Greenhouse Gas Emissions from Onion Fields Cultivated on Organic Soils under Sprinkler Irrigation in Quebec

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Abstract

Agricultural practices contribute to greenhouse gas emissions. A four year field study was conducted to quantify and compare CO_2, N_2O and CH_4 fluxes from sprinkler irrigated and non-irrigated onion fields in southern Quebec, Canada. Irrigation practices influence GHG emissions by changing the soil moisture content and thus impacting the soil microbial activity. The experimental plots were located on three organic soils with different degrees of stabilization. The static chamber method was used to obtain in-situ gas fluxes. Meteorological and soils data were also collected. Results for CO_2, N_2O and CH_4 fluxes ranged from 1 to 268 mg CO_2-C m^{-2}*hr^{-1}, -1.06 x 10^{-4} to 0.566 mg N_2O-N m^{-2} * hr^{-1} and -0.00628 to 0.00760 mg CH_4-C m^{-2} *hr^{-1}, respectively. Results showed that sprinkler irrigation had minimal effects on N_2O and CH_4 gas fluxes, however, the CO_2 fluxes increased within 24 hours of an irrigation event. In fact, CO_2 fluxes were found to be more prominently influenced by the growth stage of the plant. Higher CO_2 fluxes were observed, both, earlier and later in the season when root and leaf growth, respectively, were at their maximum. For N_2O, higher fluxes were observed primarily in the spring after snow melt and fertilizer application. As well, N_2O fluxes were influenced by heavier rainfalls (>10 mm) and wetter soils (WFPS between 70 and 100%). Organic soils for this research were predominantly methane sinks with slight increases in CH_4 flux observed following fertilizer application and soil tillage. Since greenhouse gas fluxes were sporadic and seldom linked to irrigation events, it is concluded that sprinkler irrigation had a limited impact on greenhouse gas emissions from the organic soils in this study.

Résumé

Les gaz à effet de serre provenant des pratiques agricoles sont de très importants contributeurs aux émissions globales. Une étude sur le terrain, d'une durée de quatre ans, a été menée dans le sud du Québec au Canada afin de déterminer et comparer les émissions de CO_2, N_2O et CH_4 de champs d'oignons irrigués par arrosage et non-irrigués. L'irrigation, en particulier, affecte les niveaux d'émissions en changeant l'humidité du sol et ainsi influençant l'activité microbienne du sol. L'étude a été exécutée sur trois sols organiques à différents stades de stabilisation. Les flux de gaz ont été obtenus dans le champ en utilisant la méthode de chambre statique. Les données temporal et spatial par rapport au sol ainsi que les données météorologiques de la région ont été collectionnées afin d'expliquer les résultats des émissions de gaz. Les résultats pour les flux de CO_2, N_2O et CH_4 variait de 1 à 268 mg CO_2-C $m^{-2} * hr^{-1}$, -1.06 x 10^{-4} à 0.566 mg N_2O-N $m^{-2} * hr^{-1}$ et -0.00628 à 0.00760 mg CH_4-C $m^{-2} * hr^{-1}$, respectivement. L'irrigation par arrosage avait des effets minimes sur les gaz N_2O et CH_4. L'analyse des flux de CO_2 montre que dans les 24 heures après une application d'irrigation les émissions ont augmenté. Cependant, le stade de développement de la plante avait un effet majeur sur les flux de CO_2. Une augmentation d'émissions de CO_2 a été remarquée au début de la saison quand les racines étaient à un stade maximal de développement ainsi que plus tard dans la saison quand les feuilles étaient à un stade maximal de développement. La principale hausse d'émissions de N_2O a été observée au printemps juste après la fonte des neiges et l'application des engrais. Pendant la saison d'échantillonnage, les averses de pluie (>10 mm) et les sols plus humides (espace poreux rempli d'eau entre 70 et 100%) ont provoqué des augmentations de flux de N_2O. Les sols organiques étudiés étaient principalement des puits de méthane. La production de CH_4 a augmenté légèrement après l'application d'engrais et le labour du sol. Puisque les flux de gaz à effet de serre au-delà du niveau de base étaient irréguliers et rarement lié aux événements d'irrigation, il est conclu que l'irrigation par arrosage n'avait pas d'effet majeur sur les émissions de gaz à effets de serre des sols organiques de cette étude.

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List of Abbreviations and Symbols

BD	Bulk density
BMP	Best management practice
CH_4	Methane
CH_2O	Formaldehyde
$C_6H_{12}O_6$	Glucose
CO_2	Carbon dioxide
ECD	Electron capture detector
FID	Flame ionization detector
GC	Gas chromatograph
GHG	Greenhouse gas
H_2O	Water
KCl	Potassium chloride
N_2	Nitrogen gas
NH_3	Ammonia
NH_4^+	Ammonium
NH_2CONH_2	Urea
NH_4NO_3	Ammonium nitrate
NH_2OH	Hydroxylamine
NI	No irrigation
NO	Nitric oxide
NO_2^-	Nitrite
NO_3^-	Nitrate
N_2O	Nitrous oxide
O_2	Oxygen
OH^-	Hydroxide
OM	Organic matter
S1.1	Site 1 (moderately stabilized soil) in 2012 and 2013
S1.2	Site 1 (moderately stabilized soil) in 2014

S1.3	Site 1 (moderately stabilized soil) in 2015
S2.1	Site 2 (most stabilized soil) in 2012 and 2013
S2.2	Site 2 (most stabilized soil) in 2014
S2.3	Site 2 (most stabilized soil) in 2015
S3.1	Site 3 (least stabilized soil) in 2012
S3.2	Site 3 (least stabilized soil) in 2013 and 2015
S3.3	Site 3 (least stabilized soil) in 2014
SI	Sprinkler irrigation
SOM	Soil organic matter
VWC	Volumetric water content
WFPS	Water filled pore space
WTD	Water table depth

Chapter 1 – Introduction

1.1 Problem Definition

Global emissions of greenhouse gases (GHGs) have increased by 42 % between 1990 and 2011 (Environment Canada, 2015a). According to the Intergovernmental Panel on Climate Change, the Agriculture, Forestry and Other Land Use (AFOLU) sector is responsible for just under a quarter of anthropogenic GHG emissions (Smith et al., 2014). Canada's emissions account for 1.6 % of global emissions. Around 8 % of Canada's total GHG emissions are from the agricultural sector (Agriculture and Agri-Food Canada, 2014). This excludes emissions from the use of fossil fuels or from fertilizer production. The main gases emitted by agricultural activities are carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4).

Agricultural soils can be both sources and sinks of GHGs based on different spatial and temporal environmental conditions. Carbon and nitrogen in the form of organic and inorganic material are added to the soil through multiple sources. They are released to the atmosphere in the form of CO_2, N_2O and CH_4 through multiple processes including plant respiration, decomposition of dead plant biomass and soil organic matter, and combustion (Smith et al., 2014).

In this study, GHG fluxes were collected using the in situ static chamber method and analyzed in respect to various spatial and temporal conditions. Very little research has been conducted on GHG emissions from organic soils cropped to vegetables and irrigated. Onion crops cultivated on organic soils under sprinkler irrigation are of particular interest for several reasons. Organic soils have high rates of decomposing organic matter (OM) which is the main substrate for soil microbes producing GHGs (Kasimir-Klemedtsson et al., 1997). Onions are an important crop in terms of food production and are grown worldwide (Brewster, 2008a). The emissions from this type of crop differ from those of other vegetable crops due to different required agricultural practices (i.e. fertilizer application, water application, seeding and cultivation) and plant specific properties (i.e. nitrogen uptake, microbial activity and growth stages). The use of sprinkler irrigation on organic soils was studied due to its impact on soil moisture content and gas diffusion which are known drivers of GHG emissions (Sainju et al., 2012).

Overall, this study was conducted in order to evaluate the fluxes of CO_2, N_2O and CH_4 from onion cultivated organic soils based on the degree of soil stabilization and the use of irrigation.

1.2 Objectives

i. Compare the GHG fluxes from sprinkler irrigated and non-irrigated onion fields on organic soils.
ii. Investigate how GHG fluxes differ from three organic soil sites at different states of decomposition.
iii. Assess the impact of environmental conditions and agricultural management practices on GHG fluxes from organic soils under onion production.

1.3 Scope

The research was conducted at a field in Sherrington, Quebec, and a field in Napierville, Quebec. This specific region is known for producing onions and other vegetables since the soils are high in organic matter content and nutrients. The soils derive from old wetlands that have been drained to be used for agricultural purposes. The farm chosen, was based on its unique soil type and hand moved sprinkler irrigation system. All agronomic practices were undertaken by the producer including crop type, tillage, fertilization, irrigation and harvest.

The three sites selected differed in terms of soil organic matter content state of decomposition. Crop rotation was done at the field sites each year to prevent diseases. Results from this study could be applied to fields with comparable crop type, soil type and climate. Further, application of the results could be put to practice by recommending how best to apply sprinkler irrigation to onions while avoiding GHG emissions.

Based on the accumulated information, best management practices (BMPs) to mitigate GHGs overall can be developed to increase water use efficiency and ideally reduce the emissions of GHGs from agricultural organic soils.

Chapter 2 – Literature Review

In order to accurately assess the results of this study, it is important to understand the different factors being observed.

First, it is necessary to examine how onions are produced. Soil type plays a great role in the emissions of GHGs, therefore, the basic aspects and conditions of an organic soil should be known and understood. General agricultural practices for the production of onions such as cropping, fertilization and irrigation scheduling will have an impact on emissions. By understanding the general onion production standards, results from this study could further be applied to other onion production fields.

Second, it is essential to understand how the three gases (CO_2, N_2O and CH_4) are produced within the soil and what can trigger increases in production or consumption.

Finally, a general knowledge of how the application of water can impact GHGs will give a good understanding and outlook on the results obtained in this study.

2.1 Onion production

2.1.1 Soil type

The soil type to be examined for greenhouse gas fluxes is an organic soil or more commonly termed as a muck soil. Muck soils are the result of long term plant residues that have been preserved by a high water table and thus lack oxygen (McDonald, 2010). Organic soils are commonly differentiated based on their stage of decomposition (Cowan, 2005). They range from fibrous raw peat with scarcely any decay to fine, dark, advanced decay defined as muck. Muck soils are often used in vegetable production such as onions since they have a high soil organic carbon content. Organic matter in the soil provides nutrients, improves soil structure, maintains tilth and minimizes erosion (Vickers et al. (2015). Muck soil types are high in fertility, retain moisture and supply plant nutrients (International Union of Soil Sciences, 2014). Unfortunately, due to their fine powdery characteristics, they are highly susceptible to wind erosion when dry. Management of erosion is done by wetting the soil via irrigation. A muck soil is lightweight with a bulk density (BD) between 0.2 – 0.3 g/cm^3 (Brady and Weil, 2007). Their porous nature favors

gas diffusion. In general, their carbon to nitrogen ratio is fairly high (C:N of 20:1 or higher). Additionally, muck soils are very effective in nitrification and can carry large amounts of nitrogen. Due to their dark colour they typically absorb heat more readily (Gerrard (2000); Yerima and van Ranst (2005)). However, due to their higher water retention capabilities, poorly drained organic soils may warm up slower than a well-drained lighter coloured soil (Yerima and van Ranst, 2005). Limitations that may arise when examining these types of soil are their large spatial variability and the changes in physical properties (Schwärzel et al. (2002); Kechavarzi et al. (2010)). Schwärzel et al. (2002) explained that organic soils are not completely uniform and can vary greatly.

Soil organic matter (SOM) is subject to change. The rate of decomposition and accumulation of SOM is determined by the soils properties such as texture, pH, temperature, moisture, aeration, clay mineralogy and soil biological activities. Organic matter is mainly added to the soil through crop residues and manure. When the rate of addition is less than the rate of decomposition, the SOM will diminish (Bot and Benites, 2005). Over time, wind erosion and agricultural practices such as tillage can attribute to soil subsidence and carbon loss through biochemical oxidation thus resulting in degraded muck soils (Reicosky et al., 2008). Highly productive muck soils generally have higher SOM (i.e. have not been tilled or used for agriculture extensively). Decomposition of SOM provides the main substrate for the GHG producing microorganisms (Kasimir-Klemedtsson et al., 1997).

Muck soils have a high water holding capacity, as mentioned previously, since water is retained within the organic matter. It can retain approximately 2 to 4 times its weight in water (Brady and Weil (2007); Kuntze (1972)). Vegetable production such as onions favour the use of organic soils due to their high water holding capacity. Onions require constant available water during bulb formation (George, 2011). However, although their ability to retain moisture is very high, this does not necessarily mean that there is more available water to supply the plants. Compared to mineral soils, the proportionality of unavailable water is much higher. Organic soils are very lightweight which consequently means that a given volume of organic soil will not hold as much water as the same volume of mineral soil (Brady and Weil, 2007). Kechavarsi et al. (2010) mentioned that muck soils will have differing soil properties when dry. As the soil dries it will shrink. Thus, the effect of the shrinkage will have an impact on hydraulic conductivity

characteristics. Therefore, constant monitoring of the soil available water and irrigation is necessary to optimally produce onions (Rekika et al., 2014).

2.1.2 Onion cropping practices in muck soils

Onions grow best in sand, silt or peat soils (Brewster, 2008a). Good potential moisture retention, as seen with muck soils, is ideal for onion production since they have relatively shallow root systems and require continuous available water for proper bulb formation. Often, onions cultivated in muck soils are planted on raised soil beds to avoid wet soil conditions which can be harmful to the crop's production. This type of crop can either be produced from direct seeding or by small bulbs (George, 2011). It is best to plant them when temperatures are cool. At higher latitudes it is more efficient to sow the seeds in the spring (Brewster, 2008a). The seeds should be planted as soon as the soils have begun to warm up after spring thaw. Warmer temperatures are preferred when the plant begins to mature. Seeding is done in early to mid-May. The seeds are sown at approximately 1 cm depth, 10 cm apart and in rows 30 cm apart (George, 2011). Since muck soils are highly susceptible to wind erosion, a cover crop, such as barley, is typically seeded at the same time along with the onions to avoid soil erosion and to protect the onions from wind damage (Ngouajio, 2012). According to George (2011), onion bulb maturation occurs in approximately three to five months depending on the cultivar and local climate. A study in Switzerland, which had similar sowing dates as a Quebec onion crop, examined the days of planting and the growth stages of the plant (Figure 2.1). Based on this study, seeds would germinate approximately two weeks after planting, leaf shoots would reach a maximum around July or beginning of August and bulbs would begin to grow in June and finally reach maturation near the end of August or beginning of September (Brewster (2008b); Schwartz and Cramer (2011)). Harvesting of onions is done in the fall when about 50-80 % of the crop has soft necks and the canopy has begun to collapse (Brewster, 2008a). It is done by pulling out the bulbs and allowing them to dry and cure in the field. They are left for a week or two then removed to be packed in crates or sacks (Brewster, 2008a). Since onion crop production is relatively intense, the implementation of crop rotation each year is essential in order to maintain and improve soil fertility (George, 2011).

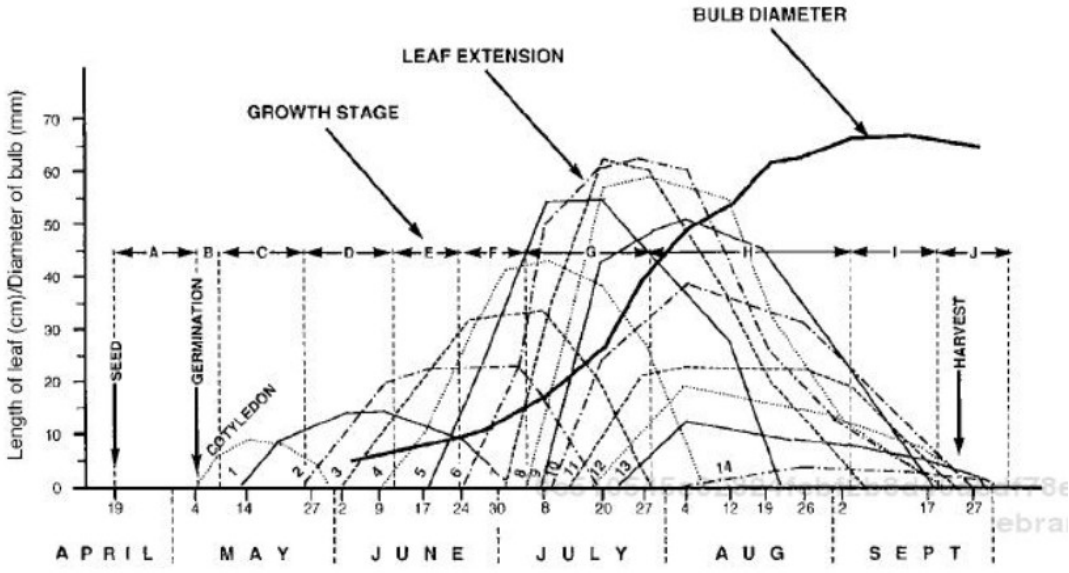


Figure 2.1: The growth stages of onion leaves and bulb as seen in Switzerland (Brewster, 2008b).

2.1.3 Onion irrigation schedule

Onions are a shallow-rooted (less than 40 cm) vegetable crop that require frequent but light irrigation (Verhallen (2009); Brewster (2008a)). They cannot tolerate continuous wet soil conditions, yet, insufficient water supply could result in reduced crop production and quality (George (2011); Rekika et al. (2014)). Constant monitoring of the soil water potential within the root zone is beneficial in establishing adequate irrigation schedules. Soil water potential and irrigation scheduling depends on the following factors: climate, soil conditions, irrigation system, crop development stage, and cultivar. The most critical period for irrigation (i.e. when the vegetable is most sensitive to water stress) is during bulb formation and enlargement (Rekika et al., 2014). Irrigation should be applied at -20 kPa or when soil moisture drops below 50 % (Rekika et al. (2014); Advisory Committee on Vegetable Crops (2000)). For organic soils, irrigation is often applied in the spring right after planting to avoid soil wind erosion. In some regions, rainfall is adequate enough to supply water to the crop and over watering could result in decreased yields (Brewster, 2008a).

Sprinkler irrigation is well suited for most row crops (Brouwer et al., 1985). It is often used for irrigating onion crops. The hand moved sprinkler system, as seen below in Figure 2.2,

can effectively meet the water requirements of an onion field. It is an overhead sprinkler system which consists of a pump unit, mainline, laterals and sprinklers. The laterals are moved in and out of the field when needed. The water discharged through the sprinklers is shot into the air and falls back down to the ground in a circular pattern (Phocaides, 2000). The water however is not uniformly distributed. A higher density of water will fall closer to the sprinkler. Therefore, the system in the field is set up so as to have the irrigation circular patterns overlapping as seen below in Figure 2.2. The application rate is the average rate at which water is sprayed on the crop and it is measured in mm/hr. When deciding an application rate, it is important to take into account the average infiltration rate of the soil. To avoid runoff, the average application rate should be less than the basic infiltration rate of the soil. The downside of this system is the loss of water through evaporation. Field studies reported that droplet evaporation losses from sprinklers ranged between 2-45 % (Uddin et al., 2010). Factors affecting evaporation loss include equipment-related factors (e.g. nozzle size, angle, operating pressure and height of the sprinkler) and climatic factors (e.g. air temperature, air friction, relative humidity, solar radiation and wind velocity).

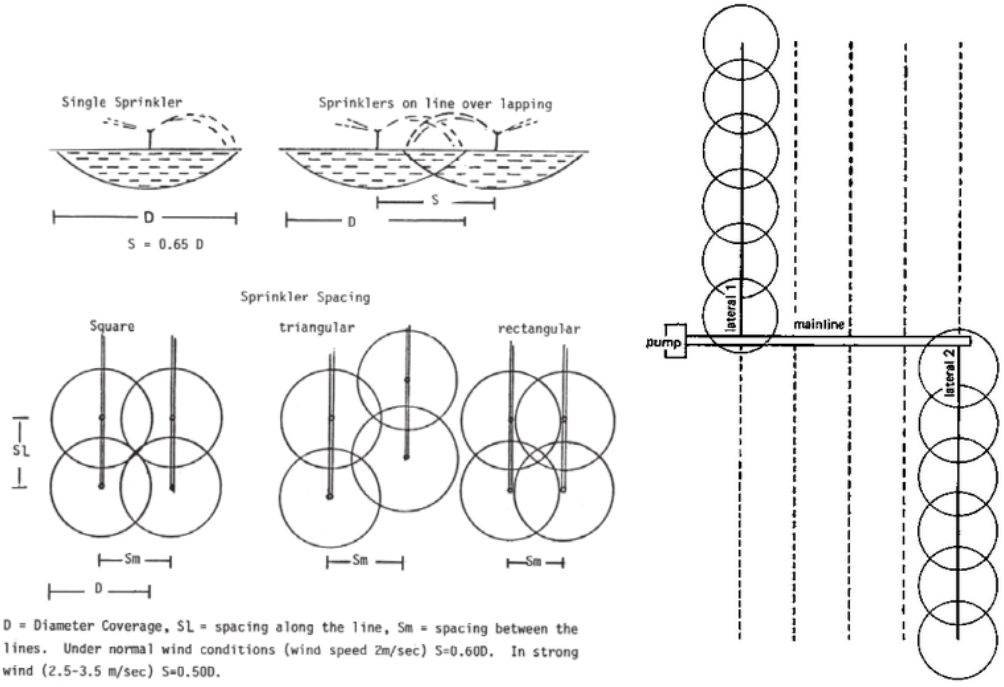


Figure 2.2: Hand moved over head sprinkler system (left) and profile of wetted soil under sprinklers (right) (Brouwer et al. (1985); Phocaides (2000)).

2.1.4 Onion fertilization schedule

Onions require a well-balanced highly fertile soil (Advisory Committee on Vegetable Crops, 2000). In muck soils a pH of 4 is sufficient (Brewster, 2008a). Nitrogen is applied as nutrients. For organic soils cropped with onions the required amount of N is approximately 157 kg/ha (Warncke et al., 2004). Generally, a part of the total nitrogen (< 56 kg/ha) can be incorporated pre-plant. The remainder of the required nitrogen is applied in one or several applications in mid to late June after the seeded onions have reached about 15 cm in height. An excess of nitrogen applied later in the season may cause delayed maturation. Phosphorus (P) and potassium (K) are often added to the soil. The recommended amounts of P and K vary depending on the nutrients already present and available in the soil. The recommended amounts of P and K range from 26-129 kg/ha of P and 50-250 kg/ha of K (Brewster, 2008a).

2.2 Carbon Dioxide (CO_2)

The total global emissions of CO_2 from soils is documented as one of the main contributors to the global carbon cycle (Schlesinger and Andrews, 2000). It is important to understand how carbon is added and retained in the soil and how it is released. Carbon inputs come from plant residues (originating from CO_2 capture during photosynthesis, minus plant respiration) that are left after harvesting the crop and from other organic residues (e.g., manure, compost, cover crops). During the decomposition process, part of the CO_2 from these inputs is released to the atmosphere (i.e., soil respiration) and the remainder is retained in the soil organic carbon pool. The soil organic carbon is a dynamic pool and susceptible to further decomposition and transformation, particularly when disturbed through agricultural practices (e.g., tillage, land leveling) and natural processes like erosion (Scharlemann et al. (2014); Rastogi et al. (2002)). These disturbances tend to stimulate soil respiration (USDA, 2012).

2.2.1 Soil Respiration

Soil respiration is the release of carbon dioxide from the soil to the atmosphere (Franzlubbers and Haney, 2006). Approximately 20% of the total emissions of CO_2 derive from

soil respiration (Rastogi et al., 2002). It includes three biological processes: microbial respiration, root respiration and faunal respiration. These processes that produce CO_2 emissions occur mainly at the soil surface or within the upper layer where most plant residue is concentrated.

Chemical oxidation, which is enhanced at higher temperatures, is the non-biological process that releases CO_2 to the atmosphere. Certain soil characteristics such as texture, temperature, moisture, pH, and available carbon (C) and nitrogen (N) content can influence the soil production and emission of CO_2.

Through the process of photosynthesis carbon dioxide is converted to soil C by the plants (Luo and Zhou, 2006). The following equation 2.1 represents the reaction:

$$6\ CO_2 + 12\ H_2O + \text{light} \rightarrow C_6H_{12}O_6 + 6\ O_2 + 6\ H_2O \quad (2.1)$$

Root respiration is the process of converting this C source back into CO_2. The following equation 2.2 represents the plant respiration also known as aerobic respiration of organic compounds (Luo and Zhou, 2006):

$$C_6H_{12}O_6 + 6\ O_2 \rightarrow 6\ CO_2 + 6\ H_2O + \text{energy} \quad (2.2)$$

Microbial respiration results from the breakdown by soil microbes of organic material such as litter fall, root mortality, application of manures and crop residues (Rastogi et al., 2002). Microbes break the C bonds of dissolved organic compounds to survive and gain energy (Li, 2007). This process requires the transfer of electrons from the dissolved organic carbon to electron acceptors. Oxygen (O_2) is the main electron receiver in the soil. It has the lowest Gibbs free energy which is the main energy reaction in living cells (Schroeder, 2000). Ionized oxygen is combined with the dissociated C through electron transfer and forms CO_2. Most soil microbes under aerobic conditions will use oxygen as an electron acceptor for producing CO_2.

According to Rastogi et al. (2002), the amount of available C for transfer to CO_2 can be defined by the carbon dynamics of a system. This in turn is determined by different carbon pools: a decomposable pool with a radiocarbon age of less than one year, a biomass pool with a radiocarbon age of 25.9 years and a chemically stabilized pool with a radiocarbon age of 2565 years. The first two pools (decomposable and biomass) create labile carbon. This type of carbon

breaks down and can be restored much faster than non-labile carbon which is created by the third pool (chemically stabilized). Labile carbon is easily broken down and is the major food source for soil microbes (Hoyle et al., 2015).

Faunal respiration is similar to microbial respiration. It is the breakdown of organic matter by the fauna that use it as a food and energy source. Microbial respiration dominates faunal respiration by a ratio of 9:1 (Blackwood et al., 2007).

2.2.2 Effect of soil moisture on CO_2 emissions

When a soil is fully aerobic, the soil microbes use O_2 as the main electron acceptor (Li, 2007). Under these circumstances, the main gas produced in the soil is CO_2. As water is added to the soil, the amount of available O_2 gradually depletes. As the soil becomes more anaerobic, less CO_2 is emitted to the atmosphere. Under anaerobic conditions, CO_2 is consumed rather than produced. For example, natural water-saturated peatlands will sequester carbon dioxide from the atmosphere and anaerobic bacteria will convert it to methane (Berglund and Berglund, 2011). When these lands are drained and cultivated it increases the aeration of the soils leading to an increase in CO_2 emissions.

Several studies examined the effect different water table depths (WTD) had on the emissions of the GHG CO_2 in situ. Most authors argued that the correlation was very poor (Rastogi et al., 2002). Lafleur et al. (2005), studied the effects of WTD on CO_2 fluxes in a dry peatland ecosystem. Previous laboratory incubation experiments found that CO_2 production was most greatly affected by moisture variations in the uppermost layers of the peat. However, since soil moisture in the upper layers is relatively invariant, WTD has little impact on soil respiration. In the field, it was found that the soil moisture in the top layers had a positive linear relation with respiration which corresponded with the laboratory results (Lafleur et al., 2005). This can perhaps be explained by the fresh detritus and high nutrient supply. A study performed on European soils in incubation found that highest CO_2 fluxes occurred at intermediate moisture content (40% to 60%) and at relatively higher temperatures (above 10 °C) (Gritsch and Zechmeister-Boltenstern, 2014). It was seen that at low temperatures (5-10 °C), moisture content had little to no effect on CO_2 flux. The final conclusions were that the gas fluxes were controlled

mainly by the variability in soil temperature and moisture. Rastogi et al.'s (2002) findings were similar to those of Gritsch and Zechmeister-Boltenstern (2014). They found that an increase in soil moisture would increase CO_2 production up to an optimum level. After it reached its peak, the carbon dioxide would decelerate in production. Rastogi et al. (2002) also tested the effects of periodic drying and wetting of soil. Their results indicated that microbes in latent state in dry soil would become active when exposed to wet conditions. This activity alongside the release of trapped air within the soil pores ultimately contributes to the increase in CO_2 flux. Xu et al. (2004), following after Rastogi et al. (2002), evaluated the impacts of soil moisture, rain pulses, and growth on the response of ecosystem respiration to temperature. Measurements were taken from a grassland in California where there is a dry season and a wet season. Congruent with previous studies, their laboratory incubation experiments proved that when soil was maintained at a constant temperature (15 °C) microbial respiration would increase with increasing soil moisture (Xu et al., 2004). However, the flux would decrease over time as the labile carbon pool would deplete. In the field, they examined the effects that a substantial rain event would have on the CO_2 in the soil pores. Prior to the rainfall events the soil would be very dry. After the rainfall there would be an immediate response in soil CO_2 flux.

2.2.3 Other factors affecting CO_2 emissions

The most critical factors affecting soil respiration are soil temperature, soil moisture, soil organic carbon, and soil texture (Lohila et al., 2003).

