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Greenhouse gas emissions from an irrigated cropping rotation with dairy manure utilization in a semiarid climate

Robert S. Dungan | April B. Leytem | David D. Tarkalson

U.S. Department of Agriculture, Agricultural Research Service, Northwest Irrigation & Soils Research Laboratory, 3793 North 3600 East, Kimberly, ID 83341, USA

Correspondence
U.S. Department of Agriculture, Agricultural Research Service, Northwest Irrigation & Soils Research Laboratory, 3793 North 3600 East, Kimberly, ID, 83341, USA.
Email: robert.dungan@usda.gov

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Abstract

This long-term study was established to increase knowledge of greenhouse gas (GHG) emissions from irrigated cropping systems utilizing dairy manure solids and compost in semiarid southern Idaho. The objective of this field study was to determine the effect of synthetic N fertilizer (urea or SuperU [enhanced-efficiency synthetic fertilizer]), composted dairy manure, dairy manure (fall or spring applied), and a control (no fertilizer or manure) on nitrous oxide (N_2O), carbon dioxide (CO_2), and methane (CH_4) emissions over the growing season. The fertilizer and manure treatments were not applied to alfalfa (*Medicago sativa* L.) (2017) but were applied to corn (*Zea mays* L.) (2018; except SuperU) and barley (*Hordeum vulgare* L.) (2019). Cumulative N_2O losses over the 3 yr ranged from 2.8 to 5.2 kg N_2O-N ha^{-1}, with the fall and spring manure emitting the greatest amounts of N_2O. Emission factors indicated that up to 0.79% of the total N applied was lost as N_2O-N during the growing seasons. Cumulative losses of CO_2 and CH_4 across the rotation were on average 12,170 kg CO_2–C ha^{-1} and –0.77 kg CH_4–C ha^{-1}, respectively, with no significant differences among the treatments. Major N_2O pulses were associated with early-season irrigation events and incorporation of fertilizer and manure, but overall fluxes tended to be the greatest when soil temperatures were higher. Dairy manure and compost applications were also found to cause rapid and significant increases in soil organic carbon (SOC) in the top 30 cm of soil under corn and barley. Despite the fact that manure does cause elevated soil N_2O emissions, it should be considered as an alternative to synthetic fertilizer use due to its ability to increase SOC and potentially help reduce the global warming potential.

1 | INTRODUCTION

Animal manure applications to croplands are beneficial because they increase soil organic carbon (SOC), improve soil health, and sustain crop nutrition with N. However, animal manures can significantly influence the amount of nitrous oxide (N_2O) and carbon dioxide (CO_2) emitted to the atmosphere from soil. Regardless of fertilizer source (synthetic vs. organic), optimal management of N is necessary to maintain crop yield and minimize ammonia and greenhouse gas (GHG) emissions. While N_2O is produced in soils through the microbial processes of nitrification and denitrification, its emissions are enhanced by the availability of mineral N. Mineral

Abbreviations: DM, dry matter; EF, emission factor; EI, emission intensity; ET, evapotranspiration; GHG, greenhouse gas; GWP, global warming potential; SOC, soil organic carbon; VWC, volumetric water content

N becomes available during the decomposition of soil organic matter, plant residues, animal manures, and other organic materials, as well as from the application of synthetic N fertilizers. Carbon dioxide emissions result from organic matter mineralization and plant root respiration, with about equal emissions from both sources (Hanson et al., 2000). Compared to CO_2, N_2O has a global warming potential (GWP) of nearly 300 times greater and is one of the most potent GHGs. In the United States, agricultural soil management accounts for approximately 75% of total N_2O emissions (direct and indirect) and 5.2% of total GHG emissions from anthropogenic sources (USEPA, 2021).

Because production of GHGs is driven by microorganisms, the gas emission rates are highly dependent upon soil organic matter properties, fertilizer type, available C and N, and soil conditions (e.g., moisture content, temperature, pH). Soil N_2O and CO_2 emissions do occur throughout the year, but higher fluxes during the growing season are related to precipitation and temperature, respectively (Almaraz et al., 2009). In semiarid southern Idaho, daily N_2O fluxes from irrigated crops were highest immediately after the first irrigation event, with CO_2 emissions increasing soon after and staying elevated for up to a few months when soil temperatures were highest (Dungan, Leytem, Tarkalson, Ippolito, & Bjorneberg, 2017; Leytem, Moore, & Dungan, 2019). Carbon dioxide emissions are elevated under warmer soil temperatures due to increased decomposition of organic matter (Oertel, Matschullat, Zurba, Zimmermann, & Erasmi, 2016). A high soil moisture content favors the activity of denitrifying bacteria when pore spaces fill with water, resulting in anaerobic conditions and production of N_2O (Keeney, Fillery, & Marx, 1979; Maag & Vinther, 1996; Skiba & Smith, 1993). The process of denitrification is stimulated when C availability increases and NO_3^-concentrations are high (Decock, 2014). Nitrification on the other hand produces N_2O as a byproduct, which occurs under aerobic conditions when soils are relatively dry (Goodroad & Keeney, 1984). In semiarid regions, nitrification is believed to be the main source of N_2O emissions as soils are rarely anaerobic for denitrification (Barton, Gleeson, Maccarone, Zúñiga, & Murphy, 2013; Galbally, Kirstine, Meyer, & Wang, 2008). However, under drying–wetting events caused by irrigation, the pulse of N_2O that occurs immediately afterwards can be a result of denitrification (Mosier, Guenzi, & Schweizer, 1986).

Methane (CH_4) is another important GHG, which has a GWP 25 times greater than CO_2. The production and consumption of CH_4 in soil (which can sometimes occur simultaneously) is facilitated by bacteria, specifically methanogens and methanotrophs, respectively (Conrad, 1996). In contrast to flooded soils, aerobic soils are a sink of atmospheric CH_4 because of microbial oxidation, and the absence of this sink in temperate soils would cause atmospheric concentrations to increase approximately 1.5 times the current rate (Ojima, Valentine, Mosier, Parton, & Schimel, 1993). The highest CH_4 oxidation rates occur in undisturbed forests and grasslands (in both temperate and tropical zones), while adoption of agricultural practices substantially reduces the oxidation rates in aerobic soils (Hütsch, 1998; Ojima et al., 1993; Priemé, Christensen, Dobbie, & Smith, 1997; Smith & Conen, 2004). In U.S. and European soils, the CH_4 oxidation rate was determined to be 60–90% lower in cropland soils than in native grassland and forest soils (Bronson & Mosier, 1993; Dobbie et al., 1996; Powlson, Goulding, Willison, Webster, & Hütsch, 1997; Robertson, Paul, & Harwood, 2000). Nitrogen fertilization is another key factor known to inhibit CH_4 oxidation in soils, specifically when NH_4^+-based fertilizers are utilized (Hütsch, 2001; Mosier, Schimel, Valentine, Bronson, & Parton, 1991; Steudler, Bowden, Melillo, & Aber, 1989)

Except for a large number of studies conducted in Colorado (Alluvione, Halvorson, & Del Grosso, 2009; Bronson & Mosier, 1993; Halvorson, Del Grosso, & Stewart, 2016; Hutchinson & Mosier, 1979; Mosier et al., 1986), very few studies have reported GHG emissions from irrigated cropping systems in other semiarid regions of the western United States (Dungan et al., 2017; Leytem et al., 2019). Given the significant contribution of agriculture to global climate change, there is a need to improve our accounting of GHG emissions and understanding of these emissions with respect to N fertilizer management, irrigation, soil type, and regional climactic differences, especially if effective mitigation strategies are to be developed. The objective of this research was to determine the effect of synthetic N fertilizer, including use of an enhanced-efficiency fertilizer (i.e., SuperU), and composted dairy manure and fall vs. spring dairy manure at typical application rates for southern Idaho on GHG losses over three growing seasons under alfalfa (*Medicago sativa* L.), silage corn (*Zea mays* L.), and barley (*Hordeum vulgare* L.). This is the second report from this ongoing long-term field study, thus an additional purpose was to compare the present results (2017−2019) with that reported by Dungan et al. (2017), which covered GHG emissions during 2013–2015.

