GREENHOUSE GAS EMISSIONS FROM IRRIGATED CROP PRODUCTION IN THE CANADIAN PRAIRIES

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 Soil Science

University of Saskatchewan

Saskatoon

By

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ABSTRACT

Irrigated agriculture in the Canadian Prairies is in a position to play a prominent role in addressing global food demands imposed by a growing world population. Particularly within Saskatchewan there is potential to see large increases in the number of irrigated hectares, due to the large irrigable land base and supply of freshwater resources. Yet, how this increase will influence the agricultural greenhouse gas (GHG) balance is not well understood. Through the quantification and comparison of GHG emissions from a typical irrigated and dryland cropping system in Saskatchewan, this research aimed to better understand the role of irrigated agriculture on GHG dynamics in this region. A field-scale analysis of irrigated soil conditions and resulting soil greenhouse gas emissions identified that soil N availability was likely the dominant factor influencing soil N_2O emissions from irrigated systems. Soil moisture was also a key factor in soil GHG fluxes, governing seasonal CH_4 uptake and episodic N_2O and CO_2 emissions. The development of system-specific GHG budgets—incorporating on-site GHG sources and sinks—identified electricity as irrigated cropping's largest contributor of global warming potential (GWP). Emissions from soil and diesel-combustion sources were less intensive under irrigated production; yet overall greenhouse gas intensity (GHGI) was greater from irrigated cropping. This research provides a first look into GHG dynamics from irrigated agriculture in Saskatchewan and identifies areas for potential mitigation as irrigated crop production expands in the Province.

ACKNOWLEDGEMENTS

I am grateful for the support and mentorship of my supervisors Drs. Warren Helgason and Richard Farrell. Appreciation is also extended to my advisory committee, Drs. Fran Whalley and Jeff Schoenau, for their valuable input and guidance.

Financial support for this work was provided by Agriculture and Agri-Food Canada through the Agricultural Greenhouse Gases Program. Personal financial support was provided by the Department of Soil Science, the College of Agriculture and Bioresources' Robson Bursary, and the Syngenta Scholarship in Sustainable Agriculture.

My research would not have been possible without the technical support provided by the members of Soil Science 5E19; the SWS Lab group; the staff at Canada-Saskatchewan Irrigation Diversification Center; and our cooperator, Garth Weiterman.

A very special thanks goes out to:

- My field-mates, especially Jannatul Ferdous and Elizabeth Edmondson, for their good company and unwavering patience for gas sampling.

- Warren Helgason, Mike Solohub, Bruce Johnson, and Dell Bayne, for introducing me to instrumentation and for sharing their valuable technical know-how.

- Rich Farrell, Frank Krijnen, and Darin Richman, for their insight and expertise with gas chromatography and greenhouse gas research.

- My father, Don David, for the much appreciated help during fall soil sampling and the numerous shop talks on the practicalities of crop production.

- And to my family and friends for their support and encouragement, I cannot say enough thanks.

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LIST OF ABBREVIATIONS

AIMM	Alberta Irrigation Management Model
AS	ammonium sulfate
CSIDC	Canada-Saskatchewan Irrigation Diversification Center
DL	dryland study site in 2012 and 2013
ET	evapotranspiration
GHG	greenhouse gas
GHGI	greenhouse gas intensity
GWP	global warming potential
HRSW	hard red spring wheat
IR12	irrigated study site in 2012
IR13	irrigated study site in 2013
LDDA	Lake Diefenbaker Development Area
MDCD	minimum detectable concentration difference
MAP	monoammonium phosphate
PET	potential evapotranspiration
PFRA	Prairie Farm Rehabilitation Administration
PMMA	polymethyl methacrylate; acrylic glass
SAFRR	Saskatchewan Agriculture, Food and Rural Revitalization
SIPA	Saskatchewan Irrigation Projects Association
SOM	soil organic matter
SWDA	Southwest Irrigation Development Area
SWE	snow water equivalent

TDH	total dynamic head
UAN	urea ammonium nitrate
VWC	volumetric water content

1. INTRODUCTION

1.1 General Introduction

World population is expected to exceed 9 billion by 2050 (Alexandratos and Bruinsma, 2012). This increase, coupled with greater food energy intake by individuals, will make meeting global food demands increasingly difficult in upcoming years. Agricultural intensification through irrigation is a method of increasing agri-food outputs to help address these demands, provided secure and sustainable water resources are available. Canada's large arable land base and substantial fresh water resources put the country in a prominent position to expand irrigated areas in an effort to strengthen the nation's food security. The Canadian Prairies, in particular, are important producers of agri-food exports and have potential to realize large increases in the number of irrigated hectares. In Saskatchewan, there are large regions of irrigable land where irrigated development has yet to occur (SIPA, 2008). Yet, transitioning from dryland (rainfed) cropping to more intensive managed irrigated cropping has potentially negative environmental impacts through elevated greenhouse gas (GHG) emissions. How the transition will affect the agricultural GHG balance is not well understood, as the GHG levels associated with irrigated agriculture have not been studied in this region.

Irrigated crop production is managed more intensively than rainfed cropping; involving supplemental water applications, and a greater reliance on fertilizer and chemical (herbicides, fungicides, and pesticides) to ensure and safeguard higher crop yields. Under dryland crop production, soils of the Canadian Prairies typically exhibit low levels of emissions with high spatial and temporal variability (Yates et al., 2006a; Rochette et al., 2008; Ellert and Janzen, 2008; Environment Canada, 2010b). However, intensification through irrigation can affect seasonal trends in soil moisture, temperature, and N availability; key factors in the production and evolution of GHG emissions from soil (Linn and Doran, 1984; Bouwman et al., 1993; Dobbie et al., 1999; Dobbie and Smith, 2003). As well as creating potentially favorable conditions for soil emissions, irrigated cropping has costs in terms of GHG emission through increased reliance on energy for pumping water and fueling additional farming operations (Schlesinger, 1999).

1.2 Research Objectives

The main purpose of this research was to explore how agricultural GHG dynamics are influenced by irrigated agriculture in the semi-arid Prairie region by comparing a typical irrigated cropping system to a typical dryland cropping system in Saskatchewan. The specific objectives of these studies were:

- to identify how emissions of soil-derived N_2O, CO_2, and CH_4 are influenced by changes in soil temperature, water status, and N rates brought about by irrigated crop management;
- to identify and quantify the sources and sinks of GHG emissions and construct and compare emission budgets for the contrasting cropping systems;
- to identify areas of focus for GHG mitigation efforts in irrigated cropping systems.

1.3 Organization of the Thesis

The research presented in this thesis is organized in manuscript format. Following this brief introduction, the Literature Review presented in Chapter 2 highlights irrigation in Saskatchewan and discusses GHG emissions trends relevant to irrigated crop production. Chapter 3 presents the first of the two research studies—a field-based investigation of how irrigation influenced soil N_2O, CO_2 and CH_4 fluxes through seasonal changes in soil conditions. The specific objectives of this study were to (a) measure and compare soil greenhouse gas emissions from typical irrigated and dryland cropping systems and to (b) determine how observed soil GHG fluxes corresponded to soil conditions as influenced by irrigated crop management, including annual fertilizer application and seasonal trends in soil temperature and moisture status. In the second study, presented in Chapter 4, cropping system-specific GHG budgets were constructed using the soil emissions obtained in Chapter 3 together with estimates of emissions associated with on-site fuel combustion and energy usage. The objective of this study was to incorporate on-site emission sources and sinks into a greenhouse gas budget for each cropping system and compare systems in terms of greenhouse gas intensity (GHGI)—the quantity of greenhouse gas emissions (expressed as CO_2 equivalents) attributed to the production of one kilogram of crop yield. Following the research studies, Chapter 5—Synthesis 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 in Chapter 6. The document is concluded with a collection of Appendices that include a detailed soil map of the research site (Appendix A), soil characteristics (Appendix B), crop details for the 2012 and 2013 production years (Appendix C), and climate data (Appendix D).

2. BACKGROUND AND REVIEW OF THE LITERATURE

2.1 Irrigation in the Canadian Prairies

2.1.1 Location and climate

The Canadian Prairies are located within the provinces of Alberta, Saskatchewan, and Manitoba and make up the northern portion of the northern Great Plains. Including the Canadian prairie provinces, the northern Great Plains include South Dakota, North Dakota, and Montana; and parts of northeastern Wyoming and northwestern Nebraska (Padbury et al., 2002). The climate of this region is continental, characterized by long, cold winters; short, warm summers with long spells of hot, dry weather; large diurnal temperature changes; frequent strong winds; and highly variable seasonal precipitation ranging from 300 to 500 mm per year, though extreme year-to-year variability is common (Padbury et al., 2002).

Saskatchewan has a relatively dry climate with extreme seasonal variability. The town of Outlook, situated at the center of Saskatchewan's large-scale irrigated cropping region—the Lake Diefenbaker Development Area (LDDA)—receives an average of 338 mm precipitation yearly (Environment Canada, 2013a). Rainfall is the dominant form of precipitation, averaging 260 mm per year. The average 78 mm of snowfall received over the winter months accumulates as snowpack and is an important source of moisture during spring melt. Due to clear skies, most areas of Saskatchewan experience greater than 2000 bright sunshine hours per year (Cote, 2006). Combined with frequent strong winds, the clear skies and low humidity allow for large potential evapotranspiration (PET) that typically exceeds precipitation, leaving a moisture deficit of between 100 and 200 mm per year (Cote, 2006; SIPA, 2008). The temperature extremes and uncertainty of precipitation are the most common crop hazards (Padbury et al., 2002), though the risk of inadequate precipitation can be alleviated with irrigation which allows for greater, more reliable crop yields (Bardak-Meyers, 1996; SIPA, 2008).

2.1.2 History

Prior to the Dominion government's Northwest Irrigation Act of 1894, which regulated the use of water for irrigation, a small number of backflood irrigation systems existed in the Cypress Hills region for the irrigation of hay meadows (Stewart, 2006). The drought of the 1930s motivated substantial irrigation development in the province—spearheaded by the Prairie Farm Rehabilitation Administration (PFRA). Approval of the South Saskatchewan River Project—which would lead to construction of Gardiner Dam and the formation of Lake Diefenbaker—saw the addition of 24,600 ha of irrigated land to provincial totals through the 1950s and 1960s. In the 40 years following the 1967 completion of Gardiner Dam, over 97,000 more hectares of land have been developed for irrigation. Expansion occurred primarily through the formation of a number of irrigation districts around Lake Diefenbaker for the cultivation of cash crops. Due to the substantial water resource provided by Lake Diefenbaker, there is a large capacity for continued irrigation development in the LDDA (SIPA, 2008).

2.1.3 Irrigation development

The majority of irrigation in Saskatchewan is privately managed. Of the 135,000 ha of irrigated crop land, 72% is managed by private irrigators while the remaining 28% is located within the province's 27 irrigation districts (SIPA, 2008). Various types of irrigation are practiced and can be categorized into two general types—surface systems or sprinkler systems.

Saskatchewan's first irrigation schemes—located in the province's southwest—were surface systems used primarily for forage production. Known as backflood systems, these schemes consisted of a series of dams and diversions designed to catch spring runoff and redirect the water into hay meadows. Over one quarter (28%) of Saskatchewan's irrigated area is still managed by backflooding, with another 18% being gravity fed, surface flood systems (SAFRR, 2003). The majority of the province's surface flood systems are located in the South West Irrigation Development Area (SWDA) and are still primarily used in forage production (SIPA, 2008).

The remaining irrigated area in Saskatchewan is managed under sprinkler irrigation (SAFRR, 2003; SIPA, 2008). Common sprinkler systems include wheel-move, high pressure center pivot, and low pressure center pivot systems. The majority of sprinkler systems are located within irrigation districts; however, the use of these systems by private irrigators is also notable. Future irrigation development in the province will be predominantly sprinkler systems (i.e., center pivots) due to the lower water requirements and better suitability for the production of a variety of cash crops. Sprinkler irrigation is also preferable to surface systems due to greater application precision and uniformity—reducing potential for runoff and erosion (SAFRR, 2003). In the short term (less than 15 years) projected expansion and district infill projects will be primarily centered around the LDDA, with additional irrigated hectares added in the SWDA in the long term (SIPA, 2008).

2.1.4 Management of irrigated cropping systems

Irrigation is defined as the practice of applying water to supplement natural precipitation, and is often accompanied by increases in fertilizer rates and cropping intensity (Bardak-Meyers, 1996). Due to the removal of soil moisture limitations, a wider range of crops can be grown with irrigation compared to dryland (rainfed) cropping conditions. In Saskatchewan, common irrigated crops include forages, cereals, oilseeds, pulses, potatoes, and horticultural crops (SAFRR, 2003). A typical irrigated cropping rotation includes a cereal, a pulse, and an oilseed, cropped over a three or four year interval.

Not to be confused with irrigated crop management, “irrigation management” refers to the planning, scheduling, and application of irrigation water. It follows that irrigation management differs with different types of irrigation systems, but the principal goal in sprinkler irrigation is to effectively supplement natural precipitation with irrigation water to achieve high production levels, while using water efficiently and effectively. Center pivot systems are not designed to apply large amounts of water at one time; rather, volumes of 15 to 20 mm are typically applied over a 36 to 48 hour period (Harms, 2011). The goal is to maintain a consistent soil moisture level, replacing water used by the crop over a certain period (i.e., one week) rather than periodically “filling up” the root zone as is practiced with surface (flood) irrigation. Thus, irrigation scheduling is an important aspect of irrigation management—preventing over-watering and minimizing moisture stress (Broner, 2005). Effective irrigation scheduling relies on accurate measurements of soil moisture status and rate of water use by the crop.

Fertilization rates are generally greater in irrigated cropping systems, which is a result of producers targeting maximum yields in the absence of moisture stress. Recommended N rates for irrigated crops range from 100 kg N ha^{-1} for cereals to greater than 175 kg N ha^{-1} for oilseeds (ICDC, 2012). A variety of synthetic fertilizers are used to achieve target N levels, including gaseous anhydrous ammonia; liquid urea ammonium nitrate (UAN); and granular sources such as urea, ammonium sulfate (AS), and monoammonium phosphate (MAP) (though AS and MAP are primarily used for their sulfur and phosphorus content, respectively). Based on fertilizer shipments to agricultural markets in Saskatchewan, the prevalent fertilizers—listed in descending order—are: urea, MAP, UAN, AS, and ammonia (Statistics Canada, 2013). Data specific to irrigated crop production was not available.

The management of irrigated crops is typically more intensive than dryland cropping, involving more chemical applications and a greater degree of soil disturbance. Additional chemical applications are used to minimize yield losses due to disease, pests, or competition from weedy species. Many producers use tillage operations to minimize soil compaction brought about by increased field traffic and irrigation in the case of fine-textured soils. In addition, the management of common irrigated crops like potato and dry bean also require a greater degree of tillage than cereals or oilseeds.

2.2 Agricultural Greenhouse Gases from Irrigated Crop Management

Greenhouse gases are atmospheric gases that absorb and emit infrared radiation, causing warming of the atmosphere and the Earth's surface—a process known as the greenhouse effect. Each GHG differs in atmospheric concentration, radiative forcing, and residence time in the atmosphere, thus contributing to warming differently. Global warming potential (GWP) is a concept used to compare the relative effects of GHG sources and sinks by converting all gases to the equivalent amount of CO_2 required to elicit a comparable warming effect over a specific period of time (typically 100 years). By convention, CO_2-equivalents are the metric used for GWP measurements (IPCC, 1996; Robertson and Grace, 2004).

Although many of these gases are produced by and emitted naturally, the magnitude of emissions is often intensified by human activity. Agriculture contributes to global warming through the fluxes of nitrous oxide (N_2O), methane (CH_4), and carbon dioxide (CO_2). The main sources of these gases are enteric fermentation (animal digestion; CH_4), manure management (N_2O and CH_4), and soils (N_2O and CO_2) (Paustian et al., 2006; Coad, 2011). Agriculture is responsible for only about 8% of Canada's total GHG emissions, however, it accounts for 23% of all CH_4 emissions and 72% of all N_2O emissions (Environment Canada, 2013b).

Compared to conventional dryland (rainfed) cropping systems, irrigated cropping systems have potential to contribute greater greenhouse gas emissions due to changes in management. Irrigated cropping typically involves more intensive fertilizer regimes, greater soil disturbance, and greater frequency of high soil moisture conditions—all of which may promote the production and evolution of greenhouse gases (Roberts and Chan, 1990; Liebig et al., 2005; Smith et al., 2008; Flynn and Smith, 2010; Sainju et al., 2012; Trost et al., 2013; Baron and Tenuta, 2014; Farrell and David, 2014). Yet, the greater productivity realized through irrigation generally results in higher biomass returns with potential to increase C sequestration and potentially offset other GHG emissions.

In Canada, field studies comparing greenhouse gas dynamics under irrigated and non-irrigated cropping systems are lacking (Environment Canada, 2010b). Very little research has been conducted in the semi-arid Prairie region, where large-scale center pivot irrigation systems are common. Information from western Canadian Prairies is extremely limited, while a small number of publications highlight studies carried out in Alberta (Hao et al., 2001; Ellert and Janzen, 2008). The bulk of the information regarding greenhouse gas emissions from intensive irrigated crop production has been gained from studies conducted elsewhere in the northern Great Plains and Colorado (Guenzi et al., 1994; Entry et al., 2002; Amos et al., 2005; Mosier et al., 2005, 2006; Liu et al., 2006; Halvorson et al., 2008, 2010a, 2010b, 2011; Alluvione et al., 2009; Sainju et al., 2010, 2012; Halvorson and Jantalia, 2011; Halvorson and Del Grosso, 2012). Although there are climatic differences across this region, irrigated cropping strategies are similar, therefore, results can be useful in approximating the effect of irrigation on greenhouse gas dynamics.

2.2.1 Nitrous oxide

In terms of global warming potential, N_2O is the most potent of the three agricultural greenhouse gases, with a GWP 298 times that of CO_2 (on a 100 year time-scale) and an atmospheric residence time of 120 years (Forster et al., 2007). Production agriculture is the principle source of anthropogenic N_2O emissions (Mosier et al., 2005; Rochette et al., 2008; Environment Canada, 2013b) and cropped soils are the single greatest contributor of N_2O

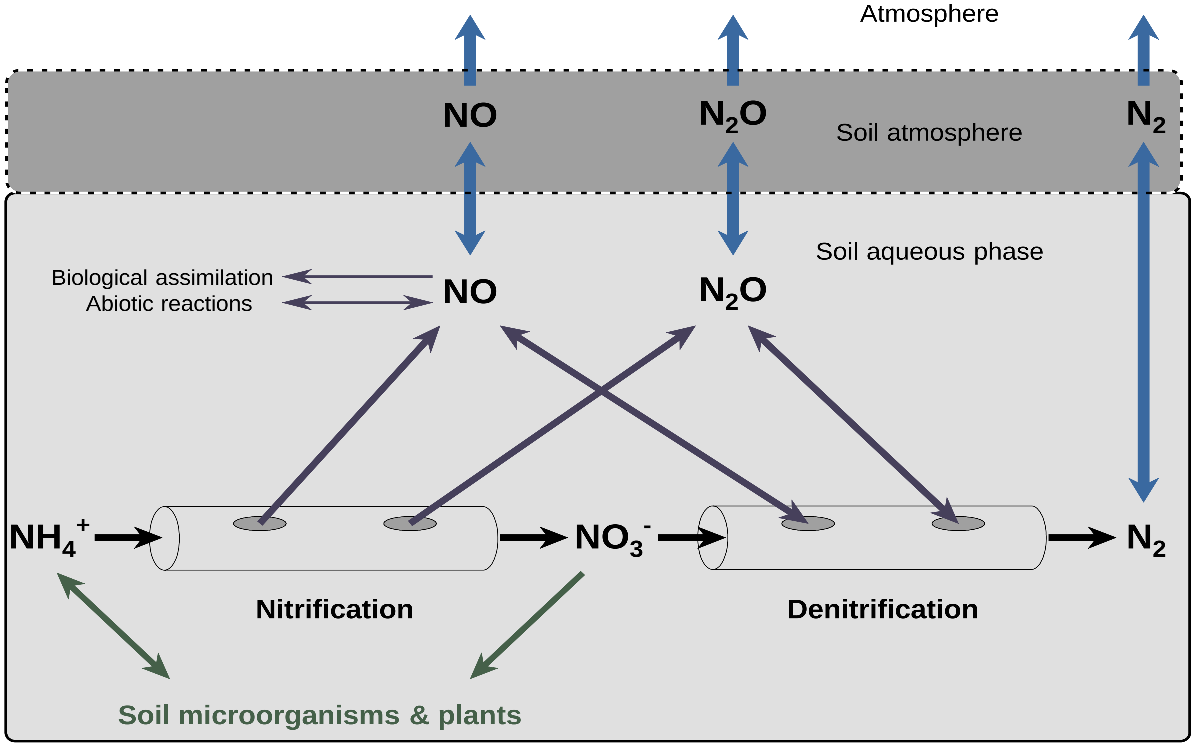

Figure 2.1. Microbially mediated pathways for the production/emission of N_2O from soils. (Davidson et al., 2000, as adapted by Farrell and David, 2014)

(Rochette et al., 2008; Environment Canada, 2013b). Nitrous oxide is produced within soils via the microbially mediated processes of nitrification and denitrification (Figure 2.1). Nitrifying bacteria are active under aerobic conditions and produce N_2O during the oxidation of ammonium (NH_4^+) to nitrate (NO_3^-). During denitrification, N_2O is produced as an intermediate during the reduction of NO_3^- under anaerobic conditions. As a result, N_2O emissions are typically associated with factors that influence microbial activity, including N availability (Bouwman, 1996; Dobbie et al., 1999), soil water content (Linn and Doran, 1984; Corre et al., 1996; Dobbie et al., 1999) and soil temperature (Dobbie and Smith, 2003). Considering that these factors are spatially and temporally variable, it is not surprising that nitrous oxide emissions from agricultural soils are inherently variable in space and time (Folorunso and Rolston, 1984; Sexstone, 1985; Parkin, 1987; Yates et al., 2007). Indeed, soil N_2O emission patterns are often characterized by small areas ('hot-spots') and brief periods ('hot moments') that account for a high percentage of the total emissions (Parkin, 1987; Groffman et al., 2009; Braker and Conrad, 2011; Butterbach-Bahl et al., 2013).

The magnitude of soil-emitted nitrous oxide is strongly governed by prevailing soil-water regimes, and in the Canadian semi-arid prairies direct emissions of N_2O tend respond to increasing levels of N fertilizer in a linear but frequently non-significant fashion (Rochette et al., 2008). However, large emissions have been observed in situations where fertilizer rates are in excess of crop requirements and soil moisture is not limiting (Izaurralde et al., 2004). This implies that emissions from irrigated soils—where moisture deficiencies are minimized by the application of irrigation water—may be a large contributor nitrous oxide in the semi-arid prairies. Irrigation is known to increase potential for N_2O emissions by increasing soil microbial activity and reducing soil oxygen status (Jambert et al., 1997).

