Impacts of organic cropping systems on greenhouse gas emissions, soil mineral nitrogen, and crop yields in field crop production in Québec - Impacts de systèmes culturaux biologiques sur les émissions de gaz à effet de serre, l'azote minéral du sol et les rendements en grandes cultures au Québec

Mémoire

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Sous la direction de :

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Résumé

L'agriculture biologique vise à améliorer la durabilité des systèmes culturaux, cependant, certaines pratiques de conservation des sols utilisées peuvent augmenter les émissions de gaz à effet de serre (GES), sous certaines conditions environnementales. L'objectif de ce projet était de déterminer l'effet de différents systèmes culturaux en grandes cultures biologiques sur les émissions de GES et sur le rendement des cultures. Un essai au champ a été réalisé à l'Institut national d'agriculture biologique, au cours de deux saisons de croissance (26 avril au 31 octobre 2019 et 29 avril au 12 novembre 2020), sur un loam sableux. Le dispositif expérimental en blocs complets aléatoire incluait deux témoins comparatifs (prairie permanente et jachère en sol nu [JSN]) et cinq systèmes culturaux combinant différentes (i) séquences culturales (orge [*Hordeum vulgare* L.]-maïs [*Zea mays* L.], soya [*Glycine max* (L.) Merr.]-blé de printemps [*Triticum aestivum* L.], maïs-soya); (ii) sources fertilisantes (fumier de poulet [FP] et/ou engrais vert en dérobée [EV] ou aucun apport); et (iii) intensités de travail primaire du sol (labour avec charrue à versoirs [LB] ou chisel [CH]). La température, la teneur en eau et les concentrations en azote minéral du sol ont été mesurées périodiquement, de même que les émissions de protoxyde d'azote (N_2O) et méthane (CH_4) à l'aide de chambres statiques à régime variable. Le système CH-EV a généré les plus faibles émissions de N_2O cumulatives en 2019 et 2020 et les systèmes LB-FP en 2019 et JSN en 2020 ont généré les émissions les plus élevées. Les émissions de CH_4 ont été équivalentes entre les différents systèmes. Le système CH-EV a minimisé les émissions de N_2O à l'échelle de la superficie sans augmenter les émissions de N_2O à l'échelle du rendement. Cependant, l'évaluation à long terme de ces systèmes culturaux est nécessaire pour déterminer les bénéfices agronomiques, économiques et environnementaux.

Abstract

Organic farming aims to enhance the sustainability of cropping systems, but some soil conservation practices implemented may increase greenhouse gas (GHG) emissions. The main objective of this study was to determine the effects of various organic cropping systems on GHG emissions and crop yields, in Québec, Canada. A field experiment was conducted at the Institut national d'agriculture biologique, over two growing seasons (26 April to 31 October 2019 and 29 April to 12 November 2020), on a sandy loam soil. The randomized complete block design included two controls (perennial forage and bare fallow [BF]) and five organic cropping systems combining different: (i) crop sequences (barley [*Hordeum vulgare* L.]- grain corn [*Zea mays* L.], soybean [*Glycine max* (L.) Merr.]- spring wheat [*Triticum aestivum* L.], grain corn-soybean); (ii) sources of fertilizers (poultry manure [PM] and/or a fall-seeded green manure [GM] or no source); and (iii) primary tillage intensities (moldboard plough [MP] or chisel plough [CP]). Soil temperature, water content, and mineral N concentrations were evaluated periodically, as well as direct nitrous oxide (N_2O) and methane (CH_4) emissions, which were quantified using non-flow-through non-steady-state chambers and gas chromatography. The lowest cumulative N_2O emissions were found in CP-GM (0.52 ± 0.11 and 3.55 ± 0.72 kg N ha^{-1} in 2019 and 0.47 ± 0.06 kg N ha^{-1} in 2020), whereas the highest emissions were found in MP-PM in 2019 (3.55 ± 0.72 kg N ha^{-1}) and BF in 2020 (1.44 ± 0.20 kg N ha^{-1}). During both years, CH_4 emissions varied from -0.65 to +0.18 kg C ha^{-1} and were similar between cropping systems. Organic cropping system CP-GM minimized the area-scaled N_2O emissions without increasing the yield-scaled N_2O emissions. However, long-term assessment is necessary to determine the agronomic, economic, and environmental benefits of these cropping systems.

Table of Content

List of Tables

List of Figures

List of Abbreviations

BF: bare fallow
C: carbon
CH: chisel
CH_4: methane
CO_2: carbon dioxide
CO_2 eq: carbon dioxide equivalent
CP: chisel plough
CT: conventional tillage
EF: emission factor
EV: engrais vert en dérobée
FP: fumier de poulet
GE: grain equivalent
GES: gaz à effet de serre
GHG: greenhouse gas
GM: green manure
GWP: global warming potential (cumulative radiative forcing of various greenhouse gases in CO_2 eq over a specific time period)
HIP: hole-in-the-pipe
JSN: jachère sol nu
LB: labour avec charrue à versoirs
MP: moldboard plough
N: nitrogen
N_2: dinitrogen
N_2O: nitrous oxide, protoxyde d'azote
NH_3: ammonia
NH_4: ammonium
NO: nitric oxide
NO_2: nitrite
NO_3: nitrate
NM: no manure
NT: no-tillage, no-till
N_{up}: aboveground crop N uptake
OM: organic matter
PF: perennial forage
PM: poultry manure
PMGM: poultry manure and fall-seeded green manure
PO_4^{3-}: phosphate
RT: reduced tillage
SE: standard error
SOC: soil organic carbon
VSWC: volumetric soil water content
WFPS: water-filled pore space

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Preface

I am first author of the scientific article inserted in this thesis (Chapter 3), written in collaboration with six co-authors: Dr. David Pelster, M.Sc. Gilles Gagné, M.Sc. Julie Anne Wilkinson, Dr. Martin Chantigny, Dr. Denis Angers, and Dr. Caroline Halde. The experimental design and treatments were primarily conceptualized by the co-authors above-mentioned. My contribution to this study on GHG emissions in organic field crops consisted of field and laboratory data collection, statistical analysis, and the writing of the scientific paper to be submitted in spring 2022 to the journal Agriculture, Ecosystems & Environment. At the date this thesis was achieved, the inserted article was still in a reviewing process with the co-authors, hence, the version of the article inserted here might differ to the version of the article to be published.

Introduction

With the perspective of rising global consumer demands for meat and biofuel in a context of climate change, it is crucial to develop agroecosystems that are highly efficient in terms of crop productivity and environmental sustainability. Thermal radiation from the Earth's surface is being absorbed by greenhouse gases (GHG) in the atmosphere (Saha et al., 2018). They, in turn, radiate it back to the surface of the Earth and, consequently, contribute to the rise of ambient air temperature. Facing a continuous increase of GHG concentrations in the atmosphere, 191 out of 197 countries have now ratified the Paris agreement with the goal to avoid a rise in the global temperature of more than 2°C relative to pre-industrial levels, and pursue the efforts to keep it below 1.5°C. However, the targets of the Paris agreement should be modified to considerably reduce GHG emissions to limit the rise of global temperature below 1.5°C if we wish to avoid catastrophic and irreversible impacts worldwide, whereas human activities have already caused an increase of approximately 1°C (IPCC, 2018). Drastic changes in human activities are thus needed, and it is urgent that all sectors from every country contribute to reduce anthropogenic GHG emissions.

Nitrous oxide (N_2O) is a potent GHG associated with ozone depletion (Ravishankara et al., 2009) and has a global warming potential (GWP) 265 times that of carbon dioxide (CO_2) on a 100-year timescale (IPCC, 2014). In agriculture, N_2O emissions may come from sources that directly produces them (e.g., direct N_2O emissions from nitrogen [N] additions to soils), or may be related to a source without being directly produced by this source (e.g., indirect N_2O emissions after transport of anthropogenic N in water bodies). Direct and indirect N_2O emissions from agriculture represent 52% and 18%, respectively, of the total anthropogenic N_2O emissions and are steadily increasing due to N fertilizer application (Tian et al., 2020). Carbon dioxide emissions in agriculture are more complex to measure accurately, and the importance of net CO_2 emissions would be lowered by C sequestration in agricultural soils. As a result, the agricultural sector's contribution is estimated to be less than 1% of global anthropogenic CO_2 emissions (Smith, 2012). However, the agricultural sector is the largest emitter of anthropogenic methane (CH_4) emissions (Saunois et al., 2020). Methane has a GWP 28 times higher than that of CO_2 to absorb heat radiation on a 100-year timescale (IPCC, 2014). During the 2008-2017 period, up to 56% of the total anthropogenic CH_4 emissions were from the agriculture and waste category, *i.e.*, from livestock enteric fermentation, manure management, rice cultivation, landfills and wastewater handling (Saunois et al., 2020). But dryland soils absorb 38 Tg CH_4 yr^{-1}, representing 6.8% of total CH_4 consumption, and consequently, constitute a major sink for CH_4.

In 2019, the agricultural economic sector accounted for 10% of Canada's total anthropogenic GHG emissions (Environment Canada, 2021). The agricultural sector contributed mainly in the form of N_2O and CH_4 emissions, with a share of 78% of and 24% of national anthropogenic emissions, respectively. Inorganic and organic N fertilizer application to soils and crop residue decomposition represented 41% of agricultural GHG emissions, mostly as N_2O emissions, while N_2O and CH_4 emissions from animal manure management and enteric fermentation contributed respectively to 41% and 13% of total agricultural emissions (Environment Canada, 2021). Environment Canada reports that because of the recent intensification of cropping systems, the increase in inorganic N fertilizers use has led to a 98% increase of the related GHG emissions between 1990 and 2019. The main driver of the increased application of inorganic N fertilizer was a shift in the industry across the country, from perennial forage crop production to annual crop production caused by an increase in grain prices. As a result, proportion of total GHG national emissions from agriculture emitted as N_2O has increased by 30%, while the proportion as CH_4 has decreased by 32% (Environment Canada, 2021). Reducing agriculture's environmental footprint through improved management and input options is thus crucial if Canada is to meet its commitments under the Paris agreement.

Similarly, in the province of Québec, 9.2% of the total provincial GHG were emitted by the agricultural sector in 2019 with 39.9% emitted as N_2O (MELCC, 2021a). These N_2O emissions were mainly related to soil and manure management, representing respectively 79% and 21% of the N_2O emissions from the agricultural sector. Methane emissions contributed to approximately 55.5% of the sector's emissions and resulted mainly from enteric fermentation (67%) and manure management (33%), while CO_2 emissions from liming, urea-based fertilizers, and C-containing fertilizers represented 4.5% of the sector's emissions. Moreover, according to MELCC (2021a), a 24.4% increase in GHG emissions related to soil management between 1990 and 2018 (2.0 to 2.5 Mt CO_2 eq) was due to an increase in conservation tillage practices and N fertilizer application. Improving farm management under the environmental conditions of eastern Canada could thus contribute to reducing the sector's environmental footprint, in the context of commitments announced by the various levels of government.

Nitrous oxide emissions are produced through nitrification and denitrification processes in soils. These processes are influenced by N and carbon (C) substrate availability, soil water saturation, climatic conditions, and environmental factors, such as soil properties (Davidson et al., 2000). Nutrient management, tillage intensity, and crop sequence influence N and C availability, soil aeration, and conditions of oxidoreduction, and thus, affect N_2O formation in the soil. Labile organic C provided in manure application is expected to enhance the activity of nitrifiers and denitrifiers and promote N_2O production (Thangarajan et al., 2013). The increased availability of N and C substrates from manure would stimulate O_2 consumption by other soil microbes, creating anoxic conditions in soil microsites that foster denitrification. Soil texture and drainage are important regulating factors in GHG emissions for their effect on soil aeration, which is also influenced by primary tillage intensities. Conventional tillage and conservation tillage practices differently alter N_2O emissions depending on the soil texture (Pelster et al., 2021; Rochette et al., 2008a). Crop sequence with various crop types requiring different amounts of N fertilizer and providing different quantities of crop residues to the soil is also influential for GHG emissions (Drury et al., 2008).

In a cool and temperate climate, such as the climate of the province of Québec, N_2O emissions are further influenced by the amount of precipitation, potential evapotranspiration, air temperature, and freeze-thaw events (Rochette et al., 2018). The province of Québec is a humid region, characterized by a ratio of growing season precipitation to potential evapotranspiration varying between 0.7 and 1.2. Grain corn (*Zea mays* L.) and soybean (*Glycine max* [L.] Merr.) are the main cash crops of the province cultivated for animal production, while spring wheat (*Triticum aestivum* L.) and barley (*Hordeum vulgare* L.) are common crops, but with a lower economic importance for the industry (ISQ and MAPAQ, 2020). Perennial forage crops are also grown in rotations of livestock-based cropping systems of the province. Dairy, swine and poultry are important animal productions for the provincial industry and contribute to N_2O and CH_4 emissions during manure management and enteric fermentation.

In European countries and United States, organic farming is increasingly being adopted (USDA, 2020; Willer et al., 2019). A similar expansion of organic farming in Canada is observed, with the highest number of organic farms recorded in the Québec province, in 2016 (Statistics Canada, 2017). In the last decade in Québec, the number of organic farms has more than doubled and now represents approximately 9% of the total number of farms. Between 2017-2018, the *Conseil des appellations réservées et des termes valorisants* (CARTV) has reported a 17% increase in certified organic cropped areas in Québec (CARTV, 2020). Organic farming aims to generate an income for farmers from food production while maintaining or improving soil fertility, and preserving biodiversity, the environment and human health (IFOAM, 2008). Strict standards of production are established to optimize agroecosystems that are ecologically, socially, and economically sustainable (CGSB, 2020; Codex Alimentarius, 1999). Recent scenarios determined the important contribution of organic farming in developing sustainable agroecosystems that reduce N inputs and N losses (Barbieri et al., 2021; Billen et al., 2021). However, an on-going debate is questioning the environmental impact of organic cropping systems, which are generally less intensive and less productive than conventional cropping systems (Baudron and Giller, 2014; Clark and Tilman, 2017). A meta-analysis comparing organic and conventional cropping systems revealed that organic systems reduced area-scaled N_2O emissions, but increased yield-scaled emissions (Skinner et al., 2014). A few recent studies demonstrated that equivalent yield-scaled N_2O emissions could be achieved in organic cropping systems by improving crop productivity and lowering area-scaled N_2O emissions (Biernat et al., 2020; Skinner et al., 2019).

In Québec, the provincial sustainable development plan aims to enhance soil conservation and soil health through a build-up of soil organic matter (OM) content and an increased use of organic N sources like cover crops and crop residues (MAPAQ, 2020). Fertilization methods in organic cropping systems are limited to organic N sources, which are strategies considered to restore soil fertility and protect the environment, while increasing farmer's income (Diacono and Montemurro, 2010; Gattinger et al., 2012). However, organic N sources also affect the activities of nitrifiers and denitrifiers in the soil and may, under certain environmental conditions, significantly impact the N_2O emissions at the system level. Soil GHG measurement in cropping systems is thus an important indicator in the development of sustainable cropping systems promoted by the provincial agri-food policies.

Given the steady expansion of organic farming in Québec and the lack of comprehension of its environmental effects, a better understanding of GHG emissions from these systems is required to develop appropriate mitigation strategies in the pedo-climatic conditions of eastern Canada. Hence, we evaluated the first two years marking the initial stage of the first long-term organic trial in the province of Québec, Canada. This study examined the effect of organic cropping systems on N_2O emissions and their driving environmental factors (soil mineral N, soil temperature, and moisture at the soil surface), on CH_4 emissions and on crop yields.

Chapter 1 Literature review

1.1 Biochemical processes involved in soil-derived greenhouse gas emissions

Soil GHG emissions are influenced by various ecosystem. Agriculture is a managed ecosystem that can strongly impact regional N and C budgets through a number of management decisions such as additions of fertilizers, tillage, selection of crop rotations, removal of crop grains and residues. While these changes to the regional nutrient budgets can affect GHG emissions, management practices can alter the soil properties, which may also affect soil GHG emissions (Butterbach-Bahl et al., 2013). An understanding of the biogeochemical processes involved in soil GHG emissions provides a starting point to interpret the various effects of agricultural practices on soil GHG emissions. The first section of this literature review describes the main biogeochemical processes that result into the production of GHG in upland soils under a cool temperate climate. The emphasis will be on N_2O and how its formation is influenced by the way different environmental factors affect the soil microbial production and consumption processes. This will be followed by an overview on how the soil environment can alter soil CH_4 fluxes. Carbon dioxide will only be briefly discussed as the change in soil C balance could not be measured during this two-year experiment.

1.1.1 Nitrous oxide

The soil ecosystem has an important role in the N cycle. When affected by human activities, it may represent a non-point source of pollution (Schlesinger and Bernhardt, 2020). Nitrogen transformation in the soil by microbes may result in nitrate (NO_3^-) leaching and in the emission of various nitrogenous gases, such as ammonia (NH_3), nitric oxide (NO), N_2O, and dinitrogen (N_2). Nitrous oxide is among the three most important GHG globally, after CO_2 and CH_4. Nitrous oxide is a powerful GHG with a GWP 265 times greater than CO_2 over a 100 year time-scale (IPCC, 2014). This gas is also the most important substance depleting stratospheric ozone (Ravishankara et al., 2009). Direct N_2O emissions from the agricultural sector represents 52% of the global anthropogenic N_2O emissions, representing a contribution of 2.5 to 5.8 Tg N yr^{-1} to the atmosphere (Tian et al., 2020). Denitrification is responsible for the returning of N_2 to the atmosphere, but both nitrification and denitrification are important processes in the global N cycle.

1.1.1.1 Microbial processes

Amino acids and simple organic-N molecules are first released from decomposing materials by microbes, starting the process of N mineralization (Schlesinger and Bernhardt, 2020). These simple organic N molecules can then be broken down into soil mineral N (primarily ammonium [NH_4^+], and NH_3) in a process called ammonification which provides primary substrates for nitrification and denitrification.

Nitrification is a microbial process in which NH_4^+ is oxidized to nitrite (NO_2^-) and NO_3^- and during which N_2O can be lost as a by-product (Fig. 1), as a result of chemical decomposition or nitrifier-denitrification (Butterbach-Bahl et al., 2013). Some of the NH_4^+ from organic forms released in the process of ammonification may undergo nitrification under the action of chemoautotrophic bacteria (*Nitrosomonas* and *Nitrobacter* genera) that fix C, or other specialized group of prokaryotes (archaea) (Butterbach-Bahl et al., 2013; Hu et al., 2015). In other cases, oxidation of NH_4^+ to NO_3^- is achieved in heterotrophic nitrification, but this process would be responsible for a much lower fraction of the N_2O produced than autotrophic nitrification (Thangarajan et al., 2013). Extractable soil mineral N (NH_4^+ and NO_3^-) indicates the net results of ammonification and nitrification, therefore, interpretation of the NH_4^+ concentrations needs to take into account the rates of both processes (Schlesinger and Bernhardt, 2020). Moreover, NH_4^+ can be taken up by plants and be lost as NH_3, especially in alkaline soils or alkaline microsites within normally acidic soils (e.g. soils with band-applied urea). Nitrate concentration in agricultural soils is also highly variable as it can be taken up by plants, lost in NO_3^- runoff or in emissions of nitrous gases during nitrification and denitrification.

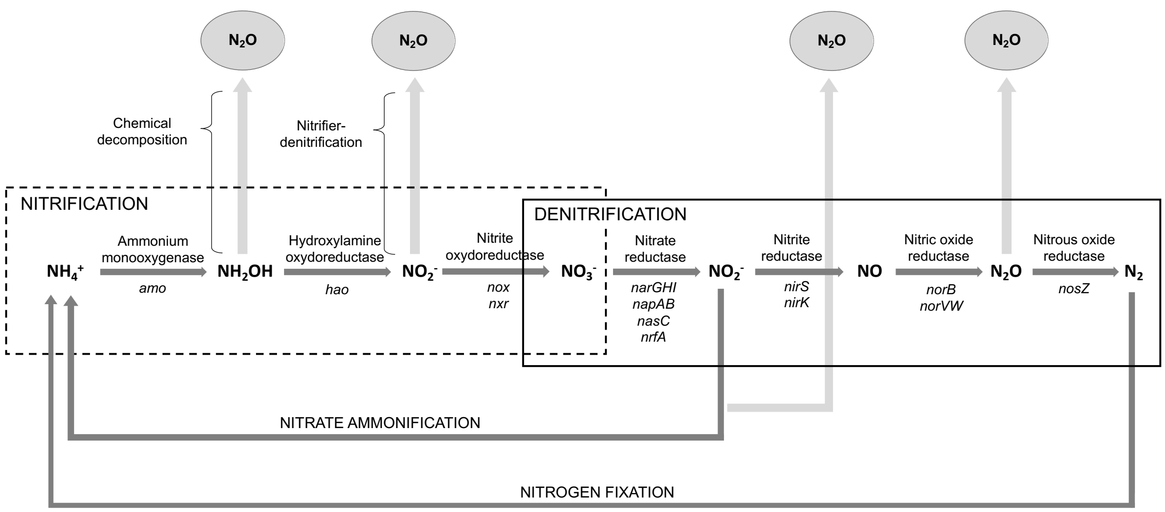

Fig. 1. Nitrification and denitrification processes, their relevant enzymes and associated genes.

During the denitrification process, microorganisms reduce NO_2^- and NO_3^- to NO, N_2O and N_2 under the action of various enzymes, as shown in Fig.1 (Butterbach-Bahl et al., 2013; Conrad, 1996; Firestone and Davidson, 1989). Consequently, there is a close relationship between soil N availability, N cycling, and N-oxide fluxes. Nitrous oxide is an intermediate in the reductive sequence of denitrification. When organic C and N-oxides are present with limited O_2 availability, the general requirements are fulfilled for denitrification to occur (Firestone and Davidson, 1989). *Pseudomonas* species are the most predominant species found in the environments where denitrification occurs, with *Alicaligenes* species being the second most important, but various microbial metabolic pathways contribute to the formation or consumption of N_2O. Currently known biotic and abiotic processes involved in N_2O formation were summarized by Butterbach-Bahl et al. (2013).

1.1.1.2 Factors influencing microbial processes

Proximal and distal factors

While O_2 and NH_4^+ availability control nitrification rates at the cellular level, O_2, NO_3^-, and organic C availability control denitrification rates (Firestone and Davidson, 1989). These cellular controllers, also defined as proximal factors, are in turn influenced by properties of the different ecosystems. The physical, chemical, and biological properties characterizing the ecosystems are in relation with environmental factors and are referred to as distal factors. Over the last decades, these relations have been studied to understand how they affect the proportions of the end products (Butterbach-Bahl et al., 2013; Chapuis-Lardy et al., 2007; Clough et al., 2005; Hu et al., 2015).

Hole-in-the-pipe model

Trace N-gas production by nitrification and denitrification is regulated through the control of process rates and the ratio of their end products (NO or N_2O). Firestone and Davidson (1989) conceptualized the "hole-in-the-pipe" (HIP) model illustrating the movement of N in the microbiological pathways using the representation of a "process pipe", through which the trace N-gas are produced, and their two main levels of regulation (Fig. 2). The HIP model describes how N_2O losses are related to the rates of the nitrification and denitrification processes, *i.e.*, the size of the pipe, and to the proportion of reactive N to end products, *i.e.*, the size of the holes in the pipes through which trace-N gases leaks.

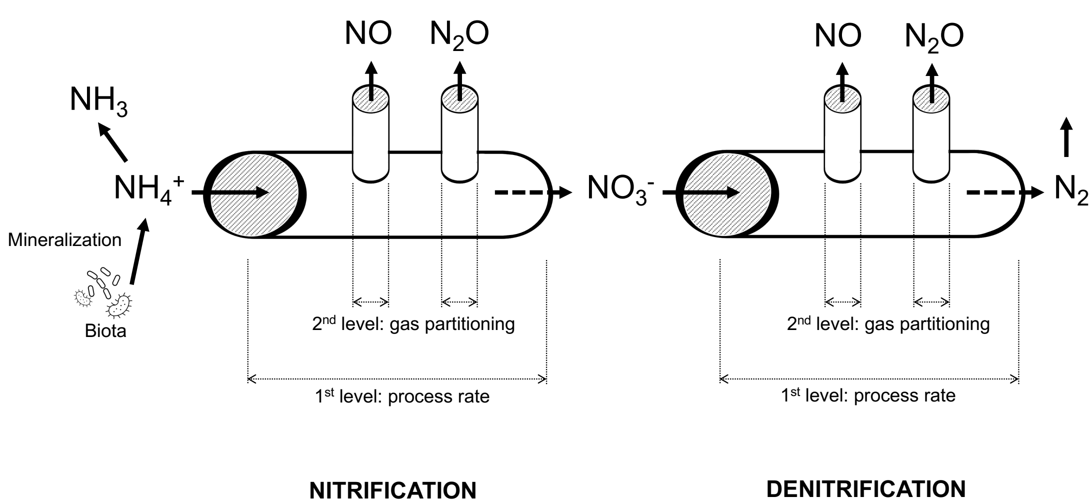

Fig. 2. The hole-in-the pipe model illustrating the microbial processes that yield nitrogenous gases during nitrification and denitrification in the soil and the two levels of regulation. Adapted from Firestone and Davidson (1989).

At the first level of regulation are the proximal factors controlling the rates of nitrification and denitrification, analogous to the rate of the N flow through the ecosystem. The size of the pipe is controlled by rates of N cycling and N availability (Davidson et al., 2000; Firestone and Davidson, 1989).

The factors controlling the N-trace gas leaks through the holes of the pipes are at the second level of regulation and control the partitioning of the N-gas produced (NO, N_2O, N_2), together with microbial consumption and diffusion rates in the soil. At this level of the HIP model, environmental factors affect the proportion of N_2O produced, quantified as a ratio with the N_2 produced in denitrifying microorganisms and soils (Firestone and Davidson, 1989). The size of the holes in the pipes is controlled by soil water content, pH, C content, NO_3^- concentration, sulfate content, and types and quantities of microbes. Soil water content would be the most important factor affecting the partitioning of the N-oxide losses from nitrification and denitrification (Davidson et al., 2000). Increasing NO_3^-, NO_2^-, or O_2 concentrations, and decreasing C availability, pH or temperature will increase the ratio of N_2O:N_2 (Firestone and Davidson, 1989; Hu et al., 2015). Factors identified as increasing the N_2O:N_2 ratio are based on the relative availability of reductants (organic C) to oxidants (N-oxide). The outcome depends on the cellular controller limiting a given environment. The higher availability of an oxidant over the availability of a reductant causes an increase in incomplete reduction, resulting in more N_2O being emitted from the soil.

Factors controlling nitrification

Several factors, cellular or environmental controllers, affect the chemoautotrophic nitrifying bacteria that oxidize NH_4^+, including pH, temperature, water potential, NH_4^+, NO_2^-, phosphate (PO_4^{3-}), and O_2 concentrations, and some allelopathic compounds (Firestone and Davidson, 1989). Although NH_4^+ is known to be the most important cellular controller in nitrification rates, O_2 is necessary for the ATP-yielding in the oxidation process, and its availability also influences nitrification rates. When O_2 is limiting during nitrification, N_2O is produced from the reduction of NO_2^- by nitrifying bacteria (Firestone and Davidson, 1989; Hu et al., 2015). The purpose of this reaction is to avoid the accumulation of NO_2^-, which may be toxic for nitrifiers. The supply of NH_4^+ and O_2 to microbes through diffusion is controlled by water, although the capacity of microbes to process the substrates may limit the process rates. Re-wetting of soils, also known as the "Birch effect", (Birch, 1958, 1959, 1960), alters the cellular controllers in the nitrification process. Re-wetting increases mineralization and nitrification rates, resulting in a large increase in N availability for nitrifiers, hence, increasing N_2O production.

Environmental factors controlling NH_4^+ availability also include temperature, plant uptake, mineralization and immobilization rates, and cation exchange capacity (Firestone and Davidson, 1989; Hu et al., 2015; Robertson, 1989). The heterogeneity of NH_4^+ distribution in microsites, the distribution of organisms and roots, together with mineralization rates, are factors that allow significant nitrification rates from nitrifying bacteria in undisturbed grassland soils, despite high competition with plant uptake and microbial immobilization (Jackson et al., 1989). An increasing pH and a decreasing O_2 availability will increase the proportion of N_2O produced as an end product and affect the size of the holes in the pipe in the HIP model. But these two factors also reduce NH_4^+ oxidation rates, therefore affecting the movement of N through the pipe. The regulation at both levels of the HIP model makes the prediction of N_2O emissions more difficult. Although rates of N_2O production are generally associated with denitrification, N_2O produced during nitrification is a constant process that may be overall significant even at a small N_2O:NO_3^- ratio, which are typically under 1% but can reach 20% (Firestone and Davidson, 1989).

The water-filled pore space (WFPS), which describes the proportion of soil pores that are filled with water, adequately describes the redox potential of a soil and is a good indicator of favourable conditions for denitrification to occur (Rochette et al., 2018). The WFPS is used for comparisons across soils of different textures as the water holding capacity highly varies among them. A WFPS between 30% and 60-70% would be optimal conditions for N_2O production via nitrification-related pathways (Hu et al., 2015).

Factors controlling denitrification

There are three influential cellular controllers in the denitrification process: O_2, N-oxides (NO_3^-, NO_2^-, NO or N_2O), and organic C availability. The preponderant importance of a controller will depend on the specific environment characterized by soil type, climatic conditions, and fertilizer type applied (Robertson, 1989). The complexity of the process regulation resides in the numerous environmental factors that regulate cellular controllers. Oxygen availability is the most common factor limiting denitrification in upland soils, and is mainly controlled by rainfall and evapotranspiration through filled pores and water films (Hu et al., 2015). Although denitrification is an anaerobic process, it also requires NO_3, which is produced during the nitrification process in aerobic conditions. Consequently, production of N_2O during denitrification is known to be high in anoxic soils, while wetland soils may allow net N_2O consumption through complete denitrification to N_2 (Firestone and Davidson, 1989). Also, plant roots may modify the soil microenvironments in several ways. For example, roots consume O_2 during respiration, thus, creating anaerobic zones. Oppositely, by removing water, they increase the diffusion of O_2 during evapotranspiration.

Previous studies reported N_2O emissions were primarily due to denitrification when the WFPS exceeded 60% WFPS and to nitrification when the WFPS was lower than 60% (Thangarajan et al., 2013). When WFPS ranges from 30% to 80%, denitrification can contribute to N_2O losses, with an overlap with nitrification N_2O losses up to about 60% WFPS (Davidson et al., 2000; Hu et al., 2015). At more than 80% WFPS levels, complete heterotrophic denitrification (*i.e.*, reduction of N_2O to N_2) starts to be the dominant process due to lower nitrification and gas diffusivity rates, and thus, less N_2O and more N_2 are produced.

The effect of WFPS on N_2O emissions also depends on nutrient availability in the soil matrix and the metabolic activity of microbial cells. As previously mentioned, re-wetting of soils induces a large increase in N and C availability from increased mineralization and nitrification rates and can also provide the substrates for denitrification, increasing N-gas production. Similarly, soil freezing-thawing stimulates substrates availability for denitrifiers by causing the lysis of microbial cells, which increases the availability of labile N and C in the soil (Ivarson and Sowden, 1970; Koponen and Martikainen, 2004; Wang and Bettany, 1993) and induce N_2O emissions during the spring (Pelster et al., 2013; Wagner-Riddle and Thurtell, 1998).

Nitrate, although being an important controller of denitrification, is less often the limiting factor in fertilized soils, especially in well-aerated coarse-textured soils (Firestone and Davidson, 1989; Rochette et al., 2018). However, the accumulation of nitrate and nitrite in fertilized soils influence N_2O reduction by inhibiting the N_2O reductase activity, and therefore, cause N_2O to be produced in a greater proportion in the ratio of the products of denitrification (N_2O:(N_2O + N_2); Firestone et al., 1979). In fertilized croplands, denitrification may be more limited by organic C than by NO_3^- although denitrification is most often limited by the presence of O_2. Organic C drives the oxidation of N-oxides in the production of N_2O during denitrification. In unfertilized soils though, denitrification may be more limited by a lack of NO_3^- than by a lack of organic C.

