Agriculture, Ecosystems and Environment 348 (2023) 108413

Contents lists available at ScienceDirect

Agriculture, Ecosystems and Environment

journal homepage: www.elsevier.com/locate/agee

Growing and non-growing season nitrous oxide emissions from a manured semiarid cropland soil under irrigation

Robert S. Dungan [a,*], April B. Leytem [a], Amber D. Moore [b], David L. Bjorneberg [a], Peter R. Grace [c], Christian Brunk [c], David W. Rowlings [c]

[a] USDA-ARS, Northwest Irrigation & Soils Research Laboratory, Kimberly, Idaho 83341, USA
[b] Oregon State University, Department of Crop and Soil Science, Corvallis, OR 97331, USA
[c] Queensland University of Technology, Centre for Agriculture and the Bioeconomy, Brisbane, QLD 4000, Australia

ARTICLE INFO

Keywords:
Emission factor
Irrigated
Manure
Nitrous oxide
Non-growing season
Semiarid

ABSTRACT

Dairy manure is used in semiarid southern Idaho to improve soil fertility, but campaigns to measure resulting nitrous oxide (N_2O) emissions over the complete year have not been conducted to date. The objective of this study was to measure N_2O fluxes throughout the growing (April to Sept) and non-growing (Oct to Mar) seasons in 2020 (sugarbeet) and 2021 (silage corn and triticale) in a field that received inorganic N fertilizer or was previously treated with dairy manure solids on an annual and biennial basis for 8 years. Gas fluxes were measured daily using automated chambers that were connected to a gas chromatograph for in situ analysis of N_2O. The N_2O emissions were found to be highly episodic and major pulses were associated with irrigation during the growing season, warming events in the winter, and soil disturbance at harvest. Emissions were greatest from soil that had received manure at the highest annual application rate of 52 Mg ha^{-1} (dry wt.), with cumulative totals of 3.6 and 3.0 kg N_2O-N ha^{-1} in 2020 and 2021, respectively. These cumulative totals were about 3-fold greater than emissions from plots treated with inorganic fertilizer or manure at 17 Mg ha^{-1} annually or 35 Mg ha^{-1} biennially. This outcome can be attributed to high concentrations of nitrate produced through mineralization of organic N in manure. Emission factors indicated that up to 1.2% of the total N applied was lost as N_2O-N, with the greatest loss from inorganic fertilizer treated soil. When breaking down the emissions by season, anywhere from 49%–63% (2020) and 37–58% (2021) of the N_2O-N emissions occurred during the non-growing season. Growing and non-growing season N_2O emissions were found to be statistically equivalent for each of the respective fertilizer or manure treatments. This finding stresses the need to also measure N_2O emissions during the non-growing season as a way to improve the accuracy of annual emission estimates.

1. Introduction

Manure from concentrated animal feeding operations is commonly applied to cropland soils for its fertilizer value and ability to improve soil quality (Rayne and Aula, 2020). While animal manures are used as a substitute for inorganic nitrogen (N) fertilizer, their subsequent effect on soil greenhouse gas (GHG) emissions has not been studied in all agroecosystems. Organic and inorganic N fertilizers are essential for maintaining crop yields; however, N application in cropping systems can enhance nitrous oxide (N_2O) emissions. Nitrous oxide is naturally produced by soil microbes during nitrification (aerobic) and denitrification (anaerobic) processes and its formation is influenced by soil mineral N availability (NH_4^+ and NO_3^-), soluble carbon (C), temperature, oxygen status, and moisture content (Galbally et al., 2008; Sanchez-Martin et al., 2008). In addition to fertilization with inorganic N, soil management practices such as irrigation and tillage, can stimulate N mineralization from organic matter, resulting in increased N_2O production when soils are wet or disturbed (Gregorich et al., 2015). According to the Draft Inventory of U.S. Greenhouse Gas Emissions and Sinks (USEPA, 2022), agricultural soil management is responsible for approximately 74% of all anthropogenic N_2O emissions and 5.3% of total GHG emissions in the U.S. It has been reported that cumulative N_2O emissions from soil with surface-applied or incorporated animal manure solids and slurry can lose up to 2% of the total N applied (Clayton et al., 1997; Dungan et al., 2021; Li et al., 2015). Given the high global warming potential of N_2O, at approximately 300, it is essential to

* Corresponding author.
E-mail address: robert.dungan@usda.gov (R.S. Dungan).

https://doi.org/10.1016/j.agee.2023.108413
Received 1 December 2022; Received in revised form 25 January 2023; Accepted 13 February 2023
Available online 24 February 2023
0167-8809/Published by Elsevier B.V.

investigate and properly quantify the influence of soil management strategies on N_2O emissions.

Few studies to date have investigated N_2O emissions in semiarid regions, specifically in irrigated cropping systems that receive cattle manure applications (Dungan et al., 2017; Halvorson et al., 2016; Hao, 2015; Leytem et al., 2019). In the western U.S., dairy manure solids are applied to cropland soils in the form of lot scrapings and compost, which are typically incorporated via tillage. Liquid dairy manure from wastewater ponds, however, is commonly applied to crops via sprinkler irrigation (Dungan, 2014). Although the effect of animal manure on soil does vary, there is strong evidence of beneficial impacts of manure on soil properties, soil fertility, and crop yields (Rayne and Aula, 2020). The benefit of using organic N sources, such as manure, is that N availability could be synchronized with plant N uptake, potentially reducing N_2O emissions (Halvorson et al., 2016). Rate, source, and timing of inorganic and organic N fertilizer are known to affect soil N_2O emissions (Asgedom et al., 2014; Leytem et al., 2019; Snyder et al., 2009). In general, increasing N fertilizer rate in excess of plant demand is associated with greater N_2O emissions, which occurs because more N is available for nitrifying and denitrifying bacteria. In southern Idaho, soil N_2O emissions were found to increase with increasing manure application rate (Leytem et al., 2019). In a related study, Dungan et al. (2021) found that there was no effect of manure application timing (i.e., fall vs. spring) on cumulative N_2O emissions, but emissions were influenced by source and were greater from non-composted than composted manure.

In many plot-scale studies, N_2O emissions are often measured using manual static (or non-steady-state) chambers and their deployment schedule is dependent upon the amount of resources available to perform the work (Charteris et al., 2020). In most situations, manual chambers are only deployed a few days per week at most with one sampling event per day (Mosier et al., 2006), but given the episodic nature of N_2O fluxes, major emission pulses can be missed (de Klein et al., 2020). To overcome this limitation of manual chambers, high frequency continuous measurement techniques, such as automated chamber systems, can improve the temporal integration of N_2O emissions and overall accuracy of emission factor estimates, which has been successfully applied in semiarid settings (Barton et al., 2015). Semiarid soils in southern Idaho are slightly alkaline, contain moderate quantities of calcium carbonate, and experience strong solar radiation and diurnal temperature fluctuations. When these extreme soil conditions are combined with irrigation and fertilization, there is great potential to influence the microbial activity and N cycling and subsequent N_2O emissions. In this study, we used automated chambers (Grace et al., 2020) to measure N_2O gas fluxes throughout the growing (April to Sept) and non-growing (Oct to Mar) season in an irrigated cropping system in Idaho, USA. The primary objective of this research was to quantify total annual N_2O emissions as affected by previous dairy manure treatment, as well as determine the effect of management, climate, and soil conditions on N_2O fluxes. We hypothesized that growing season N_2O emissions would be greater than those during the non-growing season, when more mineral N is typically available, and crops are being irrigated.