Fenn et al. (2010) found that soil temperature was more strongly correlated to CO_2 flux than soil moisture. Throughout the season, as temperature increased so did CO_2 fluxes (Fenn et al., 2010). The International Union of Soil Sciences (2014) concluded that the increase of CO_2 fluxes in correlation with soil temperature was due to an increase in decomposition rate. As soil temperature increases, photosynthesis and root respiration rates will increase as well. This leads to higher root exudation which serves as a carbon source for microbes. A higher carbon source will enhance the growth of the microbe population leading to an increase in decomposition and thus more soil respiration (International Union of Soil Sciences (2014); Xu et al. (2008)).

The growth stage of the plant has also been studied and seen to have a significant correlation with CO_2 flux (Schlesinger and Andrews, 2000). As the plant matures, emissions tend to increase. This can be explained by an increase in photosynthesis as the plant develops. Through the process of photosynthesis, carbohydrates are stored in the plant and released through the roots where they are then oxidized into CO_2 by the rhizospheric bacteria (Gerrard (2000); Curiel Yuste et al. (2007); Xu et al. (2008)). The larger the plant is, the more carbohydrates it will produce. Studies have also shown that there are higher fluxes shortly after harvest due to higher biological activity that decomposes fresh roots (Elder and Lal, 2008). This can also be explained by higher soil temperatures due to the lack of plant cover. Lee et al. (2009) observed lower CO_2 fluxes during the fallow/winter season which they justified as being due to the absence of active root respiration.

When a wetland is drained and cultivated, the soil transitions from being CO_2 sinks to sources (Elder and Lal, 2008). This is mainly caused by the subsequent subsidence that occurs in drained peat soils. The main cause of subsidence in organic fields is from aerobic decomposition of SOM. Subsidence consequently shifts C and N dynamics drastically resulting in CO_2 emissions. Through the process of cultivation and other land uses, soil organic matter content will deplete over time (Piccolo, 2012). According to Fierro and Forte (2012), most organic matter is lost within the first ten years of land use. Cultivation exposes fresh topsoil to rapid surface drying and air oxidation thus resulting in loss of carbon to the atmosphere through the production of CO_2 and enhanced mineralization (Fierro and Forte, 2012). This loss of SOM is mainly attributed to the loss of labile C. As mentioned previously, labile C is the main food source for soil microbes (Hoyle et al., 2015). Therefore, it can be justified that more mineralized soils (older muck soils) will produce CO_2 at a slower rate than more organic soils (newly cultivated muck soils) since there is less readily decomposable organic matter (Coban et al., (2015); Panosso et al. (2011)).

Studies have shown that tillage of organic soils leads to higher emissions as compared to no-till (Morris et al., 2004). Elder and Lal (2008) studied the effects of tillage on soil respiration. Tillage increases CO_2 flux by two main pathways. First by loosening and inverting soil which allows CO_2 gas to escape and O_2 to enter (i.e. increased diffusion) and second by the mixing of residues which stimulates microbial activity (Gerrard, 2000).

2.3 Nitrous Oxide (N_2O)

Nitrogen exists in the atmosphere as N_2. It is transformed into fixed N by breaking the N≡N triple bond through physical and biological processes. Agricultural and natural soils are significant contributors to the total atmospheric N_2O (Nieder and Benbi, 2008). According to Butterbach-Bahl et al. (2013), they represent 70 % of global N_2O emissions. Cultivated organic soils are a major agricultural source of these N_2O emissions. When organic soils are drained for agriculture there is an increase in the amount of available oxygen in the soil. With increasing availability of oxygen, the decomposition of organic matter is accelerated. Thus, resulting in the mineralization of large quantities of organic N (Rochette et al., 2010). Mineralization of SOM and release of N from organic and synthetic fertilizers and ensuing nitrification provides reactive, inorganic, N forms such as ammonium (NH_4^+), nitrite (NO_2^-), and nitrate (NO_3^-) (Nieder and Benbi, 2008). Denitrification subsequently follows by transforming NO_3^-, via microbial processes, into NO, N_2O and N_2 (Senbayram et al., 2012). In order to understand how it is impacted by irrigation, it is important to understand each of these processes and how they are impacted by the addition of water.

2.3.1 N_2O production in soil

Nitrous oxide emissions from the soil are mainly characterized by ‘hot spots’ and ‘hot moments’ since they depend greatly on spatio-temporal variability (Butterbach-Bahl et al., 2013). N_2O from the soil is produced by the following biological processes: denitrification, nitrification and nitrifier denitrification (Bateman and Baggs, 2005). In order to understand how it is produced, it is essential to understand the dynamics of nitrogen (N) between the soil profile and the atmosphere. The N cycle in the soil is depicted in Figure 2.3.

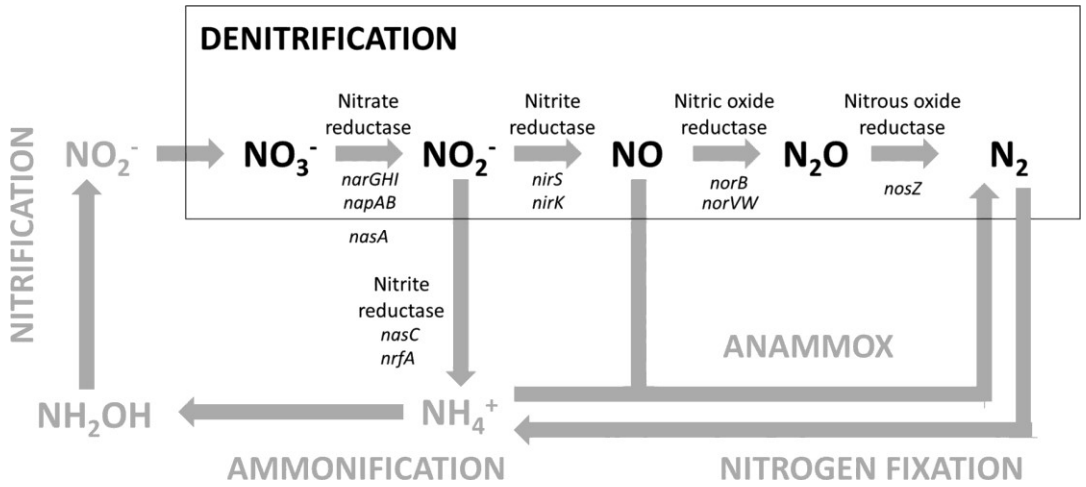


Figure 2.3: Nitrogen cycle between soil profile and atmosphere (Alvarez et al., 2014).

2.3.1.1 Nitrogen Fixation

Nitrogen fixation into reactive forms of N is completed through four major processes: lightning fixation, biological fixation, synthetic N fertilizer production and high temperature combustion fixation (Ussiri and Lal, 2013). Plant and animal matter such as crop residues and manure contribute organic N to the soil (Brady and Weil, 2007). Synthetic fertilizers contribute by adding ammonium nitrate (NH_4NO_3) or urea (NH_2CONH_2). Nitrogen (N_2) is fixed into ammonia (NH_3), ammonium ions (NH_4^+) or any organic compound by the biological reaction catalyzed by the nitrogenase enzyme which breaks the N_2 triple bond (Signor and Cerri (2013); Ussiri and Lal (2013)). This process is done so by free-living and symbiotic bacteria, archaea, and specialized plants. It can be depicted by the general equation 2.3 in which two moles of ammonia are produced from one mole of N_2.

$$N_2 + 6\ e^- + 6\ H^+ \rightarrow 2\ NH_3 \qquad (2.3)$$

2.3.1.2 Nitrogen Mineralization

Mineralization is a series of biological transformations which convert organic N into ammonium ions. It is also known as ammonification and is completed by soil microbes (Baggs

(2011); Crohn (2004)). The following equations 2.4 & 2.5 are an example of how urea is mineralized (Haynes, 1986).

$$NH_2CONH_2 + H_2O \rightarrow 2\ NH_3 + CO_2 \quad (2.4)$$

$$NH_3 + H_2O \rightarrow NH_4^+ + OH^- \quad (2.5)$$

2.3.1.3 Nitrification

Nitrification is the process by which NH_4^+ is oxidized to form nitrite (NO_2^-) and/or nitrate (NO_3^-) ions (Signor and Cerri, 2013). This transformation characteristically occurs in aerobic conditions since it requires oxygen in order to perform. There are two stages carried out by autotrophic bacteria: nitritation and nitratation. Nitritation and nitratation are achieved by the bacteria Nitrosomonas sp. and Nitrobacter sp, respectively. The following equations 2.6 & 2.7 define each step (Sayavedra-Soto and Arp (2011); Starkenburg et al. (2011)):

1) Nitritation:

$$2\ NH_4^+ + 3\ O_2 \rightarrow 2\ NO_2^- + 2\ H_2O + 4\ H^+ + \text{energy} \quad (2.6)$$

and,

2) Nitratation:

$$2\ NO_2^- + O_2 \rightarrow 2\ NO_3^- + \text{energy} \quad (2.7)$$

2.3.1.4 Denitrification

Denitrification is the microbial process of nitrate reduction that produces nitric oxide (NO), nitrous oxide (N_2O) and molecular nitrogen (N_2) (Senbayram et al., 2012). Each step is catalyzed by specific enzymes and typically occur under anaerobic conditions. The rate at which denitrification occurs in a soil is determined by oxygen availability, soil moisture, soil type, pH, nitrate concentration, and availability of labile carbon compounds. Higher denitrification rates do not necessarily lead to higher N_2O losses. It is more definitively determined by the product ratio $N_2O/(N_2O + N_2)$. This ratio determines how much of each gas will be produced and released to

the atmosphere. It is affected by the availability of C and the ratio of nitrate to available C (C:N) in arable soils. As previously mentioned, ions are created through the nitrification process. They are then reduced through the denitrification process. This is represented by the stepwise reduction of NO_3^- to N_2 (equation 2.8):

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \quad (2.8)$$

The full completion of the process will emit N_2. However, an incomplete process will result in variable fractions of emissions of NO and/or N_2O.

2.3.1.5 Nitrifier Denitrification

Nitrifier denitrification is different from coupled nitrification-denitrification in that it reduces NH_3 to N_2. NH_3 is oxidized to NO_2^- which is then reduced to NO, N_2O and finally N_2 (Wrage et al., 2001). It is carried out by autotrophic nitrifiers. The following equation 2.9 represents the transformation:

$$NH_3 \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \quad (2.9)$$

2.3.2 Effect of soil moisture on N_2O Emissions

In order to understand how irrigation will have an impact on N_2O emissions, it is important to grasp how soil moisture will have an effect on the soil microbes responsible for N_2O production. Butterbach-Bahl et al. (2013) suggest that soil moisture regulates the oxygen availability to the soil microbes. Oxygen availability will decrease as soil moisture increases. Klemedtsson et al. (2005) studied different soil properties and their effects on GHG fluxes. Their results came back inconclusive for the effect of water table depth (WTD). They were unable to find any correlation to N_2O fluxes (Klemedtsson et al., 2005). Similarly, Berglund and Berglund (2011) studied the influence of WTD and soil properties on emissions of greenhouse gases from cultivated peat soils. Their results, contrary to those of Klemedtsson et al. (2005), indicated that N_2O fluxes were greater at high water table levels (40 cm). Davidson (1992) performed a study evaluating the sources of nitric oxide (NO) and nitrous oxide (N_2O) following the wetting of dry soil. Based on his results, he suggested that N_2O fluxes were at a maximum when soil moisture

was between 50-80 %. Similarly, Butterbach-Bach et al. (2013) found that N_2O fluxes were at an optimum between 70-80 %. Above 80 %, the denitrification process will generally convert N_2O all the way to N_2 (Davidson, 1992). However, below 50 % soil moisture, denitrification will no longer occur and instead nitrification will produce more NO than N_2O. Maljanen et al. (2003) had comparable results with higher fluxes seen in the spring and early summer. They found that N_2O fluxes were highest at 80-90% water filled pore space (WFPS) and lowest at 40-70%. Rabot et al. (2015) found that fluxes increased exponentially starting at 60% WFPS. Rochette et al. (2010) noticed that the reduction of N_2O to N_2 was more rapidly inhibited when it was exposed to O_2. Therefore, after an anaerobic period, such as a heavy rainfall, the denitrifying enzymes would favour production of N_2O over N_2.

2.3.3 Other factors affecting N_2O emissions

Nitrous oxide emissions are characterized as being very erratic with high emissions during short periods of time (Flessa et al., 1998). Soil moisture is a significant factor in N_2O emissions from agricultural soils. However, there are other factors that may influence the flux as well. This includes spring thaw, C:N ratio, temperature, tillage and fertilization.

Regina et al. (2004) and Gerrard (2000) found that in the spring there was a peak of N_2O fluxes after the snow had melted. This could be explained by the production of readily available nitrogen from the freeze/thaw cycles throughout the winter (Regina et al. (2004); Gerrard (2000)). In the spring when the soil thaws, all the stored up N_2O is released and a peak in fluxes is noticed. At the same time, there are no or less plants competing for nitrogen. Therefore, there is more nitrogen available in the soil for the microorganisms (Maljanen et al., 2003). Gerrard (2000) mentions that there is an increase in mass flow that occurs after a change in pressure or temperature. An increase in mass flow coinciding with a surplus of available nitrogen will lead to higher gas emissions.

Klemedtsson et al. (2005) gathered evidence that proved that the soil C:N ratio could be used as a predictor of annual N_2O emissions. Throughout their study, gathered evidence proved that there was no release of nitrous oxide from soils with C:N ratios above 25. Similarly,

Berglund and Berglund (2011) confirmed these results. They found that N_2O fluxes were highest in soils with C:N ratios below 20.

Denitrification is extremely sensitive to rising temperatures (Butterbach-Bahl et al., 2013). Results from Elder and Lal's (2008) study revealed that N_2O flux was positively correlated with soil temperature. Decomposition of SOM which provides substrate for microorganisms generally increases with increasing temperatures. N_2O emissions are found to be more sensitive than CO_2 emissions based on the Q10. The Q10 is defined as the stimulation of denitrification following an increase in temperature by 10 °C. The microbial C and N cycles are very tightly linked. Therefore, it can be concluded that N_2O emissions are affected by the level of CO_2 emissions directly correlated to soil temperature. Soil respiration rises in accordance with rising soil temperature. Soil respiration leads to depletion of soil oxygen concentrations (Maljanen et al., 2003). Thus, the soil will become more anaerobic which is suitable for denitrification. Soil microbes responsible for other processes within the N cycle in the soil which provide the substrates for denitrification have been noted to be temperature sensitive. Therefore, as temperatures rise, there is an increase in microbial activity throughout the soil. N_2O has a more sensitive Q10 than CO_2 since the increase in microbial activity with the increase of soil temperature throughout the N cycle (finalizing with denitrification) will cause a multiplying effect of N_2O emissions.

Morris et al. (2004) found that tillage had an effect on the gas flux. In plowed fields, there are higher concentrations of readily available N. When a field is plowed a certain amount of undecomposed SOM is returned to the soil surface thus providing substrate for the nitrification/denitrification process. Tillage will also loosen up the soil aggregates resulting in a more porous soil allowing more gas diffusion. Similarly, soils with lower bulk densities are considered to be more porous and thus allow for more production and diffusion of GHGs (Tang et al., 2006). Boeckx and Van Cleemput (2001) found that tillage would enhance N_2O emissions. They also studied the effects of planting winter cover crops after tillage and found that this too would increase gas emissions.

Peak N_2O emissions are noticed shortly after fertilizers are applied (Lee et al., 2009). Fertilizers add substantial amounts of nitrogen to the soil. Inorganic and organic fertilizers will affect emissions differently (Ussiri and Lal, 2013). Ussiri and Lal (2013) concluded that organic

fertilizers such as manures would stimulate the denitrification process more than inorganic fertilizers. Organic fertilizers add NO_3^- which provides a direct component for the denitrification process. Most inorganic fertilizers are added to the soil in the form of NH_4^+ which feeds the nitrifying bacteria (Ussiri and Lal, 2013). Within four weeks NH_4^+ will mostly be nitrified. Fertilization was seen as the main driver of peaks in N_2O fluxes by Butterbach-Bahl et al. (2013). Ussiri and Lal (2013) state that a high N application to a soil that has poor drainage could potentially increase the denitrification activity. However, Flessa et al. (1998) found that application of fertilizer to organic soils did not increase N_2O fluxes by much. This suggests that mineralization of organic soil material provides enough mineral-N for producing N_2O.

2.4 Methane (CH_4)

To date, methane emissions from agricultural soils have been mainly focussed on rice field soils. CH_4 is ubiquitous in all soil types, but may be produced or consumed depending on various physical and chemical soil characteristics (Hayashi et al., 2015). It is produced through the process of anaerobic organic matter degradation by methanogenic bacteria and is oxidized aerobically by methanotrophic bacteria (Baird et al. (2009); Nedwell (1996)). Nedwell (1996) affirmed that most CH_4 that is formed anaerobically within the soil is reoxidized before it can be emitted to the atmosphere. Therefore, agricultural soils in general are mainly CH_4 sinks (i.e. soils consume CH_4) (Maljanen et al., 2004). Hayashi et al. (2015) explained that methane emissions were difficult to quantify due to their high spatiotemporal variability.

2.4.1 CH_4 production/consumption in soil

Soils are known as both a source and sink of methane. CH_4 is mainly produced in anaerobic zones of submerged soils (wetlands, peat soils, rice paddies, etc.) by methanogens and is oxidised into CO_2 by methanotrophs in aerobic zones of wetland soils and upland soils (Le Mer and Roger, 2001). Both reactions occur simultaneously within the soil. When the balance between production by methanogens and consumption by methanothrophs is positive, the environment is seen as a methane source. Contrarily, a negative balance indicates that an environment is a methane sink.

2.4.1.1 Methanogenesis

Methanogenesis is the complete mineralization of organic matter in anaerobic environments which produces CH_4 and CO_2 (Le Mer and Roger, 2001). It often occurs in environments where sulphate and nitrate concentrations are low. The process of methane production occurs by the reduction and oxidation of formaldehyde (CH_2O) into two simple hydrocarbons (Hayashi et al., 2015). Reduction results in the hydrocarbon CH_4 and oxidation produces CO_2. This reaction occurs according to the following equation 2.10:

$$2CH_2O \rightarrow CO_2 + CH_4 \quad (2.10)$$

The proportion converted into CH_4 rather than CO_2 is governed primarily by the microbial populations and their ability to gather energy by using electron acceptors such as O_2, nitrate, Fe(III), Mn(IV) and sulfate (Hayashi et al., 2015). The availability of electron acceptors will favor the conversion of organic C into CO_2 and thus reduce the production of CH_4.

Methane is also produced through acetoclastic methanogenesis (Hayashi et al., 2015). This process involves two steps following the simple equations 2.11 & 2.12:

$$2CH_2O + 2H_2O \rightarrow 2CO_2 + 4H_2 \quad (2.11)$$

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O \quad (2.12)$$

This transformation is the result of successive activity from four populations of microorganisms (Le Mer and Roger, 2001). The first is hydrolysis of biological polymers into monomers (glucides, fatty acids, amino acids) by hydrolytic microorganisms in aerobic or anaerobic environments. The second is acidogenesis from monomeric compounds and intermediary compounds formed during fermentation which produce volatile fatty acids, organic acids, alcohols, H_2 and CO_2. The third action is acetogenesis from the metabolites by syntrophic or homoacetogenic microorganisms. The fourth and last step is methanogenesis from the methanogens which use simple compounds (e.g. $H_2 + CO_2$ and acetate) as a substrate.

2.4.1.2 Methanotrophy and CH_4 Oxidation

Methane oxidation occurs in the soil primarily by two metabolic pathways using the enzymes CH_4 monooxygenase, found in methanotrophic bacteria, and NH_3 monooxygenase, found in nitrifying bacteria (Davidson and Schimel, 2009). Methanotrophy is the oxidization of CH_4 primarily done by two aerobic bacterial phyla, the *Proteobacteria* and the *Verrucomicrobia* (Dedysh and Dunfield, 2011). Most are "obligate methanotrophs", however, the presence of facultative methanotrophs has been confirmed in the recent years. "Obligate methanotrophs" make use of only methane or other chemical forms of CH_4, whereas, facultative methanotrophs can feed on either CH_4 or other multicarbon compounds as an energy source (Dedysh and Dunfield, 2011). Methanotrophs develop in the oxidized soil layer, in the aerobic rhizosphere of plants possessing an aerenchyma, and inside the roots and the submerged part of leaf sheaths. These bacteria make use of CH_4 as a carbon and energy source. It is often observed in lower oxic soil layers. According to Le Mer and Roger (2001), there are two forms of CH_4 oxidation. The first form is known as high affinity oxidation and it accounts for approximately 10 % of the total CH_4 consumption. It occurs when methane concentrations are close to the atmospheric methane concentrations (< 12 ppm). The second form is known as low affinity oxidation. This is the main action which is performed by methanotrophic bacteria in aerobic zones. It typically occurs at atmospheric CH_4 concentrations higher than 40 ppm.

2.4.1.3 Plants and fungi

In recent years, there has been great debate whether or not plants and fungi are capable of producing methane under aerobic conditions, independent of microbial activity (Jugold et al., 2012). There is known evidence which supports these speculations. Jugold et al. (2012) observed the production of methane in a soil absent of methanogenic and methanotrophic activity. Results proved that there are in fact unknown processes that can generate methane in soil and peat which are completely unrelated to methanogens. However, the exact processes are still unclear and further research is recommended.

2.4.2 Effect of soil moisture on CH_4 emissions

Methanogens are activated in a soil once it has been in an anaerobic state for a long time (e.g. several days) (Li, 2007). Under these conditions, decomposers, denitrifiers, manganese bacteria, iron bacteria and sulfur bacteria will deplete the major oxidants such as O_2, nitrate, manganese, iron, and sulfate. Correspondingly, when a soil is submerged, the size of aerobic zones diminishes thus reducing the activity of methanotrophs (Le Mer and Roger, 2001). Therefore, soils that have been submerged for an extended period of time are CH_4 sources. For example, well drained cultivated soils may become methane sources after winter when the snow has melted or during a heavy rainfall event in the summer. In poorly drained soils in forested areas, methane consumption by methanotrophs is observed to have a negative correlation with soil volumetric water content typically when it reaches 60 to 100% (Castro et al., 1993). Methanotrophy will depend on soil fertility at low water content typically between 22 to 60%. Studies done on Danish soils revealed that methane gas fluxes peaked when soils dried up after being temporarily submerged (Ambus and Christensen, 1995). This can be explained by the increased oxygenation of the soil and thus increased methane diffusion. Similarly, Jugold et al. (2012) studied the effects of repeated cycles of wetting and drying the soil. They found that after adding water to a sample soil, the latter would emit up to five times more methane than the comparable dry soil sample at same temperature. Jena et al.'s (2013) laboratory incubation experiments indicated that highest peaks of CH_4 fluxes were noticed at 100 % WFPS. Subsequently, CH_4 fluxes would decrease with a decline in soil moisture content. The longer the soil was left in a saturated condition, the more methane would be emitted.

2.4.3 Other factors affecting CH_4 emissions

Although, submersion of soils is the main contributing factor to methane emissions from agricultural soils, there are other factors that may have an influence as well.

Temperature was seen to have a significant impact in the studies performed by Le Mer and Roger (2001). They found that methanogenesis was at an optimum between 30-40 °C. Low soil temperatures seemed to influence CH_4 emissions by reducing the activity of methanogens

and other bacteria that contribute to methanogenic fermentation. Methanotrophy appeared to be less affected by variation in temperature.

The effect of N-fertilization on the soil CH_4 emissions depends on the nature and quantity of fertilizers applied (Le Mer and Roger, 2001). For example, studies have proven that ammonium and urea will usually inhibit atmospheric CH_4 oxidation whereas nitrate will not.

Soil compaction by agricultural equipment could also limit CH_4 oxidation (Le Mer and Roger, 2001). A study done on soils in Germany revealed that ploughing of a field reduced methane oxidation by 6 to 8 times when compared to a non-ploughed field. This is potentially due to the destruction of micro-aerophilic niches and of the organic matter enriched layer that develops on the surface of the soil.

Speculations have been made on the influence of plant communities (Kasimir Klemedtsson et al., 2009). Photosynthesis performed by plants provides the necessary substrates to soil microorganisms essential in methanogenesis. However, there has been no major evidence to prove this theory.

According to Le Mer and Roger (2001), variation of soil pH has an impact on CH_4 production and consumption. They found that in peat soils methane production was optimum between pH 5.5 and 7.0 and methane consumption was optimum between 5.0 and 6.5. Weslien et al. (2009) drew similar conclusions. They found that CH_4 oxidation occurred at pH levels of above 3 and the optimum pH was 5.5 (Weslien et al., 2009). Levy et al. (2012) and Taconi et al. (2007) found that methane fluxes increased with increasing acidity. Levy et al. (2012) further noted that methanotrophs and methanogens would differ in their response to pH and would have different optimum values for activity.

Chapter 3 – Materials and Methods

3.1 Site Description

The research was conducted at two commercial onion fields. One of which was located in Sherrington, Quebec, and the other in Napierville, Quebec. The location in respect to Montreal, Quebec can be seen in Figure 3.1. The distance between both fields is around 15 km and they are located approximately 60 km south of Montreal, Quebec. The soil and climatic conditions of this region are ideal for growing vegetables such as onions, carrots, lettuce, etc. The fields consist of a highly decomposed organic matter soil also known as a muck soil. Soil organic matter content for these fields range between 40 and 80 %. The onion growing season typically ranges from May to September for this region. Based on 1981-2010 measurements from Environment Canada weather stations, the mean annual daily temperature for this region is 6.6 °C and the number of frost free days is approximately 131 (Environment Canada, 2015b). The mean total annual rainfall in the region is 796 mm.

Figure 3.1: Location of Sherrington and Napierville, Quebec (Google Maps, 2015).

3.2 Sampling Strategy

Sampling and field measurements were taken at two of Vert Nature's commercial onion fields each year between 2012 and 2015. Over the four growing seasons, the field sampling location sites were changed due to the necessity of crop rotation. Since we were looking at onion crops and their effects on greenhouse gas emissions the sampling locations were changed in order to maintain the same crop. Locations for all sites from 2012 to 2015 are displayed in Figures 3.2 and 3.3. In keeping with the same methodology, soil properties were similar for each site, respectively, for all sampling seasons. All fields chosen for the experiment consisted of an organic soil. On the first field two sampling sites (1 & 2) were set up as shown in Figures 3.4 and 3.5. On the second field one sampling site (3) was set up similarly to the other two (Figures 3.4 and 3.5). Sampling plots were set up along the onion rows between the furrows. Each row of four sampling plots was separated by two onion rows which was approximately 8 m. A length of 2 m was made between sampling plots of a same row. The soil at the first field differed in organic matter content across the length. From one end to the other, the state of mineralization increased. Consequently, the end that was more mineralized had a lower soil organic matter content. The three sampling sites were chosen based on their state of mineralization and amount of soil organic matter content. Each site was categorized by their degree of soil organic matter stabilization. Soil stabilization is the increase in mineralization in order to sustain organic matter and increase carbon storage capacity (Mikutta et al., 2006). Site 3, located in Napierville, was a relatively virgin organic soil that had been recently reconditioned to be used for agricultural production. Site 1 and 2, located in Sherrington, were older soils used in terms of agriculture. Site 1 had on average a higher organic matter content then site 2 and was defined as "moderately stabilized". Thus, site 2 being more mineralized was defined as "most stabilized". Site 3 was located on a soil that was most recently transformed into agricultural fields and therefore was defined as "least stabilized". All sites were irrigated by a hand moved sprinkler irrigation system according to the crops' requirements. One row of plots was defined as the treatment row (sprinkler irrigation (SI)) and the other the control (no irrigation (NI)). When the onions were irrigated, the treatment row would receive an application of water whereas the control plots would remain dry (Figures 3.4 and 3.5). Irrigation pipes were placed in the furrows next to the irrigated onion plots (Figure 3.6).

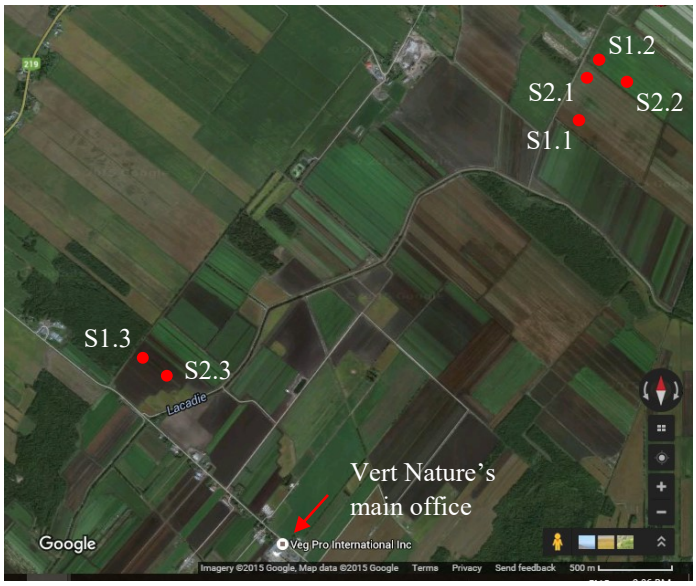


Figure 3.2: Satellite imaging with field site locations of S1.1, S1.2, S1.3, S2.1, S2.2, and S2.3 in Sherrington, Quebec (Google maps, 2015).