Soil temperature is another environmental factor that may affect N_2O production by controlling the process rate, although denitrification may be observed in soils between 2°C and 50°C (Firestone and Davidson, 1989). Increased soil temperatures stimulate denitrification by affecting enzymatic processes and by favouring respiration rates in soil, therefore depleting soil O_2 concentrations (Butterbach-Bahl et al., 2013). Moreover, the efficiency of the nitrous oxide reductase enzyme (Fig. 1) is decreased under acidic conditions, consequently increasing the N_2O:(N_2O + N_2) ratio. Hence, soil pH is negatively correlated with N_2O emissions in field experiments (Rochette et al., 2018). But environmental factors are interactive and may impact several cellular controllers at the same time. For example, plants influence denitrification through alteration of O_2 levels in soils, while the supply of C through root exudation for microbial respiration and the NO_3^- uptake by roots also interact with the denitrification process in soils.

Integration of production, consumption, and diffusion

Another important aspect considered in the HIP model is the integration of production, consumption and diffusive transport in a given environment (Davidson et al., 2000). The density of cells involved in denitrification will affect the rate of N_2O production and consumption (Firestone and Davidson, 1989). The biological consumption of proximal controllers to produce N_2O will also depend on the emission rate of the gas from the soil. The air-water distance and the diffusion rate will affect the rate of NO and N_2O movement out of the solution. Nitric oxide, an intermediate gas produced in the process of denitrification, will be influent on the net N_2O produced and will be similarly affected through production, consumption, and rates of the movement out of solution. In a dry and well-aerated soil, the more oxidized gas (NO) will be emitted before it can be consumed (Davidson et al., 2000). But in such environments, the oxidative process of nitrification dominates and the gas diffusivity is high. In wet but unsaturated soils, low diffusivity and anoxic conditions will enhance N_2O formation. In an even more water-saturated soil, anaerobic conditions will favour complete reduction to N_2 by denitrifiers.

1.1.1.3 Nitrous oxide emissions and climate change

Based on recent observations and climate projections of global warming scenarios, models predict that wet areas will experience even wetter conditions, especially in the mid- to high-latitudes (Trenberth, 2011). With warming temperatures, risks of runoff and flooding are increased in early spring, as more precipitation is received in the form of rain, while snow melts earlier. In the U.S. corn belt in Minnesota, under a humid continental climate (hot summers and cold winters with snow), the expected increase in rainfall occurring in winter and spring would affect the dissolved NO_3^- export rate, and thus, influence the direct and indirect N_2O fluxes (Griffis et al., 2017). During their 2010-2015 study, Griffis et al. (2017) observed a large interannual variability in N_2O emissions measured using tall tower observations at four heights (32, 56, 100, and 185 m above the ground) within the U.S. Corn Belt. This variability was explained by a high sensitivity from the regional emission factor (EF) to climate. The EF (fraction of applied N fertilizer emitted as N_2O-N) in the warmest year of 2012 was 7.5%, representing nearly twice the EF reported in the past. But the interannual variability in the U.S. Corn Belt was thought to be mostly due to an increase of the indirect emissions (runoff and leaching), which were approximately equivalent to direct emissions in 2012 (direct emissions 192 ± 39 Gg N_2O-N y^{-1}, indirect emissions 220 ± 75 Gg N_2O-N y^{-1}). Because of the current trends and anthropogenic product use, regional N_2O emissions are not only expected to increase in the U.S. Corn Belt and in other intensive agricultural regions, but a strong positive feedback is projected under climatic conditions that are getting warmer and wetter.

Moreover, the elevated CO_2 conditions induced by human activities are expected to differentially alter N_2O fluxes, depending on the N availability of the system (Kuzyakov et al., 2019). Under elevated atmospheric CO_2 concentrations in agricultural ecosystems rich in N, the enzyme activities were increased in the rhizosphere and denitrifiers activity was accelerated. This stimulation of microbial activity was associated with higher soil moisture conditions that enhanced soil OM mineralization and enzymatic activities, notably, NO_2 reductase (Fig. 1). In a long-term Giessen FACE experiment on permanent grassland in Germany, a 20% enrichment of ambient CO_2 concentration led to an increase in incomplete denitrification (increased N_2O:N_2 ratio) from a presumed increased nitrite reductase activity, causing the doubling of the N_2O emissions after 15 years (Moser et al., 2018). A significant increase of 4.6% in soil N_2O fluxes is expected as a result of elevated CO_2 conditions, and raises questions on the capacity of terrestrial ecosystems to mitigate climate change by acting as a sink for C (Liu et al., 2018a). Higher temperatures and wetter conditions are expected to increase N_2O formation, but the outcome of their interaction with one or two other drivers (*e.g.* CO_2, N, or ozone) are more difficult to predict (Butterbach-Bahl et al., 2013).

1.1.2 Methane

1.1.2.1 Importance of methane produced by the agricultural sector

Methane is another important GHG from the agricultural sector, the largest emitter of global anthropogenic CH_4 emissions (Saunois et al., 2020). Methane gas has a GWP 28-fold higher than CO_2 to absorb the heat radiation on a 100-year timescale (Ghosh et al., 2015). Current CH_4 atmospheric concentration reached 1856 ppb in 2018 and contributed approximately to 16% of the greenhouse effect (Saunois et al., 2020). Natural and anthropogenic sources of annual emissions are now estimated at 576 Tg CH_4 yr^{-1} by atmospheric inversion, of which about 60% (range of 55-70%) are from anthropogenic sources. The atmospheric inversion approach, or top-down approach, makes estimations based on atmospheric observations within an inverse-modelling framework in opposition with the bottom-up approach, which make estimations from land surface emissions and atmospheric chemistry, inventories of anthropogenic emissions, and data-driven extrapolations.

Anthropogenic CH_4 emissions originate mainly from agriculture and waste, fossil fuel extraction and use, and biomass and biofuel burning (Karakurt et al., 2012; Saunois et al., 2020). According to Saunois et al. (2020), agriculture and waste were responsible for close to 60% (217 Tg CH_4 annually) of total CH_4 anthropogenic emissions during the period of 2008-2017. Livestock production (enteric fermentation in ruminant animals and manure management) was the largest source of CH_4 in this category, followed by waste handling and rice cultivation (Saunois et al., 2020). Enteric fermentation and manure management represented 33% of total global anthropogenic emissions while rice cultivation represented 8%. However, soils play an important role in the global cycle with CH_4 consumption. Upland soils uptake 38 Tg CH_4 yr^{-1}, which represents 6.8% of the total CH_4 consumption (top-down approach), and thus, act as an important CH_4 sink (Saunois et al., 2020).

Upland soils are CH_4 sinks, but some CH_4 is emitted after livestock manure application (Saunois et al., 2020). Generally, untreated manure stored in anaerobic conditions is applied in the field, but various manure management systems exist (liquid/slurry treated in tanks, pits, etc.; solid in stacks or dry lots, composted, etc.). Once applied, methanogens in manure are exposed to aerobic conditions and CH_4 production is inhibited. However, in aerobic conditions, the manure decomposition would promote N_2O emissions, which would have a greater impact on global warming than CH_4 emissions.

1.1.2.2 Summary of the microbial processes

Methanogenesis and methanotrophy are two processes related to CH_4 emissions in soils, importantly interacting with the CH_4 cycle. Atmospheric CH_4 is in large part due to biogenic sources (70-80%) (Le Mer and Roger, 2001). Methanogenic bacteria live in anoxic environments such as submerged soils with low sulphate and NO_3 concentrations. They anaerobically digest OM in the reaction: $C_6H_{12}O_6 \rightarrow 3\ CO_2 + 3\ CH_4$. The interface of oxic to anoxic conditions present in various environments, including methanogenic soils and upland soils, is where methanotrophs are active and oxidize atmospheric CH_4. Methanotrophy is more efficient in often water-saturated sites, where methanogenic activity is more preserved and CH_4 concentrations are much higher. A positive correlation was found between methanogenesis and methanotrophy and this could be related to the ability of both bacterial groups to maintain their populations when exposed to unfavourable conditions (Le Mer and Roger, 2001). Methanotrophs viability is preserved in anaerobic conditions and their population would be more conserved in anaerobiosis than in aerobiosis when a C source is lacking. However, the net balance of CH_4 production and oxidation in water-saturated sites is typically positive. Factors influencing the CH_4 consumption in unsaturated oxic soils will be further discussed, as this biogeochemical process is predominant in upland soils.

1.1.2.3 Factors influencing microbial processes

Methane is used by methanotrophs as a C and energy source, and methanotroph activity is limited by the availability of O_2 in the environment (Serrano-Silva et al., 2014). Soil moisture content is an important factor in CH_4 oxidation due to the low solubility of O_2 and CH_4 in water-filled pores. Soils become a CH_4 source in situations in which O_2 diffusion is inhibited due to high water content in the soil (Khalil and Baggs, 2005; Wang and Bettany, 1995). However, the presence of aerobic microsites and anaerobic CH_4 oxidation were measured at up to 75% WFPS in a silt loam soil experiment (Khalil and Baggs, 2005). Moreover, CH_4 oxidation rates were sharply decreased under less than 30% WFPS. Using a different measurement unit, other previous studies reported a 20% water holding capacity threshold for inhibition of methanotrophic activity in dry soils (Bender and Conrad, 1995; Jäckel et al., 2001).

Despite the importance of O_2 concentration during the CH_4 oxidation process, O_2 may not a be a limiting factor in upland forests and grasslands (Sabrekov et al., 2016). Instead, the limiting factor for methanotrophic activity in these soils may be CH_4 concentration and diffusion (Serrano-Silva et al., 2014). Methane emission depends on the diffusion of dissolved CH_4 along a concentration gradient formed in deeper soil layers. Methane oxidation is promoted during this slow diffusion process as there is increased contact between the gas and methanotrophic bacteria in the lower aerobic layer of oxic soils. Soil texture and bulk density influence the diffusion of CH_4 and O_2 from the atmosphere to the soil, thus a sandy soil with high pore size will exhibit higher rates of CH_4 oxidation compared to a clayey soil (Serrano-Silva et al., 2014).

In CH_4 oxidation, there are two known types of kinetics depending on CH_4 concentrations in the environment (Le Mer and Roger, 2001; Serrano-Silva et al., 2014). The low affinity type requires CH_4 concentrations higher than 40 ppm. This type is more likely found in sediments and is the most studied since it is easily cultivated in laboratory. The second form of CH_4 oxidation, the high affinity type, can be found at concentrations lower than 40 ppm. These types of methanotrophs would be favoured in soils at CH_4 concentrations lower than 0.1 to 0.4 μmol mol^{-1}. Some soil methanotrophic bacteria have the ability to grow at high- and low- CH_4 concentrations environment as they express two types of the CH_4 monooxygenase enzyme (Szafranek-Nakonieczna et al., 2019). In Szafranek-Nakonieczna et al.'s experiment on incubated agricultural soils in Poland, the *Methylocystis* species were active across a wide range of CH_4 concentrations (0.002–10% v/v), and this could be meaningful in the context of climate change.

Positive correlations were found between the methanotrophic activity and soil pH, redox potential, moisture, and soil organic carbon (SOC; Szafranek-Nakonieczna et al., 2019). In Szafranek-Nakonieczna et al. (2019), the optimal soil pH for methanotrophs was between 5.0-6.5, but a large range of pH (between 4.5-9.0) is suitable for methanotrophic activity (Serrano-Silva et al., 2014; Szafranek-Nakonieczna et al., 2019). Furthermore, several studies revealed that CH_4 oxidation is reduced under elevated CO_2 concentrations, but clear explanations of the underlying mechanisms are still required (Serrano-Silva et al., 2014). It is hypothesized that the size or the activity of the microbial population would be affected by higher CO_2 concentrations, or that greater SOC would cause more competition for O_2 and decrease CH_4 oxidizers activity.

1.1.3 Carbon dioxide

The agricultural sector is estimated to contribute to less than 1% of global CO_2 emissions (Smith, 2012). Soil CO_2 emissions result from the decomposition of plant residues, basal respiration by microbial decomposition of soil OM, root respiration, rhizomicrobial respiration, and the 'priming effect', induced by the addition of organic amendment or root exudation (Thangarajan et al., 2013). The microbial activity is regulated by soil moisture and soil temperature in the top 5 cm, and pH (Zak et al., 1999). In a laboratory experiment with an incubated soil, 60% WFPS was the optimal level to support the maximal microbial activity (Linn and Doran, 1984). Above the 60% WFPS threshold, microbial activity decreased as conditions became more anaerobic, and below that threshold, CO_2 production decreased linearly with WFPS (between 30-70% WFPS). Different management systems may influence the C, N, and water content at the soil surface, in turn affecting microbial processes. Changes in climatic conditions also influence microbial activity, OM decomposition, and C storage in the soil ecosystem. Net CO_2 emissions from organic amendments would be neutral, as annual emissions from litter decomposition is similar to the quantity assimilated by plants (Thangarajan et al., 2013). Large variations in the

CO_2 daily fluxes are often measured in field experiments, consequently, significant differences between treatments are not often reported (Boardman et al., 2018). However, bare soils in the spring when plants have not started to grow yet can be an indicator for microbial activity in the soil. Our study will focus on the interpretation of N_2O emissions and CH_4 sequestration from agricultural upland soils, although CO_2 fluxes were quantified during this two-year experiment (Fig. S 1).

1.2 Effects of agricultural practices on soil properties and greenhouse gas emissions in upland soils

Management practices from organic and conventional cropping systems are known to differently alter the soil microbial processes responsible for GHG emissions. In conventional farming, synthetic fertilizers offer options to reduce GHG emissions (*e.g.* controlled-release fertilizers) and may be applied in combination with animal manure. Mechanisms of GHG formation in conventional cropping systems are well documented, but are still not well understood in organic cropping systems. Organic farms generally have more diverse and longer crop rotations, different crop mixtures, and larger areas covered with perennial crops. Since synthetic pesticides are prohibited in those systems, diverse and more frequent tillage operations are implemented to control weeds, impacting the soil environmental conditions and biota activity differently than in conventional farming. In organic cropping systems, N_2O emissions mainly originate from biological sources. Animal manure, biological fixation of atmospheric N_2, crop residues, and the soil reserve are the main N sources, and their slow N release and episodic nature complicate the synchronization of the N availability with crop needs. Organic amendments also influence nitrification and denitrification processes in the soil that may impact soil GHG emissions. Although management options are broadly similar within organic agriculture, the various organic farms in the province of Québec implement several cropping strategies. This section will review the possible impacts of organic cropping practices on the physical and chemical environment of soils, crop yields, and GHG emissions.

1.2.1 Crop sequence

Crop type selection, rotation length, and rotation complexity influence GHG emitted from a cropping system, in part because the N application rate varies between crops, which influences the inorganic N levels in the soil that foster denitrification. However, EF (N_2O-N emitted per N applied) may not be affected by rotation complexity. In a study conducted by Drury et al. (2008), a monoculture of continuous corn exhibited greater mean N_2O emissions (2.62 kg N ha^{-1}) than soybean (0.84 kg N ha^{-1}) and winter wheat (0.51 kg N ha^{-1}) and this was related to the soil inorganic N concentrations influenced by the N application rates (170, 0, and 83 kg N ha^{-1}, respectively) and to the amount of crop residues (incorporated with a chisel plough [CP] in fall) present in the soil. Although N_2O emissions were highest in corn, similar EF were observed among the different crops receiving the same N fertilizer, as the N rate was also the highest in corn. Similarly, in Machado et al. (2021) a diversified crop rotation (corn-corn-soybean-winter wheat) with a greater total amount of N fertilizer applied emitted 1.2 to 1.5 times more N_2O emissions than a simple crop rotation (corn-corn-soybean-soybean), but equivalent EF were obtained over complete (four-year) simple and diversified crop rotations.

Since crop residues from the previous season supply soluble C to microbes during the decomposition process, both the current and previous crops contribute to the N_2O emissions. According to Drury et al. (2008), N_2O emissions from a monoculture of continuous corn can be as much as 100% more than emissions in corn following a soybean crop, and 60% more than emissions in corn following a winter wheat crop. In Drury et al. (2008), a larger amount of crop residues would have been left on the soil after corn harvest, which provided more C than soybean and winter wheat crop residues. But in the long-term, increased OM levels in diversified crop rotation did not increase N_2O emissions (Machado et al., 2021). Machado et al. (2021) suggested the improved soil quality of diversified crop rotation promoted a greater abundance of the *nosZ* gene copy and increased the potential for complete denitrification to N_2, likely reducing the N_2O emissions from the diversified crop rotation.

A greater rotational complexity with a larger diversity of crops reduces the exposure of soils to N losses from wetting fronts, erosion, and microbial activity. The N use efficiency of a rotation is improved by the inclusion of annual crops with different N requirements, efficiency at scavenging nutrients, and timing of crop uptake (*e.g.* including winter wheat in a corn rotation) (Robertson and Vitousek, 2009). More complex rotations typically include perennials and favour the continuous presence of living plants that increases soil C through the decomposition of dead roots and aboveground residues as well as root exudation (Van Eerd et al., 2021). Increased C inputs help to build OM concentrations and fuel microbial processes, resulting into more pore space and better soil aggregation and, consequently, promoting root development and microbial activity. Longer crop rotations also enhance water infiltration through improved soil structure and aggregation. Nutrient retention is promoted by the different root systems exploring various soil depths, recycling nutrients, and mitigating nutrient losses. Organic cropping systems with a longer and more diversified crop rotation also tend to be more resilient under drought conditions than conventional systems with shorter crop rotation, as their crop yields are less variable in the long-term due to their higher water-holding capacity (Gomiero et al., 2011). Under drought conditions, organic crops have shown more resiliency than conventional cropping systems as they could achieve greater yields. This increased drought resilience may be related to improved mycorrhizal associations and to a higher water-holding capacity of organically managed soils (Lotter, 2003).

Symbiotic N fixation from legumes may bring large amounts of N into the system, but their residues may cause N losses when returned to the soil (Jensen et al., 2012). Nitrous oxide emissions from an alfalfa (*Medicago* *sativa* L.) perennial crop were comparable to those of annual crops (corn-wheat-soybean) in conventional cropping systems in the Midwest United States (Robertson et al., 2000). The rate of alfalfa N_2 fixation was high enough to sustain high productivity and soil mineral N concentrations, and consequently, may have favoured higher rates of denitrification. However, lower N_2O emissions were reported from legume annual crops than from cereal annual crops, although this difference was masked in drier growing seasons (Boardman et al., 2018; Drury et al., 2008). Gregorich et al. (2005) analyzed results from eastern Canadian studies to compare two cropping systems including legumes, either as an annual crop (soybean) or as a perennial crop (alfalfa), and they reported lower N_2O emissions in the former (1.73 kg N ha^{-1}) than the latter (2.31 kg N ha^{-1}). This could be explained by the multiple cuts of alfalfa during the growing season. After alfalfa is cut, the nodules degrade and N is released from the root system (Ta et al., 1986; Vance et al., 1979). The N released from several cuts combined with the N in the alfalfa litter fall (dead plant material) during the growing season may contribute to significant N_2O losses. Soil N_2O emissions in alfalfa (0.67 to 1.45 kg N ha^{-1}) were related to frequent cutting and harvesting of the aboveground plant biomass (Rochette et al., 2004a). Crop residues from a legume crop may contribute to the following season's emissions as large N_2O emissions were measured in the spring following incorporation of alfalfa in the following spring (Wagner-Riddle et al., 1997; Westphal et al., 2018). Similarly, N_2O emissions increased following red clover (*Trifolium pratense* L.) incorporation in Han et al. (2017a) study comparing management and two landscape positions on a well-drained loamy soil. However, seasonal cumulative emissions of the red clover-corn organic rotational phase (3.0 and 3.6 kg N_2O-N ha^{-1} for toeslope and shoulder positions, respectively) were equivalent to those of the bare fallow (BF)-corn conventional rotational phase (1.8 and 4.0 kg N_2O-N ha^{-1} for toeslope and shoulder positions, respectively).

Forage perennial cropping systems typically present lower N_2O emissions than annual cropping systems (Gregorich et al., 2005). Data extracted from field experiments in conventional cropping systems in eastern Canada showed that mean cumulative N_2O emissions of annual cropping systems were 4.5 times greater than perennial cropping systems (respectively 2.82 and 0.62 kg N_2O-N ha^{-1} yr^{-1}, means of log-transformed data). This difference in N_2O emissions between the two systems was thought to be due to lower background emissions found in the perennial systems (estimated from unfertilized crops). More recently, N_2O emissions in annual crops were about 3 times higher than fertilized perennial forage crops (grass/alfalfa mixture) in a 3-year experiment in Ontario, Canada (Abalos et al., 2016). The N_2O-N emitted per N applied was 1.3 to 3.7% higher in annual compared to perennial crops. Abalos et al. (2016) suggest that the difference in soil OM due to the distinct root systems of annual and perennial crops was the major factor explaining the reduction in N_2O emissions. Also, the longer active growth period characteristic of perennials would favour a tighter N cycling and influence soil environmental conditions affecting N_2O formation, such as soil water content, temperature and available mineral N concentrations (Gregorich et al., 2005). Similar results were reported from field experiments in organic cropping systems in Scotland and northern Germany, in comparisons of annual crop rotations with perennial forage crops including legumes in the species mixtures (Ball et al., 2014; Biernat et al., 2020). Lower N_2O emissions were generated in a grass/white clover (*Trifolium repens* L.) mixed crop (0.8 to 1.1 kg N_2O-N ha^{-1} yr^{-1}) than in annual crops like oats (*Avena sativa* L.), barley, wheat, or potato (*Solanum tuberosum* L.) (1.9 to 3.0 kg N_2O-N ha^{-1} yr^{-1}), except during a wet season when emissions from the grass/white clover mix were increased (2.8 kg N_2O-N ha^{-1} yr^{-1}) (Ball et al., 2014). Biernat et al. (2020) observed larger N_2O emissions in organic wheat cropping systems with 25 or 40% legumes in their rotation than in an extensive perennial forage crop (respectively 0.7 and 0.3 kg N_2O-N ha^{-1} yr^{-1}). Perennials are also known to alter soil environmental conditions by accumulating soil C through root biomass (Gregorich et al., 2005). Greater soil C concentrations were reported in deeper soil layer in rotations including perennial crops than in rotations with continuous annual crops after 10 to 35 years, but this would vary among crop types (Angers et al., 1999; Carter et al., 1998; Gregorich et al., 2001). Robertson et al. (2000) showed that the N_2O emissions from a conventional alfalfa perennial crop could be offset by the storage of 161 g CO_2 eq m^{-2} yr^{-1} from the unharvested plant biomass.

1.2.2 Nitrogen source

Organic and conventional cropping systems that minimize the N surplus and control available soil mineral N concentrations in the field by optimizing N use efficiency could potentially reduce N_2O emissions (Van Groenigen et al., 2010). Soil surface N balance (fertilizer N inputs minus crop N outputs) was exponentially related with direct N_2O emissions in Eagle et al. (2020) study. However, despite careful N management, N_2O emissions may still be enhanced under conditions that stimulate denitrification activity.

The effects of organic amendments on N_2O emissions depend on their composition in total N and organic C, environmental conditions (*e.g.,* soil aeration, water saturation), and soil properties (Charles et al., 2017; Rochette et al., 2018; Thangarajan et al., 2013; Zhou et al., 2017). For example, N_2O production in clay soils could be N limited whereas loamy soils may be more limited by a lack of labile C (Chantigny et al., 2010; Pelster et al., 2012). Previous studies reported that organic N sources increased N_2O emissions compared with mineral N sources in loamy soils whereas organic and mineral N sources produced equivalent or lower N_2O emissions in clay soils (Chantigny et al., 2010; Pelster et al., 2012; Zhou et al., 2017). In a clay soil, cumulative N_2O emissions were lower with pre-treated liquid swine manure treatments containing organic N only in the form of NH_4^+ than in a mineral fertilizer treatment containing 50% of the total N as NO_3^- (Chantigny et al., 2010). In contrast, greater cumulative N_2O emissions were measured for liquid swine manure than for mineral fertilizer in a loam soil. Similarly, when comparing induced N_2O emissions from animal manure application with synthetic fertilizer, a meta-analysis by Zhou et al. (2017) found greater manure-induced emissions in sandy loam, loam, and clay loam soils, compared with silty clay soils. Organic C in manure applied in a loam soil may stimulate microbial respiration, which would reduce O_2 concentration and promote N_2O production through denitrification (Thangarajan et al., 2013). Moreover, a soil with a high clay content has a greater cation exchange capacity that may limit nitrification and denitrification through increased NH_4^+ adsorption on clay particles (Jarecki et al., 2008).

Manure application also resulted in greater N_2O emissions than synthetic N fertilizer application in acidic soils, while no such difference was detected in neutral and alkaline soils (Zhou et al., 2017). The higher N_2O emissions could be due to the inhibition of the N_2O reductase activity in lower soil pH conditions, as this enzyme is determinant in the reduction of N_2O to N_2 during the denitrification process. A larger N_2O to N_2 ratio is observed under manure application in acidic soils and N_2O emissions are increased compared to neutral soils. The pH and the redox potential of a soil would also be influenced by manure application, thus complicating the understanding of underlying mechanisms involved across pH levels (Thangarajan et al., 2013).

Fertilizer rates, placement, and incorporation also influence N_2O emissions. The injection or incorporation of liquid manure to maximize soil N availability and crop N uptake and to reduce NH_3 volatilization would increase N_2O production in comparison with broadcast application in annual and perennial crops although this effect may be dependent on interannual weather variability (Abalos et al., 2016; Duncan et al., 2017). Sub-surface banding application that disposes large concentrations of C and N and covers it with soil to reduce NH_3 losses from volatilization may also enhance N_2O production (Chantigny et al., 2010). This agrees with a meta-analysis that reported increased N_2O emissions with sub-surface manure application, but not with surface manure application compared with synthetic fertilizers (Zhou et al., 2017). The contact between N and C compounds and the soil is enhanced with sub-surface manure application. Thus, the microbial activity is promoted, causing O_2 consumption and depletion and creating favourable conditions for denitrification. Furthermore, sub-surface application increase N substrate availability as less N is lost through NH_3 volatilization.

Increasing rates of N fertilizer applied were linearly related with increasing N_2O emissions in different soil textures of eastern Canada (Gregorich et al., 2005), in agreement with the assumptions of IPCC guidelines in GHG inventories (IPCC, 2019). Results from Shcherbak et al. (2014) suggests the relation may actually be non-linear. Although this was confirmed for synthetic fertilizer, it is still only assumed for manure, thus investigation is still needed. Nevertheless, the rate of application of composted manure, a more stabilized organic amendment, did not directly affect N_2O emissions (Boardman et al., 2018; Krauss et al., 2017). In a tillage and cover crop experiment on a silt loam soil, increasing application rate of composted turkey litter incorporated by disking did not increase N_2O emissions in corn (Boardman et al., 2018). The greater N_2O emissions at the highest N rate in manure-based system during the third year of the experiment were related to a build-up of N through the years, acknowledged by the increasing NO_3 concentrations measured over time. Although only weak or no correlations are generally found between N_2O daily fluxes and soil mineral N concentrations, a strong relationship exists between the soil N intensity, which is the integrated sum of NO_3-N and/or NH_4-N concentrations over time. Yao et al. (2020) showed that soil NO_3-N intensity could explain 79% of the variation in annual N_2O emissions, using an exponential model for a wide range of sites, land use (horticultural crops, field crops, grasslands, and forests), and environmental conditions.

The available mineral N concentrations, C:N ratio, water content, and nature of organic amendments further influence N_2O emissions (Charles et al., 2017; Thangarajan et al., 2013). Solid, composted or pre-treated manure characterized by a high C:N ratio would present equivalent or lower N_2O emissions than liquid manure or raw manure (not composted or pretreated) in organic and conventional farming studies (Gregorich et al., 2005; Skinner et al., 2019; Zhou et al., 2017). In Zhou et al. (2017) meta-analysis, N_2O emissions induced by raw manure and pre-treated manure (composted or digested) were 46.9% and only 2.8% higher, respectively, than N_2O emissions induced by synthetic fertilizer application. Moreover, a study on management practices in eastern Canada reported N_2O emissions from solid manure were 35% lower than emissions from liquid manure, whereas N_2O emissions from liquid manure and synthetic fertilizer were similar (Gregorich et al., 2005). Both studies are in accordance with the negative correlation found between N_2O emissions and the C:N ratio of organic amendments (Charles et al., 2017). A low C:N ratio may enhance N_2O emissions through the "priming effect", which triggers the microbial activity in the soil when easily available organic C is added and accelerates soil OM decomposition. Nitrous oxide emissions are thus greater in raw manure than pretreated manure as raw manure provides a higher availability of inorganic N compounds and easily degradable C that fuels nitrifiers and denitrifiers activity (Zhou et al., 2017). Composting manure stabilizes the OM compounds in the animal manure and reduces the priming effect. A high C:N ratio (>15) favours N immobilization and lower inorganic N substrate for nitrifiers and denitrifiers, thus reducing N_2O emissions.

However, an organic amendment with a high C:N ratio may also favour complete denitrification to N_2 and increase SOC that may accumulate over time (Thangarajan et al., 2013). The SOC and microbial biomass were increased after ten years of organic amendments in an organic farming experiment in a clay soil (Krauss et al., 2017). In this experiment, higher cumulative N_2O emissions were obtained from cattle manure compost than from cattle slurry as the N_2O emissions in these grass-clover ley-winter wheat cropping systems were probably more C than N driven in the long-term (species in the grass-clover mixture not specified).

Animal manures also differ in N mineralization patterns depending on their composition (sugar, starch, protein, uric acid, lignin, polyphenol, etc.). Poultry and pig manure tend to have a higher organic C and N degradability than cattle manure (Chadwick et al., 2000). A higher easily degradable C content (*i.e.,* dissolved organic C and volatile fatty acids) in poultry manure would enhance denitrification and would explain the greater effect size on N_2O emissions from poultry manure than from cattle manure in Zhou et al. (2017) meta-analysis. Similarly, in

Pelster et al. (2012), N_2O emissions were greater with poultry manure application than with liquid cattle or swine manure applications in a sandy loam soil, likely limited in C, but not in a silty clay soil, likely limited in N.

Cover crops may fix N biologically with the use of a legume green manure, or optimize N use by capturing the excess N from the preceding crop and preventing N losses with the use of a non-legume. Consequently, cover crops may limit N_2O emissions from direct sources (in the field) or indirect sources (from other locations reached by runoff and leaching). Cover crops help reduce indirect emissions by reducing NO_3 leaching and soil erosion, and by enhancing soil health (Van Eerd et al., 2021). A healthy soil that provides functions such as a good soil structure maintenance, water cycling, nutrient cycling, OM cycling, and temperature regulation will contribute to store C and regulate GHG emissions. After incorporation, the N captured in the cover crop residues is made available for the next crop and reduces the amount of N to be applied in the following growing season. Thus, cover crops influence direct N_2O emissions by preventing N excess in the spring when environmental conditions may favour denitrification.

However, cropping systems using cover crops are also subject to NO_3 leaching during the decomposition process of the cover crop residues. In particular, legume green manures are easily degraded as they are characterized by low C:N ratios. Legumes can contribute to N_2O emissions as a result of NO_3 build up not assimilated by legumes, or when N is released from their root exudates and from the turnover of fine roots, nodules and senesced leaf litter (Jensen et al., 2012; Rochette and Janzen, 2005). A grass/clover mixture of perennial ryegrass (*Lolium perenne* L.), white clover and red clover, and cover crop residues (winter wheat) turnover coincided with high peaks of N_2O emissions in an organic crop rotation (Brozyna et al., 2013). In Brozyna et al. (2013), annual emissions were equivalent in a manure-based and a legume-based cropping system (both systems used cover crops), on a sandy loam soil. In the manure-based cropping system, digested pig manure was applied by injection to an 8-10 cm depth (spring barley and potato) or with a trail hose (winter wheat). Equivalent N_2O emissions in the legume-based system would be due to N-rich crop residues, their supply of easily degradable C, and the accumulation of mineral N around anaerobic microsites due to the microbial respiration from material degradation. In contrast with Brozyna et al. (2013), lower N_2O emissions were obtained for two organic cropping systems including either 25% or 40% legumes within their crop rotation cycle compared to conventional cropping systems with mineral fertilizer, digestates, and pig slurry applications, on a sandy loam soil (Biernat et al., 2020). Furthermore, the 40% legume-rotation produced similar yields to the conventional crop rotation, even with a N input three times lower in the organic cropping system. In this study, the organic crop rotations could have been further optimized by minimizing field N surplus level to reduce N_2O emissions, as suggested by Eagle et al. (2020). The organic legume-based rotations did not include a fall-seeded cover crop, which could have lowered the risk of NO_3 leaching and N_2O emissions.

