Impact of organic fertilizers on crop yield, soil carbon stock and greenhouse gas fluxes from corn-soybean agroecosystems

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Abstract

Municipal organic waste transformed into organic fertilizers can replace mineral fertilizers to sustain crop production and soil organic carbon (SOC) stock but may induce soil greenhouse gas (GHG) emissions, depending on the physicochemical properties of organic fertilizer. This project evaluated the effect of three organic fertilizers on corn (*Zea mays L.*) and soybean (*Glycine max L.*) yields, SOC stock (0 – 20 cm) and soil GHG fluxes. Organic fertilizers were composted food waste (compost; 240 kg N/ha), LysteGro biosolid slurry (LysteGro; 215 kg N/ha) and liquid anaerobic digestate (digestate; 231 kg N/ha), plus a mineral fertilizer control (NPK; 170 kg N/ha). Fertilizers were applied once at the beginning of the corn-soybean rotation at the Emile A. Lods Agronomy Centre (Lods), Ste-Anne-de-Bellevue, Quebec (8.4 – 14.2 g SOC /kg, sandy loam soil) and the Elora Research Station (Elora), Elora, Ontario (21.3 – 30.8 g SOC /kg, silt loam soil). Corn and soybean yields were similar among fertilizer treatments and comparable to regional averages, indicating satisfactory agronomic performance of all organic fertilizers. The SOC stocks remained similar after one-time application of organic fertilizers. Transient effects (within one month of fertilizer application) on N_2O fluxes did not lead to any significant difference in the cumulative growing-season N_2O and CO_2 emissions. Methane fluxes were close to zero in all site-years. In addition, the long-term effect of these organic fertilizers on crop yields, SOC stock and soil N_2O emission under two future climate scenarios (RCP4.5 and RCP8.5) from 2018 – 2070 was simulated using the DayCent model after calibrating the model with two seasons of field data. DayCent predicted that digestate application would produce the highest corn silage yield (25 – 28 % higher than NPK on average) whereas compost would produce the highest soybean grain yield (2.8 – 4.4% higher than NPK) at both sites. Compost application was predicted to accrue the most SOC and have the lowest greenhouse gas intensities

(GHGI, in t CO_2-eq/t C harvested) until SOC stocks reached steady-state in 2040. While OM-rich compost appeared to be a better GHG mitigation option for the next 20 years, digestate could produce higher corn yield with lower N_2O emission until the end of 2070. In summary, these organic fertilizers met agronomic requirements in the short-term and were predicted to have superior agroecological outcomes than NPK fertilizer in the long-term.

Résumé

Les déchets organiques municipaux transformés en engrais organiques peuvent remplacer les engrais minéraux pour soutenir la production agricole et le stock de carbone organique du sol (COS), mais peuvent induire des émissions de gaz à effet de serre (GES) du sol, selon les propriétés physicochimiques des engrais organiques. Ce projet a évalué l’effet de trois engrais organiques sur les rendements du maïs (*Zea mays L.*) et du soja (*Glycine max L.*), du stock de COS (0 à 20 cm) et des flux de GES du sol. Les engrais organiques étaient les déchets alimentaires compostés (compost; 240 kg N/ha), lisier biosolides LysteGro (LysteGro; 215 kg N/ha) et digestat anaérobie liquide (digesta; 231 kg N/ha), ainsi qu’un contrôle des engrais minéraux (NPK; 170 kg N/ha). Les engrais ont été appliqués une fois au début de la rotation maïs-soja au Centre Emile A. Lods de Recherche et Agronomie (Lods), Ste-Anne-de-Bellevue (Québec) (8,4 – 14,2 g COS /kg, loam sableux) et à la Station de Recherche d’Elora (Elora), Elora (Ontario) (21,3 – 30,8 g COS /kg, loam limoneux). Les rendements du maïs et du soja étaient similaires avec chaque traitement d’engrais et proches des moyennes régionales, ce qui indique une performance agronomique satisfaisante de tous les engrais organiques. Les stocks de COS sont restés similaire après l’application exceptionnel d’engrais organiques. Les effets transitoires (pendant un mois suivant l’application des engrais) sur les flux de N_2O n’ont pas entraîné à des différences significatives entre les émissions cumulatives de N_2O et de CO_2 pendant la saison de croissance. Les flux de méthane étaient proches de zéro dans toutes les années de site. De plus, l'effet à long terme de ces engrais organiques sur les rendements des cultures, le stock de COS et les émissions de N_2O du sol sous deux scénarios climatiques futurs (RCP4.5 et RCP8.5) de 2018 à 2070 a été simulé à l'aide du modèle DayCent après calibrage du modèle avec deux saisons de données de terrain. DayCent a prédit que l’application de digestat

produirait le rendement d'ensilage de maïs le plus élevé (25 – 28 % plus élevé que le NPK en moyenne) alors que le compost produirait le rendement le plus élevé de grain de soja (2,8 – 4,4 % plus élevé que NPK) aux deux sites. DayCent prévoyait que l'application de compost accumulerait le plus de COS et qu'elle aurait les intensités de gaz à effet de serre les plus faibles (GESI, en t CO_2-eq/t C récolté) jusqu'à ce que les stocks de COS atteignent un état d'équilibre en 2040. Bien que le compost riche en matière organique semble d'être une meilleure option d'atténuation des GES pour les 20 prochaines années, le digestat pourrait produire des rendements plus élevés de maïs avec moins d'émissions de N_2O jusqu'à la fin de 2070. En résumé, ces engrais organiques répondaient aux besoins agronomiques à court terme et on prévoyait des résultats agroécologiques supérieurs à ceux des engrais NPK à long terme.

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Contribution of Authors

This manuscript-based thesis consists of Chapter 1, a general introduction outlining the research background, brief experimental method, hypotheses and objectives, written by the candidate and edited by Dr. Joann Whalen. The original experimental design is a joint effort by Dr. Joann Whalen and Dr. Maren Oelbermann at the University of Waterloo. However, due to the limited length of MSc. study (the research was originally a Ph.D. project), it was shortened and slightly modified by the candidate.

Chapter 2 is the first manuscript of literature review covering existing knowledge on the topic of my thesis, followed by a conclusion and a revisit of the hypotheses. This chapter is authored by the candidate and edited by Dr. Joann Whalen.

Chapter 3 is the second manuscript consisting of the findings from field experiments conducted at the Emile A. Lods Agronomy Centre and Elora Research Station. The candidate first-authored this chapter and was edited by Dr. Joann Whalen. The data and methodology at Elora Research Station were provided by Emmanuel Badewa, a Ph.D. candidate in Dr. Maren Oelbermann's lab at the University of Waterloo and written in its own language by the candidate.

Chapter 4 is the last manuscript consisting of the use of the DayCent model, to make predictions about future agroecological outcomes at the Emile A. Lods Agronomy Centre and Elora Research Station. The candidate first authored this chapter with Dr. Joann Whalen as the editor. The same data from Chapter 3 was used, hence Emmanuel Badewa contributed the data collected at Elora Research Station.

Chapter 5 and 6 are the general discussion and overall conclusion covering the whole thesis, outlining key findings, limitations and what is answered and unanswered. They were written by the candidate and edited by Dr. Joann Whalen.

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List of Abbreviations

C	Carbon
CO_2	Carbon dioxide
CH_4	Methane
CRCM5	5th generation Canadian regional climate model
DOC	Dissolve organic carbon
Elora	Elora Research Station
GHG	Greenhouse gas
GHGI	Greenhouse gas intensity
GWP	Global warming potential
Lods	Emile A. Lods Agronomy Research Center
MBC	Microbial biomass carbon
N	Nitrogen (element)
N_2	Nitrogen (molecular)
N_2O	Nitrous oxide
NO	Nitric oxide
NO_2^-	Nitrite
NO_3^-	Nitrate
NH_4^+	Ammonium
O_2	Oxygen
OM	Organic matter
pCO_2	Atmospheric carbon dioxide concentration
r^2	Coefficient of determination
rRMSE	relative root mean squared error
SEM	Standard error of the mean
SE_{est}	Standard error of the estimate
SOC	Soil organic carbon
VSWC	Volumetric soil water content
WFPS	Water-filled pore space

Chapter 1 – General Introduction

Increasing soil organic carbon (SOC) is crucial for soil fertility and climate change mitigation. Agricultural soil has the potential to sequester a substantial amount of carbon (C). If all arable land in the world is managed according to best conservation practices, this would result in an estimated storage of 1.55 – 1.95 Pg C/yr (Minasny et al., 2017), which is about one-third of the annual global net greenhouse gas (GHG) emissions, estimated at 4.7 Pg C/yr during 2007 – 2016 (Le Quéré et al., 2017). Therefore, increasing SOC is a key action that can tackle food insecurity and climate change simultaneously.

One way to increase SOC in agricultural soils is to apply organic fertilizers produced from municipal organic waste. In Canada, VandenBygaart et al. (2003) reported that applying organic fertilizers resulted in a large increase of 28 ± 11% in SOC versus only 7.0 ± 2.8% for mineral N fertilizer, relative to unfertilized control. This is because organic fertilizer typically supplies abundant organic C directly to soil that can take years to completely mineralize, as well as plant nutrients e.g. nitrogen (N) and phosphorus (P) that can sustain crop production and hence crop C input to soil (Diacono & Montemurro, 2011; Düring & Gäth, 2002; Yang et al., 2013). Despite these benefits, organic fertilizers can be a source of soil GHG (CO_2, CH_4 and N_2O) emissions (Thangarajan et al., 2013). The extra source of nutrients and organic matter (OM) not only support plant growth, but also modify soil microbial processes that lead to the production of GHGs at varying rates (Thangarajan et al., 2013). Applying organic fertilizers at high organic C rate promotes CO_2 emissions by increasing heterotrophic respiration, and if prolonged soil anaerobic conditions developed, CH_4 would be emitted due to methanogenesis (Paterson & Sim, 2013; Wang et al., 2003; Le Mer & Roger, 2001). A greater loss of gaseous C also means less C will be retained in the soil. On the other hand, high soluble mineral N plus labile C in the fertilizer under abundant soil moisture levels can lead to significant N_2O emissions due to increased denitrification (Senbayram et al., 2012; Thangarajan et al., 2013; Velthof et al., 2003). For instance, slurries that contain a low C:N (<5) at about 95% water content were grouped as the highest N_2O emitter (emission factor = 1.12%) among all organic fertilizers in a meta-analysis by Charles et al. (2017). Different organic fertilizers on the market can vary widely in physicochemical properties. Knowledge remains scarce as to how different organic fertilizers would differentially affect C and N fluxes in agroecosystems (Cayuela et al., 2010; Ho et al., 2015; Ho et al., 2017). Along with the constantly changing soil conditions (e.g. soil temperature and moisture) brought about by site-level weather in the short-term and climate change in the long-term, a wide difference in soil GHG emissions could arise.

This study focuses on three organic fertilizers derived from municipal organic wastes in Québec and Ontario, Canada. They are: 1. Aerobically-composted food waste (compost) produced by AIM Environmental Group; 2. LysteGro, a biosolid slurry produced from an alkaline hydrolysis process developed by the company Lystek International Inc.; 3. Liquid fraction of anaerobic digestate (digestate) from biogas digesters produced by Bio-En Power Inc. The first objective of this thesis is to evaluate how these organic fertilizers affect crop yields, SOC stock and growing-season soil GHG fluxes in two corn-soybean agroecosystems over 2 years in Québec and Ontario, Canada. Since compost supplies more organic C, followed by LysteGro and digestate, soil amended with compost is expected to have the highest overall CO_2 emission from the decomposition and respiration of the added organic C. Since digestate and LysteGro have more mineral N whereas compost has mostly organic N, I hypothesize that digestate and LysteGro will have higher crop yields due to the higher plant availability of mineral N. However, there will be greater N_2O emission in soil amended with LysteGro because it has both high mineral N and organic C concentration that will favor denitrification. Methane fluxes are expected to be insignificant compared to CO_2 and N_2O fluxes in upland annual crop fields in eastern Canada (Gregorich et al., 2005).

We do not expect SOC will show noticeable difference among fertilizer treatments within the 2-year field study (Necpálová et al., 2014). The long-term trend (2018 – 2070) of SOC stock, along with crop yield and N_2O emission will be evaluated by the DayCent model, the second objective of our study. DayCent is a biogeochemical model widely used to simulate C and N flows among the atmosphere, plants, and soil in terrestrial ecosystems at daily time-steps (Del Grosso, 2012; Necpálová et al., 2015). In Canadian agroecosystems, it has been validated repeatedly with long-term SOC and crop yield data (Chang et al. 2013; Grant et al., 2016; Guest et al., 2017). It considers climate, soil and management information (e.g. what fertilizer is applied) together, making it suitable for evaluating the effectiveness of agricultural management practices under climate change. We hypothesize that applying compost will result in the highest SOC due to its highest amount of organic C. We also expect that crop yield and N_2O emission in the farther future will increase the most for compost treatment due to the extra soil N supply in a high-SOC soil (Ding et al., 2013; Li et al., 2005). Finally, we expect that under a warmer and CO_2-rich atmosphere, crop yield will be higher (Kimball et al., 2002), SOC stock will be lower (Wiesmeier et al., 2016), and N_2O emission will be higher (Van Groenigen et al., 2011).

In summary, the objectives of my thesis are: 1) to investigate the **present-day short-term effect** of organic fertilizers on crop yield and soil GHG fluxes; and 2) to evaluate the **long-term effects** (2018 – 2070) of municipal organic wastes under climate change scenarios with DayCent. The prospects for applying organic fertilizers derived from municipal organic waste to agricultural land for crop production and as a C sequestration strategy will be discussed.

Chapter 2 - Soil carbon, fertility and greenhouse gas fluxes in agroecosystems receiving organic fertilizers – a review of concepts

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2.1 Soil carbon dynamics in agroecosystems

2.1.1 What is soil organic carbon?

Soil organic carbon (SOC) is defined as all carbon (C) derived from living biomass, non-living and decaying organic matter (OM) in soils (Stockmann et al., 2013). Soil organic carbon can be classified in several ways such as its age (based on radiocarbon dating technique), its origin (e.g. animal-, plant- or microbially-derived) and the degree of persistence (e.g. in terms of turnover times). Traditional soil models (e.g. Century, DayCent and RothC) generally classify SOC into three conceptual C pools with defined decomposition rate constants in order to represent the decomposition of OM of different stability (Figure 2-1; Zimmermann et al., 2007). More recent classifications distinguish physically-distinct and measurable C pools to facilitate the experimental validation of model mechanisms governing SOC dynamics. For instance, microbial biomass C (MBC), dissolved organic C (DOC), particulate organic C, as well as occluded SOC in soil aggregates and mineral-associated SOC are used in the Millennial model (Figure 2-1). These SOC pools have different ecological functions and decomposition rates, and therefore have different relative importance in the soil C cycle in the short-term and long-term.

2.1.2 The accumulation and loss of soil organic carbon in agroecosystems

Soil organic carbon changes with time due to continuous C inputs and losses. When the gain of C is greater than the loss of C, SOC stock increases, and vice versa. In annual crop systems, C inputs ultimately come from the C fixation of crops (and to a lesser extent from autotrophic soil microbes), mainly as root exudates of growing crops, fine root turnover and crop residues. Additional amendments such as animal manures and organic fertilizers derived from agricultural or municipal waste can also supply a substantial amount of C. The crop-derived C plus the organic C in amendments are fragmented, mixed and redistributed in the soil profile via the bioturbation and communition of soil fauna as well as flowing soil water. The feeding and defecation activities of soil fauna also contributed to the formation of occluded SOC in aggregates (Wiesmeier et al., 2019). Highly-persistent SOC such as mineral-associated SOC was shown to originate from microbial secretions and their necromass (Barré et al., 2018; Kallenbach et al., 2016; Miltner et al., 2012). For instance, exopolysaccharides secreted by microbes were found to be able to associate more readily with mineral surfaces than particulate OM (Lehmann & Kleber, 2015).

On the other hand, there are numerous ways that SOC is lost from the soil profile. The dominant pathway of SOC loss is through biological decomposition. When heterotrophs in the soil acquire and metabolize energy-rich OM, a portion of it is being respired (or fermented) to produce energy and CO_2, a gas readily diffuses out of the soil profile. Sinsabaugh et al. (2016) estimated that on average, 72% of assimilated C in soil microbes is respired, although it could vary widely from 44 – 98% depending on the organic substrate quality and soil conditions. Paul et al. (1999) estimated annual soil CO_2 emissions ranging from 790 to 960 g CO_2-C/m^2/yr from multiple corn and soybean systems in Michigan. In oxygen-depleted soils, some of the CO_2 and small organic C compounds will be converted to CH_4 via methanogenesis before escaping the soil surface (Thangarajan et al., 2013). However, CH_4 emission generally does not represent a significant C loss in non-flooded systems, with an estimated small sink capacity of -0.029 g CH_4-C/m^2/yr (net uptake) in annual crop fields in eastern Canada (Gregorich et al., 2005).

Soil erosion is another SOC loss pathway as strong wind and rain can have enough kinetic energy to carry small particles of topsoil containing SOC away from the original area. Erosion also exposes the subsoil SOC to air, making it more susceptible to decomposition, as well as reduces plant productivity that decreases C inputs, which together tends to further accelerate the loss of SOC (Berhe et al., 2012; Gregorich et al., 1998). Ketcheson and Webber (1978) reported C loss due to erosion ranging from 73.7 – 110 g C/m^2/yr over 11 years of growing continuous corn on manured and moldboard-plowed fields in Guelph, Ontario.

The final C loss pathway is the leaching loss of DOC, dissolved CO_2 as well as colloidal mineral-bound C into groundwater (Neff & Asner, 2001). The former two also stem from biological decomposition as the formation of small and soluble organic molecules as well as CO_2 during decomposition can be readily dissolved. The downward movement of soil water then carried the dissolved C out of the soil profile. Kindler et al. (2011) estimated that DOC and dissolved CO_2 leaching loss across European croplands amount to 4.1 ± 1.3 and 14.6 ± 4.8 g C/m^2/yr respectively, which is small compared to respiration C loss.

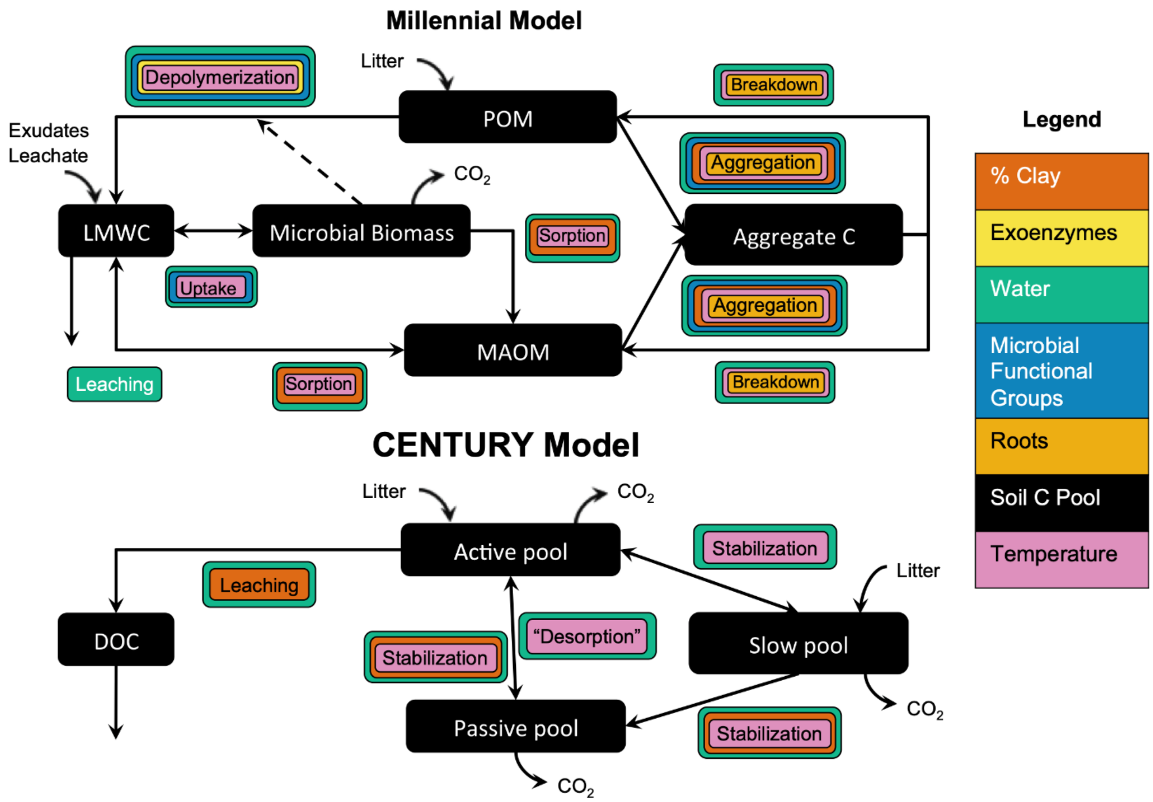

Figure 2-1. Conceptual model diagram of the Millennial (top) and Century models (bottom). The black boxes are carbon pools, and the colored boxes are fluxes. Solid arrows indicate the direction of each flux. The color legend indicates edaphic, biological, and climatic factors that may modify the rate of a given flux. Dash lines indicate controls (i.e. microbial biomass regulates the depolymerization rate). **Adapted from Abramoff et al. (2017).**

2.1.3 Organic fertilizers as a source of SOC and fertility

2.1.3.1 The potential to gain SOC following organic fertilizer application

The conversion of natural land to agricultural use generally results in substantial SOC loss due to insufficient C inputs from crops or forages relative to native vegetation. It was estimated that in eastern Canada, about 123 Tg C of SOC from ~10 Mha of cultivated land (22% of pre-agricultural SOC stock) was lost when native forest was converted to agriculture (Gregorich et al., 2005). Gregorich et al. (2005) and VandenBygaart et al. (2003) reviewed the effect of conservation management practices on SOC and reported that conservation tillage (compared to conventional tillage) had minimal effect on SOC in Canada. Certain cropping practices (e.g. full straw return and legumes in rotation with corn) have moderate but variable positive effect on SOC (2.7 ± 2.4% and 13 ± 10% respectively), whereas applying organic fertilizers led to a gain of 28 ± 11% SOC versus an increase of 7.0 ± 2.8% SOC with mineral N fertilizer, relative to an unfertilized control. This means that increasing C inputs from organic fertilizer application can potentially replenish more of the lost SOC stock.

Croplands generally accrue more SOC when they receive organic fertilizers than mineral fertilizers (Poulton et al., 2018; Tian et al., 2009). This is because it takes years for the directly added organic C from organic fertilizers to mineralize completely. For instance, Yang et al. (2013) found the extra SOC stock (up to 24.7 Mg/ha more than unfertilized control) persisted after 10 yr following a single application of different composts at high rates (75 – 300 t/ha) in a continuous corn field in southwestern Ontario. Moreover, the amount of SOC accrued over long-term is roughly proportional to the amount of organic C applied. One of the best examples is in the long-term Rothamsted experiment (1942 – 1967) where the SOC gain was nearly double from organic fertilizers (farm yard manure, vegetable compost, sewage sludge, sludge compost) applied at the 2-fold application rate compared to the single application rate (Poulton et al., 2018). Furthermore, Tian et al. (2015) suggested that applying organic fertilizers with low C:N can reduce microbial nutritional stress that helps sequester crop C via the microbial C pump, as they found that biosolid-amended soils sequestered 2 times more crop C (on top of its own organic C) than unamended control. Finally, agronomic rates of organic fertilizers generally produce comparable crop yields to mineral fertilizers, thus maintaining a similar level of crop C input to soil (Diacono & Montemurro, 2011). Therefore, organically-fertilized croplands are expected to accumulate SOC until a steady-state is reached.

2.1.3.2 Organic fertilizers supply nitrogen to both plants and microbes

Organic fertilizers also contain nitrogen (N). Aside from the immediately plant-available mineral N in soluble forms (NH_4^+ and NO_3^-), they also contain organic N that gradually becomes available by the action of soil microbes. As decomposition proceeds, the bulk of organic matrices is broken down and respired, leaving organic N compounds such as proteins from microbial residues. They are then hydrolyzed by proteases into small, soluble molecules e.g. short peptides and amino acids (Schulten & Schnitzer, 1997). These soluble organic N compounds in soil solution are further mineralized by hydrolytic enzymes to release NH_4^+ or NH_3 (Mengel, 1996). As such, crops grown with organic fertilizers that contain proportionally more organic N than mineral N generally release N over longer-term but may provide insufficient N to crops within short-term. For instance, Paul & Beauchamp (1993) found that the N recovered by corn crops following the application of liquid cattle manure (34% N in organic forms) and solid cattle manure (74% N in organic forms) were on average 18 and 5% of the total N applied respectively (compared to urea at 47% N recovered), at the Elora Research Station.

Ammonium ions (NH_4^+) are not just taken up by plants. Microbes such as autotrophic ammonia oxidizers acquire and oxidize some of the NH_4^+ (into NO_2^-) to fuel C fixation and build proteins. Nitrite oxidizers continue the oxidation chain and convert NO_2^- into NO_3^-. This whole series of oxidative reaction called nitrification requires aerobic environment and proceeds from $NH_3/NH_4^+ \rightarrow NH_2OH \rightarrow NO_2^- \rightarrow NO_3^-$ (Figure 2-2). Incomplete ammonia oxidation sometimes occurs, giving rise to gaseous intermediates i.e. NO and N_2O that could readily diffuse out of the soil profile into the atmosphere when the soil is not waterlogged (< 95% water-filled pore space)

(Figure 2-2; Caranto & Lancaster, 2017; Rabot et al., 2015). Complete nitrification results in the production of NO_3^-, another plant-available N form.

Nitrates are susceptible to leaching loss. Otherwise, when anoxic conditions develop, NO_3^- are also taken up by heterotrophic denitrifiers as terminal electron acceptors for energy production. This process (denitrification) consists of a series of reduction reactions: $NO_3^- \rightarrow NO_2 \rightarrow NO \rightarrow N_2O \rightarrow N_2$ (Figure 2-2). Ammonia oxidizers are also capable of reducing NO_2^- to NO or N_2O under low availability of O_2 via nitrifier denitrification (Figure 2-2; Zhu et al., 2013). Denitrification leads to the production of gaseous compounds in different steps i.e. NO, N_2O and N_2, in which the latter (N_2) can readily diffuse to the atmosphere even under waterlogged conditions (Wrage et al., 2001). In particular, nitrous oxide (N_2O) is a potent GHG (estimated to have 265 times the warming potential of CO_2 over 100 years) and an ozone-depleting agent (IPCC, 2014), and is arguably the most important GHG in croplands due to the large emission from fertilizer application without an accompanying sink. Gregorich et al. (2005) estimated that about 0.237 g N_2O-N/m^2/yr was emitted in manured crop fields across eastern Canada, which contributes to the highest net global warming potential (GWP) compared with CO_2 and CH_4. The factors affecting N_2O production will be discussed further in 2.2.3. Finally, the lost N (gas, leachate or exported crop N) can be replenished by the next round of organic fertilizer application, atmospheric N deposition, and nitrogen fixation of plant symbionts and free-living diazotrophs to drive the cycle again (Jensen & Hauggaard-Nielsen, 1993).

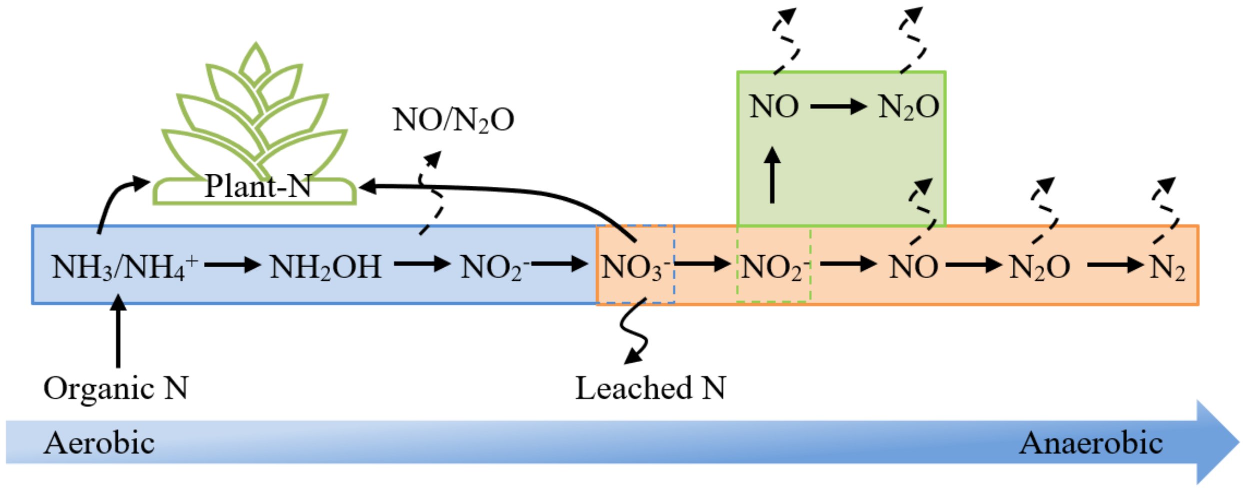

Figure 2-2. Three major nitrogen transformation processes in soil that lead to N_2O production: nitrification (blue), heterotrophic denitrification (orange), nitrifier-denitrification (green). Wavy arrows represent N losses (dashed line = gaseous and solid line = N leaching). **Adapted from reactions presented in Butterbach-Bahl et al. (2013)**.

2.2 Biophysical controls of greenhouse gas emission from agricultural soils

2.2.1 Biophysical controls of soil carbon dioxide production

All cells generate CO_2 upon respiration to produce energy with the exception of methanogenesis where CO_2 is consumed. Soil respiration can be subdivided into heterotrophic (from soil biota) and autotrophic respiration (from root cells). Heterotrophic respiration in particular is in direct association with OM decomposition and hence is more important for SOC dynamics. The basic requirement of heterotrophic respiration is the presence of electron donors and terminal electron acceptors. Small, soluble organic substrates are electron donors and molecular oxygen (O_2) is the dominant terminal electron acceptor under aerobic conditions. First, a high concentration of organic substrates in soil, such as in high SOC soils or through high inputs from crops and organic fertilizers, directly increases soil respiration. An abundance of labile C (energy-rich and readily-metabolizable C e.g. sugars and organic acids) in particular can stimulate microbes to proliferate and amplify the overall rate of respiration (Paterson & Sim, 2013; Sato & Seto, 1999; Wang et al., 2003). Second, O_2 sustains aerobic respiration, which produces considerably more CO_2 per unit time than anaerobic respiration that uses other terminal electron acceptors (Blagodatskaya et al., 2014; Sierra et al., 2017). For example, Blagodatskaya et al. (2014) found 4.5 times greater CO_2 emission under aerobic conditions (21% O_2) than anaerobic conditions (1% O_2) in an incubation study with a SOC-rich Haplic Cambisol.

The availability of organic substrates and O_2 are controlled by the constantly-changing soil moisture and temperature. An optimal level of soil moisture is necessary for microbial hydrolysis of polymeric OM to produce organic substrates, as well as for optimal substrates and O_2 diffusion (Moyano et al., 2013). The relationship of soil water content with soil CO_2 production is usually a negative-skewed unimodal curve, with the highest production occurring just below field capacity and decreases exponentially above field capacity and below wilting points (Moyano et al., 2013; Sato & Seto, 1999; Schaufler et al., 2010; Zhang et al., 2013). This is because drought slows down microbial activities and retards solute transport, whereas waterlogging reduces O_2 availability and promote the slower anaerobic metabolisms.

Most microbes are more active under higher soil temperature and hence decomposition (producing organic substrates) and respiration (producing CO_2) are also increased. Zhang et al. (2013) found that heterotrophic respiration increases exponentially with soil temperature (8 – 35 ^oC, $r^2 = 0.84$) in maize-wheat fields in northern China. Fang & Moncrieff (2001) also found a similar exponential relationship between soil CO_2 emission and soil temperature (10 – 32^oC, $r^2 = 0.90$), and that soil temperature exerted the dominant control on respiration when soil moisture is neither too dry nor too wet. In most cases, the effect of soil temperature within this range on soil respiration can be fitted by an Arrhenius exponential curve defined by a Q10 coefficient (Fang & Moncrieff, 2001).

Other soil factors that are relatively stable over short-term such as soil pH, texture and bulk density also influence soil CO_2 production. Similar to soil moisture, soil pH also has an optimal range that maximizes microbial activities and hence CO_2 production. The highest respiration rate typically occurs at around neutral pH where enzymatic processes are functioning properly (Thangarajan et al., 2013). Soil texture controls many soil processes that affect organic substrates and O_2 availability (adsorption, retention, diffusion of solutes and air), as well as influencing the extent of microbial and faunal activities. However, soil texture generally has no consistent impact on soil respiration given the soils compared are under similar agricultural management (Bouma & Bryla, 2000; Lohila et al., 2003). A higher bulk density in a given soil texture (indicating compaction) could however decrease soil respiration as it reduces O_2 availability (Jensen et al., 1996).

2.2.2 Biophysical controls of soil methane production

Net production of CH_4 from soil depends on the relative rate of two processes: methanogenesis that produces CH_4 and methanotrophy that consumes CH_4. Methanogenesis reduces CO_2 and oxidized organic C compounds including mainly acetate, which contributes to 70 – 90% of all methanogenesis, as well as other minor substrates such as formate and alcohols, into CH_4. Methanogens use H_2 and again organic C produced from fermentation as electron donors (Le Mer & Roger, 2001). Methanogenesis occurs under strictly anaerobic environment (soil redox potential < -200 mV) as methanogens are obligate anaerobes (Le Mer & Roger, 2001). Abundant labile C in the soil fuel methanogenesis by stimulating the production of CO_2

and other fermentation products (that serve as substrates) as well as decreases the soil redox potential by consuming O_2 and other oxidants. Similarly, a high soil water content reduces soil O_2 availability and hence redox potential, thereby favoring methanogenesis (Hu et al., 2001; Le Mer & Roger, 2001). Therefore, typically only flooded agroecosystems have considerable methanogenesis and CH_4 emission.

Methanotrophy oxidizes CH_4 and other reduced, one-C organic compounds (e.g. methanol) back to CO_2 using O_2 or other oxidants e.g. NO_3^-, analogous to what happen in cellular respiration (Raghoebarsing et al., 2006; Serrano Silva et al., 2014). It is carried out by methanotrophs and ammonia oxidizers occasionally, which can be active in a wider range of soil physicochemical conditions than methanogens (Le Mer & Roger, 2001; Serrano Silva et al., 2014; Stein et al., 2012). Due to the requirement of both CH_4 (produced under anaerobic conditions) and O_2 (available under aerobic conditions) as substrates in methanotrophy, if the soil has an aeration stratification where the soil surface is more aerated and the deeper, more anoxic part of the soil has abundant organic C to sustain fermentation and methanogenesis, methanotrophy would be promoted. This happens in forest floors where the highest rate of soil CH_4 uptake has been recorded but typical crop fields are a much weaker CH_4 sink (Boeckx et al.,1997; Le Mer & Roger, 2001). Moreover, as both CH_4 and O_2 diffusion is needed to sustain CH_4 oxidation, a high soil water content impeding gas movement would suppress methanotrophy (Serrano Silva et al., 2014). Furthermore, an abundance of NH_4^+ in soils (common in croplands under prolonged N fertilization) suppresses methanotrophy because of substrate inhibition of the monooxygenase enzyme in the first step of methanotrophy (Dunfield & Knowles, 1995; Hütsch et al., 1993; Serrano Silva et al., 2014).

Some studies have shown that a higher soil temperature promote methanogenesis more than methanotrophy, possibly because methanogenesis is less thermodynamically-favorable and hence need more energy to overcome the energy barrier (Serrano Silva et al., 2014). However, the trend is not always consistent and the mechanism is still under debate as both methanogenesis and methanotrophy are carried out by unique classes of microbes (rather than a broad spectrum of microbes as in respiration) that may have distinct optimal growth temperatures (Serrano Silva et al., 2014).

Most methanogens and methanotrophs work best close to neutral pH but methanogens are more sensitive and decline sharply under acidic conditions (Le Mer & Roger, 2001; Serrano Silva et al., 2014). Besides, since methanogens and methanotrophs have opposite O_2 requirements, soil factors that reduce O_2 availability e.g. high clay content, high bulk density would promote methanogenesis and hence net CH_4 production (Boeckx et al., 1997; Serrano Silva et al., 2014).

2.2.3 Biophysical controls of soil nitrous oxide production

Heterotrophic denitrification and nitrifier denitrification are the dominant processes responsible for soil N_2O production (Hu et al., 2015; Wrage-Mönnig et al., 2018; Zhu et al., 2013). Heterotrophic denitrification is a ubiquitous microbial pathway in facultative anaerobes that uses oxidized N (e.g. NO_3^- and NO_2^-) as terminal electron acceptors instead of O_2 to generate energy from organic substrates under anaerobic conditions (Wrage et al., 2001). Nitrifier denitrification is carried out by ammonia oxidizers which require NO_2^- as the primary precursor (Wrage-Mönnig et al., 2018). Ammonia oxidizers are generally autotrophic and thus do not require as much organic substrates and contribute significant N_2O emission in low SOC soils (Hu et al., 2015; Hatzenpichler, 2012). Nitrifier denitrification works best under O_2 levels slightly higher than that of heterotrophic denitrification (Figure 2-3). Zhu et al. (2013) found that nitrifier denitrification contributed the most N_2O emission under microaerobic conditions (34 – 66% N_2O under 3 and 0.5% O_2) whereas heterotrophic denitrification contributed 34 – 50% N_2O under microaerobic conditions and was the sole source at 0% O_2 in a fertilized clay loam soil. Other minor pathways such as incomplete ammonia oxidation (Caranto & Lancaster, 2017) and abiotic chemodenitrification with soil mineral reductants e.g. Fe^{2+} (Grabb et al., 2017) also contribute to N_2O production under more aerobic conditions but they will not be discussed here.

The main factors controlling N_2O production are the availability of N and C substrates as well as soil redox potential (optimal between +200 and 300 mV) (Hou et al., 2000; Wlodarczyk et al., 2005). Peak N_2O emissions often occur after N fertilizer is applied, since it increases the NH_4^+, NO_2^- or NO_3^- concentrations in the soil, either directly or through organic N mineralization, which fuel nitrifier denitrification and heterotrophic denitrification (Figure 2-2). High NO_3^- and NO_2^- also suppress N_2O to be used as the terminal electron acceptor, hence preventing its further reduction to N_2 (Wrage-Mönnig et al., 2018; Zhu et al., 2013). Organic C contributes to higher N_2O production as they are electron donors in heterotrophic denitrification and can lower the soil redox potential by consuming O_2 (Thangarajan et al., 2013). However, too much labile C in the soil can favor denitrification to N_2 (Sanchez-Martin et al., 2008).

