NO_3^- losses due to controlled drainage relative to conventional drainage systems. Jacinthe et al. (1999) reported 24% to 43% reduction in NO_3^- leaching using WTM techniques. Further reductions could be achieved if controlled drainage is kept operational during the early spring, when NO_3^--N losses is most severe (Patni et al., 1998) and drainage is not needed to optimize crop production. Operation of controlled drains might enhance denitrification and reduce excess NO_3^--N in the soil-water system in early spring, if temperatures are warm enough. This practice, however, may interfere with tillage operations in early spring.

3.3.4 Impacts of N Fertilizer Rate on NO_3^--N in the Soil Profile

Nitrogen fertilizer rate [120 kg N ha^{-1} (N_{120}) *vs* 200 kg N ha^{-1} (N_{200})] had no clear effect on soil NO_3^--N concentrations in all non-growing seasons (spring and winter) at any depth (data not presented). This general lack of significant treatment (N_{120} *vs* N_{200}) effects or even consistent trends suggests that limiting N fertilization alone might not be sufficient to overcome the problem of NO_3^--N loading in the soil-water system. The first evidence of NO_3^--N movement below the root zone for cultivated soils receiving no mineral N fertilizer or manure was presented many decades ago (Buckman, 1910). More recently, Sainju et al. (1998) reported that even with no fertilization, significant concentrations of residual NO_3^--N accumulated beyond the root zone because of continued mineralization from soil and crop residues retained on the soil surface. Nitrate -N levels tended to be greater under N_{200} than N_{120} in the middle the growing season sampling (Fig. 3.7) in all but at the deepest soil depth (0.5-0.75 m) in 2000 (Fig. 3.7 c). This trend for greater NO_3^--N concentrations under N_{200} than N_{120} treatment occurring about a month after plots received the second N fertilizer application emphasizes the notion that even moderately high rates of N application over the long term have the potential to cause greater NO_3^--N accumulation in the soil profile. Similar to our findings, Cambardella et al. (1999) found little relationship between N fertilizer rate and NO_3^--N removal in tile drainage. They concluded that although NO_3^- -N loss in drain discharge tended to increase with N fertilization rate, N mineralized from soil organic matter accounted for a significant portion of the N leached from soil or used by crops.

3.3.5 Subsurface Tile Drainage Water Quality

Since there was a significant interaction between water table (main plot) and N fertilizer (sub-plot) in most sampling dates of the 1999 and 2000 seasons with respect to NO_3^- -N levels in drainage water, interactions rather than the main treatment effects are discussed. During all three cropping seasons, drainage water from plots receiving 200 kg N ha^{-1} used with FD (FD200) contained the greatest concentration of NO_3^--N (Fig. 3.8). Leaching losses, in terms of NO_3^--N concentrations in drainage water, were often significantly less from SI plots receiving 200 kg N ha^{-1} fertilizer (SI200) than FD plots receiving 120 kg N ha^{-1} fertilizer (FD120). Although N fertilizer treatment had a significant effect on NO_3^- -N concentrations in drainage effluents in the FD system, no significant effect was observed in the SI system. These observations clearly indicate that NO_3^--N concentrations in drainage water were greatly reduced by the implementation of subirrigation, both during the cropping season and the fall months.

Controlled drainage-subirrigation reduces nitrate loading to drainage water primarily by retaining water and nitrate in the soil profile rather than allowing them to be drained away to surface waters. According to our findings, total outflow from SI were greater than from FD in 1999 (Fig. 3.3 b) and 2000 (Fig. 3.4 b), but there was no significant difference in 1998 (Fig. 3.2 b). Our results indicate that despite drain outflows from SI being mostly higher than FD, NO_3^--N concentrations were still lower under SI plots. For example, September and October 1998 and 1999 (Fig. 3.2 b and Fig. 3.3 b, respectively), drain outflow was significantly greater under SI than FD but the reverse trend was observed for NO_3^--N level in the drainage effluent (Fig. 3.8 a b, respectively). Zero drain outflow frequently observed in 2000 resulted in zero NO_3^--N outflow in these periods (Fig. 3.8 c). The effects of CD-SI on drainage outflow varies seasonally. Evans et al. (1995) have shown that during a dry period or season, CD reduces drainage outflow rates, sometimes completely eliminating outflow for some storm events. Further, they demonstrated that during wetter periods or seasons, CD may have little effect or in some cases may even increase peak outflow.

Averaged across N treatments, SI reduced seasonal mean NO_3^--N concentrations in drainage effluent by 74% in 1998, 55% in 1999, and 64% in 2000, compared to FD. Gilliam et al. (1979) reported that the reduction in NO_3^--N losses was due to reduced drain flows, rather than to reduced concentration in drainage water. Evans et al. (1995) reported a significant decrease in drainage water NO_3^- -N concentrations with CD system; however, the main reduction was of NO_3^- -N losses due to CD was achieved through the reduction in drainage volume. Contrary to these finding, Mejia and Madramootoo (1998) reported that despite greater drainage in the SI plots (0.5 m and 0.75 m below the soil surface), NO_3^--N levels from SI plots were still lower than from FD plots. Their findings suggest that the decrease in NO_3^--N associated with SI was due to enhanced denitrification and/or plant N uptake.

The lower drainage water NO_3^--N concentrations under SI plots in the present study is consistent with the conclusion that SI offers drainage water quality benefits by promoting denitrifying activity in the soil. In a previous study on this experimental site, Elmi et al. (2000) noted significantly greater denitrification rates in SI than in FD. In a lysimeter study, Kalita and Kanwar (1993) found that shallow water table depths of 0.3 to 0.6 m reduced NO_3^- -N concentrations in groundwater to levels below 10 mg NO_3^- -N L^{-1}. They postulated that these lower NO_3^- -N levels were due to enhanced denitrification resulting from saturated conditions in upper soil layers (0-0.15 m) where organic matter is greatest. Denitrification has been widely recognized as being an important removal mechanism of NO_3^- -N from the soil-water solution and controlling migration and entry of NO_3^- -N into surface water and groundwater resources (Elmi et al., 2000; Jacinthe et al., 2000; Kliewer and Gilliam, 1995).

Mean NO_3^--N concentration values obtained with this study, particularly 1998 (Fig. 3.8 a) and 1999 (Fig. 3.8 b) seasons, were markedly lower than what is commonly reported in the literature. Jaynes et al. (2001), for example, using N rates similar to ours (203 kg N ha^{-1}) reported NO_3^- -N concentrations in the tile drainage consistently exceeding the 10 mg L^{-1}. Randall and Igavarapu (1995) reported similar results for 200 kg N ha^{-1} applied to continuous corn. In New Brunswick, Canada, Milburn et al. (1997) showed tile effluent with flow-weighted average NO_3^--N concentrations between 2 and 5 mg L^{-1} but the field received only 80-105 kg N ha^{-1} in fertilizer and manure. A summary of selected studies on NO_3^--N leaching associated with corn production in various humid and temperate regions of North America compiled by Milburn and Richards (1994) showed mean annual NO_3^--N concentrations of drainage discharge ranging from 4 to 43 mg NO_3^- -N L^{-1}. Our results fall within the low end of this range (Fig. 3.8).

Significant ($P < 0.006$) differences in mass loss of NO_3^--N (kg ha^{-1}) between SI and FD plots occurred only in 1998 because of the heavy rains of June and July. In 1998, NO_3^--N losses were 7.2 kg ha^{-1} under FD and 1.5 kg ha^{-1} under SI. In 1999, total seasonal loss from FD was 3 kg ha^{-1} and 1.4 kg ha^{-1} from SI. In 2000, NO_3^- -N losses were 2.6 kg ha^{-1} under FD and 1.5 kg ha^{-1} under SI. In southwestern Ontario, Patni et al. (1996) reported 13 to 30 kg ha^{-1} loss of NO_3^- -N through subsurface tile drains in fields cropped to corn. The level at which NO_3^--N constitutes a problem is a function of water use. The losses obtained in this study may not be economically important, but may be significant in terms of NO_3^--N effects on aquatic ecosystems.

Water quality concerns are usually expressed in terms of drinking water quality guidelines of 10 mg NO_3^- -N L^{-1}. It is, however, important to highlight that water quality deterioration caused by eutrophication may occur at considerably lower concentrations of NO_3^--N than drinking water standards (Thomas et al., 1991; Evans et al., 1995). Burkholder et al. (1992) reported a depressed eelgrass (*Zostera marina*) survival when NO_3^--N exceeded values as low as 0.1 mg L^{-1}. From these observations, it is evident that NO_3^--N concentrations well-below drinking water quality guidelines may degrade surface water quality. Developing drain effluent quality guidelines for NO_3^--N concentration offers an great challenge. The difficulty is that it will be hard to define N input levels since various sources can contribute and leaching NO_3^--N greatly varies with climatic conditions which are not under farmer's control (Randal and Mulla,2001).

3.4 SUMMARY AND CONCLUSIONS

To keep food production at the same level as population growth without damaging the environment is a major challenge. In view of growing public concerns over NO_3^--N losses to water bodies, there is justification for identifying best management practices which minimize water quality problems while sustaining or enhancing food production. Integrating water table management and N input strategies can minimize the risk of NO_3^--N contamination of water resources without compromising crop yields. Averaged across all depths and N treatments, reduction of total soil NO_3^--N under SI compared to FD ranged from 55% to 74% throughout the study periods. These findings support the idea that the adoption of water table management practices may provide an economical means to offer water quality benefits by enhancing NO_3^--N removal from soil-water system through denitrification and, therefore, control migration and entry of NO_3^--N into surface- and groundwater resources.

Similar yields were obtained with SI and FD in 1999 and 2000. Yield reduction (25%) under SI in 1998 was attributed to the unusually heavy June rains coupled with the shallow water tables in the SI plots leading to ponding of water on the field on some occasions, which resulted in poor crop growth and yield. Since farmer's acceptance of a new technology largely depends on its impact on yield, this should not adversely affect farmer's acceptance of SI, as this situation could have been averted with a more rigorous management such as automating the system. Corn yield was not responsive to N fertilization rate in all three years. Hence, there was no agronomic benefits associated with the higher rate of N fertilization.

Table 3.1 Yield of corn in relation to water table management and nitrogen fertilization rate.

Years	Water table treatments		Nitrogen fertilization treatments	
	Free drainage	Subirrigation	120 kg N ha^{-1}	200 kg N ha^{-1}
		Mg ha^{-1}		
1998	8.7 a	6.7 b	7.8	7.5
1999	8.5	8.4	8.3	8.6
2000	7.3	6.7	6.77	7.03

* Values with different letters in the same row and within water table or nitrogen treatments are statistically significantly different at ($P \leq 0.05$) based analysis of variance test.

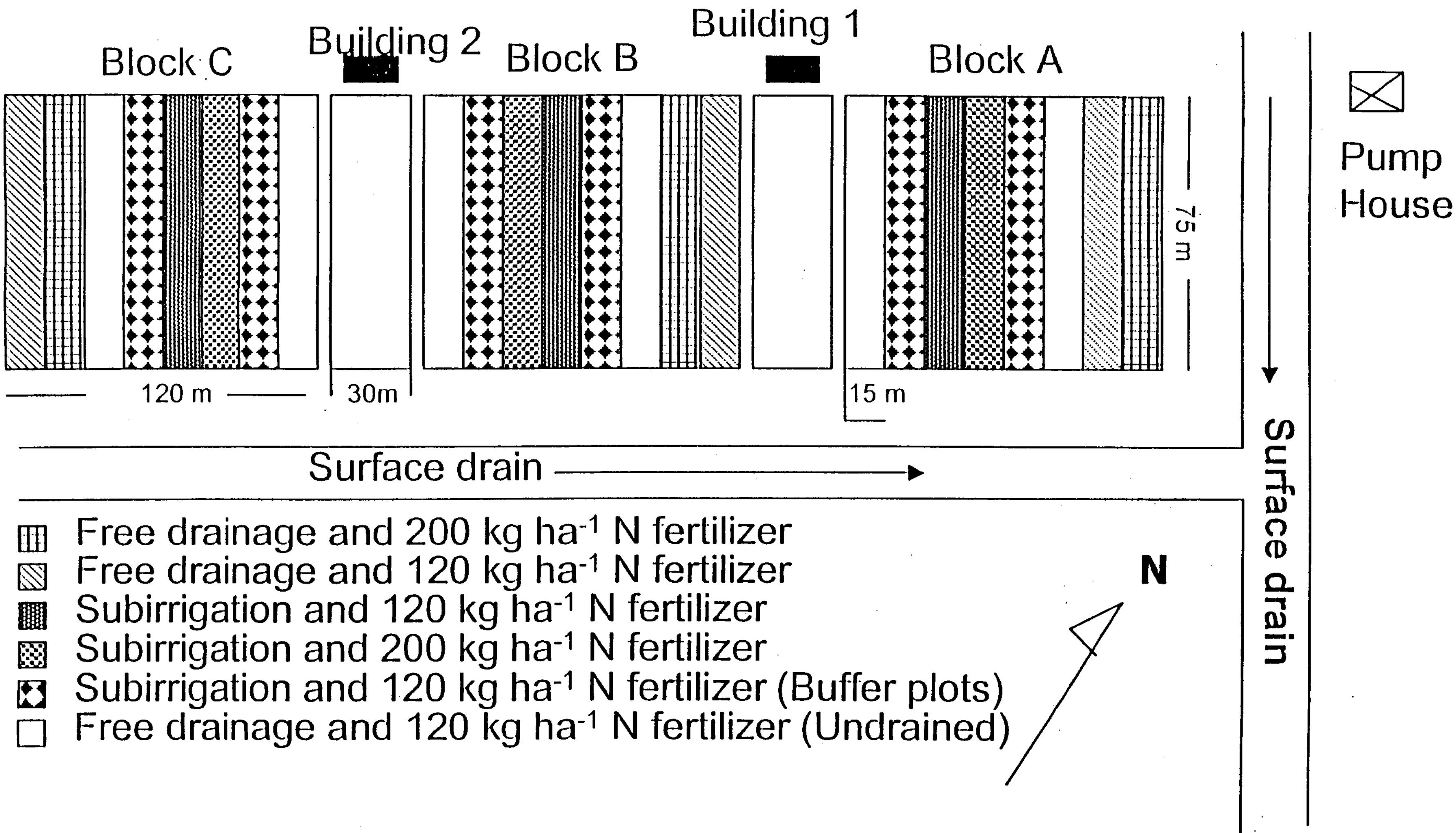

Figure 3.1: Schematic representation of the field layout.

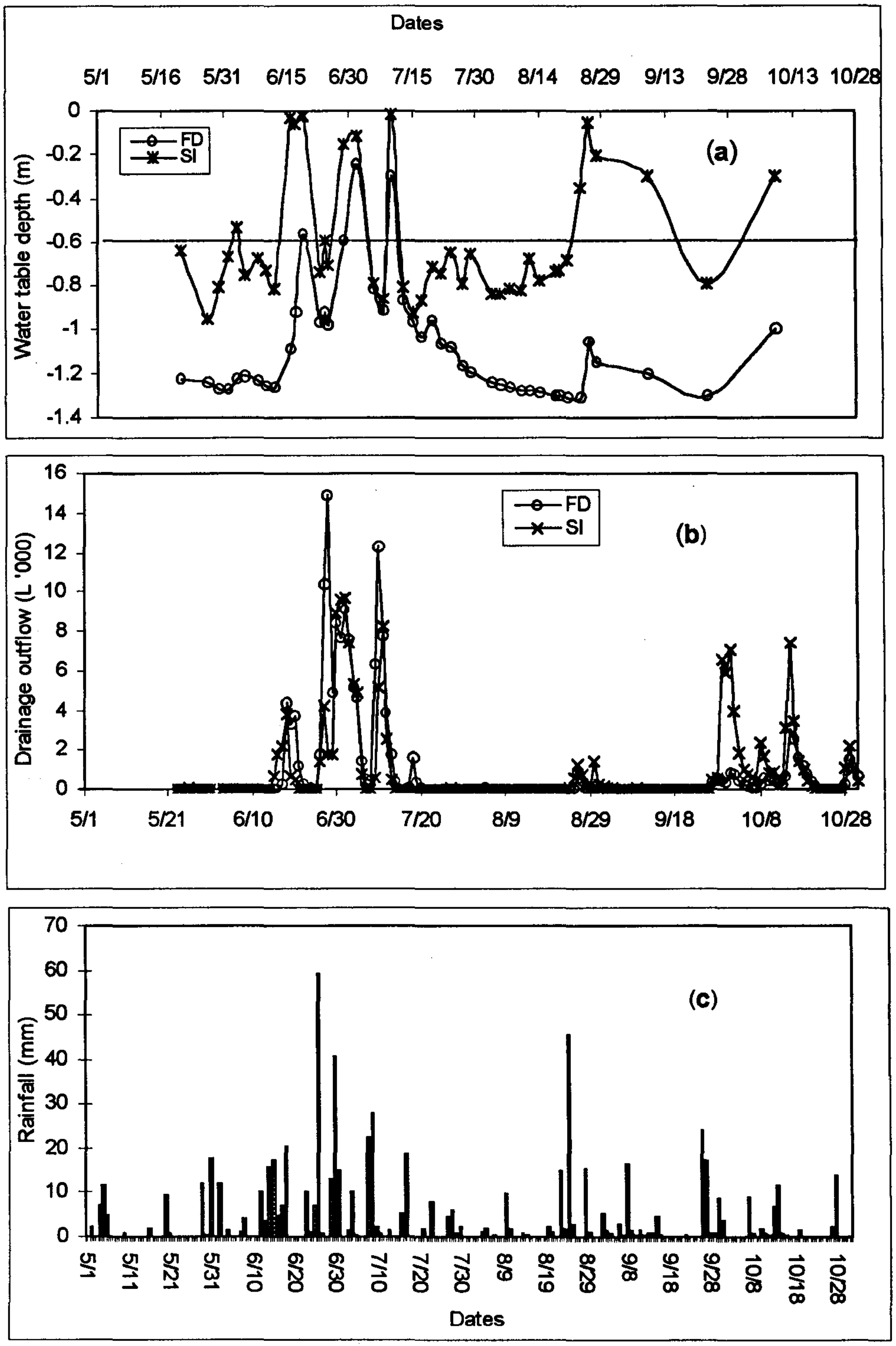

Figure 3.2: (a) water table depth (m) , (b) drainage outflow (L) under subirrigation (SI) and free drainage (FD), and (c) daily precipitation in 1998.

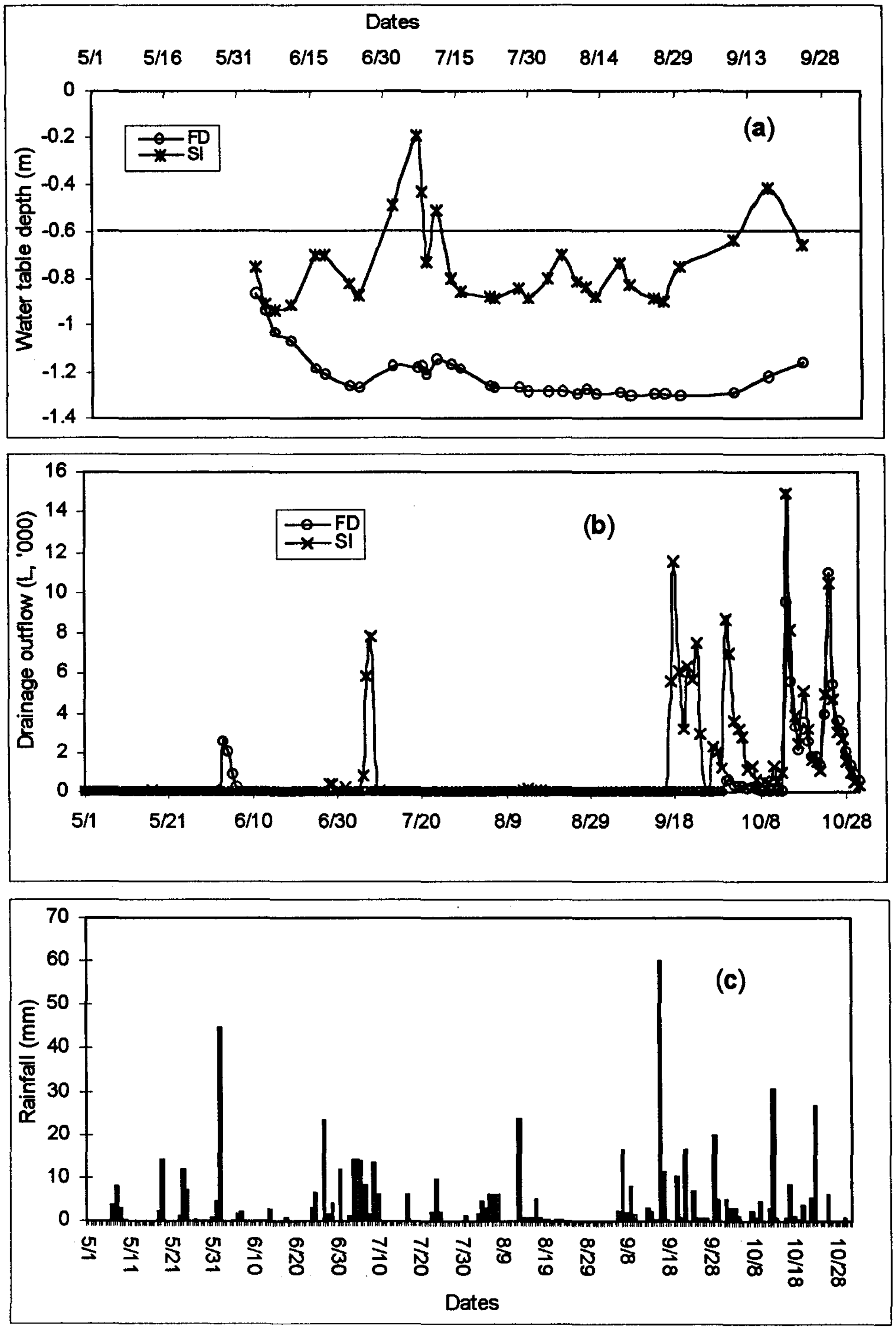

Figure 3.3: Measured (a) water table depth (m), (b) mean daily drainage outflow (L) under subirrigation (SI) and free drainage (FD), and (c) daily precipitation in 1999.

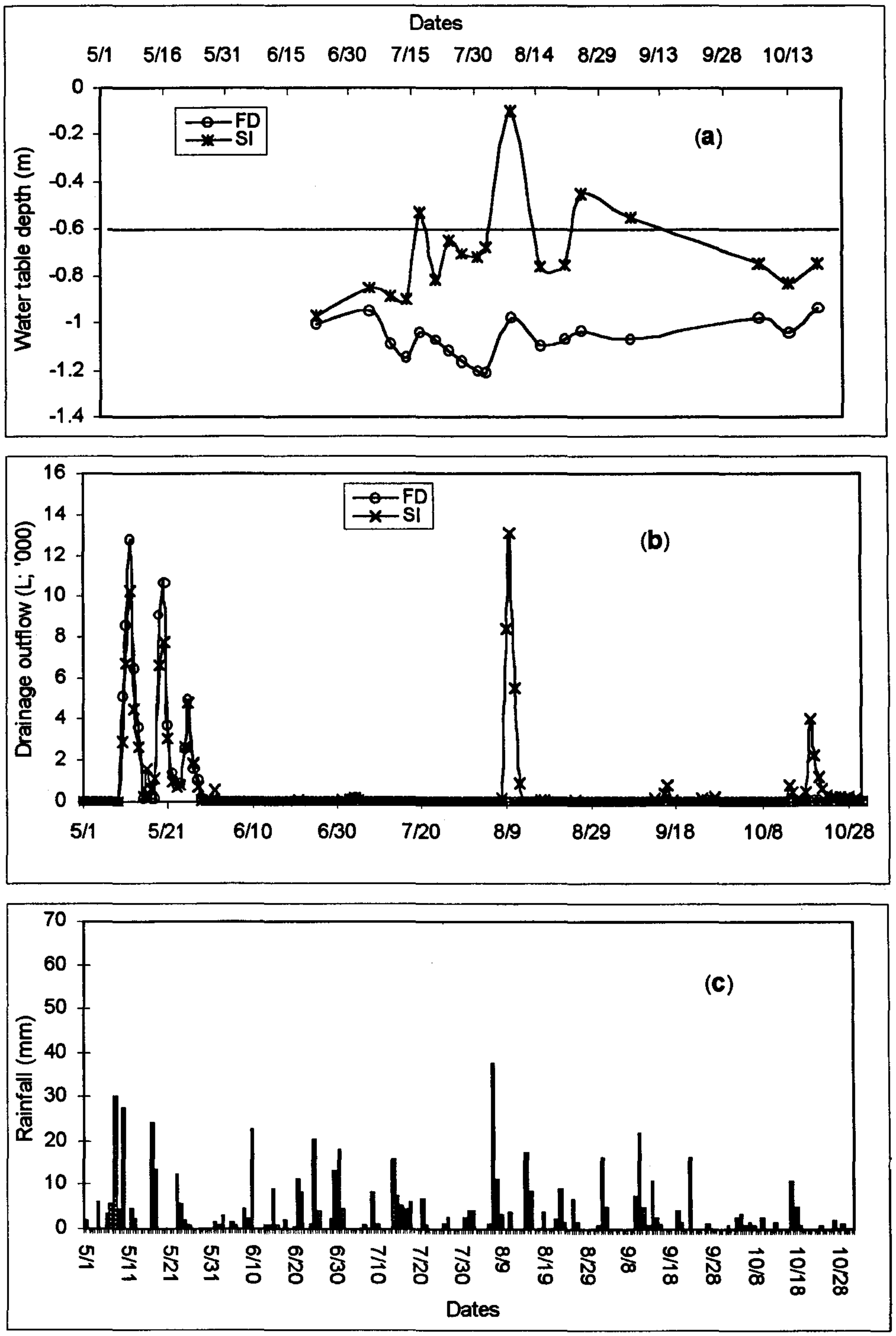

Figure 3.4: Measured (a) water table depth (m), (b) mean daily drainage outflow (L) under subirrigation (SI) and free drainage (FD), and (c) daily precipitation in 2000.