Peyrard et al. (2016) showed that legume crops and cover crops can have a limited effect on N_2O emissions considering their potential benefits. In their study, the use of cover crops in combination with legume crops allowed a reduction in the N input and contributed negligibly to cumulative N_2O emissions (<0.01 kg N_2O-N ha^{-1}). Cover cropping would be more beneficial in regions with wet springs as they contribute to dry soils in the early season (Robertson and Vitousek, 2009). However, in drier regions, a cover crop may deplete the soil moisture usually stored in the spring, therefore, no-till (NT) management would be a better option to protect against soil erosion and prevent off-season N losses. Strategies to improve the N use efficiency of a cropping system should be based on the site-specific geographical climatic conditions.

1.2.3 Tillage intensity

Soil environmental conditions such as aeration and water content are determined by soil texture and altered by tillage intensity, which, along with climatic conditions, influence N_2O production and diffusion in the soil (Rochette et al., 2018). In severe drought conditions, tillage practices may not influence GHG emissions (Boardman et al., 2018). But when the amount of precipitation received is not a limiting factor in N_2O emissions, tillage intensity may be a determinant factor. Tillage practices that favour better soil aeration and drainage conditions, and that lower soil compaction, contribute to lessen N_2O emissions. Conservation tillage practices such as reduced tillage (RT) had higher bulk density, wet aggregate stability, and soil available water capacity, compared to conventional tillage (CT) in Li et al. (2019) global meta-analysis. Whereas in Nunes et al. (2020) U.S. meta-analysis, no effect was observed on soil structure indicators (wet aggregate stability, bulk density, and soil penetration resistance), when converting from moldboard plowing to chisel plowing. The divergent results reported in meta-analyses on the effects of tillage practices on soil structure suggest that climate, soil texture, and the various combinations of agricultural practices may influence the effect of tillage intensities on soil physical properties.

Soil conservation practices may increase N_2O emissions in fine-textured soils in the temperate climate of the Québec province (Pelster et al., 2021; Rochette et al., 2008a). In Pelster et al. (2021), higher emissions were measured under RT (harrowing at 5 cm depth in the spring) than ploughing (CT; inversion tillage at 20 cm depth in the fall and harrowing at 5 cm in the spring) in a heavy clay soil, but the N_2O emissions of the two tillage intensities were equivalent in a sandy loam soil. Rochette et al. (2008a) obtained similar results comparing NT and CT (inversion tillage at 20 cm depth in the fall) in these two types of soil. In these clay soils, the RT system resulted in higher water content than the CT system, which may have enhanced denitrification, whereas in the coarser textured soils, the two tillage intensities had no distinctive effect on soil water content, likely causing no difference in denitrification.

Higher N_2O emissions under RT (skim plough and CP at 7-10 cm depth in the fall) compared with CT (moldboard plough [MP] at 15-18 cm depth in the fall) were also obtained at an organic experimental site on a calcareous clay soil in Switzerland (Krauss et al., 2017). In accordance with Pelster et al. (2021), Ball et al. (2014) reported no difference in cumulative emissions between RT (disking to 7 cm, timing not mentioned) and CT (MP and disking to 20 cm, timing not mentioned) on a sandy loam soil under conditions of 20-40% higher rainfall than average.

However, no correlation was found between soil texture and the effect of NT/RT and CT practices on N_2O emissions in the meta-analysis by Van Kessel et al. (2013). The study considered field experiments from humid and dry climatic conditions and showed the effects of tillage practices on N_2O emissions may be influenced by the climatic conditions experienced at a site. A seasonal climatic influence was observed in organic grain production systems on a silt loam soil in Missouri, USA, where tillage intensities (RT, disking at pre-planting, depth not mentioned, or NT) combined or not with a cover crop did not affect N_2O emissions during a drier season, whereas emissions increased with increased precipitation (Boardman et al., 2018).

In contrast with Pelster et al. (2021) results, RT (disking to 15 cm depth in spring) showed lower N_2O emissions than CT (ploughing at 30 cm depth in fall and roto-tilling to 15 cm depth in spring) during a humid season, in a clayey loam soil from a humid continental climate (Žurovec et al., 2017). In this experiment, the higher N_2O emissions in the CT treatment was likely due to a different timing of residues incorporation of the previous crop, alfalfa. The authors suggested a greater microbial activity in the spring and a greater response to mineral N fertilization in CT than RT plots, in the context of exceptionally wet conditions. Emissions in RT were not different than those in CT during a drier season, in the second year of the experiment. Thus, N_2O emissions are influenced by tillage practices, soil properties, and climatic conditions, but also, by the timing of residues incorporation.

1.2.4 Non-growing season nitrous oxide emissions

Crop residues after plough-down may significantly contribute to N_2O emissions in cropping systems, therefore highlighting the importance of evaluating their impacts over a full-year period and in long-term experiments. Crop residues or manure incorporation in the fall resulted in higher cumulative emissions (2.41 kg N_2O-N ha^{-1} yr^{-1}) than residues left at the soil surface (1.19 kg N_2O-N ha^{-1} yr^{-1}) on a silt loam in Ontario in winter and spring (Wagner-Riddle and Thurtell, 1998). The effect of residue incorporation on N_2O emissions depends on the soil texture and the freeze-thaw intensity the following spring (Pelster et al., 2013). An extreme freeze-thaw (+1°C to -7°C), characterized by a large fluctuation of temperatures from above freezing point, can cause higher N_2O emissions as colder temperatures may cause a greater breakdown of soil aggregates and the lysis of a larger proportion of microbial cells that release C and nutrients in the soil (Ivarson and Sowden, 1970; Koponen and Martikainen, 2004; Wang and Bettany, 1993). These thaw events occur in periods during which soil water saturation is favoured, causing conditions of restricted O_2 availability that promote denitrification of mineral N. When amended with mature crop residues in the fall, a silty clay soil may emit less N_2O than an unamended soil, as the clay soil is rich in C and the additional C from the residues would induce NO_3 immobilization (Pelster et al., 2013). Some immobilization could also occur in the amended sandy loam, but in this type of soil with a lower C content, microbial activity is more likely to be stimulated by the C added, resulting in higher emissions than in unamended soils under moderate freeze-thaw events (+1°C to -3°C, over 8 cycles of 10 days).

Moreover, perennial forage cropping systems showed lower N_2O emissions during winter and spring thaw events than annual cropping systems (Maggiotto and Wagner-Riddle, 2001; Wagner-Riddle and Thurtell, 1998). The longer active growth period of perennials and their uptake of nutrients would lessen the soil inorganic N concentrations available for denitrification. Soils with no vegetation cover nor crop residues are more subject to OM degradation, thus potentially increasing N_2O and CO_2 emissions. An alteration of the bare soil structure and N mineralization are conditions that enhance N_2O formation. Bare soils are also more exposed to spring thaw events, hydric erosion, and N leaching than agricultural soils with cover crops. According to Wagner-Riddle and Thurtell (1998), spring N_2O emissions are enhanced by field management in the previous fall (*e.g.* legume residues incorporation, manure application, fallowing), whereas the use of over-wintering cover crops, such as alfalfa or grass, is an effective strategy to mitigate N_2O emissions. Depending on the management practices, cumulative N_2O emissions could be up to 4.8 kg N ha^{-1} yr^{-1} higher if freeze-thaw emissions were included (Wagner-Riddle and Thurtell, 1998). Rochette et al. (2008c) estimated the N_2O emitted during the freeze-thaw period corresponded to 40% of the N_2O emitted during the snow-free season in eastern Canada. Furthermore, seasonal freezing could be significant globally as the total N_2O emissions may be underestimated by 17% to 28%, based on recent model estimations (Wagner-Riddle et al., 2017). However, N_2O emissions during freeze-thaw periods are not often measured due to the seasonal climate constraints.

1.2.5 Methane emissions

Although CH_4 concentration is the most influent factor on the CH_4 oxidation capacity in arable soils, land use, soil type, and soil temperature are also significant factors (Szafranek-Nakonieczna et al., 2019). Several studies suggested that net CH_4 consumption would decrease under conditions of high N fertility (Castro et al., 1994; Gregorich et al., 2005; Mosier et al., 1991). Manuring and soil structural degradation from compaction would reduce the CH_4 sink capacity, especially in the poorly drained and fine-textured soils of eastern Canada (Gregorich et al., 2005). Anaerobic conditions, compaction, and poorly drained soils are factors that can increase

CH_4 emissions as OM degradation activity by methanogens is favoured in these conditions (Gregorich et al., 2005).

In upland cultivated soils, large N inputs as well as tillage operations degrading the soil structure through compaction from the repeated passes of tractor wheels affected net CH_4 emissions, especially in fine-textured soil (Ball et al., 1999; Hansen et al., 1993). Hansen et al. (1993) reported that the accumulated CH_4 uptake was reduced by 52% under soil compaction, by 50% under fertilization (average of the mineral fertilizer and cattle slurry treatments), and by 78% under both soil compaction and fertilization, in a well-drained sandy loam. In Ball et al. (1999), tillage operations (ploughing at a 20 cm depth) reduced CH_4 oxidation rates by one-quarter compared to NT possibly by disturbing the methane-oxidizing microbial community and affecting the soil porosity through disruption of the soil structure. Similarly, methanotrophic activity was nearly two-fold higher in soils under NT in comparison with soils under CT (Szafranek-Nakonieczna et al., 2019). In this study, soils under NT exhibited greater pH and SOC, but lower NO_3-N concentrations than soils under CT.

Furthermore, reduced CH_4 emissions and oxidation are reported in fields with ammonium sulfate application (Le Mer and Roger, 2001; Serrano-Silva et al., 2014). Ammonium fertilizers tend to inhibit CH_4 oxidation since the CH_4 monooxygenase enzyme can bind and react with NH_4^+, a molecule similar to CH_4 in size and structure. However, this inhibitory effect would not occur under manure fertilization, which likely maintains larger methane-oxidizing populations than under mineral fertilization (Hütsch et al., 1993). Green manures (finely ground clover residues, species not mentioned) reduced CH_4 uptake by 43% in Nesbit and Breitenbeck (1992) laboratory experiment. This effect from GM could be due to an enhanced activity from microbial antagonisms or availability of other available substrates preferred by CH_4-oxidizing microbes.

Median rates of methanotrophy reported for cultivated soils are -4.1 g CH_4-C ha^{-1} d^{-1} (range 0.0 to -649.5 g CH_4-C ha^{-1} d^{-1}) and grassland soils are -4.9 g CH_4-C ha^{-1} d^{-1} (range -1.4 to -363.8 g CH_4-C ha^{-1} d^{-1}; Le Mer and Roger, 2001). In a Canadian study, Braun et al. (2013) reported fluxes varying from -0.25 to -0.01 g CH_4-C ha^{-1} d^{-1} in the Mixed Grassland Ecoregion of Saskatchewan, either mowed or not in early spring, to simulate moderate grazing conditions. However, well-drained soils can also be a sporadic source of CH_4 after snowmelt in spring. Wang and Bettany (1995) reported fluxes of +59 g CH_4-C ha^{-1} d^{-1} in a cultivated soil and +79 g CH_4-C ha^{-1} d^{-1} in a grassland soil, in Saskatchewan. During various studies on loamy soils in eastern Canada, cumulative growing season emissions varied from -0.70 to +0.11 kg CH_4-C ha^{-1} in fertilized and unfertilized annual grain cropping systems, with an approximative annual mean of -0.29 ± 0.30 kg CH_4-C ha^{-1} (Gregorich et al., 2005).

1.3 Environmental efficiency of agricultural practices

1.3.1 Crop productivity and greenhouse gas emission intensity

1.3.1.1 Yield-scaled emissions

The environmental impact of organic farming is being debated in the context of a global increase in food demand and climate change. Organic farming is often criticized for being less productive than other conventional farming systems. It is also implied in debates on biodiversity conservation, that greater productivity per unit area could prevent land use conversion (Baudron and Giller, 2014). Thus, yield-scaled emissions are important to consider in the assessment of the environmental efficiency and sustainability of a cropping system. Yield-scaled N_2O emissions are generally reported as the ratio of cumulative N_2O emissions of a growing season to a unit of product and are expressed in kg N_2O-N Mg^{-1} of grain or in kg N_2O-N Mg^{-1} dry matter (DM) of aboveground crop biomass. The ratio of cumulative N_2O emissions to the aboveground crop N uptake (N_{up}) is sometimes calculated to report yield-scaled emissions of different crops and cropping practices and are expressed in kg N_2O-N kg^{-1} N_{up} (Guardia et al., 2016).

Less often relayed in the literature is the "cereal unit" allocation method, developed in the 1940's and optimized continuously ever since by German scientists to report agricultural statistics (Brankatschk and Finkbeiner, 2014). A conversion factor is attributed to each crop, determined by their animal feeding value, and normalized using the animal feeding value of 1 kg of barley as a reference. The "grain equivalent" (GE) is obtained by multiplying a crop yield with its conversion factor. The GHG cumulative emissions can then be reported in kg N_2O-N GE^{-1} or kg CO_2 eq GE^{-1} (Biernat et al., 2020). With this method, the yields from different crops are put on a common basis and the comparison can be made between a wide range of crops, or between cropping systems with various crop types in their rotation. However, the conversion factors made available by Brankatschk and Finkbeiner (2014) are based on the German livestock composition and feed consumption. Therefore, calculations are needed to adapt the cereal unit allocation method for different countries.

With the yield-scaled emissions approach, the performance of the overall cropping system is assessed, therefore indirectly considering the effect of soil type, N management strategies, and climatic conditions. Organic systems generally emit less N_2O at the area scale than conventional systems (Benoit et al., 2015; Biernat et al., 2020; Petersen et al., 2006; Skinner et al., 2019; Skinner et al., 2014), but the lower productivity of organic cropping systems resulted in higher yield-scaled N_2O emissions than conventional cropping systems in Skinner et al. (2014) meta-analysis. More recent studies however reported similar yield-scaled emissions when comparing organic and conventional cropping systems (Benoit et al., 2015; Biernat et al., 2020; Skinner et al., 2019). In all three experiments, although yields remained lower in organic systems, the low cumulative N_2O emissions per unit area allowed equivalent yield-scaled emissions (units in g N_2O-N kg^{-1} N harvested, g N_2O-N Mg^{-1} DM or kg

CO_2 eq GE^{-1}) than conventional cropping system emissions. In Benoit et al. (2015), average cumulative N_2O emissions and yield-scaled emissions of an organic cropping system (8 crops) were 32% and 12% lower, respectively, than a conventional cropping system (3 crops) in a loam soil, North of France although the large variations in N_2O measurements did not allow significant differences in this experiment.

A change in the effects of cropping practices on N_2O emissions and yield-scaled emissions could be possible after several years of implementation, when the soil properties and ecosystem have stabilized. A previous meta-analysis found that in the short-term, area-scaled N_2O emissions and yield-scaled N_2O emissions in CT were equivalent to those of conservation tillage (NT or RT including shallow cultivation, shallow ploughing, CP, or zone tillage; Van Kessel et al., 2013). But after 10 years or more of experiment, both the area- and yield-scaled N_2O emissions in conservation tillage were reduced compared to CT emissions, although not significantly for yield-scaled N_2O emissions. However, the reduction of yield-scaled N_2O emissions in the long-term was significant in dry climatic zones. In these dry zones, yield-scaled N_2O emissions were 57% higher in conservation tillage than CT in short-term experiments, but 27% lower than CT after more than 10 years of implementation. Humid zones showed a higher potential to reduce yield-scaled emissions when conservation tillage practices were combined with N-fertilizer placement at a depth of 5 cm or more. However, an increase in yields may not be expected with conservation tillage practices in comparison with CT systems under conventional or organic management, even after a decade of implementation (Cooper et al., 2016; Van Kessel et al., 2013). According to a meta-analysis on tillage intensities in organic cropping systems, yields would be lowered by an average of 7.6% in RT compared to deep (>25 cm) inversion tillage (Cooper et al., 2016). However, shallow inversion tillage (<25 cm) instead of deep inversion could minimize the impact on yields with an insignificant reduction of 5.5%, while significantly increasing SOC levels and weed control. In contrast with Cooper et al. (2016), a two-year study in an organic long-term trial (tillage practices implemented for more than 10 years) in a temperate climate obtained similar yields for RT (skim plough and CP at 7–10 cm depth) and shallow inversion tillage (MP at 15–18 cm depth) (Krauss et al., 2017). However, N_2O emissions were higher in RT than shallow inversion tillage in the site-specific pedoclimatic conditions.

Reporting yield-scaled emissions may bring a more complete overview in the comparison of different cropping practices. In Brozyna et al. (2013) experiment, although similar average cumulative N_2O emissions were measured between treatments, higher winter wheat dry matter yields (+14%) and cash crop N yields (+40%) were obtained with digested pig manure than with grass-clover residues left in the field. Since the same crop was assessed in the two cropping systems, it was clear that the cropping system without manure had the greatest mean annual yield-scaled N_2O emissions (in kg N_2O-N Mg^{-1} DM). However, the environmental efficiency of cropping practices used in different crop sequences is more difficult to determine. In a long-term tillage experiment, Guardia et al. (2016) compared tillage intensities (NT, minimum tillage, and CT) on a grain legume crop (vetch, *Vicia sativa* L. var. not specified) and cereal (barley) crop, with residues from both crops left on the soil surface. Lower N_2O emissions were found for NT in vetch, but were equivalent for all tillage intensities in barley. The yield-scaled emissions (in g N_2O-N kg^{-1} N_{up}) of barley were significantly greater than vetch emissions. The interpretation of the difference in yield-scaled emissions expressed per kg N uptake from a legume and an annual cereal crop is limited though, as it does not provide a clear indication of the agronomic efficiency of the cropping systems. A legume crop such as vetch may be highly productive in terms of biomass and can fix approximately 78% of its N biomass from atmospheric N_2 (Anglade et al., 2015). Consequently, residual N in soils from vetch residue decomposition may significantly impact N losses by leaching and direct and indirect N_2O emissions. Thus, the assessment of several indicators is important when characterizing the environmental efficiency of cropping systems.

1.3.1.2 Emission factor

The inefficient use of N fertilizers is an important cause of direct and indirect N_2O losses, therefore there is a strong need to evaluate the efficiency of a cropping system. The EF can be calculated as the ratio of the difference between cumulative N_2O emissions from fertilized and unfertilized plots, with the total amount of N applied in the fertilized plot, and expressed in % of N applied (IPCC, 2019). The EF equation thus includes a correction to calculate the fraction of the N_2O emissions related to the N fertilizer inputs only, using the N_2O emissions of unfertilized plots as a control. The N_2O emissions from the mineralized OM during the growing season are considered as the background emissions of a cropping system, and are excluded in this equation.

The Intergovernmental Panel on Climate Change (IPCC) provides standard EF to forecast soil-derived N_2O emissions based on the N inputs. The EF approach is used internationally to estimate national scale emissions inventories of agricultural GHGs under the regulation of the United Nations Framework Convention on Climate Change. To estimate N_2O emissions, the IPCC suggests the use of different methods (Tier 1, Tier 2, and Tier 3) that vary in their calculation complexity and accuracy of estimation. A default EF of 1% has been set based on a large dataset, although the dataset was acknowledged to be biased toward mid-latitude and temperate regions (Rochette et al., 2018). The IPCC thus recommends the development of country-specific methodologies, and EF may be estimated for a specific region or management. The EF used for the inventory of N_2O emissions from agricultural soils in Canada were first established by Rochette et al. (2008c) and recently updated, with the addition of corrections factors for cropping systems and N source (Liang et al., 2020; Rochette et al., 2018). Research efforts are still in progress to improve the accuracy of the C and N trends estimations.

Factors impacting EF include the C:N ratio and water content of organic amendments, soil texture, soil drainage, soil C and N levels, and climatic conditions (Charles et al., 2017). Emission factors of organic amendments applied in poorly drained soils can be twice as high as those in well-drained soils, while EF of synthetic fertilizers can be as much as 7-fold higher in these conditions. The effects of soil drainage are related to climatic conditions and soil texture for their regulation of soil moisture and O_2 levels, two important factors influencing N_2O emissions. The EF found in fine-textured soils can be approximately 3 times higher than those found in coarse-textured soils (Charles et al., 2017).

In their meta-analysis on N_2O EF of organic amendments, Charles et al. (2017) showed that the IPCC EF value for organic sources could be improved, and proposed global default EF based on a fertilizer type categorization relating to their risk of N_2O emissions. The global N_2O EF estimate of organic fertilizers was approximately 32% of the EF of synthetic fertilizers (0.57% and 1.76% of N applied, respectively). Organic N sources such as crop residues and compost showed mean estimated EF of 0.19% (median 0.08%) and 0.27% (median 0.17%), respectively, and were classified as "low risk" type of fertilizer. Solid manure was classified as a "medium risk" fertilizer with a mean EF of 0.97% (median 0.24%). Liquid manure was qualified as a "high risk" fertilizer with a mean EF of 0.96% (median 0.56%). The high-risk fertilizers were associated with higher water content (5% DM), higher mineral N content (10% dry weight basis), and a low C:N ratio (<5). The low-risk fertilizers were more stabilized products characterized by a higher dry matter content (average 47%) and a higher C:N ratio (average of 18). In Zhou et al. (2017) meta-analysis on soil N_2O emissions related to animal manure application, the average N_2O EF following manure application in upland soils was estimated at 1.87% ± 0.30% of N applied, and was higher for cool temperate zones (1.95% ± 0.23%) than warm temperate zones (0.79% ± 0.12%). The EF also varied according to the type of animal manure, poultry manure showing the lowest mean EF (1.07% ± 0.39%), pig manure showing the highest mean EF (1.70% ± 0.21%), and cattle manure in between (1.35% ± 0.25%).

Organic fertilizers applied in organic farming systems are less readily available as their nutrients are released slower than those of synthetic fertilizers, and this may influence the EF estimates. In organic cropping systems with their N cycling relying on complex crop rotations, the background emissions may be greater than the emissions related to the N inputs. Moreover, the slow release of N from organic amendments may cause delayed N_2O emissions that would be included in the flux measurements of the next growing season, which may explain the very large variation in EF of organic cropping systems reported in a previous meta-analysis (ranging from 0.3 to 36% of N applied) (Skinner et al., 2014). In contrast, EF of conventional cropping systems ranged from 0.5 to 6.2%. The wide range of organic amendments (crop residues, various manure and compost types) used in the different studies of this meta-analysis may be in cause for the large EF variability. The average EF for organic and conventional arable lands and grasslands were 2.76% and 2.19%, and their medians were 3.3% and 2.4%, respectively. However, the meta-regression indicated that the total N input was determinant for the N_2O emissions of conventional systems, while emissions in organic systems were more influenced by soil properties (soil N % and soil C %, collinear variables with indistinguishable effects) and probably related to greater background emissions (Skinner et al., 2014).

Emission factors may also vary with the crop assessed in a cropping system. In Krauss et al. (2017) experiment on an organic grass-clover ley–winter wheat cropping sequence in a clay soil, EF were not different between tillage intensities (CT and RT) combined with different N sources (cattle slurry or composted cattle manure + cattle slurry), but they differed between the two crops. Among all treatments, grass-clover ley EF ranged from 0.65% to 0.74%, and winter wheat EF were between 1.64% and 1.91%. However, the use of the EF approach is questioned when assessing organic cropping systems including a grass-clover ley phase, as the mineralization following that phase may introduce an additional source of N (Skinner et al., 2019). Moreover, background emissions measured in recently established unfertilized plots to correct EF may be impacted by the land-use history of a site (Jungkunst et al., 2006). Hence, the EF approach may not provide clear explanations for single crop differences in farming systems with complex crop rotations.

1.3.1.3 Global warming potential

Global warming potential is another indicator that considers each GHG for its radiative forcing potential, and is typically expressed in CO_2 equivalent (kg CO_2 eq ha^{-1} yr^{-1}). The GWP unit is useful to report soil GHG emissions from more than one gas in agriculture. When considering net emissions from both N_2O and CH_4, composted manure GWP was lower than stockpiled manure GWP as CH_4 emissions were less important in the former due to the higher degree of aerobic conditions within the composted manure (Pattey et al., 2005). Similarly, more CH_4 emissions were reported in conventional cropping systems applying manure, slurry and synthetic fertilizer than in systems applying composted manure and slurry or exclusively synthetic fertilizers, which allowed a modest uptake in CH_4 (Skinner et al., 2019).

The GWP unit is also useful to summarize the effect of cropping practices on soil GHG emissions and soil C sequestration. Zhou et al. (2017) considered the changes in GWP induced by manure application relative to synthetic N fertilizers for SOC stocks (-1140.7 kg CO_2 eq ha^{-1} yr^{-1}) and for N_2O emissions (+419.2 kg CO_2 eq ha^{-1} yr^{-1}). The changes in GWP showed that the potential for climate change mitigation from manure application is attenuated by N_2O emissions from manure application with a net offset of 37% of the increases in SOC stocks.

Using the GWP indicator, several additional variables may also be accounted (*e.g.* fuel consumption by farm machinery), however, these supplemental variables are less often reported in agronomic studies and are more typical of life cycle analysis. Robertson et al. (2000) reported the GWP of an organic cropping system (with legume cover crops but no manure application) was reduced approximately by half compared to the GWP of a conventional cropping system due to greater soil C sequestration and the CO_2 cost of agronomic inputs (N fertilizer, lime, and fuel). Biernat et al. (2020) only measured N_2O and CH_4 emissions, but their results were consistent with those of Robertson et al. (2000). Biernat et al. (2020) found the GWP of organic cropping systems with 25 or 40% legume in their rotation were 2 to 5 times lower than conventional systems using synthetic fertilizers and manure, depending on the experimental year and proportion of legumes in the rotation.

Green manures have the potential to counterbalance the GHG emitted during their decomposition by sequestering C in the soil (Gregorich et al., 2001; Hutchinson et al., 2007; Meyer-Aurich et al., 2006). Examples of cropping practices favouring soil C storage include legume- and forage-based rotation, cover cropping, improved crop yield, RT and possibly NT (Gregorich et al., 2005). As previously mentioned, RT or the use of a GM may have a positive or a negative impact on N_2O emissions, depending on the conditions at the site. Therefore, it is important to find the right balance in the various mechanisms of GHG production and consumption as an approach for GHG mitigation in a specific pedoclimatic context.

1.3.2 Adoption of alternative practices

As most organic farms are converted from conventional farming systems, the evaluation of organic cropping systems over long-term field studies (>10 years) is necessary to provide large samples with data series, and to realistically determine how the transition to organic agriculture is expected to affect soil GHG emissions from upland soils. The effect of a broad-scale conversion to organic farming is uncertain and simulation studies are useful to estimate the effects at the region scale (Doltra et al., 2019; Hoffman et al., 2018; Lee et al., 2020). Although conversion of conventional farming systems to organic farming systems could lead to the mitigation of 0.34–1.10 Mg CO_2 eq ha^{-1} yr^{-1}, varying effects of RT and cover cropping on N_2O emissions were observed in simulation studies, ranging from a mean decrease of −0.60 kg N_2O-N ha^{-1} yr^{-1} to an increase of 0.29 kg N_2O-N ha^{-1} yr^{-1} (Lee et al., 2020).

Simulations suggest that more complexity and diversification, such as the inclusion of a perennial in the rotation, would be an effective strategy to reduce GHG emissions and energy use in grain crop production systems (Hoffman et al., 2018). Moreover, organic systems with a 2- or 3-year rotation presented the highest N_2O emissions whether at the area or at the yield scale, while N_2O emissions were the lowest in a 6-year rotation including 3 years of a perennial crop (alfalfa). However, in this study, poultry litter production and N_2O emissions from litter in 2-, 3-, and 6-year organic crop rotations would represent between 23 and 29% of the total GHG emissions. This raises the question of the reliance of organic farms on animal manure, especially when imported from conventional farms, as the nutrients in the litter originate from industrial processes (synthetic N fertilizer production, P and K mining). Organic farms could be importing significant amount of N from external sources that would not be accounted for in their environmental footprints, therefore a life cycle analysis could provide a more complete evaluation.

More experiments in organic farming are needed to improve simulation of processes related to enhancement of soil fertility and sustainable cropping systems, such as the DM and N yields of cover crops, legume biological fixation, and grass-clover residues mineralization (Doltra et al., 2019). Furthermore, more investigation in organic farming including economic analyses would help recognize the benefits of implementing new practices that impact yields and environmental footprint. Farmers might be more inclined to adopt alternative practices if financial returns are better documented. The opportunity cost (per Mg CO_2 eq) beyond which the mitigation potential increase would be limited by the profitability of a cropping system could be determined, as in Meyer-Aurich et al. (2006) study on tillage intensities. An improved understanding of the effects of organic agricultural practices on GHG emissions might also be useful for conventional farming systems to mitigate the GHG emissions of the agricultural sector.

Chapter 2 Objectives and hypotheses

2.1 General objective

Evaluate the impacts of different organic cropping systems on GHG emissions, soil mineral N, and crop yields in the province of Québec, Canada.

2.2 Specific objectives

i) Investigate the effect of organic cropping systems with different combination of tillage intensities, N sources, and crop sequence on soil temperature, soil moisture, and soil mineral N concentrations;

ii) Quantify and compare N_2O and CH_4 emissions from upland soils environment as affected by organic cropping systems (crop sequence, tillage intensity, N source);

iii) Determine the effect of organic cropping systems (crop sequence, tillage intensity, N source) on crop yield;

iv) Evaluate the efficiency of organic cropping systems (crop sequence, tillage intensity, N source) by relating the intensity of their N_2O emissions to crop productivity (N_2O per unit of product).

2.3 Hypotheses

i) Cropping systems implementing RT with a chisel plough (CP, non-inversion tillage at a 20-cm depth) combined with a fall-seeded green manure (GM) incorporated in the next spring should reduce soil mineral N availability in the spring compared to a CT practice with a moldboard plough (MP, inversion tillage at a-20 cm depth) combined with a poultry manure application (PM) in the spring.

ii) For a similar crop, organic cropping systems combining CP with only a GM as a N source should reduce N_2O emissions at the area scale relative to MP-PMGM and CP-PMGM systems due to the positive relation between PM and N_2O emissions, on a sandy loam soil.

iii) For a similar crop, organic cropping systems MP-PMGM and CP-PMGM should obtain greater yields than CP-GM cropping systems due to the favourable effect of PM on yields.

iv) For a similar crop, CP-GM systems should obtain similar or greater yield-scaled N_2O emissions than MP-PMGM and CP-PMGM systems.

v) The upland soils evaluated in this study should be a small sink for CH_4.

Chapter 3 Combining reduced tillage and green manures minimized N_2O emissions in organic cropping systems in cool humid climate

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Highlights:

- Soil N_2O and CH_4 emissions were compared for different organic cropping systems.
- Combined reduced tillage and green manuring minimized area-scaled N_2O emissions.
- Green manure and poultry manure generated similar yield-scaled N_2O emissions.
- Legume-based perennial forage crop and grain crops generated similar N_2O emissions.
- Dry conditions during spring resulted in lower N_2O emissions despite high NO_3 content.