In both irrigated and dryland cropping systems in the northern Great Plains, N_2O emissions exhibit seasonal trends. The greatest N_2O efflux is often observed in the spring during snowmelt and thawing of the soil (Nyborg et al., 1997; Lemke et al., 1998; Hao et al., 2001; Mosier et al., 2006; Dusenbury et al., 2008; Rochette et al., 2008; Ellert and Janzen, 2008; Liebig et al., 2010; Risk et al., 2013), yet the magnitude of these fluxes are smaller than those observed in Eastern Canada (Rochette et al., 2008). A study in the Parkland Region of Alberta observed that between 16% and 60% of the total estimated growing season N_2O losses may occur during spring thaw (Lemke et al., 1998). Following thaw, additional emission peaks are observed following rainfall (Dobbie et al., 1999) and/or irrigation events (Jabro et al., 2008; Sainju et al., 2012), likely due to elevated soil water content (Linn and Doran, 1984; Corre et al., 1996). The magnitude of precipitation-induced emission peaks are largest early in the growing season and diminish through the growing season, likely due to the depletion of available soil N by crop growth (Guenzi et al., 1994; Hao et al., 2001; CSIDC, 2013). Emissions during the fall season are typically very low due to reduced soil temperature, low soil N availability, and low soil water content (Liebig et al., 2005; Dusenbury et al., 2008; Sainju et al., 2012). During the winter months, N losses from soil are considered to be negligible under the snow covered, frozen soil conditions common to the Canadian Prairies (Malhi et al., 2001).

For dryland cropping systems in the Prairie region, observations suggest that N_2O production is not limited by mineral N availability but by low denitrification activity under well-aerated soil conditions (Helgason et al., 2005; Rochette et al., 2008; Environment Canada, 2010b). Irrigation creates more frequent saturated soil conditions, which may increase the frequency of anoxic soil conditions that favor denitrification (Figure 2.2). Thus, irrigation may stimulate denitrification activity similar to that observed at low-lying, moisture rich landscape positions, as observed in hummocky landscapes (Corre et al., 1996; Izaurralde et al., 2004).

2.2.2 Methane

The agriculture sector is a notable contributor of methane emissions on both a national (Environment Canada, 2013b) and global scale (Robertson and Grace, 2004). Major methane sources include ruminant digestion, animal waste, and flooded soils of rice paddies (Bronson and Mosier, 1994; Liebig et al., 2012). In semi-arid regions with well aerated soils—like those of the Canadian Prairies—CH_4 fluxes are generally small, with agricultural soils being either minor sources or minor sinks of atmospheric CH_4 (Mosier et al., 2006). Yet, the large GWP of CH_4, 25 times that of CO_2 on a 100-year time scale (Forster et al., 2007), means that even small fluxes can have a notable effect on the overall GHG source/sink of a cropping system.

Methane fluxes are primarily governed by soil oxygen status, which affects the balance between microbial processes of methanogenesis (production) and CH_4 oxidation (consumption). Under anaerobic soil conditions, CH_4 is produced through the decomposition of organic material by methanogenic microbes. Conversely, under well aerated conditions, CH_4 consumption occurs through aerobic oxidation of CH_4 by methanotrophic microorganisms. Soils of the semi-arid

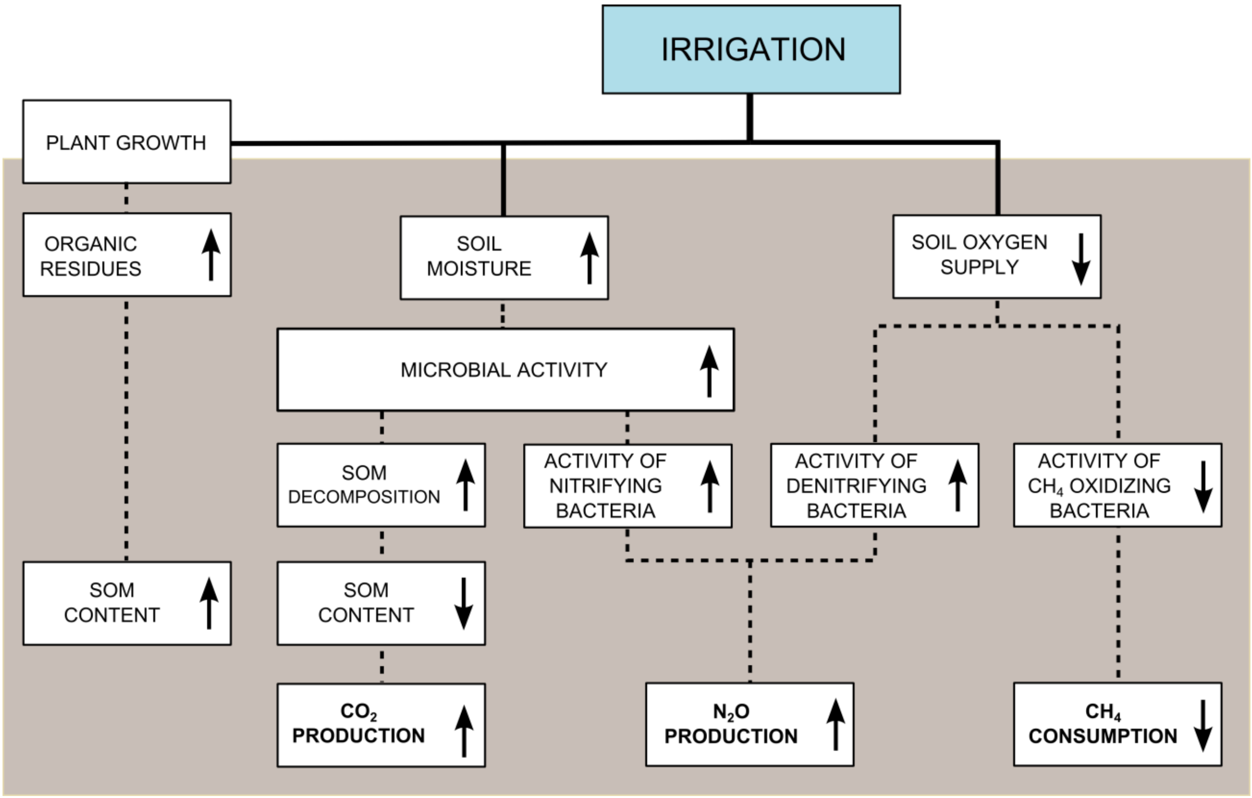

Figure 2.2. Effects of irrigation on soil processes relating to greenhouse gas emissions dynamics. Direct effects of irrigation are represented by solid connections and indirect effects are represented by dashed connections. The direction of effect (increase or decrease) is represented by the arrow within each process box. (Adapted from Trost et al., 2013)

prairies, in their natural state, act as sinks for atmospheric CH_4, due to characteristic dry and well-aerated soil conditions (Liebig et al., 2005; Fowler et al., 2009). However, natural rates of CH_4 consumption are reduced by agricultural conversion (Robertson and Grace, 2004) and cropped soils typically vary from being small sinks to minor emitters of atmospheric CH_4—with greater uptake occurring under drier soil conditions (Mosier et al., 2006; Liebig et al., 2010).

Although not well studied, CH_4 uptake by irrigated (non-flooded) agricultural soils of the northern Great Plains may be reduced due to elevated soil moisture levels and a greater reliance on N fertilizers. Irrigation application, intended to reduce water-related crop stress by increasing the plant available water content of the soil, may create temporarily anaerobic soil conditions, which in turn may increase the CH_4 'source potential' of agricultural soils. In addition, use of N fertilizers has been found to reduce CH_4 consumption by soils, in some cases, due to the inhibitory effect of NH_4^+ on CH_4 oxidation (Bronson and Mosier, 1994), however, various studies in the northern Great Plains have found no measurable influence of N fertilizer on atmospheric CH_4 uptake by irrigated soils (Mosier et al., 2006; Liu et al., 2006; Ellert and Janzen, 2008; Halvorson et al., 2011). In the semi-arid Prairie region, irrigated soils appear to retain their sink capacity for atmospheric CH_4 (Ellert and Janzen, 2008) but to a lesser extent than their dryland counterparts (Sainju et al., 2012)—likely due to elevated soil water content (Kessavalou and Mosier, 1998).

2.2.3 Carbon dioxide

Carbon dioxide is the most abundant anthropogenic greenhouse gas in the Earth's atmosphere (Forster et al., 2007). Soils can act as source or sink for CO_2—with CO_2 production resulting primarily from root respiration and the aerobic decomposition of soil organic matter by microbial communities (Sheppard et al., 1994; Buchmann, 2000; Liebig et al., 2012). Carbon sequestration in soils represents a sink for CO_2 and occurs through the incorporation of residual crop tissues (rich in C fixed from atmospheric CO_2 via photosynthesis) into long-term organic matter pools. The balance between C input (crop residue) and output (biomass harvest, soil and root respiration) dictates whether cropped soils act as a source or a sink for CO_2. Other agricultural contributions include fuel combustion; land use changes; and biomass destruction through harvest, natural decay, or burning (Liebig et al., 2012).

As with soil N_2O emissions, soil CO_2 emissions are facilitated by microbial populations; thus, they are dependent on factors such as soil temperature, moisture, and substrate availability (Jabro et al., 2008). Crop management activities such as tillage, irrigation, and fertilizer application can influence these factors and ultimately CO_2 emissions. Tillage stimulates soil respiration and CO_2 efflux by increasing C substrate availability through the physical disturbance of soil aggregates (Roberts and Chan, 1990; Ellert and Janzen, 2008). However, tillage also promotes drying of the disturbed soil, creating less favorable conditions for microbial activity (Curtin et al., 2000). Irrigation and rainfall increase soil moisture status, promoting soil CO_2 evolution via decomposition of soil organic matter (SOM) through enhanced microbial activity (Figure 2.2). In a single year comparison in North Dakota, Jabro et al. (2008) measured slightly greater CO_2 emissions (10% or 7 Mg CO_2 ha^{-1} y^{-1}) from irrigated barley compared to dryland treatments—observing that CO_2 efflux increased linearly with soil volumetric water content (VWC). This positive relationship between CO_2 flux and soil water content has also been observed in other studies (Amos et al., 2005; Sainju et al., 2010, 2012). In some cases, however, CO_2 fluxes varied with irrigation only at shorter scales and is not significantly different over the growing season across multiple years (Sainju et al., 2012). The application of N fertilizer—which promotes biomass production and greater residue returns—may enhance C sequestration or CO_2 production depending on whether residue returns are stored or quickly decomposed (Halvorson et al., 2002).

Due to the complex nature of C cycling, accurately quantifying soil CO_2 dynamics is difficult (Hanson et al., 2000). Soil organic matter levels reflect the long-term balance between additions and losses of C (Follett, 2001); thus, assessing changes in the stable stocks of C (i.e., soil organic matter pools) is a common approach to evaluating C fluxes from cropping systems (Halvorson et al., 2002; Liebig et al., 2005, 2012; Gillabel et al., 2007; Trost et al., 2013).

2.2.4 Soil organic matter

Soils are the largest C pool in terrestrial ecosystems, with over 1.55 x 10^{15} kg C stored as SOM globally (Follett, 2001). The organic C status of agricultural soils reflects the long term balance between C additions and losses of that system. Cropping systems that retain residual biomass have potential to store C in SOM, however the degree of storage is dependent on a number of factors such as climate, soil texture, degree of soil disturbance, and soil C status prior to cultivation (Liebig et al., 2005). Cropping systems that realize SOM gains typically involve minimal soil disturbance and high biomass returns—from efficient use of fertilizers and cropping with high-biomass yielding crops. Organic matter loss is enhanced by soil disturbance, which promotes decomposition (Roberts and Chan, 1990; Follett, 2001; Liebig et al., 2005). Consequently, low-till or zero-till crop management can reduce SOM losses and promote C sequestration.

Highly productive irrigated cropping systems can fix large quantities of atmospheric CO_2 in plant biomass and, through returns of residual biomass, have potential to increase C storage in SOM. Various sources have noted the potential for SOM enhancement through irrigation expansion (Follett, 2001; Smith et al., 2008; Flynn and Smith, 2010). A review of irrigation worldwide, concluded that irrigated cropping systems in arid and semi-arid regions typically realize SOM increases in of 11 to 35% compared to dryland systems, but is highly dependent on climate and initial SOM content (Trost et al., 2013). In cases where native soil C is very low, irrigation is effective in increasing total C content (Lueking and Schepers, 1985). In a southwest Nebraska study, C storage under irrigation was 25% greater than dryland treatments over 33 years of irrigation (Gillabel et al., 2007). Estimates of SOM accumulation resulting from irrigation in the northern Great Plains region of USA range from 0.32 to 0.67 Mg C ha^{-1} y^{-1} (Eve et al., 2002). Similarly, Liebig et al. (2005) report average SOM increases of 0.79 ± 0.75 Mg C ha^{-1} y^{-1} over 5 studies in Colorado, Nebraska, and Alberta. A Saskatchewan study investigating the long-term effects of irrigated crop management on soil properties found no change in SOM, in spite of consistently greater C inputs from residual crop biomass (Bardak-Meyers, 1996). The lack of SOM increase was attributed to rapid turnover, as young, labile organic matter additions act as a substrate for microorganisms. Similarly, Gillabel et al. (2007) reported 50% greater C turnover from irrigated treatments compared to dryland treatments and native vegetation reference plots in southwest Nebraska.

3. EFFECTS OF IRRIGATED CROP MANAGEMENT ON SOIL GREENHOUSE GAS EMISSIONS

3.1 Preface

Greenhouse gas emissions from agricultural soils in the Canadian Prairie region are generally low and, due to dry, well aerated soil conditions, can be quite variable (Rochette et al., 2008; Environment Canada, 2010b). Compared to dryland (rainfed) crop production, irrigated cropping has potential to contribute greater quantities of soil-derived nitrous oxide (N_2O), carbon dioxide (CO_2), and methane (CH_4) emissions as producers target higher yields by removing soil moisture limitations and applying greater amounts of N fertilizers. However, the actual GHG dynamics from irrigated soils in this region are not well understood as there have been few field-based studies in the semi-arid prairies of western Canada. The goal of this study was to identify how emissions of soil-derived N_2O, CO_2, and CH_4 are influenced by changes in soil temperature, water status, and N rates brought about by irrigated crop management. This was achieved through continuous, in-situ monitoring of soil conditions and chamber-based measurements of soil GHG flux.

3.2 Introduction

The production of greenhouse gases (GHG) in soils is a result of microbial processes; thus, a change in crop management that promotes microbial community activity can result in elevated levels of GHG emissions from these systems. In addition to water applications, irrigated crop management involves the application of plant available nutrients (especially N) at greater rates than dryland cropping—altering seasonal soil temperature, moisture, and fertility—creating potential for greater emissions of soil N_2O, CO_2, and CH_4.

Under dryland conditions in the northern Great Plains, N_2O emissions from agricultural cropping typically follows a seasonal event-based/background emission pattern—with persistent, low-magnitude baseline emissions punctuated by episodic, high-emission events (Brumme et al., 1999; Yates et al., 2006a, 2006b). The greatest N_2O efflux is usually observed in the spring during snowmelt and corresponding thawing of the soil (Nyborg et al., 1997; Lemke et al., 1998; Hao et al., 2001; Mosier et al., 2006; Dusenbury et al., 2008; Rochette et al., 2008; Ellert and Janzen, 2008; Liebig et al., 2010; Risk et al., 2013). Up to 60% of the total estimated growing season N_2O losses may occur during spring thaw events (Lemke et al., 1998). Large emissions are also associated with the first precipitation/irrigation event(s) following spring fertilization (Guenzi et al., 1994; Wagner-Riddle et al., 1996; Liebig et al., 2005; Dusenbury et al., 2008; Halvorson and Del Grosso, 2012). The magnitude of precipitation-induced emission peaks are largest early in the growing season and diminish throughout the year (Dobbie et al., 1999; Jabro et al., 2008; Sainju et al., 2012), with the low-level, background emissions pattern dominating as crop growth depletes the pool of available N (Guenzi et al., 1994; Hao et al., 2001). Indeed, low N_2O emissions occurring during the fall season are likely a result of minimal soil N availability, and low soil temperature and water content (Liebig et al., 2005; Dusenbury et al., 2008; Sainju et al., 2012). Irrigated crop management increases the potential for N_2O emissions though the combination of elevated seasonal soil moisture and high N fertilizer rates (contributing to high soil N availability) (Bouwman, 1996; Ellert and Janzen, 2008).

Cropping systems exchange large quantities of C through photosynthesis and decomposition. Highly productive irrigated cropping systems fix large quantities of atmospheric CO_2 in plant biomass that, when returned to the system as crop residues, create potential for C sequestration through soil organic matter (SOM) accumulation (Follett, 2001; Smith et al., 2008; Flynn and Smith, 2010). However, realizing this potential depends on the rate of microbial decomposition which, as a result of elevated seasonal soil moisture levels, may be more intensive in irrigated systems (Jabro et al., 2008; Trost et al., 2013). As decomposition activity increases, SOM is lost, reducing the pool of stored soil C by releasing it to the atmosphere as CO_2. A Saskatchewan study that investigated the effects of long-term irrigated crop management on soil properties found that, in spite of consistently high residue returns, the total organic matter content of irrigated soils did not increase due to rapid organic matter cycling (Bardak-Meyers, 1996).

In the semi-arid prairies, soils are natural sinks for atmospheric CH_4 due to oxidation processes facilitated by methanotrophic microbes under aerobic soil conditions (Liebig et al., 2005; Fowler et al., 2009). However, natural rates of soil CH_4 consumption are reduced by agricultural conversion of grasslands (Robertson and Grace, 2004; Fowler et al., 2009). As a result, agricultural soils can vary from being small sinks to minor sources of atmospheric CH_4, with greater uptake occurring under drier soil conditions (Mosier et al., 2006; Liebig et al., 2010). It follows that, with an increase in seasonal soil moisture, a soils natural capacity for CH_4 uptake can be diminished (Liu et al., 2006; Sainju et al., 2012). As well, some studies have identified a relationship between a reduction in CH_4 oxidation capacity and N fertilization (Bronson and Mosier, 1994; Sainju et al., 2012), likely due to competition between ammonia and methane oxidizing microbial communities (Hütsch et al., 1993). Consequently, the greater rates of N-fertilizer used in irrigated systems may mean that these systems are also characterized by lower methane uptake. However, several field scale studies in the northern Great Plains region have failed to demonstrate a reduction in CH_4 oxidation as a result of higher N fertilizer applications (Amos et al., 2005; Mosier et al., 2006; Alluvione et al., 2009; Liebig et al., 2012). Nevertheless, there remains a need for a more comprehensive understanding of how irrigation affects CH_4 uptake in soils of the semi-arid prairies.

Factors driving GHG emissions from dryland cropping systems in the Canadian Prairies are relatively well studied; however, the question remains: how are GHG dynamics altered by irrigated cropping conditions in this region? To address this question, the present study was developed to (a) measure and compare soil greenhouse gas emissions from an irrigated cropping system and a dryland cropping system typical of Saskatchewan and (b) determine how observed soil GHG fluxes correspond to soil conditions influenced by irrigated crop management including: annual fertilizer application rate, seasonal soil moisture trends, and seasonal temperature trends.

3.3 Materials and Methods

3.3.1 Site description

Situated within the South Saskatchewan River Irrigation District, the study area was located approximately 75 km south of the city of Saskatoon and 15 km north of the town of Outlook. The area consisted of three adjacent sites located in Township 31, Range 7, west of the third meridian, that included two irrigated quarter sections [i.e.. the northwest quarter of section 16 (IR12: 51°39’34”N, 106°56’31”W) and the southeast quarter of section 16 (IR13:

51°39’09”N, 106°55’51”W)] and one quarter section [i.e.. the northeast quarter of Section 8 (DL: 51°38’41”N, 106°57’16”W)] managed under dryland production (Figure 3.1). Irrigation was applied using a low pressure center pivot systems. Prior to this study, all three fields were managed by the same cooperator for eight cropping seasons.

Broadly speaking, soils of the study area are classified as Orthic Dark Brown Chernozems and the entire site has a slope classification of 2—very gently undulating with slopes of 0.5–2%. A detailed soil survey was conducted in the fall of 2011 to identify comparable areas within each field in which to establish the gas sampling transects (Appendix A). Based on this survey, the sampling transects in DL and IR12 were located in areas classified as being members of the Bradwell soil association; the sampling transect in IR13 was in an area classified as being a member of the Asquith association. Bradwell soils vary in texture from fine sandy loams to loam and contain >15% clay and >45% sand, while those of the Asquith association are restricted to sandy deposits with < 15% clay content. Although the fields differed in classification, particle size analysis of the samples—performed using a modified pipette method (Indorante, 1990)—identified the texture of the surface soil (top 30 cm) of all three sampling transects as loam textured (Appendix B.1). Average bulk densities (and standard deviation) within the tillage layer of each sampling transect were measured at 1.17 ± 0.13 Mg m^{-3} at the DL field, 1.17 ± 0.11 Mg m^{-3} at the IR12 field, and 1.18 ± 0.11 Mg m^{-3} at the IR13 field, with no significant difference among the three fields (Appendix B.2).

The DL site was planted to a wheat-canola cropping rotation, while IR12 and IR13 were managed under a wheat-dry bean-canola rotation with different phases of the rotation present in each year. During the 2012 cropping season, both IR12 and DL were in the wheat (*Triticum aestivum* L.) phase of the rotations while IR13 was in the dry bean (*Phaseolus vulgarus*) phase of the rotation (Table 3.1). Dry bean is not grown under dryland conditions, thus the IR13 site was not sampled in 2012. Likewise, site IR12 was not sampled during the 2013 cropping season; however, the site was sampled during the period from snowmelt to seeding in the spring. In 2013, sites IR13 and DL were sampled when the fields were cropped to canola (*Brassica napus*).