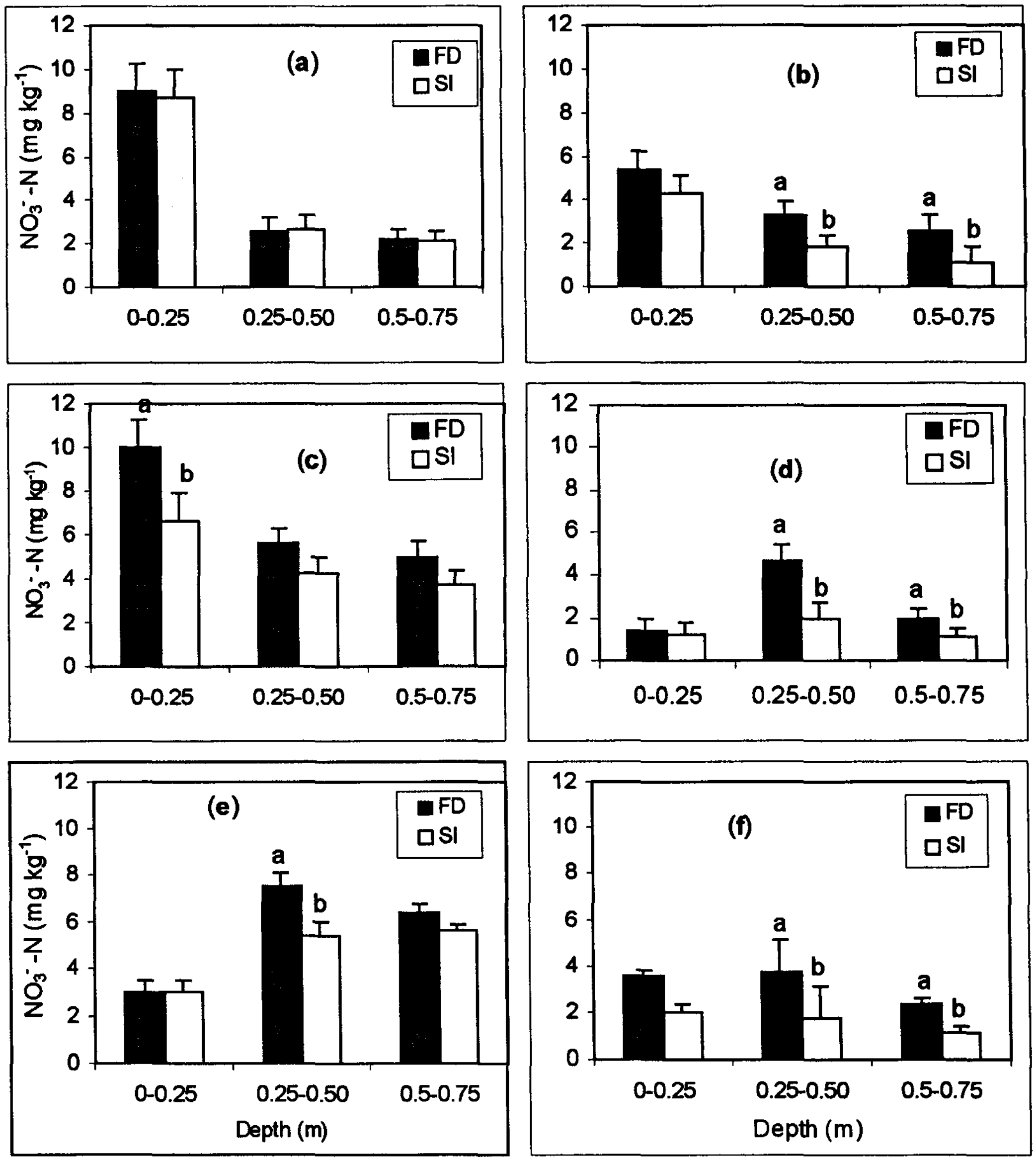

Figure 3.5: NO_3^--N concentrations (mg kg^{-1} soil) in the soil profile under free drainage (FD) and subirrigation (SI) practices in (a) spring 1998, (b) fall 1998, (c) spring 1999, (d) fall 1999, (e) spring 2000, and (f) fall 2000. Charts within same depth followed by different letters are significantly ($P < 0.05$) different. Vertical bars represent standard error of the mean ($n = 9$).

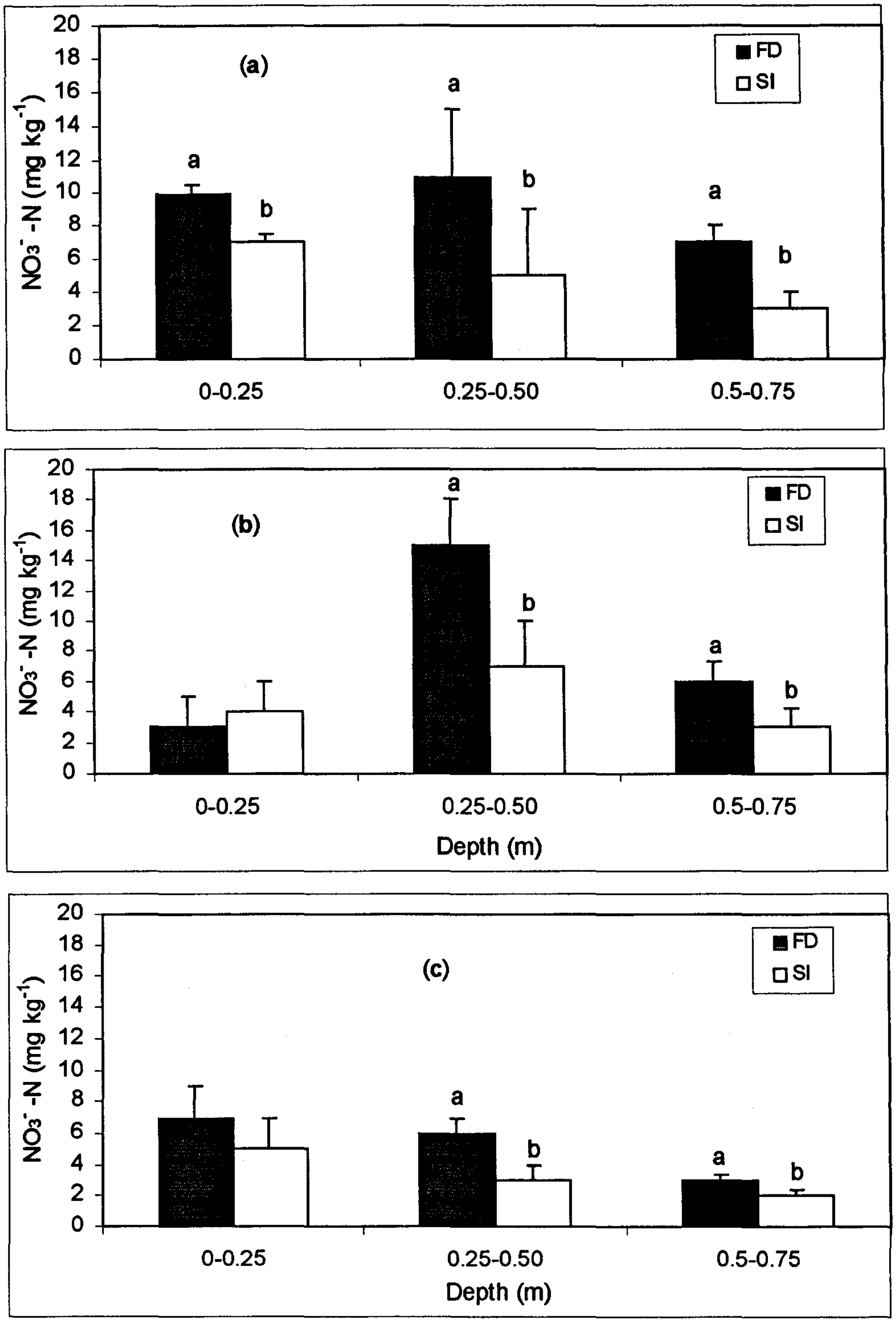

Figure 3.6: NO_3^- -N concentrations (mg kg^{-1} soil) in the soil profile under free drainage (FD) and subirrigation (SI) in summer (a) 1998, (b) 1999, and (c) 2000. Bars within same depth followed by different letters are significantly ($P \leq 0.05$) different. Vertical bars represent standard error of the mean ($n = 9$).

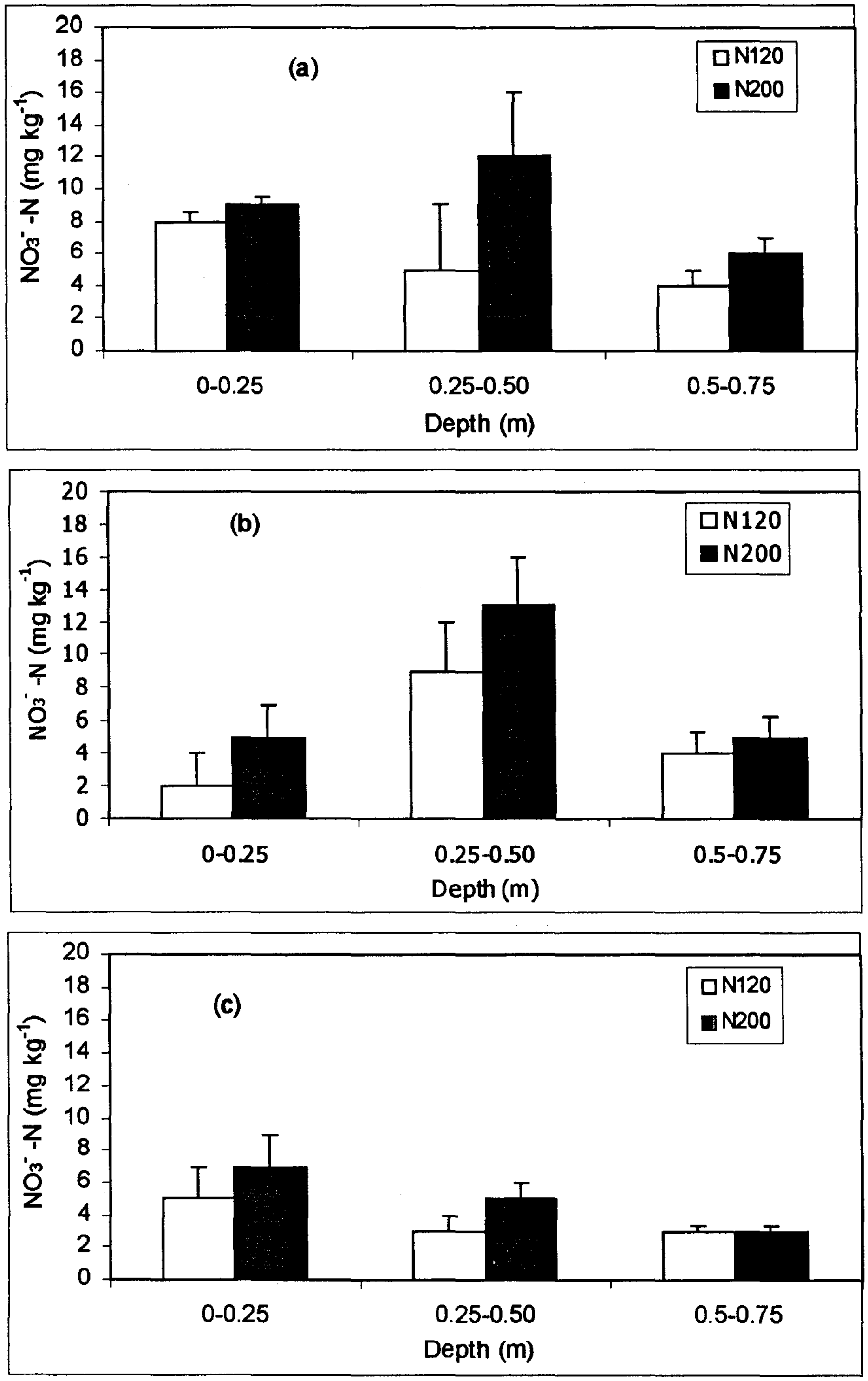

Figure 3.7: NO_3^- -N concentrations (mg kg^{-1} soil) in the soil profile under 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}) in summer (a) 1998, (b) 1999, and (c) 2000. Bars within same depth followed by different letters are significantly ($P \leq 0.05$) different. Vertical bars represent standard error of the mean (n = 9).

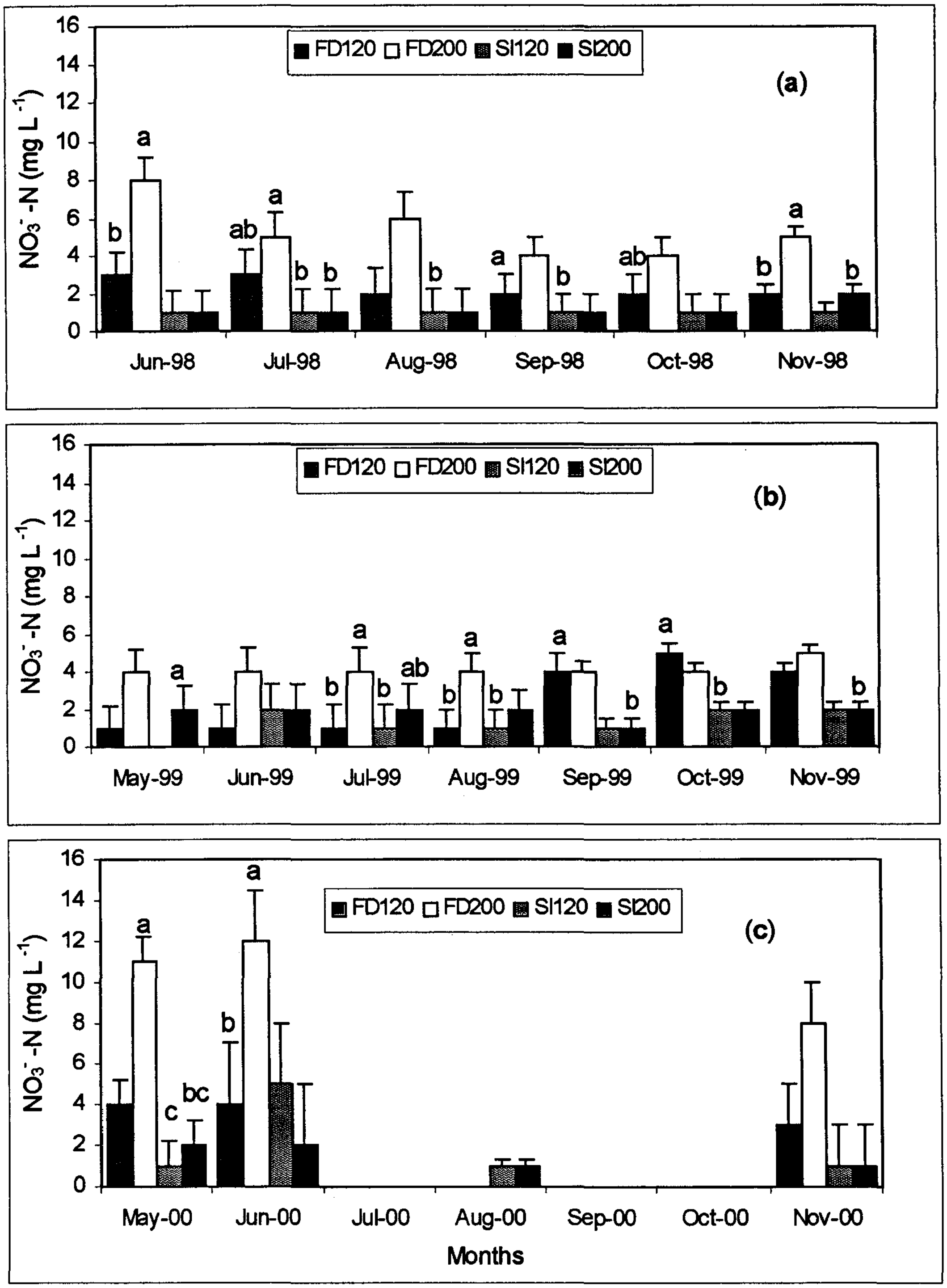

Figure 3.8: mean monthly NO_3^- -N concentration (mg L^{-1}) in subsurface drainage water under controlled drainage-subirrigation (SI) and free drainage (FD) (a) 1998, (b) 1999, and (c) 2000. Bars within same month followed by different letters are significantly ($P < 0.05$) different. Vertical bars represent standard error of the mean ($n = 3$).

PREFACE TO CHAPTER 4

In Chapter 3, we discussed the effects of water table management and N fertilization rate on corn yield, NO_3^--N concentration in the soil profile and drainage water. It was concluded that controlled drainage/subirrigation significantly reduced NO_3^--N concentration in soil-water system, probably by enhancing denitrification. Nitrous oxide emissions from denitrification is of great concern to society. Whether NO_3^--N removal by denitrification is actually beneficial to the environment without a major tradeoff depends on which denitrification gases are produced. Little is known about the proportion of denitrification gaseous end-products, namely N_2O relative to N_2O+N_2 under natural field conditions. This topic is the focus of Chapter 4, consisting of materials contained in a manuscript prepared to be submitted for publication. The format has been changed to be consistent within this thesis.

CHAPTER 4

Denitrification and Nitrous Oxide Ratio Emissions As Influenced by Water Table and Nitrogen Fertilization Application Rate

ABSTRACT

Environmental impacts of denitrification depend on two major factors: 1) the rate of N_2O production in the soil, and 2) the extent to which N_2O can be reduced to dinitrogen gas (N_2) by denitrifiers before its release to the atmosphere. Information on the ratio of N_2O to N_2O+N_2 is needed in order to devise management practices to mitigate nitrate-N (NO_3^--N) pollution without concomitantly increasing atmospheric N_2O pollution. Field experiments were conducted from 1998 to 2000 growing seasons at St- Emmanuel, Quebec, Canada, to investigate the combined impacts of water table management (WTM) and N fertilization rate on the soil denitrification (N_2O+N_2) rate, rate of N_2O production, and on the molar ratio of N_2O to N_2O+N_2. There were two water table treatments: free drainage (FD) with open drains at a 1.0 m depth from the soil surface and subirrigation (SI) with a design water table depth of 0.6 m below the soil surface, and two N fertilizer (ammonium nitrate) rates: 200 kg N ha^{-1} (N_{200}) and 120 kg N ha^{-1} (N_{120}). Denitrification rates were higher in SI than in FD plots. The N_2O fractions of denitrification constituted 35% and 11% for 1998, 19% and 18% for 1999, and 27% and 20% for 2000, under FD and SI, respectively. Denitrification rates and N_2O emissions were both only minimally affected by N fertilizer rate.

4.1 INTRODUCTION

Environmental considerations are playing an increasing role in developing management strategies that mitigate nutrient losses from agroecosystems. Currently, there is an increased emphasis on sustainable farming systems, and in particular the modification of irrigation and drainage practices to reduce pollutant levels in subsurface drain effluents. Water table management (WTM) has become a widely used technique to reduce NO_3^--N pools in the soil-water system and increase or sustain crop yield (Cooper et al., 1999; Kalita and

Kanwar, 1993; Madramootoo et al., 1993). Water table management consists of two main functions: controlled drainage (CD) or subirrigation (SI). Under CD, the drainage outlet is raised to prevent water from exiting the drain outlet. Subirrigation is achieved by installing a control mechanism at the drain outlet and supplying water to the drainage system at the outlet to maintain a desired water table depth (WTD) in the field. Raising the water table by SI increases soil water content and restricts O_2 diffusion in soil pores, thus decreasing NO_3^--N losses at the drain outlet by promoting denitrification. Drury et al. (1996) reported that a CD/SI system lowered NO_3^--N concentration in tile drainage water by 25% compared to a free drainage (FD) system. Similarly, Jacinthe et al. (1999) estimated a 40% reduction in soil NO_3^--N by denitrification in SI plots compared to FD plots.

Denitrification is the major biological process by which nitrous oxide (N_2O) enters the atmosphere. Nitrous oxide is an atmospheric trace gas, with an atmospheric lifetime of nearly 120 years and a global warming potential 320-fold that of CO_2 (IPCC, 1995). Furthermore, N_2O emissions are of serious concern because N_2O participates in the depletion of the ozone layer (Mooney et al., 1987). The depletion of the protective ozone layer increases the strength of harmful ultraviolet radiation at the earth's surface and may have detrimental effects on human and ecosystem health. However, N_2O produced in the soil may be further reduced biologically to N_2,which is carried harmlessly to the atmosphere. Understanding of the proportion of denitrification gaseous end-products is needed in order to devise management practices that reduce NO_3^--N pollution without concomitantly increasing N_2O. There is insufficient information on N_2O:N_2O+N_2 ratios under field conditions, making it difficult to identify management practices that will produce environmentally harmless N_2, rather than N_2O. Furthermore, N_2O:N_2O+N_2 ratio is variable, both spatially and temporally, and in some soils N_2O may be the dominant end product of denitrification (Granli and Bøckman, 1994; Ottow and Benckiser, 1994). This variability makes it difficult to estimate N_2O emissions from agroecosystems based on denitrification (N_2O+N_2) or N_2O emissions alone.

The objectives of this study were to examine the combined effects of water table and N fertilization rate on soil nitrate concentrations, denitrification rates, N_2O production, and N_2O:N_2O+N_2 ratios in a corn monoculture agroecosystem.

4.2 MATERIALS AND METHODS

4.2.1 Site Description and Field Management

The study was conducted on a 4.2 ha research site at St-Emmanuel, Quebec (74° 11' 15" lat., 45 ° 21' 0" long.), about 30 km southwest of the Macdonald Campus of McGill University. The soil, a Soulanges fine sandy loam [fine silty; mixed, non-acid, frigid *Humaquept*, Gleysol; (FAO classification system)], was of sedimentary origin. It overlies a clay layer to a depth of 0.5 m. Surface topography is generally flat with an average slope of less than 0.5%. In the spring of 1998 (prior to the initiation of this study) the soil contained 50 g C kg^{-1} soil in the 0-0.25 m layer, 15 g C kg^{-1} soil in the 0.25-0.55 m layer and a negligible amount of C below 0.55 m. The pH was near neutral (pH = 6.8). Experimental plots were under a conventional tillage system: moldboard- plowed to 0.20 m in fall and disked in spring which is the common practice in the region. The field was planted with corn (Pioneer hybrid 3905) at a density of 75,000 seeds ha^{-1} at a 0.75 m interrow spacing. A John Deere 700 series planter equipped with 50-mm fluted coulters completed planting on 8 May in 1998, 4 May in 1999, and 23 May in 2000. Potassium (muriate of potash, 0-0-60) was side-dressed at a rate of 90 kg K_2O ha^{-1} roughly one week before planting. In addition, the farmer applied manure (cattle slurry) to the field in spring 1998 at a rate of 20 Mg ha^{-1}. To control weeds, 1.5 kg a. i. ha^{-1} Atrazine [6-chloro-N ethyl-N (1-methylethyl)-1,3,5-traizine-2,4-diamine], 0.32 kg a. i. ha^{-1} Dicamba (3,6-dichloro-2methoxybenzoic acid), 0.32 kg a. i. ha^{-1} Bromoxynil (3,5-dibromo4-4hydroxybenzonitrile), and 1.92 kg a. i. ha^{-1} Metolachlor [2-chloro-N-(2-ethyl-6-methyl-phenyl)-N-(2-methoxy-1-methylethyl) acetamide] were applied to the field on May 13 in 1998, May 28 in1999, and June 23 in 2000.

4.2.2 Experimental Design and Field Layout

Field layout and treatment arrangements are detailed in Chapter 3. Briefly, there were two water table management treatments: FD with open drains one meter from the soil surface and SI with a design water table 0.6 m below the soil surface, factorially combined with two N fertilizer rates: 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}). Diammonium phosphate (18-46-0) was banded at planting to provide approximately 24 kg N ha^{-1} and 130 kg P_2O_5 ha^{-1}.

One month later, to reach the desired N fertilization levels, 97 kg N ha^{-1} and 178 kg N ha^{-1} were broadcast as ammonium nitrate (34-0-0) for the N_{120} and N_{200} treatments, respectively. This second application occurred on June 8, 1998; June 10, 1999; and June 20, 2000.

There were three blocks, 120 m wide and 75 m long, each containing four treatment plots, 15 m wide and 75 m length. In the middle of each plot, 75 mm diameter subsurface drain pipes were installed, at 1.0 m depth, on a 0.3 % slope. Blocks were separated by a 30m wide strip of undrained land. The SI treatment was imposed two weeks after planting and maintained until crop maturity in late September. Subirrigation was switched to FD on September. 28, 17, and 15 for 1998, 1999, and 2000, respectively.

Due to deep seepage it was difficult to maintain water tables at the desired depth. Following heavy rainfall events, pumping was stopped in SI plots and excess water drained until a 0.6 m water table depth was achieved in the field. Water table depths were measured once or twice a week during the growing season by inserting a sonic water sensor mounted on a graduated rod. Rainfall and air temperature data were obtained from an Environment Canada weather station situated 500 m from the experimental site. Soil temperature was recorded at same sampling date and location as for denitrification measurements using a water-resistant probe thermometer (Hanna instrument, HI9024/HI9025) inserted to a depth of 0.20 m below the soil surface.