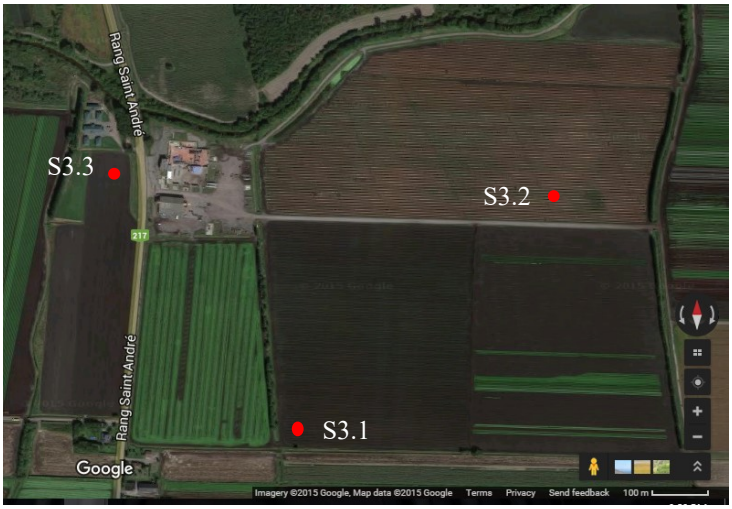


Figure 3.3: Satellite imaging with field site locations of S3.1, S3.2 and S3.3 in Napierville, Quebec (Google maps, 2015).

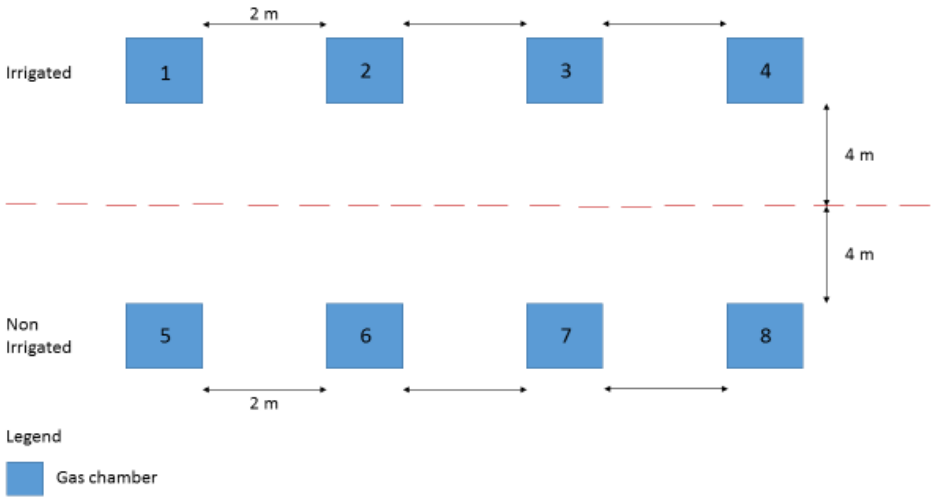


Figure 3.4: Diagram of the GHG sampling layout.

Figure 3.5: GHG sampling layout in field (Lloyd, 2014).

Figure 3.6: Sprinkler irrigation set-up in field (AGGP, 2012).

3.3 Data Collection

3.3.1 Greenhouse Gas Fluxes

CO_2, N_2O and CH_4 were measured at 24 different locations (8 locations at each site) (Figure 3.4). For each site, half of the sampling locations (4) were irrigated and the other half were not. The non-irrigated chambers acted as our control. GHG flux measurements were taken on an approximately weekly basis throughout the crop's growing season (April-October). Seeding, fertilization, harvest, first and last sampling dates as well as average temperature and total rainfall during the sampling period can be found in Table 3.1.

Table 3.1: Seeding, fertilization, harvest and sampling dates, average temperature and total rainfall.

Year	Site	Seeding	Fertilizer	Harvest	First day of sampling	Last day of sampling	Average temp. (°C)	Total rainfall (mm)
2012	S1.1	5 May	5 May	N/A	24 May	29 Aug.	20.5	168
	S2.1	5 May	5 May	N/A	24 May	29 Aug.	20.5	168
	S3.1	16 Apr.	16 Apr.	N/A	24 May	29 Aug.	20.5	168
2013	S1.1	3 May	1 May	1 Oct.	29 Apr.	17 Oct.	16.8	335
	S2.1	3 May	1 May	1 Oct.	29 Apr.	17 Oct.	16.8	335
	S3.2	20 Apr.	26 Apr.	1 Oct.	6 May	17 Oct.	16.8	335
2014	S1.2	20 May	20 May	12 Sep.	29 May	3 Nov.	16.5	292
	S2.2	20 May	20 May	12 Sep.	29 May	3 Nov.	16.5	292
	S3.3	14 May	14 May	31 Aug.	20 May	3 Nov.	16.4	328
2015	S1.3	6 May	6 May	25 Aug.	6 May	6 Oct.	18.0	428
	S2.3	6 May	6 May	25 Aug.	6 May	6 Oct.	18.0	428
	S3.2	14 May	14 May	12 Sep.	6 May	6 Oct.	18.0	428

Sampling was done at all locations once before seeding and once after harvest with a few exceptions. For 2012, the first sample was taken after seeding and the last sample was taken before harvest. The chosen technique of sampling was the static chamber method (Collier et al., 2014). This method consisted of a base placed permanently in the soil throughout the growing

season (Figure 3.7). The permanent chamber frames were made of 3.6 cm thick acrylic plastic and measured 55.6 cm x 55.6 cm x 14 cm. They were inserted 10 cm into the soil allowing 4 cm of the frame to protrude out of the soil surface. A permanent chamber frame was placed in the soil at each sampling location (i.e. 24 in total). At the time of greenhouse gas sampling a cover was placed on top of the base for the duration of one hour (Figure 3.7). The cover similarly was made of 3.6 cm thick acrylic plastic and measured 53 cm x 53 cm x 14 cm. It was vented to avoid pressure buildup and was covered with reflective material to avoid temperature buildup. The bottom of the covers were lined with cushioned tape to ensure an air tight seal within the chamber. A gas collection valve was placed on top of the chamber in order to extract the gas from within the chamber.

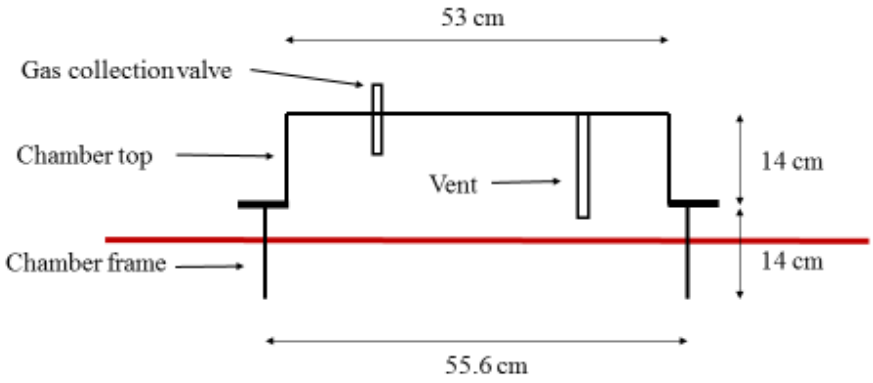


Figure 3.7: Static chamber design.

Over the course of an hour, 5 gas samples (1 sample every 15 minutes) were taken from within each chamber using the gas collection valve (Figure 3.7). The samples were obtained using a 20 mL syringe and were placed in a 12 mL vacuumed exetainer (Labco, Wycombe, UK). The samples were then analyzed for CO_2, N_2O and CH_4 gas concentrations with a gas chromatograph 450-GC System (Bruker corp., Bremen, Germany).

A flame ionization detector (FID) set at 300 °C was utilized for CO_2 and CH_4 measurements. An electron capture detector (ECD) set at 350 °C was applied for N_2O measurements. For the FID procedure, helium was used as the carrier gas with a flow rate of 30 mL min^{-1}. Argon was used for the ECD with a flow rate of 10 mL min^{-1}. The gas chromatograph

(GC) had two 30 m packed columns of 250 µm diameter. The column installed with the ECD was Hayesep D, 80/100, 2 m x 1/8 SS" (Bruker corp., Bremen, Germany). The column fitted with the FID was Hayesep A D, 80/100 Mesh, 3.6 x 1/8 SS", CP99960. Run time consisted of 4.5 minutes at a constant oven temperature of 80 °C. The integrated Bruker software Compass CDS (Version 3.0.0.68) was used to analyze the recorded data (Benslim, 2013).

The hourly fluxes for each location were obtained through several steps. The outliers were first removed from the dataset. The lower limits for N_2O, CO_2 and CH_4 were deemed to be 0.15 ppm, 300 ppm and 1.7 ppm. Once the outliers were removed, the data was converted from ppm to gas concentrations in mg/m^3 using the following constant gas law equation 3.1:

$$C_g = (C_v)*(GMW)*(P/nRT), \quad (3.1)$$

Where:

C_g = gas concentration (mg N_2O-N m^{-3}, mg CO_2-C m^{-3}, mg CH_4-C m^{-3});

C_v = volume concentration (ppm);

GMW = gram molecular weight (g);

P = atmospheric pressure (760 mmHg);

n = number of moles;

R = ideal gas constant (62.36367x10^{-3} m^3 mmHg K^{-1} mol^{-1});

T = temperature at time of GC analysis (293.15 K).

The concentrations of the five samples taken at 15 minute intervals at each sampling location were then used to calculate the gas fluxes (mg N_2O-N m^{-2} h^{-1}, mg CO_2-C m^{-2} h^{-1}, mg CH_4-C m^{-2} h^{-1}). The following equation 3.2 was used:

$$Flux = dC_g/dt * (V/A), \quad (3.2)$$

Where:

Flux = flux between any two gas concentrations over time (mg m^{-2} h^{-1});

dC_g/dt = the slope of the linear regression between any two gas concentrations over time (t) (mg m^{-3} h^{-1});

V = volume of the chamber headspace (m^3);

A = surface area of the chamber (m^2).

With the five gas concentrations collected over the course of an hour, there was a possibility of ten flux calculations for the overall flux of one sampling location. They were obtained by the ten slopes of the linear regression between any two gas concentrations over time. The resultant flux for one sampling location was obtained by taking the median of the ten derived fluxes. If a CO_2 value was found to be negative, these results were deemed as a missing value for all three gases. Any flux that did not have five gas concentrations in its flux calculation was deemed as a missing value as well. Overall, there were 158 missing values for CO_2, 168 for N_2O and 158 for CH_4. For each gas there was a total of 1240 fluxes for all four years.

3.3.2 Meteorological Data

Meteorological data were obtained from the nearest weather station located in L'Acadie, Quebec, an approximate 30 km distance north of the research site in Sherrington (Environment Canada, 2015b). Daily mean air temperature (°C) and total daily precipitation (mm) were logged for the entire sampling season between 29 April and 3 November for all years (2012-2015).

3.3.3 Soil Data

3.3.3.1 Soil Chemical Properties

Chemical soil sampling was completed on 10 and 16 May in 2012, 14 and 27 June and 5 July in 2013, 24 July in 2014, and 17 July in 2015. At each site, four composite samples were taken at three depths (0-20 cm, 20-40 cm, and 40-60 cm). Each of the composite samples consisted of five sub-soil samples taken at each of the depths between the two rows of chambers as seen in Figure 3.8. The samples were stored in Ziploc bags in a cooler and sent to the lab to be analyzed for soil organic matter content, pH, NH_4^+, NO_3^-, Ca, Mg, K, Al, and P properties. In

2015, Mn and Na were also measured. Results for the depth of 0-20 cm are displayed in Tables 3.2, 3.3, and 3.4. Results for all depths are found in Table B1. Missing values were marked as N/A.

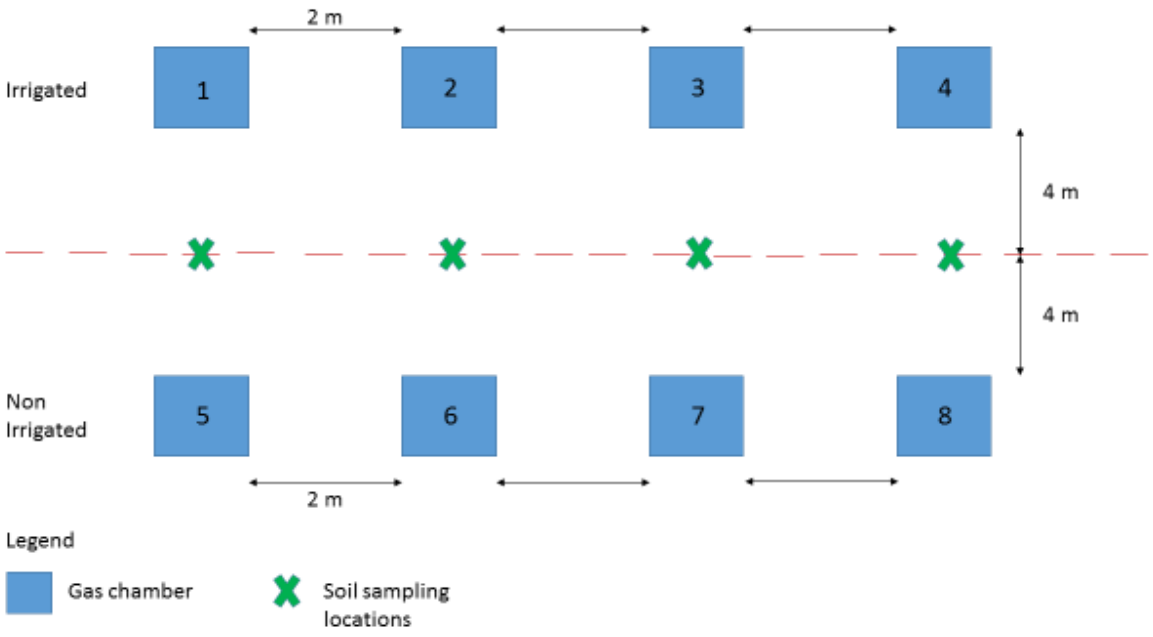


Figure 3.8: Soil sampling locations.

Soil organic matter content was measured using the Loss upon Ignition method as described by Skjemstad and Baldock (2007). Results are shown in Table 3.4. Air dried soil samples of approximately 1.3 g were first placed in crucibles. They were then dried in the oven overnight at 105 °C. The weight of the soil after drying was recorded. Finally, the samples were placed in a muffle furnace at 500 °C for 5 hours and then left to cool overnight. The final weight was recorded. The SOM was calculated using the following equations 3.3 & 3.4:

$$\text{Weight of inorganic matter} = (\text{weight of soil after 500 °C}) - (\text{weight of soil after 105 °C}) \quad (3.3)$$

$$\text{SOM (\%)} = [(\text{weight of inorganic matter}) / (\text{weight of soil after 105 °C})] * 100 \quad (3.4)$$

The method used to obtain the pH of the soil was based on the procedures developed by Hendershot et al. (2007). Results are shown in Table 3.4. Approximately 7 g of air dried soil were weighed and recorded. They were placed in plastic bottles with lids. 20 mL of double

deionized (d. d.) water was added to the soil. The suspension was mixed intermittently for 30 minutes and then left to stand for an hour. To measure the pH, the electrode was immersed in the clear supernatant and once the reading was constant it was recorded.

Nitrate and exchangeable ammonium nitrogen (NO_3^-, NH_4^+) measurements were obtained using the 2.0 KCl extraction method described by Maynard et al. (2007). Results are shown in Table 3.3. First, 5 g of air dried soil was weighed, recorded and placed in an Erlenmeyer flask. Next, 50 mL of 2.0 M KCl solution was added to the flask and placed on a shaker for 30 minutes. The samples were then filtered into bottles and analyzed for NO_3-N and NH_4-N.

Extractable Ca, Mg, K, Mn, Al, P, Mn and Na were obtained following the procedures developed by Ziadi and Sen Tran (2007). Results are shown in Table 3.2. Approximately 2.5 g of air dried soil was weighed, recorded and placed in a plastic cup. Exactly 30 mL of Mehlich-3 extracting solution was added to the cup. The samples were placed on a reciprocating shaker for 5 minutes. They were then filtered into plastic vials and stored at 4 °C until analyzed. Extractable P and K were determined using the Manual Colorimetric method. This method consisted of measuring absorbance at 845 nm. Extractable Ca, Mg, Al and Na were determined using the Flame Emission method. This involved taking 1.5 mL of the filtrate, adding approximately 40 mL of deionized water and mixing. Exactly, 1 mL of CsCl-$LaCl_3$ solution was added. The results from the flame emission were recorded. Extractable Mn was determined using the Atomic Absorption method.

Table 3.2: Soil chemical properties P, Al, Mg, Ca, K, Mn and Na (mg/kg of dry soil) at 0-20 cm depth.

Year	Site	Depth (cm)	P (mg/kg of dry soil)	Al (mg/kg of dry soil)	Mg (mg/kg of dry soil)	Ca (mg/kg of dry soil)	K (mg/kg of dry soil)	Mn (mg/kg of dry soil	Na (mg/kg of dry soil)
2012	S1.1	0-20	123	94	363	3331	628	N/A	N/A
	S2.1	0-20	124	418	415	3400	436	N/A	N/A
	S3.1	0-20	59	55	397	4121	398	N/A	N/A
2013	S1.1	0-20	123	94	363	3331	628	N/A	N/A
	S2.1	0-20	124	418	415	3400	436	N/A	N/A
	S3.2	0-20	N/A	N/A	N/A	N/A	N/A	N/A	N/A
2014	S1.2	0-20	62	331	1014	7423	337	N/A	N/A
	S2.2	0-20	171	541	1029	7140	212	N/A	N/A
	S3.3	0-20	118	259	1063	6634	315	N/A	N/A
2015	S1.3	0-20	80	237	1197	7436	369	9	255
	S2.3	0-20	98	197	1219	7706	298	8	315
	S3.2	0-20	N/A	N/A	N/A	N/A	N/A	N/A	N/A

Table 3.3: Soil chemical properties $N\text{-}NO_3$ (mg/g dry soil) and $N\text{-}NH_4$ (mg/g dry soil) at 0-20 cm depth.

Year	Site	Depth (cm)	mg N-NO3/g of dry soil	mg N-NH4/g of dry soil
2012	S1.1	0-20	0.0202	0.0193
	S2.1	0-20	0.0125	0.0159
	S3.1	0-20	N/A	N/A
2013	S1.1	0-20	0.0202	0.0193
	S2.1	0-20	0.0125	0.0159
	S3.2	0-20	0.0095	0.0104
2014	S1.2	0-20	0.0627	0.0031
	S2.2	0-20	0.0644	0.0026
	S3.3	0-20	0.0573	0.0028
2015	S1.3	0-20	0.0902	0.0019
	S2.3	0-20	0.0805	0.0017
	S3.2	0-20	0.0095	0.0104

Table 3.4: Soil chemical properties soil organic matter (%) and pH at 0-20 cm depth.

Year	Site	Depth (cm)	SOM (%)	pH
2012	S1.1	0-20	83	5.71
	S2.1	0-20	46	6.68
	S3.1	0-20	82	6.32
2013	S1.1	0-20	83	5.71
	S2.1	0-20	46	6.68
	S3.2	0-20	82	N/A
2014	S1.2	0-20	88	5.44
	S2.2	0-20	40	6.93
	S3.3	0-20	61	6.29
2015	S1.3	0-20	79	5.75
	S2.3	0-20	75	5.72
	S3.2	0-20	82	N/A

3.3.3.2 Soil Physical Properties

Physical soil sampling was completed on 10 and 16 May in 2012, 14 and 27 June and 5 July in 2013, 24 July in 2014, and 17 July in 2015. At each site, four composite and four core samples were taken at three depths (0-20 cm, 20-40 cm, and 40-60 cm). The core samples had a height of 7.7 cm and an internal diameter of 8.5 cm. Each of the composite samples consisted of five sub-soil samples taken at each of the depths between the two rows of chambers as seen in Figure 3.8. The samples were stored in a Ziploc bag in a cooler and brought back to the lab. The core samples were analyzed for bulk density and the composite samples were analyzed for particle size distribution and hydraulic conductivity. Results for all physical properties at 0-20 cm depth are displayed in Table 3.5. Results for soil physical properties at all depths (0-20, 20-40 and 40-60 cm) are displayed in Table B2.

Bulk density was measured in the lab (Brady and Weil, 2007). First, the weight of the soil sample was recorded. It was then placed in the oven for 24 hours at 105 °C. The weight of the

dry soil sample was recorded. Soil bulk density and soil porosity were calculated using the following equations 3.5 & 3.6:

$$\text{Soil bulk density (g/cm}^3\text{)} = \text{(oven dry weight of soil) / (volume of soil)} \quad (3.5)$$

$$\text{Soil porosity (\%)} = [1 - (\text{soil bulk density / soil particle density})] * 100 \quad (3.6)$$

Particle size distribution was measured using the hydrometer method (Brady and Weil (2007); Scott (2000)). First, 50 g of soil was weighed. The sample was then treated with 100 mL of sodium hexametaphosphate complex Ca^{2+}, Al^{3+}, Fe^{3+}, and other cations (5% dispersing solution). The sample was left overnight to allow the aggregates to loosen. A blank mixture was created using just 100 mL of 5 % dispersing solution and 880 mL of deionized water. Next, the soil sample mixture was mixed for 5-10 minutes using a milkshake mixer. The suspension was transferred from the dispersing cup to a 1000 mL cylinder and filled with deionized water up to the 1000-mL mark. The sample was left to equilibrate to room temperature. For each sample including the blank the following steps were taken. The plunger was inserted into the suspension and mixed for 10 plunges. As soon as the plunger was removed the hydrometer was placed in the suspension and the second timer was started. The hydrometer reading was recorded at 30 s, 40 s, 60 s and 7 hr. Temperature of the suspension was recorded after each hydrometer reading. The following equations 3.7, 3.8, 3.9 & 3.10 were used to obtain the particle density:

Corrected concentration of suspension (C_{30s}, C_{40s}, C_{60s}, and C_{7hr}) (g/L):

$$C = \text{(hydrometer reading of soil mixture)} - \text{(hydrometer reading of blank)} \quad (3.7)$$

$$\text{Percent clay (\%)} = \text{(corrected concentration at 7h / weight of sample)} * 100 \quad (3.8)$$

$$\text{Percent silt (\%)} = \text{(corrected concentration at 40s / weight of sample) - \% clay} \quad (3.9)$$

$$\text{Percent sand (\%)} = 100 - \text{\% silt - \% clay} \quad (3.10)$$

Note that most soil samples for this study were organic and, therefore, contained minimal amounts of sand, silt or clay (Table 3.5). There were only a few that were not and these were seen in the lower depths of the soil (40-60 cm). Results for these can be found in Figure B2.

Measurements of saturated hydraulic conductivity were obtained using the constant head method (Figure 3.9) (Youngs, 2001). The preparation of the sample was done by following

several steps. The disturbed soil sample was air dried and packed into the core to the according bulk density. Before placing the soil in the core, cheese cloth was placed on one side of the core to avoid soil clogging within the PVC tubes. The volume of the core was recorded. The soil samples were then saturated with deionized water using a water pump joined by a PVC tube to the bottom of the soil core. After the soil had been left to saturate for at least 30 mins, the PVC tubes were joined to the top of the core where the water could constantly flow through the saturated soil sample. Between removing the PVC tubes and rejoining them at the top, clamps were used to avoid loss of water from the saturated soil core. The height of the head of water over the bottom outlet of the set up was recorded. Water was collected at the outlet of the soil core and measured at regular time intervals. Once, a constant reading for the steady volume of change was achieved this value was recorded. The following equations 3.11, 3.12 & 3.13 were used to calculate the saturated hydraulic conductivity of the sample:

Steady flow rate, Q (cm^3/sec):

$$Q = \text{steady volume of change / time interval} \tag{3.11}$$

Steady state flux, q (cm/sec):

$$q = Q / \text{cross sectional area of core} \tag{3.12}$$

Hydraulic conductivity, K_{sat} (cm/sec):

$$K_{sat} = q / \text{total head gradient} \tag{3.13}$$

Figure 3.9: Saturated hydraulic conductivity set up in lab (Lloyd, 2015).

Table 3.5: Soil physical properties bulk density (g/cm3), porosity (%), soil type, hydraulic conductivity (cm/sec) at 0-20 cm depth.

Year	Site	Depth (cm)	Bulk Density (g/cm3)	Porosity (%)	Soil type	Hydraulic conductivity (cm/sec)
2012	S1.1	0-20	0.290	73.7	organic	0.0115
	S2.1	0-20	0.557	49.4	organic	0.0004
	S3.1	0-20	N/A	N/A	N/A	N/A
2013	S1.1	0-20	0.290	73.7	organic	0.0115
	S2.1	0-20	0.557	49.4	organic	0.0004
	S3.2	0-20	0.296	73.1	organic	0.0036
2014	S1.2	0-20	0.212	80.8	organic	0.0130
	S2.2	0-20	0.492	55.3	organic	0.0014
	S3.3	0-20	0.352	68.0	organic	0.0085
2015	S1.3	0-20	0.315	71.3	organic	0.0040
	S2.3	0-20	0.325	70.4	organic	0.0019
	S3.2	0-20	0.296	73.1	organic	0.0036

3.3.4 Soil Moisture and Temperature Measurement

Soil moisture content (%), and soil and air temperature (°C) were measured at each chamber location during the time of each gas sampling. Soil moisture content measurements were recorded at 5.8 cm depth using a hand held Theta Probe ML2x soil moisture sensor. Soil and air temperature were recorded at 6 cm depth using a 10.5 cm Hanna Checktemp temperature sensor.

3.4 Statistical Analysis

In order to compare the fluxes from each of the sites and treatments, significant differences ($p = 0.05$) were obtained using the univariate general linear model (ANOVA). A factorial design was used to test the combination of days, sites and treatments. Significant differences were analyzed using Tukey's HSD comparison test to evaluate the significant differences between each site for CO_2, N_2O and CH_4. A paired t-test was used to further compare the significant differences between the two treatments (irrigation and no irrigation) for CO_2, N_2O and CH_4. Significance was determined at $p = 0.05$ level. These tests were executed using the JMP® software from SAS (Version 11.2.0). To determine the effects of volumetric water content (VWC), soil temperature, WFPS, CO_2, N_2O and CH_4 on the GHG fluxes, a stepwise regression was executed using Microsoft Excel and the R^2 values (%) associated with each property were obtained.

Chapter 4 – Results and Discussion

4. 1 Climatic Data

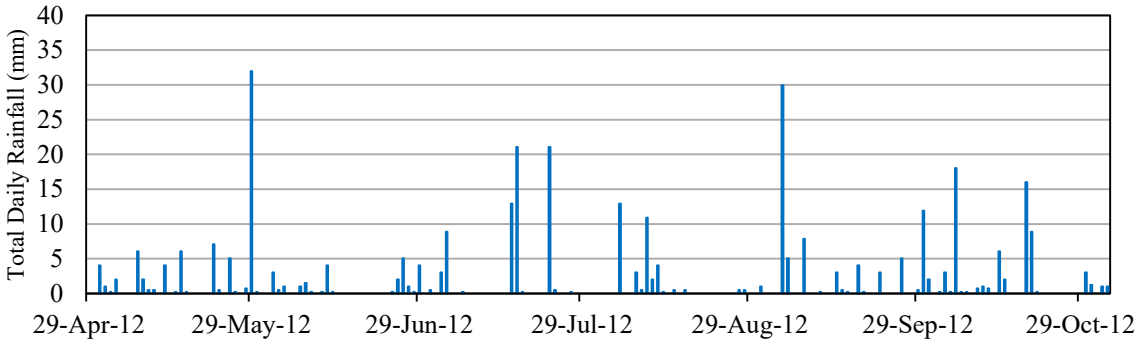
4.1.1 Rainfall

Total daily rainfall for the years 2012 to 2015 are represented in Figures 4.1 a, b, c and d (Environment Canada, 2015b). Throughout the years, rainfall occurrences and intensities differed slightly. Table 4.1 gives the monthly values for the amount of rain and the number of days it rained for all four seasons. The 2012 and 2013 seasons saw the least amount of precipitation. Days of rainfall were more frequent, but, in smaller doses. For the years 2014 and 2015, rainfall days were less frequent but at higher intensities. Overall, 2015 was the wettest sampling season and 2012 the driest. The driest months for each year were, June, 2012, July, 2013, September, 2014, and May, 2015. Subsequently, the wettest months were May, 2012, June, 2013, May, 2014, and August, 2015. The months with the most days of rainfall for each year were May, 2012, June, 2013, October, 2014, and June, 2015. The months with the least amount of days of rainfall were July, 2012, July, 2013, August, 2014, and October, 2015.

