# QUANTIFYING THE EFFECT OF SPRINKLER IRRIGATION ON GREENHOUSE GAS EMISSIONS

A Thesis Submitted to the College of

Graduate Studies and Research

In Partial Fulfillment of the Requirements

For the Degree of Master of Science

In the Department of

Chemical and Biological Engineering

University of Saskatchewan

Saskatoon

By

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# ABSTRACT

A declining area of arable land has heightened pressure to increase food production for a growing world population. The potential to enhance food production by increasing the number of irrigated farms is high on the Canadian Prairies. However, expansion of irrigated farms will likely influence agricultural greenhouse gas (GHG) emissions. Quantification and comparison of energy partitioning of surface energy fluxes, crop microclimatic modification, soil environment variation, and GHG emissions from irrigated and non-irrigated fields in the Canadian Prairies are explored in this research. The observed field data were also used to check the suitability of a regional version of a process-based GHG simulation model, the Denitrification-Decomposition (CDN-DNDC) model. It was found that irrigation alters energy partitioning noticeably, which promoted crop microclimatic modification leading to reduced vapor pressure deficit and canopy temperature. However, despite a much smaller proportion of the net radiation in non-irrigated systems being consumed by evaporation, the dryland fields did not exhibit markedly warmer soil temperatures. Soil water was found as the critical factor in influencing soil GHG emissions, and availability of soil nutrient was the dominant factor in soil $N_2O$ emissions from irrigated systems. The performance of the CDN-DNDC model to predict soil moisture under irrigation conditions during growing season was good, which allowed the model to be used to simulate different irrigated conditions. The CDN-DNDC model simulated and measured $N_2O$ emissions from irrigated and non-irrigated fields were compared, indicating that this model is suitable to assess $N_2O$ emissions from different management systems under irrigated conditions in the Canadian Prairies. According to the CDN-DNDC model, a future increase in irrigated fields will increase $N_2O$ emission. However, when crop yield is taken into consideration, there is actually a lower mean annual nitrous oxide

intensity in the irrigated field. The performance of the CDN-DNDC model was less accurate in predicting $N_2O$ emission and soil water after the spring thaw, and in predicting soil temperature with respect to irrigation. This research provides a first look at energy partitioning, crop microclimatic, and soil environment modification, as well as GHG dynamics from irrigated agricultural fields in the Canadian Prairies.

## ACKNOWLEDGEMENTS

All praises are for the Almighty Allah, the Lord of the universes, the Great and Merciful, Who deserves all credits to allow me carrying out this research work and preparing this report.

I deem it a proud privilege to express my earnest and heartfelt gratitude and appreciation to my esteemed supervisor *Dr. Warren Helgason*, for his patient guidance, intense supervision, untiring assistance, constant encouragement, valuable suggestions, constructive criticism and inestimable help during every phase of this work and preparation of this report.

I extends my sincere thanks to my committee Chair *Prof. Lope Tabil* and members *Dr. Reynald Lemke*, and *Prof. Charles Maule* for their valuable suggestions, and comments during the study period at the University.

I would like to express my heartfelt gratitude to *Dr. Ward Smith* and *Dr. Brian Grant* for providing the CDN-DNDC model for this research and their continuous help during the learning process of the model.

I would like to extend my appreciation and thankfulness to my colleague *David Cody* for his countless support in field and laboratory for experimental set up and data collection. The heartiest thanks to my other colleague *Elizabeth Edmondson* for her good company and patience during gas sampling in the field and cooperator *Garth Weiterman*.

This thesis didn't get its final format unless the support and help of my husband, *Dr. Mohammad Badrul Masud*. My special thanks and gratitude to him for continuous inspiration and criticism during data analysis, programming and report writing. I would like to thank my daughter *Ryefa* for her cute smile which remove my all stress and energized me to work further.

My sincere gratitude is due to my scholarship donor, *Ag-AgFood Canada* for funding me through the project *Agricultural Greenhouse Gas Programme (AGGP),* without which this studying at the University of Saskatchewan would not be possible. Financial support from the *Department of Biological Engineering Graduate scholarship*, and the *Saskatchewan Innovation Scholarship* also acknowledged.

Finally, I am very much grateful to my reverend parents and family members for their continuous encouragement, countless sacrifice, endless inspiration and blessing for completing this study.

# DEDICATION

This thesis is dedicated to my late Mom, late Mother-in-Law, dear husband *Mohammad Badrul Masud*, and my little angel *Ryefa Jannat Masud*

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# NOMENCLATURE

| | |
|---|---|
| *CF* | Clay fraction |
| *D* | Deep percolation (mm) |
| *DL* | Non-irrigated dryland field in 2012 and 2013 |
| *DNDC* | Denitrification-Decomposition |
| *DOC* | Dissolved organic carbon |
| $e_a$ | Actual vapor pressure (kPa) |
| *EC* | Eddy covariance |
| $e_s$ | Saturated vapor pressure (kPa) |
| *ET* | Evapotranspiration (mm) |
| *FAO* | Food and Agricultural Organization |
| *FC* | Field capacity |
| *G* | Ground heat flux (W $m^{-2}$) |
| *GHF* | Ground heat flux (W $m^{-2}$) |
| *GHG* | Greenhouse Gas |
| *H* | Sensible heat flux (W $m^{-2}$) |
| *HC* | Hydraulic conductivity (m $ha^{-1}$) |
| *I* | Irrigation (mm) |
| *IL12* | Irrigated field in 2012 |
| *IL13* | Irrigated field in 2013 |
| *LE* | Latent heat flux (W $m^{-2}$) |
| $L_{in}$ | Atmospheric longwave radiation (W $m^{-2}$) |
| $L_{net}$ | Net longwave radiation (W $m^{-2}$) |
| $L_{out}$ | Emitted longwave radiation (W $m^{-2}$) |

| | |
|---|---|
| *LWR* | Long wave radiation (W $m^{-2}$) |
| *P* | Precipitation (mm) |
| *P* | Porosity ($m^3$ $m^{-3}$) |
| *PET* | Potential evapotranspiration (mm) |
| *PWP* | Permanent wilting point |
| *Qg* | Ground heat storage (W $m^{-2}$) |
| *R* | Surface runoff (mm) |
| *RH* | Relative humidity |
| *RMSE* | Root mean square error |
| *Rn* | Net radiation (W $m^{-2}$) |
| $S_{in}$ | Incoming shortwave solar radiation (W $m^{-2}$) |
| $S_{net}$ | Net short wave radiation (W $m^{-2}$) |
| *SOC* | Soil organic carbon (kg C $kg^{-1}$) |
| $S_{out}$ | Reflected shortwave solar radiation (W $m^{-2}$) |
| *SWR* | Short wave radiation (W $m^{-2}$) |
| *T* | Air temperature (˚C) |
| *VPD* | Vapor pressure deficit (kPa) |
| *WFPS* | Water filled pore space (%) |
| $\Delta S$ | Change in soil moisture storage (mm) |

# 1. INTRODUCTION

## 1.1 BACKGROUND

In the 21st century, food demand is expected to be higher than in the 20th century, therefore agricultural production must increase. To increase crop production to a level that will satisfy the growing population, fertilizer and water may need to be applied in greater quantities. Irrigation and the application of fertilizer can increase crop yield, but may also increase greenhouse gas (GHG) emissions (Liebig et al. 2005). Much of Canada's agricultural production comes from the Canadian Prairies (Statistics Canada, 2011). Due to the vast agricultural land, this region also emits a significant amount of GHG; in 2012, Canada's share in the global context was 1.6% (Environment Canada, 2014). Therefore, GHG emission must be controlled if this region is to fulfill the world's food demand in a sustainable manner.

Globally, greenhouse gas emissions from the agricultural crop production sector are mainly $CH_4$ from flooded irrigated fields, and $N_2O$ and $CO_2$ from other types of irrigated and non-irrigated fields. Within the Prairies, where flood irrigation is infrequently practiced, $N_2O$ and $CO_2$ are the main trace GHGs. Nitrification/denitrification processes and soil respiration under wet conditions are responsible for $N_2O$ (Beauchamp, 1997) and $CO_2$ (Oberbauer et al., 1992; Davidson and Trumbore, 1995) emissions, respectively. As high soil water content can enhance the rate of denitrification as well as soil respiration, irrigation is expected to inherently influence $N_2O$ and $CO_2$ emissions. However, there are very few studies concerning the specific influence of irrigation upon GHG emissions.  As water application via irrigation is vital for agricultural production, a detailed study of how irrigation influences GHG emission, and how irrigation might improve crop yields while lowering GHG emission, is warranted.

Cropland GHG emission can be minimized by improving agronomic practices and nutrient use, by reducing tillage intensity, and by residue management. However, there is limited research that investigates effects of water management on the reduction of GHG emission (IPCC 2007). Improved water management was shown to reduce GHG emission significantly in a study in South-Eastern Queensland, Australia, by Scheer et al. (2012). The researchers irrigated wheat and cotton when the available soil water content had been depleted by 50%, 60%, and 85% and found that irrigation at 60% water depletion resulted in the lowest $N_2O$ emission. Water management, combined with other management practices, needs further research to determine its effectiveness in reducing GHG emission in the agricultural sector.

Sprinkler irrigation, a common method of irrigation in the Prairies, changes the surface energy fluxes above the crop canopy primarily via the change in the latent heat flux and sensible heat flux partitioning and the crop microclimate. This involves air, canopy, and soil surface temperatures, and the vapor pressure deficit. Irrigation increases the latent heat flux (energy released or absorbed by a body during a constant-temperature process), whereas the sensible heat flux (heat exchanged by a body that changes the temperature of the body) is decreased. This alters crop transpiration, soil temperature, and ground heat flux, which can be expected to influence soil GHG emissions. For example, soil temperature is correlated with soil GHG emission (Castaldi 2000), and variations in soil temperature occur in response to changes in the sensible heat flux. Irrigation also causes more obvious variations in the soil environment (volumetric water content, and matric potential) which may enhance GHG emissions. These variations depend on the volume of irrigation water and on the timing of irrigation. However, it is not presently understood how significant the enhancement of GHG emission is from an irrigation event, or how long this effect may last. Following irrigation, increased soil moisture tends to increase $N_2O$ emission via denitrification (Drury et al., 2003), and to increase $CO_2$ emission via soil respiration (Liu et al. 2002). When soil temperature increases, both $N_2O$ (Dinsmore et al., 2009; Schindlbacher et al., 2004) and $CO_2$ emission (Reth et al., 2005) can increase exponentially. These are common scenarios in a nutrient rich field. The variation in soil environment and its influence on soil GHG emission, and the time period of GHG emission in an irrigated agricultural are not well addressed in the literature. In this research, variations in energy partitioning, crop and soil environments, and their resulting influence on GHG emission were compared in irrigated and non-irrigated agricultural fields.

Implementation of proper management practices can reduce GHG emission. A process-based simulation model that represents the processes that usually occur in the soil can be used to predict the field condition and likewise the GHG emission. Such a process-based model can include more factors that influence the variability in GHG emissions than an empirical research method. As such, newly developed soil GHG simulation models can help to improve our understanding of the factors that affect GHG emissions in the long term, providing the capability to predict future GHG emission dynamics. Available process-oriented models like DeNitrification-DeComposition (DNDC), and Daily Century (DayCent) that can simulate GHG emission (with respect to processes like nitrification, denitrification, and mineralization) could potentially be used to investigate GHG emission and corresponding mitigation strategies for a given site. Although both DNDC and DayCent models are commonly used in Canadian agriculture (Chan et al., 2008), daily GHG emission predictions by the DayCent model are reported to be less reliable (Del Grosso et al. 2000). Furthermore, the DNDC model has been modified for regional use in Canada. As daily simulations of GHG emission were required in this research and a regional version (CDN-DNDC) of the DNDC model was available, the

DNDC model was selected for this study. The DNDC model has been used worldwide to explore GHG emission after treatments such as fertilizer (Smith et al., 2002; Li Hu et al., 2012; Kröbel et al., 2011), tillage (Smith et al., 2008), and film mulching (Han et al., 2013). The DNDC model has also been used to investigate crop yield (Zhang et al., 2002; Kröbel et al., 2011), crop growth and biomass (Kröbel et al., 2011), soil water and temperature (Smith et al., 2002 and 2008), and emission factors (Smith et al., 2012; Giltrap et al., 2013). However, the regional suitability of these models cannot be assumed. For example, Smith et al. (2002) tested the DNDC model (version 7.1) at two different agricultural sites in Canada and got a stronger agreement between predicted and actual GHG emissions in eastern Canadian agricultural fields than in western Canadian agricultural fields. This variation was primarily caused by the regional differences in the soil environment during winter. Subsequently, the DNDC model predictions have been improved for various crops and management conditions. Recently the regional version (CDN-DNDC) of DNDC model for this region has been coupled with new routines of transpiration and potential evapotranspiration (PET) along with the Food and Agricultural Organization (FAO) crop coefficient modification and biomass growth curve for Canadian crops (Grant, personal communication, 2015) for better prediction of soil moisture and crop evapotranspiration (ET). In this research the CDN-DNDC model is used in Western Canada, where soils are seasonally frozen, to predict the growing season GHG emissions. Limitations of the model for this region will be determined so that further modifications can be made. Although the DNDC model has been used to evaluate the emissions from agricultural soil after different treatments, the model has not been used to explore GHG emission versus irrigation practices. Hence, the CDN-DNDC model will be used to explore irrigation practices in Western Canada that lead to minimum GHGs emissions.

## 1.2 PROBLEM STATEMENT

To increase crop productivity and decrease GHG emission, an improved understanding of the effect of irrigation upon soil GHG emissions is essential. It is also important to understand the energy partitioning above the soil and canopy surface due to irrigation practices, which alter the crop microclimate and soil environment, subsequently influencing GHG emission. Limited research on how irrigation practice can be improved to obtain higher crop yield and lower GHG emission is present in the literature. To develop new irrigation-related mitigation strategies and to quantify the influence of irrigation on GHG emission, a GHG simulation model can play a vital role where field experiments are logistically and financially burdonsome. The GHG emission simulation model DNDC, has not been applied to examine the influence of irrigation on soil $N_2O$ emission and crop production. To the author's knowledge, this will be the first test of the DNDC model with respect to the effect of crop irrigation on GHG emissions in the Canadian Prairie region.

## 1.3 OBJECTIVES

The overall objective of this research is to assess how irrigation influences crop and soil environments to enhance GHG emissions. The specific objectives are to:

1. Assess the variation of energy partitioning, crop microclimate, soil moisture, and soil temperature during sprinkler irrigation and quantify their resulting influence on GHG emissions; and

2. Assess $N_2O$ emission using the CDN-DNDC model and explore the effect of irrigation management on soil $N_2O$ emission.

## 1.4 CONCEPTUAL MODEL

In the energy balance equation (net radiation – ground heat storage = latent heat flux + sensible heat flux), the difference between net radiation and ground heat storage is equal to the sum of the latent heat flux and the sensible heat flux. The latent heat flux represents the energy required to change the phase of a substance (e.g., liquid water to water vapor). The sensible heat flux represents the energy required to change the temperature of a substance without changing the phase (e.g., warming of the air, soil, and crop canopy in an agricultural field). Consider two adjacent fields, of which only one is under irrigation. The incoming radiation supplied to the two sites will be identical; however, the outgoing radiation, latent and sensible heat flux, can vary due to an irrigation event. When sprinkler irrigation is applied, some of the radiation is used to evaporate the water, whereas in the non-irrigated field a larger portion of the radiation is used to warm up the soil, canopy, and air. Hence, irrigated and non-irrigated fields acquire a different energy balance. As irrigation alters both the energy and water balance, it modifies the soil environment (i.e., soil moisture and temperature) and influences greenhouse gas emissions. The first objective of this research is to assess these variations and quantify their influence on GHG emission.

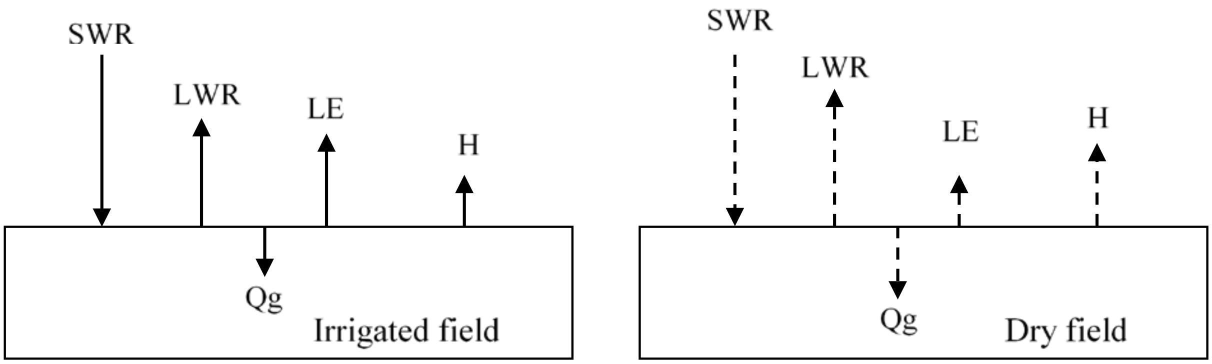


Fig. 1.1. Energy balance terms in irrigated and dry fields during irrigation practices (SWR = short wave radiation, LWR = long wave radiation, LE = latent heat flux, H = sensible heat flux, and Qg = ground heat storage).

As $N_2O$ emissions tend to increase under the conditions of moist soils, expanding the irrigated area may increase the total GHG emissions. However, the crop yield also increases when irrigation is applied, therefore, both $N_2O$ emission and crop yield must be considered. A field experiment for this goal would involve high time and cost constraints, therefore, a process-based model will represent the field condition. The second objective of this research is to evaluate the $N_2O$ emission from an irrigated agricultural field using a process-based model (CDN-DNDC model).

## 1.5 THESIS STRUCTURE

This thesis is written in manuscript style; hence, each objective is addressed in a separate chapter. Chapter 2 presents the literature review of this research. Chapter 3 presents the first of the two research studies - a field-based investigation of how irrigation influences surface energy fluxes, crop microclimate, and soil environment, resulting in changes in soil GHG emission. The specific objectives of this study were to: (a) measure and compare the latent heat flux, the sensible heat flux, the net radiation, and the ground heat flux from typical irrigated and non-irrigated cropping systems; to (b) measure and compare the canopy and soil surface temperature, and vapor pressure deficit corresponding to irrigation; and to (c) observe how soil GHG fluxes varied between the irrigated and the non-irrigated field. In the second study, presented in Chapter 4, a process-based model investigation of how irrigation influenced soil $N_2O$ emission. The objective of this study was to: (a) determine the suitability of the CDN-DNDC model by validating the model for this region; to (b) identify sensitive model parameters for this region; and to (c) determine the long-term effect of irrigation on crop yield and $N_2O$ emission. Following the research studies, Chapter 5 - Summary and Conclusions - ties Chapters 3 and 4 together and suggests areas for future research. A list of the literature cited throughout the thesis is presented at the end of the each chapter. The document is concluded with a collection of Appendices.

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# 2. REVIEW OF LITERATURE

This chapter provides an overview of previous research on soil-based emissions of major greenhouse gasses (GHGs) from agricultural cropped field under irrigated conditions along with their relation with surface energy fluxes, water balance, and management practices e.g. tillage and irrigation. This chapter also provides an overview of surface energy fluxes and crop microclimates within agricultural fields. Finally, an overview of GHG simulation models is provided. The main purpose of the literature review was to identify the gaps in previous research which are used to guide the present research.

## 2.1 GREENHOUSE GAS EMISSIONS AND AGRICULTURE

The major greenhouse gasses (GHGs) from agriculture are nitrous oxide ($N_2O$), carbon dioxide ($CO_2$), and methane ($CH_4$) (IPCC, 2001). Agricultural lands occupy about 40-50% of the Earth's land surface (consisting of cropland, managed grassland and permanent crops, agroforestry, and bio-energy crops) (IPCC, 2007). In 2010, the total land under agricultural production was 4,889 Mha, an increase of 7% (311 Mha) since 1970 (FAOSTAT, 2013). However, agricultural land area has decreased by 53 Mha since 2000 due to a decline of cropland area (Smith et al., 2014). Around 10-12% of total global anthropogenic emissions of GHGs came from agriculture (IPCC, 2014) in 2010. About 47% of total anthropogenic emissions of $CH_4$ and 58% of $N_2O$ was contributed by agriculture. The largest source of $N_2O$ is soil emissions, and the largest source of $CH_4$ is enteric fermentation, amounting to about 38% and 32%, respectively, of the total non-$CO_2$ emissions. Biomass burning (12%), rice production (11%), and manure management (7%) account for the remainder (US-EPA, 2006).

Agricultural crop and livestock production emissions increased to 14% of the total emission in 2011 as compared to the emissions from 2001. However, the emissions due to land use change and deforestation registered nearly a 10% decrease over the 2001-2010 period (FAO, 2014). Globally, agricultural GHG ($CH_4$ and $N_2O$) emissions increased 17% between 1990 and 2005 (US-EPA, 2006), which is equivalent to 58 Mt $CO_2$-eq $yr^{-1}$. Due to the higher utilization of nitrogen fertilizer and animal manure production, FAO (2003) predicted $N_2O$ emission will be 35-60% higher by 2030. The $N_2O$ emissions will also increase at least 50% by 2020 relative to 1990 (Mosier and Kroeze, 2000; US-EPA, 2006) due to the projected food demand.

The main emissions from Canadian agriculture are $N_2O$ and $CH_4$ from crop production and the animal sector. Agriculture accounts for 72% of the national $N_2O$ emissions. The main drivers of the emissions from the agricultural crop production sector are synthetic nitrogen fertilizers applied in the Prairies (NIR, 2012). That is why the crop production itself contributed 19 and 22 Mt $CO_2$eq $yr^{-1}$ in 2005 and 2010, respectively (Environment Canada, 2012). Here, $N_2O$ is the main trace of the GHGs emission whereas $CO_2$ is the second one. As flood irrigation is rarely practiced in this region, $CH_4$ production is relatively low. The details of $N_2O$ and $CO_2$ trace gas emissions are presented in the next sub-sections.

Potential greenhouse gas emission mitigations within the agricultural sector depend on sustainable development, climate change policies, and improvement of environmental quality (IPCC, 2007). There is a likelihood of higher emissions in the future due to increasing use of nitrogen fertilizer. According to IPCC, the most obvious options for GHGs mitigation within the agricultural sector is through improved agronomic practices, increased nutrient efficiency use, and better residue management. Another significant mitigation possibility is improved water management, which has received comparatively little consideration within the crop production sector.

### 2.1.1 $N_2O$ emission

Soil $N_2O$ is generated by the microbial transformation of nitrogen in soils and manures and is often enhanced in a field where available nitrogen (N) exceeds plant requirements, usually under wet conditions (Smith and Conen, 2004; Oenema et al., 2005). The overall N cycle in soil is given in Fig. 2.1.

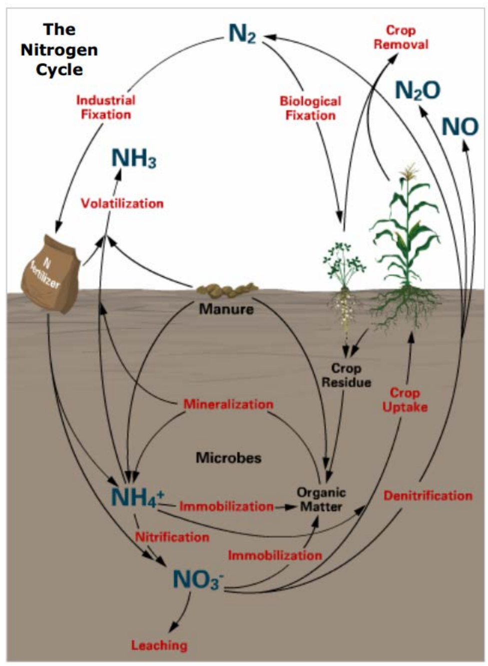



Fig. 2.1. Overall N cycle in soil (Courtney et al. 2005, with permission)

Nitrous oxide is an important trace greenhouse gas, which has 298 times the global warming potential of carbon dioxide with a contribution of 8% to the anthropogenic global warming (IPCC, 2007). In 2011, an averaged atmospheric $N_2O$ emission was 324.2 ppb, which was 5 ppb above the reported value for 2005 (IPCC, 2013). Prather et al. (2012) reported that this is an increase of 20% over the estimate for 1750 derived from ice cores. Many researchers

(Rockmann and Levin, 2005; Ishijima et al., 2007; Davidson, 2009; Syakila and Kroeze, 2011) have found that $N_2O$ emissions have been increasing since the 1950s, mainly due to the emission from soils associated with the use of synthetic and organic (manure) nitrogen fertilizer. Mosier et al. (1998) also stated that 50-60% of the anthropogenic-induced $N_2O$ emissions came from agriculture, where the main direct emission was from agricultural soils.

In an agricultural field, $N_2O$ emission can be produced from several microbial activities. The two main processes of $N_2O$ emission are nitrification under aerobic conditions, and denitrification in an oxygen deficit (anaerobic) environment. Nitrous oxide emission is highly variable due to the variability of soil carbon content, soil moisture, and nitrogen inputs. The factors that can influence $N_2O$ emission are moisture and aeration, temperature, soil and fertilizer nitrogen, soil pH and salinity, soil organic carbon (SOC), types of vegetation, and bulk density (Granli and BØckman, 1994; Stehfest and Bouwman, 2006). The processes that are responsible for $N_2O$ emission are briefly described in the following sub-sections.

Soil $N_2O$ emission increases with soil organic content and decreases with a reduction in bulk density, and an increase in soil pH (Kanerva et al., 2007; Stehfest and Bouwman, 2006). Niklaus et al. (2006) found that $N_2O$ emission can decrease if the plant species diversity increases, particularly if legumes are present. Khalil and Baggs (2005) reported that $N_2O$ emission becomes the highest in wet soils when WFPS is about 75%. They also found that 90% $N_2O$ emission comes through the denitrification process when soil micropores are primarily anaerobic.

#### 2.1.1.1 Nitrification

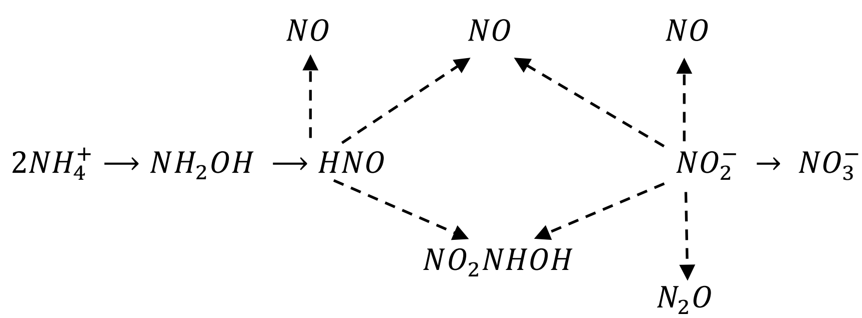
Nitrification is a microbial process that converts ammonium to $NO_2^-$ and $NO_3^-$ by the help of several bacterial activities. Nitrification is carried out by chemolithoautotrophic bacteria under aerobic condition so that the bacteria can use $O_2$ as a terminal electron acceptor. At first, $NH_4^+$ is oxidized to $NO_2^-$ by ammonia oxidizing species of the genus Nitrosomonas, and then during the second step Nitrobacter and Nitrococcus bacteria oxidizes $NO_2^-$ to $NO_3^-$ (Bremner and Blackmer, 1981; Watson et al. 1981) (Eq. 2.1 and 2.2).