Soil water content directly controls soil redox potential. The optimal water-filled pore space (WFPS) for N_2O production is generally around 80% (Butterbach-Bahl et al., 2013; Rabot et al., 2015). For instance, Dobbie & Smith (2001) found a 30-fold increase in short-term N_2O emission when WFPS was increased from 60% to 80% in a N-supplemented Gleysol. A very high soil water content, however, lowers the N_2O:N_2 emission ratio as it is a physical barrier that slows down N_2O diffusion substantially. Prolonged waterlogging promotes N_2 emission instead (Heincke & Kaupenjohann, 1999; Wrage et al., 2001). Over a broader temporal sense, the alternate drying and wetting of soils (e.g. alternate rainfall and rainless days) generally enhances the production and release of N_2O. This is due to greater production of NO_2^- and NO_3^- by nitrification during the drier, more aerated periods, which then serve as substrates for denitrification during wet periods (Burger et al., 2005; Dalal et al., 2003; Scholes et al., 1997).

Increasing soil temperature (below 40°C) generally promotes both nitrification and denitrification and hence N_2O production (Castaldi, 2000; Nieder & Benbi, 2008; Schaufler et al, 2010). A higher temperature also increases O_2 consumption and hence reduces soil redox potential and promote denitrification (Smith et al., 2003; Thangarajan et al., 2013). For instance, Dobbie & Smith (2001) found that short-term fertilizer-induced N_2O emission increased by 16-fold from 5 to 12°C and 4-fold from 12 to 18°C in an arable Gleysol, highlighting the possible strong positive effect of global warming on N_2O emission when other factors are not limiting.

Soil pH also has a strong control on N_2O production. In general, a lower pH causes more mineral N to be emitted as N_2O. The microbial mechanism behind is that low pH causes a malfunction of N_2O reductase, hindering further denitrification to N_2 (Wang et al., 2018). Soil physical properties also has a profound effect on N_2O emission. Rochette et al. (2018) found that soil clay content is positively correlated with cumulative N_2O emission in eastern Canada, and organically-fertilized fine-textured soils on average had almost 10 times the emission factor than the coarse-textured counterpart, as fine-textured soil typically has higher OM and moisture content that favor denitrification. Ruser et al. (1998) found that compacted interrow soil (bulk density = 1.56) emitted 6 – 14 times higher annual N_2O emission than uncompacted interrow soil (bulk density = 1.26) and ridge soil (bulk density = 1.05) on potato fields of the same soil type, highlighting the negative effect of compaction on O_2 availability that promotes denitrification.

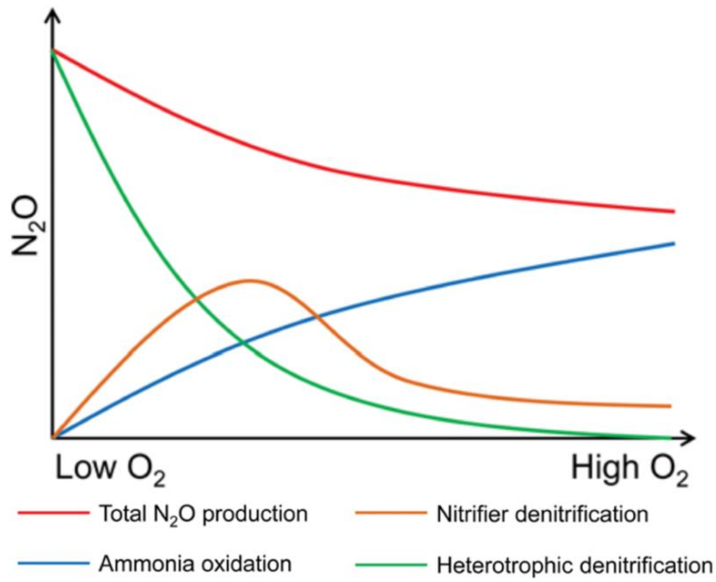

Figure 2-3. Possible changes in total soil N_2O emissions and the relative contributions of ammonia oxidation, nitrifier denitrification and heterotrophic denitrification to N_2O production along the soil O_2 gradient. **Adapted from Hu et al. (2015)**.

2.3 Agricultural management practices affecting soil greenhouse gas emissions

Many agricultural management practices affect soil GHG emissions. In this thesis, our main experimental subject was two corn-soybean fields in eastern Canada that received different organic fertilizers. Hence, the effect of applying different organic fertilizers and the different crops grown in the field on soil GHG emissions were considered. Organic C within organic fertilizers directly contribute to the amount of soil CO_2 produced so how big would an effect be depended on the quantity applied. Organic fertilizers on average had 16% lower N_2O emission (not significant) compared to mineral counterparts in eastern Canada (Rochette et al., 2018), but different fertilizer physicochemical properties can give rise to N_2O emission factor ranging from (0.02%) in compost and paper waste to (1.21%) in slurries and biosolids, a 60-fold difference

(Charles et al., 2017). The crop grown (corn vs. soybean) also exerts a major influence on soil N_2O emissions in particular. In eastern Canada, soybean fields recorded on average 48% lower growing-season N_2O emission than corn fields fertilized with 150 – 200 kg N/ha (Gregorich et al., 2005). Therefore, we expected to see different soil GHG emissions with the application of different organic fertilizers and in different cropping seasons.

2.3.1 The effect of organic fertilizers of varying properties

Organic fertilizers have a wide range of physicochemical properties. They can vary from composted solid, a solid-liquid slurry mixture, to a thin liquid containing mostly mineral nutrients. As such, they provide different amount and forms of C (labile C, polymeric C etc.) and N substrates (NH_4^+, NO_3^-, organic N etc.) that can fuel different reactions leading to the production of GHG. Also, the physical properties of fertilizer (solid, slurry, thin liquid etc.) can alter the soil environment temporarily, thereby favoring different GHG production reactions.

Organic fertilizer contains labile C as well as polymeric C that mineralizes to release C substrates gradually. In general, the higher the amount of labile C in the applied fertilizer, the stronger is the induced CO_2 production, due to the direct respiration of labile C as well as possible positive priming (Martín et al., 2012; Thangarajan et.al, 2013). Aerobic composting generally depletes labile C and the remaining C is more stable compared to those treated anaerobically (Tambone et al., 2010). For instance, Paul & Beauchamp (1989) reported the concentration of soluble C and volatile fatty acids were 7.5 – 17.4 and 3.1 – 14.4 g/kg respectively in anaerobic slurries, whereas that in composted manures were 0.2 – 0.8 and 0 g/kg. Gigliotti & Kaiser (2002) reported that municipal waste compost had about 5 times lower soluble sugar concentration in the hydrophilic fraction of DOC compared to pig slurry. Therefore, compost typically do not induce strong CO_2 spikes (Chodak et al., 2001; Grigatti et al. 2011).

However, composted solids contain much higher total/polymeric C at equivalent N-rate compared to liquid or slurry organic fertilizers as the latter contain most N in mineral forms. The gradual mineralization of the polymeric C can chronically raise CO_2 emission. Grigatti et al. (2011) found that the liquid fraction of biogas slurry produced an intense CO_2 spike that lasted about 1 d, which was not seen in composted slurry solid fraction (applied at equivalent N). However, the latter had 16% higher cumulative emission over 66 d. Simply, since respiration is a ubiquitous process, cumulative CO_2 production should linearly correlate with the amount of organic C applied given enough time, even though short-term production rate varies due to differences in the quality of OM and other factors such as the degree of mixing.

Post-application growing-season CH_4 flux is generally small in annual crop fields in eastern Canada (Gregorich et al., 2005), but we can expect that slurry fertilizers, which usually have high labile C and NH_4^+, can stimulate CH_4 emission within short-term (Ball et al., 2004; Hütsch, 2001; Rochette & Côté, 2000). The high viscosity of slurry lowers O_2 diffusion and high labile C further promote the consumption of O_2 as well as the accumulation of CO_2 and fermentation products (substrates in methanogenesis). Ammonium further inhibit the monooxygenase enzyme for CH_4 oxidation (see Section 2.2.2). For example, Ball et al. (2004) reported that only the application of cattle slurry produced up to 30 mg CH_4-C/m^2/h for 3 days, whereas all other fertilizers tested i.e. composted sludge, dry-pelleted sludge and mineral fertilizers had undetectable CH_4 emission.

Mineral N and labile C are the main substrates in nitrifier and heterotrophic denitrification. Slurries often contain both mineral N and labile C in ample quantities (Charles et al., 2017). Its viscous nature could further hinder O_2 diffusion temporarily, further promoting denitrification (Velthof et al., 2003). Abundant studies have found slurries as the highest N_2O

emitter among all organic fertilizers (Ball et al., 2004; Chantigny et al., 2013; Van Groenigen et al., 2004). A meta-analysis by Charles et al. (2017) reported that animal slurries was in the highest risk group of N_2O emission, with an estimated emission factor of 1.12%.

Waste water that contains high mineral N and minimal organic C are also classified in the high-risk group (raw mean emission factor = 1.15%) in Charles et al. (2017) but there were only 2 studies representing it. Further search into the literature reveals that the effect of these thin liquid fertilizers on N_2O emission is dependent on soil type. High N_2O emissions associated with thin liquid fertilizers typically occurred in SOC-rich clayey soils because such soils are not C-limited and are good at holding liquid whereas in SOC-depleted sandy soils they may be leached easily (Pelster et al., 2012). For instance, Fangueiro et al. (2008) found that the liquid fraction of cattle slurry had an emission factor of 0.50% whereas that of untreated slurry and composted solid fraction were 0.46% and 0.03% respectively in a 5% SOC clay loam soil. In contrast, Meijide et al. (2007) found that in a sandy loam soil with 0.74% SOC, applying the separated liquid fraction of pig slurry had 19% lower N_2O emission than the composted solid fraction.

On the other hand, high N_2O emissions following the application of organic solids (e.g. manure and compost) occur mostly in SOC-depleted sandy soils (Pelster et al., 2012; Rochette et al., 2008; Thangarajan et.al, 2013). Organic solids provide abundant C substrates that SOC-depleted soils lack, and can act as a sponge to improve soil water retention, thereby promoting N_2O production (Pelster et al., 2012). Charles et al. (2017) classified solid manure into the medium risk group with a mean emission factor of 0.35%. Composting further depletes labile C and N substrates, thereby limiting N_2O production from composts (Bernai et al, 1998; Chantigny et al., 2013). For instance, Nicholson et al. (2017) reported no emission (0 kg N_2O-N/ha, lower than unfertilized control) in a winter wheat field from the application of green/food waste compost, whereas food waste liquid digestate produced about 3 kg N_2O-N/ha (~0.5% emission factor). Charles et al. (2017) classified composts into the low-risk group with a mean emission factor of 0.0%, confirming the low N_2O emission typically associated with compost application.

2.3.2 The effect of crops

The choice of crops can affect GHG emissions in several ways. First, the living crops can exert its influence on soil processes by root exudation and nutrient uptake. Second, each cropping season is subjected to the influence of residues left from the previous season. Their mechanisms and combined effect when a field is growing corn and soybean were reviewed here. Methane will not be discussed as many studies found negligible fluxes in eastern Canada and a minimal effect of crops compared to irrigation and fertilizers.

Root exudation is a source of labile C substrates (e.g. sugars and organic acids) plus they can further cause positive priming to increase CO_2 production (Koo et al., 2006). In general, plants that have a larger biomass production (higher photosynthesis rate) and root system produce more root exudates (i.e. corn > soybean) (Kuzyakov et al., 2003). Adviento-Borbe et al. (2007) found a higher growing-season CO_2 emission in corn (3.5 - 4.0 t CO_2-C/ha) than soybean fields (2.8 – 3.2 t CO_2-C/ha) where both were growing corn the previous year. Shen et al. (2018) also found that corn had 14% higher heterotrophic CO_2 emission than soybean in plots all cropped to winter wheat the prior year. In addition, actively-growing roots produce more exudates than old roots so we can expect more CO_2 production induced by root exudation during mid-growing season than very early and late season (Gransee & Wittenmayer, 2000; Rochette & Flanagan, 1997; Sey et al., 2010).

The production of CO_2 is also inevitably affected by the decomposition of residues from the last crop. Corn residues generally have a stronger carryover effect to the next season due to the larger amount and higher persistence (Abail & Whalen, 2018; Johnson et al., 2010). Adviento-Borbe et al. (2007) reported that corn season in continuous corn versus corn following soybean had notable differences in their average CO_2 fluxes (25.4 vs. 19.2 kg CO_2-C/ha/d respectively). Behnke et al. (2018) found that soybean following corn on average had 39% higher growing-season CO_2 emission than soybean in continuous soybean. Overall, continuous corn had more than 2 times the emission than continuous soybean over 4 years. These studies exemplified the importance of residues from the previous season.

Crop N-uptake draws away mineral N that can otherwise be used by nitrifiers and denitrifiers to produce N_2O under favorable soil conditions. Corn has considerably higher crop N demand than soybean, so we can expect corn season may only have substantial N_2O emission early season, as corn N demand increases sharply afterwards (Osterholz et al., 2017). Mahmood et al. (1997) found that the presence of corn plants suppressed denitrification N loss (no differentiation of N_2O and N_2) from 20 days after germination onwards relative to unplanted controls. Soybean on the other hand was reported to emit N_2O from mid- to late season due to the degradation of dead nodules and a significant drop of N demand in reproductive stages (Ciampitti et al., 2008; Inaba et al., 2008; Yang & Cai, 2006).

Root exudates, however, enhance denitrification potential (hence N_2O production) by providing labile C that lowers soil redox potential and fuels heterotrophic denitrification (Cai et al., 2012; Henry et al., 2008; Mahmood et al., 1997; Sey et al., 2010). Thus, the N-uptake and root exudate effects are antagonistic i.e. the combined effect can go either way. Follow up to the above, Mahamood et al. (1997) further demonstrated that the addition of extra NO_3^- to corn-planted soil overshooted the denitrification N loss by more than 3 times compared to the unplanted controls, indicating that whether root exudates can enhance N_2O production depends on whether we applied excess N. Shen et al. (2018) recorded corn having 18% higher growing-season N_2O emission than soybean in the same year where both corn and soybean were fertilized at 240 kg N/ha, highlighting the effect of root exudates when N supply is abundant.

The residue effect on N_2O emission is similar to CO_2. Behnke et al. (2018) found that soybean in continuous soybean had 19% lower emission than soybean in corn-soybean rotation over 4 years. Overall, continuous corn had almost 4 times the N_2O emission than continuous soybean. Similar findings were reported in Drury et al. (2008) and Wagner-Riddle et al. (2007). Johnson et al. (2010) further found that the thaw period (next year early spring) after corn had consistently higher N_2O emission than soybean by up to 3 times, where both corn and soybean were unfertilized. All these highlight the long-lasting and positive effect of corn residue on N_2O emission. However, the effect of soybean residue on N_2O emission may be short-lived so it can be easily missed due to an abrupt end of sampling upon harvest in many studies (Kravchenko et al., 2017; Uchida & Akiyama, 2013). Thus, more studies incorporating frequent measurements after soybean harvest would be beneficial.

2.4 Climate change is a key modifier of biogeochemical processes in the long-term

2.4.1 The effect of changing climate on agroecological responses

Rising atmospheric CO_2 concentration (pCO_2) and air temperature are occurring worldwide. This alters local precipitation distribution and soil temperature in ways that are expected to influence C and N biogeochemical processes in agroecosystems. I will focus on how climate change alters crop production, SOC stock and soil N_2O emission. Crop production is the primary economic goal in any agroecosystem. Higher crop yields also lead to averted land clearance (Burney et al., 2010), it together with long-term SOC changes is an indicator of C exchange between the land and atmosphere. Nitrous oxide is considered the most important GHG in annual crop fields in eastern Canada (Gregorich et al., 2005).

Elevated pCO_2 can enhance crop production by directly increasing photosynthesis for C3 plants (e.g. soybean), as well as indirectly by water conservation via reducing stomatal conductance in C4 plants (e.g. corn) under dry conditions (Kimball et al., 2002; Long et al., 2006). In Free-Air-CO_2-Enrichment experiments, the positive effect of elevated pCO_2 in soybean grain yield is ~14% (+200 ppm CO_2) whereas for corn grain it was negligible (Long et al. 2006). The negligible effect on corn was further confirmed by the historical yield analysis by Lobell & Field (2008) who estimated no pCO_2 effect in Canada over 1961 – 2002. Second, rising temperature can either increase or decrease crop yields by altering soil moisture, as well as how close growing-season temperature is to the optimal growth temperature of a cultivar. A meta-analysis by Matiu et al. (2017) found that high temperature only significantly decreased yield in the driest 5% conditions based on a precipitation-evapotranspiration index. Bootsma et al. (2005) reported that historically, both soybean and corn yield were linearly correlated with corn heat units in Canada, meaning that a warmer and longer growing-season would increase crop yield, given that cultivars that can take advantage of this are available. Third, a higher growing-season precipitation usually benefits crop production (Lobell et al., 2011; Matiu et al., 2017). The precipitation effect is dependent on whether the area has sufficient water supply already. Moreover, a higher precipitation variability within the growing-season may negatively affect yield as indicated by the historical yield analysis in Ontario by Cabas et al. (2010). Combining the effect of pCO_2, temperature and precipitation together, He et al. (2018) projected a yield increase (2071 – 2100 relative to 1971 – 2000) of 17 and 34 % in soybean and 11 and 16% in corn under RCP4.5 and RCP8.5 respectively in southwestern Ontario. Yield increases were also predicted in earlier simulations by Brassard & Singh (2007) and Smith et al. (2013) in Quebec and Ontario. However, the crop models in these studies do not account for damage by pest and disease, which could potentially negate the positive environmental effect of climate change (St-Marseille et al., 2019).

The higher C fixation under elevated pCO_2 represents an extra C input that can increase SOC. The increased C fixed under elevated CO_2 (+200 to 350 ppm) is especially large in belowground production (+23% vs. 14% in aboveground biomass across all crops, Luo et al., 2006) and in particular in soybean it can increase by 30 – 50% (+100 to 400 ppm CO_2, Ainsworth et al. 2002). Despite the large increase in C fixation, Luo et al. (2006) reported only a non-significant 2.81% increase of SOC across all elevated CO_2 crop experiments. No SOC accumulation occurred particularly when the soil is under frequent tillage, and has optimal moisture and excess N that are conducive to rhizospheric respiration (Moran & Jastrow, 2010; Paterson et al., 2008; Peralta & Wander, 2008; Van Groenigen et al., 2014). In addition, rising temperature drives an increase in decomposition of all types of OM, but can also increase net primary production concurrently especially in cold climate (Davidson & Janssens, 2006; Ren et al., 2020; Wiesmeier et al., 2016). Overall, a meta-analysis of soil warming experiments by Yue et al. (2017) found a significant 4.3% decrease of SOC under warming (0.3 – 5^{o}C). Crowther et al. (2016) found that the gain or loss of SOC (0 – 10 cm) under warming is likely dependent on initial SOC stock size, with a larger initial stock more likely to lose SOC because the large absolute loss of SOC from faster decomposition cannot be compensated by C inputs. On the other hand, precipitation appears not to have a strong control on SOC in decadal scale (Doetterl et al., 2015). Although precipitation increases biomass production, it also increases decomposition concurrently. The high year-to-year variability of precipitation plus human interventions (harvesting and irrigation) in croplands further negate the effect of precipitation on SOC. Overall, many have cited that changes in SOC under combined climate change effect remain highly uncertain (Frey et al., 2013; Todd-Brown et al., 2014).

Elevated pCO_2 tends to increase soil N_2O emission by increasing soil moisture due to lower stomatal conductance, plus increased root exudations that lower soil redox potential (Dijkstra et al., 2012; Van Groenigen et al., 2011), despite a concurrent increase in crop N uptake by larger root systems (Kanter et al. 2016). The positive effect is particularly strong in agroecosystems (+38%) where excess soil mineral N is more common (Van Groenigen et al., 2011). Air temperature and precipitation also influence soil redox potential by controlling soil temperature and moisture as reviewed in 2.2.3. Rochette et al. (2018) found that soil N_2O emission (kg N_2O-N/ha) is positively correlated with both mean annual temperature (+0.10 kg N_2O-N/ha for every ^oC increase) and growing-season precipitation (+0.00084 kg N_2O-N/ha for every mm increase) using data collected since 1990 in eastern Canada. Combining the climate change effects, Smith et al. (2013) projected that soil N_2O emission almost doubled (2040 – 2069 relative to 1961 – 1990) across SRES scenarios (A1b, A2, B1) in rotational corn fields in Ontario.

2.4.2 Predicting agroecological responses using process-based biogeochemical models

To be able to quantitatively predict different agroecological responses (changes in crop production, SOC stock, N_2O emission) under future climate change, we must rely on models that are parameterized to reasonably reproduce the results in climate-controlled ecosystem experiments or from long-term observations spanning a period of climate change. Increasingly, ecologists have relied on parameterizing process-based models with experimental data to make predictions of future agroecological outcomes. A process-based model is built by theories and mechanisms described quantitatively that govern the flows of materials in and out of ecosystems (Cuddington et al., 2013). Since these fundamental mechanisms and processes theoretically operate in all ecosystems represented by the model, conceptually speaking, it helps us to make generalization beyond where the original experiments or observations were carried out. Process-based models also contain isolable pieces of information that have management value because we can precisely trace when and which processes (e.g. denitrification) are causing a certain response (e.g. N_2O emission) and what we can do by simulating management practices (e.g. alternative fertilizer type) in order to reinforce or dampen that response (Cuddington et al., 2013; Olander et al., 2011).

DayCent, a process-based model simulating C and N flows among the atmosphere, plants, and soil (daily time-step, 14 soil layers), is widely used to simulate crop production and SOC dynamics. It is downscaled from Century (monthly time-step, 9 soil layers), which improved the fine-scale estimation of soil conditions (e.g. temperature and moisture) and nutrient availability. As a result, DayCent has improved performance over Century (Bista et al., 2016; Congreves et al., 2015) and has been verified repeatedly with long-term SOC and crop yield data in Canadian agroecosystems (Chang et al. 2013; Grant et al., 2016; Guest et al., 2017). In addition, simulating the highly volatile GHG fluxes under a higher spatiotemporal resolution is beneficial in terms of matching the actual unit of validation measurements, and for identifying hotspots and hot moments of GHG emissions and their causes and remedies (Frolking et al., 1998). Having such information enables us to plan and screen agricultural management practices that can mitigate GHG while maintaining other desirable outcomes e.g. crop yield and SOC accrual in a specific context. For instance, De Gryze et al. (2011) used DayCent to simulate the GHG mitigation potential of a range of management practices e.g. conservation tillage, manure application and reduced N-rate (and a combination of them), in order to identify the best practices both at the regional and site level in California.

In addition, DayCent is generally considered reliable for simulating long-term agroecological responses under climate change as it was originally developed from long-term experiments in the US Great Plains and was further parameterized with CO_2-fertilization experiments (Del Grosso et al. 2005; Lee et al., 2017; Stehfest et al. 2007). Moreover, DayCent (and process-based models in general) is structured to contain easily identifiable characteristics (e.g. C:N ratio of organic fertilizers) and effect parameters (e.g. influence of WFPS on denitrification) in each model component that we can adjust to adapt the model in a specific context. This allows the application of parameterized DayCent in a wide range of situations, including in this thesis project.

2.5 Conclusions and future directions

Organic fertilizer is a source of organic C and plant nutrients that can be applied to agroecosystems. Thus, organic fertilizer is expected to support crop production and contribute to SOC accrual. However, organic fertilizers vary in their physicochemical properties (amount of water vs. dry solids, quantity and chemical form of C and N) that can lead to different levels of GHG production. Organic fertilizers that supply more organic C and mineral N (NH_4^+, NO_2^- and NO_3^-) are expected to stimulate soil CO_2 and N_2O production respectively.

This thesis work is based on a two-year biennially-fertilized corn-soybean field study that examines the effect of organic fertilizers with distinct physicochemical properties, namely compost, LysteGro and digestate, on crop yield, SOC stock and soil GHG fluxes. We evaluated both the short (immediate response) and long-term future effects (2018 – 2070) with the use of field data and the DayCent model. Specifically, I hypothesize that:

1. **Short term crop yield:** First, organic fertilizers rich in mineral nutrients (digestate and LysteGro) will promote corn yield in the first season because mineral nutrients are more plant-available in the short-term. Second, residual fertility from the OM-rich fertilizer (compost) will promote soybean yield in the second season.

   Long-term crop yield: The above yield ranking will remain in the near-term. Organic fertilizers that supply more OM (compost) will gradually increase crop production farther in time because of the built up of long-term fertility. In addition, under a more intense climate change, the crop yield of all organic fertilizer treatments will increase due to elevated pCO_2 promoting C fixation.

2. **Short-term SOC changes:** Organic fertilizers containing more organic C (compost > LysteGro > digestate) will accrue the most SOC since organic C is a direct source of SOC. However, the difference will be small since SOC tends to accrue slowly.

   Long-term SOC changes: Compost will be the best SOC accumulator as aforementioned. Under a more intense climate change, the SOC stock will decrease particularly for compost, since OM decomposition is faster.

3. **CO_2 emission**: Organic fertilizers containing more organic C (compost > LysteGro > digestate) will result in the highest cumulative soil CO_2 emission in two seasons, due

to the respiration of organic C. Cumulative CO_2 emission is not evaluated for long-term because land-atmosphere C exchange is determined from changes in SOC.

4. **Short-term N_2O emission:** Slurry organic fertilizers containing more mineral N and labile C (LysteGro) will produce the highest cumulative soil N_2O emission particularly in the first corn season. It is because mineral N and labile C are substrates required in nitrifier and heterotrophic denitrification and slurry texture reduces O_2 availability that further promotes denitrification.
5. **Long-term N_2O emission:** Organic fertilizers that accrue more SOC (compost) will become the biggest N_2O emitter farther in time, because SOC is a reserve of C and N substrates for denitrification. Under a more intense climate change, N_2O emission associated with all fertilizers will increase due to greater soil denitrification potential under elevated pCO_2 and temperature.

Combining these investigations together, we aim to use the findings to draw management recommendations concerning the optimal use of organic fertilizers with different properties, from near term to the far future.

Chapter 2 references

Abail, Z., & Whalen, J. K. (2018). Corn residue inputs influence earthworm population dynamics in a no-till corn-soybean rotation. *Applied Soil Ecology*, 127, 120-128.

Abramoff, R., Xu, X., Hartman, M., O'Brien, S., Feng, W., Davidson, E., ... & Mayes, M. A. (2018). The Millennial model: in search of measurable pools and transformations for modeling soil carbon in the new century. *Biogeochemistry*, 137(1-2), 51-71.

Adviento-Borbe, M. A. A., Haddix, M. L., Binder, D. L., Walters, D. T., & Dobermann, A. (2007). Soil greenhouse gas fluxes and global warming potential in four high-yielding maize systems. *Global Change Biology*, 13(9), 1972-1988.

Ainsworth, E. A., Davey, P. A., Bernacchi, C. J., Dermody, O. C., Heaton, E. A., Moore, D. J., ... & Curtis, P. S. (2002). A meta-analysis of elevated [CO_2] effects on soybean (Glycine max) physiology, growth and yield. *Global Change Biology*, 8(8), 695-709.

Ball, B. C., McTaggart, I. P., & Scott, A. (2004). Mitigation of greenhouse gas emissions from soil under silage production by use of organic manures or slow-release fertilizer. *Soil Use and Management*, 20(3), 287-295.

Barré, P., Quénéa, K., Vidal, A., Cécillon, L., Christensen, B. T., Kätterer, T., ... & Chenu, C. (2018). Microbial and plant-derived compounds both contribute to persistent soil organic carbon in temperate soils. *Biogeochemistry*, 140(1), 81-92.

Behnke, G. D., Zuber, S. M., Pittelkow, C. M., Nafziger, E. D., & Villamil, M. B. (2018). Long-term crop rotation and tillage effects on soil greenhouse gas emissions and crop production in Illinois, USA. *Agriculture, Ecosystems & Environment*, 261, 62-70.

Berhe, A. A., Harden, J. W., Torn, M. S., Kleber, M., Burton, S. D., & Harte, J. (2012). Persistence of soil organic matter in eroding versus depositional landform positions. Journal of Geophysical Research: *Biogeosciences*, 117(G2).

Bernai, M. P., Paredes, C., Sanchez-Monedero, M. A., & Cegarra, J. (1998). Maturity and stability parameters of composts prepared with a wide range of organic wastes. *Bioresource Technology*, 63(1), 91-99.

Bista, P., Machado, S., Ghimire, R., Del Grosso, S. J., & Reyes-Fox, M. (2016). Simulating soil organic carbon in a wheat–fallow system using the Daycent model. *Agronomy Journal*, 108(6), 2554-2565.

Blagodatskaya, E., Zheng, X., Blagodatsky, S., Wiegl, R., Dannenmann, M., & Butterbach-Bahl, K. (2014). Oxygen and substrate availability interactively control the temperature sensitivity of CO_2 and N_2O emission from soil. *Biology and Fertility of Soils*, 50(5), 775-783.

Boeckx, P., Van Cleemput, O., & Villaralvo, I. (1997). Methane oxidation in soils with different textures and land use. *Nutrient Cycling in Agroecosystems*, 49(1-3), 91-95.

Bootsma, A., Gameda, S., & McKenney, D. W. (2005). Potential impacts of climate change on corn, soybeans and barley yields in Atlantic Canada. *Canadian Journal of Soil Science*, 85(2), 345-357.

Bouma, T. J., & Bryla, D. R. (2000). On the assessment of root and soil respiration for soils of different textures: interactions with soil moisture contents and soil CO_2 concentrations. *Plant and Soil*, 227(1-2), 215-221.

Brassard, J. P., & Singh, B. (2007). Effects of climate change and CO_2 increase on potential agricultural production in Southern Québec, Canada. *Climate Research*, 34(2), 105-117.

Burger, M., Jackson, L. E., Lundquist, E. J., Louie, D. T., Miller, R. L., Rolston, D. E., & Scow, K. M. (2005). Microbial responses and nitrous oxide emissions during wetting and drying of organically and conventionally managed soil under tomatoes. *Biology and Fertility of Soils*, 42(2), 109-118.

Burney, J. A., Davis, S. J., & Lobell, D. B. (2010). Greenhouse gas mitigation by agricultural intensification. *Proceedings of the National Academy of Sciences*, 107(26), 12052-12057.

Butterbach-Bahl, K., Baggs, E. M., Dannenmann, M., Kiese, R., & Zechmeister-Boltenstern, S. (2013). Nitrous oxide emissions from soils: how well do we understand the processes and their controls?. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 368(1621), 20130122.

Cabas, J., Weersink, A., & Olale, E. (2010). Crop yield response to economic, site and climatic variables. *Climatic Change*, 101(3-4), 599-616.

Cai, Y., Ding, W., & Luo, J. (2012). Spatial variation of nitrous oxide emission between interrow soil and interrow plus row soil in a long-term maize cultivated sandy loam soil. *Geoderma*, 181, 2-10.

Caranto, J. D., & Lancaster, K. M. (2017). Nitric oxide is an obligate bacterial nitrification intermediate produced by hydroxylamine oxidoreductase. *Proceedings of the National Academy of Sciences*, 114(31), 8217-8222.

Castaldi, S. (2000). Responses of nitrous oxide, dinitrogen and carbon dioxide production and oxygen consumption to temperature in forest and agricultural light-textured soils determined by model experiment. *Biology and Fertility of Soils*, 32(1), 67-72.

Chang, K. H., Warland, J., Voroney, P., Bartlett, P., & Wagner-Riddle, C. (2013). Using DayCENT to simulate carbon dynamics in conventional and no-till agriculture. *Soil Science Society of America Journal*, 77(3), 941-950.

Chantigny, M. H., Pelster, D. E., Perron, M. H., Rochette, P., Angers, D. A., Parent, L. É., ... & Ziadi, N. (2013). Nitrous oxide emissions from clayey soils amended with paper sludges and biosolids of separated pig slurry. *Journal of Environmental Quality*, 42(1), 30-39.

Charles, A., Rochette, P., Whalen, J. K., Angers, D. A., Chantigny, M. H., & Bertrand, N. (2017). Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. *Agriculture, Ecosystems & Environment*, 236, 88-98.

Chodak, M., Borken, W., Ludwig, B., & Beese, F. (2001). Effect of temperature on the mineralization of C and N of fresh and mature compost in sandy material. *Journal of Plant Nutrition and Soil Science*, 164(3), 289-294.

Ciampitti, I. A., Ciarlo, E. A., & Conti, M. E. (2008). Nitrous oxide emissions from soil during soybean [(Glycine max (L.) Merrill] crop phenological stages and stubbles decomposition period. *Biology and Fertility of Soils*, 44(4), 581-588.

Congreves, K. A., Grant, B. B., Campbell, C. A., Smith, W. N., VandenBygaart, A. J., Kröbel, R., ... & Desjardins, R. L. (2015). Measuring and modeling the long-term impact of crop management on soil carbon sequestration in the semiarid Canadian prairies. *Agronomy Journal*, 107(3), 1141-1154.

Crowther, T. W., Todd-Brown, K. E., Rowe, C. W., Wieder, W. R., Carey, J. C., Machmuller, M. B., ... & Blair, J. M. (2016). Quantifying global soil carbon losses in response to warming. *Nature*, 540(7631), 104-108.

Cuddington, K., Fortin, M. J., Gerber, L. R., Hastings, A., Liebhold, A., O'connor, M., & Ray, C. (2013). Process-based models are required to manage ecological systems in a changing world. *Ecosphere*, 4(2), 1-12.

Dalal, R. C., Wang, W., Robertson, G. P., & Parton, W. J. (2003). Nitrous oxide emission from Australian agricultural lands and mitigation options: a review. *Soil Research*, 41(2), 165-195.

Davidson, E. A., & Janssens, I. A. (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature*, 440(7081), 165-173.

De Gryze, S., Lee, J., Ogle, S., Paustian, K., & Six, J. (2011). Assessing the potential for greenhouse gas mitigation in intensively managed annual cropping systems at the regional scale. *Agriculture, Ecosystems & Environment*, 144(1), 150-158.

Del Grosso, S. J., Mosier, A. R., Parton, W. J., & Ojima, D. S. (2005). DAYCENT model analysis of past and contemporary soil N_2O and net greenhouse gas flux for major crops in the USA. *Soil and Tillage Research*, 83(1), 9-24.

Diacono, M., & Montemurro, F. (2011). Long-term effects of organic amendments on soil fertility. In *Sustainable Agriculture Volume 2* (pp. 761-786). Springer, Dordrecht.

Dijkstra, F. A., Prior, S. A., Runion, G. B., Torbert, H. A., Tian, H., Lu, C., & Venterea, R. T. (2012). Effects of elevated carbon dioxide and increased temperature on methane and nitrous oxide fluxes: evidence from field experiments. *Frontiers in Ecology and the Environment*, 10(10), 520-527.

Dobbie, K. E., & Smith, K. A. (2001). The effects of temperature, water-filled pore space and land use on N_2O emissions from an imperfectly drained gleysol. *European Journal of Soil Science*, 52(4), 667-673.

Doetterl, S., Stevens, A., Six, J., Merckx, R., Van Oost, K., Pinto, M. C., ... & Boeckx, P. (2015). Soil carbon storage controlled by interactions between geochemistry and climate. *Nature Geoscience*, 8(10), 780-783.

Drury, C. F., Yang, X. M., Reynolds, W. D., & McLaughlin, N. B. (2008). Nitrous oxide and carbon dioxide emissions from monoculture and rotational cropping of corn, soybean and winter wheat. *Canadian Journal of Soil Science*, 88(2), 163-174.

Dunfield, P., & Knowles, R. (1995). Kinetics of inhibition of methane oxidation by nitrate, nitrite, and ammonium in a humisol. Appl. Environ. Microbiol., 61(8), 3129-3135.

Fang, C., & Moncrieff, J. B. (2001). The dependence of soil CO_2 efflux on temperature. *Soil Biology and Biochemistry*, 33(2), 155-165.

Fangueiro, D., Senbayran, M., Trindade, H., & Chadwick, D. (2008). Cattle slurry treatment by screw press separation and chemically enhanced settling: effect on greenhouse gas emissions after land spreading and grass yield. *Bioresource Technology*, 99(15), 7132-7142.

Frey, S. D., Lee, J., Melillo, J. M., & Six, J. (2013). The temperature response of soil microbial efficiency and its feedback to climate. *Nature Climate Change*, 3(4), 395-398.

Frolking, S. E., Mosier, A. R., Ojima, D. S., Li, C., Parton, W. J., Potter, C. S., ... & Flessa, H. (1998). Comparison of N 2 O emissions from soils at three temperate agricultural sites:

simulations of year-round measurements by four models. *Nutrient Cycling in Agroecosystems*, 52(2-3), 77-105.

Gigliotti, G., Kaiser, K., Guggenberger, G., & Haumaier, L. (2002). Differences in the chemical composition of dissolved organic matter from waste material of different sources. *Biology and Fertility of Soils*, 36(5), 321-329.

Grabb, K. C., Buchwald, C., Hansel, C. M., & Wankel, S. D. (2017). A dual nitrite isotopic investigation of chemodenitrification by mineral-associated Fe (II) and its production of nitrous oxide. *Geochimica et Cosmochimica Acta*, 196, 388-402.

Gransee, A., & Wittenmayer, L. (2000). Qualitative and quantitative analysis of water-soluble root exudates in relation to plant species and development. *Journal of plant nutrition and soil science*, 163(4), 381-385.

Grant, B. B., Smith, W. N., Campbell, C. A., Desjardins, R. L., Lemke, R. L., Kröbel, R., ... & Lafond, G. P. (2016). Comparison of DayCent and DNDC models: case studies using data from long-term experiments on the Canadian prairies. *Synthesis and modeling of greenhouse gas emissions and carbon storage in agricultural and forest systems to guide mitigation and adaptation*, 6, 21-57.

Gregorich, E. G., Greer, K. J., Anderson, D. W., & Liang, B. C. (1998). Carbon distribution and losses: erosion and deposition effects. *Soil and Tillage Research*, 47(3-4), 291-302.

Gregorich, E. G., Rochette, P., VandenBygaart, A. J., & Angers, D. A. (2005). Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. *Soil and Tillage Research*, 83(1), 53-72.