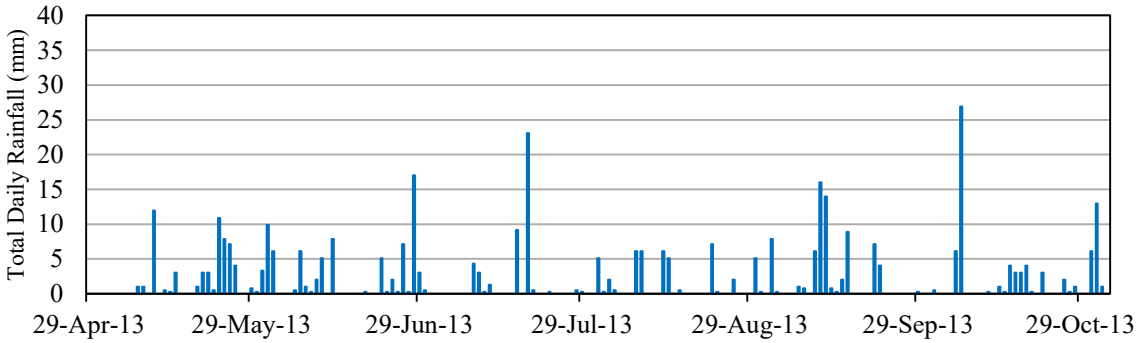
Table 4.1: Monthly and total rainfall days and amounts for the sampling period (Environment Canada, 2015b).

	2012		**2013**		**2014**		**2015**		**Historical**
Month	**Amt. (mm)**	**Rain. days**	**Amt. (mm)**	**Rain. days**	**Amt. (mm)**	**Rain. days**	**Amt. (mm)**	**Rain. days**	**30 year average (mm)**
May	73.1	19	59.7	17	105.2	14	35.0	8	85.7
June	24.6	15	74.7	19	94.5	10	62.5	14	91.2
July	68.8	10	42.7	10	76.4	11	115.3	13	107.6
Aug.	36.8	11	46.5	14	39.1	5	147.1	10	90.6
Sept.	71.9	13	69.3	14	27.2	14	68.1	10	91.0
Oct.	64.2	17	62.0	16	54.7	19	39.6	7	95.6
Total	**339.5**	**85**	**354.9**	**90**	**397.1**	**73**	**467.7**	**62**	**561.7**

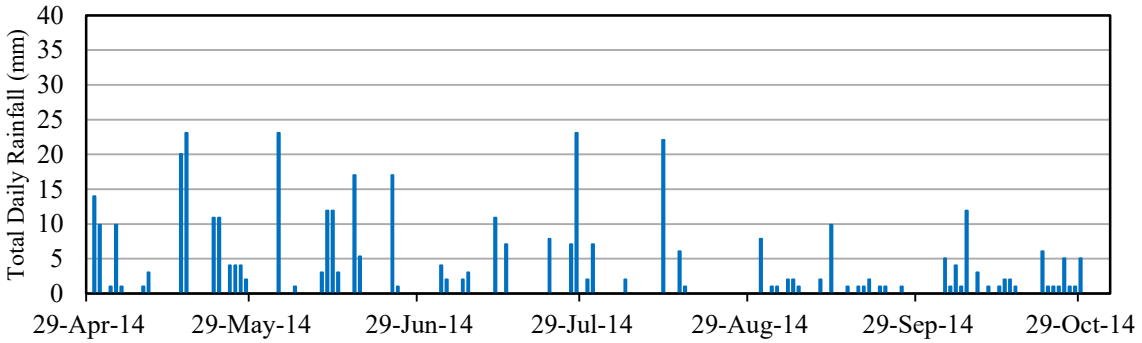
a)



b)



c)



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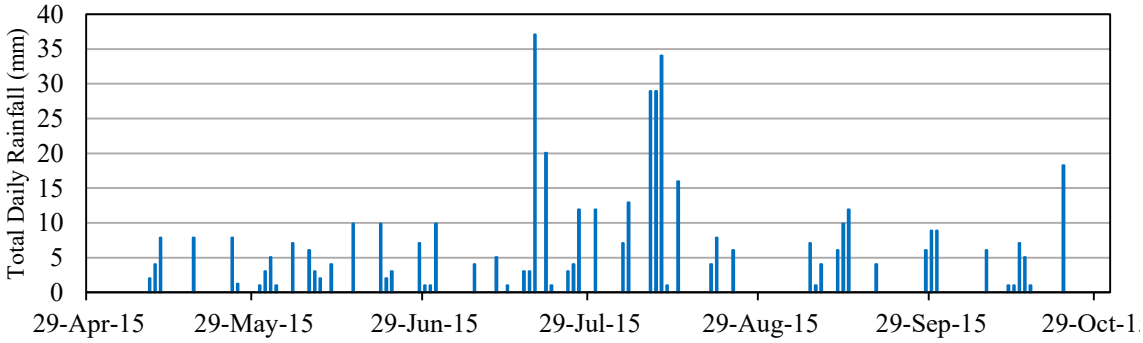


Figure 4.1: Total daily rainfall in a) 2012, b) 2013, c) 2014 and d) 2015.

4.1.2 Air Temperature

The mean daily air temperature was recorded for all sampling seasons between 29 April and 3 November (2012-2015) and is displayed in Figures 4.2 a, b, c, and d (Environment Canada, 2015b). The mean daily air temperature ranged between 3 °C to 29 °C in 2012, -1 °C to 27 °C in 2013, 2 °C to 27 °C in 2014, and -5 °C to 26 °C in 2015. Temperatures fluctuated throughout the season increasing during the months of May and June, reaching a steady point in July and August and decreasing rapidly in September and October. The coldest periods were in the spring, April and May, and in the fall during the months of September, October and November. The warmest periods of each season on average were during the months of July and August. The average mean daily temperature was 17.2 °C in 2012, 16.1 °C in 2013, 16.3 °C in 2014, and 16.4 °C in 2015 (Table 4.2).

Table 4.2: Average mean monthly air temperatures (°C) (Environment Canada, 2015b).

Month	**2012** **Air temp (°C)**	**2013** **Air temp (°C)**	**2014** **Air temp (°C)**	**2015** **Air temp (°C)**	**30 year** **average**
May	15.8	14.8	13.8	16.1	12.9
June	19.5	17.7	19.1	17.1	18.2
July	21.4	20.8	20.3	20.5	20.7
August	20.9	19.1	19.2	20.0	19.5
September	15.3	14.4	14.9	17.8	15.0
October	10.5	9.6	10.5	6.8	8.3
Overall	**17.2**	**16.1**	**16.3**	**16.4**	**15.8**

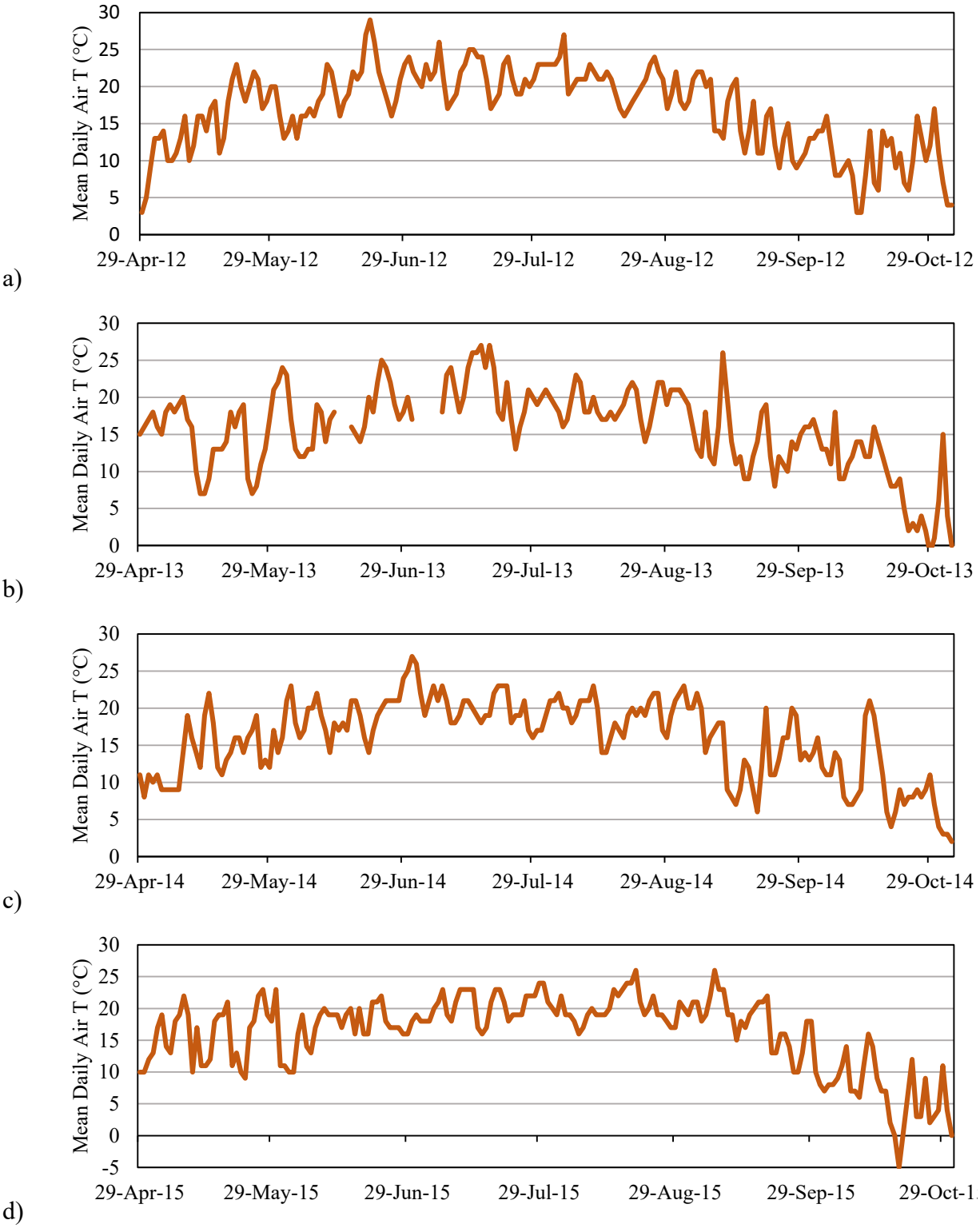


Figure 4.2: Mean daily air temperature in a) 2012, b) 2013, c) 2014, and d) 2015.

4.2 Agricultural Management Practices

4.2.1 Fertilizer Application Rates

Fertilization was applied before planting and incorporated by tillage practices. A breakdown of the N, P and K fertilizer rates and dates of application are displayed in Table 4.3.

Table 4.3: Fertilizer rates and dates of application for S1, S2, and S3.

Year	Site	Date of application	kg-N/ha	kg-P/ha	kg-K/ha
2012	S1.1	5 May, 2012	60	55	265
	S2.1	5 May, 2012	90	55	165
	S3.1	16 Apr., 2012	90	55	165
2013	S1.1	1 May, 2013	89	30	164
	S2.1	1 May, 2013	89	30	164
	S3.2	26 Apr., 2013	91	55	166
2014	S1.2	20 May, 2014	89	54	113
	S2.2	20 May, 2014	89	54	113
	S3.3	14 May, 2014	99	60	126
2015	S1.3	6 May, 2015	91	55	106
	S2.3	6 May, 2015	91	55	106
	S3.2	14 May, 2015	93	57	109

4.2.2 Irrigation Schedule

Typically in Quebec, onion crops do not require much supplemental application of irrigation in addition to rainfall. For this study, irrigation was applied in the sampling seasons of 2013 and 2014 on the days and sites listed in Table 4.4. For the 2012 season, rainfall occurred frequently and therefore irrigation was not needed. Similarly, in 2015, there was less rainfall at the beginning of the season and more in the months of July and August when irrigation is typically necessary.

Table 4.4: Irrigation schedule for the years 2013 and 2014.

Year	Site	Day	Amount (mm)	Total (mm)
2013	S1.1	13 Jun.	8	48
		16 Jun.	8	
		6 Aug.	12	
		15 Aug.	12	
		26 Aug.	8	
	S2.1	13 Jun.	8	48
		26 Jul.	16	
		16 Aug.	4	
		23 Aug.	20	
	S3.2	15 Jun.	8	58
		6 Aug.	25	
		22 Aug.	25	
2014	S1.2	9 Aug.	25	50
		27 Aug.	25	
	S2.2	9 Aug.	25	50
		27 Aug.	25	
	S3.3	9 Aug.	25	50
		27 Aug.	25	

4.3 Soil properties

4.3.1 Volumetric Water Content

The mean VWC on individual sampling dates for each site can be found in Figures 4.3 a, b, c and d. In general, the VWC at the beginning of the season was fairly high. This can be explained by snow melt and increased amounts of rainfall. As the season progressed the VWC diminished. This was seen more clearly during the warmer months of July through to September. Each year showed slightly different trends in VWC throughout the onion growing season. For

the year 2012, the mean VWC for all sites ranged between 24 to 70 % (Table 4.5). Site 2.1 (most stabilized soil) had a slightly higher VWC throughout the season. It is important to note that in the year 2012 sampling ended in August and therefore we did not see the increase in VWC in the fall that we may see in other sampling seasons. For the year 2013, the average VWC for all sites ranged from 23 to 86 % (Table 4.5). Again, site 2.1 had a slightly higher VWC throughout the season. For the year 2014, the average VWC for all sites ranged between 27 to 67 % (Table 4.5). Site 3.3 had slightly higher VWC values then the other sites during the same period of data collection. For the year 2015, soil conditions were slightly drier throughout the season compared to previous years. The average VWC for all sites ranged between 20 to 60 % (Table 4.5).

Table 4.5: Range of mean VWC at S1, S2 and S3 for the duration of the sampling period.

Year	**Site**	**Minimum**		**Maximum**	
		VWC (%)	**Date**	**VWC (%)**	**Date**
2012	S1.1	25	21 Jun.	59	31 May
	S2.1	28	21 Jun.	70	24 May
	S3.1	24	21 Jun.	64	31 May
2013	S1.1	26	7 Aug.	82	24 May
	S2.1	28	7 Aug.	86	30 May
	S3.2	22	29 Jul.	54	3 Jun.
2014	S1.2	29	3 Jul.	53	29 Jul.
	S2.2	27	3 Jul.	54	29 Jul.
	S3.3	33	3 Jul.	67	5 Jun.
2015	S1.3	27	6 Oct.	60	13 Aug.
	S2.3	21	6 Oct.	47	13 Aug.
	S3.2	20	6 Oct.	59	6 May

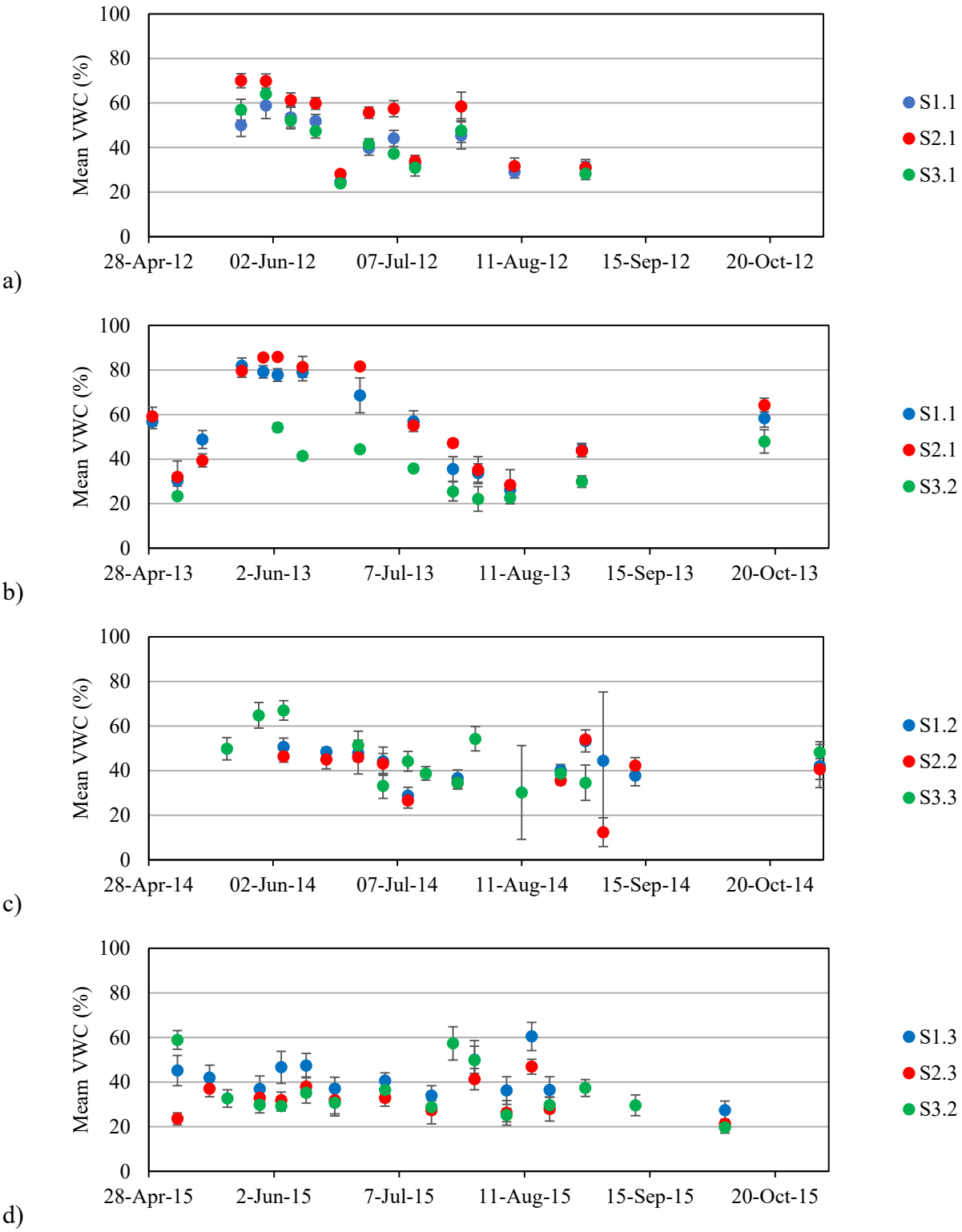


Figure 4.3: Mean volumetric water content (%) and standard deviations at S1 (blue), S2 (red) and S3 (green) in a) 2012, b) 2013, c) 2014, and d) 2015.

Mean VWC in terms of treatment methods can be seen in Figures 4.4 a and b (sprinkler irrigated (SI) and non-irrigated (NI)). Irrigation was applied at the sampling sites in the years 2013 and 2014. The sampling dates after an irrigation event for the year 2013 were 17 June, 29 July, 7 August and 27 August. For the year 2014, sampling after an irrigation event was done on 11 August and 29 August. In 2013, we see that the days preceding 17 June, 29 July, 7 August, and 27 August they irrigated variable amounts depending on the site (Table 4.4). Mean VWC was slightly higher for the irrigated plots on 7 August at S1.1 and S3.2 and 27 August at S3.2. For the other days and sites, irrigation was either of a very small dose or done several days before sampling and thus the effect was not seen. For the year 2014, they irrigated 25 mm for both days (11 and 29 August). Mean VWC results for 29 August show that there were higher values for the irrigated plots compared to the non-irrigated plots. There were no VWC measurements shown on 11 August due to experimental error.

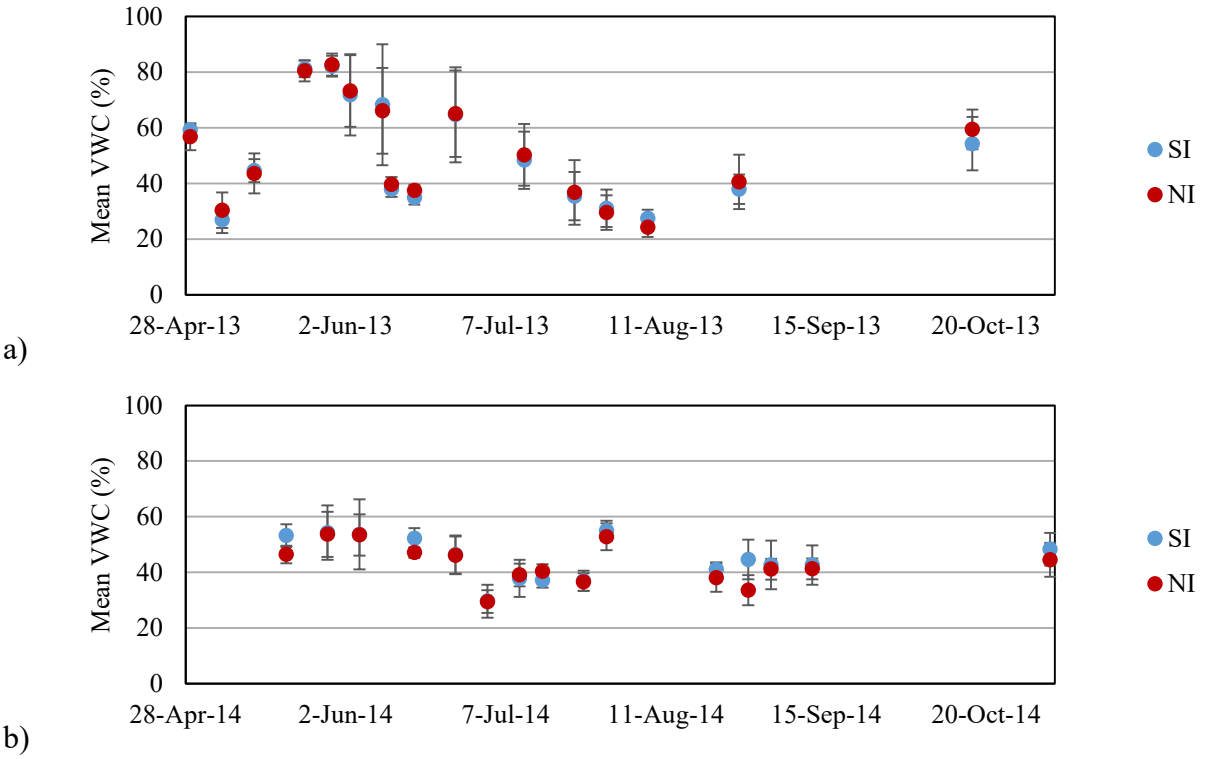


Figure 4.4: Mean volumetric water content (%) and standard deviations for SI (blue) and NI (red) in a) 2013 and b) 2014.

4.3.2 Soil Temperature

The mean soil temperatures are represented by site in Figures 4.5 a, b, c and d and by treatment in Figures 4.6 a and b. The mean soil temperatures for all sampling seasons ranged between 6.9 and 28.6 °C (Table 4.6). Soil temperatures were lower at the beginning of the season and steadily increased throughout May and June. In July and August temperatures typically did not vary much and stayed generally within 20 to 25 °C. By September temperatures began to decrease reaching a low in October and November. Between sites the temperature of the soils did not differ much. Temperatures at S3 were slightly higher for all seasons. However, on average temperatures between sites did not vary by more than 2.5 °C. Between the irrigated plots and non-irrigated plots the temperatures did not differ by much.

Table 4.6: Range of mean soil temperature (°C) at S1, S2 and S3 for the duration of the sampling period.

Year	Site	Minimum		Maximum	
		Soil T (°C)	Date	Soil T (°C)	Date
2012	S1.1	19.3	31 May	23.5	6 Jul.
	S2.1	20.0	24 May	24.0	21 Jun.
	S3.1	21.0	31 May	25.0	12 Jul.
2013	S1.1	6.5	29 Apr.	24.4	22 Jul.
	S2.1	9.5	29 Apr.	23.6	11 Jul.
	S3.2	17.8	17 Jun.	24.2	11 Jul.
2014	S1.2	7.2	3 Nov.	26.6	3 Sep.
	S2.2	8.3	3 Nov.	24.7	3 Jul.
	S3.3	12.8	3 Nov.	28.6	10 Jul.
2015	S1.3	9.9	6 May	23.6	18 Aug.
	S2.3	6.9	15 May	22.4	18 Aug.
	S3.2	9.8	6 May	23.7	28 Jul.

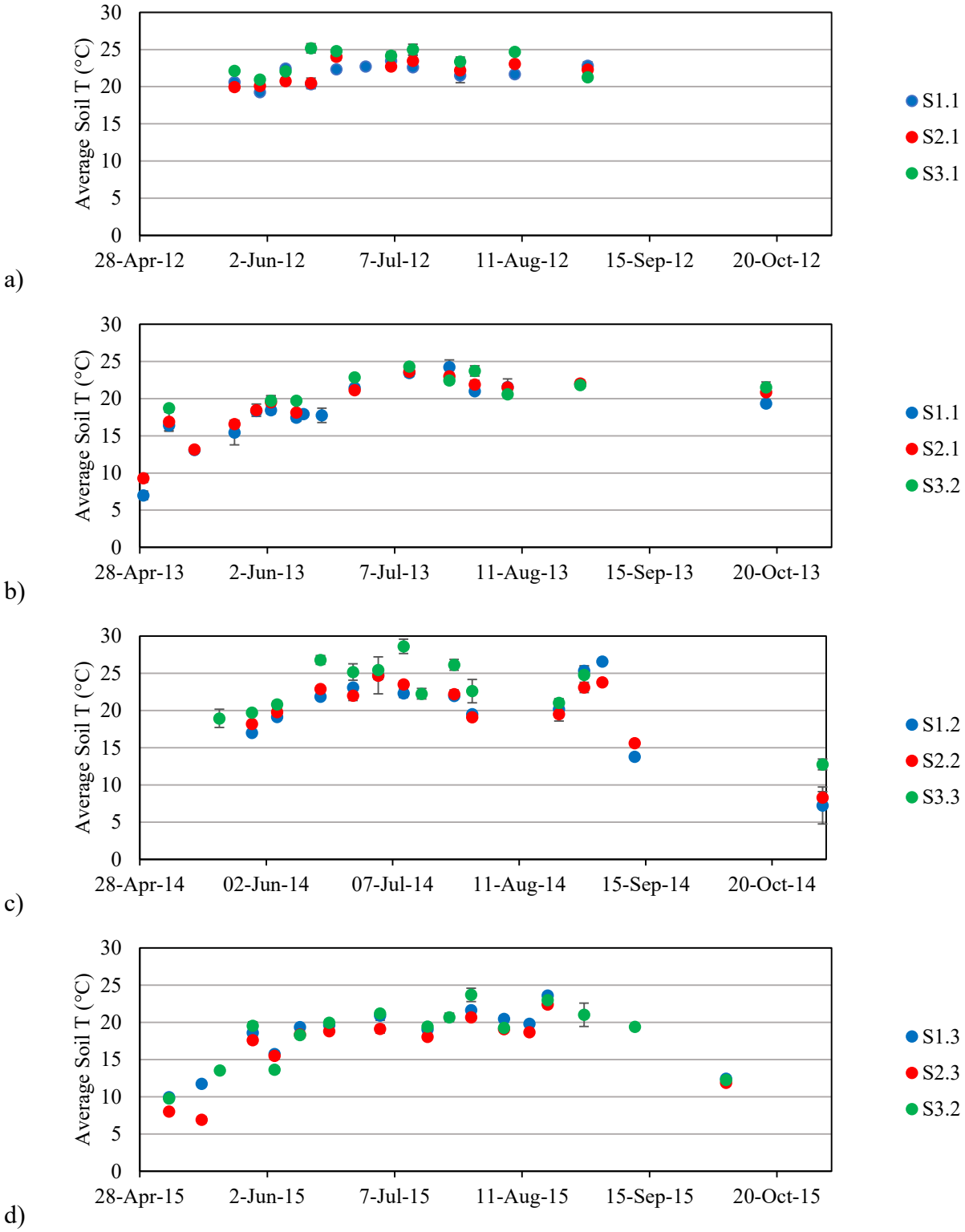


Figure 4.5: Mean soil temperature (°C) and standard deviations at S1 (blue), S2 (red) and S3 (green) in a) 2012, b) 2013, c) 2014, and d) 2015.

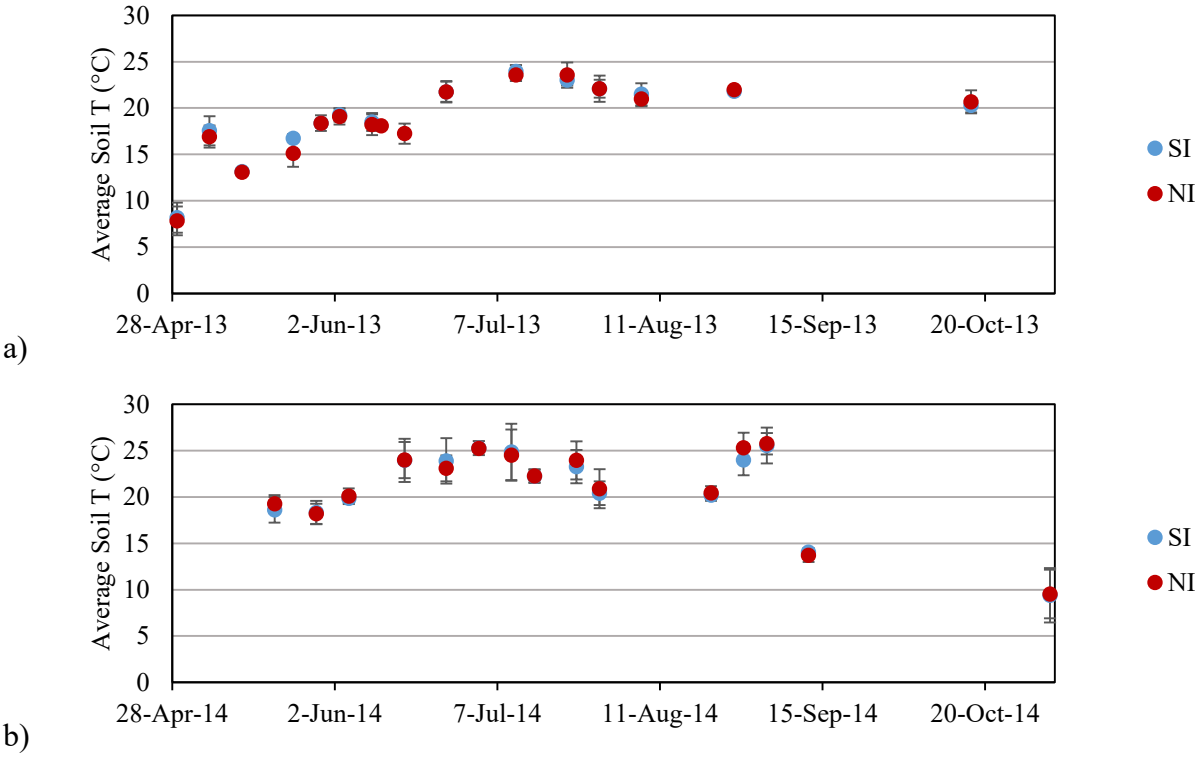


Figure 4.6: Mean soil temperature (°C) and standard deviations at SI (blue) and in NI (red) in a) 2013, b) 2014.