3.1 Résumé

Cette étude a comparé les émissions de gaz à effet de serre et le rendement de systèmes en grandes cultures biologiques au cours d'une expérimentation réalisée au champ en 2019 et 2020, sur un loam sableux, au Québec. Différents systèmes culturaux (combinaisons de séquences culturales, sources fertilisantes et intensités de travail du sol) et deux témoins (prairie permanente et jachère sol nu) ont été évalués pour leurs effets sur les émissions saisonnières de protoxyde d'azote (N_2O) et de méthane (CH_4), mesurées périodiquement à l'aide de chambres statiques. Les émissions de CH_4 ont été équivalentes entre les différents systèmes. Le système chisel-engrais vert a minimisé les émissions de N_2O à l'échelle de la superficie sans augmenter les émissions de N_2O à l'échelle du rendement, en comparaison avec des systèmes avec labour- ou chisel-fumier. L'évaluation à long terme de ces systèmes culturaux récemment établis est nécessaire pour analyser l'évolution des effets associés.

3.2 Abstract

Implementing soil conservation practices is necessary to enhance the sustainability of organic cropping systems, but some of these practices may increase soil greenhouse gas (GHG) emissions. This study examined the effects of various organic cropping systems on soil GHG emissions and crop yields, in Québec, Canada. Organic cropping systems combining different: (i) crop sequences (barley [*Hordeum vulgare* L.]-grain corn [*Zea mays* L.], soybean [*Glycine max* (L.) Merr.]-spring wheat [*Triticum aestivum* L.], and grain corn-soybean), (ii) sources of nitrogen (N) (poultry manure [PM] and/or a fall-seeded green manure [GM] or no applied N), and (iii) primary tillage intensities (moldboard plough [MP] or chisel plough [CP]) were compared to a perennial forage (PF) and a bare fallow (BF) control. During the 2019 and 2020 snow-free seasons, nitrous oxide (N_2O) and methane (CH_4) emissions, soil water content, soil temperature, and mineral N concentrations were monitored periodically on a sandy loam soil. The lowest cumulative N_2O emissions were found in CP-GM (0.52 ± 0.11 kg N ha^{-1} in 2019 and 0.47 ± 0.06 kg N ha^{-1} in 2020), whereas the highest N_2O emissions were found in MP-PM in 2019 (3.55 ± 0.72 kg N ha^{-1}) and BF in 2020 (1.44 ± 0.20 kg N ha^{-1}). For the barley-grain corn sequence, the CP-GM treatment generated N_2O emissions that were 40% to 70% lower and yields that were 33% to 51% lower than the MP-PMGM and CP-PMGM systems, which showed equivalent N_2O emissions and yields. Yield-scaled N_2O emissions were equivalent in all cropping systems. Peak N_2O daily fluxes in the PF occurred shortly after cutting in 2020. During both years, CH_4 emissions varied from -0.65 to +0.18 kg C ha^{-1} with no detectable differences among cropping systems. The CP-GM cropping system minimized area-scaled N_2O emissions without increasing yield-scaled emissions. However, this was a two-year study on a site that was recently converted

from conventional agriculture, so a long-term assessment is still necessary to determine whether the benefits associated with these cropping systems change over time.

3.3 Introduction

In European countries and North America, organic farming is increasingly adopted (Statistics Canada, 2017; USDA, 2020; Willer et al., 2019). Organic farming aims to generate an income for farmers from food production while preserving soil fertility, biodiversity, environment and human health, and thus, strict standards have been established (Canada General Standards Board [CGSB], 2020; Codex Alimentarius, 1999). Greenhouse gas (GHG) emissions from organic cropping systems are often estimated using calculation schemes (*e.g*. emission factors) based on studies in conventional systems (Muller et al., 2017), thus, more empirical studies in organic cropping systems are needed.

Direct and indirect nitrous oxide (N_2O) emissions from agriculture represent 52% of the total anthropogenic N_2O emissions and are continuously increasing due to synthetic N fertilizer application (Tian et al., 2020). Nitrous oxide is a GHG with a global warming potential (GWP) 265 times that of carbon dioxide (CO_2) on a 100-year timescale and is a potential by-product of nitrification and an intermediate in the denitrification process (IPCC, 2014). These processes are influenced by N and C substrate availability, microbial communities, soil aeration and gas diffusivity, which are in turn controlled by other soil properties (*e.g.* drainage, texture, pH) and climatic conditions (Davidson et al., 2000; Rochette et al., 2018). Crop sequence, nutrient management, and tillage intensity are known to alter soil N and C availability, soil aeration and water saturation, and thus, affect N_2O formation in the soil and its transfer to the atmosphere. Dynamics of N_2O formation are still not well understood in organic cropping systems combining different cropping practices than conventional cropping systems.

In organic cropping systems, soil N_2O emissions originate from organic N sources only (*i.e.*, animal manure, biological fixation of atmospheric N_2, soil N reserve, and crop residues). Organic N sources stimulate denitrifier activity in the soil, that may, under certain conditions, increase the N_2O emissions. Manure-induced N_2O emissions are higher than inorganic fertilizer-induced N_2O emissions on a sandy loam soil, as denitrification in these soils are thought to be limited by labile C (Han et al., 2017b; Pelster et al., 2012; Zhou et al., 2017). Soil N_2O emissions often depend on the N source that is provided, whose effect on N_2O emissions vary with soil texture and drainage conditions, among other factors (Chantigny et al., 2010; Pelster et al., 2012; Rochette et al., 2008b).

Organic crop rotations often include N_2-fixing legume cover crops as green manure (GM) as a source of N for subsequent crops. By reducing the amount of N required for the following crop, GM can prevent N_2O emissions from direct sources, such as manure application (Baggs et al., 2000). But GM may also be direct and indirect sources of N_2O (Basche et al., 2014). In particular, GM characterized by low C:N ratios are easily degraded and can cause N losses through N_2O emissions and NO_3 or soil organic N leaching during their decomposition process.

Diversified and long crop rotations are designed to retain and recycle nutrients in organic cropping systems, hence, can affect N_2O emissions. Nitrogen application rate for the different crops in rotation is variable and influence the area-scaled N_2O emissions that increase with the N rate (Rochette et al., 2018; Shcherbak et al., 2014). For example, lower N_2O emissions were reported for corn (*Zea mays* L.) when it was combined with soybean (*Glycine max* [L.] Merr.) and winter wheat (*Triticum aestivum* L.) in rotation compared to a corn-corn sequence due to higher inorganic N concentrations in the soil from high N application rate in corn (Drury et al., 2008). Crop residues may also supply soluble C to microbes during decomposition that can contribute to N_2O emissions, depending on the crop type. However, emission factors (the amount N_2O-N emitted per unit N applied) may not be affected by crop rotation complexity (Machado et al., 2021). In conventional cropping systems, the emission factor of an organic N source varies with growing season precipitation, potential evapotranspiration, and soil texture (Rochette et al., 2018). In complex organic crop rotations though, the emission factor for the N applied to individual crops can hardly be calculated as N is typically reallocated within the crop rotation (Skinner et al., 2019). However, emission factors of complete organic crop rotations may be compared (Brozyna et al., 2013).

Temporary perennial forage (PF) are generally more present in organic than conventional crop rotations (Barbieri et al., 2017) and thus influence the N_2O losses of cropping systems differently. Soil N_2O emissions in fertilized perennial crops, including legumes or not, can be two to four times lower than in fertilized annual crops; for both conventional and organic cropping systems (Abalos et al., 2016; Ball et al., 2014; Biernat et al., 2020; Gregorich et al., 2005). However, N_2O emissions from perennial forage crops may drastically increase after a plough down following or during a wet season (Macdonald et al., 2011; Abalos et al., 2016; Ball et al., 2014; Westphal et al., 2018).

Canadian organic standards promote soil fertility and biological activity by encouraging minimized tillage (CGSB, 2020), but the effect of soil conservation practices on N_2O emissions would depend on site-specific conditions and still needs investigation. In a temperate, humid climate, conservation tillage practices (no-till, non-inversion tillage), induced higher N_2O emissions compared to conventional tillage practices (inversion tillage) in fine-textured soils, whereas the N_2O emissions of the two tillage intensities were equivalent in loamy soils (Pelster et al., 2021; Rochette et al., 2008a). In contrast, a meta-analysis found no correlation between soil texture and the effect of conservation tillage and conventional tillage practices on N_2O emissions, as climate was an important driving factor interacting with tillage intensities (Van Kessel et al., 2013).

Methane (CH_4) is another GHG associated with agriculture that has a GWP 28 times higher than that of CO_2 to absorb heat radiation on a 100-year timescale (IPCC, 2014). The agricultural sector is the largest emitter of anthropogenic methane emissions, mainly from enteric fermentation, manure management and rice production (Saunois et al., 2020). However, upland soils represent a significant sink in the global CH_4 cycle, contributing to 6.8% of the total CH_4 consumption (Saunois et al., 2020).

To improve our understanding of N_2O and CH_4 fluxes in organic field crop production in a cool temperate region, we evaluated the first two years of an organic trial in the province of Québec, Canada. The study examined the effect of organic cropping systems on N_2O emissions and some of their driving environmental factors (soil mineral N, moisture, and temperature at the soil surface), as well as CH_4 emissions, and crop yields. We hypothesized that, on this sandy loam soil, (i) conventional and reduced tillage intensities would generate similar N_2O emissions, (ii) organic cropping systems with animal manure would result in greater N_2O emissions and yields than cropping systems with GM only, and (iii) GM-based systems would produce equivalent or greater yield-scaled N_2O emissions than cropping systems with animal manure.

3.4 Materials and methods

3.4.1 Research site

This experiment was conducted for two growing seasons in 2019 and 2020 at the Institut national d'agriculture biologique, located in Victoriaville, QC, Canada (46°02'N, 71°58'W, altitude 220 m). The region has a continental humid climate, with a mean annual precipitation of 896.3 mm, and a mean annual temperature of 5.3°C at the Arthabaska meteorological station (MELCC, 2021b). The soil is predominantly a Saint-Samuel sandy loam series (77.0% sand, 13.3% clay, 4.5% organic matter) and is classified as a Humic Gleysol in the Canadian classification system (Soil Classification Working Group, 1998) and as a Typic Humaquept soil in the US classification system (USDA, 1999). In 2020, the mean bulk density was 1.41 g cm^{-3}, pH 6.8 (soil:water 1:1), and CEC 12.5 (meq 100 g^{-1}; Table S 3). The site had been under conventional cash crop production (a corn-soybean rotation) for about 10 years, until a 3-year organic transition was initiated in 2016. In 2017, leveling, drainage, and liming (4 Mg ha^{-1} fine calcitic lime) were performed on the site.

3.4.2 Experimental design

The experimental units were arranged in a randomized complete block design with 4 replications (Fig. S 2-3). The treatments were cropping systems combining different crop types, tillage intensities, and organic fertilization strategies, a perennial forage (PF), and a continuous bare fallow (BF) treatment, as described in Table 1. Crop rotations implemented in 2019 were typical of organic field crop production in the region: corn, soybean, and a cereal (barley, *Hordeum vulgare* L., or spring wheat). The primary tillage intensities were a moldboard plough (MP, *i.e.*, conventional tillage), or a chisel plough (CP, *i.e.*, reduced tillage). The organic fertilization strategies were nutrient inputs from different sources, either poultry manure (PM), poultry manure and green manure (PMGM), green manure only (GM), or no manure (NM). In GM cropping systems, field pea (*Pisum sativum* L.) was fall-seeded after barley harvest. Grain corn crop was also interseeded with red clover (*Trifolium pratense* L.). Since the intercrops represented a negligible N input (< 2 kg N ha^{-1}), it was not accounted for as a N source in those cropping systems. Seven cropping systems were assessed for their GHG emissions and ancillary measurements in 2019, and eight in 2020, for a total of 28 and 32 experimental units each year, respectively. Each experimental unit was 6 m $\times$ 20 m in size.

3.4.3 Field management

A triticale (× *Triticosecale* Wittm. var. Pronghorn) and pea (var. not stated) GM was grown in 2018, the year prior to this study, in all cropping systems. This GM was incorporated in spring 2019 with primary tillage in MP cropping systems, and with shallow cultivation in the CP, PF, and BF systems. Primary tillage was conducted with a reversible MP at a 20-cm depth in the MP systems, and with a CP at a depth of 20 cm in all other systems; the PF system was not tilled after this initial chisel ploughing. After harvest, grain corn residues were either incorporated into the soil (MP) or left on the soil (CP). Cereal stubbles were mowed, shredded, and left on the ground surface in the fall. Shallow tillage at 10-cm depth or less was performed with a rototiller and a disk tiller for fertilizer incorporation, in all cropping systems, depending on the crops. The BF system was maintained bare using a rototiller (10-cm depth) 4 or 5 times over the growing season. Field management details are presented in Table S 1.

In 2019, fresh poultry manure (77.2% DM, and 28.5 g total-N kg^{-1} fresh basis) was applied evenly by hand at rates of 7.0 Mg ha^{-1}, 2.6 Mg ha^{-1}, and 2.2 Mg ha^{-1} in corn, barley, and perennial forage, respectively. In 2020, poultry manure (81.8% DM and 20.4 g total-N kg^{-1} fresh basis) application rates were 6.3 Mg ha^{-1} in spring wheat and ranged between 5.2 to 6.2 Mg ha^{-1} in grain corn to account for the N supplied by previous GM, which differed among cropping systems. Potassium sulfate (0-0-50) was distributed by hand at 0.7 kg K_2SO_4 ha^{-1} in barley, and 3.4 kg K_2SO_4 ha^{-1} in soybean, corn, and PF in 2019. In 2020, between 2.2 and 2.9 kg K_2SO_4 ha^{-1} was supplied to corn, spring wheat, soybean, and PF in 2019. Fertilizer application rates were determined using the regional recommendations for each crop. The model from Centre de développement d'agrobiologie (CDA, 1997) was used to estimate the amount of N returned to soil with GM residues. Biomass N content of GM in 2019 was considered a N source for crops grown in 2020, thus, the N input estimation includes the N from PM only in 2019, and from PM and/or GM in 2020 (Table 1). Nutrient concentration and rates applied with organic amendments are detailed in Table S 2.

Seeding rates for barley (cv. Polaris) and spring wheat (cv. Major) were 200 kg ha^{-1} and 160 kg ha^{-1}, respectively. Soybean (cv. Marula) was sown at 450,000 seed ha^{-1} in 2019, and 500,000 seed ha^{-1} in 2020, and grain corn (hybrid P8034) at 84,500 seed ha^{-1} in both years. The PF was established in the spring of 2019 with a commercial seed mixture containing: 8 kg ha^{-1} alfalfa (*Medicago sativa* L. var. not stated), 4 kg ha^{-1} red clover (var. not stated, two cuts phenotype), 2 kg ha^{-1} smooth bromegrass (*Bromus inermis* Leyss. var. Carlton), 4 kg ha^{-1} tall fescue (*Festuca arundinacea* Schreb. var. Yukon), and 2 kg ha^{-1} timothy (*Phleum pratense* L. var. not stated). Weeds were controlled mechanically using different implements (flextine harrow, rotary hoe, and finger weeder) depending on the crop stage and timing in the season (Table S1). Red clover was interseeded in all cropping systems with grain corn both years, at 10 kg ha^{-1}, at the V7-V8 corn growth stage. Field pea (var. not stated) was seeded as GM at 205 kg ha^{-1} on 28 August 2019 in GM systems.

3.4.4 Data collection and analysis

3.4.4.1 Environmental data

Rainfall was recorded for each 0.2 mm accumulated with a tipping bucket rain gauge and air temperature was recorded hourly, both from a meteorological station located at the field site (HOBO MicroRX2106, Onset, Bourne, MA, USA). Missing data were supplemented with data obtained from an Environment Canada climate station located 29.2 km from the experimental site. Concurrent with gas samplings, soil temperature (Model 11040, DeltaTrak Inc., Pleasanton, CA, USA) at 5-cm depth, and volumetric soil water content (VSWC; FieldScout TDR 150, Spectrum Technologies Inc., Aurora, IL, USA) to a 0-12 cm depth were measured. Both measurements were sampled between the rows, in the immediate vicinity of the frames used for GHG measurement (detailed below).

Volumetric soil water content was used to calculate water-filled pore space (WFPS) with the equation:

$$\mathrm{WFPS} = \frac{\mathrm{VSWC}}{1 - \frac{\mathrm{BD}}{\mathrm{PD}}}$$

where BD is the bulk density (1.59 g cm^{-3} and 1.41 g cm^{-3}, for PF and all other cropping systems, respectively, averages calculated from field measurements using the cylinder method, [Hao et al., 2008]) and PD is the mineral particle density (2.65 g cm^{-3}).

3.4.4.2 Soil sampling and analysis for mineral nitrogen

Composite soil samples (five subsamples per experimental unit) were collected at 0-20 cm depth once every week until 4 to 6 weeks after manure application, then every second week. Within 24 hours of soil collection, soil mineral N (NO_3 and NH_4) was extracted from 5 g subsamples with 25 mL of 1 *M* KCl. Soil slurries were mixed in a reciprocal shaker for sixty minutes, then centrifuged for 10 minutes at 3000 rpm before being filtered with a pre-washed (1 *M* KCl) Whatman #42 filter paper. The collected filtrates were frozen at -20°C until analysis. The extracts were analyzed with a colorimeter (Model QuickChem FIA+ 8500, LACHAT Instruments, Loveland, CO, USA) equipped with a Reagent Pump RP-100 series and AutoSampler ASX-500 series. A 20 g subsample was oven-dried at 105°C for 24 h to determine the gravimetric soil moisture. Soil inorganic N intensity (g N d kg^{-1}) was determined by computing the integrated sum of soil NH_4 and NO_3 concentrations over the complete length of the experiment (Burton et al., 2008) to assess its capacity to predict seasonal N_2O emissions.

3.4.4.3 Greenhouse gas sampling and analysis

Soil GHG fluxes were measured from 26 April to 31 October 2019, and from 29 April to 12 November 2020 using non-flow-through, non-steady-state chambers (Rochette and Bertrand, 2008). Briefly, one clear acrylic frame (0.55 m $\times$ 0.55 m $\times$ 0.14 m height) was installed to 0.10 m depth between rows in corn and soybean plots, and in PF and BF plots; for barley and wheat plots, two narrower frames (0.15 m $\times$ 0.75 m $\times$ 0.14 m height) were installed per plot, in the interrow, to adapt to the smaller interrow spacing. These frames were only removed during field operations and re-installed immediately after the operation. The frames were installed more than 1 m from the edge of the plots. Frame heights above the soil was measured after frame installation, and re-installation following field operations, to calculate the headspace volume. Chamber deployment consisted of placing an insulated acrylic, vented chamber (0.14 m height) on the frames. Each chamber was equipped with a closed-cell foam band to ensure an air-tight seal with the frame during deployment. Air samples from the headspace were collected at 0, 14, 28 and 42 minutes after deployment by inserting a needle attached to a 20 mL polypropylene syringe through a rubber septum. The 20 mL air sample was then transferred to a pre-evacuated 12-mL glass vial (Exetainer, Labco, High Wycombe, UK) with double septa (butyl rubber and silicon).

Gas sampling frequency was once per week until final harvest, except during the four-week period following spring manure application during which gas sampling frequency was twice per week, resulting in a total of 28 sampling dates in 2019 and 31 sampling dates in 2020. Gas sampling was always done between 8:00 am and 11:00 am to reduce temporal variability and to ensure that measured emissions were representative of the mean daily flux rate (Alves et al., 2012). In 2019, samples were stored for no longer than 10 days before analysis. In

2020, however, gas samples were stored in the Exetainer vials for up to 18 weeks before analysis due to restricted access to the laboratory because of Covid-19 pandemic. The samples were analyzed on a gas chromatograph (Model 3800, Varian Inc., Walnut Creek, CA, USA), equipped with an electron capture detector (N_2O) and a flame ionization detector (CH_4).

The N_2O and CH_4 fluxes were estimated from chamber concentrations versus time data, using the extended Hutchinson and Mosier flux calculation scheme with the HMR package in the R statistical program (R Core Team, 2012), as described by Venterea et al. (2020). For the plots with two narrow chambers (barley and wheat plots), the arithmetic mean of the daily fluxes determined for each chamber was computed. The non-linear model was preferred when a good fit was obtained using HMR statistical criteria, otherwise, the linear model was employed.

Cumulative area-based GHG emissions were calculated by interpolating the flux measured between the different dates and integrating the area under the curve. The amount of N_2O-N (in kg) emitted per Mg of grain yield was obtained to determine the yield-scaled N_2O emission.

3.4.4.4 Plant sampling and analysis

Crop yields were measured by harvesting central rows to avoid the edge effects. Corn yield was obtained by harvesting two rows each side of the two middle rows of an experimental unit, whereas cereals were harvested in 14 central rows, and soybean in the 4 rows. All crops were harvested with a trial-plot combine (Model Classic, Wintersteiger, Ried im Innkreis, Austria) over the complete length of the experimental unit (20 m). Grain corn, soybean, spring wheat, and barley grain N concentration was measured using an infrared analyzer (Infratec 1241, FOSS, Hilleroed, Denmark). Grain N is reported as the product of N concentration in grain by grain dry matter (DM) yield. Yield-scaled N_2O emissions in g N_2O-N kg^{-1} grain N DM is the ratio of area-scaled N_2O emissions and grain N DM.

The PF plots were established in 2019 and two cuts were collected in 2020. The first forage cut was harvested for hay, whereas the second cut was left on the soil surface as amendment. Forage aboveground biomass was sampled at mowing height (0.10 m above ground) on the day of the first and second cut, using two and four 0.25 m^2 quadrats, respectively. On the first cut, the forage aboveground biomass was dried, weighed, and ground to 2 mm before analysis (species mixture was not hand-sorted). On the second cut, the forage biomass was hand-sorted by species, dried, weighed, then ground to 2 mm, and pooled before analysis. Both forage cuts were analyzed for C, N and S concentrations by dry combustion (Model TruMac CNS-1000, LECO Corporation, St. Joseph, MI, USA). On 29 October 2019, GM and intercrops aboveground biomasses were sampled to the ground level, using three 0.25 m^2 quadrats. Plant aboveground biomass was then dried, weighed, ground to 2 mm , and analyzed with a spectrometer (Optima 3000 DV ICP-OES, Perkin Elmer, Waltham, MA, USA) for N content.

3.4.5 Statistical analyses

The statistical analyses were conducted using R version 3.6.1 GUI 1.70 (R Core Team, 2012) using the lme4 and lmerTest packages (Bates et al., 2015; Kuznetsova et al., 2017). The normality of data distribution was ascertained with the Shapiro-Wilk test and data were log-transformed when needed. For each year of the experiment, an analysis of variance (ANOVA) was conducted for the cumulative N_2O and CH_4 emissions, soil N intensities, and GWP with organic cropping systems as a fixed factor, and block as a random factor. If a significant difference was found at a P-value < 0.05, a Tukey's HSD test was performed to determine significance of differences between treatments. Simple contrasts (MP-PMGM vs CP-PMGM vs CP-GM) were used to compare crop DM yields, grain N, and yield-scaled N_2O emissions of the different cropping systems that had the same crop in an experimental year. Spring wheat DM yields, grain N, and yield-scaled N_2O emissions were not assessed in 2020 due to crop failure. Multiple linear regression analyses were performed to evaluate relationships between cumulative N_2O emissions and the environmental variables (time-weighted soil WFPS and time-weighted soil temperature, soil NO_3 and NH_4 intensities).

3.5 Results

3.5.1 Environmental conditions and greenhouse gas daily fluxes

3.5.1.1 Weather conditions

Average air temperature from 1 April to 30 November of growing seasons 2019 (10.5°C) and 2020 (11.2°C) were below the long-term 30-year normal (11.8°C) (Fig. 3). Total cumulative precipitation over the entire growing season (1 April through 30 November) in 2019 (810 mm) was similar to long-term normal (825 mm), while total growing season precipitation in 2020 (716 mm) was lower than the long-term normal mainly due to a drier than normal spring (Fig. 3). Distribution of monthly precipitation among the two years was generally consistent with long-term normal except in April and July 2019, when 200% and 50% of normal precipitation were received, respectively, and for May and June 2020 with only 31% and 18% of long-term normal precipitation.

3.5.1.2 Soil conditions

Soil NO_3 and NH_4 concentrations were the highest in cropping systems with PM applied in 2019 (MP-PM, MP-PMGM, CP-PMGM, PF-PM) and 2020 (MP-PMGM, CP-PMGM, CP-PM; Fig. 4 a-d). For the two years, NO_3 concentrations were higher than NH_4 concentrations in cropping systems with PM and/or GM. In all cropping systems with PM, NH_4 concentrations increased after PM application but remained below 8 mg NH_4-N kg^{-1} soil, except in MP-PM with corn in 2019. This latter cropping system received the highest N fertilization rate (200 kg N ha^{-1}) in 2019 and was the only one characterized by a pronounced peak of 27 mg NH_4-N kg^{-1} after PM application, followed shortly by a peak at 21 mg NO_3-N kg^{-1} soil (Fig. 4 a, c). In 2019, soil NO_3 concentrations peaked at 10 mg NO_3-N kg^{-1} in PF-PM, which received the lowest amount of N as PM (64 kg N ha^{-1}). Peaks of NO_3 concentration ranged from 25 to 30 mg NO_3-N kg^{-1} soil in crops with PM applied in 2020, and generally stayed above 10 mg NO_3-N kg^{-1} soil until mid-July (Fig. 4 b). In the CP-GM cropping system (*i.e.,* with pea incorporated in the spring of 2020), NO_3 concentrations peaked at 18 mg NO_3-N kg^{-1} soil in mid-June 2020. In 2020, small increases of 5 and 8 mg NO_3-N kg^{-1} were observed in PF-PM (no PM applied that year) following the first cut (residues harvested), and second cut (residues left in the field), respectively. Nitrate concentrations remained low (between 0 and 12 mg NO_3-N kg^{-1} soil) in both growing season in CP-NM and BF cropping systems. Concentrations in BF rose shortly after the first disk tiller pass for weed control and reached 26 mg NO_3-N kg^{-1} soil in 2019 and 19 mg NO_3-N kg^{-1} soil in 2020.

During the two-year study, soil WFPS varied over time, but with a similar pattern among treatments (Fig. 4 e-f). In 2019, WFPS ranged from 17% to 75% and was generally above 40% in all cropping systems in June after the PM application. In 2020, WFPS ranged between 5% and 71%, and was below 37% in May and June, following PM application, in all cropping systems. In 2020, PF-PM showed the lowest WFPS until the first forage cut in June, and the highest WFPS after the second forage cut in August. The MP-PM was the only cropping system where WFPS stayed between 50 and 75% in June 2019. Soil temperature at a 5 cm-depth ranged from 4°C to 33°C in both growing seasons (Fig. 4 g-h) and were similar among the various cropping systems, except PF where soil temperature was 5°C to 10°C lower than other cropping systems before the first forage cut in June 2020. Soil temperature in BF plots was generally about 5°C higher than in the other cropping systems in July 2020.

3.5.1.3 Soil nitrous oxide and methane fluxes

Daily N_2O fluxes were generally higher in 2019 than in 2020, and N_2O peaks occurred mostly when NO_3 concentrations were above 5 mg NO_3-N kg^{-1} soil and WFPS higher than 50% (Fig. 5). In 2019, the highest N_2O flux (194 g N_2O-N ha^{-1} d^{-1}) observed in corn MP-PM after PM application was concurrent with high NH_4 concentrations and WFPS, and was followed by a second peak (128 g N_2O-N ha^{-1} d^{-1}) a few weeks later, concurrent with high NO_3 concentrations. Smaller peaks (between 26.9 and 42.1 g N_2O-N ha^{-1} d^{-1}) were detected in cropping systems with PM in 2020. Apart from the PF, N_2O fluxes were the lowest in crops with no PM applied and remained below 17.8 g N_2O-N ha^{-1} d^{-1} in 2019, and below 14.4 g N_2O-N ha^{-1} d^{-1} in 2020. In 2020, four peaks up to 33.5 g N_2O-N ha^{-1} d^{-1} were observed in the PF following increases in WFPS after the forage cuts. Daily N_2O fluxes peaked at 70.0 and 23.3 g N_2O-N ha^{-1} d^{-1} in BF, in August 2019 and 2020, respectively. Over the two years, weak positive CH_4 daily fluxes were measured in all crops that did not receive PM application. Methane peaks were higher in 2019 (22.5 g CH_4-C ha^{-1} d^{-1}) than in 2020 (10.9 g CH_4-C ha^{-1} d^{-1}) (Fig. 4 i-j). In 2019, small pulses of CH_4 uptake were observed concurrently with soil temperature increases in cropping systems receiving less than 100 kg N ha^{-1} (PF-PM, MP-PMGM, and CP-PMGM). The greatest CH_4 uptake flux (-35.3 g CH_4-C ha^{-1} d^{-1}) was observed in MP-PMGM in 2020.

3.5.2 Cumulative GHG emissions, soil nitrogen intensities, and global warming potential (GWP)

The highest cumulative N_2O emissions in 2019 was more than two times greater than the highest N_2O emissions in 2020 (Table 2). In 2019, N_2O emissions were 3 to 7 times greater in corn MP-PM than barley CP-GM, soybean CP-PM, and PF-PM. Among barley crops in 2019, N_2O emissions in CP-GM were 70% lower than MP-PMGM emissions. In 2020, the highest N_2O emissions were measured in BF and PF-PM. The lowest emissions that year were in grain corn CP-GM, which were 2 to 3 times lower than spring wheat CP-PM, PF-PM and BF emissions. Among corn crops in 2020, N_2O emissions in CP-GM were almost half those in MP-PMGM and CP-PMGM, although this difference was not significant. No difference was detected among cropping systems for cumulative CH_4 emissions in each year (Table 2). Methane uptake was observed in all cropping systems except in soybean CP-PM in 2019, whereas cumulative CH_4 emissions for the BF were close to zero over the two-year study. Among the different organic cropping systems, GWP of corn MP-PM was about 4 times greater than that of soybean CP-PM, barley CP-GM, and PF-PM in 2019 (Table 2). In 2020, GWP of PF and BF were equivalent to cropping systems with PM or PMGM, and 2 to 3 times greater than GWP of cropping systems with no N input.

Higher soil inorganic N intensities (sum of NO_3-N and NH_4-N intensities) were observed in cropping systems with PM applied at rates greater than 100 kg N ha^{-1}, *i.e.*, in corn and spring wheat crops, compared to cropping systems with N fertilizer rates lower than 100 kg N ha^{-1} or without N fertilizer (Table 2). In grain corn with PM or PMGM, soil N intensities were greater than all cropping systems with barley and PF-PM in 2019, and were equivalent to those of spring wheat CP-PM, and BF in 2020. In 2020, crops with no PM applied showed the lowest soil N intensities (Table 2). Soil N intensities of cropping systems with PM in 2020 were comparable to that of BF. Simple linear regression analyses showed a positive relation between the log of cumulative N_2O emissions and soil inorganic N intensities in both year ($P < 0.05$), with a pseudo R^2 of 0.2807 (Fig. 6).

3.5.3 Crop yields and yield-scaled nitrous oxide emissions

In both years, simple contrasts showed higher crop yields in MP-PMGM and CP-PMGM than in CP-GM (Table 3). When considering a specific crop species, yield-scaled N_2O emissions were similar among cropping systems for both years. Soybean generated greater yield-scaled emissions (per kg grain DM) than grain corn in 2020 (Table 2). When expressed in g N_2O-N per kg N in grains, soybean generated the lowest yield-scaled N_2O

emissions in 2020. Simple contrasts comparing yield-scaled N_2O emissions per kg N in grains showed no significant difference among cropping systems with the same crop sequence (data not shown).

3.6 Discussion

3.6.1 Nitrous oxide emissions and driving environmental factors

Organic cropping systems with reduced tillage (CP) and GM as the N source showed the lowest N_2O cumulative emissions of all cropping systems during this two-year study. In the site-specific conditions, cropping systems with GM may have induced a slower N release from OM mineralization and better synchronization with plant uptake than manure-based (PM) systems. In manure-based systems, the timing of PM application at planting have resulted in the rapid release of available N concentrations while crops root system was not fully developed and could not efficiently take up this available N. Denitrification in manure-based systems may have been further enhanced by labile C applied with PM. In loamy soils with low to moderate C concentrations, denitrification can be limited by a lack of labile C (Chantigny et al., 2007; Chantigny et al., 2010; Pelster et al., 2012). Labile C applied with manure can also increase soil respiration rates, thus creating anaerobic conditions conducive to N_2O production via denitrification (Thangarajan et al., 2013).

