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# Combining reduced tillage and green manures minimized $N_2O$ emissions from organic cropping systems in a cool humid climate

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ABSTRACT

Developing and implementing improved management practices are necessary to enhance the sustainability of organic cropping systems. This study examined the effects of various organic cropping systems on soil greenhouse gas (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) nitrogen (N) sources (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 40–70 % lower and yields that were 33–51 % lower than the MP-PMGM and CP-PMGM systems, which showed equivalent $N_2O$ emissions and yields. Yield-scaled $N_2O$ emissions were equivalent for 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.

## 1. 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 (CGSB, 2020; Codex Alimentarius, 1999). Greenhouse gas (GHG) emissions from organic cropping systems are often estimated using calculation schemes (*e.g.* emission factors, EFs) 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 produced mainly as a by-product of nitrification or as 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,

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soil aeration and water saturation, and thus they will affect $N_2O$ formation in the soil and its transfer to the atmosphere. However, the dynamics of $N_2O$ formation in organic cropping systems combining different cropping practices than conventional cropping systems are still not well understood.

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., 2017; 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 can vary with soil texture, drainage conditions, as well as 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 $N_2O$. 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 (Basche et al., 2014).

Diversified and long crop rotations in organic cropping systems are designed to retain and recycle nutrients, which can affect $N_2O$ emissions. Variable N application rates for the different crops in rotation may 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 in rotation with soybean (*Glycine max* [L.] Merr.) and winter wheat (*Triticum aestivum* L.) 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, EFs (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 EF of an organic N source varies with growing season precipitation, potential evapotranspiration, and soil texture (Rochette et al., 2018). In complex organic crop rotations however, the EF for the N applied to individual crops is more difficult to calculate as N is typically reallocated within the crop rotation (Skinner et al., 2019). However, EFs of complete organic crop rotations may be compared (Brozyna et al., 2013).

Temporary perennial forages (PF) are generally more present in organic than conventional crop rotations (Barbieri et al., 2017) and thus influence the $N_2O$ losses of the 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 (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 $CO_2$ 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.

## 2. Materials and methods

### 2.1. 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 135 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, 2021). 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 $g^{-1}$). 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.

### 2.2. Experimental design

The experimental units were arranged in a randomized complete block design with 4 replications. 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 was interseeded with red clover (*Trifolium pratense* L.) following mechanical weeding. 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 × 20 m in size.

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

| | 2019 | | | | 2020 | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Cropping system | Crop | Timing of primary tillage | Timing of PM application | N input from PM (kg N ha$^{-1}$) | Crop | Timing of primary tillage | Timing of PM application | N input from PM (kg N ha$^{-1}$) | N input from GM (kg N ha$^{-1}$) | 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.

### 2.3. Field management

During 2018, the year prior to this study, a triticale (× *Triticosecale* Wittm. var. Pronghorn) and pea (var. not stated) GM was grown in all cropping systems. This GM was incorporated in spring 2019 with primary tillage in MP cropping systems, and with shallow cultivation (rotary tiller) 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 shallow cultivation. 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. The BF system was maintained bare using a disk-tiller (10-cm depth) or a S-tine harrow (5-cm depth) 4 or 5 times over the growing season. Field management details are presented in Table S 1.

In spring 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 spring 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 and 6.2 Mg ha$^{-1}$ in grain corn to account for the N supplied by previous GM, which differed among cropping systems. Immediately following application, the fertilizer was incorporated (to 10 cm depth) using a S-tine harrow (2019) or a disk tiller (2020). The amount of N applied from PM and/or GM are detailed in Table 1. Potassium sulfate (0−0−50) was applied by hand at locally recommended rates for each crop. Fertilizer application rates were determined using the regional recommendations for each crop. The amount of N returned to soil with GM residues was estimated by sampling GM aboveground biomasses and analyzing N content. Aboveground biomass N content of the GM in fall 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 86,500 seed ha$^{-1}$ in both years. The PF was established in the spring of 2019 with a 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.

### 2.4. Data collection and analysis

#### 2.4.1. 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$^{-3}$).

#### 2.4.2. 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 4–6 weeks after manure application, then every second week. Within 24 h of soil collection, soil mineral N ($NO_3^-$-N and $NH_4^+$-N) 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 min 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 $NH4^{+}$-N and $NO_3^-$-N concentrations over the complete length of the experiment (Burton et al., 2008) to assess its capacity to predict seasonal $N_2O$ emissions.