Grigatti, M., Di Girolamo, G., Chincarini, R., Ciavatta, C., & Barbanti, L. (2011). Potential nitrogen mineralization, plant utilization efficiency and soil CO_2 emissions following the addition of anaerobic digested slurries. *biomass and bioenergy*, 35(11), 4619-4629.

Guest, G., Kröbel, R., Grant, B., Smith, W., Sansoulet, J., Pattey, E., ... & Tremblay, G. (2017). Model comparison of soil processes in eastern Canada using DayCent, DNDC and STICS. *Nutrient Cycling in Agroecosystems*, 109(3), 211-232.

Hatzenpichler, R. (2012). Diversity, physiology, and niche differentiation of ammonia-oxidizing archaea. *Applied and environmental microbiology*, 78(21), 7501-7510.

He, W., Yang, J. Y., Drury, C. F., Smith, W. N., Grant, B. B., He, P., ... & Hoogenboom, G. (2018). Estimating the impacts of climate change on crop yields and N_2O emissions for conventional and no-tillage in Southwestern Ontario, Canada. *Agricultural Systems*, 159, 187-198.

Heincke, M., & Kaupenjohann, M. (1999). Effects of soil solution on the dynamics of N_2O emissions: a review. *Nutrient Cycling in Agroecosystems*, 55(2), 133-157.

Henry, S., Texier, S., Hallet, S., Bru, D., Dambreville, C., Chèneby, D., ... & Philippot, L. (2008). Disentangling the rhizosphere effect on nitrate reducers and denitrifiers: insight into the role of root exudates. *Environmental Microbiology*, 10(11), 3082-3092.

Hou, A. X., Chen, G. X., Wang, Z. P., Van Cleemput, O., & Patrick, W. H. (2000). Methane and nitrous oxide emissions from a rice field in relation to soil redox and microbiological processes. *Soil Science Society of America Journal*, 64(6), 2180-2186.

Hu, H. W., Chen, D., & He, J. Z. (2015). Microbial regulation of terrestrial nitrous oxide formation: understanding the biological pathways for prediction of emission rates. *FEMS microbiology reviews*, 39(5), 729-749.

Hu, R., Kusa, K., & Hatano, R. (2001). Soil respiration and methane flux in adjacent forest, grassland, and cornfield soils in Hokkaido, Japan. *Soil Science and Plant Nutrition*, 47(3), 621-627.

Hütsch, B. W. (2001). Methane oxidation in non-flooded soils as affected by crop production. *European Journal of Agronomy*, 14(4), 237-260.

Hütsch, B. W., Webster, C. P., & Powlson, D. S. (1993). Long-term effects of nitrogen fertilization on methane oxidation in soil of the Broadbalk wheat experiment. *Soil Biology and Biochemistry*, 25(10), 1307-1315.

Inaba, S., Tanabe, K., Eda, S., Ikeda, S., Higashitani, A., Mitsui, H., & Minamisawa, K. (2008). Nitrous oxide emission and microbial community in the rhizosphere of nodulated soybeans during the late growth period. *Microbes and environments*, 0812080030-0812080030.

Jensen, E. S., & Hauggaard-Nielsen, H. (2003). How can increased use of biological N 2 fixation in agriculture benefit the environment?. Plant and Soil, 252(1), 177-186.

Jensen, L. S., McQueen, D. J., & Shepherd, T. G. (1996). Effects of soil compaction on N-mineralization and microbial-C and-NI Field measurements. *Soil and Tillage Research*, 38(3-4), 175-188.

Johnson, J. M., Archer, D., & Barbour, N. (2010). Greenhouse gas emission from contrasting management scenarios in the northern Corn Belt. *Soil Science Society of America Journal*, 74(2), 396-406.

Kallenbach, C. M., Frey, S. D., & Grandy, A. S. (2016). Direct evidence for microbial-derived soil organic matter formation and its ecophysiological controls. *Nature communications*, 7, 13630.

Kanter, D. R., Zhang, X., Mauzerall, D. L., Malyshev, S., & Shevliakova, E. (2016). The importance of climate change and nitrogen use efficiency for future nitrous oxide emissions from agriculture. Environmental Research Letters, 11(9), 094003.

Ketcheson, J. W., & Webber, L. R. (1978). Effects of soil erosion on yield of corn. *Canadian Journal of Soil Science*, 58(4), 459-463.

Kimball, B. A., Kobayashi, K., & Bindi, M. (2002). Responses of agricultural crops to free-air CO_2 enrichment. In *Advances in agronomy* (Vol. 77, pp. 293-368). Academic Press.

Kindler, R., Siemens, J. A. N., Kaiser, K., Walmsley, D. C., Bernhofer, C., Buchmann, N., ... & Heim, A. (2011). Dissolved carbon leaching from soil is a crucial component of the net ecosystem carbon balance. *Global Change Biology*, 17(2), 1167-1185.

Koo, B. J., Chang, A. C., Crowley, D. E., & Page, A. L. (2006). Characterization of Organic Acids Recovered from Rhizosphere of Corn Grown on Biosolids-Treated Medium. *Communications in Soil Science and Plant Analysis*, 37(05-06), 871-887.

Kravchenko, A. N., Toosi, E. R., Guber, A. K., Ostrom, N. E., Yu, J., Azeem, K., ... & Robertson, G. P. (2017). Hotspots of soil N_2O emission enhanced through water absorption by plant residue. *Nature Geoscience*, *10*(7), 496-500.

Kuzyakov, Y., Raskatov, A., & Kaupenjohann, M. (2003). Turnover and distribution of root exudates of Zea mays. *Plant and Soil*, 254(2), 317-327.

Le Mer, J., & Roger, P. (2001). Production, oxidation, emission and consumption of methane by soils: a review. *European journal of soil biology*, 37(1), 25-50.

Lee, J., De Gryze, S., & Six, J. (2011). Effect of climate change on field crop production in California's Central Valley. *Climatic Change*, 109(1), 335-353.

Lehmann, J., & Kleber, M. (2015). The contentious nature of soil organic matter. *Nature*, 528(7580), 60.

Lobell, D. B., & Field, C. B. (2008). Estimation of the carbon dioxide (CO_2) fertilization effect using growth rate anomalies of CO_2 and crop yields since 1961. *Global Change Biology*, 14(1), 39-45.

Lobell, D. B., Schlenker, W., & Costa-Roberts, J. (2011). Climate trends and global crop production since 1980. *Science*, 333(6042), 616-620.

Lohila, A., Aurela, M., Regina, K., & Laurila, T. (2003). Soil and total ecosystem respiration in agricultural fields: effect of soil and crop type. *Plant and Soil*, 251(2), 303-317.

Long, S. P., Ainsworth, E. A., Leakey, A. D., Nösberger, J., & Ort, D. R. (2006). Food for thought: lower-than-expected crop yield stimulation with rising CO_2 concentrations. *Science*, 312(5782), 1918-1921.

Luo, Y., Hui, D., & Zhang, D. (2006). Elevated CO_2 stimulates net accumulations of carbon and nitrogen in land ecosystems: A meta-analysis. *Ecology*, 87(1), 53-63.

Mahmood, T., Ali, R., Malik, K. A., & Shamsi, S. R. A. (1997). Denitrification with and without maize plants (Zea mays L.) under irrigated field conditions. *Biology and fertility of soils*, 24(3), 323-328.

Martín, J. V., De Imperial, R. M., Calvo, R., Garcia, M. C., Leon-Cófreces, C., & Delgado, M. M. (2012). Carbon mineralisation kinetics of poultry manure in two soils. *Soil Research*, 50(3), 222-228.

Matiu, M., Ankerst, D. P., & Menzel, A. (2017). Interactions between temperature and drought in global and regional crop yield variability during 1961-2014. *PloS one*, 12(5), e0178339.

Meijide, A., Díez, J. A., Sánchez-Martín, L., López-Fernández, S., & Vallejo, A. (2007). Nitrogen oxide emissions from an irrigated maize crop amended with treated pig slurries and composts in a Mediterranean climate. *Agriculture, ecosystems & environment*, 121(4), 383-394.

Mengel, K. (1996). Turnover of organic nitrogen in soils and its availability to crops. *Plant and Soil*, 181(1), 83-93.

Miltner, A., Bombach, P., Schmidt-Brücken, B., & Kästner, M. (2012). SOM genesis: microbial biomass as a significant source. *Biogeochemistry*, 111(1-3), 41-55.

Moran, K. K., & Jastrow, J. D. (2010). Elevated carbon dioxide does not offset loss of soil carbon from a corn–soybean agroecosystem. *Environmental Pollution*, 158(4), 1088-1094.

Moyano, F. E., Manzoni, S., & Chenu, C. (2013). Responses of soil heterotrophic respiration to moisture availability: An exploration of processes and models. *Soil Biology and Biochemistry*, 59, 72-85.

Neff, J. C., & Asner, G. P. (2001). Dissolved organic carbon in terrestrial ecosystems: synthesis and a model. *Ecosystems*, 4(1), 29-48.

Nicholson, F., Bhogal, A., Cardenas, L., Chadwick, D., Misselbrook, T., Rollett, A., ... & Williams, J. (2017). Nitrogen losses to the environment following food-based digestate and compost applications to agricultural land. *Environmental Pollution*, 228, 504-516.

Nieder, R., & Benbi, D. K. (2008). Carbon and nitrogen in the terrestrial environment. *Springer Science & Business Media*.

Olander, L. P., Haugen-Kozyra, K., Del Grosso, S., Izaurralde, C., Malin, D., Paustian, K., & Salas, W. (2011). Using biogeochemical process models to quantify greenhouse gas mitigation from agricultural management projects. Nicholas Institute for Environmental Policy Solutions Report NIR, 11-03.

Osterholz, W. R., Rinot, O., Liebman, M., & Castellano, M. J. (2017). Can mineralization of soil organic nitrogen meet maize nitrogen demand?. *Plant and Soil*, 415(1-2), 73-84.

IPCC (2014). Climate Change 2014: Synthesis Report. In *Proceedings of the Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, eds., R. K. Pachauri, and L. A. Meyer. Geneva, Switzerland.

Paterson, E., & Sim, A. (2013). Soil-specific response functions of organic matter mineralization to the availability of labile carbon. *Global Change Biology*, 19(5), 1562-1571.

Paterson, E., Thornton, B., Midwood, A. J., Osborne, S. M., Sim, A., & Millard, P. (2008). Atmospheric CO_2 enrichment and nutrient additions to planted soil increase mineralisation of soil organic matter, but do not alter microbial utilisation of plant-and soil C-sources. *Soil Biology and Biochemistry*, 40(9), 2434-2440.

Paul, E. A., Harris, D., Collins, H. P., Schulthess, U., & Robertson, G. P. (1999). Evolution of CO_2 and soil carbon dynamics in biologically managed, row-crop agroecosystems. *Applied Soil Ecology*, 11(1), 53-65.

Paul, J. W., & Beauchamp, E. G. (1989). Effect of carbon constituents in manure on denitrification in soil. *Canadian Journal of Soil Science*, 69(1), 49-61.

Paul, J. W., & Beauchamp, E. G. (1993). Nitrogen availability for corn in soils amended with urea, cattle slurry, and solid and composted manures. *Canadian Journal of soil science*, *73*(2), 253-266.

Pelster, D. E., Chantigny, M. H., Rochette, P., Angers, D. A., Rieux, C., & Vanasse, A. (2012). Nitrous oxide emissions respond differently to mineral and organic nitrogen sources in contrasting soil types. *Journal of Environmental Quality*, 41(2), 427-435.

Peralta, A. L., & Wander, M. M. (2008). Soil organic matter dynamics under soybean exposed to elevated [CO_2]. *Plant and Soil,* 303(1-2), 69-81.

Poulton, P., Johnston, J., Macdonald, A., White, R., & Powlson, D. (2018). Major limitations to achieving "4 per 1000" increases in soil organic carbon stock in temperate regions: Evidence from long-term experiments at Rothamsted Research, United Kingdom. *Global Change Biology*, 24(6), 2563-2584.

Rabot, E., Cousin, I., & Hénault, C. (2015). A modeling approach of the relationship between nitrous oxide fluxes from soils and the water-filled pore space. *Biogeochemistry*, 122(2-3), 395-408.

Raghoebarsing, A. A., Pol, A., Van de Pas-Schoonen, K. T., Smolders, A. J., Ettwig, K. F., Rijpstra, W. I. C., ... & Strous, M. (2006). A microbial consortium couples anaerobic methane oxidation to denitrification. *Nature*, 440(7086), 918-921.

Ren, W., Banger, K., Tao, B., Yang, J., Huang, Y., & Tian, H. (2020). Global pattern and change of cropland soil organic carbon during 1901-2010: Roles of climate, atmospheric chemistry, land use and management. *Geography and Sustainability*.

Rochette, P., & Côté, D. (2000). CH4 fluxes and soil CH4 concentration following application of pig slurry for the 19th consecutive year. *Canadian Journal of Soil Science*, 80(2), 387-390.

Rochette, P., & Flanagan, L. B. (1997). Quantifying rhizosphere respiration in a corn crop under field conditions. *Soil Science Society of America Journal*, 61(2), 466-474.

Rochette, P., Angers, D. A., Chantigny, M. H., Gagnon, B., & Bertrand, N. (2008). N_2O fluxes in soils of contrasting textures fertilized with liquid and solid dairy cattle manures. *Canadian Journal of Soil Science,* 88(2), 175-187.

Rochette, P., Liang, C., Pelster, D., Bergeron, O., Lemke, R., Kroebel, R., ... & Flemming, C. (2018). Soil nitrous oxide emissions from agricultural soils in Canada: Exploring relationships with soil, crop and climatic variables. *Agriculture, Ecosystems & Environment*, 254, 69-81.

Ruser, R., Schilling, R., Steindl, H., Flessa, H., & Beese, F. (1998). Soil compaction and fertilization effects on nitrous oxide and methane fluxes in potato fields. *Soil Science Society of America Journal*, 62(6), 1587-1595.

Sanchez-Martin, L., Vallejo, A., Dick, J., & Skiba, U. M. (2008). The influence of soluble carbon and fertilizer nitrogen on nitric oxide and nitrous oxide emissions from two contrasting agricultural soils. *Soil Biology and Biochemistry*, 40(1), 142-151.

Sato, A., & Seto, M. (1999). Relationship between rate of carbon dioxide evolution, microbial biomass carbon, and amount of dissolved organic carbon as affected by temperature and water content of a forest and an arable soil. *Communications in Soil Science and Plant Analysis*, 30(19-20), 2593-2605.

Schaufler, G., Kitzler, B., Schindlbacher, A., Skiba, U., Sutton. M.A., & Zechmeister-Boltenstern, S. (2010). Greenhouse gas emissions from European soils under different land use: effects of soil moisture and temperature. *European Journal of Soil Science*, 61, 683-696

Scholes, M. C., Martin, R., Scholes, R. J., Parsons, D., & Winstead, E. (1997). NO and N_2O emissions from savanna soils following the first simulated rains of the season. *Nutrient Cycling in Agroecosystems*, 48(1-2), 115-122.

Schulten, H. R., & Schnitzer, M. (1997). The chemistry of soil organic nitrogen: a review. *Biology and Fertility of Soils*, 26(1), 1-15.

Serrano-Silva, N., Sarria-Guzmán, Y., Dendooven, L., & Luna-Guido, M. (2014). Methanogenesis and methanotrophy in soil: a review. *Pedosphere*, 24(3), 291-307.

Sey, B. K., Manceur, A. M., Whalen, J. K., Gregorich, E. G., & Rochette, P. (2010). Root-derived respiration and nitrous oxide production as affected by crop phenology and nitrogen fertilization. *Plant and Soil*, 326(1-2), 369-379.

Shen, Y., Sui, P., Huang, J., Wang, D., Whalen, J. K., & Chen, Y. (2018). Greenhouse gas emissions from soil under maize–soybean intercrop in the North China Plain. *Nutrient Cycling in Agroecosystems*, 110(3), 451-465.

Sierra, C., Malghani, S., & Loescher, H. W. (2017). Interactions among temperature, moisture, and oxygen concentrations in controlling decomposition rates in a boreal forest soil. *Biogeosciences*, 14(3), 703-710.

Sinsabaugh, R. L., Turner, B. L., Talbot, J. M., Waring, B. G., Powers, J. S., Kuske, C. R., ... & Follstad Shah, J. J. (2016). Stoichiometry of microbial carbon use efficiency in soils. *Ecological Monographs*, 86(2), 172-189.

Smith, K. A., Ball, T., Conen, F., Dobbie, K. E., Massheder, J., & Rey, A. (2003). Exchange of greenhouse gases between soil and atmosphere: interactions of soil physical factors and biological processes. *European Journal of Soil Science*, 54(4), 779-791.

Smith, W. N., Grant, B. B., Desjardins, R. L., Kroebel, R., Li, C., Qian, B., ... & Drury, C. F. (2013). Assessing the effects of climate change on crop production and GHG emissions in Canada. *Agriculture, Ecosystems & Environment*, 179, 139-150.

Stehfest, E., Heistermann, M., Priess, J. A., Ojima, D. S., & Alcamo, J. (2007). Simulation of global crop production with the ecosystem model DayCent. *Ecological Modelling*, 209(2-4), 203-219.

Stein, L. Y., Roy, R., & Dunfield, P. F. (2012). Aerobic methanotrophy and nitrification: Processes and connections. *eLS*.

St-Marseille, A. F. G., Bourgeois, G., Brodeur, J., & Mimee, B. (2019). Simulating the impacts of climate change on soybean cyst nematode and the distribution of soybean. *Agricultural and Forest Meteorology*, 264, 178-187.

Stockmann, U., Adams, M. A., Crawford, J. W., Field, D. J., Henakaarchchi, N., Jenkins, M., ... & Wheeler, I. (2013). The knowns, known unknowns and unknowns of sequestration of soil organic carbon. *Agriculture, Ecosystems & Environment*, 164, 80-99.

Tambone, F., Scaglia, B., D'Imporzano, G., Schievano, A., Orzi, V., Salati, S., & Adani, F. (2010). Assessing amendment and fertilizing properties of digestates from anaerobic digestion through a comparative study with digested sludge and compost. *Chemosphere*, 81(5), 577-583.

Thangarajan, R., Bolan, N. S., Tian, G., Naidu, R., & Kunhikrishnan, A. (2013). Role of organic amendment application on greenhouse gas emission from soil. *Science of the Total Environment*, 465, 72-96.

Tian, G., Chiu, C. Y., Franzluebbers, A. J., Oladeji, O. O., Granato, T. C., & Cox, A. E. (2015). Biosolids amendment dramatically increases sequestration of crop residue-carbon in agricultural soils in western Illinois. *Applied Soil Ecology*, 85, 86-93.

Tian, G., Granato, T. C., Cox, A. E., Pietz, R. I., Carlson, C. R., & Abedin, Z. (2009). Soil carbon sequestration resulting from long-term application of biosolids for land reclamation. *Journal of Environmental Quality*, 38(1), 61-74.

Todd-Brown, K. E. O., Hopkins, F., Arora, V., Hajima, T., Jones, C., Shevliakova, E., ... & Allison, S. D. (2014). Changes in soil organic carbon storage predicted by Earth system models during the 21st century. *Biogeosciences*, 11(8), 2341.

Uchida, Y., & Akiyama, H. (2013). Mitigation of postharvest nitrous oxide emissions from soybean ecosystems: a review. *Soil Science and Plant Nutrition*, 59, 477–487

Van Groenigen, J. W., Kasper, V. G., Velthof, G. L., Van den Pol-van Dasselaar, A., & Kuikman, P. J. (2004). Nitrous oxide emissions from silage maize fields under different mineral nitrogen fertilizer and slurry applications. *Plant and Soil*, 263(1), 101-111.

Van Groenigen, K. J., Osenberg, C. W., & Hungate, B. A. (2011). Increased soil emissions of potent greenhouse gases under increased atmospheric CO_2. *Nature*, 475(7355), 214-216.

Van Groenigen, K. J., Qi, X., Osenberg, C. W., Luo, Y., & Hungate, B. A. (2014). Faster decomposition under increased atmospheric CO_2 limits soil carbon storage. *Science*, 344(6183), 508-509.

VandenBygaart, A. J., Gregorich, E. G., & Angers, D. A. (2003). Influence of agricultural management on soil organic carbon: A compendium and assessment of Canadian studies. *Canadian Journal of Soil Science*, 83(4), 363-380.

Velthof, G. L., Kuikman, P. J., & Oenema, O. (2003). Nitrous oxide emission from animal manures applied to soil under controlled conditions. *Biology and Fertility of Soils*, 37(4), 221-230.

Wagner-Riddle, C., Furon, A., Mclaughlin, N. L., Lee, I., Barbeau, J., Jayasundara, S., ... & Warland, J. O. N. (2007). Intensive measurement of nitrous oxide emissions from a corn–soybean–wheat rotation under two contrasting management systems over 5 years. *Global Change Biology*, 13(8), 1722-1736.

Wang, W. J., Dalal, R. C., Moody, P. W., & Smith, C. J. (2003). Relationships of soil respiration to microbial biomass, substrate availability and clay content. *Soil Biology and Biochemistry*, 35(2), 273-284.

Wang, Y., Guo, J., Vogt, R. D., Mulder, J., Wang, J., & Zhang, X. (2018). Soil pH as the chief modifier for regional nitrous oxide emissions: New evidence and implications for global estimates and mitigation. *Global Change Biology*, 24(2), e617-e626.

Wiesmeier, M., Poeplau, C., Sierra, C. A., Maier, H., Frühauf, C., Hübner, R., ... & Schilling, B. (2016). Projected loss of soil organic carbon in temperate agricultural soils in the 21 st century: effects of climate change and carbon input trends. *Scientific Reports*, 6(1), 1-17.

Wiesmeier, M., Urbanski, L., Hobley, E., Lang, B., von Lützow, M., Marin-Spiotta, E., ... & Wollschläger, U. (2019). Soil organic carbon storage as a key function of soils-a review of drivers and indicators at various scales. *Geoderma*, 333, 149-162.

Wlodarczyk, T., Stêpniewski, W., & Brzeziñska, M. (2005). Nitrous oxide production and consumption in Calcaric Regosols as related to soil redox and texture. *International Agrophysics*, 19(3), 263.

Wrage, N., Velthof, G. L., Van Beusichem, M. L., & Oenema, O. (2001). Role of nitrifier denitrification in the production of nitrous oxide. *Soil Biology and Biochemistry*, 33(12-13), 1723-1732.

Wrage-Mönnig, N., Horn, M. A., Well, R., Müller, C., Velthof, G., & Oenema, O. (2018). The role of nitrifier denitrification in the production of nitrous oxide revisited. *Soil Biology and Biochemistry*, 123, A3-A16.

Yang, L., & Cai, Z. (2006). Effects of shading soybean plants on N_2O emission from soil. *Plant and Soil*, 283(1-2), 265-274.

Yang, X., Reynolds, W. D., Drury, C. F., Fleming, R., Tan, C. S., Denholm, K., & Yang, J. (2013). Organic carbon and nitrogen stocks in a clay loam soil 10 years after a single compost application. *Canadian Journal of Soil Science*, 94(3), 357-363.

Zhang, Q., Lei, H. M., & Yang, D. W. (2013). Seasonal variations in soil respiration, heterotrophic respiration and autotrophic respiration of a wheat and maize rotation cropland in the North China Plain. *Agricultural and Forest Meteorology*, 180, 34-43.

Zhu, X., Burger, M., Doane, T. A., & Horwath, W. R. (2013). Ammonia oxidation pathways and nitrifier denitrification are significant sources of N_2O and NO under low oxygen availability. *Proceedings of the National Academy of Sciences*, 110(16), 6328-6333.

Zimmermann, M., Leifeld, J., Schmidt, M. W. I., Smith, P., & Fuhrer, J. (2007). Measured soil organic matter fractions can be related to pools in the RothC model. *European Journal of Soil Science*, 58(3), 658-667.

Chapter 3 – Short-term effect of different organic fertilizers on crop yield and greenhouse gas fluxes in two corn-soybean fields in eastern Canada

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Abstract

Organic fertilizers made from municipal organic waste can support crop production while building soil organic carbon (SOC). However, organic fertilizers such as slurries contain abundant organic C and mineral N that can promote soil greenhouse gas (GHG) emission. We evaluated the soil GHG fluxes from two corn-soybean fields in Quebec and Ontario using static chambers over two seasons, after a one-time application of organic fertilizers with distinct properties, namely: composted food waste (compost; 15% N in mineral form), hydrolyzed biosolid slurry (LysteGro; 47% N in mineral form), liquid anaerobic digestate (digestate; 97.5% N in mineral form), plus a mineral fertilizer control (NPK). Fertilizer application supported the production of corn grain ranging from 7.9 – 9.2 t/ha at Lods and 10.2 – 13.0 t/ha at Elora, and soybean grain ranging from 1.9 – 2.3 t/ha at Lods and 4.9 – 5.4 t/ha at Elora (in dry weight), with no significant difference between treatments in both yield and SOC stocks. At Lods, substantial N_2O was emitted the first month following organic fertilizer application (contributing to 44 – 65% of corn growing-season emission), in particular from LysteGro plots. However, differences were not significant when added up to growing-season cumulative emissions for both N_2O and CO_2. We conclude that all the organic fertilizers in this study neither significantly promote nor mitigate small-scale growing-season GHG emissions, while maintaining yields and SOC stocks.

3.1 Introduction

Agricultural soils lose fertility and carbon (C) when they receive insufficient organic matter (OM). Municipalities are a potential source of OM that could be applied to croplands. For instance, the Montreal agglomeration area generated about 355,000 t of municipal organic waste in 2016, of which about 80% was landfilled or incinerated (Bureau du vérificateur général de la Ville de Montréal, 2018). Municipal organic waste is a reservoir of organic C and plant nutrients. After being treated to remove pathogens and toxic substances e.g. heavy metals, it can be used as an organic fertilizer for crop production. Many examples showed that municipal waste compost or biosolids can replace mineral fertilizer to produce economically-optimal crop yields, as reviewed by Diacono & Montemurro (2011). Moreover, Tian et al. (2009) recorded C sequestration rates ranging from 0.54 to 3.05 t C/ha/yr in 41 long-term field trials under municipal biosolids application (compared to -0.07 to 0.17 t C/ha/yr of mineral fertilizer). Thus, organic fertilizers have the potential to support crop production while enhancing SOC stock.

However, applying organic fertilizers can have immediate stimulatory effect on soil GHG emissions, namely CO_2, CH_4 and N_2O. In general, CO_2 and CH_4 emissions increase when the amount of organic C (especially labile C) applied increase, because microbes respire the organic C to produce metabolic energy and carbonaceous gases. Higher CO_2 and CH_4 emissions relative to the amount of C applied also means less C will remain in the soil. Organic fertilizers supplying abundant mineral N directly or through the rapid N-mineralization of low C:N labile OM also stimulate soil N_2O emission (Senbayram et al., 2012; Thangarajan et al., 2013). Besides, organic fertilizers can enhance N_2O production by reducing soil redox potential via increasing soil water content (e.g. direct water addition through liquid or slurry fertilizer) and reducing soil oxygen content (e.g. O_2 consumption via respiring labile C) (Thangarajan et al., 2013; Velthof et al., 2003). Therefore, how much C would stay in the soil and how much N would allocate to plant biomass, as oppose to going into the atmosphere, is directly affected by the physicochemical properties of the organic fertilizer applied.

Composted food waste (compost), LysteGro and liquid anaerobic digestate (digestate) are three organic fertilizers with distinct physicochemical properties available for on-farm use in Quebec and Ontario. We tested their effects on soil GHG emissions as well as crop yield and SOC stocks in two corn-soybean fields in Quebec and Ontario over 2 yr. Compost is an aerobically decomposed, OM-rich solid amendment that is widely used in agriculture to produce crop while enhancing long-term soil fertility by building SOC (Diacono & Montemurro, 2011). LysteGro, a fertilizer product by Lystek International Inc., is a viscous slurry of anaerobically-digested biosolids that are further hydrolyzed under heat and alkaline conditions. It contains ample quantities of both mineral N and hydrolyzed OM and is designed to be both a high-performance plant fertilizer and SOC builder (Garvey et al., 2016; Halloran, 2018). Digestate is a separated liquid fraction of anaerobically-digested organic waste, containing primarily mineral nutrients such as NH_4^+ and HPO_4^{2-} with little OM, thus making it a desirable direct substitute to synthetic mineral fertilizer (Nkoa, 2014).

We hypothesize that applying fertilizer rich in organic C (compost >> LysteGro > digestate) will result in higher CO_2 emissions over the two seasons. When LysteGro is applied, the highest N_2O emission will occur in the first season due to its abundant mineral N and OM (potentially in labile forms due to hydrolysis), the two necessary substrates in nitrification and denitrification. Methane flux will likely be very small in all three organic fertilizer treatments, same as past records in typical annual crop fields in eastern Canada (Gregorich et al., 2005). We expect that LysteGro and digestate application will similarly produce higher corn yields (the first season) than compost due to the higher plant availability of mineral nutrients compared to nutrients derived from the gradual mineralization of composted OM (Flavel and Murphy, 2006; Hargreaves et al., 2008; Masunga et al., 2016). However, compost application will support soybean (second season) growth better due to the residual fertility provided by the long-lasting OM. We expect differences in SOC stocks will not be noticeable within short-term (Necpálová et al., 2014).

3.2 Materials and Methods

3.2.1 Study sites

The two field sites are: 1. Emile A. Lods Agronomy Research Centre, Ste-Anne-de-Bellevue, Quebec (latitude: 45°25'N; longitude: 73°55'W; 39 m elevation), **abbreviated as Lods**, and 2. Elora Research Station, Elora, Ontario (latitude: 43°29'N; longitude: 80°25'W; 376 m elevation), **abbreviated as Elora** (Figure 3-1). Both sites have a humid continental temperate climate. Lods had growing-season temperature of 17.1 °C (2018) and 16.0 °C (2019) and growing-season precipitation of 386 mm (2018) and 624 mm (2019), according to meteorological data from the Ste-Anne-de-Belleuve 1 weather station (Figure 3-2; Environment Canada, 2019a). At Elora, the growing-season temperature was 16.2 °C (2018) and 15.0 °C (2019) and growing-season precipitation was 339 mm (2018) and 508 mm (2019) according to data recorded from the Elora RCS weather station (Figure 3-2; Environment Canada, 2019b). Rare missing meteorological data were replenished from weather stations within 15 km away.

The soil (0 – 20 cm) at Lods is predominantly a Gleysol of Chateauguay loam with initial bulk density of about 1.30 – 1.37 g/cm^3, pH of 5.83 – 6.26 and SOC content of 8.4 – 14.2 g/kg. The year prior to the field experiment at Lods was a fallow season. Historically, Lods cultivation methods followed conventional farming practices (spring disc harrow, fall moldboard plow and mineral fertilization), and the site was used for growing soybean, wheat and canola from 2012 onward. Prior to 2012, it was under forage production. At Elora, the soil is a Gleyed Melanic Brunisol of Woolwhich series with a silt loam texture (bulk density = 1.08 – 1.14 g/cm^3). It has a baseline pH of 7.77 – 7.86 and SOC content of 21.3 – 30.8 g/kg. The main crops grown prior to this study were soybean, canola and barley with conventional farming practices.

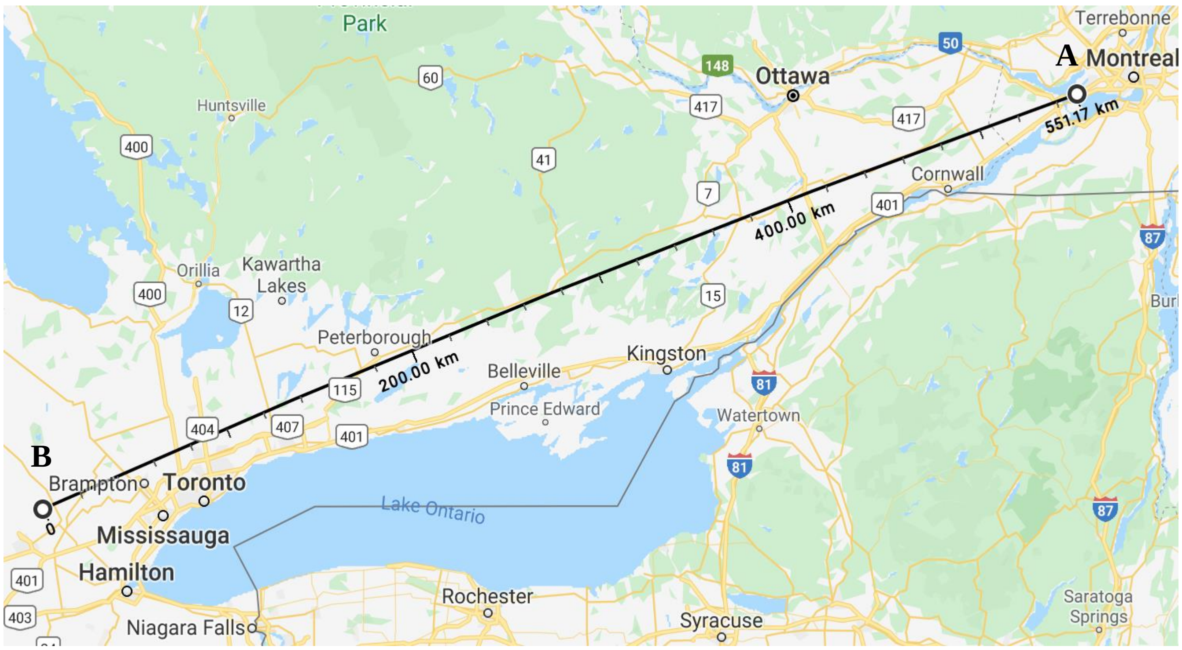

Figure 3-1. The field experiment (2018 – 2019) was conducted at A: the Emile A. Lods Agronomy Centre (Lods) in Ste-Anne-de-Bellevue, Quebec; B: the Elora Research Station (Elora) in Elora, Ontario.

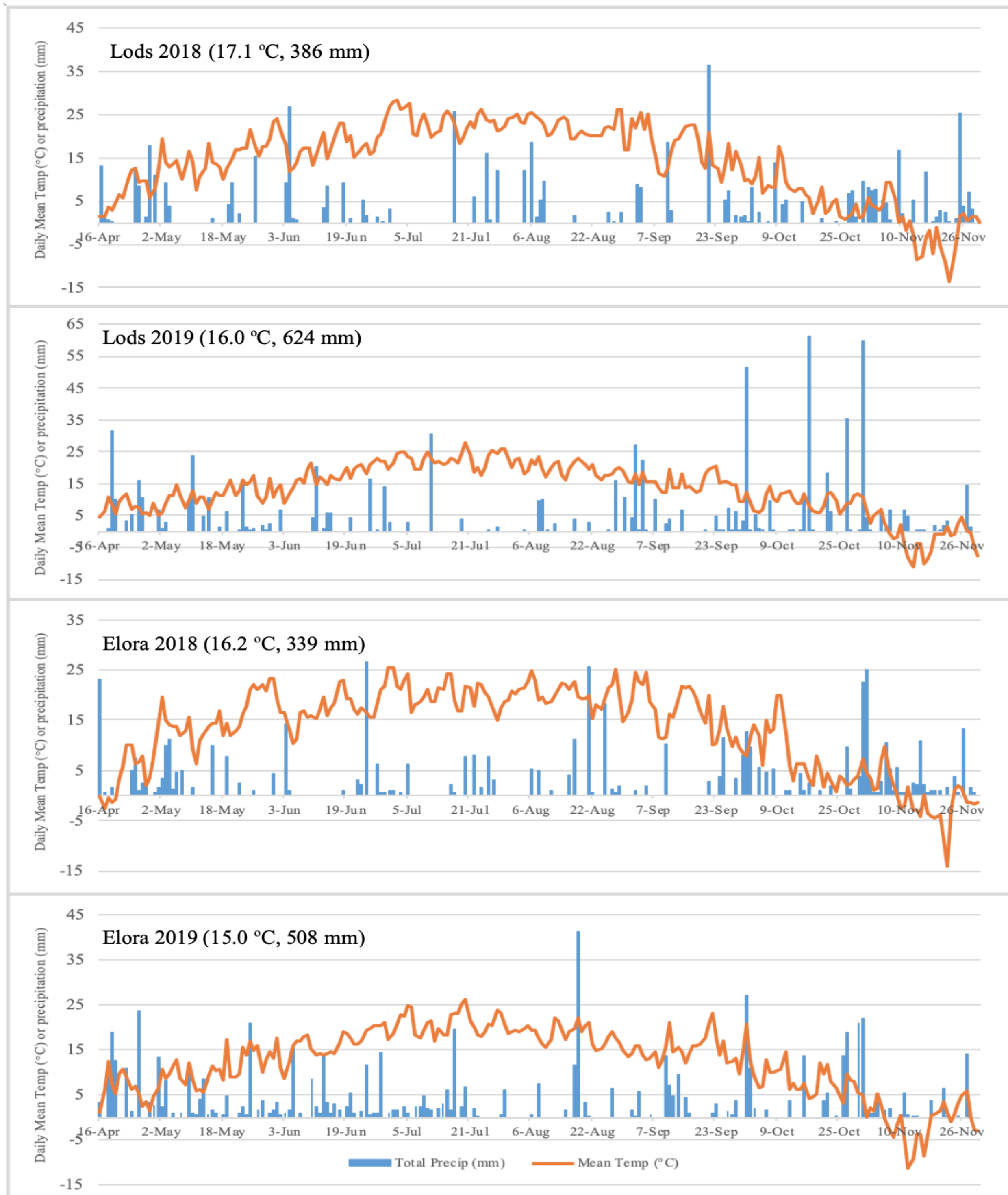

Figure 3-2. Daily mean temperature and precipitation in 2018 and 2019 at the Emile A. Lods Agronomy Centre (Lods) and the Elora Research Station (Elora). The mean growing-season temperature and total (May – Oct) is indicated in brackets.

