N$_2$O Emissions From Two Agroecosystems: High Spatial Variability and Long Pulses Observed Using Static Chambers and the Flux-Gradient Technique

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Abstract With the addition of nitrogen (N), agricultural soils are the main anthropogenic source of N$_2$O, but high spatial and temporal variabilities make N$_2$O emissions difficult to characterize at the field scale. This study used flux-gradient measurements to continuously monitor N$_2$O emissions at two agricultural fields under different management regimes in the inland Pacific Northwest of Washington State, USA. Automated 16-chamber arrays were also deployed at each site; chamber monitoring results aided the interpretation of the flux gradient results. The cumulative emissions over the six-month (1 April–30 September) monitoring period were 2.4 ± 0.7 and 2.1 ± 2 kg N$_2$O-N/ha at the no-till and conventional till sites, respectively. At both sites, maximum N$_2$O emissions occurred following the first rainfall event after N fertilization, and both sites had monthlong emission pulses. The no-till site had a larger N$_2$O emission factor than the Intergovernmental Panel on Climate Change Tier 1 emission factor of 1% of the N input, while the conventional-till site's emission factor was close to 1% of the N input. However, these emission factors are likely conservative. We estimate that the global warming potential of the N$_2$O emissions at these sites is larger than that of the no-till conversion carbon uptake. We recommend the use of chambers to investigate spatiotemporal controls as a complementary method to micrometeorological monitoring, especially in systems with high variability. Continued monitoring coupled with the use of models is necessary to investigate how changing management and environmental conditions will affect N$_2$O emissions.

Plain Language Summary Nitrous oxide (N$_2$O) is a greenhouse gas and stratospheric ozone depleting substance that is emitted by soils. Agricultural soils tend to emit more N$_2$O than natural soils due to the addition of nitrogen fertilizers. N$_2$O emissions are not well understood on the scale of individual farms, as emissions are difficult to measure at this resolution because they are irregular over time and space. This variability is due to the dependence of N$_2$O production and emission on soil properties, that is, moisture, nitrogen, and the microbiome. In this study we monitored N$_2$O emissions from two agricultural fields under different tillage regimes using two complementary methods: the flux-gradient technique and automated chambers. The flux-gradient technique measures N$_2$O emissions at the field scale, which is relevant to agronomic management. Using both techniques together improves confidence in our results, which give us information on total N$_2$O emissions from these fields, as well as the relationships between N$_2$O emissions and rainfall, temperature, and carbon dioxide respiration.

1. Introduction

Nitrous oxide (N$_2$O) has multiple negative environmental impacts, yet factors affecting N$_2$O emissions are inadequately understood and difficult to control (Butterbach-Bahl et al., 2013; Galloway et al., 2008; Groffman et al., 2000; Groffman et al., 2006). N$_2$O is a potent greenhouse gas (GHG), with 265 times the global warming potential of carbon dioxide (CO$_2$) over a 100-year time horizon (Myhre et al., 2013). Following the phase out of chlorofluorocarbons, N$_2$O has become the dominant stratospheric ozone-depleting...
substance (Ravishankara et al., 2009). \( \text{N}_2\text{O} \) is produced in both natural and managed soils as a by-product of microbial processes, but manure management and the addition of fertilizer nitrogen (N) to agricultural soils elevates \( \text{N}_2\text{O} \) production and emission in agroecosystems. In the United States, it is estimated that agricultural soil management accounts for 77% of anthropogenic \( \text{N}_2\text{O} \) (USEPA, 2017).

Although top-down and bottom-up global estimates of \( \text{N}_2\text{O} \) agree well, there is uncertainty in emission rates at regional and smaller scales (Reay et al., 2012; Syakila & Kroeze, 2012). Reay et al. (2012) call for more measurements of \( \text{N}_2\text{O} \) distributed over major agricultural systems globally to improve local emission estimates and aid in the development of mitigation strategies. Butterbach-Bahl et al. (2013) also emphasized the need for improved measurements to inform agroecological management questions, such as the potential trade-off between soil carbon sequestration and \( \text{N}_2\text{O} \) emissions (Lugato et al., 2018). Agricultural \( \text{N}_2\text{O} \) emissions can increase exponentially with increasing fertilizer application rates (Millar et al., 2018; Shcherbak et al., 2014), highlighting the need for site-specific research over linear emission factors such as those used in the Intergovernmental Panel for Climate Change's (IPCC) methodologies (IPCC, 2006).

Accurate measurements of \( \text{N}_2\text{O} \) emissions are difficult to make because of the high degree of spatial and temporal variability in emissions, or “hot spots” and “hot moments” (Mason et al., 2017; McClain et al., 2003; Molodovskaya et al., 2012; Nicolini et al., 2013). Static chambers (SC) have been used for decades to measure emissions of \( \text{N}_2\text{O} \) from soils; more recently the use of automated multichamber systems has enabled a high degree of temporal coverage relative to manual chambers, but coverage of spatial heterogeneity can still be inadequate (Barton et al., 2015; Butterbach-Bahl et al., 2013; Loescher et al., 2006). Micrometeorological techniques (MM) have high temporal resolution and integrate flux measurements over the field scale (hundreds of square meters, depending on boundary layer turbulence conditions). Furthermore, MM techniques do not disturb the soil and are nondestructive to site vegetation. However, MM requires site conditions to meet strict theoretical assumptions, require a sensitive, stable \( \text{N}_2\text{O} \) instrument with a fast response time, and have demanding power requirements.

How to optimize the use of manual or automated SC and MM for monitoring \( \text{N}_2\text{O} \) emissions is a topic of ongoing discussion in the literature (cf.Bai et al., 2019; Jones et al., 2011; Nicolini et al., 2013; Tallec et al., 2019). Nicolini et al. (2013) point out the mixed results reported in \( \text{N}_2\text{O} \) MM and SC method comparison studies. Careful placement of chambers to be within the micrometeorological flux footprint and be representative of the field-scale spatial variability have been recommended by several studies (e.g., Laville et al., 1999; Loescher et al., 2006; Pihlatie et al., 2005). Advanced SC upscaling techniques such as using footprint analyses and models have resulted in good agreement with MM methods for measuring methane fluxes (Nicolini et al., 2013), and one short-term study of \( \text{N}_2\text{O} \) emissions from sugarcane soils found good agreement between MM results and the mean of the SC measurements (Denmead et al., 2010). However, the drivers of field-scale variability in \( \text{N}_2\text{O} \) emissions, such as inorganic N content, soil moisture, and/or microbial community, are often not readily visible and thus not easy to incorporate into SC deployment or upscaling design in agronomic systems (McDaniel et al., 2017). If the goal of the monitoring is to characterize field-scale emissions, and the field is homogeneous at the macroscale, then in theory the SC location could be independent of the MM flux footprint.