4.4 Greenhouse Gases

4.4.1 Carbon Dioxide Fluxes

Over the four years, the CO_2 fluxes followed similar seasonal patterns (Figures 4.7 a, b, c and d and 4.8 a and b). On average, the mean fluxes ranged between 1 and 268 mg CO_2-C m^{-2} * hr^{-1} with large standard deviations (Table 4.7). This was due primarily to the variability within organic soils as mentioned in Chapter 2. At the beginning of the season (May), the fluxes were low. Pre-seeding samples were taken on 29 April for S1.1 and S2.1 in 2013, 20 May for S3.3 in 2014, and 6 May for all sites in 2015. No pre-seeding samples were taken at all sites in 2012, at S3.2 in 2013 and at S1.2 and S2.2 in 2014. Throughout May and June there was an exponential increase in CO_2 fluxes. A first peak of fluxes was seen near the end of May or mid-June. The fluxes decreased slightly thereafter and a second peak was noticed in July or August depending on the year. For 2012, these burst of fluxes were observed on 31 May and 12 July. In 2013, the

two peaks were noticed on 3 June and 29 July. Furthermore, for 2014, peak fluxes occurred on 26 June and 11 August. Finally, the 2015 seasonal CO_2 flux trend was similar to those of 2012 and 2013. The peaks for 2015 were observed on 29 May and around mid-July/beginning of August (different days for different sites). For all four years, the CO_2 fluxes decreased throughout the month of August and September until harvest. Post-harvest samples were taken on 17 October for the year 2013, 3 November for 2014, and 6 October for 2015. There were no post-harvest samples taken in the year 2012.

Planting was done between end of April and end of May for all four years depending on weather conditions specific to the season (i.e. after the snowmelt and when the soil started to warm up). Typically, seeds germinate approximately 2 weeks after planting (Brewster (2008b); Schwartz and Cramer (2011)). The leaves of the plant begin to sprout and normally reach a maximum growth around July or beginning of August. Bulb growth begins in June and continues to mature throughout July until the end of August or beginning of September. The days where bursts of CO_2 fluxes occurred can be related to the growth stages of the plant. This agrees with the conclusions of Schlesinger and Andrews (2000) and Curiel Yuste et al. (2007) who found that the growing season had a significant effect on CO_2 flux. The first peak was observed at the beginning of the season at the end of May or mid-June. At this point the roots had generally reached their full length. As the roots elongated there was increasing root respiration which emitted increasing amounts of CO_2 (Luo and Zhou (2006); Gerrard (2000)). The dip in CO_2 fluxes thereafter can be related to the decreasing amount of substrate available for the microbes as the labile carbon pool depleted (Xu et al., 2004). However, as the leaves of the plant began to grow, observed fluxes increased. The second burst of CO_2 fluxes was detected in July and beginning of August. As mentioned previously, this is the most critical period for leaf growth and maturity (Brewster (2008b); Schwartz and Cramer (2011)). This suggests that the peak in fluxes was due to the increase in photosynthesis as the leaves reached maturity. An increase in photosynthesis leads to an increase of carbohydrates in the plant. Consequently, this leads to an increase in root respiration and of microbial respiration in the rhizosphere due to the abundance of energy available for the microorganisms (Schlesinger and Andrews (2000); Gerrard (2000)). Throughout the month of August and September, the leaves began to collapse. A decrease in fluxes was observed which reached nearly zero after harvest. As mentioned earlier, 2014 differs from 2012, 2013, and 2015 in that the bursts were observed later. This is due to the fact that in

2014 the onions were planted later in the month of May as seen in Table 3.1. For the years 2012, 2013 and 2014, there was a statistically significant difference ($p = 0.05$) found between certain days of sampling. When observing the days that were statistically significant it can be seen that they relate back to the bursts CO_2 flux previously mentioned. Agreeing with Lee et al. (2009), the results obtained after harvest were fairly low due to the absence of vegetation and thus lack of root respiration. On 17 October, 2013, the mean flux for CO_2 ranged between 4 to 10 mg CO_2-C $m^{-2}*hr^{-1}$. Similarly, on 3 November 2014, the mean gas flux ranged from 1 to 9 mg CO_2-C $m^{-2}*hr^{-1}$. On 6 October, 2015, the fluxes were slightly higher than the other years' post-harvest results. The fluxes ranged between 30 to 34 mg CO_2-C $m^{-2}*hr^{-1}$. One explanation for this is that there may have been more substrate still leftover for the microbes since it was early in October (Elder and Lal, 2008).

The comparison of seasonal soil temperature (Figures 4.5 a, b, c and d) and CO_2 flux (Figures 4.7 a, b, c and d) showed similar trend lines. Soil temperature increased throughout the months of May, June and July. Soil temperatures began to decrease by the end of July and reached a low by October. In general, the results obtained for the years 2012 to 2015 for the muck fields showed that there were moderate positive correlations between CO_2 fluxes and soil temperature with an average R^2 of 40 % over all 4 years (Table A1 and Figure A1 a, b, c and d). This agrees with Fenn et al.'s (2010) findings that soil temperature would have an impact on the emissions of CO_2 from soils. Gerrard (2000) mentioned as well that chemical reactions would be enhanced at higher temperatures. Moreover, temperature can influence the nature and productivity of plant growth and the rate of organic matter decay which as seen previously emits CO_2 gas (International Union of Soil Sciences, 2014).

In this study, VWC was seen to have a weak correlation with CO_2 fluxes over all 4 years with an average R^2 of 14 % (Table A1). CO_2 fluxes in terms of SI and NI plots are displayed in Figures 4.8 a and b. Irrigation effects were found to be statistically significant for the 2013 season. Irrigation effects in 2014 were found to be insignificant since there were only two sampling days after an irrigation out of a total of 16 sampling days. In 2013, there were a total of four out of 16. On certain days, the irrigated plots had lower gas fluxes than the non-irrigated plots. Yet, results for VWC were also lower at the irrigated plots. This outcome was seen on 17 June, 29 July and 27 August. The reason for this may be because the samples were taken more

than two days after the application of irrigation. On 7 August, 2013, the samples were taken the day after and VWC was seen to be higher in the irrigated plots. However, the VWC was already so low on this day that the addition of water may have diminished gas diffusion by blocking the pore spaces and resulting in more anaerobic conditions (Li, 2007). In 2014, there were two days of sampling after an irrigation event, 11 and 29 August. For both days there was a clear effect of irrigation on the gas emission. The irrigated plots, consequently, had higher CO_2 fluxes than the non-irrigated. VWC measurements were taken on 29 August, 2014. Results show that VWC was indeed higher in the irrigated plots. In 2014, GHG sampling was done two days after an irrigation event and therefore we do see the effects on a small scale basis. However, as mentioned earlier, there is no statistical difference between the two irrigation treatments.

CO_2 fluxes increased with higher soil organic matter content. For this section, we looked at the average organic matter content in the top layer of the soil (0-20 cm). In 2012, site 1.1 and site 3.1 had 83 and 82 %, respectively (Table 3.4). S2.1, on the other hand, had a much lower organic matter content with an average value of 46 % (Table 3.4). Overall, site 1.1 and 3.1 had higher CO_2 fluxes when compared to site 2.1 (Figure 4.7 a). Statistical analysis proved that there was a significant difference ($p = 0.05$) between sites 1.1 and 2.1 and between sites 2.1 and 3.1. Similarly in 2013, SOM for each site, 1.1, 2.1 and 3.2, were 83, 46 and 82 %, respectively (Table 3.4). For most sampling days, site 1.1 had, on average, higher CO_2 flux results whereas site 2.1 had the lowest (Figure 4.7 b). Statistical results for 2013 show that there were significant differences ($p = 0.05$) between sites 1.1 and 2.1 and between 2.1 and 3.2. In 2014, site 1.2 had the highest organic matter content with an average value of 88 % (Table 3.4). Sites 2.2 and 3.3 had organic matter values of 40 and 61 %, respectively. In general, throughout the sampling season, site 1.2 had slightly higher CO_2 flux measurements than sites 2.2 and 3.3 (Figure 4.7 c). However, statistically there was no difference seen between sites for the sampling season of 2014. In 2015, the soil organic matter contents for each site (S1.3, S2.3 and S3.2) were fairly close in value (79, 75 and 82 %, respectively) (Table 3.4). Site 1.3 had slightly higher fluxes and site 2.3 had slightly lower fluxes (Figure 4.7 d). Nevertheless, there was no statistical difference found between sites for the year 2015.

There were large variations in results between chambers within a same site. This is due to the large spatial variation that arises more prominently in organic soils (Schwärzel et al., 2002).

These types of soil are not completely uniform and can vary from one sampling chamber to the next (two meters apart). Kechavarzi et al. (2010) explained that muck soils shrink when dry which can change the physical properties. Organic soils are similar to sponges in that they absorb water and can retain large amounts (Yerima and van Ranst, 2005). This made it difficult when sampling. When stepping around the chambers an added pressure was created in the soil and would release more gas. Each year, there was also great variability between similar sites. It was difficult to compare GHG results derived from these due to the field rotations. Different fields have varying soil properties and different microbial communities. In order to get more accurate results, it would have been best to sample every day. However, due to time and budget constraints this was not possible. Another limiting factor was the amount of irrigation applied throughout the four years. Irrigation was only done on 12 days throughout the duration of the project. This did not give enough evidence to come up with any concrete conclusions about the effects of irrigation. Due to the sampling method, it was also impossible to differentiate soil respiration from plant respiration. The chambers were placed directly along the lines of onion plants and therefore captured both soil respiration and plant respiration.

Table 4.7: Range of mean CO_2 fluxes (mg CO_2-C $m^{-2}*hr^{-1}$) for S1, S2 and S3 for the duration of the sampling period.

Year	Site	Minimum		Maximum	
		CO_2 Flux	**Date**	**CO_2 Flux**	**Date**
2012	S1.1	24	24 May	149	12 Jul.
	S2.1	15	6 Jul.	65	12 Jul.
	S3.1	17	29 Aug.	157	12 Jul.
2013	S1.1	7	29 Apr.	192	29 Jul.
	S2.1	3	24 May	164	29 Jul.
	S3.2	10	17 Oct.	138	27 Aug.
2014	S1.2	2	3 Nov.	172	24 Jul.
	S2.2	1	3 Nov.	268	11 Aug.
	S3.3	9	3 Nov.	163	26 Jun.
2015	S1.3	19	15 May	177	6 Aug.
	S2.3	2	15 May	109	28 Jul.
	S3.2	2	6 May	141	11 Sep.

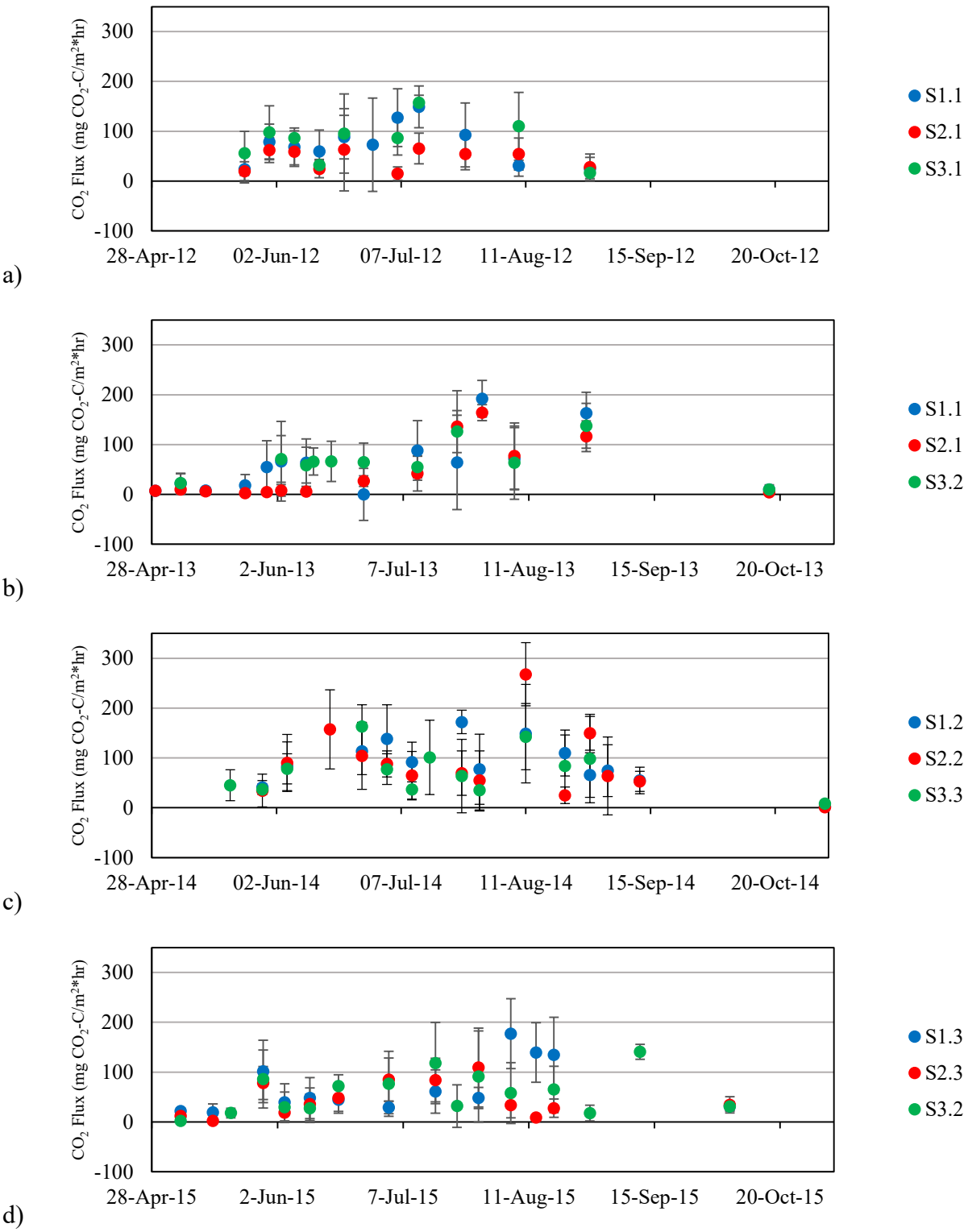
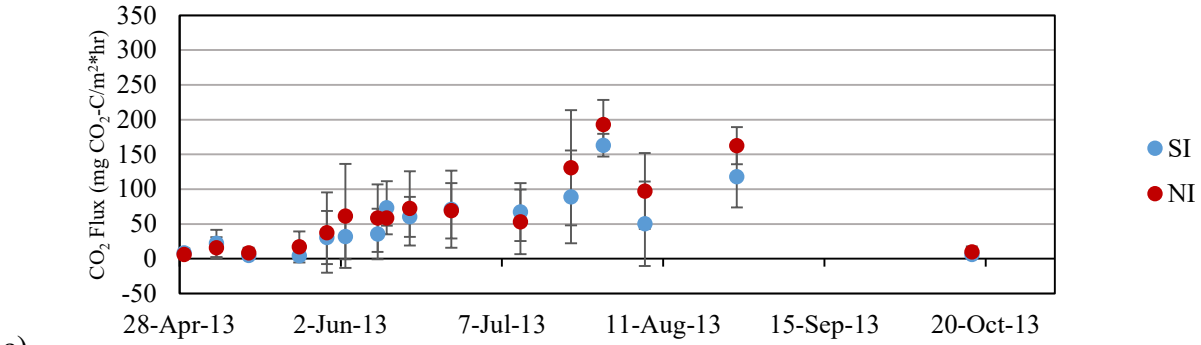
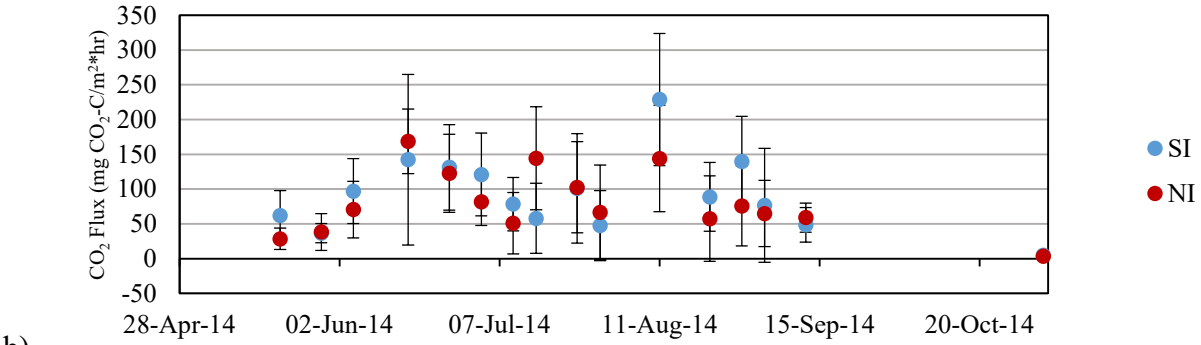


Figure 4.7: Mean CO_2 flux and standard deviations for S1 (blue), S2 (red) and S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



a)



b)

Figure 4.8: Mean CO_2 flux and standard deviations for SI (blue) and NI (red) in a) 2013 and b) 2014.

4.4.2 Nitrous Oxide Fluxes

Over the four years, N_2O fluxes followed similar, yet, slightly differing seasonal patterns. The flux values ranged from -1.06 x 10^{-4} to 0.566 mg N_2O-N m^{-2} * hr^{-1} (Table 4.8). Seasonal N_2O fluxes for sites 1, 2 and 3 are displayed in Figures 4.9 a, b, c and d and by treatments SI and NI in Figures 4.10 a, b, c and d. The rates were greater at the beginning of the season between May and June, and were at a low throughout the months of July, August, and September (Figures 4.9 a, b, c and d and 4.10 a and b). Bursts of N_2O fluxes were observed in July and August for 2012, 2014 and 2015. Post-harvest fluxes were slightly higher than the fluxes observed just prior to harvest.

Results found in this study suggest that the higher fluxes obtained in the first few sampling days of the season could be due to the coinciding effects of spring thaw and less plants competing for nitrogen (Maljanen et al. (2003); Gerrard (2000)). Rochette et al.'s (2010) study

on organic soils in southern Quebec concurrently showed that there were larger N_2O fluxes during and right after the spring thaw. This phenomenon was clearly seen in 2013 at the beginning of the season. N_2O fluxes on the first two sampling days were relatively high (Figure 4.9 b). On the third sampling day there was a sudden decrease. The same can be said for 2014 and 2015. Results from the first day or two of sampling show slightly higher fluxes of N_2O (Figure 4.9 c and d). The year 2013 showed greater fluxes due to the wetter conditions in the spring (Figure 4.9 b). For 2012, no gas samples were taken in the first few days after spring thaw.

The greater fluxes seen at the beginning of the season can also be related to the high VWC and WFPS in the soil during this period. After the snow melt and under cooler conditions, the soils in the months of May and June had a higher VWC (Figure 4.3 a, b, c, and d). Kuntze (1972) explained that muck soils have wide ranges of field capacity, wilting and available moisture. They further explained that even well drained muck soils retain a high water content. Rochette et al. (2010) who studied the emissions from organic soils in the same region found that N_2O fluxes were greater under higher soil water content. Results from our study showed similar findings to Rochette et al.'s (2010). They also coincide well with Maljanen et al.'s (2003) and Rabot et al.'s (2015) conclusions who both found that N_2O fluxes would increase exponentially at WFPS above 60 %. For the sampling seasons in 2012, 2013 and 2014, the highest N_2O fluxes were seen between 70-100 % WFPS (Figures A5 a, b, and c). Positive correlations were found for the relationship between WFPS and N_2O fluxes with average R^2 values of 44, 33 and 21 %, respectively (Table A2). In 2015, the emissions were generally much lower and WFPS rarely raised above 60 % (Figure A5 d and Table A2). For 2015, the correlation was weak due to the lower WFPS and consequently lower N_2O fluxes. The R^2 value for the linear regressions of WFPS and N_2O fluxes for the year 2015 was 8 % (Table A2).

In past research, rainfall was seen to have an effect on the N_2O emitted from the soil. Rainfall adds water to the soil, thus, increasing the VWC. Typically, when a soil goes from dry to wet there is a rapid increase in microbial activity (Davidson and Schimel (2009); Davidson (1992); Rabot et al. (2015)). In 2013 and 2014, greater amounts of rainfall were recorded in May (135 mm) and June (200 mm) (Table 4.1). Accordingly, higher N_2O fluxes were recorded in 2013 and 2014 in the months of May and June than in the same months during the 2012 and

2015 season (Figures 4.9 a, b, c and d). The total rainfall for May and June in 2012 and 2015 was 98 mm (Table 4.1). For all years except 2013, N_2O bursts were noticed later in the growing season in the months of July and August. Typically, in the months of July and August the VWC is lower and the soil is drier (Figure 4.3). When rainfall occurs, the microbial activity spikes due to higher soil moisture (Ussiri and Lal, 2013). Rabot et al. (2015) found that there would be a first peak two days after rewetting and a second peak when the soil began to dry. In 2012, a burst in flux was noted on 25 July two days after a recorded rainfall of 20 mm (Figures 4.9 a and 4.1 a). For this day, the recorded VWCs for each site sampled were 45.5 % for S1.1 and 58.4 % for S2.1 (Figure 4.3 a). In 2014, a slight peak was recorded on 29 July the day after a 23 mm rainfall (Figure 4.9 c and 4.1 c). The VWC for this day was around 55 % for all sites (Figure 4.3 c). Another peak was noticed on 22 August, 2014, that could not be associated with rainfall since there was a long dry period right before sampling. This can be explained by the findings of Maljanen et al. (2003) which state that when there are higher CO_2 emissions there will be less O_2 available. This leads to anaerobic zones which favour N_2O production. On 22 August, 2014, CO_2 fluxes were highest at S1.2 (110 mg CO_2-C/m^2*hr) and accordingly N_2O fluxes were highest at the same site (0.0499 mg N_2O-N/m^2 *hr) (Figures 4.7 c and 4.9 c). In 2015, high fluxes were recorded near the end of July and throughout the month of August on several sampling days (22 July, 28 July, 6 August, 13 August and 18 August) (Figure 4.9 d). For all sampling days, the VWC fluctuated between 25 and 60 % (Table 4.5). However, heavy rainfalls (> 10 mm), occurring right before those sampling days, suggest that rainfall had more of an impact on N_2O emissions than the level of VWC (Figure 4.1 d). Further, it is important to note that the increased amount of N_2O fluxes later in the 2015 season could be due to the very dry spring and then wet summer which causes a sudden increase in microbial activity. Similar findings were noted by Davidson (1992) in their study. Rochette et al.'s (2010) study similarly showed that there would be peaks in N_2O fluxes after a rainfall. Results from their study followed similar patterns and rarely exceeded 0.2 mg N_2O-N m^{-2} *hr^{-1}. However, their results did at times reach up to 5 mg N_2O-N m^{-2}*hr^{-1} which is much higher than the maximum flux (0.6 mg N_2O-N m^{-2} *hr^{-1}) observed in this study (Table 4.8). This can potentially be explained by the heavier rainfalls that occurred during their sampling periods. Rochette et al. (2010) explained that, following an anaerobic period, N_2O reductase can be more rapidly inhibited by exposure to O_2. Therefore, during seasons with heavy rainfalls, the denitrifying enzymes would favour production of N_2O

over N_2. Another explanation for the higher fluxes could be the different crop type. The type of crop cultivated can have slight influences on the amount of N_2O emitted due to crop specific N uptake (Thorup-Kristensen, 2001).

This study did not show significant differences ($p = 0.05$) between irrigated and non-irrigated treatments on N_2O fluxes. Theoretically, the impact of irrigation should increase N_2O emissions if we associate sprinkler irrigation with rainfall (Davidson and Schimel (2009); Davidson (1992); Rabot et al. (2015)). However, if we look at the individual sampling days, where irrigation had been applied relatively soon afterwards, we see slight differences. In 2013, irrigation was applied prior to the sampling days of 17 June, 29 July, 7 August and 27 August (Table 4.4). For 7 August 2013, it was impossible to see a difference due to missing results. For 17 June and 27 August, 2013, the N_2O fluxes for the irrigated plots were not different than the results from the non-irrigated plots (Figure 4.10 a). The irrigation done prior to these sampling days was either in very little quantity (8 mm) or was done four to five days before sampling (Table 4.4). Contrarily, on 29 July 2013, sampling was done three days after an irrigation event of 16 mm. Results for this day reveal that the irrigated plots emitted more N_2O than the non-irrigated (Table 4.4 and Figure 4.10 a). Similarly in 2014, sampling was done two days after an irrigation application of 25 mm on 11 and 29 August (Table 4.4). The N_2O fluxes were noticed to be higher in the irrigated plots for these sampling days (Figure 4.10 b). Based on these results, potential peaks in N_2O fluxes will occur within 72 hours after an irrigation of 16 mm or more.

Fertilization was applied once each season around the time of seeding. The dates and quantities of N, P and K in kg/ha of fertilization for each field for each year are displayed in Table 4.3. The effect of N application on N_2O fluxes was noticed approximately four weeks after the event (Butterbach-Bahl et al., 2013). Ussiri and Lal (2013) explained that the total nitrified NH_4^+ would occur in this estimated time frame. Rochette et al. (2010) found in their study that there was a peak in N_2O flux in May after fertilizer application. This study showed similar results. An increase in fluxes was noticed approximately a month after N application on the sampling days of 24 May, 31 May and 7 June in 2012, 30 May and 3 June in 2013, 16 June in 2014, and 20 May and 29 May in 2015.

Statistical results show that there were significant differences ($p = 0.05$) between sampling sites in 2013 and 2014. In 2013, these differences were seen between S1.1 and S2.1

and between S1.1 and S3.2. In 2014, the differences were only statistically significant between S2.2 and S3.3. The differences between sites can be related to the differing soil properties at each site. In 2012, mainly S3.1 produced more N_2O due to its smaller BD (Figure 4.9 a and Table 3.5). Theoretically, a lower BD will allow for more gas diffusion in the soil (Tang et al., 2006). Additionally, there was more N applied to S3.1 which (Table 4.3). Ussiri and Lal (2013) explained that the application of N to poorly drained soils could have a positive impact on the denitrification activity. In 2013, S1.1 produced more N_2O which had a smaller BD than S2.1 (Figure 4.9 b and Table 3.5). However, the BD of S3.2 was fairly close in range to the BD of S1.1, however, VWC throughout the season was generally higher at S1.1 (Table 3.5 and Figure 4.3 b). Furthermore, when VWC is above 80 % the denitrification process will produce more N_2 than N_2O (Kasimir Klemedtsson et al. (2009); Davidson (1992); Butterbach-Bahl et al. (2013)). This can be seen on three days in 2013 at S2.1 (3 June, 10 June, and 26 June). On the same note, Butterbach-Bahl et al. (2013) found that the production of N_2O would be optimal at 70-80 % VWC. For all three days (3, 10 and 26 June 2013) VWC was within this range at S1.1. VWC at S2.1 was higher than 80 %, as we mentioned previously, and VWC at S3.1 lower than 70 %. Therefore, both these sites produced less N_2O than S1.1 on the same day. For the year 2014, the site that produced the most N_2O was S1.2 (Figure 4.9 c). Agreeing with the notions of Tang et al. (2006) and Flessa et al. (1998), this field had a smaller BD and higher soil organic matter content than S2.2 and S3.3 (Tables 3.5 and 3.4, respectively). Higher SOM leads to an increase in decomposing organism populations which results in increased decomposition rates. The decomposition of SOM provides substrate for the denitrifying process through the mineralization of organic N (Rochette et al., 2010). In 2015, the soil properties for each site were very close in value (Tables 3.2, 3.3, 3.4 and 3.5). Therefore, overall site specific differences in N_2O fluxes was seldom observed.