### 2.4.3. 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 × 0.55 m × 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 × 0.75 m × 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 min 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 20 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. The GWP variable represents the sum of cumulative $N_2O$ and $CH_4$ emissions of a cropping system in kg $CO_{2\ eq}$ $ha^{-1}$. The GWP values used for calculation were 265 kg $CO_{2\ eq}$ for each kg $N_2O$-N and 28 kg $CO_{2\ eq}$ for each kg $CH_4$-C. The total $CO_2$ flux was not accounted for in the GWP as we measured only soil fluxes which excludes all the plant-atmosphere $CO_2$ exchanges. Typically $CO_2$ fluxes are estimated using changes in soil C over a much longer time frame. The annual EF was calculated as the difference between the total cumulative $N_2O$ emissions in a plot over two years and the average $N_2O$ emissions from unfertilized crops (barley CP-GM in 2019 and wheat CP-NM in 2020), divided by the total amount of N applied over two years divided by two. The average $N_2O$ emissions from these unfertilized crops (0.57 kg N $ha^{-1}$) are similar to $N_2O$ emissions measured in unfertilized controls from similar soils in the region (Chantigny et al., 2010; Pelster et al., 2021).

### 2.4.4. Plant sampling and analysis

Crop yields were measured by harvesting central rows to avoid the edge effects. Corn and soybean yield was obtained by harvesting four rows each side of the two middle rows of an experimental unit, whereas cereals were harvested in 14 rows in the central zone. 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 an 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 and N 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.

## 2.5. 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. Results

## 3.1. Environmental conditions and greenhouse gas daily fluxes

### 3.1.1. 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. 1). 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. 1). Distribution of monthly precipitation among the two years was generally consistent with long-term normal except in April and July 2019,

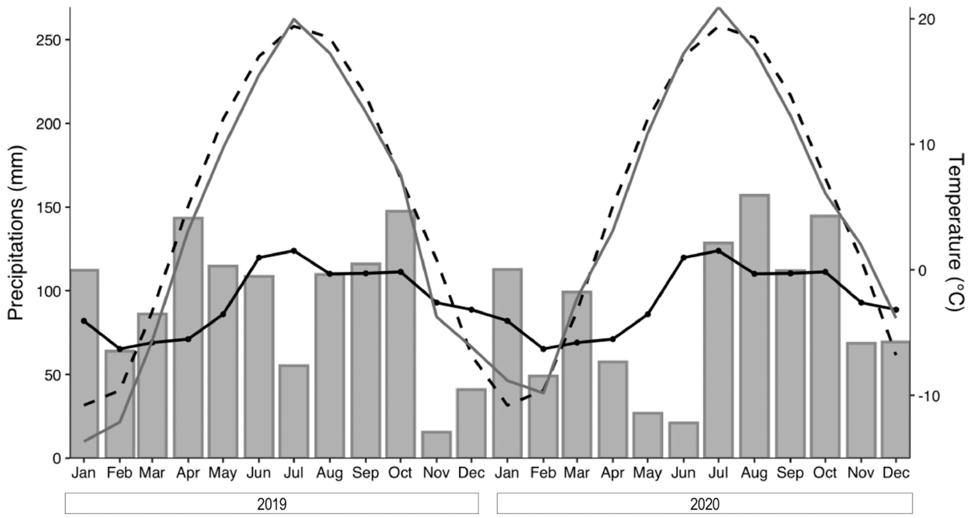



**Fig. 1.** Monthly precipitations (grey bars) and monthly mean air temperature (grey line) during the two-year study (2019 and 2020) compared with long-term normal (1981–2010) monthly precipitation (black line with dots) and monthly mean air temperature (black dashed line).

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.

### 3.1.2. Soil conditions

Soil $NO_3^-$-N and $NH_4^+$-N 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. 2a-d). For the two years, $NO_3^-$-N concentrations were higher than $NH_4^+$-N concentrations in cropping systems with PM and/or GM. In all cropping systems with PM, $NH_4^+$-N 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. 2a, c). In 2019, soil $NO_3^-$-N 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 $NO_3^-$-N 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. 2b). In