4.2.3 Sampling Strategy and Analysis

For total denitrification and nitrous oxide measurements, soil cores (0-0.15 m depth) were collected weekly from late May to July, and bi-weekly from August to October during the 1998 and 1999 growing seasons, and approximately once a month from June to October in 2000. Aluminum cylinders (55 mm diameter x 150 mm long), perforated along the sides in a 50 mm grid to enhance acetylene gas diffusion, were used to collect soil cores. On each occasion, paired soil cores were taken from a randomly selected non-wheel-tracked row. Great care was taken to minimize sampling disturbance. Samples were never taken from the same location more than once within the growing season. Soil cylinders were placed in 2 L plastic jars fitted with rubber septa for gas sampling. Samples were then incubated outdoors for 24 h to mimic field conditions. One cylinder was incubated without C_2H_2, and the other was supplied with C_2H_2 (5 % vol. vol.$^{-1}$) to block enzymatic reduction of N_2O to N_2, so that accumulated N_2O+N_2 from denitrification can be measured as N_2O (Yoshinari et al., 1977). The mole fraction of N_2O, (N_2O:N_2O+N_2 ratio), was computed as the ratio of N_2O emission without C_2H_2 to the rate of emission with C_2H_2 (Aulakh et al., 1984). In this context, N_2O production is the rate of N_2O evolved from core samples without C_2H_2, whereas denitrification rate is the N_2O produced with C_2H_2.

Total N_2O production was quantified in a fashion similar to the procedure of MacKenzie et al. (1997). Headspace gas was thoroughly mixed by inserting a syringe and pumping several times before gas sampling. About 4 mL of headspace gas were removed from the jars and injected into a gas chromatograph [GC, (5870 series II Hewlett Packard), Avondale, PA] equipped with a ^{63}Ni electron capture detector (ECD) using Ar:CH_4 (95:5) as a carrier gas, with oven and detector temperatures adjusted to 70 ° C and 400 ° C, respectively.

To assess NO_3^--N concentration in the soil, three sets of soil samples were taken at a depth of 0-0.2 m using a hand-held auger at the same time as sampling for denitrification measurements. All soil samples for NO_3^--N analysis were stored at - 4°C for one to three weeks. The samples were then mixed thoroughly and shaken with 100 mL of 1 M KCl (1:10 soil: extractant) for 60 min, filtered through Whatman # 5 filter paper, and frozen before NO_3^--N analysis. Nitrate-N was quantified using a Lachat flow injection autoanalyzer (Lachat Quickchem, Milwaukee, WI).

Soil water content was determined by oven drying soil cores at 105 ° C for 48 h. Soil bulk density (BD) at each sampling date was determined by the core method and total porosity (Pt) was calculated assuming a particle density of 2.65 Mg m^{-3}. Percent water filled pore space (%WFPS) was calculated as:

$$\%WFPS = \frac{\%H_2O}{Pt} x100 \qquad (1)$$

Where, $\%H_2O$ is the volumetric soil water content calculated as follows:

$$\%H_2O = \frac{\text{Mass } H_2O}{\text{Mass Dry Soil}} \times BD \times 100 \qquad (2)$$

4.2.4 Statistical Analysis

Analysis of variance (ANOVA) was performed separately on individual sampling dates. Data were analysed as a split plot with water table being the main plot and fertilizer N treatments as the sub-plot, using the interaction between block and the main plot (water table) as an error term. Spearman's rank correlation (a non-parametric test that does not require the data to follow a known distribution) was used to assess the relationship between denitrification gaseous products and selected soil properties. Statistical analyses were conducted using Statistical Analysis System, SAS, release 6.12 for windows (SAS Institute, Cary, NC). Differences between values were declared significant at the 5% probability level.

4.3 RESULTS AND DISCUSSION

4.3.1 Climatic Data

There was a substantial variation in timing and quantity of precipitation received during the 3 years of this study. Total seasonal (May-October) rainfall in 1998 was about 29% greater than the regional 30-year norm (Table 4.1). As much as 34% of the total seasonal rainfall in 1998 occurred in June. July 1998 was also much wetter than the norm, accounting for about 20% of the seasonal total (Table 4.1). Rainfall in 1999 was 13% higher than the norm, with most of the rainfall occurring in September and October. Similarly, the precipitation in the 2000 season was about 12% higher than the norm, with May being the wettest month and October being the driest. Mean monthly temperatures were slightly higher in 1998 and 1999 than the norm, but not in 2000 which was near the norm (Table 1). Soil temperatures (0-0.2 m depth), measured at the same time as soil samples were collected denitrification measurements, exhibited similar patterns as air temperature and were unaffected by either water table or N fertilization treatments (data not shown). In general, the field conditions during the study period were wetter and, sometimes, warmer than experienced from 1961-1990.

4.3.2 Nitrate Concentrations in the Soil Surface

A trend of greater surface (0-0.2 m) soil NO_3^--N content under FD than SI was evident at most of the sampling dates during our three year study period (Fig. 4.1). Overall, soil NO_3^--N concentrations were 31 to 52% lower under SI than FD during this study. This finding clearly indicates that maintaining the water table at a shallow depth can be a useful tool in reducing NO_3^--N accumulation in the soil. Consequently, transport of NO_3^--N into deeper soil layers may be reduced. Under certain conditions, considerable amounts of NO_3^--N may be removed from soil by denitrification. Kessavalou et al. (1996), using N^{15} techniques, found that in a wet year, 13% of the applied N was lost by denitrification during the growing season of irrigated corn. Decrease in soil NO_3^--N levels from 30% to 60%, relative to FD, have been reported under SI. For example, Fogiel and Belcher (1991) found that controlled drainage-subirrigation (CD-SI) decreased NO_3^--N loading through drainage by 25-59% over a 2-year period compared with conventional drainage. Jacinthe et al. (1999) reported 24 to 43% reductions in NO_3^--N leaching using CD-SI management techniques.

Fertilizer N rate treatments [120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200})] had no clear effects on NO_3^--N concentrations in surface soil (Fig. 4.2). This finding may suggest that managing fertilization rates alone may not be a sufficient strategy to overcome the problem of NO_3^--N loading in the soil-water system. Sainju et al. (1998) reported that even with no fertilization, significant concentrations of residual NO_3^--N accumulated beyond the root zone because of continued mineralization from soil and crop residues.

4.3.3 Effects of Water Table on Denitrification Rates

Denitrification rates in 1998 were significantly higher under SI than FD at 6 of the 15 sampling dates and not significantly different at the remaining dates (Fig. 4.3 a). On a seasonal basis, denitrification rates under SI were approximately twice as high from soils under SI (1.7 kg ha^{-1}) than soils under FD (0.74 kg ha^{-1}). The highest mean denitrification rate in 1998 was recorded on 16 June under both water table management treatments. This pulse of denitrification activity occurred shortly after N fertilizer (second application) was applied (10 June) and the plots received 20 mm of rainfall (15 June), raising WTD to about 0.50 m for the SI plots and 0.90 m for the FD plots below the surface soil. After mid July (14 July),the differences between the water table treatments were minimal (Fig. 4.3 a) although measurable denitrification fluxes were produced until the end of the growing season.

Denitrification rates in 1999 were generally less than 50 g N ha^{-1} d^{-1} throughout the cropping season except on 6 July (Fig. 4.3 b). This was particularly true for the FD system. Denitrification rates were significantly greater under SI than FD in 5 out of 15 sampling dates (Fig. 4.3 b). Seasonal denitrification rates under SI were 2-fold those under FD (1.2 kg ha^{-1} *vs* 0.6 kg ha^{-1}).

In the 2000 cropping season, the two water treatments responded differently: under FD there was only a moderate increase in denitrification on June 22 whereas under SI, denitrification peaked on August 2 through August 17 (Fig. 4.3 c). Seasonal production of N_2O+N_2 and N_2O under SI was approximately 2-fold that produced under FD (0.88 *vs* 0.40 kg ha^{-1}). Overall, denitrification rates under SI were approximately 2-fold greater than FD in all three cropping seasons studied.

Subirrigation affects denitrification by altering soil moisture regime. Greater denitrification losses under SI than FD were primarily associated with greater moisture content (Fig. 4.4), which might have created a relatively more anaerobic environment which enhanced denitrification. The influence of soil moisture content on denitrification is due to a reduction in O_2 concentration which is essential before denitrification can proceed. For example, in 1998 maximum denitrification rate coincided with the second highest soil water contents which occurred on 17 June (Fig 4.4 a). Further indication of the critical role of soil moisture in controlling denitrification can be evidenced by the lower denitrification rates in 1999 and 2000 growing seasons during which WFPS values were mostly below 60% (Fig. 4.4 b c), a critical threshold below which denitrification is limited (Bøckman and Granli, 1994; Weier et al., 1993).

4.3.4 Seasonal Trends in Denitrification

Averaged across water table treatments, denitrification rates in 1998 were about 1.5-fold and 2-fold greater than in 1999 and in 2000, respectively. However, since there were fewer sampling occasions in 2000 than in 1998 and in 1999, one must be cautious when comparing the seasons. The greater N gas production in 1998 compared to other years appears to be related to the application of manure (cattle slurry) in the spring. Ellis et al. (1998) and Chang et al. (1998) reported cattle slurry-amended fields showing significantly greater N_2O emissions than fields that had not received cattle slurry. Beauchamp et al. (1996) showed significantly greater denitrification losses when cattle manure was applied than when equivalent amount of N was applied in mineral form. Manure amendments not only enhance the N supply, but also provide a source of carbon to the denitrifying community.

The lower denitrification rates during 1999 compared to1998 were likely due to the dry conditions prevailing at the beginning of the growing season. May was 31% drier than the normal, leading to a sharp drop of water table depth in subsequent sampling dates (0.85 m in SI and 1.2 m in FD below the soil surface). Another possible explanation may be related to the inverse relationship between temperature and soil moisture. Although June and July precipitations were both slightly higher than normal (5 and 11%, respectively), these months were much warmer (3 and 1.3 ° C, respectively) than normal (Table 4.1). Drier conditions at the beginning of the season coupled with high temperatures increasing evapotranspiration losses may not favour denitrification. While not definitive, the slightly lower denitrification activity in 1999 and 2000, than 1998, may suggest that the stimulating effect of the manure amendment on N_2O production was not carried-over to the following seasons. As mentioned previously, the manure amendment was made in the spring of 1998 at 20 Mg (wet wt) rate. Further research will be needed to verify how long denitrification is stimulated after manure is applied.

4.3.5 Effects of Water Table on Nitrous Oxide Emissions

Nitrous oxide emissions were not significantly affected by water table treatment except in 1999 (Fig. 4.5), but a number of interesting trends were observed. The most striking observation in 1998 regarding N_2O production is that the maximum rate of denitrification measured on June 17 corresponded to the greatest N_2O evolution under FD but the lowest N_2O evolution under SI (Fig. 4.5 a). One plausible explanation is that SI acted as a sink for

N_2O ($N_2O \rightarrow N_2$) at a greater rate than FD as evidenced by the high N_2O+N_2 emission on June 17, 1998 (Fig. 4.3). This is a clear indication that high rates of denitrification may not necessarily be accompanied with high N_2O emissions. A marked dissimilarity among the three growing seasons with respect to N_2O emission was the fact that total seasonal N_2O emission under SI was approximately 2-fold greater than under FD in 1999 and 2000, whereas in the 1998 season the FD treatment produced more N_2O than under SI. These findings indicate that wetter soils produce higher rates of denitrification, with N_2O comprising a minor part of the total denitrification rates. Supporting this conclusion is the evidence that 1999 was the only season where the difference between SI and FD with respect to N_2O was significant on a number of occasions (Fig. 4.5 b). This is consistent with Qian et al. (1997) who showed that N_2O flux was greater in drier growing seasons.

4.3.6 Effects of N Rate on Denitrification and N_2O

Denitrification rates and N_2O emissions were only minimally affected by N fertilization throughout the study period (Figs. 4.6, 4.7). Our results differ from other studies that have reported an effect of nitrogenous fertilizers on denitrification (MacKenzie et al., 1997; Ellis et al., 1998; Sănchez et al., 2001). In soils with limited NO_3^--N content, the application of N fertilizers can stimulate denitrification. It is widely accepted that when NO_3^--N concentration is not the limiting factor, the denitrification rate follows a zero-order equation with respect to the concentration of NO_3^- -N (Müller et al., 1997; Vallego et al., 2001). We believe that soil NO_3^--N availability was not the limiting factor for denitrification activity in this experiment since neither denitrification nor N_2O emissions were correlated significantly with soil NO_3^--N (Table 4.3). It is possible that the difference in the levels of N applied in the two fertilizer treatments (80 kg N ha^{-1}) was not large enough to produce a statistically detectable effect on denitrification and N_2O emissions.

4.3.7 Nitrous Oxide and Denitrification Ratios

The ratio of N_2O:N_2O+N_2 provides a measure of microbial ability to convert NO_3^--N to N_2. The ratios were less than 1.0 during the three growing seasons ((Fig. 4.8). Exceptions were June 10 in 1998 (Fig. 4.8 a) and August 17 in 1999 (Fig. 4.8 b). Higher N_2O to N_2O+N_2 ratios would suggest that the nitrification process might have contributed to the N_2O flux from samples not treated with C_2H_2. It is interesting to note that both occasions in which N_2O:N_2O+N_2 ratios were greater than 1.0 occurred in FD plots, possibly because of the better aeration due to lower WFPS (Fig. 4.4). Granli and Bøckman (1994) suggested that increased aeration could increase the proportion of N_2O produced by denitrification. High N_2O to N_2O $+N_2$ ratios are characteristic of fairly well-aerated soils, in which N_2O can easily diffuse away from being further reduced to N_2 by denitrifying organisms (Webster and Hopkins, 1996). Under SI, the N_2O:N_2O+N_2 ratios did not exceed 0.25, indicating that N_2 formed a significant portion of the denitrification gaseous end-products throughout the study period. Other research reports indicate that the percentage of N_2O in denitrification gaseous products decreases with increasing soil water content until N_2 becomes the major gas evolved (Rolston et al., 1978). Ellis et al. (1998) urged, however, that direct comparison of N_2O and N_2O+N_2 flux should be made with caution, as C_2H_2 inhibits nitrification and may underestimate total denitrification rates.

Although the contribution of nitrification to N_2O emission cannot be quantified directly on the basis of the data obtained in this study, during the 3 years of investigation there were the only two observations (June 10 in 1998 and August 17 in 1999) of greater N_2O production from cores without C_2H_2 amendments than C_2H_2-treated cores (Fig. 4.8 a b). Mogge et al. (1998) found N_2O:N_2O+N_2 emission ratios greater than 1.0 in about 20% of all measurements they made and concluded that denitrification was the most important process producing N_2O emissions. Similarly, Henrich and Haselwandter (1997) occasionally observed higher N_2O emission from soil samples incubated without C_2H_2, but noted that these differences were not significant ($P > 0.05$).

In 1998, denitrification products contained, on average, 11% and 35% N_2O, whereas in 1999, 19% and 18% of denitrification was N_2O and in 2000, 27% and 20% of denitrification was in the form of N_2O, under FD and SI, respectively. We conclude that N_2O was not the dominant denitrification product released from soils under corn ecosystem. This conclusion is consistent with those of Daum and Schenk (1996) who found the proportion of N_2O in gaseous N losses to range from 3% to 20%. Kliewer and Gilliam (1995) have found N_2O to be a very small (2%) component of denitrification and unaffected by water table management.

Furthermore, our findings in 1998 and 1999 show that denitrification was significantly correlated with N_2O, but not with soil NO_3^--N concentrations (Table 4.3), further indicating that N_2O emitted was derived from the denitrification process rather than nitrification. There was a significant negative relationship ($r = -0.6$; $P < 0.04$ for 1998 and $r = -0.7$; $P < 0.01$ for 1999) between WFPS and soil NO_3^--N, suggesting NO_3^--N concentrations were lowest when soils were wettest. Water filled pore space was also highly positively correlated with both denitrification and N_2O emissions in 1998 and 1999 (Table 4.3). In 2000, the relationship was poor.

Further evidence of the impact of WFPS on denitrification was established by regression analysis. The regression equations are:

In 1998: Denitrification = -41 + 0.83 (WFPS); $R^2 = 0.83$*** (3)

In 1999: Denitrification = -2.5 + 0.5 (WFPS); $R^2 = 0.53$** (4)

Where denitrification = $N_2O + N_2$ emission (kg ha^{-1} $season^{-1}$). Water filled pore space poorly predicted denitrification rate in 2000, confirming the results obtained with the correlation analysis.

These results clearly demonstrate that WFPS may be a determining factor with regard to denitrification losses, with greater losses measured as the water content of the pore space rises. Maag and Vinther (1996) and Weier et al. (1993) found that N_2O to N_2 ratios decreased with increasing soil moisture. It appears that N_2O diffuses more slowly out of a wetter than a drier soil, as there is a greater possibility for its reduction under higher water content. Model estimates suggest that there is a large lag phase between N_2O production and N_2O release at the soil surface due to low gaseous diffusion under wet soils (Jury et al., 1982).

4.4 SUMMARY AND CONCLUSIONS

Environmental impacts of N_2O emitted from soils depends on the quantities produced, and the extent of its reduction to N_2 by denitrifiers before denitrification products are released into the atmosphere. Although denitrification rates were greater under SI than FD, WTM did not affect the percentage of N_2O evolved to the atmosphere. This is an indication that N_2O+ N_2 losses were greater in wetter soils, but that N_2O comprised a minor part of the total N gaseous emissions. These findings confirm the importance of assessing N_2O:N_2O+N_2 losses in order to gain better understanding of the ecological significance of denitrification. We propose that WFPS is the determining soil factor with regard to denitrification losses, with greater losses produced as the water content of the pore spaces rises. Higher WFPS in 1998 facilitated a complete reduction of denitrification process with N_2O comprising only a minor fraction of the total N gaseous emissions. We found that denitrification rates and nitrous oxide emission were not affected by N fertilizer treatments in our corn agroecosystem. Lowering N fertilizer rate alone, therefore, may not be a sufficient strategy to overcome N_2O pollution.

Table 4.1: Mean monthly precipitation and air temperature during the growing seasons of 1998, 1999 and 2000, compared to the long term (1961-1991) mean measured at Côteau-du-Lac weather station.

	Air temperature (°C)				Precipitation (mm)			
Month	1998	1999	2000	1961-1990	1998	1999	2000	1961-1990
May	16.5	15.6	13	12.4	69.6	53.2	143	76.3
June	18.4	20.2	17	17.3	230	95	111	90.1
July	20	21.5	19	20.2	128	105	86	94.6
August	19.6	18.6	19	19	101	60.2	116	94
September	15.1	17	14	14.1	89.4	169	94	90.6
October	9	7.4	8.5	7.7	53.6	107	33	76.7
Total	--	--	--	--	672	489	583	522.3
Mean	16.4	16.7	15.1	15.1	--	--	--	--

Table 4.2: Summary of analysis of variance based on seasonal average values.

Variable	Denitrification			N_2O			$N_2O:N_2O+N_2$			% WFPS		
	1998	1999	2000	1998	1999	2000	1998	1999	2000	1998	1999	2000
Model	*	NS	**	NS	NS	NS	*	NS	NS	*	*	*
Block	NS §	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	*
WTD†	*	NS	*	NS	NS	NS	NS	**	*	*	*	*
N‡	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
WTD*N	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS

† Water table depth; ‡Nitrogen fertilizer rate

§ Not significant at $P \geq 0.05$

*, ** Significant at $P \leq 0.05$ and 0.01, respectively, based on F test

Table 4.3: Correlation between seasonal average values of some selected parameters

	1998				1999				2000			
	N_2O+N_2	N_2O	NO_3^-	WFPS	N_2O+N_2	N_2O	NO_3^-	WFPS	N_2O+N_2	N_2O	NO_3^-	WFPS
N_2O+N_2	--	r = 0.7	r =0.4	r = 0.9	--	r = 0.8	r = -0.5	r = 0.7	--	r = -0.4	r = -0.1	r = 0.3
		(0.009)	(0.18)	(0.0001)		(0.001)	(0.08)	(0.01)		(0.1)	(0.8)	(0.3)
N_2O	--	--	r =0.3	r = -0.3	--	--	r = -0.5	r = 0.8	--	--	r = -0.5	r = 0.8
			(0.4)	(0.4)			(0.08)	(0.004)			(0.09)	(0.001)
NO_3^-	--	--	--	r = -0.6	--	--	--	r = -0.7	--	--	--	r = -0.6
				(0.04)				(0.01)				(0.04)

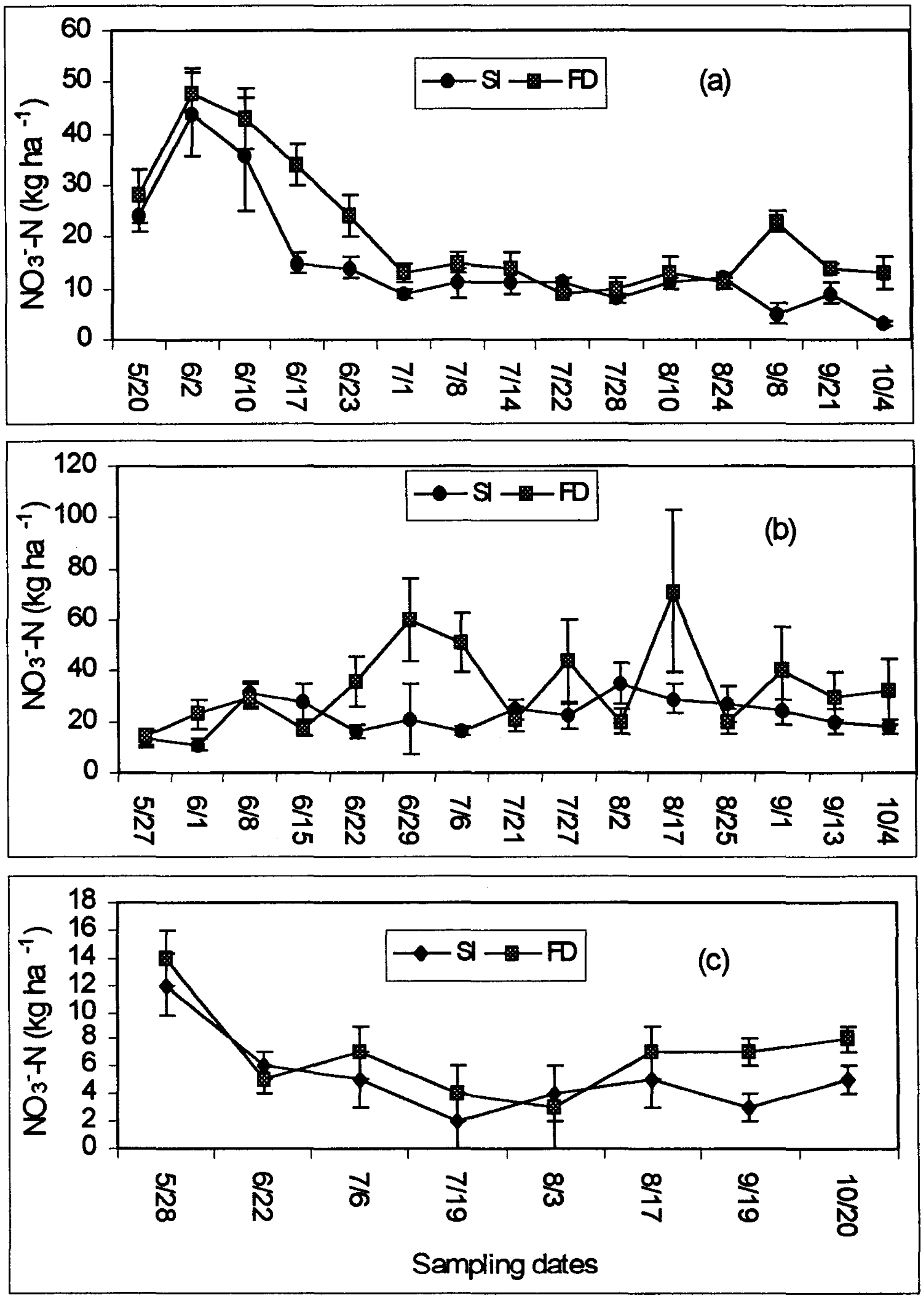

Figure 4.1: Soil surface (0-0.2 m) NO_3^--N content under subirrigation (SI) and free drainage (FD) treatments in 1998, 1999, and 2000. Vertical bars represent standard error of the mean, n = 3.

