

The Effect of Control Tile Drainage on Soil Greenhouse Gas Emissions from Agricultural Fields in the South Nation Watershed of Ontario

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A thesis submitted to the Faculty of Graduate and Postdoctoral Studies in partial fulfillment of the requirements for the PhD degree in Earth Science with Specialization in Chemical and Environmental Toxicology

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Table of Acronyms

<u>Acronym</u>	<u>Meaning</u>
BMP	Best management practice
BT	Between tile
CRC	Crop residue carbon
CTD	Control tile drainage
GHG	Greenhouse gas
OT	Over tile
R_{rh}	Rhizosphere respiration
R_s	Soil respiration
R_t	Total respiration
SOC	Soil organic carbon
ST10	Soil temperature (0-10 cm depth)
SWC20	Soil water content (0-20 cm depth)
UTD	Uncontrolled tile drainage
WTDBS	Water table depth below surface

Preface

Before you lies the thesis, “The effect of control tile drainage management on soil greenhouse gas emissions from agricultural fields in the South Nation Watershed of Ontario”. It has been written to fulfill the requirements for the PhD in Earth Science with Specialization in Chemical and Environmental Toxicology program at the University of Ottawa. I have been engaged in the requirements for this program from September 2011 to December 2015.

The basis of this thesis is an intensive two-growing season field study, which involved sampling soils, plants, and soil greenhouse gas emissions in controlled tile drained and uncontrolled tile drained agricultural fields. Environmental variables including soil temperature, air temperature, soil water content, solar radiation, precipitation, and water table depth below surface were monitored throughout the two growing seasons. The project was undertaken in the South Nation Watershed of Ontario and was in collaboration with the South Nation Conservation Authority, Agriculture and Agri-Food Canada, and the University of Ottawa. My research questions were formulated together with my two supervisors, Dr. Ian Clark and Dr. David Lapen. The fieldwork sampling was intensive with two full growing seasons, but because of the extensive amount of samples collected and analyzed, it has allowed me to answer the proposed research questions and to fill the existing knowledge gap in the literature.

This project involved the guidance and support of many individuals and it would not have been possible without them. I would like to thank Corey Flemming for all his hard work in the field with helping collect the samples. I'd like to thank my two supervisors, Dr. Ian Clark and Dr. David Lapen, for their excellent guidance and support

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I hope you enjoy reading this thesis.

Alisha Van Zandvoort

December 2015

Abstract

Controlled tile drainage (CTD) is an agricultural management practice with well-documented water quality and agronomic benefits, however, by virtue of its effect upon soil hydrology, CTD could potentially impact soil greenhouse gas (GHG: CO₂, CH₄, N₂O) emissions. This study aimed to determine whether: (1) CTD affects soil GHG emissions throughout a dry (2012) and a wet (2013) growing season for corn, soybean, and forage fields in eastern Ontario, and (2) the location in a field with respect to a tile drain (over tile (OT) versus between tile (BT)) is important in GHG emissions. Non-steady state chambers were used for sampling soil GHG emissions in order to analyze GHG fluxes, the $\delta^{13}\text{C}$ of soil-respired CO₂ (R_T), and for separating total soil respiration into its rhizosphere and soil components. There was no significant difference in average GHG emissions from CTD and UTD fields (except for 1/5 field pairs studied for N₂O) and from OT and BT locations. The means of $\delta^{13}\text{C}$ of R_T were not statistically different ($p > 0.05$) between 4/5 CTD and UTD field pairs, and between OT and BT locations in 4/5 CTD fields. The mean contributions from rhizosphere respiration and soil respiration did not differ ($p > 0.05$) in 3/4 CTD and UTD field pairs. This lack of difference in GHG emissions is believed to have resulted from their being no difference in surface soil water contents among CTD and UTD fields and among OT and BT locations. It is believed that surface soil moisture did not vary because: (1) the water table was too low in 2012 for effective water table control, and (2) significant precipitation created equally wet surface soil in 2013. In 2013, the surface soil moisture was approximately 10% greater and this may be why there was an approximate 5 kg C/ha/day greater CO₂ flux from soybean fields in 2013 than in 2012. $\delta^{13}\text{C}$ was useful for distinguishing the source of CO₂ emissions (rhizosphere versus soil respiration) in CTD fields when the crop and plant $\delta^{13}\text{C}$

signatures varied. The results are useful for helping to capture the carbon footprint of tile drainage management practices imposed at field-scale.

Chapter 1 Literature Review

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1. Greenhouse gasses from agriculture

The increase of greenhouse gas (GHG: carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O)) is caused by anthropogenic emissions from using fossil fuels and from land use and land use changes, especially agriculture (IPCC 2013).

Agricultural lands cover an estimated 40 to 50 % of the Earth's land surface (Smith *et al* 2007), and they release significant amounts of GHG to the atmosphere (Cole *et al* 1997). In 2005, agriculture was responsible for 10 to 12 % of the total global anthropogenic GHG emissions and specifically responsible for approximately 60 % and 50 %, respectively, of the total global anthropogenic emissions of N₂O and CH₄ (Smith *et al* 2007).

Agricultural soils have an important contribution in regulating atmospheric GHG concentrations because they are a source and sink of CO₂ and N₂O (Ball *et al* 1999, Nangia *et al* 2013, Hernandez-Ramirez *et al* 2009), and either a sink or a source for CH₄ (Nangia *et al* 2013, Hernandez-Ramirez *et al* 2009). Terrestrial vegetation and soils represent the second largest reservoir of carbon (C), with inputs from photosynthesis and losses through respiration (Clark 2015, Drewitt *et al* 2009, Hardy 2003). In vegetated areas, soil respiration (R_T) originates from plants through rhizosphere respiration, which is root and rhizomicrobial respiration using root derived C, and from soil through microbial respiration using soil organic matter (SOM) (Cheng 1996, Ryan & Law 2005, Rochette *et al* 1999). N₂O soil emissions are mainly produced by microbial nitrification (the oxidation of ammonium (NH₄⁺) to nitrate (NO₃⁻)) in aerobic conditions and denitrification (the reduction of nitrate to dinitrogen gas (N₂)) in anaerobic conditions (Robertson & Grace 2004, Dobbie & Smith 2006). Nitrification and denitrification can

both occur in well-drained soils because wet soils can be found inside soil aggregates (Robertson & Grace 2004). Soils can be sources or sinks for CH₄ depending on the activity of methanogens and methanotrophs (Topp & Pattey 1997). Under anaerobic conditions and in the absence of nitrate, sulfate, or ferric iron, methanogens reduce more oxidized forms of C into CH₄ thus making the soil a CH₄ source (Topp & Pattey 1997). Under aerobic conditions, soils are a CH₄ sink because methanotrophs oxidize CH₄ to CO₂ or incorporate it into the microbial biomass (Powlson *et al* 1997). In some soils, both CH₄ production and CH₄ consumption can occur (Powlson *et al* 1997).

GHG emission rates may continue to increase in the future from greater fertilizer use and increased livestock to satisfy the food demand from a growing population (Smith *et al* 2007). Agriculture is necessary for food production but while it is a considerable source of GHG emissions there are possible methods for its mitigation (Burney *et al* 2010). Thus, it is important to identify the factors affecting the production of GHG emissions in agricultural soils because this may allow for the reduction of emissions and hence contribute to climate change mitigation.

2. Soil redox reactions

The microbial generation of CO₂, CH₄ and N₂O is controlled by the reduction and oxidation (redox) of organic biomass in soils. Soil redox reactions and conditions are controlled by water saturation, oxygen availability and other associated environmental conditions. Oxidation-reduction reactions are coupled and involve the transfer of electrons from one compound to another (DeLaune & Reddy 2005). Oxidation is the removal of electrons from a compound (the electron donor), whereas reduction is the addition of electrons to a compound (the electron acceptor) (DeLaune & Reddy 2005).

Within soils, microorganisms break down hydrocarbons by oxidizing them for energy or to be incorporated into cell mass (Widdel & Rabus 2001). Microorganisms can degrade hydrocarbons under aerobic conditions, where oxygen (O_2) is the electron acceptor, or under anaerobic conditions, where electron acceptors include nitrate (NO_3^-), manganese (Mn^{4+}), ferric iron (Fe^{3+}), sulfate (SO_4^{2-}) and carbon dioxide (CO_2) (Widdel & Rabus 2001). Each species has an intrinsic redox potential (E_h), measured in millivolts (mV), which identifies its tendency to accept or donate electrons (DeLaune & Reddy 2005). A more positive E_h indicates a greater acceptance for electrons and thus a greater tendency to be reduced (DeLaune & Reddy 2005). The sequential order for the most common reduction reactions in soils is identified in the E_h potential scale (**Table 2.1**) (DeLaune & Reddy 2005). Each redox reaction becomes unstable at a critical E_h , for example, following flooding, microorganisms break down hydrocarbons using various electron acceptors beginning with O_2 (the strongest oxidizing agent that gets reduced first) followed by NO_3^- (denitrifying organisms) and Mn^{4+} , then Fe^{3+} (ferric-ion reducers), then SO_4^{2-} (sulfate reducers) and finally CO_2 (methanogens) is reduced to CH_4 (DeLaune & Reddy 2005, Diaz 2004). Deciding which electron acceptor to use, other than O_2 , is based on how available the electron acceptor is, how much competition exists for electrons from different types of microorganisms, and how much energy is produced (Diaz 2004). For example, a similar amount of energy is obtained from using NO_3^- and Fe^{3+} as electron acceptors for degrading aromatics compared to using O_2 , whereas much less energy is generated from sulfate reducers and methanogens (Diaz 2004).

Table 2.1 The redox potential (E_h) scale for soil indicating the electron acceptors with their corresponding E_h values. The arrow indicates the direction of reduction following flooding where oxygen is reduced first, followed by nitrate and oxidized manganese compounds, then ferric iron, then sulfate and finally CO_2 is reduced to CH_4 . The data was obtained from DeLaune & Reddy 2005, and Diaz 2004.

Redox condition	Electron acceptor	Respiration	Redox potential (mV)
oxidized	O_2	Aerobic respiration $\text{CH}_2\text{O} + \text{O}_2 \rightarrow \text{CO}_2 + \text{H}_2\text{O}$	800
moderately reduced	NO_3^-	Denitrification $\text{CH}_2\text{O} + \frac{4}{5}\text{NO}_3^- + \text{H}^+ \rightarrow \text{CO}_2 + \frac{2}{5}\text{N}_2 + \frac{7}{2}\text{H}_2\text{O}$	430
	Mn^{4+}	Manganese respiration $\text{CH}_2\text{O} + \text{MnO}_2 + 4\text{H}^+ \rightarrow \text{CO}_2 + 2\text{Mn}^{2+} + 3\text{H}_2\text{O}$	200
reduced	Fe^{3+}	Ferric ion reduction $\text{CH}_2\text{O} + 2\text{FeO}_2\text{H} + 8\text{H}^+ \rightarrow \text{CO}_2 + 4\text{Fe}^{2+} + 7\text{H}_2\text{O}$	100
	SO_4^{2-}	Sulfate reduction $\text{CH}_2\text{O} + \frac{1}{2}\text{SO}_4^{2-} + \text{H}^+ \rightarrow \text{CO}_2 + \frac{1}{2}\text{H}_2\text{S} + \text{H}_2\text{S}$	-270
highly reduced	CO_2	Methanogenesis $\text{CO}_2 + 4\text{H}^+ \rightarrow \text{CH}_4 + 2\text{H}_2\text{O}$	-400

Soil may be characterized by E_h , which is a measure of the soil's oxidation-reduction potential (DeLaune & Reddy 2005). E_h of a soil involves identifying the electron availability within the soil by determining the concentration of oxidants (including oxygen, nitrate, manganese, iron, sulfate and carbon dioxide) and reductants (including various organic substrates and reduced inorganic compounds) (DeLaune & Reddy 2005). For example, if the soil E_h is 0 mV then it is likely that oxygen and nitrate are not present, whereas iron and manganese are in a reduced state, and sulfate is stable with no production of sulfide (DeLaune & Reddy 2005). Anaerobic soils have low E_h values (0.2 to 0.4 V) indicating their reduced state, whereas aerobic soils have high E_h values (0.3 to 0.8 V) indicating their oxidized condition (Ponnamperuma 1972).

In the carbon cycle, CH₄ is the most reduced form of carbon, whereas acetate, alcohols, and methyl amines have an intermediate redox potential, and CO₂ is the most oxidized form (Topp & Pattey 1997). Since the carbon atom in CH₄ is fully reduced, it can only donate electrons, whereas the carbon atom in CO₂ is in its most oxidized form and, therefore, it can only accept electrons (Topp & Pattey 1997). CO₂ enters the carbon cycle by photosynthesis where it is fixed into glucose and then synthesis and degradation reactions occur to provide the required energy and materials needed for life (Topp & Pattey 1997). Under anaerobic conditions, soil microorganisms, including methanogens, can break down soil organic matter (SOM) into compounds with an intermediate redox potential (between CO₂ and CH₄) and can produce CH₄ as the final product (Topp & Pattey 1997). In aerobic conditions, CH₄ is oxidized to CO₂ by methanotrophs (methane-utilizing bacteria) and this completes the carbon cycle (Topp & Pattey 1997). The details of methanogens and methanotrophs and their reactions are in section 7.

3. Carbon isotope ¹³C

Tracing soil redox reactions and microbial gas generation is greatly assisted by the use of stable isotopes, and in particular that of carbon. Carbon is the fourth most abundant element in the universe (Clark 2015). Atmospheric CO₂ contains 1.1% of carbon-13 (¹³C) and 98.9% of carbon-12 (¹²C) (O'Leary 1988). The δ¹³C of atmospheric CO₂ is approximately -8‰ but this value is slowly decreasing mainly because of inputs of ¹³C depleted CO₂ from fossil fuel combustion (δ¹³C for fossil fuel ~-30 ‰) and by enhanced soil respiration (Clark 2015). CO₂ emissions from soil have a lower δ¹³C value than atmospheric CO₂ (Hesterberg & Siegenthaler 1991).

The ^{13}C content of CO_2 is analyzed on an isotope ratio mass spectrometer (IRMS). The mass spectrometer measures the ratio (R), where $R = \frac{^{13}\text{CO}_2}{^{12}\text{CO}_2}$, and for convenience R values are converted to values of $\delta^{13}\text{C}$ [reaction 3.1] (O'Leary 1988). Isotopic compositions are expressed using delta notation. The international standard for CO_2 samples is CO_2 from Pee Dee Belemnite (PDB) limestone and the units of $\delta^{13}\text{C}$ are parts per thousand (per mille, ‰) (Ehleringer & Osmond 1989, O'Leary 1988). A more negative $\delta^{13}\text{C}$ indicates that there is more ^{12}C (ie lighter in mass) whereas a more positive $\delta^{13}\text{C}$ has more ^{13}C (ie heavier in mass) (O'Leary 1988).

$$\text{[3.1]} \quad \delta^{13}\text{C} = \left(\frac{(^{13}\text{C}/^{12}\text{C})_{\text{sample}}}{(^{13}\text{C}/^{12}\text{C})_{\text{standard}}} - 1 \right) \times 1000$$

4. Carbon cycle and associated $\delta^{13}\text{C}$ values

Carbon exists in both inorganic (CO_2 and its hydrated forms: H_2CO_3 , HCO_3^- , and CO_3^{2-}) and organic forms (such as carbohydrate (CH_2O), humic substances and hydrocarbons) and cycles throughout the terrestrial and marine environments (Fig 4.1) (Clark 2015). The atmosphere contains 800 gigatons of carbon (GtC) as CO_2 ($P_{\text{CO}_2(\text{atm})} = 400 \text{ ppm}$, $\delta^{13}\text{C} = -8 \text{ ‰}$) and 4 GtC as CH_4 ($\delta^{13}\text{C} = -47 \text{ ‰}$) and other carbon gases (Clark 2015). Atmospheric CO_2 varies seasonally depending on photosynthesis and respiration (Clark 2015). Gross primary production (GPP) removes 121 GtC and in turn respire 120 GtC annually (Clark 2015). Terrestrial living biomass holds 800 GtC as CH_2O with an average $\delta^{13}\text{C} = -27 \text{ ‰}$ (Clark 2015). Soil is the largest terrestrial carbon pool on Earth and contains an estimated 1500 GtC (average $\delta^{13}\text{C} = -27 \text{ ‰}$) as labile organic carbon, which is available for bacterial respiration in soils, and therefore most of this CO_2 is returned back to the atmosphere (Scharlemann *et al* 2014, Clark 2015).

Over one quarter of atmospheric CO₂, about 210 GtC (ie 90 GtC from marine plus 120 GtC from terrestrial respiration), is exchanged annually from marine and terrestrial sources, giving atmospheric CO₂ a short residence time of only a few years in the atmosphere (Clark 2015). Marine ecosystems represent a significant reservoir of carbon (Clark 2015). Marine waters are near equilibrium with atmospheric P_{CO2} (Clark 2015). The shallow ocean contains approximately 1000 GtC, and 90 GtC is exchanged with the atmosphere and another 92 GtC is transferred from the atmosphere into the ocean annually (Clark 2015). The deep ocean contains 37,000 GtC (Clark 2015).

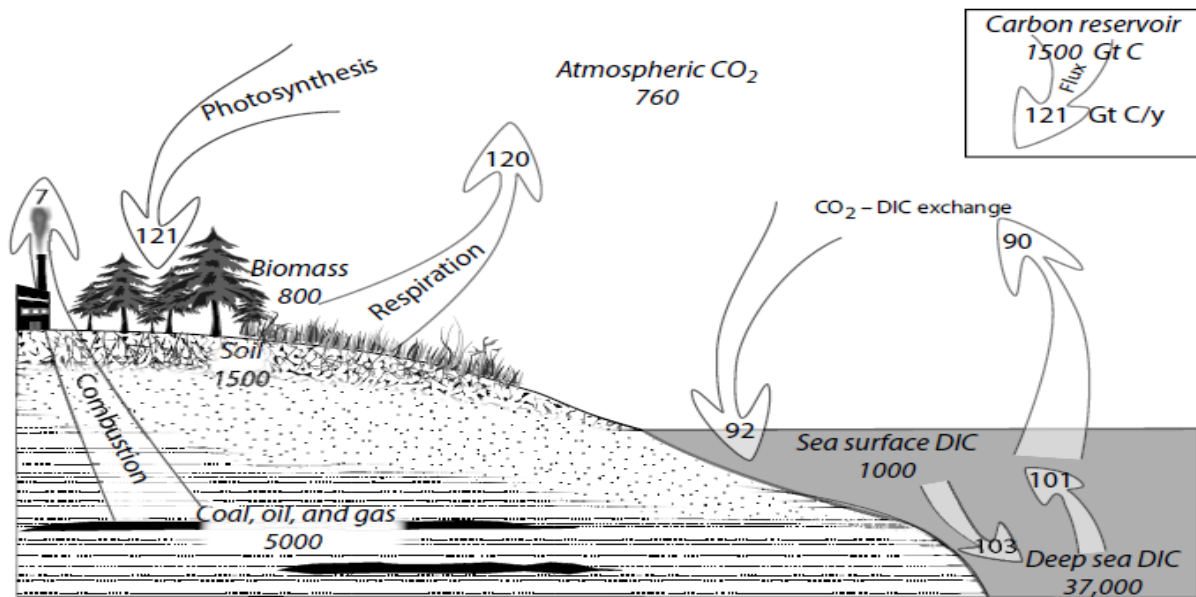


Figure 4.1 The global carbon cycle from Clark 2015 and original data source: U.S. Department of Energy; IPCC, 2007. The major carbon reservoirs are in GtC (gigatons of carbon *ie* billion tons of carbon) and fluxes are GtC/yr.

5. C₃ and C₄ photosynthesis

5.1 Mechanism

Photosynthesis is the process of carbon fixation by plants and there are three types (C₃, C₄, and CAM) among terrestrial plants (Ehleringer & Cerling 2002) of which

two (C_3 and C_4) types will be discussed as they pertain to plants used in this study. C_3 photosynthesis is the ancestral pathway for carbon fixation and is used by most plants, whereas C_4 photosynthesis is an evolutionary adaptation of C_3 photosynthesis to high light intensity, high temperature, and dryness (Ehleringer & Cerling 2002, Gowik & Westhoff 2011). The C_3 pathway is used by approximately 95% of plants and includes most terrestrial trees and shrubs, and marine plants and algae (Clark 2015). Some examples of C_3 crops are soybean, wheat, and barley, whereas C_4 plants include corn, sugar cane, and prairie and dryland grasses (Clark 2015).

C_4 plants can photosynthesize faster than C_3 plants because they use an extra biochemical pathway and special anatomy to reduce photorespiration (Wang *et al* 2012, Gowik & Westhoff 2011). C_3 photosynthesis only uses the Calvin cycle for fixing CO_2 , and this takes place inside of mesophyll cells, whereas most photosynthetic activities of C_4 plants occur in mesophyll and bundle sheath cells (Wang *et al* 2012, Gowik & Westhoff 2011). The details of the C_3 and C_4 pathways are described below and illustrated in **Fig 5.1.1**.

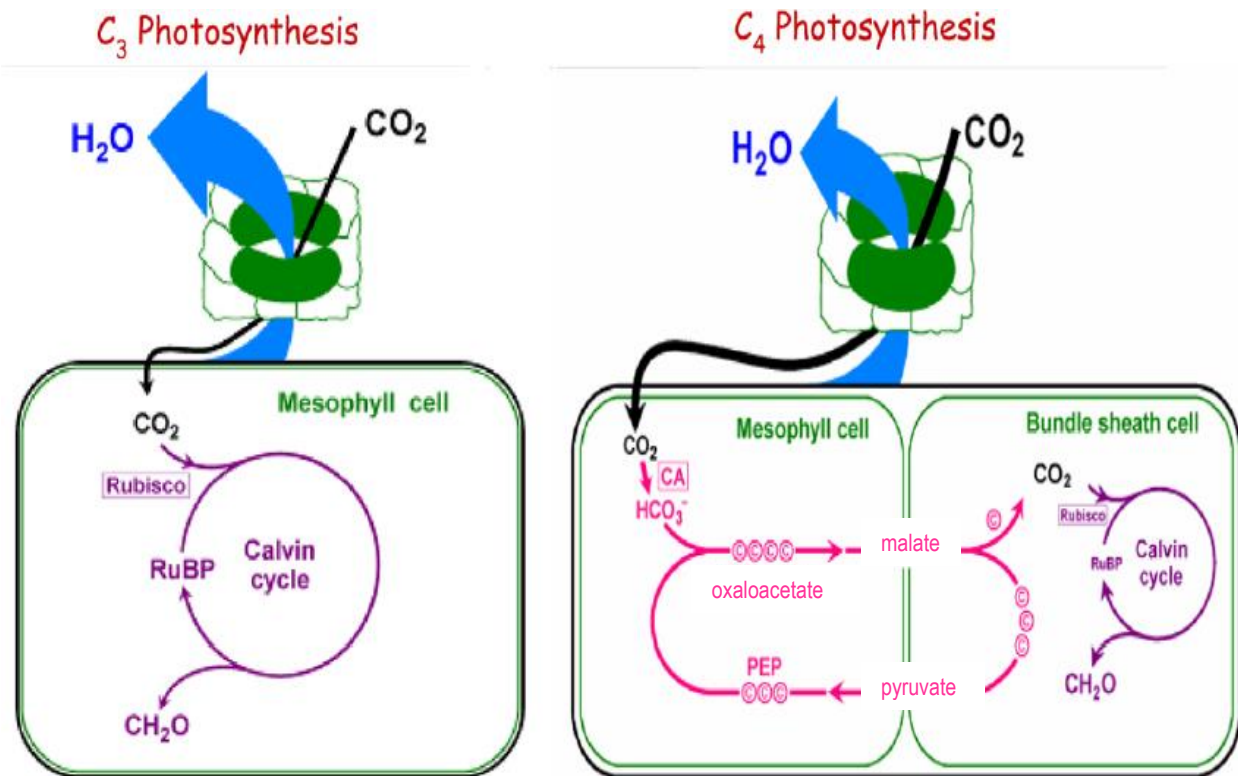


Figure 5.1.1 A diagram of C₃ and C₄ photosynthesis from Wang *et al* 2012.

Both C₃ and C₄ plants undergo the Calvin cycle for fixing CO₂ (Gowik & Westhoff 2011, Wang *et al* 2012, Ehleringer & Cerling 2002). During C₃ photosynthesis, external CO₂ is transported through the stomata in the epidermis into the internal air space (O’Leary 1988). Some CO₂ in the internal air space may diffuse back out into the atmosphere (O’Leary 1988). Internal CO₂ then diffuses to the chloroplast where carboxylation occurs in the Calvin cycle (O’Leary 1988). Ribulose-1,5-bisphosphate carboxylase/oxygenase (Rubisco) behaves as a carboxylase and combines CO₂ with ribulose-1,5-bisphosphate (RuBP) (5 C molecule) to produce two molecules of 3-phosphoglycerate (PGA) (3 C molecule) [reaction 5.1.1] (Gowik & Westhoff 2011, Wang *et al* 2012, Ehleringer & Cerling 2002). The Rubisco enzyme can also behave as an oxygenase by combining oxygen (O₂) with RuBP resulting in one molecule each of

PGA (3 C molecule) and 2-phosphoglycolate (2 C molecule) [**reaction 5.1.2**] (Gowik & Westhoff 2011, Ehleringer & Cerling 2002).

rubisco carboxylase



rubisco oxygenase



Phosphoglycolate (2 C molecule) has no metabolic purpose and is toxic for plants in high concentrations (Anderson 1971). As such, phosphoglycolate is processed in a metabolic pathway called photorespiration (Gowik & Westhoff 2011).

Photorespiration recycles the 2-phosphoglycolate back into the Calvin cycle by consuming O₂ and producing CO₂ and H₂O (Gowik & Westhoff 2011, Ehleringer & Cerling 2002). The oxygenase reaction by Rubisco is unfavourable because it is energy demanding, it leads to the production of CO₂, and it results in less net carbon fixation thus slowing the production of photosynthesis (Gowik & Westhoff 2011, Ehleringer & Cerling 2002). The concentration of [CO₂]/[O₂] in the atmosphere and the temperature (oxygenase activity increases with temperature) affects whether Rubisco will catalyze CO₂ or O₂ (Ehleringer & Cerling 2002).

C₄ photosynthesis occurs in the mesophyll cells and is followed by the C₃ Calvin cycle in the bundle sheath cells (**Fig 5.1.1**) (Wang *et al* 2012, Gowik & Westhoff 2011). The C₄ cycle concentrates CO₂ around Rubisco thereby favouring the fixing of CO₂ instead of O₂ by Rubisco, thus largely reducing the oxygenase reaction and the following photorespiratory pathway (Wang *et al* 2012, Gowik & Westhoff 2011, Ehleringer & Cerling 2002). CO₂ enters the leaf and travels into a mesophyll cell where

it is hydrated to produce bicarbonate ion (HCO_3^-) in the cytoplasm with carbonic anhydrase (CA) as catalyst [**reaction 5.1.3**] (Wang *et al* 2012). HCO_3^- reacts with the 3-carbon acid phosphoenolpyruvate (PEP) catalyzed by phosphoenolpyruvate carboxylase (PEPC) forming oxaloacetate (OAA) (4 C molecule) [**reaction 5.1.4**] (Wang *et al* 2012, Ehleringer & Cerling 2012). OAA is metabolized into malate and then diffuses into the adjacent bundle sheath cell, where it is decarboxylated and refixed in the normal C_3 pathway (Wang *et al* 2012, Ehleringer & Cerling 2012). First, malate is decarboxylated (CO_2 is removed) forming pyruvate (Wang *et al* 2012). The freed CO_2 increases the concentration of CO_2 around Rubisco and some enters the C_3 Calvin cycle (Wang *et al* 2012). Thus, CO_2 is effectively concentrated where Rubisco is located and this results in a high CO_2/O_2 ratio and as a result limited photorespiration (Ehleringer & Cerling 2012). Finally, pyruvate goes back to the mesophyll cell where it is phosphorylated by pyruvate orthophosphate dikinase (PPDK) to form PEP (Wang *et al* 2012). The regenerated PEP can then be used as the CO_2 acceptor of the C_4 cycle (Wang *et al* 2012). C_4 photosynthesis requires the additional cost of adenosine triphosphate (ATP) for the regeneration of PEP from pyruvate (Ehleringer & Cerling 2012).

carbonic anhydrase



PEPC



5.2 Advantages

Certain environmental conditions are more advantageous for C₄ photosynthesis and others for C₃ photosynthesis. C₄ photosynthesis is advantageous under low atmospheric CO₂ and/or high temperatures and/or limited water (Ehleringer & Cerling 2002, Gowik & Westhoff 2011). This is because in lower CO₂ environments and/or higher temperatures, C₃ plants have higher photorespiration rates and thus C₄ photosynthesis would be more efficient than C₃ photosynthesis (Ehleringer & Cerling 2002). C₄ plants have better water-use efficiency (the ratio of the rate of net photosynthesis to transpiration) than C₃ plants (Gowik & Westhoff 2011). C₄ plants are able to acquire enough CO₂ even when keeping their stomata more closed because of the CO₂ concentration mechanism (Gowik & Westhoff 2011), which reduces loss of water by transpiration (Long 1999). In environments with elevated CO₂ concentrations or with cool temperatures, C₃ photosynthesis is more efficient because photorespiration is reduced and the additional ATP cost of C₄ photosynthesis makes it less efficient (Ehleringer & Cerling 2002).

5.3 Isotope fractionations and $\delta^{13}\text{C}$ values

The varying C₃ and C₄ photosynthetic pathway results in differing $\delta^{13}\text{C}$ values (O'Leary 1988, Clark 2015). The $\delta^{13}\text{C}$ of plants is more depleted than the $\delta^{13}\text{C}$ of atmospheric CO₂ because during CO₂ uptake plants fix a proportionally larger amount of ¹²CO₂ and discriminate against the heavier ¹³CO₂ (O'Leary 1988, Clark 2005). This discrimination against ¹³C occurs because ¹³C is heavier than ¹²C and as a result forms stronger chemical bonds and diffuses slower (O'leary 1988). The inefficient step of CO₂ respiration in the C₃ pathway causes the $\delta^{13}\text{C}$ of C₃ plants to be much lower (~-26 to -28

‰) than atmospheric CO₂ (-8‰) because there is an approximate fractionation of 20‰ ($\epsilon^{13}\text{C}_{\text{CH}_2\text{O}-\text{CO}_2} \sim -20\text{‰}$) (Clark 2015, O'Leary 1988). In contrast, C₄ photosynthesis reduces photorespiration, and in turn accelerates CO₂ fixation, resulting in lower fractionation ($\epsilon^{13}\text{C}_{\text{CH}_2\text{O}-\text{CO}_2} \sim -6\text{‰}$) and approximate $\delta^{13}\text{C}$ values of -10 to -14 ‰ (Clark 2015, O'Leary 1988).

The analysis of $\delta^{13}\text{C}$ of total soil respiration (R_T) is a relatively new approach which uses ¹³C as a natural tracer for determining the C substrate being oxidized (Gregorich *et al* 1995, Drewitt *et al* 2009, Rochette *et al* 1999). Determining the origins of R_T is made possible by the contrast in $\delta^{13}\text{C}$ between C₃ ($\sim\delta^{13}\text{C}=-28\text{‰}$) and C₄ ($\sim\delta^{13}\text{C}=-14\text{‰}$) plants (Cheng 1996, Gregorich *et al* 1995, Drewitt *et al* 2009, Rochette *et al* 1999b). C₃ and C₄ plants affect the $\delta^{13}\text{C}$ of SOM as it resembles the $\delta^{13}\text{C}$ of the plant material from which it was derived such that SOM from C₄ plants have $\delta^{13}\text{C}$ values ranging from -12 ‰ to -14 ‰ compared to -24 ‰ to -29 ‰ for SOM from C₃ plants (Gregorich *et al* 1995, Cheng 1996). Therefore, by introducing a C₄ crop into a soil with SOM mostly originating from C₃ species or vice versa results in the SOM containing two isotopically distinct C sources (crop residue carbon (CRC) and soil organic carbon (SOC)) (Gregorich *et al* 1995, Drewitt *et al* 2009, Cheng 1996, Rochette *et al* 1999b). The isotopically distinct SOC and CRC provides a way of quantifying the origin of R_T (Cheng 1996, Gregorich *et al* 1995, Drewitt *et al* 2009, Rochette *et al* 1999b).

6. CO₂ efflux from soils

6.1 Sources and $\delta^{13}\text{C}$ of soil respiration

In agricultural ecosystems, soil organic carbon (SOC) is derived from sources including the remains of the previous years' vegetation, and the remains of the current crop along with the decomposition of its residues (Bernoux *et al* 1998). SOC is dynamic with inputs from decomposition and outputs through transfers to other pools or mineralization (Six & Jastrow 2002). This SOC turnover is quantified by the mean residence time (MRT), which is defined as the average time that carbon resides in the pool at steady state (Six & Jastrow 2002). SOC is a complex mixture containing 3 fractions with differing MRT (Paul *et al* 2001). SOC consists of an active fraction with MRT of months, a large slow intermediate fraction with MRT of years to decades, and a resistant fraction that resists decomposition for centuries to millennia (Paul *et al* 2001). While the active and slow fractions can be characterized by ¹³C, the resistant fraction cannot but it is best characterized by the radioactive isotope of carbon (radiocarbon, ¹⁴C) (Paul *et al* 2001).

Terrestrial vegetation and soils represent the second largest reservoir of carbon (C), with inputs from photosynthesis and losses through soil respiration (R_s) (Clark 2015, Drewitt *et al* 2009, Hardy 2003). R_s is the process where soils release carbon dioxide (CO₂) to the atmosphere (Pal Singh *et al* 2011). In non-vegetated areas, R_s is the result of heterotrophic respiration from organisms (including fungi, bacteria, and protozoans) decomposing SOM whereas in vegetated areas total soil respiration (R_t) originates from R_s and from rhizosphere respiration (R_{rh}), which is the sum of CO₂ respired by plant roots and by microbes that are using root derived C (including

symbiotic mycorrhizal fungi) (Hopkins *et al* 2013, Cheng 1996, Ryan & Law 2005, Rochette *et al* 1999, Pal Singh *et al* 2011, Rochette & Flanagan 1997). R_s releases approximately 10 times more CO_2 to the atmosphere than that released from fossil fuel combustion (Schlesinger 1997, Boden *et al* 2010). Therefore, the large amount of CO_2 released to the atmosphere from R_s (Rochette *et al* 1999) is of particular concern as it is contributing to its large atmospheric concentration (390.5 ppm) and its rapid rate of increase (average 2.0 ± 0.1 ppm yr^{-1} during 2002-2011) (IPCC 2013) thus promoting the greenhouse effect leading to climate change (Werth & Kuzyakov 2008).

Two primary methods for distinguishing the contributions of R_{rh} and R_s are root-exclusion and the use of ^{13}C isotope ($^{13}C/^{12}C$, $\delta^{13}C$) (Hanson *et al* 2000, Rochette *et al* 1999). The root-exclusion approach calculates R_{rh} by subtracting the CO_2 emission rates from soils without roots (R_s) from those of soils that contain roots (R_t) (Hanson *et al* 2000, Rochette *et al* 1999). If the $\delta^{13}C$ of plants varies from that of the soil it is growing in, then the ^{13}C isotope approach is valid and the R_{rh} can be quantified as the difference between ^{13}C of R_t (ie vegetated area) and ^{13}C of R_s (ie non-vegetated area) (Rochette & Flanagan 1997, Rochette *et al* 1999). The ^{13}C isotope method uses the difference in natural abundances of ^{13}C in C_3 and C_4 plants (Rochette & Flanagan 1997, Rochette *et al* 1999) as the inefficient step of CO_2 respiration in the C_3 pathway causes the $\delta^{13}C$ of C_3 plants to be much lower (~ -26 to -28 ‰) than that of C_4 plants who reduce photorespiration (~ -10 to -14 ‰) (Clark 2015, O'Leary 1988). The $\delta^{13}C$ of SOM resembles the $\delta^{13}C$ of the plant material from which it was derived such that SOM from C_4 plants have $\delta^{13}C$ values ranging from -12 ‰ to -14 ‰ compared to -24 ‰ to -29 ‰ for SOM from C_3 plants (Gregorich *et al* 1995, Cheng 1996). Many studies have used

root exclusion (eg. Rochette *et al* 1999, Irvine *et al* 2008, Ruehr & Buchmann 2010) and ^{13}C (eg. Rochette *et al* 1999, Rochette & Flanagan 1997, Werth & Kuzyakov 2008), but while both root-exclusion and ^{13}C methods have advantages and disadvantages (Hanson *et al* 2000), both approaches were found to result in similar R_{rh} values in a study (Rochette *et al* 1999).

Degradation of biomass generates a much higher CO_2 concentration in soils (P_{CO_2} between 0.003 to 0.03 atm) than in open air (0.0004 atm) (Clark 2015). This high CO_2 concentration in soils results in a much more negative $\delta^{13}\text{C}$ ($\delta^{13}\text{C}$ close to the ranges for C_3 and C_4 vegetation) (Clark 2015). Soil CO_2 has a slightly modified value because of the outward diffusion from high P_{CO_2} in soils to low P_{CO_2} in air (Clark 2015). $^{13}\text{CO}_2$ diffuses slower out of the soil than the lighter $^{12}\text{CO}_2$ molecule with a ratio of 1.0042 which results in a fractionation of 4 ‰ resulting in a slight enrichment on the remaining soil CO_2 (Hesterberg & Siegenthaler 1991, Rochette *et al* 1999, Clark 2015). In most C_3 landscapes, the soil CO_2 has $\delta^{13}\text{C}$ values in the range of -20 to -23 ‰ (Clark 2015).

6.2 Microbial respiration

Soil microbial respiration decomposes the stored living and dead biomass that was produced from photosynthesis and is a strongly exothermic reaction [**reaction 6.2.1**] (Clark 2015, Clark & Fritz 1997). Much of the biomass is oxidized to CO_2 , which is then recycled through photosynthesis (Clark & Fritz 1997). Some biomass is converted into humic substances (humic acids (HA) and fulvic acids (FA)), which are alkali-soluble acids that are less labile and more resistant and give the dark colour to soil (Clark & Fritz 1997, Clark 2015) [**reaction 6.2.2**]. HA are the most common soil-derived organic

matter substances and are high molecular weight (10,000-plus Da, unified atomic mass units), whereas FA makeup the remaining substances and are lighter weight (Clark & Fritz 1997, Clark 2015). Both HA and FA are complex aromatic carbon structures with OH and COOH functional groups (Clark 2015). HA are composed of roughly 50 to 60 % C, 30-40 % O, 5 % H, and 5 % N, whereas FA tend to have lower C contents (**Fig 6.2.1**) (Clark & Fritz 1997). HA precipitate from solution at pH less than 2 whereas FA is soluble at all pH values (Clark & Fritz 1997). Decomposition of organic matter also results in other lighter weight and more labile organic acids being produced, such as acetic (CH₃COOH), carboxylic (C₂O₂[OH]₂), formic (HCOOH), and lactic (CH₃CH[OH]COOH) acids (Clark 2015).