Long-term comparison studies have found limited agreement between the two techniques. This is often attributed to the difference in measurement footprints rather than a systematic bias; as discussed in Bai et al. (2019), SC and MM comparison studies show mixed results in terms of total emissions: in some cases, the MM measurements indicate larger emissions than SC (e.g., Bai et al., 2019; Norman et al., 1997; Wang et al., 2013), in other cases the reverse (e.g., Christensen et al., 1996; Neftel et al., 2010; Tallec et al., 2019). Similar to Bai et al. (2019), Jones et al. (2011) attributed the disagreement between their MM and SC results to different coverage of emission hot spots. The authors recommend using the two methods to complement each other: the MM results to monitor overall field-scale \( \text{N}_2\text{O} \) emissions, and SC to interpret the degree of spatial variability and environmental drivers. Similarly, Tallec et al. (2019) highlighted the usefulness of MM results for interpreting the whole-agroecosystem GHG budget and how \( \text{N}_2\text{O} \) emissions respond to environmental drivers at the field scale, and the usefulness of SC results for interpreting soil processes. The authors also list physical disturbance of the site and the need to remove and replace SC to accommodate field management as limits to using SC for defensible annual \( \text{N}_2\text{O} \) emission budgets. These logistical challenges are amplified when monitoring fluxes during spring
thaw, an important N$_2$O emission hot moment in northern agricultural sites that coincides with challenging field conditions (Flesch et al., 2018).

In this study, we measured fluxes of N$_2$O from paired agricultural fields in the inland Pacific Northwest United States using both automated static chamber systems and the flux-gradient micrometeorological technique. This intensive monitoring was part of the REgional Approaches to Climate CHange for Pacific Northwest Agriculture project, and sought to address two questions about N$_2$O emissions and climate mitigation:

1. What are the magnitudes and patterns in N$_2$O fluxes at these two sites?
2. How do the N$_2$O fluxes respond to environmental conditions and agronomic management?

The goal of answering these questions is to improve monitoring of N$_2$O emissions in this region by characterizing key temporal and spatial patterns at these sites. In addition, we critically assess uncertainties in both methods and present recommendations for strategies to investigate N$_2$O emissions from agricultural fields with high spatial variability.

2. Methods

2.1. Site Description

The two N$_2$O measurement sites were located <1 km from each other at the Washington State University Cook Agronomy Farm (CAF; 46.78°N, 117.09°W; 800 m above sea level) northeast of Pullman, WA (Figure 1). The region has a semiarid Mediterranean climate with average annual precipitation of 550 mm and an average annual temperature of 9 °C. The average high temperature is 26 °C in the summer and the average winter low is −4 °C. The soils are productive Palouse silt-loam series (fine-silty, mixed, superactive, mesic Pacific Ultic Haploxerolls). Typical springtime soil characteristics at the Cook Agronomy Farm include total soil N of 0.19 ± 0.01%, total C of 2.6 ± 0.1%, and pH of 5.2 ± 0.1 (Shrewsbury et al., 2016). The study site is representative of approximately 0.5 million hectares where dryland annual cropping is practiced in the inland Pacific Northwest (iPNW). The principal management difference between the two sites is the tillage regime. One of the sites was converted to no-tillage management in 1998 (CAF-NT), a practice implemented by ~30% of growers in the iPNW, and the other is managed with conventional reduced tillage (CAF-CT), used by ~40% of growers in the region (Bista et al., 2017). Other management differences include fertilization amount and timing, timing of sowing and harvest, and herbicide application (see Table 1). The N$_2$O measurement sites were co-located with established eddy covariance flux towers that monitored carbon dioxide, water vapor, and energy fluxes from 2011–2016 and 2012–2016 for CAF-NT and CAF-CT, respectively. Complete descriptions of the eddy covariance systems and auxiliary measurements are in Chi et al. (2016) and Waldo et al. (2016).

The monitoring results presented in this analysis are for the 2015 harvest year, 1 October 2014 to 30 September 2015; both sites were growing spring canola (Brassica napus). During the prior year, winter wheat (Triticum aestium) was grown at both sites (Table 1). The winter wheat was harvested in early August 2014 and the residue was retained at both sites, containing 28 ± 3 and 37 ± 6 kg N/ha at CAF-NT and CAF-CT, respectively. CAF-CT was tilled with a chisel plow to a depth of ~0.2 m on 27 October 2014. "Roundup ready" canola was seeded and fertilized on 4 April at CAF-CT, and 18 April at CAF-NT. A combination of anhydrous ammonia and liquid solution N fertilizers were applied at a total rate of 89 kg N/ha at CAF-NT and 100 kg N/ha at CAF-CT. At CAF-NT, the field was seeded and fertilized with a no-tillage drill, while at CAF-CT the fertilizer was deep-banded (~8 cm) with a shank applicator, and simultaneously chisel tilled. Herbicide was sprayed at CAF-NT on 30 May 2015 to terminate weeds in established canola. Both fields were harvested on 14 August 2015, and the canola residue was retained at both sites. Fertilizer was applied at a rate of 168 kg N/ha at CAF-CT on 20 August 2015 for fall planting of winter wheat. Assuming negligible atmospheric N deposition, the total N inputs for the sites as defined by the IPCC emission factor methodology (IPCC, 2006) are equal to the sum of crop residue N and fertilizer N; at CAF-NT this was 115 kg N/ha, and at CAF-CT the total N inputs for the 2015 crop year were 297 kg N/ha. It is arguable whether the N from the 20 August fertilization should be included in the budget; excluding it results in total N inputs of 138 kg N/ha.
2.2. Flux-Gradient Measurements