3.2.2 Experimental design and fertilizer treatments

The field experiment consisted of four fertilizer treatments with four replicates arranged in a randomized complete block design (Appendix 1). Experimental plots were 6×12 m^2 large at Lods and 6×15 m^2 at Elora. Fertilizer treatments consisted of three organic fertilizers: compost (applied at 12 t/ha wet weight), LysteGro (28 m^3/ha) and digestate (42 m^3/ha), plus one mineral fertilizer control (NPK). Fertilizer application rates followed the agronomic recommendations by the Ministry of Agriculture, Fisheries and Food of Québec and the Ontario Ministry of Agriculture, Food and Rural Affairs, based on potential N recovery by the first-season crop. The detailed fertilizer application rates and physicochemical properties can be found in Table 3-1. All fertilizers were applied once prior to planting corn by broadcasting on the surface and incorporating it (10 cm depth) with an offset disk harrow. One exception is that the urea of NPK at Lods was applied with a split-application (50 kg N/ha at planting, 120 kg N/ha at corn V6 stage). Corn (*Zea mays L.)* was seeded (DKC-3378RIB, 76000 seeds/ha, 75 cm inter-row spacing) after fertilizer application on 31st May 2018 at Lods. Weeds were controlled by Roundup Transorb applied on 4th July. We harvested the corn (grain + stover) on 26 Oct 2018 and moldboard plowed about a month later to incorporate the residue. At Elora, corn (*Zea mays L*.) was seeded on 25th May 2018 (DKC-3855RIB, 78500 seeds/ha, 75 cm inter-row spacing). Weeds were controlled by pre-emergence FrontierMax on 28th May and post-emergence Roundup WeatherMax on 14th June. Corn grain was harvested on 18th Oct 2018 and moldboard plowed two weeks later.

Prior to planting soybean, all plots at Lods were supplied with potassium chloride at 40 kg K_2O/ha but no fertilizer was applied at Elora. At Lods, soybean (*Glycine max L.*) was seeded (DKB 003-29, 450000 seeds/ha, 75 cm inter-row spacing) on 9th May 2019. Weeds were controlled by the application of Roundup WeatherMax and shallow inter-row cultivation on 25th June. Soybean harvest and the following field cultivation were carried out on 12th September 2019. At Elora, soybean (*Glycine max L.*) was seeded (DKB 003-29, 450000 seeds/ha, 15 cm inter-row spacing) on 12th June 2019. Soybean harvest was on 11th October 2019. Detailed management timeline can be found in Appendix 2.

Table 3-1. Organic fertilizer application rates based on the Lystek recommendations, relative to the agronomic NPK recommendation from CRAAQ (2010) and the physicochemical properties of organic fertilizer. Fertilizers were applied in May 2018 prior to planting corn at the Emile A. Lods Agronomy Centre (Lods) and the Elora Research Station (Elora).

	NPK	*Compost*	*LysteGro*	*Digestate*
Total N applied (kg/ha)	170†	240	215	231
% mineral-N (NH_4-N + NO_3-N)	All in urea form	15.2	47	97.5
Total P applied (kg P_2O_5/ha)	20 (Lods); 27 (Elora)‡	85.2	260	60.5
Total K applied (kg K_2O/ha)	67 (Lods); 55 (Elora)‡	70.8	116	75.5
Total organic C applied (kg/ha)	NA	2880	1070	92.4
*% OM (w/w)**	NA	48.1	6.25	0.23
% Water (w/w)	NA	36.9	85.8	99.3
Total C/N	NA	12	5.0	0.4

* Percent weight relative to the wet weight of the applied fertilizer.

† Lods: 50 broadcast at planting + 120 sidedress at V6 stage; Elora: 170 all at planting. Both sites used granular urea (46-0-0).

‡ P fertilizer was triple superphosphate (0-46-0) and K fertilizer was potassium chloride (0-0-60) at both sites. Both P and K fertilizers were broadcasted.

3.2.3 Field sampling and laboratory analysis

3.2.3.1 Greenhouse gas flux

We collected gas samples during the growing season (May – Oct) in 2018 and pre-season (April) plus growing season in 2019. Approximately biweekly samples were taken starting from 31^{st} May 2018 and 16^{th} April 2019 at Lods and 29^{th} June 2018 and 27^{th} April 2019 at Elora. At Lods, gas sampling schedule were adapted to catch potential post-precipitation emission peaks and we also increased sampling frequency after fertilizer addition and plowing and in 2019. Gas samples were collected using vented non-steady state gas chambers (Livingston & Hutchinson, 1995) placed in the center of each plot within inter-row space. Briefly, a heat-insulated acrylic chamber cover was placed on a permanent chamber base frame inserted about 7 cm deep into the soil, creating an enclosed headspace of approximately $(56.4 \times 56.4) \times 20$ $cm^3 = 63.6$ L. The chamber cover at Elora is circular cylinder-shaped with a radius of 16.0 cm and 30.1L of headspace volume after mounting on a circular base frame. After placing the cover, gas samples were collected at 0, 10, 20, 30 minutes mostly within 9 am – 3 pm by withdrawing headspace gas through a rubber septum on the cover using a 20 mL gas-tight syringe. We also simultaneously measured soil temperature (0 – 10 cm) using a field thermometer and volumetric soil water content (0 – 15 cm) using a portable TDR 150 spectrum (Aeldscout) during gas sampling. The gas samples were stored in pre-vacuumed 12 mL exetainers (Labco, High Wycombe, UK) sealed by Teflon-silicone septa (National Scientific Company, Rockwood, TN, USA) until analysis by a gas chromatograph 450-GC system (Bruker Corp., Bremen, Germany) with flame ionization detector (set at 300 °C) for CO_2 and CH_4 quantification and an electron capture detector (set at 350°C) for N_2O quantification.

The concentration of gases (ppmv) from the GC was first converted to mass-based concentrations (e.g. μg CO_2-C/L) by ideal gas law and then fluxes were estimated using the HMR package v1.0.0 installed in R v3.5.1 (Pedersen et al., 2010). Raw gas concentration data with abnormal values (e.g. negative concentration and time zero concentration largely different than ambient concentration) were dropped from flux estimation as suggested by Levy et al. (2011). An automatic HMR modeling approach was adopted that allows flexible flux fitting by either a HM or linear model as well as no-flux identification based on a mean squared errors best-fit criterion (Pedersen et al., 2010). Flux estimates were further screened with a post-estimation approach by standard error (SE_{est}) criteria (Appendix 3), eliminating uncertain flux estimates (0.4 – 22% CO_2, 2.4 – 31% CH_4, 0.2 – 18% N_2O flux estimates removed across site-years). The screening approach here is a substitute for screening with r^2 in linear flux estimation (Morris et al., 2013) since r^2 is not available with non-linear model. Cumulative emissions were calculated by first linearly interpolating between fluxes data points and then calculating the trapezoidal area under curve by the auc function of flux package v0.3-0 in R 3.5.1 (Jurasinski et al., 2014).

3.2.3.2 Soil organic carbon

Soil samples were collected before corn planting and harvest (0 – 10 and 10 – 20 cm) for SOC determination. Five soil subsamples within each plot were collected and then composited into one large sample. Analysis of SOC content was done using freeze-dried soil samples (< 2 mm) with an elemental analyzer (Costech 4010, Valencia, USA). The SOC content (g SOC/100g soil) was converted to SOC stock (separately for 0 – 10 and 10 – 20 cm) by the bulk density of the soil obtained from standard soil core method (Culley, 1993) using the equation:

SOC stock 10 cm deep (t C/ha ·10 cm) = SOC content × soil bulk density × 10

Where *SOC content* is in (g SOC/100g soil), *soil bulk density* is in g/cm^3, the factor 10 is for unit conversion. The 0 – 20 cm SOC stocks were obtained from summing the 0 – 10 and 10 – 20 cm SOC stocks.

3.2.3.3 Crop biomass production and nitrogen content

Corn aboveground biomass production was determined from hand-cut plant samples at the center of each plot (cut at about 5 cm above the root) occupying a quadrat area of 2.5 × 1.5 m^2 (two 2.5 m rows of corn, the quadrat area was determined from known row-spacing). Stover and grains were separated, then dried at 55 ^oC for 48 h to obtain the respective dry mass. In addition, since corn cob was discarded without being weighed, the stover biomass was adjusted by considering the cob weight to be 15% of total stover yield (Shinners & Binversie, 2007). Soybean plant samples were similarly hand-collected in 2 × 1.5 m^2 quadrats near maturity and dried to obtain the biomass. The yields in t/ha were calculated by linear extrapolation of the biomass yields within the quadrat area. To understand whether N supply was sufficient for the quality production of biomass, we also determined the stover and grain N content by an elemental analyzer (Thermo Finnigan Flash EA 1112 series, Carlo Erba, Milan, Italy) with finely ground plant samples.

3.2.4 Data Analysis

Analysis of CH_4 fluxes were not carried out because of its insignificance in terms of overall global warming potential (GWP) contribution (on average about - 0.15% of the 100-year GWP). For CO_2 and N_2O, we tested the treatment differences of 1st month cumulative emission (not available at Elora due to insufficient sampling frequency) and growing-season emission (each season and two seasons combined), analyzed separately for different sites. Normal quantile residual plots showed that the distributions of these cumulative emissions were close to normal. Bartlett's test ($\alpha = 0.05$) was used to check the homogeneity of variance. When homogeneous variances is confirmed, simple one-way ANOVA was conducted. Otherwise, treatment differences were analyzed by Welch's ANOVA. For testing combined two-season emissions, a mixed model with repeated term as year and plot as random effect was employed instead. Whenever ANOVA F-test is significant, a subsequent multiple comparison with Holm-Bonferroni adjustment was conducted ($\alpha = 0.05$) in order to control false-positive stringently. We are aware that the ANOVA approach only consider the variability of treatment replicate plots and does not consider uncertainties related to flux estimation and interpolation for cumulative emissions, which could be substantial (Kravchenko & Robertson, 2015; Levy et al., 2011; Venterea et al., 2009). Therefore, we also computed and presented standard error of flux estimates (SE_{est}) to aid the discussion on how to improve the statistical reliability of chamber-based flux estimates in future field experiment (Appendix 4). Briefly, the SE_{est} of the treatment average fluxes were computed with quadrature summation (Farrance & Frenkel, 2012; Peters, 2001). It is a good approximation for summing estimates derived from independent plots (De Nardo, 2002; personal communication with Dr. Joel Tellinghuisen). The formula for quadrature summation of uncertainties from independent variables (x, y…): ax + by… = z is given below:

$$\sigma^2_z = a^2 \sigma^2_x + b^2 \sigma^2_y \ldots$$

Where a and b are both 1 for summation and ¼ for averaging from 4 replicates, σ is standard deviation (σ^2 is variance).

The uncertainties of interpolated fluxes were assumed to take on the weighted average SE_{est} of fluxes on sampled days weighted by its number of adjacent days and are summed linearly (because of perfect correlation). This formulation of SE_{est} inflates when the number of days sampled decreases (more interpolated days), which is in line with theoretical expectation.

For crop yield (both grain and stover biomass of corn and soybean) analyses, one plot at Lods was excluded from the data analysis because it had abnormally low yield due to shading by a nearby hedgerow and frequent waterlogging (also verified in two-sided Grubbs' test to be an outlier, $p < 0.05$). Normal quantile residual plots confirmed data normality and Bartlett's test confirmed the homogeneous variance of data. Therefore, one-way ANOVA was conducted to examine the effect of fertilizer type on crop yields at $\alpha = 0.05$. For SOC stock (0 – 20 cm) analysis, it was similarly screened for outliers with Grubbs' test, checked with normal-quantile residual plots and Bartlett's test. Initial SOC stock by plot was included as a covariate for detecting the treatment effect on final SOC stock (i.e. ANCOVA) at $\alpha = 0.05$. Due to the fact that no treatment effect was found for both crop yields and SOC stock, we did not carry out post-hoc multiple comparisons. All the above statistical tests were carried out in JMP14.0.

3.3 Results

3.3.1 Temporal pattern of soil greenhouse gas fluxes and growing-season emissions

2018 Corn season

At Lods, there was no significant difference in CO_2 emission in the month following fertilizer application. At both sites, CO_2 emissions similarly peaked around July and gradually die down towards late season (Figure 3-3A and 3-4A). The cumulative growing-season CO_2 emissions among treatments were not significantly different in all cases (Table 3-2).

At Lods, N_2O spikes occurred the first month following fertilizer application (Figure 3-3B). LysteGro application produced the highest initial N_2O emission, although only significant for the comparison with NPK (Table 3-2, p = 0.017). The 1st month N_2O emission contributed 44 – 65% of the growing-season total emission among the organic fertilizer treatments. Mid- to late season emissions were consistently low (averaged 1.1 mg N_2O-N/m^2/d for the organic fertilizer treatments from July to Oct), with the exception that the split application of urea in NPK plots appeared to cause higher mid- to late season N_2O emission (Figure 3-3B). Nonetheless, no significant difference of cumulative growing-season emission was found (Table 3-2).

At Elora, the "tail" of a fertilizer-induced N_2O spike can somewhat be seen for LysteGro plots (Figure 3-4B). This higher initial flux from LysteGro plots led to marginal difference of cumulative emission compared to other treatments (ANOVA F-test p = 0.048, but no significant difference in multiple comparisons). The mid- to late season N_2O emission at Elora was generally very low (0.26 mg N_2O-N/m^2/d).

2019 Soybean season

At both sites, there was no prominent difference of CO_2 emission among treatments in the entire season (Table 3-2). At Lods, peak CO_2 emissions occurred around late July, slightly later phenologically than corn season (Figure 3-5A). At Elora, the CO_2 emission trends were similar to Lods but with one exception where a sharp spike of CO_2 occurred in LysteGro plots on 28th Aug (Figure 3-6A), but this does not result in any significant difference nonetheless.

Without N-fertilizer application in 2019, 1st month N_2O emissions contributed 12 – 26% of the overall growing-season emission at Lods and no difference was found. The two N_2O peaks at Lods coincide with field cultivation events (one in pre-planting and the other in mid-season for weed control) (Figure 3-5B). Otherwise, N_2O emissions remained low and steady mid- to late season (1.4 mg N_2O-N/m^2/d emitted averaged across all treatments from 1 month after planting onwards). At Elora, negative N_2O fluxes were measured the month prior to planting (Figure 3-

6B) and remained low in mid- to late season for all treatments (averaging 0.49 mg N_2O-N/m^2/d). At both sites, no significant difference of growing-season N_2O emission was found.

Two seasons overall

No significant treatment effect was found in all cases except for N_2O emissions at Elora where ANOVA F-test p = 0.0425 with LysteGro recorded a somewhat higher N_2O emission (but no difference in multiple comparisons). The two-season N_2O emission at Lods had marginal ANOVA F-test p = 0.071 with NPK appeared to have somewhat higher emission compared to the organic fertilizers.

Table 3-2. Average treatment CO_2 and N_2O fluxes at the Emile A. Lods Agronomy Centre (Lods) and the Elora Research Station (Elora) in 2018 and 2019. Values are in the form: estimates ± SEM, estimated by HMR v1.0.0 and linear interpolation.

	1st month emission (Lods)		*Growing-season emission (Lods)†*		*Growing-season emission (Elora) †‡*		*Lods Total*	*Elora Total*
	2018*	2019	2018	2019	2018	2019	2018 + 2019	
	CO_2 (g CO_2-C/m^2)							
NPK	29.9 ± 9.1	17.3 ± 6.3	234 ± 26	229 ± 37	250 ± 10	340 ± 61	463 ± 47	590 ± 55
Compost	58.3 ± 8.3	16.8 ± 5.0	266 ± 36	292 ± 25	314 ± 31	346 ± 7	558 ± 40	657 ± 36
LysteGro	41.2 ± 5.6	11.2 ± 1.2	258 ± 23	259 ± 21	279 ± 9	479 ± 107	515 ± 16	758 ± 102
Digestate	30.7 ± 6.2	17.9 ± 4.6	192 ± 29	269 ± 22	335 ± 41	367 ± 41	460 ± 48	702 ± 24
	N_2O (mg N_2O-N/m^2)							
NPK	20.1 ± 10.3^{B}	60.4 ± 13.8	275 ± 108	232 ± 41	20.9 ± 17.0	55.4 ± 10.7	511 ± 127	76.3 ± 19.6
Compost	53.8 ± 17.8^{AB}	21.2 ± 7.0	123 ± 40	174 ± 52	20.5 ± 4.1	53.2 ± 13.0	297 ± 42	73.7 ± 11.6
LysteGro	122 ± 31.4^{A}	33.4 ± 15.0	188 ± 30	149 ± 32	63.4 ± 13.6	39.2 ± 14.1	337 ± 51	103 ± 23.2
Digestate	41.2 ± 6.5^{AB}	42.0 ± 16.4	83 ± 21	227 ± 44	21.3 ± 4.3	55.3 ± 12.1	311 ± 58	76.6 ± 15.1

* The estimates are only compared within each column (between treatments). Values within the same column bearing the same **uppercase letter** are not different from each other in multiple comparisons. **No letter** in a column means there was no difference found in multiple comparisons ($\alpha = 0.05$).

† Growing season includes periods roughly from planting until harvest (corn:147 days at Lods and 120 days at Elora; soybean: 136 days at both sites).

‡ At Elora, the first month measurement was missing in 2018. Also, only two measurements were obtained on the first month in 2019, so first-month cumulative emission was not computed.

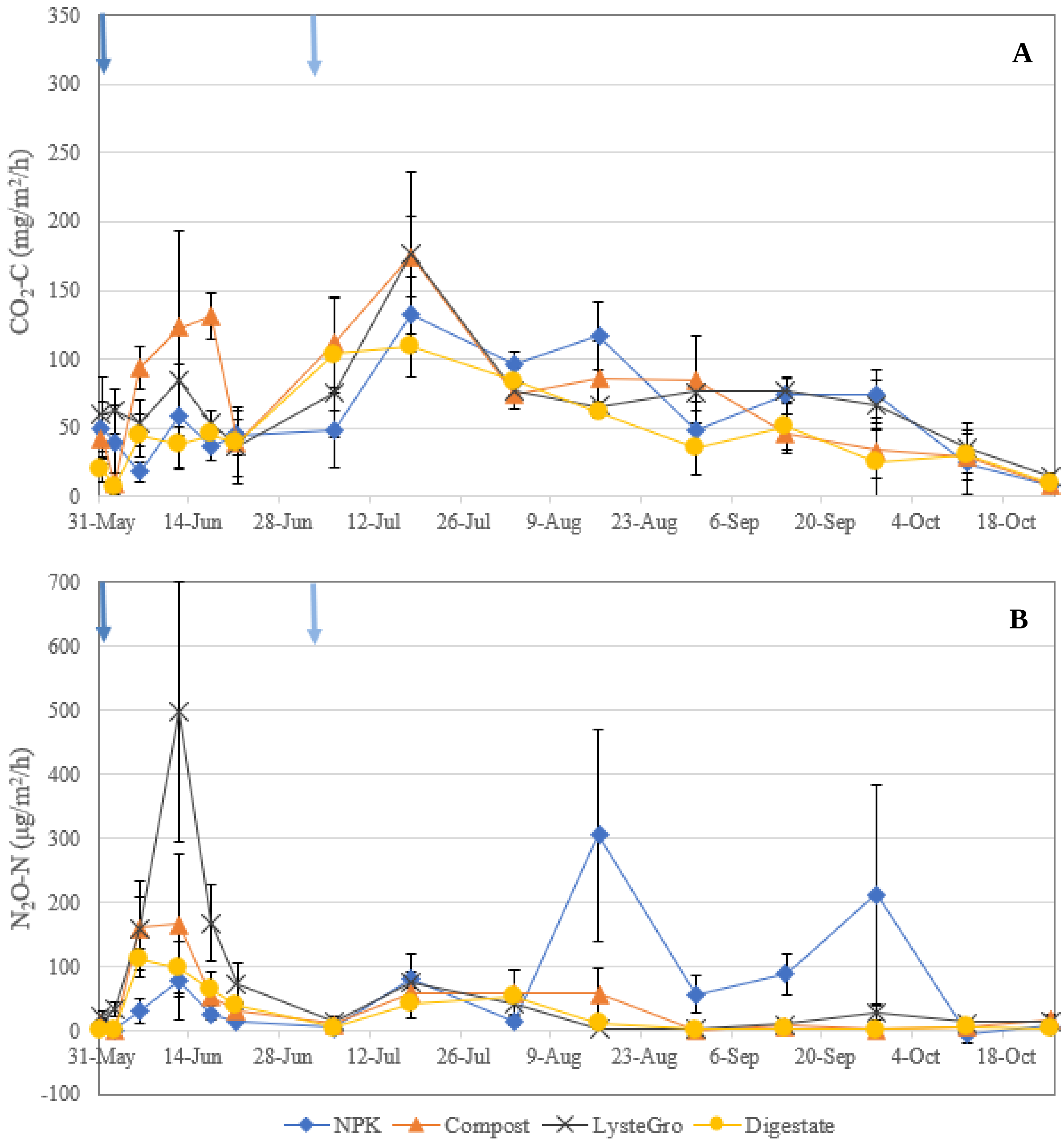

Figure 3-3. Flux time series of A: CO_2 (top) and B: N_2O (bottom) recorded at the Emile A. Lods Agronomy Centre (Lods) in 2018. The deep blue arrows indicate pre-plant fertilizer application and incorporation, the light blue arrow is the second split of sidedress urea application in NPK treatment. Error bars are SEM.

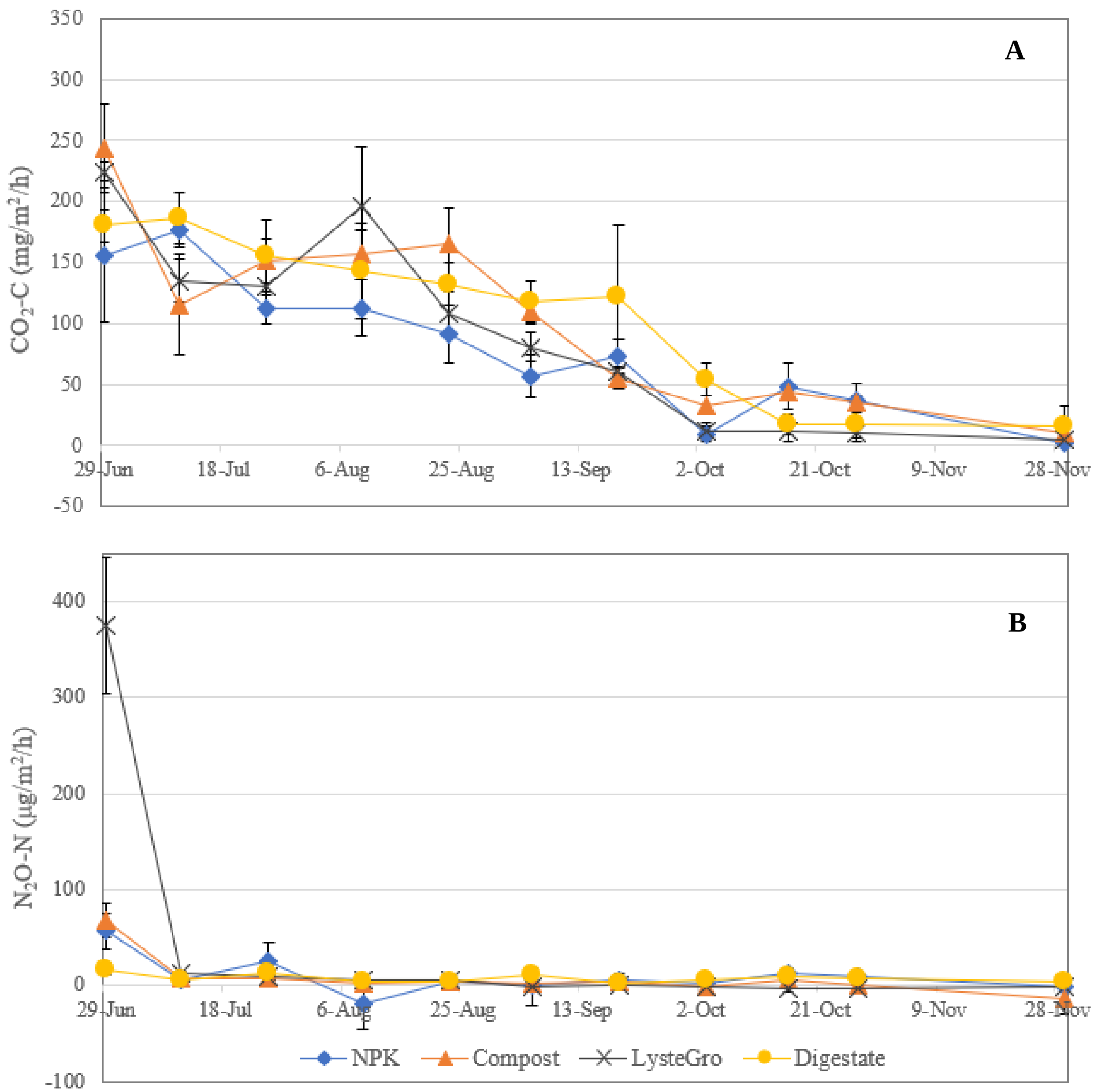

Figure 3-4. Flux time series of A: CO_2 (top) and B: N_2O (bottom) at the Elora Research Station (Elora) in 2018. Note that fertilizer application and planting occurred before the start of sampling. Error bars are SEM.

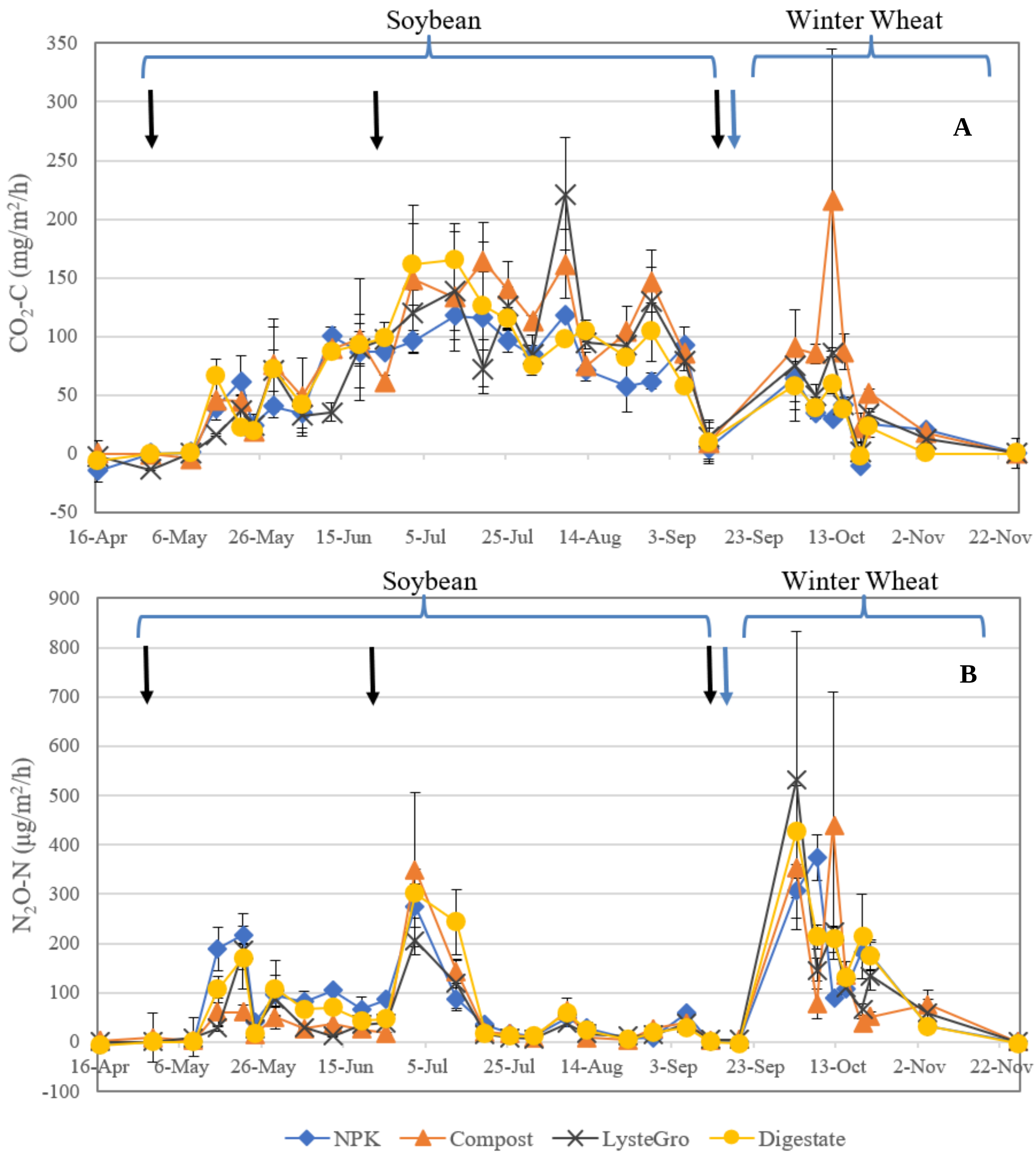

Figure 3-5. Flux time series of A: CO_2 (top) and B: N_2O (bottom) at the Emile A. Lods Agronomy Centre (Lods) in 2019. Black arrows indicate field cultivation events, deep blue arrow indicate fertilizer application and incorporation. Note that the winter wheat after soybean harvest is out of the scope of the present analysis. Error bars are SEM.

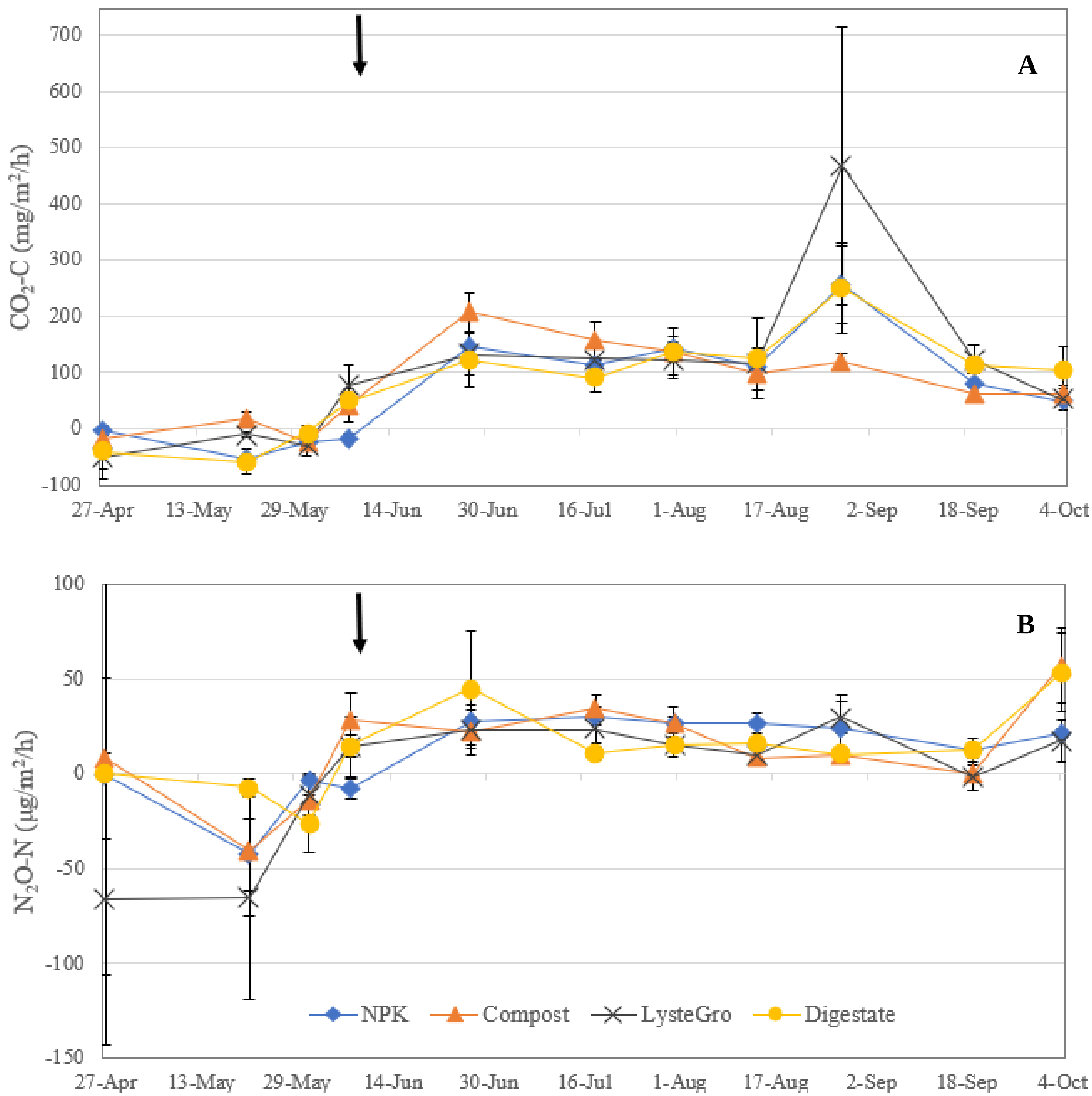

Figure 3-6. Flux time series of A: CO_2 (top) and B: N_2O (bottom) at the Elora Research Station (Elora) in 2019. Black arrows indicate field cultivation events. Error bars are SEM.

3.3.2 Crop yields

All fertilizers supported similar grain and stover biomass yields for both corn and soybean (Figure 3-7, ANOVA not significant). The average corn grain yields at Lods (8.4 t/ha dry weight) was comparable to the 5-year average of Montréal, Laval and Lanaudière region (8.0 t/ha from 2015-2019[1]) and yield at Elora (11.5 t/ha) was slightly higher than average yield typical in Southern Ontario (5-year average of 9.21 t/ha from 2015-2019[1]) (Statistics Canada, 2020). At Lods, the N concentration in corn grain was, on average 16.8 ± 0.38 g N/kg grain (n = 16) with no difference between fertilizer treatments, which is greater than the global average of 12.1 g N/kg grain in corn (Ciampitti & Vyn, 2013). Grain N data are not available at Elora.

Soybean grain yields at Lods (2.1 t/ha) were lower than the 5-year regional average (2.6 t/ha[2]) whereas Elora had a relatively high yield (5.1 t/ha) compared to the Southern Ontario 5-year average (2.9 t/ha[2]) (Statistics Canada, 2020). At Lods, soybean grain N concentration were also consistent, averaging at 53.3 ± 1.5 g N/kg grain (n = 16), which is lower than the mean value (63.4 g N/kg) reported in a global meta-analysis of soybean grain (including fertilized soybean) by Salvagiotti et al. (2008). Grain N data are not available at Elora.

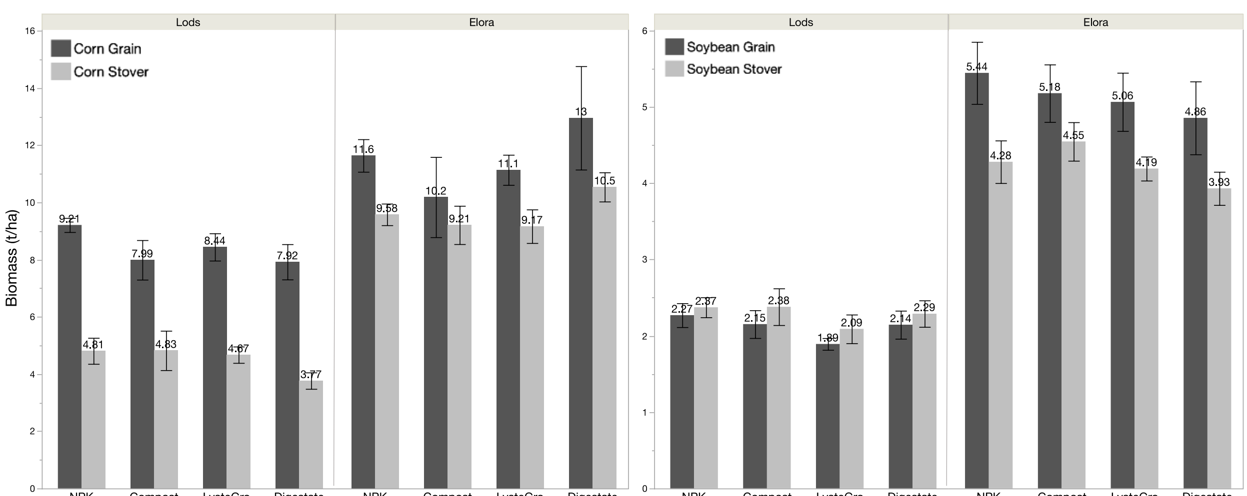

Figure 3-7. The biomass yield of corn (left) and soybean (right) at the Emile A. Lods Agronomy Centre (Lods) and the Elora Research Station (Elora). Error bars are SEM.

3.3.3 Soil organic carbon

The SOC stock (Table 3-3) ANCOVA indicates that there was no treatment effect at Lods. At Elora, treatment effect was not significant but marginal (F-test $p = 0.096$), with LysteGro and compost having somewhat higher LSmean SOC stocks. The covariate (initial SOC stock) has a significant positive relationship with final SOC stock at Elora but not at Lods. Overall, there is limited evidence over 1 season that some organic fertilizers are better than others at building SOC. The change in SOC stock within each treatment was not calculated to infer SOC accrual because short-term SOC changes could be produced by natural variation in time and space regardless of fertilizer application.

Table 3-3. Soil organic carbon stocks (0 – 20 cm) measured just before corn planting and harvest at the Emile A. Lods Agronomy Centre (Lods) and the Elora Research Station (Elora). Values are in t/ha.

Lods	**Initial SOC May 2018**	**Final SOC Oct 2018**
NPK	28.6	31.0
Compost	30.0	33.9
LysteGro	27.4	32.0
Digestate	26.9	30.7
Elora		
NPK	52.4	58.0
Compost	49.6	57.1
LysteGro	53.9	61.4
Digestate	52.8	56.3

3.4 Discussion

3.4.1 The effect of different types of organic fertilizer

Neither crop yield nor grain N concentration differed among fertilizer treatments. This rejects the hypothesis that corn yield would be higher when fertilizers containing more mineral N were applied (i.e. digestate and LysteGro) than with the OM-rich compost. Compost also did not have superior soybean yield, indicating it did not provide much residual fertility. However, the finding is still understandable considering all fertilizer treatments were applied at agronomic rate (215 – 240 kg N/ha for the organic fertilizers and 170 kg N/ha for urea), and compost, although having a high organic N content, had a low C:N (= 12) and apparently did not stimulate N immobilization. Nonetheless, the similar yields mean that these organic fertilizers can replace NPK for crop production. No significant difference in SOC stocks was observed or expected, since management effects on SOC typically only become detectable over a longer time ~ 10 yr (Necpálová et al., 2014).

No significant difference in CO_2 emissions was found across site-years (Table 3-2). This does not support the hypothesis that applying compost (containing more OM) would lead to higher CO_2 emission compared to fertilizers supplying less OM. The likely reason why cumulative CO_2 emission did not vary much is that the CO_2 produced from the decomposition of fertilizer OM was limited compared to those from native SOC and root exudates. The amount of organic C applied via compost (for the current application rate) is only about one-tenth of the amount of SOC in the first 20 cm of soil at Lods and much less at Elora. This also explains why the fertilizer effect on SOC accrual were small. We therefore conclude that the physicochemical properties of these fertilizers are not an important determinant of small-scale soil CO_2 emission when applied at agronomic rate.