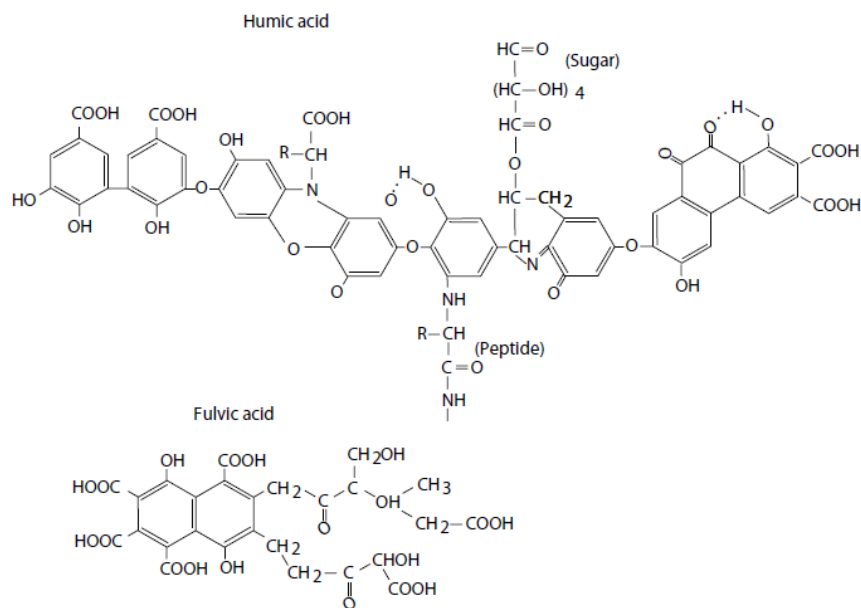
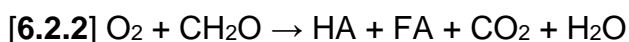


Figure 6.2.1 Structure of humic and fulvic acids as published in Clark 2015 and originally modified from Stevenson 1985 and Buffle 1977.

Microbial respiration can be traced through analysis of $\delta^{13}\text{C}$ (Clark 2015). Bacteria preferentially degrade the ^{12}C portion of a substrate and discriminate against the ^{13}C enriched portion (Clark 2015). Thus, as a particular substrate compound is consumed by the bacteria, the substrate will become enriched in ^{13}C according to a Rayleigh distillation (Clark 2015). The Rayleigh distillation equation is $R=R_0f^{(\alpha-1)}$, where in this case R_0 is the substrates initial isotope ratio ($^{13}\text{C}/^{12}\text{C}$) and R would be the ratio after a given portion of the substrate has been consumed by bacteria (Clark 2015, Clark & Fritz 1997). The fraction f is the residual substrate reservoir and α is the fractionation factor at the specific temperature (Clark 2015, Clark & Fritz 1997). Therefore, the distribution of ^{13}C reflects the rate of production and consumption of a compound, according to the Rayleigh distillation function (Clark 2015).

6.3 Autotrophic Respiration

Autotrophic respiration occurs when metabolic energy is used in the synthesis and maintenance of plant tissue (Ryan & Law 2005). Autotrophic respiration includes respiration by mycorrhizae if they receive carbohydrate from the roots (Ryan & Law 2005). When environmental conditions are favourable for growth, root respiration involves metabolism of recently fixed carbohydrates from photosynthesis, whereas during stress, such as summer drought, photosynthesis declines and stored carbohydrates are used in respiration to maintain the plant (Ryan & Law 2005, Högberg *et al* 2001). Drought greatly affects plants by reducing photosynthesis, which in turn reduces root growth and growth respiration, and eventually decreases maintenance respiration (Domec & Gartner 2003).

6.4 Factors affecting soil respiration

The quantity and chemical composition of substrate affects soil respiration. Microbial respiration is controlled by the substrate supply, which is affected by the amount of litterfall and its chemical composition, and the amount of carbohydrate transported by phloem to roots as a result of photosynthesis (Ryan & Law 2005). Soil respiration from old recalcitrant SOC is limited by its chemical composition (Sollins *et al* 1996), whether it is physically protected in soil aggregates (Swanston *et al* 2002), and whether other sources of labile carbon are available (Pendall *et al* 2004, Ryan & Law 2005). A decrease in substrate supply decreases soil respiration (Ryan & Law 2005).

Temperature, soil moisture, crop growth rate and plant tissue nitrogen concentration affect soil respiration. Temperature and soil moisture control the amount of substrate and hence control soil respiration (Pendall *et al* 2004). For example, temperature and water were found to be the primary limiting factors for soil respiration at the seasonal scale (Yuste *et al* 2007). Many studies found soil respiration to be correlated with soil temperature (eg. Rochette *et al* 1999, Datta *et al* 2013, Hesterberg & Siegenthaler 1991, Ball *et al* 1999). In dry climates, soil respiration can increase quickly after precipitation because microbes can rapidly respond to available water (Kelliher *et al* 2004). For example, 120 hours after rainfall, CO₂ flux was positively correlated with accumulated rainfall (Hernandez-Ramirez *et al* 2009). Rochette *et al* 1991 found that respiration was nine times greater 3 hours after heavy rainfall than before rainfall. Similarly, the increased respiration when dry samples are rewetted has been observed in the lab (Orchard & Crook 1983). Respiration is affected by crop growth rate as indicated by the greatest respiration occurring during the period of maximum crop

growth (Rochette *et al* 1991). Root respiration rates are directly related to temperature and tissue nitrogen concentration (Ryan *et al* 1996).

The amount of oxygen (O₂) available in the soil will affect how much CO₂ is produced from microbial respiration. If the available soil pore space for O₂ is reduced, this will reduce aerobic microbial activities, including degradation of SOM, thus resulting in less CO₂ production (Datta *et al* 2013). In contrast, if the available soil pore space for O₂ is increased, this will increase the oxidation rate of SOM thus increasing microbial respiration resulting in increased CO₂ production (Datta *et al* 2013). For example, during or immediately after heavy rainfall, there were periods of low CO₂ flux and this may be the result of either a more anaerobic environment causing a reduction in microbial activity or reduced CO₂ diffusion to the soil surface in soil with more water-filled pores (Rochette *et al* 1999, Rochette *et al* 1991, Ball *et al* 1999, Hesterberg & Siegenthaler 1991).

7. CH₄ efflux from soils

7.1 Methanogens and methanotrophs in soils

Agricultural soils play an important role in regulating atmospheric GHG concentrations because they are either a sink or a source for CH₄ (Nangia *et al* 2013, Hernandez-Ramirez *et al* 2009). Soils can be sources or sinks for CH₄ depending on the activity of methanogens and methanotrophs (*ie* bacteria) (Topp & Pattey 1997). In many terrestrial ecosystems methanogens and methanotrophs are simultaneously active (Topp & Pattey 1997). When the activities of methanogens dominate, CH₄ emissions occur, whereas when methanotrophs dominate CH₄ consumption is larger

(Topp & Pattey 1997). In some soils, both CH₄ production and CH₄ consumption can occur (Powlson *et al* 1997).

Under anaerobic conditions and in the absence of other electron acceptors with greater E_h (*ie* nitrate, oxidized manganese, ferric iron, sulfate), methanogens obtain their energy from reducing more oxidized forms of carbon into CH₄ thus making the soil a CH₄ source (**Table 2.1**) (Topp & Pattey 1997). Most methanogens reduce CO₂ (*ie* CO₂ reduction) using hydrogen as the electron donor [**reaction 7.1.1**] whereas others reduce small organic molecules (*ie* methanogenesis), such as methanol, acetate or methylamines to form CH₄ and CO₂ [**reaction 7.1.2**] (Topp & Pattey 1997, Clark 2015). Therefore, methanogens produce CH₄ through two pathways: CO₂ reduction and acetate fermentation (Avery *et al* 2003). Details of the small substrate range of methanogens and their reactions are indicated in **Table 7.1.1**. Methanogens depend on other organisms for providing their substrate and for creating reducing conditions, and as such they are rate-limited by the activities of other microbes (Topp & Pattey 1997).



Table 7.1.1 The small substrate range that methanogens use and their reactions to produce CH₄ in soils, as published in Topp & Pattey 1997.

Substrates		Products
4H ₂ + CO ₂	→	CH ₄ + 2H ₂ O
4 Formate	→	CH ₄ + 3CO ₂ + 2H ₂ O
4 (2-Propanol) + CO ₂	→	CH ₄ + 4 acetone + 2H ₂ O
Acetate	→	CH ₄ + CO ₂
4 Methanol	→	3CH ₄ + CO ₂ + 2H ₂ O
4 Methylamine + 3H ₂ O	→	3CH ₄ + CO ₂ + 4NH ₄ ⁺
2 Dimethyl sulfide + 2H ₂ O	→	3 CH ₄ + CO ₂ + H ₂ S

Under aerobic conditions, soils are a CH₄ sink because methanotrophs oxidize CH₄ to CO₂ or incorporate it into microbial biomass (Powlson *et al* 1997). Methanotrophs are physiologically diverse CH₄ oxidizing bacteria and they are found in all types of soils (Topp & Pattey 1997). CH₄ oxidation (*ie* CH₄ + O₂) results in the production of CO₂ through the sequential intermediates methanol (CH₃OH), formaldehyde (CH₂O), and formic acid (CH₂O₂) (Topp & Pattey 1997). Methanotrophs obtain energy from CH₄ oxidation and they use the intermediate CH₂O as their primary carbon source for growth (Topp & Pattey 1997). Well-drained soils (*ie* porous soils) are CH₄ sinks because atmospheric O₂ is able to diffuse into the soil thus allowing CH₄ oxidation to occur (Topp & Pattey 1997)

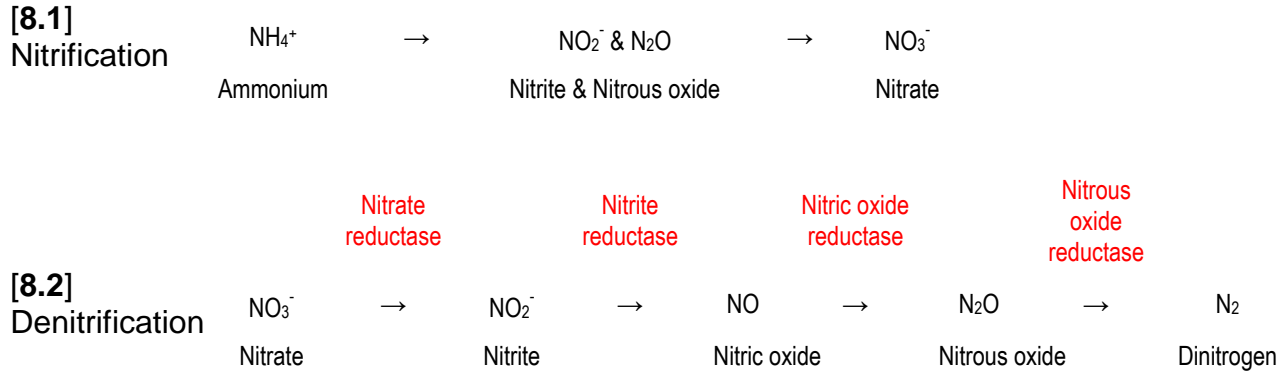
7.2 Factors affecting methane fluxes

Soil moisture and soil redox potential (E_h) affect CH₄ fluxes in soils (Topp & Pattey 1997, Hou *et al* 2000). Soils generally contain methanogens and methanotrophs and, therefore, soil moisture is a major factor affecting whether the activity of methanogens or methanotrophs will dominate (Topp & Pattey 1997). In anaerobic soil with high soil moisture, such as below the water table, methanogens are active and produce CH₄ from the decomposition of organic matter (Topp & Pattey 1997). In aerobic soil, such as above the water table, CH₄ diffuses in and it is consumed by methanotrophs through oxidation resulting in CO₂ production (Topp & Pattey 1997). As soil water content increases, CH₄ uptake decreases (Lessard *et al* 1994) because CH₄ diffuses approximately 10⁴ times slower in water than in air (Topp & Pattey 1997). For example, CH₄ fluxes were greater with greater soil moisture (Datta *et al* 2013, Mayer & Conrad 1990), and this may be due to decreased E_h or lower availability of O₂ which

promotes the population growth of methanogenic bacteria (Mayer & Conrad 1990). CH₄ emissions are strongly correlated with soil E_h, and significant CH₄ emissions only occurred when soil E_h was lower than -100 mV (**Table 2.1**) (Hou *et al* 2000).

8. N₂O efflux from soils

Agricultural soils are an important source of N₂O (Ball *et al* 1999, Nangia *et al* 2013, Hernandez-Ramirez *et al* 2009, Mosier & Kroeze 2000) and as such N₂O concentration in soils have been found to be significantly higher than in the atmosphere (Hesterberg & Siegenthaler 1991). In aerobic conditions, N₂O soil emissions are produced by microbial nitrification, which is the oxidation of ammonium (NH₄⁺) into nitrite (NO₂⁻) and then to nitrate (NO₃⁻) releasing N₂O as a byproduct [**reaction 8.1**] (Topp & Pattey 1997, Robertson & Grace 2004, Dobbie & Smith 2006). In anaerobic conditions, N₂O soil emissions are produced by denitrification, which is the reduction of nitrate (NO₃⁻) to dinitrogen gas (N₂) (Coyne 2008, Robertson & Grace 2004, Dobbie & Smith 2006). Details of the denitrification reaction are presented in **reaction 8.2** and the catalysts (*ie* nitrogen oxide reductases) are indicated in red (Coyne 2008, Robertson & Grace 2004, Dobbie & Smith 2006). N₂O is an intermediate product of denitrification, which for some environmental conditions and some denitrifiers is the end product (Cavigelli & Robertson 2000, Oehler *et al* 2007). Denitrification occurs in wet soils because the saturated conditions slow down oxygen diffusion creating a lack of oxygen (Robertson & Grace 2004). Nitrification and denitrification can both occur in well-drained soils because wet soils can be found inside soil aggregates (Robertson & Grace 2004).



N_2O soil emissions are affected by temperature, soil moisture, and soil E_h .

Contradicting results exist for the effect of soil temperature on N_2O soil emissions, for example, temperature did not affect N_2O soil emissions in one study (Hesterberg & Siegenthaler 1991), but in another study N_2O fluxes from an Eastern corn belt soil were correlated with soil temperature (Hernandez-Ramirez *et al* 2009). Studies have reported increasing N_2O emissions from agricultural soils with increasing soil moisture (eg. Dobbie *et al* 1999, Dobbie & Smith 2001, Pihlatie *et al* 2004, Ball *et al* 1999). Similarly, Datta *et al* 2013 found significantly higher N_2O emissions during the post-snow cover period due to greater soil water content, which indicates that there may be higher denitrification activity occurring in the soil. N_2O soil emissions are strongly correlated with soil E_h where N_2O emissions were not significant below +200 mV (**Table 2.1**) (Hou *et al* 2000).

9. Agricultural tile drainage management and associated effects

Agricultural lands cover an estimated 40 to 50 % of the Earth's land surface (Smith *et al* 2007) and thus contribute a significant portion to greenhouse gas emissions (IPCC 2013). Many agricultural lands have artificial subsurface tile drainage, including 17.4 million ha in the Midwestern United States (Jaynes & Isenhardt 2014) and more than

1.6 million ha in Ontario, Canada (Sunohara *et al* 2015). Artificial subsurface tile drainage is found in many agricultural areas in order to improve field drainage for crop production (Sunohara *et al* 2015, Jaynes & Isenhardt 2014, Skaggs *et al* 1994).

Artificial subsurface drainage involves placing drainage tiles, which are perforated tubes, below the soil surface throughout a field (Nangia *et al* 2013). Drainage tiles are typically located 1 m below the land surface and 15 m apart from each other (Nangia *et al* 2013). Tile laterals are approximately 4" in diameter and are the length of the field whereas the header tiles are approximately 6" in diameter and span the width of the field (Nangia *et al* 2013). For parallel drainage tiles in the field, the water table gradient towards a drainage tile results in drawdown causing a deeper water table depth below surface (WTDBS) for over tile (OT) locations compared to between tile (BT) locations, which would have a shallower WTDBS (USDA 2011, Skaggs *et al* 2005). Subsurface tile drainage systems have traditionally been managed as uncontrolled tile drainage (UTD), whereby the tile water discharges directly through a header tile and into a watercourse, including a drainage ditch or creek. Typically UTD management results in tile flow during the spring whereas tile flow decreases to zero during summer as the water table declines to the level of the tiles due to little precipitation. Therefore, problems associated with UTD management include loss of soil moisture during the summer months, an intensified loss of nutrients from the field through drainage waters, and nutrient contamination of the receiving waters that tile fields drain into.

Artificial subsurface tile drains are used in control tile drainage (CTD) management, which is a beneficial flexible agricultural management practice that manages the WTDBS by controlling the amount of water that can leave a field through

subsurface tile drains (Gilliam *et al* 1979). CTD management contrasts with UTD where there is no management of the water leaving a field through subsurface tile drains. CTD involves increasing or decreasing the WTDBS by adjusting the height of stop gates in a water flow control structure, which is connected to a header tile in the field (Sunohara *et al* 2015, Kross *et al* 2015, Cicek *et al* 2010, Mejia *et al* 2000). If all stop gates are removed, the field is UTD but if stop gates are added in the water flow control structure then the field is managed as CTD. For CTD fields, flow control structure water overflow heights are generally set at 0.4 m below the surface at planting and remain at this depth until the plants are harvested (Nangia *et al* 2013), because at this depth, plant roots have been shown to interact more with the water table (ASABE 1990). If the water table reaches a critically high level (*ie* exceeds the stop gate level of 0.4m), tile water flows over the stop gates in the water flow control structure and out of the field through a drainage tile in order to lessen the water-logging damage to crops (Cicek *et al* 2010). CTD management requires flat topography, coarser textured soils, a subsurface tile drainage system, and enough precipitation to allow for storage (Mejia *et al* 2000).

There are many benefits associated with CTD management. CTD removes excess water from the field, which allows early-season machinery operations to occur on the field (Nangia *et al* 2013, Cicek *et al* 2010, Mejia *et al* 2000) and lessens the water-logging damage to crops (Cicek *et al* 2010). CTD can elevate water tables and in turn reduce water and nutrient losses from fields, thus providing crops more access to water and nutrients during critical growth stages (Tan *et al* 1999, Cicek *et al* 2010, Sunohara *et al* 2015). CTD can increase crop yields (eg Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Kross *et al* 2015, Mejia *et al* 2000, Cicek *et al* 2010) and

reduce the amount of nutrient loss in drainage water (eg Sunohara *et al* 2015, Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Drury *et al* 1996). For example, Tan *et al* (1999) found that CTD reduced total nitrate loss by 37% in 1995-1997 and improved yields of both tomatoes (11% in 1995) and corn (64% in 1996) on a sandy loam soil in Ontario. A 4-year field study on sandy-loam soil in southern Sweden found that CTD lowered the amount of N by 65-95% in tile drainage water and increased N uptake by crops (3-14 kg/ha increase) thus resulting in greater crop yields (2-18% larger) (Wesstrom & Messing 2007). Ng *et al* (2002) did a plot study in a sandy loam soil in Southwestern Ontario and found that CTD increased corn yields by 64% and reduced total nitrate loss by 57% compared to UTD treatment. A two-year field study in eastern Ontario found that corn yields (13.8%, 6.6%) and soybean yields (8.5% and 37.3%) were greater in CTD fields with a water table set at 0.5 m than UTD fields in 1995 and 1996 respectively (Mejia *et al* 2000). Cicek *et al* (2010) used remote sensing from 2005-2008 to examine CTD on a large spatial scale (~950 ha watershed, *ie* not plot or field scale) and found an average corn and soybean grain yield increase of 3% and 4%, respectively, greater than those from UTD fields. In a clay loam soil, Drury *et al* (1996) found that the average annual nitrate loss was reduced by 43% for CTD compared to UTD fields. Furthermore, CTD can result in more uniform crop growth (Cicek *et al* 2010) and can stabilize yields from year to year by minimizing the risk of crop losses from uncertain rainfall compared to UTD fields (Mejia *et al* 2000). The impact of CTD on crop performance varies depending on soil, weather, and topography (Cicek *et al* 2010).

Although the links between CTD and increased crop yields and reduced adverse nutrient inputs in nearby waterways are well documented (eg Tan *et al* 1999, Wesstrom

& Messing 2007, Ng *et al* 2002), these benefits could result in an environmental trade-off of increased GHG emissions. CTD modifies the WTDBS and as the WTDBS gets shallower, the water filled pore space (WFPS) in the topsoil increases resulting in increased soil moisture (Dobbie & Smith 2006). With the possible increased soil moisture of CTD fields, there is the potential for the development of soil conditions that may increase GHG emissions (Nangia *et al* 2013, Elmi *et al* 2000). Increasing soil moisture can result in greater R_T (Davidson *et al* 2000) as studies indicate R_T was greatest at relatively high moisture levels and reduced under very dry and very wet soil conditions (Orchard & Cook 1983, Bowden *et al* 1998, Linn & Doran 1984, Skopp *et al* 1990). CH_4 flux may be controlled by soil moisture (Keller & Reiners 1994), where soil consumption of atmospheric CH_4 is greatest in dry soils and CH_4 production is greater with increased soil moisture (Keller & Reiners 1994, Sass *et al* 1992). CH_4 uptake is controlled by soil moisture, where decreased CH_4 uptake occurs at very low and very high soil moisture (Bowden *et al* 1998), and maximum rates of CH_4 uptake occurs at approximately 50-70% of WFPS (Nesbit & Breitenbeck 1992, Bowden *et al* 1998). Increasing soil moisture can promote denitrification resulting in greater soil N_2O efflux rates (Knowles 1982) as supported by several studies, which found N_2O emissions increased as the WTDBS got shallower (Dobbie & Smith 2006, Kliewer & Gilliam 1995, Jacinthe *et al* 2000, Elmi *et al* 2000), combined N_2O and N_2 emissions increased in a shallower water table (Elmi *et al* 2005), and N_2O emissions increased as the WFPS increased (Dobbie & Smith 2006, Dobbie & Smith 2001, Dobbie *et al* 1999, Dobbie & Smith 2003, Rabot *et al* 2015, Keller & Reiners 1994).

Within the literature on GHG emissions in CTD agricultural fields, knowledge gaps exist on GHG fluxes and their associated spatial distribution. While the spatial and temporal variability of GHG emissions in non-tile drained agricultural fields has been well documented (eg Rochette *et al* 1991, Hatfield & Parkin 2012, Stoyan *et al* 2000, Jiang *et al* 2010, Hao & Jiang 2014, Yanai *et al* 2003), the spatial distribution of GHG emissions within CTD managed fields is not well known. The difference in WTDBS for OT versus BT locations in a CTD field is expected to cause the GHG emissions to vary spatially within the field. Only a few studies have quantified the relationship between water table depth and N₂O emissions (Elmi *et al* 2005, Dobbie & Smith 2006, Nangia *et al* 2013), and only one study has compared and contrasted CO₂, CH₄, and N₂O emissions from CTD and UTD fields (Nangia *et al* 2013), but no reports were found on the effects of CTD management with respect to spatial distribution of GHG emissions. Furthermore, while studies have used $\delta^{13}\text{C}$ of R_T to reveal the source of the respired carbon (eg Drewitt *et al* 2009, Rochette *et al* 1999, Rochette *et al* 1999b), no reports have used this method to identify and compare the source of R_T throughout the growing season from CTD and UTD fields and thus to determine whether CTD affects the contribution of the carbon substrate being oxidized. No studies have used $\delta^{13}\text{C}$ to compare the spatial distribution of R_T within CTD fields. The quantitative estimates of the CO₂ produced by each source (R_s and R_{rh}) in CTD fields have not been documented but it is important in order to evaluate implications of CTD on the carbon dynamics in the soil plant system. Knowledge of GHG emissions in CTD agricultural fields is essential for properly assessing ecological impacts associated with CTD, for improving

estimates in GHG inventories, and to ultimately adopt best management practices which reduce GHG emissions from the agriculture sector.

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Chapter 2 Introduction and proposed study

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1. Background information

Agricultural lands cover an estimated 40 to 50 % of the Earth's land surface (Smith *et al* 2007) and thus contribute a significant portion to greenhouse gas (GHG: CO₂, CH₄, N₂O) emissions (IPCC 2013). Agricultural soils have an important contribution in regulating atmospheric GHG concentrations because they are a source and sink of CO₂ and N₂O (Ball *et al* 1999, Nangia *et al* 2013, Hernandez-Ramirez *et al* 2009), and either a sink or a source for CH₄ (Nangia *et al* 2013, Hernandez-Ramirez *et al* 2009). Many agricultural lands have artificial subsurface tile drains, including 17.4 million ha in the Midwestern United States (Jaynes & Isenhardt 2014) and more than 1.6 million ha in Ontario, Canada (Sunohara *et al* 2015). Control tile drainage (CTD) is a flexible agricultural management practice that manages the water table depth below surface (WTDBS) by controlling the amount of water that can leave a field through artificial subsurface tile drains (Gilliam *et al* 1979). CTD management contrasts with uncontrolled tile drainage (UTD) where there is no management of the water leaving a field through subsurface tile drains.

With the widespread use of CTD management in agriculture, it is important to understand the environmental impacts associated with it. In the literature, it is documented that CTD can increase crop yields (eg Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Kross *et al* 2015, Mejia *et al* 2000, Cicek *et al* 2010) and reduce the amount of nutrient loss in drainage water (eg Sunohara *et al* 2015, Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Drury *et al* 1996). These benefits of increased yields and reduced nutrient loss in drainage water associated with CTD could result in an environmental trade-off of increased GHG emissions. Greenhouse gasses

are currently a major concern because their increasing atmospheric concentrations over the last several centuries have resulted in climate change (IPCC 2013). CO₂ is a particular concern because it currently has the largest atmospheric concentration (390.5 ppm) compared to the other greenhouse gasses (CH₄=1.803 ppm, N₂O=0.324 ppm) and it has a rapid rate of increase (average 2.0 ± 0.1 ppm yr⁻¹ during 2002-2011) (IPCC 2013).

Within the literature, there is a lack of knowledge on GHG emissions from CTD agricultural fields. Knowledge of GHG emissions in CTD agricultural fields is essential for properly assessing ecological impacts associated with CTD, for improving estimates in GHG inventories, and to ultimately adopt best management practices to reduce GHG emissions from the agriculture sector. While one study has compared and contrasted CO₂, CH₄, and N₂O emissions from CTD and UTD fields (Nangia *et al* 2013), knowledge gaps exist regarding the effect of CTD on soil GHG fluxes and associated spatial distribution, on residue decomposition, and specifically on rhizosphere and soil respiration. No previous study has used isotopes of ¹³C for quantifying whether soil respiration varies between CTD and UTD managed fields. ¹³C is a very useful tool as it is measured very accurately using an isotope ratio mass spectrometer (IRMS) resulting in a very small standard deviation around repeated samples. Thus, with isotope measurements people can be very confident in the results.

2. The proposed study

With the existing knowledge gap in the literature regarding GHG emissions in CTD agricultural fields and their importance due to climate change, a field-scale study was designed to decrease this knowledge gap. There were two main study objectives.

The first objective was to determine whether CTD affects soil GHG emissions throughout a dry (2012) and a wet (2013) growing season for corn, soybean, and forage fields in eastern Ontario. The second objective was to determine whether location in a field with respect to a tile drain (i.e. OT vs BT) will be important in GHG emissions due to: water table mounding BT (Smedema et al., 1983), different soil properties OT (due to backfilling of trenches at time of installation), and biopores that have been shown to preferentially occur OT (Nuutinen et al., 2001).

To meet these two main objectives, the study was divided into three sections. The first section (Chapter 3 of the thesis) was designed to provide information on CO₂, CH₄, and N₂O soil emissions from CTD fields and whether OT vs BT location affected the GHG emissions within CTD fields. The specific objectives of section one were to: (1) measure CO₂, CH₄, and N₂O soil emissions between CTD and UTD fields, and (2) evaluate whether OT vs BT locations affect GHG emissions in CTD fields. The second section (Chapter 4 of the thesis) involved comparing ¹³C of R_T (i.e total soil respiration) between CTD and UTD soybean, corn, and forage fields in order to provide insights into the effects of CTD on residue decomposition and soil gas diffusivity. The specific objectives of this second part of the study were to: (1) measure the δ¹³C of R_T for CTD and UTD fields, and (2) evaluate whether OT and BT locations affect δ¹³C of R_T in CTD fields. The third part of the study (Chapter 5 of the thesis) was designed to separate total soil respiration (R_T) into its rhizosphere (R_{Rh}) and soil (R_S) components in order to identify whether CTD management affects the contribution of rhizosphere or soil respiration to total soil respiration (R_T) in corn and soybean fields. The specific objective of section three was to compare R_T, R_{Rh}, and R_S between CTD and UTD fields.

3. Field work

The field study was carried out within the ~950 ha South Nation Watershed (45.26 N, 75.18 W) in eastern Ontario, Canada (Cicek *et al* 2010) on Bainsville silt loam soil (Wicklund & Richards 1962) (**Figure 1**). This two year field-scale study (2012 – 2013) was conducted during two growing seasons, where each growing season is defined here as beginning when the stop gates were closed in CTD fields and lasting until they were reopened (**Table 1**). This study was done during the growing season because it is during this time that drainage management varies for CTD and UTD fields. This study examined two field pairs in 2012 (Field 1 & Field 2, Field 11 & Field 14) and three field pairs in 2013 (Field 1 & Field 2, Field 11 & Field 14, Field 12 & Field 13). The fields within a field pair were under a common cropping practice, in close proximity to each other, and treated the same except but varying in drainage management (one field was CTD and the other UTD). This study examined subsurface tile drained agricultural fields containing corn, soybean, and forage (timothy & alfalfa) (**Figure 2**). All fields studied are tile drained with lateral tiles (0.102 m in diameter) located at a depth of ~1 m and with a spacing of ~15 m.

Weekly samples of soil GHG emissions, monthly samples of soil (at 3 depths: 0-15, 15-30, 30-60 cm), and samples of crops just prior to harvest were collected from each field. The WTDBS was automatically measured every 15 minutes throughout the field season. SWC20, ST10, and air temperature were measured with 0.5 m of each GHG collection site. SWC20 (*ie* water volume/total volume) of the soil was measured using time-domain reflectometry (TDR) probes. ST10 and air temperature were measured using a digital thermocouple thermometer and temperature probe. The

location of GHG sampling, soil sampling, water table monitoring sites, and OT and BT locations are indicated (**Figure 3**). Two weather stations located within the study area (one in Field 14 and the other near Field 1) continuously measured precipitation (mm), soil temperature (0-15 cm depth) (°C), air temperature (°C), relative humidity (RH, %), and solar radiation (SR, W/m²) every 30 minutes.

The GHG chambers were designed based on recommendations in the literature from Rochette and Bertrand 2008 and Rochette 2011. The chamber geometry varied depending on the crop type, where soybean were rectangular chambers (0.75 x 0.15 m), corn had square chambers (0.35 x 0.35 m), and forage had cylindrical chambers (1.28 m circumference and lid height 0.32 m). The chambers were designed with adequate insulation including lids with memory foam undersides and weights on top of the lids to prevent leakage of air from chambers. The upper surface of the chamber lids were covered in reflective material to minimize air temperature change. There was a vent tube on the chambers in order to transmit pressure fluctuations. And the bases were semi-permanent (ie installed at beginning of growing season and left there) to reduce soil disturbance.

GHG samples emitted from the soil surface to the atmosphere were collected from non-steady state chambers using the method by Rochette and Bertrand 2008, which was also similarly used by Nangia *et al* 2013. The steps for soil GHG sampling are briefly described (**Figure 4**): 1. Insert chamber base at beginning of growing season to limit soil disturbance. 2. Sample between 10:00 and 2:00 (peak emission times). 3. Cap chamber with lid, insert syringe into sampling port, flush syringe to get rid of any residual air, collect syringe sample at time=0 min. 4. Sample t=0 min for all replicate

chambers within group (4 chambers per group in 2012, and 6 chambers per group in 2013). 5. Revisit chambers for successive rounds of sampling. Five GHG samples were collected per chamber, where soybean and corn were sampled at 0, 6, 12, 18, and 24 min, and forage was sampled at 0, 12, 24, 36, and 37 minutes for vegetated location and 0, 6, 12, 18, and 19 minutes for non-vegetated locations in forage fields. The first 4 GHG samples collected were sent to the GC for concentration of CO₂, CH₄, and N₂O, whereas the 5th GHG sample was sent to the IRMS for analysis of $\delta^{13}\text{C}$ of CO₂.

Greenhouse gas samples were collected from OT and BT locations, and from vegetated and non-vegetated locations within CTD and UTD fields. OT refers to samples collected from above a lateral tile whereas BT locations refers to samples coming from between the tile lateral (*ie* 7.5 m away from the OT sample) (**Figure 5**). Samples were also collected from vegetated (plants growing next to chamber) and non-vegetated sites (plants cleared for a 1 m distance around chamber) (**Figure 6**). Vegetated sites provide a measure of total soil respiration (R_T) whereas non-vegetated sites measure soil respiration (R_s) (**Figure 7**). In non-vegetated areas, R_s is the result of heterotrophic respiration from organisms decomposing soil organic matter (SOM) whereas vegetated areas provide a measure of total soil respiration (R_T), which originates from R_s and from rhizosphere respiration (R_{rh}). R_{Rh} is the sum of CO₂ respired by roots and by microbes that are using root derived carbon (Rochette *et al* 1999, Rochette & Flanagan 1997). There was 96 GHG chambers sampled during the study and approximately 7200 GHG samples collected (**Table 2**).

4. Lab work

While many soil samples were collected during the growing seasons, only a subset were analyzed for $\delta^{13}\text{C}$. The soil was prepared for $\delta^{13}\text{C}$ analysis using the method described by Brodie *et al* 2011. Briefly, the soil was dried, ground, mixed, and plant materials were removed. The soil was placed in silver capsules, weighed, acidified with HCl to remove carbonates, and packaged. The soil samples were analyzed for $\delta^{13}\text{C}$ on EA-IRMS.

The plant samples were separated into their parts (ex. pods/cobs, leaves, root, stem), dried, finely ground, mixed, and 2g from each plant for composite samples were combined. The plant samples were packed in tin capsules and analyzed for $\delta^{13}\text{C}$ on EA-IRMS.

GHG sampling involved preparation of sampling containers with extra septum, desiccant and evacuated prior to going to the field. He filled extainers were created to assess contamination by being brought to the field in the same way as our sampling containers. Reference standard gasses (1 for every 5 field samples) were prepared in the lab. The first 4 samples collected from each GHG chamber were analyzed for CO_2 , CH_4 , and N_2O concentration using the Varian CP-3800 gas chromatograph. The 5th sample collected from each GHG chamber were analyzed for $\delta^{13}\text{C}$ on a continuous flow gasbench + DeltaPlusXP IRMS at the G.G. Hatch Stable Isotope Lab, University of Ottawa.

5. Data analysis

CO₂, CH₄, and N₂O fluxes were calculated using the rate of concentration change inside the chamber during deployment (Rochette and Bertrand 2008). A linear regression model was used for each series of GHG concentration versus time data from each deployed chamber. The linear model was used because it is the safer model that usually minimizes the errors in flux calculation, whereas nonlinear models may exhibit extreme sensitivity to measurement error (Rochette and Bertrand 2008).

The $\delta^{13}\text{C}$ of CO₂ soil emissions were corrected by removing the contribution from atmospheric CO₂ that was initially inside the chamber [1]. Where, $\delta^{13}\text{C}_{\text{measured}}$ is sample's result from IRMS. $[\text{CO}_2]_{\text{measured}}$ is the calculated concentration of CO₂ in the sample at time of sampling ($[\text{CO}_2]_{\text{measured}} = (\text{slope}_{\text{CO}_2 \text{ vs time}} \times \text{time}_{\text{CO}_2 \text{ sample}}) + [\text{CO}_2]_{\text{atm}}$). $\delta^{13}\text{C}_{\text{atm}}$ is set at -8 ‰ due to value sampled from Fraserdale Ontario (Huang 2015). $[\text{CO}_2]_{\text{atm}}$ is set at 400 ppm as value sampled from study area. The $[\text{CO}_2]_{\text{emitted}}$ equals the $[\text{CO}_2]_{\text{measured}} - [\text{CO}_2]_{\text{atm}}$.

$$[1] \delta^{13}\text{C}_{\text{emitted}} = \frac{(\delta^{13}\text{C}_{\text{measured}} \times [\text{CO}_2]_{\text{measured}}) - (\delta^{13}\text{C}_{\text{atm}} \times [\text{CO}_2]_{\text{atm}})}{[\text{CO}_2]_{\text{emitted}}}$$

Rhizosphere respiration (R_{rh} =respiration from plant roots + microbes using root derived carbon) was calculated using two previously published methods: root exclusion and ¹³C isotope. One R_{Rh} flux (for both isotope and root methods) was calculated per sampling day per field. The root exclusion method involved taking the average CO₂ flux from a vegetated area minus the average CO₂ flux from a non-vegetated area (Hanson et al 2000, Rochette et al 1999, Rochette and Flanagan 1997) [2]. The ¹³C isotope method is more accurate than the root exclusion method as it takes into account the

$\delta^{13}\text{C}$ of the plant and SOC (Hanson et al 2000, Rochette et al 1999, Rochette and Flanagan 1997) [3].

$$[2] R_{rh, \text{root}} = \text{average CO}_2 \text{ flux}_{(\text{veg})} - \text{average CO}_2 \text{ flux}_{(\text{non-veg})}$$

$$[3] R_{rh, \text{iso}} = \frac{(\delta^{13}\text{C}_{\text{veg, avg}} - \delta^{13}\text{C}_{\text{non-veg, avg}})}{(\delta^{13}\text{C}_{\text{plant, avg}} - \delta^{13}\text{C}_{\text{SOC, avg}})} \times (\text{CO}_2 \text{ veg, avg})$$

The data analysis consisted of both graphical and statistical methods. The graphical method involved calculating a sampling day average per field in order to observe GHG trends throughout the growing seasons. The error bars are presented on the graphs for 1-sigma standard deviation (ie 68% of the data). For the statistical method, a growing season average was calculated per field. 2-sample t-tests were conducted and if the data was not normal then both 2-sample t-test and nonparametric (Mann-Whitney) test were done and confirmed that the same results were obtained from both tests. It is important to note that there is the possibility of type 2 errors in the design. 2-sample t-tests were done to compare whether the means of GHG (CO_2 , CH_4 , and N_2O), $\delta^{13}\text{C}$ of R_T , environmental variables (ST10, SWC20, WTDBS), and R_S & R_{Rh} differed among CTD and UTD fields on a field pair basis. 2-sample t-tests were done to determine whether means of GHG (CO_2 , CH_4 , and N_2O), $\delta^{13}\text{C}$ of R_T , and environmental variables (ST10, SWC20, WTDBS) differed among OT and BT locations within each CTD field. The 1-sigma standard deviation (ie 68% of the data) is presented around the means.

6. Thesis outline

This field study was divided into three parts. Each of these three parts is a separate chapter in this thesis, written as scientific articles for publication. Some sections within the three study chapters are repeated but the point of this is so that each chapter can be independent from the other chapters and, therefore, can be published separately. All necessary information including the specific details pertaining to each part of the study is within its respective chapter. This allows each of the three study chapters to stand on its own.

The thesis is divided into six chapters. Chapter 1 is a literature review and chapter 2 is the introduction and proposed study. Chapters 3, 4, and 5 each represent one part of the study. The first study chapter (chapter 3) is titled, "Soil CO₂, CH₄, and N₂O fluxes over and between tile drains on corn, soybean, and forage fields under tile drainage management." The second part of the study, chapter 4, is titled, "Using ¹³C isotopic analysis to assess soil carbon pools associated with tile drainage management during drier and wetter growing seasons." The final study chapter, chapter 5, is titled "Using ¹³C and root exclusion methods to separate total soil respiration into its rhizosphere and soil components in order to identify whether control tile drainage management affects rhizosphere or soil respiration in corn and soybean fields." Chapter 6 is the synthesis of the thesis and the suggestions for future research extending from the current study. I hope you enjoy learning all the findings we found from this multi-part field study.