During the first step, hydroxylamine ($NH_2OH$) and nitroxyl (NOH) are also formed, which are intermediate and unstable compounds. At this time, large amounts of molecular $O_2$ is consumed by ammonia oxidizers, causing an anaerobic microsite condition leading to a reduction of $NO_2^-$ to $N_2O$ and $N_2$ (Zart and Bock, 1998; Colliver and Stephenson, 2000).

$$2NH_4^+ + 3O_2 \xrightarrow{Nitrosomonas\ (step\ 1)} 2NO_2^- + 2H_2O + 4H^+ \quad (2.1)$$

$$\downarrow Nitrobacter\ (step\ 2)$$

$$2NO_3^-$$




(2.2)

Many researchers (Šimek., 2000; Zaman and Chang, 2004; Zaman and Nguyen, 2010) have found that sufficient soil $O_2$ levels (optimum at WFPS of 60%), adequate $NH_4^+$ concentrations, a favorable soil temperature above 5°C (optimum 25 to 35°C), and soil pH above 5 (optimum 7 to 9) control the rate of autotrophic nitrification. Zaman et al. (2007) mentioned $NH_4^+$ and $O_2$ concentrations as the most critical factor. Armstrong (1964) found that nitrification can also occur under water-logged soil conditions.

#### 2.1.1.2 Denitrification

Denitrification is a microbial process where $NO_3^-$ and $NO_2^-$ are reduced to $N_2O$ and $N_2$ through respiratory metabolism. Under anaerobic (absence of sufficient oxygen) microbial (mainly bacterial) conditions, denitrification reduces nitrate successively to nitrite and then to the gases of NO, $N_2O$, and $N_2$.

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$$

Several species of bacteria are involved in the complete reduction of nitrate to molecular nitrogen, and more than one enzymatic pathway has been identified in the reduction process. Four different reductase enzymes stimulate the complete denitrification process. Although denitrifiers are aerobic bacteria, they like to use N-oxides at low $O_2$ level (Tiedje, 1988).

Factors that influence denitrification rates are: (i) $NO_3^-$ substrate to accept electron, (ii) organic C to donate electron in an $O_2$ limited condition when soil moisture content >60% WFPS, (iii) suitable soil pH (5 to 8), (iv) soil temperature between 5 to 30°C (optimum 25°C) (Aulakh et al., 2001; Zaman et al., 2007, 2008b, 2008c, 2009). The critical regulator of denitrification is $O_2$ (Tiedje 1988). Hence, factors such as rainfall events, soil texture, and tillage, which can alter soil $O_2$ levels can influence the denitrification rate.

### 2.1.2 $CO_2$ and $CH_4$ emission

The estimated amount of global anthropogenic $CO_2$ emission from agriculture in 2005 was 5.1 to 6.1 $GtCO_2$-eq $yr^{-1}$ (10-12% of global GHGs), whereas 3.3 $GtCO_2$-eq $yr^{-1}$ emission was contributed by $CH_4$ (IPCC 2007). In agricultural fields, $CO_2$ is mainly released from burning of plant litter or microbial decay of soil organic matter (Smith, 2004b; Janzen, 2004). When organic decomposition occurs in oxygen-deprived conditions, such as from under stored manures or flooded lands, $CH_4$ is released to the atmosphere (Mosier et al. 1998). Soil carbon dynamic processes are shown in Fig. 2.2. When $CO_2$ emissions occur from the soil through respiration, they follow three different biological respiration processes: namely, microbial respiration, root respiration and faunal respiration (Edward 1975) along with one non-biological reaction i.e. chemical oxidation at a higher temperature. Factors that affect $CO_2$ emissions from soil are soil temperature, moisture, texture, pH, available C, and N content in the soil (Bunnell et al. 1977). Soil moisture is the key factor of $CH_4$ emission as soil moisture control the diffusivity of soil (US-EPA, 2010).

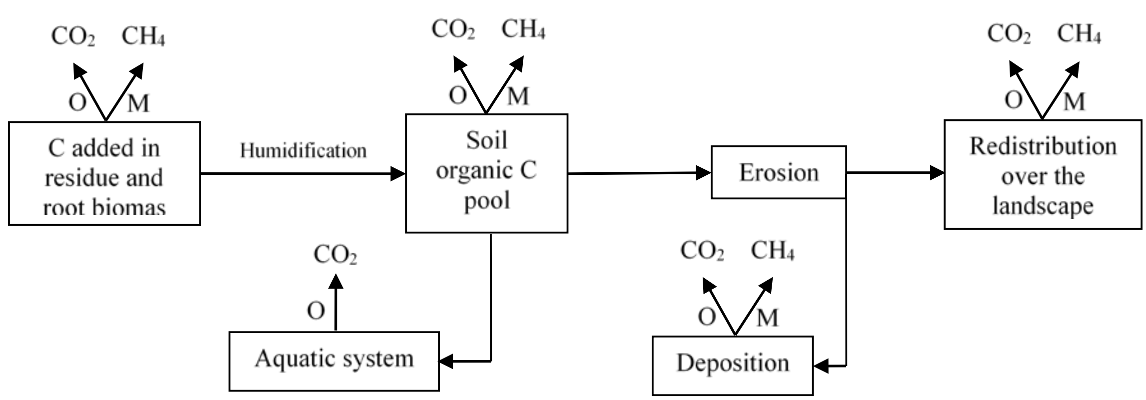



Fig. 2.2. Soil carbon dynamics processes in soil. O = Oxidation, M = Methanogenesis (modified from Lal, 2001)

Soil texture affects the formation of $CO_2$ by affecting the growth of bacteria and fungi through the supply of air and moisture. Soil texture also affects water infiltration and gas diffusion rates into soil, indirectly influencing $CO_2$ emission. $CO_2$ evolution is higher in clay soil than sandy soil (Kowalenko and Ivarson, 1978). Soil pH can affect the growth and proliferation of soil microbes. Soil $CO_2$ efflux is lower in soil at pH 3.0 than a soil with pH 4.0 (Sitaula et al., 1995). Soil $CO_2$ emission increase with the rise of soil pH up to 7.0 (Kowalenko and Ivarson, 1978). However, soil pH above 7.0 can reduce $CO_2$ emissions. Nitrogen fertilizer also affects $CO_2$ emission directly by adding nitrogen to the soil and indirectly by affecting soil pH (Katznelson and Stevenson, 1956). Though inorganic N has a relatively small effect on $CO_2$ emissions, manure application can increase $CO_2$ emission by increasing soil respiration by a factor of 2 to 3 (Rochette and Gregorich, 1998).

## 2.2 IRRIGATION IN CANADA

Agricultural production used approximately 1.7 billion cubic meters of water in Canada in 2012 (Agricultural Water Survey, 2012). Most of this water was used to irrigate agricultural field crops. The main irrigation areas are located in the provinces of Alberta and Saskatchewan, followed by Manitoba, British Columbia, and Ontario. Harker et al. (2004) reported that the Prairies use the 75% of the withdrawn water in the country in the agricultural sector, from which approximately 85% of that is used for irrigation.

In the late 1800s, irrigation development began in Alberta and Saskatchewan in order to increase production and economic benefit. In Saskatchewan, the highest expansion of irrigated land occurred in the first two decades of the $20^{th}$ century (SIPA, 2008). By 1930, expansion had slowed down due to widespread drought on the prairies, *i.e.* the ‘Dirty Thirties’. Following the drought there was interest in expanding irrigation facilities and, as a result, Lake Diefenbaker reservoir was developed as the primary source of irrigation water in Saskatchewan. The total irrigated land in Saskatchewan at the end of 1920 was 4419 hectares which had expanded to 141639 hectares by the end of the 20$^{th}$ century (SIPA, 2008). In Saskatchewan, the most popular method of irrigation is sprinkler (Ruffino, 2009) with 2,075 farms under this type of irrigation in 2010 (Statistics Canada, 2010).

## 2.3 ENERGY AND WATER BALANCE OF IRRIGATED FIELD

Irrigation plays a role in modifying the net energy budget (Eq. 2.3), and the water budget (Eq. 2.4). Along with increasing the soil water content, sprinkler irrigation also influences the surface energy fluxes and crop microclimate. Jiang et al. (2014) investigated the effect of irrigation on surface energy fluxes and temperature in Northern China and found that irrigation influenced the spatial pattern of the surface energy budget. They noted that irrigation resulted in a mean annual latent heat flux increase of 12.10 Wm$^{-2,}$ and sensible heat flux decrease of 8.85 Wm$^{-2}$ and a reduction in air temperature of 1.3 °C, across their study region. During sprinkler irrigation, the air, canopy, and soil temperatures are reduced while the atmospheric water vapor (relative humidity) increases (Tolk, et al., 1995; Liu and Kang, 2006a; Cavero, et al., 2009; Zhao, et al., 2012). This disparity comes from the variation of energy balance terms (Eq. 2.3). After an irrigation event on a sunny day most of the incoming radiation is used to evaporate water from its liquid form to vapor form (through latent heat flux); hence, sensible heat exchange between atmosphere and soil and canopy is reduced, which can lower the conductive heat flow into the soil and alter the soil temperature.

$$R_n - G - S = LE + H \tag{2.3}$$

$$I + P = ET + \Delta S + R + D \tag{2.4}$$

In the energy and water balance equations above: $R_n$ = net radiation (W $m^{-2}$); G = ground heat flux (W $m^{-2}$); S = storage heat flux (W $m^{-2}$); LE = latent heat flux (W $m^{-2}$); H = sensible heat flux (W $m^{-2}$); I = irrigation (mm); P = precipitation (mm); ET = evapotranspiration (mm); ΔS = change in soil moisture storage (mm); R = surface runoff (mm); and D = deep percolation (mm).

Water loss due to runoff and deep percolation are usually considered negligible when modern sprinkler irrigation is practiced (Thomson, 1986), so the final form of equation 2.4 simplifies to:

$$I + P = ET + \Delta S \tag{2.5}$$

Irrigation water application modifies the field water balance by altering the ET as well as soil moisture storage. Many studies have shown that after irrigation ET in the irrigated field rises due to the presence of readily available water for evaporation (Tolk et al., 1995; Cavero et al., 2009). Suna et al. (2006) found a linear relationship between irrigation and evapotranspiration in their research in North China Plain.

### 2.4 IRRIGATION AND CROP MICROCLIMATIC VARIATION

Sprinkler irrigation applies water in the cropped field, imitating a form of rainfall. During, and after sprinkler irrigation, evaporation increases from airborne droplets, canopy interception, and the wet soil surface. Hence, the crop microclimate is highly influenced by an irrigation event. During the evaporation process, droplets add water vapor to the atmosphere by exchanging heat with the air (Kohl and Wright, 1974). Tolk et al. (1995) mentioned that vapor pressure deficit (VPD) and air temperature decreased significantly during and following a sprinkler irrigation event. Many other researchers (Robinson 1970; Steiner et al. 1983; Thompson et al. 1993; Liu and Kang 2006a; Cavero et al. 2009, Yenny et al. 2013) also have found that air temperature and air vapor pressure deficit (VPD) decreased due to irrigation. Liu and Kang (2006b) reported decreases of canopy temperature of wheat of 0.3 to 2.8 °C in a sprinkler-irrigated field compared to a non-sprinkled field. When Steiner et al. (1983) compared the microclimate of maize under center pivot sprinkler and surface irrigation; they found that the daily average canopy and air temperatures of the sprinkler irrigation field were cooler than those of the surface irrigation field. Tolk et al. (1995) and Cavero et al. (2009) found for lateral move and solid set sprinkler irrigation also decreases the canopy temperature.

It has also been found that the cooling effect of sprinkler irrigation is higher during days of high evaporative demand. Yenny et al. (2013) reported that conditions of decreased air temperature lasted about 1.3 hours after an irrigation event, which is similar to the findings of other studies (Thompson et al., 1993; Tolk et al., 1995; Cavero et al., 2009).

## 2.5 IRRIGATION AND GREENHOUSE GAS EMISSION

Kulshreshtha and Junkins (2001) stated that, when irrigation development occurs, GHG emission comes from three different sources, namely direct emission, indirect emission, and induced emission (Fig. 2.3). Irrigated land requires more fertilizer input in order to achieve higher yields, so emissions are usually increased. Similarly, due to higher production in irrigated land, more crop residue stays in the field, which results in higher emissions from soil. Irrigation water can also transport nitrogen into groundwater through leaching.

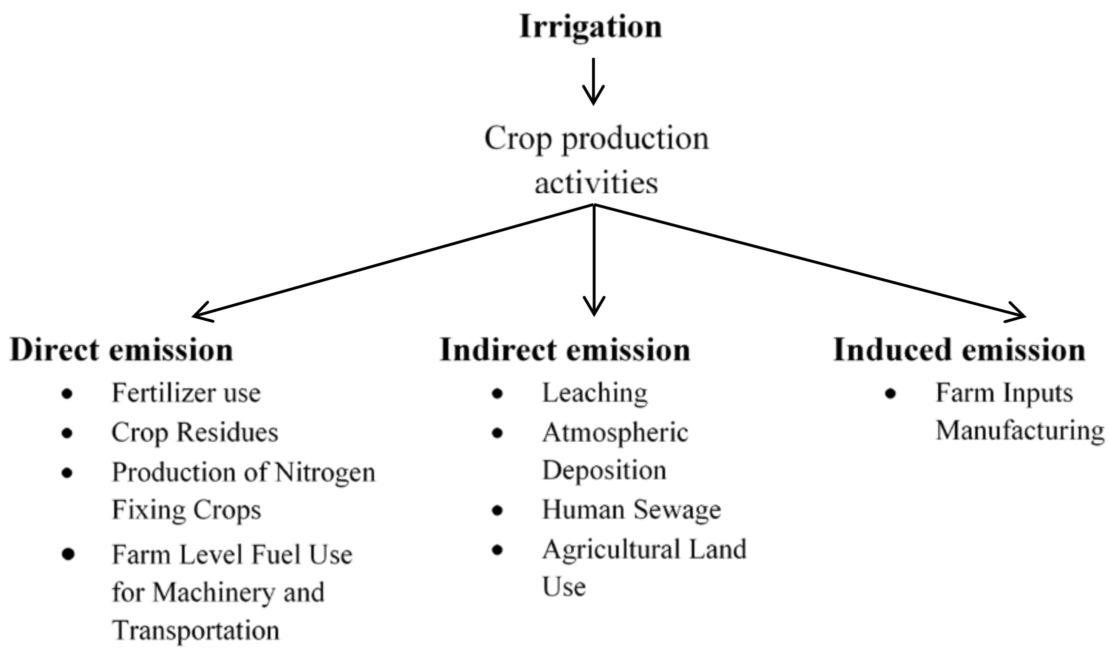



Fig. 2.3. Overview of Greenhouse Gases Emissions from irrigated agricultural crop production (Kulshreshtha and Junkins, 2001)

## 2.6 SOIL MOISTURE AND TEMPERATURE EFFECTS UPON GHG EMISSION

Carbon dioxide and $N_2O$ emissions are anticipated to increase under irrigation. Soil moisture and temperature are the vital environmental factors which influence GHG emission from the soil through the modification of soil respiration, nitrification, denitrification, and mineralization. Dry soil rewetting by irrigation or rainfall can increase soil $CO_2$ emission as it enhances respiration, C mineralization, and microbial activities, (van Gestel et al., 1993; Calderon and Jackson, 2002). $N_2O$ production can be affected by soil water content as limited $O_2$ availability from high soil water produces $N_2O$ via denitrification (McKenney et al., 2001).

An increase in the water filled portion of the pore space results in restriction of oxygen diffusion. One way of measuring the level of $O_2$ availability in the soil is by assessing the percentage water filled pore space (WFPS) (Maag and Vinther, 1999). At lower percentage of WFPS, $N_2O$ production is low due to the restricted microbial activity. When soil water content is increased, nitrification and other aerobic processes set in. The nitrification process is considered to be at its optimum between 50 and 60% WFPS (Stevens et al., 1997). Denitrification processes over takes nitrification very rapidly at WFPS above 60%, reaching its peak between 70 and 90% WFPS (Lemke et al., 1998). Ruser et al. (2006) and Drury et al. (2003) found maximum $N_2O$ fluxes from the soil with 90% WFPS, and Bouwman (1990) reported that the threshold of denitrification is 65% of field capacity (FC). Denitrification promoted by a wetting and drying cycle (Bouwman 1990, Letey et al. 1981). During wetting, more $N_2O$ emissions take place, and if after wetting soil dried very quickly then reduction of $N_2O$ to $N_2$ can be prevented (Bouwman 1990, Leteyet al. 1981). Jha et al. (2012) also found a higher denitrification rate in saturated soils versus those at field capacity.

Some studies have shown that emissions of $N_2O$ increase with rising soil temperatures. During denitrification, the ratio of $N_2O/N_2$ increases with decreasing temperatures. The rate of nitrification and denitrification is influenced by the soil temperature (Bouwman 1990). For elevating soil temperature up to 60°C or even 75°C, denitrification rates increase (Bouwman 1990). This increasing trend is exponential for soil temperatures between 0°C to 25°C (Rochette et al. 2004, Castaldi 2000). However, the optimum denitrification occurs at temperatures > 25°C while the lowest rate occurs at a temperature < 15°C (Bouwman 1990, Keeney et al. 1979). At soil temperatures between 0°C and 5°C denitrification is relatively low (Bailey and Beauchamp 1973; Knowles 1982). The optimum rate of nitrification occurs between 30°C and 35°C (Alexander 1977; Bouwman1990). Nitrification is negligible at temperatures < 5°C and > 40°C (Bouwman 1990).

Soil moisture is one important factor of $CO_2$ emission. Soil moisture increases $CO_2$ emission up to an optimum level, after which emissions decrease again. A periodic drying and wetting also influence $CO_2$ development. Orchard and Cook (1983) found that during rewetting, soil microbes start working from a latent state in dry soil, hence increasing $CO_2$ evolution. Borken et al. (1999) observed that drought reduces soil respiration, but after this, any rewetting increased the respiration by 48% to 144%. Moore and Dalva (1997) found that $CO_2$ emission exhibits a positive, linear relation with soil water content. Under dry condition soil respiration is higher during the day time than night time, whereas under wet condition soil respiration is same in both at day and at night (Grahammer et al. 1991).

Soil $CO_2$ emission is greatly influenced by soil temperature. The diurnal fluctuation of temperature can affect root respiration (Bouma et al., 1997). Sato and Seto (1999) observed that $CO_2$ emission increases exponentially with increase in incubation temperature from 4 to 40°C. Kirschbaum (1995) found that 10% soil organic carbon will be lost if mean annual temperature increases just 1°C in a region where the mean annual temperature is 5°C. He also reported that 3% soil organic carbon will be lost for the same amount of annual temperature increase in a region where the mean annual temperature is 30°C. Moore and Dalva (1997) simulate soil temperature in a laboratory incubation test to see its effect on $CO_2$ emission and found that $CO_2$ emission is 2.4 times higher at 23°C than that at 10°C.

## 2.7 RELATION OF TILLAGE AND CROPPING SYSTEM WITH GHG EMISSIONS

Soil disturbance and microbial activities lessen with the reduction of tillage intensity, which in turn reduces the emission of $CO_2$ and $N_2O$ (Lemke et al., 1999; Drury et al., 2006;

Mosier et al., 2006). Inversely, $CO_2$ emission can rise due to higher soil aeration and disruption of soil aggregates (Roberts and Chan, 1990) after increasing tillage intensity. The quality and quantity of crop residue which returns to the soil depends on the cropping system and can affect $CO_2$ and $N_2O$ emissions (Mosier et al., 2006; Sainju et al., 2010). Field management practices influence soil temperature and water, which are two important factors of GHGs emission, hence influence soil $CO_2$, $N_2O$, and $CH_4$ emissions (Parkin and Kaspar, 2003; Dusenbury et al., 2008; Liebig et al., 2010). As tillage creates a favorable condition for microbial decomposition of plant residue in the soil, tilled soil emits a higher amount of $CO_2$ than from an undisturbed (or zero tillage) soil (Rochette and Angers, 1999). The cause of low $CO_2$ emission from the no-tilled field is reduced gas diffusivity and air-filled porosity (Ball et al., 1999). They found the proportion of respired soil organic C in the 60-day period was twice in moldboard plowing field than a no-tillage field. Tillage often results in drier soil because of higher rates of soil evaporation, leading to altered GHGs emission patterns, such as an increase in $CO_2$ fluxes (Curtin et al., 2000; Al-Kaisi and Yin, 2005).

### 2.8 GREENHOUSE GAS SIMULATION MODELS

In-situ measurement of greenhouse gas emissions can provide accurate estimates of the implications of changing agricultural practices. However, considering the diversity of crop, soil, climate, and fertilizer management, it is difficult to represent a broad range of conditions during field research studies. To overcome this problem, GHG simulation models can be used.

There are many GHG simulation models e.g. Daily-time-step version of the CENTURY ecosystem model (DAYCENT) (Parton et al. 1996, 1998, 2001; Del Grosso et al. 2000), DeNitrification–DeComposition (DNDC) (Li et al. 1992a, b,1994, 1996; Li 2000), ecosys

(Grant 2001), Nitrogen LOsses from Soil Systems (NLOSS) (Riley and Matson 1998, 2000), Expert-N (Engel and Priesack 1993; Priesack et al. 2001), Water and Nitrogen Management Model (WNMM) (Li et al. 2005, 2007), FASSET (Olesen et al. 2002), and CERES-NOE (Godwin and Jones 1991; Henault et al. 2005). Among these, NGAS-DAYCENT, DNDC, ecosys and Expert-N are applicable for Canadian agriculture (Chen et al., 2008). The most popular two models for this region are DayCent and DNDC (Chen et al., 2008).

The initial soil nitrification model (NGAS) was developed by Mosier et al. (1983) and became a mechanistic model to predict daily $N_2O$ losses from semi-arid grasslands and irrigated soil after using the effort of Mosier and Parton (1985), and Parton et al. (1988a, b). Later on, NGAS-DAYCENT (Parton et al. 1996, 1998, 2001; Del Grosso et al. 2000), a daily-time-step version of CENTURY ecosystem model, was developed to simulate NO, $N_2O$ and $N_2$ and $CH_4$ emissions. Grant and Pattey (2003) described this model as relatively simple and more empirical compared to other detailed ecosystem models. Parton et al. (1996; 2001), and Del Grosso et al. (2000) have investigated the accuracy of this model in a few regions including Colorado, USA. They found that this model accurately simulates the annual mean trend of $N_2O$ emission; however, the daily observed and modeled emission had some dissimilarity.

The DNDC model, a complex simulation model, has been developed by Li et al. (1992a, b). This model predicts daily $N_2O$ emission through nitrification and denitrification process, $CO_2$ emission from the decomposition of organic matter and root respiration as well as $CH_4$ emission. To simulate emission, an hourly-time-step denitrification sub-model of DNDC is also available. With some site-specific modification, this model has widely been used in many countries for both site and regional $N_2O$ emission from agricultural fields (Li 1995; Li et al.

1996; Plant 1999; Wang et al. 1997; Zhang et al. 2002; Brown et al. 2002; Xu-ri et al. 2003; Cai et al. 2003; Smith et al. 2004; Saggar et al. 2004; Pathak et al. 2006). For different purposes like $N_2O$ emission (Smith et al., 2002), $N_2O$ emission factor (Smith et al., 2012; Giltrap et al., 2013) and crop yield (Kröbel et al., 2011) determination, this model has been improved significantly to represent Canadian agriculture. This model is poor for predicting $N_2O$ emission during spring thaw period from regions where soils are continuously frozen during winter (Smith et al., 2002, Kariyapperuma et al., 2011). Recently this model has been improved for water use in the agricultural fields in Canada by coupling new routines of transpiration and potential evapotranspiration (PET) along with FAO crop coefficient modification and biomass growth curve for Canadian crops (Grant, personal communication, 2015).

## 2.9 SUMMARY

This review reveals a lack of significant literature regarding the specific topic of investigation for this research. To date, the combined influence of irrigation on surface energy flux, crop microclimate, soil environment and GHG emissions has not been considered. Furthermore, this review points to a lack of regional application of GHG simulation models (e.g. DNDC) in irrigated fields, particularly in the seasonally frozen Canadian Prairies.

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# 3. EFFECT OF IRRIGATION UPON SURFACE ENERGY FLUX, CROP MICROCLIMATE AND SOIL ENVIRONMENT TO INFLUENCE SOIL GHG EMISSIONS

## 3.1.1 PREFACE

The Canadian Prairies has a great potential to increase crop yield by expanding irrigated farms. Irrigation drives higher crop production, however, it also can lead to higher greenhouse gas (GHG) emissions. As irrigated fields require a relatively higher amount of fertilizer than non-irrigated fields, GHG emission can increase after irrigation. The change in soil environment, created by irrigation can also produce favorable conditions for enhanced GHG emissions. However, the dynamics of soil water, temperature, energy partitioning, and crop microclimatic modifications that occur with irrigation have not previously been examined in agricultural fields in the Canadian Prairies; and thus form the goal of this study. The objective was addressed through continuous, in situ monitoring of surface energy fluxes, crop microclimate, soil conditions, and soil GHG flux over a period of two years.

## 3.1.2 ABSTRACT

Soil moisture and temperature, two important driving factors of GHGs emissions from agricultural soils, are influenced by the practice of irrigation. Irrigation not only alters soil moisture and temperature to stimulate GHG emission but also changes surface energy fluxes, which may influence the crop production. This study investigates the influence of sprinkler irrigation events upon surface energy fluxes, crop microclimate, soil moisture, soil temperature, and soil GHG emissions from adjacent irrigated and non-irrigated wheat and canola fields, located south of Saskatoon, Saskatchewan. All of the energy balance components were independently measured in each field, as well as soil moisture and soil temperature. Soil GHG emissions were measured semi-weekly using the static gas chamber method. This study observed that, following snowmelt, early season soil moisture conditions were adequate and energy fluxes were similar for both fields. However, later in the season, from mid-July through August, there was a strong decrease in soil moisture in the non-irrigated field due to a lack of rainfall. This created a sharp contrast between fields in how the available energy was partitioned. As the soil moisture in the non-irrigated field declined, the latent heat flux correspondingly decreased and more energy went into warming up the crop canopy and soil surfaces, causing larger sensible heat exchange with the atmosphere. The partitioning of the energy fluxes due to irrigation promoted crop microclimatic modification leading to reduced vapor pressure deficit and canopy temperature. However, despite a much smaller proportion of the net radiation in non-irrigated systems being consumed by evaporation, the non-irrigated fields did not exhibit markedly warmer soil temperatures. The soil GHG emissions were highest during the early season when both fields had high soil moisture and soil nutrient, but later in the season GHG emissions were minimal, due to a lower nutrient availability in irrigated field and lower soil moisture in non-irrigated field. Soil water was found as the critical factor in influencing soil GHG emissions, however, availability of soil nutrient (soil N) was the dominant factor in soil $N_2O$ emissions from irrigated systems.

**Keywords:** latent heat flux, sensible heat flux, crop microclimate, soil environment, GHG.

## 3.2 INTRODUCTION

Agricultural fields occupy about 40-50% of the earth's land surface (consisting of cropland, managed grassland and permanent crops, agro-forestry and bio-energy crops, IPCC, 2007). Only around 12% (1.5 billion ha) is used for crop production (FAO, 2012). Similarly in Canada, 36.4 million ha is used for cropland from a total of 67.5 million ha of agricultural land. As the limited agricultural area is a constraint to higher food production, supporting management activities like irrigation, and fertilizer application can increase production; but they also increase greenhouse gas (GHG) emissions (Liebig et al., 2005). The Canadian Prairies region has a potential opportunity to help fulfill the world food demand by increasing the portion of its land that is irrigated. FAO (2013) reported that just 20% of the world's irrigated croplands produced 40% of the global harvest.