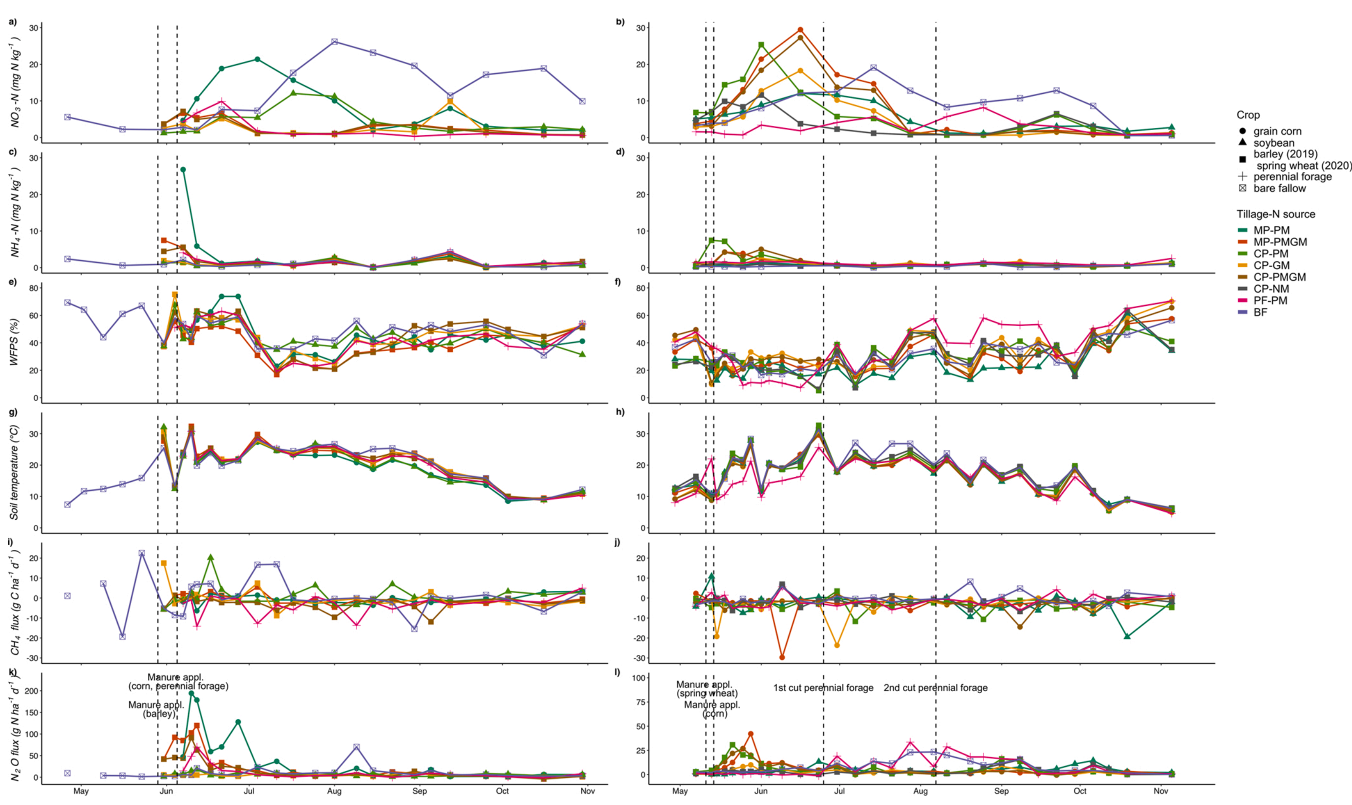



**Fig. 2.** Soil nitrate ($NO_3$-N) concentrations, ammonium ($NH_4^+$-N) concentrations, water-filled pore space (WFPS), soil temperature, methane ($CH_4$) daily fluxes, and nitrous oxide ($N_2O$) daily fluxes over time for different cropping systems in 2019 (left) and 2020 (right). Dotted lines indicate 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).

the CP-GM cropping system (i.e., with pea incorporated in the spring of 2020), $NO_3^-$-N 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 the CP-NM cropping system, while nitrate 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. 2e–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 all cropping systems during May and June, following PM application. In 2020, PF-PM tended to show the lowest WFPS before 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. 2g-h) and were similar among the various cropping systems, except PF where soil temperature was 5–10 °C cooler than other cropping systems before the first forage cut in June 2020. Soil temperature in BF plots was generally about 5 °C warmer than in the other cropping systems in July 2020.

### 3.1.3. 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 $NO_3^-$-N concentrations were above 5 mg $NO_3^-$-N $kg^{-1}$ soil and WFPS higher than 50 % (Fig. 3). 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 $NH_4^+$-N 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 $NO_3^-$-N concentrations (Fig. 2a-f, k-l). 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 tended to be 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. 2i-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.