7. Tables

Table 1 Planting, harvest, photosynthetic pathway, and key tile drainage management dates for the studied control tile drained (CTD) and uncontrolled tile drained (UTD) agricultural fields.

Field pair	CTD field: area (ha)	UTD field: area (ha)	Year	Planting date: crop type	Date stop gates were closed for CTD fields	Date stop gates were opened for CTD fields	Harvest date
1 & 2	Field 2: 2.3	Field 1: 2.0	2012	May 16: soybean	May 16	September 19	September 24
			2013	May 5: corn	May 29 ^a	September 26	November 14
11 & 14	Field 11: 4.2	Field 14: 4.1	2012	In 2011: forage	May 23	November 19	June 1, July 10, August 20, October 24
			2013	May 15: soybean	May 27	September 26	October 25
12 & 13	Field 12: 5.0	Field 13: 4.2	2013	May 15: soybean	May 27	September 26	October 25

^a Gates were left open for a longer time because farmer wanted them open due to excessive rainfall

Table 2 The number of GHG chambers (in vegetated & non-vegetated areas, and over tile & between tile locations) and the approximate number of GHG samples collected over the two year study.

Year	Field pair	Number of GHG chambers per field	Total number of times sampling chambers	Total number of GHG samples collected	# of OT chambers (V & NV)/field	# of BT chambers (V & NV)/field
2012	1 & 2	8	20	1600	4 (2V & 2NV)	4 (2V & 2NV)
	11 & 14	4	17	680	2 (1V & 1NV)	2 (1V & 1NV)
2013	1 & 2	12	15	1800	6 (4V & 2NV)	6 (4V & 2NV)
	11 & 14	12	16	1920	6 (4V & 2NV)	6 (4V & 2NV)
	12 & 13	12	10	1200	6 (4V & 2NV)	6 (4V & 2NV)
TOTAL	5	NA	78	7200	NA	NA

8. Figures



Figure 1 The field site indicated by a red star in Eastern Ontario.



Forage














Corn



Soybean

Figure 2 Photographs from three of the studied fields indicating the three crop types: forage, corn, and soybean. In the forage field, chambers deployed in a vegetated and non-vegetated area are shown. The images were taken by Corey Flemming, an MSc. student on the project.

LEGEND

-  GHG monitoring site (2013)
-  GHG monitoring site (2012 - 2013)
-  GHG monitoring site (2012)
-  Soil sampling site
-  Drainage tiles
-  Drainage ditch
-  CTD outlet
-  UTD outlet
-  CTD field
-  UTD field
-  Water table monitoring site
- OT** Over drainage tile
- BT** Between drainage tile

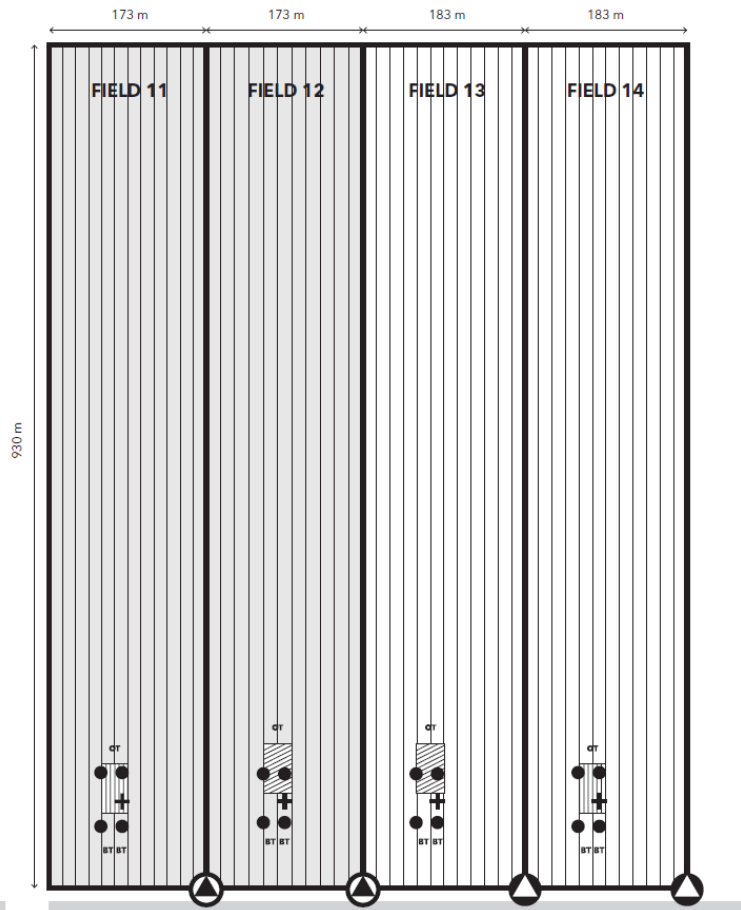
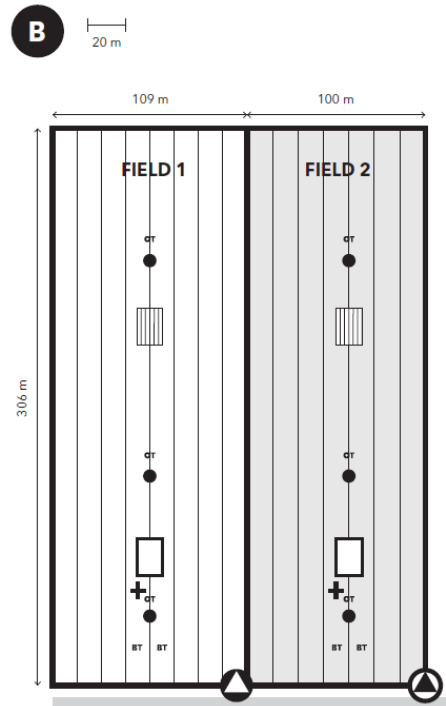


Figure 3 The location of the experimental watersheds in Eastern Ontario, Canada, (a) containing the experimental fields 1-2 (b) and 11-14 (c). The location of GHG sampling, soil sampling, and water table monitoring sites are indicated. Over drainage tile and between drainage tile locations are indicated.

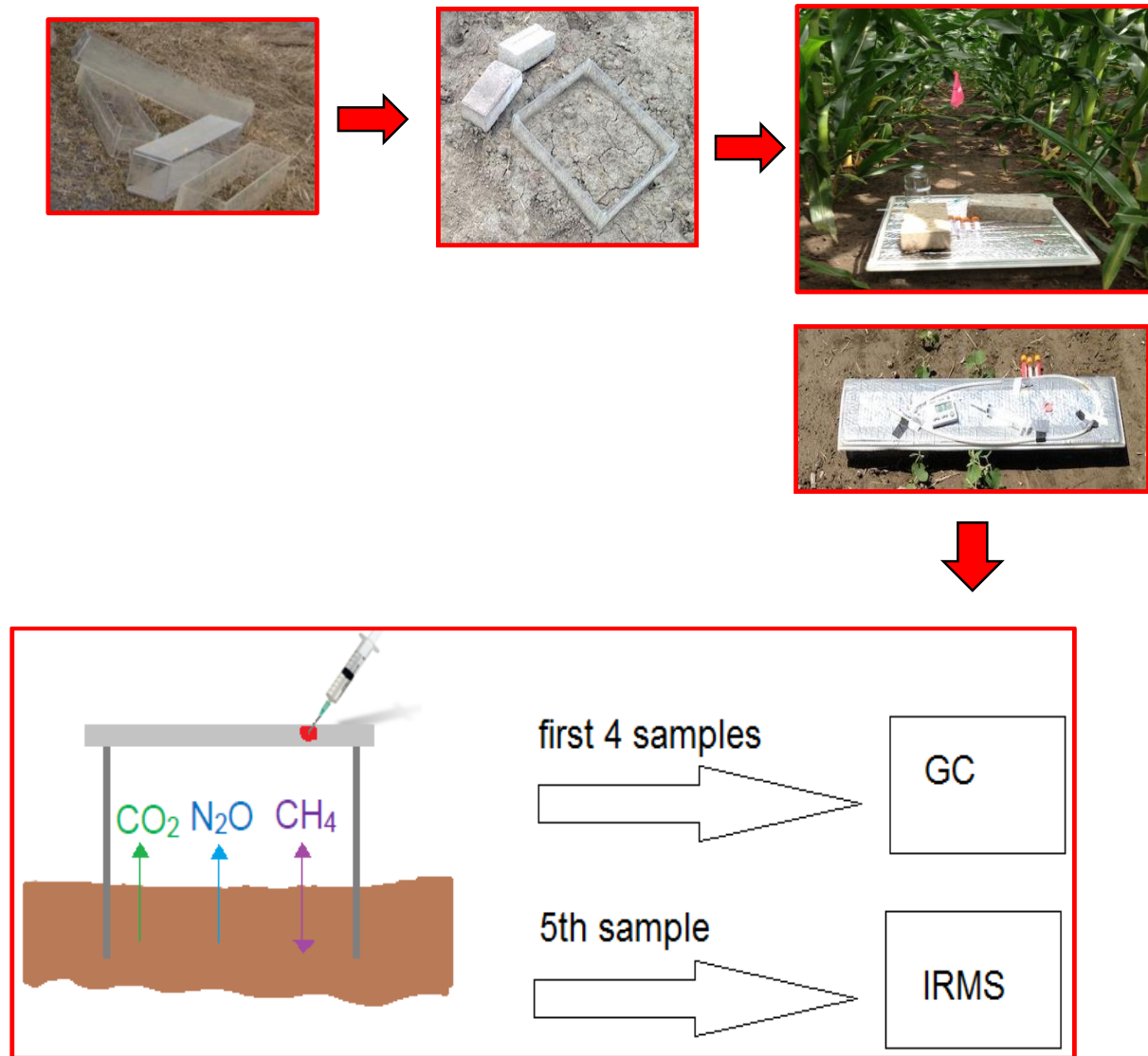


Figure 4 The procedure for soil GHG sampling.

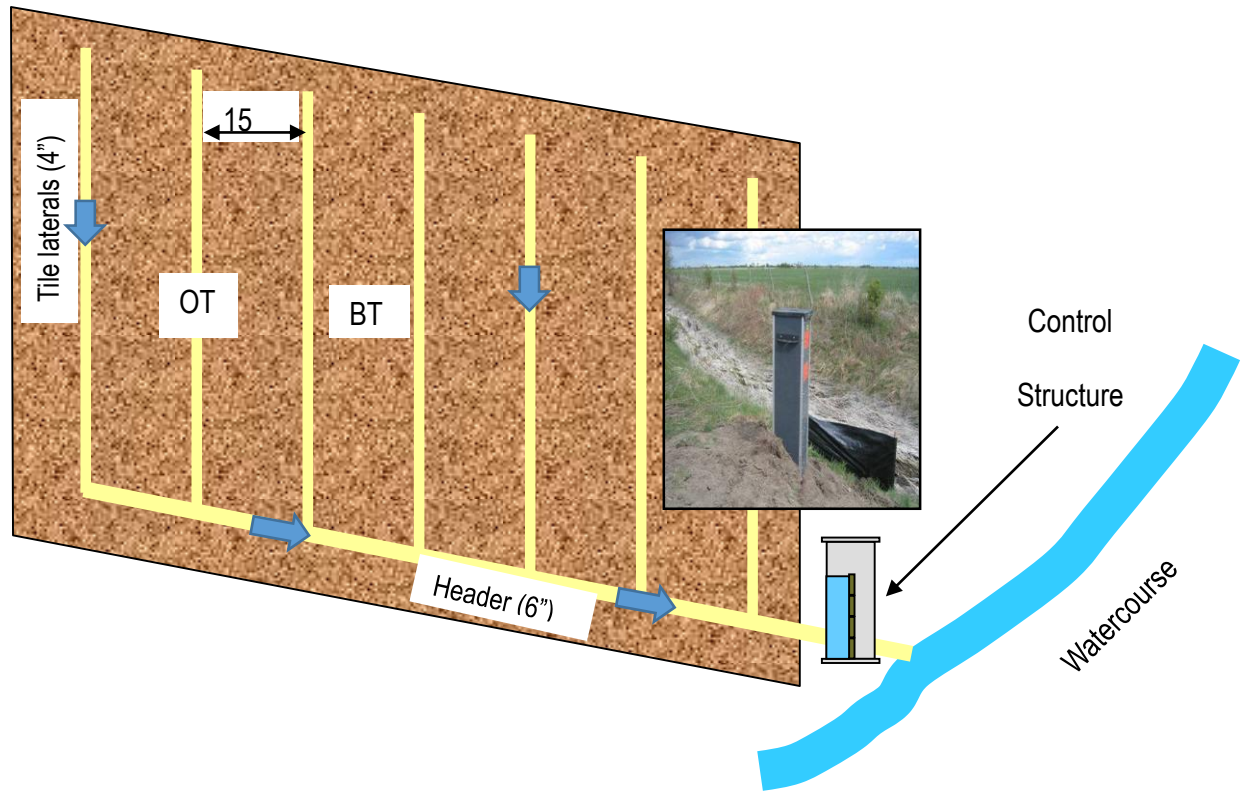


Figure 5 A diagram of a control tile drained field indicating the placement of drainage tiles, over tile (OT) and between tile (BT) locations, and the control structure located off of the field, which is connected to a watercourse. OT locations are directly over a tile lateral whereas BT locations are between a tile lateral (*ie* 7.5 m away from the tile lateral). Tile laterals are 4" in diameter whereas header tiles are 6" in diameter. The blue arrows indicate the direction of water flow. A photograph of a control structure is presented. The diagram was a collaboration from Dr. David Lapen (AAFC) and myself.

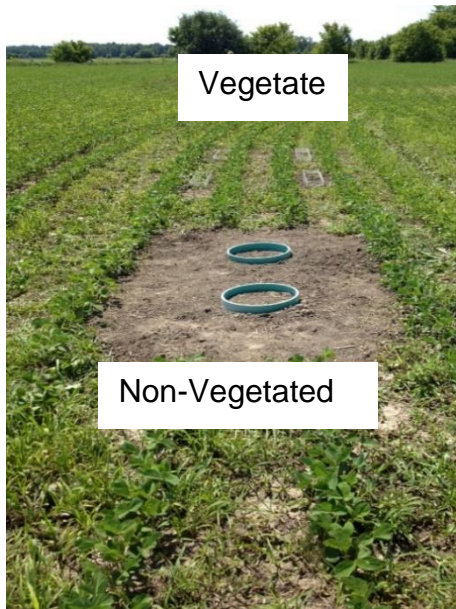


Figure 6 A photograph from the 2013 soybean field indicating chambers in a vegetated and non-vegetated location. The image was taken by Corey Flemming, an MSc. student on the project.

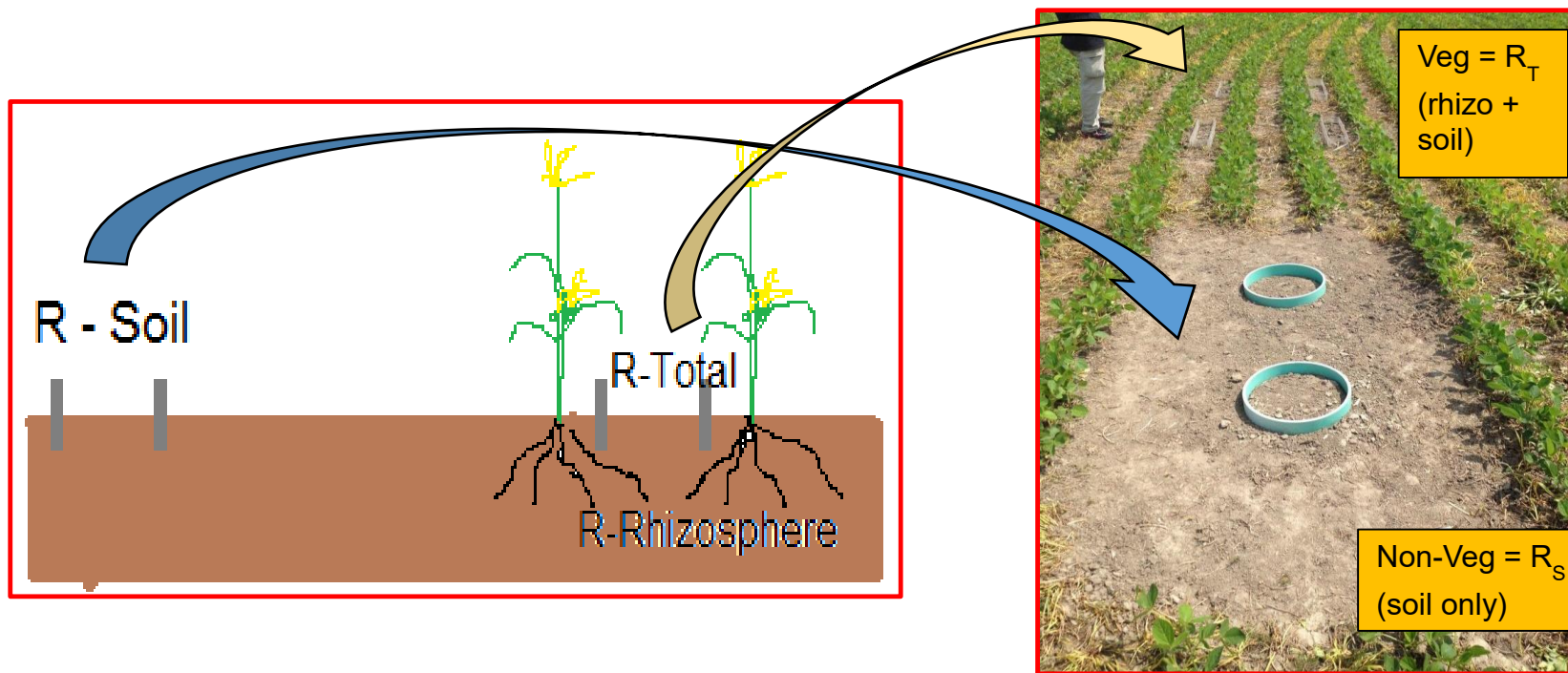


Figure 7 Greenhouse gas chambers in a vegetated area are provide a measure of total soil respiration (R_T =rhizosphere + soil contribution) whereas chambers in a non-vegetated area represent soil respiration only (R_S).

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Chapter 3 Soil CO₂, CH₄, and N₂O fluxes over and between tile drains on
corn, soybean, and forage fields under tile drainage management

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Chapter 3: Table of Contents

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Abstract

Controlled tile drainage (CTD) is an agricultural management practice that regulates the amount of water leaving a field through tile drains thus retaining water and nutrients in the field for crop use. CTD can benefit the environment and crops through improved water quality and increased yields respectively, although CTD has the potential to increase greenhouse gas (GHG: CO₂, CH₄, N₂O) emissions. However, the effects of CTD on GHG emissions, including their spatial distribution are not well-documented. This study compares chamber measurements of soil GHG emissions for silt loam soil throughout a dry (2012) and a wet (2013) growing season for corn, soybean, and forage fields under CTD and uncontrolled tile drainage (UTD) in eastern Ontario, Canada. Furthermore, this study evaluates the over tile (OT) and between tile (BT) spatial distribution of soil GHG emissions from CTD fields. CO₂ and CH₄ fluxes did not significantly differ between CTD and UTD fields ($P > 0.05$), while N₂O fluxes were also not significantly different for all field pairs studied ($P > 0.05$) except for one. Average growing season CO₂ fluxes ranged between 20.11 (Field 2, 2013) and 101.0 (Field 14, 2012) Kg C ha⁻¹ day⁻¹. Average growing season CH₄ fluxes indicated the fields were a sink for atmospheric CH₄. Average growing season N₂O fluxes ranged between 0.00255 (Field 11, 2012) and 0.0309 (Field 2, 2013) Kg N ha⁻¹ day⁻¹. Throughout the growing seasons, GHG emissions did not significantly differ between OT and BT locations within CTD fields ($P > 0.05$). There was no significant difference in mean GHG emissions from vegetated locations in soybean fields in 2012 and 2013 ($P > 0.05$). The results of this study indicate no adverse GHG emissions associated with CTD over the time span and field environments of the study, thus further supporting the use of CTD in agriculture.

1. Introduction

Carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O) are three important greenhouse gasses (GHG) as their increasing atmospheric concentrations over the last several centuries has been considered to have contributed to climate change (IPCC 2013). From 1750 to 2011, CO₂ increased by 40% from 278 ppm to 390.5 ppm, whereas CH₄ increased by 150% from 722 ppb to 1803 ppb, and N₂O increased by 20% from 271 ppb to 324.2 ppb (IPCC 2013). This increase of GHG is caused by anthropogenic emissions from using fossil fuels and from land use and land use changes, especially agriculture (IPCC 2013). Agricultural lands cover an estimated 40 to 50 % of the Earth's land surface (Smith *et al* 2007), and they release significant amounts of GHG to the atmosphere (Cole *et al* 1997). In 2005, agriculture was responsible for 10 to 12 % of the total global anthropogenic GHG emissions and specifically responsible for approximately 60 % and 50 %, respectively, of the total global anthropogenic emissions of N₂O and CH₄ (Smith *et al* 2007).

Agricultural soils have an important contribution in regulating atmospheric GHG concentrations because they are a source and sink of CO₂ and N₂O (Ball *et al* 1999, Nangia *et al* 2013, Hernandez-Ramirez *et al* 2009), and either a sink or a source for CH₄ (Nangia *et al* 2013, Hernandez-Ramirez *et al* 2009). Terrestrial vegetation and soils represent the second largest reservoir of carbon (C), with inputs from photosynthesis and losses through respiration (Clark 2015, Drewitt *et al* 2009, Hardy 2003). In vegetated areas, soil CO₂ efflux (R_T) originates from plants through rhizosphere respiration, which is root and rhizomicrobial respiration using root derived C, and from soil through microbial respiration using soil organic matter (SOM) (Cheng 1996, Ryan &

Law 2005, Rochette *et al* 1999). N₂O soil emissions are mainly produced by microbial nitrification (the oxidation of ammonium (NH₄⁺) to nitrate (NO₃⁻)) in aerobic conditions and denitrification (the reduction of nitrate to dinitrogen gas (N₂)) in anaerobic conditions (Robertson & Grace 2004, Dobbie & Smith 2006). Nitrification and denitrification can both occur in well-drained soils because wet soils can be found inside soil aggregates (Robertson & Grace 2004). Soils can be sources or sinks for CH₄ depending on the activity of methanogens and methanotrophs (Topp & Pattey 1997). Under anaerobic conditions and in the absence of nitrate, sulfate, or ferric iron, methanogens reduce more oxidized forms of C into CH₄ thus making the soil a CH₄ source (Topp & Pattey 1997). Under aerobic conditions, soils are a CH₄ sink because methanotrophs oxidize CH₄ to CO₂ or incorporate it into the microbial biomass (Powlson *et al* 1997). In some soils, both CH₄ production and CH₄ consumption can occur (Powlson *et al* 1997).

Artificial subsurface tile drainage is found in many agricultural areas in order to improve field drainage for crop production (Sunohara *et al* 2015, Jaynes & Isenhardt 2014, Skaggs *et al* 1994). For instance, in the midwestern United States, approximately 17.4 million ha of land is artificially drained (Jaynes & Isenhardt 2014). An estimated 15.3 million ha of the 170 million ha of cropland in the United States was subsurface tile drained (Skaggs *et al* 1994). In Ontario, Canada, more than 1.6 million ha of agricultural land is artificially tile drained (Sunohara *et al* 2015) out of a total of 5.1 million ha of agricultural land (OMAFRA 2011).

Controlled tile drainage (CTD) is a flexible agricultural management practice that manages the water table depth below surface (WTDBS) by controlling the amount of water that can leave a field through artificial subsurface tile drains (Gilliam *et al* 1979).

CTD involves increasing or decreasing the WTDBS by respectively removing or adding stop gates in a water flow control structure, which is connected to a header tile in the field (Sunohara *et al* 2015, Kross *et al* 2015, Cicek *et al* 2010, Mejia *et al* 2000). By controlling the amount of tile discharge, CTD can elevate water tables and in turn reduce water and nutrient losses from fields thus providing crops more access to water and nutrients during critical growth stages (Tan *et al* 1999, Cicek *et al* 2010, Sunohara *et al* 2015). CTD management contrasts with uncontrolled tile drainage (UTD) where there is no management of the water leaving a field through tile drains.

CTD is a beneficial agricultural management practice. CTD removes excess water from the field, which allows machinery operations to occur on the field (Nangia *et al* 2013, Cicek *et al* 2010, Mejia *et al* 2000) and lessens the water-logging damage to crops (Cicek *et al* 2010). If the weather conditions permit, CTD can provide the proper amount of soil moisture and nutrients during critical growth stages of plants (Cicek *et al* 2010, Mejia *et al* 2000) thus allowing CTD to increase crop yields (e.g. Delbecq *et al* 2012, Ghane *et al* 2012, Poole *et al* 2013, Skaggs *et al* 2012, Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Kross *et al* 2015, Mejia *et al* 2000, Cicek *et al* 2010) and reduce the amount of nutrient loss in drainage water (e.g. Sunohara *et al* 2015, Cooke & Verma 2012, Drury *et al* 2009, Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Drury *et al* 1996).

Although the links between CTD and increased crop yields and reduced adverse nutrient inputs in nearby waterways are well documented (e.g. Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002), these benefits could result in an environmental trade-off of increased GHG emissions. CTD modifies the WTDBS and as

the WTDBS gets shallower, the water filled pore space (WFPS) in the topsoil increases resulting in increased soil moisture (Dobbie & Smith 2006). With the possible increased soil moisture of CTD fields, there is the potential for the development of soil conditions which may increase GHG emissions (Nangia *et al* 2013, Elmi *et al* 2000). Increasing soil moisture can result in greater R_T (Davidson *et al* 2000) as studies indicate R_T was greatest at relatively high moisture levels and reduced under very dry or very wet soil conditions (Orchard & Cook 1983, Bowden *et al* 1998, Linn & Doran 1984, Skopp *et al* 1990). CH_4 flux may be controlled by soil moisture (Keller & Reiners 1994), where soil consumption of atmospheric CH_4 is greatest in dry soils and CH_4 production is greater with increased soil moisture (Keller & Reiners 1994, Sass *et al* 1992). CH_4 uptake is controlled by soil moisture, where decreased CH_4 uptake occurs at very low and very high soil moisture (Bowden *et al* 1998), and maximum rates of CH_4 uptake occurs at approximately 50-70% of WFPS (Nesbit & Breitenbeck 1992, Bowden *et al* 1998). Increasing soil moisture can promote denitrification resulting in greater soil N_2O efflux rates (Knowles 1982) as supported by several studies which found N_2O emissions increased as the WTDBS got shallower (Dobbie & Smith 2006, Kliewer & Gilliam 1995, Jacinthe *et al* 2000, Elmi *et al* 2000), combined N_2O and N_2 emissions increased in a shallower water table (Elmi *et al* 2005), and N_2O emissions increased as the WFPS increased (Dobbie & Smith 2006, Dobbie & Smith 2001, Dobbie *et al* 1999, Dobbie & Smith 2003, Rabot *et al* 2015, Keller & Reiners 1994).

Within the literature on GHG emissions in CTD agricultural fields, knowledge gaps exist on GHG fluxes and their associated spatial distribution. Only a few studies have quantified the relationship between water table depth and N_2O emissions (Elmi *et*

et al 2005, Dobbie & Smith 2006, Nangia *et al* 2013), and only one study has compared and contrasted CO₂, CH₄, and N₂O soil emissions from CTD and UTD fields (Nangia *et al* 2013), but no reports were found on the effects of CTD management with respect to spatial distribution of GHG emissions. Knowledge of GHG emissions in CTD agricultural fields is essential for properly assessing ecological impacts associated with CTD, for improving estimates in GHG inventories, and to ultimately adopt best management practices to reduce GHG emissions from the agriculture sector. Nangia *et al* 2013 compared GHG emissions from CTD and UTD fields and found no significant difference between N₂O and CH₄ soil effluxes but there were some significantly higher CO₂ fluxes associated with CTD relative to UTD fields. More studies are needed to confirm the effect of CTD on soil GHG emissions. In addition, while the spatial and temporal variability of GHG emissions in non-tile drained agricultural fields has been well documented (e.g. Rochette *et al* 1991, Hatfield & Parkin 2012, Stoyan *et al* 2000, Jiang *et al* 2010, Hao & Jiang 2014, Yanai *et al* 2003), the spatial distribution of GHG emissions within CTD managed fields is not well known. For parallel drainage tiles in the field, the water table gradient towards a drainage tile results in drawdown causing the WTDBS for over tile (OT) to be deeper whereas between tile (BT) locations would have shallower WTDBS (USDA 2011, Skaggs *et al* 2005). The difference in WTDBS for OT versus BT locations in CTD field is expected to cause soil GHG emissions to vary spatially within the field. To my knowledge, Nangia *et al* 2013 examined spatial variation of GHG emissions in CTD fields but only for one season were measurements made both BT and OT and therefore more studies are needed. As such, the proposed study

will complete the knowledge gap by providing information on soil GHG emissions from CTD fields and their spatial distribution for OT and BT locations.

The specific objectives of this paper are to: (1) compare and contrast the soil CO₂, CH₄, and N₂O flux rate in two growing seasons with contrasting amounts of precipitation from two drainage management systems (CTD and UTD) containing soybean, corn and forage; and (2) analyze the spatial (OT and BT) distribution flux rate of soil CO₂, CH₄, and N₂O emissions in CTD agricultural fields.

2. Materials and methods

2.1 Study area and drainage management

This two year study (2012 – 2013) was conducted during the growing seasons (defined here as beginning when the stop gates were closed in CTD fields and lasting until they were reopened; **Table M1**) on subsurface tile drained agricultural fields containing corn, soybean, and forage (timothy & alfalfa). For this study, comparison of GHG soil fluxes from CTD and UTD fields was done for the growing season because it is during this time that drainage management varies for CTD and UTD fields. This study examined two pairs of fields in 2012 and three in 2013 (**Fig. M1**) with each pairing having fields in close proximity to each other and under a common cropping practice but varying in drainage management (one field was CTD and the other UTD) (**Table M1**). The standard deviation is always presented in conjunction with the mean.

The fields studied are low-gradient (slope < 1%), located within the ~950 ha South Nation Watershed (45.26 N, 75.18 W) in eastern Ontario, Canada (Cicek *et al* 2010). Dominant soils are Bainsville silt loams (Wicklund & Richards 1962) with clayey

soils underneath at ~1.0 - 1.5 m (Cicek *et al* 2010). According to nearby Russell station, the 30-year normal annual precipitation (1981-2010) is 981 mm and average daily air temperature is 6.5°C (EC 2015). From June to September (approximate growing season), average 30-year precipitation (1981-2010) and average daily air temperature, according to Russel station, is 369 mm and 18°C respectively (EC 2015). Total growing season rainfall was 281 mm and 460 mm in 2012 and 2013 respectively and average soil temperature (0-10 cm depth) (ST10) during the growing season was 17.8°C ± 2.6 and 17.4°C ± 2.1 in 2012 and 2013 respectively.

All fields studied are tile drained with lateral tiles (0.102 m in diameter) located at a depth of ~1 m and with a spacing of ~15 m. The lateral tile drains connect to a header tile (0.152 m in diameter) which connects to a main outlet containing a tile drainage control structure for each field. In CTD fields, tile drainage control structures were used to manage tile drain flow by setting the water overflow depth in the control structures to 0.4 m below the surface. In UTD fields, the control structures were not used to manage tile drain flow.

2.2 Measurements of CO₂, CH₄, and N₂O soil-atmosphere flux

All soil GHG measurement sites were done in vegetated areas with sampling chambers either containing plants (forage sites) or with roots and crops growing next to chambers (corn and soybean sites). During 2012, GHG measurements for field pair 1 & 2 were made in both the upper and lower sections of each field with a BT and OT site in each section so that each field had a total of 4 GHG measurement sites. During 2012, GHG measurements for field pair 11 & 14 were made in the middle sections of the fields with one site OT and one site BT so that each field had a total of 2 GHG measurement

sites. During 2013, GHG measurements for field pairs (11-14, 12-13, and 1-2) were made in the middle section of the fields with 4 sites OT and 4 sites BT so that each field had a total of 8 GHG measurement sites.

Measurements of CO₂, CH₄, and N₂O GHG fluxes from soil to the atmosphere were carried out at weekly intervals from OT and BT locations on both fields of each field pair. All GHG measurements were sampled in between crop rows except for forage fields where forage covered the entire field. GHG sampling most frequently occurred during 1000-1400 h (peak emission times). Between 2012 and 2013, a total of 632, 629, and 632 CO₂, CH₄, and N₂O soil-atmosphere flux calculations were done from an approximate total of 2528, 2516, and 2528 collected air samples, respectively.

GHG sampling was carried out using the nonsteady-state chamber design and method described by Rochette and Bertrand 2008, which was also similarly used by Nangia *et al* 2013. Briefly, chamber bases were installed in the fields at ~10 cm depth (and ~5 cm head space) and left there for the growing season. Rectangular chambers (0.75 x 0.15 m) were used in soybean fields, and square chambers (0.35 x 0.35 m) were used in corn fields, and both types of chambers had flat lids. Forage fields had circular chambers with a 1.28 m circumference, and a lid of height 0.323 m to accommodate growing plants. Plastic chamber lids contained a rubber septa sampling port on the top and a foam gasket seal on their underside. Headspace volume and surface area of chambers were measured monthly.

GHG sampling involved capping the chamber, and using a 30 mL syringe (fitted with a 26-gauge needle) to immediately collect a 30 mL sample of the chamber

headspace at time zero. During chamber deployment, four samples were collected from each chamber's headspace and were analyzed for GHG concentration. GHG from soybean and corn fields were sampled at 0, 6, 12, and 18 minutes while forage was sampled at 0, 12, 24, and 36 minutes. The sampled air was injected into 12 mL septum-capped glass vials (Extainers, Labco Ltd.) which were prepared prior to the field by: adding an additional silicone septum on top of the rubber septum, adding 2-3 mg of magnesium perchlorate as a drying agent, and evacuating the vials (Rochette & Bertrand 2008). The samples were analyzed for CO₂, CH₄, and N₂O concentrations using a Varian CP-3800 gas chromatograph with CombiPal autosampler with helium as the carrier gas. The injection volume was 5 mL and a flame ionization detector measured CH₄ and CO₂ whereas an electron capture detector measured N₂O. Samples of reference standard gases were handled the same way as gas samples and approximately one was inserted for every five gas samples during the analysis.

CO₂, CH₄, and N₂O soil-atmosphere fluxes were calculated by performing linear regression with each series of GHG concentration versus time data from each deployed chamber. The CO₂ and CH₄ flux is calculated using [1] and the N₂O flux using [2] where: m is the linear slope of ppmv/min, P is the mean atmospheric pressure in atm during GHG sampling, R is the ideal gas constant, T is the in-situ chamber air temperature in Kelvin, V is the chamber headspace volume in L, and A is the chamber surface area in m².

$$[1] \text{ C Flux } (\mu\text{g}/\text{m}^2/\text{hr}) = m \cdot P / (R \times T) \cdot (60 \text{ min}/\text{hr}) \cdot V / A \cdot (12.011 \mu\text{g C} / \mu\text{Mol})$$

$$[2] \text{ N Flux } (\mu\text{g}/\text{m}^2/\text{hr}) = m \cdot P / (R \times T) \cdot (60 \text{ min}/\text{hr}) \cdot V / A \cdot (28.02 \mu\text{g N} / \mu\text{Mol})$$

2.3 Measurement of yields and environmental variables

Annual yield (kg/ha) for each field was calculated by plant mass * plant density, where the plant mass is taken from 5 randomly selected plants per site.

The WTDBS was measured at 15 minute intervals throughout the field season from pressure transducer water level sensors which were inserted in groundwater wells located between tiles in each field.

Soil volumetric water content (0-20 cm depth) (SWC20), soil temperature (0-10 cm depth) (ST10), and air temperature were measured within 0.5 m of each GHG sampling chamber at time of sampling (1000-1400 h). SWC20 (*ie* water volume/total volume) of the soil was measured using time-domain reflectometry (TDR) probes. ST10 and air temperature were measured using a digital thermocouple thermometer and temperature probe.

Two weather stations (one in Field 14 and the other near Field 1) measured precipitation (mm), soil temperature (0-15 cm depth) (ST15) (°C), air temperature (°C), relative humidity (RH, %), and solar radiation (SR, W/m²) every 30 minutes.

2.4 Statistical analysis

In order to compare CTD and UTD fields, GHG fluxes and environmental variables (ST10, SWC20, and WTDBS) from CTD and UTD fields were subjected to t-tests (Minitab 16). In order to compare spatial distribution of GHG within CTD fields, t-tests (Minitab 16) were done to compare OT and BT emissions. If the sample size was greater than 30 then, according to the central limit theorem, the data was considered

normal and no data transformation occurred. If $n < 30$, normality was verified graphically or by using the Anderson-Darling normality test and if normality wasn't confirmed the data was log transformed. If the log-transformed data was still not normal then non parametric Mann-Whitney tests were done in place of t-tests.

For each field, Spearman rank correlations (Minitab 16) were done among the individual CO₂ fluxes and environmental variables: SWC20 and ST10 taken next to each GHG chamber at time of GHG sampling, average daily WTDBS for GHG sampling day, sum of rainfall from 1000-1400 h on GHG sampling day, total rainfall day prior to GHG sampling, and average SR, RH, and air temperature between 1000-1400 h on GHG sampling day.

In order to compare GHG emissions from vegetated locations in soybean fields in 2012 and 2013, the fluxes were subjected to t-tests or Mann-Whitney tests when the data was not normal (Minitab 16).

3. Results and discussion

3.1 Weather conditions and crop yields

The 2012 growing season received less precipitation but had a similar mean ST15 compared to the 2013 growing season (**Fig. 3.1a**). Total rainfall during the 2013 growing season was 460 mm, which is 64% more than the 281 mm of rain in 2012 (**Fig. 3.1b**). There was 24% less and 25% greater precipitation in 2012 and 2013, respectively, compared to the average 30-year rainfall (1981-2010) for the approximate growing season (June-September = 369 mm) from nearby Russell station (EC 2015). The average ST15 during the growing season, from the weather station near Field 1,

was similar in 2012 ($17.8\text{ }^{\circ}\text{C} \pm 2.6$) and 2013 ($17.4\text{ }^{\circ}\text{C} \pm 2.1$) and was also similar to the 30-year (1981-2010) daily average air temperature for the approximate growing season (June-September = 18.4°C) from nearby Russell station (EC 2015).