Increased agricultural production in the Prairies is likely to also increase greenhouse gas emissions; mainly nitrous oxide ($N_2O$) emission. Agriculture accounts for 72% of the national $N_2O$ emissions of Canada. Crop production itself contributed 19 and 22 Mt $CO_2$e in 2005 and 2010, respectively (Environment Canada, 2012). For the purpose of minimizing greenhouse gas emission from the Prairies agricultural sector, research on different agronomic practies (i.e. reduced tillage, optimizing fertilizer application, and avoiding fall fertilizer application) has been conducted, and associcated best management practices have been promoted. Within the Prairies, the largest magnitude of $N_2O$ emissions are often observed after rainfall or irrigation events during the growing season (Lemke, 2007; Lemke and Farrell, 2008). Irrigation alters the soil moisture, which is one of the most prominent environmental factors influencing GHG emission from the soil through its effect on soil respiration, nitrification, denitrification, and mineralization. However, there has been little focus on improving irrigation management, as a

GHG mitigation strategy, and the exact mechanisms of how sprinkler irrigation may influence GHG emissions have not been previously studied.

The main purpose of irrigation is to increase crop productivity by improving soil water availability, but it can also alter the energy partitioning, which affects the temperature, water transport, and plant growth (Burba et al., 1999). Model investigations have found that irrigation decreases near-surface air temperatures, and increases relative humidity (Sacks et al. 2009). Enhanced soil moisture due to irrigation leads to greater evapotranspiration and a resulting cooling of the land surface through the repartitioning between sensible heat fluxes and latent heat fluxes (Sacks et al. 2009, Puma et al. 2010). The radiation energy above plant canopies is mainly consumed by latent and sensible heat fluxes, therefore the relative proportion of energy consumed by the evaporation process has an influence on how much radiation is available to warm the canopy and soil. Thus, it is important to know how the practice of irrigation changes the surface energy fluxes, and the crop and soil environment, in ways that can influence greenhouse gas emissions from soil. It is not presently understood how significant of an enhancement is caused by a single irrigation event, nor how long this effect may last. For instance, Tolk et al. (1995) found sprinkler irrigation resulted in a short term reduction of crop transpiration, by more than 50%, during the irrigation process, as compared to a non-irrigated field. The purpose of this research is to investigate the energy variation mechanisms due to irrigation and their resulting influence on crop microclimate and soil environment, and any associated enhancements to GHG emissions.

## 3.3 METHODOLOGY

Two adjacent agricultural lands were selected, where one was irrigated, and the other was non-irrigated. All required measurement were performed in each field to get a clear idea how irrigation changes the energy partitioning, crop microclimate, soil environment, and GHG emissions.

### 3.3.1 Experimental site

The test site was located approximately 70 km southwest (51.65N, 106.95W, elevation: 481.5. m.a.s.l.) of Saskatoon. The 30 years mean annual temperature is 3.8°C and precipitation is 348.6 mm; whereas the summer time (May to August) mean temperature is 16.13°C and precipitation is 205.4 mm (Source: Environment Canada) for this region. Field investigations were conducted from June 2012 to October 2013 on two different crops, wheat in 2012 and canola in 2013. The non-irrigated field was same for the year 2012 and 2013, but the irrigated field differed between years 2012 and 2013 (Fig. 3.1) due to crop rotation difference between irrigated and non-irrigated dryland production systems. All fields were nominally the size of ¼ section, however the actual seeded area varied between 45 and 58 ha.  In every field, instruments were installed along a 125 m transect, parallel to the direction of crop rows (green box in Fig. 3.1), to reduce the disturbance related to normal cropping operations. Due to the close location of these fields, all soil physicochemical properties were similar (Table 3.1). All three fields were managed by the same owner from 2006 to 2013; therefore, all fields were treated in a very similar manner. The specific management activities are detailed in Table 3.2.

In 2012, larger-than-normal seasonal rainfall occurred (Table 3.2); hence, the amount of irrigation applied was less than a typical amount. During the summer study period (May to

August in 2012 and 2013), the cumulative rainfall recorded was 317 mm in 2012, and 213 mm in 2013. The 2013 year can be considered to be a more normal year in terms of the relative amounts of rainfall received and irrigation applied.

Table 3.1. Physical properties of soils

| Soil feature | Year | Dry land | Irrigated field (2012) | Irrigated field (2013) |
|---|---|---|---|---|
| pH | 2011 | 8.26 | 7.65 | 8.32 |
| | 2012 | 7.05 | 7.59 | 7.22 |
| | 2013 | 8.07 | -- | 8.32 |
| EC (µS $cm^{-1}$) | 2011 | 529 | 533 | 409 |
| | 2012 | 270 | 287 | 664 |
| | 2013 | 370 | -- | 333 |
| Bulk density (g $cm^{-3}$) | | 1.17 | 1.18 | 1.17 |
| Porosity (%) | | 56 | 56 | 56 |
| Soil texture | | | Loam | |

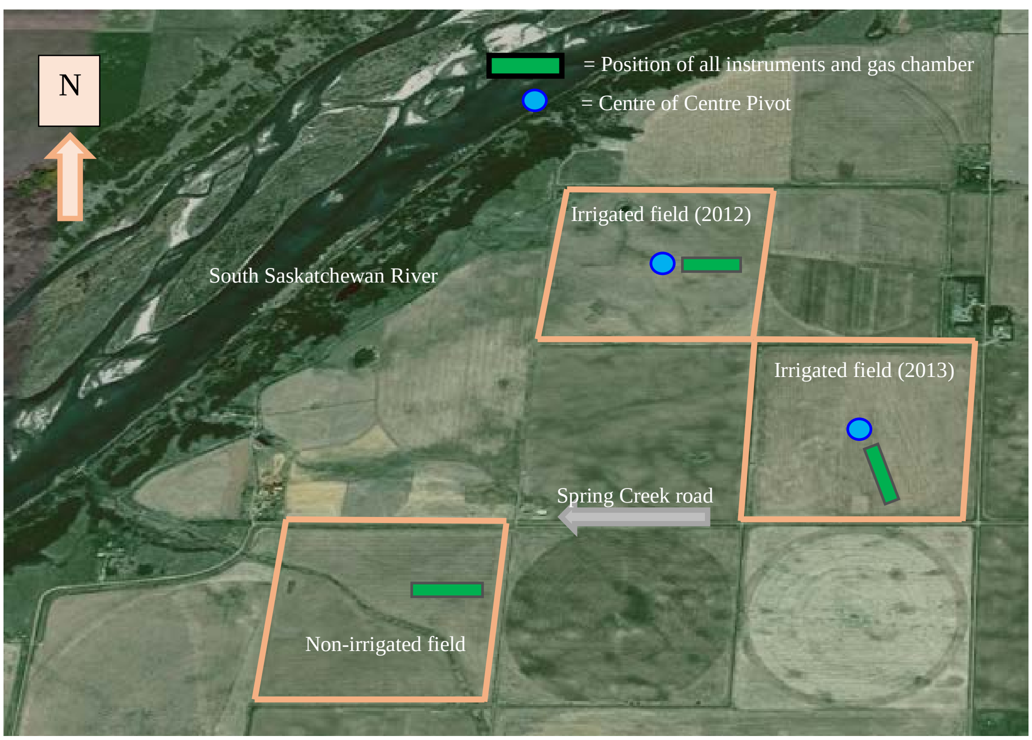


Fig. 3.1. Experimental site for field experiment (51.65N, 106.95W)

Table 3.2. Farm management for crop production during experiments (DL= non-irrigated field, IL= irrigated field)

| | 2012 | | 2013 | |
|---|---|---|---|---|
| Land type | DL | IL | DL | IL |
| Crop type | Wheat (*Triticum spp.*) | | Canola (*Brassica napus*) | |
| Variety | AC Barrie | AC Carberry | InVigor L130 | |
| Date of seeding | May 17 | May 15 | May 16 | May 15 |
| Date of swathing | Aug 29 | Aug 30 | Aug 12 | Aug 22 |
| Fertilizer (Fall, kg N $ha^{-1}$) | 67 | 100 | 78 | 138 |
| Fertilizer (Spring, kg N $ha^{-1}$) | 6 | 10 | 11 | 6 |
| Total rainfall (mm) | 321 | | 178 | |
| Total irrigation (mm) | 51 | | 127 | |

### 3.3.2 Measurements of surface energy fluxes

It was necessary to measure net radiation, latent heat flux, sensible heat flux, ground heat flux, and ground heat storage to get a clear understanding of surface energy fluxes.

As the fields are adjacent, it was assumed that the incoming radiation in both fields is the same. Hence, the net radiation (Rn) was computed using a Hukseflux four-component (Model NR01, HukselfluxUSA Inc., USA), and Hukseflux two-component net radiometer (Model RA01, HukselfluxUSA Inc., USA) placed 2 m above the ground level on the non-irrigated and irrigated fields, respectively. The four-component net radiometer recorded the incoming shortwave and longwave radiation as well as reflected shortwave and emitted longwave radiation. Moreover, two-component net radiometer records only reflected shortwave and emitted longwave radiation. From these all incoming and outgoing radiation, net radiation was determined by using the following two equations:

$$R_{\mathrm{n}} = (S_{\mathrm{in}} - S_{\mathrm{out}}) + (L_{\mathrm{in}} - L_{\mathrm{out}}) \tag{3.1}$$

$$R_{\mathrm{n}} = S_{\mathrm{net}} + L_{\mathrm{net}} \tag{3.2}$$

Where, $R_n$ is net radiation, $S_{in}$ is the incoming shortwave solar radiation, $S_{out}$ is the reflected shortwave solar radiation, $L_{in}$ is the atmospheric longwave radiation, $L_{out}$ is the surface longwave i.e. emitted longwave radiation, $S_{net}$ is net short wave radiation and $L_{net}$ is net longwave radiation.

The eddy covariance (EC) system was used to measure the turbulent fluxes of latent and sensible heat, which was placed at 3 m above the ground level. It is composed of a fast-response open-path infrared gas analyzer (LI-7500, Licor, Inc., Lincoln, NE, USA) to measure atmospheric $H_2O$ to calculate latent heat flux. It is coupled with a tridimensional sonic anemometer (CSAT-3, Campbell Scientific, Inc., Logan, UT, USA), which was used to measure wind speed. The digital signals from these instruments were sampled at 20 Hz.

EddyPro software was used to post-process the raw eddy covariance (EC) data to compute latent and sensible heat fluxes. Weekly measurements of crop height in each field was recorded throughout the growing season (May to August in 2012 and 2013) in order to account for the choosing aerodynamic roughness height.

The soil ground heat flux is the sum of the heat flux through a plate installed at a particular depth (8 cm) and soil heat storage in the layer above the plate. The soil heat flux ($G_{obs}$) was measured with two heat flux plates (HFP01, Campbell Scientific, Inc., Logan, UT, USA) buried in the ground at 8 cm depth. The values of $G_{obs}$ were obtained by averaging these two measurements. The soil heat storage was calculated by using soil temperature at 2, 4, and 8 cm depth at the same soil pit, where the heat flux plates were installed.

Fig. 3.2. Eddy covariance with all other instruments above soil ground

**3.3.3 Investigation of crop microclimatic modification**

To investigate the modification of crop microclimate after irrigation, air temperature, canopy temperature, soil surface temperature, and vapor pressure deficit (VPD) was recorded continuously throughout the study period. Air temperature and relative humidity was recorded by a Rotronics HC2_S3 temperature and relative humidity probe. This half hourly temperature and relative humidity data then processed by Eqs. 3.3 to 3.5 to calculate vapor pressure deficit.

$$VPD = e_s - e_a \tag{3.3}$$

$$e_s = 0.6108 \exp\left(\frac{17.27\,T}{T+237.3}\right) \tag{3.4}$$

$$e_a = \frac{RH}{100}\, e_s \tag{3.5}$$

Where, $e_s$ is saturated vapor pressure in kPa, $e_a$ is actual vapor pressure in kPa, and T is air temperature in ˚C.

Canopy and soil surface temperature was measured using Apogee infrared radiometers (Model SI-121; Apogee Instruments Inc, USA). The wind speed was also recorded by R.M. Young Wind Monitor (Model 05103; R.M. Young Company, USA) as well as by the sonic anemometer in eddy covariance system.

### 3.3.4 Monitoring soil water and temperature

Continuous measurements of soil water and soil temperature was performed in the field throughout the growing season to explore the effect of irrigation on soil environment. Soil volumetric water content and temperature were measured using CS650 Time Domain Reflectometer (TDR) probes at two different depths (10 cm and 25 cm). Each probe had two stainless steel rods (300 mm long × 3.2 mm diameter, and 32 mm spacing between the rods). The TDR was installed into the soil at four different locations. The distance of the measurement locations from the first chamber on each transect were 15, 45, 80 and 110 m.

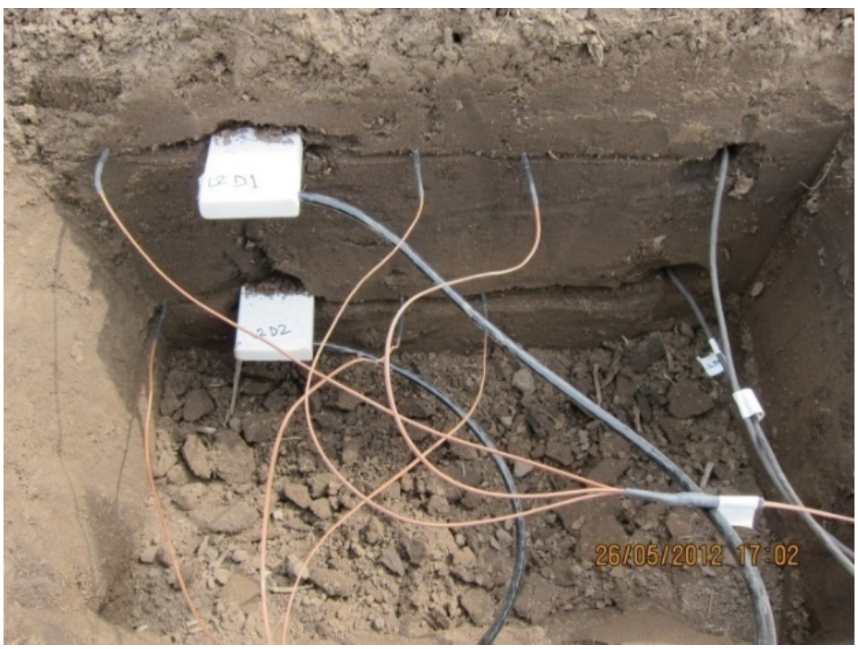


Fig. 3.3. Time domain reflectometer, heat dissipation probe, and thermocouples

### 3.3.5 Other measurement and data acquisition

Rainfall and irrigation amounts were recorded by a tipping bucket (TR-525; Texas Electronics Inc., USA) rain gauge at two different locations in each field. In 2013, standard rain gauges were also installed in both fields, and a Belfort rain gauge (Belfort 3000; Belfort Instrument, Baltimore MD) was installed in the non-irrigated field.

Campbell Scientific CR3000 micro loggers and AM16/32 relay multiplexers (Campbell Sci. Inc., Canada) were used as data loggers to collect and record all data from different sensors at 30-minute intervals.

### 3.3.6 Measurement of soil greenhouse gas emission

During the growing seasons, soil greenhouse gas emissions were measured using static acrylic chambers (22 × 45 × 10 cm). During sampling (twice a week), the chamber was sealed with a lid, and three series of gas samples were collected at fifteen, thirty, and forty-five-minute after closing the chamber. Ambient air gas samples, which were collected at the beginning and end of each sample collection, were used to determine reference values. After collection, the gas was analyzed by gas chromatography (Bruker 450 GC, Bruker Biosciences Corporation, USA) (Farrell and Elliott, 2007) to calculate the concentration of $N_2O$, $CO_2$, and $CH_4$. The raw greenhouse gas data were used to calculate fluxes using the Hutchinson and Mosier (1981) method to get the final emissions.

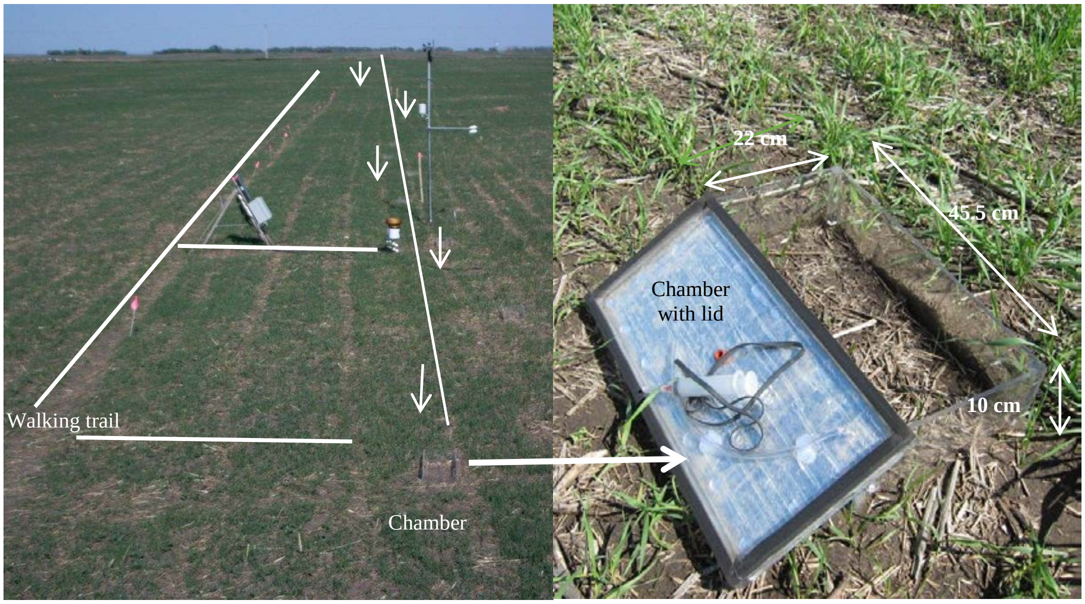


Fig. 3.4. Static gas chamber (Acrylic chamber)

## 3.4 RESULTS AND DISCUSSION

The purpose of this research was to determine the effect of irrigation on soil greenhouse gas emissions by influencing crop surface energy fluxes, crop microclimate, and soil environment. An above normal rainfall made the growing season wet in 2012 compared to 2013. As the crop water requirements for wheat (2012) and canola (2013) are similar (Govt. of Alberta, 2011), this section presents compiled results from both years together.

The measured data of a few consecutive days were combined using mean values to examine the effect of irrigation at the diurnal scale. This was compiled for three different example periods: (a. - Early S) early season of crop when there was not any irrigation or rainfall [June 21$^{st}$ to 30$^{th}$, 2013]; (b. - Mature Dry) mature stage of crop when background soil conditions were dry and irrigation was applied [July 22$^{nd}$ to 30$^{th}$, 2013], and (c.- Mature Wet) mature stage of crop when irrigation was not applied because background soil condition were wet due to the presence of rainfall [August 1st to 5th, 2012]. The effect of irrigation at daily timescales was also assessed, but in this case only the data from 2013 were considered.

### 3.4.1 Variation of surface energy fluxes

The hourly variation of energy fluxes during the example time periods are shown in Fig. 3.5. In the early season (Early S), when conditions were relatively uniform due to spring rain events, there was very little difference in any of the energy fluxes between the irrigated and non-irrigated systems. However, at the mature stage of the crop in a typical year (Mature Dry), while net radiation stayed the same in both fields, the latent and sensible heat flux changes noticeably due to the presence of irrigation. The difference in latent and sensible heat flux due to irrigation was more than 100 $Wm^{-2}$ during the day time when incoming short wave radiation was high. The reason for this is that in the irrigated field, most of the incoming radiation is used to evaporate water from soil and canopy surface. On the other hand, due to lack of moisture in the non-irrigated field, the incoming radiation mainly warms up the soil and canopy surface resulting in a high sensible heat flux in the non-irrigated field. During night time both fields had similar small fluxes. In 2012, when the field did not receive any irrigation because of a higher amount of rainfall (Mature Wet), very little difference was observed between the irrigated and non-irrigated fields, as the availability of water for evaporation was similar in both cases.

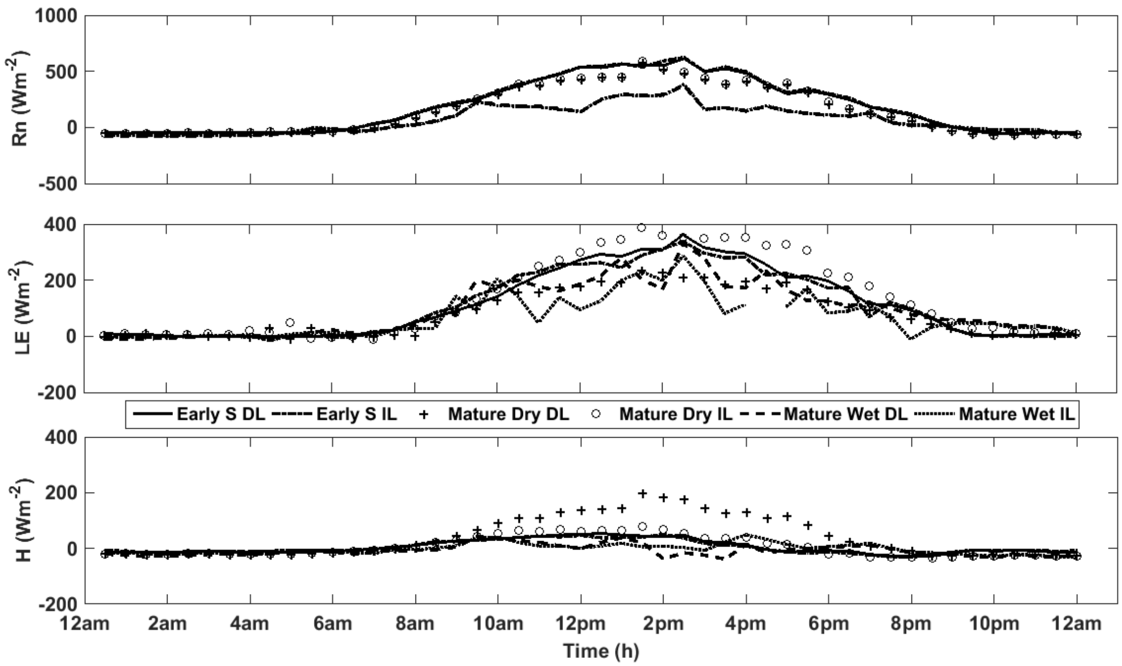


Fig. 3.5. Hourly mean surface energy fluxes (DL= Non-irrigated field, IL= Irrigated field, Rn= Net radiation, LE=Latent heat flux, H= Sensible heat flux)

In 2013, most of the irrigation was applied in July, so the latent and sensible heat fluxes from both fields were compared during this period to identify the effect of irrigation upon the energy fluxes at the daily timescale (Fig. 3.6). During and after irrigation, the irrigated field had higher latent heat fluxes, and the non-irrigated field had higher sensible heat fluxes. This trend varied slightly for the days near July 6th and 13th because both fields received rainfall and had similar fluxes. During July, the average difference of latent heat fluxes and sensible heat fluxes between irrigated and non-irrigated field was 41 W $m^{-2}$ and 13W $m^{-2}$, respectively. The highest difference in latent and sensible heat flux was observed 122 W $m^{-2}$ on 26th of July and 67 W $m^{-2}$ on 27th of July, respectively.

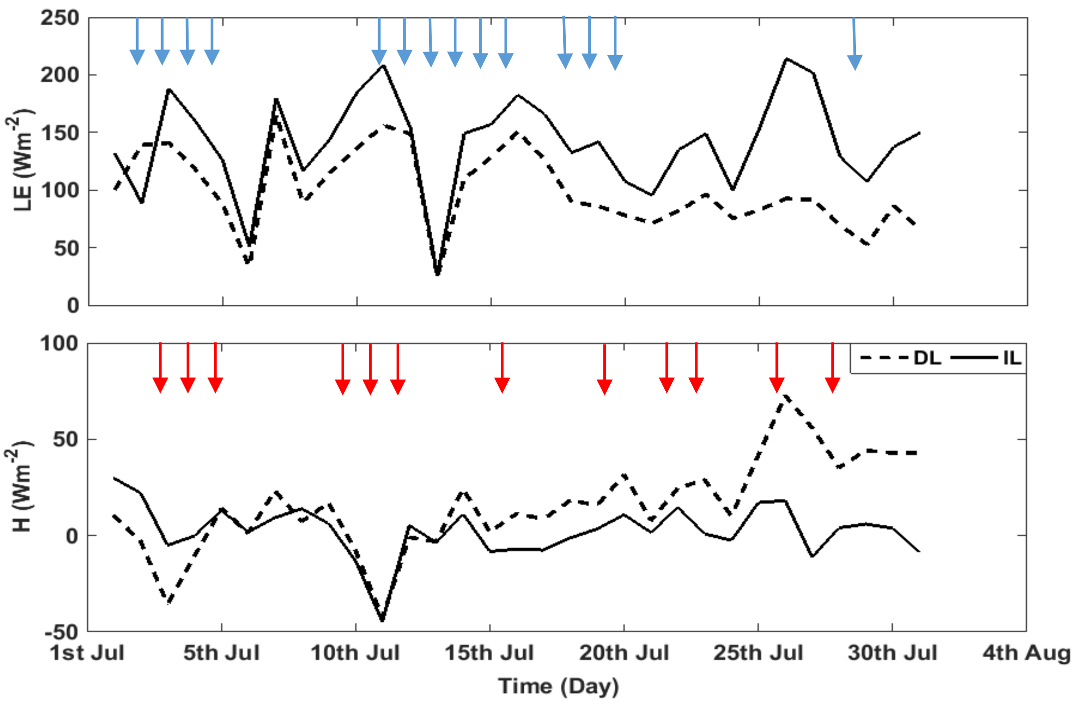


Fig. 3.6. Daily variation of latent and sensible heat flux in 2013 (DL= Non-irrigated field, IL= Irrigated field, LE=Latent heat flux, H=Sensible heat flux). Blue arrows indicate the timing of irrigation events. Red arrows indicate the timing of rainfalls

The change in sensible heat flux is driven by both a decrease in soil temperature due to the additional evaporative cooling accompanying irrigation and a shift in the energy partitioning so that more of the Rn is devoted to evaporation than to surface heating. The warming effect of sensible heat flux in non-irrigated field was different at the various times of the season. During the early season when leaf area was small, a good portion of net radiation can heat up the soil surface. However, during the late season when the leaf area almost fully covered the surface of the ground, a higher portion of radiation was used to warm the canopy. From these observations, it is found that irrigation creates a perceptible modification of the energy fluxes. This study showed that irrigation can enhance the surface energy fluxes at hourly and daily timescale. Similar variations in latent and sensible heat flux due to irrigation are reported in the literature. Kueppers and Snyder (2011) reported a variation in surface energy partitioning in their analysis of regional climate model simulations. They found that conversion of natural vegetation to irrigated agriculture can reduce net radiation by 10-30 W $m^{-2}$ and sensible heat flux by 100-350 W $m^{-2}$ and increase latent heat fluxes by 200-450 W $m^{-2}$ from May through September. Jiang et al., (2014) observed in a modelling study in a Northern China region, that conversion to irrigated agriculture leads to an increase in annual mean latent heat fluxes of 12 W $m^{-2}$, and a decrease in annual mean sensible heat fluxes of 99 W $m^{-2}$. In another model study in the U.S. Great Plain region, Huber et al., (2014) found that increased area of irrigation would increase mean latent heat flux by 89% and decrease mean sensible heat flux by 64%.

### 3.4.2 Change in crop microclimate

Figs. 3.7 and 3.8 demonstrate the hourly variation of crop microclimatic parameters (e.g. air temperature, canopy temperature, soil surface temperature, VPD, and ET) in both irrigated and non-irrigated fields. At the early stage of the crop (Early S), most of these parameters were similar in both fields except soil surface temperature, which varied due to the leaf density of crop and weed in the field. During the mature stage of the crop, provided that soil moisture conditions are comparable (Mature Wet), these parameters were similar in both fields. However, when there is a difference in moisture imposed by irrigation (Mature Dry), it noticeably influences these parameters. In the case of air temperatures the difference was relatively small, however, a large difference was observed on VPD and ET in both fields. It should be noted that these graphs have been produced by averaging the results of a few days; whereas, irrigation water was applied to the crop at different times of the day. Thus, due to the smoothing effect of averaging, the microclimatic variation immediately following irrigation will appear to be somewhat muted.