It has been proven that tillage will have an effect on the release of gases. Tillage returns SOM to the soil surface thus providing more substrate for the microorganisms (Morris et al. (2004); Elder and Lal (2008)). Additionally, it loosens up the soil aggregates providing more pores for gas to diffuse. Results from this study show concurring results. Samples were taken post-harvest after the soil had been tilled on 17 October 2013, 3 November 2014, and 6 October 2015 (Table 3.1). Samples were also taken pre-plant in the spring after the soil had been tilled in preparation for seeding. However, the effects of tillage on these days cannot be differentiated

from the joint effects of spring thaw and higher VWC. Post-harvest results show that, in 2013, the N_2O fluxes were slightly higher than those taken right before harvest (Figure 4.9 b). Results from S3.3 in 2014 show slightly higher results post-harvest than those obtained in 2013 (Figure 4.9 c). On this day, a winter cover crop (barley) had been planted. This suggests that tillage and cover crops can have potential effects on the emission of N_2O under cooler soil temperatures. Similar results were found in 2015 at S1.3 and S2.3 where the soil had been turned and a cover crop had been planted (Figure 4.9 d). This agrees with the findings of Boeckx and Van Cleemput (2001) who found that ploughing of the field and planting barley as a cover crop would enhance N_2O emissions. Rochette et al. (2010) also found an increase in flux at the end of the season in September/October. Contrary to Boeckx and Van Cleemput's (2001) and the findings from this study, they explained that the effect of drier soil and the absence of a crop increased N_2O fluxes.

The limitations encountered when sampling and analysing the results for N_2O fluxes were the same as those mentioned previously for CO_2. Parkin (1987) also mentioned that in organic soils there is high spatial variability with anaerobic zones in soil aggregates. This leads to microsites or "hot spots" for microbial populations as mentioned by (Butterbach-Bahl et al., 2013). Therefore, they concluded that there would be patchy distributions of denitrification in organic soils. These microsites vary both temporally and spatially.

Table 4.8: Range of mean N_2O fluxes (mg N_2O-N m^{-2} *hr^{-1}) for S1, S2 and S3 for the duration of the sampling period.

Year	Site	Minimum		Maximum	
		N_2O flux	**Date**	**N_2O flux**	**Date**
2012	S1.1	7.41×10^{-4}	29 Aug.	0.0111	31 May
	S2.1	4.50×10^{-4}	29 Aug.	0.0148	25 Jul.
	S3.1	-1.06×10^{-4}	29 Aug.	0.0403	7 Jun.
2013	S1.1	0.00464	27 Aug.	0.566	10 Jun.
	S2.1	0.00168	6 May	0.0571	26 Jun.
	S3.2	-0.0141	7 Aug.	0.0721	6 May
2014	S1.2	9.39×10^{-4}	3 Nov.	0.0535	5 Jun.
	S2.2	9.91×10^{-4}	3 Nov.	0.196	17 Jun.
	S3.3	0.00362	10 Jul.	0.160	3 Nov.
2015	S1.3	0.00107	16 Jul.	0.284	18 Aug.
	S2.3	0.00107	13 Aug.	0.0739	29 May
	S3.2	0.00178	28 Aug.	0.0519	6 Aug.

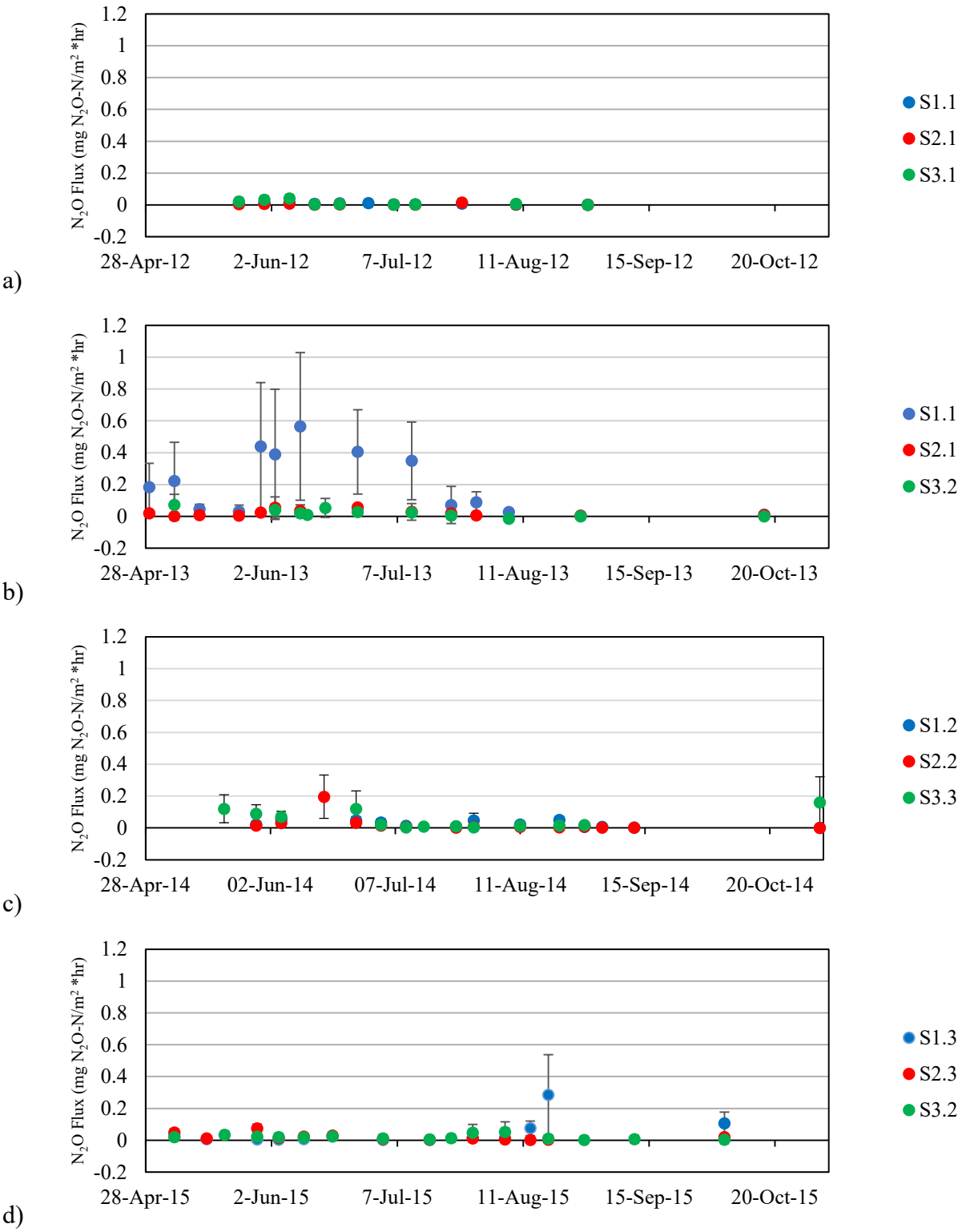
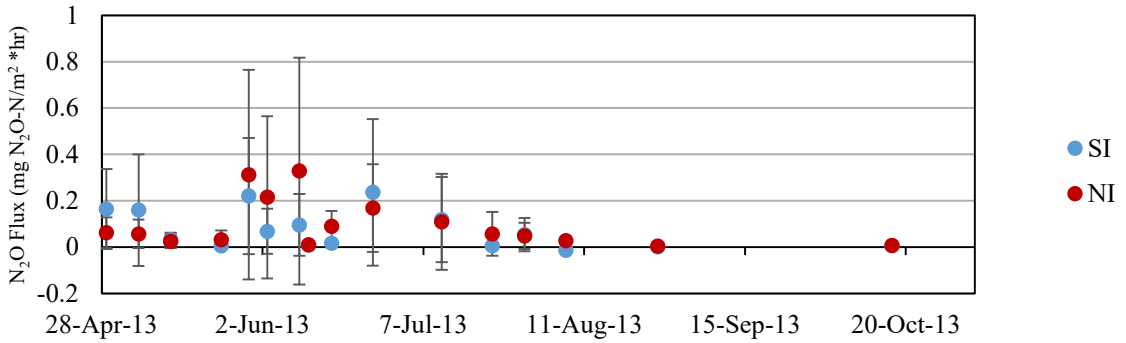
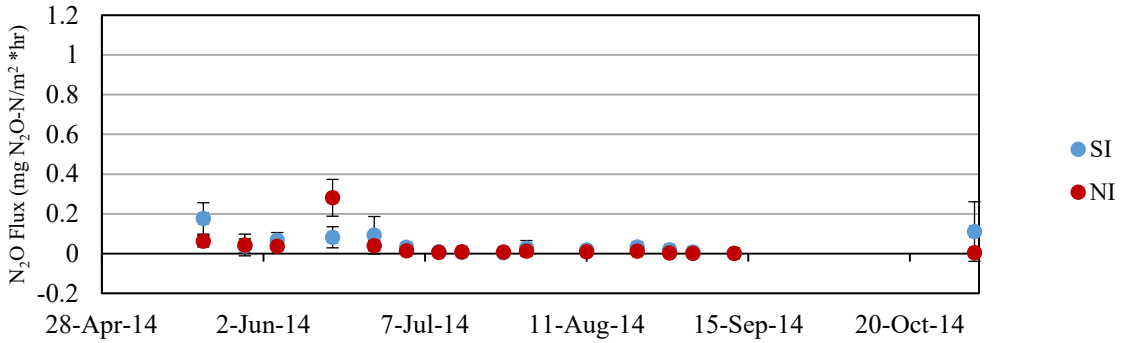


Figure 4.9: Mean N_2O fluxes and standard deviations for S1 (blue), S2 (red) and S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



a)



b)

Figure 4.10: Mean N_2O flux and standard deviations for SI (blue) and NI (red) in a) 2013 and b) 2014.

4.4.3 Methane Fluxes

The organic soils in this study primarily consumed methane. The soil acted as a sink by oxidizing CH_4. The mean gas flux collected for each site during each growing season for the years 2012 to 2015 ranged between -0.00628 and 0.00760 mg CH_4-C $m^{-2} * hr^{-1}$ (Table 4.9). On average, the results show that the years with the most CH_4 oxidation were 2014 and 2015, and the years with the most CH_4 production were 2012 and 2013 (Figures 4.11 a, b, c and d). In 2012, results showed that S2.1 emitted the most methane, and S3.1 consumed the most (Figure 4.11 a). In 2013, S2.1 emitted the most and S3.2 consumed the most (Figure 4.11 b). In 2014, S2.2 emitted the most and S1.2 consumed the most (Figure 4.11 c). Finally, in 2015, S1.3 produced the most and S3.3 consumed the most (Figure 4.11 d).

Methane production and consumption throughout each sampling season was very sporadic. Results from this study agree with Hayashi et al.'s (2015) statement that CH_4 emissions

are highly variable spatially and temporally. Correlations between the gas fluxes and soil temperature, VWC, WFPS and N_2O were all negligible (Table A3). Correlations between CH_4 and CO_2 were found to be weak (Table A3). When looking at the days with the lowest CH_4 fluxes (i.e. most consumption), we see that there were, accordingly, high CO_2 fluxes (Figure A10 a, b, c and d). However, the higher CO_2 flux could also be explained by the plant growth and not at all correlated with the CH_4 flux.

Tillage as explained by Le Mer and Roger (2001) disrupts the oxidation process. Ploughing destroys micro-niches and consequently reduces the oxidation rate. Therefore, theoretically, the CH_4 fluxes should be higher after tillage. This was seen at most of the sites in the years 2013, 2014 and 2015. Tillage was done pre-seed and post-harvest for all years. In 2013, pre-harvest and post-harvest samples for all sites were taken on 27 August and 17 October, respectively. Results for the post-harvest sampling were higher at S1.1 and S3.2 but lower at S2.1 (Figure 4.11 b). In 2014, pre-harvest and post-harvest samples for S1.2 and S2.2 were taken on 12 September and 3 November, respectively, and for S3.3 on 29 August and 3 November, respectively. Results show that pre-harvest gas fluxes were lower at S2.2 and S3.3 (Figure 4.11 c) which coincides with Le Mer and Roger's (2001) theory. Results from S1.2, however, were higher (Figure 4.11 c). In 2015, pre-harvest and post-harvest samples for S1.3 and S2.3 were taken on 18 August and 6 October, respectively. Pre-harvest and post-harvest samples for S3.2 were taken on 11 September and 6 October, respectively. Both S1.3 and S3.2 showed results that were higher post-harvest, whereas, S2.3 did not (Figure 4.11 d).

Theoretically, soil properties will have an effect on CH_4 fluxes. Le Mer and Roger (2001) found that the addition of organic matter would favour CH_4 emissions. However, results from this study show that the fields with the highest CH_4 fluxes were the soils with the least soil organic matter content (S2.1, S2.2 and S2.3) (Figures 4.11 a, b, c and d). Adhya et al. (1997) and Zhang et al. (2011) found that the application of phosphorus (P) would mitigate CH_4 production. Contrary to their findings, the sites which produced the most methane in this study had higher P content than the sites which produced the least (Table 3.2). According to Yao et al. (1999), methanogenesis is limited by available inorganic electron acceptors. Results from our study show that the sites producing the most CH_4 had, generally, higher concentrations of inorganic compounds (Al, P, etc.) (Table 3.2). The pH of the soils at each site was also examined. In

general, the sites producing more CH_4 had higher pH values (Table 3.4 and Figures 4.11 a, b, c and d). However, values were fairly close in range. These findings contradict those of Levy et al. (2012) and Taconi et al. (2007) which suggest that soils with lower pHs would produce more methane. However, Levy et al. (2012) did state that the correlation was weak. It is known that a decrease in pH will increase the potential activity of methanogenic bacterial communities (Taconi et al., 2007). Yet, it is still unclear how soil pH will affect methanotrophic bacterial communities.

Le Mer and Roger (2001) found that ammonium N-fertilizer application would reduce CH_4 oxidation. Results from this study coincide with their findings. Fertilizers were either applied at the same time as seeding or right before for most sites (Table 3.1). In 2013, pre-seeding samples were taken on 29 April. A next sampling was done post-seeding on 6 May 2013. Results from these two days show that methane oxidation was higher on 29 April 2013 from S1.1 and S2.1 (Figure 4.11 b). In 2015, pre-seeding samples were retrieved on 6 May. Post-seeding samples were taken on 15 May 2015 for sites S1.3 and S2.3 and on 20 May for S3.2 (Figure 4.11 d). Correspondingly, CH_4 oxidation was higher before the application of fertilizers at all sites (S1.3, S2.3 and S3.2).

Theoretically, WFPS should have a positive correlation with CH_4 emissions (Jena et al., 2013). However, based on the observations of Li (2007), the soil would have to be submerged for a long period of time (i.e. several days) in order to have any significant increase on observable CH_4 fluxes. Prolonged flooding events are necessary in order to completely deplete the O_2 concentration in the soil pores. Onions require O_2 for root processes and, thus, it is important to have quick drainage and O_2 diffusion into the soil pores. Therefore, it would be hypothesized that irrigation would cause CH_4 emissions to increase if WFPS nears 100 % and O_2 concentration is very low in the saturated soil zone. Statistically there was no significant difference between the irrigated and non-irrigated plots. This may be due to the lack of samples acquired after an application of water via sprinkler irrigation. However, it can mostly be explained by the fact that the amount of water applied during sprinkler irrigation did not submerge the soil for an extended period of time. When irrigated, water percolates through the soil fairly quickly or gets absorbed by organic matter allowing for O_2 diffusion. Overall, VWC and WFPS showed negligible correlations with CH_4 fluxes with average R^2 values of 16 and

19%, respectively (Table A3). Out of 13 sites that were irrigated, seven of them had higher methane fluxes from the irrigated plots (Figures 4.12 a and b). However, out of these seven sites only one of them had a higher mean VWC for the irrigated plots. More CH_4 fluxes from irrigated and non-irrigated plots would need to be sampled and analyzed in order to justify any conclusions made for these organic soils in terms of the effect of irrigation on the production of methane.

Limitations in the sampling and analysis for methane fluxes are similar to those mentioned previously for CO_2 and N_2O. Furthermore, Keppler et al. (2009) mentioned that there are certain methanotrophs that can be found inside of plants. Vegetation therefore emits methane. The static chamber, as previously mentioned, cannot differentiate GHGs produced by the soil from GHGs produced by the plant. Throughout the duration of the study, the methane emissions were very low and close to zero due to the counteracting effect between the methanogens and methanotrophs.

Table 4.9: Range of mean CH_4 fluxes (mg CH_4-C $m^{-2} * hr^{-1}$) for S1, S2 and S3 for the duration of the sampling period.

Year	Site	Minimum		Maximum	
		CH_4 flux	**Date**	**CH_4 flux**	**Date**
2012	S1.1	-0.00383	12 Jul.	0.00175	9 Aug.
	S2.1	-0.00498	12 Jul.	0.00760	6 Jul.
	S3.1	-0.00628	6 Jul.	0.00509	14 Jun.
2013	S1.1	-0.00423	3 Jun.	0.00233	7 Aug.
	S2.1	-0.00152	26 Jun.	0.00218	30 May
	S3.2	-0.00353	27 Aug.	0.00378	12 Jun.
2014	S1.2	-0.00306	3 Nov.	0.000405	11 Aug.
	S2.2	-0.00448	11 Aug.	0.00118	3 Jul.
	S3.3	-0.00555	22 Aug.	0.000442	29 Jul.
2015	S1.3	-0.00438	6 Aug.	0.00459	15 May
	S2.3	-0.00217	16 Jul.	0.000697	13 Aug.
	S3.2	-0.00330	11 Sep.	0.000401	11 Jun.

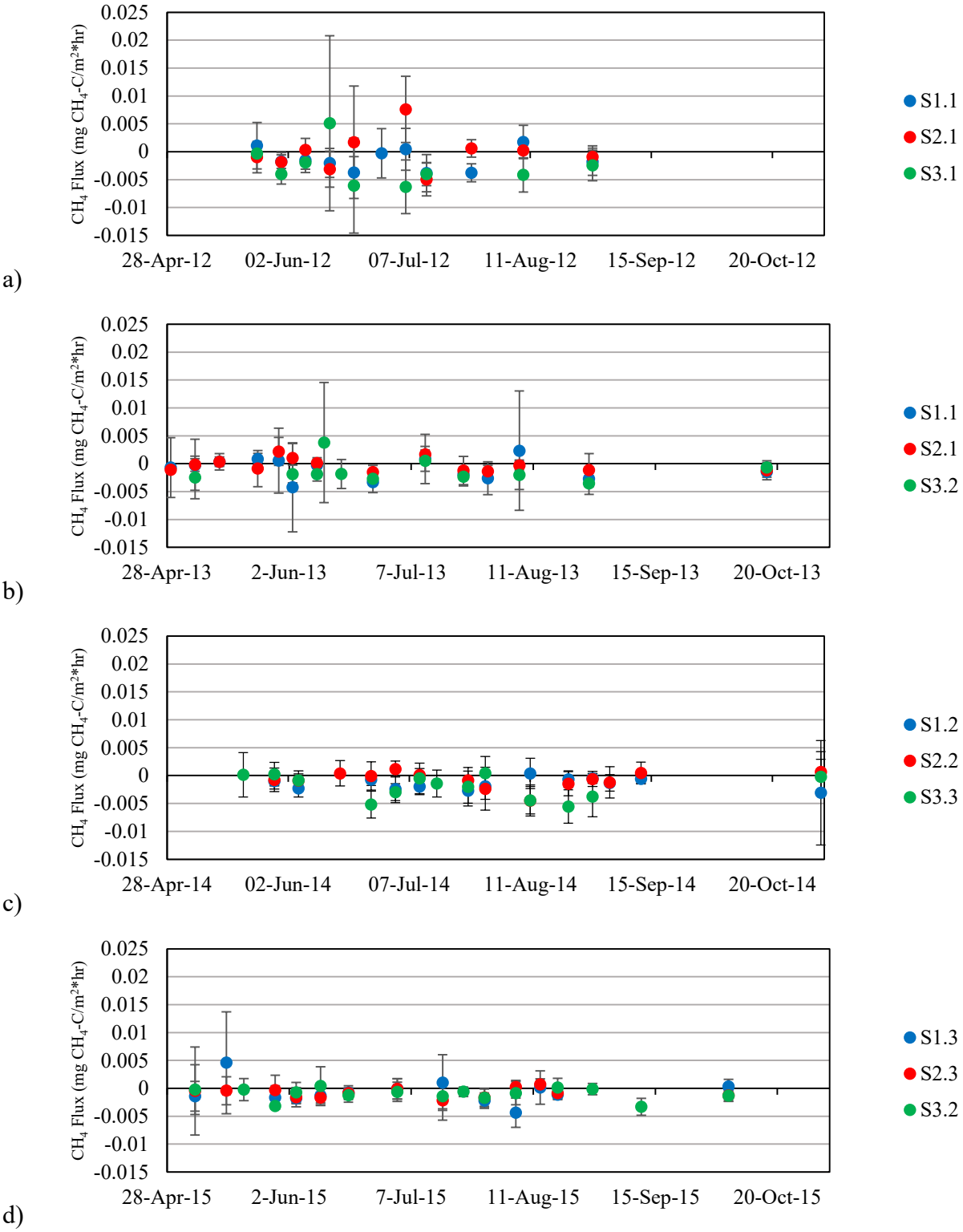
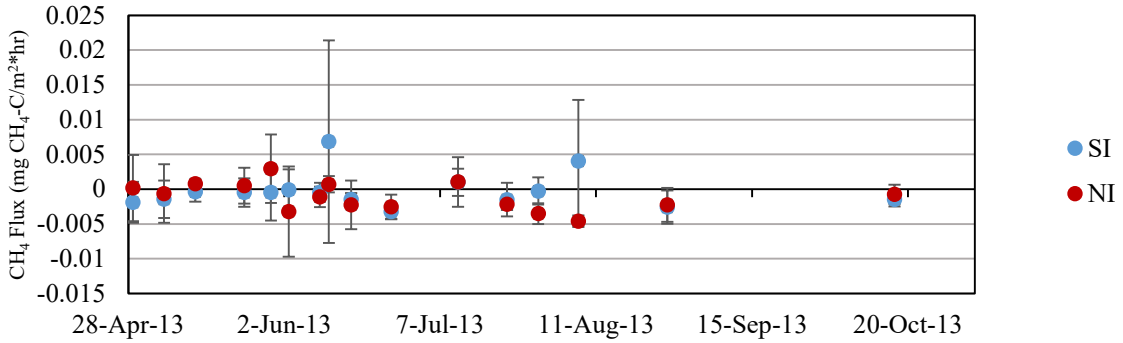


Figure 4.11: Mean CH_4 fluxes and standard deviations for S1 (blue), S2 (red) and S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.

a)



b)

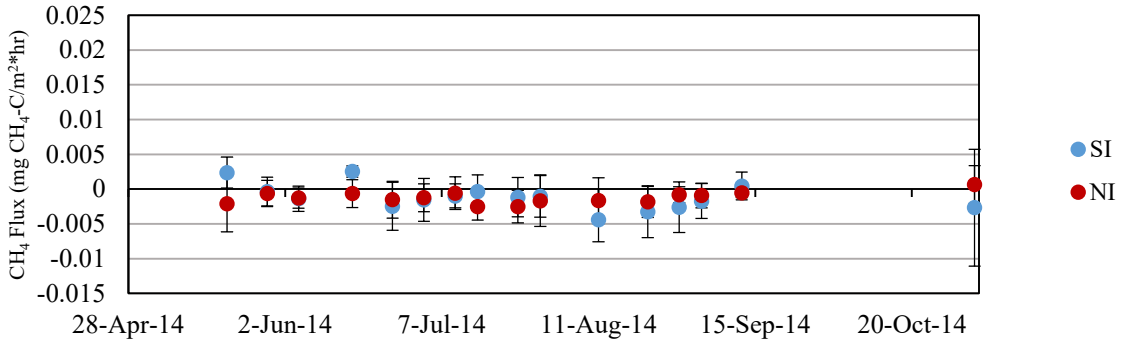


Figure 4.12: Mean CH_4 fluxes and standard deviations for SI (blue) and NI (red) in a) 2013 and b) 2014.

Chapter 5 – Summary and Conclusions

5.1 Summary

This study evaluated the emissions of CO_2, N_2O and CH_4 from organic soils under a sprinkler irrigation system. The study was conducted over four years from 2012 to 2015 on onion fields in Sherrington, Quebec, and Napierville, Quebec, two neighbouring townships. Both areas were farmed by Vert Nature a commercial vegetable producer. Sampling was done during the crop's growing season between the months of April and November on approximately a weekly basis. A total of 11, 16, 16 and 17 days of gas sampling were undertaken in 2012, 2013, 2014 and 2015, respectively. Each year varied in climatic conditions and thus irrigation schedules varied as well. In 2012 and 2015, the application of irrigation was not required. In 2013 and 2014, there were a total of six days of sampling after the application of sprinkler irrigation. All agronomic practices were undertaken by the producer in regards to normal vegetable production routines, including tillage, fertilization and irrigation. Soil and climatic conditions such as volumetric water content, soil temperature, air temperature and precipitation were collected. The preceding were used in order to explain the resulting fluxes of GHGs throughout the sampling season. Due to field rotations, sites varied from year to year. Site 1 was chosen as the moderately stabilized organic soil, site 2 the most stabilized organic soil and site 3 the least stabilized organic soil. Soil sampling and analysis for physical and chemical properties was done at each site. This in turn was used to quantify and explain the effects on GHG emissions.

5.2 Conclusions

The first objective was to compare the GHG emissions from sprinkler irrigated and non-irrigated onion fields cultivated on organic soils. There was very little difference in CO_2, N_2O and CH_4 fluxes between the sprinkler irrigated and non-irrigated plots. This may be partly due to the limitations of the number of days of irrigation, when gas sampling was done, and the amount of water applied. Statistically, the irrigation treatment was seen to have an effect on the CO_2 emissions in the year 2013 ($p = 0.05$). No significant difference was found in 2014 for CO_2 emissions. There was no significant difference between treatments for N_2O and CH_4 emissions.

The second objective was to compare the GHG emissions from three sites varying in soil organic matter content. There were statistically significant differences ($p = 0.05$) found between sites in three of the four growing seasons (2012, 2013 and 2014) for CO_2 and N_2O fluxes. Site 1 (i.e. moderately stabilized soil) and 3 (i.e. least stabilized) had significantly greater CO_2 emissions than Site 2 (i.e. most stabilized soil) in two of the four growing seasons (2012 and 2013). The amount of soil organic carbon that varied for all three sites was seen to be the leading factor influencing the CO_2 emissions. On average, sites with higher SOM produced higher CO_2 fluxes. Site 1 had significantly greater N_2O emissions than Sites 2 and 3 in one of the four growing seasons (2013). Site 2 had significantly greater N_2O emissions than Site 3 in one of the four growing seasons (2014). Bulk density was seen to have the most influence on N_2O fluxes followed by SOM. Soils with lower bulk densities are more porous and thus allow for more gas diffusion. With increasing SOM, typically, there will be an increase in soil microbial populations due to the abundance of substrate (Rochette et al., 2010). For CH_4 fluxes, no statistical differences between sites were found.

The third objective was to assess how environmental conditions and agricultural management practices influence GHG fluxes from organic soils under onion production. CO_2 fluxes increased throughout the season as the plant developed and temperatures escalated. N_2O fluxes were primarily influenced by spring thaw, rainfall and thus VWC, along with fertilizer application. For the most part, CH_4 was oxidized within the organic soils. However, management practices including fertilization and tillage slightly diminished CH_4 oxidation and thus emitted more to the atmosphere. Organic soils were found to be variant in terms of soil properties and consequently microbial communities. The results in turn were very sporadic and therefore this suggests that there were microsites which produced more GHGs than other areas in a same field.