### 3.2. 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 3–7 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 2–3 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. In annual crops, the cumulative 2-year $N_2O$ emissions were highest in the cropping system with the greatest N input in 2019 (MP-PM) and lowest in the CP-GM cropping system, which had the lowest N input overall (Fig. 4a). No difference was detected among cropping systems for cumulative $CH_4$ emissions in each year or for the 2-year cumulative emissions (Table 2, Fig. 4b). 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 2–3 times greater than GWP of cropping systems with no N input. The two-year cumulative for GWP was consistent with the 2-year cumulative $N_2O$ emissions. (Fig. 4a, c).

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

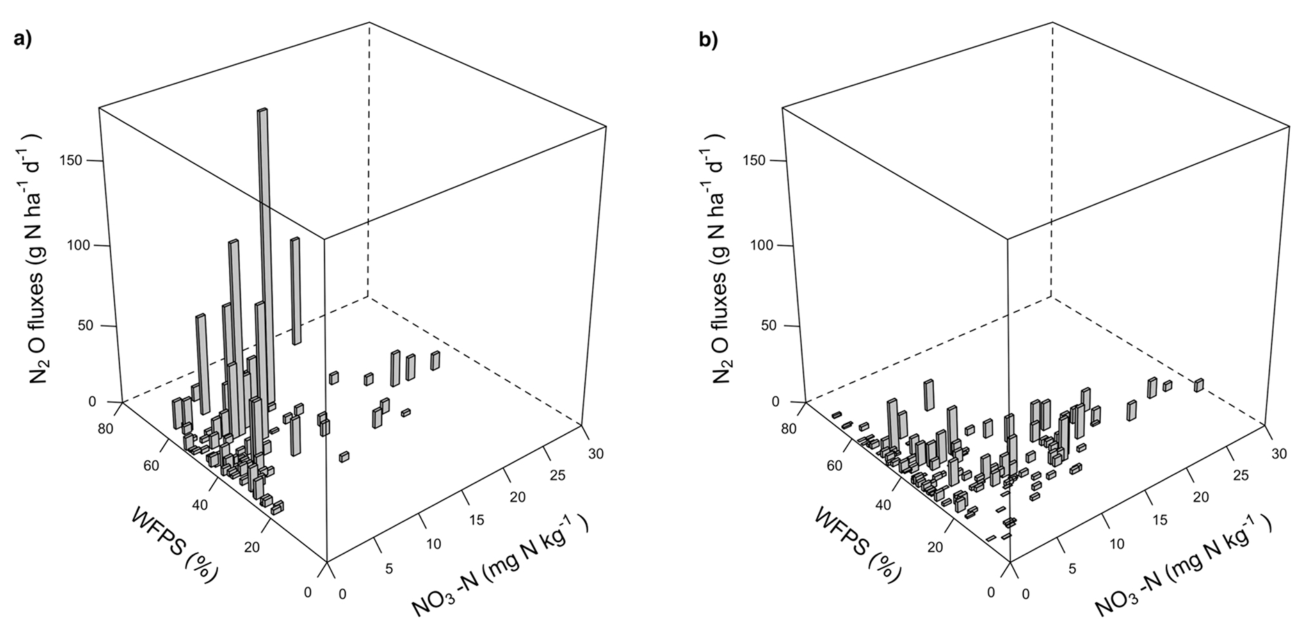



**Fig. 3.** Nitrous oxide ($N_2O$) daily fluxes as a function of soil nitrate ($NO_3^-$-N) concentrations and water-filled pore space (WFPS) in various organic cropping systems in a) 2019 and b) 2020.

**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) | | $N_2O$ emissions (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 | ab | -0.61 | ab | 236 | ab | crop failure | crop failure | | crop failure | |
| CP-GM | grain corn | 1.010 | bc | 0.47 | c | -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 | bc | -0.31 | ab | 152 | bc | crop failure | crop failure | | crop failure | |
| PF-PM[3] | perennial forage | 0.784 | c | 1.38 | a | -0.33 | ab | 357 | a | 6.65 | 0.21 | ab | – | |
| BF | bare fallow | 1.775 | a | 1.44 | a | -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.
[4] The GWP values used for calculation were 265 kg $CO_{2\ eq}$ for each kg $N_2O$-N and 28 kg $CO_{2\ eq}$ for each kg $CH_4$-C.
Different lowercase letters indicate significant differences between cropping systems within the experimental year with a *P*-value < 0.05.