Figs. 3.9 and 3.10 illustrate the daily variation of crop microclimate due to irrigation during the month of July, 2013. These figs. show that irrigation does not influence all of the microclimatic parameters at the daily time scale. Irrigation reduced the canopy temperature and vapor pressure deficit which indicates that crop heat stress and transpiration rate will be lower in the irrigated field compared to a non-irrigated field. The observed variation of soil temperature was small between these two fields, suggesting that soil water was the governing factor for soil microbial activities and other bio-chemical reactions like nitrification, denitrification, and respiration. Although the daily average soil surface temperature in both fields was similar, there was a large difference in the hourly variation of soil surface temperature. At daily scale, the average difference of canopy temperature in July 2013 was 1.54°C with the highest difference of 3.63°C observed on 7$^{th}$ of July and the lowest difference was 0.02°C on 16$^{th}$ of July. Irrigation did not appear to alter air temperature at the daily scale, but it decreased air temperature just after irrigation at the hourly scale after each irrigation event in day times (data not shown). Irrigation decreased VPD during and after second irrigation application period (13$^{th}$ to 22$^{nd}$ of July) as irrigation increased latent heat flux in this field. For the same reason, the measured ET in irrigated field was higher than that in the non-irrigated field (Fig. 3.10).

Similar observations of crop microclimatic variation due to irrigation have been reported in the literature. Liu and Kang (2006) reported from their experiment on North China Plain that air temperature at 1m height was reduced by 1.8°C under sprinkler irrigation. Cavero et al., (2009) mentioned that daytime irrigation decreased air temperature by 3.3 to 4.4°C and VPD by

1.0 to 1.2 kPa at 0.5 m below the crop canopy height. Yenny et al. (2013) found that the average decreases during irrigation were 1.8-2.1˚C for air temperature, 0.53-0.61 kPa for VPD, 3.1-3.8˚C for canopy temperature. Similar to the present study, their research found that microclimatic changes were higher in drier and warmer days.

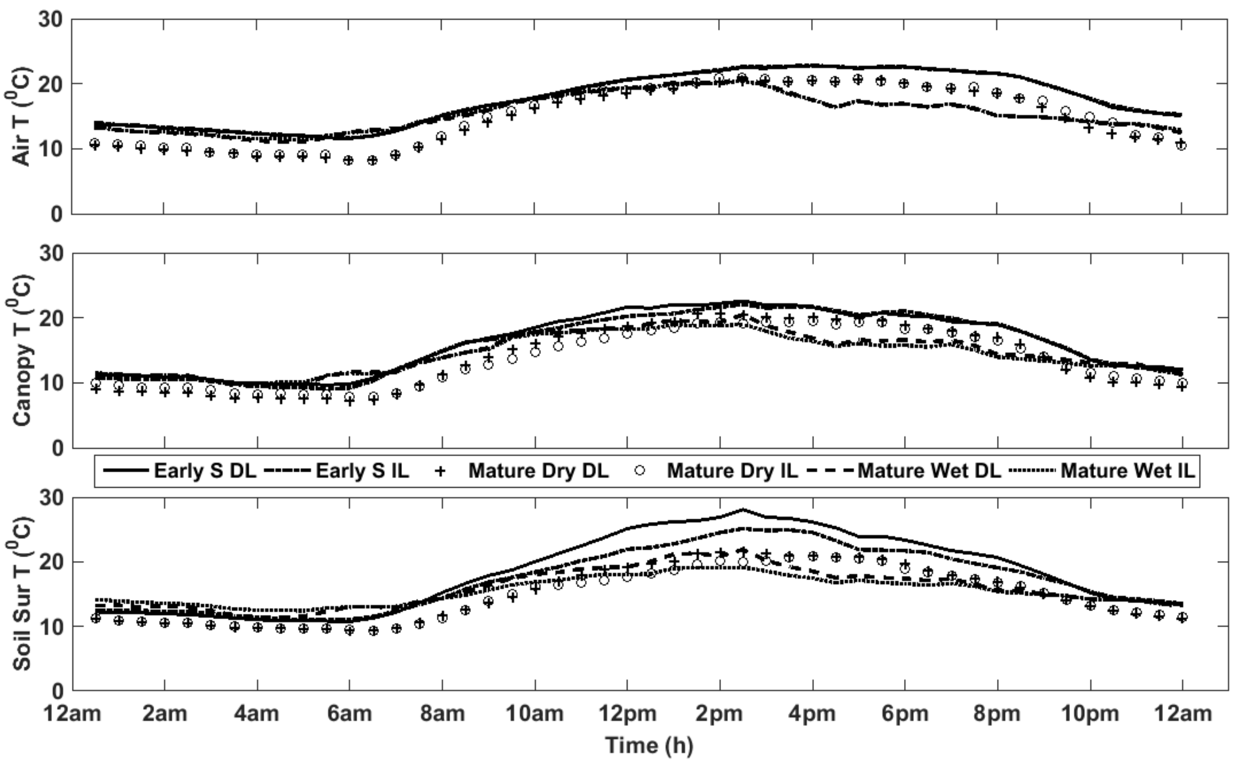


Fig. 3.7. Crop microclimatic parameter: all temperatures (DL= Non-irrigated field, IL= Irrigated field, T= Temperature, Sur=Surface)

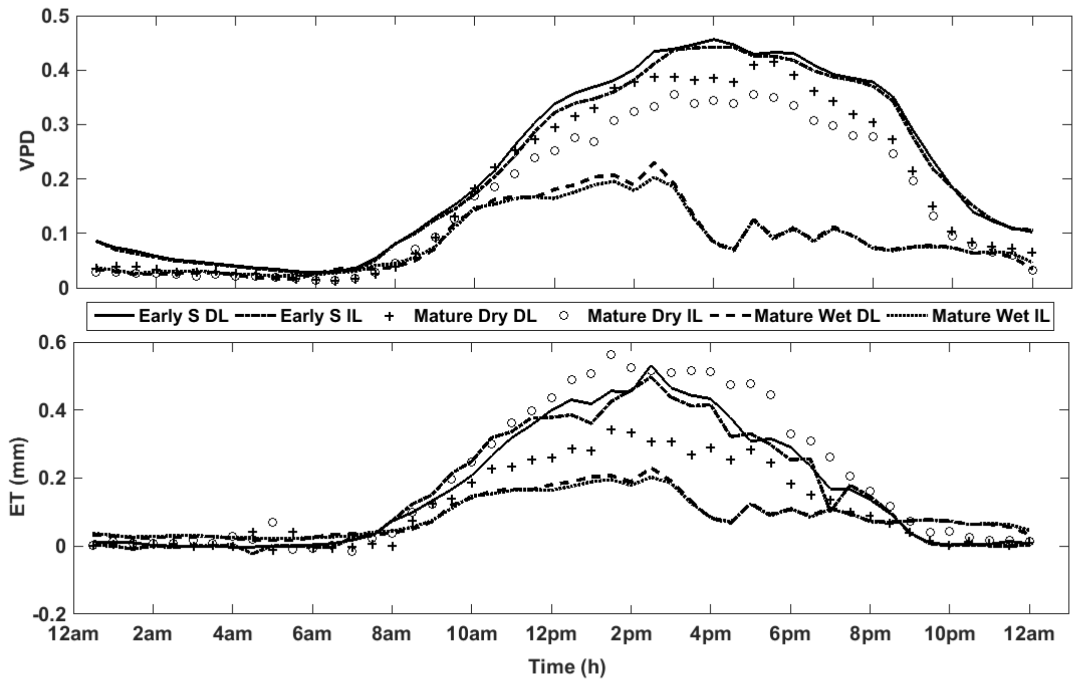


Fig. 3.8. Crop microclimatic parameter: VPD and ET (DL= Non-irrigated field, IL= Irrigated field, VPD=Vapor pressure deficit, ET=Evapotranspiration)

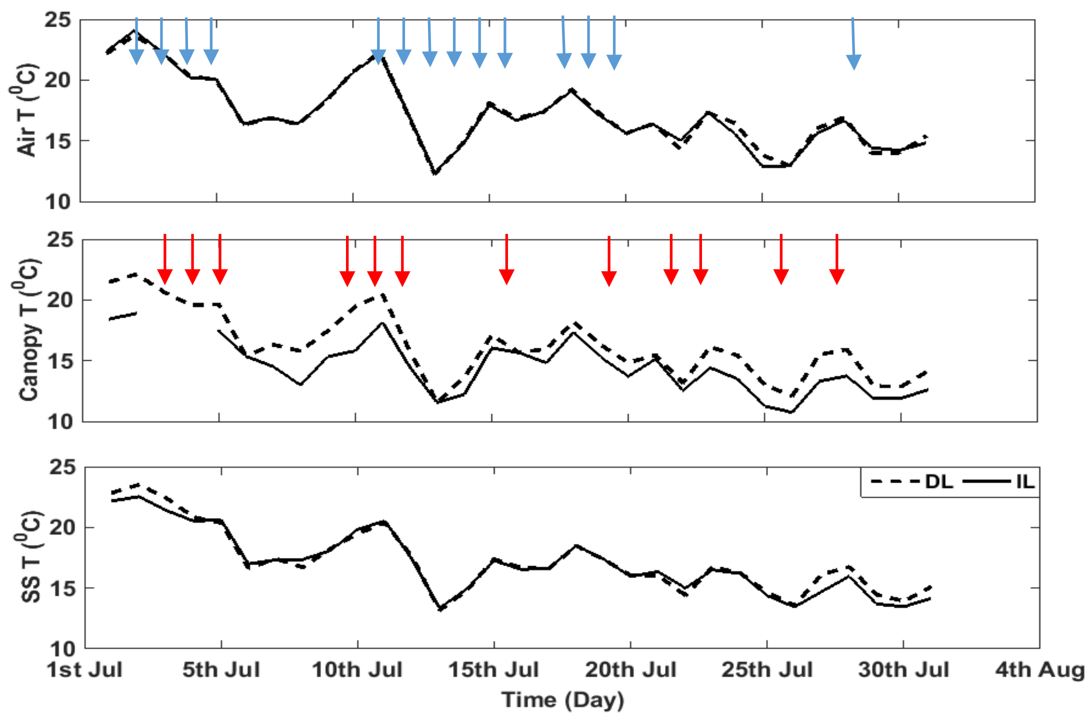


Fig. 3.9. Daily variation of air, canopy and soil surface temperature in 2013 (DL= Non-irrigated field, IL= Irrigated field, T=Temperature, SST=Soil surface temperature. Blue arrows indicate the timing of irrigation events. Red arrows indicate the timing of rainfalls)

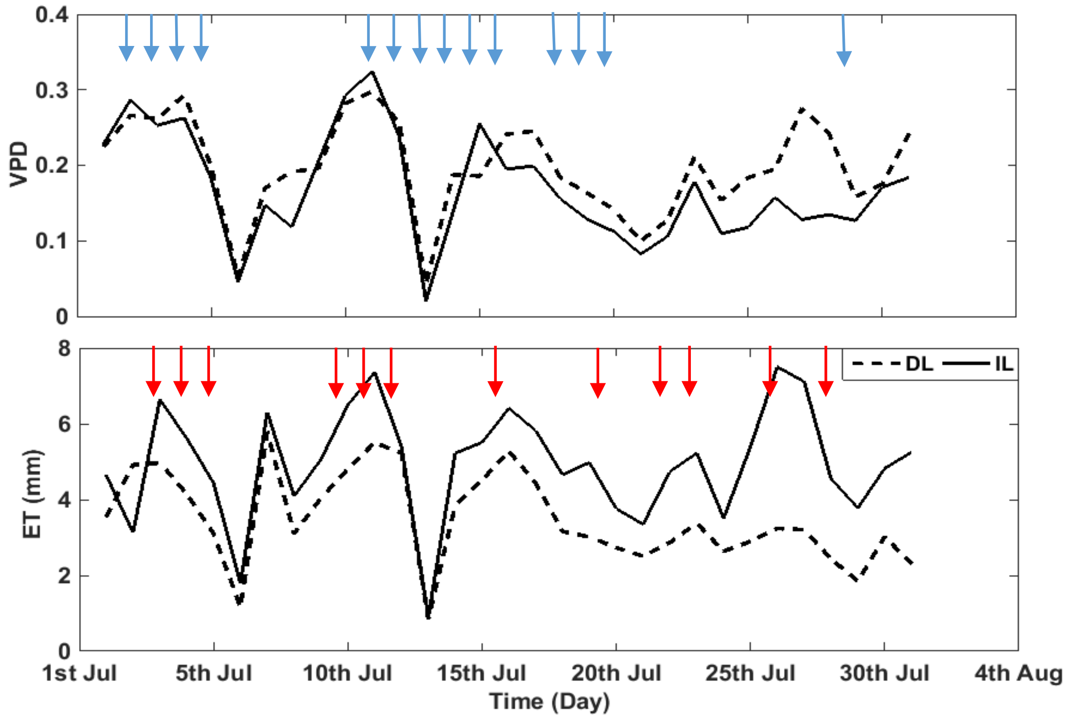


Fig. 3.10. Daily variation of vapor pressure deficit and actual evapotranspiration (ET) in 2013 (DL= Non-irrigated field, IL= Irrigated field, VPD= Vapor pressure deficit, ET= Evapotranspiration. Blue arrows indicate the timing of irrigation events. Red arrows indicate the timing of rainfalls)

### 3.4.3 Change in soil moisture

Figures 3.11 and 3.12 show the soil volumetric water content at 10 and 25 cm depth corresponding to rainfall and irrigation for the period when sensors were installed in the field. The shaded colored box represents the critical level of soil moisture for soil $N_2O$ emission, which is 60% water filled pore space (WFPS). In 2012, soil moisture at both 10 and 25 cm depths, in both fields, was similar until the first week of July. All observations were similar until the second irrigation, except for the soil moisture at 10 cm depth in the non-irrigated field. Due to irrigation, soil moisture was maintained above the critical level until the last irrigation on 14$^{th}$ of August. From the middle of July, soil moisture in the non-irrigated field was lower than this level. Moreover, at the end of the season, soil moisture in both fields was lower than the critical level. In 2013, both fields had similar early soil moisture which was above the critical level due to infiltrated snow melt water. Soil water decreased gradually in both fields until the first week of July. After that, the soil moisture in the irrigated field increased corresponding to irrigation and stayed above the zone of 60% WFPS until the second week of August. In the non-irrigated field, soil moisture rose corresponding to a big rainfall event on 6th July and gradually decreased afterward. After mid-season, when crop water requirements are high, rainfall alone was not able to increase soil moisture.

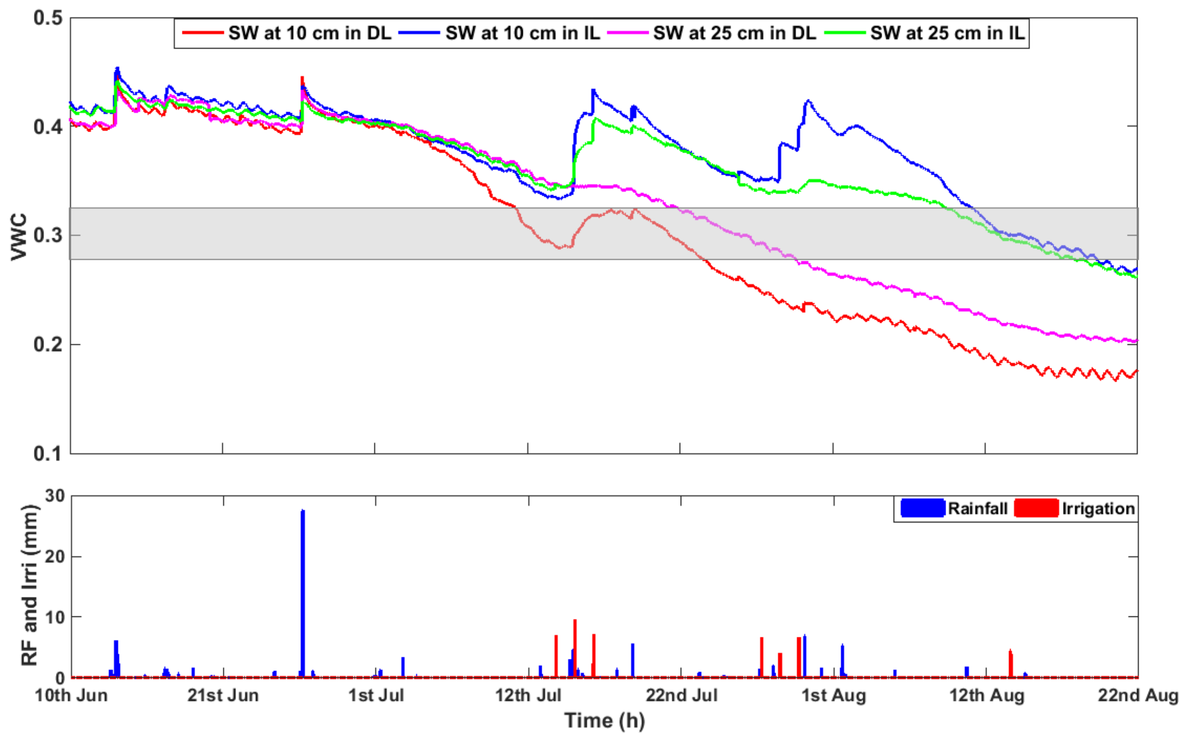


Fig. 3.11. Soil moisture in 2012, the shaded area representing the area of 60% WFPS (SW= Soil water, DL= Non-irrigated field, IL= Irrigated field, VWC=volumetric water content, RF= Rainfall, Irri= Irrigation, WFPS=Water filled pore space)

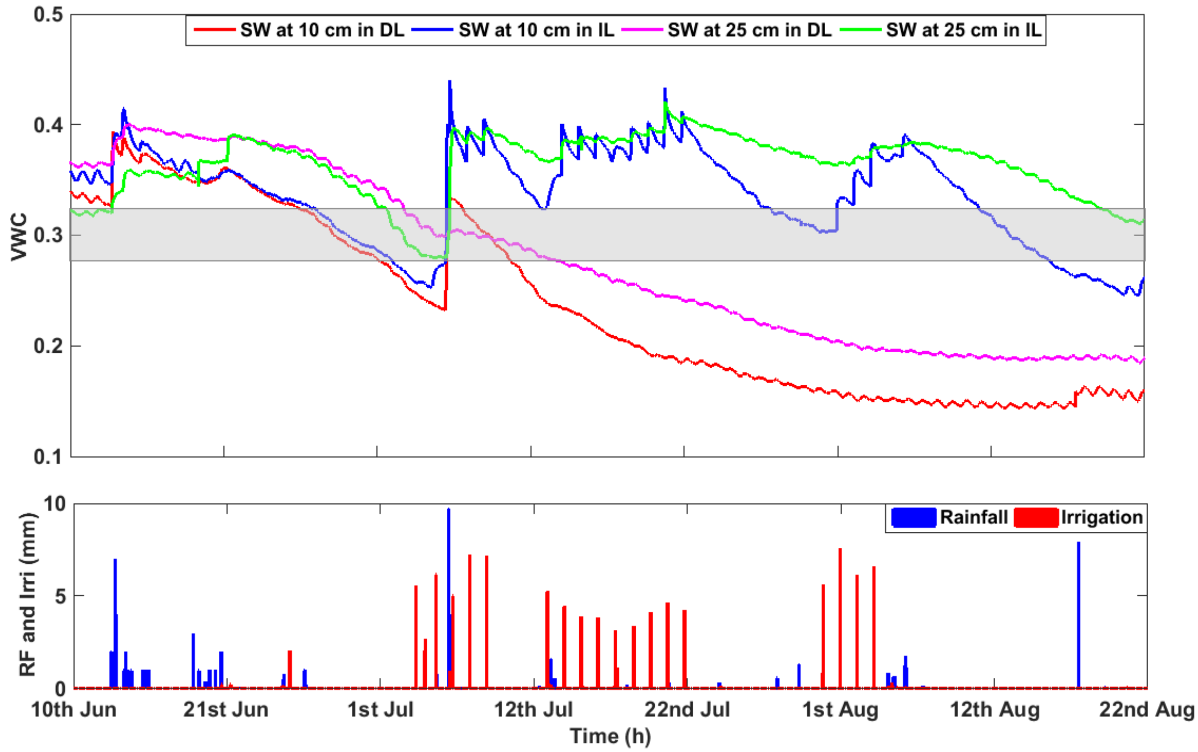


Fig. 3.12. Soil moisture in 2013, the shaded area representing the area of 60% WFPS (SW= Soil water, DL= Non-irrigated field, IL= Irrigated field, VWC=volumetric water content, RF= Rainfall, Irri= Irrigation, WFPS=Water filled pore space)

### 3.4.4 Variation of soil temperature and ground heat flux

In this section soil temperatures measured by T-type thermocouples are compared. Fig. 3.13 shows the daily variation of soil temperature between irrigated and non-irrigated fields during July 2013. The delta values were calculated by subtracting the temperatures of the irrigated field from those of the non-irrigated field. This Fig. clearly demonstrates that irrigation decreased soil temperatures, but the effect was very small. The average difference between the two fields for the month of July was 0.50˚C at 0-10 cm depth.

The lower panel of Fig. 3.13 shows the daily ground heat flux (GHF) difference between the irrigated and no-irrigated fields. Ground heat transfer was slightly higher in the non-irrigated field compared to the irrigated field. However, as the irrigated field had higher soil moisture, ground soil heat storage was higher in this field. In a modeling study Huber et al., (2014) found that irrigation changes the hourly-averaged GHF in July in a relatively small range (<1 W m$^{-2}$).

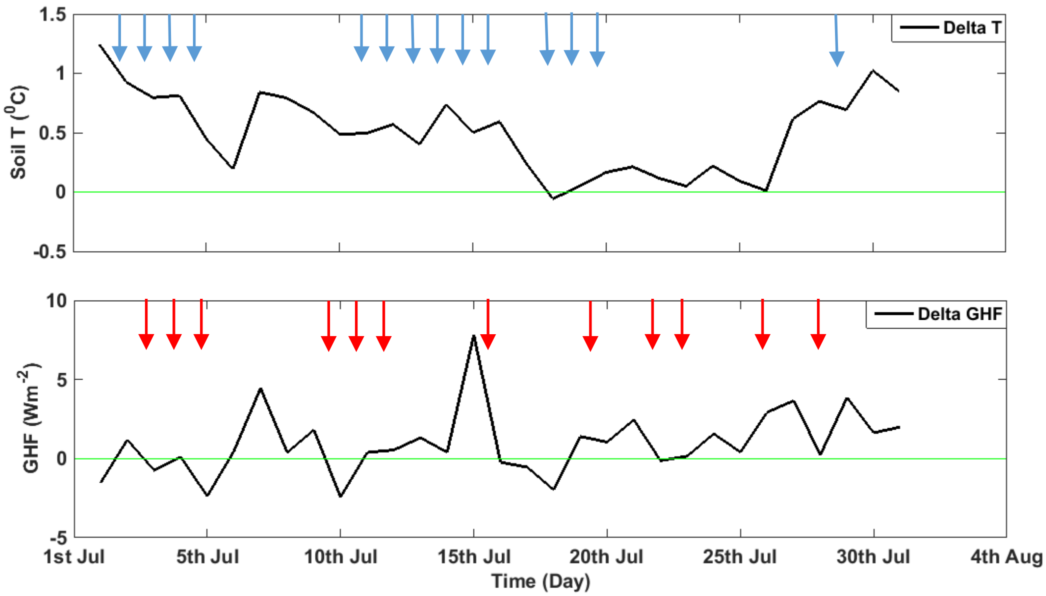


Fig. 3.13. Daily mean difference (non-irrigated- irrigated) of soil temperature (at 0-10 cm depths) and ground heat flux in 2013 (T=Temperature, GHF= Ground heat flux). Blue arrows indicate the timing of irrigation events. Red arrows indicate the timing of rainfalls.

### 3.4.5 Variation in greenhouse gas emission from soil surface

In 2012, soil moisture was high in both the irrigated and non-irrigated fields due to rainfall events on 15$^{th}$ and 25$^{th}$ of June and 16$^{th}$ of July. Similarly, irrigation events raised the moisture level of the irrigated field on 16$^{th}$ and 26$^{th}$ of July. However, in the case of GHG emission in 2012 (Fig. 3.14), the fluctuation of emissions of $N_2O$ and $CO_2$ was dissimilar to soil water fluctuations. On 15$^{th}$ June, only $N_2O$ emissions from the irrigated field increased with soil moisture. After June 15$^{th}$, $N_2O$ emissions from both the irrigated and non-irrigated fields were similar (Fig. 3.14) except for a few days when irrigated field had higher emission. In 2013, soil moisture was elevated in the irrigated and non-irrigated fields due to rainfall on 8$^{th}$

and 12th of June and 6th of July, and in the irrigated field due to irrigation on July 6th, and 13th and August 1st. Soil $N_2O$ emission from the irrigated field increased after the rainfall on 8th and 12th of June, after that, both fields had a similar trend and amount of emission (Fig. 3.15). Although there was a relatively large amount of irrigation applied in this year, the GHG emission did not increase in this field due to the irrigation.

In spring, soil GHG emission was higher in both fields due to snow melt water and fall fertilizer (David 2014). The details about seasonal cumulative GHG emission, as well as the fluctuation of emission along the transect location are explained in David (2014).