Chapter 6 – Recommendations for Future Research

i. One of the challenges in this study was the lack of irrigation applications done on onion fields in southern Quebec. Onions do not require as much irrigation as other vegetable crops. Generally in Quebec, onions do not require any irrigation if there is sufficient enough rainfall at the critical formations of the plant. Additionally, the soils investigated retain moisture longer and therefore over-irrigation must be avoided. Irrigation was mainly done to prevent erosion rather than to aid in crop development. In order to really see the effects of irrigation on an onion field, the experiment would either have to be done in a drier region or on a different type of soil. To see the effects of irrigation on the organic soils in that same region, a different crop with higher water demands, such as lettuce, would have to be investigated.

ii. The static chamber method had some challenges. For one, the samples were limited temporally. Samples were taken approximately between morning and mid-day and thus the fluxes at other times of the day were not observed. Secondly, the chambers were set up only in one area of the field and thus limited spatially. To really quantify the results from the soil, chambers would have to be set up at many more locations over the field. This is especially important when sampling from organic soils due to their variable properties. On a third note, chambers were set up in the onion rows and not in the furrows. This made it difficult to separate the GHGs produced by the plants and from the soil. The furrows are generally of more compact quality and represent almost a third of the field. To properly justify the conclusions of a certain field, chambers would have to be set up in both furrows and onion rows. When moving around the chambers, where we stepped compacted the soil underneath. There were clear paths around each chamber which consisted of denser soils which was not representative of the actual field. Finally, since organic soils are of a spongy nature, when walking on the field, fluxes of gas can be released. A method that would reduce the amount of stepping around the sampling sites would greatly improve the overall results representative of the field.

iii. Organic soils are difficult to quantify due to the wide variation in soil properties throughout one field. For this study, four soil samples two meters apart were taken at each site. The results from this study were very sporadic and each chamber showed wide ranges of results. Soil samples should be taken close to each chamber thus doubling the amount of samples. Furthermore, analysis of soil properties should include more testing for inorganic electron acceptors which could have potential impacts on certain GHGs. On the same note, NO_3-N and NH_4-N fluxes should be monitored throughout the sampling season to see if there are any great variations and to compare these with GHG outputs.

iv. Microbial populations are very important in the production of GHGs. This study, did not test for which microbial populations were abiding at the sites sampled. This made it difficult to associate what was mainly driving the GHG emissions. Furthermore, each year sites were changed due to crop rotation. Soil microbial populations can differ greatly from field to field and from site to site. A greater understanding of what type of microbes are present in soils sampled will lead to better conclusions of the main drivers of CO_2, N_2O and CH_4 emissions.

v. Soil moisture and temperature were taken directly next to the chambers. This gave an approximate reading for these measurements. Soil conditions around the chamber could potentially differ from those within. To achieve more accurate results, measurements for these parameters should have been taken directly inside the chamber where the gases were being collected.

vi. Up until now, very little research has been conducted on GHG emissions from muck soils cropped to vegetables. Results from this study could be used in predicting GHG fluxes/emissions from other muck soils producing onions in southern Quebec, regions with similar climates and soil properties. The N_2O fluxes observed in this study are comparable to those found in the study conducted by Rochette et al. (2010) on organic soils cropped to lettuce in the same region of southern Quebec. Their resultant data indicated that N_2O fluxes generally did not exceed 0.2 mg N_2O-N m^{-2} *

hr^{-1}. However, they did observe rates that exceeded 0.2 mg N_2O-N m^{-2} * hr^{-1} which could possibly be explained by the different crop and greater total rainfall throughout the sampling season. It can be concluded that, environmental conditions, soil properties and crop type will have a great impact on the rate of GHG fluxes. Based on these results, it can be said that our findings are representative and can be extrapolated to other muck soils. Still, until further research is there to confirm, there is a certain amount of uncertainty.

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Appendix A

Table A1: The R^2 values for the effects of soil temperature and VWC on the CO_2 fluxes.

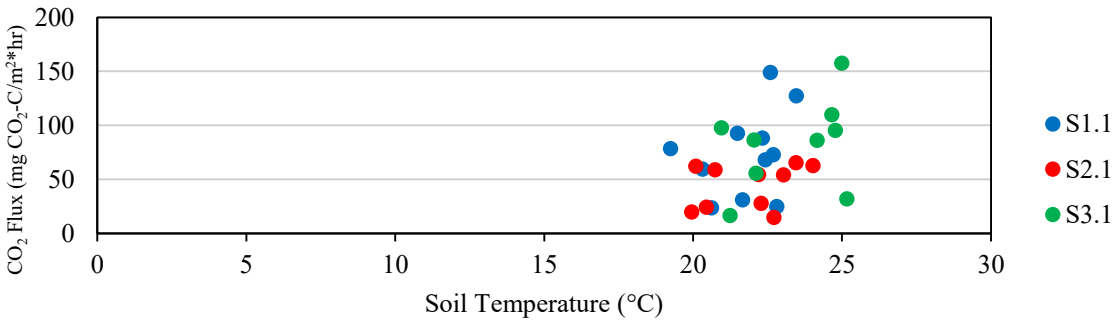
Year	2012			2013			2014			2015		
Site	S1.1	S2.1	S3.1	S1.1	S2.1	S3.2	S1.2	S2.2	S3.3	S1.3	S2.3	S3.2
Soil T (°C)	0.24	0.18	0.14	0.27	0.49	0.24	0.65	0.76	0.39	0.48	0.45	0.50
VWC (%)	0.02	0.11	0.05	0.13	0.35	0.10	0.25	0.07	0.06	0.09	0.10	0.25

Table A2: The R^2 values for the effects of soil temperature, soil volumetric water content and soil water filled pore space on the N_2O fluxes.

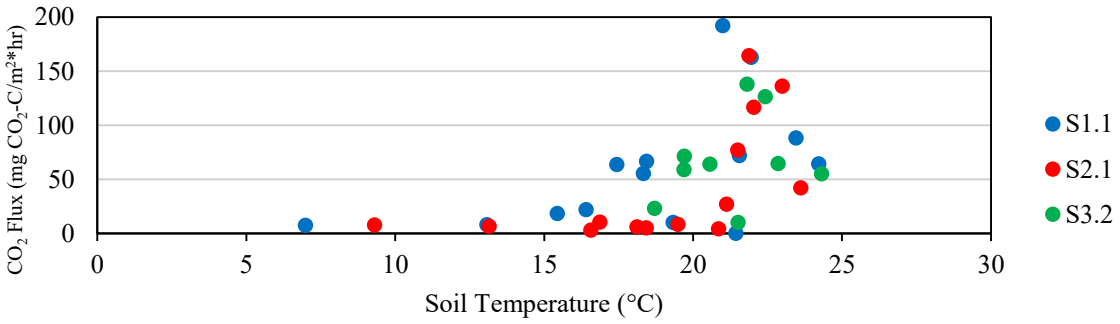
Year	2012			2013			2014			2015		
Site	S1.1	S2.1	S3.1	S1.1	S2.1	S3.2	S1.2	S2.2	S3.3	S1.3	S2.3	S3.2
Soil T (°C)	0.16	0.09	0.43	0.03	0.03	0.71	0.34	0.21	0.54	0.50	0.17	0.01
VWC (%)	0.43	0.28	0.64	0.37	0.49	0.06	0.23	0.12	0.36	0.13	0.07	0.05
WFPS (%)	0.45	0.25	0.61	0.44	0.55	0.01	0.23	0.04	0.36	0.13	0.07	0.05

Table A3: The R^2 values for the effects of soil temperature, soil volumetric water content, soil water filled pore space, N_2O flux and CO_2 flux on the CH_4 fluxes.

Year	2012			2013			2014			2015		
Site	S1.1	S2.1	S3.1	S1.1	S2.1	S3.2	S1.2	S2.2	S3.3	S1.3	S2.3	S3.2
Soil T	0.01	0.12	0.05	0.04	0.05	0.36	0.14	0.34	0.14	0.18	0.03	0.07
VWC	0.06	0.07	0.37	0.09	0.15	0.12	0.10	0.22	0.26	0.02	0.12	0.07
WFPS	0.06	0.22	0.36	0.07	0.05	0.17	0.13	0.56	0.34	0.02	0.12	0.07
N_2O	0.07	0.01	0.003	0.03	0.24	0.09	0.05	0.06	0.07	0.01	0.14	0.01
CO_2	0.30	0.19	0.39	0.06	0.19	0.17	0.21	0.48	0.66	0.23	0.13	0.57



a)



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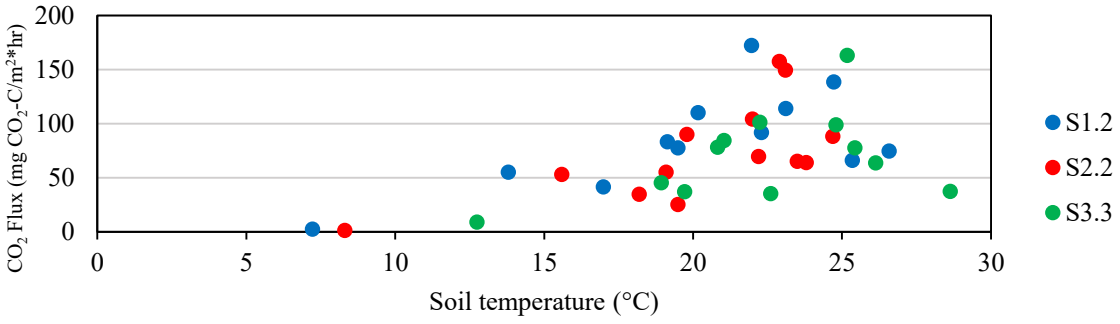
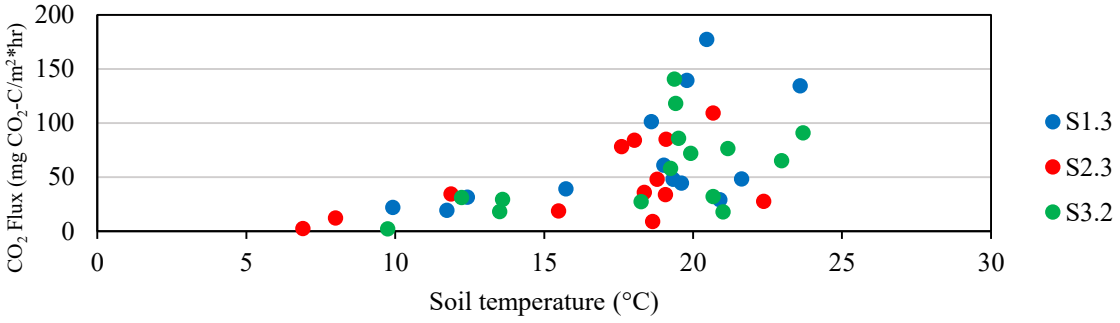
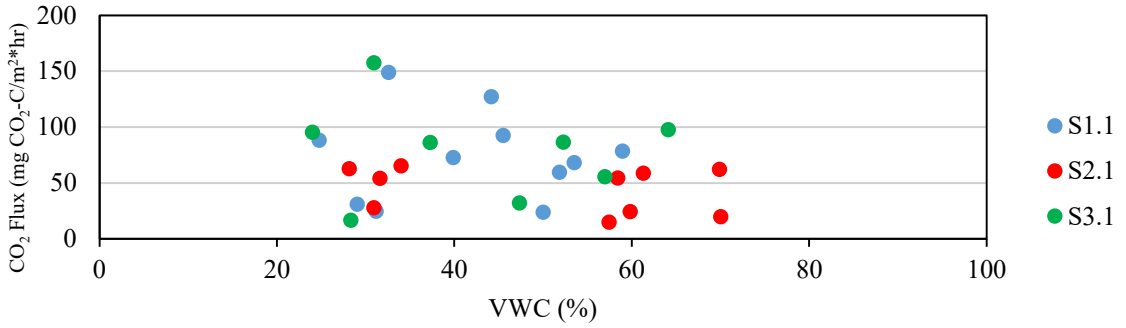


c)

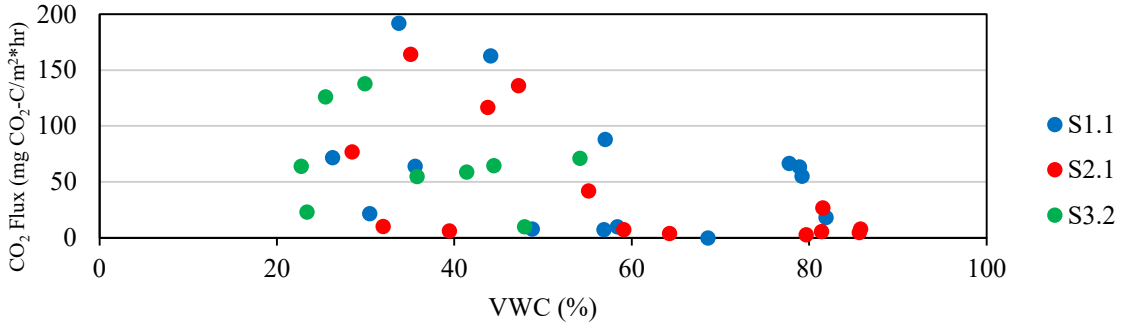


d)

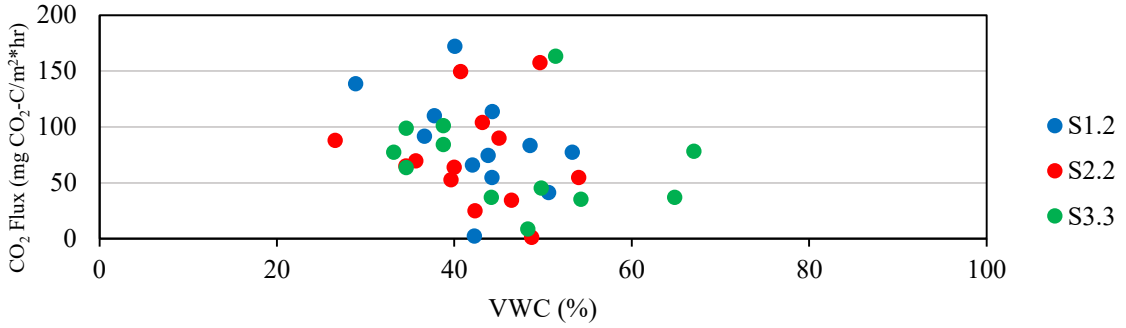
Figure A1: Scatter plot of soil temperature (°C) and CO_2 fluxes at S1 (blue), S2 (red), S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



a)



b)



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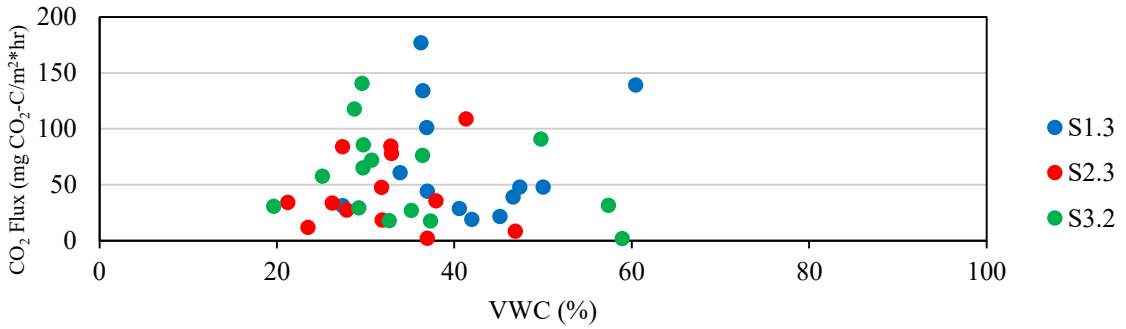


d)

Figure A2: Scatter plot of volumetric water content (%) and CO_2 fluxes at S1 (blue), S2 (red), S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.

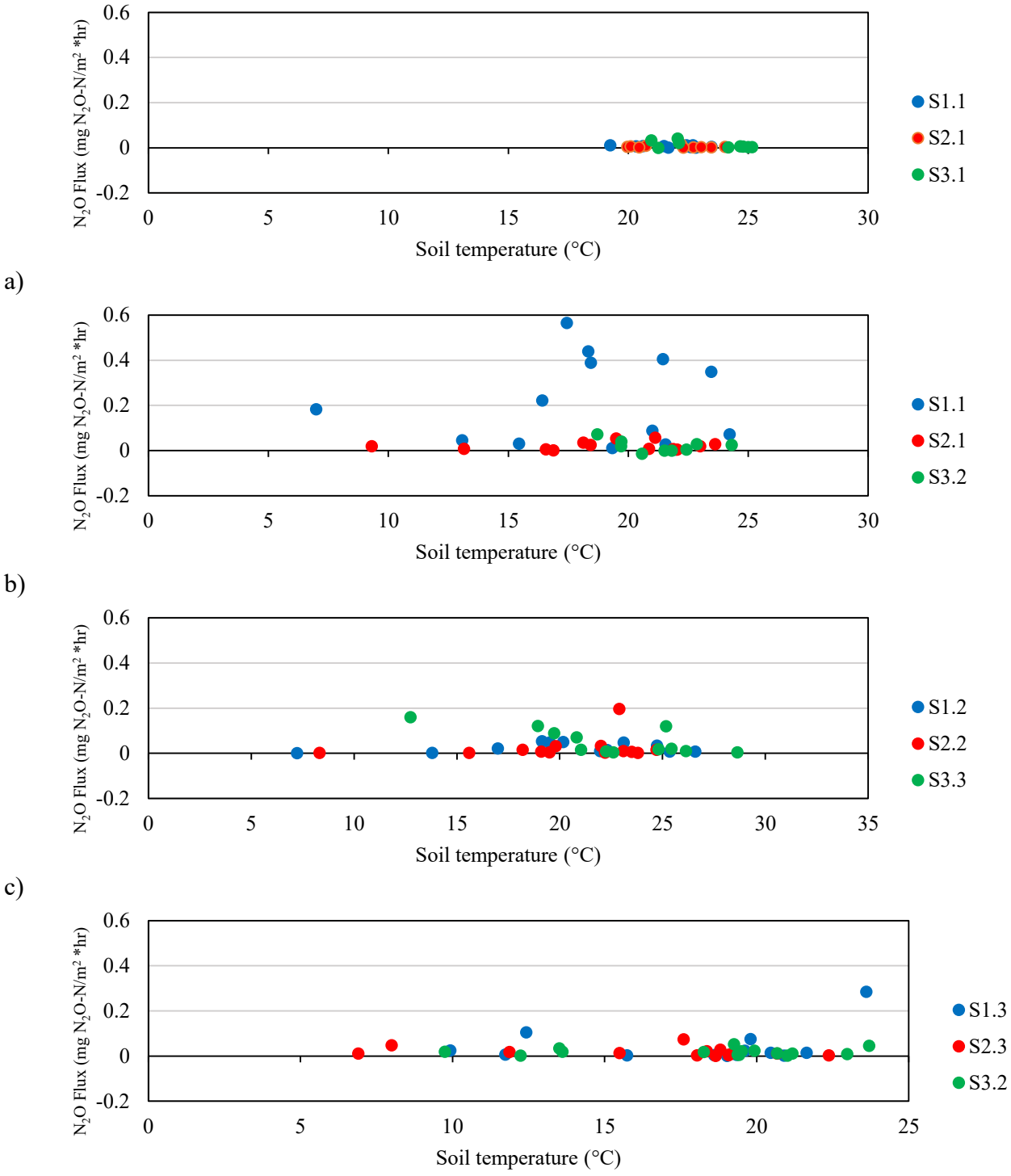
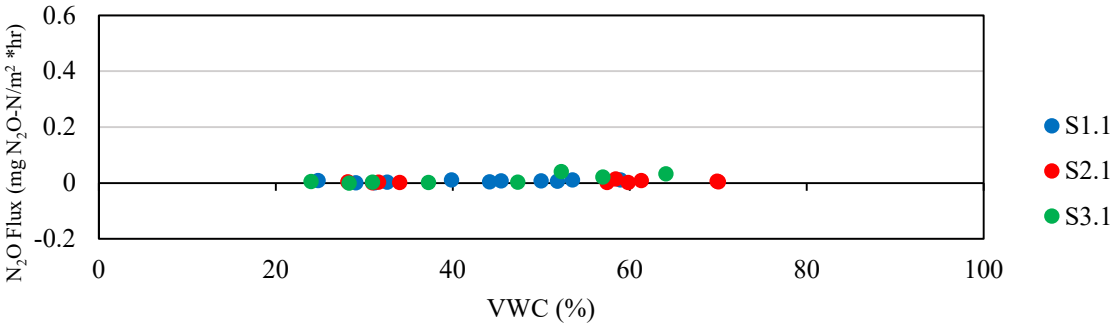
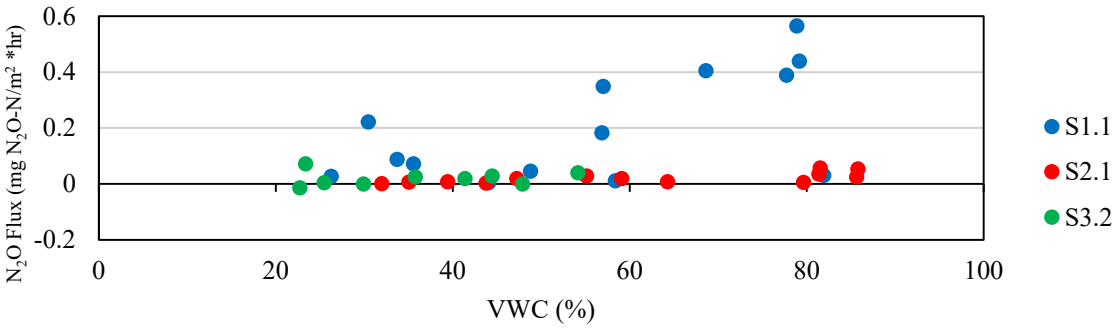


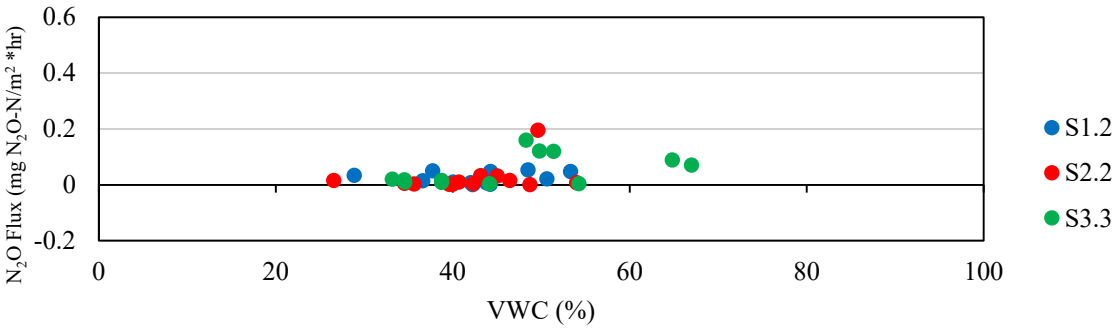
Figure A3: Scatter plot of soil temperature (°C) and N_2O fluxes at S1 (blue), S2 (red), S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



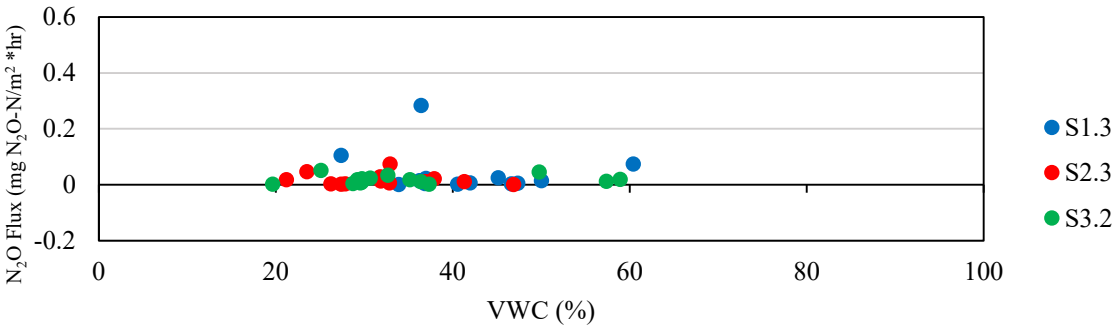
a)



b)

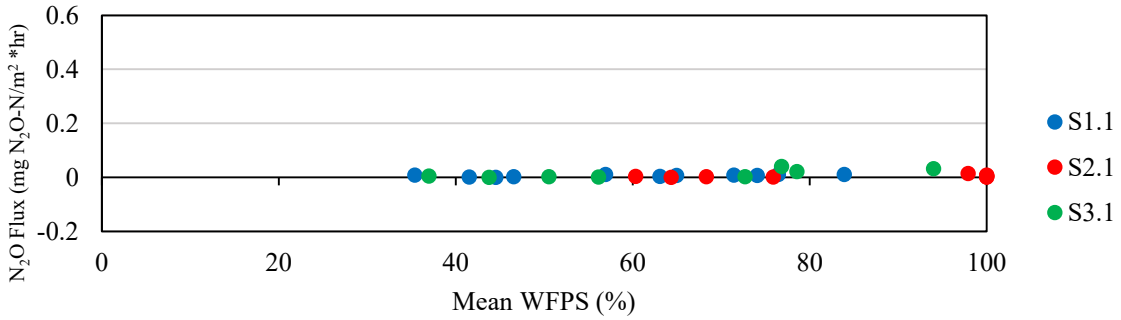


c)

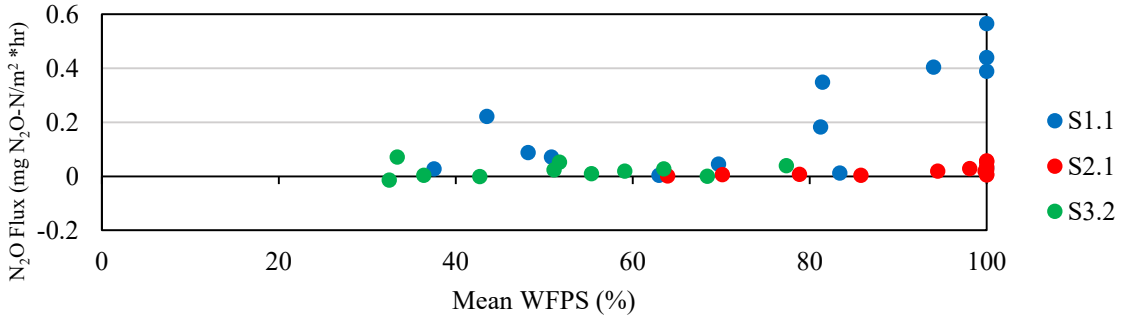


d)

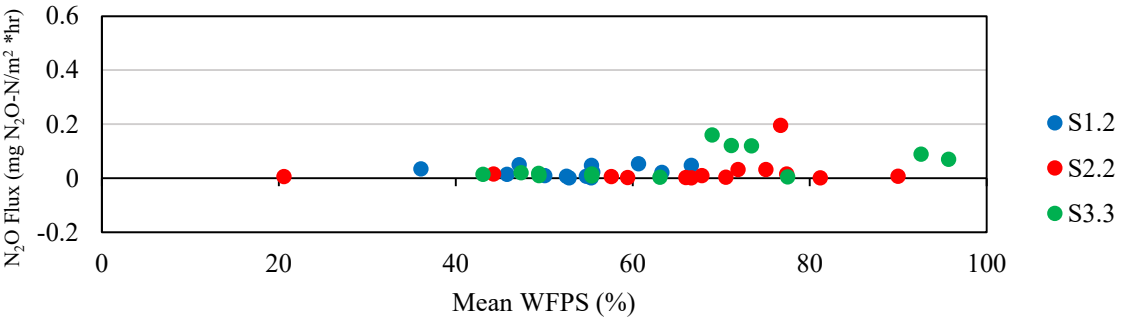
Figure A4: Scatter plot of volumetric water content (%) and N_2O fluxes at S1 (blue), S2 (red), S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



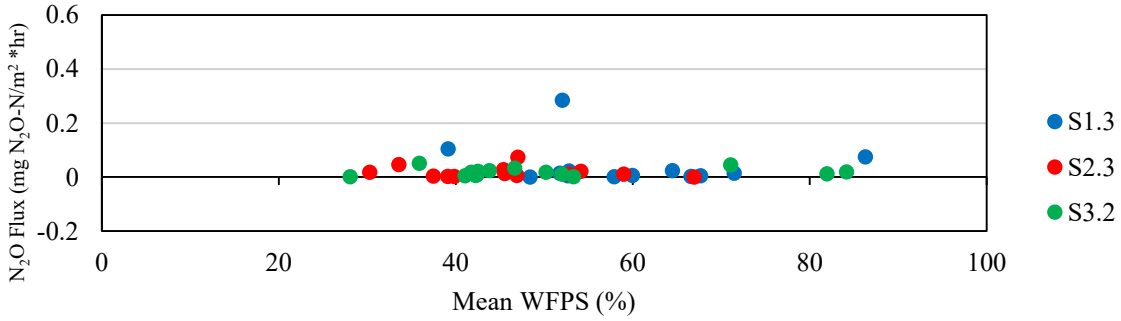
a)



b)

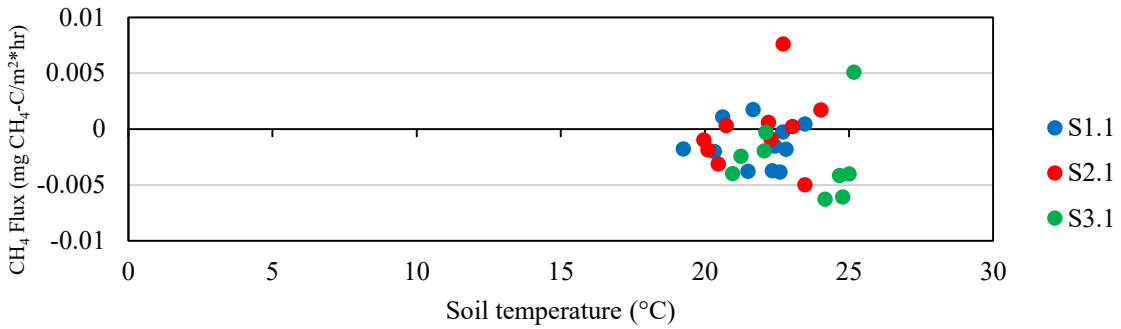


c)