Fertilizer applications occurred twice per year, during late fall and in the spring. Anhydrous ammonia (NH_3) was applied in the fall, and comprised the bulk of the target N fertilizer rate for the following growing season’s crop (see Tables 3.2 and 3.3 for rates). Granular

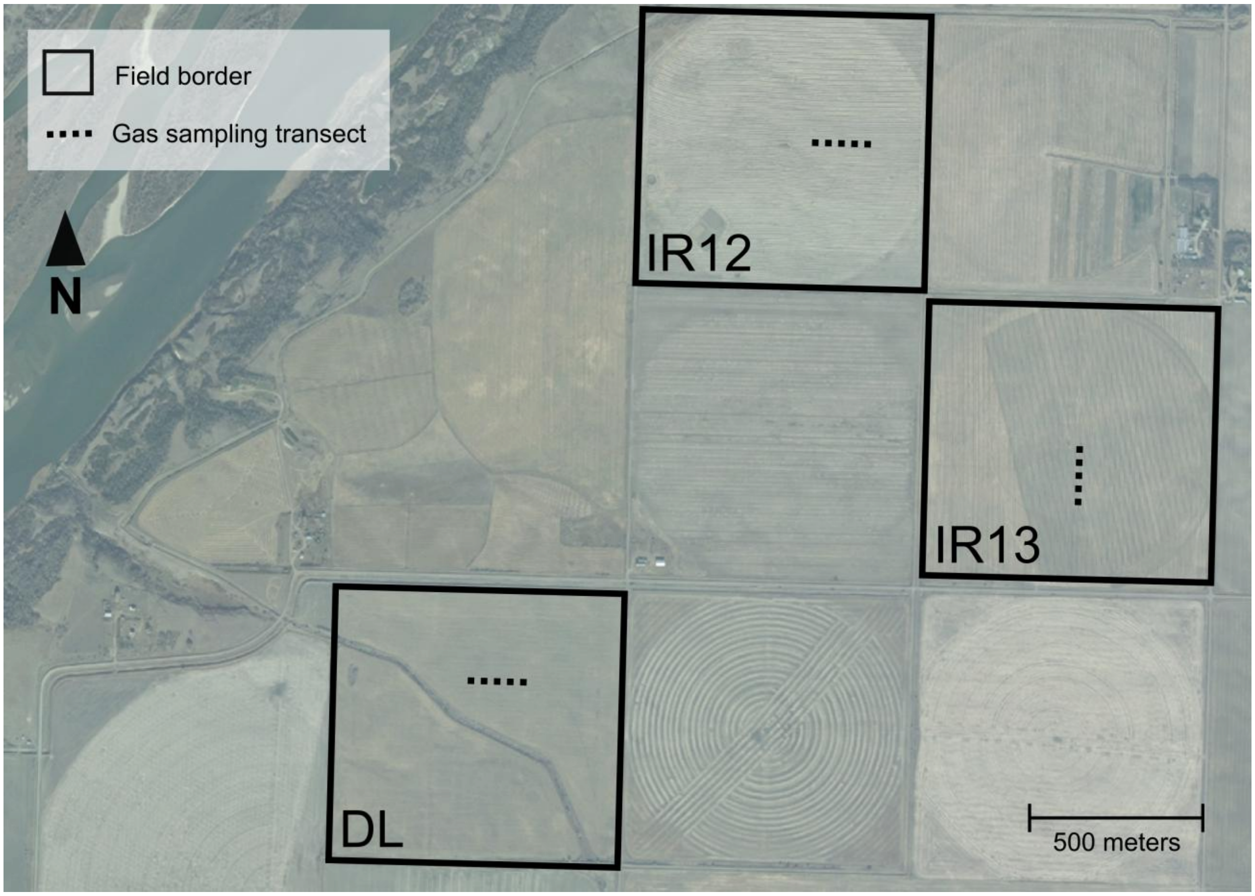

Figure 3.1 An aerial view of the study site. The northwest quarter of section 16-31-07-W3 (IR12; 51°39'34"N, 106°56'31"W) and the southeast quarter of section 16-31-07-W3 (IR13; 51°39'09"N, 106°55'51"W) are managed under irrigated crop production. The dryland study site is located at the northeast quarter of section 08-31-07-W3 (DL, 51°38'41"N, 106°57'16"W). Photo credit: FlySask (2011).

Table 3.1 Crops planted at each site during the two year study. The italicized crops (and associated sites) were investigated for each growing season.

	2012	**2013**
DL	*Wheat*	*Canola*
IR12	*Wheat*	Dry bean
IR13	Dry bean	*Canola*

fertilizers were applied at seeding to meet the remaining fertility requirements (including N, P, K, and S). Irrigation scheduling occurred at the discretion of the cooperator using the Alberta Irrigation Management Model (AIMM), which estimates irrigation requirements based on soil water content, potential evapotranspiration, and crop water requirements.

The sites were instrumented and all measurements initiated immediately after seeding in the spring of 2012; and concluding in the fall of 2013, prior to fall fertilizer application. Due to timing of project approval (fall 2011) and the logistics for site prep (i.e., inability to install equipment into frozen soil), fall and early spring GHG fluxes were not measured in the 2012 growing season. Therefore, the 2012 data do not represent a full cropping year. However, the 2013 data do encompass a complete cropping year; i.e., from fall fertilizer application (October 2012) through harvest (September 2013) and ending just prior to the next fall fertilizer application (October 2013). Greenhouse gas fluxes were measured for all three fields (DL, IR12, and IR13) from fall fertilizer application in 2012 to just prior to spring seeding in 2013. This time-frame allowed for flux comparisons during the expected peak emission period following soil thaw. Soil conditions and GHG gases were not monitored during the winter months (soil freeze-up to thaw) as losses from frozen soils during winter months are considered to be minimal in this region (Malhi et al., 2001).

3.3.2 Soil emissions measurements

Soil GHG emissions were measured using rectangular (22 cm × 45.5 cm × 15 cm; width × length × height) non-steady state vented chambers constructed from clear, 0.6-cm thick polymethyl methacrylate (PMMA). Chamber bases were installed into the soil to depth of 5-cm and, except during seeding and tillage operations, remained in position for the entire year. When installed, the chambers had a headspace volume of 10 L and covered an area of 1000 cm^2. Twenty chambers were installed in each field at 6.25-m intervals along a 125-m linear transect. In an effort to avoid sampling the same seed rows and furrows along the transect, every second chamber was offset from the transect centerline by one meter. Plants were excluded from the chamber and the crop rows disturbed by chamber installation were replanted along the outside perimeter of the chamber base. Transects were oriented parallel to the direction of seeding to minimize disruption to normal cropping operations (Figure 3.1).

Table 3.2 Site information for the irrigated (IR12) and the dryland (DL) quarter sections studied in the 2012 growing season.

	IR12		DL	
Cropping system	Irrigated		Dryland	
Location	51°39’34”N; 106°56’31”W		51°38’41”N; 106°57’16”W	
Field area (*ha*)	45		58	
	Management			
Seeding		*— date —*		*— date —*
Crop	HRSW†	15 May 2012	HRSW	17 May 2012
Variety	*AC Carberry*		*AC Barrie*	
Swathing		30 Aug 2012		29 Aug 2012
Harvest yield‡ (*kg ha^{-1}*)	3400	11 Sept 2012	2500	03 Sept 2012
	Crop inputs			
	— kg N ha^{-1} —	*— date —*	*— kg N ha^{-1} —*	*— date —*
Fertilizer — total	110		73	
Anhydrous ammonia	100	15 Oct 2011	67	15 Oct 2011
Granular (MAP)§	10	15 May 2012	6	17 May 2012
Chemical				
Herbicide	*Target/Horizon*¶	12 June 2012	*Target/Horizon*	12 June 2012
Fungicide	*Prosaro*#	11 July 2012		
	Precipitation			
	— mm —	*— date —*	*— mm —*	*— date —*
Snowpack — max SWE	27 ± 7	02 May 2012	13 ± 5	02 May 2012
Rainfall — total††	361		361	
Growing season	321		321	
Post-harvest	40		40	
Irrigation — total	79		--	
Growing season	51		--	
Post-harvest	28		--	

† Hard Red Spring Wheat

‡ Determined from yield monitor on the combine harvester and verified by the total grain yield harvested and sold by the producer.

§ Monoammonium phosphate

¶ Tank mix of Target [Metallocarboxypeptidase (23.3%) and Mecoprop-P (5.3%) and Dicamba (5.3%)] and Horizon (Clodinafop-propargyl); Syngenta Crop Protection Canada.

Mixture of Prothioconazole (18.75%) and Tebuconazole (18.75%); Bayer CropScience Canada.

†† Rain guages were not installed at the study site until May 31, 2012, therefore growing season rainfall measurements were supplimented with data from weather station at Outlook, SK (Environment Canada, 2012) for April and May.

Table 3.3 Site information for the irrigated (IR13) and the dryland (DL) quarter sections studied in the 2013 growing season.

	IR13		**DL**	
Cropping system	Irrigated		Dryland	
Location	51°39'09"N; 106°55'51"W		51°38'41"N; 106°57'16"W	
Field area (*ha*)	53		58	
	Management			
Seeding		*— date —*		*— date —*
Crop	Canola	16 May 2013	Canola	17 May 2013
Variety	*InVigor L130*[†]		*InVigor L130*	
Swathing		22 Aug 2013		12 Aug 2013
Harvest yield[‡] (*kg ha^{-1}*)	3600	09 Sept 2013	2400	08 Sept 2013
	Crop inputs			
	— kg N ha^{-1} —	*— date —*	*— kg N ha^{-1} —*	*— date —*
Fertilizer — total	146		90	
Anhydrous ammonia	140	11 Oct 2012	78	16 Oct 2012
Granular (MAP)[§]	6	13 May 2013	12	15 May 2013
Chemical				
Herbicide	*Liberty*[¶]	12 June 2013	*Liberty*	06 June 2013
Fungicide	*Astound*[#]	02 July 2013		
	Precipitation			
	— mm —	*— date —*	*— mm —*	*— date —*
Snowpack — max SWE	104 ± 22	26 Mar 2013	81 ± 33	26 Mar 2013
Rainfall — total	228		228	
Growing season	179		179	
Post-harvest	49		49	
Irrigation — total	176		--	
Growing season	127		--	
Post-harvest	49		--	

† Hybrid Canola (*Brassica napus*); Bayer CropScience Canada.

‡ Determined from yield monitor on the combine harvester and verified by the total grain yield harvested and sold by the producer.

§ Monoammonium phosphate

¶ Glufosinate ammonium; Bayer CropScience Canada.

\# Mixture of diatomacous earth, pyrimidine derivative fungicide [Cyprodinil (37.5%)], and substituted benzodioxalcarbonitrile fungicide [Fludioxonil (25.0%)]; Syngenta Crop Protection Canada.

Samples of the chamber headspace gas were drawn through a rubber septum in the chamber lid using a 20-mL syringe fitted with a 25-gauge needle and immediately injected into pre-evacuated (~0.5 kPa) 12-mL Exetainer™ vials (Labco Limited, UK) for storage and transport. Samples were collected at 15 (t_{15}), 30 (t_{30}), and 45 (t_{45}) minutes after lid closure. In addition to headspace samples, four ambient air samples were collected before and after chamber sampling to determine baseline gas concentrations (i.e., the average ambient concentration was used to assign t_0) and for calculation of the minimum detectable concentration difference (MDCD), which was used for data quality control as per Yates et al. (2006a).

Concentrations of N_2O, CO_2, and CH_4 were determined using gas chromatography (Bruker 450 GC, Bruker Biosciences Corporation USA) (Farrell and Elliott, 2007). Daily fluxes were estimated by fitting linear or exponential regression equations to the concentration *vs.* time data using the HMR model (Pedersen et al., 2010; Pedersen, 2011), a modified Hutchinson-Mosier method (Hutchinson and Mosier, 1981) implemented as an add-on package for R (R Development Core Team, 2011). Fluxes were taken as the slope of the fitted regression at t_0 as recommended by HMR, unless the concentration differences between t_0 and each subsequent time step did not exceed MDCD (Yates et al., 2006a). In cases where subsequent samples did not exceed the MDCD they were not considered significantly different and fluxes were taken as the slope of the linear regression at t_0. This method allowed for the calculation and inclusion of statistically non-significant fluxes (i.e., below the MDCD; common to CH_4 fluxes) in the dataset to minimize left censoring (Ens, 2012). Daily fluxes are reported as the median daily flux of the 20 sampling points. Cumulative annual fluxes were calculated by estimating non-sampling days by linear interpolation (Pennock et al., 2006).

The key issues related to chamber-based measurements of soil gas flux have been reviewed by Mosier (1989). In an effort to minimize the potential problems associated with these issues, the list of recommendations provided by Baker et al. (2003) were considered in the design and use of the chambers for the current study. Temperature, pressure, and humidity perturbations were minimized by constructing chambers with vented lids and employing short deployment periods. As well, chambers lids were also covered in a reflective insulation to minimize heating of the chamber headspace during sampling events. One of the major challenges associated with the chamber-based flux measurements is addressing the high spatial and temporal variation associated with soil GHG production. To minimize sampling bias associated with diurnal variations in gas fluxes (brought about by temperature fluctuations), flux measurements were made as close to mid-morning—the time of day that most closely corresponds to daily average temperature—as logistically feasible. [Note: automated chamber data from a different study indicated that, for N_2O, the flux measured at mid-day was comparable to the average flux measured during a 24-h cycle (R.E. Farrell, personal communication).] Seasonality and day-to-day variations were addressed by increasing sampling frequency during periods when fluxes were expected to be greatest and most variable (i.e., during spring thaw, following fertilizer application, after large precipitation/irrigation events). In an effort to capture the extremely high spatial variability of gas fluxes (especially N_2O), chambers were constructed with a large footprint (1000 cm^2) and twenty points were sampled from each treatment to determine daily fluxes.

3.3.3 Soil water and temperature measurements

In-situ soil sensors were used to continually monitor soil conditions at each site. Sensors were controlled using Campbell Scientific dataloggers (CR3000; Campbell Scientific Canada) programmed to collate and store sensor output data at a 30-minute interval. In each field, sensor probes were installed at a 10-cm depth at four points: 15, 45, 80, and 110 m along the gas sampling transect. Data presented are the averages of the four sampling points. Soil temperature and volumetric water content (VWC) measurements were made using CS650 water content reflectometer probes (Campbell Scientific Canada). Heat dissipation probes (CS229; Campbell Scientific Canada) were used to quantify soil matric potential. Corresponding trends in soil moisture content were observed using the CS650 and the CS229 probes; thus, because volumetric soil moisture data is generally more intuitive and more widely used in the GHG literature, only the VWC data from the CS650 are presented and discussed.

3.3.4 Ancillary data

At each site, precipitation measurements were made using a tipping bucket rain gauge (TR-525; Texas Electronics Inc, USA) connected to the CR3000 datalogger. Two gauges were installed in each field, at 20 and 85 meters along the sampling transect, with the average of the two gauges taken as the daily precipitation value. During the winter of 2012, a weighing precipitation gauge (Belfort 3000; Belfort Instrument, Baltimore MD) controlled by a CR10X datalogger (Campbell Scientific Canada) was installed at the DL site, to more accurately measure precipitation. Irrigation quantities were determined by subtracting precipitation measured at the rainfed site (DL) from the precipitation measured at the irrigated site. Snow surveys were conducted during the late winter period (mid-January until snowmelt), measuring the density and depth of accumulated snowpack to determine winter precipitation. Nitrogen fertilizer rate and source, and details pertaining to chemical applications were obtained from the cooperator. Crop yields were determined using an electronic crop yield monitor on the combine thresher and verified by the total grain yield harvested and sold by the producer. Site information for both study years is presented in Tables 3.1 and 3.2.

3.3.5 Statistical analysis

Soil GHG fluxes, especially N_2O fluxes, are characterized by highly skewed distributions; therefore, the median and interquartile range (first and third quartiles presented as Q_1 and Q_3, respectively) were used as the summary statistic to describe the flux data (Corre et al., 1996; Pennock and Corre, 2001). The correlations between soil GHG fluxes, soil temperature, and soil volumetric water content were assessed using Spearman rank correlation analysis and a significance level of 0.05. All statistical analyses were carried out in the R environment (R Development Core Team, 2011).

3.4 Results

3.4.1 Weather and soil conditions

During the present study, annual rainfall at the dryland and irrigated sites was greater in 2012 (361 mm yr^{-1}) than 2013 (228 mm yr^{-1}). The May through August rainfall in 2012 was greater than normal (309 mm *vs.* 202 mm) with the majority received during May (100 mm) and June (110 mm) (Appendix B; Table D.2). In 2013, May through August rainfall (176 mm) was slightly lower than normal (185 mm). Consequently, irrigation was applied more frequently and provided a much greater proportion of the total water received on-site (42% *vs*. 14%) in 2013 (see Tables 3.1 and 3.2). The accumulation of winter precipitation into snowpack—and its subsequent melting in early spring—provided early season soil moisture. Snowpack at the field sites was much lower in the winter of 2011/12 (13–27 mm SWE) than the winter of 2012/13 (81–104 mm SWE). In both study years snowpack was generally greater in the irrigated fields than in the dryland (rainfed) field (see Tables 3.1 and 3.2)—an unusual result considering the greater density of standing stubble remaining at the dryland site in both years (due to fall tillage on the irrigated field in 2011 and a dry bean residue *vs.* a wheat residue in the 2013 cropping year). This was especially evident in 2012 crop year when the overall snowfall was low and the snowpack on the irrigated field was double that on the dryland field (i.e., 27-mm SWE *vs*. 13-mm SWE). Although specific early season (i.e., thaw to seeding) soil moisture data was available only for the DL (Figure 3.4b) and IR12 (not shown) sites in 2013, high soil moisture (saturated and near to saturated) conditions were observed prior to seeding at all sites in both study years. However, system-specific differences were noted in the limited post-thaw period dataset. The greater snowpack and higher antecedent (fall 2012) soil moisture status at IR12 were revealed in the high soil moisture content of early spring (average VWC May 1 to 10: IR12 39% vs. DL 33%). In fact, the very wet conditions at IR12 (standing water) for the week following thaw (April 23 through May 30) made site access difficult. However, the duration of the saturated conditions at IR12 was relatively localized and not an accurate representation of the other irrigated field (IR13) at the study site which had standing water for only three days following thaw (up to April 26th). Post-melt standing water was not present at the DL site for any extended period. At the dryland site in both 2012 and 2013, soil moisture remained high (>30% VWC) for about a month following seeding (see Panel b in Figures 3.2 and 3.4, respectively), decreased steadily until mid to late summer, and then remained at between 15 and 20% VWC until after the fall harvest. Conversely, at the irrigated sites (IR12 and IR13) high soil moisture levels (>30% VWC) were maintained throughout the peak growing season (June and July) as a result of the periodic application of irrigation water (Figures 3.3b and 3.5b, respectively).

Growing season (thaw to freeze-up) air temperatures in the 2012 cropping season were similar to the 20-year normal, with the exception of October which was slightly colder (Appendix B; Table D.2). The 2013 temperatures were consistent with the 20-year normal from June through August, and October, but were warmer in May and September. The late thaw in April 2013 is reflected in the lower than average air temperatures for that month.

Seasonal trends in soil temperature were similar for both the irrigated and dryland sites in both study years. Due to study logistics, early season (i.e., thaw to seeding) soil temperatures were measured in 2013 only (DL site presented in Figure 3.4a; IR12 site not presented). Soil temperature increased rapidly following the spring thaw, reached sustained daily maxima greater

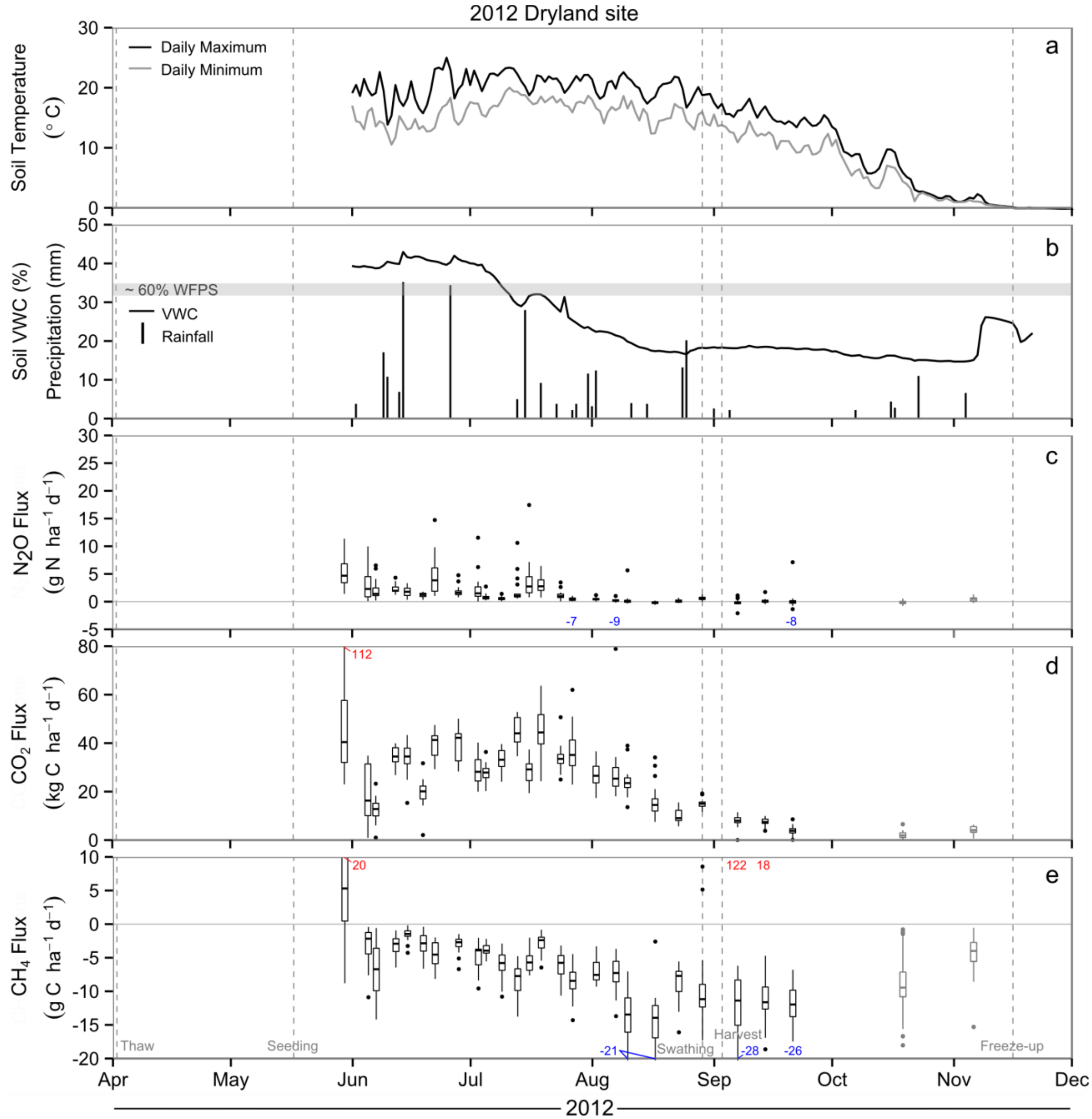

Figure 3.2 Soil emissions of N_2O (c), CO_2 (d), and CH_4 (e), in relation to seasonal soil temperature (a), precipitation and soil moisture (b) measured at the dryland site (DL) during the 2012 growing season. Emissions are presented as boxplots to represent daily variability between chambers (n=20). The grey bar on Panel b, represents the approximate range (accounting for field variability) where water filled pore space (WFPS) is at 60%. Maximum and minimum values beyond the figure boundaries are indicated in red and blue, respectively. Soil temperature and water content measurements were made at a 10-cm depth at four points along the gas sampling transect. Boxplots in grey (October 19 and November 6) occur after fall N application and are included in the 2013 crop year.