The flux-gradient (FG) technique is a micrometeorological technique assuming that turbulent mixing in the atmosphere is analogous to Fick’s law of molecular diffusion (Meredith et al., 2014). Fluxes are calculated by multiplying the molar density gradient of a trace gas by the eddy diffusivity, \( K \):

\[
F_c = -K \frac{\Delta C}{\Delta z} \tag{1}
\]

where \( F_c \) is the flux of the \( \text{N}_2\text{O} \) (nmol m\(^{-2}\) s\(^{-1}\)), \( K \) is the eddy diffusivity (m\(^2\)/s), and \( \Delta C \) is the gradient of \( \text{N}_2\text{O} \) molar density (nmol/m\(^3\)) over \( \Delta z \), the vertical gradient (m). \( K \) is a function of turbulent intensity (Baldocchi
et al., 1995; Dunn et al., 2009; Goldstein et al., 1996), and for this study was determined via the modified Bowen ratio technique (Dunn et al., 2009; Liu & Foken, 2001; Meyers et al., 1996). Using temperature as a tracer, \( K \) is calculated as

\[
K = -\frac{H}{\Delta T \rho_a C_p}
\]

where \( H \) is the sensible heat flux (W/m²), \( \Delta T \) is the difference in temperature (K) at the same heights (\( \Delta z \), m) as the \( \text{N}_2\text{O} \) measurements, \( \rho_a \) is the mass density of air (kg/m³), and \( C_p \) is the specific heat capacity of air (J kg⁻¹ K⁻¹). The sensible heat flux was calculated via the eddy covariance technique. The EC fluxes were calculated using EddyPro® version 4.2.0 (LI-COR Biosciences, Lincoln, NE, USA). The \( H \) fluxes were gap-filled using the mean diurnal variation method (Falge et al., 2001; Moffat et al., 2007). More details about the EC flux processing and gap-filling the \( H \) fluxes at these sites can be found in Waldo et al. (2016) and Chi et al. (2016).

The \( \text{N}_2\text{O} \) mixing ratio was measured at two heights using off-axis integrated cavity output spectroscopy instruments (\( \text{N}_2\text{O}/\text{CO}_2 \text{d} \), Los Gatos Research, Mountain View, CA). The gas inlets were installed at heights of 1.2 and 2.5 m above ground level at CAF-CT and 0.8 and 1.9 m at CAF-NT and were outfitted with Teflon particle filters (2 μm). The CAF-NT site had a standard model LGR instrument (part number 907-0014), while at the CAF-CT site the enhanced performance model was used (part number 913-0014); the latter controls the temperature of the optical cell to improve precision and stability of the measurements and has higher power requirements. Temperature was measured at the two heights using aspirated thermistors with a precision of ±0.01 °C (temperature sensor model number: 107-30, aspirated shield model number: 43502-30; both Campbell Scientific, Logan, UT).

The SC and FG systems shared the \( \text{N}_2\text{O} \) instrument on a 4-hr circuit starting at midnight: 3 hr were allocated to cycle through the 16 chambers, and the fourth hour was allocated to the tower. That timeline resulted in six total chamber cycles per day and two 30-min tower fluxes measured 6 times per day (0300, 0330, 0700, 0730, 1100, 1130, 1500, 1530, 1900, 1930, 2300, and 2330). Solenoid valves were used to switch between the measurement systems (Figure S1), and the gradient tubing lines were flushed at a rate of 1.6 L/min by a double-headed bypass pump (B162-AP-AA1 Double Head B-series pump, Air Dimensions Inc., Deerfield Beach, FL, USA). An example of the \( \text{N}_2\text{O} \) and temperature gradient raw time series is shown in Figure S2. More details on the combined SC-FG system can be found in Kostyanovsky et al. (2017).

### Table 1

| Date             | CAF-NT                          | CAF-CT                          |
|------------------|---------------------------------|---------------------------------|
| 6 August 2014    | Winter wheat harvest, residue ret. (37 kg/ha) | Winter wheat harvest, residue ret. (28 kg N/ha) |
| 7 August 2014    | Winter wheat harvest, residue ret. (37 kg/ha) | Field tilled                     |
| 27 October 2014  | Canola seeded and fertilized (87 kg/ha) | Canola seeded and fertilized (101 kg N/ha) |
| 4 April 2015     | Herbicide sprayed              | Canola harvested                 |
| 18 April 2015    | Canola harvested                | Canola harvested                 |
| 30 May 2015      | Herbicide sprayed              | Canola harvested                 |
| 14 August 2015   | Herbicide sprayed              | Canola harvested                 |
| 20 August 2015   | Herbicide sprayed              | Canola harvested                 |

2.3. Data Filtering, Quality Assurance, and Footprint

The \( \text{N}_2\text{O} \) mixing ratio measurements were filtered for values outside of the plausible range of \( \text{N}_2\text{O} \) ambient mixing ratios (<300 or >400 ppb). The first 60 s after switching from the chamber system to the flux-gradient system were discarded, and the first 16 s of each 60-s gradient height measurement were discarded (Figure S2). Air sample travel time lag was also considered in the calculation. The 30-min average \( \text{N}_2\text{O} \) concentration gradients (\( \Delta C \)) between the upper (even \( c_i \)) and lower (odd \( c_i \)) inlet heights were calculated as...
\[
\Delta C = \frac{1}{n-2} \sum_{i=3}^{n-1} c_i + \frac{c_i + c_{i+2}}{2} + \frac{c_{i+1} + c_{i+3}}{2} - c_{i+2}
\]  

This averaging procedure puts less weight on the terminal measurements and acts as a high-pass digital filter (Brown & Wagner-Riddle, 2017; Wagner-Riddle et al., 2007).

Zero-gradient tests were performed at each site to quantify any bias in the flux-gradient system. The inlets were co-located at the same height for three days and the resulting "gradient" in N$_2$O mixing ratios was calculated by taking the mean of the absolute values of the 36 independent data points (Figure S3). We compared the average gradients calculated during these zero-gradient tests to the theoretical minimum detection limit defined by Pattey et al. (2006):  

\[
\Delta S = \sqrt{\frac{N_h}{n} \sigma_s}
\]

where $\Delta S_{\text{min}}$ is the resolution of the sampled gradient, $N_h$ is the number of sampling heights, $\sigma_s$ is the random instrument noise (0.3 ppb on a 1-s basis), and $n$ is the number of samples taken at each level over the sampling period. The nominal value for $\Delta S_{\text{min}}$ is 0.047 ppb N$_2$O (given $N_h = 2$ sampling heights, $n =$ nominally 270 independent measurements per height over the 30-min sampling period after data filtering, and $\sigma_s$ of 0.3 ppb). The results of the zero-gradient tests were similar in magnitude to the theoretical minimum detection limit but biased slightly positive. To correct for this bias, the average result from the zero-gradient test at each site was subtracted from the measured gradient before using it to calculate an N$_2$O flux. Any offset between the aspirated thermistors used to measure the temperature gradient was addressed by calibrating the sensors to each other twice during the measurement period by co-locating the instruments.

The FG N$_2$O fluxes were filtered for conditions that violate the assumptions of the gradient technique. Periods were discarded if $|\Delta T| < 0.05 \, ^\circ\text{C}$, if the sensible heat flux ($H$) had the same sign as $\Delta T$, that is, counter-gradient fluxes, if winds were coming from behind the tower (330°–30°), friction velocity ($u_*$) < 0.13, or if the period was classified as poorest quality based on stationarity and integral turbulence characteristics tests (Foken & Wichura, 1996). Furthermore, we filtered selected 30-min flux periods with random error that exceeded the 97th percentile (fractional error greater than 2,000% and 900% for CAF-NT and CAF-CT, respectively).