Core Ideas

- Greenhouse gas emissions data is needed for irrigated cropping systems that use dairy manure.
- Growing season N_2O emissions were the greatest from fall and spring manure treatments.
- Less than 0.8% of the total N added was lost as N_2O-N.
- Cumulative CO_2 and CH_4 emissions were similar among the manure and synthetic N treatments.

TABLE 1 Physical and chemical properties of the dairy manure and compost by year. Concentrations are reported on a dry weight basis

Year	Season applied	Treatment	Moisture	Dry matter	Total C	Total N	Total P	Total K	C/N
			%	g kg^{-1}					
2017	Fall	Manure	56	438	174	12.5	4.5	25.3	14
	Fall	Compost	25	747	132	11.4	2.7	8.7	12
2018	Spring	Manure	27	727	57.8	4.1	2.1	14.0	14
	Fall	Manure	45	554	119	9.7	4.6	21.5	12
	Fall	Compost	11	886	59.9	5.2	3.0	13.7	12
2019	Spring	Manure	56	445	57.8	4.1	7.6	18.0	14

2 | MATERIALS AND METHODS

2.1 | Field site

This field site is affiliated with the USDA-ARS Greenhouse Gas Reduction through Agricultural Carbon Enhancement Network (GRACEnet) (Jawson, Shafer, Franzluebbers, Parkin, & Follett, 2005) and it is located in Kimberly, ID. Emission measurements were conducted from 2017 to 2019 during the spring, summer, and fall, which we call the "growing season" in this study. Field site details, as well as GHG emissions from 2013–2015 are reported in Dungan et al. (2017). In brief, soil at the site is a Portneuf silt loam (coarse-silty, mixed, superactive, mesic Durinodic Xeric Haplocalcid) and had not received manure applications from 2000 to 2011, until the project was initiated in late 2012. The experimental design was a randomized complete block with four replications. Each plot was 21.3 by 22.9 m and 15-m buffer strips were placed between each block.

2.2 | Soil and manure collection and analysis

Post-harvest soil samples (three cores per plot) were collected prior to manure application in the summer or fall at depths of 0–15 and 15–30 cm using an AMS 9110-AG probe (AMS Inc., American Falls, ID). After collection, the cores were processed by compositing the soil layers from each plot, followed by air drying, grinding, and passage through a fine sieve. The soil was analyzed for total C by combustion of a 50-mg sample in a FlashEA1112 NC Analyzer (CE Elantech, Lakewood, NJ). Soil organic C was determined using the Walkley–Black method (Nelson & Sommers, 1996; Souza, Morais, Matsushige, & Rosa, 2016).

Manure and compost samples were collected from each plot by randomly placing three trays (0.5 by 0.6 m) within the plots during application. Following application, samples from trays were composited and a subsample was taken from each plot. The subsamples were then freeze-dried to remove moisture, followed by grinding. Manure and compost moisture contents were gravimetrically determined by oven drying overnight at 105 ˚C. Total C and N was determined by combustion of a 50-mg sample in a FlashEA1112 NC Analyzer (CE Elantech). Total P and K were determined via digestion of a 0.5-g sample with nitric/perchloric acid and measurement by inductively coupled plasma optical emission spectrometry (PerkinElmer Optima 7300 DV, PerkinElmer, Waltham, MA). The manure and compost properties are listed in Table 1.

2.3 | Synthetic fertilizer and manure treatments

The treatments were: (a) no fertilizer (control); (b) granular urea (urea); (c) SuperU [stabilized granular urea with urease {N-[n-butyl]-thiophosphoric triamide) and nitrification (dicyandiamide) inhibitors}; (d) composted dairy manure applied in the fall with supplemental granular urea applied in the spring (compost); (e) dairy manure applied in the fall (fall manure); and (f) dairy manure applied in the spring (spring manure). Prior to this study, the last manure and synthetic fertilizer applications occurred in spring 2014, then the field was in alfalfa production from 2015 to 2017 without any nutrient additions. In fall 2017, manure and compost were applied, followed by manure and synthetic fertilizer in spring 2018. This application cycle was repeated once again in the fall and spring of 2018 and 2019, respectively.

The manure and compost were applied at 56 and 34 Mg ha^{-1} (dry wt. basis), which are rates typical for this region, while synthetic fertilizers were applied at agronomic rates based on spring soil N test data. Table 2 summarizes the amount of manure C and N and synthetic N for each treatment and the timing of the applications. In spring 2018, only monoammonium phosphate and potassium chloride were applied to the urea and SuperU (24 kg N, 53 kg P, and 209 kg K ha^{-1}) and compost (7 kg N, 15 kg P, and 35 K ha^{-1}) plots with application rates based on soil test data and University of Idaho recommendations (Brown, Horneck, & Moore, 2010). In spring 2019, urea was applied to the urea and compost plots (55 kg N ha^{-1}), while SuperU (94 kg N ha^{-1}) was applied to

TABLE 2 Average amount of manure C and N and synthetic fertilizer N added to the plots by treatment and year

Year	Control	Urea	SuperU	Compost[a]	Fall manure	Spring manure
	kg C or N ha^{-1}					
			Manure C			
2017	–	–	–	3,331	9,335	–
2018	–	–	–	1,798	8,081	5,143
2019	–	–	–	–	–	3,161
			Manure N			
2017	–	–	–	289	670	–
2018	–	–	–	157	660	365
2019	–	–	–	–	–	224
			Synthetic fertilizer N			
2017	–	–	–	–	–	–
2018[b]	–	24	24	7	–	–
2019	–	55	94	55	–	–

[a]Urea applied in the spring after compost application in the previous fall.
[b]Nitrogen from monoammonium phosphate in 2018; no urea or SuperU was applied.

corresponding plots as well; P as triple superphosphate was also applied to the urea and SuperU plots at a rate of 99 kg P ha^{-1}. After broadcasting the manure, compost, and fertilizers, the treatments were incorporated 15 cm into the soil on the same day using a tandem disk (control plots were also disked), followed by roller harrowing.

2.4 | Cropping system

Alfalfa (Dyna-Gro, Grandstand), corn (Pioneer, P9188R), and barley (MillerCoors, Moravian 69) were planted on 16 Apr. 2015, 17 May 2018, and 13 Apr. 2019, with row spacings of 19.1, 76.2, and 17.8 cm, respectively. Respective seeding rates were 34, 32, and 124 kg seed ha^{-1}, which is approximately equal to 1.5×10^7, 8.6×10^4, and 4.1×10^6 seeds ha^{-1}. Irrigation water was applied two to three times per week using a lateral-move irrigation system, with rates based on estimated crop water use data from the Bureau of Reclamation AgriMet web site (https://www.usbr.gov/pn/agrimet/). The crop evapotranspiration (ET) was determined daily by inputting data from Agrimet into the Kimberly–Penman ET model (Wright, 1982).

Herbicide applications at label rates for alfalfa occurred on 3 Mar. 2017 (Metribuzen [Loveland Products, Inc., Loveland, CO] and Gramoxone [Syngenta International Ag, Basel, Switzerland]), those for corn on 15 May 2018 (Makaze Glyphosate [Loveland Products, Inc.] and Diflexx [Bayer Crop Science, Monheim am Rhein, Germany]) and 20 June 2018 (Makaza Glyphosate [Loveland Products, Inc.]), and for barley on 13 May 2019 (Axial Star [Syngenta International Ag] and Affinity BroadSpec [Dupont, Willington, DE]) and 21 Aug. 2019 and 21 Sept. 2019 (Roundup PowerMax [ScottsMiracle-Gro, Marysville, OH]).