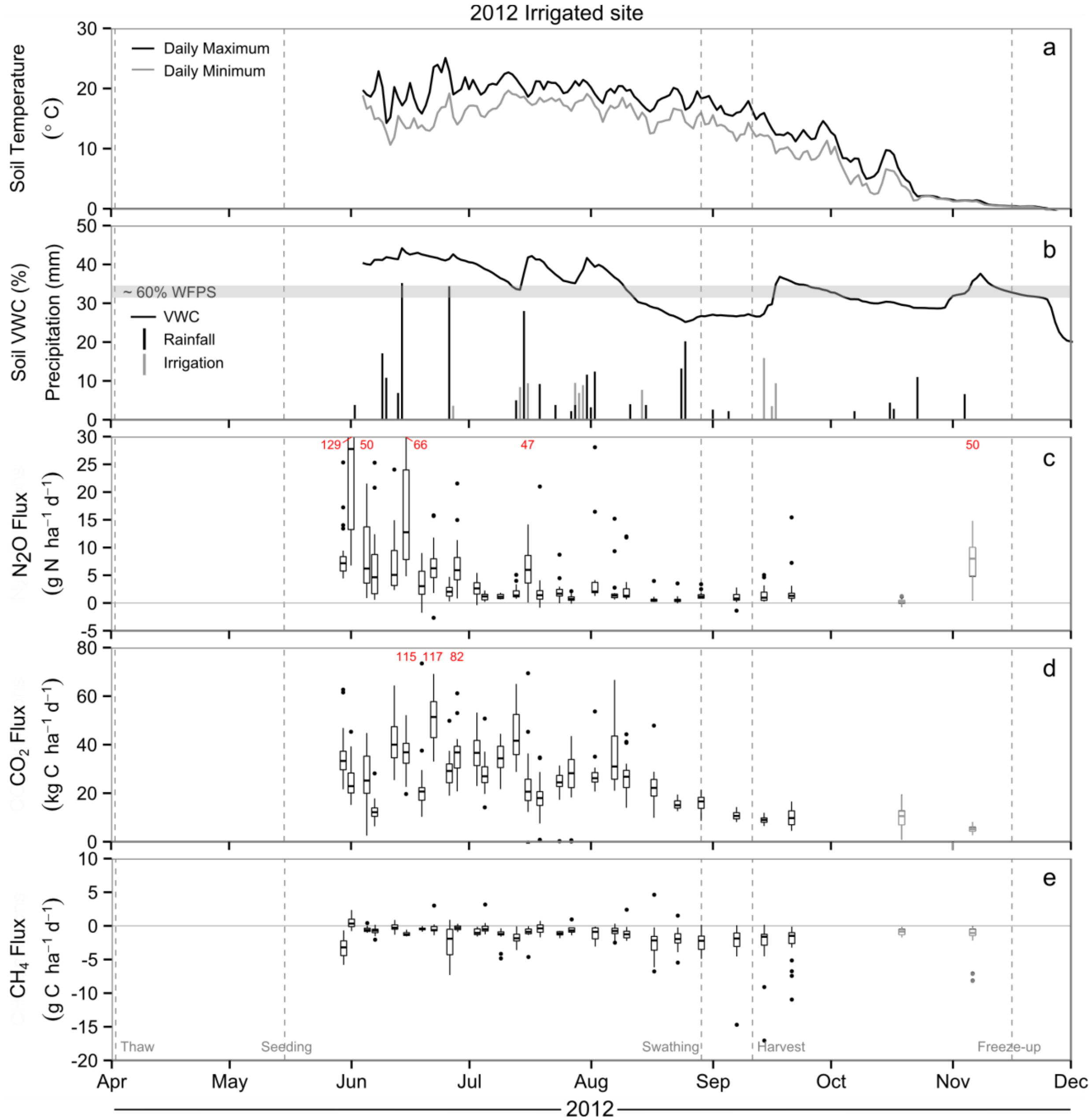

Figure 3.3 Soil emissions of N_2O (c), CO_2 (d), and CH_4 (e), in relation to seasonal soil temperature (a), precipitation and soil moisture (b) measured at the irrigated site (IR12) during the 2012 growing season. Emissions are presented as boxplots to represent daily variability between chambers (n=20). The grey bar on Panel b, represents the approximate range (accounting for field variability) where water filled pore space (WFPS) is at 60%. Values exceeding figure boundaries are indicated in red. Soil temperature and water content measurements were made at a 10-cm depth at four points along the gas sampling transect. Boxplots in grey (October 19 and November 6) occur after fall N application and are included in the 2013 crop year.

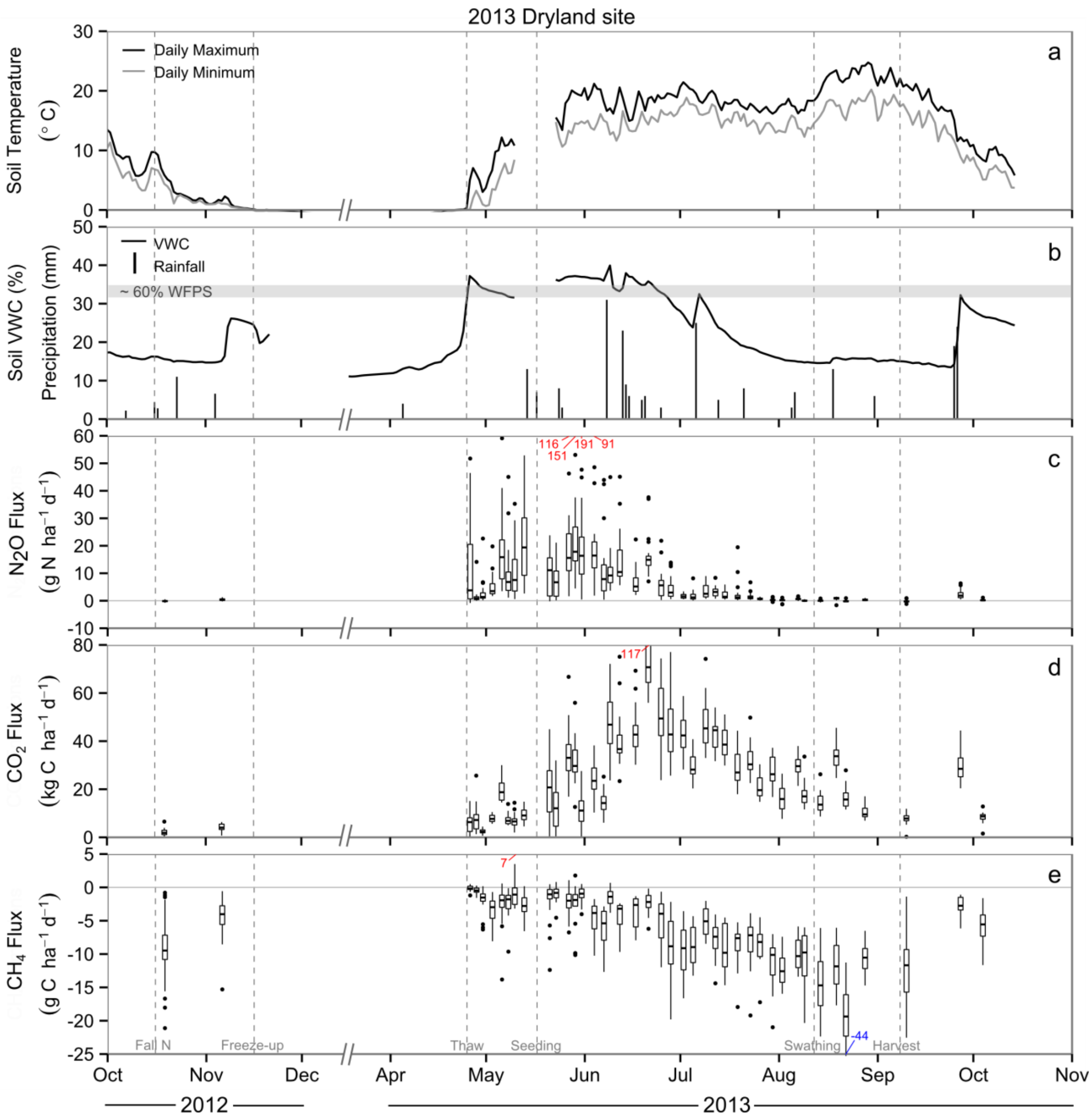

Figure 3.4 Soil emissions of N_2O (c), CO_2 (d), and CH_4 (e) in relation to seasonal soil temperature (a), precipitation and soil moisture (b) measured at the dryland site (DL) during the 2012/2013 cropping year. Emissions are presented as boxplots to represent daily variability between chambers (n=20). The grey bar on Panel b, represents the approximate range (accounting for field variability) where water filled pore space (WFPS) is at 60%. Maximum and minimum values beyond the figure boundaries are indicated in red and blue, respectively. Soil temperature and water content measurements were made at a 10-cm depth at four points along the transect. Missing soil temperature and water content data during mid-May corresponds to sensor removal for crop seeding. Soil conditions and emissions were not monitored during the winter months (soil freeze-up to thaw).

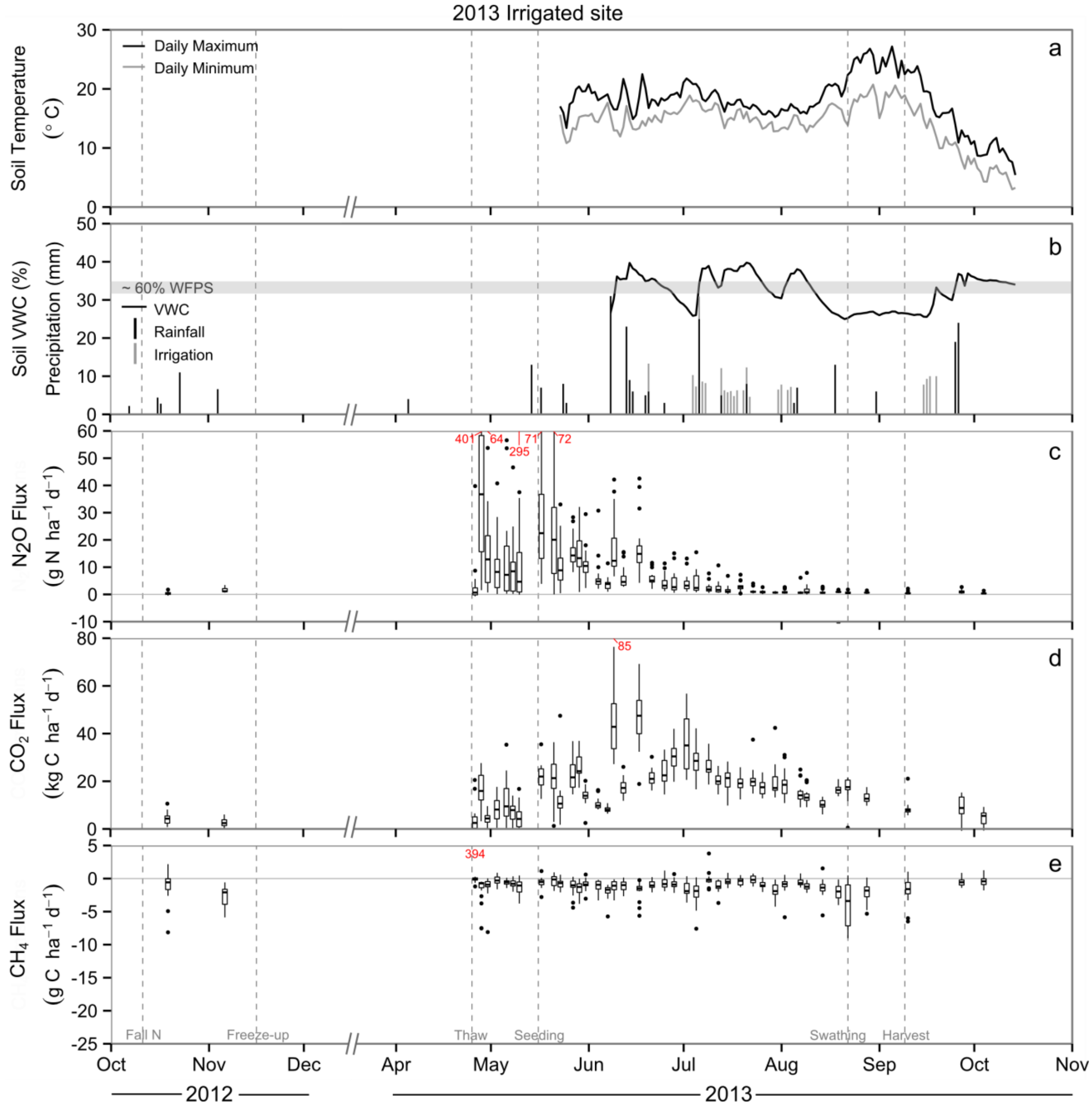

Figure 3.5 Soil emissions of N_2O (c), CO_2 (d), and CH_4 (e) in relation to seasonal soil temperature (a), precipitation and soil moisture (b) measured at the irrigated site (IR13) during the 2012/2013 cropping year. Emissions are presented as boxplots to represent daily variability between chambers (n=20). The grey bar on Panel b, represents the approximate range (accounting for field variability) where water filled pore space (WFPS) is at 60%. Maximum values beyond the figure boundaries are indicated in red. Soil temperature and water content measurements were made at a depth of 10 cm at four points along the sampling transect. Soil water content and temperature monitoring began on May 22. Soil conditions and emissions were not monitored during the winter months (soil freeze-up to thaw).

than 15ºC within 30 days, and maintained warm temperatures (with daily maximums ≥ 20ºC) throughout the late summer, after which temperatures decreased steadily until November freeze-up. In general, mid-summer soil temperatures were between 15 and 20ºC, though notable year-to-year differences were observed. In 2012, snowmelt and soil thaw occurred at the beginning of April, while a late April thaw occurred in 2013; thus, spring soil temperature increase in 2012 would have occurred much less rapidly than was observed in 2013. Peak soil temperatures were recorded in late June in 2012, but occurred between late August and early September in 2013. Very warm soil temperatures were measured in both cropping systems during the entire two to three week period between swathing and harvest in 2013—with minimum daily soil temperatures exceeding 20ºC on multiple days. In both years, the average range between minimum and maximum daily temperatures was similar for the irrigated and dryland sites (3.0 ± 0.3ºC in 2012; 3.5 ± 0.4ºC in 2013).

Though soil temperature trends are similar between cropping systems, differences were more apparent over shorter periods (3 to 5 weeks) during the cropping season. In 2012, daily maximum soil temperature and daily soil temperature ranges were greater (> 1ºC difference) at the DL site from late June through to late August. Similarly, the DL site had higher and more variable soil temperatures (i.e., greater daily minimum, maximum, mean, and range) than the IR12 site from mid-September to mid-October. Although the soils at the DL and IR12 thawed (daily soil temperature > 0 ºC) on the same day (23 Apr 2013), soil at the DL site were warmer prior to and after thaw. The minimum, maximum, and average daily soil temperatures differences from mid-April to mid-May were 1.3 ± 0.8ºC, 1.2 ± 0.8ºC, 1.4 ± 0.8ºC, respectively. In the 2013 growing season, soil temperatures (daily max, mean, and range) were greater in the DL system from late-July to late-August, while temperatures (daily min, max, mean, and range) were greater at the IR13 site from late-August to mid-September. At the end of the 2013 season (mid-September to mid-October), the irrigated cropping system had a greater daily soil temperature range (4.1 ± 1.3ºC *vs.* 3.0 ± 0.8ºC) and lower minimum daily soil temperatures (8.1 ± 4.1ºC *vs.* 9.3 ± 3.8ºC). Minimal differences in soil temperature (< 0.5 ºC) were observed outside of the periods presented.

Applications of fertilizer-N differed both between years (reflecting the different fertility requirements for wheat and canola) and cropping systems (with the irrigated sites receiving greater inputs of fertilizer-N). In the 2012 season, the wheat crops received 73 kg N ha^{-1} for the

DL site and 110 kg N ha^{-1} for the IR12 site (Table 3.2). In 2013, the N rates for canola were 90 and 146 kg N ha^{-1} for the DL and IR13 sites, respectively (Table 3.3). Despite a large difference in fertilizer-N application, in 2012 there was no difference in available soil N (i.e., NO_3 + NH_4) in surface (0–15 cm) soils along the sampling transects in the irrigated (15.4 ± 1.8 kg N ha^{-1}) and dryland (13.1 ± 0.7 kg N ha^{-1}) sites at fall soil sampling (Appendix B; Table B.7). However, after the 2013 cropping season there were greater quantities of plant available N remaining in the surface soils at the DL site (35.1 ± 14.8 kg N ha^{-1}) than at the irrigated site (11.7 ± 1.7 kg N ha^{-1}) (Appendix B; Table B.8).

3.4.2 Nitrous oxide emissions

Daily N_2O emissions demonstrated strong seasonal trends in both the irrigated and dryland cropping systems, with the largest emissions occurring early in the growing season. In 2012, median daily N_2O emissions from both fields were greatest during the period immediately after seeding and the spring application of N-fertilizer (Figures 3.2c and 3.3c). At the irrigated site (IR12), the largest median daily flux (28 g N_2O-N ha^{-1} d^{-1})—together with the widest range in values along the transect (7–129 g N_2O-N ha^{-1} d^{-1})—occurred on June 1st, two weeks after seeding. The smallest median daily flux (0 g N_2O-N ha^{-1} d^{-1}) occurred late in the season on October 19th. At the dryland site (DL), the largest median daily flux (5 g N_2O-N ha^{-1} d^{-1}) also occurred about two weeks after seeding (May 30th), but with a magnitude that was about one-fifth that at the irrigated site. Again, the smallest median daily flux (0 g N_2O-N ha^{-1} d^{-1}) occurred later in the season and on multiple days between August 27th and October 19th. In 2012, variability in N_2O emissions along the transect at the DL site was greatest on July 16th—with individual fluxes ranging from 1 to 17 g N_2O-N ha^{-1} d^{-1} (*median* = 3 g N_2O-N ha^{-1} d^{-1})—following a significant precipitation event during which the field received 28 mm of rain (see Figure 3.2; Panels b and c). Small negative fluxes (i.e., influx to the soil) were observed on several days, but were always associated with “low flux” days; i.e., days with a median flux between 0 and 1 g N_2O-N ha^{-1} d^{-1} (Figures 3.2c and 3.3c).

In 2013, N_2O emissions were greatest—and exhibited the highest degree of within transect variability—immediately following the onset of snowmelt and soil thaw (Figure 3.4c and 3.5c). This was most notable in the irrigated cropping system at site IR13. The largest median daily flux in 2013 (36 g N_2O-N ha^{-1} d^{-1}) occurred at IR13 on April 28th (Figure 3.5c).

The within transect variability was also greatest on April 28th, with emissions from the individual gas sampling chambers (*n* = 20) ranging from 2 to 401 g N_2O-N ha^{-1} d^{-1}. As was the case in 2012, the smallest median daily fluxes at IR13 (0 g N_2O-N ha^{-1} d^{-1}) occurred late in the season (i.e., 19 Oct 2012; 10 Sept 2013; and 04 Oct 2013). At the DL site, the largest median daily flux (19 g N_2O-N ha^{-1} d^{-1}) occurred on May 13th (Figure 3.4c). Conversely, low flux days (median ≤ 2 g N_2O-N ha^{-1} d^{-1}) were quite common during the late summer and continued to soil freeze-up (November 16). Within transect variability was greatest at the DL site about two weeks after seeding and spring fertilizer application (i.e., on May 31st)—with fluxes ranging from 1 to 191 g N_2O-N ha^{-1} d^{-1} and a median flux of 16 g N_2O-N ha^{-1} d^{-1} (which was the second largest median flux recorded at the site).

In addition to the daily flux calculations, cumulative annual N_2O emissions were calculated for the dryland and both irrigated site (Table 3.4). For logistical reasons, the 2012 sampling season did not include the period from the fall 2011 fertilizer application through the 2012 spring thaw and, thus could not be used to develop emission coefficients for inventory purposes. Nevertheless, the cumulative growing season emissions were calculated in order to facilitate a comparison between dryland and irrigated wheat production (Table 3.5). Cumulative growing season emissions in 2012 (115 days; May 30th to September 21st) were greater for the irrigated cropping system (IR12; *median* = 490 g N ha^{-1}; Q_1 and Q_3 = 361 and 624 g N ha^{-1}, respectively) than the dryland cropping system (DL; *median* = 147 g N ha^{-1}; Q_1 and Q_3 = 120 and 168 g N ha^{-1}, respectively).

The 2013 crop year included a complete crop cycle, from the fall of 2012 fertilizer application through the 2013 spring thaw, the 2013 growing season (seeding to harvest), and the post-harvest period leading up to the fall 2013 fertilizer application. Cumulative growing season emissions in 2013 (141 days; May 17th to October 4th; see Table 3.6) were comparable for the irrigated canola production system (*median* = 533 g N ha^{-1}; Q_1 = 398 g N ha^{-1}; Q_3 = 596 g N ha^{-1}) and the dryland system (*median* = 548 g N ha^{-1}; Q_1 = 445 g N ha^{-1}; Q_3 = 677 g N ha^{-1}). Cumulative yearly emissions (351 days; 19 Oct 2012 to 04 Oct 2013) from the irrigated canola production system (*median* = 962 g N ha^{-1}; Q_1 = 844 g N ha^{-1}; Q_3 = 1502 g N ha^{-1}) were only about 10 percent greater than those from the dryland production system (*median* = 871 g N ha^{-1}; Q_1 = 679 g N ha^{-1}; Q_3 = 1050 g N ha^{-1}). The cumulative yearly values were subsequently used in calculating the C-footprint of the two production systems (see Chapter 4).

Table 3.4 Cumulative annual soil fluxes for the 2013 crop year (351 days; 19 Oct 2012 to 04 Oct 2013) from the irrigated (IR13) and dryland (DL) sites. Values presented are the median and interquartile range. Positive values represent GHG emissions from the soil to the atmosphere. Negative values represent atmospheric GHG uptake by the soil.

	IR13			**DL**		
	Q_1	*median*	*Q_3*	*Q_1*	*median*	*Q_3*
	g N_2O-N ha^{-1}					
Nitrous oxide	844	962	1502	679	871	1050
	kg CO_2-C ha^{-1}					
Carbon dioxide	2705	2803	2932	3614	3820	4473
	g CH_4-C ha^{-1}					
Methane	-365	-226	-102	-1618	-1260	-1082

Table 3.5 Cumulative growing season soil fluxes for the 2012 cropping season (115 days; May 30 to September 21) from the irrigated (IR12) and dryland (DL) sites. Values presented are the median and interquartile range. Positive values represent GHG emissions from the soil to the atmosphere. Negative values represent atmospheric GHG uptake by the soil.

	IR12			DL		
	Q_1	*median*	*Q_3*	*Q_1*	*median*	*Q_3*
	g N_2O-N ha^{-1}					
Nitrous oxide	361	490	624	120	147	168
	kg CO_2-C ha^{-1}					
Carbon dioxide	2587	2889	3107	2547	2918	3050
	g CH_4-C ha^{-1}					
Methane	-230	-170	-119	-975	-812	-696

Table 3.6 Cumulative growing season soil fluxes for the 2013 cropping season (141 days; May 17th to October 4th) from the irrigated (IR13) and dryland (DL) sites. Values presented are the median and interquartile range. Positive values represent GHG emissions from the soil to the atmosphere. Negative values represent atmospheric GHG uptake by the soil.

	IR13			DL		
	Q_1	*median*	*Q_3*	*Q_1*	*median*	*Q_3*
	g N_2O-N ha^{-1}					
Nitrous oxide	398	533	596	445	548	677
	kg CO_2-C ha^{-1}					
Carbon dioxide	2291	2396	2525	3314	3500	4088
	g CH_4-C ha^{-1}					
Methane	-232	-70	-111	-1412	-1035	-841

Spring emissions during the 2013 crop year (25 days; April 23rd to May 17th) were measured for all three sites (DL, IR12, and IR13). The DL site had the lowest emissions (*median* =161 g N ha^{-1}; Q_1 and Q_3 = 97 and 270 g N ha^{-1}, respectively) followed by the IR13 site with emissions over twice as high (*median* = 354 g N ha^{-1}; Q_1 and Q_3 = 2181 and 554 g N ha^{-1}, respectively). The fluxes measured at the IR12 site (*median* = 1674 g N ha^{-1}; Q_1 and Q_3 = 954 and 2347 g N ha^{-1}, respectively) were substantially greater than both other sites at over 10 and 4.5 times greater than DL and IR13, respectively.