2. Materials and methods

2.1. Site description

The N_2O monitoring was conducted in 2020–2021 on an established long-term study that began in 2013 to evaluate the effects of dairy manure application rate and timing on nutrient cycling. The field was located at the USDA-ARS, Northwest Irrigation & Soils Research Laboratory in Kimberly, Idaho, USA (latitude 42.551302° and longitude −114.353983°; elevation 1187 m). This region of southern Idaho has a semiarid climate, consisting of hot dry summers and cool wet winters, with mean annual precipitation of 229 mm and mean annual low and high temperatures of 1.7 °C and 15.6 °C, respectively. Given that most rainfall occurs between Oct and May, irrigation is necessary during the late spring and summer months. Soil at the site is classified as a Portneuf silt loam (coarse-silty, mixed, superactive, mesic Durixerollic Calciorthids) and had the following initial characteristics in the top 30 cm in fall 2012: clay, 315 g kg^{-1}; silt, 538 g kg^{-1}; sand, 147 g kg^{-1}; pH, 7.85; total C, 13.7 g kg^{-1}; organic C, 8.0 g kg^{-1}; and total N, 0.8 g kg^{-1}.

2.2. Soil sampling and analysis

Spring soil sampling occurred in late March or early April with 6 subsamples per plot composited at the 0–30 cm depth. Soils were air-dried, ground, and passed through a 2-mm sieve (US no. 10, Fisher Scientific Co., Hampton, NH, USA) before analysis. Soil NH_4-N and NO_3-N were determined by extraction with 2 mol L^{-1} KCl (5 g soil in 50 mL of 2 mol L^{-1} KCl), shaken for 2 h, filtered and analyzed for NH_4-N and NO_3-N using QuickChem Methods 12–107–06–2-A (NH_4) and 12–107–04–1-B (NO_3) on a Lachat QuickChem 8500 Flow Injection Analysis System (Hach Company, Loveland, CO, USA). Olsen P was determined as bicarbonate-extractable P following Olsen (1954). Soil organic C (SOC) was determined by dichromate oxidation on a microplate spectrophotometer (Bierer et al., 2021). Total C and N were determined by combustion of a 50-mg sample in a FlashEA1112 (CE Elantech, Lakewood, NJ, USA). pH and electrical conductivity (EC) were determined on a 1:1 slurry (Miller et al., 2013). Soil chemical characteristics prior to planting in 2020 and 2021 are presented in Table 1.

2.3. Experimental design

The field study was initiated in 2013 with a 4-year rotation of hard red wheat (*Triticum aestivum* L.), potato (*Solanum tuberosum* L.), malt barley (*Hordeum vulgare* L.), and sugarbeet (*Beta vulgaris* L.). The crop rotation was repeated in 2017, followed by a spring and fall planting of silage corn (*Zea mays* L.) and triticale (× *Triticosecale* Whittmack) in 2021, respectively. There were four replications per treatment, which were arranged in a randomized complete block design and each experimental plot was 18.3 × 12.2 m. From 2012–2020, the treatments consisted of a control (no nutrient application), inorganic fertilizer, and solid dairy manure (obtained from local open-lot dairies) applied in the fall on a biennial or annual schedule at average rates of 17, 35, and 52 Mg ha^{-1} (dry wt.). The manure application rates were representative of what is typically applied in this region of Idaho. Inorganic fertilizer applications were determined via pre-plant soil test nutrient status and University of Idaho nutrient recommendations (Brown et al., 2010; Moore et al., 2009). After broadcasting the manure and fertilizer, it was incorporated within 24 h of application to 15 cm using a tandem disk, followed by roller harrowing. All plots received tillage at the same time for consistency.

The last manure application was fall of 2019, but all plots were soil sampled prior to planting in 2020 and 2021 to determine their nutrient status. Manure N and inorganic fertilizer N application rates, two years before (i.e., 2018–2019) and two years during N_2O monitoring (i.e., 2020–2021), are presented in Table 2. The crops present during N_2O monitoring in 2020 and 2021 were sugarbeet and silage corn/triticale, respectively. The sugarbeets were planted on 7 May and harvested on 8 Oct in 2020; silage corn was planted on 13 May and harvested on 11 Sept in 2021; and triticale was planted on 27 Sept 2021 and harvested the following spring in 2022. The crops were irrigated using a solid set sprinkler system with irrigation application rates determined based on crop water use estimates using the Washington State University Irrigation Scheduler (http://irrigation.wsu.edu/index.php). Crop water use was estimated as evapotranspiration (ET) calculated based on the reference alfalfa ET based on the standardized ASCE Penman-Monteith method and mean crop coefficients for the crop being grown (Allen et al., 1998). After harvest, all plant sub-samples were dried in a constant-temperature forced-draft oven at 60 °C, then weighed daily until the mass stabilized to determine the dry matter fraction. Crop

Table 1
Chemical characteristics of soils at 0–30 cm prior to planting.

Year	Treatment	pH	EC	Total C	Total N	SOC	NO_3-N	NH_4-N	Olsen P
			dS m^{-1}	g kg^{-1}			mg kg^{-1}		
2020	Fert	8.0 a†	0.6 d	13.2 d	1.0 d	8.1 d	11.9 d	3.3 a	13 d
	17 A	7.8 bc	1.3 bc	17.3 c	1.3 c	11.4 c	31.9 bc	2.2 b	52 c
	35 B	7.9 b	1.0 c	16.6 c	1.3 c	11.0 c	25.8 c	2.2 b	48 c
	35 A	7.7 cd	1.5 b	19.8 b	1.7 b	14.4 b	38.7 b	2.6 ab	108 b
	52 A	7.6 d	2.6 a	24.1 a	1.9 a	17.3 a	68.7 a	2.5 ab	178 a
2021	Fert	8.0 a	1.2 c	14.1 d	0.9 d	10.2 d	20.5 d	3.3 a	15 d
	17 A	8.0 a	1.6 bc	18.4 c	1.4 bc	13.7 c	34.7 c	2.3 b	38 c
	35 B	8.0 a	2.0 b	18.3 c	1.3 c	13.3 c	41.7 bc	2.0 b	41 c
	35 A	8.0 a	1.4 bc	21.0 b	1.6 b	17.4 b	44.9 b	2.4 b	102 b
	52 A	8.0 a	2.9 a	26.8 a	2.1 a	22.4 a	76.7 a	2.5 ab	165 a

†Mean values within a column and year followed by the same letter are not significantly different ($P < 0.05$).

Table 2
The amount of inorganic fertilizer N and manure N applied to the plots in the years before (2018–2019) and during (2020–2021) the measurement of N_2O fluxes.

Year	Fertilizer	17A†	35B	35A	52A
			kg manure N ha^{-1}		
2018	–	196	389	386	583
2019	–	243	–	485	734
2020	–	–	–	–	–
2021	–	–	–	–	–
			kg fertilizer N ha^{-1}		
2018	266	236	250	190	144
2019	40	56	–	–	–
2020	90	–	–	–	–
2021	45	–	–	–	–

†Dry manure rate in Mg ha^{-1}; A = annually applied, B = biennially applied.

yields on a dry matter basis are presented in Table 3.