The flux contribution footprint for each 30-min FG measurement was determined using the Flux Footprint Prediction model developed by Kljun et al. (2015). The measurement height of the sonic anemometer at each site was used as the reference height (1.91 and 2.21 m at CAF-NT and CAF-CT, respectively). Canopy height, used to determine roughness length and displacement height, was calculated by averaging measurements of crop height taken at four points around the flux tower every two weeks during the growing season.

### 2.4. Chamber Measurements

An array of 16 static automated chambers (LI-8100, LiCor Biosciences, Lincoln, NE) was installed at each site. The LI-8100 instrumentation has a carbon dioxide (CO$_2$) analyzer and a multiplexer to control the sample gas flow from the 16 chambers. The chambers were arranged in four parallel lines approximately 20 m apart, with approximately 2 m between each chamber, covering a total grid area of ~360 m$^2$. The N$_2$O mixing ratio within the chamber headspace was measured with the same integrated cavity output spectroscopy instruments used to measure N$_2$O for the flux-gradient system (section 2.2). The N$_2$O flux out of the soil was calculated as

\[
F_{\text{N}_2\text{O}} = \frac{\Delta N_{\text{N}_2\text{O}}}{\Delta t} \times \frac{V}{A}
\]

where $\Delta N_{\text{N}_2\text{O}}/\Delta t$ is the change in N$_2$O molar density (nmol/m$^3$) observed in the chamber during each sample period ($\Delta t$), determined from a best fit linear or exponential regression of the instrument output versus time during each sampling period; $V$ is the volume (m$^3$) of the chamber system including tubing and instrument cavities; and $A$ (m$^2$) is the surface area of the chamber surface. The effective noise level (3$\sigma$) of the N$_2$O
instrument is 0.3 ppb on a 1-s basis which yields a minimum flux detection limit of 0.008 nmol N₂O m⁻² s⁻¹ for the chamber method. The chamber system is described in more detail by Kostyanovsky et al. (2017).

2.5. Uncertainty Parameterization

2.5.1. Random Uncertainty in the Flux Gradient Measurements

Uncertainty in the N₂O fluxes calculated with the flux-gradient technique was determined by propagating the random error associated with each of the component measurements: ΔC, H, ΔT, and Δz. Errors in ρa and Cp were deemed negligible. The random error in the N₂O concentration gradient (EΔC) was calculated as

\[ E_{\Delta C} = \sqrt{\frac{\sigma_{z_1}^2}{n_1} + \frac{\sigma_{z_2}^2}{n_2}} \]

where \( \sigma_{z_1} \) and \( \sigma_{z_2} \) are the standard deviations of the N₂O concentration measurements at heights \( z_1 \) and \( z_2 \), respectively, and \( n_1 \) and \( n_2 \) are the number of observations at the two heights. The uncertainty in \( H \) was parameterized as the variance of the covariance (Finkelstein & Sims, 2001). The random error in the temperature gradient was calculated similar to the N₂O concentration gradient, using the measured standard deviations. Error in the gradient separation height, \( \Delta z \), due to “droopiness” of the lines after weather events, etc., was estimated to be 5%, or 5–6 cm. The random error of the N₂O flux was calculated by combining the fractional random error of each component source in quadrature and multiplying by the N₂O flux:

\[ E_{\phi_c} = \sqrt{\left( \frac{\partial F}{\partial \Delta C} E_{\Delta C} \right)^2 + \left( \frac{\partial F}{\partial H} E_H \right)^2 + \left( \frac{\partial F}{\partial \Delta T} E_{\Delta T} \right)^2 + \left( \frac{\partial F}{\partial \Delta z} E_{\Delta z} \right)^2} \]

This error was determined for each 30-min flux-gradient measurement period. The 30-min random errors were propagated to daily mean flux random error by summing in quadrature.

2.5.2. Systematic Uncertainty in the Flux Gradient Measurements

To parameterize the systematic uncertainty of the N₂O flux gradient measurements, we looked at two major sources: uncertainty due to one-point sampling (\( U_{op} \)) and uncertainty in the eddy diffusivity, \( K \). Previous eddy covariance studies of N₂O fluxes have reported that most of the systematic uncertainty can be attributed to one-point sampling (Kroon et al., 2010; Tallec et al., 2019; Wang et al., 2013). We calculated \( U_{op} \) as

\[ U_{op} = \sqrt{\frac{20 \zeta}{TU} \sqrt{\sigma_{N_2O_{z_1}}^2 + \sigma_{N_2O_{z_2}}^2}} \]

where \( \zeta \) is the measurement height (m), \( T \) is the averaging time (s), \( U \) is the wind speed (m/s), and the \( \sigma_{N_2O}^2 \) terms are the variances in N₂O molar densities measured at each height.

To parameterize the systematic uncertainty in \( K \), we compared the values of \( K \) calculated using the modified Bowen ratio technique using temperature as a tracer (equation (2)), with \( K \) calculated using similarity theory ("K aerodynamic," hereafter "\( K_a \);" cf. Brown & Wagner-Riddle, 2017):

\[ K_a = \frac{u_\ast (z_2 - z_1)}{\ln \left( \frac{z_2 - d}{z_1 - d} \right)^{-\psi_{z_2}} + \psi_{z_1}} \]

where \( u_\ast \) is the friction velocity, \( \kappa \) is the von Karman constant (\( \kappa = 0.40 \)), \( z_2 \) and \( z_1 \) are the two gradient measurement heights, \( d \) is the zero-plane displacement height, and \( \psi_{z_2} \) and \( \psi_{z_1} \) are the integrated Monin-Obukhov similarity functions for heat for each sampling height. The similarity functions depend on stability conditions, categorized by \( \zeta = (z - d) L^{-1} \), and can be calculated using the Businger-Dyer relationships (Businger et al., 1971; Dyer & Hicks, 1970):

\[ \psi = \begin{cases} 2 \ln \left( \frac{1 + x^2}{2} \right), & x = (1 - 15 \zeta)^{1/4} \quad \text{Unstable}, \zeta < -0.01 \\ -4.7 \zeta, & \text{Stable}, \zeta > 0.01 \\ 0, & \text{Neutral}, -0.01 < \zeta < 0.01 \end{cases} \]

where \( L \) is the Monin-Obukhov length:
\[ L = \frac{(-u^2 T_v)}{w T_v} \]

where \( T_v \) is the virtual temperature, \( g \) is the acceleration due to gravity, and \( w T_v \) is the kinematic virtual temperature flux.