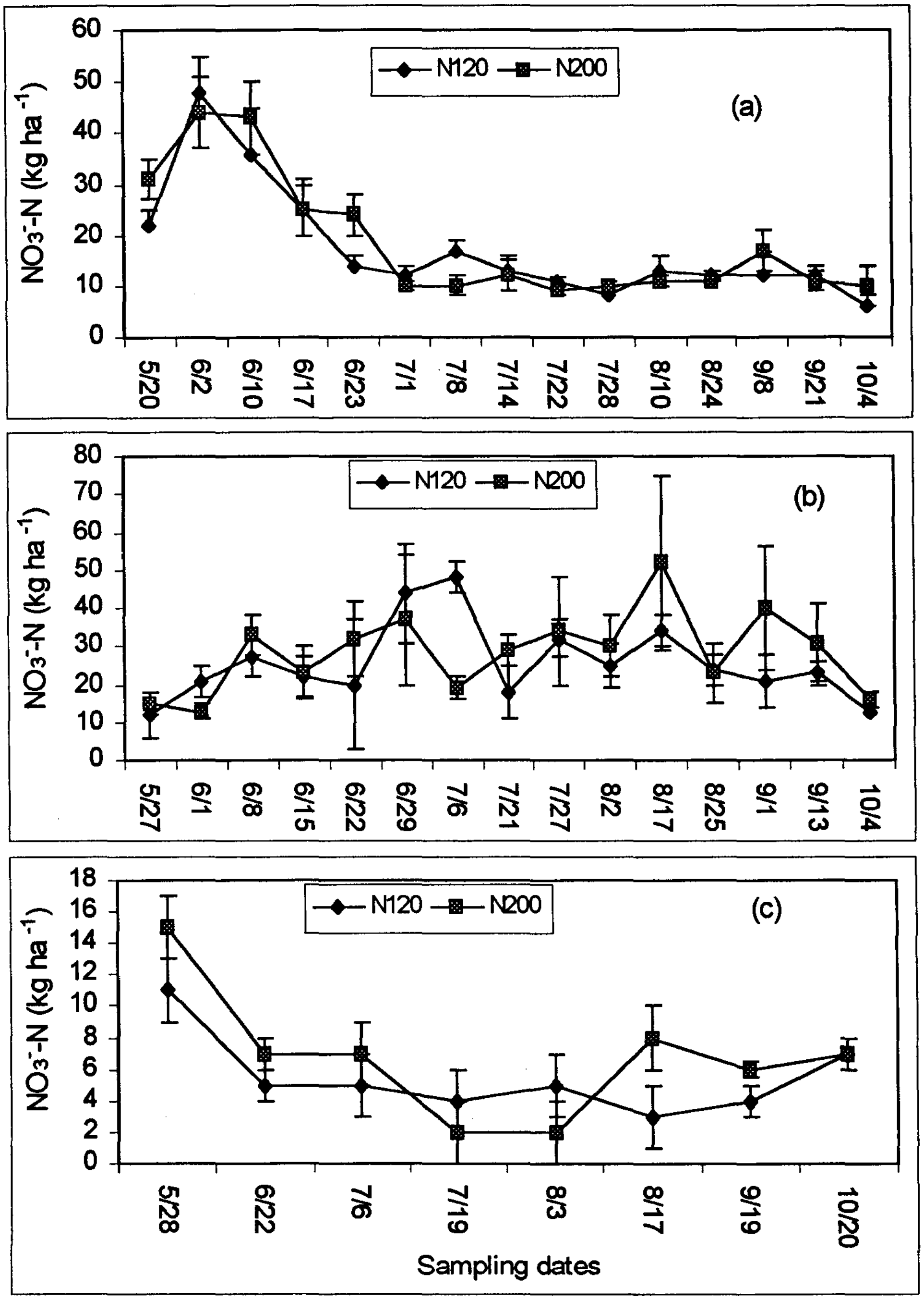

Figure 4.2: Soil surface (0-0.2 m) NO_3^--N content under 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}) application rates in 1998, 1999, and 2000. Vertical bars represent standard error of the mean, n = 3.

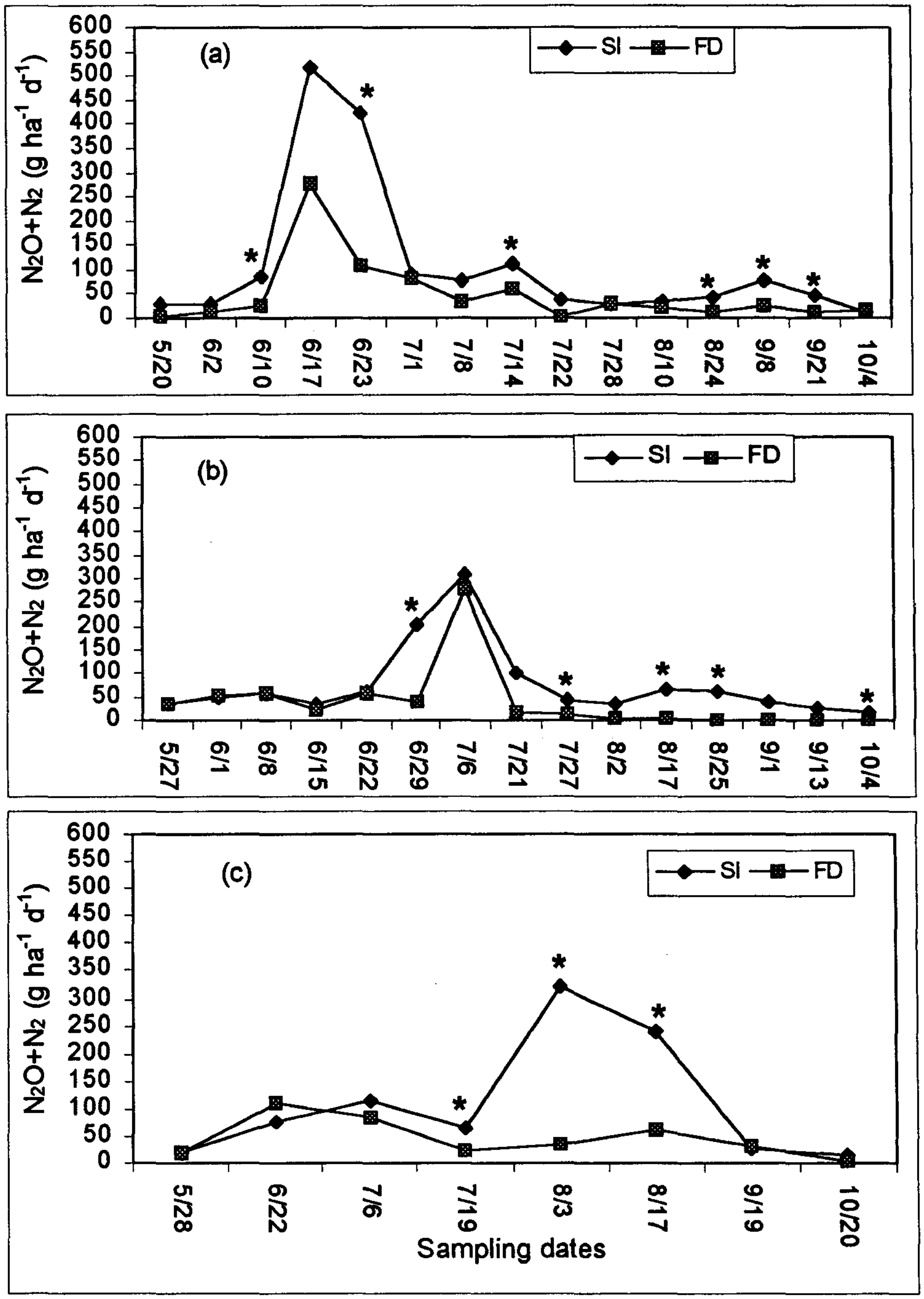

Figure 4.3: Soil surface (0-0.15 m) denitrification rates under subirrigation (SI) and free drainage (FD) treatments in (a)1998, (b) 1999, and (c) 2000. Asterisks indicate the differences between the two points on the same sampling dates are significant at ($P \leq 0.05$).

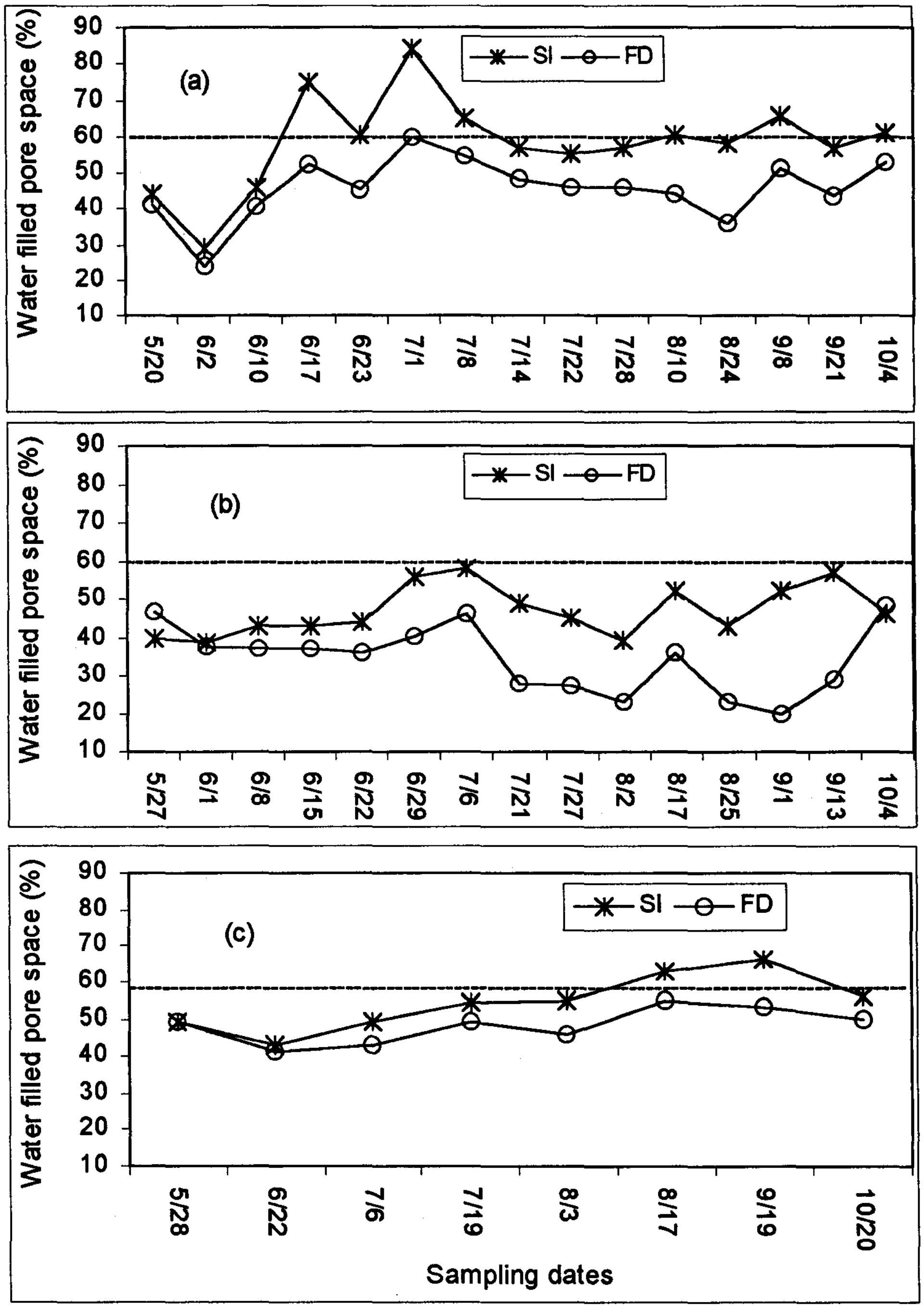

Figure 4.4: Mean soil water content expressed in percentage of water filled pore space (%WFPS) under free drainage (FD) and subirrigation (SI) treatments in (a) 1998, (b) 1999, and (c) 2000.

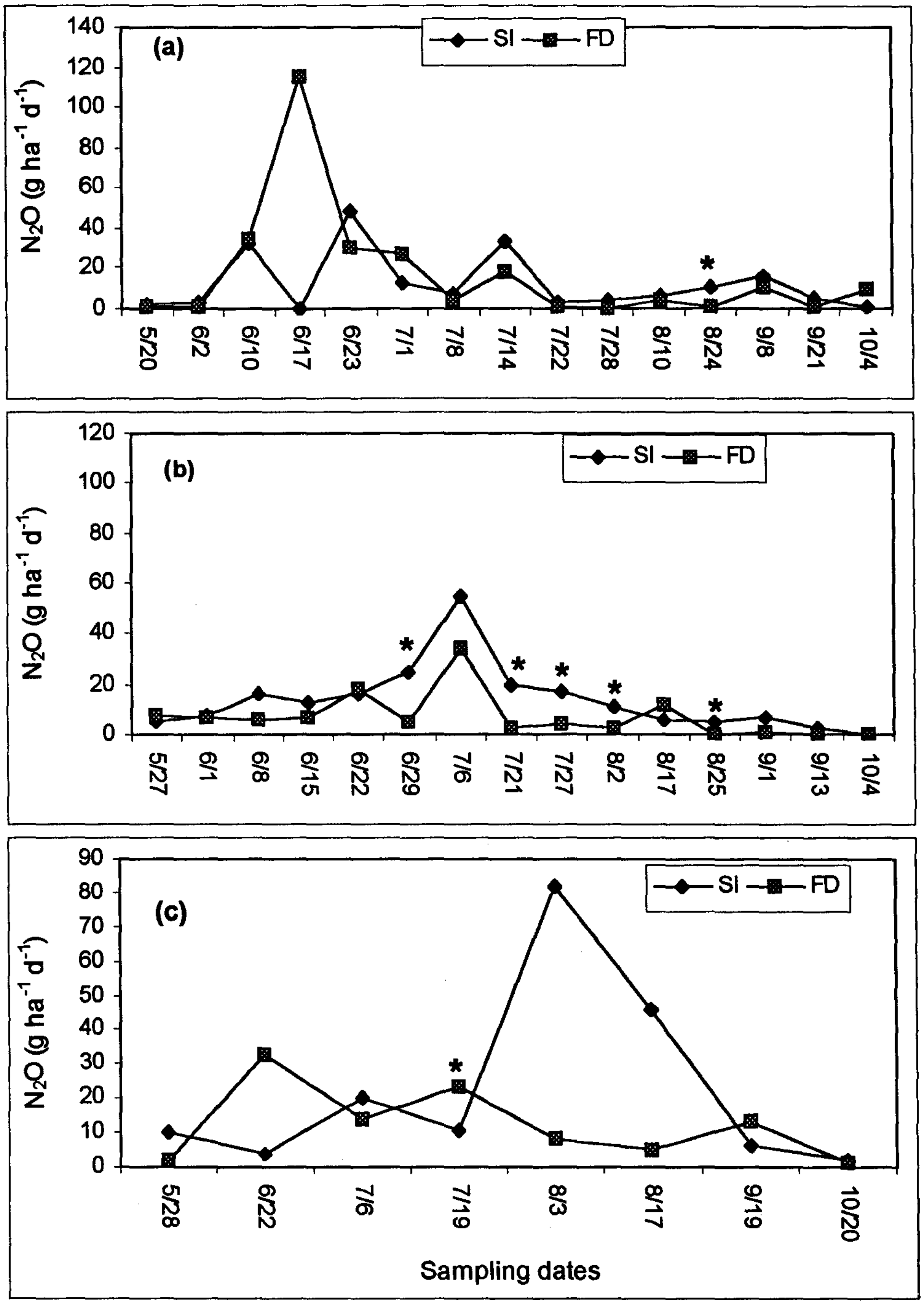

Figure 4.5: Nitrous oxide emission at the soil surface (0-0.15 m) as influenced by subirrigation (SI) and free drainage (FD) treatments in (a)1998, (b) 1999, and (c) 2000. Asterisks indicate the differences between the two points on the same sampling dates are significant at ($P \leq 0.05$)

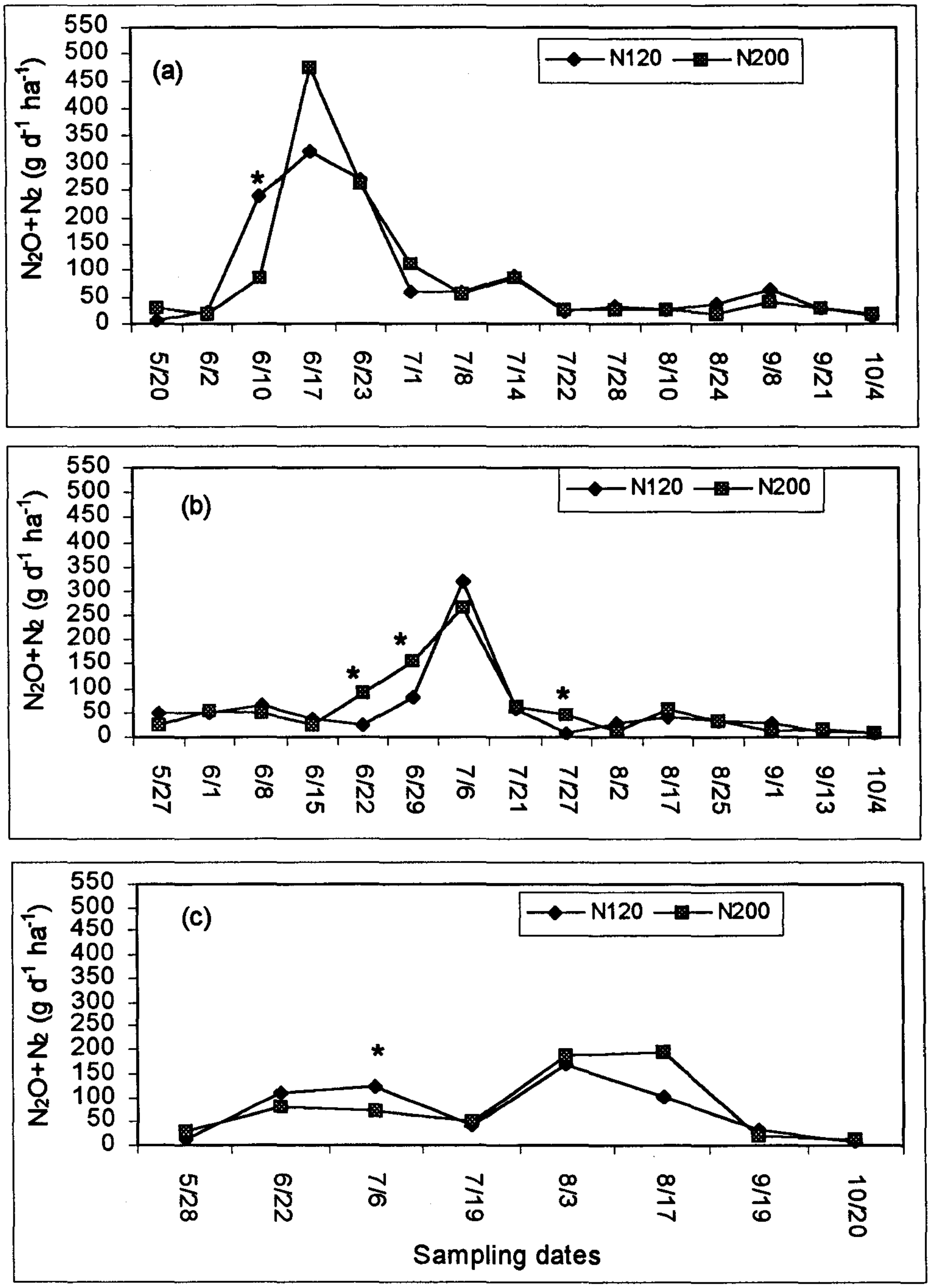

Figure 4.6: Soil surface (0-0.15 m) denitrification rates under 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}) N application rate treatments in (a)1998, (b) 1999, and (c) 2000. Asterisks indicate the differences between the two points on the same sampling dates are significant at ($P \leq 0.05$).

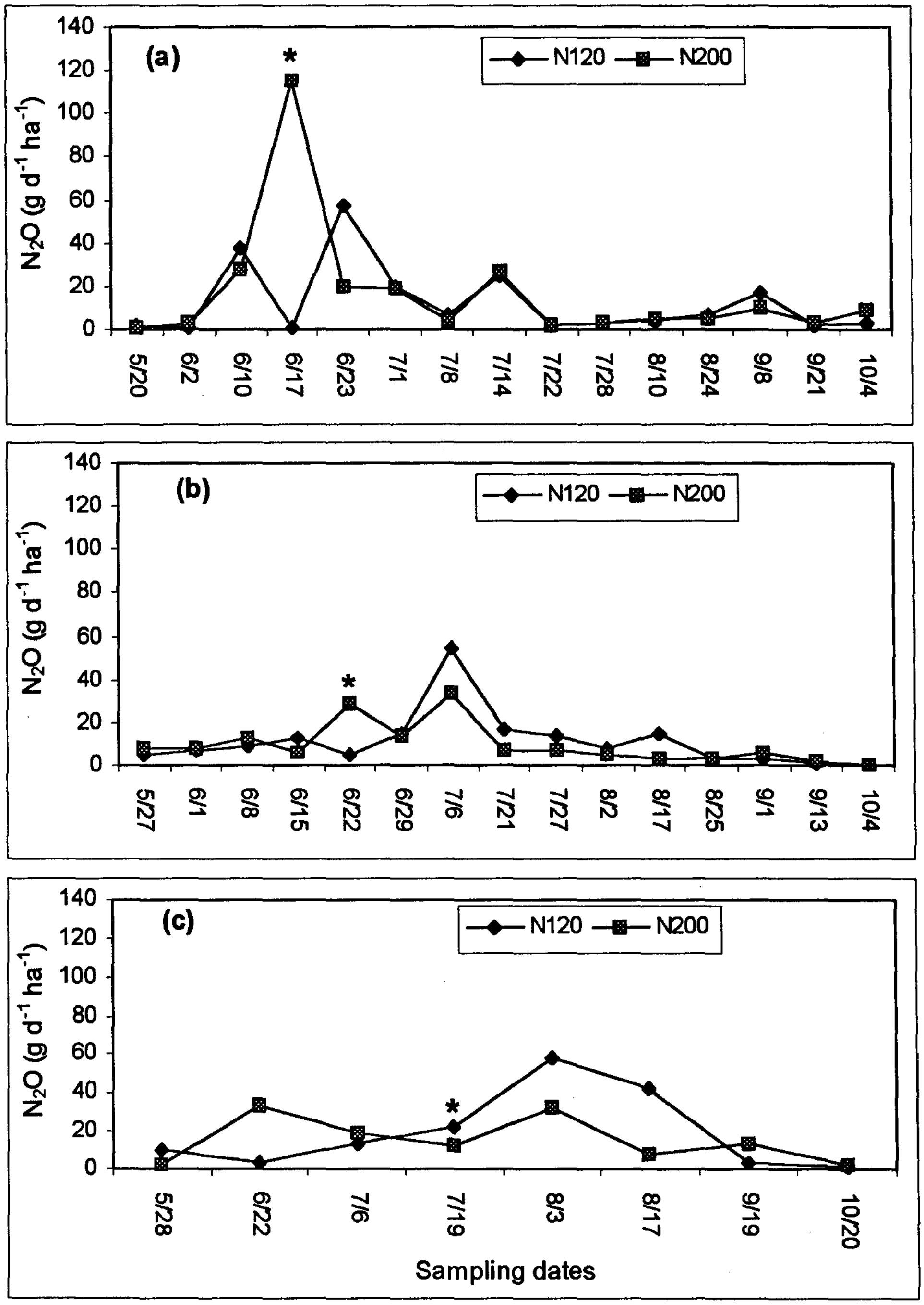

Figure 4.7: Nitrous oxide emission at the soil surface (0-0.15 m) under 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}) N fertilizer treatments in (a)1998, (b) 1999, and (c) 2000. Asterisks indicate the differences between the two points on the same sampling dates are significant at ($P \leq 0.05$).

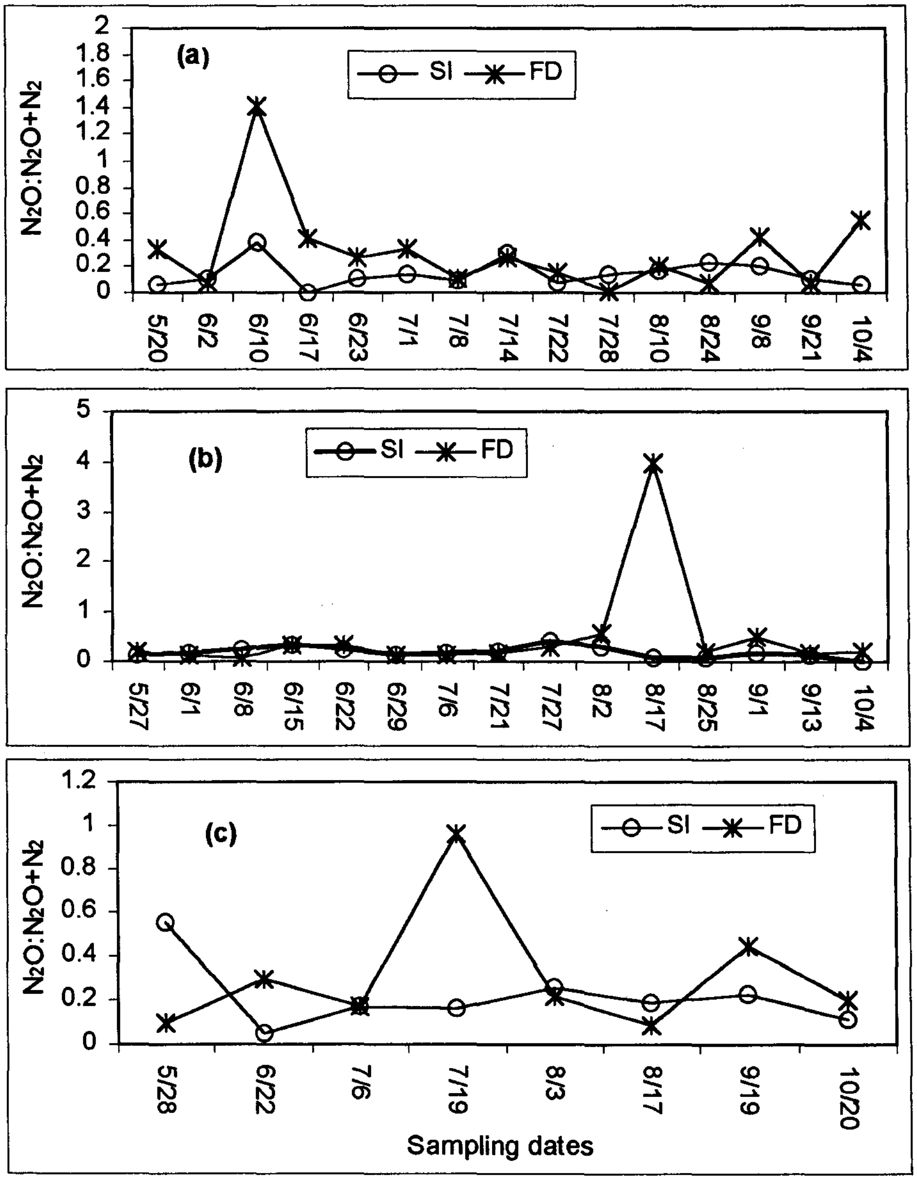

Figure 4.8: Mean N_2O:N_2O+N_2 ratios at the soil surface (0-0.15 m) as influenced by subirrigation (SI) and free drainage (FD) treatments in (a)1998, (b) 1999, and (c) 2000.