2.4. Measurement of N_2O fluxes

Nitrous oxide emissions were measured daily throughout 2020 (sugarbeet) and 2021 (silage corn/triticale), except for a total of 79 and 53 d, respectively, to accommodate planting and harvesting operations. In 2020, the plots that had received the following treatments were monitored: i) inorganic fertilizer (Fert); ii) dairy manure applied biennially at 35 Mg ha^{-1} (35B); and iii) dairy manure applied annually at 17 Mg ha^{-1} (17A) and 52 Mg ha^{-1} (52A). The same treatments were monitored in 2021, except that measurements for 35B were terminated on 29 April prior to planting operations, then on 25 May, N_2O measurements were commenced in plots that had previously received dairy manure annually at 35 Mg ha^{-1} (35A). Our original intention was to move the chambers from 35B to 35A at the beginning of 2021, but that did not happen due to frozen soil conditions. The change from 35B to 35A was implemented to evaluate an additional rate of annually-applied manure.

The N_2O fluxes were determined using an automated system as described in detail by Rowlings et al. (2012). This system consisted of 12 pneumatically-operated vented non-steady-state chambers, which were linked to an automated gas sampling system and a gas chromatograph for in situ analysis of N_2O. The chamber frame (0.5 m × 0.5 m × 0.15 m) was manufactured with stainless steel and acrylic glass was used to cover the frame and lid openings, except the lids were also covered with reflective insulation to minimize internal heating during their closure. The chamber, which contained a rubber seal along the bottom of the frame, was secured to a stainless steel base that was inserted into the soil to a depth of 0.1 m. One base (0.25 m^2) was installed per plot and it was positioned to cover both the row and inter-row space. The lids were opened during irrigation events to ensure all chambers received the same amount of water. Plants within the chamber were allowed to grow until they reached the height of the lid, then they were trimmed back to ground level until senescence occurred. In one chamber per group, soil moisture sensors (MAS-1, Meter Group, Pullman, WA, USA) were used to monitor moisture at 0–5 cm in the area immediately outside the chamber and resistance temperature sensors (Omega Engineering Inc., Norwalk, CT, USA) were used to monitor the ambient temperature within the chamber. The tubing and sensor cables were all routed to a temperature-controlled trailer, which housed all of the instrumentation.

The N_2O concentrations were determined using a gas chromatograph (model 8610, SRI Instruments, Torrance, CA, USA) equipped with an electron capture detector. To reduce interference from carbon dioxide and moisture on N_2O measurement, a pre-column containing silica coated with sodium hydroxide (Part no. AR169, Alpha Resources, LLC., Stevensville, MI, USA) was installed ahead of the analytical column and changed when the majority of the material was saturated as determined by color change. A full measurement cycle for flux determination occurred over a 60-min period when the lids were closed on 4 chambers. During this time, each chamber was sequentially sampled for 3 min followed by a single-point calibration using a certified gas standard of 0.5 ppm N_2O (Airgas, Durham, NC, USA). This process was repeated at 15-min intervals, thus each chamber was sampled 4 times over the closure period. The lids were then opened and remained in that position for 120 min before commencement of the next cycle. A total of 8 flux measurements were obtained for each chamber per day or 56 fluxes per week. Because the chamber system is configured with three groups of four chambers, we were only able to measure N_2O emissions from three of four replicate plots.

Hourly N_2O fluxes were calculated from the slope of the linear regression of concentrations versus time during the chamber lid closure period. The raw data were processed using an Auto GHG System Flux Calculator (FluxNet 3.3) developed by the Queensland University of Technology (Personal communication, D.W. Rowlings, 2019). The data were corrected for air temperature, atmospheric pressure, and ratio of chamber volume to surface area. Fluxes were set as missing values and not gap-filled if the coefficient of determination was < 0.7. On average, 8.5% of the fluxes were screened out based on this criteria. The detection limit of the system is ~2 μg N_2O-N m^{-2} h^{-2}, but values below this were kept in the dataset as opposed to setting them to zero (Grace et al.,

Table 3
Average crop yields by treatment and year, on a dry weight basis.

Treatment	Sugarbeet (2020)	Silage Corn (2021)
	Mg ha^{-1}	
Fert	21.4 ab†	22.8 b
17A	21.0 b	23.0 ab
35B	22.6 a	23.1 ab
35A	20.4 b	23.8 ab
52A	20.2 b	24.5 a

†Mean values within a column followed by the same letter are not significantly different ($P < 0.05$).

2020). The average daily N_2O flux was calculated by averaging eight or fewer emission measurements for each day, while cumulative N_2O fluxes were calculated by summing the average daily fluxes.

2.5. Calculating emission factors

Nitrous oxide emission factors (EFs) were determined by normalizing the cumulative emission value for each year (minus emissions from non-fertilized control soils) to the mass of N applied (Table 2). For Fert, the EF calculation for each year was based on the amount of inorganic fertilizer N applied in 2020 (90 kg N ha^{-1}) or 2021 (45 kg N ha^{-1}). For manure, the 2020 EF calculation was based on the amount of manure N applied in 2019 (243 kg N ha^{-1} in 17A and 734 kg N ha^{-1} in 52A). For manure treatments in 2021, we wanted to estimate an EF representing the "legacy" effect of manure additions, as manure N will continue to mineralize over multiple years (Tarkalson et al., 2018). In this instance, the EF calculation was based on the amount of manure N applied in 2019, minus the N removal by the sugarbeet roots (117 and 162 kg N ha^{-1} for 17A and 52A, respectively). For this calculation, we did not subtract out N_2O emissions from 2020, as they were quite low compared to the manure N added, thus did not have a significant effect on the final values. Since we did not monitor N_2O emissions from control soils in the present study, we used an average annual value of 350 g N_2O-N ha^{-1} that was obtained from our previous work at this field site (Leytem et al., 2019).

2.6. Statistical analysis

Emissions data were statistically analyzed using the generalized linear mixed model (GLIMMIX) procedure of SAS (SAS Institute, Inc., 2019, Cary, NC) including treatment as a main effect and block as a random effect. The data were analyzed separately for each year and treatment effects were assessed using LSMEANS. Statements of statistical significance were based on a P-value < 0.05.

3. Results and discussion

3.1. Precipitation, soil characteristics, and crop response

During the emissions campaign, precipitation totals were 166 and 208 mm and irrigation totals were 772 and 608 mm in 2020 and 2021, respectively. Respective growing and non-growing season precipitation totals were 33 and 133 mm in 2020 and 18 and 190 mm in 2021. In the manured plots, available N ($NO_3^- + NH_4^+$) in preplant soil samples ranged, on average, from 28 to 71 mg kg^{-1} (109–276 kg N ha^{-1}) in 2020 and 37–79 mg kg^{-1} (144 and 308 kg N ha^{-1}) in 2021 and was positively correlated to the manure application rate (Table 1). All plots previously treated with manure had significantly more available NO_3^- than Fert during both years, with treatment 52A having the greatest NO_3^- concentrations near 70 mg NO_3^- kg^{-1}. The NH_4^+ concentrations were an order of magnitude lower than NO_3^- across all treatments, ranging from 2.0 to 3.3 mg NH_4^+ kg^{-1}, and were only slightly, but significantly elevated in the Fert, 35A, and 52A treatments. Compared to inorganic N fertilizer, organically-bound N is gradually released over time as a result of mineralization; however, the relatively high rates of manure that were repeatedly applied over 8 years have allowed for excessive mineral N to build in the top 30 cm of soil. This is supported by the fact that SOC concentrations were significantly greater in the manured soils as compared to Fert (Table 1). Despite the potential soil health benefits of increased organic matter, the long-term annual application of dairy manure in this study negatively affected sugarbeet yield, as yields in 35A and 52A were significantly lower than in 35B (Table 3). Conversely, silage corn yields were not significantly affected by manure rate, but they were numerically greater in 52A.