### 2.5.3. Uncertainty in the Chamber Measurements

The uncertainty in the daily \( \text{N}_2\text{O} \) emissions measured by the chambers was estimated by calculating the 95% confidence interval of the arithmetic mean chamber \( \text{N}_2\text{O} \) emission averaged between individual chambers and over the six daily observation cycles (both space and time). Since neither data set followed a normal, log-normal, or other discernable distribution, we used a nonparametric Chebyshev-type inequality to calculate the 95% confidence interval of each chamber data set (Singh et al., 2006).

### 2.6. Determining Cumulative \( \text{N}_2\text{O} \) Emissions and Relation of \( \text{N}_2\text{O} \) to \( \text{CO}_2 \)

Cumulative \( \text{N}_2\text{O} \) emissions for both sites and both methods were determined by integrating the daily mean \( \text{N}_2\text{O} \) emissions over the measurement period. Gaps in the daily \( \text{N}_2\text{O} \) emissions time series were filled via linear interpolation. This gap-filling strategy was selected for simplicity. Uncertainties were propagated from daily to cumulative fluxes for both the chamber and flux-gradient results by summing in quadrature. To compare the relative impact of \( \text{N}_2\text{O} \) and \( \text{CO}_2 \) fluxes in these systems, we converted the \( \text{N}_2\text{O} \) emissions to \( \text{CO}_2 \)-eq using the global warming potential factor of 265 kg \( \text{N}_2\text{O} \) per kg \( \text{CO}_2 \) (Myhre et al., 2013).

### 3. Results

#### 3.1. \( \text{N}_2\text{O} \) Monitoring Results

Emissions of \( \text{N}_2\text{O} \) had the same general trend over time at both sites: they increased from baseline levels after canola was planted and fertilized in the spring to peak levels in mid-May, coinciding with a heavy precipitation event that took place from 12 to 16 May 2015 (Figure 2). Maximum daily \( \text{N}_2\text{O} \) emission values measured at CAF-NT by the FG and SC systems were 202 ± 65 and 54 ± 26 g \( \text{N}_2\text{O} \)-N ha\(^{-1}\) d\(^{-1}\) on 21 and 24 May, respectively. Maximum daily \( \text{N}_2\text{O} \) emission values measured at CAF-CT by the FG and SC methods were 94 ± 48 and 136 ± 67 g \( \text{N}_2\text{O} \)-N ha\(^{-1}\) d\(^{-1}\) on 14 and 19 May, respectively. \( \text{N}_2\text{O} \) emissions then decreased to a baseline close to zero (<14 g \( \text{N}_2\text{O} \)-N ha\(^{-1}\) d\(^{-1}\) at CAF-NT and <3.5 g \( \text{N}_2\text{O} \)-N ha\(^{-1}\) d\(^{-1}\) at CAF-CT) from late June through August, punctuated by a brief spike in \( \text{N}_2\text{O} \) emissions due to a rainfall event on 10 July 2015 (observed in each data set except for the CAF-NT FG measurements). Emissions increased moderately in September, coinciding with rainfall events. The SC results for 1 October 2014 to 1 April 2015 show that both sites had measurable \( \text{N}_2\text{O} \) emissions during the winter; larger \( \text{N}_2\text{O} \) emissions were measured by SC at the CAF-NT site than the CAF-CT site (Figure 2). Most chambers at both sites had a diurnal pattern of elevated afternoon \( \text{N}_2\text{O} \) emissions (Figures S4 and S5). No clear diurnal pattern is discernible in the FG results (Figure S6), but the lack of temporal coverage, especially over the periods of 0700 and 1900, likely contributed to this (Figure S7).

We compared the SC and FG results at the two sites separately for periods of elevated and baseline fluxes (Table 2; cf. Tallec et al., 2019). We defined 15–29 July 2015 as the baseline period and 12–26 May 2015 as the emission pulse period. We used evaporation as a proxy for soil moisture in comparing the two periods at the two sites. Evaporation (\( E \)) was parameterized via partitioning the latent heat flux (\( LE \)) measured via eddy covariance into evaporation and transpiration (see Chi et al., 2016). The CAF-CT site saw a larger difference in \( E \) between the midsummer baseline period and the early summer emission pulse than CAF-NT did. The drop in \( E \) at CAF-CT, indicating a drop in soil moisture relative to CAF-NT, is consistent with previous results at these two sites: during the 2013 crop year (both sites growing garbanzo beans) CAF-NT had consistently higher soil water content than CAF-CT at 18- and 30-cm depths. This difference can be attributed to tillage management; no-till management can reduce evaporation and increase precipitation infiltration (Chi et al., 2016; Chi et al., 2017). However, the differences in \( \text{N}_2\text{O} \) emission magnitude between the FG and SC results at both sites make it difficult to interpret the impact of field-scale soil moisture effects on \( \text{N}_2\text{O} \) emissions.
Figure 2. Time series of N$_2$O fluxes. (a) CAF-NT FG, (b) CAF-NT chambers, (c) CAF-CT FG, and (d) CAF-CT chambers. (e) Precipitation and (f) temperature for the 2015 crop year. Management activities are indicated by $T =$ tillage, planting, $F =$ fertilization, $S =$ herbicide spray application, and $H =$ harvest.

Table 2
Summary Statistics (Mean, Coefficient of Variation (CV), Number of Independent Observations (N), and Lower 5% and Upper 95% Intervals) During a Two-Week Baseline Emission Period (15–29 July 2015) and During Two Weeks of the Emission Pulsed (12–26 May 2015) at Both the No-Tillage and Conventional Tillage Sites (CAF_NT and CAF-CT, Respectively)