Crop harvest dates were 5 May, 10 July, and 15 Sept. 2017 for alfalfa; 24 Sept. 2018 for corn silage; and 6 Aug. 2019 for barley grain. For alfalfa, a Swift Machine and Welding (Swift Current, SK, Canada) custom forage harvester was used to harvest a 1.5 by18 m area from each plot. All plant material was weighed on a load cell platform and subsamples were collected from the platform. The remaining bulk alfalfa was cut, baled, and removed within 1 wk. For corn silage, a tractor-mounted Kemper (Stadtlohn, Germany) Champion C 1200 Universal Forage Harvester was used to harvest a 1.2 by 18 m area from each plot. The plant material was transferred to a Maize Harvester Haldrup (Ilshofen, Germany) M-63 unit that contains a load cell platform to collect and weigh the cut plant material and an auger port to collect plant subsamples. The remaining bulk corn was harvested and removed from the field within 1 wk. For barley grain, an Almaco (Nevada, IA) PMC20 Plot Master Combine was used to harvest a 1.5 by 18 m area. All grain was collected in sacks and later weighed and subsamples collected. The remaining bulk barley grain and straw were harvested and removed from the field within a week. All plant subsamples were dried in a constant-temperature forced-draft oven at 60 °C for 48 h, then reweighed to determine the dry matter fraction. Crop yields on a dry matter basis are presented in Table 3.

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

Treatment	Alfalfa, 2017	Corn silage, 2018	Barley grain, 2019
	kg ha^{-1}		
Control	14,339a[a]	23,240a	5,379c
Urea	16,148a	24,043a	7,387b
SuperU	15,478a	23,660a	8,123ab
Compost	16,300a	24,499a	8,484ab
Fall manure	17,040a	26,621a	7,890ab
Spring manure	16,158a	22,935a	8,776a

[a]Mean values within a column followed by the same lowercase letter are not significantly different at a .05 probability level.

2.5 | Gas emission measurements

Nitrous oxide, CO_2, and CH_4 flux measurements were conducted using a vented, non-steady-state, closed chamber technique as described by Dungan et al. (2017). In brief, during each measurement event, rectangular chambers (78.5 by 40.5 by10 cm) were fitted to aluminum anchors that were set 10 cm into the soil and sealed using a water channel. Duplicate anchors were placed 1 m apart in each plot and were set lengthwise with the crop row to cover the row and interrow space. The anchors were only removed during tillage, fertilizer application and harvest, but otherwise remained in the plots at all times. Alfalfa, corn, and barley within the anchor area were cut so that they did not extend above the water channel of the anchor.

Gas samples from the chamber were collected at 0-, 15-, and 30-min intervals using a 30-ml polypropylene syringe with a stopcock attached to the chamber. The syringes were stored in a cooler without ice packs until transported to the laboratory, where 25 ml was injected into pre-evacuated 12-ml Exetainer vials with gray butyl rubber septa (Labco Limited, Lampeter, UK). The samples were analyzed for GHGs using an Agilent Technologies (Santa Clara, CA) model 7890A gas chromatograph equipped with a GC 120 autosampler and electron capture, thermal conductivity and flame ionization detectors to quantify N_2O, CO_2, and CH_4, respectively. Gas fluxes were determined from a linear or nonlinear increase in the concentration within the chamber headspace over time (Hutchinson & Mosier, 1981). For estimated cumulative emissions, daily gas emissions between sampling days were generated using the adjacent sampling dates and linear regression values.

Chamber gas samples were usually collected two times per week, which was initiated at 1000 h. The samples were collected during the following growing periods each year: 2017 (29 March–30 November; DOY: 88–334); 2018 (3 April–21 September; DOY: 93–264); and 2019 (17 April–19 November; DOY: 107–323). Soil volumetric water content (VWC) at 0–15 cm was measured at the time of gas sampling by manually placing a model CS630 time domain reflectometry probe (Campbell Scientific, Inc., Logan, UT) into the soil immediately outside the gas-flux measurement area.

2.6 | Weather station

An onsite weather station with Campbell Scientific, Inc. sensors was located at the Northwest corner of the field in a control plot to measure soil temperature (model 109), heat flux (model HFP01), volumetric water content and electrical conductivity (model CS650), air temperature and relative humidity (model HMP50), barometric pressure (model CS106), wind speed, and direction (model 034B), and solar radiation (model LI200X). Precipitation data was obtained from the Kimberly Idaho AgriMet weather station (TWFI), which is 3.8 km Northeast of the field site.

2.7 | Statistical analysis

Cumulative N_2O-N, CO_2–C, and CH_4–C emissions, average growing season fluxes, crop yields, soil organic carbon (SOC) levels, net N_2O-N emission losses, and N_2O-N emission intensities were statistically analyzed using PROC GLM in SAS (SAS Institute, 2004). Values were log transformed when needed to achieve normality. Mean comparisons were performed using the Ryan–Einot–Gabriel–Welsch multiple range test at an α level of .05. Pearson correlation coefficients (r) were calculated using PROC CORR in SAS to determine relationships between the GHG fluxes and VWC or SOC. Statements of statistical significance were declared at $P < .05$.

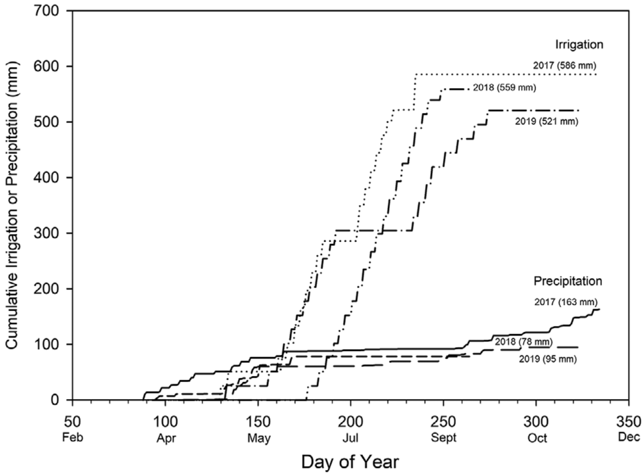

FIGURE 1 Cumulative precipitation and irrigation amounts (mm) during the 2017, 2018, and 2019 growing seasons

3 | RESULTS

3.1 | Environmental conditions

Precipitation accumulations and irrigation totals during the growing seasons are presented in Figure 1. The precipitation accumulations were 163, 78, and 95 mm, while irrigation totals were 586, 559, and 521 mm in 2017, 2018, and 2019, respectively. The precipitation events generally occurred at the beginning of the growing season in April through mid-June, then at the onset of the fall season near mid-September and later. Irrigation water was applied beginning on 10 May, 26 June, and 13 May in 2017, 2018, and 2019, respectively. Irrespective of year, the soil volumetric water content (VWC) was at about 0.2 $m^3 m^{-3}$ prior to the first irrigation event, then it hovered near 0.3 $m^3 m^{-3}$ throughout the remainder of the irrigation season (Figures 2, 3, and 4). The VWC trends and values at 0–15 cm were similar in all experimental plots (data not shown), indicating that there was little influence of the fertilizer and manure treatments on soil–water retention. Soil temperatures at 6 cm below the surface (at the time of gas sampling) were lowest at the beginning of the growing season, then steadily increased to their maximum between 20 and 23 °C near the end of July and early August and tapered off thereafter (Figures 2, 3, and 4). Average soil temperatures at 6 cm during the growing seasons were 16, 18, and 14 °C, with maximum temperatures of 34, 32, and 26 °C in 2017, 2018, and 2019, respectively.