PREFACE TO CHAPTER 5

Chapter 4 addressed denitrification rates and N_2O emission at the soil surface (0-0.15 m). Denitrification occurrence below the soil surface layers is of interest since it can alleviate NO_3^--N pollution of underground water with little chance of contributing N_2O emissions to the atmosphere. In Chapter 5, we report findings of a two year research (1999-2000) at St. Emmanuel site in which the extent of denitrification and N_2O in subsurface soil (0-0.15, 0.15-0.30, and 0.30-0.45 m) was investigated as influenced by water table and N fertilization rate. Sampling frequency was less intense than when only one soil depth (0-0.15 m) was sampled as reported in Chapter 4, because of the heavy labor commitment required by soil core sampling and time constraints for the incubations.

Materials contained in this Chapter are being prepared for publication. The format has been changed to be consistent within this thesis.

CHAPTER 5

Surface and Subsurface Nitrous Oxide Emission and Denitrification Ratios From Sandy Loam Soil in a Corn Monoculture Field

ABSTRACT

Nitrous oxide (N_2O) production in subsurface soils is only poorly understood because most research into denitrification has concentrated on the upper soil layers (0-0.15 m). This study, undertaken during the 1999 and 2000 cropping seasons, was designed to examine the effects of water table management (WTM) and N application rate on subsurface (0-0.45 m depth) soil N_2O emissions from a corn (*Zea mays* L.) field. There were two water table treatments: free drainage (FD) with open drains at 1.0 m from the soil surface and subirrigation (SI) with a water table depth of 0.6 m below the soil surface factorially combined with two N fertilizer (ammonium nitrate) rates: 200 kg N ha^{-1} (N_{200}) and 120 kg N ha^{-1} (N_{120}). During both growing seasons greater denitrification rates were measured in SI plots than FD plots, particularly in the surface soil (0-0.15 m) and at the intermediate (0.15-0.30 m) soil depth. Denitrification rates under SI were 2.7-fold and 2-fold those of FD in 1999 and 2000, respectively. Denitrification rate and N_2O emissions were unaffected by N rate at any soil depth. Overall, half of the denitrification occurred at the 0.15-0.30 m and 0.30-0.45m soil layers combined under both water treatment. Consequently, sampling of the 0-0.15 m soil layer alone may not give an accurate estimation of denitrification losses. Greater denitrification rates under the SI treatment were not accompanied with greater N_2O emissions, as ratios of N_2O:N_2O+N_2 were lower in SI than FD plots. The reduced N_2O production under SI was caused by a more complete reduction of N_2O to N_2 which resulted in a decrease in the N_2O:N_2O+N_2 ratio. The N_2O:N_2O+N_2 ratio was not affected by N rate.

5.1 INTRODUCTION

Contamination of water resources by nitrate-N (NO_3^--N) has been widely documented as a serious problem in many areas (Randall and Mulla, 2001; Patni et al., 1998; Milburn et al., 1997; Spalding and Exner, 1993). Significant leaching of NO_3^--N is associated with conditions that allow NO_3^--N accumulation in the soil profile. When NO_3^--N is translocated to a lower depth of soil profile, it becomes unavailable for plant uptake and a danger to the quality of the underlying water systems. In addition, Hatfield et al. (1999) estimated that most of NO_3^--N passing through the root zone is intercepted by the tile drains and moved as discharge into surface waters. Strategies to reduce NO_3^--N pollution should therefore seek to prevent accumulation of NO_3^--N in the soil profile.

Water table management (WTM) has been proposed as a best management practice for bioremediation of NO_3^--N-contaminated soils by enhancing denitrification (Jacinthe et al., 2000; Kliewer and Gilliam, 1995). Nitrous oxide (N_2O) emissions resulting from denitrification activities have been a topic of increasing concern because N_2O has a well-documented role in stratospheric ozone depletion and contributes to the atmospheric greenhouse gas effect (Duxbury et al., 1982). Yet, microbial nitrogen (N) transformations in subsurface soils as influenced by water table are poorly understood as most research on denitrification has concentrated on the upper soil layer (0-0.15 m).

Although it is often assumed that denitrification in the top 0.15 m of the soil to be truly representative of the overall rate of denitrification process, some researchers have reported increased denitrification in subsoils where soluble C is not limited (Ryan et al., 1998; Jarvis and Hatch, 1994). Simmons et al. (1992) speculated that during leaching event, water soluble organic C can be translocated to lower soil depths with the percolating water and stimulate denitrification. If soluble C (dissolved organic carbon; DOC) percolates in the soil solution to the lower depth in the soil profile, denitrification with depth can be an important mechanism reducing the loading of NO_3^--N in the saturated zone (Lind and Eland, 1989).

The extent of denitrification below the surface soil layers is of great interest since it can alleviate contamination of ground and surface waters by NO_3^--N. Whether NO_3^--N removal by denitrification is beneficial to the environment depends on the partitioning of denitrification into N_2O and N_2. If significant amounts of N_2O are produced, then WTM practices may trade water pollution for air pollution. It has been hypothesized (Arah et al., 1991) that N_2O:N_2O+N_2 ratios decreases with depth, suggesting the environmentally harmless gas (N_2) is the dominant end product of denitrification. Despite growing recognition that denitrification in subsurface soil might ameliorate NO_3^--N pollution without concomitantly increasing N_2O, the effects of water table depth and N fertilization management practices on subsurface N_2O production have not been investigated under field conditions.

Furthermore, our present understanding of the N_2O:N_2O+N_2 ratios under WTM is based mainly on laboratory studies (e. g., Kliewer et al., 1995; Jacinthe et al., 2000), and it is questionable whether these results can be extrapolated to field conditions, where many factors which influence the production and reduction of N_2O cannot be easily controlled.

This study was designed to: (1) examine the effects of WTM and N fertilization rate on N_2O production in different soil layers (0-0.15, 0.15-0.30, and 0.30-0.45 m depth below the soil surface) from a corn field during the two cropping seasons, and (2) test, under field conditions, the hypothesis that the N_2O:N_2O+N_2 ratio decreases with soil depth.

5.2 MATERIALS AND METHODS

5.2.1 Field Management and Experimental Design

Field operations and experimental layout are described in detail in Chapters 3 and 4. Briefly, we conducted this study on a privately owned corn (*Zea mays* L.) field (4.2 ha) located at St-Emmanuel near Côteau-du-Lac, Quebec (74° 11' 15" lat., 45° 21' 0" long). The soil is classified as a Soulanges fine sandy loam (fine silty; mixed, non-acid, frigid *Humaquept*, Gleysol, according to the FAO classification system). The fine sandy loam soil (0-0.25 m) was underlain by layers of sandy clay loam (0.25-0.55 m) and clay (0.55-1.0 m), and the clay layer impeded the natural drainage. In the spring of 1998 (prior to the initiation of this field experiment) the soil contained 50 g C kg^{-1} soil (fresh wt) in the 0-0.25 m layer, 15 g C kg^{-1} soil (fresh wt) and a negligible amount of C below 0.55 m. The pH was near neutral (6.8). The farmer applied manure (cattle-slurry) to the field in the spring of 1998 at a rate of 20 Mg ha^{-1} (wet wt). Primary tillage after harvest consisted of moldboard plowing to a depth of 0.15 -0.20 m. Secondary tillage consisted of discking before planting.

There were two water table management treatments: free drainage (FD) and subirrigation (SI), factorially combined with two fertilizer rates: 120 kg N ha^{-1} (N_{120}) and 200 kg N ha^{-1} (N_{200}). The field was planted with corn (Pioneer hybrid 3905) at a density of 75,000 plants ha^{-1} with 0.75 m and 0.15 m inter and intra-row spacings, on 4 May in 1999 and 23

May in 2000. Potassium (muriate of potash, 0-0-60) was broadcast at a rate of 90 kg K_2O ha^{-1} roughly one week before planting. Diammonium phosphate (18-46-0) was broadcast at planting to provide approximately 24 kg N ha^{-1} and 130 kg P_2O_5 ha^{-1}. One month later, to reach the desired levels of the N fertilization, 97 kg N ha^{-1} and 178 kg N ha^{-1} were surface applied as ammonium nitrate (34-0-0) for the N_{120} and N_{200} treatments, respectively. This second application occurred on June 10, 1999; and June 20, 2000.

Treatments were laid out in a split plot design fashion with water table as main plot and N fertilization rate as subplot. The water table treatments were established in 30 m wide and 75 m long plots, and each main plot was split into two 15 x 75 m subplot. The water table treatments were replicated in 3 blocks, and fertilizer treatments were assigned randomly to subplots. Blocks were separated by a 30 m wide strip of undrained land. The SI treatment was imposed only after all field operations were completed and maintained until crop maturity in late September. Rainfall and air temperature data were obtained from an Environment Canada weather station situated 500 m from the experimental site.

5.2.2 Denitrification and Nitrous Oxide

To assess the relative proportion of N_2O and N_2 emissions in the surface and subsurface soil, three incremental depths (0-0.15, 0.15-0.30, 0.3-0.45 m) were sampled simultaneously in pairs, 6 times in 1999 and 8 times in 2000. Due to the heavy labor commitment required by the soil core sampling and time constraints related to incubating cores, it was only feasible to collect one pair of samples from each treatment plot for each sampling date. Soil cores were sampled in non-wheel tracked rows. Samples were never taken from the same location more than once within the growing season. Care was taken to avoid cross contamination between sampling depths.

On each sampling date, aluminum cylinders (50 mm diam x 150 mm long) were used to collect soil cores from randomly selected locations in the middle rows of each plot. The cylinders were perforated along the sides in a 50 mm grid to enhance acetylene (C_2H_2) gas diffusion. Samples were placed in 2 L plastic jars fitted with rubber stoppers for gas sampling. One hundred mL of the headspace in the jars were removed from one sample of each pair and replaced with 100 mL of C_2H_2 to give a 5 % (vol. vol.$^{-1}$) concentration. Samples were incubated outdoors overnight to mimic field conditions. The consequence of C_2H_2 treatment is the total inhibition of the nitrate reductase which reduces N_2O to N_2 and the suppression of nitrification process, so that accumulated N_2O (N_2O+N_2) could be measured (Yoshinari et al., 1977). The second sample was incubated without C_2H_2. Total N_2O production was quantified in a fashion similar to the procedure of MacKenzie et al. (1997) as detailed in chapter 4.

The soil core method used in this study assumes that N_2O measured in the soil cores would eventually be lost to the atmosphere. As a consequence, the N_2O measured in soil cores from deeper depths may be overestimated and, hence, $N_2O{:}N_2O+N_2$ may be inflated. An estimate of the mole fraction of N_2O ($N_2O{:}N_2O+N_2$) was computed using the following equation:

$$\frac{N_2O\text{-}N \text{ without } C_2H_2}{N_2O\text{-}N \text{ with } C_2H_2} = \frac{N_2O}{N_2O+N_2} \tag{1}$$

The average values of N_2O flux from denitrification in field conditions multiplied by this coefficient allows the calculation of potential N_2O emissions. In this context, N_2O production is the rate of N_2O emitted from soil core samples incubated without C_2H_2, whereas denitrification rate is the N_2O produced with C_2H_2.

5.2.3 Dissolved Organic Carbon

Three soil samples were collected from each plot using a hand-held auger before planting and after harvest. Samples were combined to make composite samples. To extract water soluble organic carbon (i.e., dissolved organic carbon), 10 g of field moist subsample were shaken in 100 mL of deionized distilled water for one hr, centrifuged at 10,000 rpm for 10 min, and then filtered through Whatman # 5 paper. Samples were analyzed for organic C using a Shimadzu TOC-5000A Total Organic C (TOC) analyzer (Shimadzu Scientific Instruments, Inc., Columbia, MD).

5.2.4 Soil Parameters

Soil water content was determined by oven drying soil cores at 105 ° C for 48 h. Soil bulk density, BD, (Mg m^{-3}) was determined by knowing the volume of the cylinder and dry weight of the soil. Total porosity was calculated assuming a particle density of 2.65 Mg m^{-3}. Percent water filled pore space (%WFPS) was calculated as:

$$\%WFPS = \frac{\%H_2O}{Porosity} x100 \quad (2)$$

Where, $\%H_2O$ is percentage of volumetric soil water content and calculated as following:

$$\%H_2O = \frac{\text{Mass } H_2O}{\text{Mass dry soil}} \text{ x BD x } 100 \quad (3)$$

5.2.5 Statistical Analysis

Analysis of variance (ANOVA) was performed for each sampling date. Main treatment effects and interactions were considered significant if $P < 0.05$. Differences among means were evaluated using Fisher's Protected LSD test. Statistical analysis were performed for each depth. Repeated measures analysis was conducted on denitrification and N_2O with depth being the repetition factor. Relationships between denitrification rates and selected soil parameters were examined using Spearman's rank correlation coefficients. All statistical analysis were conducted using Statistical Analysis System for windows, (SAS Institute, Cary, NC).

5.3 RESULTS AND DISCUSSION

5.3.1 Weather Conditions

Based on a 30-year average, the climate at the experimental site has a monthly mean temperature of 15 ° C and a mean precipitation of 522.2 mm during the growing season (May-October). Total seasonal rainfall in1999 was 13% higher than normal with almost half (47%) of the rainfall occurring in September and October. In 2000 it was about 12% higher than normal, with May being the wettest month (two-fold the norm) and October the driest (less than half the norm) of this growing season. Mean monthly temperatures at the site during the growing season (May-October) were 1.6 ° C higher in 1999 than the norm, while 2000 followed the norm.

5.3.2 Effects of Water Table on Denitrification and N_2O Production

Summary of the analysis of variance is presented in Table 5.1. Based on seasonal averages, there was no significant interaction between the two treatment factors, therefore, main effects were examined independently. Greater denitrification rates were measured in SI plots than FD plots, particularly at the surface soil (0-0.15 m) and intermediate (0.15-0.30 m) depths during both growing seasons (Tables 5.2, 5.3). Averaged across the three depths, seasonal denitrification losses under SI were 2.7-fold for 1999 and 2-fold for 2000 that of FD. In 1999 maximum denitrification rate was produced on July 7 under both SI and FD treatments (Table 5.2). For the 2000 cropping season, the two water treatments responded differently: under FD denitrification peaked on June 22 whereas under SI denitrification peaked from August 3 to August 17 (Table 5.3). All of these peaks occurred following the second N fertilizer application. These observations are consistent with Koops et al. (1996), who estimated that denitrification rates from the 0-0.40 m soil layer increased two-fold after N fertilizer application. Similarly, Velthof et al. (1996) reported that N_2O losses from deeper layers were most significant after application of N fertilizer.

Differences between SI and FD with regard to denitrification rates were primarily associated with a higher proportion of WFPS in SI relative to FD (Fig. 5.1). High WFPS may have restricted O_2 diffusion to denitrifying microorganisms, thereby enhancing denitrification. For example, WFPS in FD plots dropped below 40% from July to September, 1999 (Fig. 5.1 a), corresponding to the lowest denitrification and N_2O production measured during the two growing seasons (Tables 5.2, 5.3). Overall, water table significantly affected WFPS in all but the deepest depth in the 2000 growing season (Table 5.1). Soil moisture was also significantly affected by the depth of sampling, with the highest moisture measured at the intermediate depth (0.15-0.30 m) and lowest in the shallowest depth (0-0.15 m) (Fig. 5.1). The lower moisture content at the surface soil layer may be due to evaporation.

5.3.3 Effects of N Fertilizer Rate on Denitrification and N_2O Production

Although it has been widely reported (MacKenzie et al., 1997; Ellis et al., 1998) that the application rate of nitrogenous fertilizers has a significant influence on denitrification, we were unable to confirm this conclusion as differences in denitrification rates between N_{120} and N_{200} treatments were not significant throughout both growing seasons (Tables 5.1, 5.4, 5.5). One plausible explanation may be that there was sufficient NO_3^--N in all fertilizer treatments for denitrification to occur. If this is correct, managing N fertilizer rate alone may not be a promising strategy to mitigate N_2O air pollution.

5.3.4 Changes in Denitrification Rates and N_2O Production in the Soil Profile

Most of denitrification (90%) occurred in the 0-0.15 m and 0.15-0.30 m soil layers, with much less denitrification activity detected at 0.30-0.45 m soil layer. These findings were consistent in both water treatments. In only one sampling date (July 7 in 1999) did we find more denitrification at the 0.30-0.45 m depth than other soil depths in SI plots (Table 5.2). At this sampling date and depth, denitrification was about 6.5 times greater than the rest of all sampling dates combined. High N_2O sample variations are unfortunately common in these kinds of studies. It is possible that the core contained a small hot spot of activity associated with plant residue or the manure applied in the preceding year (spring, 1998). Parkin (1987) and Gold et al. (1998) demonstrated that hot-spots of denitrification activity are associated with patches of organic C in the soil profile creating anaerobic microsites, with high denitrification activity. Extensive review of N_2O studies by Mosier et al. (1996) concluded that it is not the measurement technique that provides most of the uncertainty in N_2O flx values but rather the diverse combinations of physical, chemical, and biological factors that control gas fluxes.

Enhanced denitrification in the 0.15-0.30 m soil layer indicates the existence of favorable conditions for denitrification at depth below 0.15 m. Consequently, we propose that sampling of the 0-0.15 m soil layer alone may not give an accurate estimation of denitrification losses. As noted previously, the field was moldboard plowed to a depth of 0.20m. It is plausible that the moldboard plowing turned under plant residues creating a layer of organic matter accumulation in the 0.15 -0.30 m depth, which supported higher microbial activity in that depth.

Seasonal N_2O emissions were estimated by summing averages for the three sampling depths. Nitrous oxide emissions in 1999 were 41 g d^{-1} ha^{-1} under SI and 25 g d^{-1} ha^{-1} under FD; 35 g d^{-1} ha^{-1} under N_{120} and 31 g d^{-1} ha^{-1} under N_{200}. In 2000, N_2O emissions were slightly higher than the 1999 growing season with 52 g d^{-1} ha^{-1} for SI, 63 g d^{-1} ha^{-1} for FD, 48 g d^{-1} ha^{-1} for N_{120}, and 56 g d^{-1} ha^{-1} for N_{200}. These values are within the lower portion of Williams et al. (1992), who evaluated available data of N_2O emissions from different land areas and found N_2O emissions ranging from 2.4 to 136 g d^{-1} ha^{-1} in agricultural fields. These losses may be small enough to have little economic or agronomic importance, but significant in terms of their effects on atmospheric pollution.

Although both denitrification and nitrous oxide production decreased with depth, measurable denitrification occurred at depths lower than 0.15 m. McCarty and Bremner (1992) who measured denitrification in surface (0-0.25 m) and subsurface (1.5 to 2 m) soils reported very low rates of subsoil N_2O in response to added NO_3^- -N, but addition of both NO_3^- -N and glucose stimulated denitrification activity. They concluded that the low rate of denitrification in subsoils was due to a lack of available organic carbon (C). In our study, however, vertical distribution of dissolved organic carbon (DOC) concentrations were not affected by the depth from which soils were sampled. In spite of the fact that the relationship between soil organic C and denitrification was established decades ago (Bremner and Shaw, 1958), the critical level of DOC needed for denitrification to proceed is not yet well-defined. Lack of similar studies makes any comparison difficult; however, high concentrations of DOC found at all the depths sampled (Fig. 5.2) may suggest that DOC in this soil was sufficiently high to support sustained denitrification. Mean DOC concentrations were generally higher in FD than SI, but the difference was not significant in all seasons. These consistent trends appear to suggest that DOC in SI plots might have been metabolized at a greater rate than in FD plots.

5.3.5 Nitrous Oxide-to-Denitrification Ratio

The N_2O:N_2O+N_2 ratio varied between the two growing seasons and provided an interesting contrast. The difference between FD and SI with regard to N_2O:N_2O+N_2ratio was not significant at any depth in 1999 (Fig. 5.3 a), but differed significantly in 2000 at all but the uppermost depth (Fig. 5.3 b). Although N rate had no significant effect on N_2O:N_2O+N_2 ratio (Table 5.1 and Fig. 5.3 c d), an interesting trend was observed with respect to the depth of sampling. The trend was the N_2O:N_2O+N_2 ratio increasing with depth under the N_{200} treatment (Fig. 5.3 c), with the 0.30-0.45 m depth producing nearly twice that of the upper layers.

Based on seasonal values averaged across all depths, the N_2O fraction in 1999 accounted for 28% under SI and 43% under FD, whereas in 2000 it was 39% in SI and 58% in FD. As shown in Tables 5.2 and 5.3, the decrease in N_2O production under SI did not result from lower denitrification, but instead was caused by more complete reduction of N_2O to N_2 which resulted in a decrease in the N_2O:N_2O+N_2 ratio. This is clear evidence that higher denitrification rates under SI than FD do not necessarily add to concerns over global atmospheric N_2O loadings, due to the increased likelihood of N_2O undergoing further microbial reduction to N_2. However, values of N_2O:N_2O+N_2 ratios obtained in our study are greater than the findings of Kliewer and Gilliam (1995) who reported the N_2O fraction accounting for only 2% of denitrification. In their study, water tables were set at a much shallower depth than ours (up to 0.15m), which apparently promoted a complete reduction of N_2O to N_2. For a practical purpose, such shallow water tables can only be recommended during non growing season, in order not to interfere with tillage and other field operations.

5.3.6 Trends of N_2O:N_2O+N_2 Ratio in the Soil Profile

The largest fraction of N_2O was contributed by the 0.30-0.45 m depth (Fig. 5.3). It is interesting to note that the N_2O:N_2O+N_2 ratio was close to 1.0 under the FD and N_{200} treatments at 0.3-0.45 m depth, suggesting that N_2O was the dominant N gaseous product in that soil layer. Results show that the shift from N_2 to N_2O as the predominant product of denitrification in the 0.30-0.45 m soil layer (Fig. 5.3) did not result from an increase in N_2O

production with depth (Table 5.2, 5.3). It was due to a sharp decrease in denitrification at the 0.30-0.45 m depth without a corresponding decrease in N_2O at the equivalent depth, leading to a greater N_2O:N_2O+N_2 ratio. This notion is corroborated by the strong positive correlation between denitrification and N_2O at the 0.30-0.45 m depth in both growing seasons ($r = 0.9$, $P < 0.0018$, $n = 12$ for 1999 and $r = 0.8$, $P < 0.0025$, $n = 12$ for 2000). This may provide further evidence that the increase in the N_2O:N_2O+N_2 ratio with depth was not due to the interference of C_2H_2 with nitrification. It may be related to incomplete C_2H_2 inhibition of N_2O-reductase and, hence N_2 was produced, leaving a portion of unaccounted for N_2O. Jacinthe et al. (2000) reported concentrations of N_2O remaining high at the depths lower than 40 m of soil columns and indicated that the activity of N_2O-reductase was inherently low and that N_2O was not being reduced at a significant rate. The effectiveness of C_2H_2 inhibition varies with soil type and achieving a uniform distribution of C_2H_2 to effectively inhibit N_2O takes longer in soils with clay content than other soils (Ryden et al., 1987). Supporting this argument is the fact that clay content in our soil increased with depth and the 0.30-0.45 m layer was a sandy clay loam.

It is important to point out, however, that with the soil core method, it is assumed that N_2O measured would instantly be lost to the atmosphere. As a consequence, N_2O measured in cores from deeper depths may be overestimated and, hence, N_2O:N_2O+N_2 ratio may be inflated. This explanation suggests that the diffusion of the N_2O generated at a certain depth in the soil profile to the surface soil is a critical factor which determines the final emission at the soil surface. Nitrous oxide produced near the soil surface would probably have readily diffused out of the soil into the atmosphere, whereas N_2O produced at deeper depths may have taken longer to diffuse from the soil providing more opportunity for reduction to N_2O before reaching the atmosphere (Arah et al., 1991). In our study, because N_2O was produced at 0.30-0.45 m depth, the time required to diffuse from this depth to the soil surface may be significant and, in the process, N_2O could further be reduced to N_2.

5.4 SUMMARY AND CONCLUSIONS

Although denitrification activity decreased with depth, the soil layer at 0.15-0.30 m, contributed substantially to the total N loss through denitrification. The assumption that denitrification in the top 0.15 m of soil is representative of the overall rate of denitrification from the soil may not always be true. Denitrification and N_2O production were higher in SI than FD but were unaffected by N fertilizer treatments. We also found that the SI treatment acted as a sink for N_2O. Ratios of N_2O:N_2O+N_2 were consistently lower in SI plots than FD, but not affected by N rate. During the 1999 growing season, the N_2O comprised 28% under SI and 43% under FD. In 2000, the proportion of denitrification products emitted as N_2O were 39% under SI and 58% under FD. Water table management may be used to reduce NO_3^--N pollution of surface and groundwater, without increasing N_2O.

Table 5.1: Summary of analysis of variance for 1999 and 2000 seasons.