|                  | CAF-NT |                  | CAF-CT |                  |
|------------------|--------|------------------|--------|------------------|
|                  | FG     | SC               | $E$    | $T_{air}$ (°C)   | FG     | SC               | $E$    | $T_{air}$ (°C)   |
|                  | (ng N$_2$O-N m$^{-2}$ s$^{-1}$) | (ng N$_2$O-N m$^{-2}$ s$^{-1}$) | (mm m$^{-2}$ d$^{-1}$) |           | (ng N$_2$O-N m$^{-2}$ s$^{-1}$) | (ng N$_2$O-N m$^{-2}$ s$^{-1}$) | (mm m$^{-2}$ d$^{-1}$) |           |
| Baseline (15–29 July) |        |                  |        |                  |        |                  |        |                  |
| Mean 2.00        | 2.00   | 0.51             | 1.32   | 20.0             | 0.25   | 2.98             | 0.69   | 20.2             |
| CV 58%           | 113%   | 90%              | 26%    | 1920%            | 90%    | 104%             | 28%    |                  |
| N 92             | 1344   | 670              | 670    | 68               | 1344   | 670              | 670    |                  |
| 5th % −17.0      | 0      | 0.17             | 12.0   | −6.42            | 0.11   | 1.11             | 0.03   | 11.6             |
| 95th % 18.6      | 1.65   | 3.64             | 29.1   | 8.67             | 8.12   | 2.08             | 30.1   |                  |
| Emission pulse (12–26 May) |        |                  |        |                  |        |                  |        |                  |
| Mean 74.1        | 42.2   | 1.94             | 14.6   | 47.7             | 133    | 1.98             | 14.8   |                  |
| CV 143%          | 126%   | 98%              | 29%    | 64%              | 125%   | 108%             | 29%    |                  |
| N 30             | 1282   | 670              | 594    | 130              | 1342   | 670              | 462    |                  |
| 5th % 5.95       | 1.63   | 0.26             | 8.39   | 6.35             | 3.99   | 0.03             | 8.60   |                  |
| 95th % 95.95     | 197    | 5.58             | 21.4   | 105              | 501    | 6.40             | 21.6   |                  |

Statistics are reported for the N$_2$O fluxes measured with the flux gradient method (FG), and the static chamber systems (SC), for latent heat flux (LE), and for air temperature ($T_{air}$), proxies for soil moisture and temperature.
emissions during either period. During the baseline period, it is likely that conditions other than moisture (e.g., substrate availability) affected production and emission of N$_2$O.

3.2. Measurement Uncertainty

3.2.1. Flux-Gradient Random and Systematic Uncertainties

Random measurement error ($E_{FC}$) accounted for most of the cumulative error at both sites (Figure 3). Propagating $E_{FC}$ over the six-month monitoring period resulted in ±29% and ±7.6% of the cumulative flux for CAF-NT and CAF-CT, respectively. The latter is comparable to random error magnitudes reported in Brown and Wagner-Riddle (2017), while the former is more similar to the error ranges reported by Phillips et al. (2007). We attribute the difference in random error to the different instruments used: the high-performance instrument at CAF-CT was better able to resolve N$_2$O gradients and recorded less noise than the instrument at CAF-NT. Random errors in the measurement of the N$_2$O gradient ($E_{ΔC}$) was the largest contributor to $E_{FC}$ at both sites: the median contributions of $E_{ΔC}$ to $E_{FC}$ for all 30-min FG calculation periods were 98% and 91% at CAF-NT and CAF-CT, respectively.

Systematic uncertainty due to one-point sampling was a smaller source of uncertainty than due to characterization of $K$ ($U_k$), the latter of which notably did not follow the same exponential decay in fractional uncertainty with an increase in flux magnitude (Figure S8). We chose not to include the $U_k$ in our overall uncertainty calculation. Due to the nonideal hilly topography of our study locations (Figure S9), we argue that using a tracer is a more direct method of characterizing the eddy diffusivity, violating fewer assumptions than similarity theory does. See Figure S10 for a direct comparison of the two $K$ terms. It would be
appropriate to include $U_k$ if both the modified Bowen ratio method with temperature as a tracer ($K_T$) and similarity theory ($K_a$) were equally well suited for our sites.

The systematic uncertainty due to one-point sampling ($U_{op}$) was more important to the overall uncertainty of the 30-min $N_2O$ fluxes at CAF-CT than at CAF-NT. The median contributions of fractional $U_{op}$ and random measurement fractional error to the total (sum of random and systematic) error for all 30-min FG calculation periods were 45% and 47%, respectively, at CAF-CT, and 22% and 77%, respectively, at CAF-NT. The $U_{op}$ error had a larger impact on smaller-magnitude fluxes (Figure 3). The propagated $U_{op}$ error of the cumulative fluxes over the six-month monitoring period represented ±0.3% of the cumulative flux for CAF-NT and ±0.6% for CAF-CT, more than an order of magnitude smaller than the cumulative uncertainty from random measurement error.

The results of the FG zero gradient tests indicated slight positive biases at both sites: 0.042 ± 0.022 ppb/m at CAF-NT and 0.021 ± 0.010 ppb/m at CAF-CT (Figure S3). These values are similar in magnitude to the average theoretical minimum detection limits calculated with equation (7) of 0.040 ± 0.006 and 0.032 ± 0.006 ppb/m at CAF-NT and CAF-CT, respectively. The lower level of noise at the CAF-CT site is consistent with the advantages of the enhanced performance model LGR instrument deployed at that site.

### 3.2.2. Chamber Spatial Variability

At both sites, hot spot chambers had disproportionate impacts on the mean upscaled $N_2O$ emissions. Each ~20-cm-diameter chamber comprised 6.25% (1/16th) of the total SC measurement area. The emissions measured at one Group 3 chamber at CAF-NT comprised 25% of the total $N_2O$ emissions (Figure 4b) at that site, and the two highest-emitting chambers accounted for 33% of total emissions. At CAF-CT, the highest emissions measured by a Group 1 chamber comprised almost 17% of the total emissions at that site (Figure 4d), and the two highest measurement sites accounted for 27% of total emissions. Furthermore, the difference between the mean and median illustrate the non-Gaussian distribution of the SC measurements, an indication of systematic uncertainty. The SC cumulative mean and median at CAF-NT are 3.9 and 3.2 kg $N_2O$-N ha$^{-1}$ yr$^{-1}$, respectively, and 5.6 and 4.5 kg $N_2O$-N ha$^{-1}$ yr$^{-1}$, respectively, at CAF-CT. While we cannot ignore...
the hot spot chambers, these results highlight the difficulty in characterizing field-scale emissions from SC measurements.

The average spatial coefficient of variation (CV) between 1 October 2014 and 30 September 2015 between all the chambers at CAF-NT was 147%; CAF-CT had a similar average annual CV of 143%. The average CVs during the elevated emission period from 1 April to 1 July 2015 at CAF-NT and CAF-CT were lower than the annual average: 121% and 104%, respectively (Figure S11). The annual average CV values for CO2 were 96% and 84% for CAF-NT and CAF-CT (not shown). The high CV values indicate that more extensive chamber coverage would be needed to accurately characterize emissions: in a controlled emission study Loescher et al. (2006) found in a system with a CV of 0.8 (the highest CV considered), ≥20 chamber locations were needed to be within 50% of the true mean emission (α = 0.1).