There were differences in N_2O emission at Lods initially in the 1st month and marginally at Elora over the sampling period (also attributable to the "tail" from initial N_2O spike). As predicted, LysteGro, the biosolid slurry induced the strongest initial spikes, presumably because they contributed more mineral N and labile OM that was metabolized by denitrifiers. However, the evidence that LysteGro application resulted in higher growing-season N_2O emission from one to two seasons is small. At both sites, LysteGro plots may even have lower N_2O emissions in 2019, although not significant (Table 3-2). Therefore, we can say that overall there is evidence that organic fertilizer physicochemical properties play some roles in very short-term N_2O emission but not for one to two growing-season emission. Although biosolids and slurries have been found to stimulate N_2O emission in numerous occasions (e.g. Ball et al., 2004; Chantigny et al., 2013; Van Groenigen et al., 2004) including in a meta-analysis by Charles et al. (2017), this trend may not hold when a field fertilized one-time at agronomic rate is concerned. Longer-term study may be needed to discover if there is small effect that accrues over time that is important for agricultural management decision.

3.4.2 What influences the temporal dynamics of CO_2 and N_2O fluxes?

Soil temperature is known to positively correlates with decomposition or respiration rate (Fang & Moncrieff, 2001; Zhang et al., 2013), so it is no surprise that the trends of CO_2 emission followed quite closely with air temperature trends (Figure 3-2 to Figure 3-6). On the other hand, there was no obvious association of precipitation with CO_2 emission except in a few occasions where the re-occurrence of rainfall following a dry period appeared to stimulate CO_2 emission (e.g. 17 Jul 2018 and 8 Aug 2019 at Lods), similar to what was reported in Pelster et al. (2012). The lack of association is probably because soil moisture is secondary to the availability of organic substrates in controlling respiration. More secretion of root exudates during active vegetative growth likely contributed to the higher emissions around July comparing to August and early September (Gransee & Wittenmayer, 2000; Rochette & Flanagan, 1997; Sey et al., 2010), despite temperature and precipitation (as well as soil temperature and moisture, data not shown) remained similar in this span for both sites.

Nitrous oxide fluxes appeared not very responsive to either air temperature and rainfall and responded strongly only after fertilizer application and field cultivation. Soybean season had comparable overall N_2O emission to corn season at Lods despite aboveground corn biomass were harvested and no N fertilizer was applied before planting soybean. This is in contrast to many studies which found lower N_2O emissions associated with soybean than corn season (Behnke et al., 2018; Drury et al., 2008; Gregorich et al., 2005; Johnson et al., 2010), and in Mosier et al. (2006) and Wagner-Riddle et al. (2007), they even reported several times lower N_2O emission in 0N-fertilized soybean following fertilized corn. The reason we saw high N_2O emission in soybean season is likely because of the much higher growing-season rainfall in 2019 than 2018, which promotes denitrification (Rochette et al., 2018). Abundant and consistent rainfalls throughout the growing season did not create sharp N_2O spikes, but rather, it raised the background N_2O emission (Figure 3-3 vs. Figure 3-5). This is also true at Elora, where N_2O emissions in 2019 from mid- to late season were noticeably higher (0.26 mg N_2O-N/m^2/d in 2018 and 0.49 mg N_2O-N/m^2/d in 2019). The above highlights the strong influence of factors other than fertilizer application in our experiment. Nevertheless, our infrequent sampling might not capture the full details of emission trends over time. The frequency and timing of gas sampling is crucial as factors such as precipitation and soil moisture govern the transport and release of gas into the atmosphere, not just gas production.

3.4.3 Limitation of findings and potential source of improvements

Chamber-based GHG measurement has been a popular bottom-up approach to estimate land-based emission. However, in this study, we can see that both the SEM (Table 3-2, 3-3) and SE_{est} (Appendix 4) are very high. This large uncertainty causes only large differences to be detectable by statistical methods, which is known as the issue of effect size inflation. It could cause problem if identifying a potential risk factor is of concern, thus there is a strong need to address this issue. There are numerous sources of high-magnitude uncertainties associated with chamber GHG fluxes (Kravchenko & Robertson, 2015; Levy et al., 2011; Rochette & Eriksen-Hamel, 2008). Many studies only considered treatment replicate variability (can be conceived as a kind of spatial variability) and adopt an ANOVA approach, ignoring the large uncertainties with measurement and estimation methods nested under it. In our study, we noted high SE_{est} in most estimates even after screening (Appendix 4), which means including it would make the already non-significant differences "even more non-significant". The SE_{est} are even higher than SEM in some cases at Elora where the sampling was infrequent (since the formulation of SE_{est} here penalized low sampling frequency). To tackle this, a higher number of data points per time-series (in our case only 4) can directly reduce SE_{est}, we therefore advocate increasing the number of gas samples whenever resource permits, especially when in most cases researchers ignored estimation-level uncertainties (hence a lower inherent SE_{est} would help improve the validity of ANOVA results).

We only tried to interpret our GHG findings as small-scale phenomena. Generalizing beyond this scale is problematic due to the uncertain representativeness of the small chamber to the experimental plot (only one 0.32 or 0.08 m^2 chamber within a plot of size 72 or 90 m^2).

Greenhouse gas especially N_2O fluxes are known to have very high spatial variability (Chadwick et al., 2014). For instance, Morris et al. (2013) found that N_2O fluxes had essentially no spatial correlation for chambers more than 1 m apart from each other. Therefore, if the flux estimates are meant for extrapolation or application to larger scales (e.g. used to validate a site or regional-level model), we suggest adding more chambers per plot or having smaller plot size and more replicates to address both representativeness and spatial variability (i.e. increasing the density of chambers).

Other sources of uncertainty are related to the timing and frequency of sampling in capturing the true temporal variability of GHG fluxes. Of course, whenever resource permits, a more frequent sampling leads to a better estimate of true emission. Parkin (2008) found that a sampling interval of below 7 days is required in order to make sure the measured flux falls within ± 20% of the expected flux (collected by automated chambers every 6 h). In our study, extracting a biweekly GHG dataset from the full (ca. weekly) dataset of Lods 2019 resulted in a -8.9 to +8.7% differences in growing-season N_2O emission estimates across the treatments and a -0.4 to +10.6% differences in growing-season CO_2 emission estimates, substantially altering the magnitude of differences between treatments. When high-frequency measurement is not possible due to resource constraint, it is important to at least pay attention to capture probable peaks as well as the"troughs" that follow, given the episodic nature of N_2O flux in particular. In summary, it is clear that there is still a gap between chamber-based GHG fluxes and properly informing larger-scale management decision.

3.5 Conclusion

The present study evaluated the soil GHG fluxes associated with the application of three organic fertilizers: composted food waste (compost), hydrolyzed biosolid slurry (LysteGro) and liquid anaerobic digestate (digestate), on two corn-soybean fields in Quebec and Ontario. We also measured crop yields and SOC stocks but found no evidence of significant differences among fertilizer treatments. The initial N_2O fluxes differed among fertilizer treatments, which was the highest in LysteGro plots. We attributed this to its high mineral N and hydrolyzed OM content in slurry form, which could promote denitrification. However, no significant difference was found for both N_2O and CO_2 when added up to one to two growing season cumulative emissions. We concluded that for the application rates used in this study, the effect of organic fertilizer physicochemical properties was small on one to two seasons GHG fluxes in the two corn-soybean fields, regardless of different soils and other site characteristics.

Chapter 3 references

Ball, B. C., McTaggart, I. P., & Scott, A. (2004). Mitigation of greenhouse gas emissions from soil under silage production by use of organic manures or slow-release fertilizer. *Soil Use and Management,* 20(3), 287-295.

Behnke, G. D., Zuber, S. M., Pittelkow, C. M., Nafziger, E. D., & Villamil, M. B. (2018). Long-term crop rotation and tillage effects on soil greenhouse gas emissions and crop production in Illinois, USA. *Agriculture, Ecosystems & Environment,* 261, 62-70.

Bureau du vérificateur général de la Ville de Montréal (2018). Organic Waste Management. In *Bureau du vérificateur général de la Ville de Montréal 2017*. Retrieved from: http://www.bvgmtl.ca/wp-content/uploads/2018/06/AR2017_EN-FullVersion_Web.pdf

Chadwick, D. R., Cardenas, L., Misselbrook, T. H., Smith, K. A., Rees, R. M., Watson, C. J., ... & Dhanoa, M. S. (2014). Optimizing chamber methods for measuring nitrous oxide emissions from plot-based agricultural experiments. *European Journal of Soil Science,* 65(2), 295-307.

Chantigny, M. H., Pelster, D. E., Perron, M. H., Rochette, P., Angers, D. A., Parent, L. É., ... & Ziadi, N. (2013). Nitrous oxide emissions from clayey soils amended with paper sludges and biosolids of separated pig slurry. *Journal of Environmental Quality*, 42(1), 30-39.

Charles, A., Rochette, P., Whalen, J. K., Angers, D. A., Chantigny, M. H., & Bertrand, N. (2017). Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. *Agriculture, Ecosystems & Environment*, 236, 88-98.

Ciampitti, I. A., & Vyn, T. J. (2013). Grain nitrogen source changes over time in maize: A review. *Crop Science*, 53(2), 366-377.

Culley, J.L.B. (1993). Density and compressibility. In *Soil Sampling and Method of Analysis* (Carter, M.R., Ed.), pp. 529-630Lewis Publisher, New York, USA.

De Nardo, L. (2002). On the Propagation of Statistical Errors. HERMES Internal Note, 02-008.

Diacono, M., & Montemurro, F. (2011). Long-term effects of organic amendments on soil fertility. In *Sustainable Agriculture Volume 2* (pp. 761-786). Springer, Dordrecht.

Drury, C. F., Yang, X. M., Reynolds, W. D., & McLaughlin, N. B. (2008). Nitrous oxide and carbon dioxide emissions from monoculture and rotational cropping of corn, soybean and winter wheat. *Canadian Journal of Soil Science*, 88(2), 163-174.

Environment Canada. (2019a). Daily meterological summaries for Ste-Anne-de-Belleuve 1. Ottawa, ON, Canada: Atmospheric Environment Branch, Environment Canada.

Environment Canada. (2019b). Daily meterological summaries for Elora RCS. Ottawa, ON, Canada: Atmospheric Environment Branch, Environment Canada.

Fang, C., & Moncrieff, J. B. (2001). The dependence of soil CO_2 efflux on temperature. Soil *Biology and Biochemistry*, 33(2), 155-165.

Farrance, I., & Frenkel, R. (2012). Uncertainty of measurement: a review of the rules for calculating uncertainty components through functional relationships. *The Clinical Biochemist Reviews*, 33(2), 49.

Flavel, T. C., & Murphy, D. V. (2006). Carbon and nitrogen mineralization rates after application of organic amendments to soil. *Journal of Environmental Quality*, 35(1), 183-193.

Gransee, A., & Wittenmayer, L. (2000). Qualitative and quantitative analysis of water-soluble root exudates in relation to plant species and development. *Journal of Plant Nutrition and Soil Science*, 163(4), 381-385.

Gregorich, E. G., Rochette, P., VandenBygaart, A. J., & Angers, D. A. (2005). Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. *Soil and Tillage Research*, 83(1), 53-72.

Hargreaves, J. C., Adl, M. S., & Warman, P. R. (2008). A review of the use of composted municipal solid waste in agriculture. *Agriculture, Ecosystems & Environment*, 123(1-3), 1-14.

Johnson, J. M., Archer, D., & Barbour, N. (2010). Greenhouse gas emission from contrasting management scenarios in the northern Corn Belt. *Soil Science Society of America Journal*, 74(2), 396-406.

Jurasinski, G., Koebsch, F., Guenther, A., & Beetz, S. (2014).flux: Flux rate calculation from dynamic closed chamber measurements. R package version 0.3-0. https://CRAN.R-project.org/package=flux

Kravchenko, A. N., & Robertson, G. P. (2015). Statistical challenges in analyses of chamber-based soil CO_2 and N_2O emissions data. *Soil Science Society of America Journal*, 79(1), 200-211.

Levy, P. E., Gray, A., Leeson, S. R., Gaiawyn, J., Kelly, M. P. C., Cooper, M. D. A., ... & Sheppard, L. J. (2011). Quantification of uncertainty in trace gas fluxes measured by the static chamber method. *European Journal of Soil Science*, 62(6), 811-821.

Livingston, G. P., & Hutchinson, G. L. (1995). Enclosure-based measurement of trace gas exchange: applications and sources of error. In: Matson, P.A. and Harris, R.C., Eds., *Biogenic trace gases: measuring emissions from soil and water*. Blackwell Science Ltd., Oxford, UK. 14–51.

Masunga, R. H., Uzokwe, V. N., Mlay, P. D., Odeh, I., Singh, A., Buchan, D., & De Neve, S. (2016). Nitrogen mineralization dynamics of different valuable organic amendments commonly used in agriculture. *Applied Soil Ecology*, 101, 185-193.

Morris, S. G., Kimber, S. W. L., Grace, P., & Van Zwieten, L. (2013). Improving the statistical preparation for measuring soil N_2O flux by closed chamber. *Science of the Total Environment*, 465, 166-172.

Mosier, A. R., Halvorson, A. D., Reule, C. A., & Liu, X. J. (2006). Net global warming potential and greenhouse gas intensity in irrigated cropping systems in northeastern Colorado. *Journal of Environmental Quality*, 35(4), 1584-1598.

Necpálová, M., Anex, R. P., Kravchenko, A. N., Abendroth, L. J., Del Grosso, S. J., Dick, W. A., ... & Sawyer, J. E. (2014). What does it take to detect a change in soil carbon stock? A regional comparison of minimum detectable difference and experiment duration in the north central United States. *Journal of Soil and Water Conservation*, 69(6), 517-531.

Nkoa, R. (2014). Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review. *Agronomy for Sustainable Development*, 34(2), 473-492.

Parkin, T. B. (2008). Effect of sampling frequency on estimates of cumulative nitrous oxide emissions. *Journal of environmental quality*, 37(4), 1390-1395.

Pedersen, A. R., Petersen, S. O., & Schelde, K. (2010). A comprehensive approach to soil-atmosphere trace-gas flux estimation with static chambers. *European Journal of Soil Science*, 61(6), 888-902.

Pelster, D. E., Chantigny, M. H., Rochette, P., Angers, D. A., Rieux, C., & Vanasse, A. (2012). Nitrous oxide emissions respond differently to mineral and organic nitrogen sources in contrasting soil types. *Journal of environmental quality*, 41(2), 427-435.

Peters, C. A. (2001). Statistics for analysis of experimental data. Environmental engineering processes laboratory manual, 1-25.

R Development Core Team. (2017). R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria

[URL: http://www.R-project.org]

Rochette, P., & Eriksen-Hamel, N. S. (2008). Chamber measurements of soil nitrous oxide flux: are absolute values reliable?. *Soil Science Society of America Journal*, 72(2), 331-342.

Rochette, P., & Flanagan, L. B. (1997). Quantifying rhizosphere respiration in a corn crop under field conditions. *Soil Science Society of America Journal*, 61(2), 466-474.

Rochette, P., Liang, C., Pelster, D., Bergeron, O., Lemke, R., Kroebel, R., ... & Flemming, C. (2018). Soil nitrous oxide emissions from agricultural soils in Canada: Exploring relationships with soil, crop and climatic variables. *Agriculture, Ecosystems & Environment*, 254, 69-81.

Salvagiotti, F., Cassman, K. G., Specht, J. E., Walters, D. T., Weiss, A., & Dobermann, A. (2008). Nitrogen uptake, fixation and response to fertilizer N in soybeans: A review. *Field Crops Research*, 108(1), 1-13.

Senbayram, M., Chen, R., Budai, A., Bakken, L., & Dittert, K. (2012). N_2O emission and the $N_2O/(N_2O+N2)$ product ratio of denitrification as controlled by available carbon substrates and nitrate concentrations. *Agriculture, Ecosystems & Environment*, 147, 4-12.

Sey, B. K., Manceur, A. M., Whalen, J. K., Gregorich, E. G., & Rochette, P. (2010). Root-derived respiration and nitrous oxide production as affected by crop phenology and nitrogen fertilization. *Plant and Soil*, 326(1-2), 369-379.

Shinners, K. J., & Binversie, B. N. (2007). Fractional yield and moisture of corn stover biomass produced in the Northern US Corn Belt. *Biomass and Bioenergy*, 31(8), 576-584.

Statistics Canada. Estimated areas, yield and production of principal field crops by Small Area Data Regions, in metric and imperial units. https://www150.statcan.gc.ca/t1/tbl1/en/cv.action?pid=3210000201#timeframe

Thangarajan, R., Bolan, N. S., Tian, G., Naidu, R., & Kunhikrishnan, A. (2013). Role of organic amendment application on greenhouse gas emission from soil. *Science of the Total Environment*, 465, 72-96.

Tian, G., Granato, T. C., Cox, A. E., Pietz, R. I., Carlson, C. R., & Abedin, Z. (2009). Soil carbon sequestration resulting from long-term application of biosolids for land reclamation. *Journal of Environmental Quality*, 38(1), 61-74.

Van Groenigen, J. W., Kasper, V. G., Velthof, G. L., Van den Pol-van Dasselaar, A., & Kuikman, P. J. (2004). Nitrous oxide emissions from silage maize fields under different mineral nitrogen fertilizer and slurry applications. *Plant and Soil*, 263(1), 101-111.

Velthof, G. L., Kuikman, P. J., & Oenema, O. (2003). Nitrous oxide emission from animal manures applied to soil under controlled conditions. *Biology and Fertility of Soils*, 37(4), 221-230.

Venterea, R. T., Spokas, K. A., & Baker, J. M. (2009). Accuracy and precision analysis of chamber-based nitrous oxide gas flux estimates. *Soil Science Society of America Journal*, 73(4), 1087-1093.

Wagner-Riddle, C., Furon, A., Mclaughlin, N. L., Lee, I., Barbeau, J., Jayasundara, S., ... & Warland, J. O. N. (2007). Intensive measurement of nitrous oxide emissions from a corn–soybean–wheat rotation under two contrasting management systems over 5 years. *Global Change Biology*, 13(8), 1722-1736.

Zhang, Q., Lei, H. M., & Yang, D. W. (2013). Seasonal variations in soil respiration, heterotrophic respiration and autotrophic respiration of a wheat and maize rotation cropland in the North China Plain. *Agricultural and Forest Meteorology*, 180, 34-43.

Connecting Paragraph

The findings of Chapter 3 indicated that there was no prominent advantages or drawbacks in terms of yield, SOC stock and GHG emission for the organic fertilizers applied at agronomic rates over short-term. However, it does not mean that this will remain as the status quo with time. Very often, agroecological responses, such as changes in SOC, are small but accumulating. Long-term field experiment is an option. However, they are often hard to organize logistically and financially due to great time and resources requirements. In the recent decades, soil ecologists and model developers have made great strides in synthesizing quantitative knowledge from many different experiments into a mathematical program that can be used for ecological predictions and experimentation. In the next chapter, we will explore the use of DayCent model to make long-term predictions about the post-application effects of these same organic fertilizers.

Chapter 4 – Applying organic fertilizers under climate change: DayCent predictions of crop yield, soil carbon and nitrous oxide emission (2018 – 2070) in two corn-soybean fields in eastern Canada

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Abstract

Organic fertilizers produced from municipal organic waste can replace mineral fertilizers in crop production. However, organic fertilizers varied widely in properties and may trigger different agroecological responses e.g. changes in crop yield, SOC stock and soil N_2O emission. We examined the long-term (2018 – 2070) effect of applying three organic fertilizers biennially, namely composted food waste (compost), LysteGro biosolid slurry and liquid digestate (digestate), plus a mineral fertilizer control (NPK) on corn and soybean yields, SOC stock and soil N_2O emission under climate change scenarios (RCP4.5 and RCP8.5 radiative forcing) with the DayCent model. DayCent was parameterized with field data collected in two corn-soybean fields (Lods and Elora) in eastern Canada. At both sites, DayCent predicted digestate application would produce the highest corn silage C yield (25 – 28 % higher than NPK) whereas compost would produce the highest soybean grain C yield (2.8 – 4.4% higher than NPK). Within the simulation period, all organic fertilizer mitigated climate change in terms of greenhouse gas intensity (GHGI, in t CO_2-eq/t C harvested) and compost application had the lowest GHGI before 2040 due to the quickest SOC accrual, although the mitigation diminished after SOC reached steady-state. Consistently higher GHGI were found in RCP8.5 than RCP4.5 particularly

for compost treatment due to a smaller SOC stock and higher N_2O emission. There were cross-fertilizer model biases particularly where the N_2O emissions with compost and digestate were over-estimated compared to measured calibration and validation datasets. However, this would reinforce compost application as the best mitigation option in the near-term and may indicate that digestate can be a relatively time-insensitive mitigation option that improve crop yields while lowering N_2O emissions.

4.1 Introduction

The use of organic fertilizers derived from municipal organic waste in croplands has gained tremendous interest in eastern Canada in order to divert organic waste from landfills (Bureau du vérificateur général de la Ville de Montréal, 2018). Organic fertilizers can sustain crop production via the provision of plant nutrients and build soil organic carbon (SOC) stock via direct organic matter (OM) addition and promoting crop growth. Numerous studies found that organic fertilizers are capable of producing economical crop yields similar to mineral fertilizers (Diacono & Montemurro, 2011). Tian et al. (2009) found that applying municipal biosolids annually resulted in SOC accrual rates ranging from 0.54 to 3.05 t C/ha/yr from 41 long-term field trials, compared to -0.07 to 0.17 t C/ha/yr under mineral fertilization. However, the exact realizable SOC accrual is hard to predict as it depends on numerous factors such as organic fertilizer chemical properties (e.g. OM content), crop growth (C inputs via roots and residues), as well as soil conditions and climate. In particular, many studies projected that a warmer world would promote SOC loss (Davidson & Janssen, 2006; Wiesmeier et al., 2016). Therefore, the effect of climate change has to be considered when predicting future SOC stocks.

Moreover, the N in organic fertilizers can trigger soil N_2O emission under the right chemical forms and soil conditions. Nitrous oxide emission is a major radiative offset of soil carbon sequestration as SOC-rich soils tend to emit more N_2O (Ding et al., 2013; Gu et al, 2017; Li et al., 2005). Besides, organic fertilizers such as animal slurries and biosolids are rich in mineral N and labile C, which can fuel nitrifier and heterotrophic denitrification and promote N_2O emission (Thangarajan et.al, 2013; Velthof et al., 2003). In a meta-analysis, the emission factor of slurries averaged 1.12%, which is higher than the 1% emission factor of IPCC Tier I method (Charles et al., 2017). In addition, there is evidence that N_2O emission will increase in a warmer world (Dobbie & Smith, 2001; Schaufler et al, 2010; Thangarajan et al., 2013). Thus, it is necessary to consider N_2O emission from organic fertilizer application.

These local agroecological outcomes (crop yield, SOC accrual and N_2O emission) can be combined into a more comprehensive climate change impact indicator. Greenhouse gas intensity (GHGI), calculated as the global warming potential (GWP) from changes in SOC stock, N_2O and CH_4 emission divided by crop yield, is a measure to assess yield-scaled climate change impact (Mukumbuta et al., 2017; Sainju, 2016; Yang et al., 2015; Zhang et al., 2016). It is a “global” climate change impact indicator, because it factors in averted land clearance under higher yields (Burney et al., 2010; Van Groningen et al., 2010). Methane emission was not considered in this study due to its very low contribution to GWP budget in annual crop fields of eastern Canada (Gregorich et al., 2005).

Predictions about agroecological outcomes can be made by process-based models, such as DayCent, which simulates C and N flows among plants, soils and the atmosphere in terrestrial ecosystems at daily time-steps (Del Grosso et al., 2012). In Canadian agroecosystems, DayCent was validated with long-term crop yield (Chang et al., 2013; Guest et al., 2017; Sansoulet et al.,

2014) and SOC data (Congreves et al., 2015; Smith et al. 2012). It has also been used for simulating N_2O emissions, albeit not perfectly. However, this can potentially be rectified by model parameterization (Grant et al., 2016; Necpálová et al., 2015; Smith et al., 2008). DayCent was parameterized with CO_2 fertilization experiments in its past development and together with the simulation of soil temperature and moisture dynamics and agronomic practices (e.g., fertilizer application, tillage, crop rotations), it allows users to evaluate the effectiveness of management practices under different climate change scenarios (De Gryze et al., 2011; Weiler et al., 2017). However, the effect of different organic fertilizers on agroecological outcomes is seldom evaluated with DayCent. A main reason is that organic fertilizer is applied via the OMAD function in DayCent, where a pure OM is assumed to be applied. This OM is also coarsely characterized, consisting only of C:N ratio and lignin content to determine its subsequent fate. These two factors represent only a very limited aspect of organic fertilizer as many physical (e.g. solid vs viscous slurry vs. liquid) and chemical properties (e.g. amount of dissolved organics and mineral nutrients) are ignored. This may present a problem while simulating GHG fluxes because it is crucial to know how much soluble C and N is available over time in order to precisely calculate GHG production (Frolking et al., 1998). This study thus used a novel approach to represent a more complete profile of organic fertilizer physicochemical properties using multiple DayCent functions (see Materials and Methods).

In this study, we predicted the effect of three organic fertilizers on two biennially-fertilized corn-soybean fields in eastern Canada from 2018 – 2070 using DayCent. The fertilizers were aerobically-composted food waste (compost), LysteGro biosolid slurry and liquid fraction of anaerobic digestate (digestate). Climate scenarios produced from the 5th generation Canadian regional climate model (CRCM5) forced by RCP4.5 and RCP8.5 were incorporated to examine the long-term agroecological outcomes under different climate change intensities. As the three fertilizers bear distinct properties, they were expected to produce different agroecological outcomes. We hypothesized that compost application would accrue the most SOC due to its highest organic C content. This would in turn promote higher crop yields and N_2O emission farther in time. LysteGro and digestate would produce higher crop yields and N_2O emissions in the near-term because they contain more mineral N and labile OM (Flavel & Murphy, 2006). Finally, we expect greater crop yields and N_2O emission but less SOC accrual in a warmer and CO_2-rich world (Jastrow et al., 2005; Kimball et al, 2002; Van Groenigen et al., 2011).

4.2 Materials and Methods

4.2.1 Study sites and data requirement

The data in this study came from two field experiments (from 2018 – 2019), one located at Emile A. Lods Agronomy Research Centre (Lods) in Ste-Anne-de-Bellevue, Quebec (latitude: 45°25'N; longitude: 73°55'W; 39 m elevation) and the other located at Elora Research Station (Elora) in Elora, Ontario (latitude: 43º29'N; longitude: 80º25'W; 376 m elevation). The initial soil properties, experiment treatments and data collection method were previously described in Chapter 3. Briefly, corn (2018) and soybean (2019) were grown and the fields were fertilized in the corn season with four fertilizer treatments in quadruplicates (compost, LysteGro, digestate, plus a mineral fertilizer control (NPK) consisting of urea, triple superphosphate and potassium chloride). Fertilizer application rates were based on the agronomic recommendation by the Ministry of Agriculture, Fisheries and Food of Québec and the Ontario Ministry of Agriculture, Food and Rural Affairs. Physicochemical properties and application rates of the fertilizers can be found in Table 3-1.

The input dataset for DayCent consists of the site-level daily maximum and minimum air temperature, daily precipitation, a schedule of the agronomic management (e.g. planting, fertilization, plowing) and initial soil properties such as soil texture (% clay and sand), bulk density and pH. Soil hydraulic parameters i.e. field capacities, wilting points and saturated hydraulic conductivities at different depths were estimated using the soil water characteristic model developed by Saxton & Rawls (2006) with soil texture, OM content and bulk density. All the aforementioned soil properties except OM content (i.e. SOC) were assumed to be constant throughout the simulation. In addition, we used harvested aboveground biomass C and N, SOC stock (0 – 20 cm) measured before planting and harvest, volumetric soil water content (0 – 15 cm) (VSWC), soil temperature (0 – 10 cm) and soil N_2O fluxes measured approximately biweekly to calibrate and validate the DayCent model. The details of model setup, calibration and validation can be found in the subsequent sections.

4.2.2 DayCent modeling

4.2.2.1 Model description

We used the DayCent model (DailyDayCent vDec2016) that simulates crop growth as a function of crop cultivar-specific production potential, solar radiation, temperature, soil moisture and nutrient uptake (determined from the soil nutrient concentration and root biomass) as well as atmospheric CO_2 level (Del Grosso et al. 2008). Dead organic matter (senescent crop biomass, fertilizer OM) is partitioned into recalcitrant structural and labile metabolic litter pools based on its C:N ratio and lignin content (Figure 4-1). Decomposition of litter results in respiration CO_2 loss and further partitioning into three SOC pools (active, slow and passive) controlled by litter type and soil texture, with each SOC pool having a specific decomposition rate constant and a defined range of C:N ratio. Decomposition rate is further modified by field cultivation events, soil temperature and moisture (Del Grosso et al., 2008; Del Grosso et al., 2011). Decomposition rate is also used as a proxy for labile C availability that controls the rate of denitrification in N_2O production model. Together with NO_3^- availability, gas diffusivity (based on soil texture and bulk density), soil water and temperature status, the rate of N_2O production from denitrification is determined (Figure 4-1). DayCent also simulates N_2O loss from nitrification with NH_4^+ as the precursor. Nitrification rate is determined by soil NH_4^+, water content, temperature and pH. The mole fraction of N_2O in nitrification N gas loss is controlled by gas diffusivity and a precipitation pulse multiplier (Figure 4-1) (Parton et al., 2001).

4.2.2.2 Model setup and initialization

To run DayCent, basic input parameters were set up as NELEM = 2 and EDEPTH = 0.2 for simulating C and N dynamics at 20 cm depth, and IDEF = 3 for adopting a unimodal decomposition rate-soil water function. Then, model initialization was carried out to match the measured baseline soil conditions at both sites. Briefly, the initial ecosystem was set as a forest (i.e. IVAUTO = 3) for native vegetation simulations of about 3000 years to achieve equilibrium between SOC pools. The forest was adopted from the Harvard Forest in Savage et al. (2013), a temperate mixed hardwood forest in close geographical location. Additional forest fire events were included at Elora because of the warmer and dryer summer than Lods. The forest clearance and first cultivation at Lods was set at 1750 A.D. based on the history of agriculture along St. Lawrence River (Jones, 1942) whereas at Elora it was set at 1860 A.D. based on information provided by researchers at Elora. At Lods, the farther past consisted of low-yielding varieties of crops such as wheat, pea, barley and potato managed with organic methods i.e. manure application and fallow (Jones, 1942) whereas the more recent history was simulated as having a corn-soybean-spring wheat rotation with urea fertilization and heavier field cultivation types (influenced by post-World War II Green Revolution). The early management history at Elora consisted of pasture, followed by low-yielding varieties of wheat, soybean and barley with manure application. More recent history was simulated as having continuous corn and corn-soybean-spring wheat rotation with urea fertilization (Dayamurthi, 1997). Matching initial soil conditions was achieved by adjusting the productivity of the forest "prdx(2)" as well as field cultivation decomposition parameters "clteff" (see Appendix 5 for details of history simulation). The initialization yielded fractions of total SOC as approximately 61% passive SOC, 30% slow SOC and 1.8% active SOC at Lods and 72% passive SOC, 24% slow SOC and 1.5% active SOC at Elora. The ratio of active SOC simulated in 2017 (52 vs. 79 g C/m^2 at Lods and Elora respectively) corroborates with the microbial biomass C (63 vs. 84 g C/m^2 at Lods and Elora respectively) measured in soil samples (0 – 20 cm) collected in October, 2018.

After the history simulations, the current management schedule in 2018 – 19 (Appendix 2) were used in calibration and validation runs. The corn and soybean grown were set up as C6 and SYBN2 in DayCent. At Lods, soybean harvest was followed with a round of Fall-applied organic fertilizers, which was exclusively included in the calibration run to provide extra data for model calibration (for all future simulations, it will remain as biennially-fertilized corn-soybean rotation). Since DayCent represents OM addition (OMAD) as adding "pure" OM but in reality each organic fertilizer contains both OM and inorganic nutrients, each organic fertilizer application was thus modeled as an OMAD event, a FERT event for adding mineral N and an additional IRRI event for water addition through liquid fertilizers, all on the same day. Parameters related to the added fertilizers were set up according to the known properties and application rates in Table 3-1, except for lignin content which we did not measure and thus incorporated into the subsequent model calibration exercise. The properties and composition of the fertilizers are assumed to be constant throughout any simulation in this study.

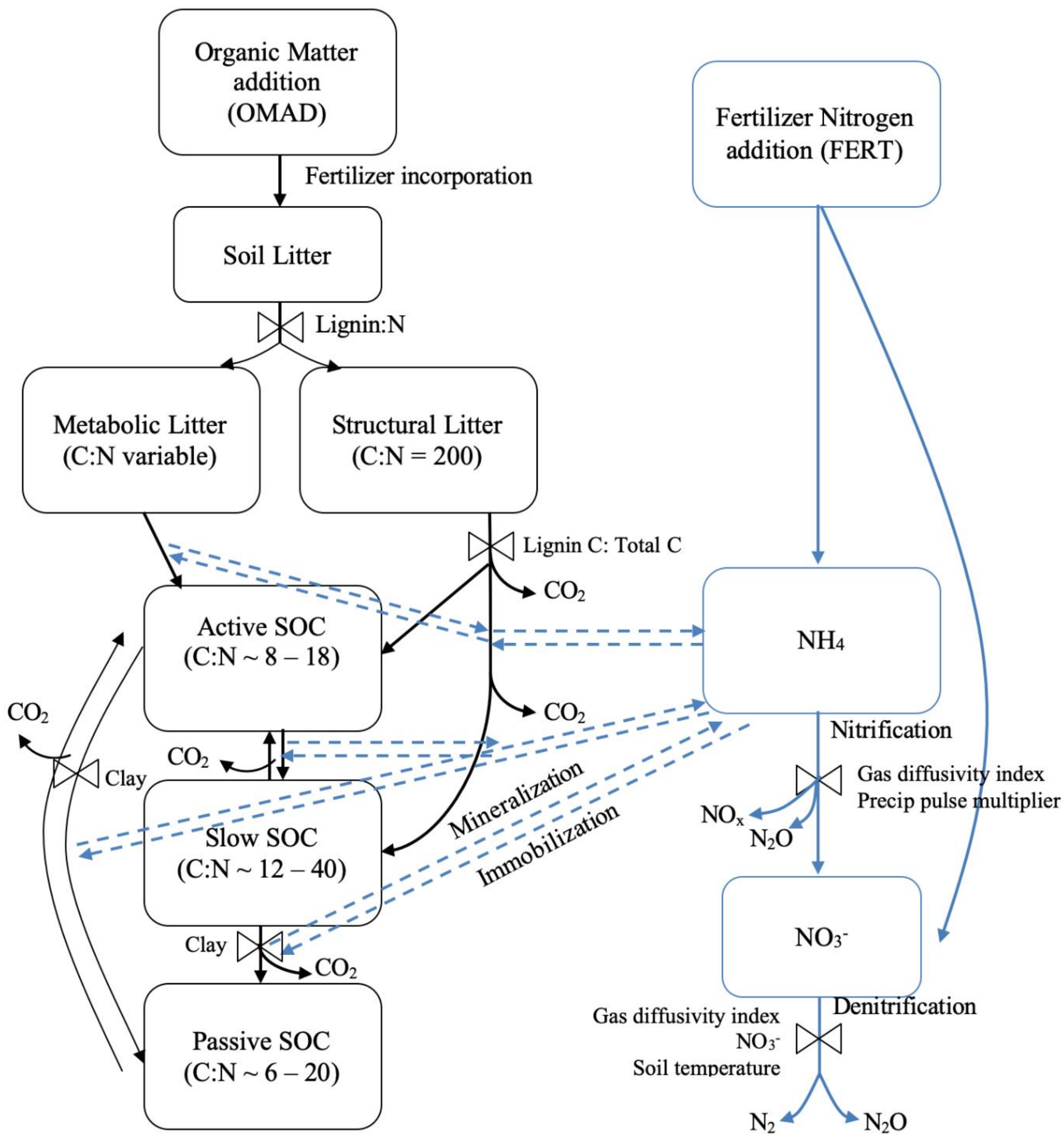


Figure 4-1. An overview of the DayCent conceptual model linking organic fertilizer application (via OMAD and FERT) to different SOC pools and N_2O emission. Black arrows indicate the flows of OM. Blue arrows indicate the flows of mineral-N. Dashed arrows are the flows of N via mineralization or immobilization.

4.2.2.3 Model calibration

DayCent was calibrated according to the recommended workflow in Del Grosso et al. (2011). The calibration problem was divided into a sequential multi-stage optimization problem, the first step calibrates the model to improve fit to aboveground crop biomass C and N as well as measured VSWC. The second step calibrated the model with measured SOC stocks and soil temperature. The third step improved the fit of model to measured soil N_2O fluxes at individual sampling dates as well as monthly cumulated emission of N_2O (linearly-interpolated monthly emissions were only compiled in months with at least 4 measured flux data points).

The calibration exercise was accomplished with algorithmic inverse modelling using PEST (version 13.0), which minimizes a weighted least-square function of the residuals of simulated vs. measured outputs by the Gauss-Levenberg-Marquardt gradient search algorithm (Doherty and Hunt, 2010). The approach was adopted from Necpálová et al. (2015, 2018) and was used in several other studies such as Rafique et al. (2014, 2015) and Gaillard et al. (2018). The model input parameters to be calibrated (Appendix 6) was decided based on which observation groups have poor default fit and the parameters contributing to it, as well as a consideration of what other studies have used. The whole parameter set to be calibrated were divided into two classes, first is a “site-specific” class that was calibrated site-by-site as some parameters are circumstantial. This includes “characteristics” parameters controlling crop and soil water outputs i.e. the first stage of calibration. For instance, crop productivity index “prdx(1)” that encompasses the genetic potential of the specific cultivar and how far it is from the optimal planting density was included in this class. The second class contains generalizable, “effect” parameters in the second and third stages of calibration (SOC, soil temperature, N_2O

fluxes) and are related to various effects on decomposition rate (e.g. cultivation and temperature), the availability of NO_3^- and the sensitivity of N_2O production to different soil environmental variables. The second-class parameters were calibrated by the dataset from Lods only as the dataset is more complete and measurements were more frequent.