Variable	Denitrification		N_2O		N_2O:N_2O+N_2		% WFPS	
	1999	2000	1999	2000	1999	2000	1999	2000
------0-0.15 m------								
Model	NS	*	NS	NS	NS	NS	*	*
Block	NS	NS	NS	NS	NS	NS	NS	*
WTM[†]	NS	*	NS	NS	NS	*	*	*
N[‡]	NS	NS	NS	NS	NS	NS	NS	NS
WTM*N	NS	NS	NS	NS	NS	NS	NS	NS
------0.15-0.30 m------								
Model	*	NS	NS	NS	NS	NS	*	*
Block	NS	NS	NS	NS	NS	NS	NS	NS
WTD	*	*	NS	NS	NS	**	*	***
N	NS	NS	NS	NS	NS	NS	NS	*
WTM*N	NS	NS	NS	NS	NS	NS	NS	NS
------0.30-0.45 m------								
Model	*	NS	NS	NS	NS	NS	*	NS
Block	NS	NS	NS	NS	NS	NS	NS	NS
WTM	*	*	NS	NS	NS	*	*	NS
N	NS	NS	NS	NS	NS	NS	NS	NS
WTM*N	NS	NS	NS	NS	NS	NS	NS	NS

[†] Water table management; [‡]N fertilization rate treatment

NS Not significant at $P \leq 0.05$

* , **, *** Significant at $P \leq 0.05$, 0.01and 0.001, respectively

Table 5.2: Denitrification (N_2O+N_2) and Nitrous oxide (N_2O), g ha^{-1} d^{-1}, in surface and subsurface soils as affected by free drainage (FD) and subirrigation (SI) water treatments in the 1999 cropping season

WTM†	Depth (m)	Sampling dates 06/9	07/7	07/28	08/26	09/18	10/20	Mean Season	Total Season
Denitrification									
FD‡	0.15	57	275a*	5	1	2	4	57	344a
	0.30	62	52b	6	0.5	0.7	3	21	124b
	0.45	8	15b	2	0.5	0.6	3	5	29c
SI §	0.15	59	308	52	63	24	9	86	515
	0.30	51	144	100	76	14	12	66	397
	0.45	13	353	21	14	3	5	68	409
N_2O									
FD	0.15	6	42	4	0.2	0.2	0.6	9	53
	0.30	43	22	3	0.1	0.7	1	12	70
	0.45	12	11	2	0	0	1	4	26
SI	0.15	16	62	17	8	0	2	18	105
	0.30	20	30	12	11	1	1	13	75
	0.45	4	48	4	2	2	1	10	61

† Water table management; ‡ Free drainage; § Subirrigation

* Different letters within column the same treatment or factor indicate significant depth differences ($P \leq 0.05$)

Table 5.3: Denitrification (N_2O+N_2) and Nitrous oxide (N_2O), g ha^{-1} d^{-1}, in surface and subsurface soils as affected by free drainage (FD) and subirrigation (SI) water treatments in the 2000 cropping season

WT†	Depth (m)	5/28	6/22	7/6	7/19	8/3	8/17	9/19	10/20	Mean	Total
						Sampling dates					
-----Denitrification-----											
FD‡	0.15	21	111a*	83	24	36a	61	29	5	46	395a
	0.30	14	35b	80	27	18b	157	27	11	46	401a
	0.45	1	9b	32	20	10b	20	12	8	14	125b
SI §	0.15	18	77	114	66a	321a	240	26	17	110	971a
	0.30	31	85	88	91a	210ab	77	33	19	79	682a
	0.45	1	9	19	20b	90 b	45	17	9	26	235b
-----N_2O-----											
FD	0.15	2	33	14b	23	8	5	13	1	12	109
	0.30	4	9	65a	16	13	13	15	5	18	154
	0.45	0.3	3	0.5	17	14	27	10	6	13	100
SI	0.15	10	4ab	20	11	82	46	5	2	23	193
	0.30	3	0.292	55	12	52	39	6	3	22	189
	0.45	0.5	2b	13	13	67	30	14	4	18	144

† Water table; ‡ Free drainage; § Subirrigation

* Different letters within column of the same treatment or factor indicate significant depth differences ($P \le 0.05$).

Table 5.4: Total denitrification (N_2O+N_2) and nitrous oxide (N_2O), g ha^{-1} d^{-1}, in surface and subsurface soil as affected by nitrogen fertilization rate, 120 kg N ha^{-1} (N_{120}) vs 200 kg N ha^{-1} (N_{200}) in the 1999 cropping season.

N rate†	Depth	Sampling dates							
	(m)	06/9	07/7	07/28	08/26	09/18	10/20	Mean	Total
-----Denitrification-----									
N_{120}‡	0.15	67	318a*	36	33	21	8	81	483a
	0.30	58	64b	37	46	16	10	39	231b
	0.45	8	35b	7	4	1	5	10	60b
N_{200} §	0.15	50	265	25	32	5	0.208	64	382
	0.30	54	132	70	31	8	0.208	49.2	300
	0.45	13	333	16	11	3	3b	63	379
-----N_2O-----									
N_{120}	0.15	9	62a	13	5	0.2	0.4	15	90
	0.30	48	24b	10	5	1	1	15	89
	0.45	8	15b	2	1	0	1	5	27
N_{200}	0.15	13	41	7	3	0	2	11	66
	0.30	15	28	5	6	1	1	9	56
	0.45	8	45	3	2	5	1	11	64

† N rate treatments; ‡120 kg N ha^{-1}; § 200 kg N ha^{-1}

* Different letters within column of the same treatment or factor indicate significant depth differences ($P \leq 0.05$).

Table 5.5: Total denitrification (N_2O+N_2) and Nitrous oxide (N_2O), g ha^{-1} d^{-1}, in surface subsurface soil affected by nitrogen fertilization rate, 120 kg N ha^{-1} (N_{120}) vs 200 kg N ha^{-1} (N_{200}) in the 2000 cropping season

N†	Depth (m)	Sampling dates 5/28	6/22	7/6	7/19	8/3	8/17	9/19	10/20	Mean	Total
Denitrification											
N_{120}‡	0.15	11b*	110a	123	41a	170	103	281	66	113	905a
	0.30	37a	40b	113	52a	81	157	293	91	108	864a
	0.45	1b	8b	24	15b	41	24	54	64	29	231b
N_{200}§	0.15	28a	79	74a	52	187	198	172	107	112	981
	0.30	8b	80	55ab	63	147	77	200	149	97	868
	0.45	1b	8	17b	20	60	40	180	72	50	447
N_2O											
N_{120}	0.15	10	3	15	22	58	43	3	1	19	164
	0.30	2	5	68	16	27	25	3	4	19	167
	0.45	1	1	16	10	28	18	5	3	10	91
N_{200}	0.15	2ab	34	19	12	32	8	14	2	15	136
	0.30	0.17	11	52	13	38	25	19	4	21	183
	0.45	0.3b	3	20	19	53	39	19	6	20	318

† N rate treatments; ‡120 kg N ha^{-1}; § 200 kg N ha^{-1}

* Different letters within column of the same treatment or factor indicate significant depth differences ($P \leq 0.05$).

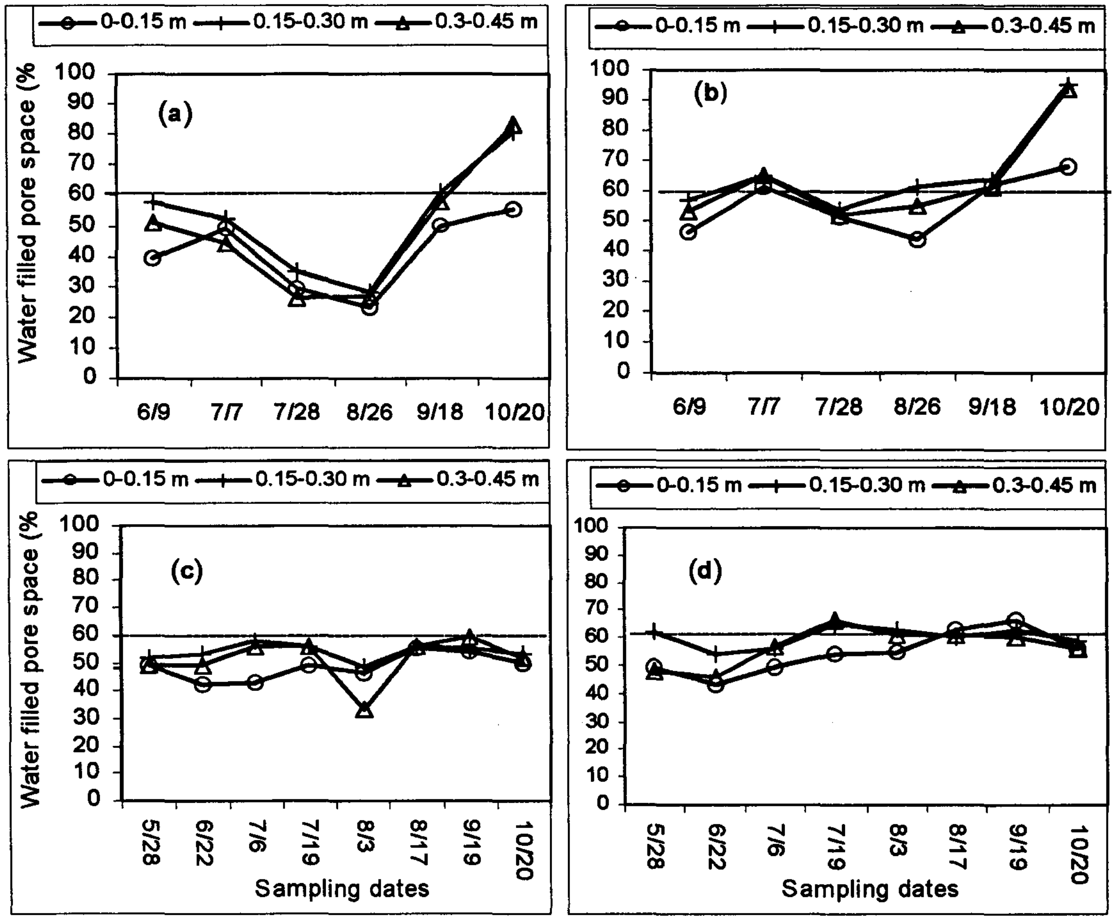

Figure 5.1: Mean soil water content expressed in percentage of water filled pore space (%WFPS) under (a) and (c) free drainage (FD) in 1999 and 2000, respectively, and (b) and (d) subirrigation (SI) 1999 and 2000, respectively, at three depths.

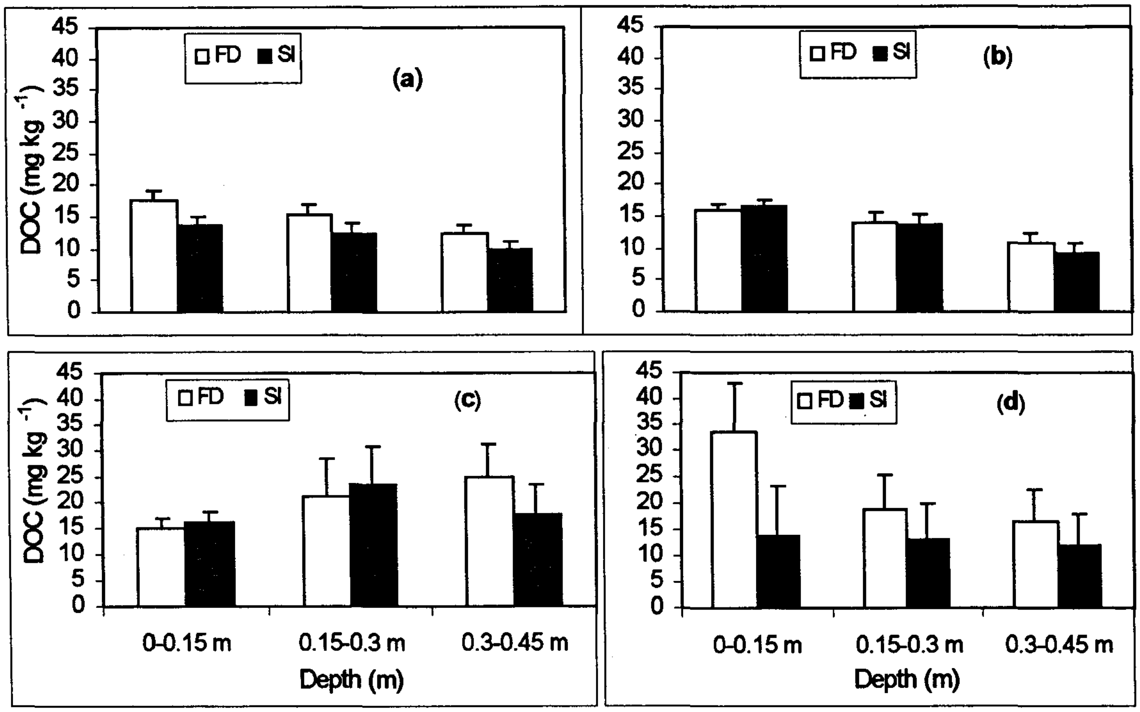

Figure 5.2: Dissolved organic carbon (DOC) concentration (mg kg^{-1} soil) in the soil profile in (a) spring 1999, (b) fall 1999, (c) spring 2000, and (d) fall 2000.

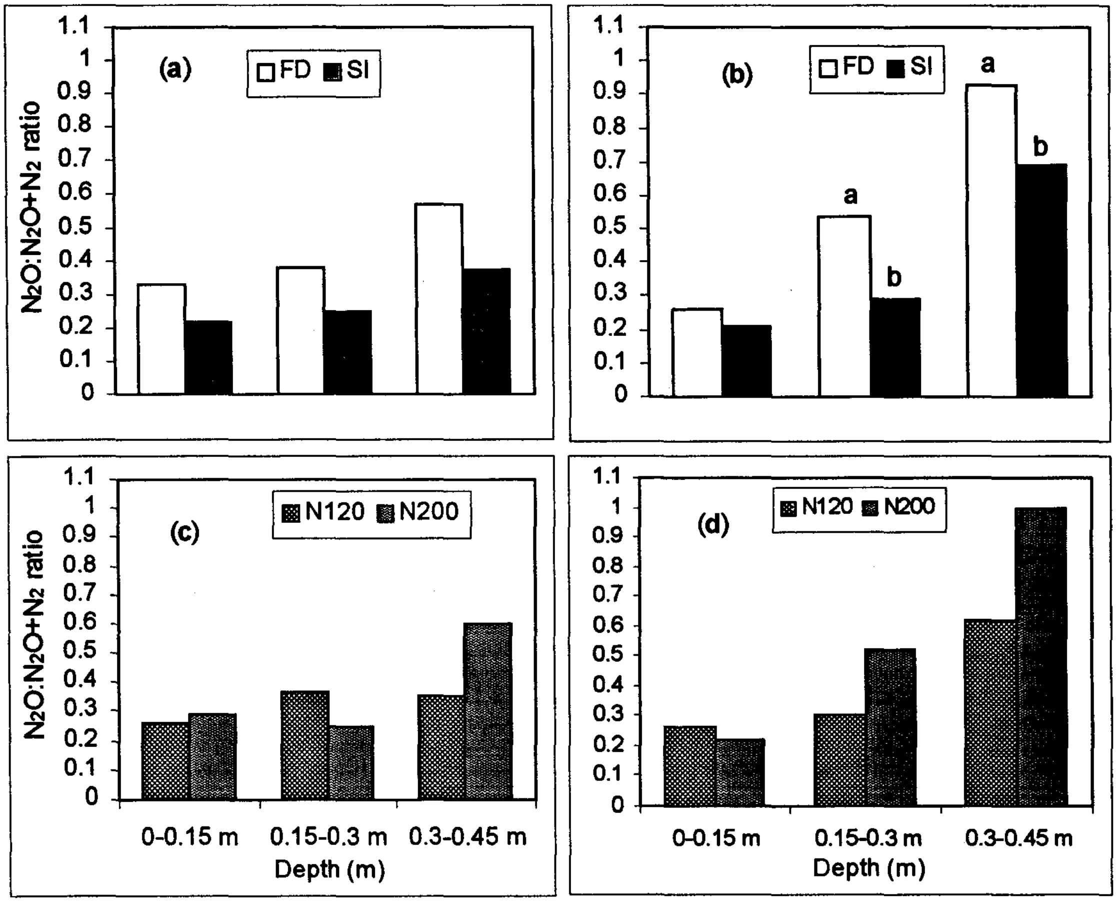

Figure 5.3: Seasonal average of the proportion of N_2O and denitrification ratios with different depths in (a) 1999, (b) 2000 under free drainage (FD) and subirrigation (SI) treatments and in (c) 1999 and (d) 2000 under 120 N kg ha^{-1} and 200kg N ha^{-1} treatments . Bars of different depths but within same treatment having different letters are significantly (P < 0.05) from each other, n = 36.

PREFACE TO CHAPTER 6

Concerns for environmental quality have stimulated much interest in developing various management strategies that mitigate nutrient losses to the environment. Pervious Chapters (Chapters 3-5) have emphasized water table and N fertilizer management. A number of other management strategies have been proposed to mitigate the impact of agricultural production on the environment. The focus of Chapter 6 is on soil and crop residue management. The need to reduce soil erosion and improve soil quality has prompted a growing interest in conservation tillage systems, including no-till (NT) and reduced tillage (RT) as alternatives to conventional tillage (CT) systems. The purpose of this paper is to evaluate the effects of three tillage systems on NO_3^--N accumulation in the soil profile, denitrification rates, N_2O:N_2O+N_2 ratios, following 10 years of continuous corn grown under three different tillage systems with crop residue.

CHAPTER 6

Denitrification and N_2O:N_2O+N_2 Ratios in the Soil Profile As Influenced By Tillage Systems Under Continuous Corn Production

ABSTRACT

Soil and water quality deterioration due to conventional farming systems continue to raise environmental and agronomic concerns. There is a growing interest in the adoption of conservation tillage systems, including no-till (NT) and reduced tillage (RT), as alternatives to conventional tillage (CT) systems. A two year study to investigate possible environmental consequences of three tillage systems was conducted on a 2.4 ha field located at Macdonald Research Farm, McGill University, Montreal. The soil was a sandy loam (0.5 m depth) underlain by a clay layer. Treatments consisted of a factorial combination of CT, RT, and NT with the presence or absence of crop residue. Soil NO_3^--N concentrations tended to be lower in RT than NT and CT tillage treatments. Denitrification and N_2O emissions were similar among tillage systems. Large rates of N_2O production were measured in the subsurface (0.15-0.45 M) soil, suggesting that a significant portion of emitted N_2O may be missed if only soil surface gas flux measurements are made. Nitrous oxide mole fraction (N_2O:N_2O+N_2) was higher in the drier season of 1999 under CT, with the ratio exceeding 1.0 in some soil layers. Dissolved organic carbon concentrations remained high in all soil depths sampled, but was not affected by tillage system.

6.1 INTRODUCTION

Ecological and environmental issues have come to the forefront of societal concerns. The impact of these concerns in relation to agriculture is growing. Traditional farming systems for intensive production of agricultural lands can seriously degrade the quality or health of soil and water resources. For example, use of conventional tillage (CT) systems can accelerate the depletion of soil organic matter (SOM) and lead to the deterioration of soil structure, resulting in a severe soil erosion (Hussain et al., 1999; Martel and MacKenzie, 1980). Soil erosion is a major form of environmental degradation which, if preventive or remedial actions are not taken, may ultimately threaten the long-term sustainability of food production capacity. Monreal et al. (1998) noted that decrease in SOM content and subsequent erosion resulting from excessive tillage reduces water holding capacity of the soil, increases surface water runoff and, consequently, decreases soil fitness and productivity. Furthermore, soil particles associated with runoff water may carry plant nutrients such as P and N, that contaminate water and contribute to the eutrophication of aquatic ecosystems. Water quality measurements at 300 locations in major U.S rivers showed that suspended sediments and nutrients associated with runoff from agroecosystems are the most damaging nonpoint source pollutants (Smith et al., 1987). As a result, the use of water resources for drinking, irrigation and recreation may be impaired. This relatively new environmental concern has added to the urgency of developing more efficient soil and water management techniques.

There is a growing interest in the adoption of conservation tillage systems, including no-till (NT) and reduced tillage (RT), primarily because they have been shown to be significantly more water efficient (Lindwall and Anderson, 1981), to improve soil and water quality (Hussain et al., 1999; Logan et al., 1987), and to reduce production costs due to lower fuel and labor inputs (Uri et al., 1999; Lindwall and Anderson, 1981). In recent years, the adoption of NT and different forms of RT systems have grown steadily in Canada and throughout the world. Crop management, nowadays also involves leaving crop residue on the soil surface rather than merely eliminating or reducing tillage operations.

The presence of crop residue on the soil surface protects soil against raindrop impacts, reduces runoff and erosion, improves surface water quality, and enhances water infiltration to benefit crops during dry or low rainfall periods (Ogden et al., 1999). However, the continuing increase in acreage under NT and RT raises concerns about the impact of these practices on the quality of surface and underground waters. Formation of macropores coupled with reduced surface runoff in NT/RT fields can increase downward movement of water containing nitrate-N (NO_3^--N) and other agrochemicals to subsurface tile drains or ground water. While the likelihood of NO_3^--N leaching is increased under NT and RT systems, published studies of NO_3^--N losses from NT and different forms of RT have given somewhat divergent results (Kanwar et al. 1997; Randall and Iragavarapu, 1995).

In addition, soils under NT and/or /RT retain greater moisture than those under CT. This may enhance denitrification which can be a significant source of N_2O (MacKenzie et al., 1998; Mummey et al., 1998; Burton et al., 1997; Rice and Smith, 1982), a potent greenhouse gas thought to be involved in the depletion of the ozone layer. Denitrification is a major biological process in soil, which produces N_2 and N_2O in proportions that vary widely (Bergsma et al., 2002). Subsoil denitrification has been suggested as an important mechanism to remove excess NO_3^--N leached from agricultural soils before leaching to groundwater or discharged to surface aquifers via subsurface drainage (Sotomayor and Rice, 1996). These authors also recognized the scarcity of the information about denitrification in subsoil environments. Furthermore, the proportion of denitrification evolved as N_2O (N_2O mole fraction; N_2O/N_2O+N_2) could have widespread significance for the global N_2O budget. We are unaware of any published study that has investigated how N_2O mole fraction in surface and subsurface soil environments is impacted by different tillage systems under natural field conditions.

The main objectives of this study were to assess the effects of long term tillage practices on (1) NO_3^--N distribution in the soil profile, (2) denitrification and N_2O emissions in subsurface (0-0.45 m) soils, and (3) to estimate the ratio of N_2O:N_2O+N_2 produced during two growing seasons.

6.2 MATERIALS AND METHODS

This study, undertaken in 1999 and 2000, was conducted on a 2.4-ha site at McGill University's research farm on Macdonald Campus, Ste-Anne-de-Bellevue, Quebec. The soil was mostly of the St. Damase series (Typic Endoaquent; Humic Gleysol). The upper soil layer (about 0.28 m) was a sandy loam, underlain by a sand layer (mean thickness about 0.18 m), with clay beginning at a mean depth of 0.46 m (Burgess, 2000).

The study consisted of three tillage systems: no-till (NT), reduced tillage (RT), and conventional tillage (CT) factorially combined with two residue treatments: with and without.

Conventional tillage plots were moldboard-plowed to 0.20 m in the fall and spring, RT plots were offset-disked to 0.15 m in the fall and spring, and NT plots were not tilled at any time. Field layout and treatment arrangement are depicted in Fig. 6.1. Treatments were laid out in a randomized complete block design. The study site consists of 18 plots, half (9 plots) with residue and planted to corn harvested for grain corn, and the other half without residue and planted to corn harvested as silage. Plots were 18 m x 80 m, and drained by a subsurface drainage system to a depth of 1.0 m below the soil surface. In this study, only plots with residue were included. The grain-corn plots were harvested with a combine that removed only grain, leaving all residues on the plots.

Corn (Funk 4120 hybrid) was planted in rows spaced 0.76 m apart. All plots received: at seeding, diammonium phosphate (18-46-0), banded 50 mm below and 50 mm laterally from the seeds to provide 40 kg N ha^{-1} and 102 kg P_2O_5 ha^{-1}. The field was seeded on 6 May in 1999 and on 8 May in 2000. Ammonium nitrate (34-0-0) and muriate of potash (0-0-60) were top-dressed 2-3 weeks later to provide an additional 140 kg N ha^{-1} and 148 kg K_2O ha^{-1}. The second application occurred on 4 June in 1999 and on 9 June in 2000.

6.2.1 Measurements of N_2O and mineral N in soil

Sampling strategy and analytical procedures were the same as detailed in chapters 3 through 5. Briefly, to assess the relative proportion of N_2O and N_2 emissions in the surface and subsurface soil, three incremental depths (0-0.15, 0.15-0.30, 0.3-0.45 m) were sampled simultaneously in pairs. Due to the heavy labor commitment required by the coring procedure and time constraints related to incubating cores, it was only feasible to collect one pair of samples from each treatment plot for each sampling date. Soil cores were sampled in non-wheel-tracked rows. Samples were never taken from the same location more than once within a growing season. Care was taken to avoid cross contamination between sampling depths with careful cleaning of each drilling depth site.

Denitrification and N_2O production rates were measured using the core incubation method, in the presence and absence of acetylene, respectively. Minimally disturbed soil core samples (50 mm diameter x 150 mm long) were incubated in a 2 L mason jars. For denitrification measurements, 100 ml headspace air was replaced by acetylene (5% v/v) to inhibit N_2O reduction. The second sample was incubated without C_2H_2 to estimate N_2O emissions. Dinitrogen (N_2) fluxes were estimated by subtracting the N_2O fluxes measured with and without C_2H_2 present.

Soil samples (three samples per plot) for NO_3^--N analysis were taken prior to planting in the spring (April) and shortly after harvest in fall (October) from 0-0.25 m, 0.25-0.50 m, and 0.50-0.75 m depth increments using hand-held auger sampling probe. Replicate samples were then thoroughly mixed and moist subsamples of 10 g were shaken with 100 mL of 1 M KCl for 60 min. The soil suspensions were filtered through Whatman # 5 filter papers. Nitrate-N was quantified using a Lachat flow injection autoanalyzer (Lachat Quickchem, Milwaukee, WI) according to Keeney and Nelson (1982). The detection limit was 0.05 mg L^{-1}.