CTD fields had similar yields to UTD yields except for a few exceptions (**Table 3.1a**). In 2012, CTD Field 11 had a greater yield for OT and BT compared to UTD Field 14. In 2012, UTD Field 1 had a larger OT yield compared to CTD Field 2. In 2013, UTD Field 14 for BT locations had a larger yield than CTD Field 11. In the literature, CTD has mostly been beneficial by increasing crop yields (Delbecq *et al* 2012, Ghane *et al* 2012, Poole *et al* 2013, Skaggs *et al* 2012, Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Kross *et al* 2015, Mejia *et al* 2000), but some studies agree with the findings of this study in that there is no significant effect of CTD on yields (Cooke & Verma 2012, Drury *et al* 2009), while some other studies show that for some sites UTD resulted in a greater yield than CTD fields (Ghane *et al* 2012, Drury *et al* 2009). For the current study, there were no consistent patterns between CTD and yields, where in some instances yields increased on CTD fields, others were similar, and in some cases yields decreased on CTD fields, which was also seen in Cooke & Verma 2012. A study found that at the field scale, some observed yields from CTD and UTD fields were not significantly different from each other ($p>0.05$), but when evaluating crop responsiveness to CTD at a watershed scale over 4 years, CTD observed corn and soybean yields were 3 and 4 % greater, respectively (Cicek *et al* 2010). Therefore, it is not surprising in this study that there were similar yields in CTD and UTD fields with some exceptions.

3.2 Environmental variables in CTD vs UTD fields

3.2.1 Water table depths

There was a significant difference ($p < 0.05$) between average WTDBS between all CTD and UTD field pairs from data measured continuously every 15 minutes throughout the 2012 and 2013 growing seasons (**Table 3.2a**). In the 2013 growing season, CTD fields 2, 11, and 12 had a smaller average WTDBS (0.966 ± 0.406 , 1.070 ± 0.446 , 1.266 ± 0.415 respectively) compared to their UTD field pairs (1.115 ± 0.301 , 1.277 ± 0.380 , 1.413 ± 0.318) (**Fig. 3.2a**). In 2013, the daily average WTDBS, for the days that GHG's were sampled on, indicates a significant difference ($p < 0.05$) between CTD and UTD fields where CTD fields have a shallower WTDBS (**Table 3.2b**). In 2012, there was no significant difference ($p > 0.05$) between daily average WTDBS for GHG sampling days between CTD and UTD fields (**Table 3.2b**). In the dry 2012 growing season, the WTDBS only reached 1 m and shallower in late May to early June for field pair 1 and 2, and did not reach this level in Field 11 and 14 (**Fig. 3.2b**). In 2012, when the WTDBS was ≤ 1 m the CTD Field 2 generally had a shallower WTDBS than UTD Field 1. The CTD fields spent a greater percentage of time during the growing season with WTDBS equal to and shallower than 1 m compared to UTD fields (**Fig. 3.2c**). Field 11 and 14 in 2012 had 0 % of the growing season with a WTDBS equal to or shallower than 1 m. Therefore, the findings that CTD fields had significantly shallower WTDBS in the wet growing season, and they spent more time with WTDBS equal to and shallower than 1 m compared to UTD fields were as expected.

3.2.2 Soil moisture

SWC20 measured next to GHG chambers during GHG sampling is compared for CTD and UTD fields (**Table 3.2b**). There was no significant difference ($p>0.05$) between SWC20 in all CTD and UTD fields for both years of study. It was expected that a shallower WTDBS in CTD fields would result in a greater SWC20 compared to UTD fields, but this was not the case. It is hypothesized that there could have been greater root water uptake (RWU) by plants in CTD fields causing the measured SWC20 to have no significant difference in CTD and UTD fields. This suggestion is supported by findings in the literature that indicate RWU is sensitive to fluctuations in the WTDBS (Li *et al* 2015, Askri *et al* 2014, De Silva *et al* 2008). For example, Askri *et al* 2014 indicate that decreasing the WTDBS during summer increases the RWU in date palms (a fruit tree). Simulation results indicated that decreasing the WTDBS along with higher temperatures, which increased potential evapotranspiration, caused 40% greater RWU in the wet season compared to the dry season (De Silva *et al* 2008). RWU in Chinese tamarisk was sensitive to WTDBS, where RWU increased when the WTDBS decreased up until the WTDBS reached a minimum value (Li *et al* 2015). During GHG sampling in 2013, SWC20 was approximately 10% greater than in 2012, which was expected due to the greater amount of precipitation in 2013.

3.2.3 Soil temperature

ST10 measured next to GHG chambers, during the same day and time that GHG were sampled, is compared for CTD and UTD fields (**Table 3.2b**). There was no significant difference ($p>0.05$) between ST10 in all CTD and UTD fields for both years of study. The findings for no difference in ST10 between CTD and UTD fields are as

expected since we wouldn't expect a difference in ST10 as there was no significant difference ($p>0.05$) between SWC20 in all CTD and UTD fields. ST10 during GHG sampling was similar in 2012 and 2013, and thus the study can focus on how soil moisture affects GHG emissions over the two years as soil temperature was similar.

3.3 CO₂, CH₄, and N₂O soil fluxes in CTD vs UTD fields

Average growing season CO₂ fluxes ranged between 20.11 (Field 2, 2013) and 101.0 (Field 14, 2012) Kg C ha⁻¹ day⁻¹ in vegetated areas (**Table 3.3a**). Throughout the growing seasons, CO₂ fluxes did not differ between CTD and UTD fields (**Fig. 3.3a**), which is supported from t-tests indicating no significant differences (**Table 3.3a**). In 2013, CO₂ fluxes tended to be greatest from mid-June to mid-July. In 2012 forage, CO₂ fluxes gradually increased until cuttings and then were much lower after cuttings.

Average growing season CH₄ fluxes ranged between -0.00098 (Field 14, 2013) and -0.00040 (Field 14, 2012) Kg C ha⁻¹ day⁻¹, in vegetated areas, indicating the fields were a sink for CH₄ (**Table 3.3a**). According to t-tests, there were no significant differences among CH₄ fluxes between CTD and UTD fields (**Table 3.3a**).

Average growing season N₂O fluxes ranged between 0.00255 (Field 11, 2012) and 0.0309 (Field 2, 2013) Kg N ha⁻¹ day⁻¹ in vegetated areas (**Table 3.3a**). According to t-tests, there were no significant differences among N₂O fluxes between CTD and UTD fields except for the N₂O flux from CTD Field 12 and UTD Field 13 in 2013 (**Table 3.3a**).

There is believed to be no significant impact of CTD on GHG emissions because, eventhough WTDBS was smaller in CTD fields, SWC20 and ST10 were not significantly

different between CTD and UTD fields and, therefore, there was no driving factor for difference in soil GHG emissions. Furthermore, there was no significant difference in yields between CTD and UTD fields thus further supporting why there would be no difference in GHG emissions among differences in drainage management.

3.4 Spatial distribution of CO₂, CH₄, and N₂O soil fluxes in CTD fields

Throughout the growing seasons, CO₂ fluxes did not differ for OT and BT locations within CTD fields (**Fig. 3.4a**), which is supported from t-tests indicating no significant differences (**Table 3.4a**). N₂O and CH₄ emissions also did not significantly vary for OT and BT emissions in all CTD fields studied ($p > 0.05$) (**Table 3.4a**).

It was surprising that no spatial variation of GHG emissions existed in CTD fields. It is known in the literature that the water table gradient towards a drainage tile results in drawdown causing the WTDBS for OT to be deeper whereas BT locations would have a shallower WTDBS (USDA 2011, Skaggs *et al* 2005). While the WTDBS was only measured for BT locations within CTD fields, it was predicted that the difference in the expected WTDBS for OT versus BT locations in CTD fields would cause the GHG emissions to vary spatially within the fields but this was not observed. It is believed that there was no significant difference in OT and BT spatial distribution of GHG emissions in CTD fields because there was no significant difference ($p > 0.05$) between OT and BT locations for ST10 and SWC20 in all CTD fields during GHG sampling days (**Table 3.4b**). ST10 and SWC20 for OT and BT locations within CTD fields were similar and this is believed to be why the GHG emissions from OT and BT were also similar.

3.5 Soil CO₂ fluxes and environmental variables

For each field, Spearman Rank correlations were examined among the individual soil CO₂ fluxes and environmental variables (**Table 3.5a**). The environmental variables significantly potentially influenced by drainage management practice (SWC20, ST10, and WTDBS) were found to be more influenced in CTD fields with 80, 100, and 60 % of fields being significantly correlated compared to 60, 60, and 40 % in UTD fields. ST10 had a strong positive correlation (0.5 and greater) in all but two observed fields (field 1 and 14 in 2012). SWC20 was weakly to moderately negatively correlated (-0.509 to -0.340) in all fields except fields 1, 2, and 14 in 2012. In 2013 SWC20 and ST10 was significantly correlated with CO₂ flux in all studied fields. RH and air temperature, for almost all fields, were positively correlated with soil CO₂ emissions.

As there was no significant difference in GHG emissions between CTD and UTD fields, and spatially within CTD fields, spearman rank correlations were done to examine what was driving the CO₂ emissions (*ie* the GHG with the largest GHG flux). The main drivers for soil CO₂ emissions appear to be RH, air temperature, and ST10, which are significantly positively correlated with CO₂ in many fields, and SWC20 which is significantly negatively correlated with CO₂ in many fields. Thus, it is not surprising that there was no significant difference in GHG emissions between CTD and UTD fields, and spatially within CTD fields, because there was no significant difference in the driving factors between CTD and UTD fields and between OT and BT locations within CTD fields.

3.6 Soil GHG fluxes in wet (2013) vs dry (2012) growing seasons

Because of the greater total rainfall in the 2013 growing season (460 mm) compared to the 2012 growing season (281 mm), it was expected that GHG emissions from vegetated locations in soybean fields in 2012 (fields 1 & 2) would vary from the emissions in 2013 (fields 11-14). Mean soil CO₂ emission during the 2013 growing season from vegetated locations in soybean fields was greater (28.81 ± 8.00 kg C/ha/day) than in 2012 (24.08 ± 9.23 kg C/ha/day), although it was not a significant difference (t-test; $p > 0.05$) (**Fig 3.6a**). There was no significant difference in mean soil N₂O emission from vegetated locations in soybean fields in 2012 (0.0114 ± 0.0228 kg N/ha/day) and 2013 (0.00657 ± 0.00443 kg N/ha/day) (t-test; $p > 0.05$) (**Fig 3.6a**). Similarly, there was no significant difference in mean CH₄ emission from vegetated locations in soybean fields in 2012 (-0.000582 ± 0.000963 kg C/ha/day) and 2013 (-0.000598 ± 0.000600 kg C/ha/day) (**Fig 3.6a**), and according to a Mann-Whitney test, the median CH₄ emissions from 2012 and 2013 did not significantly vary ($p = 0.798$).

Average SWC₂₀ in 2013 was greater than in 2012, but average ST₁₀ was similar in both growing seasons. Soybean fields had an average SWC₂₀ of 19.39 ± 7.63 % in 2012, compared to 33.28 ± 4.36 % in 2013. Soybean fields had an average ST₁₀ of 20.98 ± 3.29 °C in 2012, which was similar to 19.26 ± 3.27 °C in 2013. It is believed that GHG emissions from vegetated locations in soybean fields did not vary between 2012 and 2013 growing seasons because soil temperature was similar and SWC₂₀ only varied by approximately 10%.

4. Conclusion

The demonstrated benefits of controlled tile drainage (CTD) include increased yields and reduced nutrient loss in tile drainage water. The potential for increased soil emissions of CO₂, CH₄ and N₂O due to greater soil saturation with CTD has been evaluated through an extensive monitoring of soil gas fluxes in paired fields (CTD and uncontrolled tile drainage (UTD)) over two growing seasons with contrasting rainfall records. For both the comparatively wet and dry years, the measured soil CO₂ and CH₄ fluxes did not significantly differ between CTD and UTD fields ($P>0.05$). With the exception of one field pair, N₂O fluxes were also not significantly different for field pairs studied ($P>0.05$). When examining spatial distribution of measurements over the drainage tiles (OT) and between tile (BT) locations in CTD fields, there was no significant difference in GHG emissions ($P>0.05$). CO₂ emissions had significant positive correlations with relative humidity, air temperature, and soil temperature (0-10 cm depth) (ST10), and a significant negative correlation with soil water content (0-20 cm depth) (SWC20) in many fields. There was no significant difference in mean GHG emissions from vegetated locations in soybean fields in 2012 and 2013 ($P>0.05$).

It was expected that the significantly smaller WTDBS in CTD fields compared to UTD fields would result in greater SWC20 in CTD fields, but this study indicated that there was no significant difference in SWC20 between CTD and UTD fields and spatially within CTD fields. Therefore, as there was no significant difference in SWC20 and ST10 between CTD and UTD fields and spatially within fields, there was no driving factor for differences in GHG emissions. Thus, the positive findings of this study regarding GHG emissions further supports the use of CTD in agriculture.

5. Figures & tables

Table M1 Planting, harvest, photosynthetic pathway, and key tile drainage management dates for the studied control tile drained (CTD) and uncontrolled tile drained (UTD) agricultural fields.

Field pair	CTD field: area (ha)	UTD field: area (ha)	Year	Planting date: crop type	Date stop gates were closed for CTD fields	Date stop gates were opened for CTD fields	Harvest date
1 & 2	Field 2: 2.3	Field 1: 2.0	2012	May 16: soybean	May 16	September 19	September 24
			2013	May 5: corn	May 29 ^a	September 26	November 14
11 & 14	Field 11: 4.2	Field 14: 4.1	2012	In 2011: forage	May 23	November 19	June 1, July 10, August 20, October 24
			2013	May 15: soybean	May 27	September 26	October 25
12 & 13	Field 12: 5.0	Field 13: 4.2	2013	May 15: soybean	May 27	September 26	October 25

^a Gates were left open for a longer time because farmer wanted them open due to excessive rainfall

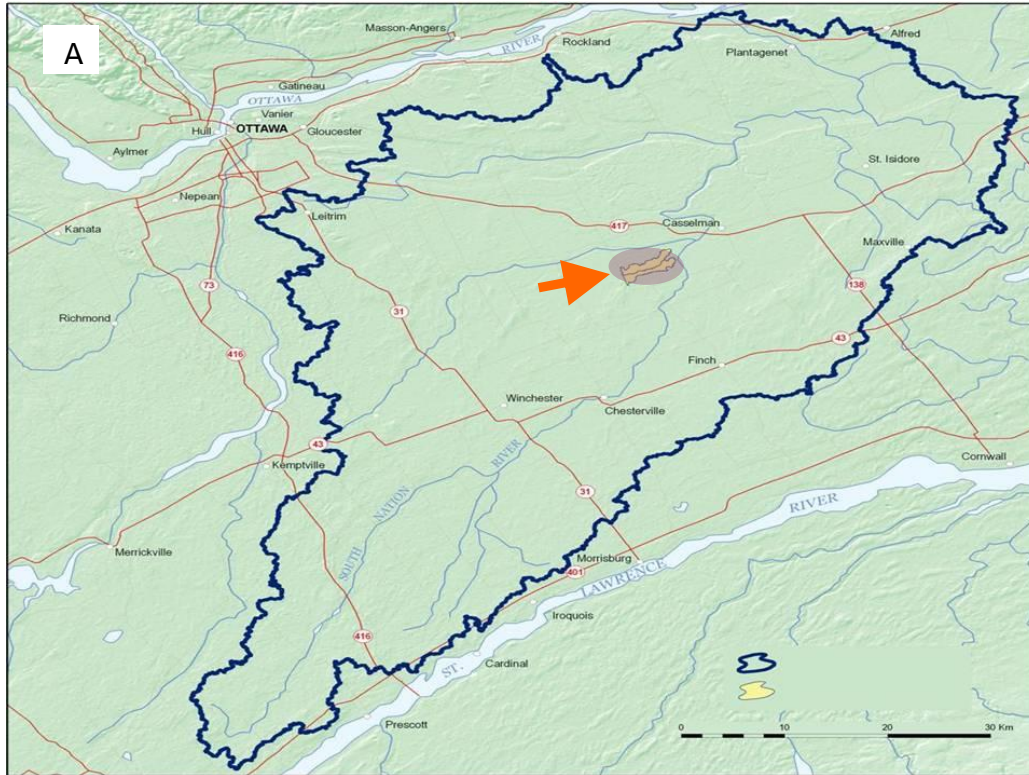


Figure M1 The agriculturally productive 3,900 km South Nation Watershed in Eastern Ontario, Canada (A), and the location of the fields in this study (B). The arrow in (A) indicates the location of the fields within the watershed (source David Lapen AAFC) and (B) is a Google Earth Maps image of the fields (Map data: Google, 2015).

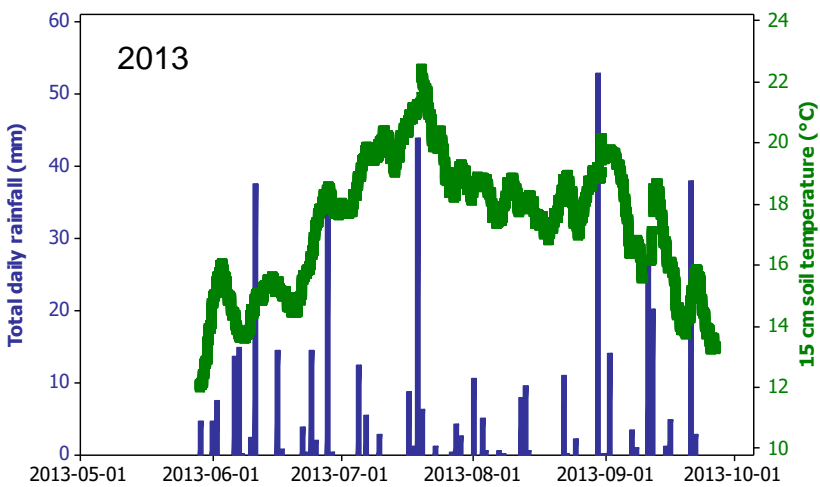
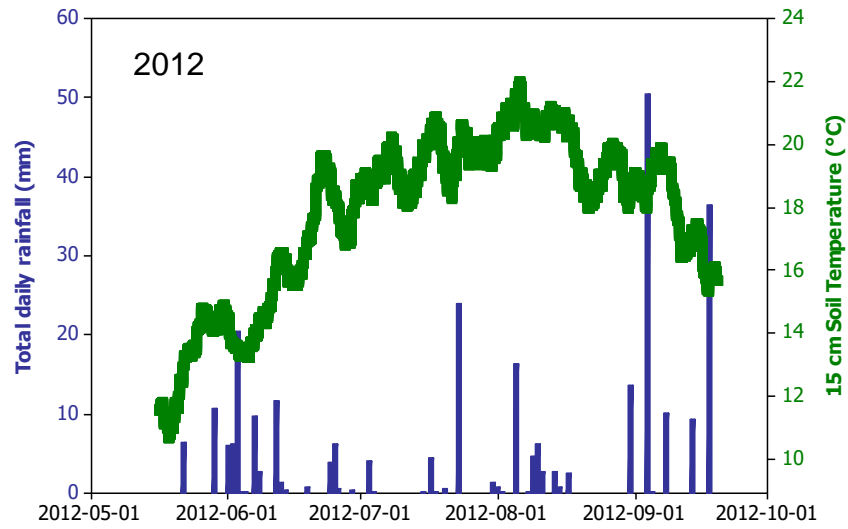


Figure 3.1a Total daily rainfall and soil temperature (0 - 15 cm depth) measured every 30 minutes, in 2012 and 2013, from the weather station located near Field 1.

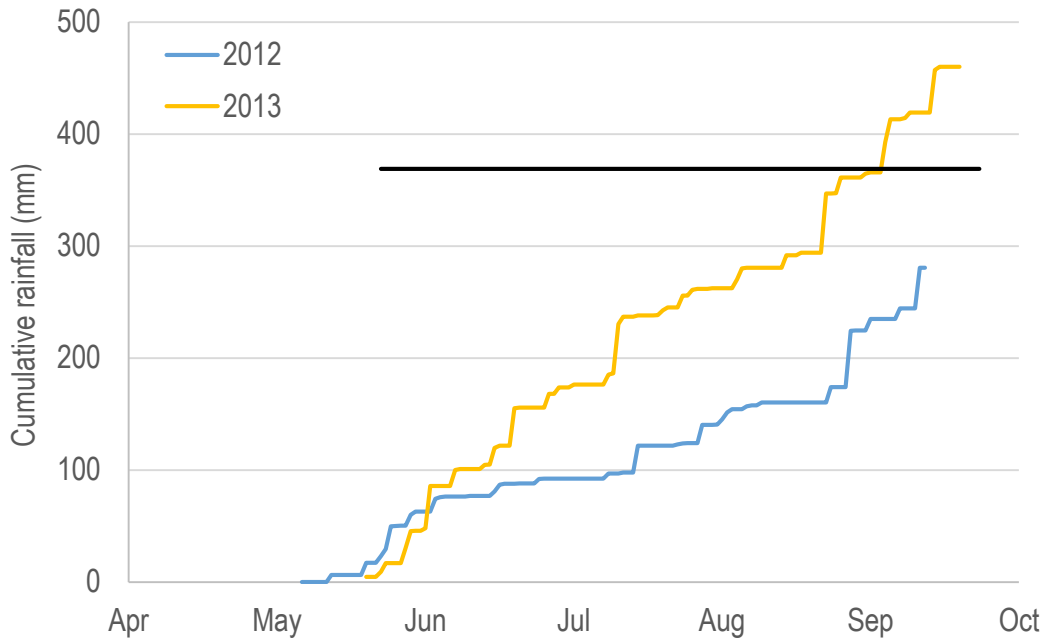


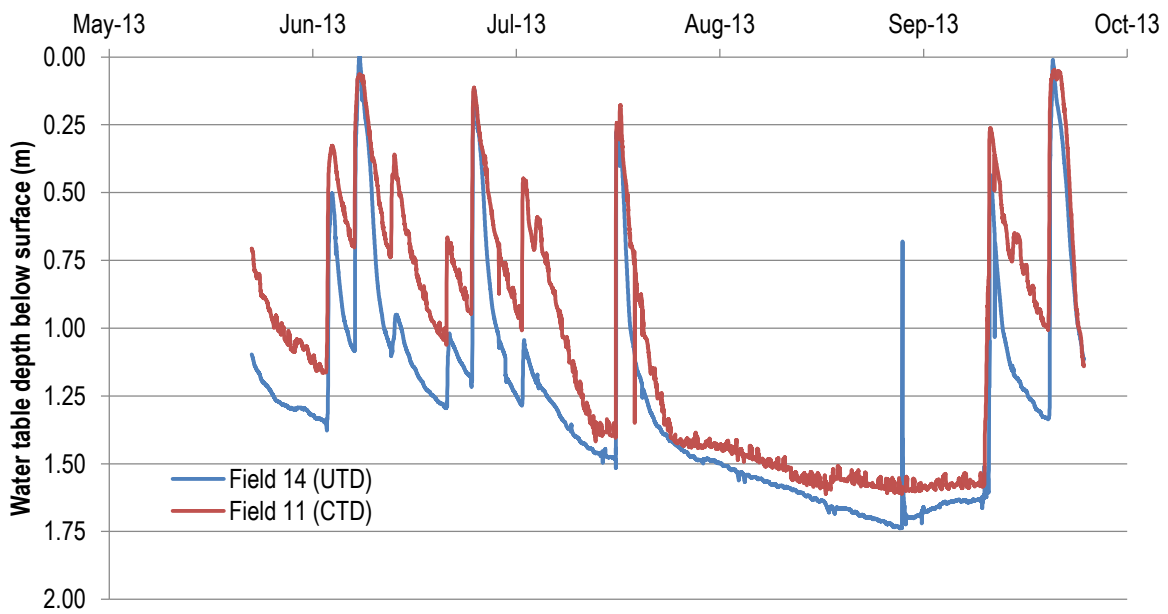
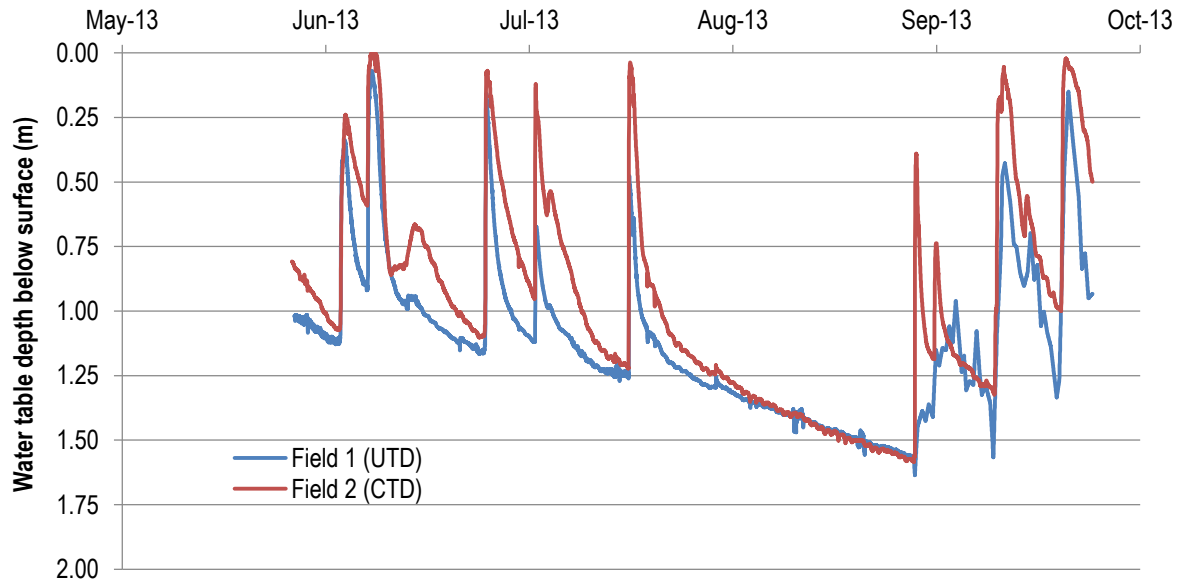
Figure 3.1b Cumulative rainfall for the 2012 and 2013 growing seasons (according to specific dates from Field 1 & 2’s growing seasons), from the weather station located near Field 1. The black line represents the average 30-year rainfall (1981-2010) for the approximate growing season (June-September = 369 mm) from nearby Russell station (EC 2015).

Table 3.1a Yields (kg/ha) for over tile and between tile locations within CTD and UTD fields during the 2012 and 2013 growing seasons. Shading indicates no data available.

Year	Field pair	Between Tile		Over Tile	
		UTD Yield	CTD Yield	UTD Yield	CTD Yield
2012	1 & 2	3329	3249	3750	3075
	11 & 14	7498	9323	7863	8864
2013	1 & 2	12017	12197		
	11 & 14	4901	4266	3718	3735
	12 & 13	4255	4260	4129	4289

Table 3.2a Average \pm standard deviation for water table depth below surface (m) for uncontrolled tile drained (UTD) and control tile drained (CTD) field pairs in 2012 and 2013. P values are presented for t-tests.

Field Pair	UTD	CTD	P value
2012			
Field 1 & 2	1.573 \pm 0.417	1.593 \pm 0.407	<0.05
Field 11 & 14	2.690 \pm 0.628	2.491 \pm 0.672	<0.05
2013			
Field 1 & 2	1.115 \pm 0.301	0.966 \pm 0.406	<0.05
Field 11 & 14	1.277 \pm 0.380	1.070 \pm 0.446	<0.05
Field 12 & 13	1.413 \pm 0.318	1.266 \pm 0.415	<0.05



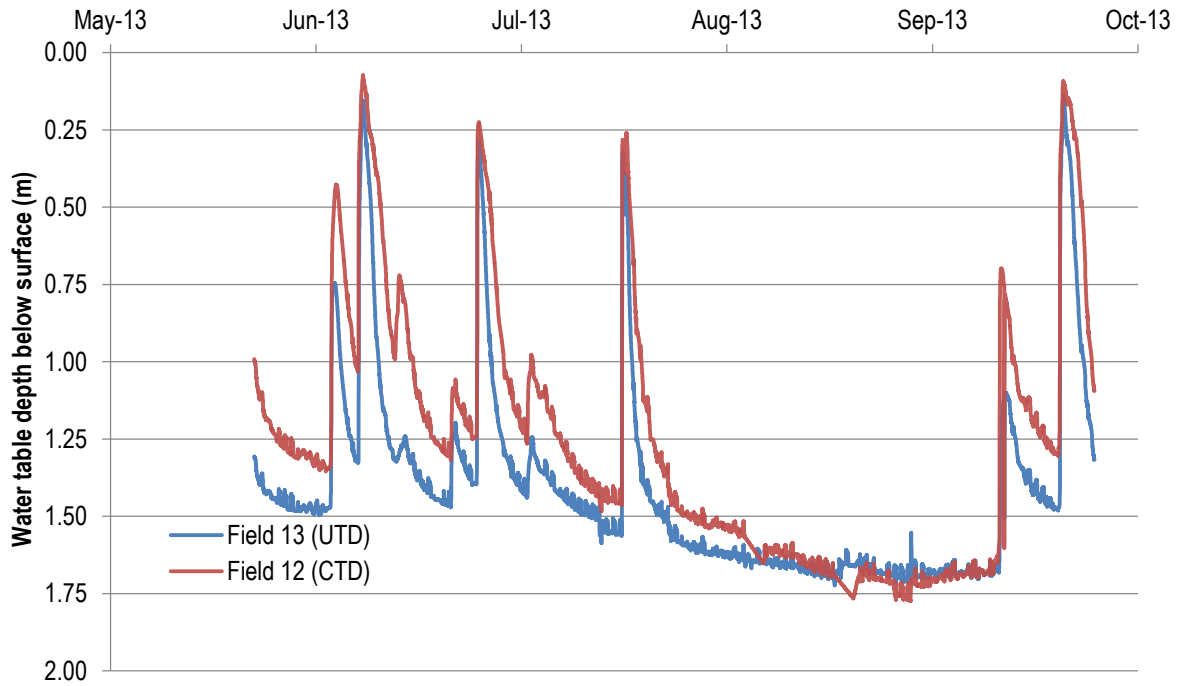


Figure 3.2a Water table depth below surface, measured every 15 minutes, for field pairs 1 (UTD) and 2 (CTD), 11 (CTD) and 14 (UTD), and 12 (CTD) and 13 (UTD) during the 2013 growing season.

Table 3.2b Average \pm standard deviation for soil moisture (0-20 cm depth), soil temperature (0-10 cm depth), and daily water table depth below surface for control tile drained (CTD) and uncontrolled tile drained (UTD) field pairs from greenhouse gas sampling days during the 2012 and 2013 growing seasons. P values are presented for t-tests.

Year	Field pair	Soil moisture (%)			Soil temperature (°C)			Daily water table depth below surface (m)		
		CTD	UTD	P value	CTD	UTD	P value	CTD	UTD	P value
2012	1 & 2	20.49 \pm 8.04	20.65 \pm 8.67	0.942	20.30 \pm 3.36	20.03 \pm 3.11	0.739	1.61 \pm 0.46	1.62 \pm 0.44	0.923
	11 & 14	22.19 \pm 8.55	24.05 \pm 8.14	0.444	17.60 \pm 5.78	17.15 \pm 5.17	0.779	2.65 \pm 0.59	2.84 \pm 0.51	0.157
2013	1 & 2	32.20 \pm 4.05	31.78 \pm 4.50	0.326	18.33 \pm 4.00	19.05 \pm 4.76	0.242	1.10 \pm 0.31	1.21 \pm 0.18	0.002
	11 & 14	33.80 \pm 4.51	33.34 \pm 5.16	0.552	19.34 \pm 3.62	19.41 \pm 3.52	0.912	1.20 \pm 0.32	1.40 \pm 0.19	0.000
	12 & 13	31.57 \pm 3.85	31.64 \pm 4.12	0.918	19.32 \pm 2.90	19.36 \pm 2.82	0.928	1.48 \pm 0.19	1.57 \pm 0.11	0.001

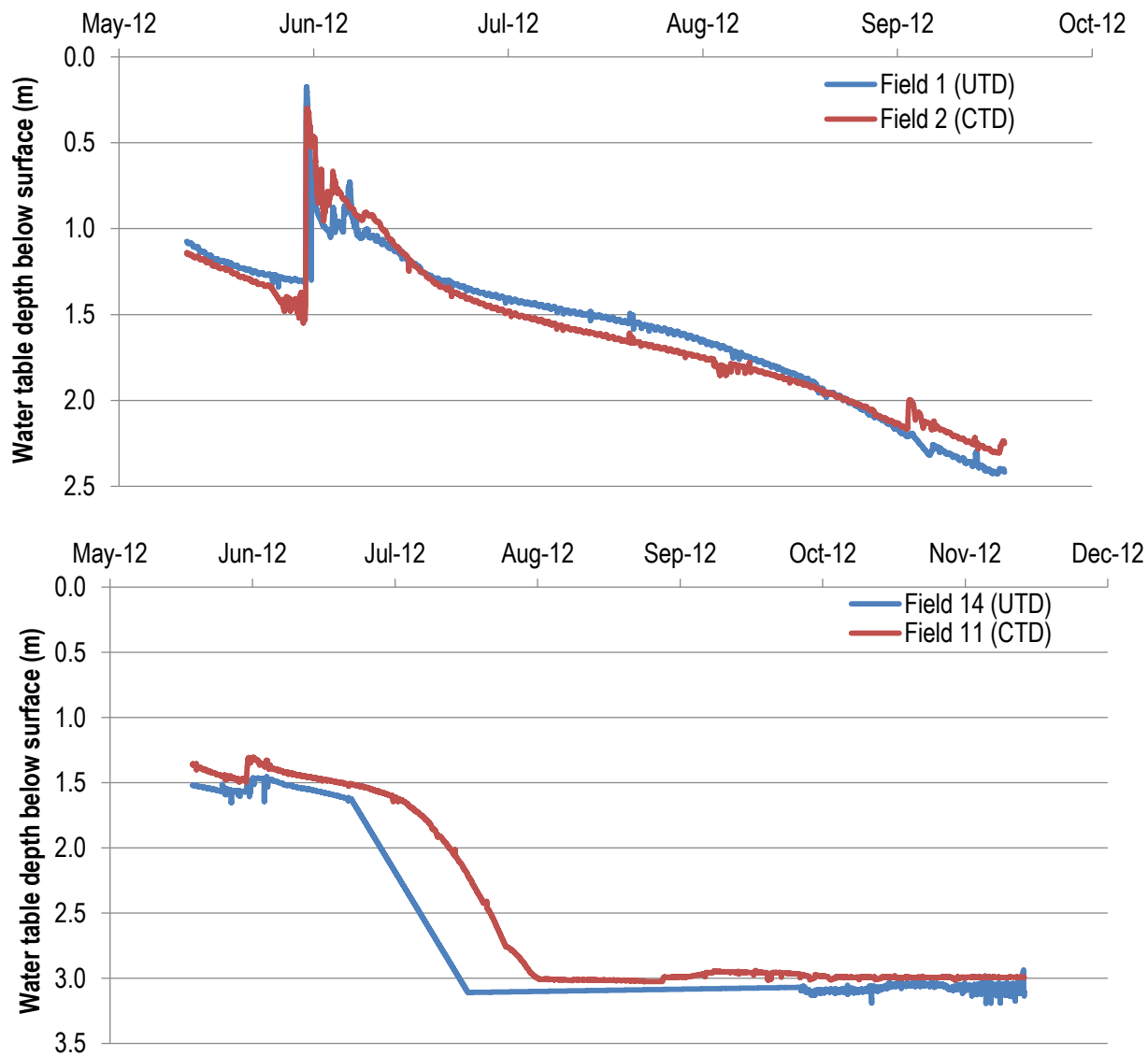


Figure 3.2b Water table depth below surface, measured every 15 minutes, for field pairs 1 (UTD) and 2 (CTD), and 11 (CTD) and 14 (UTD) during the 2012 growing season.

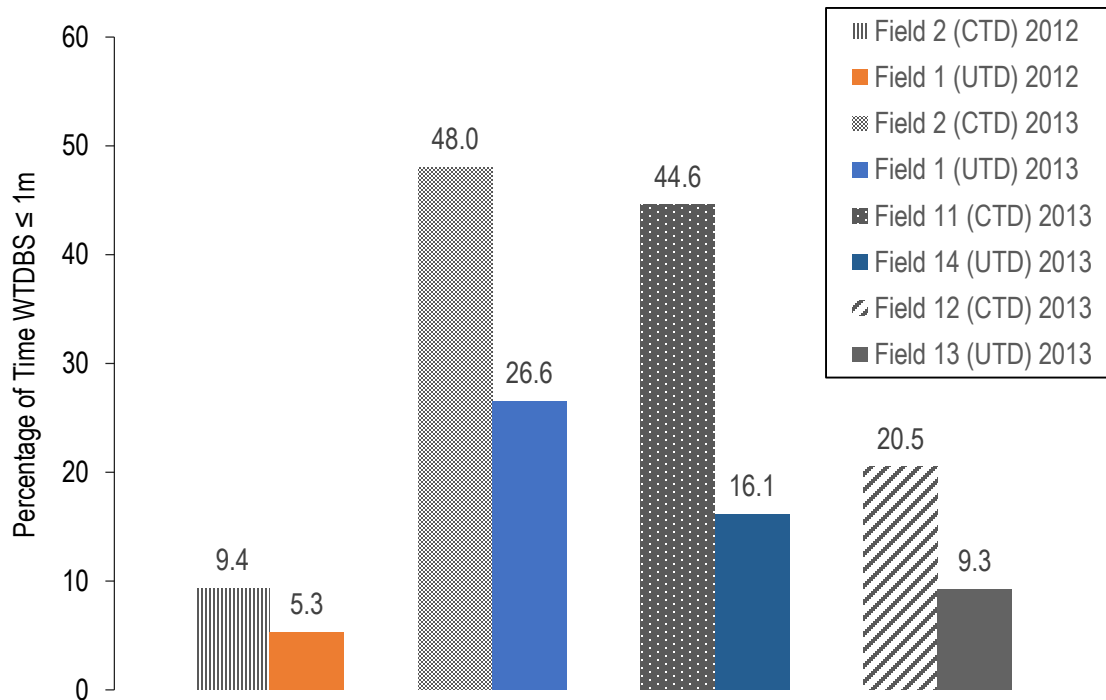


Figure 3.2c The percentage of time each field spent with a water table depth below surface (WTDBS) at 1 m or shallower during the 2012 and 2013 growing seasons. The CTD fields are indicated with a patterned fill. Note that Field 11 and 14 in 2012 had 0 % of the growing season with a WTDBS equal to or shallower than 1 m.

Table 3.3a Average daily soil GHG fluxes (CO₂, CH₄, and N₂O) from vegetated locations in CTD and UTD fields during the 2012 and 2013 growing season. T-tests were conducted among field pair fluxes except for Field 11 and 14 N₂O in 2012 where the data was not normal even when log-transformed, and therefore a non-parametric Mann-Whitney test was done. P values with a * indicate significance at 0.05 level. Shading indicates field pairs.