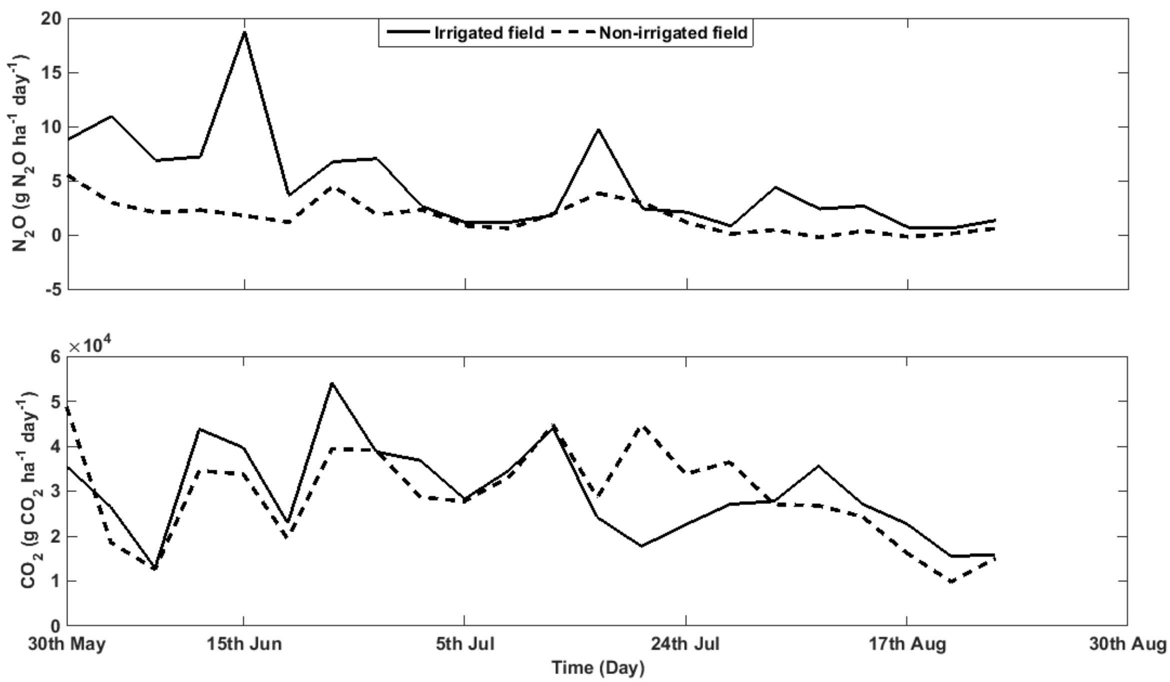


Fig. 3.14. $N_2O$ and $CO_2$ emission from irrigated and non-irrigated field in 2012

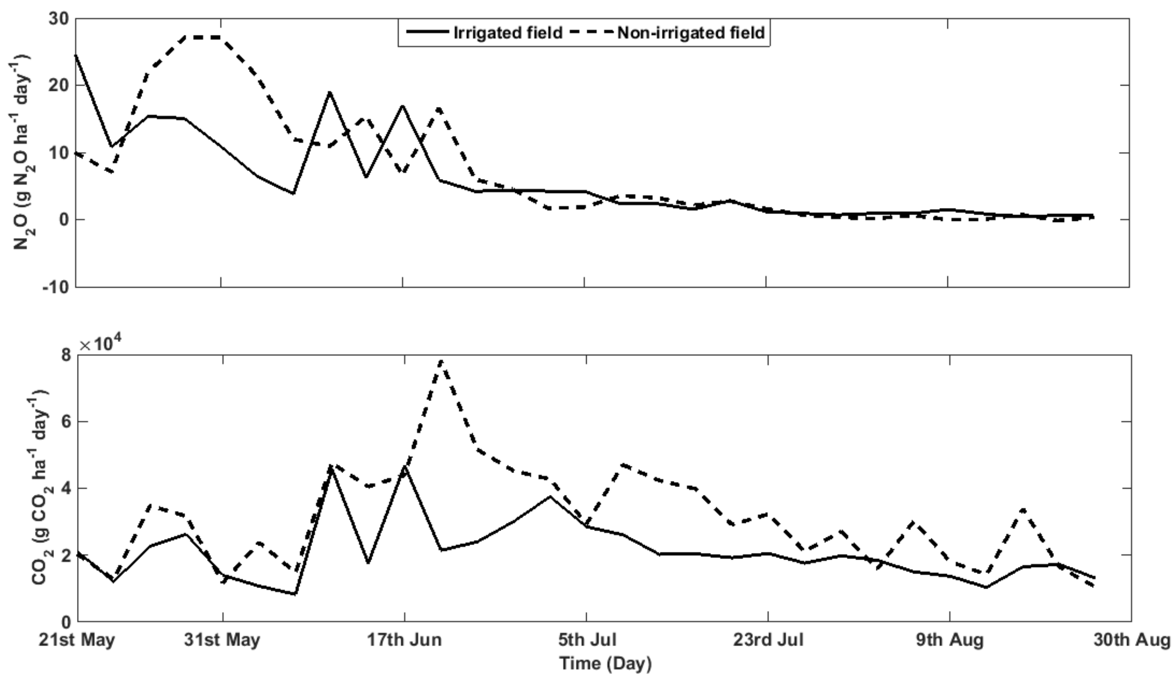


Fig. 3.15. $N_2O$ and $CO_2$ emission from irrigated and non-irrigated field in 2013

## 3.5 SUMMARY AND CONCLUSIONS

This study concludes that irrigation exerts a strong influence upon the variation in latent and sensible heat fluxes, as compared to a non-irrigated field. On a clear sky day, the difference of latent or sensible heat fluxes between irrigated and non-irrigated field can be as high as 100-200 W $m^{-2}$. In July in 2013, a monthly mean difference of latent and sensible heat flux between irrigated and the non-irrigated field was 41 and 13 W $m^{-2}$, respectively. It demonstrates how the availability of water in an agricultural field, as controlled through irrigation, could control the surface energy partitioning. Consequently, latent heat flux in the irrigated field increased the relative humidity and decreased the crop heat stress, as compared to the non-irrigated field. In the non-irrigated field, a greater proportion of the net radiation was available to exert a warming effect upon the canopy and soil. During the typical month of July 2013, mean monthly canopy and soil surface temperature was 1.5 and 0.5$^{\circ}$C higher in the non-irrigated field, respectively. Also, VPD was decreased in irrigated field after irrigation and ET was increased. On average, ET from the irrigated field was 1.54 mm higher in July 2013 compared to the non-irrigated field.

This research indicates that irrigation exerts a much stronger influence on soil moisture conditions than those of soil temperature. Despite the changes in energy partitioning caused by irrigation, soil temperatures were only modified slightly. Conversely, irrigation maintained soil moisture levels at or above 60% WFPS for most of the growing season, thereby providing more favorable conditions for denitrification.

Due to snow melt water and early season rainfall events, irrigation was not required early in the season. Therefore, GHG emissions during this period were not likely enhanced by irrigation practices, except via differences in fertilization amounts. Later in the season, when irrigation is typically applied, GHG emission from both irrigated and non-irrigated fields were low. The mechanisms of emission were not the focus of this research, but the low late season emissions, where irrigation practices were most likely to have an influence, may have been due to low available nitrogen levels.

In this study, soil environmental parameters and energy fluxes were monitored continuously, however, GHG fluxes were only sampled periodically. Thus, an opportunity for future research may be to employ an automated GHG soil emission measurement system to better understand the influences of irrigation at shorter timescales.

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# 4. EVALUATING $N_2O$ EMISSIONS FROM IRRIGATED AGRICULTURAL FIELDS IN WESTERN CANADA WITH CDN-DNDC MODEL

### 4.1.1 PREFACE

The high number and large size of farms in the Canadian Prairies makes it an important region for agriculture in Canada (Statistics Canada, 2014). In 2014, Canadian agriculture was responsible for 8% of the total national greenhouse gas (GHG) emissions and 70% of the national nitrous oxide ($N_2O$) emission (NIR, 2016). Hence, there is a need to control GHG emission, particularly $N_2O$ emission, from Prairie agriculture. As soil water exerts a dominant control upon $N_2O$ emissions from nitrogen (N) rich agricultural fields, proper water management is essential to mitigate $N_2O$ emission. To develop new irrigation-related mitigation strategies and to quantify the influence of irrigation on GHG emission, a GHG simulation model can play a vital role where field experiments are logistically and financially burdonsome. The GHG emission simulation model Denitrification-Decomposition (DNDC), has not been applied to examine the influence of irrigation on soil $N_2O$ emission and crop production. The goal of this study was to assess how a regional version of the DNDC model (CDN-DNDC) could predict $N_2O$ emission under irrigation condition. The objective was addressed through a two-year continuous field experiment to validate the CDN-DNDC model followed by sensitivity test and long-term scenario development.

### 4.1.2 ABSTRACT

Denitrification-Decomposition (DNDC) is a well-known robust process-based model for simulating $N_2O$ emissions from agricultural soils. This model has been extensively used to explore $N_2O$ emissions under various fertilizer and tillage practices; however, it has not been used to explore the effect of irrigation in the Canadian Prairie. As soil moisture is one of the most important driving factors of $N_2O$ emissions, the regional version of the model, the CDN-DNDC model is used to investigate the effect of irrigation, to identify the opportunity of the irrigation management practices as a viable GHG mitigation technique. To validate the model for local conditions a field experiment was conducted by instrumenting two adjacent irrigated and non-irrigated fields located near Saskatoon, Saskatchewan, Canada during the 2012 and 2013 growing seasons. Soil GHG emissions were manually sampled semiweekly using static vented chambers, and were complemented by automated measurements of soil moisture, soil temperature, and local meteorological variables. The model was validated by comparing the simulated soil moisture, soil temperature, and $N_2O$ emissions with field observations, confirming that the model is suitable to use under local conditions. This study found that, with the exception of its inferior ability to simulate the soil water and $N_2O$ emissions during the spring thaw period, the model is generally suitable for use in this region. A parameter sensitivity test identified the clay fraction as more sensitive than both the soil water holding characteristics and the soil hydraulic conductivity. The findings of a long-term (11-yr) simulation found that irrigation increased the total $N_2O$ emissions over the study period; however, once the increased crop yields are taken into consideration, the emission intensity is actually lower in the irrigated field as compared to non-irrigated field.

**Keywords:** soil water, soil temperature, $N_2O$ emission , CDN-DNDC, irrigation.

## 4.2 INTRODUCTION

One of the main sources of greenhouse gas (GHG) emissions at the global scale is agricultural production, accounting for 15-30% of the total anthropogenic emission, (IPCC 2007; Tubiello et al. 2013). Among all trace GHGs ($CO_2$, $N_2O$, and $CH_4$), $N_2O$ is particularly noted for its high global warming potential (GWP), which is 298 times more than that of carbon dioxide (IPCC, 2007). Hence, GHG mitigation strategies should focus on reduction of $N_2O$ emissions. Agricultural soil emission accounts for approximately 46-52% of the global anthropogenic $N_2O$ flux (Mosier et al., 1998; Olivier et al., 1998; Kroeze et al., 1999). The biological processes of nitrification and denitrification are the primary causes of $N_2O$ emissions from the soil, with denitrification playing the dominant role (Conrad 1996, Bockman and Olfs 1998, Stevens and Laughlin 1998). These processes are influenced by nitrogen fertilizer, soil moisture, temperature, crop type, soil organic carbon (SOC) content, soil pH, tillage and soil texture (Dobbie et al., 1999; Stehfest and Bouwman, 2006; IPCC, 2007; Metay et al., 2007). Elevated soil moisture alleviates crop water stress but also enhances microbial activities, which in turn influence supplied mineral N, crop N uptake, and the abiotic soil conditions that control $N_2O$ emissions from soils; however, knowledge of how soil moisture can be managed to reduce emissions is scarce.

The most common mitigation option for reducing $N_2O$ emission is improved fertilizer management (controlling the amount of fertilizer and timing of application) followed by improved tillage practices (IPCC 2001, Duke 2006). Although fertilizer application has been considered the primary cause of $N_2O$ emissions (Mosier 1994), the leading driver of the emission in N-rich soil is soil moisture (Weitz et al. 2001). During the growing season, high magnitude pulses of $N_2O$ emission are commonly observed after precipitation events (Lemke, 2007). Major N losses often occur during the first week after applying N fertilizer, and additional N losses continue over the following three weeks (Inselbacher et al. 2011). For spring fertilizer application, higher emissions due to irrigation or precipitation can be observed for thirty days following fertilizer application (Wei et al. 2010). Hence, in order to reduce $N_2O$ emission from spring fertilizer application; proper irrigation management can play a vital role to control soil moisture and emission flux.

In order to reduce $N_2O$ emissions, it is critical to manage the water content of the 0-10 cm depth of soil. As soil moisture increases; more available $NH_4^+$ is converted to $NO_3^-$ (an important form of nitrogen for plant uptake) through the nitrification process. This conversion is rapid when water filled pore space (WFPS) is 50-60% (Stevens et al., 1997). When soil moisture increases further, $NO_3^-$ can be converted into $N_2O$, NO or $N_2$ flux through the denitrification process, or $NO_3^-$ losses can occur through leaching. However, N loss can be minimized by controlling soil moisture through proper irrigation management. Varying the frequency and volume of irrigation applications can have a notable effect upon $N_2O$ emissions. Scheer et al., (2014) tested the DayCent model for simulating $N_2O$ emission from different irrigation treatments in a fertilized agricultural field (cotton-wheat rotation) in Australia. They observed that more frequently applied irrigation, of optimal volume, potentially reduced the $N_2O$ intensity. Goescherl (2013) applied full and deficit irrigation to a manure-amended corn field in Nebraska, and reported that different irrigation levels do not significantly alter $N_2O$ emissions on a daily basis; however, in the case of cumulative emission, full irrigation emits a greater amount than deficit irrigation. However, the effect of the irrigation upon soil moisture, and specifically how irrigation management can be used as a GHG mitigating tool, has not yet been examined.

To investigate irrigation management as a mitigation tool to reduce $N_2O$ emissions, process oriented GHG simulation models may be useful because these models simulate $N_2O$ emissions by considering soil biophysical processes like nitrification and denitrification as well as irrigation. There are various process-oriented GHG simulation models for North American agricultural fields, such as DeNitrification-DeComposition (DNDC), and Daily Century (DayCent). Parton et al. (1996 and 2001) and Del Grosso et al. (2000) investigated the accuracy of the DayCent model in a few regions in the USA. They found that this model accurately simulates annual mean trend of $N_2O$ emission but observed some dissimilarity between the daily measured and modeled emission. After development by Li et al. in 1992, the DNDC model has extensively been improved to investigate the effect of various treatments like fertilizer (Smith et al., 2002; Li Hu et al., 2012; Kröbel et al., 2011) and tillage (Smith et al., 2008). With some site-specific modification, this model also has widely been used in many countries for both site and regional $N_2O$ emission from agricultural fields (Li 1995; Li et al. 1996; Zhang et al. 2002; Xu-ri et al. 2003; Smith et al. 2004; Pathak et al. 2006). However, the DNDC model has not been used to evaluate the $N_2O$ emission from irrigated agricultural conditions.

The purpose of this research is to investigate $N_2O$ emissions simulated by the DNDC model under irrigated and non-irrigated conditions in the Canadian Prairies. The results of this research demonstrate how the DNDC model can be used to estimate $N_2O$ emissions from irrigated and non-irrigated agricultural crops.  The study also identifies areas where the model needs further improvements for application in the water management sector of Prairies agriculture.

## 4.3 METHODOLOGY

The first step in this research was to validate the DNDC model under local conditions using data from a two-year (2012 and 2013) field experiment in Saskatchewan, Canada. The required driving data and input parameters were collected from the experimental site. After validation, the sensitivity of the model to certain soil environmental parameters was examined.

### 4.3.1 Experimental site and field data collection

The field experiment was conducted in adjacently-located irrigated and non-irrigated fields, which were cropped to wheat in 2012 and canola during 2013. The test site is located approximately 70 km southwest (51.65N, 106.95W, elevation: 481.5. m.a.s.l.) of Saskatoon. The study area has a 30 years mean annual temperature 3.8°C and annual precipitation of 348.6 mm. During the growing season (May to August) mean air temperature is 16.1°C and precipitation is 205 mm (Source: Environment Canada). The selected non-irrigated field (DL) had a wheat-canola crop rotation for the year 2012 and 2013; however, the crop rotation in the irrigated field was wheat-dry bean-canola. In order to match the examined crops between irrigated and non-irrigated fields, different irrigated fields were used in 2012 (IL12) and 2013 (IL13) (Fig. 4.1). All required instruments (Table 4.1) were installed in the direction of crop rows (green box in Fig. 4.1) in order to reduce the disturbance related to normal cropping operations. All instruments and gas chambers were placed along a 125m transect located approximately at the middle of the center and side of the fields. In the irrigated field, all chambers and instruments received irrigation water at the same time as all instruments were set along the direction of the center pivot sprinkler system. Due to the close location of these fields, all soil physicochemical properties were similar (Appendix A.1).

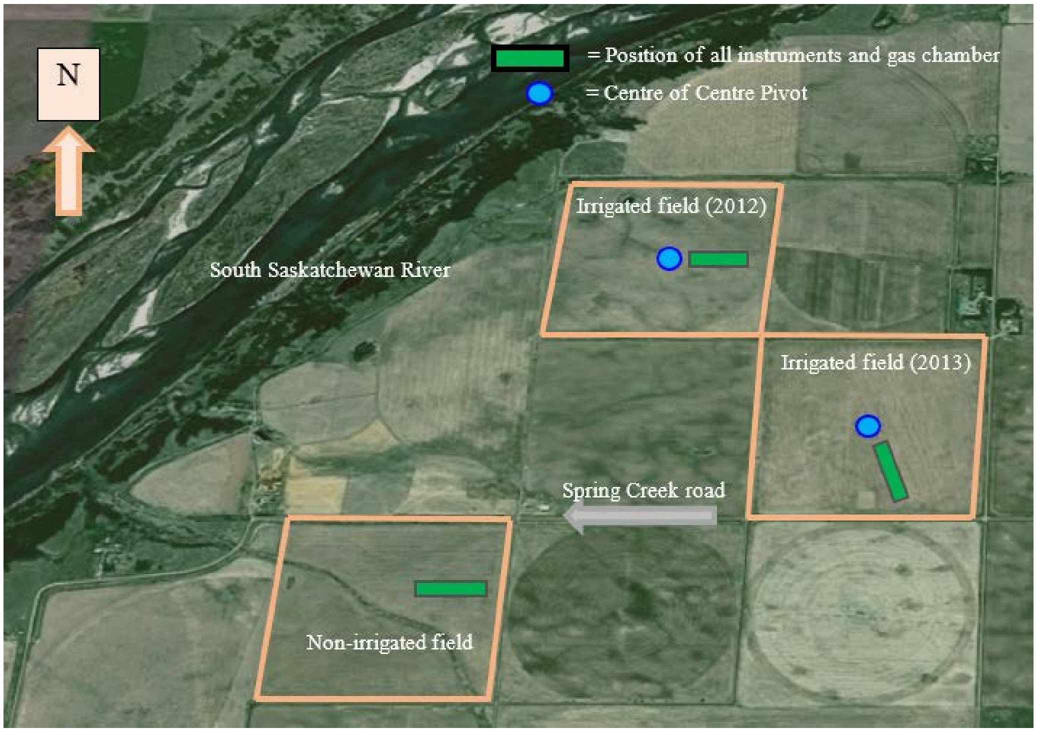



Fig. 4.1. Experimental site, near Outlook, SK and beside South Saskatchewan River (51.65N, 106.95W)

Table 4.1. Instruments used in the field for continuous in situ measurements

| Instruments | Variables measured by the instrument |
|---|---|
| Hukseflux NR01 four-component radiometer Hukseflux RA01 two-component radiometer | Incoming and outgoing radiation |
| R.M. Young 05103 Wind Monitor | Wind speed and direction |
| Rotronics HC2S3 temperature and relative humidity probe | Air T and RH |
| Texas Electronics TE525 tipping bucket rain gauge | Rainfall and Irrigation |
| Campbell Scientific CS650 Time Domain Reflectometer (TDR) | Soil VWC and T |
| Campbell Scientific CS229 heat dissipation probe | Soil water matric potential |
| T-type thermocouples (home-built) | Soil T |

#### 4.3.1.1 Soil moisture and temperature monitoring

Continuous measurements of volumetric soil water and temperature at 10 cm depth were recorded at four different locations in each field using CS650 Time Domain Reflectometer (TDR) probes. In order to convert volumetric water content to water filled pore space (WFPS), the soil bulk density of each field was measured by the core sampling method. The WFPS at field capacity (FC) and permanent wilting point (PWP) was determined from in situ soil-water-retention-curve, which was developed by using the recorded VWC and matric potential data.

#### 4.3.1.2 Direct measurement of $N_2O$ emission from fields

In the field, $N_2O$ emissions were sampled using static acrylic chambers (22 × 45 × 10 cm). Along each measurement transect, twenty chambers were installed in the direction of seeding at 6.25 m spacing. After installing the chambers in the field just after seeding, all plants from inside the chambers were removed, and the disturbed plants surrounding each chamber were replanted. During semi-weekly sampling, the chamber was sealed using a lid and rubber gasket. The first sample was collected at fifteen minutes after closing the chamber, and the second and third samples were collected at thirty and forty-five minutes, respectively. An additional eight ambient air gas samples were collected from the outside of chamber just before and after sample collection to determine reference values. The samples were collected using a 20 mL syringe with a 20 ga needle, which was evacuated into pre-vacuumed tubes containing desiccants to absorb any moisture in the sample for storage and transport from the field to the laboratory. In the laboratory, the gas samples were analyzed by gas chromatography (Bruker 450 GC, Bruker Biosciences Corporation, USA) (Farrell and Elliott, 2007) to calculate the concentration of $N_2O$, $CO_2$, and $CH_4$. From these analyzed raw data, the daily $N_2O$ fluxes were determined by the Hutchinson and Mosier (1981) method. These daily emissions were used to evaluate the model for the study site.

#### 4.3.1.3 Meteorological data collection

The DNDC model requires an input climate file containing daily values of the incoming solar radiation, maximum and minimum air temperature, relative humidity, precipitation, and wind speed for each day of the year. At the field site, all of these were measured and recorded on an half-hourly basis and were subsequently converted into daily format during post-processing. The required instruments (Table 4.1) for these input variables were installed in each irrigated and non-irrigated field (Fig. 4.2). During 2013, standard rain gauges were also included in both fields at four different locations, and a Belfort (Belfort 3000; Belfort Instrument, Baltimore MD) weighing type precipitation gauge was also used to ensure the accuracy of rainfall. CR3000 micro-loggers (Campbell Scientific Inc., Canada) were used to sample all instruments at 5 sec intervals and to store mean or summed values of all data at 30-minute intervals.

Fig. 4.2. Instruments above soil ground in field

### 4.3.2 The DNDC model

The DNDC model was initially developed by Li et al. in 1992 and was first used in the USA to simulate $N_2O$ emission from agricultural soil in 1995 (US EPA, 1995). The DNDC model consists of six different sub-models, namely: soil climate, plant growth, decomposition, denitrification, nitrification, and fermentation. The first three sub-models predict soil temperature, moisture, pH, redox potential (Eh), and substrate concentration ($NH_4^+$, $NO_3^-$, DOC (Dissolved organic carbon)). These stimulate the last three sub-models to predict emissions of carbon dioxide ($CO_2$), methane ($CH_4$), ammonia ($NH_3$), nitric oxide (NO), nitrous oxide ($N_2O$) and dinitrogen ($N_2$) from the plant-soil systems. The model has a good bridge between the C and N biogeochemical cycles and uses the information of soil moisture and temperature to predict $N_2O$ emission and other N related loss (Appendix A.2). After initial development, the DNDC model has undergone several developments for different regions and conditions (Saggar et al., 2004, Li et al., 2000; Stange et al., 2000, Kröbel et al., 2011, Han et al., 2014). Recently the DNDC model has been updated for local use for Canadian crops by coupling new routines of transpiration and potential evapotranspiration (PET) along with FAO crop coefficient modification and updated biomass growth curves for wheat and canola. This newly updated CDN-DNDC model (DNDC 9.5) was used in this research.

### 4.3.3 Model parameterization

The CDN-DNDC model requires a number of user-set parameters for simulating soil moisture, soil temperature, $N_2O$ emission and other N losses. Among these input parameters, most were obtained from field experiments, whereas some others were used as the model default value (Table 4.2). The water-field-pore-space (WFPS) at field capacity (FC) and permanent wilting point (PWP) was matched with the field observations by manually calibrating the model for the best simulation of soil moisture. The daily meteorological data file was prepared for 11 years from 2003 to 2013 to initialize the soil organic matter and nutrient pools in the model in an equilibrium condition so that the model stabilized the soil nitrogen and carbon. From 2003 to 2011 (the spin-up period), the meteorological data for Outlook was collected from Environment Canada's website; and for 2012 and 2013, in situ measured meteorological data was used. During the spin-up period of the model, no irrigation was added. The irrigation was only added for the experimental period. The irrigation events were included in the form of rainfall in the climate file for both 2012 and 2013.

The CDN-DNDC model quantifies soil moisture and temperature at different depths (1, 5, 10, 20, 30, 40, and 50 cm) as well as $N_2O$ fluxes at daily time steps. Therefore, the measured daily $N_2O$ fluxes, soil moisture, and temperature from both irrigated and non-irrigated fields were employed to test the applicability of the CDN-DNDC model in the study area. The fertilizer application was performed on a split basis. The amount of fertilizer N applied in fall was 140 kg $ha^{-1}$ and 78 kg $ha^{-1}$ and in spring was 6 kg $ha^{-1}$ and 12 kg $ha^{-1}$ in irrigated and non-irrigated field, respectively for the canola crop in 2013. During the spin-up period the time of application, amount, and type of fertilizer for wheat and canola was the same in both fields (i.e. for canola 78 kg N $ha^{-1}$ fall fertilizer and 12 kg N $ha^{-1}$ spring fertilizer and for wheat 78 kg N $ha^{-1}$ fall fertilizer and 6 kg N $ha^{-1}$ spring fertilizer).

### 4.3.4 Model sensitivity and evaluation

The CDN-DNDC model simulates soil moisture and $N_2O$ emission in a comprehensive manner which includes the combined effects of weather, soil and farm management activities. The modeled soil moisture and emission will vary when any of the driving factors change. The model performance under varied input parameters was tested through a sensitivity analysis in order to determine which parameters have the greatest effect on the predicted $N_2O$ emission.

Model validation differs from model sensitivity analysis because model validation includes a comparison of model output with observed data whereas the sensitivity analysis does not compare the field data. Simulated seasonal $N_2O$ flux, total N loss, and total water loss sensitivities were evaluated with CDN-DNDC for hydraulic conductivity (HC), porosity (P), soil organic carbon (SOC) and clay fraction (CF) (Table 4.3). These parameters were set to several values while all other model parameters and inputs were held constant at standard values. The baseline/default values of input parameter were the standard value, which was used in the validation test for the local climatic condition.

The model was evaluated using correlation of coefficient ($R^2$) and root mean square error (RMSE). Root mean square error is considered as a best overall measure of model performance as it summarizes the mean difference in the units of observed and predicted values (Willmott 1982).

Table 4.2. Soil and crop parameterization for the CDN-DNDC model

| Parameter type | Value | | Note |
|---|---|---|---|
| | Wheat | Canola | |
| Climate parameter | | | In situ measured field data along with collected data from nearby meteorological station |
| Climate data type | 5 | | |
| Total simulated year | 11 (*Spin up period 9 years, validation period 2 years*) | | |
| Soil parameter | | | |
| Soil Texture | Loam | | |
| Bulk Density | 1.16 | | (g $cm^{-3}$) |
| Soil pH | 7.60 | | |
| Hydraulic conductivity (HC) | 0.02502* | | (m $ha^{-1}$) |
| SOC at Surface | 0.02* | | (kg C $kg^{-1}$) |
| Clay fraction | 0.20* | | |
| Field capacity | 0.70 | | WFPS |
| Wilting point | 0.35 | | WFPS |
| Porosity | 0.55 | | ($m^3$ $m^{-3}$) |
| Soil CEC | 0* | | |
| Base saturation | 0* | | |
| Bulk density (>50 cm) | 1.78 | | (g $cm^{-3}$) |
| Initial N concentration (at surface) | | | |
| Nitrate | 0.5* | | (mg N $kg^{-1}$) |
| Ammonium | 0.05* | | |
| Crop parameter | | | |
| Plant time | 5 16 | 5 16 | Month day |
| Harvest time | 8 28 | 8 28 | Month day |
| Ground Residue | 0.80 | 0.80 | Leaves+stems in the field |
| Maximum Yield | 1800 | 1700 | kg C $ha^{-1}$ |
| Initial biomass * | 12.50 | 12.50 | |
| Biomass partitions | 0.37/0.48 /0.15 | 0.28/0.51/0.21 | Grain/(leaf+stem)/root |
| C/N ratio | 15/55/40 | 9/50/50 | Grain/(leaf+stem)/root |
| Thermal degree days (TDD) | 1637 | 1697 | °C |
| Water requirement | 270 | 120 | kg water/kg dry matter |
| N fixation | 1* | 1 | |
| Optimum temp | 21 | 21 | °C |
| Fertilizer parameter | | | |
| Fall fertilizer in preceding year | 140 (78) | 140 (78) | Irrigated field (non-irrigated field) in kg N $ha^{-1}$ |
| Spring fertilizer | 10 (6) | 10 (12) | |
| Irrigation parameter | | | |
| Amount of precipitation | 418.7 | 279.8 | mm |
| Amount of irrigation | 76.45 | 141.3 | mm |

* = Default values

Table 4.3. Baseline and alternative values of input parameters for the sensitivity test

| Scenario | Descriptions |
|---|---|
| Baseline of input parameters | HC *(m/hr)* 0.025000 *(Test Default)*; P 0.55 *(Test Default)*; SOC *(kg C/kg)* 0.02 *(Test Default)*; and CF 0.20 *(Test Default)* |
| Change in HC | Decrease by 0.01*(Test 2)* and 0.02 *(Test1)* and increase by 0.01 *(Test 4)* and 0.02 *(Test 5)* |
| Change in porosity | Decrease by 0.05 *(Test 2)* and 0.1 *(Test1)* and increase by 0.05 *(Test 4)* and 0.102 *(Test 5)* |
| Change in SOC | Decrease by 0.02 *(Test 2)* and 0.04 *(Test1)* and increase by 0.02 *(Test 4)* and 0.04 *(Test 5)* |
| Change in CF | Decrease by 0.2 *(Test 2)* and 0.4 *(Test1)* and increase by 0.2 *(Test 4)* and 0.402 *(Test 5)* |

After evaluation, the CDN-DNDC model was simulated for two different management scenarios (irrigated and non-irrigated) over 11 years to assess the long-term effect of fertilizer and irrigation management on $N_2O$ emission over wheat-canola crop rotation. Irrigation application was based on available soil moisture (ASM) within the upper 50 cm depth of soil. First, the CDN-DNDC model was simulated for the non-irrigation condition, then based on the simulated soil moisture, irrigation applications were prescribed to remove any moisture deficits. When the soil moisture declined below 50% of ASM, irrigation was applied (depth of each application was 10 mm). The application of N fertilizer (ammonium nitrate) was specified at 150 kg in the irrigated field and 100 kg in the non-irrigated field. This amount of fertilizer was applied in the spring (May $1^{st}$) of each simulation year.