3.2 | Crop yields

Alfalfa and corn silage yields were statistically similar among the synthetic N fertilizer and manure treatments, as well as with the control (Table 3). The alfalfa was planted in 2015 and 2017 was the 3rd year of growth without the addition of any soil amendments and nutrients since 2014. The yield comparison *P* values for the alfalfa and corn silage were .08 and .71, respectively. The only crop that showed treatment differences was the barley, where the grain yield was lowest in the control and highest for spring manure. Barley grain yields from the urea and spring manure treatments were not significantly different from SuperU, compost, and fall manure. The lack of alfalfa and corn silage yield response from the N source

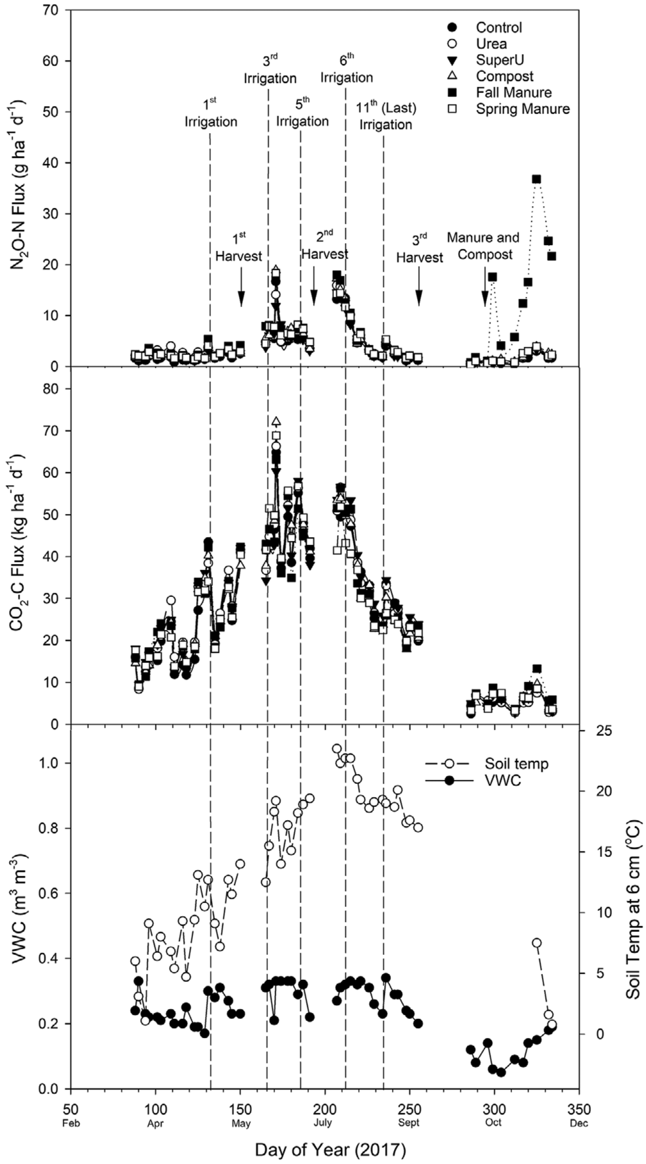

FIGURE 2 Average daily N_2O-N (top panel) and CO_2-C (middle panel) fluxes from the control, urea, SuperU, compost, fall manure, and spring manure plots during the 2017 growing season under alfalfa. Synthetic fertilizer and manure application rates can be found in Table 1. The bottom panel presents the average volumetric water content (VWC) and soil temperature (6 cm) at the time of gas sampling. Data gaps in the figure indicate time periods when the chambers were removed from the field for planting and harvesting

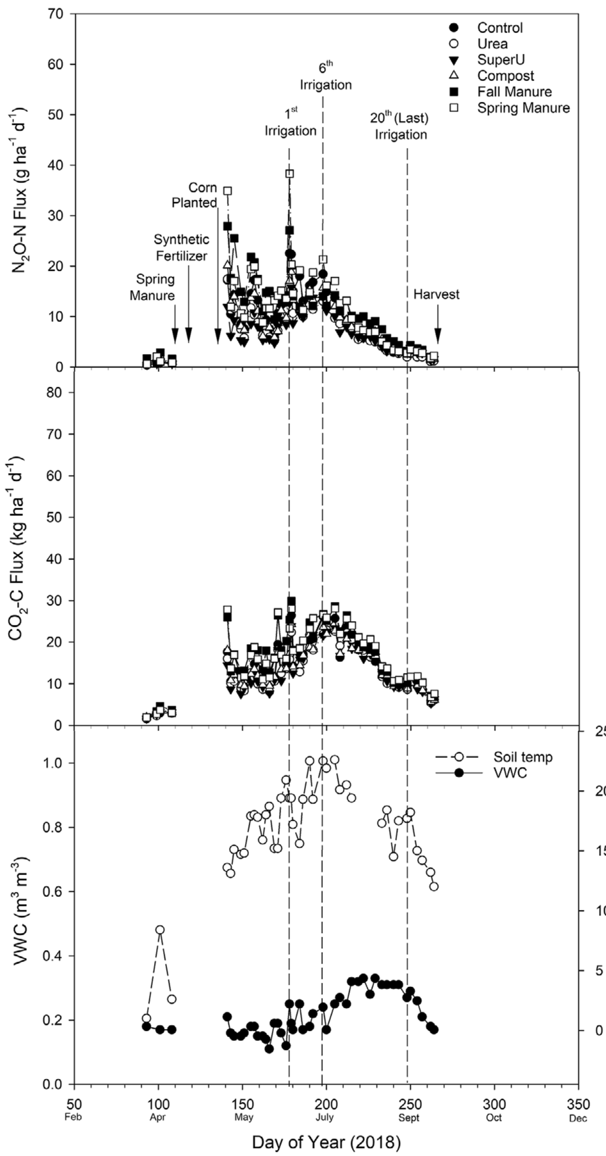

FIGURE 3 Average daily N_2O-N (top panel) and CO_2-C (middle panel) fluxes from the control, urea, SuperU, compost, fall manure, and spring manure plots during the 2018 growing season under corn. Synthetic fertilizer and manure application rates can be found in Table 1. The bottom panel presents the average volumetric water content (VWC) and soil temperature (6 cm) at the time of gas sampling. Data gaps in the figure indicate time periods when the chambers were removed from the field for planting and harvesting

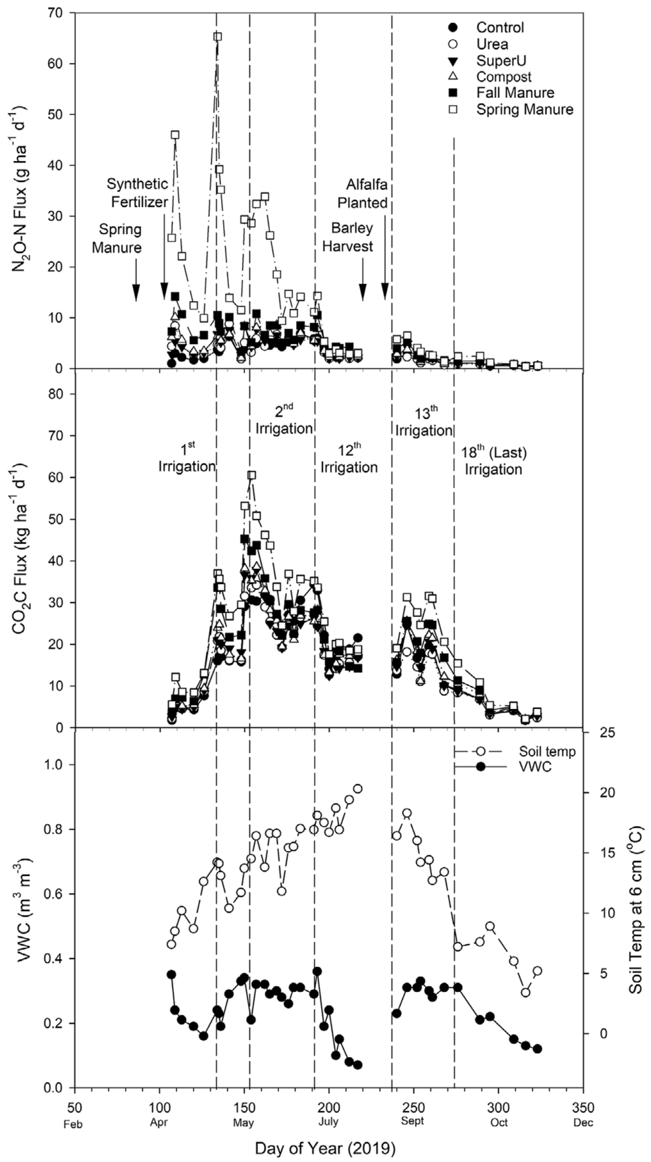

FIGURE 4 Average daily N_2O-N (top panel) and CO_2-C (middle panel) fluxes from the control, urea, SuperU, compost, fall manure, and spring manure plots during the 2019 growing season under barley. Synthetic fertilizer and manure application rates can be found in Table 1. The bottom panel presents the average volumetric water content (VWC) and soil temperature (6 cm) at the time of gas sampling. Data gaps in the figure indicate time periods when the chambers were removed from the field for planting and harvesting

treatments compared to the control was likely caused by N mineralization during the growing season.