Drainage Management: Field ID	Average Flux						
	CO ₂ ± Stdev. kg C ha ⁻¹ day ⁻¹	P Value	CH ₄ ± Stdev. kg C ha ⁻¹ day ⁻¹	P Value	N ₂ O ± Stdev. kg N ha ⁻¹ day ⁻¹	P Value	
2012							
Soybean	UTD: 1	26.53 ± 9.93		-0.00043 ± 0.00150		0.0088 ± 0.0159	
	CTD: 2	25.41 ± 9.10	0.618	-0.00086 ± 0.00149	0.229	0.0104 ± 0.0243	0.755
Forage	CTD:11	85.40 ± 40.0		-0.00061 ± 0.00119		0.00255 ± 0.00295	
	UTD: 14	101.0 ± 53.3	0.256	-0.00040 ± 0.00144	0.598	0.00462 ± 0.00581	0.359
2013							
Corn	UTD: 1	20.27 ± 7.39		-0.000759 ± 0.000579		0.0237 ± 0.0791	
	CTD: 2	20.11 ± 8.11	0.891	-0.000849 ± 0.000594	0.313	0.0309 ± 0.0840	0.559
Soybean	CTD:11	29.47 ± 8.20		-0.000406 ± 0.000625		0.00907 ± 0.00768	
	UTD: 14	28.09 ± 9.66	0.347	-0.00098 ± 0.00264	0.069	0.00779 ± 0.00699	0.288
	CTD: 12	29.89 ± 9.18		-0.00084 ± 0.00213		0.00653 ± 0.00519	
	UTD: 13	31.57 ± 8.52	0.261	-0.00067 ± 0.00214	0.642	0.00407 ± 0.00335	0.001*

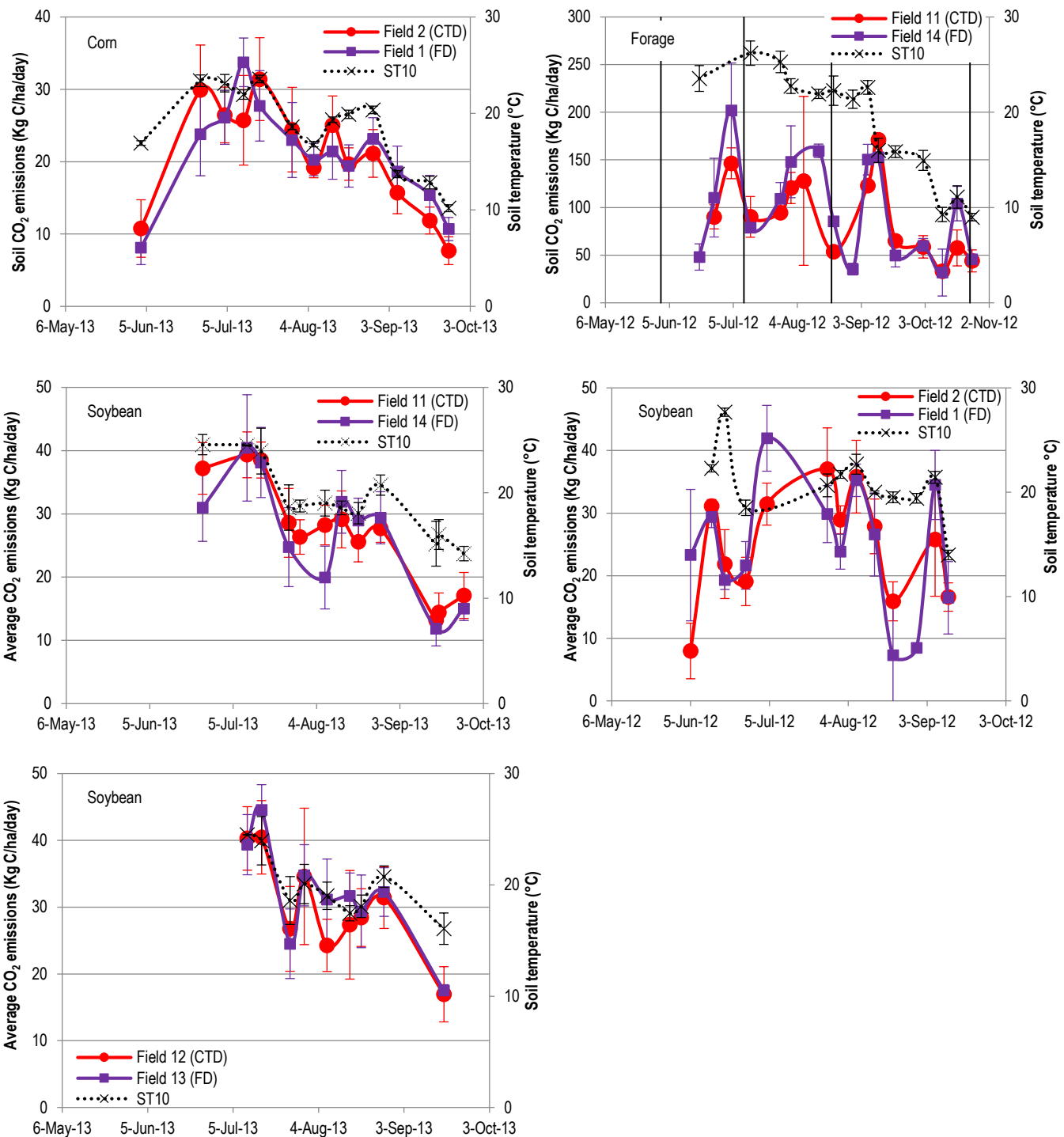


Figure 3.3a Average CO₂ soil emissions and soil temperature (0-10 cm depth) (ST10) during the 2012 and 2013 growing seasons, from vegetated sites, in CTD and UTD fields. Vertical lines within the forage graph represent cuttings on June 1, July 10, August 20, and October 24, 2012. In 2012, the forage CO₂ measurements consisted of

plants within the chamber and that is why the magnitude of respiration is much higher compared to the other crops. The crop is indicated for each field pair and the error bars indicate standard deviations.

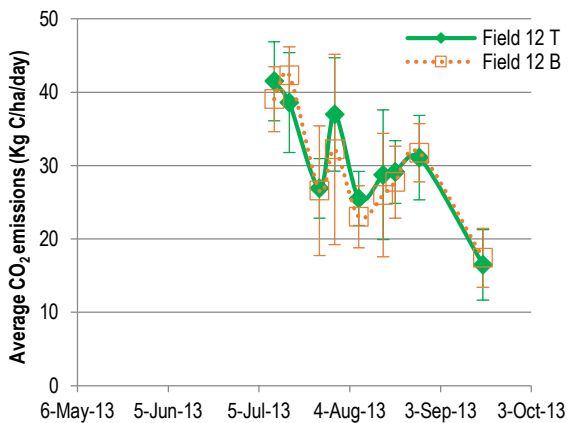
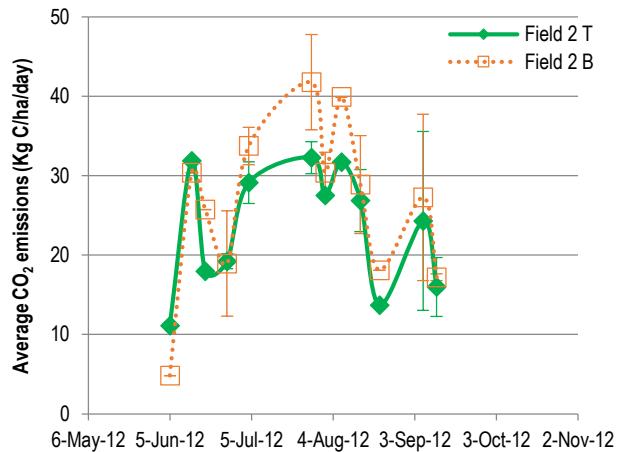
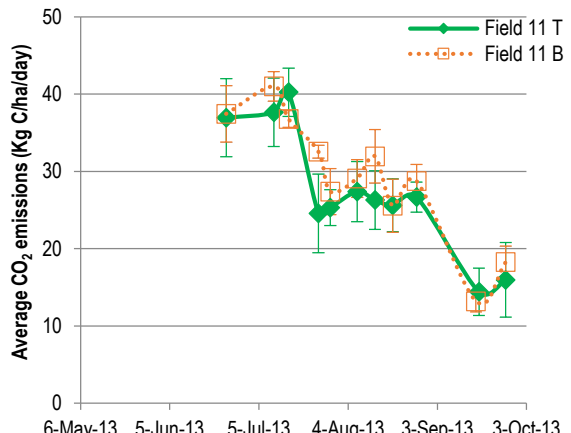
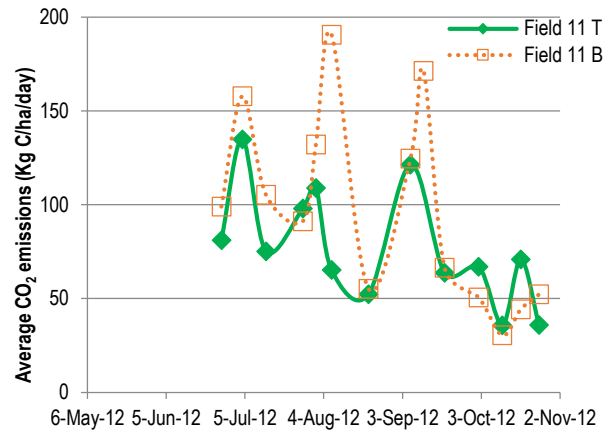
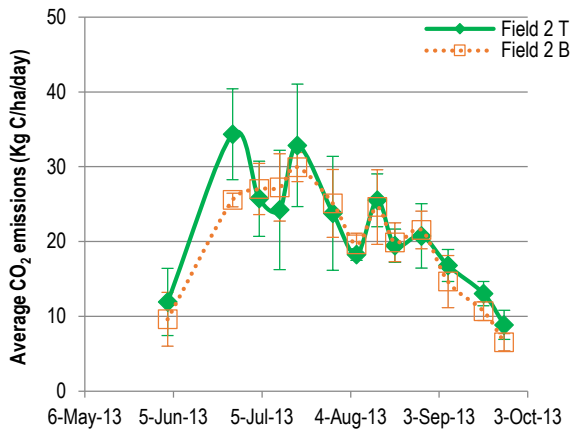


Figure 3.4a Average soil CO₂ emissions, during the growing season, from over tile (T) and between tile (B) locations in vegetated sites within CTD fields in 2012 and 2013. Error bars indicate standard deviations. Note that for Field 11 2012 there are no error bars because there is only one value for each T and B measurement (no averages). Also, Field 2 2012 has two measurements for some data points and one value for others where no error bar is present.

Table 3.4a Average daily soil GHG fluxes (CO₂, CH₄, and N₂O) from over tile (OT) and between tile (BT) vegetated locations in CTD fields during the 2012 and 2013 growing season. P-values from t-tests were conducted comparing OT and BT emissions for each field except for Field 2 2012 N₂O and CH₄ where the data was not normal even when log-transformed and, therefore, a non-parametric Mann-Whitney test was done. P values with a * indicate significance at 0.05 level. Shading indicates an OT and BT comparison.

Field ID: location	Average Flux						
	CO ₂ ± Stdev.		CH ₄ ± Stdev.		N ₂ O ± Stdev.		
	kg C ha ⁻¹ day ⁻¹	P Value	kg C ha ⁻¹ day ⁻¹	P Value	kg N ha ⁻¹ day ⁻¹	P Value	
2012							
Soybean	2: OT	24.05 ± 7.46	0.296	-0.00107 ± 0.00197	0.930	0.0083 ± 0.0201	0.293
	2: BT	27.1 ± 10.2		-0.000650 ± 0.000620		0.0114 ± 0.0273	
Forage	11: OT	77.7 ± 30.6	0.370	0.00022 ± 0.00580	0.420	0.00191 ± 0.00170	0.416
	11: BT	92.3 ± 48.6		-0.001114 ± 0.000747		0.00281 ± 0.00353	
2013							
Corn	2: OT	20.66 ± 8.41	0.562	-0.000876 ± 0.000698	0.239	0.036 ± 0.108	0.484
	2: BT	19.67 ± 7.98		-0.000712 ± 0.000591		0.0245 ± 0.0446	
Soybean	11: OT	28.67 ± 7.90	0.183	-0.000418 ± 0.000520	0.334	0.00728 ± 0.00635	0.203
	11: BT	30.87 ± 6.73		-0.000528 ± 0.000487		0.00940 ± 0.00832	
	12: OT	30.55 ± 8.99	0.654	-0.00110 ± 0.00291	0.278	0.00686 ± 0.00516	0.708
	12: BT	29.57 ± 9.56		-0.000553 ± 0.000666		0.00639 ± 0.00536	

Table 3.4b Average \pm standard deviation for soil moisture (0-20 cm depth) and soil temperature (0-10 cm depth) for over drainage tile and between drainage tile locations in control tile drained (CTD) agricultural fields throughout the 2012 and 2013 growing seasons. P values are presented for t-tests.

Year and crop	CTD field#	Soil moisture (%)			Soil temperature (°C)		
		Over tile	Between tile	P value	Over tile	Between tile	P value
2012							
Soybean	2	20.51 \pm 8.27	22.33 \pm 8.96	0.467	20.76 \pm 3.56	20.81 \pm 3.48	0.961
Forage	11	24.12 \pm 7.51	24.69 \pm 6.72	0.805	18.01 \pm 6.35	17.61 \pm 6.03	0.876
2013							
Corn	2	32.05 \pm 3.32	32.39 \pm 4.65	0.558	18.34 \pm 4.07	18.39 \pm 4.02	0.951
Soybean	11	33.41 \pm 4.24	33.33 \pm 4.18	0.902	19.39 \pm 3.86	19.40 \pm 3.80	0.989
Soybean	12	31.47 \pm 3.62	31.67 \pm 4.08	0.764	18.54 \pm 2.35	19.15 \pm 2.73	0.340

Table 3.5a Spearman rho value with its associated P value below, from Spearman Rank correlations among soil CO₂ fluxes and environmental variables. The environmental variables examined were: SWC20 and ST10 taken next to GHG chambers at time of GHG sampling, average daily WTDBS for GHG sampling day, sum of rainfall from 1000-1400 h on GHG sampling day, total rainfall day prior to GHG sampling, and average SR, RH, and air temperature between 1000-1400 h on GHG sampling day. Shading indicates a significant P value (P<0.05). A * indicates no ranking could be done as all values in list were identical.

Variable	2013						2012			
	Field 1	Field 2	Field 11	Field 14	Field 12	Field 13	Field 1	Field 2	Field 11	Field 14
	UTD	CTD	CTD	UTD	CTD	UTD	UTD	CTD	CTD	UTD
SWC20	-0.469	-0.435	-0.340	-0.536	-0.362	-0.470	-0.106	-0.041	-0.509	-0.191
	0.000	0.000	0.002	0.000	0.002	0.000	0.577	0.825	0.026	0.383
ST 10	0.648	0.801	0.777	0.752	0.654	0.632	0.346	0.503	0.725	0.382
	0.000	0.000	0.000	0.000	0.000	0.000	0.061	0.003	0.000	0.072
WTDBS	0.389	0.255	-0.229	0.347	0.024	0.107	-0.041	-0.272	-0.557	0.123
	0.000	0.012	0.041	0.002	0.844	0.377	0.831	0.132	0.003	0.517
rainfall 1000-1400h	0.176	0.280	0.185	0.393	0.402	0.327	0.166	0.182	*	0.324
	0.091	0.006	0.100	0.000	0.000	0.005	0.381	0.318	*	0.080
rainfall day prior	0.032	0.228	-0.271	-0.138	-0.297	-0.303	0.101	-0.019	-0.052	0.006
	0.759	0.025	0.015	0.222	0.011	0.010	0.596	0.916	0.797	0.976
SR	-0.556	-0.545	0.206	0.196	0.090	0.105	0.210	0.351	0.368	-0.168
	0.000	0.000	0.066	0.081	0.450	0.383	0.265	0.049	0.059	0.375
RH	0.442	0.588	0.218	0.290	0.494	0.542	0.504	0.479	0.252	0.393
	0.000	0.000	0.052	0.009	0.000	0.000	0.005	0.006	0.204	0.032
air temperature	0.685	0.704	0.688	0.696	0.533	0.623	0.482	0.390	0.705	0.325
	0.000	0.000	0.000	0.000	0.000	0.000	0.007	0.027	0.000	0.080

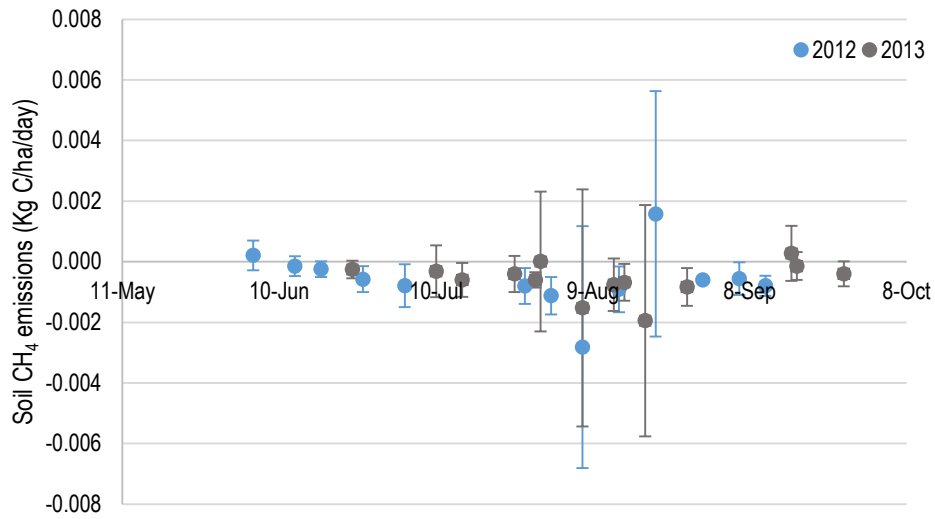


Figure 3.6a Average soil CO₂, N₂O, and CH₄ emissions from vegetated locations in soybean fields in 2012 (Field 1 & 2) and 2013 (Field 11-14) growing seasons. Error bars indicate standard deviations.

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Chapter 4 Using ^{13}C isotopic analysis to assess soil carbon pools associated with tile drainage management during drier and wetter growing seasons

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Abstract

Controlled tile drainage (CTD) is an agricultural management practice with well-documented water quality and agronomic benefits, however, by virtue of its effect upon soil hydrology, CTD could potentially affect soil respiration. Firstly, the $\delta^{13}\text{C}$ of soil-respired CO_2 (R_T) was analyzed from CTD and uncontrolled tile drained (UTD) fields cropped with corn, soybean, and forage to test whether CTD affects R_T through altering residue decomposition or gas transport. Secondly, we examined whether CTD imparted an over tile (OT) and between tile (BT) locational effect on R_T . The means of $\delta^{13}\text{C}$ of R_T were not statistically different ($p>0.05$) between four of the five CTD and UTD field pairs. The means of $\delta^{13}\text{C}$ of R_T did not differ ($p>0.05$) between OT and BT locations in four of the five CTD fields. This lack of difference in mean $\delta^{13}\text{C}$ of R_T is believed to have resulted from their being effective indifference in surface soil water contents among CTD and UTD fields and among OT and BT locations. These findings suggest that there may be no requirement for systematically locating soil GHG sampling sites with respect to location of tile drains in field, at least for medium textured soils such as those examined. The results are also useful for helping to capture the carbon footprint of tile drainage management practices imposed at field-scale.

1. Introduction

Soil respiration is a significant source of atmospheric CO₂ (IPCC, 2013; Rochette et al., 1999a; Smith et al., 2007). Soil respiration originates from plants through rhizosphere respiration and soils through microbial respiration (Cheng, 1996; Rochette et al., 1999a; Ryan and Law, 2005). Soil respiration is controlled by factors that affect the transport of CO₂ from soil to atmosphere, and the rate that CO₂ is produced within the soil, where its rate is strongly influenced by nutrient availability, soil temperature, and soil moisture (Kreba et al., 2013; Phillips et al., 2010; Schwen et al., 2015). Gaseous diffusion is a primary transport mechanism associated with gas exchange from soil to atmosphere (Jin and Jury, 1996; Marshall, 1959; Moldrup et al., 2004) and it is largely governed by concentration gradients and the physical disposition of the soil (Ball et al., 1997; Moldrup et al., 2013; Tuli et al., 2005).

Agricultural lands cover a significant amount of the Earth's land surface (Smith et al., 2007), making them a large contributor to atmospheric CO₂. In light of climate change implications, it is becoming increasingly important to understand and quantify the carbon footprint of agricultural management practices broadly (Paustian et al., 2000; West and Marland, 2002). Agricultural management practices can have a variable impact on soil CO₂ emissions (Al-Kaisi and Yin 2005; Lal, 2004; Paustian et al., 1997, 2000). For example, no-till or reduced tillage practices can decrease soil respiration rates compared to intensive or conventional tillage practices (Al-Kaisi and Yin 2005; Paustian et al., 2000; Sánchez et al., 2002; Schwen et al., 2015). Reducing bare fallow in crop rotations, by either increasing the cropping intensity or by using cover crops, can also reduce soil respiration (Al-Kaisi and Yin 2005; Paustian et al., 1997, 2000). López-López et al. (2012)

indicate that a mineral supplement can decrease soil respiration whereas organic fertilizers can increase soil respiration. Mordhorst et al. (2014) found that soil compaction significantly reduces soil respiration by reducing the soil pore network.

Controlled tile drainage (CTD) is an agricultural management practice that regulates tile drain discharge (Gilliam et al., 1979) with well-documented water quality and agronomic benefits (Mejia et al., 2000; Ng et al., 2002; Sunohara et al., 2015; Wesstrom and Messing, 2007). However by virtue of its effect upon soil and groundwater hydrology, CTD could potentially augment GHG emissions to the atmosphere (Elmi et al., 2000; Nangia et al., 2013). For example, Nangia et al. (2013) found significantly higher bulk CO₂ emissions from soil surfaces in cropped CTD fields, in relation to paired uncontrolled tile drained (UTD) fields; but that study did not reveal the source and nature of the respired carbon or whether gas diffusion was directly affected by CTD.

We are currently unaware of studies that have analyzed the effects of CTD on the origin of soil respiration and soil gas diffusivity. A useful tool in this regard is the stable isotope $\delta^{13}\text{C}$, which can be used as a natural tracer to analyze alterations in gas transport and the dynamics of soil organic carbon (SOC); including identification of carbon substrates being oxidized (Cerling et al., 1991; Clark, 2015; Drewitt et al., 2009; Gregorich et al., 1995; Phillips et al., 2010; Rochette et al., 1999a). By measuring $\delta^{13}\text{C}$ of total soil respiration (R_T ; rhizosphere and microbial sources combined) from cropped agricultural fields, rather than from areas containing only heterotrophic soil emissions (i.e., soil without crops), we can study contributions from soil and plant respiration and thus evaluate whether CTD has an overall effect on R_T .

Using $\delta^{13}\text{C}$ as a natural tracer is made possible by the varying C_3 and C_4 photosynthetic pathways, which cause C_3 and C_4 plants to have differing $\delta^{13}\text{C}$ values (Cheng, 1996; Drewitt et al., 2009; Gregorich et al., 1995; Rochette et al., 1999b). The $\delta^{13}\text{C}$ values of C_3 and C_4 plants are more depleted than atmospheric CO_2 because during CO_2 uptake, plants fix a proportionally larger amount of $^{12}\text{CO}_2$ and discriminate against the heavier $^{13}\text{CO}_2$. C_3 plants have a considerably more depleted $\delta^{13}\text{C}$ (~ -26 to -28 ‰) than atmospheric CO_2 (~ -8 ‰) because of the inefficient step of CO_2 respiration that results in an approximate 20 ‰ fractionation. In contrast, C_4 photosynthesis reduces photorespiration and increases CO_2 fixation, which results in a lower fractionation of approximately -6 ‰ and a more enriched $\delta^{13}\text{C}$ (~ -10 to -14 ‰) compared to C_3 plants (Clark, 2015; O'Leary, 1988).

The $\delta^{13}\text{C}$ of R_T can be used to identify the carbon substrate being oxidized because the $\delta^{13}\text{C}$ of plants affect the $\delta^{13}\text{C}$ of soil organic matter (SOM). The $\delta^{13}\text{C}$ of SOM resembles the $\delta^{13}\text{C}$ of the plant material from which it was derived such that soils derived from C_4 plants have a more enriched $\delta^{13}\text{C}$ (~ -12 ‰ to -14 ‰) compared to soils derived from C_3 plants (~ -24 ‰ to -29 ‰) (Cheng, 1996; Gregorich et al., 1995). By growing a C_4 crop in a soil derived from C_3 plants or vice versa results in isotopically distinct values for the crop and the SOC thus providing a means to quantify the source of R_T (Cheng, 1996; Drewitt et al., 2009; Gregorich et al., 1995; Rochette et al., 1999b). Identifying the source of R_T is critical for assessing how field management practices sequester, emit, and cycle soil carbon; especially within the context of gauging the environmental footprint of a BMP.

The $\delta^{13}\text{C}$ of R_T can be used to analyze the dynamics of SOC because the $\delta^{13}\text{C}$ of R_T is affected by biological changes in the produced CO_2 and soil physical changes that alter gas exchange with the atmosphere (Cerling et al., 1991; Clark, 2015; Phillips et al., 2010). At a diffusive steady-state, the $\delta^{13}\text{C}$ of R_T equals the $\delta^{13}\text{C}$ produced within the soil from either plants or soil (Cerling et al., 1991). Changes in the environment can alter the amount of CO_2 produced and the gas diffusivity through the soil thus creating non-steady state conditions where the $\delta^{13}\text{C}$ of R_T does not equal the $\delta^{13}\text{C}$ produced within the soil (Phillips et al., 2010). Therefore the $\delta^{13}\text{C}$ of R_T can identify whether soil physical changes occurred that affected CO_2 production or gas transport.

While studies have used $\delta^{13}\text{C}$ of R_T to reveal the sources of respired carbon (Drewitt et al., 2009; Rochette et al., 1999a, 1999b); as far as we are aware, there are no reports that have used $\delta^{13}\text{C}$ of R_T to reveal whether CTD affects the carbon substrates being oxidized and/or the soil physical conditions that govern CO_2 production and gas transport. We hypothesize that CTD will alter residue decomposition and soil gas diffusivity through impacts on soil hydrology (e.g., decreased water table depth and increased soil water contents). We also hypothesize that the location in a field with respect to a tile drain will be important in this regard due to water table mounding (Smedema et al., 1983) that can occur between tile drains, different soil chemical/physical properties over tiles (trenches backfilled with disturbed soil during time of tile installation), and biopores that can preferentially occur over tile drains (in relation to between them) (Nuutinen et al., 2001).

The objectives of this study are to: (1) measure the $\delta^{13}\text{C}$ of R_T in CTD and UTD corn, soybean, and perennial forage fields during two growing seasons with contrasting precipitation records; (2) compare whether environmental variables such as soil temperature, soil water content, and water table depth vary for CTD and UTD fields; and (3) evaluate differences in $\delta^{13}\text{C}$ of R_T between and over tile.

2. Materials and methods

2.1 Study fields and weather

The fields studied are located within experimental watersheds (45.26 N, 75.18 W) in eastern Ontario, Canada (**Figure 1**; see Sunohara et al., 2015 for details). The fields are low-gradient (slope < 1%). Soil at the site is classified as a Bainsville silt loam (Wicklund and Richards, 1962). The 30-year normal annual precipitation (1981-2010) at the site is 981 mm and average daily air temperature is 6.5°C (EC, 2015). From June to September (approximate growing season), cumulative average 30-year rainfall (1981-2010) and average daily air temperature is 369 mm and 18 °C, respectively (EC, 2015). Total growing season rainfall was 281 mm and 460 mm in 2012 and 2013, respectively and average soil temperature (0-15 cm depth) during the growing season was 17.8°C ± 2.6 and 17.4°C ± 2.1 in 2012 and 2013, respectively.

This two year study (2012 – 2013) was conducted during the growing season on subsurface tile drained agricultural fields under corn (*Zea mays*), soybean (*Glycine max*), and forage (*Phleum pratense* and *Medicago sativa* timed together) (**Figure 1**). This study examined two pairs of fields in 2012 and three in 2013 with each pairing under a common cropping-soil management practice but varying in tile drainage management practice (UTD and CTD field pairing) (**Table 1**). Tillage practices for soybean in 2012 (Fields 1 &

2) consisted of fall mouldboard ploughing (to max ~0.3 m depth), and spring cultivation using a chisel plough. For soybean in 2013 (Fields 11-14) and forage in 2012 (Fields 11 & 14) there was no tillage but herbicide was sprayed for Fields 11-14 in 2013 to kill the forage crop from the prior year. Row spacing for corn and soybean was approximately 0.5 m with typical seeding rates and standard mineral fertilizer rates (~170 Kg N/ha for corn and ~10 kg N/ha for soybean) for the region (Ontario Ministry of Food and Rural Affairs, 2009; Sunohara et al., 2015). The crop history and the associated photosynthetic pathway for fields 11-14 for the five years prior to the study was corn (C₄: 2007-2010) and sudan grass (C₄: 2011), whereas for fields 1 and 2 it was corn (C₄: 2007, 2009, 2010) and soybean (C₃: 2008, 2011).

Tiles in the fields were 0.102 m diam., spaced 15-17m apart, and had a nominal field depth of ~1 m. The lateral tile drains connected to a header tile (0.152 m in diameter) which were in turn connected to a main outlet that drained into a drainage ditch. For both CTD and UTD fields, in-line water flow control structures (Agri Drain Corp., Adair, IA) were installed (see Sunohara et al., 2015, 2014). Water flow control was employed for CTD fields by setting structure stopgates (stoplogs) to a flow control depth of roughly 0.6 m below the surface; when water levels in field were shallower than this depth, tile flow to the receiving stream occurred. Flow control was imposed between roughly spring planting and near time of harvest in fall (**Table 1**). For CTD fields, tile drainage control was not imposed during the non-growing season. For UTD fields, tile drains were allowed to drain unimpeded year round (no stopgates were used to control flow in the in-line structures).

2.2 Soil-respired CO₂ and isotope measurements

R_T sampling was carried out using the nonsteady-state chamber design and method described by Rochette and Bertrand (2008), which was also similarly used by Nangia et al. (2013). Briefly, chamber bases were installed between crop rows (corn and soybean) except for forage fields where forage covered the entire field and thus the chambers contained plants. Chamber bases were installed at ~10 cm depth (and ~5 cm head space) and left there for the growing season (with routine checks to contact integrity with soil). Rectangular chambers (0.75 x 0.15 m) were used in soybean fields, and square chambers (0.35 x 0.35 m) were used in corn fields, and both types of chambers had flat lids. Forage fields had circular chambers with a 1.28 m circumference, and a lid of height 0.32 m to accommodate growing plants. Plastic chamber lids contained a rubber septa sampling port on the top and a foam gasket seal on their underside. Headspace volume and surface area of chambers were measured monthly.

Sampling of R_T was carried out between 1000-1400 h (peak emission times) at weekly intervals from over tile (OT) and between (BT) locations on both fields of each field pair (**Figure 1**); sample numbers varied marginally due to logistics or weather. In field pair 1-2, GHG measurements were made in one area of the field in 2013 (total of 12 chambers per field) and two areas of the field in 2012 (total of 8 chambers per field) (**Figure 1**). In fields 11-14, GHG measurements were made in one area of the field, where in 2012 field pair 11 & 14 had a total of 4 chambers per field and in 2013 fields 11-14 had a total of 12 chambers per field (**Figure 1**).

R_T sampling involved capping the chamber, and using a 30 mL syringe (fitted with a 26-gauge needle) to immediately collect a 30 mL sample of the chamber headspace at

time zero. During chamber deployment, four samples were collected from each chamber's headspace and were analyzed for CO₂ concentration. Samples of R_T were analyzed for CO₂ concentration, in order to obtain the concentration at time of isotope sample so that isotope samples could be corrected for atmospheric CO₂. R_T from soybean and corn fields were sampled at 0, 6, 12, and 18 minutes whereas forage was sampled over a longer time at 0, 12, 24, and 36 minutes to ensure that the larger forage chambers had enough time for concentrations to increase to a measurable amount. A fifth R_T sample from each chamber was collected from soybean and corn fields at 24 minutes and from forage fields at 37 minutes and these were analyzed for δ¹³C of R_T. In addition, two air samples were collected and analyzed for CO₂ concentration. The sampled air was injected into 12 mL septum-capped glass vials (Extainers, Labco Ltd.), which were prepared prior to the field by adding an additional silicone septum and 2-3 mg of magnesium perchlorate before evacuating the vials (Rochette and Bertrand, 2008).

The samples that were analyzed for CO₂ concentrations were done using the Varian CP-3800 gas chromatograph with CombiPal autosampler with helium as the carrier gas. The injection volume was 5 mL and a flame ionization detector was used. Samples of reference standard gases were handled the same way as gas samples and approximately one was inserted for every five gas samples during the analysis. CO₂ concentration fluxes were calculated using the rate of concentration change inside the chamber during deployment (Rochette and Bertrand, 2008; Rochette and Hutchinson, 2005).

The R_T samples that were collected for analysis of δ¹³C were analyzed on a continuous flow gasbench + DeltaPlusXP IRMS at the G.G. Hatch Stable Isotope Lab,

University of Ottawa, Ontario, Canada. Isotopic compositions are expressed using delta notation [1]. The international standard for CO₂ samples is CO₂ from Pee Dee Belemnite (PDB) limestone (Ehleringer and Osmond, 1989) and the δ units are parts per thousand (‰).

$$[1] \delta^{13}\text{C} = \left[\left(\frac{^{13}\text{C}/^{12}\text{C}_{\text{sample}}}{^{13}\text{C}/^{12}\text{C}_{\text{standard}}} \right) - 1 \right]$$

The results for δ¹³C of R_T were corrected by removing the contribution from atmospheric CO₂ that was initially inside the chamber by using the mass balance formula [2]. δ¹³C_{measured} is the sample's result obtained from the IRMS, and [CO₂]_{measured} is the calculated concentration of CO₂ in the sample at the time of sampling. [CO₂]_{measured} is calculated using the equation of a line (y=mx+b) [3] where, slope_{CO₂ vs time} is the slope (ppmv/min) obtained from graphing the CO₂ concentrations against their respective sampling time for each chamber deployed. Time_{CO₂ sample} is the time that the measured sample was collected (i.e., R_T from soybean and corn fields were sampled at 24 minutes while forage was sampled at 37 minutes). δ¹³C_{atm} is set at -8 ‰ because according to Fraserdale, Ontario (49°53'N, 81°34'W), which is the closest data to our site, the δ¹³C_{atm} for the approximate growing season (May to end of September) was -8.54 ± 0.70 ‰ (for 2009 and 2010) (Huang, 2015), and thus δ¹³C_{atm} was set at -8 ‰ for the current study because the difference between -8 ‰ and -8.54 ‰ was insignificant for the resulting calculations. [CO₂]_{atm} is set at 400 ppm for this study because the average of two air samples collected in 2013 in the South Nation Watershed was 393.3 ± 3.6 ppm and thus was rounded to 400 ppm as the difference between 393 and 400 ppm was insignificant

for the resulting calculations. $[\text{CO}_2]_{\text{emitted}}$ is calculated as $([\text{CO}_2]_{\text{measured}} - [\text{CO}_2]_{\text{atm}})$ in order to give how much CO_2 was emitted from the soil.

$$\text{[2]} \delta^{13}\text{C}_{\text{emitted}} = [(\delta^{13}\text{C}_{\text{measured}} \times [\text{CO}_2]_{\text{measured}}) - (\delta^{13}\text{C}_{\text{atm}} \times [\text{CO}_2]_{\text{atm}})] / [\text{CO}_2]_{\text{emitted}}$$

$$\text{[3]} [\text{CO}_2]_{\text{measured}} = (\text{slope}_{\text{CO}_2 \text{ vs time}} \times \text{time}_{\text{CO}_2 \text{ sample}}) + [\text{CO}_2]_{\text{atm}}$$

2.3 Isotopic analysis of soils and plants

Soil samples (0-15 cm, 15-30 cm, and 30-60 cm depth) were collected once a month from all fields with a hand auger (0.076 m length and 0.047m ID cores) for analysis of $\delta^{13}\text{C}$ of SOM. Soil sampling for field pair 1 and 2 in 2012 and 2013 was made at three roughly equidistant locations down the middle of each field from both OT and BT locations (**Figure 1**). Soil sampling for Fields 11-14 for 2012 and 2013 were made from two OT and two BT locations (**Figure 1**). SOM was prepared for $\delta^{13}\text{C}$ analysis using the method described by Brodie et al. (2011). Briefly, the soil was dried, ground, placed in silver capsules and acidified with HCl to remove carbonates. Soil samples were analyzed for $\delta^{13}\text{C}$ on the IRMS at the G.G. Hatch Stable Isotope Lab, University of Ottawa.

Just prior to harvest, plant samples were randomly collected from OT and BT locations from all fields. The plants were separated into their parts with soybeans separated into beans and rest of plant in 2012 compared to pods, leaves, root, and stem in 2013. Corn was separated into cob, leaves, seeds, root, and stem. Forage was either the whole plant (i.e., plant with roots) or just the plant (i.e., plant with no roots). The plants were dried at 80 °C, and finely ground (<100 mesh). Two grams from each plant sample were mixed for composite samples and the plant samples were analyzed for $\delta^{13}\text{C}$ on the IRMS at the G.G. Hatch Stable Isotope Lab, University of Ottawa. In fields 1 and 2 2012,

1 and 2 2013, and 11-14 2013, each sample analyzed for $\delta^{13}\text{C}$ was a composite sample from a mixture of 30 plants, 4 plants, and 6 plants, respectively. In field 11 and 14 2012, each forage sample (plant with no roots) was collected from an operational area of 0.38 m², where samples analyzed for $\delta^{13}\text{C}$ from 1st, 2nd, 3rd, and 4th cut were a composite from plants collected in 3, 2, 2, and 3 operational areas respectively. In addition, from the 4th forage cut, one forage sample (plant with roots) for OT and one for BT (each a composite of 2 plants) were analyzed per field.

2.4 Measurements of environmental variables

Each water table monitoring site (**Figure 1**) consisted of a 3 m deep groundwater well composed of perforated PVC pipe (0.05 m ID) wrapped in geotextile cloth and located midway between tile drains. Each groundwater well contained a level logger that consisted of a pressure transducer water level sensor and datalogger combination (WL16 Water Level Loggers, Global Water Instruments, Inc., Gold River, CA), which was used to measure the water table depth below surface (WTDBS) at 15 minute intervals throughout the growing season.

Soil volumetric water content (0-20 cm depth) (SWC20), soil temperature (0-10 cm depth) (ST10), and air temperature were measured within 0.5 m of each GHG sampling chamber at time of sampling (1000-1400 h). SWC20 of the soil was measured using the ESI (Environmental Sensors Inc., Moisture Point, Model MP-917, Victoria, BC) time domain reflectometry (TDR) system. ST10 and air temperature were measured using a Fluke 52-2 thermocouple thermometer and temperature probe (Fluke Corporation, WA).

Two weather stations (HOBO Weather Station Data Logger, Onset Computer Corp., MA), located in experimental Field 14 and Field 1 measured precipitation, soil temperature (0-15 cm depth) (ST15), and air temperature at 30 minute intervals throughout the growing season.

2.5 Statistical analysis

Two sample t-tests (Minitab 17) were used to determine if the means of $\delta^{13}\text{C}$ of R_T among CTD and UTD fields were different on a field pair basis. Also, two sample t-tests were used to compare mean differences for environmental variables (ST10, SWC20, and WTDBS) among CTD and UTD fields in the same way. In addition, t-tests were used to compare means of OT and BT $\delta^{13}\text{C}$ of R_T within each CTD field.

3. Results and discussion

3.1 Weather conditions and environmental variables

While the air temperature was similar during both studied growing seasons, the 2012 growing season was drier whereas the 2013 growing season was wetter (**Figure 2**). Total rainfall was 460 mm and 281 mm during the 2013 and 2012 growing seasons, respectively. There was 24% less and 25% greater precipitation in 2012 and 2013, respectively, compared to the average 30-year rainfall (1981-2010) for the approximate growing season (June-September = 369 mm) (EC, 2015) (**Figure 3**). The average ST15 for the growing season, from the weather station in Field 1, was similar in 2012 ($17.8\text{ }^\circ\text{C} \pm 2.6$) and 2013 ($17.4\text{ }^\circ\text{C} \pm 2.1$).