3.2. Average daily N_2O fluxes and effect of environmental factors

The average daily N_2O fluxes during calendar years 2020 and 2021 are presented in Figs. 1 and 2, respectively, along with average soil temperatures, volumetric water contents (VWCs), and irrigation and precipitation amounts. In 2020, the effect of manure treatment was evident during both the growing and non-growing seasons, with the largest N_2O pulses occurring in 52A (Fig. 1 a). In early winter, N_2O pulses were associated with warmer than normal ambient temperatures and associated freeze-thaw events, along with periodic rainfall. During the 2020 growing season, the largest N_2O pulses occurred in June and early July and were associated with the commencement of irrigation. The greatest average daily flux of 91 g N_2O-N ha^{-1} d^{-1} occurred during this time, immediately after the second irrigation event. In late fall after harvest, large N_2O pulses were noted in 35B and 52A, which could be related to the soil disturbance that occurs during sugarbeet harvest and subsequent disking to 15 cm. In addition, during sugarbeet harvest, the aboveground biomass is returned to the field and recycled through the soil. This biomass is readily degraded, providing a fresh source of mineral N that can be used by nitrifying and denitrifying bacteria.

Unlike 2020, large N_2O emission pulses in 2021 occurred only during the growing season when the crops were being irrigated (Fig. 2a). In fact, the first 9 major pulses occurred in direct synchronization with the corresponding irrigation events, with each pulse lasting for nearly one week. The largest pulses occurred in 52A, followed by 35A, which is consistent with the amount of available preplant NO_3^- in those treatments. The greatest average daily fluxes occurred during the third irrigation event and were 57 and 125 g N_2O-N ha^{-1} d^{-1} in 35A and 52A, respectively. In 52A, the 9 emission pulses accounted for approximately 55% of the total annual cumulative emissions. In comparison to sugarbeet in 2020, a post-harvest spike in N_2O emissions did not occur in 2021 under corn and it could be attributed to the fact that there were no major soil disturbances associated with corn harvest and tillage was not performed after harvest. Triticale was also planted six days after corn harvest and emerged in late Sept, thus it likely used some of the residual mineral N, making it less available for N_2O production.

The average daily N_2O fluxes in the present study were within ranges reported in semiarid irrigated croplands, both with and without livestock manure treatments (Cui et al., 2012; Ellert and Janzen, 2008; Ghimire et al., 2017; Heller et al., 2010; Liu et al., 2011; Mosier et al., 2006). It is well known that production and emission of N_2O are regulated by availability of substrates (C and N), as well as soil temperature and moisture (Liu et al., 2010; Smith, 1980; Wagner-Riddle et al., 2007). Although semiarid soils are substantially aerobic, making nitrification the main source of N_2O emissions, the use of irrigation in cropping systems can shift soils to become anaerobic and forcing denitrification to occur more regularly (Galbally et al., 2008). Leytem et al. (2019) reported that growing season N_2O emissions in an irrigated semiarid soil were greatest when the VWC was near 0.3 m^3 m^{-3}. In the present study, average growing season VWCs ranged from 0.17 to 0.28 m^3 m^{-3} (2020) and 0.12–0.29 m^3 m^{-3} (2021). A VWC of 0.28 m^3 m^{-3} is approximately a water-filled pore space (WFPS) of 60%, assuming representative bulk and particle densities of 1.4 and 2.65 g cm^{-3} found in our silt loam soil (data not shown). Water-filled pore space is a proxy for soil aeration status and when the WFPS is high, then diffusion of O_2 into the soil is impeded. A WFPS of 60% is considered the lower limit for denitrification, but it increases with increasing soil water content, while a WFPS of 60% is the upper limit for nitrification (Linn and Doran, 1984). However, when the WFPS is greater than 80%, then consumption by denitrification may even decrease soil N_2O fluxes because N_2O is reduced to N_2 (Veldkamp et al., 1998). In soils treated with cattle manure, it was found that denitrification was the principal source of N_2O when the WFPS ranged from about 58–80% (Akiyama et al., 2004). This information suggests that denitrification was likely responsible for most N_2O production immediately after irrigation in the present study, but as the soils dried out between irrigation events, then nitrification could be the

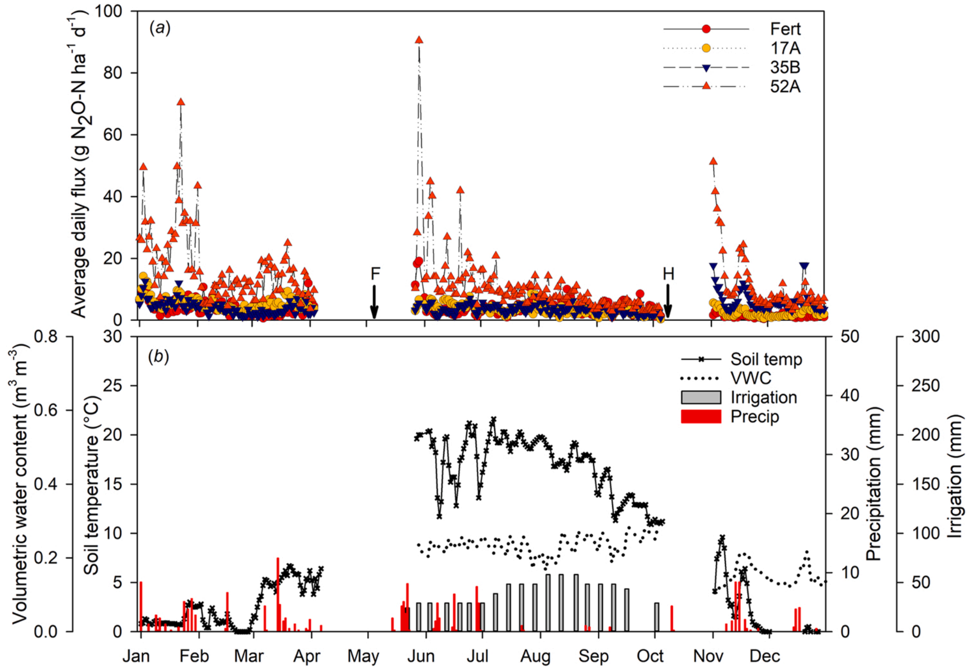

Fig. 1. Temporal pattern of (*a*) average daily N_2O-N fluxes from the fertilized and manured plots during the 2020 emissions campaign and (*b*) soil temperature and volumetric water content (VWC), as well as precipitation and irrigation totals per event. The downward black arrows in Fig. 1a indicate the dates that inorganic fertilizer (F) was applied and plant harvest (H) occurred.