In this study, cumulative N_2O emissions were in the range of growing season N_2O emissions typically found in loamy soils of the region under similar climatic conditions (Table 4). In agreement with past studies (Boardman et al., 2018; Drury et al., 2008; Gregorich et al., 2005), cumulative N_2O emissions in our study were lower in the unfertilized annual legume crop than the fertilized grain crops, but were equivalent among fertilized and unfertilized crops in 2020, as particularly dry conditions occurred at time of N application (May-June).

Peak N_2O fluxes generally occurred shortly after PM application and were related to increased NH_4 and NO_3 concentrations. In 2019, the highest N_2O daily fluxes were measured in the weeks following PM application, when soil inorganic N were high and WFPS levels were between 40-80%, creating ideal conditions for nitrification and denitrification (Davidson et al., 2000). By contrast, the low WFPS values measured in May and June 2020 would explain that N_2O emissions were limited despite high NO_3 concentrations, because aerobic conditions inhibits soil denitrification (Davidson et al., 2000).

Both nitrification and denitrification processes were likely involved in N_2O emissions in manure-based systems. Nitrification contribution to GM-based system N_2O emissions may have been more limited by NH_4 availability, as indicated by the lower available NH_4 concentrations in the spring. The first N_2O peak observed in manure-based systems occurred simultaneously with a peak in NH_4 concentrations at WFPS lower than 60%, suggesting that this N_2O peak was related to nitrification. A substantial portion of PM N is present as uric acid that is rapidly hydrolyzed to NH_4 in soil, as reflected by the rapid rise in NH_4 concentrations following PM applications. The oxidation of NH_4 could result in N_2O production and loss as a byproduct during nitrification (Davidson et al., 2000). This was followed by increased NO_3 concentrations, which could have been denitrified when WFPS increased above 60%, resulting in the second N_2O peak. In 2019, the second peak in N_2O fluxes observed in MP-PM was likely promoted by increased labile N and C availability after a rewetting event that increased the soil water content from 60 to 76% WFPS. Rewetting of dried soils are known to stimulate nitrification, a process known as the "Birch effect" (Birch, 1958, 1960), which can also increase N_2O emissions (Liu et al., 2018b; Mumford et al., 2019). Similar patterns of two consecutive N_2O peaks related to a rise in NH_4 and then to NO_3 concentrations have been previously reported (Brozyna et al., 2013; Pelster et al., 2021).

The positive linear relation found between cumulative N_2O emissions and soil N intensity in this study is consistent with the linear relation generally found in various situations (Yao et al., 2020), but also specifically in manure-based cropping systems of the region (Pelster et al., 2021). In our organic cropping systems, the increase in cumulative N_2O emissions for each g NH_4-N kg^{-1} was about 6 times greater than for each g of NO_3-N kg^{-1}. The important influence of NH_4 concentration to N_2O emissions and the fluctuating aerobic-anaerobic conditions found during the study suggest that nitrifier denitrification or coupled nitrification-denitrification may have been another pathway for N_2O emissions (Butterbach-Bahl et al., 2013; Wrage-Mönnig et al., 2018). In a laboratory experiment, manure addition to soil was associated with increased soil NO_2 concentrations, the substrate for nitrifier denitrification, which tend to increase N_2O production with decreasing water content (Wrage et al., 2004). Results from our study are consistent with results from Hung et al. (2021) and Pan et al. (2018) who reported an important contribution of nitrifier denitrification on N_2O emissions in manure-based cropping systems.

3.6.2 Yield-scaled nitrous oxide emissions, yields, and nitrogen uptake

When comparing across the same crop sequence (barley-grain corn) and N source (PMGM), N_2O emissions, yields, and yield-scaled N_2O emissions were similar across tillage intensities (MP and CP) for both years, consistent with previous findings in loamy soils (Ball et al., 2014; Pelster et al., 2021; Rochette et al., 2008a; Van Kessel et al., 2013). For the same crop sequence (barley-grain corn) and tillage intensity (CP), N_2O emissions and crop yields were lower with GM than PMGM as the N source, however the yield-scaled N_2O emissions were similar between these N sources for both years. This indicates that the 40% to 70% reduction in N_2O emissions was enough to compensate for the 35% to 50% lower yields in the GM-based system, resulting in a similar impact on product-related N_2O emissions compared to the more intensive PMGM systems. Our results are consistent with those of Biernat et al. (2020), who compared manure-based conventional crop rotations with GM-based organic crop rotations. However, our results are not consistent with those of Brozyna et al. (2013), who compared manure-based to GM-based organic crop rotations. In Brozyna et al. (2013), the supply of labile C present in cover crop residues and grass-clover in GM systems may explain that similar N_2O emissions were found between the two systems. Moreover, yield-scaled N_2O emissions were the highest in their GM systems, since the highest yields were obtained with the animal manure treatment.

The annual grain crop yields during our two-year study were generally consistent with regional average yields from conventional and organic cropping systems, except for CP-GM cropping system and for spring wheat (FADQ, 2021a, 2021b). All cropping systems with a spring wheat crop (CP-PM and CP-NM) experienced crop failure due to weeds pressure. Yields in the CP-GM system were lower than average regional yields by 25% and 65% for grain corn and barley, respectively. In our recently implemented organic cropping systems, reduced tillage and conventional tillage generated equivalent yields, which is consistent with previous meta-analyses, although reduced tillage was found to result in lower yields under a humid climate after ten years (Cooper et al., 2016; Van Kessel et al., 2013). After ten years, in loamy soils, similar area-scaled and yield-scaled N_2O emissions in reduced and conventional tillage, would still be obtained (Van Kessel et al., 2013). Over time, lower yields in reduced tillage cropping systems with a high crop residues retention and a low soil disturbance may be compensated by the positive impact on soil physical properties such as increases in aggregate size and water-stable aggregates in comparison to conventional tillage systems (Li et al., 2019). Reduced tillage and GM-based organic cropping systems may be less productive, but may promote soil conservation, prevent point source pollution, and improve soil structure (Chivenge et al., 2007; Diacono and Montemurro, 2010).

Annual legume crops lessened the N_2O emitted per unit N produced, which improved the N balance of cropping systems over a complete rotation. In 2019, the annual legume grain crop (soybean) was up to 12 times more efficient than annual non-legume grain crops (corn, cereals) when considering the yield-scaled N_2O emissions per exported N unit. Similar to our study in 2020, previous studies also reported yield-scaled emissions per grain N in a legume crop were about half of the yield-scaled emissions from an annual non-legume grain crop (Guardia et al., 2016; Malhi and Lemke, 2008). In scenarios exploring the feasibility of expanding organically cropped areas, an increase in legume use would be necessary to partly compensate for the necessary change in food diet and N source (Barbieri et al., 2021; Billen et al., 2021). Hence, increasing the use of legumes in crop rotations could be one of the strategies to improve agroecosystems sustainability while minimizing the N_2O-N emitted per N harvested.

3.6.3 Perennial forage nitrous oxide emissions

Cumulative N_2O emissions from fertilized and unfertilized perennial crops are generally lower than from fertilized annual grain crops (Abalos et al., 2016; Gregorich et al., 2005; Jensen et al., 2012). In our study, PF N_2O emissions were lower than manure-fertilized grain corn in 2019, but were equivalent to those of all fertilized annual grain cropping systems in 2020, in accordance with Robertson et al. (2000) who compared a legume perennial crop (alfalfa) and fertilized annual grain crops. Perennial forage N_2O emissions in our organic cropping systems were equivalent to soybean N_2O emissions in both years, in accordance to Rochette et al. (2004a) for loamy soils, where N_2O emissions from alfalfa were equal to or greater than for soybean. Higher N_2O emissions in a pure stand of legume perennial crop (alfalfa) than in a legume annual crop (soybean) were also reported in Gregorich et al. (2005).

Keeping soil covered with a PF crop during our two-year study lowered soil N intensities compared to BF, however, it did not lower N_2O cumulative emissions (Table 2). Increased available NO_3 concentrations observed in BF in July and August in both years were likely related to enhanced N mineralization following shallow cultivation for weed control. The available NO_3 concentrations (between 10 and 26 mg NO_3-N kg^{-1} soil) in BF did not favor N_2O peaks larger than 70.0 g N_2O-N ha^{-1} d^{-1} likely because WFPS levels on those dates were below 55%. In 2020, the peaks of N_2O daily fluxes in PF were likely due to the combination of increased NO_3 availability and WFPS greater than 50% (Chantigny et al., 2013; Linn and Doran, 1984; Rochette et al., 2004b). The high NO_3 concentrations found in PF plots in spring 2019 were likely derived from the decomposition of applied PM and GM (*i.e.*, the triticale and pea incorporated in spring 2019) while the PF root system was still developing and unable to efficiently take up all the available N and water in the top . However, once established, perennial forage roots can efficiently absorb nutrients and water for most of the growing season, thus abating soil mineral N concentrations and therefore the N_2O emissions (Abalos et al., 2016).

Cumulative N_2O emissions in PF with PM applied in the first (establishment) year were similar to those following the two cuts in the second year. The increases in N_2O daily fluxes following the two forage cuts in 2020 were similar to a past study (Rochette et al., 2004a). As previously reported, cutting a legume forage can cause the nodules to degrade, releasing N from the root system (Ta et al., 1986; Vance et al., 1979). As the PF contained approximately 87% legumes, the cuts likely caused N to be released from the roots as well as from the crop residues left on soil surface at the second cut, causing the intensified N_2O emissions. Furthermore, increased N availability from nodules senescence would be enhanced in dry periods, as experienced in 2020 in our study, as drought stress may induce a greater N accumulation in nodules (Thilakarathna et al., 2016). Some specific species combinations in perennial grasslands can mitigate N_2O emissions due to a complementarity in root morphology and an improved total biomass productivity (Abalos et al., 2014). In the present study, the PF cropping system could be optimized by increasing the proportion of *Festuca arundinacea* (Schreb.) and *Phleum pratense* (L.) in the species mixture, as these species helped mitigate N_2O emissions in Abalos et al. (2014).

3.6.4 Methane emissions and global warming potential

The GWP was the lowest in GM-based or unfertilized cropping systems due to their lower N_2O emissions, which is consistent with a previous study in organic and conventional cropping systems (Biernat et al., 2020). Incorporating legume residues into soils reduced CH_4 uptake by 43% compared to a control without residues in Nesbit and Breitenbeck's (1992) laboratory experiment, but this effect was not observed in GM-based systems of our field experiment. Similar net CH_4 emissions were observed in fertilized and unfertilized cropping systems during this two-year study, in contrast with findings from Castro et al. (1994); Hansen et al. (1993); Mosier et al. (1991), reporting a decrease in net CH_4 consumption under N fertilization. Ammonium fertilizers tend to inhibit CH_4 oxidation since the CH_4 monooxygenase can bind and react with NH_4, a molecule similar to CH_4 in size and structure (Le Mer and Roger, 2001; Serrano-Silva et al., 2014). However, this inhibitory effect would not occur under manure fertilization, which likely maintains larger methane-oxidizing populations than under mineral fertilization (Hütsch et al., 1993).

3.7 Conclusion

Our study highlighted the potential of organic cropping systems combining reduced tillage and a GM to minimize N_2O emissions by preventing the accumulation of available soil N in the spring, in the specific context of our study. Overall, the dry conditions in spring 2020 induced low cumulative N_2O emissions in fertilized cropping systems despite high available NO_3 concentrations. Conventional and reduced tillage intensities resulted in equivalent area-based and yield-scaled N_2O emissions. On this loamy soil, the GM-based cropping system generated lower crop yields than manure-based cropping systems, but similar yield-scaled N_2O emissions were measured. Once established, the legume-based PF required no additional N inputs, however, a release of inorganic N after cutting caused pulses of N_2O that resulted in cumulative N_2O emissions equivalent to annual cereal crops. All cropping systems were small sinks for CH_4 and CH_4 cumulative emissions were similar between cropping systems.

Thus, on a sandy loam soil, in a cool temperate climate, farmers could include GM in their crop rotation to reduce their manure application rates in the spring, and their related N_2O emissions. But farmers should also be careful when determining the proportion of leguminous species in their PF species mixture, however, determining the adequate proportion was beyond the scope of this study. Soil N_2O emissions measurement over a full-year period and in long-term experiments is needed to better understand the longer term fertility effects of organic cropping systems, particularly in GM-based systems, and how that may affect N_2O emissions. An economic analysis would clarify whether the adoption of GM-based cropping systems could be profitable for farmers despite the lower yields.

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3.9 References

Abalos, D., Brown, S.E., Vanderzaag, A.C., Gordon, R.J., Dunfield, K.E., Wagner-Riddle, C., 2016. Micrometeorological measurements over 3 years reveal differences in N_2O emissions between annual and perennial crops. Global Change Biol. 22, 1244-1255. https://doi.org/10.1111/gcb.13137

Abalos, D., Deyn, de G.B., Kuyper, T.W., Van Groenigen, J.W., 2014. Plant species identity surpasses species richness as a key driver of N_2O emissions from grassland. Global Change Biol. 20, 265-275. https://doi.org/10.1111/gcb.12350

Alves, B.J.R., Smith, K.A., Flores, R.A., Cardoso, A.S., Oliveira, W.R.D., Jantalia, C.P., Urquiaga, S., Boddey, R.M., 2012. Selection of the most suitable sampling time for static chambers for the estimation of daily mean N_2O flux from soils. Soil Biol. Biochem. 46, 129-135. https://doi.org/10.1016/j.soilbio.2011.11.022

Baggs, E., Watson, C., Rees, R., 2000. The fate of nitrogen from incorporated cover crop and green manure residues. Nutr. Cycl. Agroecosyst. 56, 153-163. https://doi.org/10.1023/A:1009825606341

Ball, B.C., Griffiths, B.S., Topp, C.F.E., Wheatley, R., Walker, R.L., Rees, R.M., Watson, C.A., Gordon, H., Hallett, P.D., McKenzie, B.M., Nevison, I.M., 2014. Seasonal nitrous oxide emissions from field soils under reduced tillage, compost application or organic farming. Agric. Ecosyst. Environ. 189, 171-180. https://doi.org/10.1016/j.agee.2014.03.038

Barbieri, P., Pellerin, S., Nesme, T., 2017. Comparing crop rotations between organic and conventional farming. Sci. Rep. 7, 13761. https://doi.org/10.1038/s41598-017-14271-6

Barbieri, P., Pellerin, S., Seufert, V., Smith, L., Ramankutty, N., Nesme, T., 2021. Global option space for organic agriculture is delimited by nitrogen availability. Nat. Food 2, 363-372. https://doi.org/10.1038/s43016-021-00276-y

Basche, A.D., Miguez, F.E., Kaspar, T.C., Castellano, M.J., 2014. Do cover crops increase or decrease nitrous oxide emissions? A meta-analysis. J. Soil Water Conserv. 69, 471-482. https://doi.org/10.2489/jswc.69.6.471

Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 67. https://doi.org/10.18637/jss.v067.i01

Baudron, F., Giller, K.E., 2014. Agriculture and nature: Trouble and strife? Biol. Conserv. 170, 232-245. https://doi.org/10.1016/j.biocon.2013.12.009

Biernat, L., Taube, F., Loges, R., Kluß, C., Reinsch, T., 2020. Nitrous oxide emissions and methane uptake from organic and conventionally managed arable crop rotations on farms in northwest Germany. Sustainability 12. https://doi.org/10.3390/su12083240

Billen, G., Aguilera, E., Einarsson, R., Garnier, J., Gingrich, S., Grizzetti, B., Lassaletta, L., Le Noë, J., Sanz-Cobena, A., 2021. Reshaping the European agro-food system and closing its nitrogen cycle: The potential of combining dietary change, agroecology, and circularity. One Earth 4, 839-850. https://doi.org/10.1016/j.oneear.2021.05.008

Birch, H.F., 1958. The effect of soil drying on humus decomposition and nitrogen availability. Plant Soil 10, 9-31.

Birch, H.F., 1960. Nitrification in soils after different periods of dryness. Plant Soil 12, 81-96.

Boardman, D.L., Easterby, S., Clark, K.M., Kitchen, N.R., Staples, J.S., Reinbott, T.M., Kremer, R.J., 2018. Do tillage, cover crops, and compost management within organic grain cropping affect greenhouse gas emissions? Agron. J. 110, 1893-1904. https://doi.org/10.2134/agronj2018.01.0023

Brozyna, M.A., Petersen, S.O., Chirinda, N., Olesen, J.E., 2013. Effects of grass-clover management and cover crops on nitrogen cycling and nitrous oxide emissions in a stockless organic crop rotation. Agric. Ecosyst. Environ. 181, 115-126. https://doi.org/10.1016/j.agee.2013.09.013

Burton, D.L., Gillam, K.M., Zebarth, B.J., MacLeod, J.A., 2008. Effect of split application of fertilizer nitrogen on N_2O emissions from potatoes. Can. J. Soil Sci. 88, 229-239. https://doi.org/10.4141/CJSS06007

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? Phil. Trans. R. Soc. B 368, 20130122. https://doi.org/10.1098/rstb.2013.0122

Castro, M.S., Peterjohn, W.T., Melillo, J.M., Steudler, P.A., Gholz, H.L., Lewis, D., 1994. Effects of nitrogen fertilization on the fluxes of N_2O, CH_4, and CO_2 from soils in a Florida slash pine plantation. Can. J. For. Res. 24, 9-13. https://doi.org/10.1139/x94-002

CDA, 1997. Engrais verts et cultures intercalaires. Centre de développement d'agrobiologie, Sainte-Élisabeth-de-Warwick.

CGSB, 2020. Organic production systems: General principles and management standards. Canada General Standards Board: Standards Council of Canada, Ottawa, ON, CA.

Chantigny, M.H., Angers, D.A., Rochette, P., Bélanger, G., Massé, D., Côté, D., 2007. Gaseous nitrogen emissions and forage nitrogen uptake on soils fertilized with raw and treated swine manure. J. Environ. Qual. 36, 1864-1872. https://doi.org/10.2134/jeq2007.0083

Chantigny, M.H., Pelster, D.E., Perron, M.H., Rochette, P., Angers, D.A., Parent, L., Massé, D., Ziadi, N., 2013. Nitrous oxide emissions from clayey soils amended with paper sludges and biosolids of separated pig slurry. J. Environ. Qual. 42, 30-39. https://doi.org/10.2134/jeq2012.0196

Chantigny, M.H., Rochette, P., Angers, D.A., Bittman, S., Buckley, K., Massé, D., Bélanger, G., Eriksen-Hamel, N., Gasser, M.O., 2010. Soil nitrous oxide emissions following band-incorporation of fertilizer nitrogen and swine manure. J. Environ. Qual. 39, 1545-1553. https://doi.org/10.2134/jeq2009.0482

Chivenge, P., Murwira, H., Giller, K., Mapfumo, P., Six, J., 2007. Long-term impact of reduced tillage and residue management on soil carbon stabilization: Implications for conservation agriculture on contrasting soils. Soil Tillage Res. 94, 328-337. https://doi.org/10.1016/j.still.2006.08.006

Codex Alimentarius, 1999. Guidelines for the production, processing, labelling and marketing of organically produced foods. Adopted 1999. Revisions 2001, 2003, 2004 and 2007. Amendments 2008, 2009, 2010, 2012 and 2013. Codex Alimentarius Commission, Rome, Italy.

Cooper, J., Baranski, M., Stewart, G., Nobel-de Lange, M., Bàrberi, P., Fließbach, A., Peigné, J., Berner, A., Brock, C., Casagrande, M., Crowley, O., David, C., Vliegher, A.D., Döring, F.T., Dupont, A.I., Entz, M., Grosse, M., Haase, T., Halde, C., Hammerl, V., Huiting, H., Leithold, G., Messmer, M., Schloter, M., Sukkel, W., van der Heijden, G.A.M., Willekens, K., Wittwer, R., Mäder, P., 2016. Shallow non-inversion tillage in organic farming maintains crop yields and increases soil C stocks: a meta-analysis. Agron. Sustain. Dev. 36, 22. https://doi.org/10.1007/s13593-016-0354-1

Davidson, E.A., Keller, M., Erickson, H.E., Verchot, L.V., Veldkamp, E., 2000. Testing a conceptual model of soil emissions of nitrous and nitric oxides. Bioscience 50, 667-680. https://doi.org/10.1641/0006-3568(2000)050[0667:TACMOS]2.0.CO;2

Diacono, M., Montemurro, F., 2010. Long-term effects of organic amendments on soil fertility. A review. Agron. Sustain. Dev. 30, 401-422. https://doi.org/10.1051/agro/2009040

Drury, C., Yang, X., Reynolds, W., McLaughlin, N., 2008. Nitrous oxide and carbon dioxide emissions from monoculture and rotational cropping of corn, soybean and winter wheat. Can. J. Soil Sci. 88, 163-174. https://doi.org/10.4141/CJSS06015

FADQ, 2021a. Rendements de référence 2020 en assurance récolte. Assurance récolte - Rendements de référence. Financière agricole du Québec. https://www.fadq.qc.ca/ (accessed 10 March 2022).

FADQ, 2021b. Rendements réels de l'année 2019 en assurance récolte. Assurance récolte - Rendements réels. Financière agricole du Québec. https://www.fadq.qc.ca/ (accessed 10 March 2022).

Firestone, M., Smith, M., Firestone, R., Tiedje, J., 1979. The influence of nitrate, nitrite, and oxygen on the composition of the gaseous products of denitrification in soil. Soil Sci. Soc. Am. J. 43, 1140-1144.

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 Tillage Res. 83, 53-72. https://doi.org/10.1016/j.still.2005.02.009

Guardia, G., Tellez-Rio, A., García-Marco, S., Martin-Lammerding, D., Tenorio, J.L., Ibáñez, M.Á., Vallejo, A., 2016. Effect of tillage and crop (cereal versus legume) on greenhouse gas emissions and Global Warming Potential in a non-irrigated Mediterranean field. Agric. Ecosyst. Environ. 221, 187-197. https://doi.org/10.1016/j.agee.2016.01.047

Han, Z., Walter, M.T., Drinkwater, L.E., 2017. N_2O emissions from grain cropping systems: a meta-analysis of the impacts of fertilizer-based and ecologically-based nutrient management strategies. Nutr. Cycl. Agroecosyst. 107, 335-355. https://doi.org/10.1007/s10705-017-9836-z

Hansen, S., Mæhlum, J.E., Bakken, L.R., 1993. N_2O and CH_4 fluxes in soil influenced by fertilization and tractor traffic. Soil Biol. Biochem. 25, 621-630. https://doi.org/10.1016/0038-0717(93)90202-M

Hao, X., Bal, B.C., Culley, J.L.B., Carter, M.R., Parkin, G.W., 2008. Soil density and porosity. In: Carter, M.R., Gregorich, E.G. (Eds.), Soil Sampling and Methods of Analysis. CRC Press, Boca Raton, FL, pp. 743-760.

Hung, C.-Y., Ejack, L., Whalen, J.K., 2021. Fall-applied manure with cover crop did not increase nitrous oxide emissions during spring freeze-thaw periods. Appl. Soil Ecol. 158, 103786. https://doi.org/10.1016/j.apsoil.2020.103786

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 Biol. Biochem. 25, 1307-1315. https://doi.org/10.1016/0038-0717(93)90045-D

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

Jensen, E.S., Peoples, M.B., Boddey, R.M., Gresshoff, P.M., Hauggaard-Nielsen, H., Alves, B.J., Morrison, M.J., 2012. Legumes for mitigation of climate change and the provision of feedstock for biofuels and biorefineries. A review. Agron. Sustain. Dev. 32, 329-364. https://doi.org/10.1007/s13593-011-0056-7

Kuznetsova, A., Brockhoff, P.B., Christensen, R.H., 2017. lmerTest package: tests in linear mixed effects models. J. Stat. Softw. 82, 1-26. https://doi.org/10.18637/jss.v082.i13

Le Mer, J., Roger, P., 2001. Production, oxidation, emission and consumption of methane by soils: A review. Eur. J. Soil Biol. 37, 25-50. https://doi.org/10.1016/S1164-5563(01)01067-6

Lessard, R., Rochette, P., Gregorich, E.G., Pattey, E., Desjardins, R.L., 1996. Nitrous oxide fluxes from manure-amended soil under maize. J. Environ. Qual. 25, 1371-1377. https://doi.org/10.2134/jeq1996.00472425002500060029x

Li, Y., Li, Z., Cui, S., Jagadamma, S., Zhang, Q., 2019. Residue retention and minimum tillage improve physical environment of the soil in croplands: A global meta-analysis. Soil Tillage Res. 194, 104292. https://doi.org/10.1016/j.still.2019.06.009

Linn, D.M., Doran, J.W., 1984. Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. Soil Sci. Soc. Am. J. 48, 1267-1272. https://doi.org/10.2136/sssaj1984.03615995004800060013x

Liu, S., Schloter, M., Brüggemann, N., 2018. Accumulation of NO_2^- during periods of drying stimulates soil N_2O emissions during subsequent rewetting. Eur. J. Soil Sci. 69, 936-946. https://doi.org/10.1111/ejss.12683

MacDonald, J.D., Rochette, P., Chantigny, M.H., Angers, D.A., Royer, I., Gasser, M.-O., 2011. Ploughing a poorly drained grassland reduced N_2O emissions compared to chemical fallow. Soil Tillage Res. 111, 123-132. https://doi.org/10.1016/j.still.2010.09.005

Machado, P.V.F., Farrell, R.E., Deen, W., Voroney, R.P., Congreves, K.A., Wagner-Riddle, C., 2021. Contribution of crop residue, soil, and fertilizer nitrogen to nitrous oxide emissions varies with long-term crop rotation and tillage. Sci. Total Environ. 767, 145107. https://doi.org/10.1016/j.scitotenv.2021.145107

Malhi, S.S., Lemke, R., 2008. Tillage, crop residue and N fertilizer effects on crop yield, nutrient uptake, soil quality and nitrous oxide gas emissions in a second 4-yr rotation cycle. Soil Tillage Res. 96, 269-283. https://doi.org/10.1016/j.still.2007.06.011

MELCC, 2021b. Normales climatiques du Québec 1981-2010. Gouvernement du Québec. https://www.environnement.gouv.qc.ca/climat/normales/ (accessed 6 January 2022).

Mosier, A., Schimel, D., Valentine, D., Bronson, K., Parton, W., 1991. Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. Nature 350, 330-332. https://doi.org/10.1038/350330a0

Muller, A., Schader, C., El-Hage Scialabba, N., Brüggemann, J., Isensee, A., Erb, K.-H., Smith, P., Klocke, P., Leiber, F., Stolze, M., Niggli, U., 2017. Strategies for feeding the world more sustainably with organic agriculture. Nature communications 8, 1290. https://doi.org/10.1038/s41467-017-01410-w

Mumford, M.T., Rowlings, D.W., Scheer, C., De Rosa, D., Grace, P.R., 2019. Effect of irrigation scheduling on nitrous oxide emissions in intensively managed pastures. Agric. Ecosyst. Environ. 272, 126-134. https://doi.org/10.1016/j.agee.2018.11.011

Nesbit, S.P., Breitenbeck, G.A., 1992. A laboratory study of factors influencing methane uptake by soils. Agric. Ecosyst. Environ. 41, 39-54. https://doi.org/10.1016/0167-8809(92)90178-E

Nunes, M.R., Karlen, D.L., Moorman, T.B., 2020. Tillage intensity effects on soil structure indicators - A U.S. meta-analysis. Sustainability 12, 2071. https://doi.org/10.3390/su12052071

Pan, H., Ying, S., Liu, H., Zeng, L., Zhang, Q., Xu, J., Li, Y., Di, H., Liu, Y., 2018. Microbial pathways for nitrous oxide emissions from sheep urine and dung in a typical steppe grassland. Biol. Fertil. Soils, 54, 717-730. https://doi.org/10.1007/s00374-018-1297-2

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. J. Environ. Qual. 41, 427-435. https://doi.org/10.2134/jeq2011.0261

Pelster, D.E., Chantigny, M.H., Royer, I., Angers, D.A., Vanasse, A., 2021. Reduced tillage increased growing season N_2O emissions from a fine but not a coarse textured soil under the cool, humid climate of eastern Canada. Soil Tillage Res. 206, 104833. https://doi.org/10.1016/j.still.2020.104833

R Core Team, 2012. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/ (accessed 6 January 2022).

Robertson, G.P., Paul, E.A., Harwood, R.R., 2000. Greenhouse gases in intensive agriculture: Contributions of individual gases to the radiative forcing of the atmosphere. Science 289, 1922-1925.

Rochette, P., Angers, D.A., Bélanger, G., Chantigny, M.H., Prévost, D., Lévesque, G., 2004a. Emissions of N_2O from alfalfa and soybean crops in eastern Canada. Soil Sci. Soc. Am. J. 68, 493-506. https://doi.org/10.2136/sssaj2004.4930

Rochette, P., Angers, D.A., Chantigny, M.H., Bertrand, N., 2008a. Nitrous oxide emissions respond differently to no-till in a loam and a heavy clay soil. Soil Sci. Soc. Am. J. 72, 1363-1369. https://doi.org/10.2136/sssaj2007.0371

Rochette, P., Angers, D.A., Chantigny, M.H., Bertrand, N., Côté, D., 2004b. Carbon dioxide and nitrous oxide emissions following fall and spring applications of pig slurry to an agricultural soil. Soil Sci. Soc. Am. J. 68, 1410-1420. https://doi.org/10.2136/sssaj2004.1410

Rochette, P., Angers, D.A., Chantigny, M.H., Gagnon, B., Bertrand, N., 2008b. N_2O fluxes in soils of contrasting textures fertilized with liquid and solid dairy cattle manures. Can. J. Soil Sci. 88, 175-187. https://doi.org/10.4141/CJSS06016

Rochette, P., Bertrand, N., 2008. Soil-surface gas emissions. In: Carter, M.R., Gregorich, E.G. (Eds.), Soil sampling and methods of analysis. CRC Press, Boca Raton, FL, pp. 851-861.

Rochette, P., Liang, C., Pelster, D., Bergeron, O., Lemke, R., Kroebel, R., MacDonald, D., Yan, W., Flemming, C., 2018. Soil nitrous oxide emissions from agricultural soils in Canada: Exploring relationships with soil, crop and climatic variables. Agric. Ecosyst. Environ. 254, 69-81. https://doi.org/10.1016/j.agee.2017.10.021

Rochette, P., van Bochove, E., Prévost, D., Angers, D.A., Côté, D., Bertrand, N., 2000. Soil carbon and nitrogen dynamics following application of pig slurry for the 19th consecutive year II. Nitrous oxide fluxes and mineral nitrogen. Soil Sci. Soc. Am. J. 64, 1396-1403. https://doi.org/10.2136/sssaj2000.6441396x

Saunois, M., R. Stavert, A., Poulter, B., Bousquet, P., G. Canadell, J., B. Jackson, R., A. Raymond, P., J. Dlugokencky, E., Houweling, S., K. Patra, P., Ciais, P., K. Arora, V., Bastviken, D., Bergamaschi, P., R. Blake, D., Brailsford, G., Bruhwiler, L., M. Carlson, K., Carrol, M., Castaldi, S., Chandra, N., Crevoisier, C., M. Crill, P., Covey, K., L. Curry, C., Etiope, G., Frankenberg, C., Gedney, N., I. Hegglin, M., Höglund-Isaksson, L., Hugelius, G., Ishizawa, M., Ito, A., Janssens-Maenhout, G., M. Jensen, K., Joos, F., Kleinen, T., B. Krummel, P., L. Langenfelds, R., G. Laruelle, G., Liu, L., MacHida, T., Maksyutov, S., C. McDonald, K., McNorton, J., A. Miller, P., R. Melton, J., Morino, I., Müller, J., Murguia-Flores, F., Naik, V., Niwa, Y., Noce, S., O'Doherty, S., J. Parker, R., Peng, C., Peng, S., P. Peters, G., Prigent, C., Prinn, R., Ramonet, M., Regnier, P., J. Riley, W., A. Rosentreter, J., Segers, A., J. Simpson, I., Shi, H., J. Smith, S., Paul Steele, L., F. Thornton, B., Tian, H., Tohjima, Y., N. Tubiello, F., Tsuruta, A., Viovy, N., Voulgarakis, A., S. Weber, T., Van Weele, M., R. Van Der Werf, G., F. Weiss, R., Worthy, D., Wunch, D., Yin, Y., Yoshida, Y., Zhang, W., Zhang, Z., Zhao, Y., Zheng, B., Zhu, Q., Zhu, Q., Zhuang, Q., 2020. The global methane budget 2000-2017. Earth Syst. Sci. Data 12, 1561-1623. https://doi.org/10.5194/essd-12-1561-2020

Serrano-Silva, N., Sarria-Guzmàn, Y., Dendooven, L., Luna-Guido, M., 2014. Methanogenesis and methanotrophy in soil: A review. Pedosphere 24, 291-307. https://doi.org/10.1016/S1002-0160(14)60016-3

Shcherbak, I., Millar, N., Robertson, G.P., 2014. Global metaanalysis of the nonlinear response of soil nitrous oxide (N_2O) emissions to fertilizer nitrogen. Proc. Natl. Acad. Sci. 111, 9199-9204. https://doi.org/10.1073/pnas.1322434111

Skinner, C., Gattinger, A., Krauss, M., Krause, H.-M., Mayer, J., van der Heijden, M.G.A., Mäder, P., 2019. The impact of long-term organic farming on soil-derived greenhouse gas emissions. Sci. Rep. 9, 1-10. https://doi.org/10.1038/s41598-018-38207-w

Soil Classification Working Group, 1998. The Canadian System of Soil Classification, third ed. Agriculture and Agri-Food Canada, Publication 1646.