There was a significant difference ($p < 0.05$) for the average WTDBS between all CTD and UTD field pairs, on a field pair basis for each growing season (**Table 2**). In the

2013 growing season, CTD fields 2, 11, and 12 had a smaller average WTDBS compared to their UTD field pairs. On GHG sampling days, there was no significant difference in the daily average WTDBS between CTD and UTD field pairs in 2012, whereas in the wetter 2013 growing season, there was a significant difference ($p < 0.05$) where CTD fields had a shallower WTDBS (**Table 3**).

SWC20 and ST10 measured within 0.5 m of each GHG sampling chamber on sampling days at time of sampling were not significantly different ($p > 0.05$) between all CTD and UTD field pairs for both years of study (**Table 3**). Similarly, means of SWC20 and ST10 were not significantly different ($p > 0.05$) among OT and BT locations in all CTD fields during GHG sampling times (**Table 4**). It was expected that a shallower WTDBS in CTD fields would result in a greater SWC20 compared to UTD fields, but this was not the case. Mean SWC20 values in CTD fields 2, 11, and 12 in 2013 were 32.20 ± 4.05 , 33.80 ± 4.51 , and 31.57 ± 3.85 respectively compared to UTD fields 1, 14, and 13 in 2013 were 31.78 ± 4.50 , 33.34 ± 5.16 , and 31.64 ± 4.12 respectively.

We suggest three hypotheses as possible explanations for why SWC20 did not vary among CTD and UTD field pairs. First, it is hypothesized that there could have been greater root water uptake by crops in CTD fields thus causing no significant difference in SWC20 among CTD and UTD fields. This contention is supported in the literature in that a shallower water table depth has been shown to increase transpiration rates and root water uptake by vegetation (Askri et al., 2014; De Silva et al., 2008; Li et al., 2015). Ng et al. (2002) found 50% greater transpiration rates from corn in a CTD treatment, which contained a shallower water table depth, than corn in an UTD treatment. Furthermore, Mejjia et al. (2000) suggested that higher water tables in CTD fields allowed greater water

uptake by crops resulting in greater yields in CTD plots compared to UTD plots. Although in the current study, there was no significant difference in biomass among the CTD and UTD field pairs in 2012 and 2013 (Kross et al., 2015), thus likely refuting this first hypothesis of greater crop water uptake in CTD fields. Two other viable hypotheses are proposed. In 2012, the water table depths were greater than 1.6 m (see **Table 2**), which may have been too deep to see a management effect thus resulting in similar surface soil water contents among CTD and UTD fields. In the 2013 growing season, there was 25% greater precipitation compared to the average 30-year rainfall (1981-2010) for the approximate growing season (June-September) (EC, 2015), and perhaps the precipitation was frequent enough that surface soils were not deficit but rather were wet to a point that provided for limited differentiation between CTD and UTD field pairs.

3.2 Soil organic matter and plant $\delta^{13}\text{C}$

The $\delta^{13}\text{C}$ of SOM was consistent over time for CTD and UTD fields. The average $\delta^{13}\text{C}$ of SOM over both years of study was -22.25 ± 1.09 ‰ for Field 1 and 2, and -24.79 ± 0.58 ‰ for Fields 11-14. Average $\delta^{13}\text{C}$ of SOM remained relatively constant temporally in each field throughout the growing seasons indicating that the plants did not affect the $\delta^{13}\text{C}$ of SOM (**Table 5**). As the $\delta^{13}\text{C}$ of SOM resembles the $\delta^{13}\text{C}$ of the plant material from which it was derived, the SOM of Field 1 and 2 had greater carbon inputs from previous C_4 plants compared to Fields 11-14, which are in the range for SOM from C_3 plants (-24 ‰ to -29 ‰) (Cheng, 1996; Gregorich et al., 1995).

The average $\delta^{13}\text{C}$ of corn, soybean and forage for all fields and years of study were -12.37 ± 0.89 ‰, -27.80 ± 0.64 ‰, and -29.09 ± 0.97 ‰ respectively. The $\delta^{13}\text{C}$ values of

the plants were expected since the results for corn were in the common range for C₄ plants (~-11 to -15 ‰) whereas forage and soybean were in the common range for C₃ plants (~-26 to -30 ‰) (O'Leary, 1988). Each part of the plant had a similar $\delta^{13}\text{C}$ value, and the $\delta^{13}\text{C}$ values of soybean plants and corn plants are presented in **Table 6** and **Table 7** respectively. Each part of the plant had a similar $\delta^{13}\text{C}$ value as expected. Rochette et al. (1999a) found the $\delta^{13}\text{C}$ of corn leaves ($-13.44 \pm 1.34\text{‰}$), stems ($-12.12 \pm 0.93\text{‰}$), and roots ($-13.65 \pm 0.86\text{‰}$) had similar $\delta^{13}\text{C}$ values. The forage plants had an average $\delta^{13}\text{C}$ of $-28.87 \pm 1.07 \text{‰}$ in Field 11 and $-29.31 \pm 0.86 \text{‰}$ in Field 14.

3.3 $\delta^{13}\text{C}$ of soil-respired CO₂

The mean $\delta^{13}\text{C}$ of R_T did not differ ($p>0.05$) between CTD and UTD field pairs throughout the two growing seasons, except for one field pair (11 & 14 in 2012) (**Table 8**) where CTD Field 11 resulted in a more enriched $\delta^{13}\text{C}$ of R_T compared to UTD Field 14 (**Figure 4**). The significant difference in $\delta^{13}\text{C}$ of R_T between CTD Field 11 and UTD Field 14 in 2012 ($p<0.05$) may have been the result of different proportions and sizes of timothy and alfalfa within the chambers (both C₃ plants). The $\delta^{13}\text{C}$ of R_T can be used to reveal the source of the respired carbon (e.g., Drewitt et al., 2009; Rochette et al., 1999a, 1999b), and as the $\delta^{13}\text{C}$ of R_T did not significantly vary between CTD and UTD fields, except for one field pair, this suggests that CTD did not affect the contribution of the carbon substrate being oxidized during these contrasting growing seasons.

The absence of a statistically significant difference between the $\delta^{13}\text{C}$ of R_T from CTD and UTD fields was not expected. The $\delta^{13}\text{C}$ of R_T is affected by biological changes in the produced CO₂ and physical changes that alter gas transport (Cerling et al., 1991;

Clark, 2015; Phillips et al., 2010). The outward diffusion from high concentration of CO₂ in soils to low concentration of CO₂ in the atmosphere results in an approximate fractionation of 4.4 ‰ because heavier ¹³CO₂ diffuses slower out of the soil than the lighter ¹²CO₂ (Cerling et al., 1991; O'leary, 1988; Phillips et al., 2010). At a diffusive steady-state, the δ¹³C of R_T equals the δ¹³C produced within the soil (Cerling et al., 1991), but at non-steady state conditions caused by environmental changes, such as changes in soil moisture, the amount of CO₂ produced and the gas diffusivity through the soil are affected, which may affect the magnitude of fractionation (Phillips et al., 2010). Soils with a high moisture content have reduced CO₂ diffusion, reduced invasion of atmospheric CO₂, and greater rates of biological CO₂ production thus resulting in greater CO₂ concentrations and more depleted δ¹³C of R_T in relation to similar soils with low moisture (Cerling et al., 1991; Denny, 1993; Phillips et al., 2010). As supported through greenhouse experiments and modeling, the δ¹³C of R_T was found to be more depleted in high soil moisture treatments compared to low soil moisture and was likely influenced by gas transport (Phillips et al., 2010, 2008). As such, it was expected that the reduced drainage in CTD fields would create higher soil moisture conditions causing reduced CO₂ diffusion, reduced invasion of atmospheric CO₂, and greater rates of biological CO₂ production thus resulting in greater R_T concentrations and more depleted δ¹³C of R_T in relation to UTD fields with lower soil moisture.

The lack of a significant difference in means of δ¹³C of R_T between CTD and UTD fields is believed to have resulted from effectively indifferent surface soil water contents from bulk measurements (including OT and BT) among CTD and UTD field pairs. It was expected that CTD would increase SWC₂₀ by virtue of reducing drainage, as supported

in Nangia et al. (2013) who observed significantly higher soil moisture (0-30 cm depth) in CTD fields relative to UTD fields. Similarly, Ng et al. (2002) observed greater soil moisture content (0-120 cm depth) in CTD treatment than UTD treatment. However, plant transpiration is a complex process and its rate depends on several factors in the soil (e.g., soil water), plant (e.g., root-shoot ratio, leaf area and number of stomata, leaf cuticle structure, stomatal movement), and atmosphere (e.g., light, temperature, humidity, wind) (Novák et al., 2005; Zhang et al., 2008), but transpiration is often limited by soil water content in the root zone (Denmead and Shaw, 1962). For example, under conditions of water stress, atmospheric factors affected maize transpiration whereas under conditions with high soil water content, atmospheric factors had almost no effect on transpiration rate (Novák et al., 2005). Therefore, the retained water in CTD fields supports more transpiration, and this net result of greater transpiration may have resulted in surface soil water contents (i.e., SWC20) in CTD fields being effectively similar to those of UTD fields. Thus, this is believed to be one reason why we observed a dampening of measurable differences in SWC20 among CTD and UTD fields. Although in the current study, there was no significant difference in biomass among the CTD and UTD field pairs in 2012 and 2013 (Kross et al., 2015), thus likely refuting this hypothesis of greater crop water uptake in CTD fields. Two other viable hypotheses are proposed. In 2012, the water tables may have been too deep to see a management effect thus resulting in similar surface soil water contents among CTD and UTD fields. In the 2013 growing season, perhaps the precipitation was frequent enough that surface soils were not deficit but rather were wet to a point that provided for limited differentiation between CTD and UTD field pairs.

The $\delta^{13}\text{C}$ of R_T varied temporally throughout the growing season in all fields (**Figure 4**). The isotopic composition of R_T is not constant in soils but rather it changes when the concentration of CO_2 in soils change (Cerling et al., 1991). The concentration of CO_2 in soils changes over time as it is mainly affected by the soil respiration rate, its permeability, and the amount of atmospheric CO_2 that enters the soil (Rochette et al., 1999a). Therefore, in this study, changes in the $\delta^{13}\text{C}$ of R_T throughout the growing seasons are the result of changes in rates of biological CO_2 production that were caused by either a change in soil respiration rate and/or a change in the soil's permeability. A change in the soil's permeability may have been caused by a change in soil moisture that in turn affected CO_2 diffusion and the amount of atmospheric CO_2 entering the soil.

Within each field, there are clear trends of increasing contributions of the crops as a source for decomposition while the plants mature (**Figure 4**). In field pair 1 and 2, 2013, the $\delta^{13}\text{C}$ of R_T begins at its most depleted value (~ -25 ‰), which resembles the $\delta^{13}\text{C}$ of the soil, and then becomes gradually more enriched, due to the increasing contribution from corn at ~ -12 ‰, until mid-July where it remains relatively stable (~ -17 ‰) for the remainder of the growing season. For soybean (fields 1 & 2 in 2012, and fields 11-14 in 2013), the $\delta^{13}\text{C}$ of R_T generally begins more enriched due to contribution from enriched soil (~ -22 and ~ -25 ‰, respectively) then decreases as more inputs from depleted soybean (~ -27 ‰) and finally more enriched at the end of the growing season due to greater inputs from soil. For forage, there is a lot of sample to sample variation in $\delta^{13}\text{C}$ of R_T because the values are based on at most two samples and sometimes only one. It appears that at the beginning of the season the $\delta^{13}\text{C}$ of R_T reaches a max enrichment just after the cutting on July 10 and August 20 (enriched due to greater contribution from soil

at $\delta^{13}\text{C}$ \sim -25 ‰ and less from forage $\delta^{13}\text{C}$ \sim -29 ‰), but no change after October 24 cut. Also while there appears to be an enrichment in $\delta^{13}\text{C}$ of R_T on September 11 this is just based on one sample.

The $\delta^{13}\text{C}$ of R_T was used to determine whether CTD management had a tile location effect on R_T . It was expected that the location in a field with respect to a tile drain would alter residue decomposition and soil gas diffusivity through impacts on soil properties including altered soil hydrology due to water table mounding (Smedema et al., 1983) that can occur between tile drains, different soil chemical/physical properties over tiles (trenches backfilled with disturbed soil during time of tile installation), and biopores that can preferentially occur over tile drains (in relation to between them) (Nuutinen et al., 2001). Contrary to expectation, the means of $\delta^{13}\text{C}$ of R_T did not differ between OT and BT locations in all CTD fields ($p > 0.05$) except for Field 11 in 2012, where BT had a more enriched $\delta^{13}\text{C}$ of R_T compared to OT (**Figure 5**). The observed difference in $\delta^{13}\text{C}$ of R_T from OT and BT locations within one CTD forage field is believed to be because of significant variations in emissions due to random distribution of two crops (alfalfa and timothy) being present in the field. It is believed that there was no tile location effect on $\delta^{13}\text{C}$ of R_T in CTD fields because there was no significant difference ($p < 0.05$) in SWC20 and ST10 between OT and BT locations in all CTD fields during GHG sampling times (**Table 4**). It is hypothesized that SWC20 did not vary for OT and BT locations in CTD fields because of three possible hypotheses. Firstly, there could have been greater root water uptake in BT locations, as root water uptake has been shown in the literature to be sensitive to fluctuations in the water table depth below surface (Askri et al., 2014; De Silva et al., 2008; Li et al., 2015). Secondly, the water table depths may have been too deep in

2012 to see an OT and BT management effect. Thirdly, perhaps the precipitation was frequent enough that OT and BT surface soils were not deficit but rather wet to a point that provided for limited differentiation. Apparently similar soil physical conditions (e.g., ST10 and SWC20) among OT and BT locations within CTD fields reflects minimal impact of tile location on CO₂ diffusion, invasion of atmospheric CO₂, or rates of CO₂ production. This result suggests, for drier and wetter seasons, that sampling during growing seasons could be accomplished anywhere in the field, and that preferential sampling between and over tile is unnecessary.

4. Conclusion

Within the present study, two main research hypotheses were addressed. Firstly, the $\delta^{13}\text{C}$ of soil-respired CO₂ (R_T) was analyzed from controlled tile drained (CTD) and uncontrolled tile drained (UTD) fields cropped with corn, soybean, and forage to test whether CTD affects R_T through altering residue decomposition or gas transport. Secondly, we examined whether CTD imparted an over tile (OT) and between tile (BT) locational effect on R_T .

We expected that the reduced drainage in CTD fields would create higher soil water contents resulting in greater rates of biological CO₂ production and more depleted $\delta^{13}\text{C}$ of R_T in relation to UTD fields. We expected a more depleted $\delta^{13}\text{C}$ of R_T in CTD fields compared to UTD fields because of reduced CO₂ diffusion and reduced invasion of atmospheric CO₂. Contrary to expectation, the means of $\delta^{13}\text{C}$ of R_T were not statistically different ($p>0.05$) between four CTD and UTD field pairs but it was different for one field pair. This finding of a significant difference in $\delta^{13}\text{C}$ of R_T between a CTD and UTD forage

field pair may have been the result of significant variations in emissions due to random distribution of two crops (alfalfa and timothy) being present in the field. The lack of difference in mean $\delta^{13}\text{C}$ of R_T between CTD and UTD field pairs is believed to have resulted from their being effective indifference in surface soil water contents among CTD and UTD fields. Three possible reasons explain why this may have occurred. Firstly, there is potential that reduced drainage in CTD fields could have supported greater root water uptake by crops thus causing no significant difference in surface soil water contents among CTD and UTD fields. Secondly, the water table was too low during 2012 for effective water table control, and so both treatments (i.e., CTD and UTD) were indifferent with respect to surface soil water contents. Thirdly, during the wetter year (i.e., 2013), precipitation may have been significant enough to promote equally wet surface soil condition among both CTD and UTD fields. As there was no significant difference in biomass among the CTD and UTD field pairs in 2012 and 2013 (Kross et al., 2015), the first hypothesis is likely refuted thus leaving the two other hypotheses as viable options. These explanations may have had variable effect on the lack of difference in $\delta^{13}\text{C}$ of R_T among OT and BT locations in all but one CTD field. These findings suggest that there may be no requirement for systematically locating GHG sampling sites with respect to location of tile drains in field, at least for medium textured soils such as those examined.

While there were no significant differences in $\delta^{13}\text{C}$ of R_T among CTD and UTD fields, and among OT and BT locations within CTD fields, it is noteworthy that we measured weekly fluxes during the growing season (~planting to harvest) when CTD was employed. By sampling weekly, we could have missed important emitting or non-emitting events. Also, soil water content was sampled at time of sampling and perhaps we could

have missed some important transient events. By managing field drainage, there is potential for higher water tables and higher soil water contents that could create favourable conditions for R_T ; yet our experimental findings indicate that CTD did not affect R_T . While our study was done during growing seasons that were approximately 24% drier and 25% wetter than the 30-year normal for the area, it would be worthwhile to examine whether CTD affects $\delta^{13}C$ of R_T in a growing season with average rainfall.

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7. Tables

Table 1 Planting, harvest, photosynthetic pathway, and key tile drainage management dates for the studied control tile drained (CTD) and uncontrolled tile drained (UTD) agricultural fields.

Field pair	CTD field: area (ha)	UTD field: area (ha)	Year	Planting date: crop type	Date stop gates were closed for CTD fields	Date stop gates were opened for CTD fields	Harvest date
1 & 2	Field 2: 2.3	Field 1: 2.0	2012	May 16: soybean	May 16	September 19	September 24
			2013	May 5: corn	May 29 ^a	September 26	November 14
11 & 14	Field 11: 4.2	Field 14: 4.1	2012	In 2011: forage	May 23	November 19	June 1, July 10, August 20, October 24
			2013	May 15: soybean	May 27	September 26	October 25
12 & 13	Field 12: 5.0	Field 13: 4.2	2013	May 15: soybean	May 27	September 26	October 25

^a Gates were left open for a longer time because farmer wanted them open due to excessive rainfall

Table 2 Average \pm standard deviation for water table depth below surface (m) for uncontrolled tile drained (UTD) and control tile drained (CTD) field pairs in 2012 and 2013. P values are presented for t-tests.

Field Pair	UTD	CTD	P value
2012			
Field 1 & 2	1.573 \pm 0.417	1.593 \pm 0.407	0.000
Field 11 & 14	2.690 \pm 0.628	2.491 \pm 0.672	0.000
2013			
Field 1 & 2	1.115 \pm 0.301	0.966 \pm 0.406	0.000
Field 11 & 14	1.277 \pm 0.380	1.070 \pm 0.446	0.000
Field 12 & 13	1.413 \pm 0.318	1.266 \pm 0.415	0.000

Table 3 Average \pm standard deviation for soil moisture (0-20 cm depth), soil temperature (0-10 cm depth), and daily water table depth below surface for control tile drained (CTD) and uncontrolled tile drained (UTD) field pairs from greenhouse gas sampling days during the 2012 and 2013 growing seasons. P values are presented for t-tests.

Year	Field pair	Soil moisture (%)			Soil temperature (°C)			Daily water table depth below surface (m)		
		CTD	UTD	P value	CTD	UTD	P value	CTD	UTD	P value
2012	1 & 2	20.49 \pm 8.04	20.65 \pm 8.67	0.942	20.30 \pm 3.36	20.03 \pm 3.11	0.739	1.61 \pm 0.46	1.62 \pm 0.44	0.923
	11 & 14	22.19 \pm 8.55	24.05 \pm 8.14	0.444	17.60 \pm 5.78	17.15 \pm 5.17	0.779	2.65 \pm 0.59	2.84 \pm 0.51	0.157
2013	1 & 2	32.20 \pm 4.05	31.78 \pm 4.50	0.326	18.33 \pm 4.00	19.05 \pm 4.76	0.242	1.10 \pm 0.31	1.21 \pm 0.18	0.002
	11 & 14	33.80 \pm 4.51	33.34 \pm 5.16	0.552	19.34 \pm 3.62	19.41 \pm 3.52	0.912	1.20 \pm 0.32	1.40 \pm 0.19	0.000
	12 & 13	31.57 \pm 3.85	31.64 \pm 4.12	0.918	19.32 \pm 2.90	19.36 \pm 2.82	0.928	1.48 \pm 0.19	1.57 \pm 0.11	0.001

Table 4 Average \pm standard deviation for soil moisture (0-20 cm depth) and soil temperature (0-10 cm depth) for over drainage tile and between drainage tile locations in control tile drained (CTD) agricultural fields throughout the 2012 and 2013 growing seasons. P values are presented for t-tests.

Year and crop	CTD field#	Soil moisture (%)			Soil temperature (°C)		
		Over tile	Between tile	P value	Over tile	Between tile	P value
2012							
Soybean	2	20.51 \pm 8.27	22.33 \pm 8.96	0.467	20.76 \pm 3.56	20.81 \pm 3.48	0.961
Forage	11	24.12 \pm 7.51	24.69 \pm 6.72	0.805	18.01 \pm 6.35	17.61 \pm 6.03	0.876
2013							
Corn	2	32.05 \pm 3.32	32.39 \pm 4.65	0.558	18.34 \pm 4.07	18.39 \pm 4.02	0.951
Soybean	11	33.41 \pm 4.24	33.33 \pm 4.18	0.902	19.39 \pm 3.86	19.40 \pm 3.80	0.989
Soybean	12	31.47 \pm 3.62	31.67 \pm 4.08	0.764	18.54 \pm 2.35	19.15 \pm 2.73	0.340

Table 5 Monthly average \pm standard deviation for $\delta^{13}\text{C}$ (‰) of soil organic carbon for all studied fields throughout the 2012 and 2013 growing seasons. The soil is from 0-60 cm depth. Note that Fields 11-14 in 2013 were sampled on October 1st not in September.

Month	Field 1		Field 2		Field 11		Field 14		Field 12	Field 13
	2012	2013	2012	2013	2012	2013	2012	2013	2013	2013
June	-22.42 \pm 1.32	-22.65 \pm 0.36	-22.68 \pm 1.30	-21.39 \pm 0.70	-25.32 \pm 0.41	-24.60 \pm 0.40	-25.14 \pm 0.25	-24.09 \pm 0.15	-25.33 \pm 0.30	-24.47 \pm 0.79
July	-22.36 \pm 0.71	-23.20 \pm 0.72	-22.16 \pm 0.92	-23.32 \pm 1.36	-24.75 \pm 0.26	-24.36 \pm 0.30	-24.55 \pm 0.14	-25.08 \pm 0.85	-24.64 \pm 0.59	-25.12 \pm 0.27
August	-22.08 \pm 0.53	-22.03 \pm 0.41	-21.47 \pm 0.95	-21.37 \pm 0.49	-24.97 \pm 0.37	-24.47 \pm 0.52	-24.80 \pm 0.42	-24.64 \pm 0.28	-24.96 \pm 0.28	-25.11 \pm 0.28
September	-22.14 \pm 0.66	-21.78 \pm 0.80	-22.44 \pm 1.00	-21.30 \pm 0.90	-25.11 \pm 0.44	-25.15 \pm 0.37	-23.92 \pm 1.27	-24.52 \pm 0.34	-24.61 \pm 0.67	-24.52 \pm 0.18
Total	-22.32 \pm 1.04	-22.41 \pm 0.80	-22.34 \pm 1.20	-21.84 \pm 1.22	-25.09 \pm 0.42	-24.65 \pm 0.49	-24.71 \pm 0.74	-24.58 \pm 0.58	-24.88 \pm 0.55	-24.81 \pm 0.53

Table 6 Average \pm standard deviation for $\delta^{13}\text{C}$ (‰) of soybean plants from Field 1 and 2 prior to harvest in 2012, and from Field 11-14 prior to harvest in 2013. In 2012, soybean pods were separated from the rest of the plant. In 2013, the plant was divided into its pods, leaves, root, and stem for analysis.

Part of Soybean Plant	2012		2013			
	Field 1	Field 2	Field 11	Field 12	Field 13	Field 14
Plant	-27.77 \pm 0.06	-27.06 \pm 0.52	ND	ND	ND	ND
Pods	-27.66 \pm 0.04	-27.09 \pm 0.40	-27.43 \pm 0.05	-27.65 \pm 0.32	-28.21 \pm 0.66	-28.10 \pm 0.90
Leaves	ND	ND	-28.64 \pm 0.37	-28.78 \pm 0.16	-28.96 \pm 0.17	-28.87 \pm 0.01
Root	ND	ND	-27.53 \pm 0.05	-27.40 \pm 0.34	-27.67 \pm 0.18	-27.54 \pm 0.52
Stem	ND	ND	-27.81 \pm 0.12	-27.40 \pm 0.03	-27.07 \pm 0.12	-27.45 \pm 0.32
Total	-27.71 \pm 0.07	-27.07 \pm 0.38	-27.85 \pm 0.53	-27.81 \pm 0.64	-27.98 \pm 0.79	-27.99 \pm 0.73

ND = not determined

Table 7 Average \pm standard deviation for $\delta^{13}\text{C}$ (‰) of five different parts of corn plants from Field 1 and 2 prior to harvest in 2013.












Part of Corn Plant	$\delta^{13}\text{C}$ (‰)			
	Field 1		Field 2	
Cob	-12.06	\pm 0.29	-11.28	\pm 0.18
Leaves	-14.07	\pm 1.56	-12.25	\pm 0.41
Seeds	-12.28	\pm 0.01	-11.73	\pm 0.08
Root	-12.91	\pm 0.52	-12.19	\pm 0.03
Stem	-12.92	\pm 1.17	-12.05	\pm 0.07
Total	-12.84	\pm 1.00	-11.90	\pm 0.41

Table 8 Average \pm standard deviation for $\delta^{13}\text{C}$ (‰) of CO_2 emissions from soil (combined rhizosphere and microbial sources) for control tile drained (CTD) and uncontrolled tile drained (UTD) fields during the 2012 and 2013 growing seasons. P values are presented for t-tests.

Field Pair	UTD	CTD	P value
2012			
Field 1 & 2	-21.08 \pm 2.05	-21.59 \pm 1.84	0.291
Field 11 & 14	-26.78 \pm 1.36	-23.86 \pm 3.60	0.000
2013			
Field 1 & 2	-19.01 \pm 2.99	-19.01 \pm 2.86	0.994
Field 11 & 14	-25.02 \pm 1.13	-24.57 \pm 1.24	0.102
Field 12 & 13	-24.78 \pm 1.16	-25.21 \pm 0.98	0.096

8. Figures

LEGEND

-  GHG monitoring site (2013)
-  GHG monitoring site (2012 - 2013)
-  GHG monitoring site (2012)
-  Soil sampling site
-  Drainage tiles
-  Drainage ditch
-  CTD outlet
-  UTD outlet
-  CTD field
-  UTD field
-  Water table monitoring site
- OT** Over drainage tile
- BT** Between drainage tile

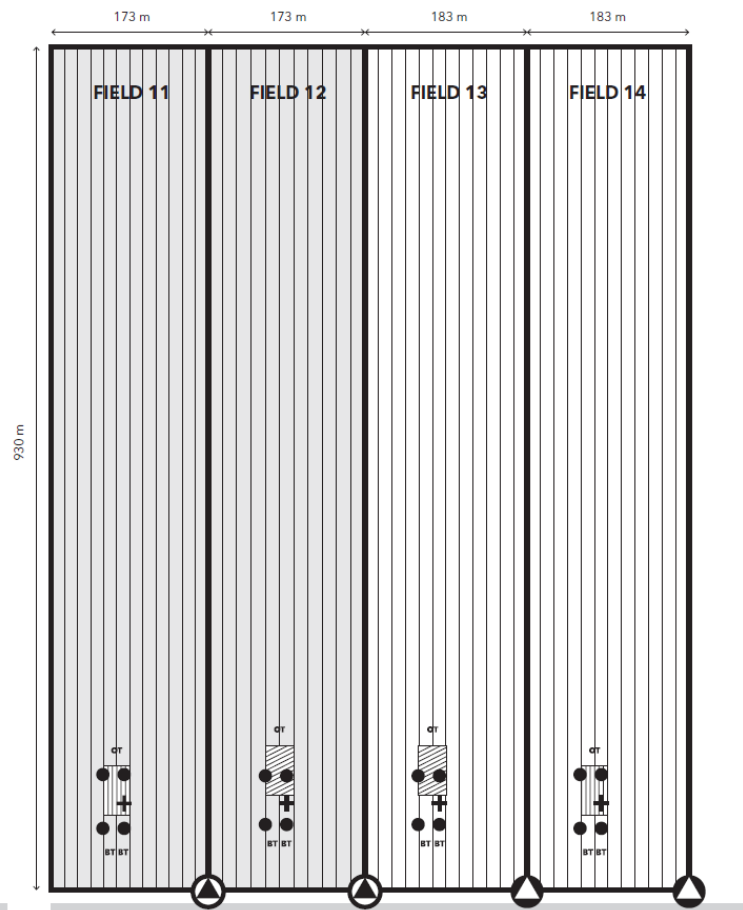
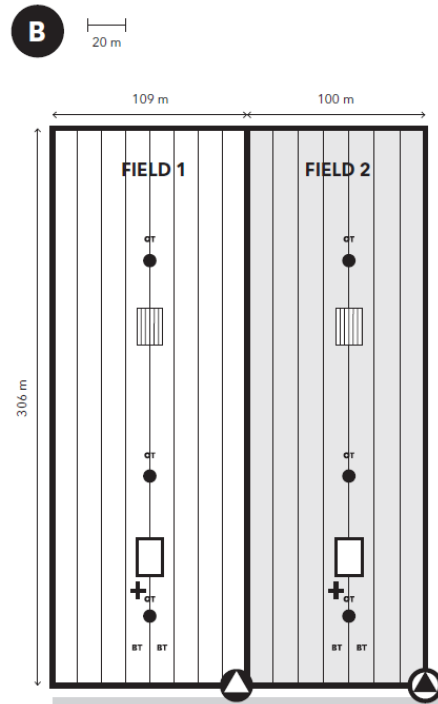


Figure 1 The location of the experimental watersheds in Eastern Ontario, Canada, (a) containing the experimental fields 1-2 (b) and 11-14 (c). The location of GHG sampling, soil sampling, and water table monitoring sites are indicated. Over drainage tile and between drainage tile locations are indicated.

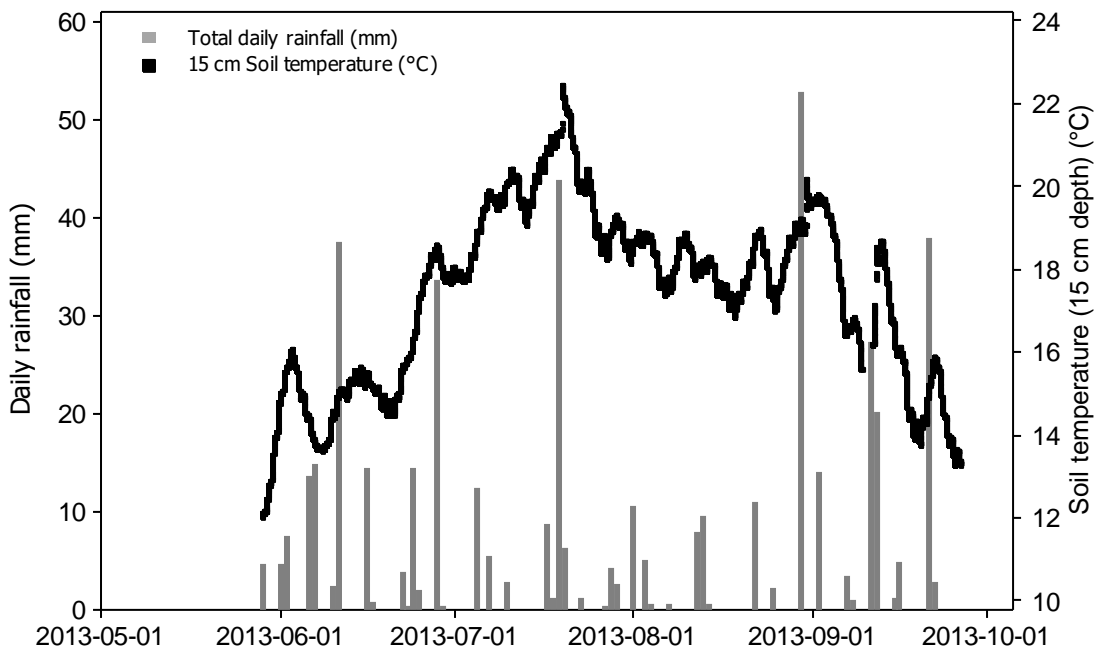
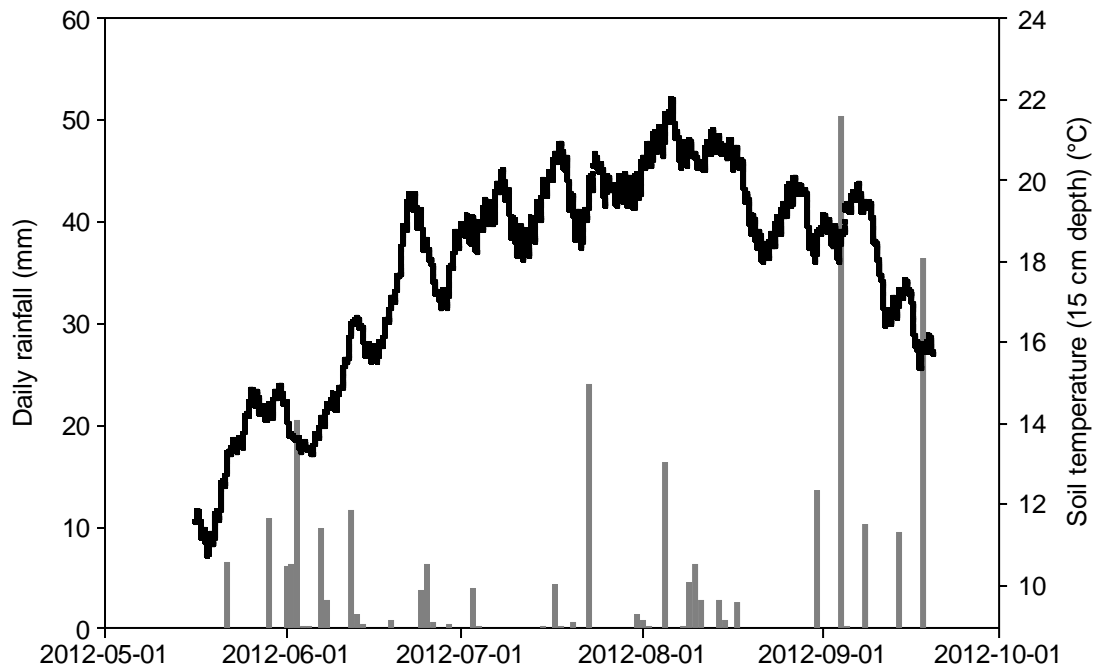


Figure 2 Total daily rainfall and soil temperature (0 - 15 cm depth) in 2012 and 2013, from the weather station in Field 1.

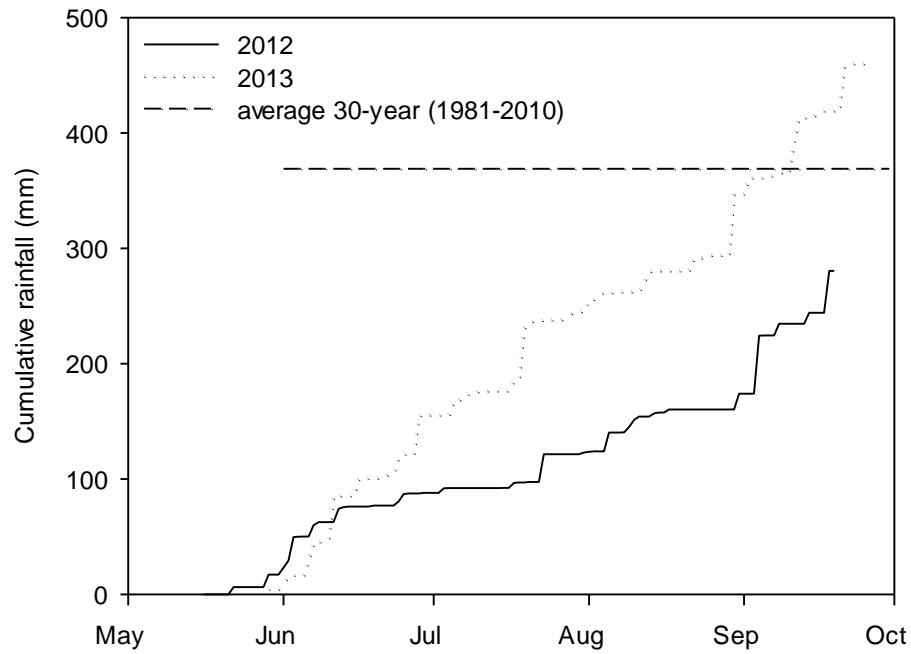


Figure 3 Cumulative rainfall for the 2012 and 2013 growing seasons (according to field pair 1 and 2) from a weather station in Field 1. The cumulative average 30-year rainfall (1981-2010) for the approximate growing season (June-September) is presented (EC, 2015).

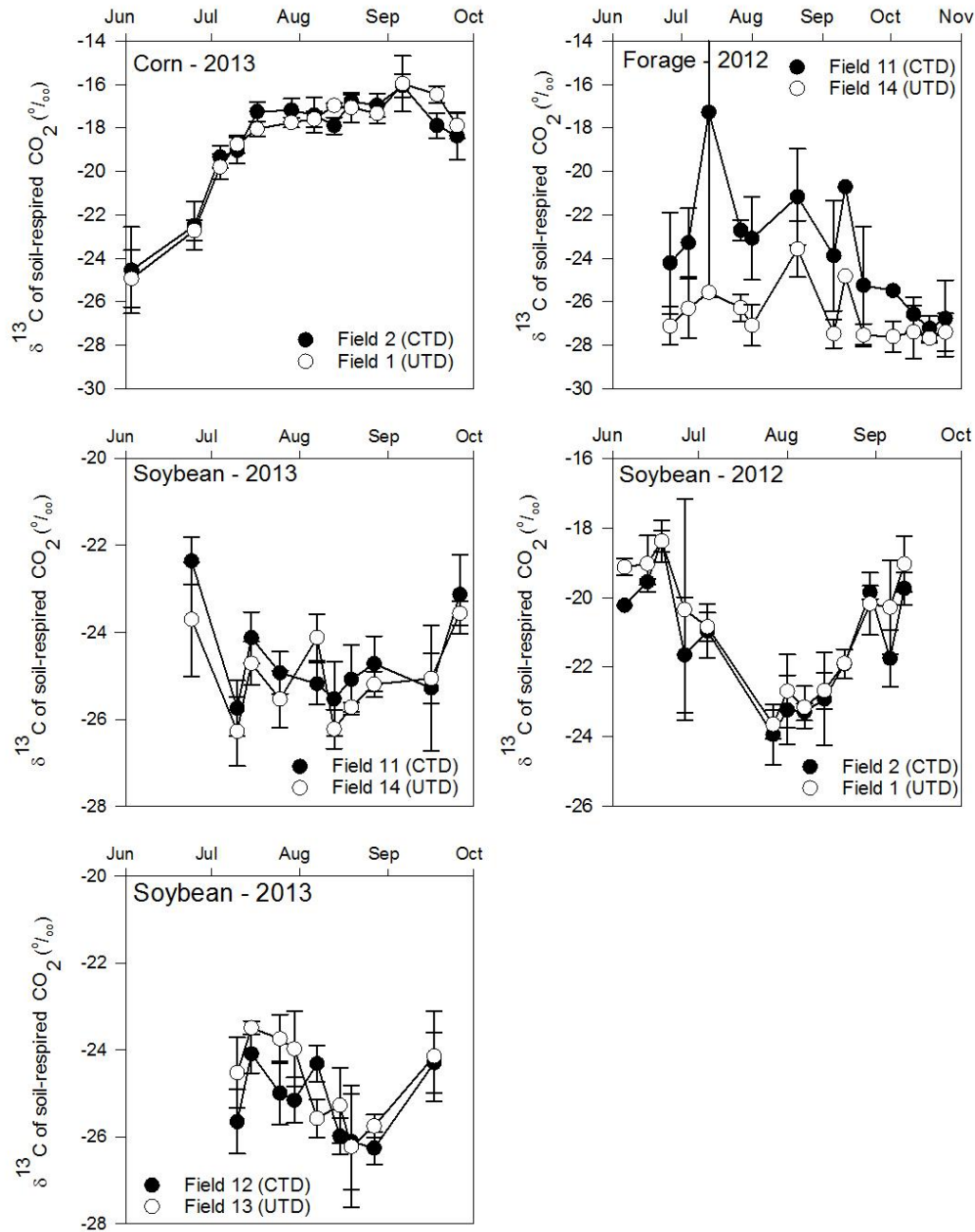


Figure 4 Average $\delta^{13}\text{C}$ of CO_2 emissions from soil (combined rhizosphere and microbial sources) for control tile drained (CTD) and uncontrolled tile drained (UTD) agricultural fields throughout the 2012 and 2013 growing seasons. Error bars indicate standard deviations.