## 4.4 RESULTS AND DISCUSSION

### 4.4.1 Validation of the model

The validation of the CDN-DNDC model was performed with two different questions in mind: a) how well does the model predict the dynamics of soil moisture and temperature under irrigated and non-irrigated situations; and b) how well does the $N_2O$ emission predicted by the model represent the observed local emission pattern?

#### 4.4.1.1 Simulation of soil moisture and temperature

Figures 4.3-4.6 illustrate the magnitude and temporal patterns of the observed and modeled soil moisture and temperature in response to local rainfall and irrigation. In 2012, all sensors were placed in the field in the first week of June; hence, there is no measured soil moisture and temperature in the early season (Figs. 4.3 and 4.4). Missing values in measured soil moisture and temperature in 2013 (Figs. 4.5 and 4.6) indicate the period when all the sensors were removed from the fields for seeding and other field operations at the beginning of the season and re-installed after all operations. It should be noted that the TDR sensors are not suitable for measuring soil moisture when ice is present in the soil pores; therefore, these data have been excluded during the winter and pre-soil-thaw period.

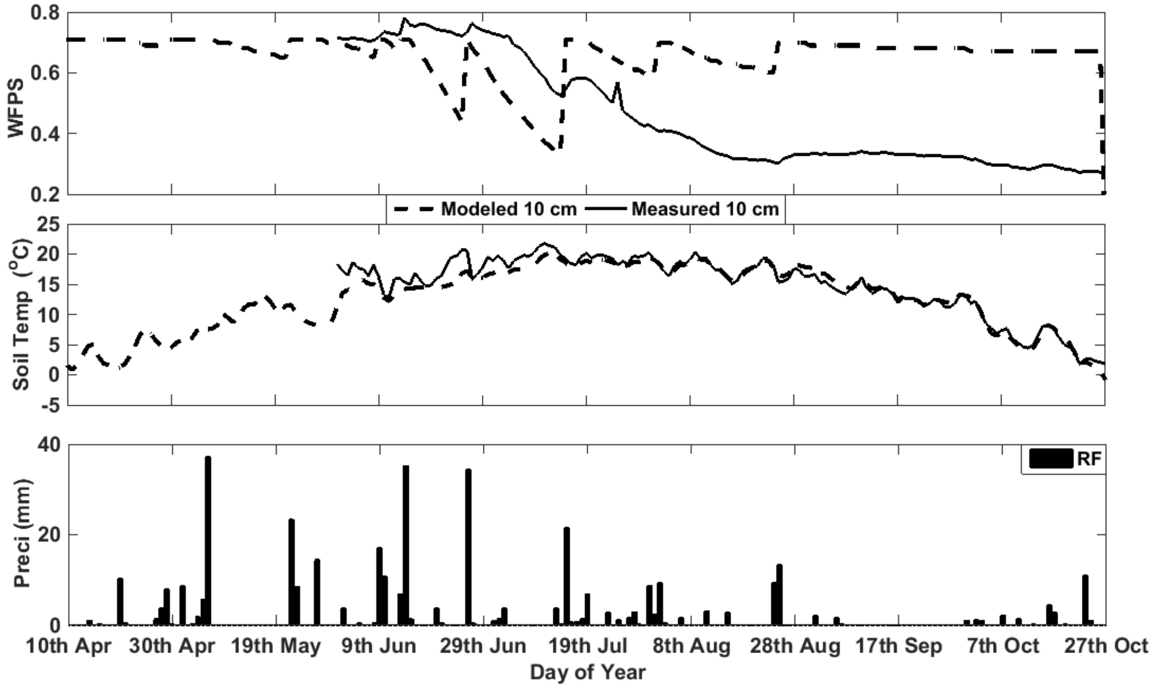



Fig. 4.3. Soil water and temperature in non-irrigated field in 2012 (RF=Rainfall, WFPS= Water filled pore space)

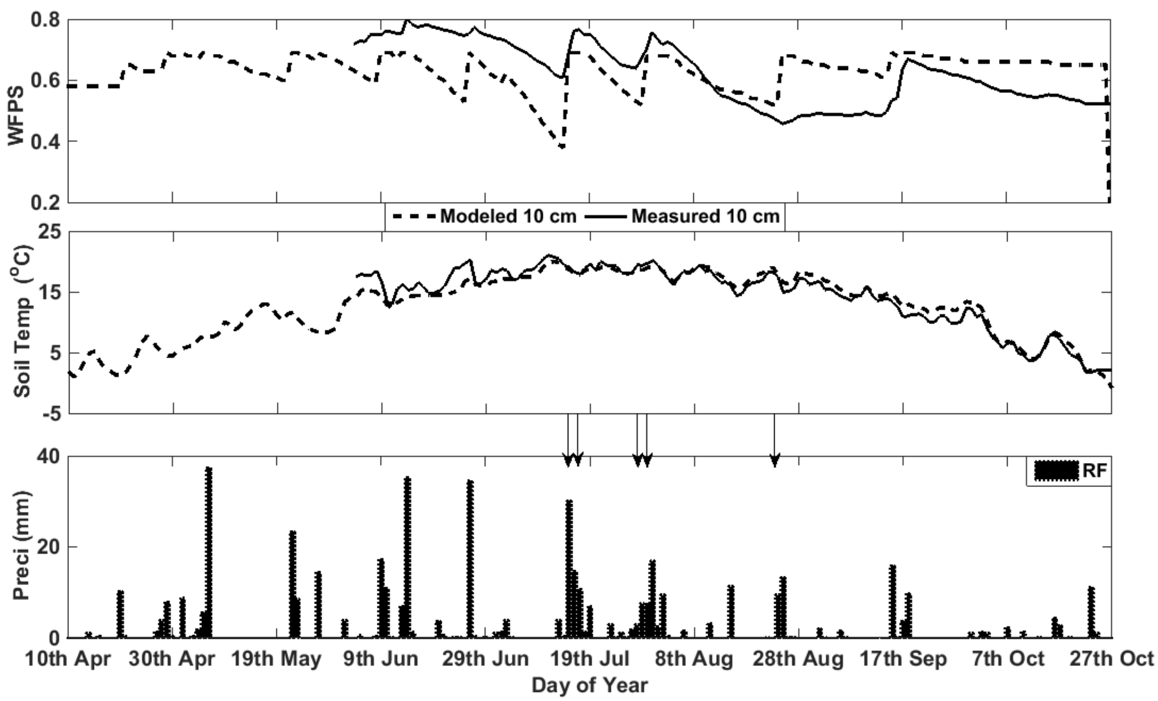



Fig. 4.4. Soil water and temperature in irrigated field in 2012 (RF=Rainfall, WFPS= Water filled pore space). Arrow shows the dates of each irrigation application

Early season soil moisture in both irrigated and the non-irrigated field was near FC (i.e. about 70% WFPS) at the beginning of both growing seasons; mainly caused by infiltrated snowmelt water. The 2012 growing season was especially wet due to above-normal rainfall. Consequently, soil moisture at 10 cm depth in both fields was near field capacity until the first week of July. Following this, soil moisture in the non-irrigated field decreased gradually, whereas the soil moisture in the irrigated field was above 50% WFPS due to additional irrigation. At the end of the season ($27^{th}$ of Oct) soil moisture in the non-irrigated field was around 30% and in the irrigated field around 50% WFPS. However, at the beginning of the season in 2013, both fields had around the same (68% WFPS) amount of soil moisture. The 2013 growing season was drier, and a reduction in soil moisture in both fields began in the middle of June. After that, the soil moisture in the non-irrigated field declined slowly except for a few days, when there were large rainfall events. Soil moisture in the irrigated field was maintained between 70 and 50% WFPS during and after a few days of irrigation. At the fruit ripening stage ($2^{nd}$ week of August) soil moisture in the irrigated field was above 60% WFPS and was below 30% WFPS in the non-irrigated field.

The ability of the model to match the observed soil moisture was variable. During the 2012 growing season, modeled soil moisture was under-predicted during the early season and over-predicted during the late season. The irrigated, and non-irrigated field had the same temporal pattern of soil moisture until the irrigation event on $13^{th}$ July. The model started reducing soil moisture from 70% WFPS on June $15^{th}$ and reached 55% WFPS by June $24^{th}$ with the association of the low amount of rainfall. Modeled soil moisture reached field capacity (70% WFPS) on $25^{th}$ and $26^{th}$ of June due to a big rainfall event and then followed a rapid reduction (near 40% WFPS) until $13^{th}$ of July. After July $13^{th}$, soil moisture was above 60% in the irrigated field due to additional irrigation on the field except for few days, and above 55% WFPS in the non-irrigated field. During the 2013 growing season, the model predicted soil moisture was similar to observed soil moisture. However, the amount of the modeled soil water in the non-irrigated field was higher than the measured soil water. In this year modeled soil moisture was around 70% WFPS in May and started to decline in June. However, due to rainfall, soil moisture was above 50% WFPS in June in both fields. Following this, soil moisture was maintained above 60% WFPS in the irrigated field due to applied irrigation, and it dropped below 50% WFPS in the non-irrigated field due to lack of rainfall.

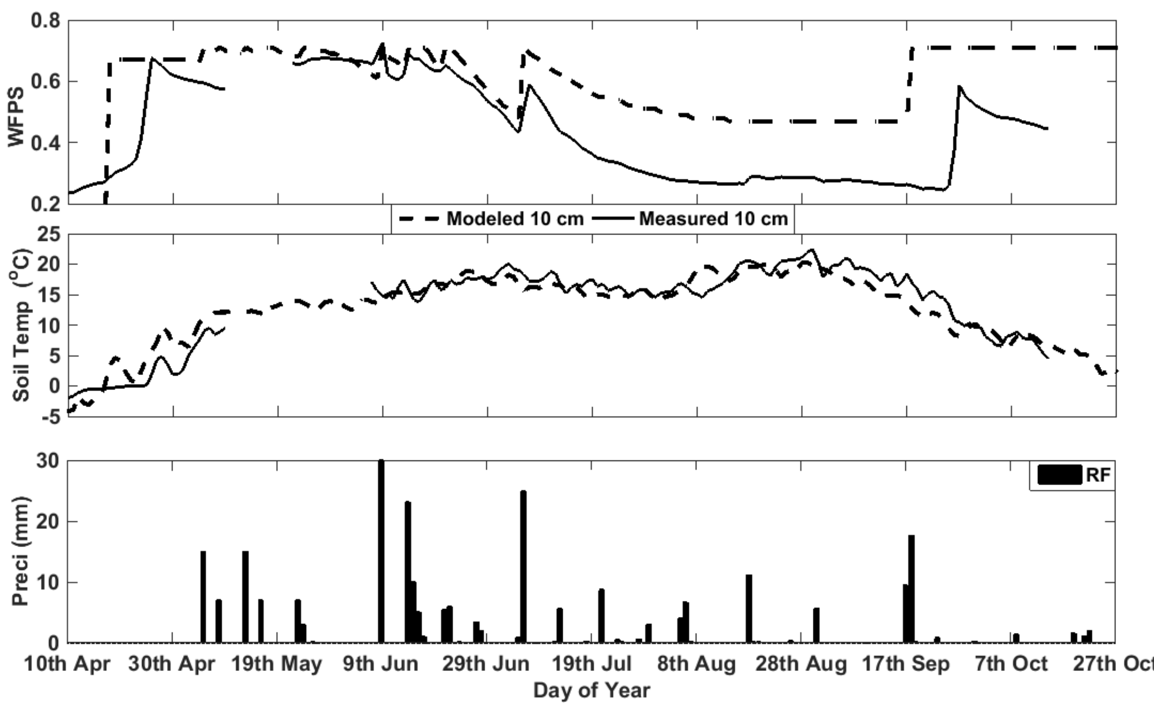



Fig. 4.5. Soil water and temperature in non-irrigated field in 2013 (RF=Rainfall, WFPS= Water filled pore space)

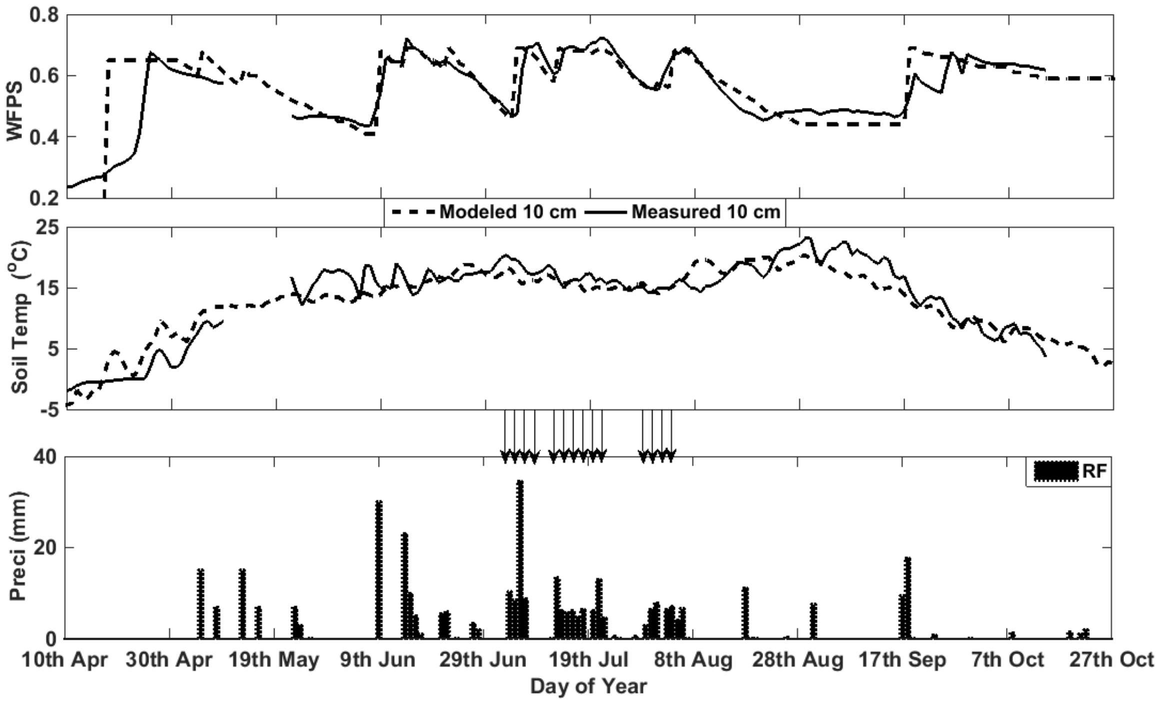



Fig. 4.6. Soil water and temperature in irrigated field in 2013 (RF=Rainfall, WFPS= Water filled pore space). Arrow shows the dates of each irrigation application

Generally, the temporal pattern of soil moisture provided by the model was good and properly followed the observed trend; however, the magnitude of these fluctuations differed from the observed. The model tended to predict sharp changes in soil moisture associated with precipitation or irrigation events. However, actual soil moisture fluctuations related to precipitation events were more gradual. Later in both growing seasons, the model predicted that the non-irrigated field increased soil moisture following rainfall, whereas the observed soil moisture stayed approximately at the same level. Correlation between observed and modeled soil moisture was good in 2013 in both irrigated ($R^2$ = 0.81 and RMSE = 0.04) and non-irrigated field ($R^2$ = 0.75 and RMSE = 0.10) and poor in 2012 in both irrigated ($R^2$ = 0.11 and RMSE = 0.11) and non-irrigated ($R^2$ = 0.10 and RMSE = 0.17) field. The model is incapable of increasing soil moisture from snow melt water in the same manner as the observed field condition. Hence, at the 25 cm depth, there was a difference in simulated soil water between irrigated and non-irrigated field early in the season in 2013 (Appendix A.3), whereas measured soil water in both fields was similar.

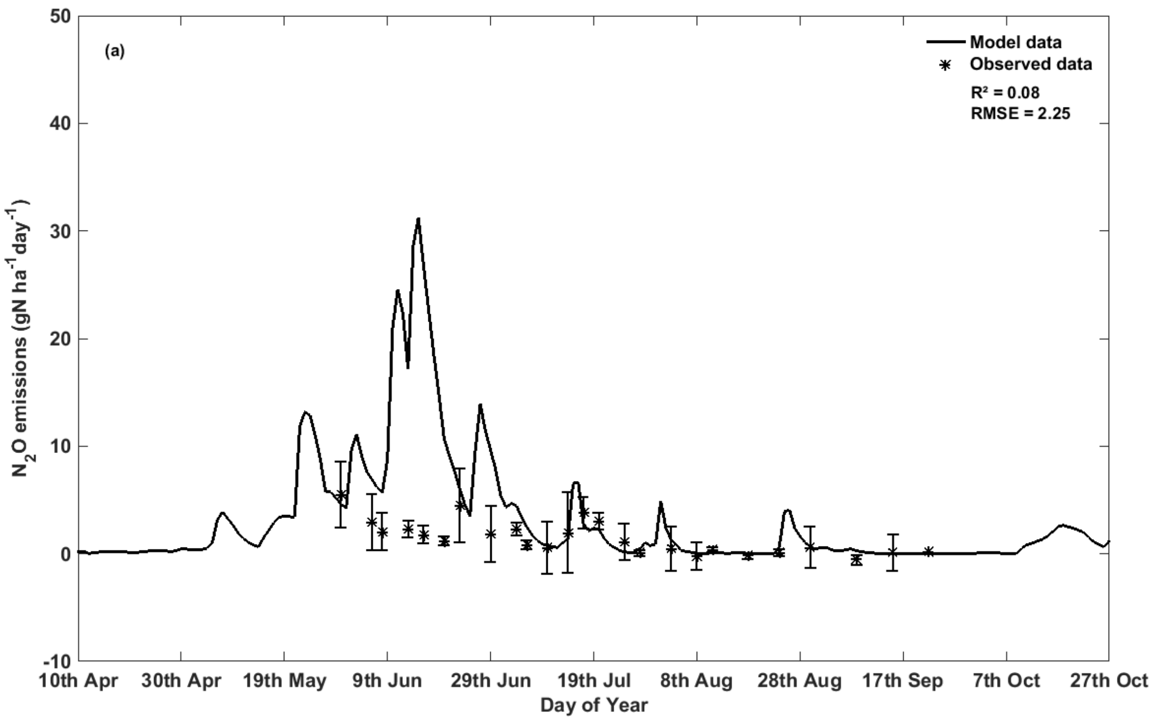
The DNDC model assumes homogeneous soil conditions for all depths, whereas in the study fields, the soil layers were observed to be heterogeneous (Appendix A.4). For example, the DNDC model has only one value of porosity for all depths; however, in field conditions, the bulk density generally increases with depth, decreasing the porosity. Because of the simple model structure, it has been demonstrated to be relatively insensitive to changes in the parameterized hydraulic conductivity of the soil (Krobel et al., 2010). The difference between the observed and modeled soil moisture in this study is likely due to a combined effect of all of the aforementioned factors. Similar reasons have contributed to differences between observed and simulated soil moisture throughout the literature. Zhang et al. (2002b) pointed out that the DNDC model estimated dynamics of soil water quite well in natural systems. However, problems occurred when the model was applied to systems under varying irrigation conditions. They concluded that the deviation of simulated soil moisture under different irrigation conditions may have resulted from poorly represented soil heterogeneity. They also observed that the simulations were better for deeper layers than for shallow layers, and their simulated average soil moisture of the whole soil profile (0-50 cm) was better than individual layers. Li et al. (2006a, 2006b) stated that the changes in the water discharge simulation of the DNDC model (through 90% decreased water conductivity for the discharge layer) yielded improved modeling results. Nevertheless, discrepancies still occurred between measurements and simulations. Beheydt et al. (2007) found that the DNDC model underestimated WFPS for the different investigated sites (in different agricultural fields in Belgium) in their study. Similarly, Smith et al. (2008) observed that the DNDC model under-predicted soil water by 17% average relative error at the Elora research station in Ontario. They mentioned that the reason for this presumably originated from the tipping bucket hydraulic routines. With this model structure, water in the soil profile is quickly drained to field capacity following rainfall events. Krobel et al. (2010) used DNDC for modeling water dynamics in Northern China and found that neither the default nor the optimized DNDC was able to satisfactorily reproduce the soil water dynamics.

The observed and modeled soil temperatures (at 10 cm depth) are shown in the middle panel of Fig.s 4.3-4.6. The 2012 field observations showed that the highest temperature occurred near mid-July and decreased afterward in both fields. During the growing season, soil temperature stayed between 14 to 22°C in both fields. However, the non-irrigated field had a slightly higher temperature compared to the irrigated field. In the case of 2013’s growing season, the lowest temperature was observed in July, and the highest temperature was recorded in the last week of August and the first week of September. In this year, soil temperature was between 14 to 20°C in both fields during the growing season. Although the air temperature was very similar in both fields, both years' field observations showed that irrigation slightly decreased the daily average soil temperature.

The soil temperature simulated by the DNDC model is linked with air temperature, and the air temperature used to drive the model was the same for both fields. Hence, the CDN-DNDC model predicted identical soil temperatures in both irrigated and non-irrigated fields. The model captured the observed seasonal trend of soil temperature. However, the shorter time-scale dynamics of soil temperature were poorly represented by the model. Correlation between observed and modeled soil temperature was good in 2012 in both irrigated ($R^2$ = 0.54 and RMSE = 1.40) and non-irrigated ($R^2$ = 0.51 and RMSE = 1.80) fields. During 2013, correlations were poor in both the irrigated ($R^2$ = 0.24 and RMSE = 2.14) and non-irrigated ($R^2$ = 0.43 and RMSE = 1.64) fields. When the DNDC simulated and observed soil temperature was compared in Woodslee in Southern Ontario, Smith et al. (2008) found that it under-predicted soil temperatures by 7% average relative error. Kariyapperuma (2011) found good agreement between measured and simulated soil temperature at 5 cm depth after spring thaw at the Elora research station in Ontario. However, they noticed that the simulated soil temperature in the deeper soil (greater than 5 cm depth) was higher than the measured soil temperature. Balashov et al. (2014) validated DNDC for soil temperature in the northwestern region of Russia and conclude that the efficiency in predicting seasonal dynamics of soil temperature is poor.

#### 4.4.1.2 Simulation of soil $N_2O$ emission

The comparison of daily simulated and observed soil $N_2O$ emissions for the year 2012 and 2013 are presented in Fig. 4.7 and 4.8. Emissions from both fields generally follow a similar pattern in which the largest emissions are observed following snowmelt and early season rainfall, followed by much lower late season fluxes. In 2012, emission measurement began on May 30$^{th}$; hence, the emissions immediately following snowmelt were not recorded for this year. Owing to the greater amount of fall-applied N, emissions from the irrigated field were higher than the non-irrgated field. The largest observed daily emission in 2013 was 67 gN ha$^{-1}$ day$^{-1}$ in the irrigated field on April 28$^{th}$. In 2012, both fields had the highest emission in the weeks following seeding. In 2013, early in the season emission was above 10 gN ha$^{-1}$ day$^{-1}$ and later in the season was below 10 gN $ha^{-1}$ $day^{-1}$, with both fields experiencing fluxes of similar magnitude. The total emission measured during the growing season in 2012 was 539 and 141 gN $ha^{-1}$ from the irrigated and non-irrigated field, respectively. Whereas in 2013, total annual emission was 1147 gN $ha^{-1}$ from the irrigated field and 983 gN $ha^{-1}$ from the non-irrigated field. As the measurement of emission was started from May 30th in 2012, the emissions calculated using a similar time period for 2013 would be 609 and 842 gN $ha^{-1}$ from the irrigated and non-irrigated field, respectively. This results indicates that the $N_2O$ emission is highly variable from day to day and between chamber locations (error bars in Figs. 4.7 and 4.8). The details concerning the emission patterns, including the daily variation in emission along the transect, are explained in David (2014). David (2014) hyopthesises that (a) in the early season, both irrigated, and non-irrigated field experience high emissions due to wet soil conditions from snow-melt water or rainfall as well as high levels of soil nutrients (i.e. available N) from fertilizer and crop residue, and (b) later in the season, emissions are limited in the irrigated field by low nutrient levels and are limited in the non-irrigated by low moisture levels.