3.3 | Nitrous oxide fluxes

Under alfalfa in 2017, baseline N_2O fluxes (avg. 2.1 g N_2O-N ha^{-1} d^{-1}) prior to the first irrigation event and harvest were similar (Figure 2), with the last manure/compost and synthetic N fertilizer treatments having been applied in fall 2013 and spring 2014. The N_2O fluxes peaked on Day 171 shortly after the third irrigation event, with an average flux of 16.2 g N_2O-N ha^{-1} d^{-1} among all plots. A second N_2O emission pulse of similar magnitude occurred on Day 207, which was 16 d after the second alfalfa harvest. A minor N_2O pulse occurred after the 11th irrigation event (last of season), followed by a decline to background levels. Fall manure and compost were applied to the plots on Day 297 and after which, the N_2O fluxes peaked on Days 299 and 325 at 17.6 and

TABLE 4 Average daily flux of N_2O-N, CO_2-C, and CH_4-C from the experimental plots during the growing seasons by treatment and year

Treatment	2017[a]	2018[b]	2019
		g N_2O-N ha^{-1} d^{-1}	
Control	3.5b[c]	9.1ab	3.0b
Urea	3.9b	7.2b	3.5b
SuperU	3.5b	6.8b	3.7b
Compost	4.0b	8.4ab	4.2b
Fall manure	6.6a	11.4a	5.6b
Spring manure	4.1b	11.3a	14.1a
		kg CO_2–C ha^{-1} d^{-1}	
Control	26.1a	13.8ab	17.4b
Urea	27.1a	12.7b	16.1b
SuperU	27.2a	12.7b	17.0b
Compost	26.7a	13.3ab	18.0b
Fall manure	26.9a	16.6a	20.6
Spring manure	26.2a	16.4a	25.4a
		g CH_4–C ha^{-1} d^{-1}	
Control	−1.1a	−1.1ab	−1.0a
Urea	−1.1a	−1.0a	−1.1a
SuperU	−1.0a	−1.1ab	−1.1a
Compost	−1.0a	−1.1ab	−1.4a
Fall manure	−1.0	−1.4b	−1.1a
Spring manure	−1.0a	−1.4b	−0.7a

[a]Treatments not applied in 2017, but last applied in fall 2013 and spring 2014 (see Table 1).
[b]Urea and SuperU not applied in 2018; N addition was via use of monoammonium phosphate (see Table 1).
[c]Emission values within a column followed by the same lowercase letter are not significantly different at the .05 probability level.

36.8 g N_2O-N ha^{-1} d^{-1}, respectively, in the fall manure only. When comparing the average N_2O fluxes over the growing season under alfalfa, the fall manure was significantly greater than all other treatments (Table 4).

Daily N_2O-N fluxes under corn production in 2018 are presented in Figure 3. Background emissions were monitored between Days 93 and 101 before the addition of spring manure and synthetic fertilizer, with an average daily flux of 1.1 g N_2O-N ha^{-1} d^{-1}. Fluxes were not measured between Days 101 and 141, due to field preparation, resulting in the highest flux measurements from all treatments on Day 141. The lowest flux on this day was 11.9 g N_2O-N ha^{-1} d^{-1} from the SuperU plots, while the greatest flux was 34.9 g N_2O-N ha^{-1} d^{-1} from the spring manure. Afterwards, the emissions briefly declined, followed by a pulse on Day 155 which was associated with a rainfall event of 9.1 mm the previous day (data not shown). After a minor pulse on Day 178, the daily fluxes steadily decreased to an average of 1.7 g N_2O-N ha^{-1} d^{-1} on Day 264 immediately before the corn was harvested. The average flux over the growing season was approximately 11.4 g N_2O-N ha^{-1} d^{-1} from both the fall and spring manure, which were significantly greater than average fluxes from urea and SuperU (Table 4).

The N_2O-N flux measurements under barley production in 2019 are presented in Figure 4. Twenty-one and 5 d after manure and fertilizer application, respectively, the N_2O flux from spring manure was 25.7 g N_2O-N ha^{-1} d^{-1}, whereas from the control, urea, SuperU, compost, and fall manure fluxes were lower at 1.0, 4.4, 2.8, 6.2, and 7.3 g N_2O-N ha^{-1} d^{-1}, respectively. On Day 109, the emissions peaked from all treatments, with the highest flux from spring manure at 46 g N_2O-N ha^{-1} d^{-1} and second highest flux from fall manure at 14.2 g N_2O-N ha^{-1} d^{-1}. The maximum flux from spring manure (65.3 g N_2O-N ha^{-1} d^{-1}) occurred on Day 134, which coincides with the first irrigation event. After the second irrigation event on Day 156 the emissions began to increase, but mainly from the spring manure, reaching 33.8 g N_2O-N ha^{-1} d^{-1} on Day 162. The N_2O emissions then declined thereafter, with a slight increase coinciding with the 12th irrigation event on Day 192. Once the barley was harvested (Day 218) and alfalfa seed was sown (Day 233), emission measurements were resumed until Day 323. The average N_2O fluxes over the growing season ranged from 3.0 g N_2O-N ha^{-1} d^{-1} from the control plots to 14.1 g N_2O-N ha^{-1} d^{-1} from the spring manure plots, but only the emissions from spring manure were significantly greater than the other treatments (Table 4).

3.4 | Carbon dioxide fluxes

Similar to daily N_2O fluxes under alfalfa, the CO_2 fluxes from each treatment tracked each other closely, with low variation between the treatment fluxes (Figure 2). Emission pulses during the growing season tended to occur near some irrigation events, soil temperature increases, and after fall manure/compost were applied. The maximum CO_2 fluxes occurred 4 d after the third irrigation event on Day 170, which also coincided with a rapid increase in soil temperature. The emissions began to taper off after Day 210 when the soil temperature also began to decrease and reached background levels near Day 286. The average CO_2 flux across all plots over the growing season was 26.7 kg CO_2–C ha^{-1} d^{-1} and there was no statistical difference between average fluxes when compared by treatment (Table 4).

Under corn production, the CO_2–C fluxes were about twofold lower than under alfalfa (Figure 3). Emission pulses likely occurred sometime during synthetic fertilizer and spring manure application, as noted by higher fluxes on Day 141. However, it is not known if pulses occurred between Days 108 and 141, as no gas samples were collected during this time. While emissions peaked briefly from all treatments on Day 179, the emissions were highest on Day 205 (avg. 25.2 kg ha^{-1} d^{-1}), then declined thereafter to an average flux of 6.5 kg ha^{-1} d^{-1} on the last sampling day (i.e., 264). The average flux over the growing season from fall manure (16.6 kg CO_2-C ha^{-1} d^{-1}) and spring manure (16.4 kg CO_2-C ha^{-1} d^{-1}) were statistically similar and significantly greater than the average fluxes from urea and SuperU (12.7 kg CO_2–C ha^{-1} d^{-1}) (Table 4).

The CO_2 emissions under barley also followed a similar trend as when under alfalfa, although the two crops were managed very differently. During the growing season, the largest emission pulses were associated with the first and second irrigation events, with respective fluxes being the greatest from the spring manure treatment at 37.0 and 60.5 kg CO_2–C ha^{-1} d^{-1}. The CO_2 emissions declined until the barley was harvested on Day 218, which then increased slightly from all treatments after the alfalfa was planted (Day 233) and 13th irrigation event (Day 234). After Day 261, the emissions started to decline to background levels with an average CO_2 flux of 3.1 kg CO_2-C ha^{-1} d^{-1} on Day 323, the last measurement of the year. Average fluxes over the growing season ranged from 16.1 kg CO_2–C ha^{-1} d^{-1} (control) to 25.4 kg CO_2–C ha^{-1} d^{-1} (spring manure) and the flux from the spring manure treatment was significantly greater than all but the fall manure treatment (Table 4).