A Spearman rank correlation analysis identified positive correlations between daily N_2O flux and VWC during the growing season (seeding to post-harvest) for the dryland site in both years (2012: $r_s = 0.76$, $p < 0.001$; 2013: $r_s = 0.97$, $p < 0.001$), and the irrigated site in 2012 (IR12: $r_s = 0.89$, $p < 0.01$). The 2013 irrigated site demonstrated a moderate but non-significant correlation (IR13: $r_s = 0.44$, $p < 0.1$).

3.4.3 Carbon dioxide

Soil-derived CO_2 emissions reflect both root and microbial respiration (Rochette et al., 1999; Curtin et al., 2000; Ellert and Janzen, 2008; Liebig et al., 2010) and, in general, followed a clear seasonal trend that paralleled crop growth (Figures 3.2d–3.5d). That is, emissions were lowest prior to seeding, increased to maxima during June and July, and then decreased steadily through to the onset of soil freeze-up.

In 2012, the magnitude of the CO_2 fluxes measured at the irrigated (IR12) and dryland (DL) sites were comparable—with peak emissions (40 to 50 kg CO_2-C ha^{-1} d^{-1}) occurring during June and July (Figures 3.2d and 3.3d). Cumulative growing season emissions were of similar magnitude for both the DL site (*median* = 2918 kg CO_2-C ha^{-1} 115-d^{-1}; Q_1 and Q_3 = 2547 and 3050 kg CO_2-C ha^{-1} 115-d^{-1}, respectively) and the IR12 site (*median* = 2889 kg CO_2-C ha^{-1} 115-d^{-1}; Q_1 and Q_3 = 2587 and 3107 kg CO_2-C ha^{-1} 115-d^{-1}, respectively). The largest median daily flux at the IR12 site (41 kg CO_2-C ha^{-1} d^{-1}) occurred on July 13th; at the DL site, the largest median daily flux (44 kg CO_2-C ha^{-1} d^{-1}) occurred on July 13th and 19th. At both sites, the fall months (September, October, and November) were characterized by low median daily emissions (< 10 kg CO_2-C ha^{-1} d^{-1}).

During the 2013 crop year (351 days; 19 Oct 2012 to 04 Oct 2013), CO_2 emissions from the DL site were greater than those from the IR13 site, with cumulative annual emissions of 3820 kg CO_2-C ha^{-1} (*median*; Q_1 = 3614 kg CO_2-C ha^{-1}; Q_3 = 4473 kg CO_2-C ha^{-1}) compared to 2803 kg CO_2-C ha^{-1} (*median;* Q_1 = 2705 kg CO_2-C $ha^{-1;}$ Q_3 = 2932 kg CO_2-C ha^{-1}) at the IR12 site. Growing season emissions (141 days; May 17th to October 4th) from the DL site (*median* = 3500 CO_2-C ha^{-1}; Q_1 = 3314 kg CO_2-C ha^{-1}; Q_3 = 4088 kg CO_2-C ha^{-1}) were greater than emissions from both sites in 2012, while emissions from the IR13 site (*median* = 2396 CO_2-C ha^{-1}; Q_1 = 2291 kg CO_2-C ha^{-1}; Q_3 = 2525 kg CO_2-C ha^{-1}) were lower. Emissions were greatest at the DL site from June through August—with the largest median daily emissions occurring on June 21st (71 kg CO_2-C ha^{-1} d^{-1}). Median daily emissions for the IR13 site peaked on June 17th (47 kg CO_2-C ha^{-1} d^{-1}).

As was observed with N_2O emissions, 2013 cumulative spring CO_2 emissions were greatest from the IR12 site at 390 kg CO_2-C ha^{-1} 25-d^{-1} (*median*; Q_1 = 319 kg CO_2-C ha^{-1} 25-d^{-1}; Q_3 = 450 kg CO_2-C ha^{-1} 25-d^{-1}). Emissions from DL and IR13 were 152 kg CO_2-C ha^{-1} 25-d^{-1} (*median;* Q_1 = 132 kg CO_2-C ha^{-1} 25-d^{-1}; Q_3 = 187 kg CO_2-C ha^{-1} 25-d^{-1}) and 240 kg CO_2-C ha^{-1} 25-d^{-1} (*median;* Q_1 = 194 kg CO_2-C ha^{-1} 25-d^{-1}; Q_3 = 276 kg CO_2-C ha^{-1} 25-d^{-1}), respectively.

A Spearman rank correlation analysis on growing season (seeding to post-harvest) data identified positive moderate correlations between daily CO_2 flux and VWC for the IR12 site in 2012 (r_s = 0.37, $p < 0.05$), the DL site in 2013 (r_s = 0.33, $p < 0.05$), and the DL site in 2012 (though not significant; r_s = 0.42, $p < 0.1$). Carbon dioxide flux and daily soil temperature maximum were correlated at both fields in 2012 (DL: r_s = 0.43, $p < 0.05$; IR12: r_s = 0.47, $p < 0.001$). Carbon dioxide fluxes were not correlated with any other measured parameters at the IR13 site.

3.4.4 Methane

Methane emissions from soils are generally associated with water saturated conditions, conditions which may be intensified by the application of irrigation water. Volumetric water content and growing season (seeding to post-harvest) CH_4 fluxes were positively correlated (Spearman rank correlation, $p < 0.001$) at both sites in both years (2012: DL r_s = 0.85, IR12 r_s = 0.60; 2013: DL r_s = 0.92, IR13 r_s = 0.71). However, prolonged saturated conditions are rare with proper irrigation management. The data obtained at the two irrigated sites (IR12 and IR13) indicate that these soils were actually small sinks for atmospheric CH_4 (Figures 3.4e–3.5e), with greater uptake (measured flux < 0) under drier conditions. Soils at the dryland (DL) site also were sinks for atmospheric CH_4 (Figures 3.2e–3.3e), though CH_4 uptake (i.e., influx *vs*. efflux) measured at the dryland site was greater than that at the irrigated sites in both years of the study—particularly during the period from August through October. Aside from the spring period (April and May), CH_4 uptake from the dryland system exceeded -5 g CH_4-C ha^{-1} d^{-1} for the majority of both cropping seasons, with periods in July and August exceeding -10 g CH_4-C ha^{-1} d^{-1}. At the dryland site, median daily uptake peaked on August 17th (-14 g CH_4-C ha^{-1} d^{-1}) in 2012 and on August 22nd (-19 g CH_4-C ha^{-1} d^{-1}) in 2013. At the irrigated sites, daily fluxes typically ranged from 0 to -2 g CH_4-C ha^{-1} d^{-1}. Peak methane uptake in 2012 (-2 g CH_4-C ha^{-1} d^{-1}) was measured on multiple days from mid-August through mid-September. In 2013, peak uptake (-4 g CH_4-C ha^{-1} d^{-1}) occurred on August 22nd. Methane emissions (i.e., measured flux > 0) were occasionally measured from individual chambers along the sampling transects; however, only once was there a positive median daily flux; i.e., 30 May 2012 (5 g CH_4-C ha^{-1} d^{-1}) at the DL site.

Cumulative growing season CH_4 uptake in the 2012 season (115 days; May 30th to September 21st) was greater from the dryland site (DL; *median* = -812 g CH_4-C ha^{-1}; Q_1 and Q_3 = -975 and -696 g CH_4-C ha^{-1}) than the irrigated site (IR12; *median* = -170 g CH_4-C ha^{-1}; Q_1 and Q_3 = -230 and -119 g CH_4-C ha^{-1}). In 2013, growing season (141 days; May 17th to October 4th) uptake was also greater from DL (*median* = -1035 g CH_4-C ha^{-1}; Q_1 and Q_3 = -1412 and -841 g CH_4-C ha^{-1}) compared to the IR13 site (*median* = -170 g CH_4-C ha^{-1}; Q_1 and Q_3 = -232 and -111 g CH_4-C ha^{-1}). Median cumulative annual flux (351 days; 19 Oct 2012 to 04 Oct 2013) was -1260 g CH_4-C ha^{-1} (Q_1 and Q_3 = -1618 and -1082 g CH_4-C ha^{-1}) and -226 g CH_4-C ha^{-1} (Q_1 and Q_3 = -365 and -102 g CH_4-C ha^{-1}) for the DL and IR13 sites, respectively.

3.5 Discussion

Seasonal trends in soil temperature differed very little between the irrigated and dryland cropping systems. The differences that were observed occurred on shorter time scales (i.e., weeks to months) and varied year to year. Soil temperature is closely tied to air temperature, thus the soil temperature trends in 2012 season were likely typical of the normal. Similarly, the early season and late season months in 2013 were colder and warmer than normal, respectively. Considering that the thermal properties of soil are largely dependent on soil water content (i.e., wetter soil requires more energy to warm than dry soil), it is not surprising that, in both years, soil temperatures were slightly higher and more variable under dry land conditions, aside from the period between late-August and mid-September when soil temperatures were greater at the irrigated site (IR13). The elevated temperatures observed during this period were likely a combination of the low soil moisture status, a dark and unshaded soil surface (the crop had been recently swathed), and warmer than normal weather. Due to more frequent tillage, the IR13 site had a lower albedo (darker soil surface) than the DL site (which had a small "duff layer" made up of residue from the previous crop) which would absorb more solar radiation. In both years irrigation events maintained soil moisture levels above 30% VWC until 2–3 weeks prior to swathing (to allow the crop to dry and mature for harvest). Although these high soil moisture conditions create potential for greater GHG emissions (i.e., greater N_2O and CO_2 emissions; lower CH_4 uptake), results suggest that the implications may not be as severe as expected.

Nitrous oxide is perhaps the most important GHG associated with cropping systems, as small quantities can elicit large effects in terms of radiative forcing. The greatest daily emissions were observed during snow melt and spring thaw, consistent with observations from dryland cropping (Nyborg et al., 1997; Lemke et al., 1998; Dusenbury et al., 2008; Liebig et al., 2010) and irrigated cropping (Mosier et al., 2006; Sainju et al., 2012) in the northern Great Plains. Due to fall fertilizer applications in both the irrigated and dryland systems, large amounts of soil N were potentially available for transformation and loss during high soil moisture conditions at spring thaw (Hao et al., 2001). Spring emissions were greater from the irrigated system (IR13), reflecting the greater rates of fall-applied anhydrous ammonia (140 kg N ha^{-1} *vs.* 78 kg N ha^{-1}; Table 3.1). At seeding, the remaining crop N requirements were fulfilled with the application of granular fertilizer; thus, available soil N levels were expected to be high which likely contributed to the elevated N_2O emissions observed during the month following seeding.

After the peak N_2O emissions period in spring, small increases in flux variability and median daily flux were observed at elevated soil moisture levels caused by rainfall and irrigation events—a trend commonly observed in other studies (Dobbie et al., 1999; Mosier et al., 2005; Liu et al., 2006; Jabro et al., 2008). In both cropping systems during the 2012 and 2013 seasons, median daily flux steadily declined through June and remained low from July onward despite periods of high soil temperature and moisture at the irrigated sites. The low emission magnitude, coupled with the small amounts of soil N remaining in the soil after the growing season (most notably in 2013), suggests that N_2O production was limited by N availability in the irrigated system.

Previous research has noted that soil N_2O production in the Canadian Prairies is not limited by mineral N availability, but by low denitrification activity under well-aerated soil conditions (Rochette et al., 2008)—suggesting that high rates of N_2O production would be observed if the moisture limitation was removed. While, in the present study, this appears to hold true for the dryland cropping system, results from the irrigated cropping system suggest otherwise. Although small peaks in N_2O emissions were observed following irrigation events in both study years, periods of high soil moisture during the summer months did not elicit a large N_2O emission response (see Figures 3.3 and 3.5), as would be expected if mineral N availability were non-limiting. At 56% soil porosity—a conservative estimate based on measured bulk density and a soil particle density of 2.65 g cm^{-3} (Blake, 2008)—the optimal moisture levels for N_2O production is expected to occur at around 39% VWC or 70% water filled pore space (Linn and Doran, 1984). Soil moisture levels up to and exceeding 40% VWC were observed during multiple periods throughout both study years (Figures 5.2 through 5.5; Panel b) without large N_2O fluxes (i.e., > 5 kg N_2O-N ha^{-1} d^{-1}), suggesting that mineral N availability, rather than low denitrification activity, may be limiting N_2O production in these irrigated production systems.

The results of the correlation analysis support these observations. The strong correlation between daily N_2O fluxes and soil VWC content observed in the dryland system, coupled with the greater quantities of soil N available after harvest (in 2013, at least), suggests that soil N_2O production in this system was limited by low soil moisture (low potential for denitrification activity). In contrast, a much weaker correlation was observed in the irrigated cropping system, as high VWC conditions from July onward did not stimulate elevated N_2O emissions. This was likely due to a reduction in soil-N availability later in the growing season resulting from crop uptake. The N use efficiency of the 2013 irrigated canola crop was greater than that of the dryland, with a lower proportion of fertilizer-N lost as N_2O (0.5% *vs.* 0.8% of applied N; Appendix C.2).

The fall 2012/spring 2013 comparison of the three study sites (DL, IR12, and IR13) highlights the range in magnitude of N_2O emissions that can occur under relatively similar site conditions within the same study year (spatial variation). The much higher emissions observed at the IR12 site were likely a result of the saturated soil conditions (high soil moisture from fall irrigation and additional water from melting of the winter snowpack), as fall applied N-fertilizer rates at the other sites were either equal to or greater than those at IR12 (DL and IR12 = 78 kg N ha^{-1}; IR13 = 90 kg N ha^{-1}) and available soil N at fall sampling (prior to anhydrous ammonia application) did not differ between the three fields. Yet, crop residue degradation may have played a role in elevated emissions. Even though both the IR12 and DL sites were cropped to wheat in 2012 (IR13 was cropped to dry beans), the combination of greater biomass yields (3530 ± 510 kg ha^{-1} *vs.* 2830 ± 490 kg ha^{-1}) and incorporation of residues in the fall of 2012 (DL residues were not incorporated) may have provided a source of additional N (total of ~38 kg N ha^{-1} in biomass residue, though only a fraction would likely contribute to N_2O). Indeed, Gregorich et al. (2005) found that incorporation of stubble residues in the fall can lead to higher N_2O emissions than if residues are left on soil surface.

The observed seasonal CO_2 trends followed crop growth trends, which is not surprising considering the chamber-based methodology used. Carbon dioxide fluxes measured by plant-excluded chambers capture the combination of soil microbial respiration and root respiration products, and, although relative contributions are difficult to separate, estimates suggest that up to 50% of soil CO_2 emissions can be attributed to plant root respiration (Rochette et al., 1999; Curtin et al., 2000). In addition to this over-arching seasonal trend, elevated soil moisture levels appeared to stimulated CO_2 emissions responses, with greater flux variability and median daily fluxes observed after large precipitation events. Most notable are the high-volume rainfall received during mid-June and early July 2013; elevated emissions were measured at both the DL and IR13 sites (Figures 3.4 and 3.5). A similar response was observed in 2012, after the mid-July precipitation events at the DL and IR12 sites, and the combined irrigation/precipitation in late-July/early-August at IR12. It is well understood that elevated soil moisture (under aerobic conditions) promotes soil respiration, and a positive correlation between soil moisture and CO_2

evolution has been observed in other irrigated studies (Ellert and Janzen, 2008; Jabro et al., 2008; Sainju et al., 2012; Trost et al., 2013). In the present study, correlations between CO_2 flux and VWC were weak and limited to the irrigated site in 2012 (IR12) and the dryland site in 2013 (DL). This poor correlation is likely due to periodic low-flux days where soil water content is high (>70% WFPS) creating low oxygen/anoxic soil conditions.

In a single-year, irrigated-dryland comparison in North Dakota, greater overall CO_2 emissions were observed from irrigated treatments due to the elevated soil water content brought about by irrigation activities (Jabro et al., 2008). In contrast, the cumulative annual CO_2 emissions in the present study were greater from the DL compared to IR13, even though soil moisture at IR13 was greater. Crop rotation effects were likely responsible for this difference as the irrigated cropping systems included dry bean (*Phaseolus vulgarus*) in their rotation. Consider that in 2012, the IR13 site was cropped to dry beans while the DL was cropped to wheat. Compared to dry bean, wheat crops produce greater residual biomass (Gan et al., 2009) with a wider C to N ratio, providing a large—and highly labile—pool of C for microbial degradation, resulting in greater CO_2 emissions. In the 2012 season, both fields were grown on canola residue, which explains the comparable emissions observed during this study year.

The capacity for atmospheric CH_4 uptake (oxidation) by soils was greatly diminished with irrigation, likely due to the elevated soil moisture conditions as others have observed (Mosier et al., 2006; Liebig et al., 2010; Sainju et al., 2012). The increased soil disturbance associated with irrigated cropping (i.e., management of dry bean, tillage in some years at fall N application) may have also contributed to lower CH_4 uptake, as disturbance has shown clear negative effects on soil oxidation in some cases (Alluvione et al., 2009), however other studies in the northern Great Plains have shown no effects from disturbance (Mosier et al., 2006; Liebig et al., 2012). In both cropping systems during both study years, greater CH_4 uptake was observed during periods of low soil moisture, supported by the strong positive correlation between CH_4 flux and VWC. Due to the design of the study, it was difficult to determine if the greater N fertilizer rates applied to the irrigated cropping systems played a role in inhibiting CH_4 uptake. Results from other irrigated cropping studies in the northern Great Plains indicate that N fertilizer application generally has no influence on atmospheric CH_4 uptake (Mosier et al., 2006; Liu et al., 2006; Ellert and Janzen, 2008; Halvorson et al., 2011), thus an inhibitory effect is unlikely the case in this study.

3.6 Conclusions

Certainly, compared to dryland cropping systems, irrigated crop management results in changes to seasonal soil conditions. In addition to supplemental water application, producers enhance soil fertility (especially N) with greater rates of fertilizer, adopt a more aggressive tillage regime (i.e., tillage before fall N applications, inter-row tillage for weed control, etc.), and grow slightly different suite of crops which are often not suitable for rainfed cropping conditions (i.e., potato, dry bean, other pulse crops). In the present study, the most notable change in soil conditions brought about by irrigation was the elevated soil N levels and moisture trends. Elevated soil moisture appeared to influence the flux dynamics of all three agricultural greenhouse gases, leading to lower CH_4 uptake, and periodic increases in CO_2 and N_2O emissions. The greater fertilizer rates common to irrigated crop management creates potential for high N_2O emissions, however this may be realized in short lived, episodic bursts of N_2O emission without substantially increasing cumulative N_2O emissions from irrigated cropping systems—as was observed in the 2013 season. Aside from the periods immediately following soil thaw and seeding, daily N_2O emissions were not drastically different between the dryland and irrigated sites—despite much greater N fertilization under irrigated management. Due to the removal of soil moisture limitation, nitrous oxide emissions dynamics from irrigated cropping appear to be limited by N availability rather than low denitrification activity—underscoring the importance of effective fertility management in irrigated cropping systems.

4. NET GLOBAL WARMING POTENTIAL AND GREENHOUSE GAS INTENSITIES OF A TYPICAL IRRIGATED AND DRYLAND CROPPING SYSTEM IN THE SEMI-ARID CANADIAN PRAIRIES

4.1 Preface

As discussed in Chapter 3, soils of irrigated cropping systems have potential for greater N_2O and CO_2 emissions and lower atmospheric CH_4 uptake than their dryland (rainfed) counterparts, due to greater fertilizer rates and wetter soil conditions. However, when assessing the effects of irrigation on the agricultural greenhouse gas balance, it is important to consider all sources of greenhouse gas (GHG) emissions and their global warming potential (GWP). In addition to soil emissions, irrigated cropping may contribute greater GHG emissions due to the more intensive crop management that characterize these systems. Irrigated crop management requires energy for pumping water, additional field operations (i.e., tillage, spraying), and relies on supplemental fertilizer and chemical applications to safeguard prospective yields—all of which have an associated cost in respect to greenhouse gas emissions. The goal of this study was to construct and compare emission budgets for a dryland and an irrigated cropping system typical of Saskatchewan, incorporating only on-site sources of greenhouse gas emissions. Considering the differing levels of productivity between systems, emissions have been yield-scaled and compared in terms of greenhouse gas intensity (GHGI).

4.2 Introduction

As global food demand rises with a growing world population [a projected increase of 2.25 billion over the next 40 years (Alexandratos and Bruinsma, 2012)], expansion of irrigated crop land within the Canadian Prairies—particularly in Saskatchewan—will become increasingly important in addressing national and global food security (Madramootoo and Fyles, 2011). Yet, how this expansion will impact the region's agricultural greenhouse gas inventory is not well understood. Crop production plays a major role in the release of N_2O, CO_2, and CH_4 to the atmosphere through soil emissions and emissions associated with management activities. Compared to dryland production systems, irrigated cropping requires more intensive management and has a greater reliance on energy—both electrical and fossil fuel-based—for pumping water, pivot operation, and additional cropping operations. Thus, when assessing the net GHG balance of these cropping systems, emissions associated with energy usage must be considered.

Energy is used to produce and/or transport all agricultural inputs; thus, each input has an associated GHG “cost”. The combination of the GHG embodied within the inputs and the GHGs released during crop production make up the GHG “cost” of crop system outputs (i.e., grain yield). Greenhouse gas intensity (GHGI) is used to express the total GHG emissions per unit output, and for agricultural cropping systems is expressed as CO_2-equivalents (CO_2-eq) per kg seed yield. Irrigated crop production relies on a number of inputs, including fertilizer, chemicals (herbicide, fungicide, pesticide, etc.), irrigation water, and fuel (for machinery). Inefficiencies in use and/or losses of inputs represent GHG release without benefit to crop production, and increases the GHGI of the system. Consider irrigation water inputs; energy is used to pump water from the source (i.e., Lake Diefenbaker) into canals and then from the canal onto the field. During this process, water can be lost via evaporation (from the canal, crop, or soil), deep percolation, and runoff, resulting in GHG emissions without benefit to crop production. Similarly, N-fertilizer losses through runoff, leaching, denitrification, and volatilization—as well as creating potential for N_2O emissions—represent GHG losses associated with the production and transport of the fertilizer. Although some degree of inefficiencies and losses are inevitable, minimizing their magnitude is important to the sustainability of irrigated crop production. Few studies have aimed at quantifying the net GHG balance of irrigated cropping systems in the semi-arid Prairie region, thus, focusing on the major on-site sources and sinks for greenhouse gas emissions is an important first step in better understanding the GHG dynamics of these systems. The conceptual schematic presented Figure 4.1 highlights the major GHG sources/sinks for irrigated crop production.