3.3. Implications of Spatial Variability on Comparing FG and SC Results

The spatial coverage of the FG and SC systems and their ability to characterize patchy emissions is important when comparing results from the two systems. The soil surface area enclosed by each chamber was ~318 cm², for a total measured surface area of 0.51 m² at each site. A more liberal way to characterize the SC measurement area is by the dimensions of the area encompassed by the four-chamber by four-chamber grid:
measurements. The uncertainty is the propagated random error in the 30-minute
measurement period. SHadowed areas indicate uncertainty. For SC,
uncertainty is the propagated standard error of the daily measurements. For FG,
the uncertainty is the propagated random error in the 30-minute
measurements.

![Cumulative N2O emissions](image)

**Figure 6.** Cumulative N2O emissions at the no-till site (CAF-NT, green), and the conventional till site (CAF-CT, blue) with SC (solid lines) and the FG method (dashed line). Shaded areas indicate uncertainty. For SC, uncertainty is the propagated standard error of the daily measurements. For FG, the uncertainty is the propagated random error in the 30-minute measurements.

~360 m². Using the flux footprint parameterization (Kljun et al., 2015), we estimate that the FG systems were integrating flux signals over a cumulative area of ~2,500 m² (Figure 5; integrating up to the 80% flux contribution extent). There is uncertainty in footprint parameterization (cf. Heidbach et al., 2017); other footprint models (Kljun et al., 2004; Kormann & Meixner, 2001) indicate a larger footprint area (7,800 m² at CAF-NT and 10,000 m² at CAF-CT; Table S1). Conversely, the use of the sonic anemometer height (1.91 and 2.21 m at CAF-NT and CAF-CT, respectively) rather than the geometric mean of the gradient inlet heights (1.41 and 1.53 m, respectively), could cause the footprint area to be overpredicted. Although the chamber variability also challenges the MM method requirement of a homogenous emission source over the footprint, we argue that the source should be considered homogenous at the integration scale of hundreds of square meters. Because of the large spatial variability observed in the SC results, and the greater spatial coverage of the FG technique, we will focus on the FG results when interpreting field-scale cumulative emissions at these sites.

4. Discussion

4.1. Cumulative N2O Emissions

The cumulative emissions measured by the FG systems over the 1 April–30 September monitoring period were similar for both sites: 2.4 ± 0.7 and 2.1 ± 0.2 kg N2O-N/ha at the no-till and conventional-till sites, respectively (Figure 6 and Table S1). The cumulative emissions measured by the SC systems over the same six-month period were 1.9 ± 0.14 and 4.8 ± 0.33 kg N2O-N/ha at CAF-NT and CAF-CT. For the period from 1 October 2014 through 30 September 2015, the SC system observations yielded 3.9 ± 0.22 and 5.6 ± 0.34 kg N2O-N/ha at CAF-NT and CAF-CT. The FG and SC cumulative emission magnitudes agree better at CAF-NT than CAF-CT, perhaps because of chamber location relative to the FG footprint, and the patterns of spatial variability at the two sites. The NW–SE wind pattern and hence measurement footprint of the flux gradient system has better overlap with all 16 chambers at CAF-NT than CAF-CT (Figure 5). Furthermore, at CAF-CT, there was a gradient in emission strength with SC group (Figure S13), indicating higher emissions in up slope than downslope locations. Although both sites had sloping topography (Figure S9), the chambers spanned a steeper gradient at CAF-CT than CAF-NT: 12.5% and 8% grades, respectively. Because soil moisture, chemistry, and other characteristics were not measured at the chamber sites in this study, we are not able to quantify these effects and explain the cause of the spatial pattern. Another factor that could have influenced the hierarchy in chamber emissions at CAF-CT is the potential overapplication of fertilizer. The CAF-CT site is a private, working farm, and we were not able to remove the chambers before planting and fertilization at this site, necessitating the grower to maneuver around the field trailer and other research equipment. The area avoided by the grower was hand-planted and fertilized days after, and the difference in management is visible in the satellite image (Figure 5). Similarly, the placement of the chambers relative to the spacing of the fertilizer bands was not considered in the study design. These potential departures from characterizing nominal field conditions with the SC in our study highlight the value of quantifying soil chemistry, in addition to the need for close collaboration with growers when conducting research on working agricultural fields.

The FG cumulative emissions correspond to 2.1% ± 0.6% and 1.5% ± 0.1% of the N inputs at CAF-NT and CAF-CT, respectively (Table S1), higher than the IPCC Tier 1 emission factor (EF) at CAF-NT but comparable at CAF-CT. If the 20 August fertilization in preparation for winter wheat seeding is included in the EF calculation at CAF-CT, it is 0.7% ± 0.07%, smaller than the IPCC Tier 1 EF by 33%. Evidence suggest that these EF estimates are likely conservative. First, they do not include the period of the crop year between 1 October and 1 April, a time which the SC results show that there were appreciable N2O fluxes (Figure 2).
Second, the comparison between $K_T$ and $K_a$ (Figure S10) indicated that $K_T$ was systematically lower than $K_a$, suggesting that within the range of uncertainty $K_T$ is more likely biased low than high. Lastly, many N$_2$O studies define the emission factor as a function of the fertilizer N input alone. Using that parameterization raises the EFs to 2.8% and 2.1% for CAF-NT and CAF-CT per the FG results.

The N$_2$O emission magnitudes observed in this study fall within the range of emissions reported from other systems that receive a similar amount of fertilizer N input (Figure S14). As discussed in Huang et al. (2014), there is large range in N$_2$O emissions reported in the literature, but the amount of N fertilizer applied tends to be the most important driver of emissions ($R^2 = 0.48$ in the Huang regression, $R^2 = 0.37$ including results from this study). Huang et al. (2014) included results from five studies that measured N$_2$O emissions over the growing season only (duration of three to six months). This may cause estimates of cumulative annual N$_2$O emissions to be biased low, as the results from this study illustrate.

### 4.2. Relationship Between N$_2$O Emissions and Environmental Variables

We observed extended periods of elevated N$_2$O emissions following spring rains. Rain can flush existing N$_2$O from soil pores (Huang et al., 2014), and the added moisture can create anaerobic microsites in soil pores that are hot spots for denitrification, leading to elevated N$_2$O production and emission (Pihlatie et al., 2005). The emission events following the 12–16 May rainfall at both sites lasted for approximately a month (Figure 2), indicating that the rainfall lead to enhanced production of N$_2$O at these sites.

Conversely, the one-day spike in emissions after the 10 July rainfall event was likely a physical flush of N$_2$O-rich air from the soil pore space.