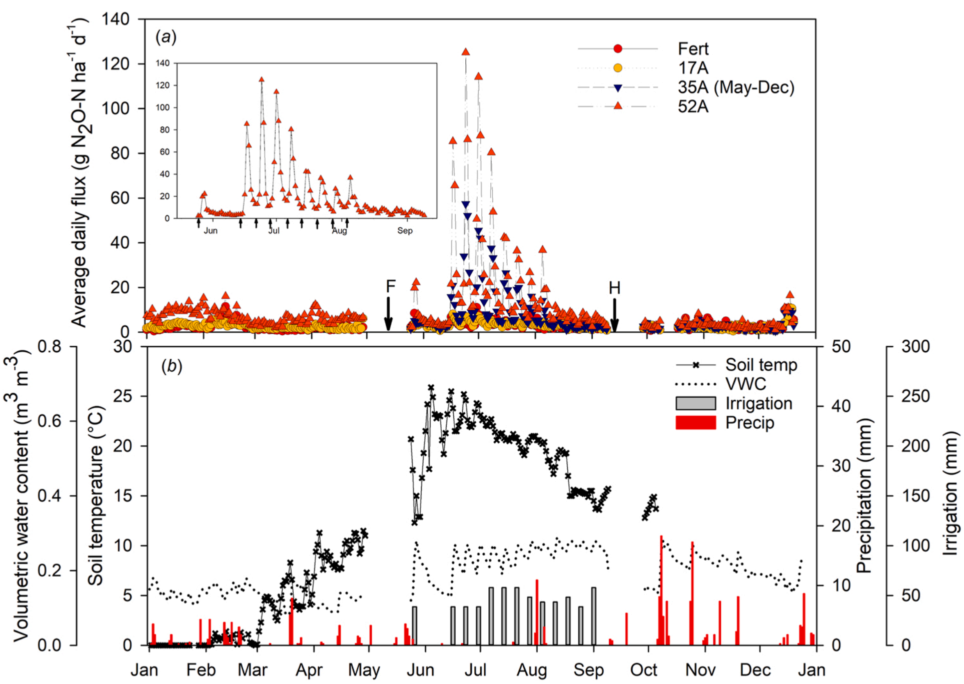

Fig. 2. Temporal pattern of (*a*) average daily N_2O-N fluxes from the fertilized and manured plots during the 2021 emissions campaign and (*b*) soil temperature and volumetric water content (VWC), as well as precipitation and irrigation totals per event. The inset figure in Fig. 2a is showing the relationship between the timing of the irrigation events (upward black arrows) and N_2O-N fluxes in plots previously treated with 52 Mg ha^{-1} of manure per year for 8 years. The downward black arrows in Fig. 2a indicate the dates that inorganic fertilizer (F) was applied and plant harvest (H) occurred.

dominant process with some denitrification occurring in anaerobic microsites.

In semiarid climates, like that found in southern Idaho, winters can be wet and cold, thus freeze-thaw conditions can persist for many weeks during the non-growing season. During the non-growing season in cold climates, denitrification processes tend to be the dominant source of N_2O emissions, especially during thawing events (Ejack and Whalen, 2021; Risk et al., 2013). However, N_2O can also be produced through nitrifier-denitrification (i.e., nitrite reduction by ammonia oxidizers) when conditions are suboptimal for denitrification (Kool et al., 2011; Wrage et al., 2001). Freeze-thaw cycles enhance N_2O emissions due to increased anaerobiosis and substrate availability, change in the structure and activity of denitrifying enzymes, and release of trapped N_2O as ice or snow cover thaws (Burton and Beauchamp, 1994; Goodroad and Keeney, 1984; Ruan and Robertson, 2016; Teepe et al., 2001; Wagner-Riddle et al., 2017). In the latter case, compared to the physical release of ice-trapped N_2O, it was concluded that most N_2O emitted during spring thaws was a result of de novo denitrification (Risk et al., 2013). There is also some evidence to suggest that N in fall-applied manure is biologically acted upon in frozen soils (Ejack and Whalen, 2021), as N_2O losses during the non-growing season under thawing conditions are considerable from manured soils (Chantigny et al., 2016; Kariyapperuma et al., 2012; Wagner-Riddle and Thurtell, 1998). In a study using soils from the present field site, Cassity-Duffey et al. (2018) reported that N mineralization was occurring down to temperatures as low as − 14 to 4 °C, with mineralization being three times greater in manured versus control soils at these temperatures. Unfortunately, efforts to mitigate GHG emissions by using livestock manure to increase organic C storage in soils are likely to be offset by the stimulation of higher N_2O emissions (Gu et al., 2017). Denitrification may occur more intensively when soils receive organic amendments, which provide more labile C for heterotrophic bacteria, resulting in the depletion of O_2 and creation of anaerobic microsites within aggregates that favor N_2O production (Gu et al., 2017; Li et al., 2016; Smith et al., 2003).

3.3. Annual, growing season, and non-growing season N_2O emissions

In 2020, annual cumulative emissions were 995, 1101, 1237, and 3609 g N_2O-N ha^{-1}, on average, from Fert, 17A, 35B, and 52A, respectively (Table 4). When considering N_2O-N emissions during the growing season, they accounted for 51%, 42%, 37%, and 37% of the respective annual emissions. In 2021, annual cumulative emissions were 884, 855, and 2995 g N_2O-N ha^{-1}, on average, from Fert, 17A, and 52A, respectively (Table 4). Growing season emissions accounted for 44%, 42%, and 63% of the cumulative emissions from the respective treatments. Therefore, anywhere from 49%–63% (2020) and 37–58% (2021) of the N_2O-N emissions occurred during the non-growing season, which is a substantial fraction of the annual emissions. When comparing the growing and non-growing season emissions for each treatment and year, it was found that there were no significant differences ($P \geq 0.11$; data not shown). This outcome contradicts our hypothesis and demonstrates the importance of measuring non-growing season N_2O emissions in semiarid irrigated cropping systems. Although large N_2O emission events can be expected during warmer growing seasons, non-growing season emissions can account for a significant fraction of the total annual emissions, which may exceed 50% and have been reported to be as high as 88% of annual emissions in cold climates (Cambareri et al., 2017; Chantigny et al., 2016; Liu et al., 2019; Wagner-Riddle et al., 2007; Wagner-Riddle and Thurtell, 1998). In an arid irrigated cropping system in western China, Lv et al. (2021) found that non-growing season emissions ranged from 300 to 1700 g N_2O-N ha^{-1}, accounting for 28–37% of the annual emissions. In a rainfed cropping system in humid northeastern China, N_2O during the non-growing season ranged from 150 to 220 g N_2O-N ha^{-1}, corresponding to 11–21% of the annual emissions (Chen et al., 2016). Because N_2O emissions are highly episodic and occur throughout the year (Charteris et al., 2020), measurement campaigns should be conducted during both growing and non-growing seasons to accurately determine annual emissions from agricultural fields (Ellert and Janzen, 2008). This may be particularly important when applying an N source post-harvest, as was done with fall-applied manure in 2019, which may have contributed to the 1.3- to 2-fold increase in non-growing season N_2O emissions in 2020 compared to 2021. However, winter measurements may not be as important when soil is frozen and covered with snow (Maljanen et al., 2007; Ruan and Robertson, 2016), but that is a relatively small portion of the total cropland in the U.S.

For plots that had received manure annually at 35 Mg ha^{-1} (i.e., 35A), we could not compare growing and non-growing season emissions as we did not have emissions for the complete year of 2021 (Table 4). It should be noted, however, that growing season N_2O-N emissions from 35A in 2021 were 2-fold greater than from 35B in 2020. This outcome could be attributed to the fact that 35A had received two times more dairy manure than 35B from 2012 to 2019, as well as other factors such as available N, crop, organic matter content, moisture content, and temperature. Although the average NO_3-N concentration was 1.7-fold greater in 35A (2021) versus 35B (2020), soil temperatures during peak fluxes were also generally higher in 2021 (19–26 °C) when 35A was measured compared to 2020 (11–22 °C) when 35B was measured. Higher temperatures in 2021 could have been more favorable to the denitrifying bacteria and diffusion of N_2O produced and trapped in subsoils.