Statistics Canada, 2017. Growing opportunity through innovation in agriculture. Minister of Industry, Government of Canada. https://www150.statcan.gc.ca/n1/en/pub/95-640-x/2016001/article/14816-eng.pdf (accessed 6 January 2022).

Ta, T.C., Macdowall, F.D.H., Faris, M.A., 1986. Excretion of nitrogen assimilated from N_2 fixed by nodulated roots of alfalfa (*Medicago sativa*). Can. J. Bot. 64, 2063-2067. https://doi.org/10.1139/b86-270

Thangarajan, R., Bolan, N.S., Tian, G., Naidu, R., Kunhikrishnan, A., 2013. Role of organic amendment application on greenhouse gas emission from soil. Sci. Total Environ. 465, 72-96. https://doi.org/10.1016/j.scitotenv.2013.01.031

Thilakarathna, M.S., McElroy, M.S., Chapagain, T., Papadopoulos, Y.A., Raizada, M.N., 2016. Belowground nitrogen transfer from legumes to non-legumes under managed herbaceous cropping systems. A review. Agron. Sustain. Dev. 36, 1-16. https://doi.org/10.1007/s13593-016-0396-4

Tian, H., Xu, R., Canadell, J.G., Thompson, R.L., Winiwarter, W., Suntharalingam, P., Davidson, E.A., Ciais, P., Jackson, R.B., Janssens-Maenhout, G., Prather, M.J., Regnier, P., Pan, N., Pan, S., Peters, G.P., Shi, H., Tubiello, F.N., Zaehle, S., Zhou, F., Arneth, A., Battaglia, G., Berthet, S., Bopp, L., Bouwman, A.F., Buitenhuis, E.T., Chang, J., Chipperfield, M.P., Dangal, S.R.S., Dlugokencky, E., Elkins, J.W., Eyre, B.D., Fu, B., Hall, B., Ito, A., Joos, F., Krummel, P.B., Landolfi, A., Laruelle, G.G., Lauerwald, R., Li, W., Lienert, S., Maavara, T., MacLeod, M., Millet, D.B., Olin, S., Patra, P.K., Prinn, R.G., Raymond, P.A., Ruiz, D.J., van der Werf, G.R., Vuichard, N., Wang, J., Weiss, R.F., Wells, K.C., Wilson, C., Yang, J., Yao, Y., 2020. A comprehensive quantification of global nitrous oxide sources and sinks. Nature 586, 248-256. https://doi.org/10.1038/s41586-020-2780-0

USDA, 1999. Soil taxonomy : A basic system of soil classification for making and interpreting soil surveys. United States Department of Agriculture, Washington, DC.

USDA, 2020. 2017 Census of agriculture: 2019 Organic survey. United States Department of Agriculture, National Agricultural Statistics Service.

Van Kessel, C., Venterea, R., Six, J., Adviento-Borbe, M.A., Linquist, B., Groenigen, K.J., 2013. Climate, duration, and N placement determine N_2O emissions in reduced tillage systems: a meta-analysis. Global Change Biol. 19, 33-44. https://doi.org/10.1111/j.1365-2486.2012.02779.x

Vance, C.P., Heichel, G.H., Barnes, D.K., Bryan, J.W., Johnson, L.E., 1979. Nitrogen fixation, nodule development, and vegetative regrowth of alfalfa (*Medicago sativa* L.) following harvest. Plant Physiol. 64, 1-8. https://doi.org/10.1104/pp.64.1.1

Venterea, R.T., Petersen, S.O., de Klein, C.A.M., Pedersen, A.R., Noble, A.D.L., Rees, R.M., Gamble, J.D., Parkin, T.B., 2020. Global Research Alliance N_2O chamber methodology guidelines: Flux calculations. J. Environ. Qual. 49, 1141-1155. https://doi.org/10.1002/jeq2.20118

Wagner-Riddle, C., Thurtell, G., Kidd, G., Beauchamp, E., Sweetman, R., 1997. Estimates of nitrous oxide emissions from agricultural fields over 28 months. Can. J. Soil Sci. 77, 135-144. https://doi.org/10.4141/S96-103

Wagner-Riddle, C., Furon, A., Mclaughlin, N.L., Lee, I., Barbeau, J., Jayasundara, S., Parkin, G., Von Bertoldi, P., Warland, J., 2007. Intensive measurement of nitrous oxide emissions from a corn–soybean–wheat rotation under two contrasting management systems over 5 years. Global Change Biol. 13, 1722-1736. https://doi.org/0.1111/j.1365-2486.2007.01388.x

Westphal, M., Tenuta, M., Entz, M.H., 2018. Nitrous oxide emissions with organic crop production depends on fall soil moisture. Agric. Ecosyst. Environ. 254, 41-49. https://doi.org/10.1016/j.agee.2017.11.005

Willer, H., Schaack, D., Lernoud, J., 2019. Organic farming and market development in Europe and the European Union. The world of organic agriculture. Statistics and emerging trends 2019. Research Institute of Organic Agriculture FiBL and IFOAM - Organics International, pp. 217-254.

Wrage, N., Velthof, G.L., Laanbroek, H.J., Oenema, O., 2004. Nitrous oxide production in grassland soils: assessing the contribution of nitrifier denitrification. Soil Biol. Biochem. 36, 229-236. https://doi.org/10.1016/j.soilbio.2003.09.009

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 Biol. Biochem. 123, A3-A16. https://doi.org/10.1016/j.soilbio.2018.03.020

Yao, Z., Liu, C., Zheng, X., Butterbach-Bahl, K., Pelster, D.E., 2020. Soil N intensity as a measure to estimate annual N_2O and NO fluxes from natural and managed ecosystems. Curr. Opin. Environ. Sustain. 47, 1-6. https://doi.org/10.1016/j.cosust.2020.03.008

Zebarth, B.J., Rochette, P., Burton, D.L., 2008a. N_2O emissions from spring barley production as influenced by fertilizer nitrogen rate. Can. J. Soil Sci. 88, 197-205. https://doi.org/10.4141/CJSS06006

Zebarth, B.J., Rochette, P., Burton, D.L., Price, M., 2008b. Effect of fertilizer nitrogen management on N_2O emissions in commercial corn fields. Can. J. Soil Sci. 88, 189-195. https://doi.org/10.4141/CJSS06010

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. Glob. Change Biol. 23, 4068-4083. https://doi.org/10.1111/gcb.13648

3.10 Tables

Table 1. Organic cropping systems tested during the two-year study on a sandy loam soil.

Cropping system	2019 Crop	2019 Timing of primary tillage	2019 Timing of PM application	2019 N input from PM (kg N ha^{-1})	2020 Crop	2020 Timing of primary tillage	2020 Timing of PM application	2020 N input from PM (kg N ha^{-1})	2020 N input from GM (kg N ha^{-1})	2020 N input from PM and GM (kg N ha^{-1})
MP-PM[1]	grain corn	spring, fall	spring	200	soybean	fall	-	-	-	-
MP-PMGM	barley	spring	spring	74	grain corn	spring	spring	113	117	230
CP-PM[1]	soybean	fall	-	-	spring wheat	fall	spring	129	-	129
CP-GM	barley	spring	-	-	grain corn	spring	-	-	87	87
CP-PMGM	barley	spring	spring	74	grain corn	spring	spring	121	85	206
CP-NM[2]	soybean	fall	-	-	spring wheat	fall	-	-	-	-
PF-PM	perennial forage	-	spring	63	perennial forage	-	-	-	-	-
BF	bare fallow	-	-	-	bare fallow	-	-	-	-	-

Tillage: moldboard plough (MP), chisel plough (CP); Fertilization: poultry manure (PM), fall-seeded pea green manure (GM), poultry manure and fall-seeded pea green manure (PMGM), no poultry manure nor green manure (NM). Crop sequence: perennial forage (PF), bare fallow (BF).

[1] Poultry manure was applied in at least one cropping-system-year for MP-PM and CP-PM.

[2] No GHG and soil mineral N measurements were performed in 2019 as it was identical to CP-PM cropping system.

Table 2. Soil nitrogen (N) intensity, area-scaled nitrous oxide (N_2O) emissions, area-scaled methane (CH_4) emissions, global warming potential (GWP), grain yields, yield-scaled N_2O emissions in grain dry matter (DM), and yield-scaled N_2O emissions in grain N DM.

Cropping system	Crop	Soil N intensity (g NO_3-N and NH_4-N d kg^{-1} dry soil)		N_2O emission (kg N ha^{-1})		CH_4 emissions (kg C ha^{-1})		GWP (kg $CO_{2\,eq}$ ha^{-1})		Yield (Mg grain DM ha^{-1})	Yield-scaled N_2O emissions (g N_2O-N kg^{-1} grain DM)		Yield-scaled N_2O emissions (g N_2O-N kg^{-1} grain N DM)	
	2019													
MP-PM[1]	grain corn	1.500	b	3.55	a	-0.07	a	939	a	5.68	0.63	a	9.06	a
MP-PMGM	barley	0.542	cd	1.73	ab	-0.21	a	453	ab	1.84	0.96	a	8.51	a
CP-PM[1]	soybean	0.851	c	0.86	bc	0.18	a	233	b	2.76	0.31	a	0.72	b
CP-GM	barley	0.495	d	0.52	c	-0.22	a	134	b	0.91	0.60	a	5.00	a
CP-PMGM	barley	0.505	d	1.17	abc	-0.42	a	299	ab	1.54	0.78	a	6.92	a
CP-NM[2]	soybean	-		-		-		-		-	-		-	
PF-PM	perennial forage	0.491	d	1.07	bc	-0.44	a	267	b	-	-		-	
BF	bare fallow	2.417	a	1.53	abc	0.08	a	402	ab	-	-		-	
	2020													
MP-PM[1]	soybean	1.088	bc	0.84	abc	-0.65	a	204	abc	2.68	0.32	a	0.75	b
MP-PMGM	grain corn	1.631	a	0.88	abc	-0.60	ab	217	abc	8.29	0.11	b	1.52	a
CP-PM[1]	spring wheat	1.316	ab	0.96	bc	-0.61	ab	236	ab	crop failure	crop failure		crop failure	
CP-GM	grain corn	1.010	bc	0.47	a	-0.57	ab	109	c	5.29	0.09	b	1.49	ab
CP-PMGM	grain corn	1.452	ab	0.78	abc	-0.53	ab	191	abc	8.02	0.10	b	1.48	ab
CP-NM	spring wheat	0.756	c	0.61	ab	-0.31	ab	152	bc	crop failure	crop failure		crop failure	
PF-PM[3]	perennial forage	0.784	c	1.38	c	-0.33	ab	357	a	6.65	0.21	ab	-	
BF	bare fallow	1.775	a	1.44	c	-0.04	b	380	a	-	-		-	
	***P*-value**													
2019		< 0.001		< 0.001		0.063		< 0.001		-	0.026		< 0.001	
2020		< 0.001		< 0.001		0.032		< 0.001		-	< 0.001		0.014	

Tillage: moldboard plough (MP), chisel plough (CP); Fertilization: poultry manure (PM), fall-seeded pea green manure (GM), poultry manure and fall-seeded pea green manure (PMGM), no poultry manure nor green manure (NM); Crop sequence: perennial forage (PF), bare fallow (BF).

[1] Poultry manure was applied at least in one cropping system-year in MP-PM and CP-PM.

[2] No greenhouse gas and soil mineral N measurements were performed in 2019 as it was identical to CP-PM cropping system.

[3] PF yields are expressed in Mg aboveground biomass DM.

Different lowercase letters indicate significant differences between cropping systems within the experimental year with a *P*-value < 0.05.

Table 3. Grain yields and yield-scaled nitrous oxide (N_2O) emissions comparisons with simple contrasts for three organic cropping systems in 2019 and 2020.

Contrasts	Yield (Mg grain DM ha^{-1}) comparisons					Yield-scaled (g N_2O-N kg^{-1} grain DM) comparisons				
	estimate	SE	df	t.ratio	*P*-value	estimate	SE	df	t.ratio	*P*-value
2019										
Barley MP-PMGM vs CP-PMGM	0.343	0.312	9	1.097	0.3012	0.174	0.259	14.3	0.670	0.5137
Barley CP-PMGM vs CP-GM	0.720	0.312	9	2.306	0.0466	0.180	0.207	14.3	0.866	0.4007
Barley MP-PMGM vs CP-GM	1.063	0.312	9	3.402	0.0078	0.353	0.238	14.3	1.483	0.1598
2020										
Corn MP-PMGM vs CP-PMGM	0.327	0.893	9	0.367	0.7224	0.009	0.024	13.9	0.395	0.6986
Corn CP-PMGM vs CP-GM	3.188	0.893	9	3.572	0.0050	0.005	0.022	13.9	0.215	0.8327
Corn MP-PMGM vs CP-GM	3.516	0.893	9	3.938	0.0034	0.014	0.023	13.9	0.609	0.5527

Tillage: moldboard plough (MP), chisel plough (CP); Fertilization: fall-seeded pea green manure (GM), poultry manure and fall-seeded pea green manure (PMGM). SE: Standard error. Df: degree of freedom.

Table 4. Nitrous oxide (N_2O) emissions of organic cropping systems in the present two-year study and of different conventional cropping systems in previous studies on loamy soils[1] in similar climatic conditions in eastern Canada.

Crop	N_2O emissions organic cropping systems in the present study (kg N_2O-N ha^{-1})	N_2O emissions conventional cropping systems in eastern Canada (kg N_2O-N ha^{-1})	References
Barley	0.5 – 1.7	0.6 – 8.1	(Rochette et al., 2008a; Wagner-Riddle et al., 1997; Zebarth et al., 2008a)
Spring wheat and winter wheat	0.6– 1.0	0.2 – 2.2	(Drury et al., 2008; Machado et al., 2021; Pelster et al., 2012; Pelster et al., 2021; Wagner-Riddle et al., 2007)
Corn	0.5 – 3.6	0.4 – 6.0	(Chantigny et al., 2010; Lessard et al., 1996; Machado et al., 2021; Pelster et al., 2021; Rochette et al., 2008b; Rochette et al., 2000; Wagner-Riddle et al., 1997; Wagner-Riddle et al., 2007; Zebarth et al., 2008b)
Soybean	0.8 – 0.9	0.2 – 3.1	(Drury et al., 2008; Machado et al., 2021; Pelster et al., 2021; Rochette et al., 2004a; Wagner-Riddle et al., 1997; Wagner-Riddle et al., 2007)
Perennial forage	1.1 – 1.4	0.2 – 1.5	(Chantigny et al., 2007; MacDonald et al., 2011; Rochette et al., 2004a; Wagner-Riddle et al., 1997)

[1] Studies in silt loam soils were included and studies in clay loam soils were not included.

3.11 Figures

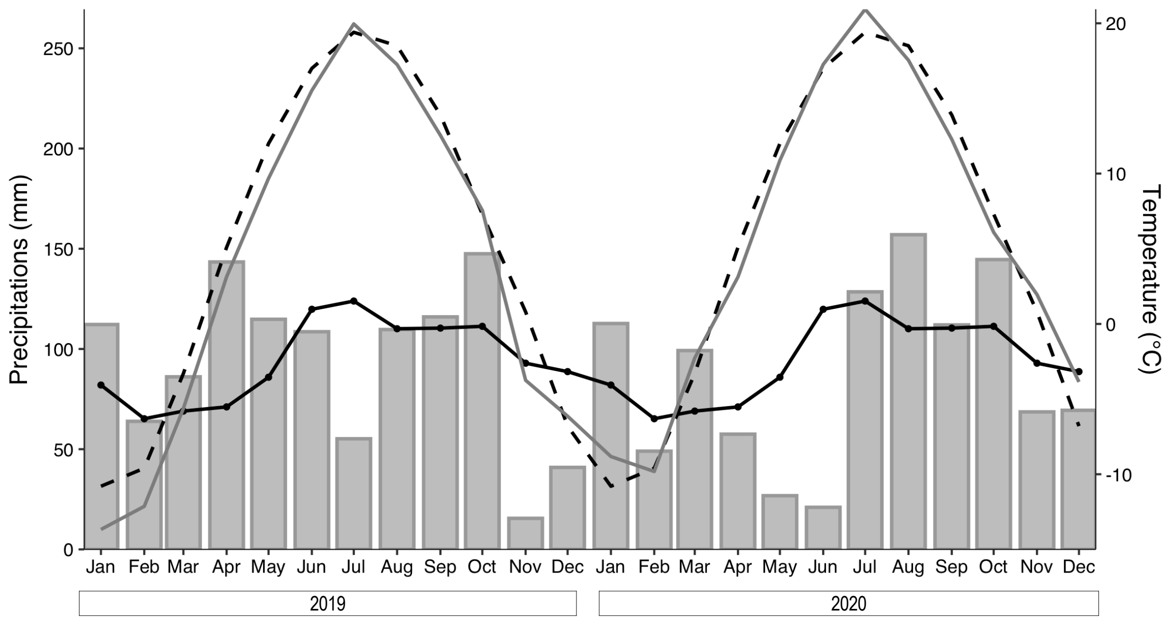


Fig. 3. Monthly precipitations (grey bars) and monthly mean air temperature (grey line) during the two-year study (2019 and 2020) compared with long-term normal (1981-2010) monthly precipitation (black line with dots) and monthly mean air temperature (black dashed line).

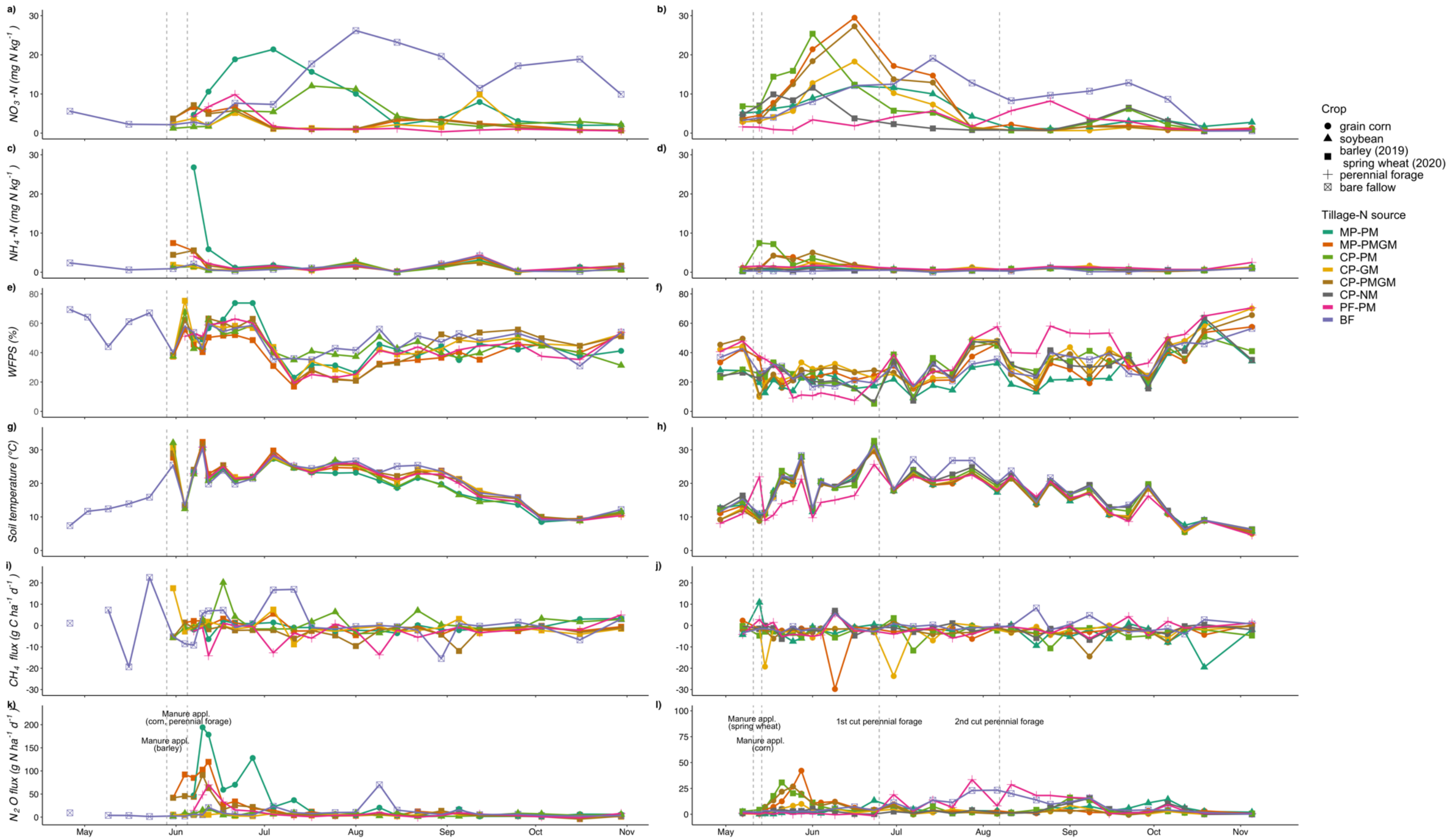


Fig. 4. Soil nitrate (NO_3) concentrations, ammonium (NH_4) concentrations, water-filled pore space (WFPS), soil temperature, methane (CH_4) daily fluxes, and nitrous oxide (N_2O) daily fluxes over time for different cropping systems in 2019 (left) and 2020 (right). Dotted lines indicate poultry manure (PM) applications and perennial forage cuts. Tillage: moldboard plough (MP), chisel plough (CP); Fertilization: poultry manure (PM), fall-seeded pea green manure (GM), poultry manure and fall-seeded pea green manure (PMGM), no poultry manure nor green manure (NM); Crop sequence: perennial forage (PF), bare fallow (BF).

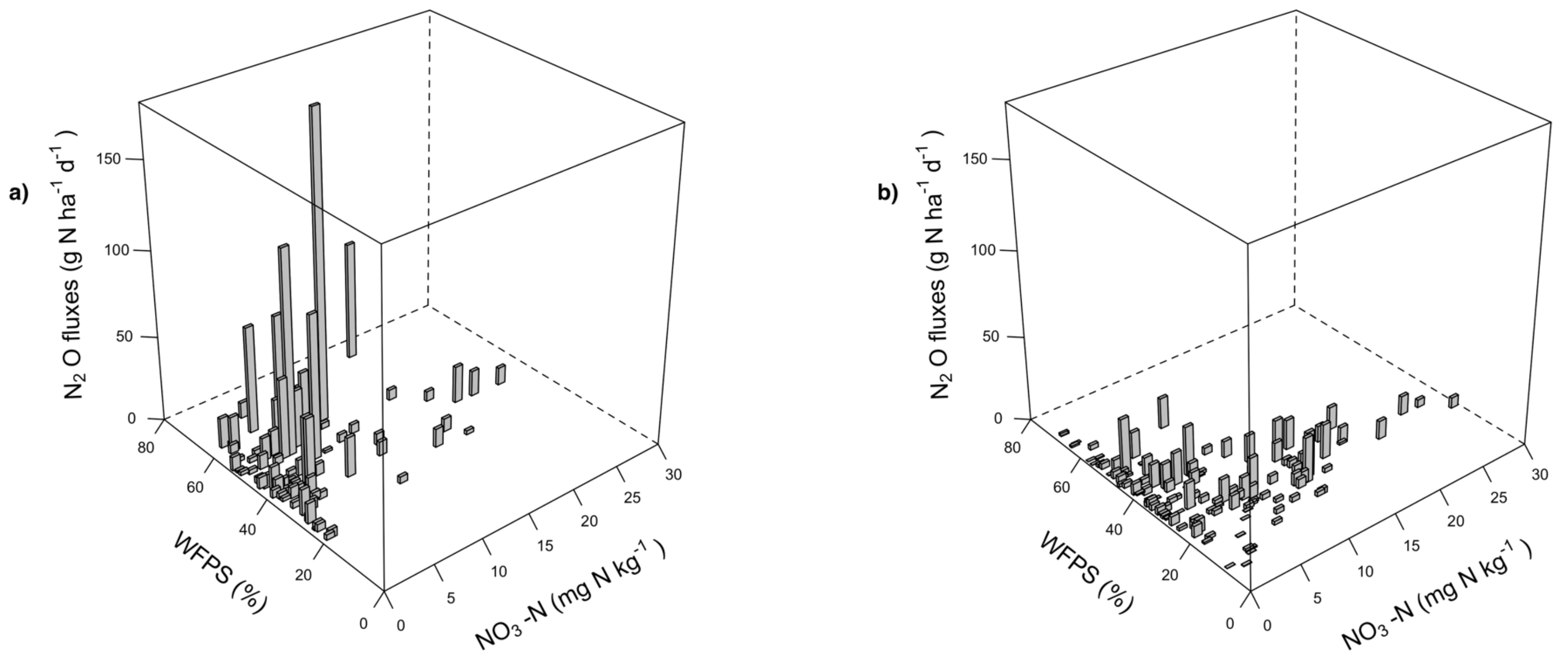
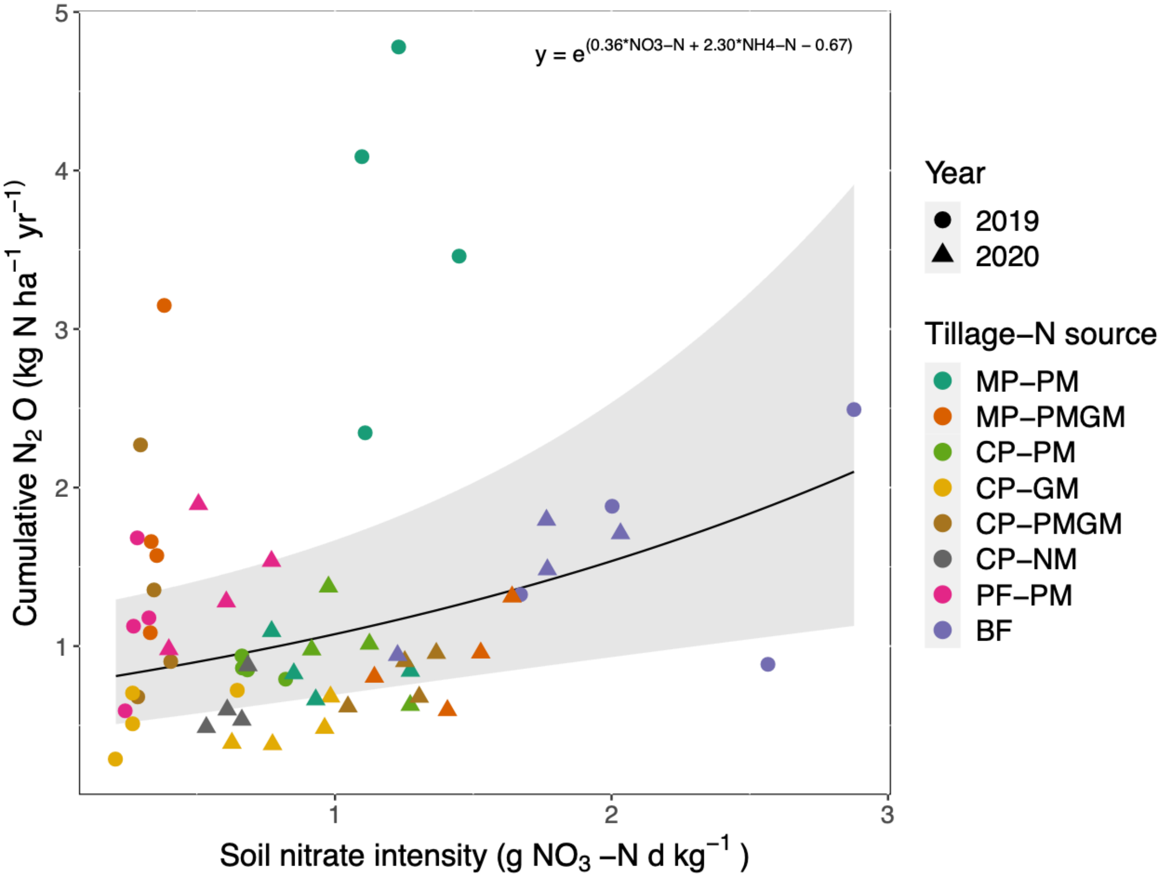


Fig. 5. Nitrous oxide (N_2O) daily fluxes as a function of soil nitrate (NO_3) concentrations and water-filled pore space (WFPS) in various organic cropping systems in a) 2019 and b) 2020.



	Estimate	SE	*P*-value
Y-intercept	-0.6652	0.3163	0.1053
NO_3-N	0.3565	0.1151	0.0031
NH_4-N	2.1168	1.0709	0.0530

Fig. 6. Cumulative nitrous oxide (N_2O) emissions versus soil nitrate (NO_3) intensities in different cropping systems and table of simple linear regressions for soil NO_3 and ammonium (NH_4) intensities in a two-year study. Tillage: moldboard plough (MP), chisel plough (CP); Fertilization: poultry manure (PM), fall-seeded pea green manure (GM), poultry manure and fall-seeded pea green manure (PMGM), no poultry manure nor green manure (NM); Crop sequence: perennial forage (PF), bare fallow (BF). SE: Standard error.

3.12 Supplementary material

Table S 1. Dates of the different field operations in a two-year study.