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 years ($P < 0.05$), with a pseudo $R^2$ of 0.2807 (Fig. 5). A large variation was observed in the $N_2O$ EF between the different cropping systems (−0.08 % to 1.14 %, Fig. 4d) with PF showing the highest EF. Emission factors were equivalent between CP-PM, MP-PMGM, CP-PMGM and CP-GM cropping systems which were lower than MP-PM and PF-PM cropping systems.

### 3.3. 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). However, 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).

## 4. Discussion

### 4.1. Nitrous oxide emissions and driving environmental factors

Organic cropping systems with reduced tillage (CP) and GM as the N source showed the lowest cumulative $N_2O$ 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 (Fig. 2a-d). In manure-based systems, the timing of PM application at planting resulted in the rapid release of available N concentrations while the crop root system was not fully developed, preventing the crop from efficiently taking 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, 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 a past study (Gregorich et al., 2005), cumulative $N_2O$ emissions in our study tended to be lower in the unfertilized annual crop than the fertilized grain crops in 2019, but were similar among fertilized and unfertilized crops in 2020, as particularly dry conditions occurred at time of N application (May-June). Dry conditions favored low $N_2O$ emissions despite N availability (Fig. 2) in the fertilized crops, likely because dry conditions can limit nitrification (Maag and Vinther, 1996) and aerobic conditions