A limited number of field studies have measured N_2O emissions from irrigated cropping systems using cattle manure solids in the western U.S. In a field trial under continuous corn in semiarid Colorado, Halvorson et al. (2016) reported that N_2O-N emissions (averaged over 3 years) during the growing season from urea and dairy manure were 795 and 819 g ha^{-1} and during the non-growing season were 179 and 193 g ha^{-1}, respectively. Based on these results, approximately 80% of the N_2O emissions occurred during the growing season, regardless of the N source. In the Colorado study, dairy manure was applied at an average rate of 41 Mg ha^{-1} over the experimental period, resulting in an average application rate of 412 kg N ha^{-1} y^{-1}. While the manure application rate fell within rates used in the present study, urea was applied at a much higher rate of 179 kg N ha^{-1} y^{-1}. Despite some of these differences, our annual cumulative N_2O emissions from Fert, 17A, and 35B were similar to those obtained by Halvorson et al. (2016). Additional field studies were conducted in southern Idaho that addressed growing season GHG emissions (with some overlap into the non-growing season) where inorganic N fertilizer was compared to dairy manure at an application rate of 52 Mg ha^{-1}. In a corn-barley-3 × alfalfa rotation, N_2O emissions were 3-fold greater from non-composted manure than urea (Dungan et al., 2021, 2017). During the first four years at the present field site, cumulative N_2O emissions were up to 3-fold greater from plots that received annual dairy manure application when compared to inorganic N fertilizer (Leytem et al., 2019). The cumulative growing season N_2O emissions from these Idaho studies ranged from 370 to 1200 g N ha^{-1} (avg = 700) from inorganic N and 460–3500 g N ha^{-1} (avg = 1500) from dairy manure when applied at 52 Mg ha^{-1}.

3.4. N_2O emission factors

The total N lost as N_2O-N emissions during the 2-year campaign was

Table 4
Average cumulative growing season, non-growing season, and annual N_2O-N losses by year and as a percentage of cumulative N applied.

Treatment	Growing		Non-growing		Annual		Relative loss#	
	2020	2021	2020	2021	2020	2021	2020	2021
	g N_2O-N ha^{-1}						%	
Fert	510b†	385b	485b	489bc	995b	884b	0.72a	1.19a
17A	458b	361b	643b	494b	1101b	855b	0.31b	0.40b
35B	458b	-	779b	-	1237b	-	-	-
35A	-	914ab	-	211‡c	-	1125§b	-	-
52A	1344a	1861a	2266a	1094a	3609a	2955a	0.44b	0.46b

†Mean values within a column followed by the same letter are not significantly different ($P < 0.05$).
‡Emissions from 29 Sept to 20 Dec 2021, not the complete non-growing season.
§Emissions from 25 May to 20 Dec 2021.
#Relative loss was calculated as follows: [(Annual N_2O-N loss from the treatments) - (Control N_2O-N emissions [350 g N ha^{-1}])/(Total N applied)].

1879, 1956, and 6564 g ha^{-1} from Fert, 17A, and 52A, respectively (Table 4). Treatment 52A lost nearly 3.4-fold more N than either Fert or 17A, which was a significantly greater amount. In 2020, the EFs for Fert, 17A, and 52A were 0.72%, 0.31%, and 0.44%, respectively. In 2021, the EFs were slightly greater than in 2020, with the greatest EF being for Fert at 1.2%, while both manure treatments were still less than 0.5%. In the 2019 refinement of the Intergovernmental Panel on Climate Change (IPCC) Guidelines for National Greenhouse Gas Inventories, updates for calculating N_2O emissions from managed soils were included. Default EFs for N additions from fertilizer and organic amendments had an aggregated default value of 1% with the option to disaggregate based on climate with an EF of 0.5% for all N inputs in dry climates (IPCC, 2019). The EFs for fertilizer in the present study (0.72 and 1.19) were in line with the aggregated default value in the 2019 refinement but were greater than the disaggregated value of 0.5%, although they were in the uncertainty range (0–1.1%). The EFs for the manure treatments ranged from 0.31 to 0.46 and were less than the aggregated default value but similar to the disaggregated default value, falling within the uncertainty range. The "legacy" EFs were not significantly different than the EFs determined the previous year when manure inputs had occurred. This suggests that the effects of manure on subsequent year N_2O emissions may be underestimated with current Tier 2 methodologies, unless these effects are accurately captured via soil C mineralization.

Overall, the EFs in the present study were similar to those obtained in previous local studies conducted during the growing season, where EFs ranged from 0.05–0.8% in urea- and manure-treated soils (Dungan et al., 2021, 2017). When an emissions campaign was conducted during 2013–2016 at the same field site as the present study, growing season EFs were reported to be about 0.2% (Leytem et al., 2019). In arid cropland studies that have used inorganic fertilizer and livestock manure (cattle and/or sheep), EFs ranged from 0.04–0.15% (growing season only) in irrigated cotton (Kuang et al., 2018) and 0.5–1.0% (growing and non-growing season) in irrigated cotton-maize-wheat (Lv et al., 2021). In a semiarid region of south Alberta, Canada, the N-scaled EF was 0.09% (growing season only) for rainfed barley receiving cattle manure, but the EF was found to be 10-fold greater when the cattle manure was anaerobically digested (Thomas and Hao, 2017). Although we did not measure N_2O emissions from unfertilized soil in the present study, background emissions are present, thus a proportion of N_2O emitted is not directly derived from the application of inorganic or organic N fertilizer. In fact, growing season cumulative N_2O emissions from unfertilized soils were reported to range from 230 to 590 g N ha^{-1} which amounted to 50% of the N_2O-N emissions from urea-treated soil over a 4-year period (Leytem et al., 2019).

4. Conclusions

Many cropland studies have focused their efforts on measuring GHG emissions during the growing season, since this is when most emissions are expected to occur due to higher soil temperature, moisture, and nutrient availability. Using an automated chamber system in the present study, we were able to effectively measure N_2O emissions during the non-growing season as well, when ambient temperatures often go below freezing. The N_2O emissions were found to be highly episodic and major pulses were associated with irrigation during the growing season, warming events in the winter, and soil disturbance at harvest. Soils previously treated with dairy manure at the highest application rate, produced the greatest cumulative emissions when compared to manure applied at lower rates or soils treated with inorganic fertilizer. This outcome can be attributed to high concentrations of preplant NO_3^- in the top 30 cm of soil, produced through mineralization of organic N in manure. Given that crop yields were similar among the different N treatments, N inputs via manure application could be dramatically reduced, thus providing an effective means of minimizing N_2O emissions without affecting soil fertility. To emphasize this point, cumulative N_2O emissions were 3.4-fold lower from 17A than 52A over the two years. In fact, use of inorganic N fertilizer could be completely replaced with 17 Mg ha^{-1} of manure without any crop yield penalty, plus there are potential climate benefits of avoiding nitrification of synthetically-derived ammonium and increasing SOC. Most importantly, this study demonstrated that growing and non-growing season N_2O emissions were equivalent for each of the different treatments. This finding stresses the need to measure N_2O emissions during the non-growing season in semiarid climates as a way to improve the accuracy of annual emission estimates.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

The authors are grateful to Sheryl Verwey, Chad McKinney, and Myles Miller for their assistance with this field study. Mention of trade names or commercial products in the publication is solely for The purpose of providing specific information and does not imply recommendation or endorsement by the USDA. The USDA is an equal opportunity provider and employer.