	2019									2020								
Treatment	Crop	Inversion tillage (MP)	Non-inversion tillage (CP)	Shallow cultivation[3]	Mechanical weeding	PM application	Crop sowing	Crop harvesting	Green manure sowing	Crop	Inversion tillage (MP)	Non-inversion tillage (CP)	Shallow cultivation[3]	Mechanical weeding	PM application	Crop sowing	Crop harvesting	Green manure sowing
MP-PM[1]	grain corn	June-5, Nov 28	-	June-5	June-10, 18, 25, July 2, 8	June-5	June-6	Nov-11	-	soybean	Nov-2	-	May-14	May-27, June-12, 23, July-2, 15	-	May-27	Oct-8	-
MP-PMGM	barley	May-29	-	May-29, Aug-28	June-4	May-29	May-30	Aug-27	Aug-28 (pea)	grain corn	May-13	-	May-14	May-22, 25, June-16, 26	May-14	May-22	Oct-28	June-26 (red clover interrow)
CP-PM[1]	soybean	-	Oct-29	June-5	June-10, 25, July 5, 10, 15	-	June-6	Oct-22	-	spring wheat	-	Oct-29	May-11, 12 Aug-28	May-15	May-11	May-12	Aug-27	May 15, June-10 (red clover) Sept-17 (oat)
CP-GM	barley	-	-	May-29, Aug-28	June-4	-	May-30	Aug-27	Aug-28 (pea)	grain corn	-	May-12	May-14	May-22, 25, June-16, 26	-	May-22	Oct-28	June-26 (red clover interrow)
CP-PMGM	barley	-	-	May-29, Aug-28	June-4	May-29	May-30	Aug-27	Aug-28 (pea)	grain corn	-	May-12	May-14	May-22, 25, June-16, 26	May-14	May-22	Oct-28	June-26 (red clover interrow)
CP-NM[2]	soybean	-	Oct-29	June-5	June-10, 25, July 5, 10, 15	-	June-6	Oct-22	-	spring wheat	-	Oct-29	May-11, 12 Aug-28	May-15	-	May-12	Aug-27	May 15, June-10 (red clover) Sept-17 (oat)
PF-PM	perennial forage	-	-	June-5	-	June-5	June-6	-	-	perennial forage	-	-	-	-	-	-	June-25, (harvested) Aug-6 (residues left on soil)	-
BF	bare fallow	-	-	May-29, July-3, July-24, Sept-30	-	-	-	-	-	bare fallow	-	-	June-2, July-14	-	-	-	-	-

Tillage: moldboard plough (MP; Varimaster 2, Kuhn, Saverne, Grand Est, FR) and chisel plough (CP; Bluebird GH, Rabe, Bad Essen, Lower Saxony, DE); Fertilization: poultry manure (PM), fall-seeded pea green manure (GM), poultry manure and fall-seeded pea green manure (PMGM), and no poultry manure nor green manure (NM). Crop sequence: perennial forage (PF) and bare fallow (BF).

[1] Poultry manure was applied in at least one cropping system-year for MP-PM and CP-PM.

[2] No greenhouse gas and soil mineral N measurements were performed in 2019 as it was identical to CP-PM cropping system.

[3] Shallow cultivation: S-tine harrow (unknown model and manufacturer), rototiller (GS 81 155, Kverneland group, Kvernaland, Western Norway, NO), and disk tiller (Diskomat 3,5N, Farmet, Česká Skalice, Náchod, CZE).

Table S 2. Application rates, nutrient concentration, and nutrient input from poultry manure, and nitrogen (N) input from a fall-seeded pea green manure in a two-year study.

	2019								2020								
Treatment	Crop	Rate applied ($Mg\ ha^{-1}$)	Nutrient concentration ($kg\ Mg^{-1}$)			Nutrient input ($kg\ ha^{-1}$)			Crop	Rate applied ($Mg\ ha^{-1}$)	Nutrient concentration ($kg\ Mg^{-1}$)			Nutrient input ($kg\ ha^{-1}$)			N input from GM ($kg\ ha^{-1}$)
			N	P_2O_5	K_2O	N	P_2O_5	K_2O			N	P_2O_5	K_2O	N	P_2O_5	K_2O	
MP-PM[1]	grain corn	7.0	28.5	22.5	14.0	199.5	157.5	98	soybean	-	-	-	-	-	-	-	-
MP-PMGM	barley	2.6	28.5	22.5	14.0	74.1	58.5	36.4	grain corn	5.2-5.9	20.4	19.7	12.9	113.2	109.3	71.6	117
CP-PM[1]	soybean	-	-	-	-	-	-	-	spring wheat	6.3	20.4	19.7	12.9	128.5	124.1	81.3	-
CP-GM	barley	-	-	-	-	-	-	-	grain corn	-	-	-	-	-	-	-	87
CP-PMGM	barley	2.6	28.5	22.5	14.0	74.1	58.5	36.4	grain corn	5.7-6.2	20.4	19.7	12.9	121.4	117.2	76.8	85
CP-NM[2]	soybean	-	-	-	-	-	-	-	spring wheat	-	-	-	-	-	-	-	-
PF-PM	perennial forage	2.2	28.5	22.5	14.0	62.7	49.5	30.8	perennial forage	-	-	-	-	-	-	-	-
BF	bare fallow	-	-	-	-	-	-	-	bare fallow	-	-	-	-	-	-	-	-

Tillage: moldboard plough (MP) and chisel plough (CP); Fertilization: poultry manure (PM), fall-seeded pea green manure (GM), poultry manure and fall-seeded pea green manure (PMGM), no poultry manure nor green manure (NM). Crop sequence: perennial forage (PF), bare fallow (BF).

[1] Poultry manure was applied in at least one cropping system-year for MP-PM and CP-PM.

[2] No greenhouse gas and soil mineral N measurements were performed in 2019 as it was identical to CP-PM cropping system.

Conclusion

In the specific context of this two-year study, organic cropping systems combining RT and a GM minimized N_2O emissions by preventing the accumulation of available soil N in the spring, confirming our first and second hypotheses. Conventional and RT intensities resulted in equivalent N_2O emissions and yields on this sandy loam soil. Our third and fourth hypotheses were also validated as the GM-based cropping system generated lower yields than manure-based cropping systems, but similar yield-scaled N_2O emissions were produced on this loamy soil. Overall, the dry conditions in spring 2020 induced low cumulative emissions in fertilized cropping systems despite high available NO_3 concentrations. All cropping systems were small sinks for CH_4 and CH_4 cumulative emissions were similar between cropping systems, which confirms our fifth hypothesis.

Organic crop rotations include more temporary forage crops that impact the N_2O losses compared to annual crops. Once established, in 2020, the legume-based PF reduced external N inputs however this did not reduce N_2O emissions compared to the annual crops. Inorganic N concentrations released after cutting the PF resulted in high N_2O emissions, which may have been minimized with a lower legume proportion in the PF species mixture. Some grass/legume species combination could further mitigate N_2O emissions, however, the use of different proportions and species mixtures is beyond the scope of this study.

Thus, on a sandy loam soil, farmers could include GM in their crop rotation to reduce their N_2O emissions. But farmers should also be careful when determining the proportion of leguminous species in their PF species mixture. Growing GM in combination with RT could help build up the soil fertility, that may eventually contribute to provide N to the growing crops. Reduced tillage and GM cropping practices may serve objectives of soil conservation, help prevent non-point source pollution, and improve soil structure by a high crop residues retention and a low soil disturbance. Thus, longer term evaluation of organic cropping systems over one or more complete crop rotations is necessary to better understand the longer term fertility effects of organic cropping systems, in particular, GM-based systems, and how N_2O emissions may be affected.

Further research are needed in organic cropping systems to 1) better understand the mechanisms underlying GHG mitigation in organic cropping systems; 2) determine the potential of different cover crops and perennial crop species either in pure stand or in species mixtures for GHG mitigation; 3) determine the profitability for farmers to implement reduced tillage and include more legume annual and/or perennial crops in a rotation; 4) provide large dataset for modelling purposes; 5) inform the policy makers about the benefits of agricultural operations on GHG mitigation.

Bibliography

Abalos, D., Brown, S.E., Vanderzaag, A.C., Gordon, R.J., Dunfield, K.E., Wagner-Riddle, C., 2016. Micrometeorological measurements over 3 years reveal differences in N_2O emissions between annual and perennial crops. Global Change Biol. 22, 1244-1255. https://doi.org/10.1111/gcb.13137

Abalos, D., Deyn, de G.B., Kuyper, T.W., Van Groenigen, J.W., 2014. Plant species identity surpasses species richness as a key driver of N_2O emissions from grassland. Global Change Biol. 20, 265-275. https://doi.org/10.1111/gcb.12350

Alves, B.J.R., Smith, K.A., Flores, R.A., Cardoso, A.S., Oliveira, W.R.D., Jantalia, C.P., Urquiaga, S., Boddey, R.M., 2012. Selection of the most suitable sampling time for static chambers for the estimation of daily mean N2O flux from soils. Soil Biol. Biochem. 46, 129-135. https://doi.org/10.1016/j.soilbio.2011.11.022

Angers, D.A., Edwards, L.M., Sanderson, J.B., Bissonnette, N., 1999. Soil organic matter quality and aggregate stability under eight potato cropping sequences in a fine sandy loam of Prince Edward Island. Can. J. Soil Sci. 79, 411-417. https://doi.org/10.4141/S98-033

Anglade, J., Billen, G., Garnier, J., 2015. Relationships for estimating N_2 fixation in legumes: incidence for N balance of legume-based cropping systems in Europe. Ecosphere 6, 1-24. https://doi.org/10.1890/ES14-00353.1

Baggs, E., Watson, C., Rees, R., 2000. The fate of nitrogen from incorporated cover crop and green manure residues. Nutr. Cycl. Agroecosyst. 56, 153-163. https://doi.org/10.1023/A:1009825606341

Ball, B.C., Griffiths, B.S., Topp, C.F.E., Wheatley, R., Walker, R.L., Rees, R.M., Watson, C.A., Gordon, H., Hallett, P.D., McKenzie, B.M., Nevison, I.M., 2014. Seasonal nitrous oxide emissions from field soils under reduced tillage, compost application or organic farming. Agric. Ecosyst. Environ. 189, 171-180. https://doi.org/10.1016/j.agee.2014.03.038

Ball, B.C., Scott, A., Parker, J.P., 1999. Field N_2O, CO_2 and CH_4 fluxes in relation to tillage, compaction and soil quality in Scotland. Soil Tillage Res. 53, 29-39. https://doi.org/10.1016/S0167-1987(99)00074-4

Barbieri, P., Pellerin, S., Nesme, T., 2017. Comparing crop rotations between organic and conventional farming. Sci. Rep. 7, 13761. https://doi.org/10.1038/s41598-017-14271-6

Barbieri, P., Pellerin, S., Seufert, V., Smith, L., Ramankutty, N., Nesme, T., 2021. Global option space for organic agriculture is delimited by nitrogen availability. Nat. Food 2, 363-372. https://doi.org/10.1038/s43016-021-00276-y

Basche, A.D., Miguez, F.E., Kaspar, T.C., Castellano, M.J., 2014. Do cover crops increase or decrease nitrous oxide emissions? A meta-analysis. J. Soil Water Conserv. 69, 471-482. https://doi.org/10.2489/jswc.69.6.471

Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 67. https://doi.org/10.18637/jss.v067.i01

Baudron, F., Giller, K.E., 2014. Agriculture and nature: Trouble and strife? Biol. Conserv. 170, 232-245. https://doi.org/10.1016/j.biocon.2013.12.009

Bender, M., Conrad, R., 1995. Effect of CH_4 concentrations and soil conditions on the induction of CH_4 oxidation activity. Soil Biol. Biochem. 27, 1517-1527.

Benoit, M., Garnier, J., Billen, G., Tournebize, J., Gréhan, E., Mary, B., 2015. Nitrous oxide emissions and nitrate leaching in an organic and a conventional cropping system (Seine basin, France). Agric. Ecosyst. Environ. 213, 131-141. https://doi.org/10.1016/j.agee.2015.07.030

Biernat, L., Taube, F., Loges, R., Kluß, C., Reinsch, T., 2020. Nitrous oxide emissions and methane uptake from organic and conventionally managed arable crop rotations on farms in northwest Germany. Sustainability 12. https://doi.org/10.3390/su12083240

Billen, G., Aguilera, E., Einarsson, R., Garnier, J., Gingrich, S., Grizzetti, B., Lassaletta, L., Le Noë, J., Sanz-Cobena, A., 2021. Reshaping the European agro-food system and closing its nitrogen cycle: The potential of combining dietary change, agroecology, and circularity. One Earth 4, 839-850. https://doi.org/10.1016/j.oneear.2021.05.008

Birch, H.F., 1958. The effect of soil drying on humus decomposition and nitrogen availability. Plant Soil 10, 9-31.

Birch, H.F., 1959. Further observations on humus decomposition and nitrification. Plant Soil 11, 262-286.

Birch, H.F., 1960. Nitrification in soils after different periods of dryness. Plant Soil 12, 81-96.

Boardman, D.L., Easterby, S., Clark, K.M., Kitchen, N.R., Staples, J.S., Reinbott, T.M., Kremer, R.J., 2018. Do tillage, cover crops, and compost management within organic grain cropping affect greenhouse gas emissions? Agron. J. 110, 1893-1904. https://doi.org/10.2134/agronj2018.01.0023

Brankatschk, G., Finkbeiner, M., 2014. Application of the Cereal Unit in a new allocation procedure for agricultural life cycle assessments. J. Clean. Prod. 73, 72-79. https://doi.org/10.1016/j.jclepro.2014.02.005

Braun, M., Bai, Y., McConkey, B., Farrell, R., Romo, J.T., Pennock, D., 2013. Greenhouse gas flux in a temperate grassland as affected by landform and disturbance. Landscape Ecol. 28, 709-723. https://doi.org/10.1007/s10980-013-9878-9

Brozyna, M.A., Petersen, S.O., Chirinda, N., Olesen, J.E., 2013. Effects of grass-clover management and cover crops on nitrogen cycling and nitrous oxide emissions in a stockless organic crop rotation. Agric. Ecosyst. Environ. 181, 115-126. https://doi.org/10.1016/j.agee.2013.09.013

Burton, D.L., Gillam, K.M., Zebarth, B.J., MacLeod, J.A., 2008. Effect of split application of fertilizer nitrogen on N_2O emissions from potatoes. Can. J. Soil Sci. 88, 229-239. https://doi.org/10.4141/CJSS06007

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? Phil. Trans. R. Soc. B 368, 20130122. https://doi.org/10.1098/rstb.2013.0122

Carter, M.R., Gregorich, E.G., Angers, D.A., Donald, R.G., Bolinder, M.A., 1998. Organic C and N storage, and organic C fractions, in adjacent cultivated and forested soils of eastern Canada. Soil Tillage Res. 47, 253-261.

CARTV, 2020. Portail BioQuébec, état et évolution du secteur biologique. Conseil des appellations réservées et des termes valorisants du Québec, Montréal, QC. https://www.portailbioquebec.info/ (accessed 6 January 2022).

Castro, M.S., Peterjohn, W.T., Melillo, J.M., Steudler, P.A., Gholz, H.L., Lewis, D., 1994. Effects of nitrogen fertilization on the fluxes of N_2O, CH_4, and CO_2 from soils in a Florida slash pine plantation. Can. J. For. Res. 24, 9-13. https://doi.org/10.1139/x94-002

CDA, 1997. Engrais verts et cultures intercalaires. Centre de développement d'agrobiologie, Sainte-Élisabeth-de-Warwick.

CGSB, 2020. Organic production systems: General principles and management standards. Canada General Standards Board: Standards Council of Canada, Ottawa, ON, CA.

Chadwick, D., John, F., Pain, B., Chambers, B., Williams, J., 2000. Plant uptake of nitrogen from the organic nitrogen fraction of animal manures: a laboratory experiment. J. Agric. Sci. 134, 159-168. https://doi.org/10.1017/S0021859699007510

Chantigny, M.H., Angers, D.A., Rochette, P., Bélanger, G., Massé, D., Côté, D., 2007. Gaseous nitrogen emissions and forage nitrogen uptake on soils fertilized with raw and treated swine manure. J. Environ. Qual. 36, 1864-1872. https://doi.org/10.2134/jeq2007.0083

Chantigny, M.H., Pelster, D.E., Perron, M.H., Rochette, P., Angers, D.A., Parent, L., Massé, D., Ziadi, N., 2013. Nitrous oxide emissions from clayey soils amended with paper sludges and biosolids of separated pig slurry. J. Environ. Qual. 42, 30-39. https://doi.org/10.2134/jeq2012.0196

Chantigny, M.H., Rochette, P., Angers, D.A., Bittman, S., Buckley, K., Massé, D., Bélanger, G., Eriksen-Hamel, N., Gasser, M.O., 2010. Soil nitrous oxide emissions following band-incorporation of fertilizer nitrogen and swine manure. J. Environ. Qual. 39, 1545-1553. https://doi.org/10.2134/jeq2009.0482

Chapuis-Lardy, L., Wrage, N., Metay, A.L., Chotte, J.-L., Bernoux, M., 2007. Soils, a sink for N_2O? A review. Global Change Biol. 13, 1-17. https://doi.org/10.1111/j.1365-2486.2006.01280.x

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. Agric. Ecosyst. Environ. 236, 88-98. https://doi.org/10.1016/j.agee.2016.11.021

Chivenge, P., Murwira, H., Giller, K., Mapfumo, P., Six, J., 2007. Long-term impact of reduced tillage and residue management on soil carbon stabilization: Implications for conservation agriculture on contrasting soils. Soil Tillage Res. 94, 328-337. https://doi.org/10.1016/j.still.2006.08.006

Clark, M., Tilman, D., 2017. Comparative analysis of environmental impacts of agricultural production systems, agricultural input efficiency, and food choice. Environ. Res. Lett. 12, 064016. https://doi.org/10.1088/1748-9326/aa6cd5

Clough, T.J., Sherlock, R.R., Rolston, D.E., 2005. A review of the movement and fate of N_2O in the subsoil. Nutr. Cycl. Agroecosyst. 72, 3-11. https://doi.org/10.1007/s10705-004-7349-z

Codex Alimentarius, 1999. Guidelines for the production, processing, labelling and marketing of organically produced foods. Adopted 1999. Revisions 2001, 2003, 2004 and 2007. Amendments 2008, 2009, 2010, 2012 and 2013. Codex Alimentarius Commission, Rome, Italy.

Conrad, R., 1996. Soil microorganisms as controllers of atmospheric trace gases (H_2, CO, CH_4, OCS, N_2O, and NO). Microbiological reviews 60, 609-640. https://doi.org/10.1128/MR.60.4.609-640.1996

Cooper, J., Baranski, M., Stewart, G., Nobel-de Lange, M., Bàrberi, P., Fließbach, A., Peigné, J., Berner, A., Brock, C., Casagrande, M., Crowley, O., David, C., Vliegher, A.D., Döring, F.T., Dupont, A.I., Entz, M., Grosse, M., Haase, T., Halde, C., Hammerl, V., Huiting, H., Leithold, G., Messmer, M., Schloter, M., Sukkel, W., van der Heijden, G.A.M., Willekens, K., Wittwer, R., Mäder, P., 2016. Shallow non-inversion tillage in organic farming maintains crop yields and increases soil C stocks: a meta-analysis. Agron. Sustain. Dev. 36, 22. https://doi.org/10.1007/s13593-016-0354-1

Davidson, E.A., Keller, M., Erickson, H.E., Verchot, L.V., Veldkamp, E., 2000. Testing a conceptual model of soil emissions of nitrous and nitric oxides. Bioscience 50, 667-680. https://doi.org/10.1641/0006-3568(2000)050[0667:TACMOS]2.0.CO;2

Diacono, M., Montemurro, F., 2010. Long-term effects of organic amendments on soil fertility. A review. Agron. Sustain. Dev. 30, 401-422. https://doi.org/10.1051/agro/2009040

Doltra, J., Gallejones, P., Olesen, J.E., Hansen, S., Frøseth, R.B., Krauss, M., Stalenga, J., Jończyk, K., Martínez-Fernández, A., Pacini, G.C., 2019. Simulating soil fertility management effects on crop yield and soil nitrogen dynamics in field trials under organic farming in Europe. Field Crops Res. 233, 1-11. https://doi.org/10.1016/j.fcr.2018.12.008

Drury, C., Yang, X., Reynolds, W., McLaughlin, N., 2008. Nitrous oxide and carbon dioxide emissions from monoculture and rotational cropping of corn, soybean and winter wheat. Can. J. Soil Sci. 88, 163-174. https://doi.org/10.4141/CJSS06015

Duncan, E., Dell, C., Kleinman, P., Beegle, D., 2017. Nitrous oxide and ammonia emissions from injected and broadcast-applied dairy slurry. J. Environ. Qual. 46, 36-44. https://doi.org/10.2134/jeq2016.05.0171

Eagle, A.J., McLellan, E.L., Brawner, E.M., Chantigny, M.H., Davidson, E.A., Dickey, J.B., Linquist, B.A., Maaz, T.M., Pelster, D.E., Pittelkow, C.M., Kessel, C., Vyn, T.J., Cassman, K.G., 2020. Quantifying on-farm nitrous oxide emission reductions in food supply chains. Earth's Future 8. https://doi.org/10.1029/2020EF001504

Environment Canada, 2021. National Inventory Report 1990-2019: Greenhouse gas sources and sinks in Canada. Environment and Climate Change Canada, Gatineau, QC.

FADQ, 2021a. Rendements de référence 2020 en assurance récolte. Assurance récolte - Rendements de référence. Financière agricole du Québec. https://www.fadq.qc.ca/ (accessed 10 March 2022).

FADQ, 2021b. Rendements réels de l'année 2019 en assurance récolte. Assurance récolte - Rendements réels. Financière agricole du Québec. https://www.fadq.qc.ca/ (accessed 10 March 2022).

Firestone, M., Davidson, E., 1989. Microbiological basis of NO and N_2O production and consumption in soil. Exch. Trace Gases Between Terr. Ecosyst. Atmos. 47, 7-21.

Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliessbach, A., Buchmann, N., Mäder, P., Stolze, M., Smith, P., Scialabba, N.E.-H., 2012. Enhanced top soil carbon stocks under organic farming. Proc. Natl. Acad. Sci. 109, 18226-18231. https://doi.org/10.1073/pnas.1209429109

Ghosh, A., Patra, P.K., Ishijima, K., Umezawa, T., Ito, A., Etheridge, D.M., Sugawara, S., Kawamura, K., Miller, J.B., Dlugokencky, E.J., Krummel, P.B., Fraser, P.J., Steele, L.P., Langenfelds, R.L., Trudinger, C.M., White, J.W.C., Vaughn, B., Saeki, T., Aoki, S., Nakazawa, T., 2015. Variations in global methane sources and sinks during 1910-2010. Atmos. Chem. Phys. 15, 2595-2612. https://doi.org/10.5194/acp-15-2595-2015

Gomiero, T., Pimentel, D., Paoletti, M.G., 2011. Environmental impact of different agricultural management practices: conventional vs. organic agriculture. Crit. Rev. Plant Sci. 30, 95-124. https://doi.org/10.1080/07352689.2011.554355

Gregorich, E.G., Drury, C.F., Baldock, J.A., 2001. Changes in soil carbon under long-term maize in monoculture and legume-based rotation. Can. J. Soil Sci. 81, 21-31. https://doi.org/10.4141/S00-041

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 Tillage Res. 83, 53-72. https://doi.org/10.1016/j.still.2005.02.009

Griffis, T.J., Chen, Z., Baker, J.M., Wood, J.D., Millet, D.B., Lee, X., Venterea, R.T., Turner, P.A., 2017. Nitrous oxide emissions are enhanced in a warmer and wetter world. Proc. Natl. Acad. Sci. 114, 12081-12085. https://doi.org/10.1073/pnas.1704552114

Guardia, G., Tellez-Rio, A., García-Marco, S., Martin-Lammerding, D., Tenorio, J.L., Ibáñez, M.Á., Vallejo, A., 2016. Effect of tillage and crop (cereal versus legume) on greenhouse gas emissions and Global Warming Potential in a non-irrigated Mediterranean field. Agric. Ecosyst. Environ. 221, 187-197. https://doi.org/10.1016/j.agee.2016.01.047

Han, Z., Walter, M.T., Drinkwater, L.E., 2017a. Impact of cover cropping and landscape positions on nitrous oxide emissions in northeastern US agroecosystems. Agric. Ecosyst. Environ. 245, 124-134. https://doi.org/10.1016/j.agee.2017.05.018

Han, Z., Walter, M.T., Drinkwater, L.E., 2017b. N_2O emissions from grain cropping systems: a meta-analysis of the impacts of fertilizer-based and ecologically-based nutrient management strategies. Nutr. Cycl. Agroecosyst. 107, 335-355. https://doi.org/10.1007/s10705-017-9836-z

Hansen, S., Mæhlum, J.E., Bakken, L.R., 1993. N_2O and CH_4 fluxes in soil influenced by fertilization and tractor traffic. Soil Biol. Biochem. 25, 621-630. https://doi.org/10.1016/0038-0717(93)90202-M

Hao, X., Bal, B.C., Culley, J.L.B., Carter, M.R., Parkin, G.W., 2008. Soil density and porosity. In: Carter, M.R., Gregorich, E.G. (Eds.), Soil Sampling and Methods of Analysis. CRC Press, Boca Raton, FL, pp. 743-760.

Hoffman, E., Cavigelli, M.A., Camargo, G., Ryan, M., Ackroyd, V.J., Richard, T.L., Mirsky, S., 2018. Energy use and greenhouse gas emissions in organic and conventional grain crop production: Accounting for nutrient inflows. Agric. Syst. 162, 89-96. https://doi.org/10.1016/j.agsy.2018.01.021

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 Microbiol. Rev. 39, 729-749. https://doi.org/10.1093/femsre/fuv021

Hung, C.-Y., Ejack, L., Whalen, J.K., 2021. Fall-applied manure with cover crop did not increase nitrous oxide emissions during spring freeze-thaw periods. Appl. Soil Ecol. 158, 103786. https://doi.org/10.1016/j.apsoil.2020.103786

Hutchinson, J.J., Campbell, C.A., Desjardins, R.L., 2007. Some perspectives on carbon sequestration in agriculture. Agric. For. Meteorol. 142, 288-302. https://doi.org/10.1016/j.agrformet.2006.03.030

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 Biol. Biochem. 25, 1307-1315. https://doi.org/10.1016/0038-0717(93)90045-D

IFOAM, 2008. Principles of organic agriculture. IFOAM Organics International, Bonn, Germany. https://www.ifoam.bio/resource-library (accessed 10 March 2022).

IPCC, 2014. Climate Change 2014: Synthesis report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Core Writing Team, Pachauri, R.K., and Meyer, L.A. (Eds.). IPCC, Geneva, Switzerland.

IPCC, 2018. Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty. Masson-Delmotte, V., Zhai, P., Pörtner, H.-O., Roberts, D., Skea, J., Shukla, P.R., Pirani, A., Moufouma-Okia, W., Péan, C., Pidcock, R., Connors, S., Matthews, J.B.R., Chen, Y., Zhou, X., Gomis, M.I., Lonnoy, E., Maycock, T., Tignor, M. and Waterfield, T. (Eds.). In Press.

IPCC, 2019. 2019 Refinement to the 2006 IPCC Guidelines for national greenhouse gas inventories. Calvo Buendia, E., Tanabe, K., Kranjc, A., Baasansuren, J., Fukuda, M., Ngarize S., Osako, A., Pyrozhenko, Y., Shermanau, P. and Federici, S. (Eds). IPCC, Geneva, Switzerland.

ISQ, MAPAQ, 2020. Profil sectoriel de l'industrie bioalimentaire au Québec - Édition 2019. Gouvernement du Québec, Québec, Canada.

Ivarson, K.C., Sowden, F.J., 1970. Effect of frost action and storage of soil at freezing temperatures on the free amino acids, free sugars and respiratory activity of soil. Can. J. Soil Sci. 50, 191-198. https://doi.org/10.4141/cjss70-027

Jäckel, U., Schnell, S., Conrad, R., 2001. Effect of moisture, texture and aggregate size of paddy soil on production and consumption of CH_4. Soil Biol. Biochem. 33, 965-971. https://doi.org/10.1016/S0038-0717(00)00248-0

Jackson, L.E., Schimel, J.P., Firestone, M.K., 1989. Short-term partitioning of ammonium and nitrate between plants and microbes in an annual grassland. Soil Biol. Biochem. 21, 409-415. https://doi.org/10.1016/0038-0717(89)90152-1

Jarecki, M.K., Parkin, T.B., Chan, A.S., Hatfield, J.L., Jones, R., 2008. Greenhouse gas emissions from two soils receiving nitrogen fertilizer and swine manure slurry. J. Environ. Qual. 37, 1432-1438. https://doi.org/10.2134/jeq2007.0427

Jensen, E.S., Peoples, M.B., Boddey, R.M., Gresshoff, P.M., Hauggaard-Nielsen, H., Alves, B.J., Morrison, M.J., 2012. Legumes for mitigation of climate change and the provision of feedstock for biofuels and biorefineries. A review. Agron. Sustain. Dev. 32, 329-364. https://doi.org/10.1007/s13593-011-0056-7

Jungkunst, H.F., Freibauer, A., Neufeldt, H., Bareth, G., 2006. Nitrous oxide emissions from agricultural land use in Germany— a synthesis of available annual field data. J. Plant Nutr. Soil Sci. 169, 341-351. https://doi.org/10.1002/jpln.200521954

Karakurt, I., Aydin, G., Aydiner, K., 2012. Sources and mitigation of methane emissions by sectors: A critical review. Renewable Energy 39, 40-48. https://doi.org/10.1016/j.renene.2011.09.006

Khalil, M., Baggs, E., 2005. CH_4 oxidation and N_2O emissions at varied soil water-filled pore spaces and headspace CH_4 concentrations. Soil Biol. Biochem. 37, 1785-1794. https://doi.org/10.1016/j.soilbio.2005.02.012

Koponen, H.T., Martikainen, P.J., 2004. Soil water content and freezing temperature affect freeze–thaw related N_2O production in organic soil. Nutr. Cycl. Agroecosyst. 69, 213-219. https://doi.org/10.1023/B:FRES.0000035172.37839.24

Krauss, M., Ruser, R., Müller, T., Hansen, S., Mäder, P., Gattinger, A., 2017. Impact of reduced tillage on greenhouse gas emissions and soil carbon stocks in an organic grass-clover ley - winter wheat cropping sequence. Agric. Ecosyst. Environ. 239, 324-333. https://doi.org/10.1016/j.agee.2017.01.029

Kuznetsova, A., Brockhoff, P.B., Christensen, R.H., 2017. lmerTest package: tests in linear mixed effects models. J. Stat. Softw. 82, 1-26. https://doi.org/10.18637/jss.v082.i13

Kuzyakov, Y., Horwath, W.R., Dorodnikov, M., Blagodatskaya, E., 2019. Review and synthesis of the effects of elevated atmospheric CO_2 on soil processes: No changes in pools, but increased fluxes and accelerated cycles. Soil Biol. Biochem. 128, 66-78. https://doi.org/10.1016/j.soilbio.2018.10.005

Le Mer, J., Roger, P., 2001. Production, oxidation, emission and consumption of methane by soils: A review. Eur. J. Soil Biol. 37, 25-50. https://doi.org/10.1016/S1164-5563(01)01067-6

Lee, J., Necpálová, M., Six, J., 2020. Biophysical potential of organic cropping practices as a sustainable alternative in Switzerland. Agric. Syst. 181, 102822. https://doi.org/10.1016/j.agsy.2020.102822

Lessard, R., Rochette, P., Gregorich, E.G., Pattey, E., Desjardins, R.L., 1996. Nitrous oxide fluxes from manure-amended soil under maize. J. Environ. Qual. 25, 1371-1377. https://doi.org/10.2134/jeq1996.00472425002500060029x

Li, Y., Li, Z., Cui, S., Jagadamma, S., Zhang, Q., 2019. Residue retention and minimum tillage improve physical environment of the soil in croplands: A global meta-analysis. Soil Tillage Res. 194, 104292. https://doi.org/10.1016/j.still.2019.06.009

Liang, C., MacDonald, D., Thiagarajan, A., Flemming, C., Cerkowniak, D., Desjardins, R., 2020. Developing a country specific method for estimating nitrous oxide emissions from agricultural soils in Canada. Nutr. Cycl. Agroecosyst. 117, 145-167. https://doi.org/10.1007/s10705-020-10058-w

Linn, D.M., Doran, J.W., 1984. Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. Soil Sci. Soc. Am. J. 48, 1267-1272. https://doi.org/10.2136/sssaj1984.03615995004800060013x

Liu, S., Ji, C., Wang, C., Chen, J., Jin, Y., Zou, Z., Li, S., Niu, S., Zou, J., 2018a. Climatic role of terrestrial ecosystem under elevated CO_2: a bottom-up greenhouse gases budget. Ecol. Lett. 21, 1108-1118. https://doi.org/10.1111/ele.13078

Liu, S., Schloter, M., Brüggemann, N., 2018b. Accumulation of NO_2^- during periods of drying stimulates soil N_2O emissions during subsequent rewetting. Eur. J. Soil Sci. 69, 936-946. https://doi.org/10.1111/ejss.12683

Lotter, D.W., 2003. Organic Agriculture. J. Sustainable Agric. 21, 59-128. https://doi.org/10.1300/J064v21n04_06

MacDonald, J.D., Rochette, P., Chantigny, M.H., Angers, D.A., Royer, I., Gasser, M.-O., 2011. Ploughing a poorly drained grassland reduced N_2O emissions compared to chemical fallow. Soil Tillage Res. 111, 123-132. https://doi.org/10.1016/j.still.2010.09.005

Machado, P.V.F., Farrell, R.E., Deen, W., Voroney, R.P., Congreves, K.A., Wagner-Riddle, C., 2021. Contribution of crop residue, soil, and fertilizer nitrogen to nitrous oxide emissions varies with long-term crop rotation and tillage. Sci. Total Environ. 767, 145107. https://doi.org/10.1016/j.scitotenv.2021.145107

Maggiotto, S.R., Wagner-Riddle, C., 2001. Winter and spring thaw measurements of N_2O, NO and NO_x fluxes using a micrometeorological method. Water, Air and Soil Pollution: Focus 1, 89-98. https://doi.org/10.1023/A:1013174214957

Malhi, S.S., Lemke, R., 2008. Tillage, crop residue and N fertilizer effects on crop yield, nutrient uptake, soil quality and nitrous oxide gas emissions in a second 4-yr rotation cycle. Soil Tillage Res. 96, 269-283. https://doi.org/10.1016/j.still.2007.06.011

MAPAQ, 2020. Agir, pour une agriculture durable : Plan 2020-2030. Gouvernement du Québec, Québec, Canada.