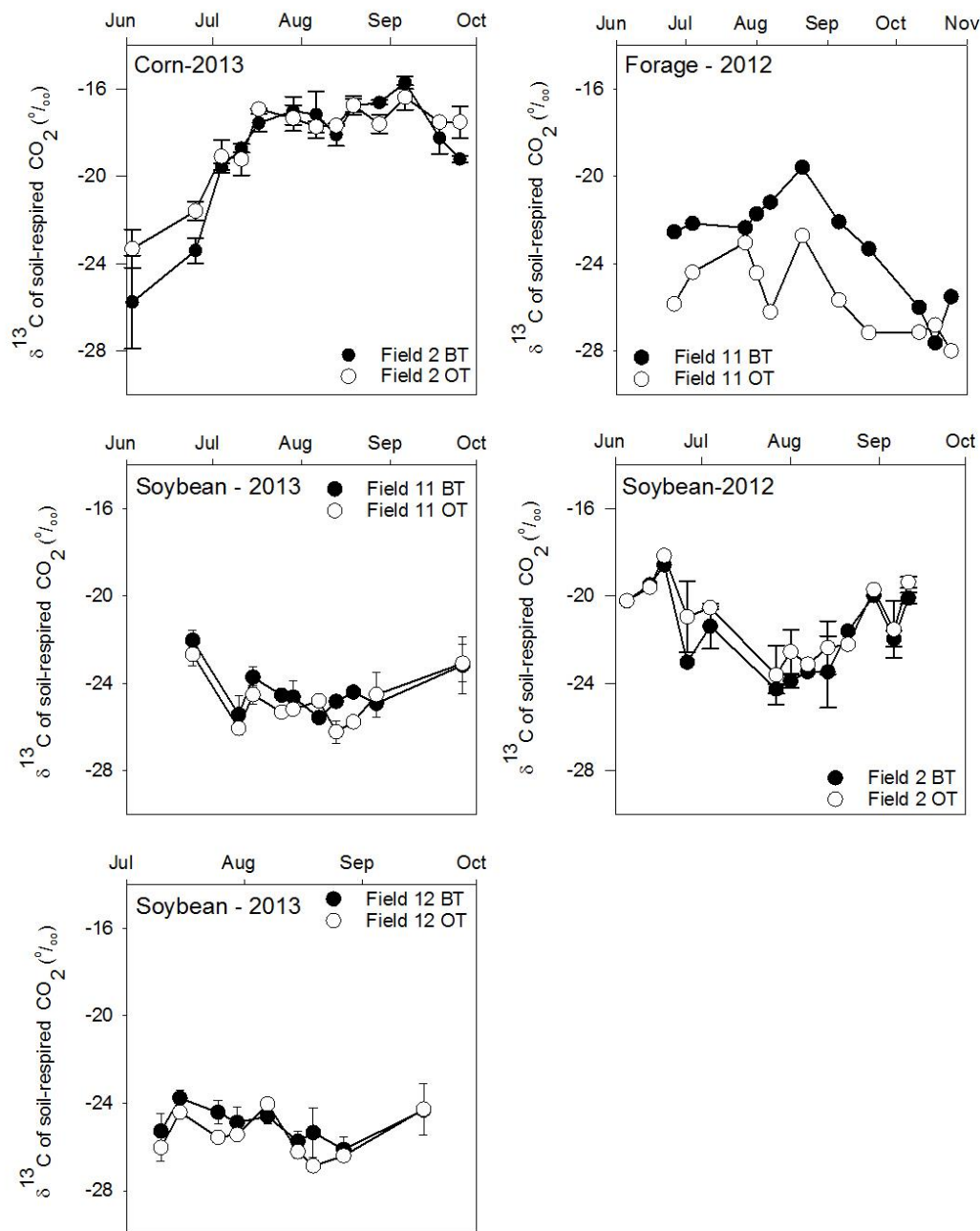


Figure 5 Average $\delta^{13}\text{C}$ of CO_2 emissions from soil (combined rhizosphere and microbial sources) for over drainage tile (OT) and between drainage tile (BT) locations in control tile drained agricultural fields throughout the 2012 and 2013 growing seasons. Error bars indicate standard deviations. Note that forage in 2012 has no error bars as only one sample was measured for each OT and BT location.

Chapter 5 Identifying whether control tile drainage affects the contributions of rhizosphere and soil respiration to total soil respiration in corn and soybean fields

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Chapter 5: Table of Contents

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Abstract

Controlled tile drainage (CTD) management is common in agricultural fields but its effect on residue decomposition is unknown. Quantitative estimates of the CO₂ produced by both rhizosphere respiration (R_{rh}) and soil respiration (R_s) in CTD fields are required in order to properly assess the environmental effects of CTD. In order to determine whether CTD affects the contribution of soil carbon pools (R_{rh} and R_s) to total soil respiration (R_T), soil CO₂ efflux from four field pairs, cropped with corn and soybean, were sampled from vegetated and non-vegetated areas throughout two growing seasons. Both the root-exclusion approach (root) and the natural abundance isotope ¹³C approach (iso) were used to separate R_T into its R_{rh} and R_s components. Within corn fields, there was no significant difference (p>0.05) for R_s, R_T, and R_{rh} between CTD and UTD fields. Similarly, there was no significant difference (p>0.05) for R_s, R_T, and R_{rh} between CTD and UTD soybean fields for two field pairs. One field pair resulted in CTD having a significant increase in the contribution of R_{rh} over R_s, although R_T remained the same. Thus, only one field pair out of the four pairs studied found that CTD did affect the contribution of R_{rh} and R_s but R_T remained the same. The estimated R_{rh} ranged up to 20.48 kg C/ha/day within corn fields, up to 33.75 kg C/ha/day within soybean fields in the wet growing season, and up to 23.23 kg C/ha/day within soybean fields in the dry growing season. In conclusion, CTD has not significantly impacted the carbon emissions in the soil-plant system, thus further supporting the use of CTD in agriculture.

1. Introduction

Terrestrial vegetation and soils represent the second largest reservoir of carbon (C), with inputs from photosynthesis and losses through soil respiration (R_s) (Clark 2015, Drewitt *et al* 2009, Hardy 2003). R_s is the process where soils release carbon dioxide (CO_2) to the atmosphere (Pal Singh *et al* 2011). In non-vegetated areas, R_s is the result of heterotrophic respiration from organisms decomposing soil organic matter (SOM) whereas in vegetated areas total soil respiration (R_T) originates from R_s and from rhizosphere respiration (R_{rh}), which is the sum of CO_2 respired by roots and by microbes that are using root derived C (Cheng 1996, Ryan & Law 2005, Rochette *et al* 1999, Pal Singh *et al* 2011, Rochette & Flanagan 1997). R_s releases approximately 10 times more CO_2 to the atmosphere than that released from fossil fuel combustion (Schlesinger 1997, Boden *et al* 2010). Therefore, the large amount of CO_2 released to the atmosphere from R_s (Rochette *et al* 1999) is of particular concern as it is contributing to its large atmospheric concentration (390.5 ppm) and its rapid rate of increase (average 2.0 ± 0.1 ppm yr^{-1} during 2002-2011) (IPCC 2013) thus promoting the greenhouse effect leading to climate change (Werth & Kuzyakov 2008).

Agricultural lands cover an estimated 40 to 50 % of the Earth's land surface (Smith *et al* 2007) and thus contribute a significant portion to greenhouse gas emissions (IPCC 2013). Many agricultural lands have artificial subsurface tile drainage, including 17.4 million ha in the Midwestern United States (Jaynes & Isenhardt 2014) and more than 1.6 million ha in Ontario, Canada (Sunohara *et al* 2015) are artificially drained. Artificial subsurface tile drainage is used in controlled tile drainage (CTD) management where the amount of water that can leave a field through artificial subsurface tile drains is

controlled (Gilliam *et al* 1979). CTD management contrasts with uncontrolled tile drainage (UTD) where there is no management of the water leaving a field through subsurface tile drains.

CTD involves increasing or decreasing the water table depth below surface (WTDBS) by adjusting the height of stop gates in a water flow control structure (Sunohara *et al* 2015, Kross *et al* 2015, Cicek *et al* 2010, Mejia *et al* 2000). By controlling the amount of tile discharge, CTD improves field drainage which has been shown to increase crop yields (eg. Delbecq *et al* 2012, Ghane *et al* 2012, Poole *et al* 2013, Skaggs *et al* 2012, Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Kross *et al* 2015, Mejia *et al* 2000, Cicek *et al* 2010) and reduce the amount of nutrient loss in drainage water (eg. Sunohara *et al* 2015, Cooke & Verma 2012, Drury *et al* 2009, Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Drury *et al* 1996). CTD may affect CO₂ emissions released to the atmosphere compared to UTD fields as one study found some significantly higher CO₂ fluxes associated with CTD fields (Nangia *et al* 2013) whereas another study found no change in CO₂ emissions (Van Zandvoort 2015). Furthermore, a study found that the source of CO₂ emissions does not vary between CTD and UTD fields (Van Zandvoort 2015), but quantitative estimates of the CO₂ produced by R_s and R_{rh} in CTD fields have not been documented. It is important to provide quantitative estimates of the CO₂ produced by each source (R_s and R_{rh}) in CTD fields in order to evaluate implications of CTD on the C dynamics in the soil plant system.

Two primary methods for distinguishing the contributions of R_{rh} and R_s are root-exclusion and the use of ¹³C isotope (¹³C/¹²C, δ¹³C) (Hanson *et al* 2000, Rochette *et al*

1999). The root-exclusion approach calculates R_{rh} by subtracting the CO_2 emission rates from soils without roots (R_s) from those of soils that contain roots (R_T) (Hanson *et al* 2000, Rochette *et al* 1999). If the $\delta^{13}C$ of plants varies from that of the soil it is growing in, then the ^{13}C isotope approach is valid and the R_{rh} can be quantified as the difference between ^{13}C of R_T (*ie* vegetated area) and ^{13}C of R_s (*ie* non-vegetated area) (Rochette & Flanagan 1997, Rochette *et al* 1999). The ^{13}C isotope method uses the difference in natural abundances of ^{13}C in C_3 and C_4 plants (Rochette & Flanagan 1997, Rochette *et al* 1999) as the inefficient step of CO_2 respiration in the C_3 pathway causes the $\delta^{13}C$ of C_3 plants to be much lower (~ -26 to -28 ‰) than that of C_4 plants who reduce photorespiration (~ -10 to -14 ‰) (Clark 2015, O'Leary 1988). The $\delta^{13}C$ of SOM resembles the $\delta^{13}C$ of the plant material from which it was derived such that SOM from C_4 plants have $\delta^{13}C$ values ranging from -12 ‰ to -14 ‰ compared to -24 ‰ to -29 ‰ for SOM from C_3 plants (Gregorich *et al* 1995, Cheng 1996). Many studies have used root exclusion (eg. Rochette *et al* 1999, Irvine *et al* 2008, Ruehr & Buchmann 2010) and ^{13}C (eg. Rochette *et al* 1999, Rochette & Flanagan 1997, Werth & Kuzyakov 2008), but while both root-exclusion and ^{13}C methods have advantages and disadvantages (Hanson *et al* 2000), in a study both approaches were found to result in similar R_{rh} values (Rochette *et al* 1999).

The objectives of this study are to: (a) determine the contributions of R_{rh} and R_s to R_T in CTD fields throughout two growing seasons, with contrasting precipitation records, cropped with corn and soybean, and (b) compare R_{rh} results from two approaches, the root exclusion and ^{13}C isotope method, and (c) evaluate implications of CTD on the C dynamics in the soil plant system.

2. Materials and methods

2.1 Site description

The fields studied are low-gradient (slope < 1%), located within the ~950 ha South Nation Watershed (45.26 N, 75.18 W) in eastern Ontario, Canada (Cicek *et al* 2010). This two year study (2012 – 2013) was conducted during the growing seasons (defined here as beginning when the stop gates were closed in CTD fields and lasting until they were reopened; **Table M1**) on subsurface tile drained agricultural fields containing corn and soybean. This study was done during the growing season because it is during this time that drainage management varies for CTD and UTD fields. This study examined one field pair in 2012 and three in 2013 (**Fig. M1**) with each pairing having fields in close proximity to each other and under a common cropping practice but varying in drainage management (one field was CTD and the other UTD) (**Table M1**). The standard deviation is always presented in conjunction with the mean.

Soil at the site is Bainsville silt loams (Wicklund & Richards 1962) with clayey soils underneath at ~1.0 - 1.5 m (Cicek *et al* 2010). According to nearby Russell station, the 30-year normal annual precipitation (1981-2010) is 981 mm and average daily air temperature is 6.5°C (EC 2015). From June to September (approximate growing season), average 30-year precipitation (1981-2010) and average daily air temperature, according to Russel station, is 369 mm and 18 °C respectively (EC 2015). Total growing season rainfall was 281 mm and 460 mm in 2012 and 2013 respectively and average soil temperature (0-10 cm depth) (ST10) during the growing season was 17.8°C ± 2.6 and 17.4°C ± 2.1 in 2012 and 2013 respectively.

All fields studied are tile drained with lateral tiles (0.102 m in diameter) located at a depth of ~1 m and with a spacing of ~15 m. The lateral tile drains connect to a header tile (0.152 m in diameter) which connects to a main outlet containing a tile drainage control structure for each field. In CTD fields, tile drainage control structures were used to manage tile drain flow by setting the water overflow depth in the control structures to 0.4 m below the surface. In UTD fields, the control structures were not used to manage tile drain flow.

2.2 Field sampling of soil-respired CO₂

Soil-respired CO₂ sampling was carried out using the nonsteady-state chamber design and method described by Rochette & Bertrand 2008, which was also similarly used by Nangia *et al* 2013. Briefly, chamber bases were installed in the fields at ~10 cm depth (and ~5 cm head space) and left there for the growing season. Rectangular chambers (0.75 x 0.15 m) were used in soybean fields, and square chambers (0.35 x 0.35 m) were used in corn fields, and both types of chambers had flat lids. Plastic chamber lids contained a rubber septa sampling port on the top and a foam gasket seal on their underside. Chambers were located in vegetated and non-vegetated areas. In vegetated areas, chambers were located between crop rows with roots and crops growing next to chambers. In non-vegetated areas, chambers were located between crop rows and the crops (including roots) were removed by hand for an approximate 1 m distance around the chamber. The CO₂ flux within vegetated locations provides a value for R_T (*ie* $R_{rh} + R_s$) whereas non-vegetated areas give a measure of R_s . Headspace volume and surface area of chambers were measured monthly.

Sampling of CO₂ from soil-atmosphere was mostly carried out between 1000-1400 h (peak emission times) at weekly intervals from over tile (OT) and between tile (BT) locations on both fields of each field pair. CO₂ soil-atmosphere sampling involved capping the chamber, and using a 30 mL syringe (fitted with a 26-gauge needle) to immediately collect a 30 mL sample of the chamber headspace at time zero. During chamber deployment, four samples were collected from each chamber's headspace and were analyzed for CO₂ concentration. GHG from soybean and corn fields were sampled at 0, 6, 12, and 18 minutes. A fifth CO₂ sample from each chamber was collected at 24 minutes and these were analyzed for $\delta^{13}\text{C}$ of CO₂. The sampled air was injected into 12 mL septum-capped glass vials (Extainers, Labco Ltd.) which were prepared prior to the field by: adding an additional silicone septum on top of the rubber septum, adding 2-3 mg of magnesium perchlorate as a drying agent, and evacuating the vials (Rochette & Bertrand 2008).

2.3 Analyses of CO₂ efflux samples

2.3.1 CO₂ concentration

The samples of CO₂ from soil-atmosphere that were collected for analysis of CO₂ concentration were run on the Varian CP-3800 gas chromatograph with CombiPal autosampler with helium as the carrier gas. The injection volume was 5 mL and a flame ionization detector was used. Samples of reference standard gases were handled the same way as gas samples and approximately one was inserted for every five gas samples during the analysis.

CO₂ concentration fluxes were calculated by performing linear regression with each series of GHG concentration versus time data from each deployed chamber. The

CO₂ flux is calculated using [1] where: m is the linear slope of ppmv/min, P is the mean atmospheric pressure in atm during GHG sampling, R is the ideal gas constant, T is the in-situ chamber air temperature in Kelvin, V is the chamber headspace volume in L, and A is the chamber surface area in m².

$$[1] \text{ C Flux } (\mu\text{g}/\text{m}^2/\text{hr}) = m \cdot P / (R \times T) \cdot (60 \text{ min}/\text{hr}) \cdot V / A \cdot (12.011 \mu\text{gC}/\mu\text{Mol})$$

2.3.2 $\delta^{13}\text{C}$ of CO₂

The CO₂ samples that were collected for analysis of $\delta^{13}\text{C}$ were analyzed on a continuous flow gasbench + DeltaPlusXP IRMS at the G.G. Hatch Stable Isotope Lab, University of Ottawa. Isotopic compositions are expressed using delta notation [2]. The international standard for CO₂ samples is CO₂ from Pee Dee Belemnite (PDB) limestone (Ehleringer & Osmond 1989) and the δ units are parts per thousand (‰).

$$[2] \delta^{13}\text{C} = [(^{13}\text{C}/^{12}\text{C}_{\text{sample}}) / (^{13}\text{C}/^{12}\text{C}_{\text{standard}})] - 1$$

The results for $\delta^{13}\text{C}$ of CO₂ were corrected by removing the contribution from atmospheric CO₂ that was initially inside the chamber by using the mass balance formula [3]. $\delta^{13}\text{C}_{\text{measured}}$ is the sample's result obtained from the IRMS, and $[\text{CO}_2]_{\text{measured}}$ is the calculated concentration of CO₂ in the sample at the time of sampling.

$[\text{CO}_2]_{\text{measured}}$ is calculated using the equation of a line (*ie* $y=mx+b$) [4] where, $\text{slope}_{\text{CO}_2 \text{ vs time}}$ is the slope (ppmv/min) obtained from graphing the CO₂ concentration against its respective sampling time for each chamber deployed. $\text{Time}_{\text{CO}_2 \text{ sample}}$ is the time that the measured sample was collected (*ie* 24 minutes for soybean and corn fields). $\delta^{13}\text{C}_{\text{atm}}$ is set at -8 ‰ because according to Fraserdale, Ontario (49°53'N, 81°34'W), which is the

closest data to our site, the $\delta^{13}\text{C}_{\text{atm}}$ for the approximate growing season (May to end of September) was $-8.54 \pm 0.70 \text{ ‰}$ (for 2009 and 2010) (Huang 2015), and thus $\delta^{13}\text{C}_{\text{atm}}$ was set at -8 ‰ for the current study because the difference between -8 ‰ and -8.54 ‰ was insignificant for the resulting calculations. $[\text{CO}_2]_{\text{atm}}$ is set at 400 ppm for this study because the average of two air samples collected in 2013 in the South Nation Watershed was $393.3 \pm 3.6 \text{ ppm}$ and thus was rounded to 400 ppm as the difference between 393 and 400 ppm was insignificant for the resulting calculations. $[\text{CO}_2]_{\text{emitted}}$ is calculated as $([\text{CO}_2]_{\text{measured}} - [\text{CO}_2]_{\text{atm}})$ in order to give how much CO_2 was emitted from the soil.

$$\text{[3]} \quad \delta^{13}\text{C}_{\text{emitted}} = [(\delta^{13}\text{C}_{\text{measured}} \times [\text{CO}_2]_{\text{measured}}) - (\delta^{13}\text{C}_{\text{atm}} \times [\text{CO}_2]_{\text{atm}})] / [\text{CO}_2]_{\text{emitted}}$$

$$\text{[4]} \quad [\text{CO}_2]_{\text{measured}} = (\text{slope}_{\text{CO}_2 \text{ vs time}} \times \text{time}_{\text{CO}_2 \text{ sample}}) + [\text{CO}_2]_{\text{atm}}$$

2.4 Methods for estimating rhizosphere respiration

Two documented approaches in the literature, the root-exclusion method and the isotope ^{13}C method, were used to separate R_{T} into its R_{rh} and R_{s} components. The root-exclusion method involves calculating an estimated R_{rh} (referred to as $R_{\text{rh,root}}$) by subtracting the CO_2 emission rates from soils without roots (*ie* non-vegetated area; R_{s}) from those of soils that contain roots (*ie* vegetated area; R_{T}) (Hanson *et al* 2000, Rochette *et al* 1999). The isotope ^{13}C method involves calculating an estimated R_{rh} (referred to as $R_{\text{rh,iso}}$) as the difference between ^{13}C of R_{T} (*ie* vegetated area) and ^{13}C of R_{s} (*ie* non-vegetated area) (Rochette & Flanagan 1997, Rochette *et al* 1999). First, the R_{rh} fraction is calculated using [5] from which the $R_{\text{rh,iso}}$ is then estimated using [6]

(Rochette *et al* 1999). If the $\delta^{13}\text{C}$ of plants varies from that of the soil it is growing in, then the ^{13}C isotope approach is valid (Rochette & Flanagan 1997, Rochette *et al* 1999).

$$[5] R_{rh,frac} = R_{rh}/R_T$$

$$= (\delta^{13}\text{C}_{soil\ CO_2\ efflux\ veg} - \delta^{13}\text{C}_{soil\ CO_2\ efflux\ non-veg}) / (\delta^{13}\text{C}_{plant} - \delta^{13}\text{C}_{soil})$$

$$[6] R_{rh,iso} = R_{rh,frac} * R_T, \text{ where } R_T \text{ is } \text{CO}_2 \text{ efflux from vegetated sites (Kg C/ha/day)}$$

2.5 Isotopic analysis of soils and plants

Soil samples were collected with a hand auger approximately every 3 to 4 weeks from OT and BT locations at 3 depths (0-15 cm, 15-30 cm, and 30-60 cm) in all fields. In fields 11-14 in 2013, soil was collected in the same area of the field as the CO_2 chambers from 2 BT locations on opposite sides of the piezometer nest and laterally 7.5 m away from 2 OT locations. In field 1 and 2 2012, soil was collected from 3 locations (top, middle, bottom of field) for BT and OT locations. In field 1 and 2 2013, soil was collected from 2 locations (bottom and top of field) with a BT location close to the piezometer nest and an OT location 7.5 m away.

While many soil samples were collected during the growing seasons, only a subset were analyzed for $\delta^{13}\text{C}$ of SOM. In 2013, a total of 6 soil samples (OT and BT locations at 3 depths each: 0-15 cm, 15-30 cm, 30-60 cm) per field were analyzed each month (June-September) for fields 1-2 and 11-14. In fields 1 & 2 2012, a total of 18 soil samples (3 sites OT and BT locations at 3 depths each: 0-15 cm, 15-30 cm, 30-60 cm) per field were analyzed in June, 6 soil samples (OT and BT locations at 3 depths each: 0-15 cm, 15-30 cm, 30-60 cm) per field were analyzed each in July and August, and 4

soil samples (OT and BT location at 2 depths each: 0-30 cm, 30-60 cm) per field were analyzed in September.

The soil was prepared for $\delta^{13}\text{C}$ analysis using the method described by Brodie *et al*/2011. Briefly, the soil was dried, ground, placed in silver capsules and acidified with HCl to remove carbonates. Soil samples were analyzed for $\delta^{13}\text{C}$ on the IRMS at the G.G. Hatch Stable Isotope Lab, University of Ottawa.

Plant samples were collected just prior to harvest from OT and BT locations. The plant samples were separated into their parts, where in 2012 beans were separated from the rest of the soybean plant. In 2013, soybean was separated into pods, leaves, root, and stem whereas corn was separated into cob, leaves, seeds, root, and stem. In field 1 and 2 2013, a sample of stem, leaves, root, cob, and seeds was submitted for OT and BT locations (where each sample is a composite sample from a mixture of 4 plants) for a total of 10 analyzed samples for $\delta^{13}\text{C}$ per field. In field 11-14 2013, a sample of pods, leaves, root, stem were submitted for OT and BT locations (where each sample is a composite sample from a mixture of 6 plants) for a total of 8 analyzed samples per field. In field 1 and 2 2012, one OT and BT sample for plant and beans (where each sample is a composite sample from a mixture of 30 plants) was submitted per field for a total of 4 analyzed samples per field.

The plants were dried at 80 °C and finely ground using a ball mill. Equal amounts from each plant sample (2 g) were mixed for composite samples. Plant samples were analyzed for $\delta^{13}\text{C}$ on the IRMS at the G.G. Hatch Stable Isotope Lab, University of Ottawa.

2.6 Measurements of environmental variables

The water table depth below surface (WTDBS) was measured at 15 minute intervals throughout the field season from pressure transducer water level sensors which were inserted in groundwater wells located between tiles in each field.

Soil volumetric water content (0-20 cm depth) (SWC20), soil temperature (0-10 cm depth) (ST10), and air temperature were measured within 0.5 m of each GHG sampling chamber at time of sampling (1000-1400 h). SWC20 (*ie* water volume/total volume) of the soil was measured using time-domain reflectometry (TDR) probes. ST10 and air temperature were measured using a digital thermocouple thermometer and temperature probe.

Two weather stations (one in Field 14 and the other near Field 1) measured precipitation (mm), soil temperature (0-15 cm depth) (ST15) (°C), air temperature (°C), and solar radiation (SR, W/m²) every 30 minutes.

2.7 Statistical analysis

In order to compare $R_{rh,iso}$ and $R_{rh,root}$ estimates, t-tests (Minitab 16) were done for each field. For comparing CTD and UTD management within each field pair, t-tests (Minitab 16) were done to compare R_s , R_T , $R_{rh,iso}$, and $R_{rh,root}$. Pearson correlations were done for R_s and R_T with SWC20 and ST10. Pearson correlations were done for $R_{rh,iso}$ and $R_{rh,root}$ with the following environmental variables: SWC20 and ST10 taken next to GHG chambers at time of GHG sampling, daily average WTDBS for GHG sampling day, and average SR and air temperature between 1000-1400 h on GHG sampling day.

3. Results and discussion

3.1 Weather conditions

While the temperature during both studied growing seasons was similar to the normal average for the area, the 2012 growing season was drier whereas the 2013 growing season was wetter than the normal average. The 2012 growing season received less precipitation but had a similar mean ST15 compared to the 2013 growing season (**Fig. 3.1a**). Total rainfall was 460 mm and 281 mm respectively during the 2013 and 2012 growing seasons (**Fig. 3.1b**). There was 24% less and 25% greater precipitation in 2012 and 2013, respectively, compared to the average 30-year rainfall (1981-2010) for the approximate growing season (June-September = 369 mm) from nearby Russell station (EC 2015). The average ST15 during the growing season, from the weather station near Field 1, was similar in 2012 ($17.8\text{ }^{\circ}\text{C} \pm 2.6$) and 2013 ($17.4\text{ }^{\circ}\text{C} \pm 2.1$) and was also similar to the 30-year (1981-2010) daily average air temperature for the approximate growing season (June-September = $18.4\text{ }^{\circ}\text{C}$) from nearby Russell station (EC 2015). By having similar temperatures but one wet growing season and one dry growing season provided a very interesting scenario for comparing soil respiration results from both growing seasons.

3.2 Soil organic matter and plant $\delta^{13}\text{C}$

The average $\delta^{13}\text{C}$ of SOM from June to September was $-22.25 \pm 1.09\text{ }_{\text{‰}}$ for Field 1 and 2 over both years of study, and $-24.73 \pm 0.54\text{ }_{\text{‰}}$ for Fields 11-14 in 2013. Average soil $\delta^{13}\text{C}$ values remained relatively constant in each field throughout the growing seasons indicating that the plants did not affect the $\delta^{13}\text{C}$ of SOM (**Table 3.2a**). The $\delta^{13}\text{C}$ of SOM resembles the $\delta^{13}\text{C}$ of the plant material from which it was derived

(Gregorich *et al* 1995, Cheng 1996). Soil $\delta^{13}\text{C}$ from Fields 1 and 2 was slightly more enriched than the other fields and was mostly C_3 with some C_4 plant inputs as it was neither in the SOM from C_4 plants range ($\delta^{13}\text{C}$ values from -12‰ to -14‰) or C_3 plants (-24‰ to -29‰) (Gregorich *et al* 1995, Cheng 1996). Soil $\delta^{13}\text{C}$ from Field 11 to 14 were in the range for SOM from C_3 plants (-24‰ to -29‰) (Gregorich *et al* 1995, Cheng 1996). The crop history in Fields 11 to 14 for the five previous years to the study were C_4 plants and, therefore, the SOM from Field 11-14 must be from older contributions.

The average $\delta^{13}\text{C}$ of corn and soybean for all fields and years of study were $-12.37 \pm 0.89\text{‰}$ and $-27.80 \pm 0.64\text{‰}$ respectively. Each part of the plant had a similar $\delta^{13}\text{C}$ value, and the $\delta^{13}\text{C}$ values of soybean plants and corn plants are presented in **Table 3.2b and Table 3.2c** respectively. The $\delta^{13}\text{C}$ plant values are as expected because soybean is in the range for C_3 plants (~ -26 to -28‰), whereas corn is in the C_4 range (-10 to -14‰) (Clark 2015, O'Leary 1988). It was not surprising that each part of the plant had a similar $\delta^{13}\text{C}$ value, as this was also seen in the literature. For example, Rochette *et al* 1999 found the $\delta^{13}\text{C}$ of corn leaves ($-13.44 \pm 1.34\text{‰}$), stems ($-12.12 \pm 0.93\text{‰}$), and roots ($-13.65 \pm 0.86\text{‰}$) had a similar $\delta^{13}\text{C}$ value.

3.3 Soil respiration and environmental variables

3.3.1 Corn fields

Within corn fields (*ie* Fields 1 and 2 2013), R_T ranged from 6.74 to 35.05 kg C/ha/day while the contribution from the estimated R_{rh} ranged up to 20.48 kg C/ha/day for isotope method and up to 16.16 kg C/ha/day for root-exclusion method (**Fig. 3.3a**).

In CTD Field 2 throughout the growing season, the estimated R_{rh} contributed an average 63 % and 17 % to R_T from the isotope method and the root-exclusion method respectively. According to Rochette *et al* 1999, the isotope ^{13}C method for estimating R_{rh} is valid only if the ^{13}C content of soil organic carbon (SOC) and plant C differs. In the case of Field 1 and 2 2013, the SOC (average= -21.84 ± 1.22 ‰) and the plant C (average= -11.90 ± 0.41 ‰) differed in ^{13}C content and thus both the isotope and root-exclusion methods were used to estimate R_{rh} . The estimates of R_{rh} obtained using both methods (^{13}C and root-exclusion) were significantly different in CTD Field 2 (t-test; $p < 0.05$) but were similar in UTD Field 1 (t-test; $p > 0.05$). This contrasts with Rochette *et al* 1999, who found the root-exclusion and isotope methods resulted in similar R_{rh} estimates within a maize field. The root-exclusion approach resulted in some negative R_{rh} in Field 2 and thus the isotope method seemed to provide more accurate results. When comparing CTD and UTD management, there was no significant difference ($p > 0.05$) between R_s , R_T , and R_{rh} in CTD and UTD corn fields. Therefore, CTD did not affect the contribution of soil carbon pools to R_T within a corn field, thus indicating no observed effects of CTD on residue decomposition.

R_s and R_T in corn Field 1 and 2 2013 were driven by soil moisture and soil temperature. A Pearson correlation indicated that R_s in corn Field 1 and 2 had a significant moderate negative correlation with SWC20 ($r = -0.501$, $p < 0.05$) and a significant strong positive correlation with ST10 ($r = 0.735$, $p < 0.05$). A Pearson correlation indicated that R_T in corn Field 1 and 2 had a significant strong negative correlation with SWC20 ($r = -0.565$, $p < 0.05$) and a significant strong positive correlation with ST10 ($r = 0.606$, $p < 0.05$). R_s was the dominant contributor to R_T and, therefore, both

are correlated with soil moisture and soil temperature. These findings are supported in the literature because most ecosystem-scale C cycle models determine R_s by the amount of SOC (fresh litter + SOM) and the temperature and soil moisture levels, as temperature and soil moisture affect decomposition rates (Hopkins *et al* 2013).

Similarly, Zhao *et al* 2013 found that soil respiration was best predicted when soil temperature and soil water content were combined rather than separating out those factors. Within Field 1 and 2, SOC is available as it is an agricultural field and thus, as expected, R_s is correlated with soil temperature and soil moisture.

When examining what was driving R_{rh} in corn Field 1 and 2 2013, it was not significantly correlated with any of the examined variables. $R_{rh,iso}$ in corn was not significantly correlated with SR ($r=0.309$, $p>0.05$), WTDBS ($r=0.027$, $p>0.05$), SWC20 ($r=-0.034$, $p>0.05$), or ST10 ($r=-0.070$, $p>0.05$). Similarly, $R_{rh,root}$ was not significantly correlated with SR ($r=0.179$, $p>0.05$), WTDBS ($r=0.225$, $p>0.05$), SWC20 ($r=-0.090$, $p>0.05$), or ST10 ($r=-0.137$, $p>0.05$). It was also found in the literature that R_{rh} from maize plants did not vary across a temperature gradient (Vicca *et al* 2010). It was suggested in the literature that root respiration is a primary function of gross-primary productivity (GPP), as GPP can explain most patterns of root respiration within a growing season (Hopkins *et al* 2013). R_{rh} is linked to photosynthesis (*ie* GPP) because photosynthesis affects the amount of C transported by phloem to the plant roots (Högberg *et al* 2001, Moyano *et al* 2008, Gomez-Casanovas *et al* 2012, Hopkins *et al* 2013). In the case of this study, GPP was not measured although it is expected that R_{rh} would correlate with GPP.

3.3.2 Soybean fields in wet growing season

Within the soybean fields in 2013 (*ie* Fields 11-14), R_t ranged from 13.21 to 45.54 kg C/ha/day while the contribution from the estimated $R_{rh,root}$ ranged up to 33.75 kg C/ha/day (**Fig. 3.3b**). Throughout the growing season in CTD fields, the estimated $R_{rh,root}$ contributed an average 11 % to R_T in CTD Field 11 and 62 % to R_T in CTD Field 12. According to Rochette *et al* 1999, the isotope ^{13}C method for estimating R_{rh} is valid only if the ^{13}C content of SOC and plant C differs. In the case of Fields 11 to 14 in 2013, the SOC from June-September (average= -24.73 ± 0.54 ‰) and the plant C (average= -27.90 ± 0.65 ‰) were similar in ^{13}C content and thus only the estimates from the root-exclusion method were used to estimate R_{rh} .

CTD and UTD management were compared for soybean fields in 2013. There was no significant difference ($p>0.05$) for R_s , R_T , and $R_{rh,root}$ between CTD Field 11 and UTD Field 14. When comparing CTD Field 12 and UTD Field 13, there was a significant increase ($p<0.05$) in the contribution of R_{rh} over R_s , although R_T remained the same. The mean R_s was smaller in CTD Field 12 (12.57 ± 4.86 kg C/ha/day) than in UTD Field 13 (28.18 ± 8.37 kg C/ha/day). In contrast, the mean $R_{rh,root}$ was much greater in CTD Field 12 (21.06 ± 6.88 kg C/ha/day) than in UTD Field 13 (5.41 ± 6.86 kg C/ha/day). It is unsure why CTD Field 12 had a smaller R_s contribution compared to UTD Field 13. A possible explanation could be a decreased amount of active decomposers in CTD Field 12. CTD Field 12 had a greater $R_{rh,root}$ which may possibly be because of greater GPP on sampling days even though there was no difference in final yields between CTD Field 12 (OT= 4289kg/ha, and BT= 4260 kg/ha) and UTD Field 13 (OT= 4129 kg/ha and BT= 4255 kg/ha). Therefore, CTD did affect the contribution of soil carbon pools to respiration within one studied soybean field pair but not in the other studied field pair.

Within soybean fields 11-14 in 2013, R_T was correlated with soil moisture and soil temperature in all four fields, whereas R_s was correlated with soil moisture in two fields and soil temperature in three fields. R_T in soybean fields 11-14 in 2013 has a significant strong negative correlation with SWC20 ($r=-0.772$, $p<0.05$) and a significant strong positive correlation with ST10 ($r=0.836$, $p<0.05$). R_s in Field 11 and 14 2013 has a significant strong negative correlation with SWC20 ($r=-0.624$, $p<0.05$) and a significant strong positive correlation with ST10 ($r=0.740$, $p<0.05$). R_s in Field 12 was significantly strongly positively correlated with ST10 ($r=0.715$, $p<0.05$) and not significantly correlated with soil moisture ($r=-0.335$, $p=0.378$). R_s in Field 13 was not significantly correlated with ST10 ($r=0.286$, $p>0.5$) or SWC20 ($r=-0.539$, $p>0.05$).

When examining what was driving $R_{rh,root}$ in soybean fields 11-14, it was correlated with air temperature in two fields and soil temperature in three fields. In Field 11 & 14, $R_{rh,root}$ had a significant moderate positive correlation with air temperature ($r=0.436$, $p<0.05$) and ST10 ($r=0.465$, $p<0.05$), but it was not significantly correlated with SR ($r=-0.156$, $p>0.05$), WTDBS ($r=-0.082$, $p>0.05$), or SWC20 ($r=-0.240$, $p>0.05$). In Field 12, $R_{rh,root}$ was significantly strongly correlated with ST10 ($r=0.816$, $p<0.05$) whereas in Field 13 there was no significant correlation with ST10 ($r=-0.290$, $p>0.05$). In Field 12 & 13, $R_{rh,root}$ was not significantly correlated with SR ($r=-0.224$, $p>0.05$), WTDBS ($r=-0.126$, $p=0.619$), SWC20 ($r=-0.264$, $p>0.05$), or air temperature ($r=0.292$, $p>0.05$).