References

Akiyama, H., McTaggart, I.P., Ball, B.C., Scott, A., 2004. N2O, NO, and NH3 emissions from soil after the application of organic fertilizers, urea and water. Water, Air, Soil Pollut. 156, 113–129.

Allen R.G., L.S. Pereira, D. Raes, M. Smith. 1998. Crop evapotranspiration-Guidelines for computing crop water requirements-FAO Irrigation and drainage paper 56. Fao, Rome. 300:D05109.

Asgedom, H., Tenuta, M., Flaten, D.N., Gao, X., Kebreab, E., 2014. Nitrous oxide emissions from a clay soil receiving granular urea formulations and dairy manure. Agron. J. 106, 732–744.

Barton, L., Wolf, B., Rowlings, D., Scheer, C., Kiese, R., Grace, P., Stefanova, K., Butterbach-Bahl, K., 2015. Sampling frequency affects estimates of annual nitrous oxide fluxes. Sci. Rep. 5, 1–9.

Bierer, A.M., Leytem, A.B., Rogers, C.W., Dungan, R.S., 2021. Evaluation of a microplate spectrophotometer for soil organic carbon determination in south-central Idaho. Soil Sci. Soc. Am. J. 85, 438–451.

Brown B., J. Hart, D. Horneck, A. Moore. 2010. Nutrient management for field corn silage and grain. University of Idoho Extension.

Burton, D.L., Beauchamp, E.G., 1994. Profile nitrous-oxide and carbon-dioxide concentrations in a soil subject to freezing. Soil Sci. Soc. Am. J. 58, 115–122.

Cambareri, G., Drury, C., Lauzon, J., Salas, W., Wagner-Riddle, C., 2017. Year-round nitrous oxide emissions as affected by timing and method of dairy manure application to corn. Soil Sci. Soc. Am. J. 81, 166–178.

Cassity-Duffey, K.B., Moore, A., Satterwhite, M., Leytem, A., 2018. Nitrogen mineralization as affected by temperature in calcareous soils receiving repeated applications of dairy manure. Soil Sci. Soc. Am. J. 82, 235–242.

Chantigny, M.H., Rochette, P., Angers, D.A., Goyer, C., Brin, L.D., Bertrand, N., 2016. Nongrowing season N2O and CO_2 emissions—temporal dynamics and influence of soil texture and fall-applied manure. Can. J. Soil Sci. 97, 452–464.

Charteris, A.F., Chadwick, D.R., Thorman, R.E., Vallejo, A., de Klein, C.A., Rochette, P., Cárdenas, L.M., 2020. Global research alliance N2O chamber methodology guidelines: Recommendations for deployment and accounting for sources of variability. J. Environ. Qual. 49, 1092–1109.

Chen, Z., Ding, W., Xu, Y., Müller, C., Yu, H., Fan, J., 2016. Increased N2O emissions during soil drying after waterlogging and spring thaw in a record wet year. Soil Biol. Biochem. 101, 152–164.

Clayton, H., McTaggart, I., Parker, J., Swan, L., Smith, K., 1997. Nitrous oxide emissions from fertilised grassland: A 2-year study of the effects of N fertiliser form and environmental conditions. Biol. Fertil. Soils 25, 252–260.

Cui, F., Yan, G., Zhou, Z., Zheng, X., Deng, J., 2012. Annual emissions of nitrous oxide and nitric oxide from a wheat–maize cropping system on a silt loam calcareous soil in the North China Plain. Soil Biol. Biochem. 48, 10–19.

Dungan, R.S., 2014. Estimation of infectious risks in residential populations exposed to airborne pathogens during center pivot irrigation of dairy wastewaters. Environ. Sci. Technol. 48, 5033–5042.