Both N$_2$O emission “spikes” of approximately one day and “pulses” lasting weeks (coined by Cai et al. (2013)) have been observed in other N$_2$O monitoring studies. Studies have reported emission spikes following rainfall lasting less than one day in agroecosystems (Barton et al., 2011; Glenn et al., 2012; Huang et al., 2014), spikes lasting two to six days following fertilization (Jones et al., 2011), elevated N$_2$O emissions lasting on the order of two weeks following fertilization and rainfall (Cai et al., 2013; Tallec et al., 2019; Zhou et al., 2013), and emission pulses lasting two months in grassland systems following tillage (Cowan et al., 2016). In the context of these prior studies, the monthlong periods of sustained elevated emissions observed at both cropland sites in this study are important for monitoring, understanding, and predicting future N$_2$O emissions from these systems.

Both the conventional tillage and no-tillage site N$_2$O emissions responded similarly to temperature. Linear regressions between mean daily temperature and mean daily N$_2$O fluxes from 1 April to 11 May 2015 measured by the FG system for each site have similar slopes (2.14 and 2.23 at CAF-NT and CAF-CT, respectively; Figure 7), indicating that emissions respond similarly to changes in temperature, absent other environmental drivers. We truncated the data set at 11 May 2015 to minimize the influence of the 12–15 May precipitation event (affecting soil moisture) on the measured N$_2$O fluxes.

While CO$_2$ respiration is not a direct driver of N$_2$O production and emission, the same environmental conditions (e.g., soil temperature, moisture, and microbial activity) can have similar influences on the production and emission of both CO$_2$ and N$_2$O (Knox et al., 2016; Morin et al., 2014). Linear regressions between N$_2$O emission and CO$_2$ respiration in the winter (1 October 2014 to 1 March 2015) have similar slopes (0.19 and 0.25 at CAF-NT and CAF-CT, respectively), but the relationship is stronger at CAF-NT than CAF-CT, with an adjusted $R^2$ value of 0.55 versus 0.26 (Figure S15). Figures S16–S20 illustrate how the relationships between N$_2$O and CO$_2$ emissions break down by individual chamber and by winter and growing season time periods.
4.3. Implications of N$_2$O Emission Results for Overall N and GHG Budgets

The emission rates observed in this study are comparable to literature values (Huang et al., 2014; Figure S14). However, we are interested in increasing our knowledge of local N$_2$O emission behavior. Prior to this study, there was only one published article reporting direct N$_2$O emission measurements in the iPNW region (Cochran et al., 1981), which found that <0.1% of fertilizer N was emitted as N$_2$O. A more recent study used a process-based model to simulate N$_2$O emissions for several representative areas and management practices in the iPNW, including scenarios co-located with the CAF-NT and CAF-CT sites (Stöckle et al., 2012). The simulation results found these systems were sources of N$_2$O at a rate of 0.72 to 0.98 kg N$_2$O-N ha$^{-1}$ yr$^{-1}$, and had emission factors of 0.71% to 1.1% of applied fertilizer N. These simulation study results agree reasonably well with the IPCC Tier 1 estimates, and with the conservative EF calculated for CAF-CT but are lower than the upper EF estimates of this study by a factor of 2.

The difference between the N$_2$O emission results from this study and previous best estimates has important implications when propagated to the GHG budget of the systems. Typical rates of increase in soil organic carbon after converting to no-tillage management for continuous cropping systems have been reported at 25 g C m$^{-2}$ yr$^{-1}$ (West & Post, 2002), 20 g C m$^{-2}$ yr$^{-1}$ (Brown & Huggins, 2012), and 16–23 g C m$^{-2}$ yr$^{-1}$ (Stöckle et al., 2012). The global warming potential of N$_2$O emissions characterized in this study is on the high end of typical soil C uptake, equivalent to 27 ± 8 g C m$^{-2}$ yr$^{-1}$ per the FG results at CAF-NT, and 24 ± 2 g C m$^{-2}$ yr$^{-1}$ per the FG results at CAF-CT. The rate of C uptake in soils after converting to no-till lasts 15–20 years with diminishing returns, while N$_2$O emissions persist over time.

5. Summary and Conclusions

In this study, automated SC and the FG technique were used to monitor N$_2$O emissions over two canola fields under different tillage regimes in the inland Pacific Northwest. An array of 16 automated static chambers was operated at each site over the entire 2015 crop year, while results from the FG measurements were available from 1 April to 30 September 2015. Maximum emissions were observed at both sites in response to multiple rain events following fertilization, and the elevated emissions lasted on the order of one month at both sites. Ambient temperature as well as CO$_2$ respiration were positively correlated with N$_2$O emissions at both sites. The FG results indicated that total emissions were 2.4 ± 0.7 and 2.1 ± 0.2 kg N$_2$O-N/ha, respectively, from the no-till and conventional till sites. This translates to N$_2$O emission factors of 2.1% and 1.5% respectively, likely conservative estimates due to the six-month duration of the measurements and other biases discussed in section 4.1.

The results of this study highlight the limitations inherent in characterizing field-scale fluxes with SC point measurements. We reemphasize the recommendation of other groups (cf. Jones et al., 2011; Tallec et al., 2019): rely on micrometeorological methods or other methods that integrate over larger spatial scales to answer questions about field-scale processes (i.e., total cumulative N$_2$O emissions) and use SC to validate the observed temporal patterns. The SC measurements in tandem with co-located measurements of environmental variables can answer questions about biophysical drivers of N$_2$O production and emission useful for parameterizing models. There are plans to continue monitoring N$_2$O fluxes at paired conventional tillage and no-tillage sites as part of the USDA Long-Term Agroecosystem Research program at Cook Agronomy Farm, and this future work should include measurements of soil texture, moisture, mineral N, and other factors that affect N$_2$O production and emission co-located with SC to aid in the interpretation and upscaling of both SC and FG results. Identifying the N$_2$O production pathways and parameterizing N$_2$O:N$_x$ product ratios for these systems should be prioritized to determine the gaseous loss term of the N budget (cf. Butterbach-Bahl et al., 2013; Wragge et al., 2001). If denitrification is a dominant production pathway, total losses of N as N$_2$O + N$_2$ could be an agronomically important part of the N budget in this region. For systems such as the ones in this study with high spatial heterogeneity, combining measurements and process-based models may be the best strategy to estimate field-scale and regional emissions of N$_2$O. Future SC studies investigating drivers could be used to improve the parameterization of N$_2$O dynamics in process-based models such as CropSyst (Stöckle et al., 2003), and FG or eddy covariance field-scale results could be used to constrain the cumulative emissions.
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