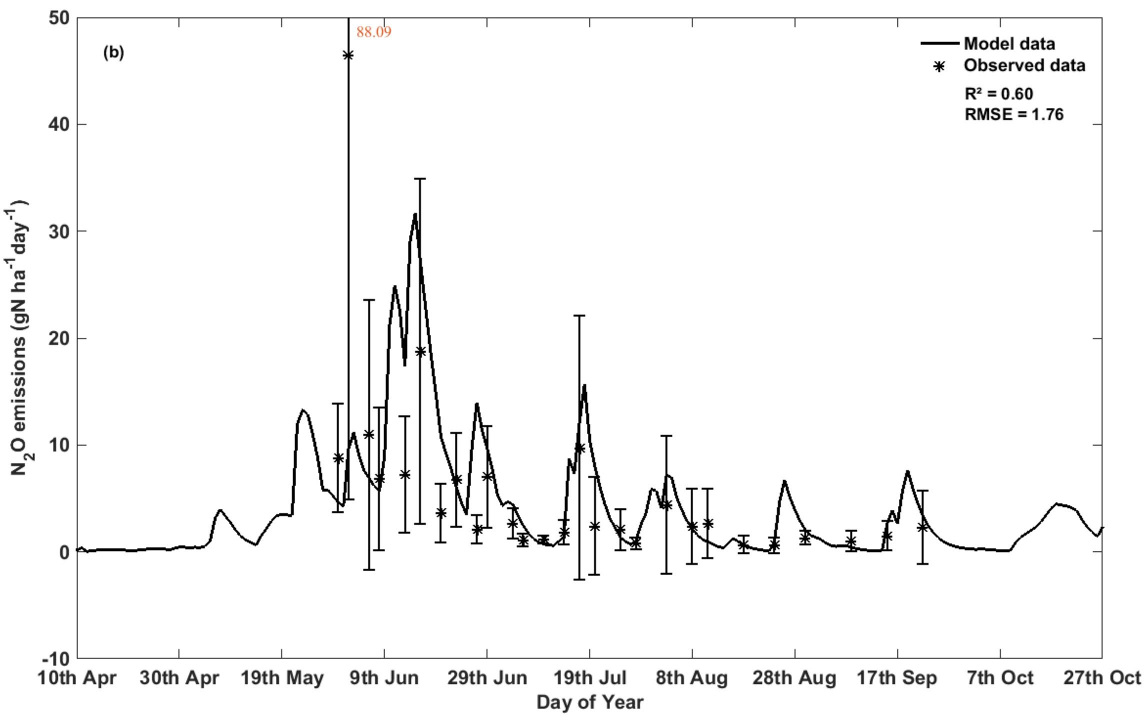
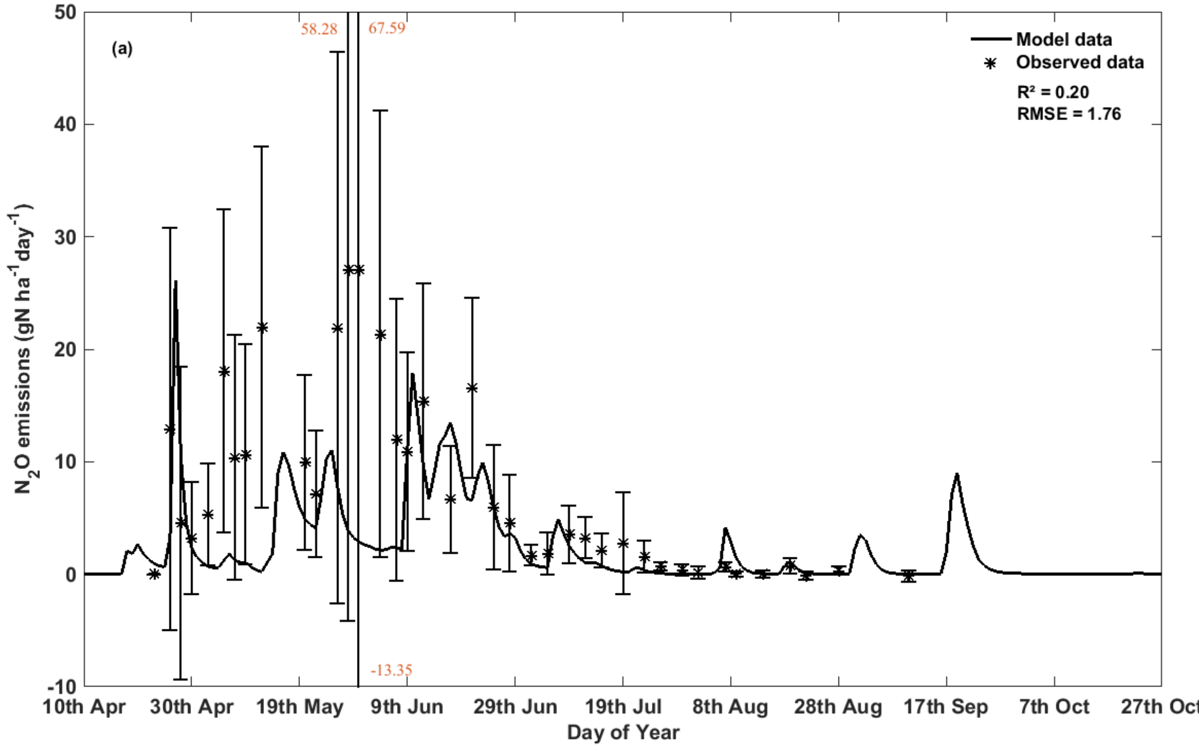




Fig. 4.7. Measured and simulated $N_2O$ emission in 2012 (a. Non-irrigated field, b. Irrigated field). Error bar in each measured point indicates the standard deviation of measured emission from 20 chambers in that day



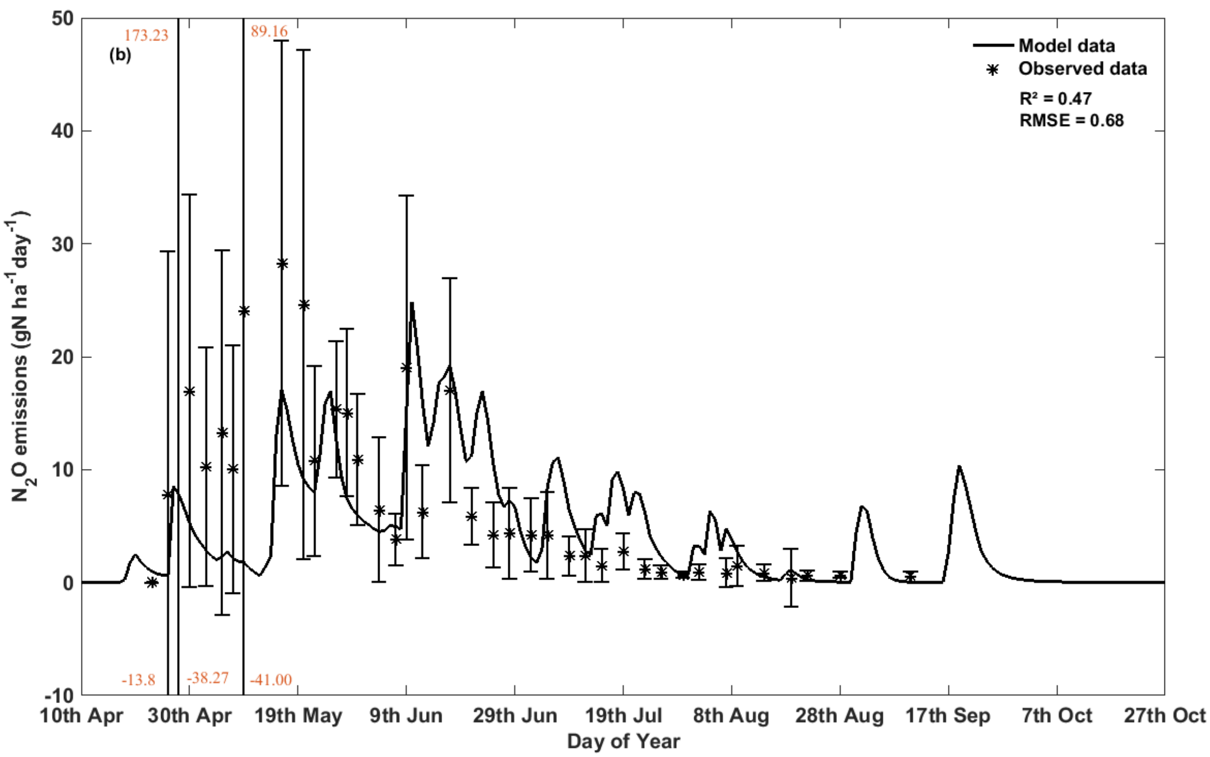




Fig. 4.8. Measured and simulated $N_2O$ emission in 2013(a. Non-irrigated field, b. Irrigated field). Error bar in each measured point indicates the standard deviation of measured emission from 20 chambers in that day

The model predicted emissions from both irrigated and non-irrigated field which were similar in short-term fluctuations and the overall emission pattern. The model predicts higher emission in the irrigated field due to the higher amount of fall fertilizer. The model predicted the highest emissions after seeding in both growing season and the lowest emission in the late season except the spring emission from the non-irrigated field in 2013. In 2012, the highest simulated emission was 32 and 31 gN $ha^{-1}$ $day^{-1}$ in the irrigated and non-irrigated field, respectively on June 14th corresponding to 35 mm rainfall on June 13th. The total simulated emission in growing season in 2012 was 666 and 563 gN $ha^{-1}$ from irrigated and non-irrigated field, respectively. In 2013, the highest predicted emission by the model was found on June 10th in irrigated field amounting to 25 gN $ha^{-1}$ $day^{-1}$, and on April 26th in the non-irrigated field amounting 26 gN $ha^{-1}$ $day^{-1}$. Total simulated emission in the growing season of 2013 was 717 gN $ha^{-1}$ from the irrigated field and 355 gN $ha^{-1}$ from the non-irrigated field. When the high variability of field-measured $N_2O$ emissions are considered (error bars in Figures 4.7 and 4.8), it can be seen that the simulated emission often lies within the range of the measured emission. Notable exceptions include to the 2012 emissions from the non-irrigated field, and a few emission days in the irrigated field during July and August, 2013. In all cases, the model struggles to correctly match the measured emission during the spring thaw.

Overall, the prediction of the temporal pattern of daily $N_2O$ fluxes by the model was adequate, particularly later in the growing season. The simulated emission pattern was more realistic in the irrigated field; however, the timing of the peak and magnitude was different between the observed and simulated daily fluxes. Fig. 4.7 and 4.8 clarify that, from the end of June to the end of the season, the difference between modeled and measured emission is small in the non-irrigated field and the irrigated field in 2012. The model tended to predict lower emission at this time perhaps due to the lower amount of remaining N fertilizer in the soil and followed the same hypothesis of emission from irrigated and non-irrigated fields (David, 2014). There was a good correlation between measured vs. modeled daily fluxes in the irrigated field ($R^2$=0.60 and RMSE=1.76 gN $ha^{-1}$ $day^{-1}$ in 2012, and $R^2$=0.47 and RMSE=0.68 gN $ha^{-1}$ $day^{-1}$ in 2013). However, there was a poor correlation between modeled and measured emissions in the non-irrigated field in both 2012 ($R^2$=0.08 and RMSE=2.25 gN $ha^{-1}$ $day^{-1}$) and 2013 ($R^2$=0.19 and RMSE=1.76 gN $ha^{-1}$ $day^{-1}$). The difference of measured and simulated fluxes for both fields is likely related to the model's limitation in the proper prediction of spring emission. An irregular $N_2O$ emission was found in Ontario, Canada by Kariyapperuma et al. (2011). They found a large discrepancy between simulated and observed fluxes regarding the magnitude and timing due to the soil's freeze-thaw mechanism in that region.

The measured and simulated available soil $NH_4$ and $NO_3$ at 0 to 60 cm depths after the growing season in 2013 were compared (Fig. 4.9). In 2013, the total available measured soil N (sum of $NH_4$ and $NO_3$) at the end of the growing season was higher in the non-irrigated field (55 kg N $ha^{-1}$) than in the irrigated field (42 kg N $ha^{-1}$). Similarly, the available soil N at the depth of 0 to 90 cm was 69 kg N $ha^{-1}$ in irrigated field and 99 kg N $ha^{-1}$ in the non-irrigated field. The values of available soil N provides some evidence for the hypothesis of $N_2O$ emission: i.e. the non-irrigated field could be rich in nutrient at the end of the season. In this case, the model also follows the same hypothesis. Hence, soil available N ($NH_4$ and $NO_3$) was 49 kg N $ha^{-1}$ in irrigated field and 73 kg N $ha^{-1}$ in the non-irrigated field. Fig. 4.9 also showed that the model tended to predict an opposite trend to the measured soil $NH_4$ and $NO_3$ i.e. the model underestimated the $NH_4$ and overestimated the $NO_3$.

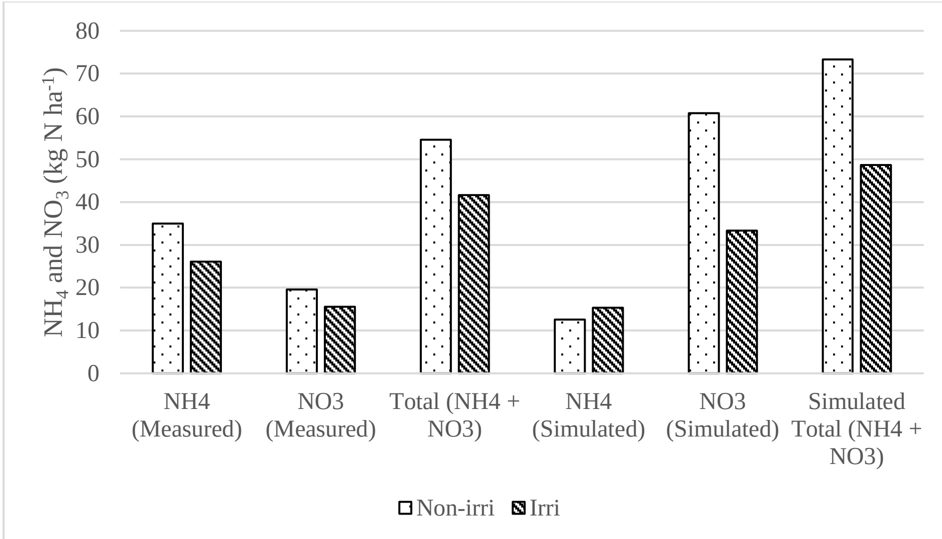



Fig. 4.9. Available soil nutrient at the end of the growing season at 0 to 60 cm depth

Measured and simulated crop yield was compared in Table 4.4 for both the irrigated and non-irrigated field. The model input parameter for expected maximum yield, was kept the same in both irrigated and non-irrigated field, which was 4500 kg ha$^{-1}$ for wheat and 4200 kg ha$^{-1}$ for canola. The crop yield predicted by the model was lower than the actual production in the case of the irrigated condition in 2012 (wheat) and was higher than the actual production in the rest of the cases. Interestingly, the crop yield in 2012 was predicted to be the same in both the irrigated and non-irrigated field, as rainfall was high in this year.

Table 4.4. Model predicted crop production

| Year | Non-irri field | Non-irri field | Irrigated field | Irrigated field |
|---|---|---|---|---|
| | Measured (kg ha$^{-1}$) | Simulated (kg ha$^{-1}$) | Measured (kg ha$^{-1}$) | Simulated (kg ha$^{-1}$) |
| 2012 | 2500 | 2713 | 3400 | 2713 |
| 2013 | 2400 | 2465 | 3600 | 3978 |

### 4.4.2 Sensitivity test

The sensitivities of CDN-DNDC-predicted soil moisture and $N_2O$ emissions to the adjustable soil parameters were examined by independently changing one parameter at a time while fixing all other input parameters. The results of the sensitivity test (Table 4.5) illustrate which parameters are expected to have the greatest effect on soil moisture and $N_2O$ emission, as well as reducing total N loss and improving crop yields.

Regarding soil moisture, the CDN-DNDC model was very sensitive to soil porosity, which acted to decrease water leaching and increase the crop yield. Simulated water loss through leaching decreased from 16 mm at a porosity of 0.45 to 5.28 mm at a porosity of 0.65. However, this amount of water saving increased the simulated crop yield from 2988 kg $ha^{-1}$ (porosity = 0.45) to 3168 kg $ha^{-1}$ (porosity = 0.65). The CDN-DNDC model is also sensitive to the specified clay fraction. Increasing the clay fraction increased leached water loss from 4 mm at CF 0.16 to 18 mm at CF 0.24 and decreased crop yield from 3310 kg $ha^{-1}$ at CF 0.16 to 2965 kg $ha^{-1}$ at CF 0.24. Increases the SOC decreases the water loss through leaching from 16 mm at SOC 0.016 to 5 mm at SOC 0.024. Increasing SOC also increases crop yield from 2888 kg $ha^{-1}$ at SOC 0.016 to 3343 kg $ha^{-1}$ at SOC 0.024. Although hydraulic conductivity is a key parameter to control the water movement between soil depths, the CDN-DNDC model proved to be insensitive to changes in this parameter with respect to crop production. However, the highest amount of water loss by leaching was found at lower HC.

Table 4.5. Sensitivity of the CDN-DNDC model to HC, porosity, CF, and SOC

| | | Water leaching (mm) | Total N loss* (kg N $ha^{-1}yr^{-1}$) | N leach (kg N $ha^{-1}yr^{-1}$) | $N_2O$ emission (kg N $ha^{-1}yr^{-1}$) | Yield (kg $ha^{-1}yr^{-1}$) | $R^2$ (soil water at 10 cm depth) | $R^2$ ($N_2O$ emission) |
|---|---|---|---|---|---|---|---|---|
| HC | Test 1 (0.005 m/hr) | 20.21 | 12.64 | 6.64 | 0.71 | 3077.5 | 0.78 | 0.60 |
| | Test 2 (0.015 m/hr) | 20.21 | 12.64 | 6.64 | 0.71 | 3077.5 | 0.78 | 0.53 |
| | Test Default (0.025 m/hr) | 10.24 | 11.27 | 6.65 | 0.65 | 3105 | 0.80 | 0.60 |
| | Test 4 (0.035 m/hr) | 11 | 11.6 | 6.97 | 0.65 | 3100 | 0.80 | 0.60 |
| | Test 5 (0.045 m/hr) | 11.52 | 11.94 | 7.31 | 0.65 | 3095 | 0.80 | 0.61 |
| Porosity | Test 1 (0.45) | 15.85 | 18.6 | 15.5 | 0.57 | 2987.5 | 0.82 | 0.41 |
| | Test 2 (0.50) | 16.08 | 15.91 | 11.48 | 0.69 | 3055 | 0.78 | 0.55 |
| | Test Default (0.55) | 10.24 | 11.27 | 6.65 | 0.65 | 3105 | 0.80 | 0.60 |
| | Test 4 (0.60) | 4.95 | 7.23 | 2.36 | 0.65 | 3150 | 0.80 | 0.62 |
| | Test 5 (0.65) | 5.28 | 6.92 | 1.36 | 0.66 | 3167.5 | 0.68 | 0.53 |
| Clay Fraction | Test 1 (0.16) | 3.89 | 9.15 | 4.6 | 0.66 | 3310 | 0.78 | 0.59 |
| | Test 2 (0.18) | 3.93 | 7.84 | 3.24 | 0.66 | 3237.5 | 0.79 | 0.59 |
| | Test Default (0.20) | 10.24 | 11.27 | 6.65 | 0.65 | 3105 | 0.80 | 0.60 |
| | Test 4 (0.22) | 17.19 | 13.04 | 8.57 | 0.63 | 3005 | 0.79 | 0.60 |
| | Test 5 (0.24) | 18.36 | 11.33 | 6.8 | 0.63 | 2965 | 0.80 | 0.60 |
| Soil organic carbon (SOC) | Test 1 (0.016) | 16.22 | 13.4 | 8.67 | 0.6 | 2887.5 | 0.79 | 0.61 |
| | Test 2 (0.018) | 13.93 | 12.9 | 8.31 | 0.62 | 2987.5 | 0.79 | 0.60 |
| | Test Default (0.020) | 10.24 | 11.27 | 6.65 | 0.65 | 3105 | 0.80 | 0.60 |
| | Test 4 (0.022) | 8.82 | 10.25 | 5.73 | 0.68 | 3212.5 | 0.80 | 0.59 |
| | Test 5 (0.024) | 5.16 | 7.76 | 3.35 | 0.69 | 3342.5 | 0.80 | 0.58 |

*Total N loss = the sum of N leaching, N runoff, $N_2O$ flux, NO flux, $N_2$ flux, and $NH_4$ flux

Soil $N_2O$ emissions were most sensitive to SOC, with emissions decreasing with a decreasing SOC (Table 4.5). Total soil N loss was sensitive to SOC followed by porosity. Neither soil $N_2O$ emission nor total N loss was substantively sensitive to HC and CF.

### 4.4.3 Long-term management scenario

The validation test showed that the CDN-DNDC model has the potential to predict the dynamics of soil moisture and temperature at 10 cm depths and daily $N_2O$ emissions under different management and climate scenarios (irrigated and non-irrigated conditions). The validation experiment did identify model limitations concerning simulation of soil water, soil temperature, and $N_2O$ emission during the spring period. However, the model performance was deemed to be acceptable for irrigated conditions during the bulk of the growing season. As the purpose of this research is to investigate $N_2O$ emissions simulated by the DNDC model under irrigated and non-irrigated conditions in the Canadian Prairies, a long-term scenario by the CDN-DNDC model has been performed to identify differences in $N_2O$ emission from irrigated cropping systems. The aforementioned model deficiency concerning spring emissions should similarly affect the irrigated and non-irrigated systems, and would not be expected to introduce any bias. To test the long-term impacts, two long-term (11 yr) alternative management scenarios were constructed on wheat-canola crop rotation: (1) an irrigated management system with a fertilizer application rate of 150 kg N ha $^{-1}$yr$^{-1}$, (2) a non-irrigated management system with an N fertilizer application rate of 100 kg N ha$^{-1}$yr$^{-1}$. The rest of the model's driving variables (i.e., climate, soil and farm management) were kept constant for the observed values and model default data. The DNDC model was run for 11 yr with each of the scenarios with the past climate data (2003-2013).

Nitrogen losses were dominated by nitrate leaching, N runoff, and $NH_3$, $N_2O$, and $N_2$ trace gas flux (Table 4.6). In this scenario, modeled nitrate leaching demonstrated large interannual variability corresponding to the amount of precipitation early in the season, resulting in extremely high values in some years. The long-term scenario test demonstrate a mean nitrate leaching 31.94 kg N ha$^{-1}$ and 9.21 kg N ha$^{-1}$ in the irrigated and non-irrigated management system. As the irrigated system had higher input water, water losses through leaching are also higher in the irrigated field than the non-irrigated field. Because of a higher amount of fertilizer in the irrigated field, $N_2O$ emissions were greater in the irrigated field (a mean annual $N_2O$ flux of 0.93 kg N $ha^{-1}$ and 0.65 kg N $ha^{-1}$ in the irrigated and non-irrigated management system, respectively). Although, irrigated management system increased $N_2O$ emission, crop production was also high in the irrigated system. Hence, if the amount of crop yield is taken into consideration, then the emission was not large from the irrigated system. Hence, when the maximum nitrous oxide intensity, which is the amount of $N_2O$ emission per unit crop yield, was calculated it was found that this nitrous oxide intensity was 0.56 and 0.93 g N $ha^{-1}$ in irrigated and non-irrigated field, respectively. The dry condition in the non-irrigated management system causes high $NH_4^+$ in the field as conversion of $NO_3^-$ from $NH_4^+$ is limited due to lack of sufficient soil water. Hence, $N_2O$ flux is found to be higher in the non-irrigated field in some years, because of having relatively high amount of rainfall.  From the literature, it is found that the DNDC model could over predict $N_2O$ flux under dry conditions (Frolking et al. 1998). In the present study, most of the N losses (i.e. total N loss or N leaching) occured due to the higher volume of precipitation after fertilizer application as well as early in the season.

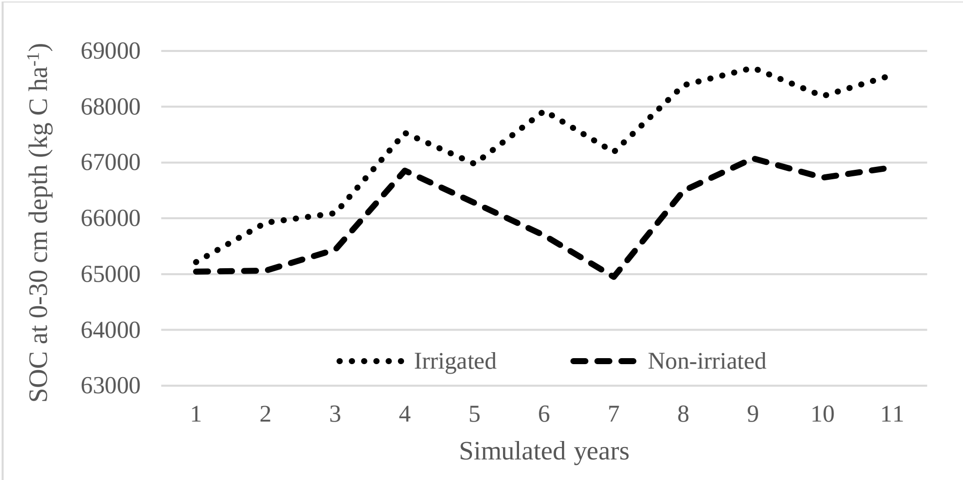



Fig. 4.10. Long time effect on SOC under irrigated and non-irrigated condition

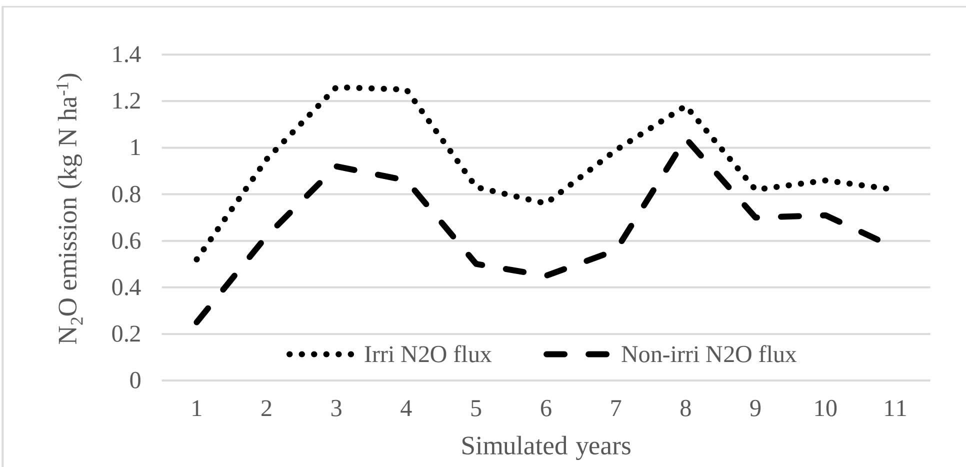



Fig. 4.11. Long time effect on soil $N_2O$ emission under irrigated and non-irrigated condition

Table 4.6. Cumulative flux N trace gasses, and other N losses in kg N $ha^{-1}$ $yr^{-1}$ over 11-year simulations for an irrigated wheat–canola compared to non-irrigated wheat-canola crop rotation system

| | N uptake | $N_2O$ flux | $NO_3^-$ Leach | NO flux | $N_2$ flux | $NH_3$ flux | N runoff | Total N loss | Litter N | Mineralization | Water Leach | Water runoff | Yield |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *kg N $ha^{-1}$ $yr^{-1}$* | | | | | | | | | | *mm* | | *kg $ha^{-1}$* |
| **Irrigated management system** | | | | | | | | | | | | | |
| Mean | 179.42 | 0.93 | 31.94 | 0.29 | 0.25 | 3.38 | 2.88 | 39.67 | 68.19 | 75.84 | 22.20 | 24.05 | 3244 |
| Max | 254.48 | 1.26 | 132.64 | 0.35 | 0.32 | 4.52 | 9.38 | 138.75 | 102.61 | 84.40 | 107.02 | 59.36 | 3960 |
| Min | 78.79 | 0.82 | 0.00 | 0.20 | 0.17 | 2.20 | 0.11 | 3.20 | 35.18 | 70.60 | 0.00 | 2.53 | 2143 |
| **Non-irrigated management system** | | | | | | | | | | | | | |
| Mean | 139.20 | 0.65 | 9.21 | 0.21 | 0.15 | 2.59 | 2.93 | 15.74 | 52.29 | 69.85 | 6.77 | 23.81 | 2550 |
| Max | 254.47 | 1.04 | 35.88 | 0.27 | 0.23 | 3.79 | 8.25 | 42.85 | 101.66 | 81.8 | 26.54 | 59.36 | 3923 |
| Min | 27.48 | 0.25 | 2.64 | 0.14 | 0.05 | 1.56 | 0.1 | 2.12 | 12.07 | 64.7 | 0 | 2.53 | 605 |

This simulation showed that there is the scope for increasing crop production and reducing $N_2O$ emissions and other N related losses through modified fertilizer and irrigation management. This simulation exercise suggests that irrigated wheat-canola rotations can increase crop yield sustainably. Future research should evaluate the economic and environmental costs and benefits of modified management options and examine how farmers can best be encouraged to adopt climate-friendly management strategies.

### 4.4.4 Model application and extension

There are some discrepancies in estimating $N_2O$ emission from the irrigated agricultural fields in Saskatchewan, mainly because of the cold winter climate. Modification of the model is still required to (i) improve the simulation of soil moisture, and soil temperature immiediately following the snow melt period, (ii) improve the simulation of $N_2O$ emissions from snow melt water earlier at the growing season, (iii) improve the simulation of soil temperature with irrigation. The development of this model with more detailed processes will further improve the model performance for irrigated condition in the study area. Once the model is accurately calibrated and validated after further modifications, it can be utilized for improving $N_2O$ emission estimates, identifying $N_2O$ mitigation strategies, identifying changes in C and N dynamics under long-term cropping system under irrigation, and identifying best fertilizer and irrigation management for this region.