Dungan, R.S., Leytem, A.B., Tarkalson, D.D., Ippolito, J.A., Bjorneberg, D.L., 2017. Greenhouse gas emissions from an irrigated dairy forage rotation as influenced by fertilizer and manure applications. Soil Sci. Soc. Am. J. 81, 537–545.
Dungan, R.S., Leytem, A.B., Tarkalson, D.D., 2021. Greenhouse gas emissions from an irrigated cropping rotation with dairy manure utilization in a semiarid climate. Agron. J. 113, 1222–1237.
Ejack, L., Whalen, J.K., 2021. Freeze-thaw cycles release nitrous oxide produced in frozen agricultural soils. Biol. Fertil. Soils 57, 389–398.
Ellert, B.H., Janzen, H.H., 2008. Nitrous oxide, carbon dioxide and methane emissions from irrigated cropping systems as influenced by legumes, manure and fertilizer. Can. J. Soil Sci. 88, 207–217.
Galbally, I.E., Kirstine, W.V., Meyer, C., Wang, Y.P., 2008. Soil–atmosphere trace gas exchange in semiarid and arid zones. J. Environ. Qual. 37, 599–607.
Ghimire, R., Norton, U., Bista, P., Obour, A.K., Norton, J.B., 2017. Soil organic matter, greenhouse gases and net global warming potential of irrigated conventional, reduced-tillage and organic cropping systems. Nutr. Cycl. Agroecosystems 107, 49–62.
Goodroad, L., Keeney, D., 1984. Nitrous oxide emissions from soils during thawing. Can. J. Soil Sci. 64, 187–194.
Grace, P.R., van der Weerden, T.J., Rowlings, D.W., Scheer, C., Brunk, C., Kiese, R., Butterbach-Bahl, K., Rees, R.M., Robertson, G.P., Skiba, U.M., 2020. Global Research Alliance N2O chamber methodology guidelines: considerations for automated flux measurement. J. Environ. Qual. 49, 1126–1140.
Gregorich, E., Janzen, H.H., Helgason, B., Ellert, B., 2015. Nitrogenous gas emissions from soils and greenhouse gas effects. In: Advances in agronomy, 132. Elsevier,, pp. 39–74.
Gu, J., Yuan, M., Liu, J., Hao, Y., Zhou, Y., Qu, D., Yang, X., 2017. Trade-off between soil organic carbon sequestration and nitrous oxide emissions from winter wheat-summer maize rotations: Implications of a 25-year fertilization experiment in Northwestern China. Sci. Total Environ. 595, 371–379.
Halvorson, A.D., Del Grosso, S.J., Stewart, C.E., 2016. Manure and inorganic nitrogen affect trace gas emissions under semi-arid irrigated corn. J. Environ. Qual. 45, 906–914.
Hao, X., 2015. Nitrous oxide and carbon dioxide emissions during the nongrowing season from manured soils under rainfed and irrigated conditions. Geomicrobiol. J. 32, 648–654.
Heller, H., Bar-Tal, A., Tamir, G., Bloom, P., Venterea, R.T., Chen, D., Zhang, Y., Clapp, C. E., Fine, P., 2010. Effects of manure and cultivation on carbon dioxide and nitrous oxide emissions from a corn field under Mediterranean conditions. J. Environ. Qual. 39, 437–448.
IPCC, 2019 (Published). In: Calvo Buendia, E., Tanabe, K., Kranjc, A., Baasansuren, J., Fukuda, M., Ngarize, S., Osako, A., Pyrozhenko, Y., Shermanau, P., Federici, S. (Eds.), 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. IPCC, Switzerland.
Kariyapperuma, K.A., Furon, A., Wagner-Riddle, C., 2012. Non-growing season nitrous oxide fluxes from an agricultural soil as affected by application of liquid and composted swine manure. Can. J. Soil Sci. 92, 315–327.
de Klein, C.A., Harvey, M.J., Clough, T.J., Petersen, S.O., Chadwick, D.R., Venterea, R.T., 2020. Global research alliance N2O chamber methodology guidelines: Introduction, with health and safety considerations. J. Environ. Qual. 49, 1073–1080.
Kool, D.M., Dolfing, J., Wrage, N., Van Groenigen, J.W., 2011. Nitrifier denitrification as a distinct and significant source of nitrous oxide from soil. Soil Biol. Biochem. 43, 174–178.
Kuang, W., Gao, X., Gui, D., Tenuta, M., Flaten, D.N., Yin, M., Zeng, F., 2018. Effects of fertilizer and irrigation management on nitrous oxide emission from cotton fields in an extremely arid region of northwestern China. Field Crops Res. 229, 17–26.
Leytem, A.B., Moore, A.D., Dungan, R.S., 2019. Greenhouse gas emissions from an irrigated crop rotation utilizing dairy manure. Soil Sci. Soc. Am. J. 83, 137–152.
Li, J., Luo, J., Shi, Y., Lindsey, S., Houlbrooke, D., Ledgard, S., 2015. Nitrous oxide emissions from dairy farm effluent applied to a New Zealand pasture soil. Soil Use Manag. 31, 279–289.
Li, X.X., Sorensen, P., Olesen, J.E., Petersen, S.O., 2016. Evidence for denitrification as main source of N2O emission from residue-amended soil. Soil Biol. Biochem. 92, 153–160.
Linn, D.M., Doran, J.W., 1984. Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. Soil Sci. Soc. Am. J. 48, 1267–1272.
Liu, C., Zheng, X., Zhou, Z., Han, S., Wang, Y., Wang, K., Liang, W., Li, M., Chen, D., Yang, Z., 2010. Nitrous oxide and nitric oxide emissions from an irrigated cotton field in Northern China. Plant Soil 332, 123–134.
Liu, C., Hu, X., Zhang, W., Zheng, X., Wang, R., Yao, Z., Cao, G., 2019. Year-round measurements of nitrous oxide emissions and direct emission factors in extensively managed croplands under an alpine climate. Agric. For. Meteorol. 274, 18–28.
Liu, Y.T., Li, Y.E., Wan, Y.F., Chen, D.L., Gao, Q.Z., Li, Y., Qin, X.B., 2011. Nitrous oxide emissions from irrigated and fertilized spring maize in semi-arid northern China. Agric. Ecosyst. Environ. 141, 287–295.
Lv, J., Yin, X., Dorich, C., Olave, R., Wang, X., Kou, C., Song, X., 2021. Net field global warming potential and greenhouse gas intensity in typical arid cropping systems of China: A 3-year field measurement from long-term fertilizer experiments. Soil Tillage Res. 212, 105053.
Maljanen, M., Kohonen, A.R., Virkajarvi, P., Martikainen, P.J., 2007. Fluxes and production of N2O, CO_2 and CH4 in boreal agricultural soil during winter as affected by snow cover. Tellus Ser. B-Chem. Phys. Meteorol. 59, 853–859.
Miller, R.O., Gavlak, R., Horneck, D., 2013. Soil, Plant and Water Reference Methods for the Western Region, fourth ed. Colorado State University,, Fort Collins, CO, USA, p. 155.
Moore, A., Stark, J., Brown, B., Hopkins, B., 2009. Sugar beets. Southern Idaho Fertilizer Guide. Sugar beets. Curr. Inf. Ser. 1174.
Mosier, A.R., Halvorson, A.D., Reule, C.A., Liu, X.J., 2006. Net global warming potential and greenhouse gas intensity in irrigated cropping systems in northeastern Colorado. J. Environ. Qual. 35, 1584–1598.
Olsen S.R. Estimation of available phosphorus in soils by extraction with sodium bicarbonate: US Department of Agriculture, 1954.
Rayne, N., Aula, L., 2020. Livestock manure and the impacts on soil health: a review. Soil Syst. 4, 64.
Risk, N., Snider, D., Wagner-Riddle, C., 2013. Mechanisms leading to enhanced soil nitrous oxide fluxes induced by freeze–thaw cycles. Can. J. Soil Sci. 93, 401–414.
Ruan, L., Robertson, G.P., 2016. Reduced snow cover increases wintertime nitrous oxide (N2O) emissions from an agricultural soil in the Upper U.S. Midwest. Ecosystems 20, 917–927.
Sanchez-Martin, L., Vallejo, A., Dick, J., Skiba, U.M., 2008. The influence of soluble carbon and fertilizer nitrogen on nitric oxide and nitrous oxide emissions from two contrasting agricultural soils. Soil Biol. Biochem. 40, 142–151.
Smith, K., 1980. A model of the extent of anaerobic zones in aggregated soils, and its potential application to estimates of denitrification 1. J. Soil Sci. 31, 263–277.
Smith, K.A., Ball, T., Conen, F., Dobbie, K.E., Massheder, J., Rey, A., 2003. Exchange of greenhouse gases between soil and atmosphere: interactions of soil physical factors and biological processes. Eur. J. Soil Sci. 54, 779–791.
Snyder, C.S., Bruulsema, T.W., Jensen, T.L., Fixen, P.E., 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agric., Ecosyst. Environ. 133, 247–266.
Tarkalson, D.D., Bjorneberg, D.L., Lentz, R.D., 2018. Effects of manure history and nitrogen fertilizer rate on sugar beet production in the Northwest US. Crop. Forage Turfgrass Manag. 4.
Teepe, R., Brumme, R., Beese, F., 2001. Nitrous oxide emissions from soil during freezing and thawing periods. Soil Biol. Biochem. 33, 1269–1275.
Thomas, B.W., Hao, X., 2017. Nitrous oxide emitted from soil receiving anaerobically digested solid cattle manure. J. Environ. Qual. 46, 741–750.
Veldkamp, E., Keller, M., Nuñez, M., 1998. Effects of pasture management on N2O and NO emissions from soils in the humid tropics of Costa Rica. Glob. Biogeochem. Cycles 12, 71–79.
Wagner-Riddle, C., Thurtell, G., 1998. Nitrous oxide emissions from agricultural fields during winter and spring thaw as affected by management practices. Nutr. Cycl. Agroecosystems 52, 151–163.
Wagner-Riddle, C., Furon, A., Mclaughlin, N.L., Lee, I., Barbeau, J., Jayasundara, S., Parkin, G., Von Bertoldi, P., Warland, J., 2007. Intensive measurement of nitrous oxide emissions from a corn-soybean-wheat rotation under two contrasting management systems over 5 years. Glob. Change Biol. 13, 1722–1736.
Wagner-Riddle, C., Congreves, K.A., Abalos, D., Berg, A.A., Brown, S.E., Ambadan, J.T., Gao, X., Tenuta, M., 2017. Globally important nitrous oxide emissions from croplands induced by freeze–thaw cycles. Nat. Geosci. 10, 279–283.
Wrage, N., Velthof, G.L., Van Beusichem, M.L., Oenema, O., 2001. Role of nitrifier denitrification in the production of nitrous oxide. Soil Biol. Biochem. 33, 1723–1732.