3.3.3 Soybean fields in dry growing season

Within soybean fields in 2012 (*ie* fields 1-2), R_T ranged from 7.30 to 44.17 kg C/ha/day while the contribution from the estimated R_{rh} ranged up to 23.23 kg C/ha/day

(for isotope method) and up to 10.47 kg C/ha/day (for root-exclusion method) (**Fig. 3.3b**). Throughout the growing season in CTD Field 2, the estimated R_{rh} contributed an average 63 % and 17 % to R_T from the isotope method and the root-exclusion method respectively. It was possible to use natural abundance of ^{13}C for estimating the contribution of R_{rh} to R_T in soybean Fields 1 and 2 in 2012 because the SOC ($-22.33 \pm 1.11 \text{ ‰}$) and the plant C ($-27.39 \pm 0.43 \text{ ‰}$) differed in ^{13}C content. The R_{rh} results obtained from both methods (^{13}C and root-exclusion) were not significantly different ($p>0.05$) between CTD Field 2 and UTD Field 1. When comparing CTD and UTD management, there was no significant difference ($p>0.05$) for R_s , R_T , and R_{rh} between CTD Field 2 and UTD Field 1. Therefore, CTD did not affect the contribution of soil carbon pools to R_T within a soybean field, thus indicating no observed effects of CTD on residue decomposition.

Pearson correlation analyses involving R_s and R_T from Field 1 and 2 2012 did not provide any significant findings except that R_s had a significant weak negative correlation with WTDBS ($r=-0.450$, $p<0.05$). R_s and R_T in corn Field 1 and 2, 2012, did not have any significant correlations with SWC20 ($r=0.413$, $p>0.05$; $r=-0.022$, $p>0.05$) or ST10 ($r=0.373$, $p>0.05$; $r=0.328$, $p>0.05$) respectively. R_T was not significantly correlated with SR ($r=0.372$, $p>0.05$).

When examining what was driving R_{rh} in soybean Field 1 and 2 in 2012, there was no significant correlations involving $R_{rh,iso}$ but significant correlations existed with $R_{rh,root}$. $R_{rh,iso}$ was not significantly correlated with SR ($r=0.036$, $p>0.05$), WTDBS ($r=-0.112$, $p>0.05$), SWC20 ($r=-0.070$, $p>0.05$), ST10 ($r=0.079$, $p>0.05$), or air temperature ($r=-0.179$, $p>0.05$). $R_{rh,root}$ was not significantly correlated with SR ($r=0.049$, $p>0.05$),

WTDBS ($r=0.313$, $p>0.05$), or ST10 ($r=0.038$, $p>0.05$). $R_{rh,root}$ had a significant weak negative correlation with SWC20 ($r=-0.464$, $p=0.039$) and a moderate positive correlation with air temperature ($r=0.592$, $p=0.006$).

The wetter growing season in 2013 resulted in greater average R_{rh} emissions from CTD soybean fields compared to the dry 2012 growing season. Mean R_{rh} from CTD soybean fields was greater in 2013 (mean $R_{rh,root}$ for Field 11 & 12 combined; 10.99 ± 10.61) than in 2012 (mean $R_{rh,iso}$ for Field 2; 3.71 ± 2.97).

4. Conclusion

Controlled tile drainage (CTD) management is common in agricultural fields but its effect on residue decomposition is unknown. Quantitative estimates of the CO_2 produced by both rhizosphere respiration (R_{rh}) and soil respiration (R_s) in CTD fields are required in order to properly assess the environmental effects of CTD. In order to determine whether CTD affects the contribution of soil carbon pools (*ie* R_{rh} and R_s) to total soil respiration (R_T), soil CO_2 efflux from four field pairs, cropped with corn and soybean, were sampled from vegetated and non-vegetated areas throughout two growing seasons. Both the root-exclusion approach (root) and the natural abundance isotope ^{13}C approach (iso) were used to separate R_T into its R_{rh} and R_s components. Within corn fields, there was no significant difference ($p>0.05$) for R_s , R_T , and R_{rh} between CTD and UTD fields. Similarly, there was no significant difference ($p>0.05$) for R_s , R_T , and R_{rh} between CTD and UTD soybean fields for two field pairs. One field pair resulted in CTD having a significant increase in the contribution of R_{rh} over R_s , although R_T remained the same. We conclude that CTD has not significantly impacted the carbon emissions in the soil-plant system, thus further supporting the use of CTD in agriculture.

5. Figures & tables

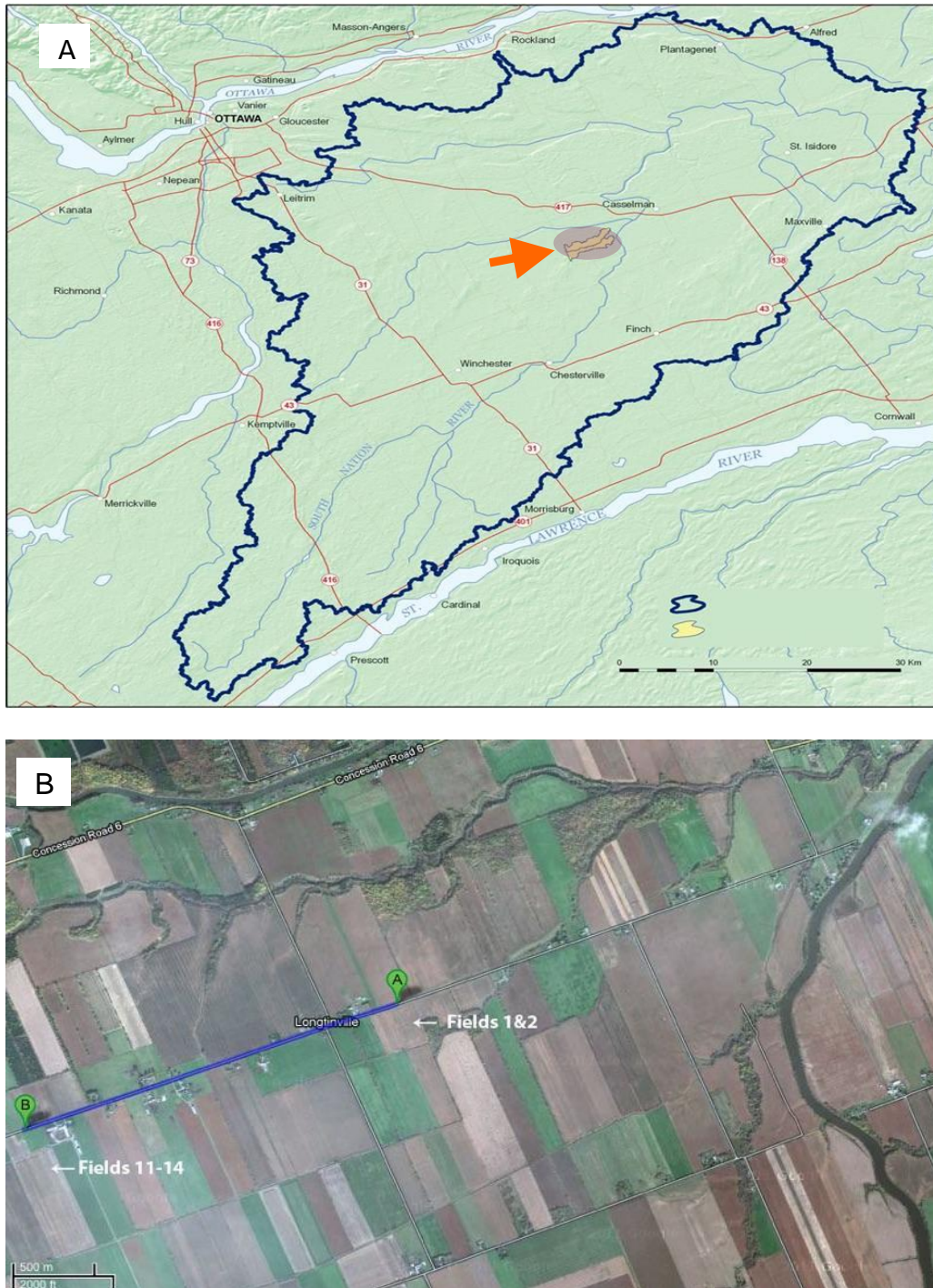


Figure M1 The agriculturally productive 3,900 km South Nation Watershed in Eastern Ontario, Canada (A), and the location of the fields in this study (B). The arrow in (A) indicates the location of the fields within the watershed (source David Lapen AAFC) and (B) is a Google Earth Maps image of the fields (Map data: Google, 2015).

Table M1 Planting, harvest, photosynthetic pathway, and key tile drainage management dates for the studied control tile drained (CTD) and uncontrolled tile drained (UTD) agricultural fields.

Field pair	CTD field: area (ha)	UTD field: area (ha)	Year	Planting date: crop type	Date stop gates were closed for CTD fields	Date stop gates were opened for CTD fields	Harvest date
1 & 2	Field 2: 2.3	Field 1: 2.0	2012	May 16: soybean	May 16	September 19	September 24
			2013	May 5: corn	May 29 ^a	September 26	November 14
11 & 14	Field 11: 4.2	Field 14: 4.1	2012	In 2011: forage	May 23	November 19	June 1, July 10, August 20, October 24
			2013	May 15: soybean	May 27	September 26	October 25
12 & 13	Field 12: 5.0	Field 13: 4.2	2013	May 15: soybean	May 27	September 26	October 25

^a Gates were left open for a longer time because farmer wanted them open due to excessive rainfall

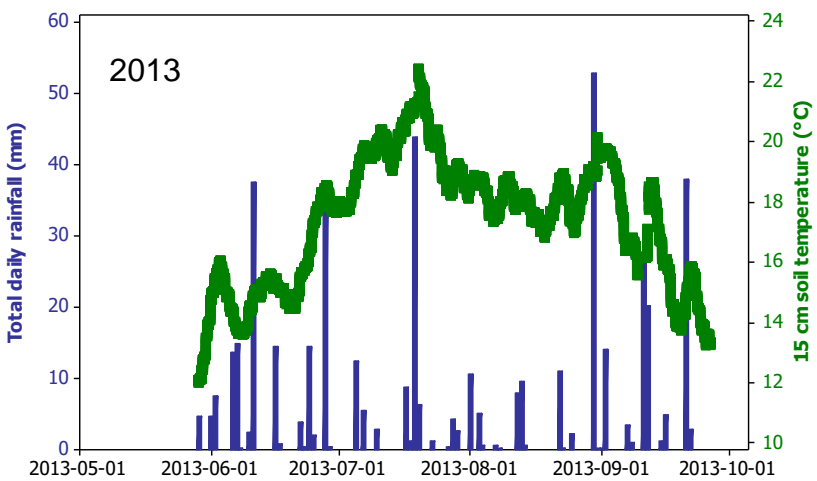
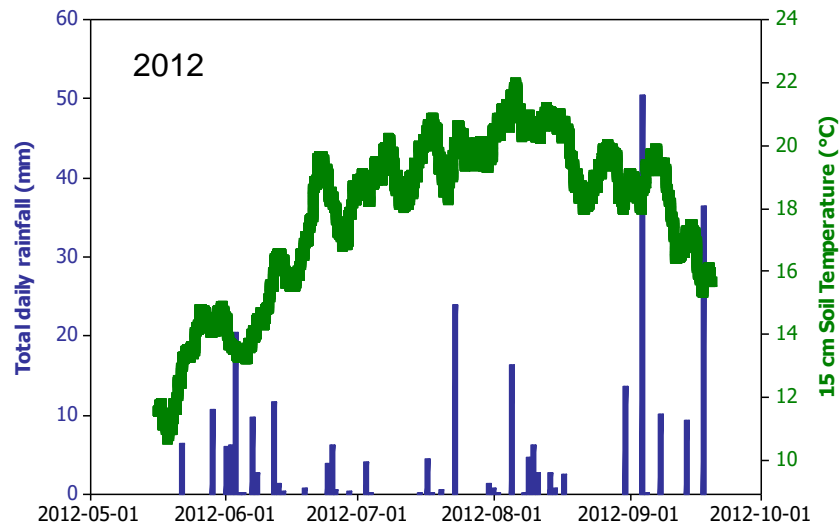


Figure 3.1a Total daily rainfall and soil temperature (0 - 15 cm depth) measured every 30 minutes, in 2012 and 2013, from the weather station located near Field 1.

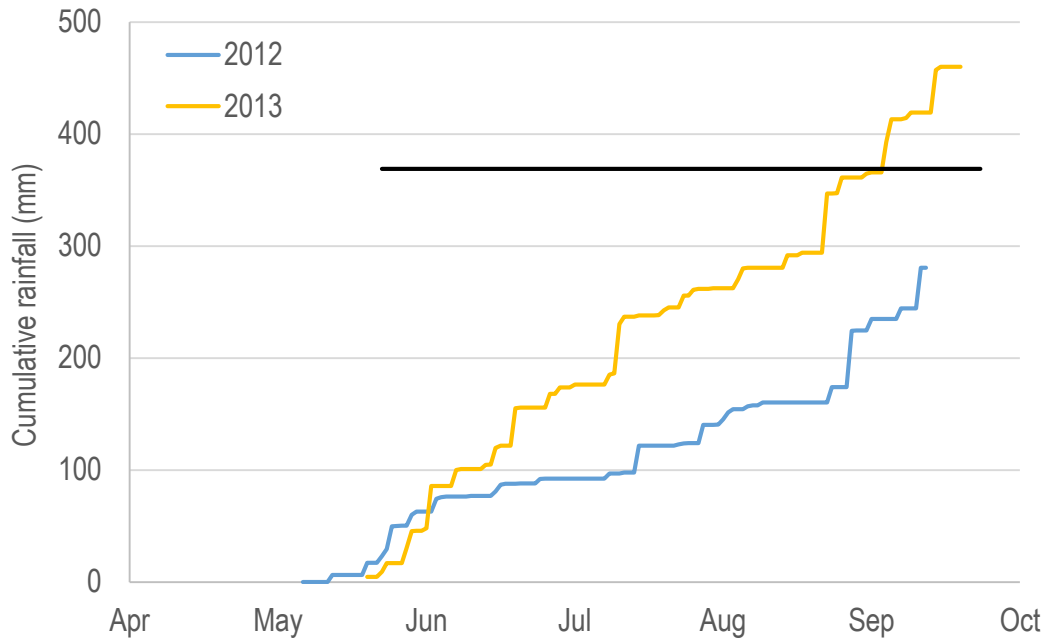


Figure 3.1b Cumulative rainfall for the 2012 and 2013 growing seasons (according to specific dates from Field 1 & 2's growing seasons), from the weather station located near Field 1. The black line represents the average 30-year rainfall (1981-2010) for the approximate growing season (June-September = 369 mm) from nearby Russell station (EC 2015).

Table 3.2a Monthly average \pm standard deviation for $\delta^{13}\text{C}$ (‰) of soil organic carbon for all studied fields throughout the 2012 and 2013 growing seasons. The soil is from 0-60 cm depth. Note that Fields 11-14 in 2013 were sampled on October 1st not in September.

Month	Field 1		Field 2		Field 11		Field 14		Field 12	Field 13
	2012	2013	2012	2013	2012	2013	2012	2013	2013	2013
June	-22.42 \pm 1.32	-22.65 \pm 0.36	-22.68 \pm 1.30	-21.39 \pm 0.70	-25.32 \pm 0.41	-24.60 \pm 0.40	-25.14 \pm 0.25	-24.09 \pm 0.15	-25.33 \pm 0.30	-24.47 \pm 0.79
July	-22.36 \pm 0.71	-23.20 \pm 0.72	-22.16 \pm 0.92	-23.32 \pm 1.36	-24.75 \pm 0.26	-24.36 \pm 0.30	-24.55 \pm 0.14	-25.08 \pm 0.85	-24.64 \pm 0.59	-25.12 \pm 0.27
August	-22.08 \pm 0.53	-22.03 \pm 0.41	-21.47 \pm 0.95	-21.37 \pm 0.49	-24.97 \pm 0.37	-24.47 \pm 0.52	-24.80 \pm 0.42	-24.64 \pm 0.28	-24.96 \pm 0.28	-25.11 \pm 0.28
September	-22.14 \pm 0.66	-21.78 \pm 0.80	-22.44 \pm 1.00	-21.30 \pm 0.90	-25.11 \pm 0.44	-25.15 \pm 0.37	-23.92 \pm 1.27	-24.52 \pm 0.34	-24.61 \pm 0.67	-24.52 \pm 0.18
Total	-22.32 \pm 1.04	-22.41 \pm 0.80	-22.34 \pm 1.20	-21.84 \pm 1.22	-25.09 \pm 0.42	-24.65 \pm 0.49	-24.71 \pm 0.74	-24.58 \pm 0.58	-24.88 \pm 0.55	-24.81 \pm 0.53

Table 3.2b Average \pm standard deviation for $\delta^{13}\text{C}$ (‰) of soybean plants from Field 1 and 2 prior to harvest in 2012, and from Field 11-14 prior to harvest in 2013. In 2012, soybean pods were separated from the rest of the plant. In 2013, the plant was divided into its pods, leaves, root, and stem for analysis.

Part of Soybean Plant	2012		2013			
	Field 1	Field 2	Field 11	Field 12	Field 13	Field 14
Plant	-27.77 \pm 0.06	-27.06 \pm 0.52	ND	ND	ND	ND
Pods	-27.66 \pm 0.04	-27.09 \pm 0.40	-27.43 \pm 0.05	-27.65 \pm 0.32	-28.21 \pm 0.66	-28.10 \pm 0.90
Leaves	ND	ND	-28.64 \pm 0.37	-28.78 \pm 0.16	-28.96 \pm 0.17	-28.87 \pm 0.01
Root	ND	ND	-27.53 \pm 0.05	-27.40 \pm 0.34	-27.67 \pm 0.18	-27.54 \pm 0.52
Stem	ND	ND	-27.81 \pm 0.12	-27.40 \pm 0.03	-27.07 \pm 0.12	-27.45 \pm 0.32
Total	-27.71 \pm 0.07	-27.07 \pm 0.38	-27.85 \pm 0.53	-27.81 \pm 0.64	-27.98 \pm 0.79	-27.99 \pm 0.73

ND = not determined

Table 3.2c Average \pm standard deviation for $\delta^{13}\text{C}$ (‰) of five different parts of corn plants from Field 1 and 2 prior to harvest in 2013.

Part of Corn Plant	$\delta^{13}\text{C}$ (‰)	
	Field 1	Field 2
Cob	-12.06 \pm 0.29	-11.28 \pm 0.18
Leaves	-14.07 \pm 1.56	-12.25 \pm 0.41
Seeds	-12.28 \pm 0.01	-11.73 \pm 0.08
Root	-12.91 \pm 0.52	-12.19 \pm 0.03
Stem	-12.92 \pm 1.17	-12.05 \pm 0.07
Total	-12.84 \pm 1.00	-11.90 \pm 0.41

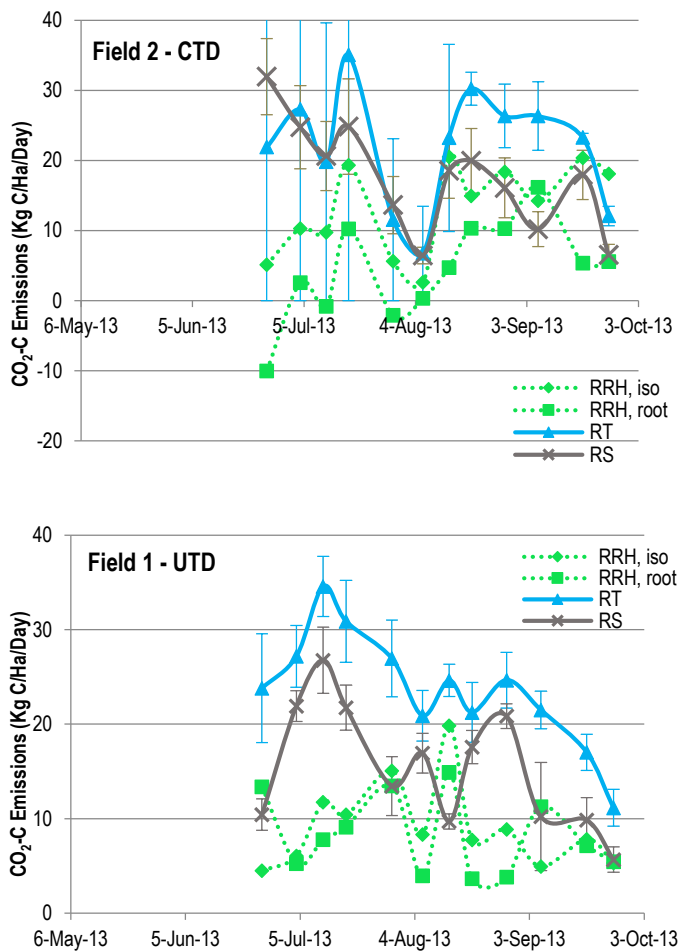


Figure 3.3a Total soil respiration (RT), along with its contributions from rhizosphere (RRH) and soil (RS) respiration, in CTD and UTD corn fields during the 2013 growing season. Estimates of RRH were obtained from the ¹³C isotope (RRH,iso) and the root-exclusion (RRH,root) method. Vertical bars represent \pm SD.

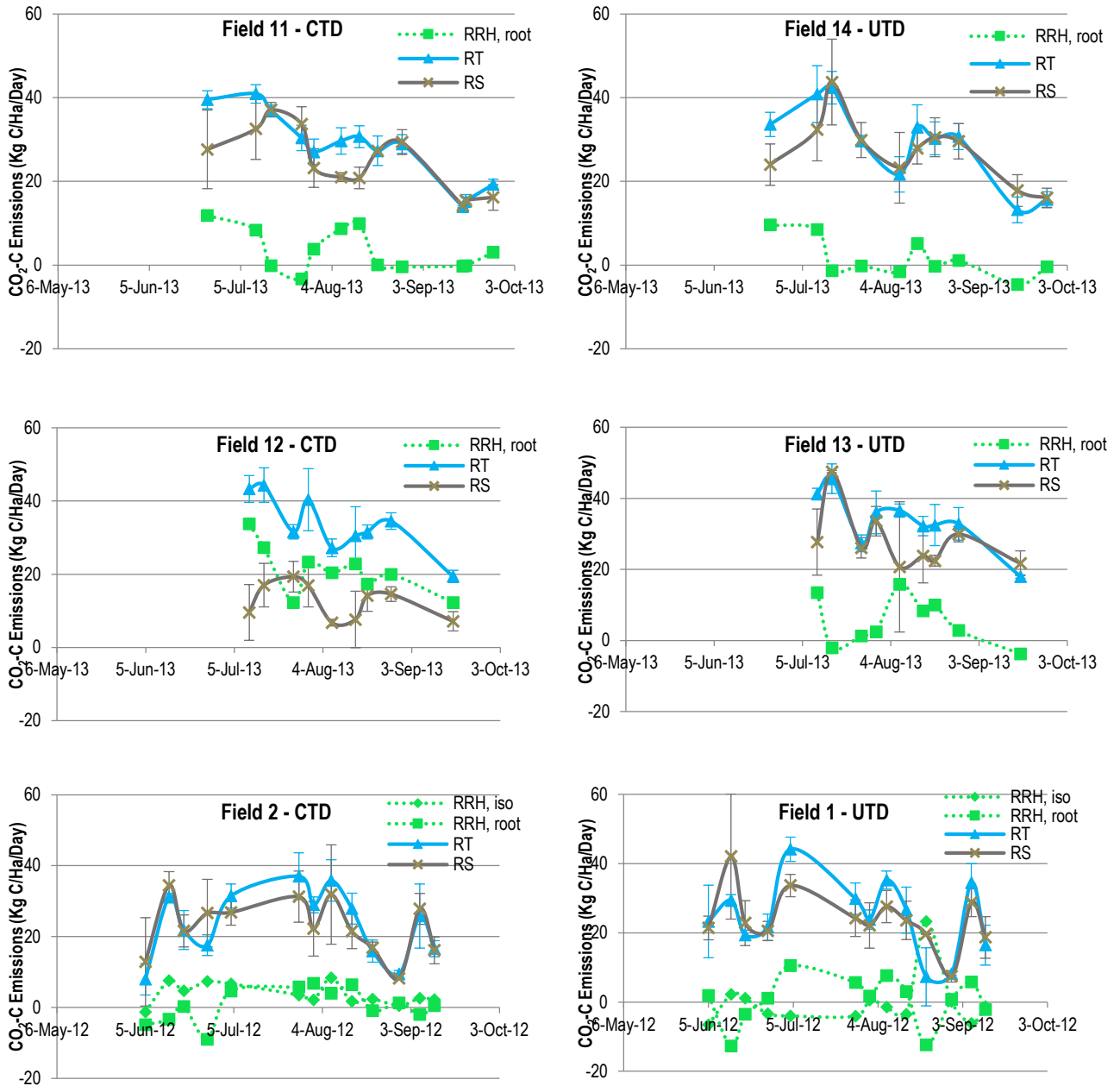


Figure 3.3b Total soil respiration (RT), along with its contributions from rhizosphere (RRH) and soil (RS) respiration, in each CTD and UTD soybean field pair during the 2013 and 2012 growing season. Estimates of RRH were obtained from the ¹³C isotope (RRH,iso) and the root-exclusion (RRH,root) method. Vertical bars represent \pm SD.

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Chapter 6 Synthesis and suggestions for future research extending from the current study

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1. Summary of results

This study was undertaken to determine the impact, if any, of controlled tile drainage (CTD) management on soil greenhouse gas (GHG: CO₂, CH₄, N₂O) emissions from agricultural fields in the South Nation watershed of Ontario. To rigorously evaluate the impact of this drainage water management scheme, a statistically-rigorous study was designed and carried out. The intensiveness of this GHG study, within the South Nation watershed of Ontario, is highlighted by the very large number of GHG samples collected. An approximate total of 5970 soil GHG samples were collected over the two growing seasons (Field 1-2 2012 ~ 930, Field 11 & 14 2012 ~ 700, Field 1-2 2013 ~ 1610, Field 11-14 2013 ~ 2730), where approximately 1/5 of those samples were collected for analysis of $\delta^{13}\text{C}$ of CO₂. Through this intensive sampling, comprehensive data were obtained that were used to explain what was happening in CTD and uncontrolled tile drainage (UTD) fields throughout two growing seasons. The findings from this study were divided into three parts and the results from each of those parts are summarized below.

1.1 First part of the study

Since the effects of CTD on GHG emissions and their distribution relative to tile drains are not well-documented, it was examined in Chapter 3 whether CTD had an effect on soil GHG effluxes and whether there was a tile location effect on GHG emissions. Chamber measurements of soil GHG emissions were compared for CTD and UTD managed corn, soybean, and forage fields containing silt loam soil throughout a dry and a wet growing season in eastern Ontario, Canada. Furthermore, this study evaluated the over tile (OT) and between tile (BT) spatial distribution of soil GHG

emissions from CTD fields. CO₂ and CH₄ soil fluxes did not significantly differ between CTD and UTD fields ($p>0.05$), while N₂O soil fluxes were also not significantly different for all field pairs studied ($p>0.05$) except for one. Throughout the growing seasons, soil GHG emissions did not significantly differ between OT and BT locations within CTD fields ($p>0.05$). The results of this study indicate no adverse soil GHG emissions associated with CTD over the time span and field environments of the study. When comparing the dry (2012) versus wet (2013) growing season, the surface soil moisture was approximately 10% greater in 2013 and this may be why there was an approximate 5 kg C/ha/day greater CO₂ flux from soybean fields in 2013 than in 2012. The results of this study differ from a previous study, where Nangia et al 2013 found significantly higher CO₂ emissions from CTD fields during some years of study. During the current study, there was no difference in surface soil moisture between CTD and UTD fields whereas in Nangia et al 2013 the surface soil moisture did differ between CTD and UTD fields thus causing a difference in CO₂ emissions in their study.

1.2 Second part of the study

The second part of the study (Chapter 4) involved using ¹³C of soil respiration to infer the contribution from soil carbon pools in order to provide insights into the effects of CTD on residue decomposition. CTD has the potential to increase total soil respiration (R_T) by creating wet soil conditions that favour the decomposition of soil organic carbon (SOC) and crop residue carbon (CRC). Isotope values ($\delta^{13}C$) of R_T can identify the effects of CTD on residue decomposition. This study compared $\delta^{13}C$ of R_T from chamber samples collected in silt loam soil throughout a dry and a wet growing season for corn, soybean, and forage fields under CTD and UTD management in

eastern Ontario, Canada. Furthermore, this study evaluated the $\delta^{13}\text{C}$ of R_T samples collected from chamber located OT and BT in CTD fields. Values for $\delta^{13}\text{C}$ of R_T were not significantly different ($p>0.05$) between CTD and UTD fields for all field pairs studied except for one. Within CTD fields, the $\delta^{13}\text{C}$ of R_T did not significantly vary ($p>0.05$) from OT and BT locations except for one field. It is believed that Field 11 compared to Field 14, and Field 11 BT versus OT had a more enriched $\delta^{13}\text{C}$ of R_T because of different contributions of timothy and alfalfa within the chambers. Field 11 and Field 11 BT chambers could have contained a greater amount of timothy compared to alfalfa plants. Timothy plants have a shallower root system compared to the deep tap root of alfalfa. Thus chambers containing more timothy plants would create a greater contribution of respiration from SOC ($\delta^{13}\text{C} = -25 \text{ ‰}$) and less from plant carbon ($\delta^{13}\text{C} = -29 \text{ ‰}$) thus making the $\delta^{13}\text{C}$ of R_T more enriched. The results of this study indicate no significant effect of CTD on residue decomposition. Therefore, CTD management did not affect the contribution of the carbon substrate being oxidized and it also did not result in any change in the concentration of CO_2 in soils as the soil respiration rate, the soil porosity, and the amount of atmospheric CO_2 entering the soil in CTD and UTD fields were similar. Thus, the positive findings of this study, indicating that CTD management does not affect the source of residue decomposition, further supports the use of CTD in agriculture.

1.3 Third part of the study

CTD management is common in agricultural fields but its effect on rhizosphere respiration (R_{rh}) and soil respiration (R_s) is unknown. Quantitative estimates of the CO_2 produced by both R_{rh} and R_s in CTD fields are required in order to properly assess the

environmental effects of CTD. In order to determine whether CTD affects the contribution of soil carbon pools (R_{rh} and R_s) to total soil respiration (R_T), soil CO_2 efflux from four field pairs, cropped with corn and soybean, were sampled from vegetated and non-vegetated areas throughout two growing seasons (Chapter 5). Both the root-exclusion approach (root) and the natural abundance isotope ^{13}C approach (iso) were used to separate R_T into its R_{rh} and R_s components. Within corn fields, there was no significant difference ($p>0.05$) for R_s , R_T , and R_{rh} between CTD and UTD fields. Similarly, there was no significant difference ($p>0.05$) for R_s , R_T , and R_{rh} , between CTD and UTD soybean fields for two field pairs. One field pair resulted in CTD having a significant increase in the contribution of R_{rh} over R_s , although R_T remained the same. Thus, only one field pair out of the four pairs studied found that CTD did affect the contribution of R_{rh} and R_s but R_T remained the same. The estimated R_{rh} ranged up to 20.48 kg C/ha/day within corn fields, up to 33.75 kg C/ha/day within soybean fields in the wet growing season, and up to 23.23 kg C/ha/day within soybean fields in the dry growing season. In conclusion, CTD has not significantly impacted the carbon emissions in the soil-plant system, thus further supporting the use of CTD in agriculture.

1.4 Overall summary of this study

Previous to this study, it was documented in the literature that CTD can increase crop yields (eg. Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Kross *et al* 2015, Mejia *et al* 2000, Cicek *et al* 2010) and can reduce the amount of nutrient loss in drainage water (eg. Sunohara *et al* 2015, Tan *et al* 1999, Wesstrom & Messing 2007, Ng *et al* 2002, Drury *et al* 1996). It was thought that these benefits of increased yields and reduced nutrient loss in drainage water associated with CTD could result in an

environmental trade-off of increased soil GHG emissions. Within the literature, there was a lack of knowledge on soil GHG emissions from CTD agricultural fields. Thus, the proposed study examined whether CTD affected soil GHG emissions and their spatial distribution, whether CTD affected residue decomposition, and specifically whether CTD affected the contribution of rhizosphere or soil respiration.

Compilation of the information obtained from the three parts of this study results in an overall positive finding for the effects of CTD on soil GHG emissions. CO₂ and CH₄ fluxes did not significantly differ between CTD and UTD fields ($p>0.05$), while N₂O fluxes were also not significantly different for all field pairs studied ($p>0.05$) except for one. Values for $\delta^{13}\text{C}$ of R_T were not significantly different between CTD and UTD fields for all field pairs studied ($p>0.05$) except for one. There was no significant difference for R_s, R_T, and R_{rh} between CTD and UTD fields ($p>0.05$) except for one field pair where R_s and R_{rh} significantly varied ($p<0.05$). When examining spatial distribution of OT and BT locations in CTD fields, there was no significant difference in soil GHG emissions ($p>0.05$), and in the $\delta^{13}\text{C}$ of R_T ($p>0.05$) except for one field. The results of this study indicate no significant adverse effect of CTD on soil GHG emissions and their spatial distribution, on residue decomposition, and on R_{rh} and R_s emissions over the time span and field environments of the study. Thus, the positive findings of this study regarding soil GHG emissions further supports the use of CTD in agriculture. This study was presented at the 41st International Association of Hydrogeologists conference in 2014.

2. Suggestions for future research

SOC is dynamic with inputs from decomposition and outputs through transfers to other pools or mineralization (Six & Jastrow 2002). This SOC turnover is quantified by

the mean residence time (MRT), which is defined as the average time that carbon resides in the pool at steady state (Six & Jastrow 2002). SOC is a complex mixture containing three fractions with differing MRT (Paul *et al* 2001). SOC consists of an active fraction with MRT of months, a large slow intermediate fraction with MRT of years to decades, and a resistant fraction that resists decomposition for centuries to millennia (Paul *et al* 2001). While the active and slow fractions can be characterized by ^{13}C , the resistant fraction cannot but it is best characterized by the radioactive isotope of carbon (radiocarbon, ^{14}C) (Paul *et al* 2001). Therefore, a suggestion for further studying SOC dynamics within CTD agricultural fields is to use ^{14}C .

^{14}C has both a natural and artificial source (Paul *et al* 1997, Trumbore 2000). Natural ^{14}C is produced in the upper atmosphere, at an approximately constant rate over decades to centuries, through cosmic rays bombarding with atmospheric nitrogen (N_2) (Paul *et al* 1997, Trumbore 2000). Artificial ^{14}C was produced by thermonuclear (H) bomb testing in the 1950s and early 1960s resulting in a large spike in atmospheric ^{14}C (Trumbore 2000, Kramer & Gleixner 2006, Paul *et al* 1997). The bomb testing increased atmospheric ^{14}C concentration up to 1000 ‰ compared to its concentration prior to 1950 (Kramer & Gleixner 2006). After 1963, the concentration of atmospheric ^{14}C declined to near background levels through mixing with terrestrial and marine carbon pools as nuclear weapon testing was banned (Trumbore 2000, Kramer & Gleixner 2006).

The use of ^{14}C as an isotopic tracer for evaluating SOC dynamics is possible because ^{14}C can be detected in all ecosystems (Trumbore 2000, Kramer & Gleixner 2006, Paul *et al* 1997). ^{14}C can be exchanged between all reservoirs because it gets

oxidized to $^{14}\text{CO}_2$ in the atmosphere, which gets assimilated into plants by photosynthesis and then into soils through decomposition (Trumbore 2000, Kramer & Gleixner 2006, Paul *et al* 1997, Collins *et al* 1999). The ^{14}C content of soil organic matter (SOM) resembles the plants that it was derived from because little isotopic discrimination occurs as plant residues are transformed into SOM (Paul *et al* 1997).

Dr Willard Libby is credited with discovering radiocarbon dating (Libby 1963). Samples for ^{14}C analyses are measured using an accelerator mass spectrometer (AMS) and the results are reported as the per mille (‰) deviation from a reference standard [2.1] (Trumbore 2000, Paul *et al* 1997, Libby 1963). The reference standard for ^{14}C is a 1950 oxalic acid standard and it has a set ^{14}C value of zero ($\Delta^{14}\text{C}=0$) (Paul *et al* 1997, Trumbore 2000). A sample containing a high ^{14}C concentration (*ie* positive $\Delta^{14}\text{C}$) indicates that bomb produced ^{14}C is present and, therefore, the sample is young (Kramer & Gleixner 2006, Trumbore 2000). In contrast, samples with negative $\Delta^{14}\text{C}$ values indicate old carbon (*ie* natural ^{14}C) as it has had enough time for a significant amount of radioactive decay to have occurred (^{14}C half-life = 5730 yr) (Kramer & Gleixner 2006, Trumbore 2000). Thus, ^{14}C can be used as a tracer on a time scale of years to decades because radioactive decay on this time scale is negligible (Trumbore 2000, Kramer & Gleixner 2006).

$$\text{[2.1] } \Delta^{14}\text{C} = \left[\frac{(^{14}\text{C}/^{12}\text{C})_{\text{sample}}}{(^{14}\text{C}/^{12}\text{C})_{\text{standard}}} - 1 \right] \times 1000$$

^{14}C data from SOM and soil respiration can be used for determining soil carbon dynamics (Trumbore 2000). The carbon contribution from new plants and from old recalcitrant parts affect the ^{14}C content of SOM and in turn soil respiration (Kramer &

Gleixner 2006). By measuring ^{14}C of SOM and soil respiration, the age of the soil carbon pools and their turnover times can be identified, and the source of soil respiration can be separated into recent and old carbon sources (Hopkins *et al* 2013, Trumbore 2000, Kramer & Gleixner 2006). For example, Kramer & Gleixner 2006, used ^{14}C to determine the contribution of recent plant material and older SOM content to soil respiration in agricultural soils. Paul *et al* 1997 used ^{14}C -dating techniques to measure the age of SOM and the turnover rate for a range of sites in the North American grasslands. Trumbore 2000 studied ^{14}C of soil respiration and SOM in forest ecosystems to identify the turnover times of SOM and to separate soil respiration. ^{14}C can also be used to separate total soil respiration (R_T) into its contributions from rhizosphere (R_{rh}) and soil (R_s) (eg. Hopkins *et al* 2013, Borken *et al* 2006). Furthermore, the ^{14}C of soil respiration can help to explain the causes for variability in soil respiration within a season and from one year to the next (Trumbore 2000). Thus, there are many advantages for using ^{14}C to study soil carbon dynamics.

By combining soil GHG effluxes with ^{13}C tracer, in the current study, has provided a great deal of information on soil GHG emissions and their associated SOC dynamics in CTD agricultural fields. The natural abundance of ^{13}C was successfully used as a signal to follow SOC dynamics. However, there is a missing link in this study, which is dating and identifying the turnover times of the SOC pools. Therefore, it is suggested to use the useful but expensive ^{14}C tracer to date and identify MRT of the SOC pools and thus to have data from both carbon tracers to complete this study. Both the stable isotope ^{13}C and the radioactive isotope ^{14}C are appropriate tracers as they are both present in atmospheric CO_2 and as a result they get incorporated into plants

and soils (Collins *et al* 1999). Therefore, in combination with soil GHG effluxes and ¹³C isotope measurements, ¹⁴C can provide additional valuable information to the study through dating techniques.

3. References

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