To avoid overfitting to the calibration dataset, which has its own uncertainties and limits of measurement scale, certain confidence is still placed on the default parameters. A conservative approach with smaller parameter ranges was used (generally about ± 20% of the initial value), compared to those used in other studies e.g. Rafique at al. (2014) and Necpálová et al. (2015). We argued that large parameter ranges, which allows a re-calculation of the relative importance of model processes, is not suitable in a non-big data setting. Only uncertain characteristics parameters such as "prdx(1)", as well as parameters related to N_2O emission were given wider ranges because of the tendency of the default model to underestimate N_2O emission by at least 5 times. This conservative approach forces PEST to make fine adjustment to multiple parameters, which mimics what happens in nature i.e. agroecological outcomes are, theoretically-speaking, the culmination of a multitude of interacting processes and characteristics. This avoids the pitfall of adjusting a few very sensitive parameters for a "quick fix", whether manually or algorithmically, that may lead to ungeneralizable values fitting to a very limited circumstance.

Second, a "feedback mechanism" was incorporated into calibration to avoid tradeoffs in fitting a certain category of variables at the cost of fitting another variable more poorly (Del Grosso et al., 2008). In PEST, this was realized by giving large objective function weight (100) to variable category of primary focus at the current stage (e.g. crop C and N and VSWC at the first stage) but still giving weight (10) to other variable categories. Moreover, observations from different fertilizer treatments were generally given equal weights in the PEST exercise. A few candidate calibrated parameter sets were produced by slightly different weighting strategies addressing certain observations that had unreasonably poor fit and the one that has the best performance in validation was kept at the end. The Lods-calibrated parameters from stage 2 and 3 were imported for simulations at Elora except for the site-specific stage 1 crop and water parameters which were recalibrated. Lastly, as preliminary future simulations resulted in unrealistically high soybean yield at Elora in some occasions (> 5 t C/ha grain yield ~ 10 t/ha grain biomass yield), FSDETH parameters were activated to simulate shading effect to increase shoot death by adjusting FSDETH(3) to 0.2 and FSDETH(4) to 600 g C/m^2, based on the maximum attainable soybean yield reported in Grassini et al. 2014. The calibration workflow can be visualized in Figure 4-2.

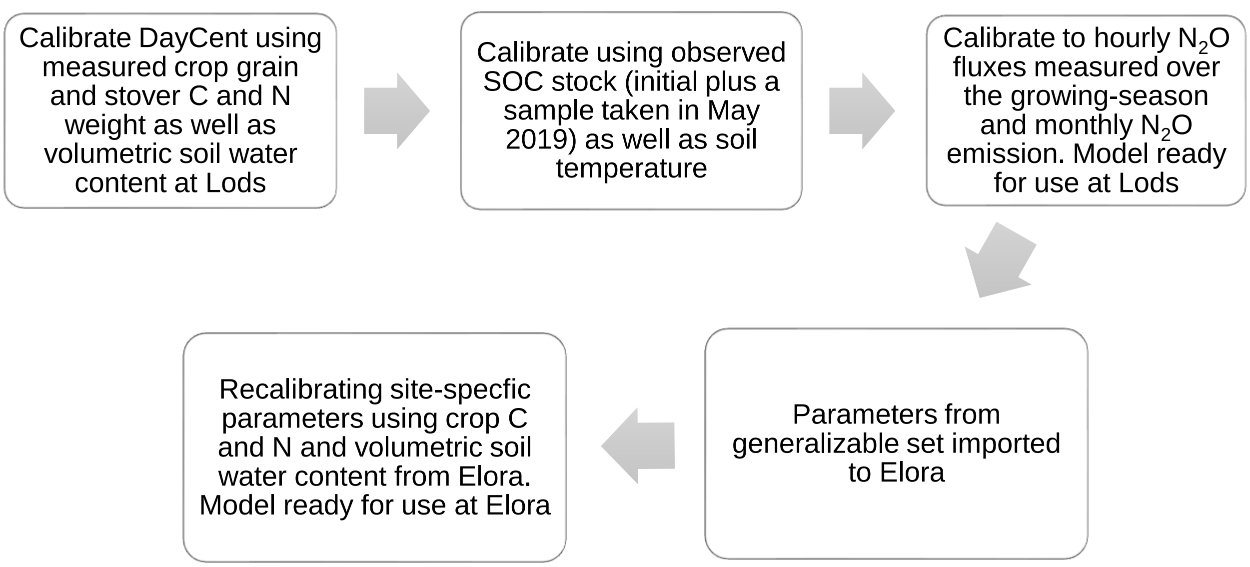


Figure 4-2. Calibration workflow for DayCent (DailyDayCent vDec2016) using PEST v13.0.

4.2.2.4 Model performance evaluation and validation

Performance of the calibrated model was evaluated with measured datasets at Lods and Elora (as validation) (n.b. each measured value is the mean of 4 replicates). The data categories to be evaluated were, aboveground biomass production in weight of C and N, VSWC, soil temperature, SOC stock, individual N_2O fluxes, monthly N_2O emissions (calculated from linear interpolation on months with at least 4 data points). Elora had missing measurements of C and N content for its crop tissues so the grain C and N content measured at Lods were used directly to calculate grain C and N weight at Elora. Stover generally has a higher variability of C and N content influenced by factors such as physiological age (Hanway, 1962), so stover C and N were not evaluated at Elora. For observations that occurred only a few times (crop yield and SOC), percentage difference (using measured values as denominator) between simulated and measured values were used to evaluate the fit. Coefficient of determination (r^2) and relative root mean squared error (rRMSE) were used to evaluate all other variables that were regularly measured. Individual N_2O fluxes were also plotted in time-series to facilitate graphical evaluation due to possible shift of timing of emission peaks or troughs.

$$rRMSE = \frac{\sqrt{\frac{1}{N}\sum_{i=1}^{N}(y_i - y'_i)^2}}{|\bar{Y}|}$$

Where N is the number of observations, $|\bar{Y}|$ is the grand mean of the absolute value of measured variables, y_i is individual measured value and y'_i is the corresponding simulated value.

4.2.3 Future simulation and climate scenarios

Future simulations (2020 – 2070) were directly extended from 2018 – 2019 current runs, rather than assuming a new start with equalized SOC stocks across treatments. The future management scenario was modified from the 2018-19 management, and consisted of disk harrow to prepare seed bed on 10th May, fertilizer application and incorporation on 14th May (corn season only), planting on 15th May for both crops and moldboard plow two weeks after harvest to incorporate residues. Harvesting of silage corn (grain + 90% of stover biomass removed) occurred on 15th Sept and soybean grain-only harvest was scheduled on 1st Oct.

Future climate scenarios of daily maximum and minimum temperature and daily precipitation were produced by CRCM5 (downscaled from CanESM2) under the radiative forcing of RCP4.5 and RCP8.5, which was further bias-corrected with measured site-level climate data (1981 – 2010) via quantile-mapping (Grenier et al., 2015). Daily mean temperature (for producing climate summary) was calculated by averaging daily maximum and minimum temperature (Grenier et al., 2015). The CO_2 fertilization scenarios were constructed from annual pCO_2 data in the RCP database (Meinshausen et al., 2011). However, reading in exact annual CO_2 ppm values is not supported in DayCent. Instead, DayCent has a built-in function of linear pCO_2 ramp up to incorporate CO_2 fertilization of free-air CO_2 experiments. We found that the CO_2 trajectories from both RCPs were very close to a linear CO_2 increase ($r^2 > 0.98$) so this was considered a good approximation. A base pCO_2 is set at 416 ppm in 2020, with final pCO_2 of 524 ppm for RCP4.5 and 677 ppm for RCP8.5 in 2070.

Table 4-1. Climate statistics computed from future climate scenarios of 2020 – 2070 generated by the 5th generation Canadian Regional Climate Model (CRCM5) under RCP4.5 or RCP8.5; measured weather data in 2018 and 2019 (recent) and from 1981 – 2010 (historical) from weather stations near Lods and Elora. Values for 2020 – 2070 climate statistics are in the form: RCP4.5; RCP8.5.

	Mean annual temperature (ºC) RCP4.5; RCP8.5	***Mean growing-season temperature (ºC)***	***Mean annual total precipitation (mm)***	***Mean growing-season total precipitation (mm)***	****Interannual variability of growing-season precipitation (mm)***
Lods (2020 – 2070)	8.86; 9.33	18.1; 18.8	1052; 1052	534; 533	116; 113
Lods (2018 - 2019)	6.53	16.6	971	506	NA
Lods (1981 – 2010)	6.61	15.9	1029	542	111
Elora (2020 – 2070)	8.87; 9.34	17.1; 17.8	911; 911	485; 483	107; 103
Elora (2018 – 2019)	6.48	15.6	985	448	NA
Elora (1981 – 2010)	6.62	15.0	893	495	118

* Interannual variability of growing-season precipitation is the standard deviation of annual growing-season precipitation. It was not calculated for 2018 – 2019 since there is only 2 years.

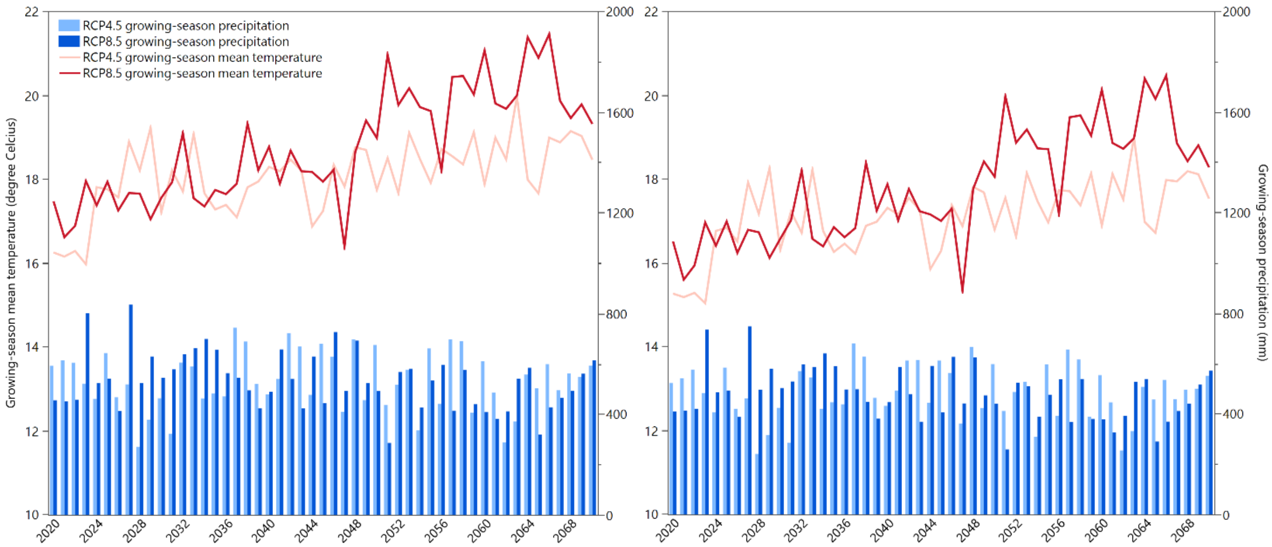


Figure 4-3. Simulated growing-season mean temperature and precipitation from 2020 – 2070 at Lods (left) and Elora (right). The data from were generated by the 5th generation Canadian Regional Climate Model (CRCM5) under the radiative forcing of RCP4.5 and RCP8.5.

4.2.4 Model outputs analysis

Basic model outputs (crop yields, SOC stock and N_2O emission) were analyzed descriptively by their averages and trends. To compute GHGI, first we calculated the on-site GHG balance by combining the GWP associated with changes in SOC stocks and N_2O emission together (see equation below). We considered the climate change impact of changes in SOC by assuming that an increase (or decrease) in the slow and passive SOC pools (som2c(2) and som3c in DayCent, the more stabilized SOC pools) resulted in avoided (or promoted) CO_2 emission (Baldock et al., 2010; Lal, 2013). These pools were chosen because they are more resistant to changing management practice compared to just total SOC (De Gryze et al., 2011) and that they are more closely associated with actual carbon sequestration, which is infeasible to estimate within the scope of this study (Powlson et al., 2011). The total N_2O emitted from 2018 – 2070 were converted to CO_2-equivalent (CO_2-eq) by considering a 265-time warming potential of 1 kg of N_2O vs. 1 kg of CO_2 over 100 years (IPCC, 2014). The approach is similar to Gu et al. (2017) and Wang et al. (2011), although they used total SOC stock instead of stable SOC stock.

$$\textit{On-site GHG balance} = (-44/12 \times \Delta \textit{ stable SOC}) + (265 \times 44/28 \times N_2O\textit{-N emission})$$

Where *on-site GHG balance* is in t CO2-eq/ha, *stable SOC* is in t C/ha, and *N_2O-N emission* is in t N/ha. The on-site GHG balance was converted to GHGI (in t CO2-eq/t C harvested) by dividing it with the total crop C yield (silage C and grain C in corn and soybean year respectively). The on-site GHG balance and GHGI of organic fertilizer treatments were compared to NPK control in order to infer post-application GHG mitigation relative to the scenario of applying mineral fertilizer.

4.3 Results

4.3.1 DayCent model performance

The default DayCent consistently produced much lower yields than measured yields. The greatest improvement was seen for crop variables at Lods except that corn stover N were underestimated (-22 to -44%). At Elora, the calibrated model was not able to simulate the high corn grain N yield measured (-25 to -33%), but otherwise it had good performance (±10%) for other grain variables (Table 4-2). At both sites, grain yields (both corn and soybean) were slightly underestimated for NPK relative to other fertilizer treatments.

Soil temperature and VSWC were simulated reasonably well in both default (rRMSE < 0.475) and calibrated model (rRMSE < 0.458). However, there was a slight tendency for DayCent to over-predict high VSWC and under-predict low soil temperature (Figure 4-4).

The default model consistently underestimated SOC stocks, which was improved significantly in the calibrated model in all treatments (Table 4-2). Calibration decreased the tendency of SOC depletion but it kept the SOC ranking of fertilizer treatments as produced by the default model.

The calibrated model produced better fits of N_2O emission all-around as the default model severely underestimated the cumulative growing-season N_2O emission by up to about 20 times at Lods and 5 times at Elora (Table 4-2, Figure 4-5). At both sites, the calibrated model allocated most N_2O emissions to the corn season, resulting in reasonable simulated growing-season N_2O emission in 2018 but underestimated that in 2019 at Lods, while having an over-estimation of N_2O emission in 2018 and reasonable ranges in 2019 at Elora (Table 4-2). However, widespread model bias among fertilizer treatments can be seen. The calibrated model generally produced the best fit for LysteGro treatment (-48 to +3.7 %) except for an over-estimation at Elora in 2019 where DayCent cannot generate negative N_2O fluxes to match the measured negative fluxes (Figure 4-5C). The calibrated model moderately over-estimated N_2O emission (+48 to 185%) for compost and severely over-estimated (+134 to 186%) for digestate in the corn season (Table 4-3). The over-estimation occurred mainly during measured high emission periods in compost and low emission periods in digestate (Figure 4-5A & C). In contrast, there were notable underestimation of N_2O emission for NPK treatment particularly at Lods (-46 to -71%), where split application was practiced (Table 4-3, Figure 4-5). The N_2O time-series plots showed fairly good temporal fluxes agreements between measured and simulated emission peaks (off by only a few days).

Table 4-2. Measured crop biomass variables and SOC stock (0 – 20 cm) in 2018 corn and soybean seasons at Lods and Elora, compared to simulated values by default and calibrated DayCent (vDec2016). In brackets are percentage differences compared to measured values.

	Lods			Elora (validation)		
2018	Measured	Default	Calibrated	Measured	Default	Calibrated
Corn grain C (g/m²)						
NPK	397	160 (-60%)	353 (-11%)	502	110 (-78%)	486 (-3.0%)
Compost	345	163 (-53%)	350 (+1.4%)	440	104 (-76%)	471 (+7.0%)
LysteGro	368	160 (-56%)	354 (-3.8%)	486	109 (-78%)	503 (+3.4%)
Digestate	346	163 (-53%)	358 (+3.6%)	565	121 (-79%)	545 (-3.6%)
Corn grain N						
NPK	14.8	7.3 (-51%)	12.7 (-14%)	18.8	6.0 (-68%)	13.5 (-28%)
Compost	13.3	7.3 (-45%)	12.5 (-5.8%)	17.0	5.8 (-66%)	12.7 (-25%)
LysteGro	13.9	7.26 (-48%)	12.8 (-7.9%)	18.4	6.0 (-68%)	13.8 (-25%)
Digestate	13.5	7.3 (-46%)	12.9 (-4.4%)	22.1	6.3 (-72%)	14.8 (-33%)
Corn stover C						
NPK	189	96.4 (-49%)	187 (-1.2%)	NA*		
Compost	205	98.3 (-52%)	185 (-9.8%)	NA		
LysteGro	197	96.4 (-51%)	187 (-5.1%)	NA		
Digestate	159	97.9 (-38%)	189 (+19%)	NA		
Corn stover N						
NPK	5.8	1.8 (-68%)	3.2 (-44%)	NA		
Compost	5.4	1.8 (-66%)	3.2 (-41%)	NA		
LysteGro	4.1	1.8 (-56%)	3.2 (-22%)	NA		
Digestate	4.8	1.8 (-62%)	3.3 (-33%)	NA		
SOC May 2018 (g C/m²) †						
NPK	2867	2817	2838	5237	5300	5309
Compost	3004	2817	2838	4958	5300	5309
LysteGro	2740	2817	2838	5392	5300	5309
Digestate	2691	2817	2838	5276	5300	5309
SOC Oct 2018						
NPK	3099	2683 (-13%)	2824 (-8.9%)	5797	5245 (-9.5%)	5384 (-7.1%)
Compost	3386	2800 (-21%)	2950 (-16%)	5712	5392 (-5.6%)	5540 (-3.0%)
LysteGro	3200	2729 (-15%)	2869 (-10%)	6139	5302 (-14%)	5442 (-11%)
Digestate	3069	2689 (-12%)	2834 (-7.7%)	5628	5258 (-6.6%)	5399 (-4.1%)

(Continue) 2019		Lods			Elora (validation)	
	Measured	Default	Calibrated	Measured	Default	Calibrated
Soybean grain C (g/m^2)						
NPK	114	46.4 (-59%)	106 (-7.2%)	273	64.1 (-77%)	252 (-7.7%)
Compost	108	52.6 (-51%)	110 (+2.0%)	259	68.4 (-74%)	257 (-0.7%)
LysteGro	94.0	50.2 (-47%)	109 (+16%)	251	66.5 (-74%)	254 (+1.3%)
Digestate	106	48.1 (-55%)	108 (+1.5%)	241	65.2 (-73%)	253 (+5.1%)
Soybean grain N						
NPK	12.6	7.8 (-38%)	11.0 (-13%)	30.3	9.5 (-69%)	24.2 (-20%)
Compost	11.2	8.7 (-22%)	11.4 (+1.5%)	27.0	10.1 (-63%)	24.8 (-8.0%)
LysteGro	9.97	8.4 (-16%)	11.3 (+13%)	26.7	9.8 (-63%)	24.5 (-8.2%)
Digestate	10.9	8.1 (-26%)	11.2 (+2.3%)	24.7	9.6 (-61%)	24.4 (-1.2%)
Soybean stover C						
NPK	101	93.8 (-7.1%)	85.7 (-15%)	NA*		
Compost	103	106 (+3.3%)	89.4 (-13%)	NA		
LysteGro	90.6	101 (+12%)	88.2 (-2.7%)	NA		
Digestate	99.1	97.3 (-1.8%)	87.2 (-12%)	NA		
Soybean stover N						
NPK	2.0	3.4 (+70%)	2.1 (+7.2%)	NA		
Compost	2.0	3.8 (+90%)	2.2 (+11%)	NA		
LysteGro	1.8	3.6 (+100%)	2.1 (+21%)	NA		
Digestate	1.8	3.5 (+94%)	2.2 (+20%)	NA		
SOC May 2019 (g C/m^2) †						
NPK	3229	2748 (-15%)	2836 (-12%)	NA		
Compost	3097	2895 (-6.5%)	3000 (-3.1%)	NA		
LysteGro	3146	2799 (-11%)	2892 (-8.1%)	NA		
Digestate	3193	2756 (-14%)	2846 (-11%)	NA		

* NA means measurement not available.

† This serves as the baseline SOC stock in DayCent. Since there was no significant difference of measured SOC among treatments ($p > 0.05$), we simulate them at the site-level and used the average of the four treatment SOC.

Table 4-3. Growing-season N_2O emission in 2018 – 2019 estimated from linear interpolation of N_2O fluxes measured at Lods and Elora, compared to simulated values by default and calibrated DayCent (vDec2016). In brackets are percentage differences compared to measured values.

	Lods			**Elora (validation)**		
	Measured	Default	Calibrated	Measured	Default	Calibrated
Growing-season N_2O emission (g N/ha)						
2018	**31 May to 25 Oct**			**29 June to 27 Oct ***		
NPK	2750	196 (-93%)	1479 (-46%)	209	142 (-32%)	406 (+94%)
Compost	1230	214 (-83%)	1815 (+48%)	205	127 (-38%)	585 (+185%)
LysteGro	1880	224 (-88%)	1950 (+3.7%)	634	144 (-77%)	509 (-20%)
Digestate	834	233 (-72%)	1954 (+134%)	213	168 (-21%)	609 (+186%)
2019	**29 April to 12 Sept**			**21 May to 4 Oct**		
NPK	2323	57.5 (-98%)	667 (-71%)	554	94.1 (-83%)	662 (+20%)
Compost	1742	84.6 (-95%)	862 (-55%)	532	119 (-78%)	849 (+60%)
LysteGro	1495	70.9 (-95%)	780 (-48%)	392	106 (-73%)	759 (+94%)
Digestate	2274	60.7 (-97%)	717 (-69%)	554	98.8 (-82%)	705 (+27%)

* This sampling period began a month after fertilizers were applied. Thus, the accuracy of simulated values was strongly affected by the correct simulation of the timing of fertilizer-induced emission spikes.

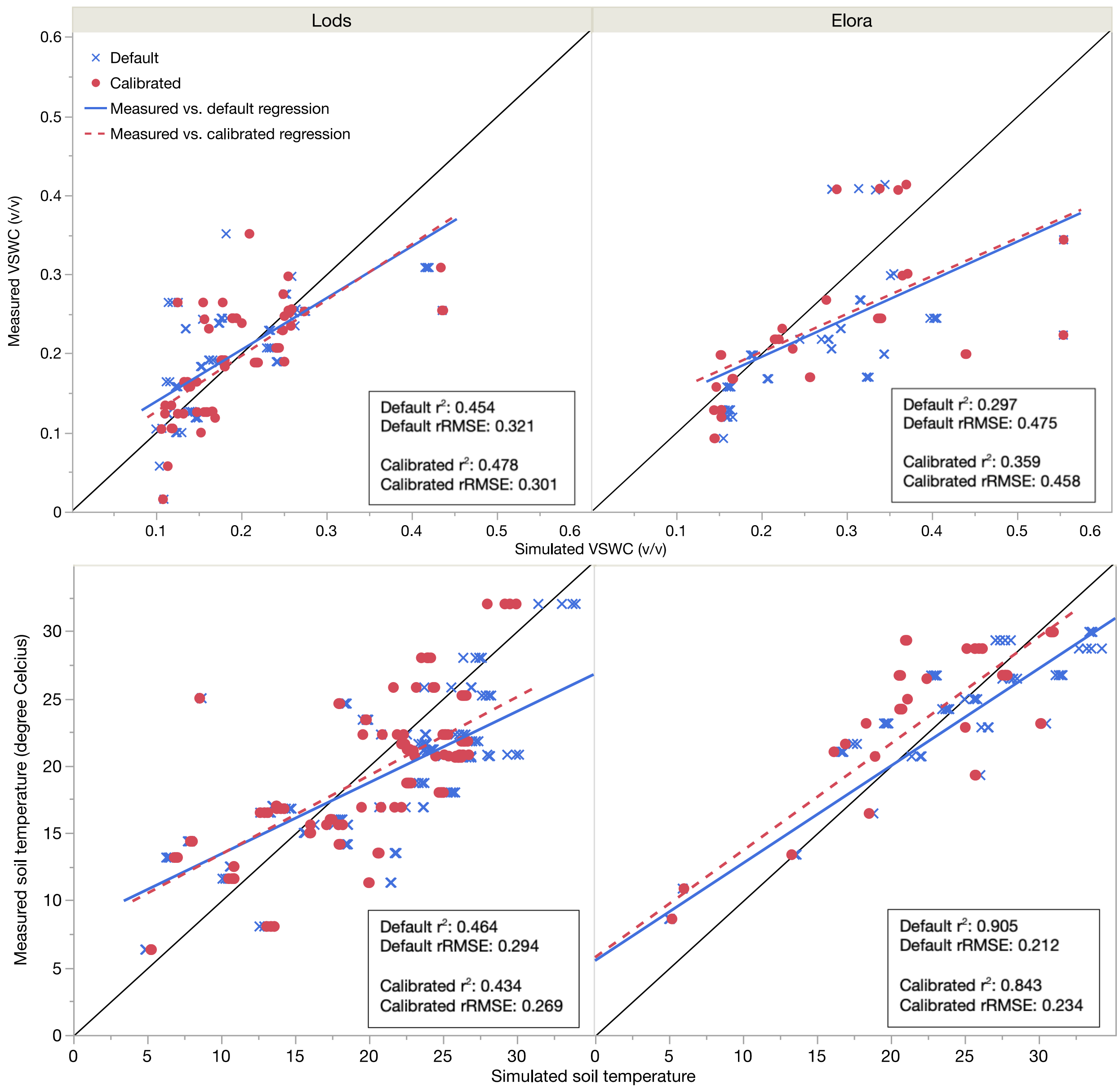


Figure 4-4. Measured vs. simulated VSWC (top) and soil temperature (bottom) for Lods (left) and Elora (right). Fertilizer treatments were pooled together since there were only very small difference in the simulated values.

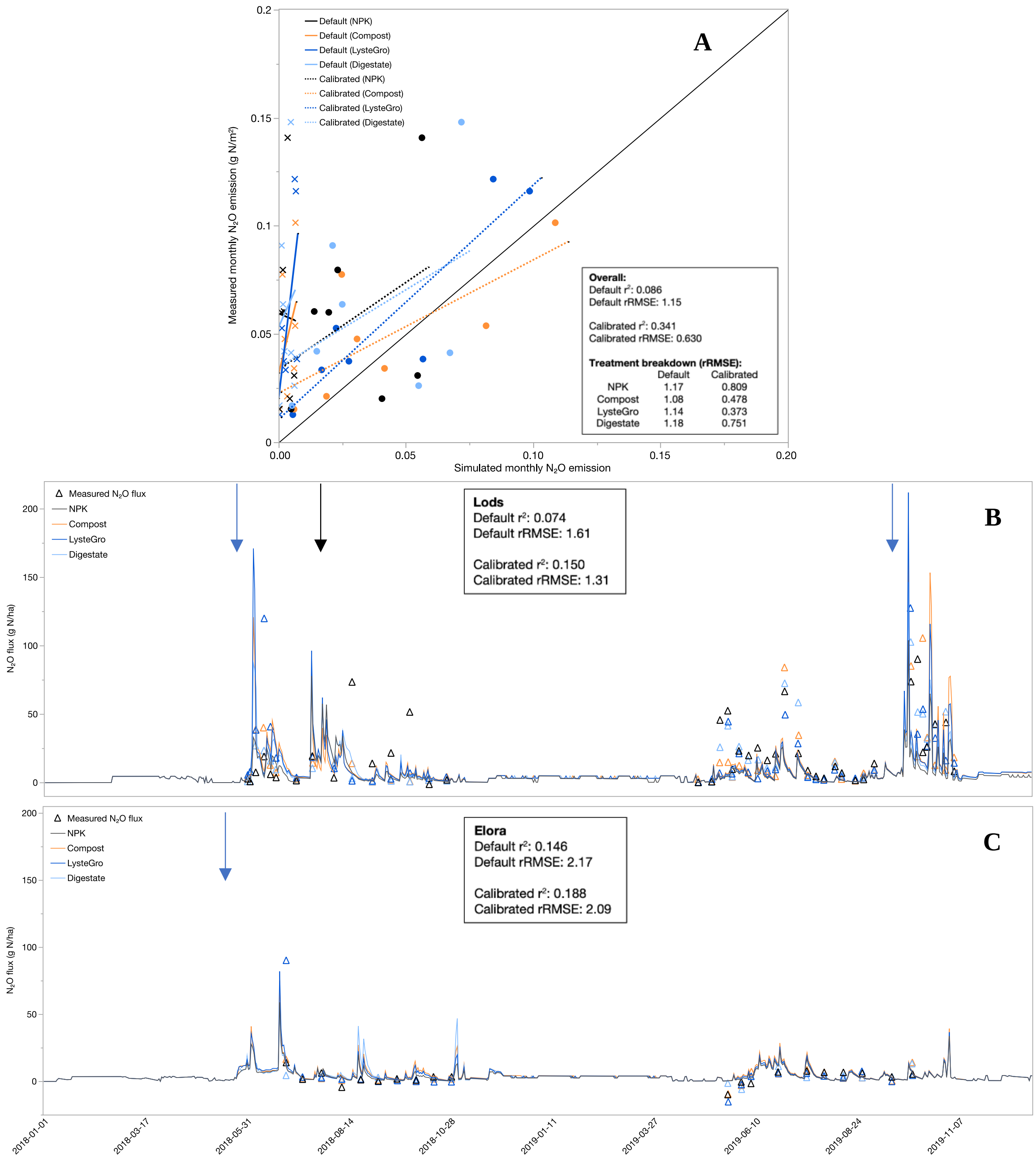


Figure 4-5. Measured vs. simulated A: monthly N_2O emission (Lods only since Elora did not have any month with at least four flux measurements), measured vs. simulated N_2O flux time-series at B: Lods and C: Elora (bottom). Blue arrows are the dates of fertilizer application. Black arrow is the second-split of urea application for NPK treatment at Lods.

4.3.2 Simulated long-term crop yield trends

DayCent predicted NPK treatment to produce the lowest corn silage yield (averaged 539 at Lods and 788 at Elora in g C/m^2) whereas digestate application would produce the highest yield consistently (25 and 28% higher than NPK at Lods and Elora respectively; Figure 4-6). LysteGro and compost application were predicted to produce similar corn yield but compost has the fastest rising yield trends at both sites which compensated the lower yield in earlier years. At both sites, the yield gaps between all organic fertilizers and NPK appear to widen further in time (Figure 4-6). DayCent predicted 1.9 and 4.5% higher corn yields under RCP8.5 than RCP4.5 at Lods and Elora respectively (calculation excluded the abnormally-low yield in 2028 under RCP4.5). The temporal trends of N yield (not shown) were similar to C except that the up-and-down fluctuation is smaller i.e. a higher C production resulted in proportionally smaller increase in N uptake (+1.0% N yield at Lods and +4.0% N yield at Elora under RCP8.5; Table 4-4).

Soybean grain yields were predicted to be very close between fertilizer treatments at both sites, with compost treatment maintained a small yield advantage consistently (4.4 and 2.8% higher grain C yield than NPK at Lods and Elora respectively; Figure 4-6). Similarly, DayCent predicted RCP8.5 would produce much higher soybean yields than RCP4.5 at Lods (+8.4% in C yield; +4.0% in N yield) but less so at Elora (+3.9%; +2.2%) where soybean yields are close to maximum attainable yield (Figure 4-6; Table 4-4).

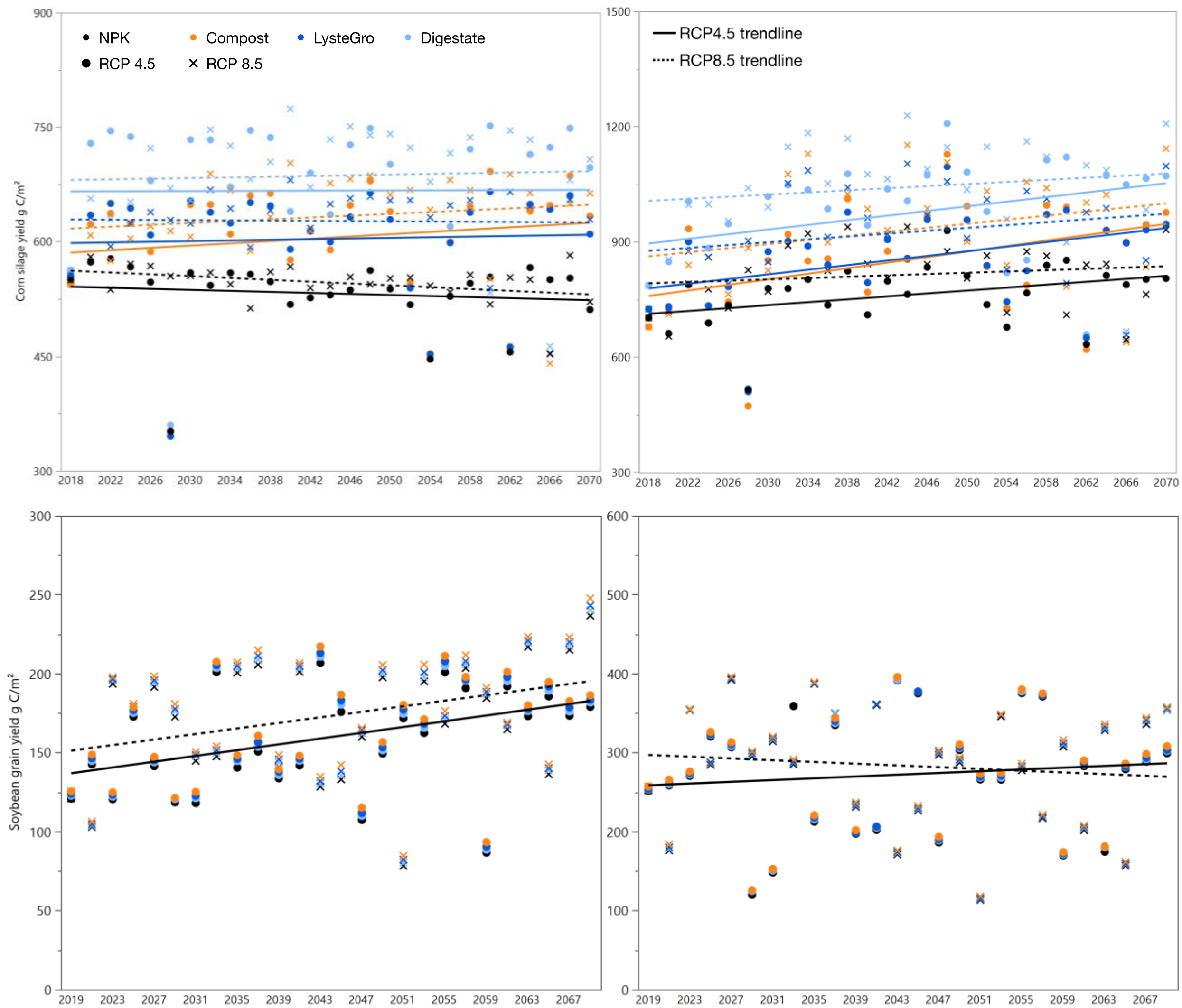


Figure 4-6. Corn silage yield (grain + 90% stover) (top) and soybean grain yield (bottom) at the biennially-fertilized corn-soybean fields at Lods (left) and Elora (right) simulated by DayCent (vDec 2016) under RCP4.5 and RCP8.5. For reference, the measured biomass C wt% in this study were 42% for corn silage and 50% for soybean grain.

Table 4-4. Predicted average crop N yield and annual N_2O emission from 2018 – 2070 at Lods and Elora by DayCent (vDec 2016). All values are in the form RCP4.5; RCP8.5.

	Lods		Elora	
g N/m²	Mean	Std. deviation*	Mean	Std. deviation
Corn silage N †	RCP4.5; RCP8.5			
NPK	15.2; 15.3	0.5; 0.5	16.1; 16.6	0.7; 0.9
Compost	16.6; 16.8	0.8; 0.8	16.7; 17.6	1.4; 1.6
LysteGro	16.8; 16.9	0.8; 0.7	17.4; 18.1	1.1; 1.3
Digestate	18.6; 18.6	1.6; 1.2	20.0; 20.7	1.5; 1.6
RCP average	16.8; 16.9	1.6; 1.4	17.5; 18.3	2.0; 2.0
Corn annual N_2O emission				
NPK	0.219; 0.227	0.022; 0.022	0.156; 0.167	0.024; 0.032
Compost	0.308; 0.326	0.044; 0.056	0.231; 0.249	0.033; 0.051
LysteGro	0.290; 0.303	0.037; 0.043	0.206; 0.221	0.030; 0.045
Digestate	0.275; 0.286	0.024; 0.026	0.193; 0.204	0.028; 0.038
RCP average	0.273; 0.286	0.047; 0.053	0.196; 0.210	0.041; 0.051
Soybean grain N				
NPK	13.4; 14.0	1.6; 1.8	25.2; 25.8	4.7; 4.7
Compost	14.0; 14.6	1.6; 1.9	26.0; 26.5	4.7; 4.6
LysteGro	13.8; 14.3	1.6; 1.8	25.6; 26.2	4.7; 4.7
Digestate	13.7; 14.2	1.6; 1.8	25.5; 26.0	4.7; 4.7
RCP average	13.7; 14.3	1.6; 1.9	25.6; 26.1	4.7; 4.7
Soybean annual N_2O emission				
NPK	0.134; 0.147	0.020; 0.026	0.134; 0.146	0.021; 0.029
Compost	0.179; 0.192	0.023; 0.030	0.172; 0.182	0.029; 0.034
LysteGro	0.156; 0.170	0.021; 0.028	0.155; 0.165	0.025; 0.032
Digestate	0.145; 0.159	0.021; 0.028	0.147; 0.158	0.023; 0.031
RCP average	0.153, 0.167	0.027; 0.024	0.152; 0.163	0.028; 0.034

* The standard deviation of annual crop yield or N_2O emission from 2018 – 2070

† For corn silage N yield under RCP4.5, the calculation excluded the year 2028 which was predicted to have abnormally-low yield due to a combination of low growing-season rainfall and high temperature leading to significant moisture stress.