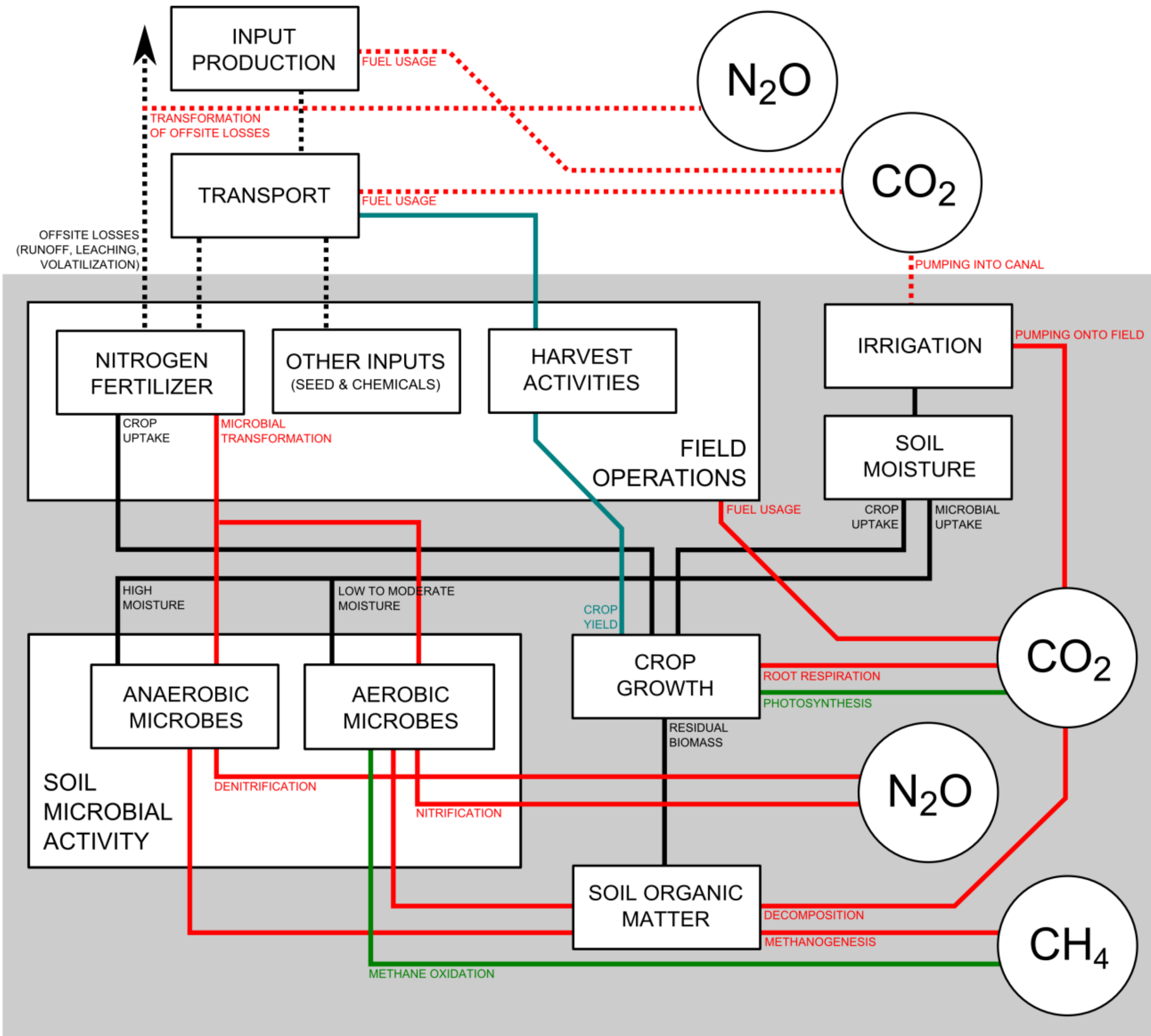

Figure 4.1 A conceptual schematic of greenhouse gas (GHG) sources and sinks for irrigated crop production. Greenhouse gases are presented in circles and sources/sinks of GHG are presented in white boxes. The budget boundaries of the present study (i.e., “the farm-gate”) are indicated by the grey box. Flows occurring within and outside of the budget boundaries are indicated by solid lines and dotted lines, respectively. Greenhouse gas emissions are represented by red lines while reductions (GHG uptake) are represented in green. Black lines indicate a connection/relationship. The processes or conditions associated with each flow are noted in the corresponding color scheme.

Diesel fuel is the dominant energy source used to power agricultural machinery (Follett, 2001). In Canada, 25 billion liters of diesel is consumed annually, primarily for powering agricultural equipment and transport trucks (Boehm et al., 2006). Carbon dioxide is the main gas emitted from diesel combustion (2663 g CO_2 per L), with minor emissions of CH_4 and N_2O (0.15 and 1.1 g per L, respectively) contributing to a total emission factor of 2995 g CO_2-eq per L diesel (Environment Canada, 2010b). On-farm use of diesel fuel accounts for emissions of roughly 8.5 Mt of CO_2-eq annually (Environment Canada, 2010a).

Three phase electricity and combustion engines are the most common power sources used to pump irrigation water. Electricity is the preferred method in areas with access to suitable grid-supplied power, while fossil fuel powered combustion engines are used at less accessible sites. With the exception of electricity generated via nuclear or renewable resources (i.e., wind, tidal, hydro, or solar), energy used on-site has an associated cost in terms of CO_2 emissions. The extent of the emissions is dependent on fuel source and method of electricity generation. Electricity is the most common power source in the Canadian Prairies, where the majority of irrigated agriculture is concentrated into irrigation districts and is accessible to the grid. Methods of generating electricity differs between regions, thus, each province has specific emission multipliers (Environment Canada, 2010b). Consider the Prairie Provinces for example; electricity production in Saskatchewan and Alberta is dominated by coal and natural gas combustion resulting in high GHG intensities (710 and 880 g CO_2-eq kWh^{-1}, respectively) when compared to the national average (220 g CO_2-eq kWh^{-1}); whereas in Manitoba, nearly all electricity (97%) is generated via hydro, resulting in a low overall GHGI for electricity production (10 g CO_2-eq kWh^{-1}).

In targeting higher yields, irrigated crop producers typically rely on greater rates of N fertilizer. Targeted fertility rates are largely dependent on crop type, but it is not uncommon for N rates to exceed those of dryland production by 50 to 100% [based on rates recommended by ICDC (2012) and Saskatchewan Ministry of Agriculture (2013)]. The correlation between N fertilizer rate and N_2O emissions is well known; with higher fertilizer rates favoring greater emissions (Bouwman, 1996; Environment Canada, 2011; van Groenigen et al., 2011; Kim et al., 2013). Soil emission trends may be further complicated by the change in soil moisture regime brought about by irrigation. Soil moisture stimulates soil microbial activity and strongly regulates soil aeration (Linn and Doran, 1984). Saturated conditions favor anaerobic microbial activity—specifically, methanogenic and denitrifiying microbial communities—which promotes emissions of CH_4 and N_2O (Corre et al., 1996; Izaurralde et al., 2004). Under aerobic soil conditions, elevated moisture levels stimulate decomposition and nitrification activity, leading to the release of CO_2 and N_2O from the soil (Robertson and Grace, 2004; Ellert and Janzen, 2008) as well as a reduction in CH_4 uptake by methane oxidizing bacteria in soils (Liu et al., 2006; Sainju et al., 2012). Yet, the greater productivity realized through irrigation may act to offset some of these emissions.

Greater biomass production allows for greater returns of C as crop residue. If residue returns can be maintained and incorporated into the SOM pool, there is potential for C sequestration and GHG mitigation within these systems (Follett, 2001; Flynn and Smith, 2010). Whether the residual biomass C is incorporated to the soil C pool or lost to the environment is largely dependent on the degree of decomposition—a function of soil moisture and disturbance—in the system. In some cropping systems, even though C inputs are high, total soil C may not accumulate due to rapid SOM cycling (McGill and Cole, 1981; Jantalia and Halvorson, 2011; Liebig et al., 2012). Such has been the case for some irrigated systems in the Canadian Prairies (Bardak-Meyers, 1996), although research data on SOM dynamics and sequestration potential under irrigation is scarce (Follett, 2001). Irrigated crop management typically involves a greater degree of soil disturbance than dryland management—a result of including tillage-intensive crops like potatoes and beans in the crop rotation. Producers also use mechanical tillage to alleviate soil compaction, as a form of weed control, and to prepare the seedbed for planting.

A field-level analysis incorporating all on-site contributions to the net global warming potential is required to understand the influence of irrigated cropping on radiative forcing. Two methods are commonly used to calculate the net global warming potential (GWP) of GHG emissions from cropping systems, one based on soil organic C status and the other on soil respiration. Both methods include emissions source such as soils and fuel usage for cropping activities (and others, depending on the scope of the study), the difference lies in how the net CO_2 balance is determined. Calculation of net GWP by the soil organic carbon method (GWP_C) considers the soil C sequestration rate as a sink for GHG emissions and does not include measurements of soil respiration (Robertson et al., 2000; Robertson and Grace, 2004; Mosier et al., 2005, 2006; Sainju et al., 2014). Net GWP calculations based on respiration (GWP_R) treat soil respiration as a source of CO_2 emissions and the residues of the previous year's crop as a CO_2 sink (Mosier et al., 2006). As discussed by Sainju et al. (2014), each method has potential drawbacks; GWP_C depends on a long-term quantitative measurement of C sequestration rate—because SOM changes little from year-to-year—and GWP_R requires accurate quantification of soil respiration and crop residues returned in the previous cropping year. Studies that have compared the two calculation methods suggest that sink potentials estimated by GWP_R are greater than those estimated using GWP_C (Mosier et al., 2006; Sainju et al., 2014). Overestimating a sink potential can be chancy, especially when comparing alternative cropping systems as the present study does, as the analysis is likely to favor the cropping system with greater yearly biomass yields (i.e., irrigated crop production).

The objective of this study was to determine the effect of irrigated cropping on the agricultural greenhouse gas balance, relative to conventional, dryland (rainfed) crop production. This was achieved by constructing system-specific GHG emission budgets for irrigated and dryland crop production and comparing these two systems in terms of global warming potential GWP and GHGI.

4.3 Materials and Methods

4.3.1 Study site

The study area, consisting of one irrigated quarter section and one dryland quarter section (IR13 and DL, respectively) is located approximately 75 km south of the city of Saskatoon and is within the South Saskatchewan River Irrigation District (Figure 4.2). The irrigated site has been managed under irrigation since the installation of the center pivot irrigation system in 1979. During the early 1990s, the pivot was converted to a lower pressure system (30 m of head; 42 psi) with drop tubes and rotating spray plate sprinklers. Irrigation water was supplied by a 3-stage vertical turbine pump (Berkeley Pump Co, USA) powered by a 29 kW (40 horsepower) induction motor (General Electric, USA), with a capacity of 3.6 m^3 min^{-1} (950 US gal min^{-1}) at a total dynamic head (TDH) of 44 m (62 psi).

Both fields were managed by the same producer, and were planted to hybrid Canola (*Brassica napus*, InVigor L130; Bayer CropScience Canada) in 2013. The study year encompassed one full crop year—from fall fertilizer application in October 2012 through

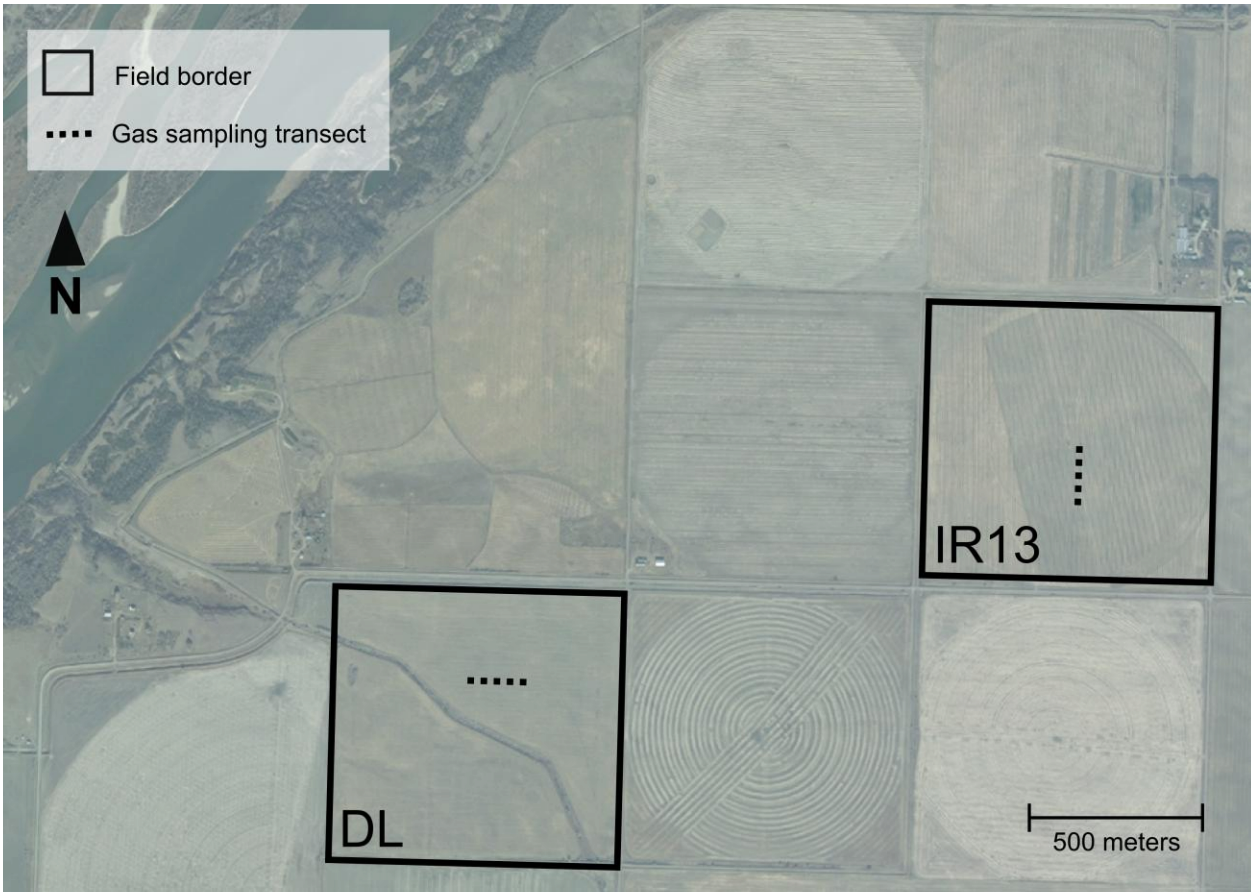

Figure 4.2 An aerial view of the study site. The southeast quarter of section 16 (IR13; 51°39’09”N, 106°55’51”W) is managed under irrigated crop production. The dryland study site is located at the northeast quarter of section 8 (DL, 51°38’41”N, 106°57’16”W). Photo credit: FlySask (2011).

October 2013. Crop yields were determined using an electronic crop yield monitor installed on the combine thresher and verified by the total grain yield harvested and sold by the producer. The close proximity of the two fields meant that rainfall and climatic conditions at the two sites were comparable, and that variations in soil texture were minimal. The soils at both sites are classified Orthic Dark Brown Chernozemic with a loam Ap horizon (Appendix B). Site management and additional soil characteristics were summarized in Chapter 3 (Table 3.3) and Appendix B, respectively.

4.3.2 GHG budget boundaries

This investigation was limited to on-site sources and sinks of GHG emissions, including emissions from soils, fuel consumed during farming operations, and the emissions associated with the production of the electricity used for irrigation operations. These boundaries were chosen to reflect the GHG sources and sinks that are within the control of an individual producer at “the farm gate” (i.e., on-site). Consequently, emissions associated with fertilizer and chemical production, off-site transportation, and the delivery of irrigation water to the site (via canal) were not included. The budget boundaries of the current study are represented by grey box in the conceptual schematic presented in Figure 4.1.

4.3.3 Soil emissions

Soil emission of N_2O and CH_4 were quantified using vented rectangular chambers (22 cm × 45.5 cm × 15 cm; width × length × height) constructed of 6-mm PMMA (headspace volume = 10 L; soil surface area = 1000 cm^2). Twenty chambers were installed at a spacing of 6.25m along a 125-m linear transect in the fall of 2012. Transects were oriented in the direction of seeding with every second chamber offset from the transect centerline by one meter. Plants were excluded from the chamber and disturbed seed rows were replanted along the outsider perimeter of the chamber base. Gas sampling occurred from fall anhydrous ammonia application in 2012 (19 Oct 2012) to freeze-up (24 Nov 2012). During the winter months (December through March), gas sampling was not conducted as N losses are assumed to be negligible during this period (Malhi et al., 2001). Sampling resumed again at the start of soil thaw in spring 2013 (26 Apr 2013) and continued up to fall anhydrous application in 2013 (04 Oct 2013). Sampling frequency was greatest during the spring months when the highest and most variable soil emissions were expected to occur.

Headspace air samples were collected through rubber septa using a 20 mL syringe fitted with a 25-gauge needle, and immediately transferred into pre-evacuated (0.5 kPa) 12-mL Exetainer^{TM} vials (Labco Limited, UK) for transport to the lab and subsequent analysis by gas chromatography (Bruker 450 GC, Bruker Biosciences Corporation, USA) (Farrell and Elliott, 2007). Samples were collected at 15-min intervals during a 45-min deployment period following closure of the chamber lids. In addition to the chamber gas samples, four ambient air samples were collected before and after chamber sampling to determine baseline gas concentrations and the minimum detectable concentration difference (MDCD) (Yates et al., 2006a). Daily fluxes were taken as the slope of the fitted regression at t_0 as recommended by the HMR model (Pedersen et al., 2010; Pedersen, 2011), a modified Hutchinson-Mosier method available as an add-on package for the R environment (R Development Core Team, 2011). However, if the concentration differences between each subsequent time step did not exceed MDCD (Yates et al., 2006a) the fluxes were taken as the slope of the linear regression at t_0. This allowed for the calculation and inclusion of statistically non-significant fluxes (i.e., below the MDCD; common to CH_4 fluxes) in the dataset to minimize left censoring (Ens, 2012). Cumulative annual fluxes were calculated using measured median daily flux rates and estimates of non-sampling days made by linear interpolation (Pennock et al., 2006).

The recommendations proposed by Baker et al. (2007) for minimizing potential problems related to chamber techniques for gas flux measurement [summarized by (Mosier, 1989)] were considered in the design and operation of the chambers for this study. The actions taken to minimize temperature and pressure perturbations and to account for spatial and temporal variability are outlined in Chapter 3, Section 3.3.2.

4.3.4 Net carbon exchange

In quantifying net CO_2 exchange in this study, the relative difference in SOM stocks between the irrigated and dryland cropping systems were evaluated. Before the installation of the irrigation system, both fields were subject to similar cropping and management history, and if we assume—as others have (Gillabel et al., 2007)—that SOM content was at equilibrium when irrigation began and has not changed under dryland conditions, the rate of change can be attributed to irrigated cropping. This method was preferred over a GWP_R-type analysis for two reasons. First, due to differences in crop rotation between systems, the 2013 canola crop was grown on different residues (irrigated grown on bean residue, dryland grown on wheat residue). Different residue composition (i.e., C:N ratio) can be expected to have a large effect on soil respiration and resulting CO_2 dynamics (as was observed in Chapter 3), making single-year comparisons weak. Secondly, as noted previously, the GWP_R method lends to greater estimates of sink potential (Mosier et al., 2006; Sainju et al., 2014), which could potentially overstate the importance of irrigation to CO_2 sequestration. Since the purpose of the present study is to compare the effect of irrigation management on GHG dynamics, it is better to take a conservative approach and risk underestimating sink potential, so that prospective mitigation opportunities cannot be overstated.

To quantify SOM stocks, soil samples were collected adjacent to each gas sampling chamber at 0–15 cm and 15–30 cm depths. Samples from every four chambers were bulked, and subsampled for analysis. Carbon content was determined by combustion analysis (LECO C632; LECO Corporation, USA) using subsamples that were pretreated to remove carbonates (Harris et al., 2001). Difference in SOM between cropping systems was attributed to irrigated crop management over the 35 years since center pivot irrigation began at the study site and was expressed as average gain/loss of C per year.

4.3.5 Emissions from cropping operations

Emissions from on-site fuel and electricity usage were estimated using emission factors from the Canada's National Inventory Report (Environment Canada, 2010a). A record of diesel fuel usage, provided by the producer, was applied to the "Off-road Diesel" multiplier of 2.99 kg CO_2-eq L^{-1}. Electricity usage was determined from the pre-existing electricity meter, and applied to the Saskatchewan-specific multiplier for electricity generation—0.71 kg CO_2-eq kWh^{-1} (Environment Canada, 2010c).

4.4 Results

No measurable difference in SOM was found between cropping systems, with SOM quantities of the DL and IR13 site measured at 1.78 ± 0.44% and 1.76 ± 0.25% at 0–15 cm, and 1.18 ± 0.17% and 1.02 ± 0.20% at 15–30 cm, respectively (Appendix B.5). Thus, the yearly rate of SOM accumulation was zero (based on the change in SOM stocks over 35 years of irrigated cropping).

On-site diesel fuel usage for the dryland cropping system totaled 3410 L for two fertilizer operations (anhydrous ammonia application in the fall of 2012 and granular application in spring 2013), seeding, harrow packing (following seeding), a single herbicide application, and crop swathing and harvest. The irrigated system received an additional chemical application (fungicide) and required more fuel at harvest for swathing and harvest (due to greater biomass). Thus, total fuel usage for the irrigated cropping system was slightly higher at 3760 L. The resulting emissions were 176 and 211 kg CO_2-eq ha^{-1} for the dryland and irrigated systems, respectively (Table 4.1). A total of 31970 kWh of energy was required to operate the pivot in 2013, which applied 176 mm of water, and contributed 429 kg CO_2-eq ha^{-1} to the irrigated systems net GWP total.

Soils of the cropping systems were sources of N_2O emissions and small sinks for atmospheric CH_4. Cumulative annual emissions (351 days; 19 Oct 2012 to 04 Oct 2013) from the dryland cropping system were 871 g of N_2O-N (*median*; Q_1 = 679 g N ha^{-1}; Q_3 = 1050 g N ha^{-1}), contributing 408 kg CO_2-eq ha^{-1}. Methane uptake was 1260 g CH_4-C ha^{-1} (*median*; Q_1 and Q_3 = -1618 and -1082 g CH_4-C ha^{-1}) over the same period, and represented a reduction of 42 kg CO_2-eq ha^{-1}. In the irrigated cropping system, cumulative median annual N_2O emissions were 962 g N ha^{-1} (Q_1 and Q_3 = 844 g N ha^{-1} and 1502 g N ha^{-1}, respectively) and cumulative median annual CH_4 emissions were -226 g CH_4-C ha^{-1} (Q_1 and Q_3 = -365 and -102 g CH_4-C ha^{-1}, respectively). Nitrous oxide emissions contributed 450 kg CO_2-eq ha^{-1} to the net GWP, while CH_4 uptake reduced the net GWP by 8 kg CO_2-eq ha^{-1}.

Considering all emission sources and sinks within "the farm gate", the 2013 net GWP for the irrigated cropping system was twice that of the dryland cropping system at 1082 and 542 kg CO_2-eq ha^{-1}, respectively (Table 4.1). However, in addition to greater emissions per hectare, irrigated cropping also produces greater yields per hectare. Thus, to truly compare the relative

Table 4.1 Emission sources contributing to the net global warming potential (GWP) in the irrigated and dryland cropping system in 2013. Values in parenthesis represent uptake of atmospheric greenhouse gases. Both systems were planted to hybrid Canola (*Brassica napus*, InVigor L130; Bayer CropScience Canada). Values in parentheses represent negative emissions or atmospheric GHG uptake.

	CO_2			N_2O[§]	CH_4[¶]	Net GWP
	Δ Soil C[†]	Fuel[‡]	Electricity[‡]	Soil	Soil	
	kg CO_2*-eq ha*$^{-1}$					
Dryland cropping	--	176	--	408	(42)	542
Irrigated cropping	0	211	429	450	(8)	1082

[†] The dryland cropping system is used as the reference for change in soil C for the irrigated system.
[‡] The small amounts of N_2O and CH_4 produced during fuel combustion and electricity production have been included.
[§] GWP based on 100-year time horizon using IPCC (2007) multiplier of 298.
[¶] GWP based on 100-year time horizon using IPCC (2007) multiplier of 25.

