



Compost and soil moisture effects on seasonal carbon and nitrogen dynamics, greenhouse gas fluxes and global warming potential of semi-arid soils

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Abstract

Purpose An 8-week incubation study was conducted to monitor soil inorganic nitrogen (N), dissolved organic carbon (DOC), greenhouse gases (GHG) [CO₂, N₂O and CH₄] and cumulative global warming potential (GWP) in dryland soil.

Methods Soil was amended with variable rates of compost (zero, 15, 30 and 45 dry Mg ha⁻¹) and soil moistures [5% (dry), 7% (normal) and 14% (wet) water filled pore space (WFPS)] and experienced biweekly temperature transitions from 5 °C (late winter) to 10 °C (early spring) to 15 °C (late spring) to 25 °C (early summer).

Results The addition of 30 and 45 Mg ha⁻¹ compost enhanced N mineralization with 13% more soil inorganic N (7.49 and 7.72 μg Ng⁻¹ day⁻¹, respectively) during early summer compared with lower compost rates. Normal and wet soils had 35% more DOC in the late spring (an average of 34 μg g⁻¹ day⁻¹) compared to the dry WFPS, but transitioning from late spring to early summer, DOC at all soil WFPS levels increased. Highest rates of compost were not significant sources of GHG with normal soil WFPS, compared with lower compost rates. Carbon dioxide emissions increased by 59 and 15%, respectively, as soil WFPS increased from dry to normal and normal to wet. Soils with normal WFPS were the most effective CH₄ sink.

Conclusion One-time application of high compost rates to dryland soils leads to enhanced N and C mineralization under normal soil moisture and warmer temperature of the summer but will not pose significant global warming dangers to the environment through GHG emissions since soils are rarely wet.

Keywords Dryland soil · Compost · Nutrient mineralization · Greenhouse gas emission

Introduction

Soil organic amendments such as composted feedlot manure (henceforth referred to as “compost”), are important and sometimes, the only sources of nutrients available to dryland organic winter wheat (*Avena sativa* L.) farmers in the

Northern High Plains (NHP) ecoregion of the United States (Larney and Angers 2012). This ecoregion experiences very high variability in precipitation and temperatures during distinct annual seasons (winter, spring, summer and fall). Such variability in weather conditions has significant impact on soil organic matter (SOM) mineralization, nutrient retention and potential losses to greenhouse gas (GHG) emissions. More information on the interactive effects of seasonal availability of soil water and compost-derived organic matter mineralization is critical to formulate appropriate recommendations for farmers and assess the impact on global warming potential of management practices, especially since agriculture contributes 12% of global GHG emissions (Linquist et al. 2012). Concurrent monitoring of soil C and N dynamics with GHG emissions under controlled laboratory conditions can also help better understand the suitability of compost for dryland crop production in semi-arid climates (Beare et al. 2009; Mikha et al. 2005).

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Prior research on interactive effects of compost-amended soils with temperature and/or soil moisture in semi-arid climates has produced variable results. Reddy and Crohn (2013) observed up to 6% increase in soil salinity, which restricted plant nutrient assimilation and negatively affected crop yields when 10–15 Mg ha⁻¹ of compost was applied every 3–4 years on clay loam soils. Reeve et al. (2011) observed that one-time application of a high rate of compost (50 Mg ha⁻¹) to semi-arid nutrient-poor soils resulted in elevated soil available phosphorous (P), soil organic C and microbial biomass C, 16 years following the application. Rodionow et al. (2006) observed higher microbial activity in thin films of water surrounding soil particles in the winter, contrary to several reports of slow decomposition rates and slow microbial activity during winter. Rewetting of dry soils, especially by summer rains, has triggered immediate soil microbial activity (Austin et al. 2004) and caused temporary increases in mineral and labile organic N and C and GHG pulses (Norton et al. 2008).

Studies on mechanisms of soil nutrient dynamics and GHG emissions from soils amended with compost often employ laboratory incubations (Beare et al. 2009; Mikha et al. 2005). Using laboratory incubation studies, Hernandez-Ramirez et al. (2009) demonstrated that soils receiving frequent applications of compost emit more N₂O compared with soils amended with inorganic N fertilizer following rewetting. Wang et al. (2016a) observed that CO₂ fluxes are best correlated with temperature and N fertilizer inputs. Wang et al. (2016b) observed that N₂O fluxes depend on soil NH₄ concentrations, followed by temperature and moisture. Denmead et al. (2010) found that high soil total C results in greater N₂O fluxes following rewetting of weathered acidic soils planted to sugarcane (*Saccharum officinarum*). Short-term laboratory incubations can provide rapid responses and useful insights into mechanisms of soil organic matter transformations (Belay-Tedda et al. 2009).