## 4.5 CONCLUSIONS

The study indicates that the regional version (CDN-DNDC) of the DNDC model is capable of quantitatively capturing the major aspects of $N_2O$ emission from irrigated agricultural fields in Western Canada. This model successfully simulated the soil water under irrigated condition in a fairly typical year (2013). The model also simulated the temporal pattern of the $N_2O$ emission for both irrigated and non-irrigated condition. However, there were some discrepancies between observed and simulated daily fluxes, soil moisture, and soil temperature, indicating that DNDC does not capture all processes occurring in the field, mainly during the spring thaw period. Further improvement of the CDN-DNDC model is required in order to investigate the influence of varying irrigation volume upon $N_2O$ emission.

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# 5. SUMMARY AND CONCLUSIONS

The question of how irrigation influences soil GHG emissions, and how it can be managed to reduce GHG emissions, is examined in this study. Specifically, this research examined the effects of irrigation on energy partitioning, crop microclimate, soil environment, and GHG ($N_2O$) emission. Furthermore, a regional version of the Denitrification-Decomposition (CDN-DNDC) model was validated and used to predict GHG emissions occurring under irrigated conditions in the Canadian Prairies.

## 5.1 SUMMARY

Chapter 3 describes the effect of irrigation on surface energy flux, crop microclimate, soil environment, and soil GHG emission. The chapter explained the findings of a two-year experiment on adjacent fields, one irrigated and one non-irrigated, where continuous measurement of energy fluxes, meteorological data, canopy and soil temperatures, soil water status, were conducted, along with periodic chamber based GHG emissions measurements. These findings improve our understanding of how irrigation stimulates soil GHG emissions by changing the overall crop and soil environments.

The latent and sensible heat fluxes were greatly influenced by irrigation, indicating that irrigation alters energy partitioning, causing the crop microclimate to evolve. The modification of crop microclimate due to irrigation is more noticeable at short time scales (hourly variations) than for long time scale (daily variations). Among all microclimatic parameters the variation in vapor pressure deficit (VPD) was high in both short and long time scale. The observations also showed that irrigation reduced the canopy temperature in the irrigated field. The modification of temperature and VPD helps to minimize the transpiration loss.

Soil GHG emissions are driven by soil moisture and temperature. As the variation in soil temperature between irrigated and non-irrigated fields at this site was minimal, this study identified soil moisture as the major factor in GHG emission in this region. The difference in soil moisture between irrigated and non-irrigated fields started at the end of June, however, soil $N_2O$ emission was virtually same in both fields in 2012 and 2013 at this time. In both fields, peak $N_2O$ emissions were observed during spring thaw due to abundant soil moisture from snow melt and the presence of fertilizer from fall applications. As most of the spring emission occurred before and within two weeks of seeding, managing the fall fertilizer provides a great opportunity to reduce GHG emission as well as N loss. Previous research has shown that avoiding the application of fertilizer in the fall (Lemke, 2007) and managing the timing of fertilizer applications (Roberts, 2007) are the best methods of reducing $N_2O$ emission. Therefore, proper management of fertilizer application and soil moisture control together will help to minimize GHG emission and maximize crop yield sustainably.

In the agricultural fields of the Canadian Prairies, the prominent GHG is $N_2O$ in terms of amount emitted from the field and global warming potential. Field studies that evaluate mitigation strategies to control $N_2O$ emission from irrigated agricultural fields have a large time and cost requirement. Therefore, a GHG simulation model was used as a viable alternative in this research. The process-based GHG simulation model is able to simulate $N_2O$ emission for different management scenarios. Chapter 4 describes the use of a recently updated GHG simulation model, CDN-DNDC, to simulate soil moisture and temperature, crop yield, crop water and N uptake, and $N_2O$ emission.

The CDN-DNDC model was validated to ensure its suitability for the research site. The validation test showed that the model adequately represented the temporal pattern of soil water, temperature, and soil $N_2O$ emission. However, some discrepancies in the simulated soil water and $N_2O$ emission indicated that the model was limited in its consideration of all processes. The model had insufficient complexity to adequately simulate the springtime soil water, soil temperature, and soil $N_2O$ emission during the thaw period. Overall, this study found good agreement between measured and simulated soil water and soil $N_2O$ emission in 2013 and soil temperature in 2012. As described in chapter 3, irrigation alters the soil temperature but not by much. However, the CDN-DNDC model showed no effect on soil temperature due to irrigation (chapter 4). The sensitivity test indicated that the model was sensitive to porosity, clay fraction, and soil organic carbon. Hydraulic conductivity, a key parameter of soil water movement, was insensitive to soil water and $N_2O$ emission in the CDN-DNDC model.

The validation test showed that the CDN-DNDC model was suitable for this region and for long-term simulations of $N_2O$ emission. Hence, a long-term CDN-DNDC scenario of a wheat-canola crop rotation was developed for irrigated and non-irrigated conditions. The long-term scenario identified that irrigated cropping increased $N_2O$ emission. However, proper management of irrigation (for example - by scheduling irrigations to reduce the number of consecutive days when the soil water is above 60% WFPS) can help to reduce $N_2O$ emission compared to a non-irrigated cropping system and can maximize crop production significantly. In the long-term scenario, the result of simulated $N_2O$ emission in the individual year showed that without control, increasing the amount of input water in the field during the growing season can increase N loss through leaching and can decrease crop production. Irrigation increased crop water and N uptake and promoted uniformity of soil moisture, which ultimately helped to improve crop production and decrease $N_2O$ emission. The Canadian Prairie experiences unexpected weather patterns and variable climate from year to year. Hence, this regional CDN-DNDC model will help to identify the total amount of emission and N loss corresponding to crop yield for different management strategies of irrigation.

## 5.2 CONCLUSIONS

The findings of this two year field experiment have established that irrigation markely influences energy flux partitioning, resulting in a unique microclimate for the irrigated field. A larger amount of the available radiation energy is consumed by evaporating the readily availabe water provided by the irrigation system. In the non-irrigated system, there is more energy available to warm the soil and canopy, resulting in much larger sensible heat exchange with the atmosphere. Although irrigation was responsible for maintaining constently high soil moisture contents through the growing season, the largest GHG emissions were found in the spring thaw period in both fields. Late season GHG emmissions were relatively small in both fields, and only exhibited a very minor influence of irrigation.

The validation results of the research showed that the regional version of the DNDC model (CDN-DNDC) is suitable for use in the study region. However, the model has not been developed to accurately predict spring thaw $N_2O$ emission in the Canadian Prairies. The updated CDN-DNDC model predicted soil moisture persuasively for both non-irrigated and irrigated conditions. The model was also capable of predicting the $N_2O$ emission pattern with irrigation and rainfall with some distinction in emission magnitude. The model predicted that N loss occurred mainly through leaching which had not been measured in the field experiment during irrigation. The simulated soil water and $N_2O$ emissions were sensitive to soil organic carbon, clay fraction, and porosity. A long-term scenario showed that irrigation increases $N_2O$ emission as well as total N loss. However, by considering the yield corresponding to total N loss, the relative $N_2O$ emission was not high. Although the mean annual $N_2O$ emission was higher in the irrigated field than the non-irrigated field, the mean annual nitrous oxide intensity in the irrigated field was actually lower than that in the non-irrigated field. Careful management of irrigation and fertilizer application can increase crop production in a sustainable manner to supply food for a growing population.

## 5.3 RECOMMENDATIONS

Energy fluxes, and crop and soil environments vary throughout the day. Hence, GHG emission will also vary at short time scales due to rapid changes in soil moisture and nutrients. In this research, GHG emission from soil was measured twice a week, sometimes just before or after irrigation, and sometimes a few days before and after irrigation. The measurement of short time fluctuations in GHG emission at different times in a day, and on consecutive days after irrigation is still needed for a clear understanding of soil GHG emission from an irrigated field.

Although surface flux partitioning varied with irrigation, soil GHG emission did not vary significantly with irrigation when maximum fertilizer was applied in the previous year's fall season. Correspondingly, during irrigation periods, there was a small difference in GHG emission between irrigated and non-irrigated conditions. To get a more accurate idea of how much GHG emission increased due to irrigation, similar studies should be carried out for spring fertilizer application rather than fall fertilizer application.

In the field experiment, soil nutrients were measured at the end of the growing season to determine how much nutrient remained in the ground. However, frequent measurements of soil nutrient during the growing season allowed us to correlate changes in soil temperature and water with available soil N at different stages of crop growth in this region. This procedure can help to determine the amount of irrigation water necessary to control GHG emission.

Two years was not long enough to see the crop rotation effect on GHG emission in the irrigated field. As crop rotation is an alternative management practice to reduce GHG emission (Campbell et al. 2014), a multiyear experiment with different crop rotations could suggest how GHG emission could be further decreased.

The validation test in chapter 4 showed that to measure net seasonal emission accurately, it was important to measure soil $N_2O$ emission daily. Therefore, in future, the CDN-DNDC model simulated $N_2O$ emission should be compared with the measured daily $N_2O$ emission, which can be done by using an automated chamber instead of a static chamber. Along with the field experiment, a laboratory experiment should conducted to measure N loss through leaching, because it was found that the CDN-DNDC model-simulated N loss through leaching was higher than the N loss in the form of $N_2O$ emission in the irrigated field. Further study with the CDN-DNDC model should be performed at a regional scale to estimate the total regional $N_2O$ emission for irrigated fields in the Canadian Prairies.

This CDN-DNDC model should be updated for the Canadian Prairie spring thaw period because higher $N_2O$ emission occurred in the spring thaw. When the model can appropriately simulate the Prairie spring thaw, new mitigation options can be applied to reduce spring $N_2O$ emission.

## 5.4 REFERENCES


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# APPENDIX

## Appendix A.1

Soil physicochemical properties in both irrigated and non-irrigated field

Table A.1.1. Soil physical properties of the sites

| Soil feature | Non-irrigated field | Irrigated field |
|---|---|---|
| pH | 7.05 | 7.22 |
| EC | 270 | 664 |
| Bulk density | 1.17 | 1.17 |
| Porosity | 0.56 | 0.56 |
| Soil texture | Loam | |
| VWC at FC | 0.3353 | 0.4242 |
| VWC at PWP | 0.2139 | 0.2519 |

Table A.1.2. VWC and WFPS at FC and PWP based on crop-water-retention curve (VWC=volumetric water content, WFPS= water filled pore space, P= porosity)

| | FC<br>WFPS = (VWC/P) | PWP<br>WFPS = (VWC/P) |
|---|---|---|
| DL 2012 | 0.4531/0.56=0.81 | 0.2457/0.56=0.44 |
| DL 2013 | 0.3353/0.56=0.60 | 0.2139/0.56=0.38 |
| IL 2012 | 0.4984/0.56=0.89 | 0.2468/0.56=0.44 |
| IL 2013 | 0.4242/0.56=0.76 | 0.2519/0.56=0.45 |

**Appendix A.2**

Subsections in the Denitrification-Decomposition (DNDC) model, adeopted from DNDC manual (Version 9.5)

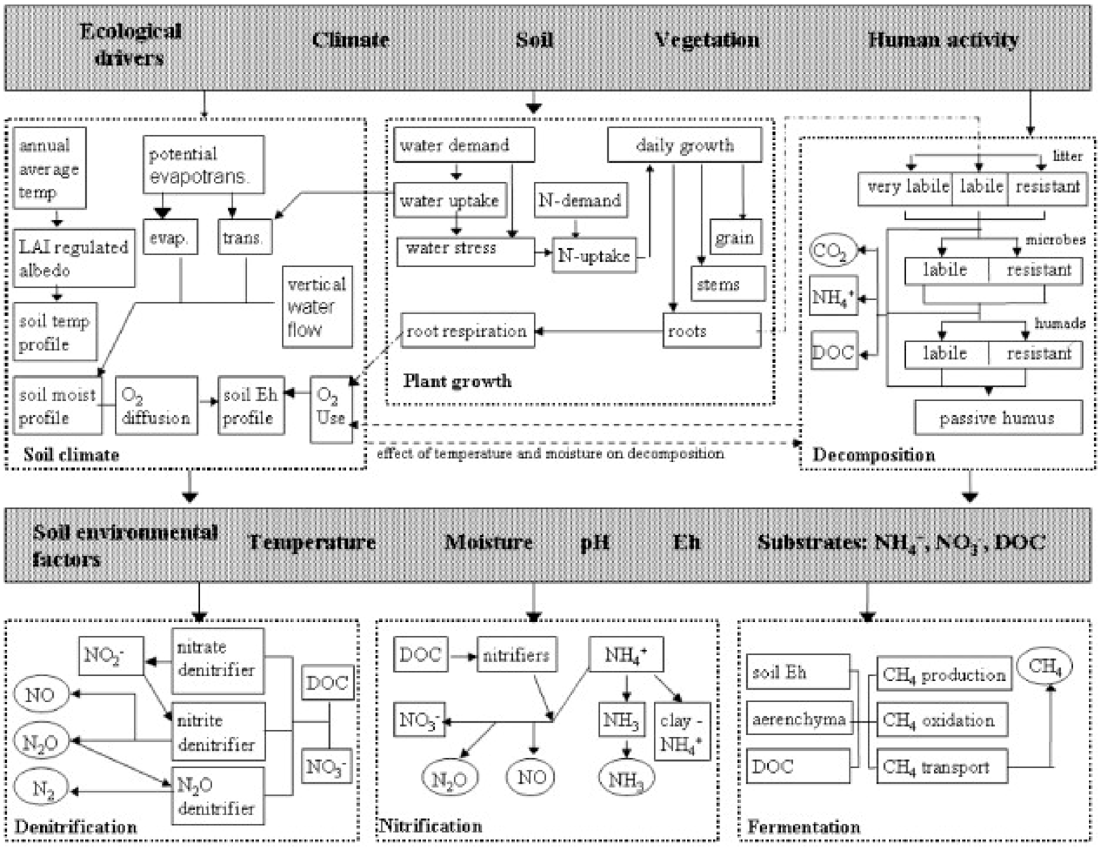


Fig. A.2.1. DNDC model subsections (the bridge between the C and N biogeochemical cycles)

## Appendix A.3

Comparison of observed soil moisture and temperature at 25 cm depth and simulated soil moisture and temperature at 30 cm depth.

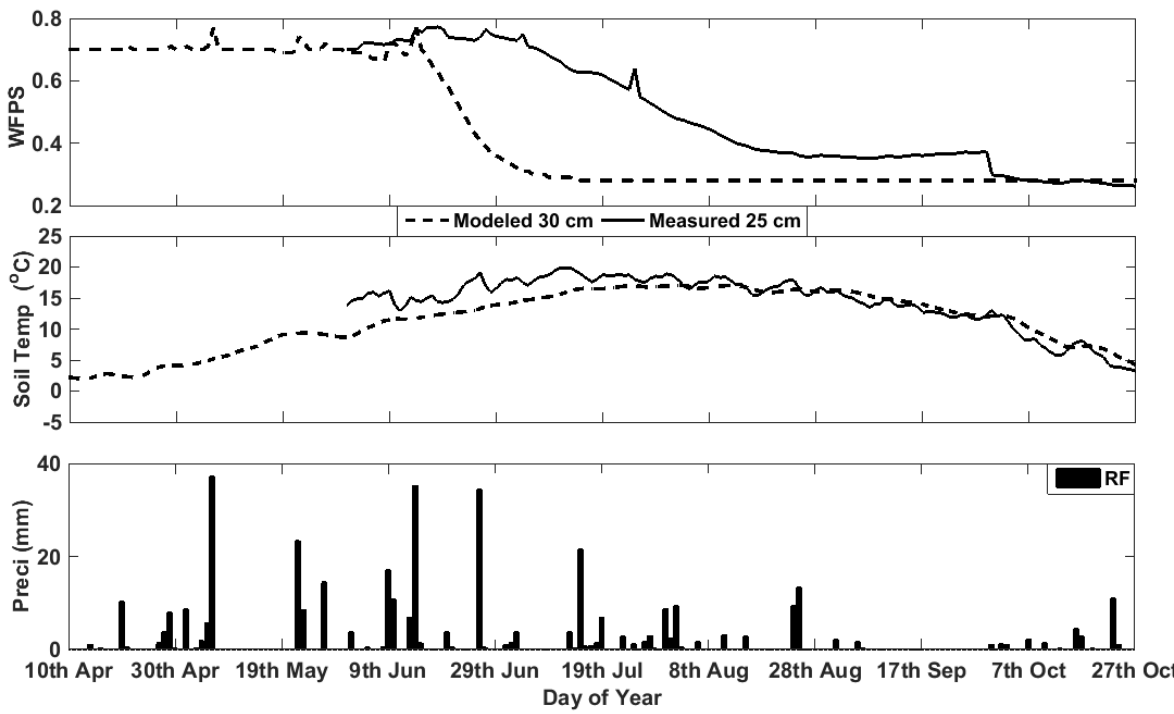


Fig. A.3.1. Measured and simulated soil moisture and temperature in 2012 in non-irrigated field

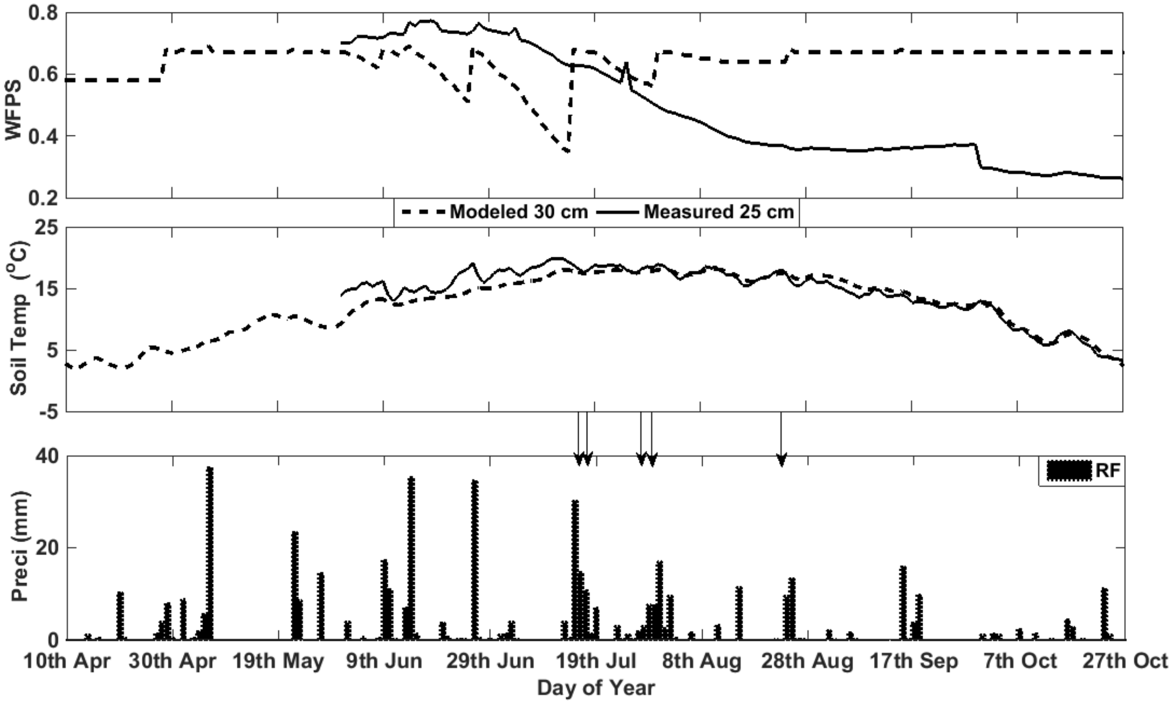


Fig. A.3.2. Measured and simulated soil moisture and temperature in 2012 in irrigated field

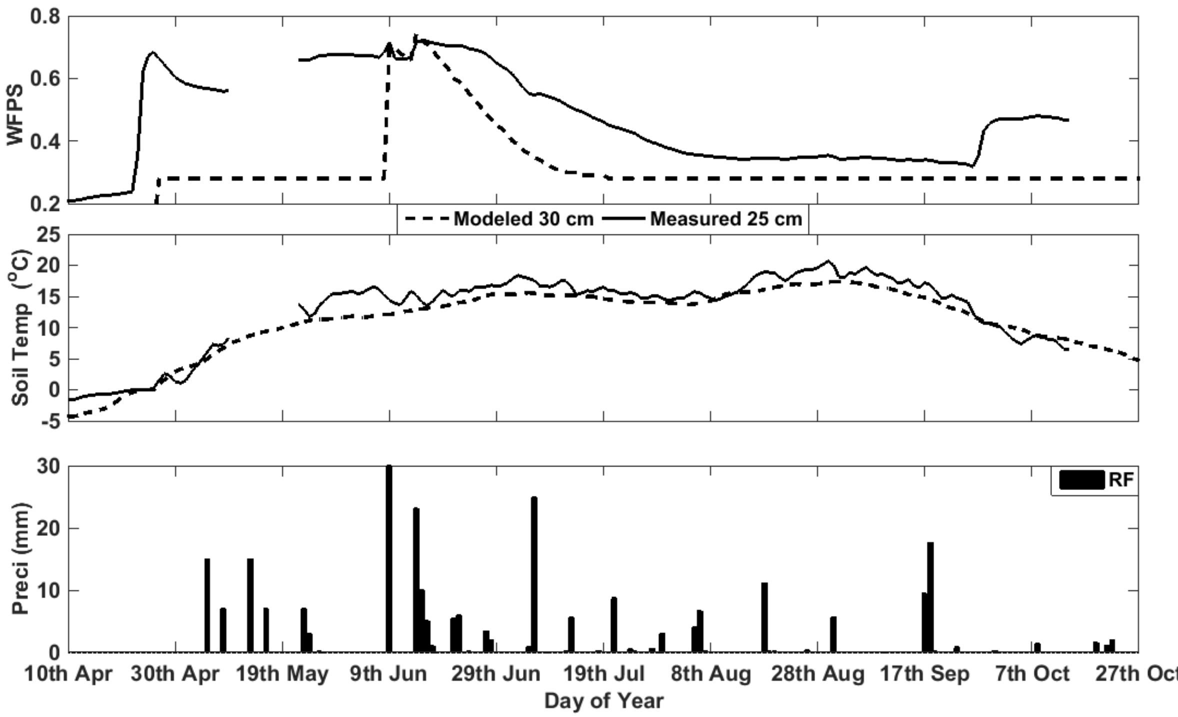


Fig. A.3.3. Measured and simulated soil moisture and temperature in 2013 in non-irrigated field

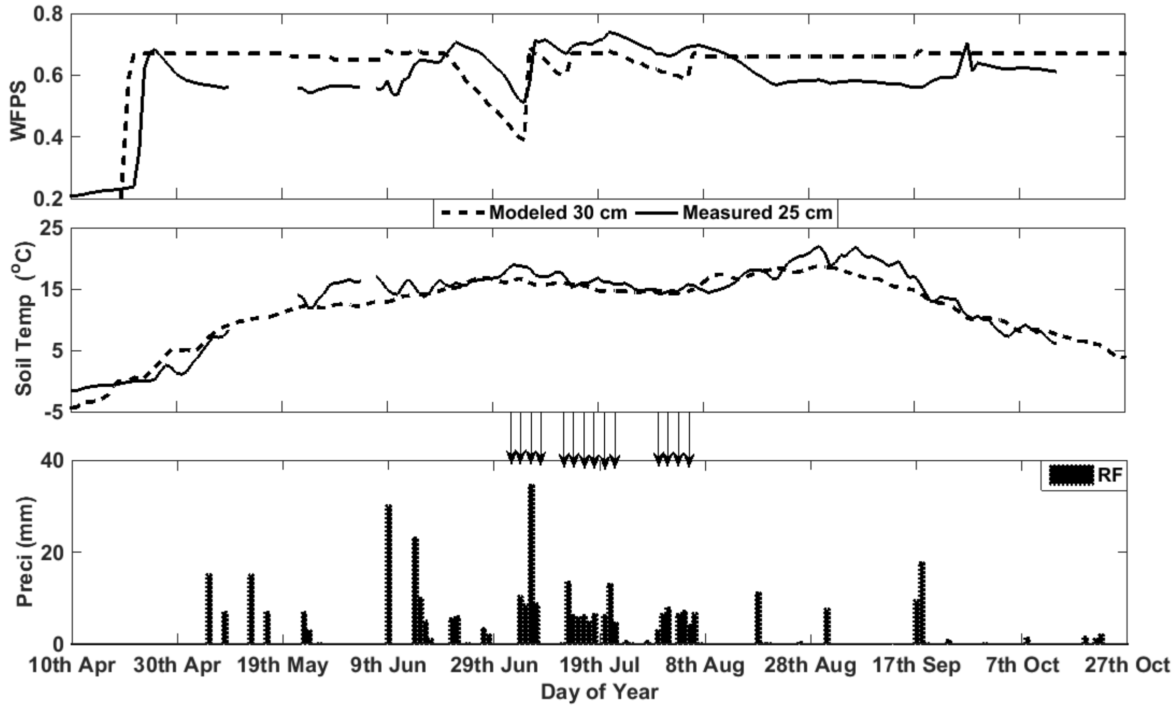


Fig. A.3.4. Measured and simulated soil moisture and temperature in 2013 in irrigated field

**Appendix A.4**

Soil particle size and texture at the experimental sites

Table A.4.1. Soil particle size and texture at the field sites. Values presented are the mean and standard deviation of the number of samples indicated (DL= non-irrigated field, IL12= irrigated field in 2012, IL13= irrigated field in 2013)

| **Sites (name)** | **Depth** | **Sand** | **Silt** | **Clay** | **n** | **Texture** |
|---|---|---|---|---|---|---|
| | — *cm* — | ——— | *%* | ——— | | |
| DL | 0 - 15 | 45.9 ± 3.9 | 33.5 ± 5.0 | 20.6 ± 1.3 | 5 | Loam |
| | 15 - 30 | 50.6 ± 1.6 | 27.8 ± 1.8 | 21.6 ± 1.0 | 4 | Loam |
| | 30 - 60 | 41.5 ± 4.2 | 32.5 ± 3.8 | 26.1 ± 0.5 | 3 | Loam |
| | 60 - 90 | 65.2 ± 10.4 | 17.0 ± 6.7 | 17.8 ± 4.9 | 4 | Sandy Loam |
| | 90 -120 | 69.9 ± 21.4 | 15.8 ± 10.4 | 14.3 ± 11.1 | 3 | Sandy Loam |
| IL12 | 0 - 15 | 33.6 ± 5.8 | 47.7 ± 5.2 | 18.7 ± 1.4 | 4 | Loam |
| | 15 - 30 | 39.4 ± 7.6 | 41.0 ± 8.1 | 19.7 ± 1.5 | 4 | Loam |
| | 30 - 60 | 38.5 ± 3.1 | 36.6 ± 2.6 | 24.9 ± 1.5 | 4 | Loam |
| | 60 - 90 | 55.3 ± 17.0 | 23.5 ± 11.7 | 21.2 ± 5.4 | 4 | Sandy Clay Loam |
| | 90 -120 | 67.6 ± 16.8 | 16.2 ± 11.2 | 16.2 ± 6.0 | 4 | Sandy Loam |
| IL13 | 0 - 15 | 34.7 ± 4.7 | 45.3 ± 4.9 | 20.0 ± 0.9 | 5 | Loam |
| | 15 - 30 | 39.1 ± 4.7 | 40.3 ± 8.1 | 20.6 ± 1.3 | 5 | Loam |
| | 30 - 60 | 49.4 ± 21.5 | 28.9 ± 16.6 | 21.7 ± 5.3 | 3 | Loam |
| | 60 - 90 | 57.3 ± 20.7 | 23.4 ± 12.8 | 19.3 ± 11.9 | 4 | Sandy Loam |
| | 90 -120 | 81.7 | 8.8 | 9.5 | 1 | Loamy Sand |