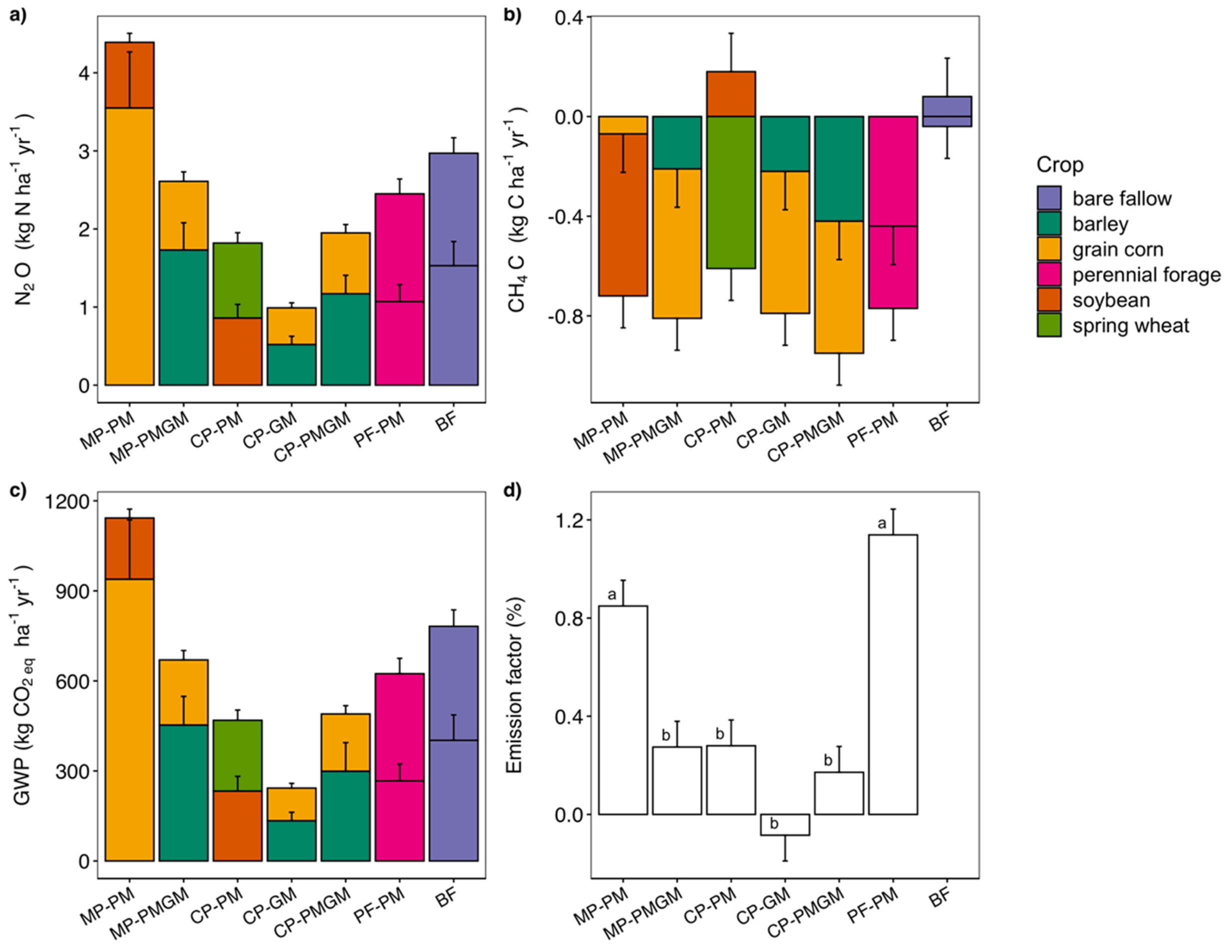


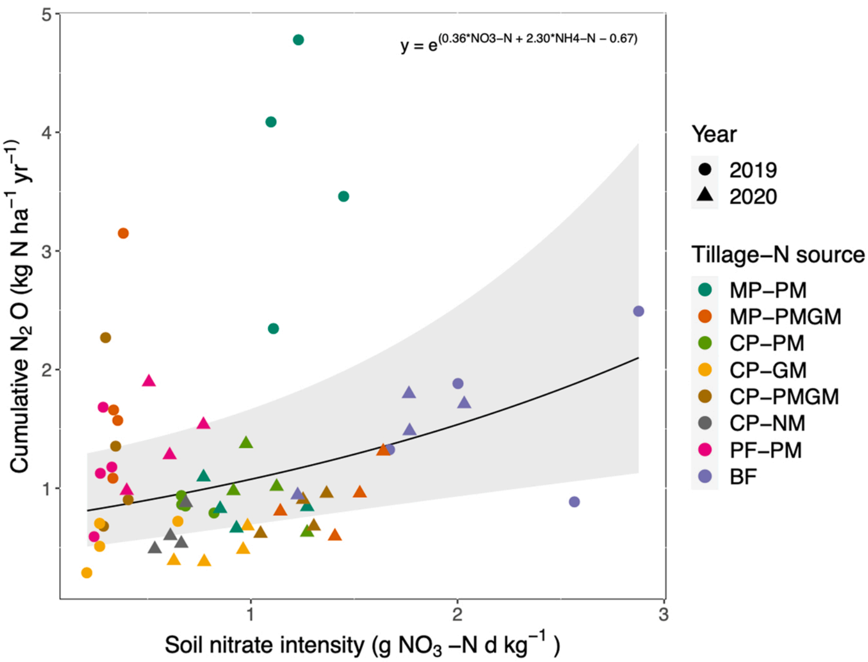

**Fig. 4.** Cumulative nitrous oxide ($N_2O$) emissions, cumulative methane ($CH_4$) emissions, global warming potential (GWP), and emission factors (EFs) over the two years of the study. a, b, c) Stacked bars show the values for 2019 (bottom) and 2020 (top); d) Bars show the EFs of the cropping systems over two years. 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). The GWP values used for calculation were 265 kg $CO_{2\ eq}$ for each kg $N_2O$-N and 28 kg $CO_{2\ eq}$ for each kg $CH_4$-C. Error bars: standard error. Different lowercase letters indicate significant differences between cropping systems over the two years of the study with a P-value < 0.05.

inhibits denitrification in soils (Davidson et al., 2000). This is also shown in Fig. 3, where the highest emissions over the two years occurred when WFPS exceeded 50 % in conjunction with soil $NO_3^-$-N above 5 mg kg$^{-1}$ and where emissions were consistently low when WFPS was less than 30 %, regardless of soil $NO_3^-$-N concentration.

Peak $N_2O$ fluxes generally occurred shortly after PM application and were related to increased $NH_4^+$-N and $NO_3^-$-N 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 % and 80 % (Fig. 2a, c, e, k), 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 why $N_2O$ emissions were limited despite high $NO_3^-$-N concentrations (Fig. 2b, f, l). In accordance with Davidson et al. (2000), the highest $N_2O$ emissions occurred when inorganic N concentrations were high and soil water content between 55 % and 65 % WFPS (Fig. 3). At higher water content, $N_2O$ emissions from soil decrease because gas diffusivity is reduced, increasing rates of $N_2O$ reduction to $N_2$ (Balaine et al., 2016).