6.2.2 Dissolved Organic Carbon

Three soil samples were collected from each plot using a hand-held soil auger before planting and after harvest to 0-0.15, 0.15-0.30, and 0.30-0.45 m depth increments. Samples from within each treatment plot were combined to make a composite samples. To extract water soluble organic carbon (i.e., dissolved organic carbon), a 10 g of field-moist subsample was shaken in 100 mL of deionized distilled water for 1 hr, centrifuged at 10,000 rpm for 10 min, and then filtered through Whatman # 5 paper. Samples were analyzed for organic C using a Shimadzu TOC-5000A Total Organic C (TOC) analyzer (Shimadzu Scientific Instruments, Inc., Columbia, MD).

6.2.3 Soil Parameters

Soil moisture content was determined by oven drying soil cores at 105 ° C for 48 h. Soil bulk density, BD (Mg m^{-3}) was determined by the core method and total porosity was calculated assuming a particle density of 2.65 Mg m^{-3}. Percent water filled pore space (%WFPS) was calculated as:

$$\mathrm{WFPS} = \frac{\%\mathrm{H_2O}}{\mathrm{Porosity}} \times 100 \qquad (1)$$

Where, $\%H_2O$ is the volumetric soil water content in percentage calculated as follows:

$$\%\mathrm{H_2O} = \frac{\mathrm{Mass\ H_2O}}{\mathrm{Mass\ dry\ soil}} \times \mathrm{BD} \times 100 \qquad (2)$$

6.2.4 Statistical Analysis

Analysis of variance (AVOVA) was performed for each sampling date and depth. Differences among means were evaluated using Sheffe's multiple comparison test. Data was analysed as a randomized complete block design. All statistical analysis were conducted using the General Linear Model (GLM) procedure of the Statistical Analysis System, (SAS Institute, Cary, NC).

6.3 RESULTS AND DISCUSSION

6.3.1 Climatic Data

Total seasonal (May-October) rainfall in 1999 was near the 30-year norm (Table 6.1). May and August were the driest months during the 1999 growing season, each receiving only slightly more than half the normal rainfall. Rainfall in the 2000 growing season was 40 mm (8%) greater than the norm, with May being the wettest month followed by August, and October was the driest. It is interesting to note that the two driest months in 1999 (May and August) corresponded to the wettest months in 2000 (Table 6.1). Average mean monthly temperature was 2 ° C and 0.7 ° C higher than the norm in 1999 and 2000, respectively (Table 6.1).

6.3.2 Nitrate-N concentrations in the soil profile

Residual NO_3^--N levels in the soil profile were not affected by tillage (Fig. 6.2). The only exception was spring 1999 at the 0.25-0.50 m soil layer when NO_3^--N levels were significantly greater under RT than CT and NT (Fig. 6.2 a). Although the effects of tillage on soil NO_3^--N levels were not obvious, the general trend, except in the fall of 1999 (Fig. 6.2 b), was for NO_3^--N concentration in the soil profile to be lower in NT than RT or CT systems (6. 2 a c d). In a continuous corn production, Randall and Iragavara (1995) measured higher subsurface drain flow under NT than CT, but total NO_3^--N losses were higher under CT. This may suggest that greater infiltration and preferential flow under NT may result in greater drain discharge, while enhanced denitrification under NT system may decrease the amount of leachable NO_3^--N in the soil solution.

No clear trend of either increase or decrease in NO_3^--N with soil depth was obvious in 1999 (Fig. 6.2 a b). The opposite was true in 2000 where NO_3^--N decreased with depth in all tillage systems (Fig. 6.2 c d). Perhaps the most striking finding with respect to NO_3^--N concentrations in the soil profile is the sharp decline in NO_3^--N in fall 2000 (Fig. 6.2 d), compared to fall 1999 (Fig. 6.2 b). This is an important indication that NO_3^--N was removed from the soil solution before it could be leached down to the groundwater or reach to the surface waters via subsurface drainage systems. Enhanced denitrification losses (discussed below) was likely to be the primary reason for this reduction of NO_3^--N in the fall because of wet conditions during the preceding growing season.

6.3.3 Denitrification Rates and N_2O Production

Denitrification rates and N_2O production from NT and RT systems were in general similar or slightly higher than those under CT management in both seasons and at all depths (Figs. 6.3 - 6.6). This finding differs from that of Fan et al. (1997) who reported greater denitrification rates under NT soils than under CT in a corn field in southwestern Quebec. This led them to conclude that corn production should be carried out under CT, if reduced N_2O emission is required. Similarly, Staley et al. (1990) found that under NT, denitrification was increased when compared with CT. They postulated that the difference was in part due to the presence of a greater amount of oxidizable C in the surface soils under NT. In contrast, Robertson et al. (2000) noted that N_2O fluxes under NT were not different than those under conventional soil management. Despite this contradiction, the general consensus is that because of higher moisture and organic matter content, and higher microbial populations, NT produces higher rates of denitrification, depending on prevailing climatic conditions at the time of measurements. For example, Mummey et al. (1998) suggested that NT management in periods or regions that are relatively warm and wet may result in N_2O emission rates similar or less than those under CT and NT may thus be a viable means to reduce N_2O emissions while enhancing soil quality. These authors documented that in drier periods or regions, N_2O emissions were greater under NT because of increased soil moisture content.

At the beginning of the 1999 growing season, denitrification rates were episodic at the soil surface (0-0.15 m depth), but such a high variability disappeared after August (Fig. 6.3 a). At the deeper soil layers, N_2O production remained low (< 10 g d^{-1} ha^{-1}) at all sampling dates and showed no significant difference between tillage systems (Fig. 6.3 b c). Nitrous oxide production varied widely between treatments most notably at the September 14 sampling date at the 0-0.15 m depth (Fig. 6.4 a) and on June 10 sampling date at the 0.15-0.30 m depth (Fig. 6.4 b). On average, N_2O production from these two sampling dates was greater than denitrification, suggesting nitrification might have contributed to N_2O. This is possible especially that N_2O production was measured from soil cores incubated without acetylene block. It is also interesting to note that both of these observations of N_2O production greater than denitrification were recorded under the CT system. The greater N_2O production measured in CT plots may be attributed to conditions of greater oxidation created by the tillage.

The 2000 growing season differed from 1999 in two main respects: 1) denitrification rates were 5- to 10-fold greater than in 1999 regardless of the tillage treatment, and 2) on a seasonal average, the greatest denitrification rate was measured under CT at the 0.15-0.30 m depth. An extremely high denitrification rate was measured on June 14 (Fig. 6.5 b). Analysis of the data without this value resulted in no significant difference between tillage treatments at any depth. Nitrous oxide production rates were smaller than denitrification rates for all measurements and depths, suggesting that N_2 formed a large proportion of the denitrification gaseous end-products, irrespective of tillage system (Figs. 6.5-6.6).

Mean daily N_2O production over the sampling periods and depths was 9 g d^{-1} N_2O -N ha^{-1} for CT, 5 g d^{-1} N_2O -N ha^{-1} for RT, and 6 g d^{-1} N_2O -N ha^{-1} for NT in 1999. In the 2000 season, the values were much greater; 20 g d^{-1} N_2O -N ha^{-1} for CT, 15 g d^{-1} N_2O -N ha^{-1} for

RT, and 27 g d^{-1} N_2O -N ha^{-1} for NT. These rates compare well to those measured by Cates and Keeney (1987), who reported mean daily N_2O flux values ranging from 8 to 12 g d^{-1} N_2O -N ha^{-1}. Burton and Beauchamp (1994) estimated denitrification rates of 200 g d^{-1} N_2O -N ha^{-1} under both CT and NT subsequent to broadcasting 150 kg N ha^{-1} fertilizer. A striking finding with this study is the general trend that the greatest denitrification and N_2O production rates were measured at the 0.15-0.30 m depth under CT, whereas denitrification rates were highest near the soil surface in NT and RT systems. It is likely that the moldboard plowing led to plant residues accumulating in the 0.15 -0.30 m depth, resulting in greater organic C which supported higher microbial activity at that depth. This is consistent with Becker et al. (1990) who stated that maximum denitrification in the field can be expected near the bottom of the plow layer.

Nitrous oxide is produced near the soil surface as well as in underlying horizons (Cates and Keeney, 1987; Li et al., 2002). We hypothesized that long-term NT and/or RT practices would increase DOC concentrations in subsurface soils and hence denitrification capacity. This did not happen and the differences in DOC between tillage systems were not significant at any depth (Fig. 6.7), except in Spring 2000 at the 0-0.15 m and 0.30-0.45 m depths soil layers, when RT showed significantly greater DOC than either NT or CT (Fig. 6.7 c). This finding is consistent with Parkin and Meisinger (1989), who reported that surface soil straw management had no effect upon subsurface soil denitrification. An interesting aspect of denitrification in subsurface soil is that this denitrification is unlikely to add to concerns over global atmospheric N_2O concentrations due to the further reduction of N_2O during diffusion up the soil profile. Sahrawat and Keeney (1986) noted that denitrification proceeds rapidly to N_2 in wet soils emitting little N_2O. McCarty and Bremner (1993) found that no DOC leached from the surface soil into subsoil during the decomposition of freshly added plant tissue. They concluded that the DOC was rapidly metabolized by the microbial community in the surface soil. In our study, however, concentrations of DOC remained high at all depths (Fig.6.7) suggesting that there was probably some C movement through the coarse-textured soil to the lower depth. Starr and Gillham (1993) monitored organic C movement in sandy soil in Ontario. They confirmed that organic C movement actually took place resulting in increased denitrification rates in aquifers where the groundwater table was less than 2-3 m. In an Australian clay soil under pasture or an annual crop, Weier et al. (1993) showed that denitrification up to 1.0 m below the soil surface was possible when available C was not limiting. However, this may only occur after periods of heavy rainfall, when soluble C could percolate with the soil solution to lower depths in the soil profile.

6.3.3 Role of Dissolved Organic C in Subsurface Denitrification Rate

While the critical role of dissolved organic C as a source of energy and nutrients to the denitrifying community is widely recognized, the threshold below which denitrification is limited is not clearly defined. In a lysimeter study, Brye et al. (2001) found DOC concentrations ranging from 45 to 82 mg L^{-1} under a chisel-plowed field. Gambrell et al. (1975) reported that in poorly drained subsoils, a soluble C concentration of 12-15 mg L^{-1} was sufficient to act as an effective potential energy source for denitrification. Similarly, Obenhuber and Lowrance (1991) suggested that significant denitrification occurs with 10 mg L^{-1} DOC, below which concentration little or no denitrification can be expected. The DOC concentrations in our study ranged from 13 to 47 mg kg^{-1}. All these findings from different agroecosystems and climatic cycles suggest that soil denitrification studies which rely only on surface soil conditions will fail to account for the production or emission of the N_2O throughout the soil profile.

6.3.4 Ratio of N_2O:N_2O+N_2 in the Soil Profile

With regard to the ratios of N_2O:N_2O+N_2, the two seasons differed and provided interesting contrasts. Although there was considerable scatter in the relationship between water filled pore space (WFPS) and denitrification and N_2O emission, N_2O was the dominant end-product under CT, especially at the upper soil layers in 1999 (Fig. 6.8 a) where WFPS were mostly below 30% (Fig. 6.9). In the wetter season of 2000, WFPS were mostly above 50% (Fig. 6.10) and N_2O formed only 36% of denitrification end-products under CT, suggesting more complete reduction of N_2O to N_2. In the drier season of 1999, the overall N_2O:N_2O+N_2 ratios were 1.0 for CT, 0.60 for NT, and 0.36 for RT. In the wetter season of

2000, the N_2O:N_2O+N_2 ratios were 0.38 for CT, 0.41 for RT, and 0.49 for NT. The N_2O:N_2O+N_2 ratio under NT and RT remained relatively constant in both growing season (Fig. 6.8). The shift from N_2O to N_2 under CT as the predominant product of denitrification in the wetter soil indicates that the sink for N_2O provided by denitrifying organisms was greater than under drier conditions. Other researchers have also noted that continuously wet soils where denitrification proceeds rapidly to N_2 emit little N_2O (Sahrawat and Keeney, 1986). Webster and Hopkins (1996) reported the N_2O:N_2O+N_2 ratio to be 50% for the drier soil and between 18 to 21% for the wetter soil. All these findings lead us to conclude that knowledge of the denitrification gaseous end-products, namely N_2O:N_2O+N_2 ratio, is necessary in order to accurately assess environmental consequences of the denitrification process.

6.3.5 CONCLUSIONS

Tillage had little or no effect on denitrification and N_2O production rates in the soil profile. Nitrate-N levels in the soil profile were lower in the wetter season (2000) than the drier season (1999). We concluded that enhanced denitrification was the primary reason for the reduction of NO_3^--N in the soil solution under higher soil moisture content. The net emission of N_2O from soil and, hence, its environmental impacts depends on the rate of N_2O formation and the consumption of N_2O ($N_2O \rightarrow N_2$) during denitrification. The N_2O:N_2O+N_2 ratios varied seasonally and were affected by tillage systems in the drier year in the upper soil layers (0-0.30 m). In the drier season of 1999, the N_2O:N_2O+N_2 ratios were greater under CT than under NT and RT. In the wetter season of 2000, the N_2O:N_2O+N_2 ratios were similar among the three tillage systems. These findings indicate that NT and RT systems did not contribute greater N_2O emissions to the atmosphere than CT as suggested by some in the literature. Across soil depths, dissolved organic C levels in the soil profile were not consistently affected by tillage system and, thus, was not considered to be a primary reason for the lower denitrification at the deeper depth.

Table 6.1: Mean monthly precipitation and air temperature during the 1999 and 2000 growing seasons measured at Macdonald Campus Research Weather Station, compared to the long term mean(1961-1991)[a].

Month	Air temperature (°C)			Precipitation (mm)		
	1999	2000	1961-1991	1999	2000	1961-1991
May	15.3	13.0	12.9	40.8	133.3	68
June	20.8	16.7	18	111	86.0	83
July	21.6	19.2	20.8	100	81.2	86
August	19.1	19	19.4	55	125.5	100
September	17.1	13.7	14.5	100.1	84.0	87
October	7.6	8.6	8.3	90.6	29	75
Mean	17.0	15.0	15.7	---	---	---
Total	---	---	---	497	539	499

[a] Normals were not available for the Macdonald Weather Station. Values for the normal were obtained from Dorval International Airport, about 10 km east of the field site.

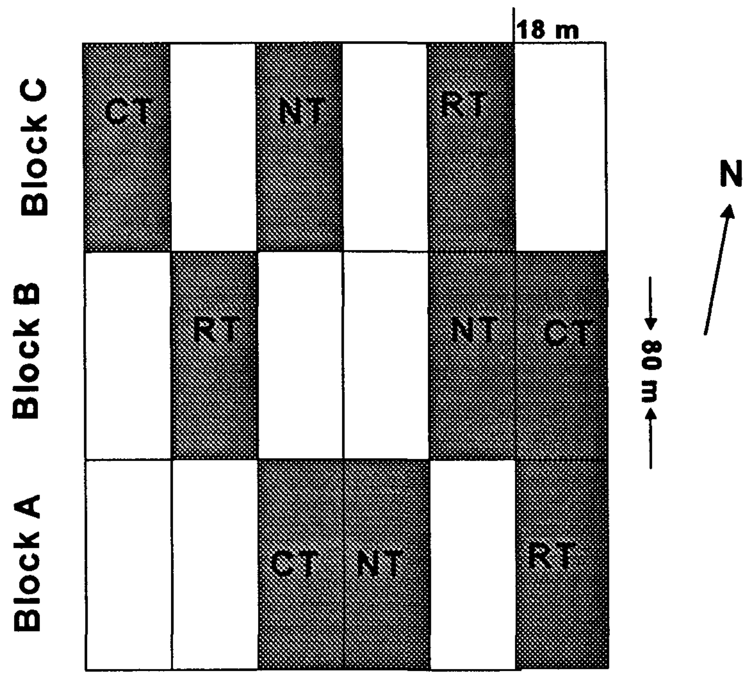

Figure 6.1: Schematic representation of the field lay out.

Only shaded areas, which were tillage treatments with crop residues left on the soil surface, have been included in this study. Corn in these plots was harvested as a grain corn. Blank areas represented plots without crop residue with corn harvested as a sillage.

CT: Conventional tillage; NT: No-till; RT: Reduced tillage

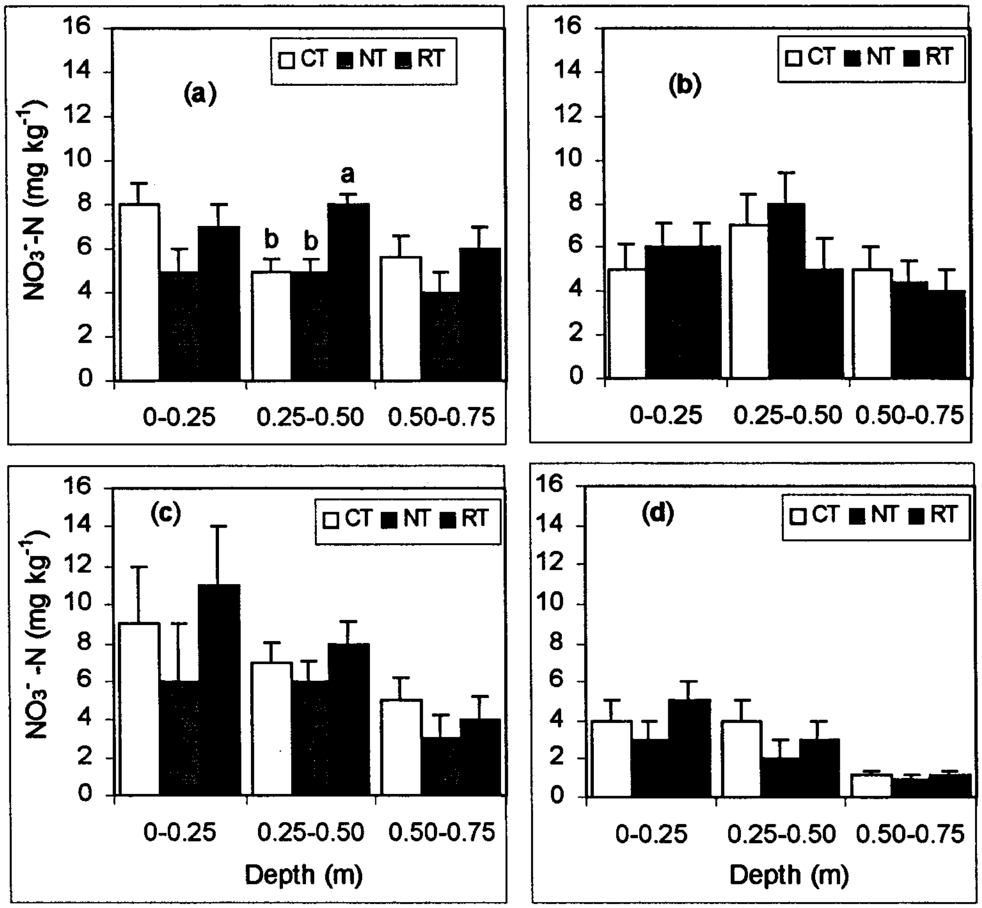

Figure 6.2: NO_3^--N concentrations (mg kg^{-1} soil) in the soil profile under conventional tillage (CT), reduced tillage (RT) and no-tillage (NT) systems in (a) spring 1999, (b) fall 1999, (c) spring 2000, and (d) fall 2000. Bars within same depth followed by different letters are significantly ($P < 0.05$) different. Vertical bars are standard error of the mean ($n = 9$).

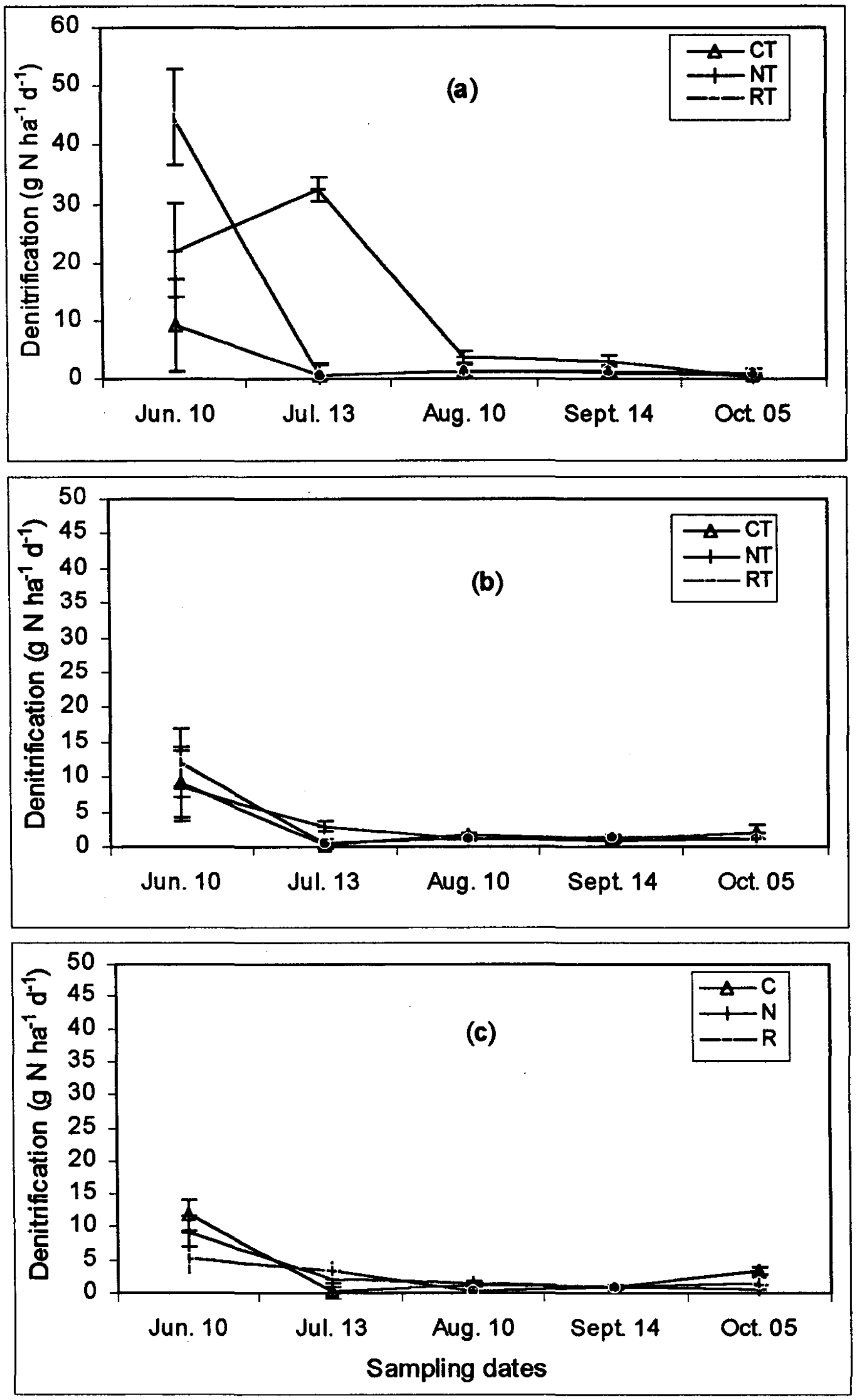

Figure 6.3: Denitrification rates under conventional tillage (CT), reduced tillage (RT), and no-tillage (NT) systems in 1999 cropping season with (a) 0-0.15 m, (b) 0.15-0.30, and (c) 0.30-0.45 m depths. Vertical bars indicate standard error of the mean (n = 3).

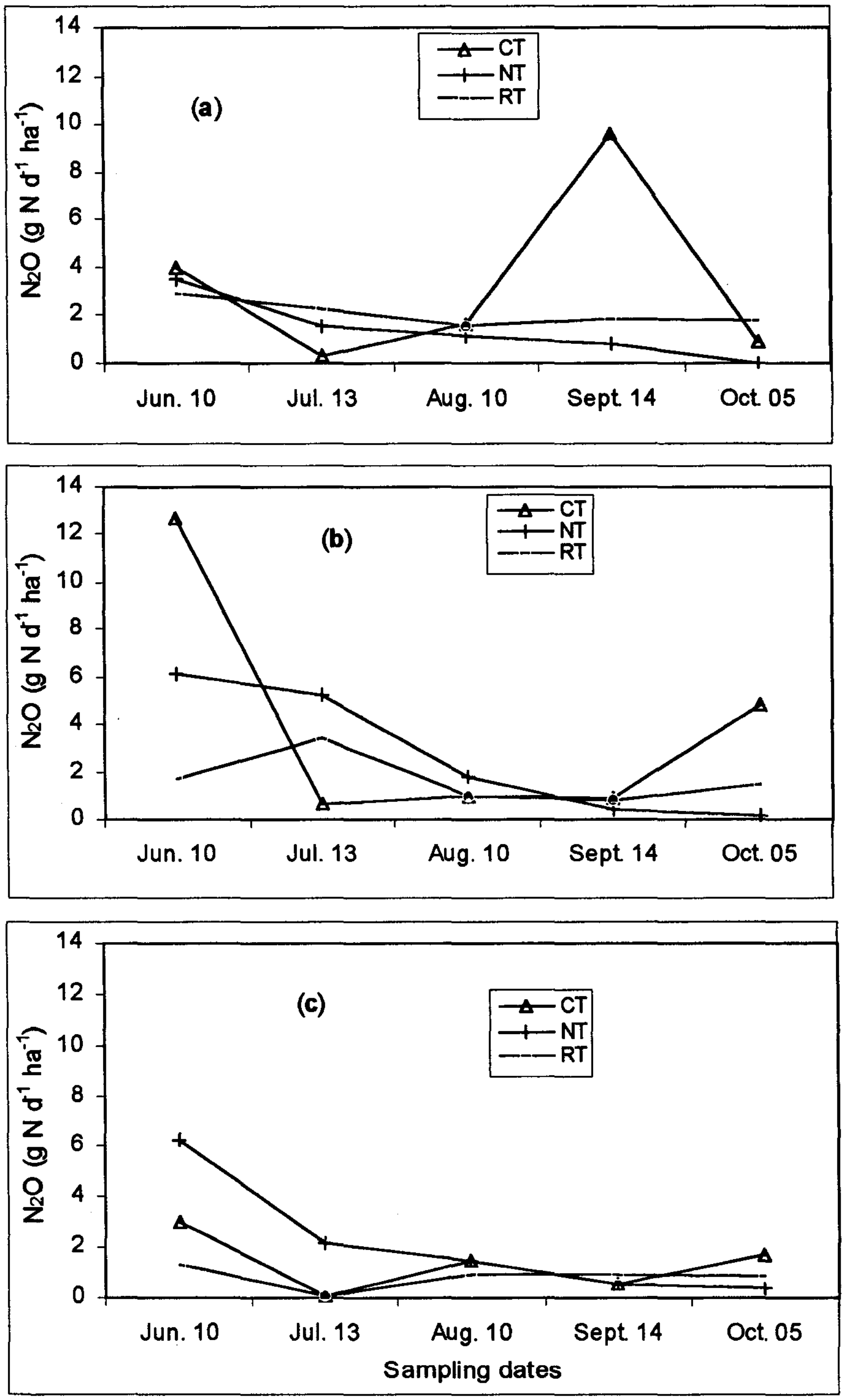

Figure 6.4: Nitrous oxide (N_2O) production under conventional tillage (CT), reduced tillage (RT), and no-tillage (NT) systems in 1999 cropping season with (a) 0-0.15 m, (b) 0.15-0.30, and (c) 0.30-0.45 m depths. Vertical bars indicate standard error of the mean (n = 3).