3.5 | Methane fluxes

Methane emissions were negative regardless of crop and treatment, with the average flux among all treatments over the growing season being −1.0, −1.2, and −1.1 g CH_4–C ha^{-1} d^{-1} under alfalfa (2017), barley (2018), and corn (2019), respectively (Table 4). When broken down by treatment, there were no statistical differences between the average fluxes under alfalfa and barley. Under corn though, the consumption of CH_4 was significantly greater in the fall and spring manure vs. urea ($P < .05$). Methane production on the other hand, generally only occurred 1 to 2 d per growing season, where positive fluxes ranged from 0.5 to 2.3 g CH_4–C ha^{-1} d^{-1} (data not shown).

3.6 | Cumulative greenhouse gas emissions

Cumulative N_2O, CO_2, and CH_4 emissions over the growing seasons in 2017, 2018, and 2019 are presented in Table 5. No manure and fertilizer treatments had been applied since spring 2014, but treatments were once again applied for this study starting in fall 2017 (Table 1). However, under alfalfa in 2017, the residual effect of the last fall manure application in 2013 was likely evident, as the cumulative N_2O emission of 1.6 kg N_2O-N ha^{-1} was significantly greater than emissions from all other treatments (1.9-fold on average). In 2018, the cumulative N_2O emission of 2.0 kg ha^{-1} from the spring manure was significantly greater than from the urea, SuperU, and compost plots at about 1.2 kg N_2O-N ha^{-1}. In 2019, the spring manure produced the highest cumulative N_2O emission of 2.3 kg N_2O-N ha^{-1}, which was significantly greater than all other treatments by 3.2-fold on average. Interestingly, although the cumulative N_2O emission from the control was the lowest numerically, it was not significantly different from the other treatments, except the spring manure. Cumulative CO_2–C emissions were statistically not different among the treatments in 2017, as the treatments had not been applied as stated above. However, in 2018, the fall and spring manure emissions were significantly greater than the other treatments, while in 2019 only the spring manure produced the greatest cumulative emission. Regarding cumulative CH_4 emissions, the average values among the treatments were −0.30, −0.20, and −0.27 kg CH_4–C ha^{-1} under alfalfa, corn, and barley, respectively (Table 5). There were no significant differences between the cumulative CH_4 emissions for any of the treatments during any year.

TABLE 5 Average cumulative N_2O-N, CO_2–C, and CH_4–C emissions from the experimental plots during the growing seasons by treatment and year

Treatment	2017[a]	2018[b]	2019[c]	Total loss
	kg N_2O-N ha^{-1}			
Control	0.77b[d]	1.5abc	0.56b	2.8b
Urea	0.87b	1.2c	0.64b	2.7b
SuperU	0.79b	1.1c	0.69b	2.6b
Compost	0.89b	1.4bc	0.75b	3.1b
Fall manure	1.6a	1.9ab	1.0b	4.5a
Spring manure	0.91b	2.0a	2.3a	5.2a
	kg CO_2–C ha^{-1}			
Control	6,059a	2,228b	3,382b	11,669a
Urea	6,253a	2,063b	3,106b	11,422a
SuperU	6,333a	2,046b	3,279b	11,658a
Compost	6,145a	2,161b	3,483b	11,789a
Fall manure	6,327a	2,711a	3,870ab	12,908a
Spring manure	6,052a	2,730a	4,792a	13,574a
	kg CH_4–C ha^{-1}			
Control	–0.30a	–0.20a	–0.25a	–0.75a
Urea	–0.30a	–0.20a	–0.25a	–0.75a
SuperU	–0.27a	–0.20a	–0.29a	–0.76a
Compost	–0.30a	–0.20a	–0.29a	–0.79a
Fall manure	–0.33a	–0.19a	–0.28a	–0.80a
Spring manure	–0.32a	–0.20a	–0.26a	–0.78a

[a]Treatments not applied in 2017, but last applied in fall 2013 and spring 2014 (see Table 1). Values based on gas samples collected from 29 March–30 November.
[b]Urea and SuperU not applied in 2018; N addition was via use of monoammonium phosphate (see Table 1). Values based on gas samples collected 3 April–21 September.
[c]Values based on gas samples collected 17 April–19 November.
[d]Emission values within a column followed by the same lowercase letter are not significantly different at the .05 probability level.

3.7 | Nitrous oxide emission factors and emission intensities

The net N_2O emission losses as a percentage of total N applied (i.e., emission factor, EF) are presented in Figure 5. In 2018, −1.0, −1.5, and −0.02% of the N applied was lost as N_2O-N from the urea, SuperU, and compost treatments, respectively. Negative EFs in this case mean that more N_2O was emitted from the control plots vs. those treated with urea, SuperU, and compost. From the fall and spring manure plots, only 0.07 and 0.15% of the N applied was lost, respectively. The N_2O losses associated with the compost, fall manure, and spring manure plots, while low, were significantly greater than from the urea plots. In 2019, all of the N_2O EFs were positive and soils treated with spring manure lost 0.79% of the N applied, a significantly greater amount than all other treatments.

Nitrous oxide emissions per crop yield, also referred to as emission intensity (EI), were calculated for each growing season and expressed as g N_2O-N lost per megagram of crop dry matter (DM) yield (Figure 6). The highest EI for alfalfa was 93 g N_2O-N Mg DM^{-1} from the fall manure plots, which was significantly greater than all other treatments by 1.7-fold on average. Under corn, the EI for silage with spring manure was 88 g N_2O-N Mg DM^{-1} and significantly greater than all other treatments by 1.4- to 1.9-fold, except fall manure. The EI for barley grain with spring manure was the highest across all crops at 265 g N_2O-N Mg DM^{-1} and significantly greater than the other treatments by 2.5- to 3.1-fold during the 2019 growing season.

3.8 | Soil organic carbon

The SOC levels were measured each fall (post-harvest) to track annual changes. At 0–30 cm, the SOC levels ranged from 41.4 to 76.0 Mg C ha^{-1} with the fall and spring manure plots having significantly higher values than the other treatments (Table 6). The annual increase in SOC was calculated by linear regression of SOC levels from 2017 to 2019 and only the regressions for compost (r^2 = .52, P = .008), spring manure (r^2 = .66, P = .001), and fall manure (r^2 = .84, P < .0001) were determined to be significant. When

TABLE 6 Soil organic carbon (SOC) levels at 0–30 cm for each growing season by treatment, as well as a determination of net SOC levels over the 3-yr study

Treatment	SOC 2017	SOC 2018	SOC 2019	Linear regression Slope	r^2	*P* value	Net SOC[a]
	Mg ha^{-1}						Mg ha^{-1}
Control	41.4b[b]	43.7b	44.1c	1.34	.10	.306	–
Urea	42.8b	43.8b	45.0c	1.09	.15	.221	–
SuperU	42.0b	44.9b	43.8c	0.91	.31	.308	–
Compost	44.8b	49.2b	51.4c	3.30	.52	.008	9.9
Fall manure	52.1a	56.6a	76.0a	12.0	.66	.001	35.9
Spring Manure	55.2a	56.7a	64.3b	4.58	.84	<.0001	13.8

[a]Net SOC was calculated by multiplying the regression slope by three, but only for treatments where the *P* value was < .05.
[b]Mean values within a column followed by the same lowercase letter for each depth are not significantly different at a .05 probability level.