Both nitrification and denitrification processes were likely involved in the $N_2O$ emissions from the manure-based systems. The contribution of nitrification to GM-based system $N_2O$ emissions may have been more limited by $NH_4^+$ availability, as indicated by the lower $NH_4^+$-N concentrations in the spring than manure-based systems. The first $N_2O$ peak observed in manure-based systems occurred simultaneously with a peak in $NH_4^+$-N 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 was rapidly hydrolyzed to $NH_4^+$ in soil, as reflected by the rapid rise in $NH_4{}^+$-N 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 $NO_3^-$-N concentrations, which could have been denitrified when the WFPS increased above 60 %, resulting in the second $N_2O$ peak as denitrification tends to be the predominant source of $N_2O$ when WFPS exceeds 60 % (Bateman and Baggs, 2005). 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., 2018; Mumford et al., 2019). Similar patterns of two consecutive $N_2O$ peaks related to a rise in $NH_4^+$-N and then to $NO_3^-$-N 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 $NH_4^+$-N concentration




| | 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. 5.** 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.

**Table 3**
Grain yields and yield-scaled nitrous oxide ($N_2O$) emissions (2019 and 2020) comparisons with simple contrasts for three organic cropping systems with the same crop for each year.

| 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.

on $N_2O$ emissions found from our simple linear regression analyses (Fig. 5) and the fluctuating aerobic-anaerobic conditions observed during the study (Fig. 2e), suggest that nitrifier denitrification or coupled nitrification-denitrification may have been the prevalent pathway for $N_2O$ emissions (Butterbach-Bahl et al., 2013; Wrage-Mönnig et al., 2018). Also, a laboratory experiment found that manure addition to soil was associated with increased soil $NO_2$ concentrations, which is the substrate for nitrifier denitrification that tends 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.

The GWP was lowest in the GM-based or unfertilized cropping systems due to their lower $N_2O$ emissions, which is consistent with a previous study comparing organic and conventional cropping systems (Biernat et al., 2020). Differences in the 2-year cumulative $N_2O$ emissions and GWP between the different cropping systems may be related to soil N availability and crop N needs under the influence of environmental conditions. It is clear from Fig. 2a, that the poultry manure applied to grain corn in 2019 largely contributed to the high $N_2O$ emissions and GWP observed in MP-PM. Thus, even though the MP-PMGM and CP-PMGM had higher N inputs than the MP-PM, the dry conditions in spring 2020, when the poultry manure was applied to the MP-PMGM and CP-PMGM, likely caused their lower GWP. These results are consistent with Rochette et al. (2018) who reported that growing season precipitation is a key influence on $N_2O$ EFs.

The highest EF found in our legume-based PF system (1.1 %) was similar to the mean EF of fertilized non-legume PF systems (1.2 %) reported in Gregorich et al. (2005) study. The EFs in our organic cropping systems with annual crops (−0.08 % to 0.85 %) were also consistent with the Charles et al. (2017) meta-analysis, who reported a mean EF of

**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., 2021, 2012; 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, 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., 2004; Wagner-Riddle et al., 1997; Wagner-Riddle et al., 2007) |
| Perennial forage | 1.1 – 1.4 | 0.2 – 1.5 | (Chantigny et al., 2007; Rochette et al., 2004; Wagner-Riddle et al., 1997) |

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

0.97 % for solid manure (median 0.24 %) and a weighted mean EF of 0.57 % for all organic sources. However, our manure-based cropping systems were lower than the EF related to animal manure application in Zhou et al. (2017) meta-analysis for cool temperate zones (1.95 % ± 0.23 %). The EFs of our manure-based cropping systems with high N inputs were equivalent to the EF of the GM-based system with a lower N input, which is consistent with the Brozyna et al. (2013) study, who reported an EF of 0.60 % for their manure-based and GM-based cropping systems over a four-year organic crop rotation (spring barley, grass-clover, potato and winter wheat). 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 large variation in EF of single crops generally observed in organic cropping systems (Skinner et al., 2014). Only the first two years of our cropping system rotations were assessed and the difference seen in the EFs of our cropping systems could be attenuated after a complete crop rotation, as the N applied might be reallocated within the crop rotation. Moreover, in our study, background emissions were estimated from two unfertilized non-legume crops and were used in the EF calculation of all cropping systems. The EF of CP-GM cropping system resulted in a negative value, very close to zero, indicating that its $N_2O$ emissions were only related to background emissions.