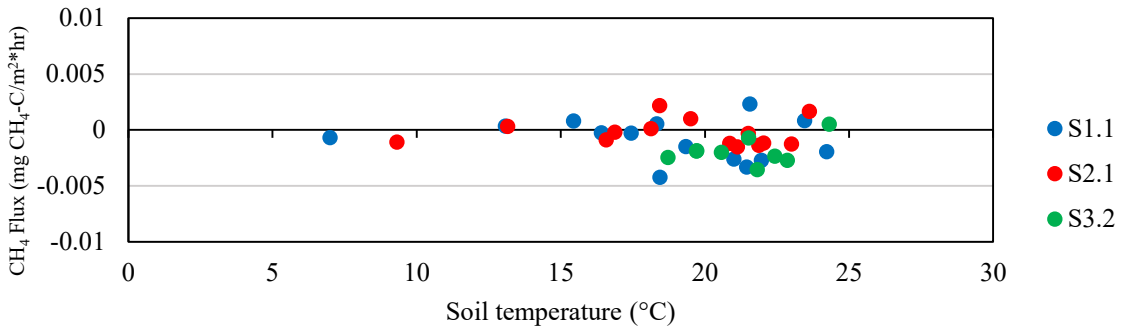


d)

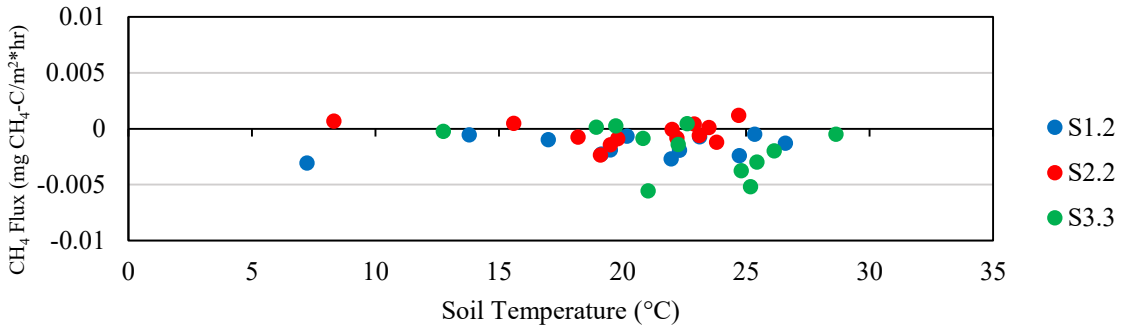
Figure A5: Scatter plot of soil water filled pore space (%) and N_2O fluxes at S1 (blue), S2 (red), S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



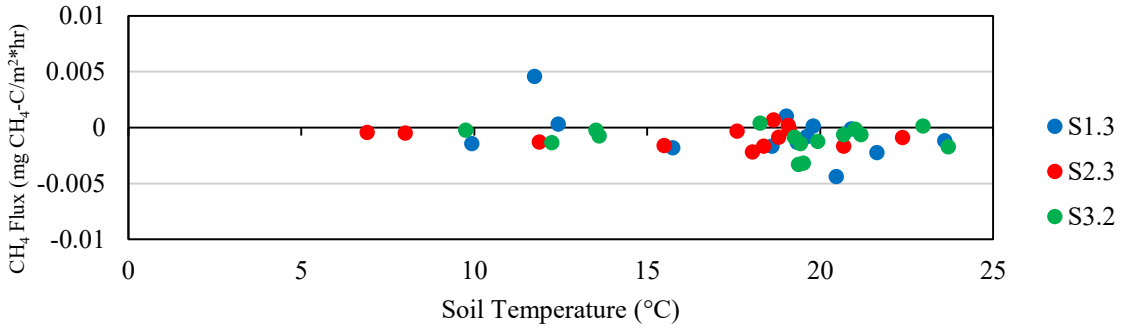
a)



b)

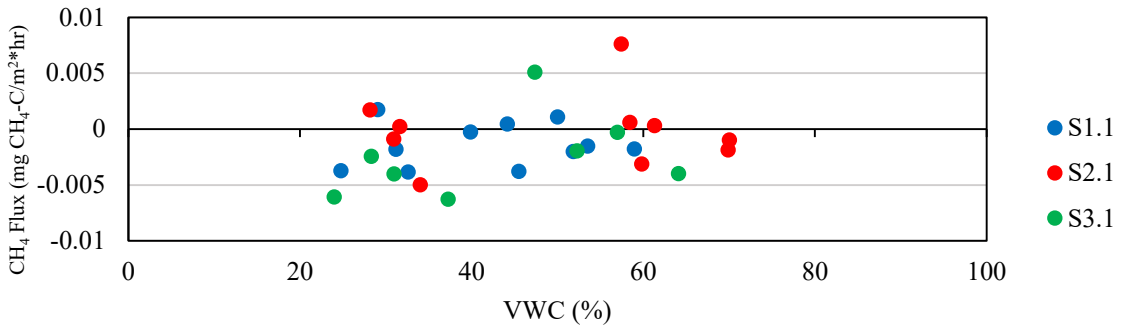


c)

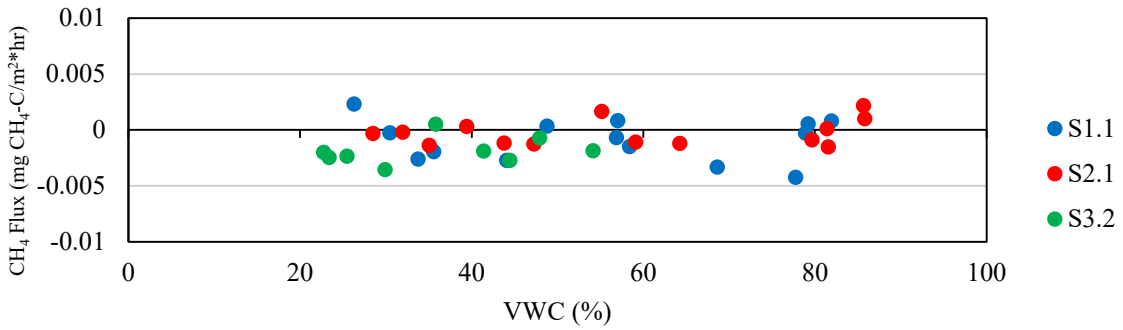


d)

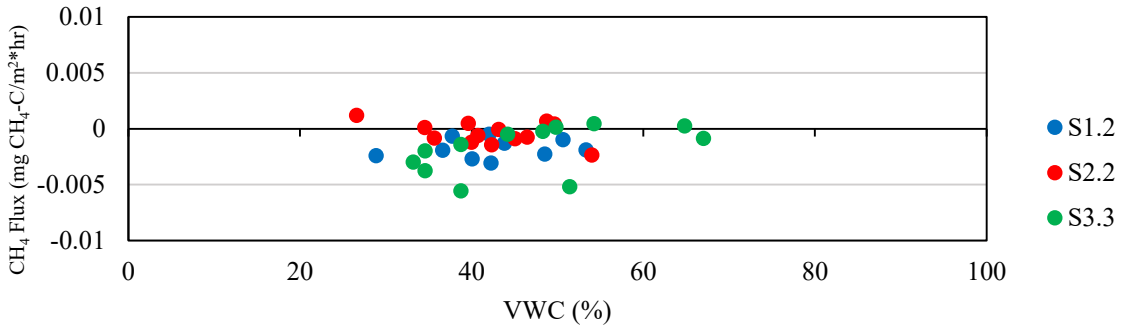
Figure A6: Scatter plot of soil temperature (°C) and CH_4 fluxes at S1 (blue), S2 (red), and S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



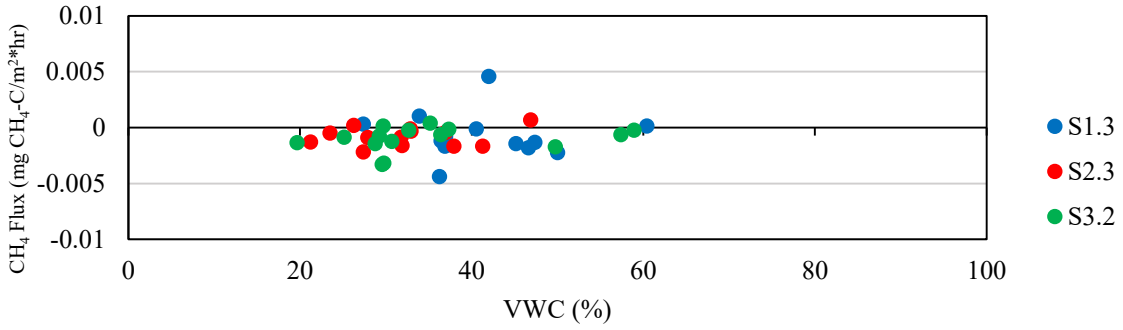
a)



b)

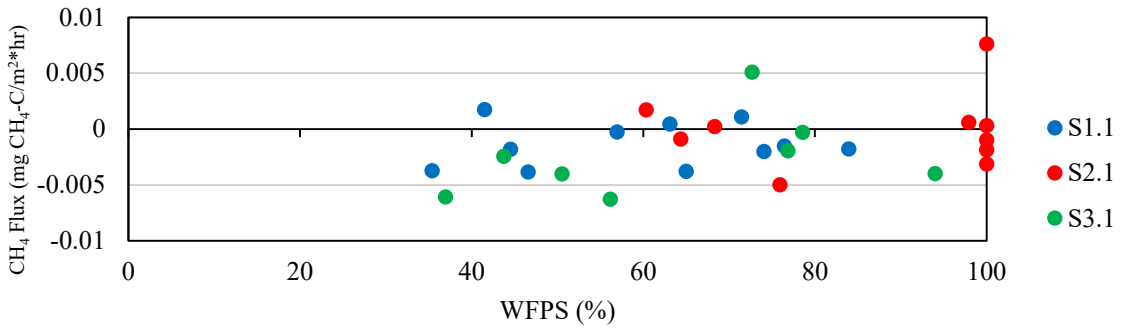


c)

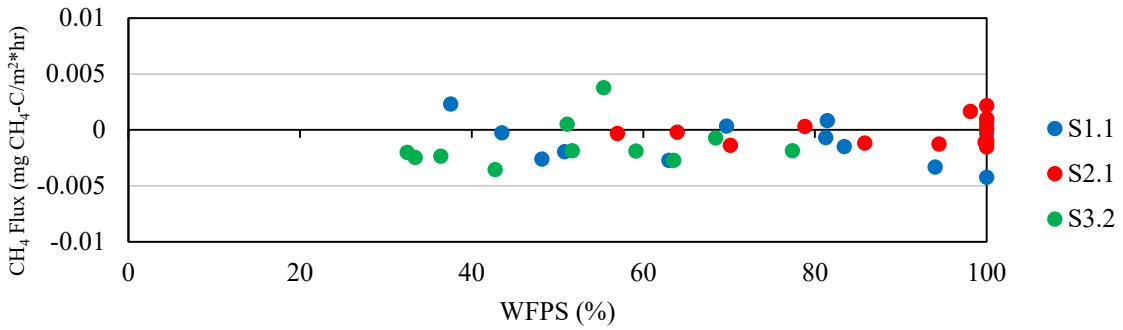


d)

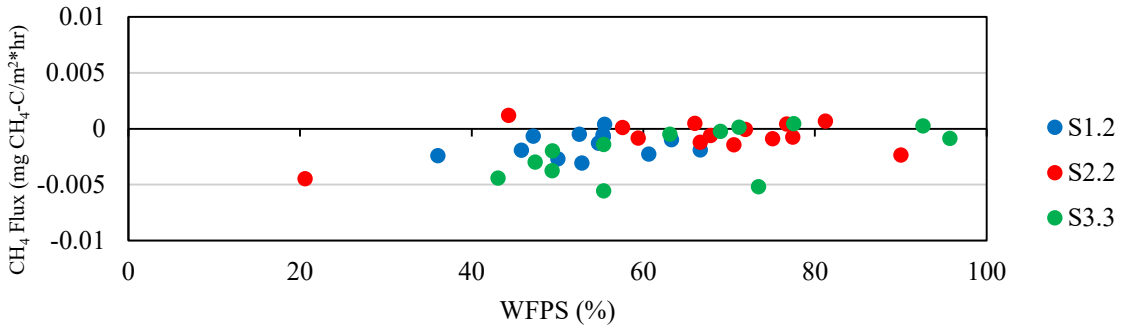
Figure A7: Scatter plot of soil volumetric water content and CH_4 fluxes at S1 (blue), S2 (red), and S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



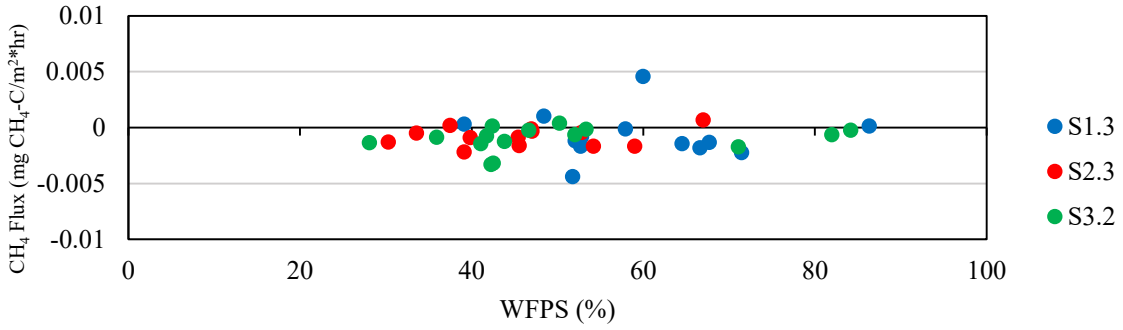
a)



b)

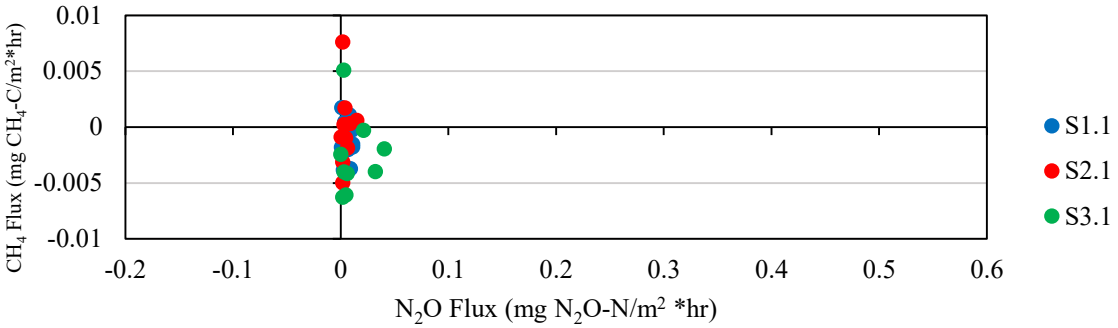


c)

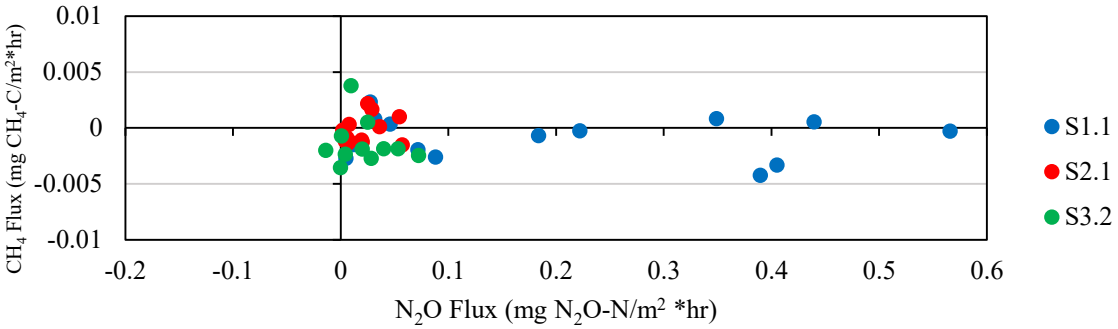


d)

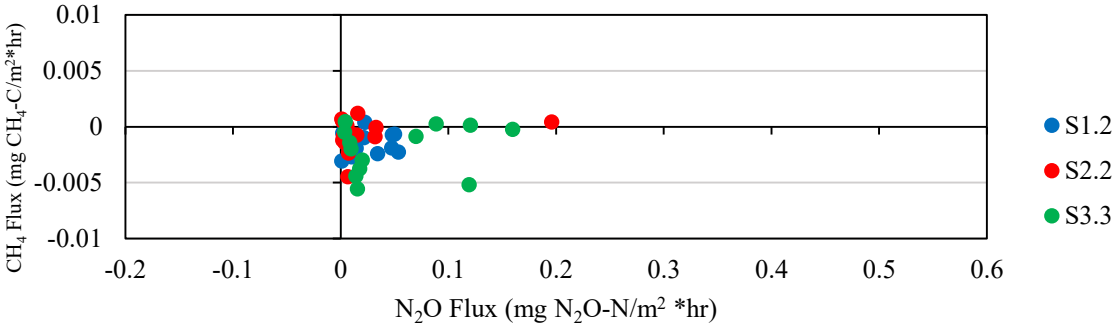
Figure A8: Scatter plot of soil water filled pore space and CH_4 fluxes at S1 (blue), S2 (red), and S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



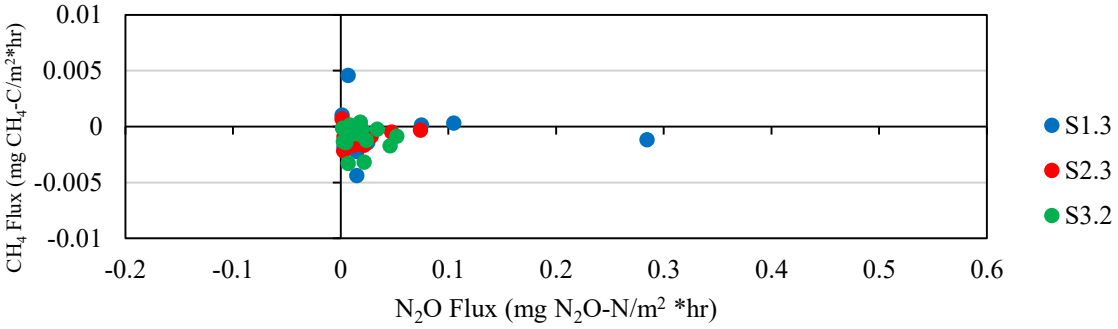
a)



b)

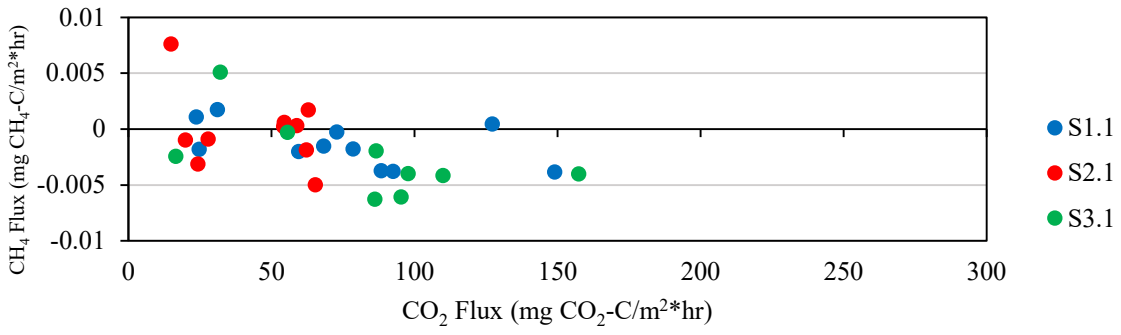


c)

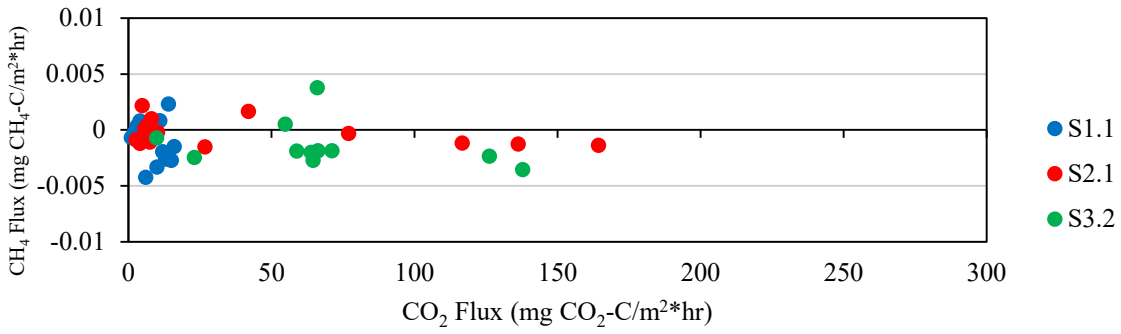


d)

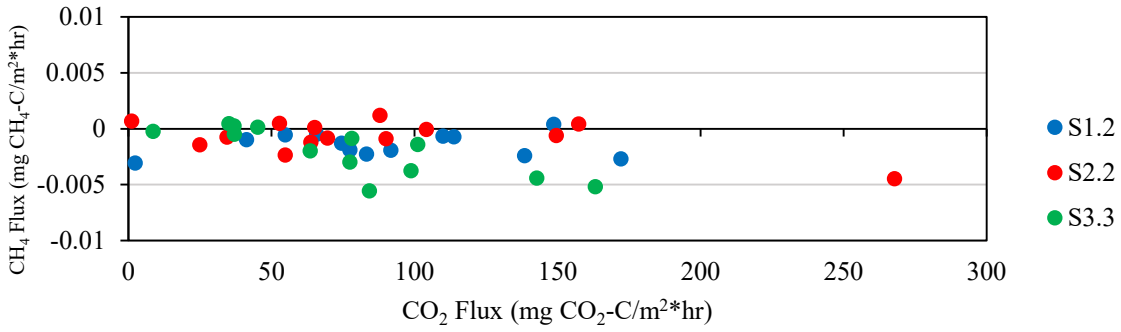
Figure A9: Scatter plot of N_2O fluxes and CH_4 fluxes at S1 (blue), S2 (red), and S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.



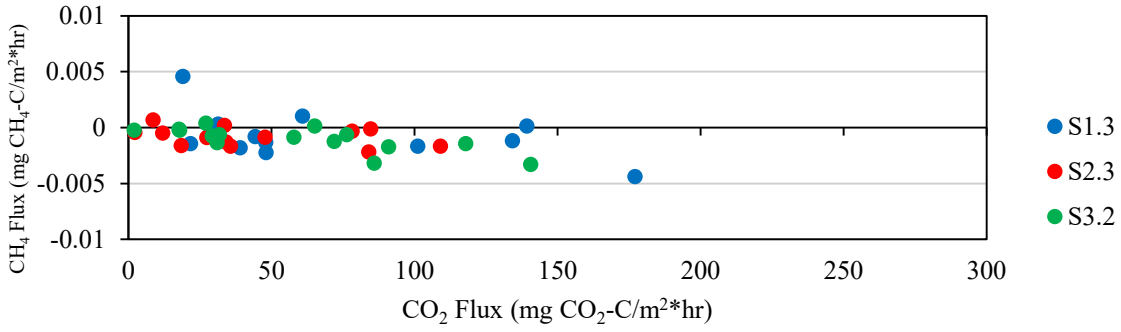
a)



b)



c)



d)

Figure A10: Scatter plot of CO_2 fluxes and CH_4 fluxes at S1 (blue), S2 (red), and S3 (green) in a) 2012, b) 2013, c) 2014 and d) 2015.

Appendix B

Table B1: Chemical soil properties at 0-20 cm, 20-40 cm and 40-60 cm for P, Al, Mg, Ca, K, Mn, Na, Organic Matter Content, pH, NO3 and NH4 in 2012, 2013 2014 and 2015.

Year	Site	Depth (cm)	P (mg/kg of dry soil)	Al (mg/kg of dry soil)	Mg (mg/kg of dry soil)	Ca (mg/kg of dry soil)	K (mg/kg of dry soil)	Mn (mg/kg of dry soil	Na (mg/kg of dry soil)	Organic Matter Content (%)	pH	mg N-NO3/g of dry soil	mg N-NH4/g of dry soil
2012	S1.1	0-20	123	94	363	3331	628	N/A	N/A	83	5.71	0.0202	0.0193
		20-40	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0402	0.0259
		40-60	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0761	0.0293
	S2.1	0-20	124	418	415	3400	436	N/A	N/A	46	6.68	0.0125	0.0159
		20-40	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0142	0.0168
		40-60	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0202	0.0160
	S3.1	0-20	59	55	397	4121	398	N/A	N/A	82	6.32	N/A	N/A
		20-40	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
		40-60	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
2013	S1.1	0-20	123	94	363	3331	628	N/A	N/A	83	5.71	0.0202	0.0193
		20-40	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0402	0.0259
		40-60	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0761	0.0293
	S2.1	0-20	124	418	415	3400	436	N/A	N/A	46	6.68	0.0125	0.0159
		20-40	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0142	0.0168
		40-60	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0202	0.0160
	S3.2	0-20	N/A	N/A	N/A	N/A	N/A	N/A	N/A	82	N/A	0.0095	0.0104
		20-40	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0064	0.0076
		40-60	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0049	0.0075
2014	S1.2	0-20	62	331	1014	7423	337	N/A	N/A	88	5.44	0.0627	0.0031
		20-40	56	328	987	6855	168	N/A	N/A	90	5.26	0.0353	0.0039
		40-60	40	332	1067	6039	117	N/A	N/A	92	5.08	0.0230	0.0029
	S2.2	0-20	171	541	1029	7140	212	N/A	N/A	40	6.93	0.0644	0.0026
		20-40	100	463	904	4515	95	N/A	N/A	22	6.84	0.0218	0.0014
		40-60	13	443	443	2968	66	N/A	N/A	4	7.48	0.0096	0.0011
	S3.3	0-20	118	259	1063	6634	315	N/A	N/A	61	6.29	0.0573	0.0028
		20-40	108	408	722	4519	211	N/A	N/A	45	5.74	0.0169	0.0027
		40-60	49	589	330	1774	111	N/A	N/A	9	5.60	0.0077	0.0020
2015	S1.3	0-20	80	237	1197	7436	369	9	255	79	5.75	0.0902	0.0019
		20-40	71	215	1187	7193	355	10	261	82	5.70	0.0813	0.0019
		40-60	70	190	1080	6563	359	10	249	82	5.61	0.0971	0.0020
	S2.3	0-20	98	197	1219	7706	298	8	315	75	5.72	0.0805	0.0017
		20-40	89	182	1207	7774	306	8	325	80	5.71	0.0896	0.0018
		40-60	74	56	1044	6412	282	8	276	73	5.55	0.1188	0.0025
	S3.2	0-20	N/A	N/A	N/A	N/A	N/A	N/A	N/A	82	N/A	0.0095	0.0104
		20-40	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0064	0.0076
		40-60	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.0049	0.0075

Table B2: Physical soil properties at 0-20, 20-40 and 40-60 cm for bulk density, porosity, soil classification and hydraulic conductivity in 2012, 2013, 2014 and 2015.

Year	Site	Depth (cm)	Bulk Density (g/cm3)	Porosity (%)	Soil type	Hydraulic conductivity (cm/sec)
2012	S1.1	0-20	0.290	0.7	organic	0.0115
		20-40	0.316	0.7	organic	0.0193
		40-60	0.159	0.9	organic	0.0259
	S2.1	0-20	0.557	0.5	organic	0.0004
		20-40	0.720	0.3	organic	0.0003
		40-60	1.778	0.3	loam	0.0001
	S3.1	0-20	N/A	N/A	N/A	N/A
		20-40	N/A	N/A	N/A	N/A
		40-60	N/A	N/A	N/A	N/A
2013	S1.1	0-20	0.290	0.7	organic	0.0115
		20-40	0.316	0.7	organic	0.0193
		40-60	0.159	0.9	organic	0.0259
	S2.1	0-20	0.557	0.5	organic	0.0004
		20-40	0.720	0.3	organic	0.0003
		40-60	1.778	0.3	loam	0.0001
	S3.2	0-20	0.296	0.7	organic	0.0036
		20-40	0.328	0.7	organic	0.0037
		40-60	0.203	0.8	organic	0.0090
2014	S1.2	0-20	0.212	0.8	organic	0.0130
		20-40	0.216	0.8	organic	0.0142
		40-60	0.174	0.8	organic	0.0267
	S2.2	0-20	0.492	0.6	organic	0.0014
		20-40	0.498	0.5	organic	0.0016
		40-60	1.284	0.5	sandy clay	0.0007
	S3.3	0-20	0.352	0.7	organic	0.0085
		20-40	0.321	0.7	organic	0.0007
		40-60	0.884	0.7	sandy clay loam	0.0008
2015	S1.3	0-20	0.315	0.7	organic	0.0040
		20-40	0.272	0.8	organic	0.0079
		40-60	0.186	0.8	organic	0.0046
	S2.3	0-20	0.325	0.7	organic	0.0019
		20-40	0.206	0.8	organic	0.0010
		40-60	0.193	0.8	organic	0.0041
	S3.2	0-20	0.296	0.7	organic	0.0036
		20-40	0.328	0.7	organic	0.0037
		40-60	0.203	0.8	organic	0.0090