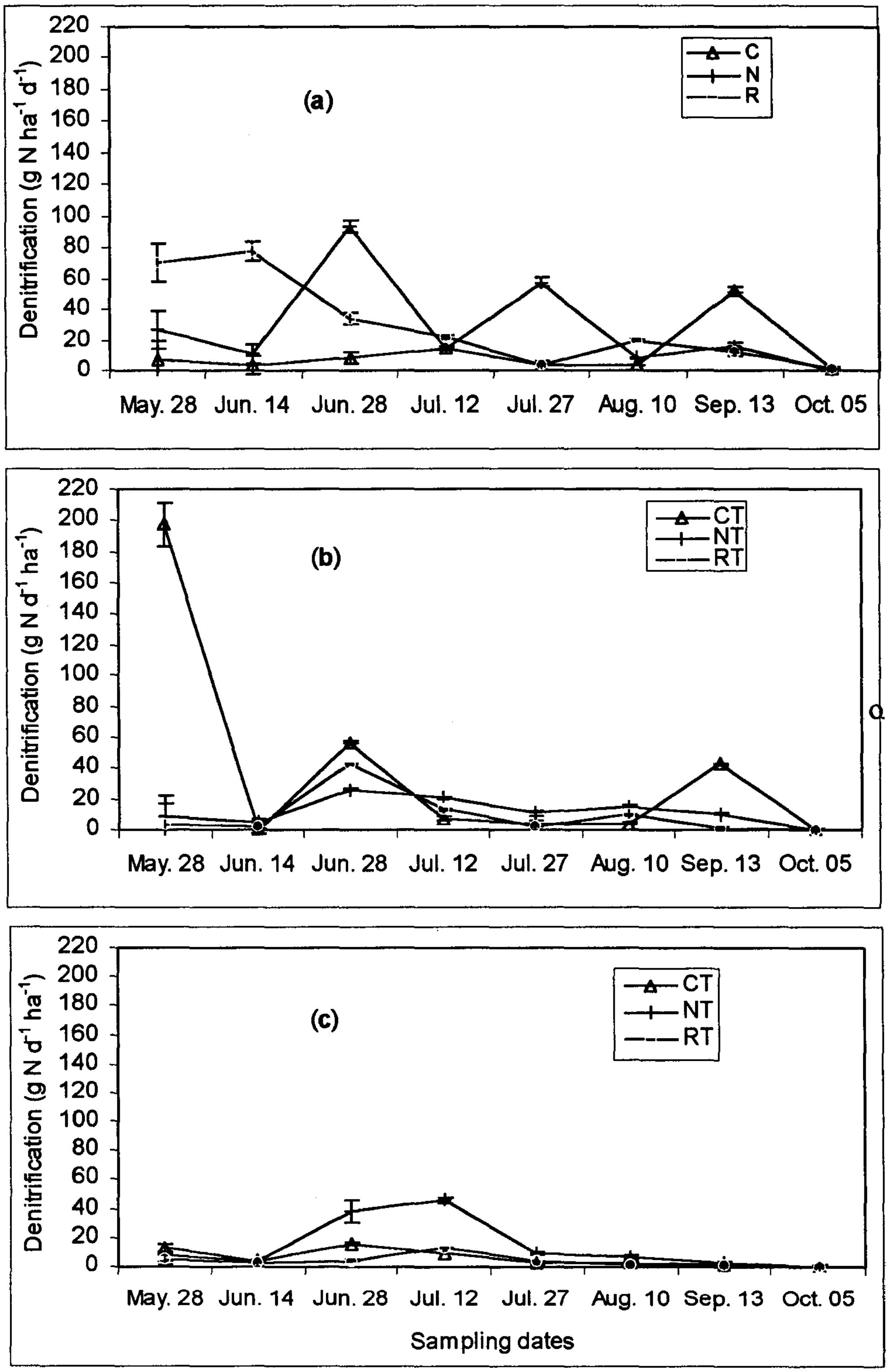

Figure 6.5: Denitrification rates under conventional tillage (CT), reduced tillage (RT), and no-tillage (NT) systems in 2000 cropping season with (a) 0-0.15 m, (b) 0.15-0.30, and (c) 0.30-0.45 m depths. Vertical bars indicate standard error of the mean (n = 3)

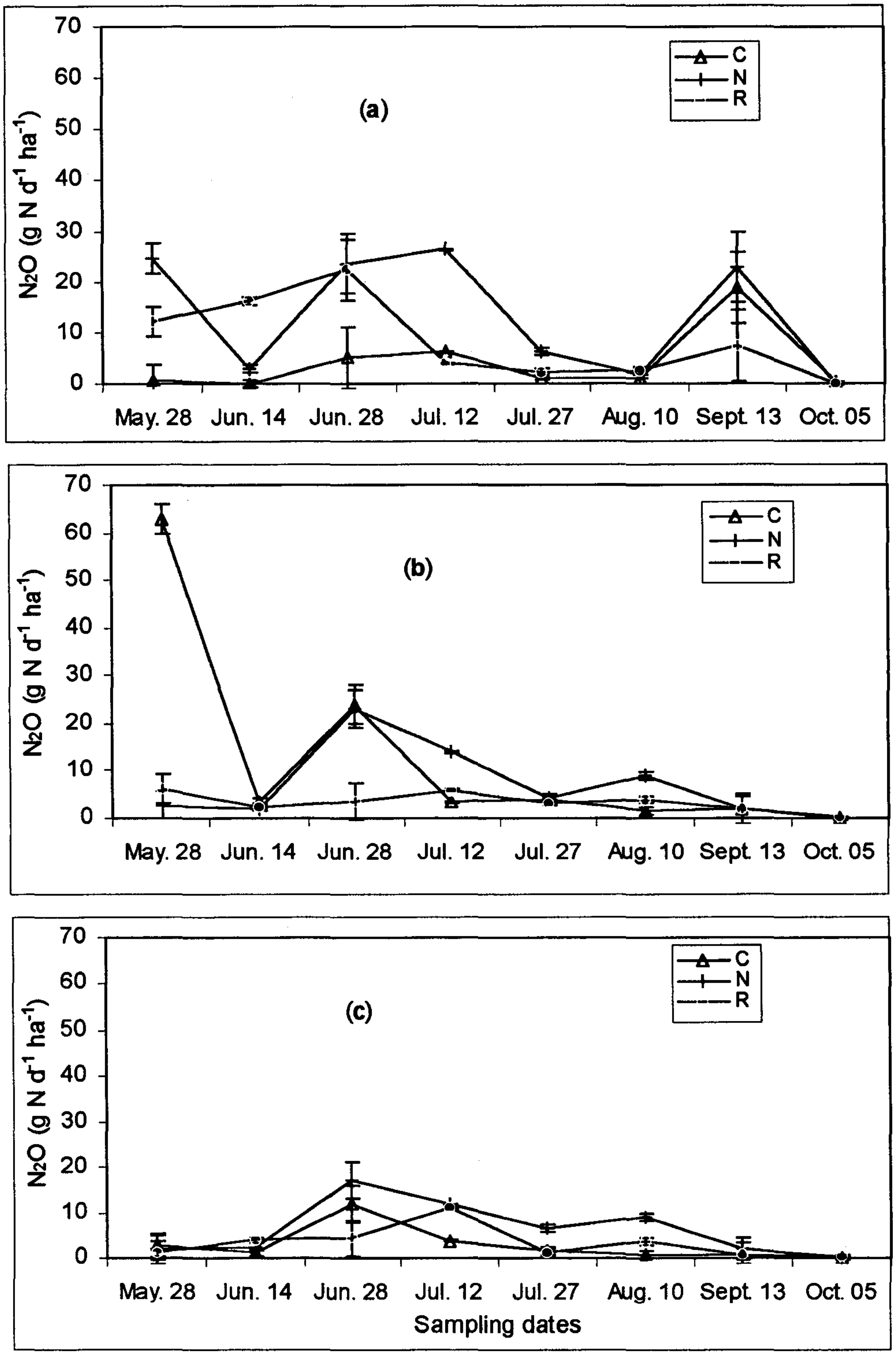

Figure 6.6: Nitrous oxide production under conventional tillage (CT), reduced tillage (RT), and no-tillage (NT) systems in 2000 growing season with (a) 0-0.15 m, (b) 0.15-0.30, and (c) 0.30-0.45 m depths. Vertical bars indicate standard error of the mean (n = 3).

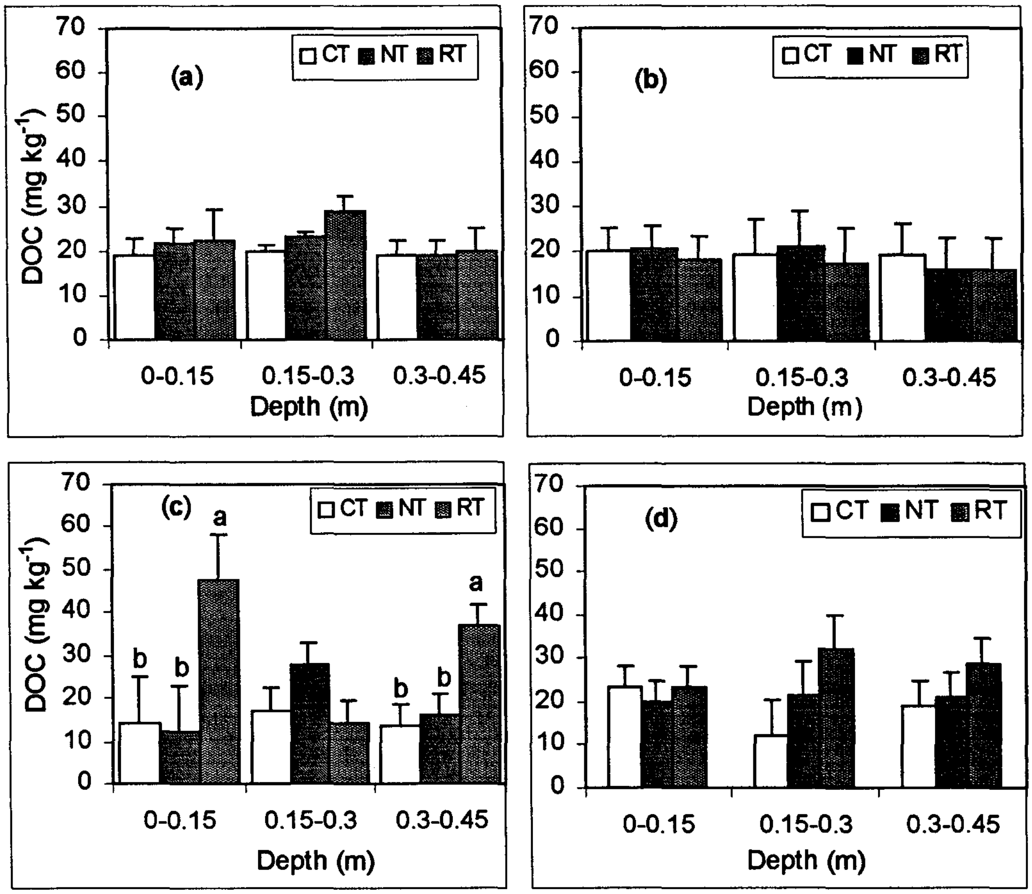

Figure 6.7: Dissolved organic carbon (DOC) concentration (mg kg^{-1} soil) in the soil profile in (a) spring 1999, (b) fall 1999, (c) spring 2000, and (d) fall 2000 as influenced conventional (CT), reduced tillage (RT), and no-tillage (NT) systems. Charts of the same depth followed by different letter are significantly (P<0.05) different. Vertical bars are standard error of the mean (n = 9).

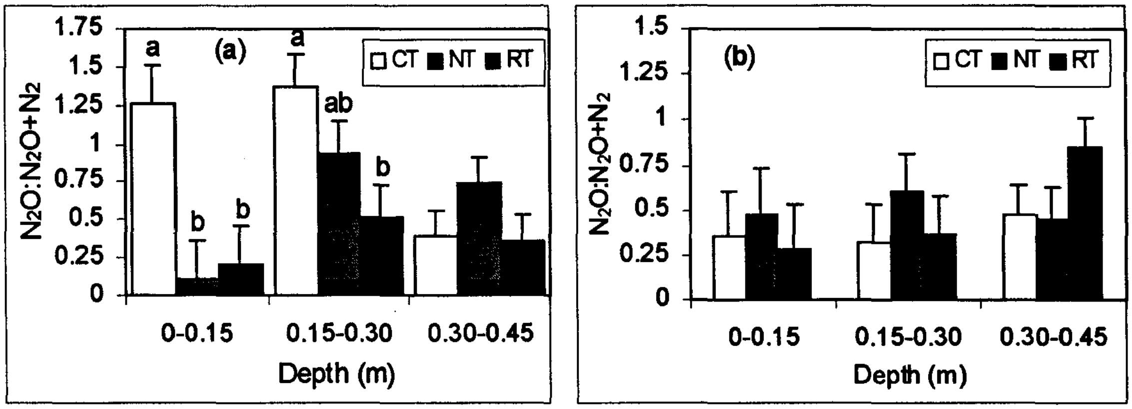

Figure 6.8: Seasonal average of N_2O:N_2O+N_2 ratio with three incremental depths in (a) 1999 and (b) 2000.

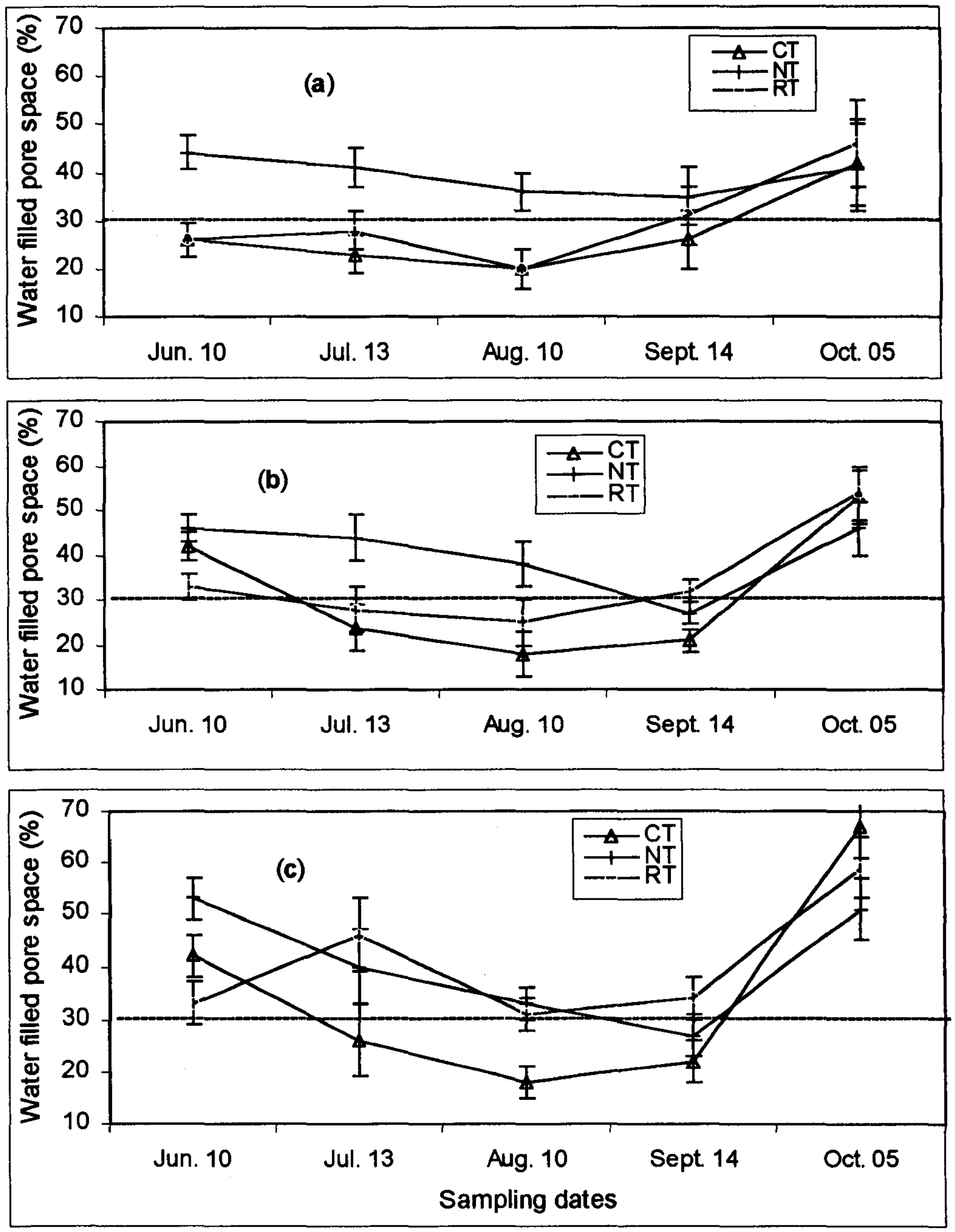

Figure 6.9: Water filled pore space percentage (%WFPS) under conventional tillage (CT), reduced tillage (RT), and no-tillage (NT) systems in 1999 cropping season with (a) 0-0.15 m, (b) 0.15-0.30, and (c) 0.30-0.45 m depths. Vertical bars indicate standard error of the mean (n = 3).

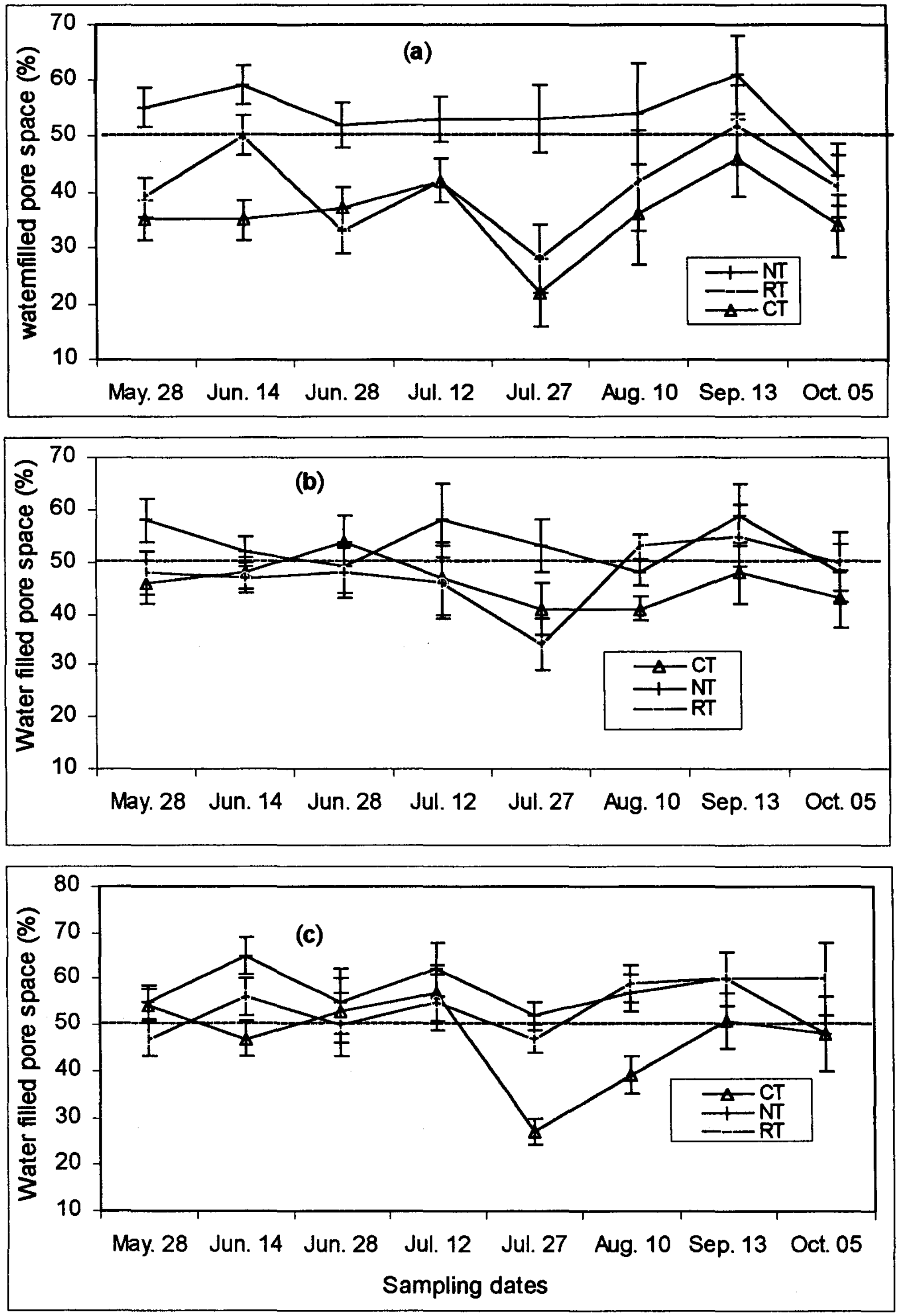

Figure 6.10: Water filled pore space percentage (%WFPS) under conventional tillage (CT), reduced tillage (RT), and no-tillage (NT) systems in 2000 cropping season with (a) 0-0.15 m, (b) 0.15-0.30, and (c) 0.30-0.45 m depths. Vertical bars indicate standard error of the mean (n = 3).

CHAPTER 7

GENERAL CONCLUSIONS

To maintain food production at the same level as population growth without damaging the environment offers a special challenge. Pressure is growing to identify best management practices that ensure sustainable food production while minimizing negative environmental impacts. A number of management strategies have been proposed to mitigate the impact of agricultural production on the environment.

From these field studies, the following conclusions could be drawn:

1) Water table control, via subirrigation, had little effect on corn yield. Yield reduction under SI in 1998 was attributed to the unusually abundant rainfall in June 1998 coupled with the shallow water tables in the SI plots, which led to ponding of water on the field on some occasions resulting in poor crop growth and yield.

2) Corn yield was not responsive to N fertilization rate in any of the three seasons studied. Hence, there was no agronomic benefits associated with the higher rate of N fertilization.

3) Subirrigation decreased soil NO_3^--N and subsurface tile drainage NO_3^--N concentrations by enhancing denitrification. Subirrigation increased denitrification rates by up to three fold, compared to FD.

4) Although denitrification rates and N_2O were greater under SI than FD, water table management did not affect the percentage of N_2O in total denitrification (N_2O:N_2O+N_2) evolving to the atmosphere at the soil surface (0-0.15 m depth). This is clear evidence that higher denitrification rates under SI than FD may not necessarily add to concerns over global atmospheric N_2O loadings, due to the increased likelihood of N_2O undergoing further microbial reduction to N_2.

5) We found that denitrification rates and N_2O emission were not affected by N fertilizer

CHAPTER 8

CONTRIBUTIONS TO THE ADVANCEMENT OF KNOWLEDGE

This study provides new insights on denitrification, N_2O, and N_2O:N_2O+N_2 ratios in the soil profile under southwestern Quebec soil conditions. The following are the contributions of this study to the advancement of knowledge in the field of nitrogen dynamics in agricultural soils.

1) To the best of my knowledge, no other published study on denitrification has examined N_2O mole fraction (N_2O:N_2O+N_2) in the soil profile (0-0.45 m) as influenced by water table depth under field conditions. Hence, my work advances the understanding of the impacts of water table management regimes on N_2O mole fraction dynamics within agroecosystems. This may lead to the development of strategies that can minimize N_2O flux to the atmosphere.

2) While effects of tillage systems on denitrification have been investigated sufficiently, no field results had been published on the variations in the N_2O:N_2O+N_2 ratio with depth (0-0.45 m) under different tillage systems.

3) This study showed evidence of previously unknown or poorly documented aspects of N dynamics in the soil-water system. These include: **a)** adopting water table management technology reduces NO_3^--N in the soil profile and, hence NO_3^--N movement to water bodies without a concomitant increase in N_2O, and **b)** significant denitrification occurred at soil layers deeper than 0.15 m below the soil surface, suggesting that the assumption of surface (0-0.15 m) denitrification measurements to be highly representative needs to be re-evaluated.

4) Overall, findings of these field trials will contribute toward building a scientific basis for understanding the relative environmental significance of N transformations below the soil surface (0.0.15 m).

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