4.3.3 Simulated long-term SOC stock trends

At both sites, a clear pattern emerged as compost >> LysteGro > digestate > NPK in terms of SOC accrual (Figure 4-7). At Lods, compost application was predicted to accrue 15% and 13% of initial SOC stock over the simulation period under RCP 4.5 and 8.5 respectively, while others incurred a net loss (Table 4-5). Overall, 7.4 and 6.9% of the total applied C from compost became SOC under RCP4.5 and RCP8.5. At Elora, the tendency of SOC accrual was noticeably higher. Compost treatment was predicted to gain 25 and 28% of initial SOC under RCP4.5 and RCP8.5 respectively (10 and 9.6% of total C applied). LysteGro plots also gained an appreciable amount of SOC, even higher than that of compost at Lods. At both sites, most SOC accrual occurred before 2040 and plateaued quickly afterwards (Figure 4-7; Table 4-5). The SOC curves under RCP8.5 at Elora however peaked at around 2037, followed by a declining trend instead of reaching steady-state (Figure 4-7). Under RCP8.5, compost and LysteGro had less final SOC than RCP4.5 whereas for NPK and digestate the difference was minimal.

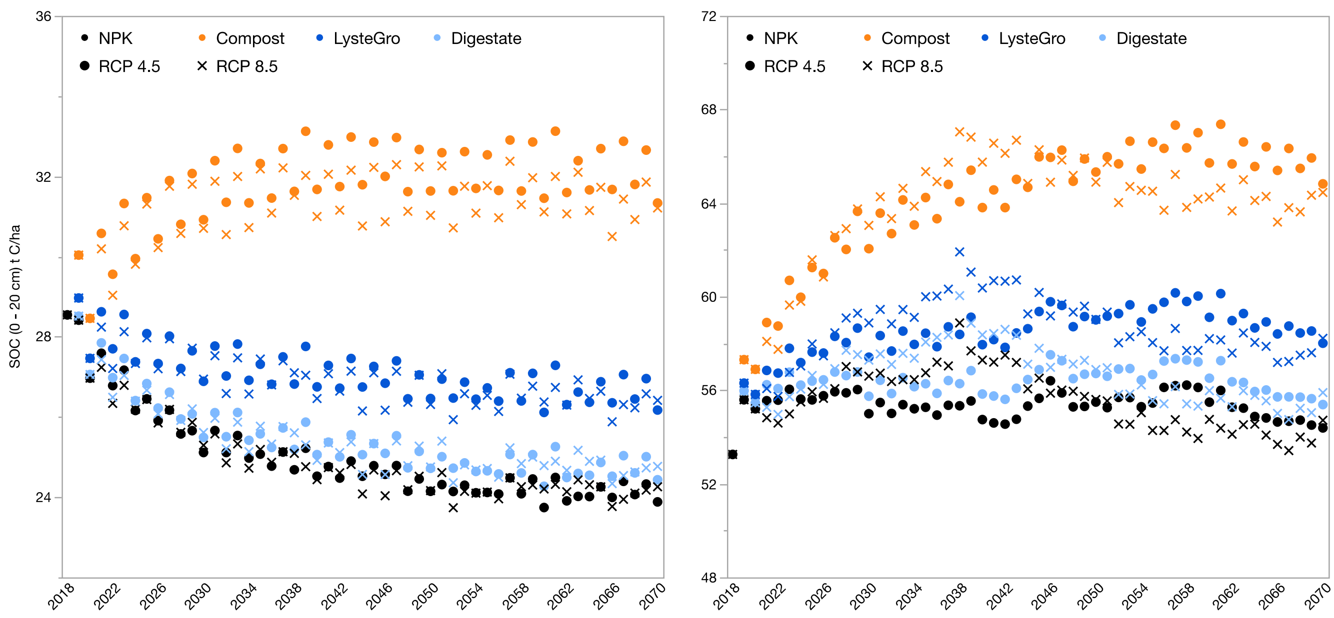


Figure 4-7. The annual SOC stock (0 – 20 cm) trends from 2018 – 2070 at Lods (left) and Elora (right) simulated by DayCent (vDec2016) under RCP4.5 and RCP8.5.

Table 4-5. Predicted rate of change of SOC stocks in t C/ha/yr in the period 2018 – 2040 and 2041 – 2070, final SOC stock, net SOC accrued/lost at the end of 2070, organic fertilizer-derived SOC (in t C/ha) by DayCent (vDec2016). The initial SOC stock were 28.4 at Lods and 53.1 t C/ha at Elora. All values are in the form RCP4.5; RCP8.5.

	2018 – 2040 SOC rate of change	*2041 – 2070 SOC rate of change*	**Final SOC stock**	*Net SOC Accrued/Lost*	*Organic fertilizer-derived SOC in 2070*	*final SOC minus organic fertilizer-derived SOC*
Lods	RCP4.5; RCP8.5					
NPK	-0.16; -0.16	-0.014; -0.010	24.5; 24.5	-4.02; -4.03	NA*	24.5;24.5
Compost	0.19; 0.16	-0.005; 0.004	32.9; 32.3	4.32; 3.73	5.73; 5.27	27.1; 27.0
LysteGro	-0.05; -0.06	-0.010; -0.004	27.2; 27	-1.34; -1.50	1.63; 1.52	25.6; 25.5
Digestate	-0.13; -0.14	-0.010; -0.007	25.3; 25.2	-3.29; -3.31	0.14; 0.13	25.1; 25.1
RCP average	-0.0375; -0.05	-0.010; -0.004	27.5; 27.3	-1.08; -1.27	NA	NA
Elora	RCP4.5; RCP8.5					
NPK	0.07; 0.18	0.002; -0.084	54.9; 54.8	1.63; 1.55	NA	54.9; 54.8
Compost	0.50; 0.58	0.050; -0.043	66.3; 65.4	13.1; 12.2	8.14; 7.45	58.2; 57.9
LysteGro	0.22; 0.33	0.020; -0.069	59; 58.7	5.76; 5.50	2.59; 2.38	56.4; 56.4
Digestate	0.12; 0.23	0.009; -0.075	56.3; 56.3	3.00; 3.05	0.22; 0.20	56; 56.1
RCP average	0.23; 0.33	0.020; -0.068	59.1; 58.8	5.87; 5.55	NA	NA

* Not applicable.

4.3.4 Simulated long-term trends of annual N_2O emission

DayCent predicted average annual N_2O emissions in the following order: compost > LysteGro > digestate > NPK over the simulation period (Table 4-4; Figure 4-8). The long-term trend of N_2O emission appears to be rising for compost at Lods and all treatments at Elora (basicially all the treatments that have accrued SOC). Besides, annual N_2O emission was predicted to increase by 6.3 and 7.7% under RCP8.5 than RCP4.5 at Lods and Elora respectively, with compost treatment rises particularly more (Table 4-4).

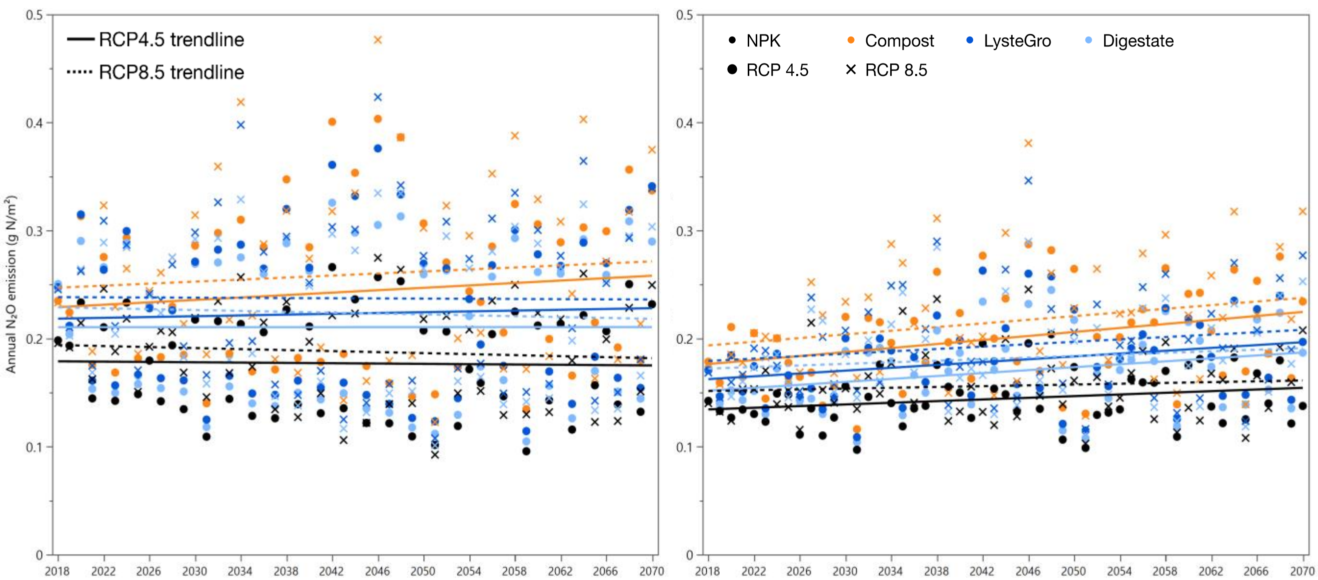


Figure 4-8. Simulated annual N_2O emission at Lods (left) and Elora (right) by DayCent (vDec 2016) under RCP4.5 and RCP8.5.

4.3.5 On-site greenhouse gas balance and yield-scaled greenhouse gas intensities

At both sites, compost application was predicted to have the lowest on-site GHG balance (-41% at Lods and -230% at Elora compared to NPK, averaged across RCPs) and GHGI (-47% and -226%) particularly before 2040 (Table 4-6). LysteGro and digestate in general have similar on-site GHG balance compared to NPK but both have lower lifetime GHGI than NPK. However, the on-site GHG balance and GHGI of all the organic fertilizer treatments appear to increase more quickly (or decrease more slowly) relative to NPK, resulting in less mitigation at the end of 2070 compare to near-term 2040 (Table 4-6). However, in absolute term, GHGI only increased for compost treatment at Lods and all treatments at Elora (i.e. all the treatments that accrued SOC until 2040). Besides, DayCent predicted RCP8.5 to have higher on-site GHG balance (6.0 % at Lods and 18% at Elora) and GHGI (1.4 % at Lods and 11% at Elora) than RCP4.5 in 2070.

Table 4-6. On-site GHG balance (in t CO_2-eq/ha) calculated from combining avoided/promoted CO_2 emission based on simulated changes in stable SOC stock (som2c2 + som3c in DayCent) and total N_2O emission by DayCent (vDec 2016). The GHGI (t CO_2-eq/ t C harvested) was calculated by dividing on-site GHG balance with total crop yields. Values are in the form RCP4.5; RCP8.5.

	2018 - 2040 GHG balance	***Difference relative to NPK***	***2018 – 2040 GHGI***	***2018 – 2070 overall GHG balance***	***Difference relative to NPK***	***2018 – 2070 Overall GHGI***
Lods	RCP4.5; RCP8.5					
NPK	28.5; 30.8	NA*	0.355; 0.364	53.1; 55.3	NA	0.288; 0.288
Compost	15.5; 19.7	-13; -11.1	0.177; 0.211	47.8; 52.5	-5.3; -2.8	0.232; 0.242
LysteGro	25.9; 29.3	-2.6; -1.5	0.293; 0.313	56.3; 59.7	3.2; 4.4	0.275; 0.278
Digestate	29.8; 32.6	1.3; 1.8	0.309; 0.326	58.7; 61.4	5.6; 6.1	0.265; 0.267
RCP average	24.9; 28.1	-4.8; -3.6	0.283; 0.303	54.0; 57.2	1.2; 2.6	0.265; 0.269
Elora						
NPK	10.8; 6.6	NA	0.094; 0.05	31.6; 35.5	NA	0.115; 0.121
Compost	-9.4; -12.1	-20.2; -18.7	-0.076; -0.086	13.9; 20.9	-17.7; -14.6	0.046; 0.064
LysteGro	3.9; 0.1	-6.9; -6.5	0.032; 0.001	27.9; 32.8	-3.7; -2.7	0.092; 0.101
Digestate	9.2; 5.2	-1.6; -1.4	0.067; 0.033	33.7; 37.1	2.1; 1.6	0.101; 0.105
RCP average	3.6; -0.1	-9.6; -8.9	0.029; -0.001	26.8; 31.6	-6.4; -5.2	0.089; 0.098

* Not applicable

4.4 Discussion

4.4.1 DayCent performance and limitations

Although calibration improved model fit in almost all variables, it did not improve the performance equally well for all fertilizer treatments. This corroborates with the findings of Toonsiri (2017) and Yue et al. (2019) that the performance of DayCent varied for different types of fertilizer. A closer inspection revealed that similar fertilizer rankings of model outputs can be seen in both default and calibrated model (e.g. digestate produced the highest corn yield and corn-season N_2O emission; compost sustained the highest soybean yield, soy-season N_2O emission and SOC stock at both sites, Table 4-2 and 4-3). This suggests that these rankings were likely generated by mechanisms deep-rooted in DayCent, rather than an artefact of calibration. These model-produced rankings unfortunately did not always corroborate with the measured rankings, although it must be noted that in this study, almost none of the measured variables were significantly different among fertilizer treatments at $\alpha = 0.05$, except some differences were observed in initial N_2O emission after fertilizer application. Nonetheless, the model fit to N_2O emissions were the poorest among all measured variables.

Default DayCent underestimated all N_2O emissions (Table 4-3 and Figure 4-5). DayCent has a limited number of adjustable parameters to control N_2O emission (Appendix 6). The sensitivity of denitrification to soil moisture (*wfpsdnitadj*) and NO_3^- (*no3_n2o* parameters) were the only parameters that can reasonably be adjusted to improve emission estimates, based on the fact that default DayCent simulated zero denitrification (data not shown). However, this did not rectify the ranking among fertilizers as it drove up all emissions nearly equally, which resulted in an over-estimation of corn-season N_2O emissions in digestate and compost. Part of the lack-of-fit in default DayCent is likely due to inherent model formulation not rectifiable by calibration e.g.

the inability to simulate increased gas diffusion following field cultivation and the limited representation of fertilizer properties, considering the default model underestimated N_2O emissions mainly during the “hot periods” after fertilizer application and field cultivation, whereas the calibrated model over-estimated the “cold periods” instead. In addition, DayCent allocated most N_2O emissions to the corn season, inconsistent with reality (Table 4-3). The reason is an underestimation of N contributed from native soil OM mineralization and an over-estimation of N contributed from fertilizers, which also explains the underestimated corn grain N uptake under high-yielding conditions at Elora (Table 4-2). A closer inspection found that initialization yielded baseline NH_4^+ and NO_3^- about 3 times lower than measured (data not shown), similar problems were reported in Grant et al. (2016) and Yue et al. (2019). Multiple causes could be attributed e.g. an underestimation of NH_4^+ holding capacity and an over-estimated N-loss other than N_2O, which points to a need of further investigation.

Toonsiri (2017) also found that liquid fish emulsion and cyano-fertilizer (< 0.066% organic C, total C:N < 1.5), comparable to digestate, had over-estimated N_2O emission relative to solid organic fertilizers in DayCent. This bias may be because DayCent assumes all NH_4^+ must be in the top 15 cm of soils, whereas in reality the NH_4^+ in “watery” digestate could be deposited at greater depths where it is more shielded from tillage (Clough et al., 2005; Meijide et al., 2007), nitrifying activities are lower, and further reduction of N_2O to N_2 is more likely (n.b. DayCent does not allow gas re-consumption once produced) (Clough et al., 2005; Grant et al., 2016). On the other hand, the over-estimation of N_2O emission from compost application may be caused by the inability to accurately represent the stability of composted OM. Although there is an option in this DayCent version to add composted OM as slow SOC directly, preliminary simulations found that this option was “too slow”, which resulted in substantially retarded crop growth. On the contrary, DayCent underestimated the N_2O emission of NPK treatment (granular urea as N) relative to the organic fertilizers at both sites. Many studies observed higher N_2O emissions following granular urea application compared to many other fertilizers (Asgedom et al., 2014; Halvorson & Del Grosso, 2012; López-Fernández et al., 2007). A possible explanation for the underestimation is that applying granular urea produces "localized" concentrated N spots that can greatly stimulate N_2O emission (Tenuta & Beauchamp, 2000). DayCent has no functionality to simulate the physical form of fertilizers or application methods that entails a localized N concentration effect, any fertilizer is basically applied as a homogeneous "smear". These inherent model shortcomings and biases for different fertilizers suggest more testing and reformulation of DayCent is required, as parameterizing the model separately for each and every fertilizer is arguably not ideal. Also, a consideration of these biases is necessary when interpreting agricultural management implication.

4.4.2 Predicted yield and environmental performance of the organic fertilizers

Corn yield was predicted to be the highest for digestate at both sites. This high corn yield conforms with many studies that reported liquid digestate to have a higher short-term fertilizing performance than organic solids and even mineral fertilizers occasionally (Alburquerque et al., 2012; Cavalli et al., 2016; Panuccio et al., 2019; Simon et al., 2015). Despite having the highest corn yield, digestate application did not accrue SOC because silage harvest removed most corn biomass. On the other hand, compost treatment saw a consistent edge on soybean yield and the fastest rise of corn yield over time, coinciding with its highest OM content and SOC accrual. Abundant long-term studies confirmed that the amount of SOC accrued were positively correlated with the amount of organic C applied (Hemmat et al., 2010; Monaco et al., 2008;

Nardi et al., 2004; Poulton et al., 2018). Building SOC to improve long-term crop productivity is also widely documented (Benbi & Brar, 2009; Lal, 2013). However, it must be noted that DayCent (a Century model descendant) has not incorporated a non-linear SOC saturation mechanism, which may lead to an over-prediction of SOC accrual at already high SOC levels (Abramoff et al., 2018).

At both sites, average annual N_2O emission was predicted to be the highest for compost treatment. The emission also increased the quickest over time, concomitantly with the rising yield trend, highlighting N_2O as a negative tradeoff to SOC accrual. This corroborates with findings that increasing SOC could lead to more N_2O emission (Ding et al., 2013; Gu et al, 2017; Li et al., 2005). However, recall the model over-estimated the N_2O emission of compost application, which indicate that actual emission might be lower (though it does not reject a rising N_2O trend). Studies have found that composted organic solids typically do not lead to large N_2O emission compared to liquid and mineral N-rich fertilizer, particularly in soils rich in SOC (e.g. at Elora) where microbes are not limited by organic C availability (Meijide et al., 2007; Meng et al., 2005; Petersen et al., 2008; Rochette et al., 2008; Thangarajan et.al, 2013). A meta-analysis by Zhou et al. (2017) further add evidence that N_2O emissions higher than mineral fertilizers were only associated with "nutrient-dense" raw cattle and poultry manure rather than pre-treated (composted or anaerobically-digested) manure. These cast some doubt on the consistent model-predicted N_2O emission ranking of compost (over-estimated) > LysteGro > digestate (over-estimated) > NPK (underestimated) at both sites that varied widely in SOC stock. Nonetheless, most studies comparing organic fertilizers were short-term, more long-term studies (> 20 years) of fertilizer-induced N_2O emission using multiple fertilizer types (not just one organic vs. one mineral fertilizer) would be beneficial for further verification of the DayCent predictions.

Greenhouse gas intensity is the most comprehensive indicator of post-application climate change impact in this study. Compost was predicted to have much lower GHGI than NPK both before 2040 and 2070, without even considering the model bias aforementioned. Both LysteGro and digestate had lower GHGI than NPK mainly because of their higher crop yields, as they generally have similar or even higher on-site GHG balance than NPK (Table 4-6). In all cases, the differences of GHGI between the organic fertilizer treatments and NPK diminished (less mitigation) in 2070 relative to 2040, particularly for treatments that gained more SOC (i.e. compost). This is because DayCent predicted N_2O emissions to increase more than crop yield under higher SOC levels, probably because crop growth has more stringent biological and physical constraints. Very few long-term experiments monitoring N_2O emission under consistent organic vs. mineral fertilizer application exist, and even fewer that concurrently report long-term crop yield trends (Zhang et al., 2020; Zhou et al., 2017) so the reliability of this prediction is hard to evaluate. Nonetheless, when model biases are taken into account, the apparent smaller mitigation in 2070 than 2040 is likely over-stated, thus compost could still be a suitable mitigation option beyond 2070. Similarly, digestate may mitigate more GHG than predicted in Table 4-6 after accounting for the model biases, and given that its increase in GHGI over time was much smaller, it could be used as a long-lasting mitigation option.

4.4.3 Management implication under climate change and future research needs

From Table 4-6, we can see that a more severe climate change (RCP8.5) was predicted to increase GHGI (particularly for compost treatment) relative to RCP4.5. The DNDC simulations by Smith et al. (2013) across several annual crop fields in Canada also found GHGI to be generally higher under more severe climate change scenarios. This is mainly because a higher temperature drive SOC loss and increase N_2O emission under wet climate (Dijkstra et al., 2012; Li et al., 2020), while the crop and SOC benefits derived from elevated pCO_2 cap earlier (Broberg et al., 2019; Gill et al., 2002). What is worth mentioning is that real climate is changeable unlike climate scenarios, it is possible to curb the intensifying positive feedback of global warming-land GHG emissions given that intensive mitigation practices are implemented early on (Melillo et al., 2017; Wiesmeier et al., 2016). In this view, applying the OM-rich compost as an early mitigation option should be prioritized because its mitigation potential will be penalized under a warmer world. In contrast, the more inorganic fertilizers (i.e. digestate) may be applied as a time-insensitive mitigation strategy as they work by enhancing crop yield (more averted land clearance) and shaving down N_2O emission.

Last but not least, management decision should further take into account pre-application impacts e.g. anaerobic digestion usually lead to less impact operationally from gas capture compared to composting (Haight, 2005; Levis and Barlaz, 2011). Also, we need to be aware that fertilizer choice is not the only variable we can manipulate. Building high SOC may be able to allow additional opportunities of high-yielding practices and cultivars that only rich soil can support (Benbi & Brar, 2009; Garratt et al. 2018; Johnston et al., 2009). This could potentially reduce impact further by increasing crop yields and building even more SOC (Tian et al., 2015; Table 4-5), thereby forming a positive feedback loop with reduced efforts.

4.5 Conclusion

The present study compared the agroecological outcomes of applying three organic fertilizers: composted food waste, LysteGro biosolid slurry, and liquid digestate. We predicted using DayCent model, the long-term trend (2018 – 2070) of corn and soybean yields, SOC stock and soil N_2O emission in two fields in Quebec and Ontario under the biennial application of these fertilizers. DayCent predicted digestate application would produce the highest corn silage yield whereas compost would produce the highest soybean yield as well as the fastest SOC accrual. These contributed to lower GHGI than NPK, in particular, compost application showed strong mitigation potential before 2040. The mitigation potential of all organic fertilizers diminished farther in time as well as under RCP8.5 compared to RCP4.5, indicating the positive-feedback of climate change-land GHG emissions. A management practice that maximizes mitigation early on (i.e. compost application) has its merits to curb this positive-feedback. Lastly, there were cross-fertilizer model biases, particularly where the N_2O emissions with compost and digestate were over-estimated compared to measured dataset, indicating the need of model improvement. Nonetheless, accounting this bias would reinforce compost application as the best mitigation option in near-term and make digestate a probable long-term mitigation option via enhancing crop yield while lowering N_2O emission.

Chapter 4 references

Alburquerque, J. A., De la Fuente, C., Campoy, M., Carrasco, L., Nájera, I., Baixauli, C., ... & Bernal, M. P. (2012). Agricultural use of digestate for horticultural crop production and improvement of soil properties. *European Journal of Agronomy*, 43, 119-128.

Asgedom, H., Tenuta, M., Flaten, D. N., Gao, X., & Kebreab, E. (2014). Nitrous oxide emissions from a clay soil receiving granular urea formulations and dairy manure. *Agronomy Journal*, 106(2), 732-744.

Baldock, J. A., Sanderman, J., & Farquharson, R. (2010). Capturing carbon in Australian soils: potential and realities. In *Proceedings of the 19th World Congress of Soil Science* (pp. 1-6).

Benbi, D. K., & Brar, J. S. (2009). A 25-year record of carbon sequestration and soil properties in intensive agriculture. *Agronomy for Sustainable Development*, 29(2), 257-265.

Broberg, M. C., Högy, P., Feng, Z., & Pleijel, H. (2019). Effects of elevated CO2 on wheat yield: non-linear response and relation to site productivity. *Agronomy*, *9*(5), 243.

Burney, J. A., Davis, S. J., & Lobell, D. B. (2010). Greenhouse gas mitigation by agricultural intensification. *Proceedings of the National Academy of Sciences*, 107(26), 12052-12057.

Cavalli, D., Cabassi, G., Borrelli, L., Geromel, G., Bechini, L., Degano, L., & Gallina, P. M. (2016). Nitrogen fertilizer replacement value of undigeted liquid cattle manure and digestates. *European Journal of Agronomy*, 73, 34-41.

Chang, K. H., Warland, J., Voroney, P., Bartlett, P., & Wagner-Riddle, C. (2013). Using DayCENT to simulate carbon dynamics in conventional and no-till agriculture. *Soil Science Society of America Journal*, 77(3), 941-950.

Charles, A., Rochette, P., Whalen, J. K., Angers, D. A., Chantigny, M. H., & Bertrand, N. (2017). Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. *Agriculture, Ecosystems & Environment*, 236, 88-98.

Chodak, M., Borken, W., Ludwig, B., & Beese, F. (2001). Effect of temperature on the mineralization of C and N of fresh and mature compost in sandy material. *Journal of Plant Nutrition and Soil Science*, 164(3), 289-294.

Clough, T. J., Sherlock, R. R., & Rolston, D. E. (2005). A review of the movement and fate of N_2O in the subsoil. *Nutrient Cycling in Agroecosystems*, 72(1), 3-11.

Congreves, K. A., Grant, B. B., Campbell, C. A., Smith, W. N., VandenBygaart, A. J., Kröbel, R., ... & Desjardins, R. L. (2015). Measuring and modeling the long-term impact of crop management on soil carbon sequestration in the semiarid Canadian prairies. *Agronomy Journal*, 107(3), 1141-1154.

Davidson, E. A., & Janssens, I. A. (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature*, 440(7081), 165-173.

Dayamurthi, S (1997). The effects of tillage and croping on the dynamics of soil organic matter as determined by 13C natural abundance. [Doctoral dissertation, The University of Guelph]. National Library of Canada.

Del Grosso, S. J., Halvorson, A. D., & Parton, W. J. (2008). Testing DAYCENT model simulations of corn yields and nitrous oxide emissions in irrigated tillage systems in Colorado. *Journal of Environmental Quality*, 37(4), 1383-1389.

Del Grosso, S. J., Parton, W. J., Adler, P. R., Davis, S. C., Keough, C., & Marx, E. (2012). DayCent model simulations for estimating soil carbon dynamics and greenhouse gas fluxes from agricultural production systems. In *Managing Agricultural Greenhouse Gases* (pp. 241-250).

Del Grosso, S. J., Parton, W. J., Keough, C. A., & Reyes-Fox, M. (2011). Special features of the DayCent modeling package and additional procedures for parameterization, calibration, validation, and applications. In *Methods of introducing system models into agricultural research*, 2, 155-176.

Diacono, M., & Montemurro, F. (2011). Long-term effects of organic amendments on soil fertility. In *Sustainable Agriculture* Volume 2 (pp. 761-786). Springer, Dordrecht.

Dijkstra, F. A., Prior, S. A., Runion, G. B., Torbert, H. A., Tian, H., Lu, C., & Venterea, R. T. (2012). Effects of elevated carbon dioxide and increased temperature on methane and nitrous oxide fluxes: evidence from field experiments. *Frontiers in Ecology and the Environment*, 10(10), 520-527.

Ding, W., Luo, J., Li, J., Yu, H., Fan, J., & Liu, D. (2013). Effect of long-term compost and inorganic fertilizer application on background N_2O and fertilizer-induced N_2O emissions from an intensively cultivated soil. *Science of the Total Environment*, 465, 115-124.

Dobbie, K. E., & Smith, K. A. (2001). The effects of temperature, water-filled pore space and land use on N_2O emissions from an imperfectly drained gleysol. *European Journal of Soil Science*, 52(4), 667-673.

Doherty, J.E., Hunt, R.J. (2010). Approaches to Highly Parameterized Inversion: A Guide to Using PEST for Groundwater-Model Calibration. *U.S. Geological Survey Scientific Investigations Report* 2010–5169. U.S. Geological Survey, Reston, Virginia (p. 59).

Gaillard, R. K., Jones, C. D., Ingraham, P., Collier, S., Izaurralde, R. C., Jokela, W., ... & Ruark, M. D. (2018). Underestimation of N_2O emissions in a comparison of the DayCent, DNDC, and EPIC models. *Ecological Applications*, 28(3), 694-708.

Garratt, M. P., Bommarco, R., Kleijn, D., Martin, E., Mortimer, S. R., Redlich, S., ... & van Gils, S. (2018). Enhancing soil organic matter as a route to the ecological intensification of European arable systems. *Ecosystems*, 21(7), 1404-1415.

Gill, R. A., Polley, H. W., Johnson, H. B., Anderson, L. J., Maherali, H., & Jackson, R. B. (2002). Nonlinear grassland responses to past and future atmospheric CO_2. *Nature*, *417*(6886), 279-282.

Grant, B. B., Smith, W. N., Campbell, C. A., Desjardins, R. L., Lemke, R. L., Kröbel, R., ... & Lafond, G. P. (2016). Comparison of DayCent and DNDC models: case studies using data from long-term experiments on the Canadian prairies. *Synthesis and modeling of greenhouse gas emissions and carbon storage in agricultural and forest systems to guide mitigation and adaptation*, 6, 21-57.

Grassini, P., Torrion, J. A., Cassman, K. G., Yang, H. S., & Specht, J. E. (2014). Drivers of spatial and temporal variation in soybean yield and irrigation requirements in the western US Corn Belt. *Field Crops Research*, 163, 32-46.

Gregorich, E. G., Rochette, P., VandenBygaart, A. J., & Angers, D. A. (2005). Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. *Soil and Tillage Research*, 83(1), 53-72.

Grenier, P., De Elia, R., & Chaumont, D. (2015). Chances of Short-Term Cooling Estimated from a Selection of CMIP5-Based Climate Scenarios during 2006–35 over Canada. *Journal of Climate*, 28(8), 3232-3249.

Gu, J., Yuan, M., Liu, J., Hao, Y., Zhou, Y., Qu, D., & Yang, X. (2017). Trade-off between soil organic carbon sequestration and nitrous oxide emissions from winter wheat-summer maize rotations: Implications of a 25-year fertilization experiment in Northwestern China. *Science of the Total Environment*, 595, 371-379.

Haight, M. (2005). Assessing the environmental burdens of anaerobic digestion in comparison to alternative options for managing the biodegradable fraction of municipal solid wastes. *Water Science and Technology*, 52(1-2), 553-559.

Halvorson, A. D., & Del Grosso, S. J. (2012). Nitrogen source and placement effects on soil nitrous oxide emissions from no-till corn. *Journal of Environmental Quality*, 41(5), 1349-1360.

Hanway, J. J. (1962). Corn Growth and Composition in Relation to Soil Fertility: II. Uptake of N, P, and K and Their Distribution in Different Plant Parts during the Growing Season 1. *Agronomy Journal*, 54(3), 217-222.

Hartmann, M. D., Parton, W. J., Del Grosso, S. J., Hendryx, J., Hilinski, T., Kelly, R., ... & McKeon, R. (2018). DayCent Ecosystem Model. User Manual, Scientific Basis, and Technical Documentation.

Hemmat, A., Aghilinategh, N., Rezainejad, Y., & Sadeghi, M. (2010). Long-term impacts of municipal solid waste compost, sewage sludge and farmyard manure application on organic carbon, bulk density and consistency limits of a calcareous soil in central Iran. *Soil and Tillage Research*, 108(1-2), 43-50.

Johnston, A. E., Poulton, P. R., & Coleman, K. (2009). Soil organic matter: its importance in sustainable agriculture and carbon dioxide fluxes. Advances in agronomy, 101, 1-57.

Jones, R. L. (1942). French-Canadian Agriculture in the St. Lawrence Valley, 1815-1850. *Agricultural History*, 16(3), 137-148.

Kimball, B. A., Kobayashi, K., & Bindi, M. (2002). Responses of agricultural crops to free-air CO_2 enrichment. In *Advances in agronomy* (Vol. 77, pp. 293-368). Academic Press.

Lal, R. (2013). Intensive agriculture and the soil carbon pool. *Journal of Crop Improvement*, 27(6), 735-751.

Lee, J., De Gryze, S., & Six, J. (2011). Effect of climate change on field crop production in California's Central Valley. *Climatic Change*, 109(1), 335-353.

Li, C., Frolking, S., & Butterbach-Bahl, K. (2005). Carbon sequestration in arable soils is likely to increase nitrous oxide emissions, offsetting reductions in climate radiative forcing. *Climatic Change*, 72(3), 321-338.

Li, L., Zheng, Z., Wang, W., Biederman, J. A., Xu, X., Ran, Q., ... & Zhou, S. (2020). Terrestrial N2O emissions and related functional genes under climate change: A global meta-analysis. *Global Change Biology*, *26*(2), 931-943.

López-Fernández, S., Diez, J. A., Hernaiz, P., Arce, A., García-Torres, L., & Vallejo, A. (2007). Effects of fertiliser type and the presence or absence of plants on nitrous oxide emissions from irrigated soils. *Nutrient Cycling in Agroecosystems*, 78(3), 279-289.

Meijide, A., Díez, J. A., Sánchez-Martín, L., López-Fernández, S., & Vallejo, A. (2007). Nitrogen oxide emissions from an irrigated maize crop amended with treated pig slurries and composts in a Mediterranean climate. *Agriculture, Ecosystems & Environment*, 121(4), 383-394.

Meinshausen, M., Smith, S. J., Calvin, K., Daniel, J. S., Kainuma, M. L. T., Lamarque, J. F., ... & Thomson, A. G. J. M. V. (2011). The RCP greenhouse gas concentrations and their extensions from 1765 to 2300. *Climatic Change*, 109(1-2), 213.

Melillo, J. M., Frey, S. D., DeAngelis, K. M., Werner, W. J., Bernard, M. J., Bowles, F. P., ... & Grandy, A. S. (2017). Long-term pattern and magnitude of soil carbon feedback to the climate system in a warming world. *Science*, 358(6359), 101-105.

Monaco, S., Hatch, D. J., Sacco, D., Bertora, C., & Grignani, C. (2008). Changes in chemical and biochemical soil properties induced by 11-yr repeated additions of different organic materials in maize-based forage systems. *Soil Biology and Biochemistry*, 40(3), 608-615.

Nardi, S., Morari, F., Berti, A., Tosoni, M., & Giardini, L. (2004). Soil organic matter properties after 40 years of different use of organic and mineral fertilisers. *European Journal of Agronomy*, 21(3), 357-367.

Necpálová, M., Anex, R. P., Fienen, M. N., Del Grosso, S. J., Castellano, M. J., ... & Barker, D. W. (2015). Understanding the DayCent model: Calibration, sensitivity, and identifiability through inverse modeling. *Environmental Modelling & Software,* 66, 110-130.

Necpálová, M., Lee, J., Skinner, C., Büchi, L., Wittwer, R., Gattinger, A., ... & Mayer, J. (2018). Potentials to mitigate greenhouse gas emissions from Swiss agriculture. *Agriculture, Ecosystems & Environment*, 265, 84-102.

IPCC (2014). Climate Change 2014: Synthesis Report. In *Proceedings of the Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, eds., R. K. Pachauri, and L. A. Meyer. Geneva, Switzerland.

Panuccio, M. R., Papalia, T., Attinà, E., Giuffrè, A., & Muscolo, A. (2019). Use of digestate as an alternative to mineral fertilizer: effects on growth and crop quality. *Archives of Agronomy and Soil Science*, 65(5), 700-711.

Parton, W. J., Holland, E. A., Del Grosso, S. J., Hartman, M. D., Martin, R. E., Mosier, A. R., ... & Schimel, D. S. (2001). Generalized model for NOx and N_2O emissions from soils. Journal of Geophysical Research: *Atmospheres*, 106(D15), 17403-17419.

Pelster, D. E., Chantigny, M. H., Rochette, P., Angers, D. A., Rieux, C., & Vanasse, A. (2012). Nitrous oxide emissions respond differently to mineral and organic nitrogen sources in contrasting soil types. *Journal of Environmental Quality*, 41(2), 427-435.

Petersen, S. O., Schjønning, P., Thomsen, I. K., & Christensen, B. T. (2008). Nitrous oxide evolution from structurally intact soil as influenced by tillage and soil water content. *Soil Biology and Biochemistry*, 40(4), 967-977.

Poulton, P., Johnston, J., Macdonald, A., White, R., & Powlson, D. (2018). Major limitations to achieving "4 per 1000" increases in soil organic carbon stock in temperate regions: Evidence from long-term experiments at Rothamsted Research, United Kingdom. *Global Change Biology*, 24(6), 2563-2584.

Powlson, D. S., Whitmore, A. P., & Goulding, K. W. (2011). Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. *European Journal of Soil Science*, 62(1), 42-55.

Rafique, R., Kumar, S., Luo, Y., Kiely, G., & Asrar, G. (2015). An algorithmic calibration approach to identify globally optimal parameters for constraining the DayCent model. *Ecological Modelling*, 297, 196-200.

Rafique, R., Kumar, S., Luo, Y., Xu, X., Li, D., & Zhang, W. (2014). Estimation of greenhouse gases (N_2O, CH_4 and CO_2) from no-till cropland under increased temperature and altered precipitation regime: a DAYCENT model approach. *Global and Planetary Change*, 118, 106-114.

Rochette, P., Angers, D. A., Chantigny, M. H., Gagnon, B., & Bertrand, N. (2008). N_2O fluxes in soils of contrasting textures fertilized with liquid and solid dairy cattle manures. *Canadian Journal of Soil Science*, 88(2), 175-187.

Sainju, U. M. (2016). A global meta-analysis on the impact of management practices on net global warming potential and greenhouse gas intensity from cropland soils. *PloS one*, 11(2), e0148527.

Sansoulet, J., Pattey, E., Kröbel, R., Grant, B., Smith, W., Jégo, G., ... & Tremblay, G. (2014). Comparing the performance of the STICS, DNDC, and DayCent models for predicting N uptake and biomass of spring wheat in Eastern Canada. *Field Crops Research*, 156, 135-150.

Savage, K. E., Parton, W. J., Davidson, E. A., Trumbore, S. E., & Frey, S. D. (2013). Long-term changes in forest carbon under temperature and nitrogen amendments in a temperate northern hardwood forest. *Global Change Biology*, 19(8), 2389-2400.