MELCC, 2021a. Inventaire québécois des émissions de gaz à effet de serre en 2019 et leur évolution depuis 1990. Ministère de l'Environnement et de la Lutte contre les changements climatiques, Québec, Canada.

MELCC, 2021b. Normales climatiques du Québec 1981-2010. Gouvernement du Québec, Québec, Canada. https://www.environnement.gouv.qc.ca/climat/normales/ (accessed 6 January 2022).

Meyer-Aurich, A., Weersink, A., Janovicek, K., Deen, B., 2006. Cost efficient rotation and tillage options to sequester carbon and mitigate GHG emissions from agriculture in eastern Canada. Agric. Ecosyst. Environ. 117, 119-127. https://doi.org/10.1016/j.agee.2006.03.023

Moser, G., Gorenflo, A., Brenzinger, K., Keidel, L., Braker, G., Marhan, S., Clough, T.J., Müller, C., 2018. Explaining the doubling of N_2O emissions under elevated CO_2 in the Giessen FACE via in-field ^{15}N tracing. Global Change Biol. 24, 3897-3910. https://doi.org/10.1111/gcb.14136

Mosier, A., Schimel, D., Valentine, D., Bronson, K., Parton, W., 1991. Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. Nature 350, 330-332. https://doi.org/10.1038/350330a0

Muller, A., Schader, C., El-Hage Scialabba, N., Brüggemann, J., Isensee, A., Erb, K.-H., Smith, P., Klocke, P., Leiber, F., Stolze, M., Niggli, U., 2017. Strategies for feeding the world more sustainably with organic agriculture. Nature communications 8, 1290. https://doi.org/10.1038/s41467-017-01410-w

Mumford, M.T., Rowlings, D.W., Scheer, C., De Rosa, D., Grace, P.R., 2019. Effect of irrigation scheduling on nitrous oxide emissions in intensively managed pastures. Agric. Ecosyst. Environ. 272, 126-134. https://doi.org/10.1016/j.agee.2018.11.011

Nesbit, S.P., Breitenbeck, G.A., 1992. A laboratory study of factors influencing methane uptake by soils. Agric. Ecosyst. Environ. 41, 39-54. https://doi.org/10.1016/0167-8809(92)90178-E

Pan, H., Ying, S., Liu, H., Zeng, L., Zhang, Q., Xu, J., Li, Y., Di, H., Liu, Y., 2018. Microbial pathways for nitrous oxide emissions from sheep urine and dung in a typical steppe grassland. Biol. Fertil. Soils, 54, 717-730. https://doi.org/10.1007/s00374-018-1297-2

Pattey, E., Trzcinski, M.K., Desjardins, R.L., 2005. Quantifying the reduction of greenhouse gas emissions as a result of composting dairy and beef cattle manure. Nutr. Cycl. Agroecosyst. 72, 173-187. https://doi.org/10.1007/s10705-005-1268-5

Pelster, D.E., Chantigny, M.H., Rochette, P., Angers, D.A., Laganière, J., Zebarth, B., Goyer, C., 2013. Crop residue incorporation alters soil nitrous oxide emissions during freeze—thaw cycles. Can. J. Soil Sci. 93, 415-425. https://doi.org/10.1139/CJSS2012-043

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. J. Environ. Qual. 41, 427-435. https://doi.org/10.2134/jeq2011.0261

Pelster, D.E., Chantigny, M.H., Royer, I., Angers, D.A., Vanasse, A., 2021. Reduced tillage increased growing season N_2O emissions from a fine but not a coarse textured soil under the cool, humid climate of eastern Canada. Soil Tillage Res. 206, 104833. https://doi.org/10.1016/j.still.2020.104833

Petersen, S.O., Regina, K., Pöllinger, A., Rigler, E., Valli, L., Yamulki, S., Esala, M., Fabbri, C., Syväsalo, E., Vinther, F.P., 2006. Nitrous oxide emissions from organic and conventional crop rotations in five European countries. Agric. Ecosyst. Environ. 112, 200-206. https://doi.org/10.1016/j.agee.2005.08.021

Peyrard, C., Mary, B., Perrin, P., Véricel, G., Gréhan, E., Justes, E., Léonard, J., 2016. N_2O emissions of low input cropping systems as affected by legume and cover crops use. Agric. Ecosyst. Environ. 224, 145-156. https://doi.org/10.1016/j.agee.2016.03.028

R Core Team, 2012. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/ (accessed 6 January 2022).

Ravishankara, A.R., Daniel, J.S., Portmann, R.W., 2009. Nitrous oxide (N_2O): the dominant ozone-depleting substance emitted in the 21st century. Science 326, 123-125. https://doi.org/10.1126/science.1176985

Robertson, G.P., 1989. Nitrification and denitrification in humid tropical ecosystems: potential controls on nitrogen retention. Miner. Nutr. Trop. For. Savanna Ecosyst. 9, 55-69.

Robertson, G.P., Paul, E.A., Harwood, R.R., 2000. Greenhouse gases in intensive agriculture: Contributions of individual gases to the radiative forcing of the atmosphere. Science 289, 1922-1925.

Robertson, G.P., Vitousek, P.M., 2009. Nitrogen in agriculture: balancing the cost of an essential resource. Annu. Rev. Environ. Resour. 34, 97-125. https://doi.org/10.1146/annurev.environ.032108.105046

Rochette, P., Angers, D.A., Bélanger, G., Chantigny, M.H., Prévost, D., Lévesque, G., 2004a. Emissions of N_2O from alfalfa and soybean crops in eastern Canada. Soil Sci. Soc. Am. J. 68, 493-506. https://doi.org/10.2136/sssaj2004.4930

Rochette, P., Angers, D.A., Chantigny, M.H., Bertrand, N., 2008a. Nitrous oxide emissions respond differently to no-till in a loam and a heavy clay soil. Soil Sci. Soc. Am. J. 72, 1363-1369. https://doi.org/10.2136/sssaj2007.0371

Rochette, P., Angers, D.A., Chantigny, M.H., Bertrand, N., Côté, D., 2004b. Carbon dioxide and nitrous oxide emissions following fall and spring applications of pig slurry to an agricultural soil. Soil Sci. Soc. Am. J. 68, 1410-1420. https://doi.org/10.2136/sssaj2004.1410

Rochette, P., Angers, D.A., Chantigny, M.H., Gagnon, B., Bertrand, N., 2008b. N_2O fluxes in soils of contrasting textures fertilized with liquid and solid dairy cattle manures. Can. J. Soil Sci. 88, 175-187. https://doi.org/10.4141/CJSS06016

Rochette, P., Bertrand, N., 2008. Soil-surface gas emissions. In: Carter, M.R., Gregorich, E.G. (Eds.), Soil sampling and methods of analysis. CRC Press, Boca Raton, FL, pp. 851-861.

Rochette, P., Janzen, H.H., 2005. Towards a revised coefficient for estimating N_2O emissions from legumes. Nutr. Cycl. Agroecosyst. 73, 171-179. https://doi.org/10.1007/s10705-005-0357-9

Rochette, P., Liang, C., Pelster, D., Bergeron, O., Lemke, R., Kroebel, R., MacDonald, D., Yan, W., Flemming, C., 2018. Soil nitrous oxide emissions from agricultural soils in Canada: Exploring relationships with soil, crop and climatic variables. Agric. Ecosyst. Environ. 254, 69-81. https://doi.org/10.1016/j.agee.2017.10.021

Rochette, P., van Bochove, E., Prévost, D., Angers, D.A., Côté, D., Bertrand, N., 2000. Soil carbon and nitrogen dynamics following application of pig slurry for the 19th consecutive year II. Nitrous oxide fluxes and mineral nitrogen. Soil Sci. Soc. Am. J. 64, 1396-1403. https://doi.org/10.2136/sssaj2000.6441396x

Rochette, P., Worth, D.E., Desjardins, R.L., Lemke, R.L., McConkey, B.G., Pennock, D.J., Wagner-Riddle, C., 2008c. Estimation of N_2O emissions from agricultural soils in Canada. I. Development of a country-specific methodology. Can. J. Soil Sci. 88, 641-654. https://doi.org/10.4141/CJSS07025

Sabrekov, A.F., Glagolev, M., Alekseychik, P., Smolentsev, B., Terentieva, I., Krivenok, L., Maksyutov, S., 2016. A process-based model of methane consumption by upland soils. Environ. Res. Lett. 11, 075001. https://doi.org/10.1088/1748-9326/11/7/075001

Saha, S., Minhas, P.S., Choudhary, R., 2018. Monitoring greenhouse gas fluxes in agro-ecosystems. In: Bal, S.K., Mukherjee, J., Choudhury, B.U., Dhawan, A.K. (Eds.), Advances in crop environment interaction. Springer Singapore, Singapore, pp. 25-50.

Saunois, M., R. Stavert, A., Poulter, B., Bousquet, P., G. Canadell, J., B. Jackson, R., A. Raymond, P., J. Dlugokencky, E., Houweling, S., K. Patra, P., Ciais, P., K. Arora, V., Bastviken, D., Bergamaschi, P., R. Blake, D., Brailsford, G., Bruhwiler, L., M. Carlson, K., Carrol, M., Castaldi, S., Chandra, N., Crevoisier, C., M. Crill, P., Covey, K., L. Curry, C., Etiope, G., Frankenberg, C., Gedney, N., I. Hegglin, M., Höglund-Isaksson, L., Hugelius, G., Ishizawa, M., Ito, A., Janssens-Maenhout, G., M. Jensen, K., Joos, F., Kleinen, T., B. Krummel, P., L. Langenfelds, R., G. Laruelle, G., Liu, L., MacHida, T., Maksyutov, S., C. McDonald, K., McNorton, J., A. Miller, P., R. Melton, J., Morino, I., Müller, J., Murguia-Flores, F., Naik, V., Niwa, Y., Noce, S., O'Doherty, S., J. Parker, R., Peng, C., Peng, S., P. Peters, G., Prigent, C., Prinn, R., Ramonet, M., Regnier, P., J. Riley, W., A. Rosentreter, J., Segers, A., J. Simpson, I., Shi, H., J. Smith, S., Paul Steele, L., F. Thornton, B., Tian, H., Tohjima, Y., N. Tubiello, F., Tsuruta, A., Viovy, N., Voulgarakis, A., S. Weber, T., Van Weele, M., R. Van Der Werf, G., F. Weiss, R., Worthy, D., Wunch, D., Yin, Y., Yoshida, Y., Zhang, W., Zhang, Z., Zhao, Y., Zheng, B., Zhu, Q., Zhu, Q., Zhuang, Q., 2020. The global methane budget 2000-2017. Earth Syst. Sci. Data 12, 1561-1623. https://doi.org/10.5194/essd-12-1561-2020

Schlesinger, W.H., Bernhardt, E.S., 2020. Biogeochemistry: An analysis of global change, fourth edition. Academic Press, London.

Serrano-Silva, N., Sarria-GuzmáN, Y., Dendooven, L., Luna-Guido, M., 2014. Methanogenesis and methanotrophy in soil: A review. Pedosphere 24, 291-307. https://doi.org/10.1016/S1002-0160(14)60016-3

Shcherbak, I., Millar, N., Robertson, G.P., 2014. Global metaanalysis of the nonlinear response of soil nitrous oxide (N_2O) emissions to fertilizer nitrogen. Proc. Natl. Acad. Sci. 111, 9199-9204. https://doi.org/10.1073/pnas.1322434111

Skinner, C., Gattinger, A., Krauss, M., Krause, H.-M., Mayer, J., van der Heijden, M.G.A., Mäder, P., 2019. The impact of long-term organic farming on soil-derived greenhouse gas emissions. Sci. Rep. 9, 1-10. https://doi.org/10.1038/s41598-018-38207-w

Skinner, C., Gattinger, A., Muller, A., Mäder, P., Flieβbach, A., Stolze, M., Ruser, R., Niggli, U., 2014. Greenhouse gas fluxes from agricultural soils under organic and non-organic management - a global meta-analysis. Sci. Total Environ. 468-469, 553-563. https://doi.org/10.1016/j.scitotenv.2013.08.098

Smith, P., 2012. Agricultural greenhouse gas mitigation potential globally, in Europe and in the UK: what have we learnt in the last 20 years? Global Change Biol. 18, 35-43. https://doi.org/10.1111/j.1365-2486.2011.02517.x

Soil Classification Working Group, 1998. The Canadian System of Soil Classification, third ed. Agriculture and Agri-Food Canada, Publication 1646.

Statistics Canada, 2017. Growing opportunity through innovation in agriculture. Minister of Industry, Government of Canada.

Szafranek-Nakonieczna, A., Wolińska, A., Zielenkiewicz, U., Kowalczyk, A., Stępniewska, Z., Błaszczyk, M., 2019. Activity and identification of methanotrophic bacteria in arable and no-tillage soils from Lublin Region (Poland). Microb. Ecol. 77, 701-712. https://doi.org/10.1007/s00248-018-1248-3

Ta, T.C., Macdowall, F.D.H., Faris, M.A., 1986. Excretion of nitrogen assimilated from N_2 fixed by nodulated roots of alfalfa (*Medicago sativa*). Can. J. Bot. 64, 2063-2067. https://doi.org/10.1139/b86-270

Thangarajan, R., Bolan, N.S., Tian, G., Naidu, R., Kunhikrishnan, A., 2013. Role of organic amendment application on greenhouse gas emission from soil. Sci. Total Environ. 465, 72-96. https://doi.org/10.1016/j.scitotenv.2013.01.031

Thilakarathna, M.S., McElroy, M.S., Chapagain, T., Papadopoulos, Y.A., Raizada, M.N., 2016. Belowground nitrogen transfer from legumes to non-legumes under managed herbaceous cropping systems. A review. Agron. Sustain. Dev. 36, 1-16. https://doi.org/10.1007/s13593-016-0396-4

Tian, H., Xu, R., Canadell, J.G., Thompson, R.L., Winiwarter, W., Suntharalingam, P., Davidson, E.A., Ciais, P., Jackson, R.B., Janssens-Maenhout, G., Prather, M.J., Regnier, P., Pan, N., Pan, S., Peters, G.P., Shi, H., Tubiello, F.N., Zaehle, S., Zhou, F., Arneth, A., Battaglia, G., Berthet, S., Bopp, L., Bouwman, A.F., Buitenhuis, E.T., Chang, J., Chipperfield, M.P., Dangal, S.R.S., Dlugokencky, E., Elkins, J.W., Eyre, B.D., Fu, B., Hall, B., Ito, A., Joos, F., Krummel, P.B., Landolfi, A., Laruelle, G.G., Lauerwald, R., Li, W., Lienert, S., Maavara, T., MacLeod, M., Millet, D.B., Olin, S., Patra, P.K., Prinn, R.G., Raymond, P.A., Ruiz, D.J., van der Werf, G.R., Vuichard, N., Wang, J., Weiss, R.F., Wells, K.C., Wilson, C., Yang, J., Yao, Y., 2020. A comprehensive quantification of global nitrous oxide sources and sinks. Nature 586, 248-256. https://doi.org/10.1038/s41586-020-2780-0

Trenberth, K.E., 2011. Changes in precipitation with climate change. Clim. Res. 47, 123-138. https://doi.org/10.3354/cr00953

USDA, 1999. Soil taxonomy : A basic system of soil classification for making and interpreting soil surveys. United States Department of Agriculture, Washington, DC.

USDA, 2020. 2017 Census of agriculture: 2019 Organic survey. United States Department of Agriculture, National Agricultural Statistics Service.

Van Eerd, L., Congreves, K., Arcand, M., Lawley, Y., Halde, C., 2021. Soil health and management. In: Krzic, M., Walley, F.L., Diochon, A., Paré, M.C., Farell, R.E. (Eds.), Digging into Canadian soils: An introduction to soil science. Canadian Society of Soil Science, Pinawa, MB, pp. 463-517.

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. Eur. J. Soil Sci. 61, 903-913. https://doi.org/10.1111/j.1365-2389.2009.01217.x

Van Kessel, C., Venterea, R., Six, J., Adviento-Borbe, M.A., Linquist, B., Groenigen, K.J., 2013. Climate, duration, and N placement determine N_2O emissions in reduced tillage systems: a meta-analysis. Global Change Biol. 19, 33-44. https://doi.org/10.1111/j.1365-2486.2012.02779.x

Vance, C.P., Heichel, G.H., Barnes, D.K., Bryan, J.W., Johnson, L.E., 1979. Nitrogen fixation, nodule development, and vegetative regrowth of alfalfa (*Medicago sativa* L.) following harvest. Plant Physiol. 64, 1-8. https://doi.org/10.1104/pp.64.1.1

Venterea, R.T., Petersen, S.O., de Klein, C.A.M., Pedersen, A.R., Noble, A.D.L., Rees, R.M., Gamble, J.D., Parkin, T.B., 2020. Global Research Alliance N_2O chamber methodology guidelines: Flux calculations. J. Environ. Qual. 49, 1141-1155. https://doi.org/10.1002/jeq2.20118

Wagner-Riddle, C., Congreves, K.A., Abalos Rodriguez, D., Berg, A.A., Brown, S.E., Ambadan, J.T., Gao, X., Tenuta, M., 2017. Globally important nitrous oxide emissions from croplands induced by freeze-thaw cycles. Nature Geoscience 10, 279-283. https://doi.org/10.1038/NGEO2907

Wagner-Riddle, C., Thurtell, G., Kidd, G., Beauchamp, E., Sweetman, R., 1997. Estimates of nitrous oxide emissions from agricultural fields over 28 months. Can. J. Soil Sci. 77, 135-144. https://doi.org/10.4141/S96-103

Wagner-Riddle, C., Thurtell, G.W., 1998. Nitrous oxide emissions from agricultural fields during winter and spring thaw as affected by management practices. Nutr. Cycl. Agroecosyst. 52, 151-163. https://doi.org/10.1023/A:1009788411566

Wagner-Riddle, C., Furon, A., Mclaughlin, N.L., Lee, I., Barbeau, J., Jayasundara, S., Parkin, G., Von Bertoldi, P., Warland, J., 2007. Intensive measurement of nitrous oxide emissions from a corn–soybean–wheat rotation under two contrasting management systems over 5 years. Global Change Biol. 13, 1722-1736. https://doi.org/0.1111/j.1365-2486.2007.01388.x

Wang, F.L., Bettany, J.R., 1993. Influence of freeze-thaw and flooding on the loss of soluble organic carbon and carbon dioxide from soil. J. Environ. Qual. 22, 709-714. https://doi.org/10.2134/jeq1993.00472425002200040011x

Wang, F.L., Bettany, J.R., 1995. Methane emission from a usually well-drained prairie soil after snowmelt and precipitation. Can. J. Soil Sci. 75, 239-241. https://doi.org/10.4141/cjss95-033

Westphal, M., Tenuta, M., Entz, M.H., 2018. Nitrous oxide emissions with organic crop production depends on fall soil moisture. Agric. Ecosyst. Environ. 254, 41-49. https://doi.org/10.1016/j.agee.2017.11.005

Willer, H., Schaack, D., Lernoud, J., 2019. Organic farming and market development in Europe and the European Union. The world of organic agriculture. Statistics and emerging trends 2019. Research Institute of Organic Agriculture FiBL and IFOAM - Organics International, pp. 217-254.

Wrage, N., Velthof, G.L., Laanbroek, H.J., Oenema, O., 2004. Nitrous oxide production in grassland soils: assessing the contribution of nitrifier denitrification. Soil Biol. Biochem. 36, 229-236. https://doi.org/10.1016/j.soilbio.2003.09.009

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 Biol. Biochem. 123, A3-A16. https://doi.org/10.1016/j.soilbio.2018.03.020

Yao, Z., Liu, C., Zheng, X., Butterbach-Bahl, K., Pelster, D.E., 2020. Soil N intensity as a measure to estimate annual N_2O and NO fluxes from natural and managed ecosystems. Curr. Opin. Environ. Sustain. 47, 1-6. https://doi.org/10.1016/j.cosust.2020.03.008

Zak, D.R., Holmes, W.E., MacDonald, N.W., Pregitzer, K.S., 1999. Soil temperature, matric potential, and the kinetics of microbial respiration and nitrogen mineralization. Soil Sci. Soc. Am. J. 63, 575-584. https://doi.org/10.2136/sssaj1999.03615995006300030021x

Zebarth, B.J., Rochette, P., Burton, D.L., 2008a. N_2O emissions from spring barley production as influenced by fertilizer nitrogen rate. Can. J. Soil Sci. 88, 197-205. https://doi.org/10.4141/CJSS06006

Zebarth, B.J., Rochette, P., Burton, D.L., Price, M., 2008b. Effect of fertilizer nitrogen management on N_2O emissions in commercial corn fields. Can. J. Soil Sci. 88, 189-195. https://doi.org/10.4141/CJSS06010

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. Glob Chang Biol 23, 4068-4083. https://doi.org/10.1111/gcb.13648

Zúrovec, O., Sitaula, B.K., Cústović, H., Zúrovec, J., Dörsch, P., 2017. Effects of tillage practice on soil structure, N_2O emissions and economics in cereal production under current socio-economic conditions in central Bosnia and Herzegovina. PloS one 12, e0187681. https://doi.org/10.1371/journal.pone.0187681

Appendix A. Carbon dioxide fluxes

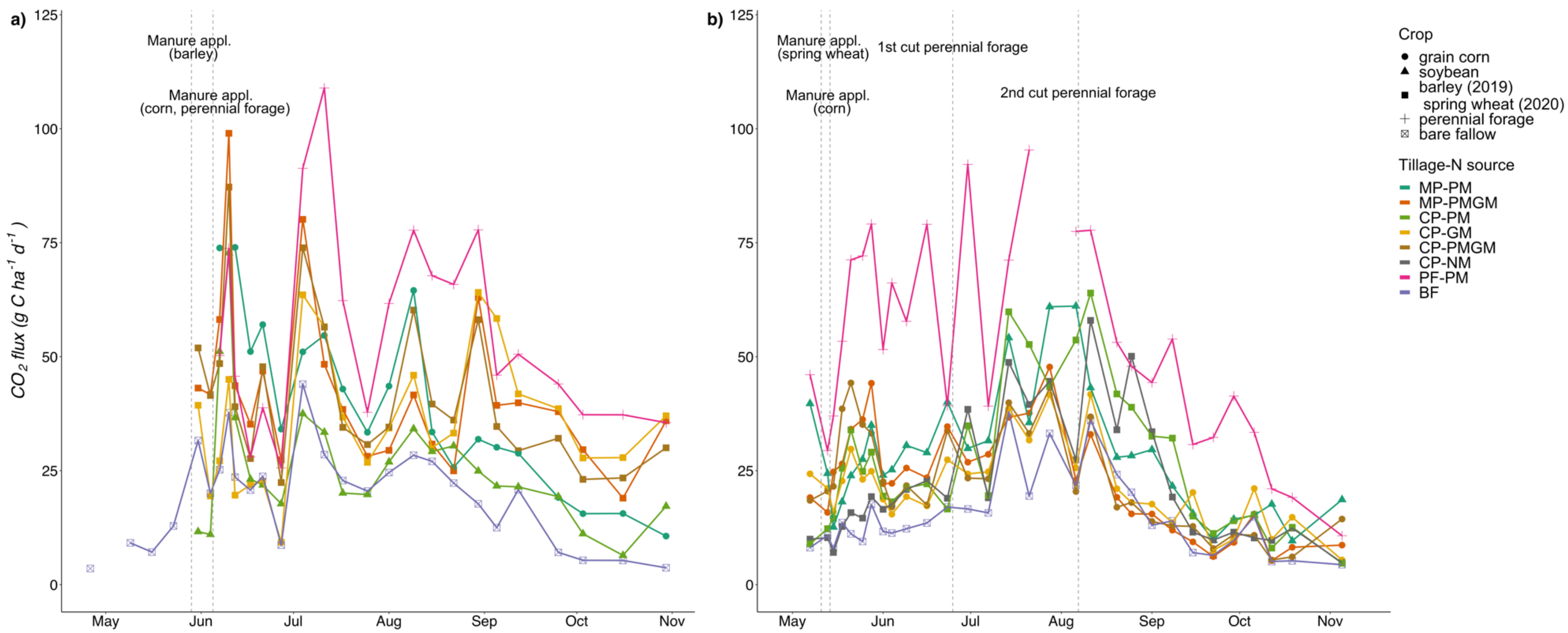


Fig. S 1. Carbon dioxide (CO_2) daily fluxes over time for different cropping systems in a) 2019 and b) 2020. Dotted lines indicate poultry manure applications and perennial forage cuts. Tillage: moldboard plough (MP), chisel plough (CP); Fertilization: poultry manure (PM), fall-seeded pea green manure (GM), poultry manure and fall-seeded pea green manure (PMGM), no poultry manure nor green manure (NM); Crop sequence: perennial forage (PF), bare fallow (BF).

Appendix B. Aerial photo of the field experiment

Fig. S 2. Aerial photo of the field experiment at Institut national d’agriculture biologique, Victoriaville, QC, Canada, on 21 July 2020. Photo credit: CETAB+.

Appendix C. Experimental design

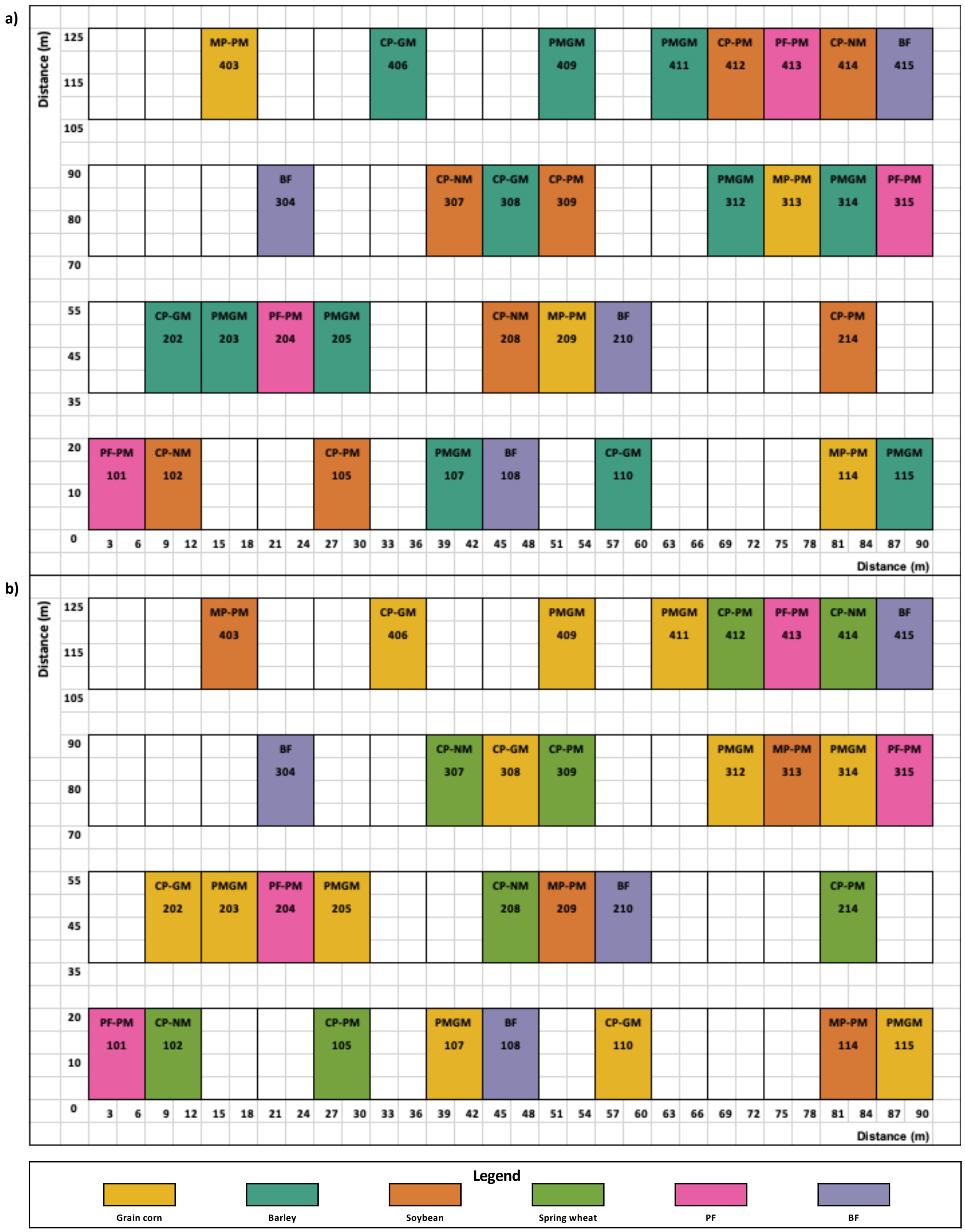


Fig. S 3. Experimental design in a two-year study, showing the different crops in the cropping systems in a) 2019 and b) 2020. Tillage: moldboard plough (MP), chisel plough (CP); Fertilization: poultry manure (PM), fall-seeded pea green manure (GM), poultry manure and fall-seeded pea green manure (PMGM), no poultry manure nor green manure (NM); Crop sequence: perennial forage (PF), bare fallow (BF).

Appendix D. Soil analysis

Table S 3. Soil nutrient Mehlich 3 analysis from AgroEnviroLab in 2018, 2019, and 2020 and bulk density from field measurements in 2020.

	Arithmetic mean of the four blocks[1]		
	2018	2019	2020
pH (water 1:1)	6.1	6.2 ± 0.1	6.8 ± 0.2
Buffer pH	6.6	6.7 ± 0.1	7.0 ± 0.1
Organic matter (%)	4.5	4.2 ± 0.7	4.2 ± 0.7
P (kg P ha^{-1})	223	183 ± 33.6	185 ± 39
K (kg K ha^{-1})	102	70 ± 8.1	92 ± 19
CEC (meq 100 g^{-1})	16.3	13.7 ± 1.4	12.5 ± 1.4
Granulometry (%)			
Sand	77.0	-	-
Silt	13.3	-	-
Clay	9.7	-	-
Bulk density[2] (g cm^{-3})			
All cropping systems except perennial forage	-	-	1.41 ± 0.13
Perennial forage	-	-	1.59 ± 0.09

[1] Soil sampling with a soil probe at a 0-20 cm depth in September 2018, May 2019, and April 2020. One composite sample for each block (4 blocks) in 2018. One composite sample for each experimental unit of the long-term experiment (15 experimental units per block) in 2019 and 2020.

[2] Bulk density with the cylinder method: average of five field measurements in each experimental units in 2020 (6 May, 9 June, 13 August, 16 September, and 22 October). Soil sampling at a 5-10 cm depth; Cylinder length: 5.0 cm; cylinder diameter: 6.3 cm.