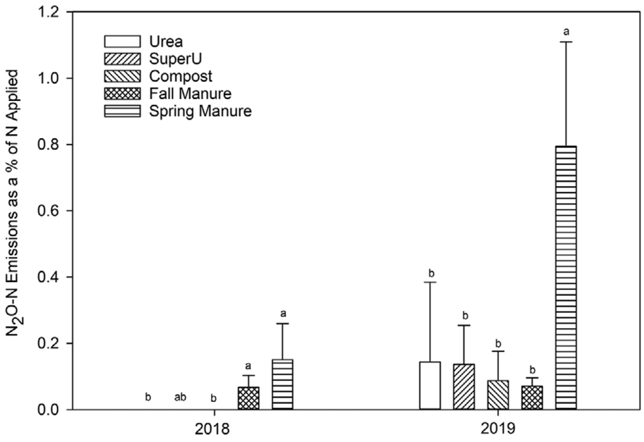

FIGURE 5 Net N_2O-N emission losses as a percentage of total N applied for urea, SuperU, compost, fall manure and spring manure plots in 2018 and 2019. Values were calculated by subtracting the cumulative N_2O-N emissions of the control from that of the synthetic fertilizer and manure treatments (Table 5), then dividing by the appropriate N rate in Table 1. The absence of vertical bars indicates that the values were less than zero. Lowercase letters above the bars indicate significant difference between the treatments at a 0.05 probability level for that year only

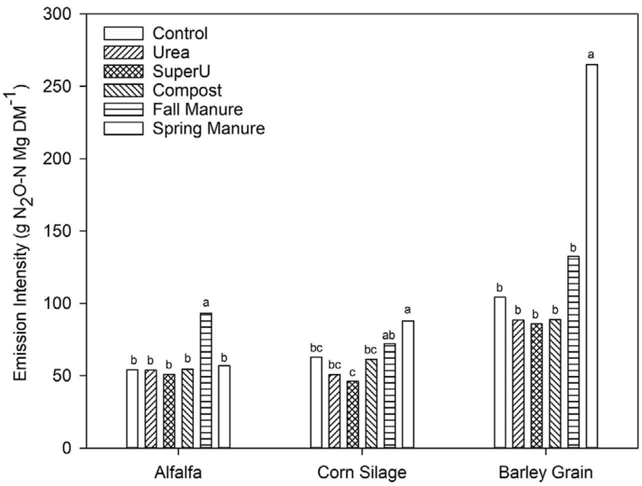

FIGURE 6 Nitrous oxide emission intensities for each growing season presented as g N_2O-N per megagram of dry matter (DM) yield for alfalfa, corn silage, and barley grain. Lowercase letters above the vertical bars indicate significant difference between the treatments at a 0.05 probability level

multiplying the regression slopes by 3, the net SOC increase in the compost, fall manure, and spring manure plots over 3 yr was calculated to be 9.9, 35.9, and 13.8 Mg SOC ha^{-1}, respectively. When net SOC is converted to a CO_2 eq. basis, these treatments respectively gained 36.3, 131.8, and 50.4 Mg CO_2 equivalent ha^{-1}.

4 | DISCUSSION

Dairy manure solids in composted and non-composted forms are often applied to agricultural soils as a substitute for synthetic N fertilizers, but documented impacts on trace gas fluxes in the semiarid western United States are limited. While there is a demonstrated effect of N source on GHG emissions from irrigated soils in Colorado (Halvorson et al., 2016), only two studies to date have investigated emissions from cropping systems in Idaho (Dungan et al., 2017; Leytem et al., 2019). Average daily fluxes of N_2O, CO_2, and CH_4 in the present study were within ranges from our previous study as reported by Dungan et al. (2017) and others in the literature (Alluvione et al., 2009; Ellert & Janzen, 2008; Halvorson, Del Grosso, & Alluvione, 2010a; Heller et al., 2010; Leytem et al., 2019; Rochette, Angers, Chantigny, Gagnon, & Bertrand, 2008). In 2017, the N_2O fluxes and cumulative emission were greatest from the fall manure treatment, which is interesting since the last manure application occurred in 2013. This could be due to a manure carryover effect, although one would also expect to see an increased flux of CO_2 from this treatment. The manure applications, which were resumed after the alfalfa was terminated, did begin to have a positive effect on emissions in subsequent years. During the previous crop rotation, cumulative N_2O emissions were 53% lower with SuperU than with granular urea under corn (Dungan et al., 2017), with similar results reported by Halvorson, Del Grosso, and Francesco (2010b) under no-till corn. However, in the previous rotation, there was no effect of SuperU vs. urea under barley (Dungan et al., 2017) which was the same trend as seen in the present study. When summing the cumulative N_2O emissions from all three growing seasons, the fall and spring manure treatments produced the greatest emissions, which is an expected result given the well-known effect that animal manure additions have on soil N_2O emissions.

Over a 4-yr period in a wheat (*Triticum aestivum* L.)–potato (*Solanum tuberosum* L.)–barley–sugarbeet (*Beta vulgaris* L.) rotation in southern Idaho, Leytem et al. (2019) found that respective cumulative emission losses of N_2O were 1.3- and 3.0-folder greater from plots that received annual applications of dairy manure at 18 and 52 Mg ha^{-1} vs. plots receiving synthetic N fertilizer. In the preceding crop rotation of corn–barley–alfalfa, plots treated with spring or fall manure had cumulative N_2O emissions that were 1.9-fold greater than those treated with urea, while plots treated with fall compost had emissions similar to that of urea (Dungan et al., 2017). Synthetic N fertilizers, such as urea, are also a driver of increased N_2O fluxes (Abalos, Sanz-Cobena, Misselbrook, & Vallejo, 2012; Delgado & Mosier, 1996) and in the Idaho studies mentioned above, cumulative emissions were 1.5- to 2.0-fold greater from urea plots than control plots. In the present study, respective total N_2O losses over the rotation from fall and spring manure plots were 1.7- and 2.0-fold greater than urea-treated plots; however, the emissions were not significantly different between the urea and control plots. While these emission trends are similar to that in our first study, the urea-treated plots in that study did emit about 1.5-fold more N_2O than control plots over the course of the 3-yr study (Dungan et al., 2017). The difference here is likely due to the fact that, in the present study, the corn and barley followed 3 yr of alfalfa growth, thus the alfalfa was a N source and it boosted N_2O emissions in the control plots. Alfalfa can provide an additional 67–112 kg of available N per hectare beyond what is detected by spring soil sampling when high tissue-N concentrations are released during decomposition (Brown et al., 2010). Wagner-Riddle, Thurtell, Kidd, Beauchamp, and Sweetman (1997) reported that the highest estimated annual N_2O emissions were in agricultural fields after alfalfa was terminated by tillage incorporation in the fall and when liquid dairy manure was applied to fallow fields. Similarly, Johnson, Weyers, Archer, and Barbour (2012) found that the largest N_2O flux occurred during spring thaw in a conventionally managed system when the alfalfa had been plowed under the previous fall.

In addition to N inputs, other factors that can cause N_2O emission pulses during the growing season include environmental factors and management practices. Irrigation and rainfall have been documented to cause N_2O pulses, especially when associated with N applications (Hutchinson & Mosier, 1979; Liu et al., 2011; Westphal, Tenuta, & Entz, 2018). Because N fertilizer was not added to alfalfa in the present study, it is possible that N released during the temporary senescence of nodules and roots after harvest (Brophy & Heichel, 1989; Vance, Heichel, Barnes, Bryan, & Johnson, 1979) was partly responsible for the N_2O emission pulses. Following harvest, the released N can be nitrified to NO_3^- when soil pore spaces were not saturated; however, upon irrigation the pore spaces become anaerobic as they fill with water, then N_2O is produced via denitrification. The large pulses of N_2O generally occurred when the VWC was slightly >0.3 m^3 m^{-3}, which is approximately a water-filled pore space of 57% (assuming respective bulk and particle densities of 1.4 and 2.65 g cm^{-3}). A water-filled pore space of 60% is considered the lower limit for denitrification, but it increases with increasing soil water content and reaches a maximum at saturation, whereas nitrification reaches it maximum at 60% water-filled pore space (Linn & Doran, 1984). In semiarid regions, nitrification is probably the main source of N_2O emissions from soils that are predominantly aerobic, but the addition of irrigation water to support agricultural production has shifted the soils to become anaerobic more often and invoking denitrification (Barton et al., 2008; Galbally et al., 2008).