### 4.2. *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 40–70 % reduction in $N_2O$ emissions was enough to compensate for the 35–50 % 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 the corn and barley CP-GM cropping system and for spring wheat (FADQ, 2021a; b). Our organic PMGM cropping systems had similar yields compared to regional averages, even though a yield gap of 7–26 % with conventional cereal yields is typically reported (Wilbois and Schmidt, 2019). According to Seufert et al. (2012), the yield gap can be up to 40 % for wheat, which is higher than for other crops, but still less than the spring wheat (CP-PM and CP-NM) in this study that experienced crop failure due to weeds pressure. As seen in Schrama et al. (2018) though, the yield gap between organic and conventional cropping systems may be reduced after about ten years, likely as a result of a higher nutrient efficiency and improved soil structure (higher organic matter concentrations and soil aggregation), so it is feasible that yields at our study site may increase as well.

In our recently implemented organic cropping systems, reduced tillage and conventional tillage generated equivalent yields, which is consistent with previous meta-analyses (Cooper et al., 2016; Van Kessel et al., 2013), although reduced tillage was found to result in lower yields under a humid climate after ten years (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). However, reducing $N_2O$ emissions through the use of organic cropping systems that also reduce yields implies that more land would be needed to grow sufficient food. This would likely cause changes to land use and the loss of existing natural areas that could cause negative impacts on regional GHG emissions, as well as other detrimental environmental effects. The concept of either using land for food production or ecosystem services or combining the two within a single land use is discussed in depth by Baudron and Giller (2014) who conceptualize this balance as either "land sharing and land sparing". In the context of climate change it is likely that both conceptual approaches will be needed, and cropping systems that offer more ecosystems services will need to be balanced against those that focus more on production, depending on the location (Wilbois and Schmidt, 2019).

Annual legume crops lessened the $N_2O$ emitted per unit N produced compared to annual non-legume crops, 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 required 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 strategy to improve agroecosystem sustainability while minimizing the $N_2O$-N emitted per N harvested.

### 4.3. 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 also equivalent to soybean $N_2O$ emissions in both years, consistent with Rochette et al. (2004) where $N_2O$ emissions from alfalfa were equal to or greater than for soybean on a loamy soil. 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 cumulative $N_2O$ emissions (Fig. 4). Increased available $NO_3^-$-N concentrations observed in BF in July and August in both years were likely related to enhanced N mineralization following shallow cultivation for weed control. However, these high $NO_3^-$-N 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 %. The high $NO_3^-$-N concentrations found in PF plots in spring 2019 meanwhile, 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, which resulted in emissions of 70 g $N_2O$-N $ha^{-1}$ $d^{-1}$ one week after manure application. 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 cumulative emissions during the second year but were likely due to different mechanisms. In the first year, the higher emissions occurred early in the spring, shortly after fertilization while the crop was still small, whereas in the second year, the emission peaks occurred later in the year, immediately following the first and second cuts. Thus, in 2019 the high $NO_3^-$-N immediately following fertilizer application, combined with low N uptake combined with high spring moisture caused the peak emissions, while in 2020, the peak $N_2O$ daily fluxes in PF after the forage cuts (Fig. 2) were likely due to decomposition of organic material released from the cutting of the grasses, resulting in increased $NO_3^-$-N availability in combination with WFPS greater than 55 % (Chantigny et al., 2013; Linn and Doran, 1984; Rochette et al., 2004). The increased $N_2O$ daily fluxes following the two forage cuts in 2020 were similar to a past study (Rochette et al., 2004). 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 intensified $N_2O$ emissions. Furthermore, increased N availability from nodule 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 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).

## 5. 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 on a sandy loam soil. Overall, the dry conditions in spring 2020 induced low cumulative $N_2O$ emissions in fertilized cropping systems despite high available $NO_3^-$-N concentrations. Conventional and reduced tillage intensities resulted in equivalent area based and yield-scaled $N_2O$ emissions. For similar yield-scaled $N_2O$ emissions, the manure-based cropping systems generated greater crop yields than the GM-based cropping system, and thus, would allow more land to be spared for other ecosystem services unless yields in GM-based cropping system improve over time. 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 $CH_4$, with similar cumulative emissions between cropping systems.

Thus, on a sandy loam soil, in a cool temperate climate, GM could be included in crop rotation to reduce manure application rates in the spring, and related $N_2O$ emissions. However, care should be taken when determining the proportion of leguminous species in their PF species mixture, although 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 also clarify whether the adoption of GM-based cropping systems could be profitable for farmers despite the lower yields, particularly in the first years following organic transition.

## Declaration of Competing Interest

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## Data Availability

Data will be made available on request.

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## Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2022.108205.

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