Table 4.2 Greenhouse gas intensities (GHGI) of the irrigated and dryland cropping system in the 2013 cropping season. Both sites were planted to hybrid Canola (*Brassica napus*, InVigor L130; Bayer CropScience Canada).

	Soil	Field operations[†]	Irrigation[‡]	Total
	kg CO_2*-eq kg*$^{-1}$*canola yield*[§]			
Dryland cropping	0.152	0.074	--	0.226
Irrigated cropping	0.123	0.059	0.119	0.301

[†] Emissions from diesel fuel combustion by agricultural machinery.
[‡] Emissions associated with the production of electrical energy used for pumping irrigation water and operating the irrigation system.
[§] Yields obtained from the electronic yield monitor on the combine harvester and verified by the total grain yield harvested and sold by the producer.. Yields were 2400 and 3600 kg ha^{-1} for the dryland and irrigated systems, respectively.

effect of irrigation management on radiative forcing, emission must be considered in terms of yield. The average canola yield from the irrigated study site was 3600 kg ha^{-1}—1.5 times greater than the 2400 kg ha^{-1} yield produced under dryland conditions. However, despite the greater productivity, the GHGI of the irrigated cropping system remained greater than the dryland system (301 g CO_2-eq kg^{-1}canola seed *vs.* 266 g CO_2-eq kg^{-1}canola seed; Table 4.2).

4.5 Discussion

The 2013 growing season produced better-than-average yields under both irrigated (3600 kg ha^{-1} *vs.* 3000 kg ha^{-1}) and dryland conditions (2400 kg ha^{-1} *vs.* 1900 kg ha^{-1}) [average yields obtained from ICDC (2012) and Saskatchewan Ministry of Agriculture, (2013)]. Annual precipitation was typical for the region (*20 year normal* = 338 mm; IR13 = 332 mm and DL = 309 mm; see Table 3.3) and was received primarily as rainfall (228 mm) during the early summer months. As a result, irrigation quantities (127 mm) were also typical for the 2013 season. The irrigation quantities applied in 2013 were well within seasonal irrigation estimates for canola from the past decade (*mean* = 121 mm; *95% confidence interval* = 58 mm and 183 mm) [calculated using calculated using AIMM climate data from 2004 to 2013 (Alberta Agriculture and Rural Development, 2014)]. Details of the calculation are provided in Appendix E.

Soil N_2O emissions were notable contributors to GWP and GHGI in these cropping systems. Compared with the few irrigated canola studies conducted in this region, the 962 g N_2O-N ha^{-1} 351 d^{-1} (2.74 g N_2O-N ha^{-1} d^{-1}) measured from the irrigated cropping system was on the low end of the values reported by Hao et al. (2001) (2.50 to 15.64 g N_2O-N ha^{-1} d^{-1}) and CSIDC (2013) (1675 to 3310 g N_2O-N ha^{-1} from June to mid-October; approximately 140 days). Consistent with other studies, soil N_2O emissions strongly influenced the net GWP trend from both cropping systems (Robertson and Grace, 2004; Mosier et al., 2006). Nitrogen availability appears to be the limiting factor in soil N_2O emissions in this irrigated system (see Chapter 3); thus, reductions in soil emissions may be achieved through more closely managed fertility regimes. Yet, the lower intensities for soil GHG emissions observed with irrigated cropping suggests that, when compared to dryland production, soil emissions are already well managed under irrigated crop production. A rough calculation of N use efficiency at the site suggests that around 0.5% of the N applied to the irrigated system was lost as N_2O, lower than the 0.8% lost from the dryland cropping system (Appendix C.2). Additional reductions may be realized by reducing or eliminating N fertilization in the fall, leading to lower quantities of soil N available for microbial transformation at spring thaw—the period where greatest N_2O emissions occur (Hao et al., 2001; Burton et al., 2008).

Methane uptake by the soils of the cropping systems contributed small reductions in net GWP for each system; however, these reductions are minor relative to the on-site emission sources. Thus, as Ellert and Janzen (2008) have stated previously, "e*fforts to reduce net emissions of greenhouse gases from the soil might better focus on N_2O sources rather than on CH_4 sinks*".

Irrigation has potential to increase SOM levels and mitigate CO_2 emission through high biomass yields and residue returns (Follett, 2001). A recent review of GHG dynamics under irrigated cropping found that in semi-arid regions irrigation increased SOM by 11 to 35% (Trost et al., 2013). However, in the present study, emission offsets in the form of SOM storage were not realized, as no measurable difference in SOM was found in the top 30 cm of soil after 35 years of irrigated cropping (Appendix B.5). Similarly, Bardak-Meyers (1996) found no increase in SOM content of irrigated soils at multiple sites within the same region and attributed the lack of C sequestration to rapid C turnover due to the addition of young, labile organic matter which is readily decomposed. Gillabel et al. (2007) observed C accumulation under irrigation in Nebraska (arid climate) and proposed that C storage occurred through stabilization and physical protection inside soil microaggregates. The researchers also noted faster cycling of C under irrigated conditions, due to an increase in decomposition favored by the removal of soil moisture limitation (Gillabel et al., 2007). The lack of C storage observed in the present study, in spite of the high residue returns, is likely due to rapid turnover expedited by a change in soil moisture regime and increased soil disturbance, resulting from irrigated crop management. Certainly, many studies have documented the negative response soil disturbance has to C storage (Follett, 2001; Robertson and Grace, 2004; Alluvione et al., 2009; Liebig et al., 2012; Trost et al., 2013). Producers may realize soil C gains by reducing tillage intensity, however, much of the disturbance occurring is inherent to the crops grown under a typical irrigated cropping rotation (i.e., hilling of potatoes, inter-row cultivation of dry beans). Thus, the GHG mitigation opportunities represented by soil C storage may be limited in irrigated cropping systems in this region.

Irrigated crop production is expected to have a greater fuel requirement than dryland cropping, due to additional field operations (i.e., spraying, tillage) and greater quantities of biomass to cut and harvest. In 2013, diesel fuel associated GHG emissions were greater from irrigated cropping by 20%. Relative fuel usage during typical growing seasons, like 2013 (i.e., low incidence of disease, no need for re-seeding), is not likely to fluctuate drastically due to similarities in field operations between cropping systems.

In this single year comparison, irrigated cropping demonstrated greater net GWP and GHGI than conventional dryland cropping. However, the source-by-source comparison indicates that the difference is entirely due to the energy consumed for irrigation (pumping and moving the pivot), the single largest contributor to the net GWP (429 kg CO_2-eq per ha). The high GHG emissions associated with electricity are due to the fossil-fuel-intensive methods of electricity production in Saskatchewan. Currently, greater than 75% of the electricity in Saskatchewan is produced through the combustion of coal, natural gas, and refined petroleum products (Environment Canada, 2010c), which releases large quantities of CO_2 in the process. Considering that irrigation quantities applied in 2013 were typical of the 10-year normal, electricity usage was also typical. Based on the mean seasonal precipitation (May through September) 95% confidence interval, electricity usage for the irrigation system at the study site would typically be between 10564 kwh and 33548 kwh and would contribute between 143 kg CO_2-eq ha^{-1} and 454 kg CO_2-eq ha^{-1} to the net GWP. Clearly, electricity is a substantial source of emissions for irrigated cropping systems and large reductions in GHGI could be realized by reducing on-site electricity requirements.

Individual producers have little control over electricity generation that occurs off-site, however producers have the opportunity to make changes to their irrigation application systems (i.e., pivots) that decrease electricity requirements. A reduction in electricity usage may be achieved by irrigating with a lower water pressure requirement. With a conversion to a lower sprinkler operating pressure (10 m TDH; 15 psi), estimated energy use may be reduced by as much as 25% (440 kWh ha^{-1} *vs.* 600 kWh ha^{-1}) based on 2013 irrigation requirements (Alberta Agriculture and Rural Development, 2008). This reduction in energy input would reduce pumping associated emissions by over 100 kg CO_2-eq ha^{-1}. Further reductions in electricity associated GHG reductions may be realized through lower irrigation requirements via increased application efficiency. Existing pivots can be retrofit to low energy precision application (LEPA) irrigation (Lyle and Bordovsky, 1981), which uses low hanging applicators to deliver irrigation water at lower pressures and greater application efficiencies than spray irrigation [efficiencies of 95-98% *vs.* 90% (Schneider, 2000)]. However, the cost of a LEPA conversion combined with changes in management (planting in a circle, changes to row spacing) may be a limitation for adoption by producers.

The most effective means of reducing pumping-associated GHGI is to generate electricity using less emission intensive energy sources. In contrast to Saskatchewan's coal-dominated electricity production, Manitoba generates over 97% of its electricity through hydro and, as a result, has a very low emission intensity—10 g CO_2-eq kWh^{-1} *vs.* 710 g CO_2-eq kWh^{-1} (Environment Canada, 2010c). Consider the present cropping system comparison if it was located in Manitoba instead of Saskatchewan, a reasonable proposition considering Manitoba's similar climate and growing conditions. The GWP for pumping irrigation water would be only 10 kg CO_2-eq ha^{-1}, lowering overall GHGI of the irrigated cropping system to 0.164 CO_2-eq kg^{-1} canola. At this rate, irrigated crop production would be less GHG intensive than dryland cropping. Although hydroelectricity production is not without environmental consequence, the example highlights the potential for less emission-intensive food production through irrigation if more sustainable sources of electricity are used.

Drawing system boundaries is necessary when assessing net GWP and GHGI of productions systems. The boundaries of the current study did not included the production and transport of inputs to these agricultural systems, which are obvious sources of GHG emissions for agricultural cropping systems. Results of a full C cycle analysis for cropping agriculture in the US (West and Marland, 2002) demonstrate that, when compared to on-site sources, the production of chemical are a relatively minor source of emissions (13 and 56 kg CO_2-eq ha^{-1} for conventional till wheat and corn, respectively). In contrast, the production of nitrogen fertilizer can be a substantial source of emissions (267 and 383 kg CO_2-eq ha^{-1} for conventional till wheat and corn, respectively). However, whereas the emissions associated with these industrial processes are beyond the producer's control, including such sources becomes important at a policy-making level.

4.6 Conclusions

In the present study, soils were the largest contributor to GHGI in both cropping systems, while the GHGI of field operations (on-site diesel fuel usage) was the lowest. Irrigated cropping demonstrated a greater overall GHGI, yet, this was entirely a result of the electricity used for operating the irrigation system, as energy generation in Saskatchewan is largely fossil fuel based. Considering soils and field operations only, the GHGI of the irrigated system was lower than its dryland counterpart, suggesting that irrigated cropping has potential to produce agri-food outputs at a lower GHGI on a smaller area of land, if more environmentally responsible modes of electricity generation are adopted.

Although this single-year, single site comparison is useful in gaining insight into the major sources and sinks of GHG in Saskatchewan cropping systems, it is important to acknowledge the limitations associated with this comparison so that the scope is not overstated. In Saskatchewan, a variety of crop rotations are grown on different soil types using a number of different management practices (i.e., soil moisture regime, fertility regime, tillage, etc.). The interaction of these factors combined with the high year-to-year weather variability common to the Canadian Prairies can result in large differences in net GWP and GHGI between fields and growing years. Multi-year studies incorporating complete cropping rotations, differing management practices, and the variety of center-pivots irrigation systems in operation are crucial for accurate quantification of agricultural GHG emissions from these cropping systems.

5. SYNTHESIS AND CONCLUSIONS

5.1 Summary of findings and suggestions for mitigation

As Canadian agriculture becomes increasingly important as a producer of agri-food exports, irrigation will become more prevalent in Saskatchewan crop production schemes. Understanding the dynamics of GHG emissions—and major contributors to the net GWP—from irrigated cropping is the first step in developing effective GHG mitigation practices for these systems.

Chapter 4 identified that, overall, the irrigated cropping system studied had greater GHG emissions than the dryland cropping system—contributing twice the amount of emissions per hectare of land. Yet, when system productivity was considered, some emission sources demonstrated a lower GHGI under irrigated crop management; namely soil emissions and emissions from diesel combustion. The greatest contributor to net GWP and GHGI in irrigated systems is energy used for pumping, which reflects the heavy reliance on fossil fuel for electricity generation in Saskatchewan. If a less GHG intensive source of electricity was used for pumping, irrigated crop production could provide greater yields at a lower GHGI than dryland agriculture. Unfortunately, producers have little control over electricity production, leaving few options for the mitigation of GHG emissions associated with pumping irrigation water. Producers could conceivably use alternative, on-site generated sources of energy for pumping (West and Marland, 2002), however, these are also often fossil fuel-based and may realize only marginal benefits for GHG reductions. A promising alternative may be solar powered (photovoltaic) irrigation systems (Ahmed, 2013), yet the relatively new application and high cost of these systems is a large deterrent to their adoption. For producers, the most feasible method of reducing emissions associated with pumping may be to decrease electricity usage by reducing the pressure requirement for irrigation—retrofitting existing systems with low pressure applicators/sprinklers. For example, with the irrigation system in the present study, 25% reductions in energy requirements could be achieved by a conversion to a lower pressure requirement (10 m TDH from 30 m TDH), which could reduce emissions by 100 CO_2-eq ha_{-1} (based on 2013 data).

Soils are important contributors to GHG fluxes in agricultural cropping systems, thus GHG mitigation efforts most feasible for producers may be in managing soil emissions. Nitrous oxide is the chief GHG emissions from agricultural soils; representing fertilizer losses and large contributions to radiative forcing. Results of Chapter 3 suggest that soil N availability may be driving N_2O emission from irrigated systems. Indeed, others have concluded that N availability is an accurate predictor of N_2O emissions (Robertson and Grace, 2004; Trost et al., 2013). Thus, the largest gains in mitigation may be realized from fertility management practices that more closely correspond to plant requirements. A rough calculation of N use efficiency at the site suggests that around 0.5% of the N applied to the irrigated system was lost as N_2O, lower than the 0.8% lost from the dryland cropping system (Appendix C.2). Although these losses are low, they must be considered in context of the high yields experienced in 2013. Losses are likely to be more substantial under less favorable production years (wetter conditions favors runoff or leaching, slower growing/lower biomass crop would use fewer nutrients leaving more soil N available for transformations, etc.). Echoing the recommendations of others: eliminating or reducing fall-applied fertilizer can help reduce N_2O emissions throughout the Canadian Prairies, by reducing the amount of N available during the peak emission period at spring thaw (Hao et al., 2001). It is widely accepted that fall fertilizer applications, when compared to spring applications, are subject to more opportunities for losses; yet many producers practice fall nitrogen application, accepting fertilizer losses as the cost of saving time during the busy spring seeding period. Although these N losses may be inconsequential to a producer, losses as N_2O have notable effects in terms of GWP. Fertigation—the application of fertilizer with irrigation water—may be a feasible alternative; saving time and operations in the spring, and allowing producers to match application with crop demand, ultimately minimizing the amount of soil N available for microbial transformations. Recent work on irrigated potato cropping in Manitoba found that N fertilizer timing that more closely matched crop requirements (lower pre-planting N rate, greater N rate at hilling) resulted in lower overall N_2O emissions [Parsonage (2014) in Baron and Tenuta (2014)]. Potato yields under fertigation treatment produced greater yields than split application treatments, with only slightly higher overall N_2O emissions. In conjunction with new variable-rate irrigation technology, fertigation can be used for precision application of fertilizers, further reducing the potential for N_2O-N losses. Currently, fertigation is a rare practice in Saskatchewan and requires further investigation to confirm its potential for reducing GHG emissions in this region.

The high productivity of irrigated cropping and resulting potential for SOM gains has been touted as a source of GHG reduction. In a recent review of irrigated agriculture, Trost et al. (2013) report increases in SOM of between 11 to 35% under irrigated production in semi-arid regions. However, an increase in soil C storage was not observed in this study, likely due to the high C turnover through decomposition, promoted by soil disturbance and elevated moisture regime (McGill and Cole, 1981; Bardak-Meyers, 1996; Gillabel et al., 2007). Mitigation of GHG emission through SOM storage in these systems is an optimistic—and arguably unrealistic—expectation. Although the adoption of no-till or reduced-tillage practices in these highly productive systems can conceivably increase SOM levels (Follett, 2001; Entry et al., 2002; Alluvione et al., 2009), many producers depend on high-value, tillage-intensive crops like potato and dry bean in their crop rotation. Thus, the low disturbance requirement for realizing SOM increases in irrigated systems is not practical for the majority of crop producers in Saskatchewan.

5.2 Conclusions

The results of this two-year investigation highlight that irrigated cropping in Saskatchewan—although currently having a greater GHGI intensity—has potential to produce lower GHG emission per unit crop yield on a smaller area of land than dryland cropping if less emission intensive forms of electricity production are realized. Emissions from soils of irrigated cropping system (namely N_2O emissions) appear to be already well managed, however, improvements in fertility management practices (i.e., reducing/eliminating fall-applied N; adoption of fertigation) may lower soil N_2O emissions and improve N use efficiency—especially during poor growing seasons where the potential for losses is more substantial. Managing soil N losses and reducing electricity usage appear to be a producer's best option for mitigating on-site emissions from irrigated production, as the inherent potential for GHG reductions via atmospheric CH_4 uptake and SOM increases are low in these systems.

As the irrigated land base in Saskatchewan expands, total agricultural GHG emissions can be expected to increase. Due to the presently greater GHGI of irrigated systems, this increase in total GHG emissions will occur at a greater rate than the increase in agri-food outputs. However, considering that the demand for agri-food exports will continue to rise, intensification via irrigation may be preferable to extensification. To produce comparable outputs under dryland production would require 1.5 times the area of land when compared to irrigated (based on 2013 yields). This would necessitate the conversion of additional land, either pasture or forested lands, to crop production which has an associated "cost" in terms of greenhouse gas emissions. Land use changes to agricultural cropping results in losses of soil C (as CO_2), increases in soil N_2O emissions (due to increased N inputs), and a reduction in CH_4 oxidation capacity (Paustian et al., 2000, 2006; Smith et al., 2008; Environment Canada, 2010b; Flynn and Smith, 2010). Intensification through irrigated crop production is clearly the favorable option, and will become increasingly so as better fertility and irrigation management practices are developed through additional study of these systems and their role on GHG dynamics.

5.3 Future research

This body of research provides a first look into the greenhouse gas dynamics of irrigated cropping systems in Saskatchewan. The Canadian Prairies are well known for high year-to-year variability in climate, and unexpected weather patterns can negatively affect the GHGI of irrigated systems through ineffective crop inputs, greater soil emissions, and reduced productivity. Although the studies presented here highlight typical greenhouse gas emission trends over two seasons, long-term, multiple year investigations are crucial for accurate comparisons of these systems.

Multiple year studies are needed for examining crop rotation effects on greenhouse gas dynamics, as rotation effects are evident in this study as well as others (Halvorson et al., 2008; Alluvione et al., 2009; Sainju et al., 2012; Farrell and David, 2014). Typical irrigated cropping rotations involve a different set of crops than their dryland counter parts (i.e., dry bean, potato, etc.); thus, an accurate system-to-system comparison should encompass a complete crop rotation.

Understanding the factors driving soil emission is important for GHG modelling, developing accurate GHG inventory estimates, and developing effective management practices that mitigate GHG emissions. The results of Chapter 3 suggest that soil N availability is the primary factor influencing soil N_2O emissions—especially in conjunction with high moisture soil conditions. Managing soil N fertility to more closely match crop uptake will help reduce soil N transformation and losses. A more detailed investigation assessing soil N levels and moisture in relation to soil N_2O emissions throughout the cropping season would be valuable in predicting soil N_2O emissions and developing N fertilizer management strategies to reduce emissions. Fertigation may be a valuable tool in managing soil N fertility, but is relatively rare in Saskatchewan. Beneficial management practices for soil fertility may be realized by exploring fertigation application timing and crop nutrient demands.

This study investigated irrigated crop management practices typical to Saskatchewan; yet, throughout the province, producers manage a variety of crops under different center-pivot irrigation systems. Expanding the GHG budget investigation to include additional irrigated cropping systems would increase the scope of this work. Modern, low-pressure system are more energy efficient than older high-pressure systems—even those that have been retrofit for low-pressure applications. Applicator technology is constantly improving to maximize irrigation application efficiencies, and as a result, the variability in energy requirements between irrigated cropping systems may be high. A study including a representative set of crop types and management practices would improve emission accounting for irrigated cropping systems.

Manual, chamber-based GHG measurements are very labor intensive and time consuming. Accurately quantifying soil gas fluxes are limited to short time intervals—typically less than one hour—two to three times per week. Daily and season emissions are upscaled from these short, mid-day sampling intervals, affecting the accuracy and precision of these estimations. Recent developments in instrumentation—namely micrometorological techniques and automated chamber sampling—allow for semi-continuous measurements of GHG fluxes from these systems. Although the high cost of these systems is somewhat limiting, the improved accuracy and sensitivity of daily and seasonal measurements will aid in future field-scale accounting of soil GHG fluxes.

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APPENDICES

APPENDIX A. DETAILED SOIL SURVEY OF THE STUDY AREA

A detailed soil survey was conducted at the study area in the fall of 2011 to determine comparable sites within each field for establishing gas sampling transects. Soil identification and preparation of the accompanying map was conducted by Marc St. Arnaud.

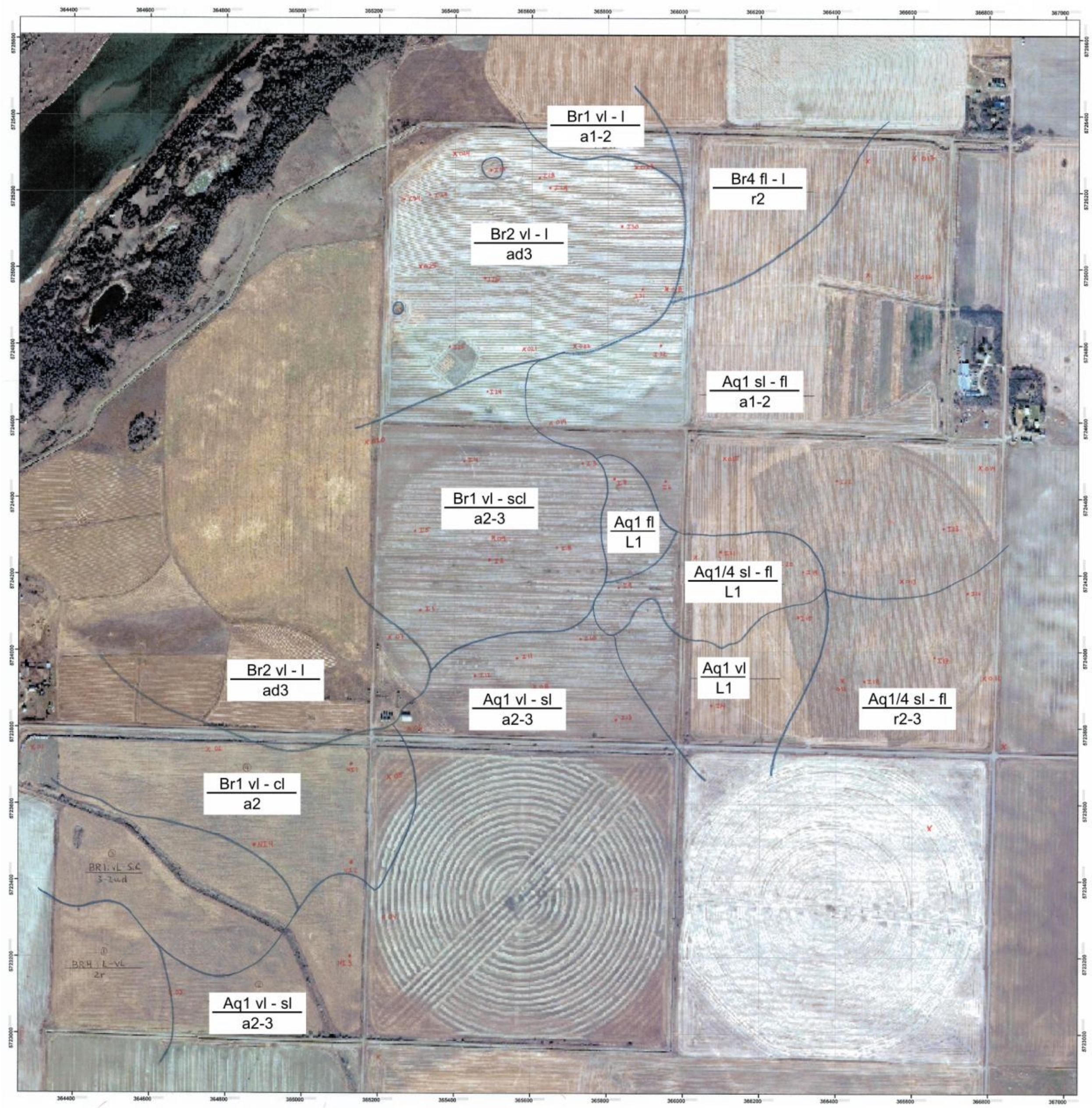

Figure A.1 An aerial view of the study site overlain by soil association designations, as determined by Marc St. Arnaud from visual inspection of soil pits. Symbols and abbreviations represented are the standard map units used by the Canadian Soil Survey. Photo credit: FlySask (2011).