Denitrification in soils not only requires NO_3^-, high moisture and anaerobic conditions, but soluble (and labile) C which, in this particular case, can be produced by irrigation of soils recently treated with manure. The lack of N_2O pulses during the 2019 growing season, from soils treated with compost and manure the previous fall, likely occurred because soluble C leached deeper into the soil profile during the wetter winter months and/or was used by microorganisms during freeze–thaw events. Surprisingly though, the fall manure plots were determined to have significantly higher SOC levels in the top 30 cm than in spring manure plots, although SOC levels in both treatments were significantly greater than all other treatments. A higher SOC level should provide more labile C for utilization by heterotrophic denitrifying bacteria, thus resulting in the creation of anaerobic microsites within soil aggregates as oxygen is consumed, favoring N_2O production and emissions (Gu et al., 2017; Parkin, 1987). However, denitrification occurs more readily when oxygen availability is low due to high soil moisture contents and to a lesser extent when labile C stimulates heterotrophic microbial activity (Thomas et al., 2017). In the present study, there were weak but significant correlations during each year between the daily N_2O

fluxes and soil moisture readings at the time of gas sampling ($r = -.23$–$.34$, $P < .005$; data not shown). The correlations were somewhat stronger when comparing the average daily N_2O flux from each growing season to corresponding SOC levels at 0–30 cm ($r = .49$–$.64$; $P < .02$; data not show).

In contrast to N_2O emissions, CO_2 fluxes during the growing season followed a distinct parabolic trend that can be attributed to increased root activity (i.e., growth and exudation) and enhanced organic matter decomposition that occur under warmer conditions (Adviento-Borbe et al., 2010; Alluvione et al., 2009). As anticipated, the fall/spring and spring manure produced the greatest cumulative CO_2 losses in 2018 and 2019, which was a 1.3- and 1.5-fold increase over the urea-treated plots, respectively. Despite the fact that manure was not added to the plots in 2017, the cumulative CO_2 losses were substantially greater than the other years across all treatments, which is likely a result of the extensive root system produced by the alfalfa in its 3rd year of growth.

The vast majority of daily CH_4 fluxes were negative and fell within ranges reported in other irrigated cropping studies (Alluvione et al., 2009; Delgado & Mosier, 1996; Ellert & Janzen, 2008). The predominant CH_4 sink is destruction in the troposphere, but aerobic and dry soils also act as the only known terrestrial sink and remove about 5.8% of globally produced CH_4 per year (Hütsch, 2001). While arable soils are an important sink, in the present study, CH_4 oxidation only mitigated ≤0.16% of the GHGs emitted over the growing season when calculated on a CO_2 equivalent basis (data not shown). Even though irrigation has been shown to dramatically reduce CH_4 uptake in soils (Bronson & Mosier, 1993), the soils in the present study dry out at a high enough rate that CH_4 oxidation occurs largely unimpeded without positive emissions. However, correlations between CH_4 flux and VWC were positive under irrigation-intensive alfalfa and corn ($r = .30$ and $.51$, $P < .001$), demonstrating that less CH_4 was consumed at higher soil moisture contents (data not shown). With respect to N source influences, only the average daily CH_4 fluxes in 2018 from the spring and fall manure treatments were found to be greater (i.e., more negative) than the control by 1.4-fold. Despite this slight bump in CH_4 consumption on average, cumulative CH_4 emissions for each growing season and total losses over the rotation were not significantly different among the treatments. Methane uptake rates are generally not influenced by manure applications, although use of NH_4^+–based fertilizers has been shown to inhibit CH_4 oxidation (Chan & Parkin, 2001; Lessard, Rochette, Gregorich, Desjardins, & Pattey, 1997; Powlson et al., 1997).

Negative EFs occurred only in urea, SuperU, and compost plots in 2018, which was because cumulative N_2O emissions from the control were slightly greater than these treatments. During the same growing season, the proportional losses of applied N from the fall and spring manure treatments were 0.07 and 0.15%, respectively. In 2019, the greatest loss of applied N was from the spring manure treatment at 0.79% and all other treatments were at ≤0.14%. The N_2O EFs from our manure-treated plots were comparable to EFs of 0.27% (Bell et al., 2016) and ranges of 0.09–0.55% (Webb, Thorman, Fernanda-Aller, & Jackson, 2014) and 0.13–0.18% (Leytem et al., 2019) reported from cropping systems using cattle manure. The fact that spring manure had the greatest EF was not expected, as the N has lower availability compared to synthetic N fertilizer because it is tied up in organic form until it is released upon mineralization, which generally occurs over the growing season. In contrast, Dungan et al. (2017) reported that the N_2O EF from urea-treated plots (i.e., 0.2%) was about 6.0-fold greater on average than from manure-treated plots under corn production, but there were no significant differences between EFs the following year under barley. Regardless of year and treatment, EFs from the present study were found to be at the lower end of values from synthetic and organic fertilizer studies (Snyder, Bruulsema, Jensen, & Fixen, 2009) and were less than the Tier 1 EF of 1.0% (uncertainty range of 0.01–1.8%) proposed by the Intergovernmental Panel on Climate Change (IPCC) to estimate direct N_2O emissions from managed soils (IPCC, 2006). The default EF of 1.0% accounts for N additions from synthetic fertilizers, organic amendments and crop residues and mineralization of N in soil organic matter and equates to a GWP of 4.9 kg CO_2 kg^{-1} of N applied.

Emission intensities are useful in that they can provide information to help reduce agricultural climate impacts through intensification (Snyder et al., 2009). The EIs in the present study ranged from 46 to 265 g N_2O-N Mg DM^{-1} and were similar to values reported by Westphal et al. (2018) and Gelfand, Shcherbak, Millar, Kravchenko, and Robertson (2016) in conventional and organic production systems with a soybean–wheat–alfalfa or corn–soybean–wheat rotation, respectively. While the EIs were always numerically greater in the fall and spring manure treatments, the greatest difference occurred with barley grain, where the EI for the spring manure treatment was nearly 3.0-fold greater than from the other treatments. Synthetic N fertilizer and compost tended to have the lowest EIs and their crop yields were not significantly different than the manure treatments. While EIs are higher from the manure treatments, the advantage of using these organic fertilizers is the GWP offset that occurs as a result of increasing SOC.

5 | CONCLUSIONS

In summary, fall and spring manure applications produced the greatest cumulative N_2O emissions across all growing seasons compared to synthetic N fertilizer and composted dairy manure, which is likely due to increased availability of labile C and N as a result of these treatments. Nitrous oxide fluxes increased as soil temperatures increased, and major pulses were associated with early-season irrigation events and incorporation of N sources. Only when under barley production with spring manure were the N_2O pulses found to be substantially greater than the other treatments, with 76% of the total cumulative emissions occurring just 65 d into the growing season as compared to 41–52% from the other treatments. Compared to the urea treatment, the enhanced-efficiency fertilizer SuperU, was not found to reduce N_2O emissions as was found in our previous study under corn. This suggests that SuperU effectiveness is not consistent, but we cannot rule out the influence of year-to-year differences between climate and soil conditions as being partly or solely responsible for this outcome. Regardless of the treatment, the total loss of N_2O-N as a percentage of N applied was relatively low at ≤0.79% during the growing season. Carbon dioxide emissions were not found to be significantly different among the treatments across all growing seasons, nor were the CH_4 emissions. While CH_4 emissions were negative throughout the study, the offsetting of GWP was only of very minor significance. Because dairy manure will continue to be used in cropping systems in southern Idaho, approaches to minimize GHG emissions and/or offset the GWP are in need of continued investigation. Despite greater N_2O emissions from manured soils, one potential benefit is the accumulation of organic C in the soil profile, which could be used to reduce the GWP. Additional research is needed to understand the long-term stability of manure-derived organic C in the profile of semiarid irrigated soils.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

ORCID

Robert S. Dungan https://orcid.org/0000-0002-7560-5560
April B. Leytem https://orcid.org/0000-0001-5976-402X
David D. Tarkalson https://orcid.org/0000-0003-4670-1868

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