Saxton, K. E., & Rawls, W. J. (2006). Soil water characteristic estimates by texture and organic matter for hydrologic solutions. *Soil Science Society of America Journal*, 70(5), 1569-1578.

Schaufler, G., Kitzler, B., Schindlbacher, A., Skiba, U., Sutton. M.A., & Zechmeister-Boltenstern, S. (2010). Greenhouse gas emissions from European soils under different land use: effects of soil moisture and temperature. *European Journal of Soil Science*, 61, 683-696

Šimon, T., Kunzová, E., & Friedlová, M. (2015). The effect of digestate, cattle slurry and mineral fertilization on the winter wheat yield and soil quality parameters. *Plant, Soil and Environment*, 61(11), 522-527.

Smith, W. N., Grant, B. B., Campbell, C. A., McConkey, B. G., Desjardins, R. L., Kröbel, R., & Malhi, S. S. (2012). Crop residue removal effects on soil carbon: Measured and inter-model comparisons. *Agriculture, Ecosystems & Environment*, 161, 27-38.

Smith, W. N., Grant, B. B., Desjardins, R. L., Kroebel, R., Li, C., Qian, B., ... & Drury, C. F. (2013). Assessing the effects of climate change on crop production and GHG emissions in Canada. *Agriculture, Ecosystems & Environment*, 179, 139-150.

Smith, W. N., Grant, B. B., Desjardins, R. L., Rochette, P., Drury, C. F., & Li, C. (2008). Evaluation of two process-based models to estimate soil N_2O emissions in Eastern Canada. *Canadian Journal of Soil Science*, 88(2), 251-260.

Tenuta, M., & Beauchamp, E. G. (2000). Nitrous oxide production from urea granules of different sizes. *Journal of Environmental Quality*, 29(5), 1408-1413.

Thangarajan, R., Bolan, N. S., Tian, G., Naidu, R., & Kunhikrishnan, A. (2013). Role of organic amendment application on greenhouse gas emission from soil. *Science of the Total Environment*, 465, 72-96.

Tian, G., Granato, T. C., Cox, A. E., Pietz, R. I., Carlson, C. R., & Abedin, Z. (2009). Soil carbon sequestration resulting from long-term application of biosolids for land reclamation. *Journal of Environmental Quality*, 38(1), 61-74.

Toonsiri, P. (2017). Effects of agricultural management on greenhouse gas emissions, carbon and nitrogen sequestration, and DAYCENT simulation accuracy. Doctoral dissertation, Colorado State University, Fort Collins, Colorado.

Van Groenigen, J. W., Velthof, G. L., Oenema, O., Van Groenigen, K. J., & Van Kessel, C. (2010). Towards an agronomic assessment of N_2O emissions: a case study for arable crops. *European Journal of Soil Science*, 61(6), 903-913.

Wang, W., Dalal, R. C., Reeves, S. H., Butterbach-bahl, K. L. A. U. S., & Kiese, R. (2011). Greenhouse gas fluxes from an Australian subtropical cropland under long-term contrasting management regimes. *Global Change Biology*, 17(10), 3089-3101.

Weiler, D. A., Tornquist, C. G., Parton, W., Pereira dos Santos, H., Santi, A., & Bayer, C. (2017). Crop biomass, soil carbon, and nitrous oxide as affected by management and climate: a DayCent application in Brazil. *Soil Science Society of America Journal*, 81(4), 945-955.

Wiesmeier, M., Poeplau, C., Sierra, C. A., Maier, H., Frühauf, C., Hübner, R., ... & Schilling, B. (2016). Projected loss of soil organic carbon in temperate agricultural soils in the 21 st century: effects of climate change and carbon input trends. *Scientific Reports*, 6(1), 1-17.

Yang, B., Xiong, Z., Wang, J., Xu, X., Huang, Q., & Shen, Q. (2015). Mitigating net global warming potential and greenhouse gas intensities by substituting chemical nitrogen fertilizers with organic fertilization strategies in rice–wheat annual rotation systems in China: A 3-year field experiment. *Ecological Engineering*, 81, 289-297.

Yue, Q., Cheng, K., Ogle, S., Hillier, J., Smith, P., Abdalla, M., ... & Pan, G. (2019). Evaluation of four modelling approaches to estimate nitrous oxide emissions in China's cropland. *Science of the Total Environment*, 652, 1279-1289.

Zhang, X., Fang, Q., Zhang, T., Ma, W., Velthof, G. L., Hou, Y., ... & Zhang, F. (2020). Benefits and trade-offs of replacing synthetic fertilizers by animal manures in crop production in China: A meta-analysis. *Global Change Biology*, 26(2), 888-900.

Zhou, M., Zhu, B., Wang, S., Zhu, X., Vereecken, H., & Brüggemann, N. (2017). Stimulation of N_2O emission by manure application to agricultural soils may largely offset carbon benefits: A global meta-analysis. *Global Change Biology*, 23(10), 4068-4083.

Chapter 5 – General Discussion

5.1 Field experiment – the short-term effect of organic fertilizer application

Were there differences among the fertilizers tested? We found no significant effects of one-time application of the organic fertilizers on crop yield, SOC stock and growing-season CO_2 and N_2O emissions in two seasons. These findings are not unexpected considering we only applied the fertilizer once at agronomic rates (no excessive OM or insufficient N applied). However, the non-significant effect over short-term does not rule out the possibility of longer-term effect. It is particularly well-known that SOC changes slowly that is easily masked by natural spatial heterogeneity over short-term (Necpálová et al., 2014). Therefore, continuing the experiment over a longer term can help discover possible differences worth noting, considering the model findings in Chapter 4 did show some differences between the fertilizers, and that management advice to farmers should gear towards a longer-term management practice. Knowing whether the current application rates are optimal is also important if the best use of these organic fertilizers is of concern. For instance, yield-scaled N_2O emission (a type of GHGI) changes with application rates, reaching a minimum and increases rapidly when additional N input struggle to further increase yield (Hoben et al. 2011; Qin et al., 2012; Van Groenigen et al., 2010). If longer-term experiment were to be conducted, it would also help detect the effect of slightly different application rates.

The difference of GHG fluxes (if any) were obscured by numerous sources of uncertainties as discussed in Chapter 3, from flux estimation and interpolation to inherent spatio-temporal variability. With large uncertainties, it means that there is the issue of effect size inflation whenever we detect a significant effect, so reducing the uncertainty is of prime importance when potential risk factors (even small ones) are to be identified. Increasing the number of gas samples per sampling event can reduce estimation-level uncertainty. More frequent gas sampling is needed if obtaining reliable estimates of cumulative emission is required. Spatial variability can be accounted for by either having more replicates using smaller plot or more chamber (subsample) per replicate, depending on the cost, which also addresses the problem of representativeness. With one fixed-location small chamber per plot, it is unclear whether the chambers are representative of the "plot" in a traditional ANOVA sense. So, we can only infer that the chamber is representative of the small-scale area covered by the chamber itself. It then become clear that there is a disconnect between small-scale (< 1 m^2) flux measurements by gas chamber and large-scale (> 500 m^2) measurements by eddy covariance that is not well-adapted for replicated field experiment (McDaniel et al., 2017; Molodovskaya et al., 2011). Sophisticated spatial flux model taking into account soil heterogeneity (validated by high-density and high-frequency measurements) is likely needed to interpolate and extrapolate chamber measurements for informing field- to regional-scale agricultural practices.

What drives soil N_2O fluxes in field? We did not specifically investigate the potential mechanisms driving soil N_2O emission in this MSc. project. Nonetheless, if the experiment is longer and there are more differences detected, it may be useful to know what contributes to the differences between fertilizers from a management perspective. To understand the drivers of soil N_2O emission, intensively measuring soil mineral N (NH_4^+, NO_2^- and NO_3^-) concentration along with soil moisture and temperature just before and after fertilizer application and after plowing (i.e. baseline and hot moments) is beneficial, as sharp changes in N_2O fluxes occur during these instances (Charteris et al., 2020). Occasional mineral N measurements in-season adopted in many studies (we also collected monthly samples, although not used in this thesis) may decouple from actual N supply to microbes. Basically, a low mineral N measured at a particular isolated moment and location does not equate low N supply to microbes, it could be because microbes were taking up mineral N quickly due to other soil drivers (which we did not exhaustively measured), or just due to fine-scale heterogeneity in the soil. Given that N_2O fluxes during hot moments contribute to cumulative N_2O emission substantially (e.g. >44% cumulative emissions come from 1st month in our study and 51% from <7% time in Molodovskaya et al. (2012)), I argue that intensive, spatially- and temporally-clustered measurements around potential "hot moment" is superior to infrequent regular interval measurements in order to reveal the underlying drivers of N_2O emission relevant for agricultural management.

5.2 DayCent simulations – long-term effect of organic fertilizer application

How reliable is our simulation findings and how to improve it? Although DayCent is generally considered suitable for making long-term predictions (particularly SOC), a good marriage between simulation findings and management decision is still limited by the coverage of calibration and validation dataset, as well as inherent model shortcomings. The approach of calibration used was to constrain DayCent parameters based on a more complete 2-year dataset with more frequent measurements in one site (Lods), and to validate the model with data from another site (Elora). The limitation of this approach is that the model is likely adapted to a very narrow agro-climate condition, which limits the reliability of model predictions under long-term changing climate. This should either be overcome by having more diverse calibration and validation datasets covering a wider range of climate conditions, or conducting the existing field experiment for a much longer period of time (> 20 years). The former is arguably much less costly and therefore similar research in the future should be conducted under the premise of more data sharing, much like the basis of collaborative research network e.g. FLUXNET and

GRACENET. However, having long-term experimental data under consistent agricultural management is still important for verifying emergent (non-linear) trends occurring at a later time, such as when SOC stock saturation or hitting the yield cap would occur. This would help tune the functionalities of model SOC pools and ecophysiological parameters such as microbial carbon use efficiency and crop death rate under biophysical constraints that are relevant for long-term predictions. I suggest that more agricultural field research should consider having long-term consistent treatment as the main effect. If a subsequent experiment is proposed, the new treatment practice can simply be included as subplots.

DayCent has inherent model shortcomings that need to be overcome. The lack-of-fit of simulated soil N_2O emissions and its cross-fertilizer biases (compost and digestate over-estimated, NPK underestimated) should be another prime target for improving the applicability of model findings. Suggestions of improvement were made in Chapter 4 including representing a more complete profile of fertilizer physicochemical properties, incorporating the effect of tillage on gas exchange and accounting localized N concentration effect brought by different physical forms of fertilizer and application methods. In addition, our simulations and other studies such as Grant et al. (2016) and Yue et al. (2019) showed that DayCent underestimated the mineral N contributed by native soil OM for both plant growth and N_2O production. Aside from improving model formulation, it is perhaps also important to have an unfertilized control so that information about the inherent N supply from soil OM mineralization can be used to constrain the model.

Chapter 6 - Final Conclusions

This project evaluated the effect of three organic fertilizers: composted food waste (compost; 15% N in mineral forms), LysteGro biosolid slurry (LysteGro; 47% N in mineral forms) and liquid anaerobic digestate (digestate; 97.5% N in mineral forms), plus a mineral fertilizer control (NPK), on corn and soybean yields, SOC stock (0 – 20 cm) and soil GHG fluxes on two corn-soybean fields in Quebec and Ontario over two seasons. The key finding of the field study was that organic fertilizers can replace NPK fertilizer in the short-term, since it produced similar crop yield and did not alter the SOC stock or stimulate growing-season CO_2 and N_2O emissions. The year-to-year variability in a particular site was similar, indicating that site-specific factors other than fertilization exerted a predominant control on cumulative soil GHG emissions in our experiment.

Since it is difficult to make generalization about the management decision of organic fertilizers from a short-term field study, the long-term effects were evaluated with the DayCent model. At both sites, DayCent predicted that digestate application would produce the highest corn silage C yield whereas compost would produce the highest soybean grain C yield. Compost is the best GHG mitigation option until 2040 due to the predicted highest SOC accrual, but once a steady-state SOC level is reached, greater decomposition of this large SOC stock is coupled with increased N_2O emission which gradually increase the GHGI. This implies that the carbon credit from organic fertilizers like compost must be adjusted considering the steady-state of SOC level. This finding needs to be incorporated into predictive models used by environmental ministries to calculate carbon credits for the agricultural sector. Digestate application may support crop N demands without increasing the N_2O emissions compared to NPK fertilizer, making it a good substitute for NPK fertilizer over long-term. Still, the prediction that SOC loss and N_2O emissions will increase (in particular for OM-rich compost) under the RCP 8.5 is alarming and suggests that farmers, farm advisors and agricultural ministries need to insist upon judicious application of any fertilizer to maximize GHG mitigation early on, by achieving the desired agroecological outcomes of low GHG emissions, high crop yields and SOC stocks.

Other References

Cayuela, M. L., Oenema, O., Kuikman, P. J., Bakker, R. R., & Van Groenigen, J. W. (2010). Bioenergy by-products as soil amendments? Implications for carbon sequestration and greenhouse gas emissions. *Gcb Bioenergy*, 2(4), 201-213.

Chang, K. H., Warland, J., Voroney, P., Bartlett, P., & Wagner-Riddle, C. (2013). Using DayCENT to simulate carbon dynamics in conventional and no-till agriculture. *Soil Science Society of America Journal*, 77(3), 941-950.

Charles, A., Rochette, P., Whalen, J. K., Angers, D. A., Chantigny, M. H., & Bertrand, N. (2017). Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. *Agriculture, Ecosystems & Environment*, 236, 88-98.

Charteris, A. F., Chadwick, D. R., Thorman, R. E., Vallejo, A., de Klein, C. A., Rochette, P., & Cárdenas, L. M. (2020). Global Research Alliance N2O Chamber Methodology Guidelines: Recommendations for deployment and accounting for sources of variability. *Journal of Environmental Quality*.

Del Grosso, S. J., Parton, W. J., Adler, P. R., Davis, S. C., Keough, C., & Marx, E. (2012). DayCent model simulations for estimating soil carbon dynamics and greenhouse gas fluxes from agricultural production systems. In *Managing Agricultural Greenhouse Gases* (pp. 241-250).

Diacono, M., & Montemurro, F. (2011). Long-term effects of organic amendments on soil fertility. In *Sustainable Agriculture Volume 2* (pp. 761-786). Springer, Dordrecht.

Ding, W., Luo, J., Li, J., Yu, H., Fan, J., & Liu, D. (2013). Effect of long-term compost and inorganic fertilizer application on background N_2O and fertilizer-induced N_2O emissions from an intensively cultivated soil. *Science of the Total Environment*, 465, 115-124.

Düring, R. A., & Gäth, S. (2002). Utilization of municipal organic wastes in agriculture: where do we stand, where will we go?. *Journal of Plant Nutrition and Soil Science*, 165(4), 544-556.

Grant, B. B., Smith, W. N., Campbell, C. A., Desjardins, R. L., Lemke, R. L., Kröbel, R., ... & Lafond, G. P. (2016). Comparison of DayCent and DNDC models: case studies using data from long-term experiments on the Canadian prairies. *Synthesis and modeling of greenhouse gas emissions and carbon storage in agricultural and forest systems to guide mitigation and adaptation*, 6, 21-57.

Gregorich, E. G., Rochette, P., VandenBygaart, A. J., & Angers, D. A. (2005). Greenhouse gas contributions of agricultural soils and potential mitigation practices in Eastern Canada. *Soil and Tillage Research*, 83(1), 53-72.

Guest, G., Kröbel, R., Grant, B., Smith, W., Sansoulet, J., Pattey, E., ... & Tremblay, G. (2017). Model comparison of soil processes in eastern Canada using DayCent, DNDC and STICS. *Nutrient Cycling in Agroecosystems*, 109(3), 211-232.

Ho, A., Ijaz, U. Z., Janssens, T. K., Ruijs, R., Kim, S. Y., de Boer, W., ... & Bodelier, P. L. (2017). Effects of bio-based residue amendments on greenhouse gas emission from agricultural soil are stronger than effects of soil type with different microbial community composition. *Gcb Bioenergy*, 9(12), 1707-1720.

Hoben, J. P., Gehl, R. J., Millar, N., Grace, P. R., & Robertson, G. P. (2011). Nonlinear nitrous oxide (N_2O) response to nitrogen fertilizer in on-farm corn crops of the US Midwest. *Global Change Biology, 17*(2), 1140-1152.

Kimball, B. A., Kobayashi, K., & Bindi, M. (2002). Responses of agricultural crops to free-air CO_2 enrichment. In *Advances in agronomy* (Vol. 77, pp. 293-368). Academic Press.

Le Mer, J., & Roger, P. (2001). Production, oxidation, emission and consumption of methane by soils: a review. *European Journal of Soil Biology*, 37(1), 25-50.

Le Quéré, C., Andrew, R. M., Friedlingstein, P., Sitch, S., Pongratz, J., Manning, A. C., ... & Boden, T. A. (2017). Global carbon budget 2017. *Earth System Science Data Discussions*, 1-79.

Li, C., Frolking, S., & Butterbach-Bahl, K. (2005). Carbon sequestration in arable soils is likely to increase nitrous oxide emissions, offsetting reductions in climate radiative forcing. *Climatic Change*, 72(3), 321-338.

McDaniel, M. D., Simpson, R. R., Malone, B. P., McBratney, A. B., Minasny, B., & Adams, M. A. (2017). Quantifying and predicting spatio-temporal variability of soil CH_4 and N_2O fluxes from a seemingly homogeneous Australian agricultural field. *Agriculture, Ecosystems & Environment*, 240, 182-193.

Minasny, B., Malone, B. P., McBratney, A. B., Angers, D. A., Arrouays, D., Chambers, A., ... & Field, D. J. (2017). Soil carbon 4 per mille. *Geoderma*, *292*, 59-86.

Molodovskaya, M., Singurindy, O., Richards, B. K., Warland, J., Johnson, M. S., & Steenhuis, T. S. (2012). Temporal variability of nitrous oxide from fertilized croplands: hot moment analysis. *Soil Science Society of America Journal*, 76(5), 1728-1740.

Molodovskaya, M., Warland, J., Richards, B. K., Öberg, G., & Steenhuis, T. S. (2011). Nitrous oxide from heterogeneous agricultural landscapes: source contribution analysis by eddy covariance and chambers. *Soil Science Society of America Journal*, *75*(5), 1829-1838.

Necpálová, M., Anex, R. P., Fienen, M. N., Del Grosso, S. J., Castellano, M. J., ... & Barker, D. W. (2015). Understanding the DayCent model: Calibration, sensitivity, and identifiability through inverse modeling. *Environmental Modelling & Software*, 66, 110-130.

Paterson, E., & Sim, A. (2013). Soil-specific response functions of organic matter mineralization to the availability of labile carbon. *Global Change Biology*, 19(5), 1562-1571.

Qin, S., Wang, Y., Hu, C., Oenema, O., Li, X., Zhang, Y., & Dong, W. (2012). Yield-scaled N_2O emissions in a winter wheat–summer corn double-cropping system. *Atmospheric Environment*, *55*, 240-244.

Senbayram, M., Chen, R., Budai, A., Bakken, L., & Dittert, K. (2012). N2O emission and the N2O/(N2O+N2) product ratio of denitrification as controlled by available carbon substrates and nitrate concentrations. *Agriculture, Ecosystems & Environment*, 147, 4-12.

Van Groenigen, K. J., Osenberg, C. W., & Hungate, B. A. (2011). Increased soil emissions of potent greenhouse gases under increased atmospheric CO_2. *Nature*, 475(7355), 214-216.

Van Groenigen, J. W., Velthof, G. L., Oenema, O., Van Groenigen, K. J., & Van Kessel, C. (2010). Towards an agronomic assessment of N_2O emissions: a case study for arable crops. *European Journal of Soil Science*, *61*(6), 903-913.

VandenBygaart, A. J., Gregorich, E. G., & Angers, D. A. (2003). Influence of agricultural management on soil organic carbon: A compendium and assessment of Canadian studies. *Canadian Journal of Soil Science*, 83(4), 363-380.

Velthof, G. L., Kuikman, P. J., & Oenema, O. (2003). Nitrous oxide emission from animal manures applied to soil under controlled conditions. *Biology and Fertility of Soils*, 37(4), 221-230.

Wang, W. J., Dalal, R. C., Moody, P. W., & Smith, C. J. (2003). Relationships of soil respiration to microbial biomass, substrate availability and clay content. *Soil Biology and Biochemistry*, 35(2), 273-284.

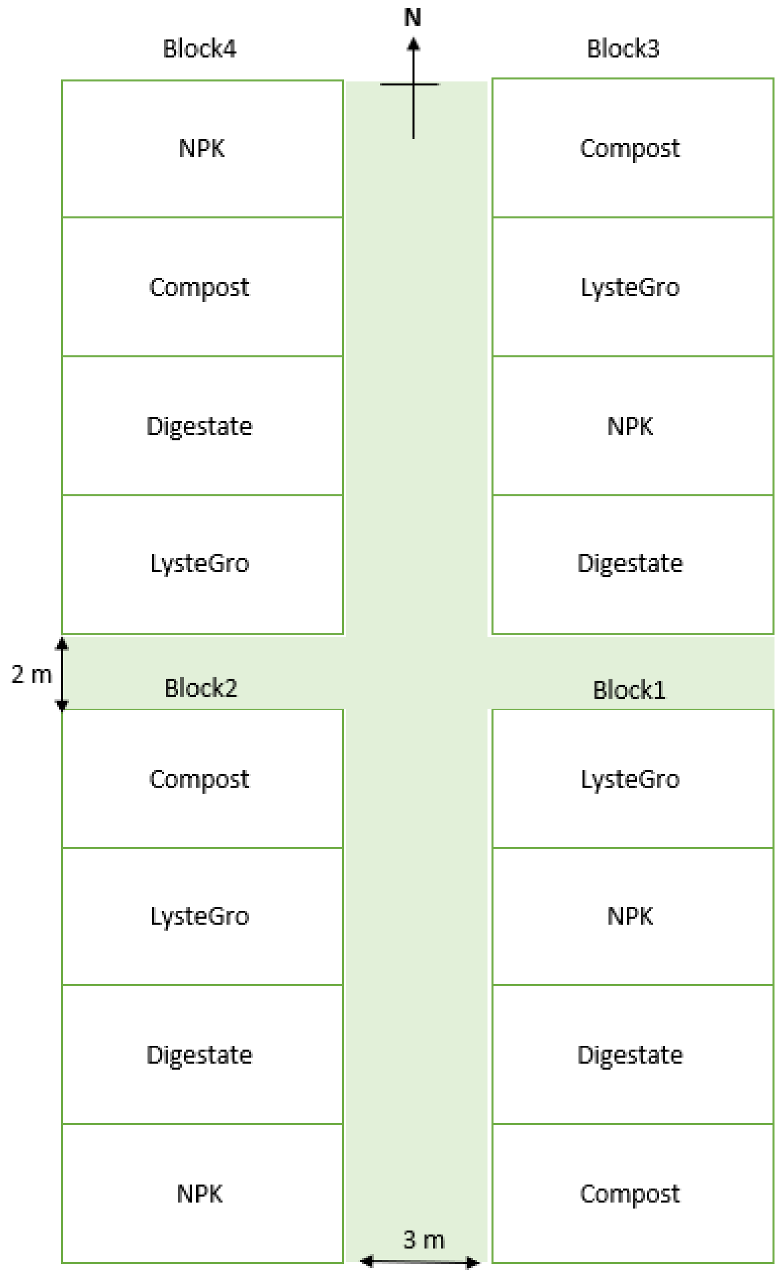
Wiesmeier, M., Poeplau, C., Sierra, C. A., Maier, H., Frühauf, C., Hübner, R., ... & Schilling, B. (2016). Projected loss of soil organic carbon in temperate agricultural soils in the 21 st century: effects of climate change and carbon input trends. *Scientific Reports*, 6(1), 1-17.

Yang, X., Reynolds, W. D., Drury, C. F., Fleming, R., Tan, C. S., Denholm, K., & Yang, J. (2013). Organic carbon and nitrogen stocks in a clay loam soil 10 years after a single compost application. *Canadian Journal of Soil Science*, 94(3), 357-363.

Yue, Q., Cheng, K., Ogle, S., Hillier, J., Smith, P., Abdalla, M., ... & Pan, G. (2019). Evaluation of four modelling approaches to estimate nitrous oxide emissions in China's cropland. *Science of the Total Environment*, 652, 1279-1289.

Appendices

Appendix 1. General field layout of the field experiment at the Emile A. Lods Agronomy Centre (Ste-Anne-de-Bellevue, Quebec) and the Elora Research Station (Elora, Ontario). Green-shaded area is plant-less buffer zone to separate the blocks. The plot size is 6*12 m^2 at Lods and 6*15 m^2 at Elora

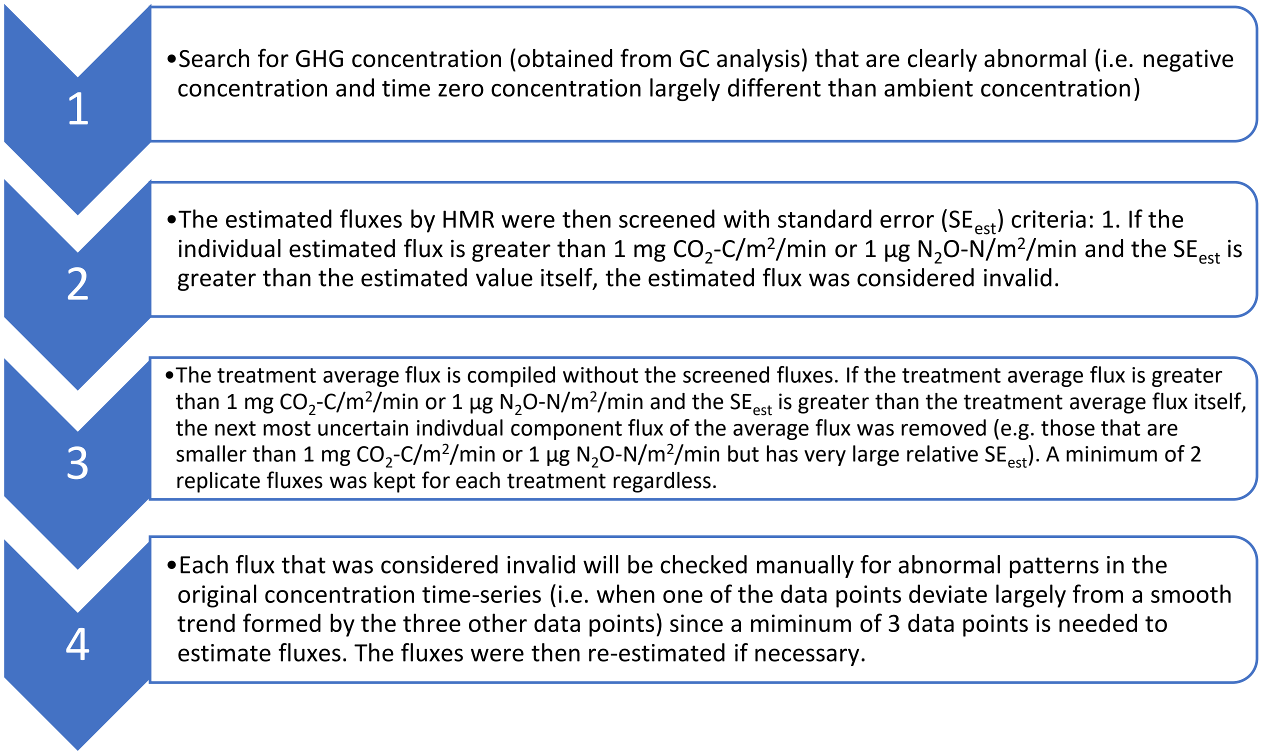


Appendix 2. Management schedule in 2018 (corn) and 2019 (soybean) at the Emile A. Lods Agronomy Centre (Ste-Anne-de-Bellevue, Quebec) and the Elora Research Station (Elora, Ontario).

2018		
Date	**Lods**	**Elora**
22 May		prepare soil for seeding (disc harrow)
24 May		All Organic fertilizer application: Composted food waste: 12t/ha LysteGro: 28000L/ha Digestate: 42000L/ha
25 May	prepare soil for seeding (disc harrow)	NPK application (170 kg N/ha; 27 kg P_2O_5/ha; 55 kg K_2O/ha) Fertilizer incorporation Seeding corn (78,500 /ha of DeKalb DKC-3855RIB; 75 cm inter-row spacin)
28 May		Herbicide application: Frontier Max (dimethenamid-p /Banvel (dicamba)/Atrazine tankmix (720/165/554 g active ingredient /ha)
30 May	Fertilizer application: NPK: K hand surfaced applied at 67 kg/ha (600g of 0-0-60 per plot) Composted food waste: 12t/ha LysteGro: 28000L/ha Digestate: 42000L/ha	
31 May	NPK: Urea at 50kg N/ha; 20 kg P_2O_5 /ha Fertilizer incorporation Seeding corn (DKC-3378-RIB, 76000 seeds per hectare, 75 cm inter-row spacing)	
14 June		Herbicide application: Roundup Weathermax (glyphosate) 1350g/ha
4 July	Herbicide application: Roundup Transorb 3L/ha	
5 July	Fertilization with urea, sidedress (120 kg N/ha)	
18 Oct		Corn harvest (grain only)

26 Oct	Corn harvest (leaving root and aboveground stem of about 5 cm)	
1 Nov		Moldboard plowing 20 cm deep
30 Nov	Moldboard plowing 20 cm deep	
2019		
Date	**Lods**	**Elora**
8 May	Field Cultivation (disc harrow) K fertilizer addition (64.5 kg KCl/ha)	
9 May	Seeding soybean (DKB 003-29, 450000 seeds per hectare, 75cm inter-row spacing)	
7 June		Field Cultivation (disc harrow)
12 June		Seeding soybean (DKB 003-29, 450000 seeds per hectare, 15cm inter-row spacing)
25 June	Herbicide application (Roundup WeatherMax); Shallow inter-row cultivation	
12 Sept	Soybean harvest (grain only), stalk residue chopped and disc harrowed	
25 Sept	Application of urea (20 kg N/ha) LysteGro: 15600 L/ha Digestate: 21800 L/ha Fertilizer incorporation	
27 Sept	Application of composted food waste (6 t/ha) and Fertilizer incorporation; winter wheat planting	
11 Oct		Soybean harvest (grain only), stalk residue chopped and disc harrowed

Appendix 3. Screening approach for eliminating uncertain data-points and estimated greenhouse gas fluxes.



Appendix 4. Standard error of estimate (SE_{est}) associated with CO_2 emission (g CO_2-C/m^2) and N_2O emission (mg N_2O-N/m^2), taking into account the uncertainties of flux estimation and linear interpolation.

	1st month emission (Lods)		*Growing-season emission (Lods)†*		*Growing-season emission (Elora) †‡*	
CO_2	2018	2019	2018	2019	2018	2019
NPK	5.6	2.6	18	14	23	33
Compost	5.5	3.7	29	21	20	23
LysteGro	4.1	2.3	18	16	19	31
Digestate	3.8	4.1	11	15	30	32
N_2O	2018	2019	2018	2019	2018	2019
NPK	5.2	7.3	57	15	16.1	13.3
Compost	6.7	7.3	16	33	3.7	11.4

LysteGro	18	11.1	25	17	13.8	16.3
Digestate	3.7	6.8	8	20	12.8	18.2

Appendix 5. DayCent parameters of initial model setup and history simulations plus assumed field management.

Parameters/management	Lods	Elora
NELEM	2	
EDEPTH	0.2	
IDETH	3	
IVAUTO	3 (forest)	
prdx(2)	0.34	0.57
Forest plow out (year)	1750	1860
Early history after plow out	1751 – 1900: wheat fallow, pea, barley, potato with manure application 1901 – 1947: corn-soybean-winter wheat with manure application	1861 – 1931: grazing pasture; 1932 – 1966: spring wheat-soybean-barley with manure application
Recent history	1948 – 2017: corn-soybean-spring wheat with urea fertilization	1967 – 1979: continuous corn with urea fertilization 1980 – 2017: corn-soybean-spring wheat with urea fertilization
clteff	Generally, the clteff of active and slow SOC was downregulated but the clteff of passive SOC was upregulated to reduce the equilibrium pool size of passive SOC*	

***** clteff adjustment were numerous and so difficult to document all. Interested researchers can request the model file directly.

Appendix 6. DayCent parameters included in calibration by PEST v13.0.

Parameters	Ranges	Initial value	Final value	Remarks
Site-specific parameters*				
Field capacity (0 – 10 cm) Lods †	0.24 – 0.3	0.270	0.264	0 – 10 cm refers to DayCent soil layer 1 – 3
Field capacity (10 – 20 cm) Lods †	0.24 – 0.3	0.284	0.24	10 – 20 cm refers to DayCent soil layer 4
Field capacity (0 – 10 cm) Elora †	0.33 – 0.39	0.359	0.378	
Field capacity (10 – 20 cm) Elora †	0.325 – 0.385	0.355	0.325	
Wilting points (0 – 10 cm) Lods †	0.12 – 0.15	0.142	0.15	
Wilting points (10 – 20 cm) Lods †	0.13 – 0.16	0.150	0.16	
Wilting points (0 – 10 cm) Elora †	0.15 – 0.18	0.168	0.15	
Wilting points (10 – 20 cm) Elora †	0.15 – 0.18	0.167	0.15	
Ksat (0 – 10 cm) Lods †	0.0004 – 0.0006	0.000501	0.0006	
Ksat (10 – 20 cm) Lods †	0.0003 – 0.0005	0.000388	0.0003	
Ksat (0 – 10 cm) Elora †	0.00075 – 0.0009	0.000826	0.0009	

Ksat (10 – 20 cm) Elora †	0.0006 – 0.00075	0.000684	0.00071	
fwloss(4)	0.6 – 1.0	0.8	0.644 (Lods); 0.997 (Elora)	
dmpflux	6.4E-6 – 8E-6	8E-6	8E-6 (Lods); 6.4E-6 (Elora)	
fleach(3)	0.2 – 0.5	0.45	0.313 (Lods); 0.200 (Elora)	
minlch	0.08 – 0.12	0.1	0.08 (Lods); 0.12 (Elora)	
prdx(1) corn	0.8 – 2.0	0.9	1.02 (Lods); 2.0 (Elora)	
prdx(1) soy	0.4 – 0.8	0.5	0.44 (Lods); 0.8 (Elora)	
ppdf(1) corn	25 – 35	30	27 (Lods); 25 (Elora)	
ppdf(1) soy	22 – 32	27	26.8 (Lods); 22 (Elora)	
himax corn Lods ‡	0.63 – 0.70	0.66	0.63	
himax soy Lods ‡	0.5 – 0.58	0.52	0.558	
himax corn Elora ‡	0.55 – 0.70	0.55	0.670	C content of grain borrowed from Lods
himax soy Elora ‡	0.55 – 0.60	0.58	0.562	C content of grain borrowed from Lods
efrgrn(1) corn Lods ‡	0.7 – 0.75	0.73	0.75	
efrgrn(1) soy Lods ‡	0.81 – 0.88	0.85	0.81	
efrgrn(1) corn Elora ‡	0.6 – 0.8	0.63	0.8	N content of grain borrowed from Lods
efrgrn(1) soy Elora ‡	0.81 – 0.88	0.88	0.88	N content of grain borrowed from Lods
pramn(1,2) corn	50 – 62.5	62.5	50 (Lods); 62.5 (Elora)	
pramn(1,2) soy	23 – 35	30	33.7 (Lods); 23.0 (Elora)	
snfxmx(1) soy	0.03 – 0.045	0.0375	0.045 (both)	
frtc(1) corn	0.4 – 0.5	0.5	0.4	Elora only due to overestimation of root biomass
frtc(1) soy	0.35 – 0.4	0.4	0.35	Elora only due to overestimation of root biomass
frtc(2) corn	0.05 – 0.1	0.1	0.05	Elora only due to overestimation of root biomass
frtc(2) soy	0.05 – 0.1	0.1	0.05	Elora only due to overestimation of root biomass
frtc(4) corn	0.16 – 0.24	0.2	0.16	Elora only due to overestimation of root biomass

wscoeff(1) corn	0.34 – 0.378	0.378	0.34	Elora only due to overestimation of root biomass
wscoeff(1) soy	0.34 – 0.378	0.378	0.34	Elora only due to overestimation of root biomass
Generalizable parameters*				
dmp	0.0025 – 0.0035	0.003	0.0025	
astlig Compost	0.05 – 0.3	0.2	0.3	Ranges inferred from knowledge about how much labile OM it likely has
astlig LysteGro	0.05 – 0.2	0.15	0.2	Ranges inferred from knowledge about how much labile OM it likely has
clteff(1)K	4 – 12	10	4	K = Moldboard plow to incorporate crop residue
clteff(2)K	4 – 12	10	4	
clteff(3)K	1 – 10	1	1	
clteff(4)K	4 – 12	10	4	
clteff(1)I	4 – 8	6.67	8	I = Offset disk for fertilizer incorporation
clteff(2)I	4 – 8	6.67	8	
clteff(3)I	1 – 6	1	6	
clteff(4)I	4 – 8	6.67	4	
clteff(1)D	2.5 – 5	3.41	2.5	D = field cultivation to prepare seedbed
clteff(2)D	2.5 – 5	3.41	5	
teff(1)	12 – 18	15.40	12	
teff(2)	9.5 – 14	11.75	9.66	
teff(3)	24 – 36	29.70	35.4	
dec2(2)	12 – 24	18.5	24	
dec3(2)	9 – 13	11	10.67	
dec4	0.003 – 0.004	0.0035	0.004	
dec5(2)	0.1 – 0.3	0.2	0.3	
maxnitamt	0.391 – 0.450	0.391	0.426	
wfpsdnitadj	1.16 – 1.74	1.45	1.16	
n2n2oadj	0.5 – 0.75	0.617	0.623	
netmn_to_no3	0.1 – 0.35	0.3	0.35	
no3_n2o_x	0.63 – 1.17	0.899	0.63	Specific to Daycent vDec2016
no3_n2o_y	1.05 – 1.95	1.501	1.95	Specific to Daycent vDec2016
no3_n2o_step	2.05 – 3.8	2.927	2.05	Specific to Daycent vDec2016
no3n2oslope	4.2 – 7.8	5.992	4.2	Specific to Daycent vDec2016

* Site-specific parameters in this study were the those calibrated separately for the two sites. Generalizable parameters were parameterized from Lods dataset.

† Initial values were estimated by the soil hydrologic algorithm developed by Saxton and Rawls (2006).

‡ Initial values calculated based on C and N allocation of actual harvested samples. Note however, himax is a “maximum” grain harvest index rather than the actual harvest index.