APPENDIX B. SOIL CHARACTERISTICS OF THE FIELD SITES

Fall soil sampling was conducted to determine soil characteristics of the field site. In the fall of 2011, six samples were collected to a depth of 100 cm using a dutch auger in the area of each field where the gas sampling transects were to be established in the spring of 2012. Due to logistics, sampling occurred after anhydrous ammonia was applied to the dryland and 2012 irrigated (IR12) study sites. Samples were analyzed for pH; EC; total C; total N; and extractable nitrate, ammonia, and phosphate.

Following harvest in the 2012 and 2013 cropping season, samples were collected using a hydraulic-powered soil coring machine (Giddings Machine Company, USA). Soil was collected to a depth of 120 cm at points adjacent to each gas sampling chamber using a 45 mm diameter coring tube. At some sampling points, soil could only be collected as deep as 90 cm due to an underlying layer of course gravel. Samples from four adjacent points were collated into bulk samples representing five, 25-m blocks along the transect. Bulked samples were subsampled for laboratory determination of soil texture; pH; EC; organic C; total C; total N; and extractable nitrate, ammonia, and phosphate.

Samples for bulk density determination were collected in the fall of 2013, using aluminum cylinders measuring 7.5-cm tall and 7.5-cm in diameter. Cylinders were tapped into the soil using a block of wood and a mallet. Samples were collected within and below the tillage layer, approximately 0-15 cm and >15 cm, respectively.

B.1 Soil texture

Table B.1 Soil particle size and texture at the dryland site (DL) as determined by modified pipette method (Indorante, 1990). Values presented are the mean and standard deviation of the number of samples indicated.

Depth	Sand	Silt	Clay	n	Texture
— *cm* —		%			
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

Table B.2 Soil particle size and texture at the 2012 irrigated site (IR12) as determined by modified pipette method (Indorante, 1990). Values presented are the mean and standard deviation of the number of samples indicated.

Depth	Sand	Silt	Clay	n	Texture
— *cm* —		%			
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

Table B.3 Soil particle size and texture at the 2013 irrigated site (IR13) as determined by modified pipette method (Indorante, 1990). Values presented are the mean and standard deviation of the number of samples indicated.

Depth	Sand	Silt	Clay	n	Texture
— *cm* —		%			
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

B.2 Bulk density

Table B.4 Soil bulk density at the dryland (DL), 2012 irrigated (IR12), and 2013 irrigated (IR13) sites within and below the tillage layer. Values presented are the mean and standard deviation (n = 5).

Depth		DL	IR12	IR13
	— *cm* —		*Mg m^{-3}*	
Within tillage layer	0 -15	1.17 ± 0.13	1.17 ± 0.11	1.18 ± 0.11
Below tillage layer	>15	1.34 ± 0.06	1.46 ± 0.06	1.43 ± 0.04

B.3 Extractable nitrate, ammonium and phosphate

Table B.5 Fall 2011 soil sampling results for extractable nitrate (NO_3^-), ammonium (NH_4^+) and phosphate (PO_4) at the dryland (DL), 2012 irrigated (IR12), and 2013 irrigated (IR13) sites. Concentrations were determined by liquid extract analysis using a Segmented Flow Autoanalyzer (Technicon, Denmark). Values presented are mean and standard deviation (n = 6). Nitrate and ammonia extracts were prepared using a 2 M KCl solution. Extractions for phosphate analysis were prepared using a Kelowna solution (Bates and Richards, 1993). Note: Samples were collected at DL and IR12 after anhydrous ammonia application.

Depth	DL[†]	IR12[‡]	IR13[§]
		Nitrate	
cm		*kg NO_3-N ha^{-1}*	
0-25	8.5 ± 2.6	16.4 ± 4.0	2.6 ± 0.6
25-50	4.1 ± 1.6	4.1 ± 0.9	3.6 ± 3.1
50-75	6.1 ± 8.3	3.5 ± 1.6	3.6 ± 2.5
75-100	2.6 ± 1.5	10.0 ± 6.0	1.8 ± 1.3
		Ammonium	
cm		*kg NH_4-N ha^{-1}*	
0-25	116.2 ± 34.0	129.8 ± 68.5	23.1 ± 4.5
25-50	78.7 ± 23.1	48.5 ± 18.4	20.8 ± 11.4
50-75	88.7 ± 21.4	39.0 ± 25.0	17.8 ± 9.0
75-100	70.4 ± 16.3	41.1 ± 26.6	16.6 ± 11.3
		Phosphate	
cm		*kg PO_4-P ha^{-1}*	
0-25	9.4 ± 2.6	16.7 ± 5.6	9.2 ± 4.5
25-50	6.6 ± 6.6	6.0 ± 1.9	4.8 ± 3.4
50-75	5.3 ± 5.3	5.0 ± 1.3	2.3 ± 1.8
75-100	4.8 ± 4.8	6.9 ± 5.2	1.6 ± 1.8

[†] Anhydrous ammonia applied prior to sample collection at a rate of 73 kg N ha^{-1}.
[‡] Anhydrous ammonia applied prior to sample collection at a rate of 100 kg N ha^{-1}.
[§] Sampling occurred before anhydrous ammonia application.

Table B.6 Fall 2012 soil sampling results for extractable nitrate (NO_3^-), ammonium (NH_4^+) and phosphate (PO_4) at the dryland (DL), 2012 irrigated (IR12), and 2013 irrigated (IR13) sites. All samples were collected prior to fall anhydrous ammonia application. Concentrations were determined by liquid extract analysis using a Segmented Flow Autoanalyzer (Technicon, Denmark). Values presented arethe mean and standard deviation (n = 5). Nitrate and ammonia extracts were prepared using a 2 M KCl solution. Extractions for phosphate analysis were prepared using a Kelowna solution (Bates and Richards, 1993).

Depth	DL	IR12	IR13
		Nitrate	
cm		*kg NO_3-N ha^{-1}*	
0-15	1.5 ± 0.2	5.6 ± 1.4	4.6 ± 1.2
15-30	0.2 ± 0.1	1.2 ± 0.6	0.3 ± 0.1
30-60	0.6 ± 0.3	7.2 ± 2.9	1.4 ± 0.7
60-90	0.8 ± 0.1	6.3 ± 2.1	2.0 ± 1.7
90-120	0.8 ± 0.7	2.6 ± 0.2	0.7 ± 0.5
		Ammonium	
cm		*kg NH_4-N ha^{-1}*	
0-15	11.6 ± 0.5	9.8 ± 0.4	12.1 ± 0.3
15-30	10.1 ± 1.5	9.8 ± 0.3	8.8 ± 1.0
30-60	17.3 ± 1.7	14.3 ± 2.1	16.3 ± 1.7
60-90	22.1 ± 8.2	13.8 ± 3.4	16.5 ± 1.9
90-120	11.5 ± 1.5	12.3 ± 0.3	14.4 ± 2.3
		Phosphate	
cm		*kg PO_4-P ha^{-1}*	
0-15	7.6 ± 1.1	12.1 ± 2.4	11.3 ± 2.8
15-30	8.3 ± 3.8	8.8 ± 0.4	8.9 ± 1.6
30-60	21.3 ± 9.5	37.2 ± 18.9	32.7 ± 8.3
60-90	42.3 ± 11.3	65.1 ± 7.8	44.2 ± 25.8
90-120	42.5 ± 37.9	30.0 ± 10.1	18.4 ± 5.2

Table B.7 Fall 2013 soil sampling results for extractable nitrate (NO_3^-), ammonium (NH_4^+) and phosphate (PO_4) at the dryland (DL) and 2013 irrigated (IR13) sites. All samples were collected prior to fall anhydrous ammonia application. Concentrations were determined by liquid extract analysis using a Segmented Flow Autoanalyzer (Technicon, Denmark). Values presented are the mean and standard deviation (*n* = 5). Nitrate and ammonia extracts were prepared using a 2 M KCl solution. Extractions for phosphate analysis were prepared using a Kelowna solution (Bates and Richards, 1993).

Depth	DL	IR13
	Nitrate	
cm	*kg NO_3-N ha^{-1}*	
0-15	2.4 ± 1.1	1.4 ± 0.2
15-30	1.1 ± 0.7	0.5 ± 0.1
30-60	2.5 ± 1.6	1.0 ± 0.1
60-90	2.7 ± 1.1	1.1 ± 0.2
90-120	3.9 ± 1.9	1.1 ± 0.3
	Ammonium	
cm	*kg NH_4-N ha^{-1}*	
0-15	32.7 ± 13.7	10.5 ± 1.5
15-30	28.9 ± 16.6	9.9 ± 1.5
30-60	60.0 ± 36.8	17.6 ± 2.2
60-90	60.9 ± 20.5	18.7 ± 3.1
90-120	68.8 ± 26.3	19.7 ± 7.2
	Phosphate	
cm	*kg PO_4-P ha^{-1}*	
0-15	2.8 ± 0.8	4.1 ± 1.0
15-30	0.5 ± 0.1	1.1 ± 0.3
30-60	3.0 ± 1.2	2.3 ± 1.1
60-90	3.7 ± 1.2	1.5 ± 1.1
90-120	1.9 ± 1.2	0.8 ± 1.4

B.4 Total carbon and nitrogen

Table B.8 Percent total soil C at depth for the dryland (DL), 2012 irrigated (IR12), and 2013 irrigated (IR13) sites determined by combustion analysis (TruMac CNS; LECO Corporation, USA). Values presented are the mean and standard deviation (n = 6 for 2011, n = 5 for 2012 and 2013).

Depth	DL	IR12	IR13
		2011	
—— *cm* ——		%	
0-25	2.00 ± 0.28	1.58 ± 0.78	1.47 ± 0.38
25-50	1.14 ± 0.15	1.25 ± 0.37	1.48 ± 0.77
50-75	2.04 ± 0.55	1.37 ± 0.39	1.62 ± 0.74
75-100	2.17 ± 1.10	1.92 ± 0.67	1.18 ± 0.49
		2012	
—— *cm* ——		%	
0-15	1.93 ± 0.38	1.71 ± 0.15	1.67 ± 0.31
15-30	1.27 ± 0.17	1.09 ± 0.10	1.18 ± 0.36
30-60	1.80 ± 0.12	1.98 ± 0.68	1.51 ± 0.33
60-90	1.40 ± 0.48	1.97 ± 0.40	1.14 ± 0.39
90-120	1.70 ± 0.96	1.40 ± 0.64	0.77 ± 0.22
		2013	
—— *cm* ——		%	
0-15	2.22 ± 0.17	--	1.74 ± 0.11
15-30	1.23 ± 0.28	--	1.25 ± 0.28
30-60	1.82 ± 0.38	--	1.64 ± 0.48
60-90	2.41 ± 0.55	--	1.14 ± 0.42
90-120	1.86 ± 0.58	--	0.70 ± 0.26

Table B.9 Percent total soil N at depth for the dryland (DL), 2012 irrigated (IR12), and 2013 irrigated (IR13) sites determined by combustion analysis (TruMac CNS; LECO Corporation, USA). Values presented are the mean and standard deviation (*n* = 6 for 2011, *n* = 5 for 2012 and 2013).

Depth	DL	IR12	IR13
		2011	
cm		%	
0-25	0.24 ± 0.04	0.17 ± 0.10	0.12 ± 0.02
25-50	0.13 ± 0.02	0.12 ± 0.01	0.08 ± 0.03
50-75	0.11 ± 0.07	0.06 ± 0.02	0.04 ± 0.02
75-100	0.03 ± 0.02	0.04 ± 0.03	0.01 ± 0.01
		2012	
cm		%	
0-15	0.20 ± 0.03	0.18 ± 0.01	0.17 ± 0.03
15-30	0.13 ± 0.02	0.11 ± 0.01	0.11 ± 0.01
30-60	0.09 ± 0.02	0.09 ± 0.01	0.08 ± 0.01
60-90	0.03 ± 0.01	0.04 ± 0.01	0.03 ± 0.01
90-120	0.09 ± 0.11	0.03 ± 0.01	0.02 ± 0.01
		2013	
cm		%	
0-15	0.25 ± 0.03	--	0.18 ± 0.02
15-30	0.14 ± 0.03	--	0.09 ± 0.01
30-60	0.09 ± 0.03	--	0.04 ± 0.03
60-90	0.03 ± 0.03	--	0.02 ± 0.03
90-120	0.03 ± 0.02	--	0.01 ± 0.01

B.5 Organic carbon

Table B.10 Percent soil organic C in the top 30 cm at the dryland (DL), 2012 irrigated (IR12), and 2013 irrigated (IR13) sites determined by combustion analysis (LECO C-632; LECO Corporation, USA). Samples were pretreated via HCl fumigation to remove carbonates. Values presented are the mean and standard deviation (n = 5). Same letters following values indicated no significant difference among fields within each depth (P > 0.05), determined by an analysis of variance.

Depth	**DL**	**IR12**	**IR13**
cm	%		
0-15	1.78 ± 0.44 *a*	1.66 ± 0.14 *a*	1.76 ± 0.25 *a*
15-30	1.18 ± 0.17 *a*	0.91 ± 0.15 *b*	1.02 ± 0.20 *ab*

B.6 Electrical conductivity and pH

Table B.11 Soil electrical conductivity determined on a 2:1 water:soil solution using a portable conductivity meter (Accumet AP85; Fischer Scientific Inc, Canada). Values presented are the mean and standard deviation (*n* = 6 for 2011, *n* = 5 for 2012 and 2013).

Depth	DL	IR12	IR13
		2011	
cm		*μS cm^{-1}*	
0-25	529 ± 56	533 ± 208	409 ± 88
25-50	400 ± 63	459 ± 116	423 ± 73
50-75	501 ± 145	668 ± 460	381 ± 114
75-100	472 ± 190	581 ± 156	381 ± 128
		2012	
cm		*μS cm^{-1}*	
0-15	270 ± 85	287 ± 160	664 ± 210
15-30	199 ± 15	199 ± 52	236 ± 35
30-60	273 ± 43	303 ± 93	284 ± 48
60-90	277 ± 36	466 ± 319	268 ± 64
90-120	268 ± 97	964 ± 629	224 ± 40
		2013	
cm		*μS cm^{-1}*	
0-15	370 ± 69	--	333 ± 60
15-30	319 ± 80	--	277 ± 28
30-60	366 ± 56	--	265 ± 33
60-90	412 ± 107	--	259 ± 45
90-120	405 ± 81	--	202 ± 24

Table B.12 Soil pH determined on a 2:1 water:soil solution using a portable pH meter (Accumet AP85; Fischer Scientific Inc, Canada). Values presented are the mean and standard deviation (n = 6 for 2011, n = 5 for 2012 and 2013).

Depth	DL			IR12			IR13		
				2011					
cm				*pH*					
0-25	8.26	±	0.29	7.65	±	0.59	8.32	±	0.20
25-50	8.76	±	0.25	8.60	±	0.13	8.56	±	0.23
50-75	8.92	±	0.46	8.86	±	0.32	8.89	±	0.29
75-100	9.11	±	0.32	8.93	±	0.24	9.10	±	0.25
				2012					
cm				*pH*					
0-15	7.05	±	0.40	7.59	±	1.05	7.22	±	0.37
15-30	8.19	±	0.04	7.41	±	0.41	7.50	±	0.47
30-60	8.27	±	0.91	8.26	±	0.34	8.41	±	0.30
60-90	9.26	±	0.08	9.04	±	0.3	9.12	±	0.21
90-120	9.55	±	0.17	9.40	±	0.46	9.51	±	0.37
				2013					
cm				*pH*					
0-15	8.07	±	0.48	--			8.32	±	0.49
15-30	8.80	±	0.22	--			8.74	±	0.44
30-60	9.27	±	0.20	--			9.21	±	0.26
60-90	9.40	±	0.07	--			9.30	±	0.33
90-120	9.53	±	0.09	--			9.34	±	0.23

APPENDIX C. CROP SAMPLING AND ANALYSIS

During the 2012 and 2013 study years, crop samples were collected prior to swathing to determine seed and biomass yield, C content, and N content. Crop samples were collected from one square meter areas adjacent to each chamber, for a total of 20 samples per field. Cloth bags were used to transport samples from the field to the processing facility at the university, where they remained in cloth bags to be air dried. Seed and biomass were separated with portable thresher.

C.1 Crop yield and biomass carbon and nitrogen

Table C.1 Yield and percent C and N from wheat (*Triticum aestivum* L.) samples collected at the dryland (DL) and irrigated (IR12) sites in 2012. Carbon and N content were determined by combustion analysis (TruMac CNS; LECO Corporation, USA). Seed and biomass samples were ground prior to analysis. Values presented are mean and standard deviation (n = 20).

	DL			IR12		
Plant tissue			**Yield**			
			kg ha^{-1}			
Seed	1060	±	260	2220	±	320
Biomass	2830	±	490	3530	±	510
			Carbon			
			%			
Seed	43.2	±	0.6	43.2	±	0.1
Biomass	43.6	±	0.3	42.3	±	0.3
			Nitrogen			
			%			
Seed	2.9	±	0.1	3.2	±	0.1
Biomass	0.6	±	0.1	1.1	±	0.1

Table C.2 Yield and percent C and N in Canola (*Brassica napus*) samples collected at the dryland (DL) and irrigated (IR13) sites in 2013. Carbon and N content were determined by combustion analysis (TruMac CNS; LECO Corporation, USA). Biomass samples were ground prior to analysis. Seed samples were analyzed whole. Values presented are mean and standard deviation (n = 20).

	DL			IR13		
Plant tissue			**Yield**			
			kg ha^{-1}			
Seed	2020	±	490	3130	±	760
Biomass	5960	±	780	6950	±	1210
			Carbon			
			%			
Seed	53.5	±	6.5	57.7	±	2.7
Biomass	41.2	±	0.9	42.9	±	1.0
			Nitrogen			
			%			
Seed	3.3	±	0.4	3.8	±	0.3
Biomass	0.4	±	0.2	0.5	±	0.2

C.2 Crop N use efficiency

Table C.3 Nitrogen partitioning within the dryland (DL) and irrigated (IR13) cropping systems in 2013, represented as a percent of N applied. Fertilizer N was applied at 90 and 146 kg ha^{-1} for DL and IR13, respectively. These values represent an estimate, as the sum of total percentages exceed 100% due to unaccounted inputs (i.e., residue returns, atmospheric N deposition) and losses (leaching, volatilization, complete denitrification).

	DL	IR13	DL	IR13
	Measured		**Normalized**†	
	% N applied			
Canola seed	74.1 ± 0.9	81.5 ± 1.4	73.0 ± 0.9	76.9 ± 1.3
Canola biomass	26.5 ± 1.5	23.8 ± 1.2	26.1 ± 1.4	22.5 ± 1.2
N_2O emissions	1.0‡	0.7‡†	1.0†	0.6†
Total	101.5 ± 2.3	105.9 ± 2.6	100.0 ± 2.3	100.0 ± 2.4

† Normalized to total fertilizer-N applied.
‡ Calculated from median cumulative annual N_2O emissions, therefore no standard deviation is presented.

APPENDIX D. CLIMATE NORMALS

Table D.1 Total monthly rainfall for the growing season at the study site and 20-year mean monthly rainfall at the Outlook weather station. Incomplete site-specific data has been supplemented with measurements from the Outlook weather station. Standard deviations were not reported for the historic values.

Month	1981–2010	2012†	2013‡
	mm		
May	39.0	100§	31
June	63.9	110	85
July	56.1	56	40
August	42.8	43	29
September	32.8	5	43
October	12.6	27	0

† Measured using tipping bucket rain gauge (TR-525; Texas Electronics Inc, USA)
‡ Measured using weighing precipitation gauge (Belfort 3000; Belfort Instrument, Baltimore MD)
§ Value obtained from Outlook weather station

Table D.2 Mean monthly air temperature (± sd) for the growing season at the study sites and 20-year mean monthly air temperatures at the Outlook weather station. Incomplete site-specific data has been supplemented with measurements from the Outlook weather station.

		DL	IR12	DL	IR13
Month	1981–2010	2012		2013	
	°C				
April	5.3 ± 1.9	4.5†	4.5†	-1.1	
May	11.5 ± 1.6	10.8†	10.8†	13.6	15.2
June	16.1 ± 1.5	17.0	16.0	16.1	16.0
July	18.9 ± 1.4	20.3	20.1	17.3	18.2
August	18.0 ± 2.0	18.0	18.0	18.5	18.2
September	12.3 ± 1.7	13.2	13.2	15.2	15.3
October	5.1 ± 1.7	2.2	2.3	4.5†	4.5†

† Value obtained from Outlook weather station

APPENDIX E. DETERMINATION OF NORMAL IRRIGATION REQUIREMENT

Irrigation requirement was taken as the moisture deficit remaining after seasonal precipitation and soil moisture reserves were subtracted from crop water use [seasonal crop evapotranspiration (ET)]. Seasonal crop ET was estimated for the 2004 through 2013 growing seasons (473mm at 95% probability of non-exceedance) using a canola specific crop coefficient curve—constructed in the linear FAO style outlined in Allen et al. (2007) using Canola production details from the Canola Council of Canada (2014)—and seasonal reference ET [calculated from AIMM climate data using the ASCE-EWRI standardized Penman-Monteith method (Allen et al., 2007)]. Growing season (May through September) precipitation was obtained from the AIMM climate data record (μ = 275 mm; 95% CI = 206 mm, 345 mm; assumed 90% effective) and soil moisture reserves (105 mm) were estimated from the cooperators management practices (seasonal moisture at seeding = 40% VWC, soil moisture at swathing = 25% VWC, and a managed soil moisture depth of 0.7 m).