The objective of this study was to monitor the soil C and N dynamics, GHG emissions, and cumulative global warming potential (GWP) of dryland soil amended with different rates of compost (0, 15, 30 and 45 dry Mg ha⁻¹) and soil moisture contents [5% water filled pore space (WFPS) (dry), 7% (normal) and 14% (wet)] and transitioning through the various seasons of the year. We hypothesized that, because the highest compost rate is assumed to contain more N and C, the highest soil moisture content will cause it to release more available soil N and C during the summer temperature, but will improve soil nutrient holding capacity enough to reduce losses to the atmosphere as GHG.

Materials and methods

Type and site of experiment

The experiment was a laboratory incubation study conducted at laboratories and the greenhouse facility of the Plant Sciences department at the University of Wyoming. The incubated soil was collected from 0 to 15 cm depth at a research site located at the Sustainable Agricultural Research and Extension Center (SAREC), near Lingle, Wyoming, USA (42° 7' 45.92" N LONG, 104°23'24.88" W LAT, 1276 m above sea level). The soil of this site is silty clay loam with alkaline pH and CaCO₃ content between 1 and 3%.

The research site has been managed under wheat fallow rotation for over 15 years using tillage to 15-cm depth performed up to six times per year for the past 12 years (Bista et al. 2017; Norton et al. 2012). The site receives 300–400 mm year⁻¹ of rainfall and has 125 frost-free days. Average temperatures range between – 11 °C in winter and 32 °C in summer.

Experimental design and treatments

The study was established in a factorial completely randomized design with three replications. Treatments included one of 0, 15, 30 and 45 Mg ha⁻¹ dry weight of compost matched with one of 5, 7 and 14% water filled pore spaces (WFPS). The soil 5, 7 and 14% WFPS which represented dry, normal and wet field soil moisture conditions, respectively. Since about 11% of the total N in compost (see Table 1) is mineralized during the first year (Eghball 2000) in drylands, the amount of total N supplied by the compost equaled 0, 15, 30 and 45 kg N ha⁻¹, respectively. The soil and compost were analyzed for physical and chemical properties before the start of the experiment.

Table 1 Physical and chemical properties of soil and compost

Properties	Compost	Soil
Moisture (%)	1.97	3.8
Texture	Not determined	Silty clay loam
pH (1:1 soil: water ratio)	8.46	7.80
Bulk density (g cm ⁻³)	0.98	1.37
Total N (g kg ⁻¹)	9.08	1.72
Total C (g kg ⁻¹)	85.7	18.4
TOC (g kg ⁻¹)	76.3	13.5
IC (g kg ⁻¹)	9.38	4.96
PMN (mg kg ⁻¹)	68.2	21.1
Olsen P (mg kg ⁻¹)	36.2	23.5



Mass of soil needed to fill specimen cups and centrifuge tubes for compost application was calculated based on field bulk density as follows:

weight of soil = Bulk density

$$* (\text{filling volume of specimen cup/centrifuge tube}). \quad (1)$$

After obtaining the mass of soil, amount of compost to be added was calculated on mass per volume basis as follows.

Assuming 45 Mg ha⁻¹ compost application:

1 ha = 2 * 10⁶kg soil (based on the assumption that one hectare furrow slice has 2 Mkg soil).

If 45 Mg compost = 1 ha soil

$$\begin{aligned} 45 * 10^6 \text{ g} &= 2 * 10^6 \text{ kg soil} \leftrightarrow 45 * 10^6 \text{ g} \\ &= 2 * 10^9 \text{ g soil} \leftrightarrow 0.0225 \text{ g} = 1 \text{ g soil} \end{aligned}$$

The amount of compost to be added to the mass of soil in Eq. 1 above is calculated as follows:

$$\text{Amount of compost (g)} = \frac{\text{weight of soil(g)} * 0.0225(\text{g})}{1(\text{g})}. \quad (2)$$

Moisture treatments were applied with deionized (DI) water before the addition of compost. Together, there were 12 amendment-by-moisture treatments with three replicates at the start of the incubation. No water was added to the soil again after the first water addition. The same soil samples were subjected to temperature increments from the start to the end of the experiment as explained below.

To measure soil C and N dynamics (part 1) concurrently with GHG emissions (part 2), the experiment was deployed as two separate incubations (one for soil nutrients and one for GHG). Jars containing centrifuge tubes (explained in part 1 below) or specimen cups (explained in part 2 below) were kept in temperature-controlled dark incubators (BOD low temperature incubator, VWR International, LLC and Percival Intellus Control System Model E36HO, Percival Scientific Inc, Perry, IA, USA) for a period of 8 weeks. During the first 2 weeks, the temperature was 5 °C and afterwards, the temperature was increased every 2 weeks to 10 °C, 15 °C and 25 °C to simulate late winter, early spring, late spring and summer temperature conditions. Soil and gas indices were calculated as averages per day under each temperature regime.

Determination of soil physical and chemical properties

Before compost application, the soil was analyzed for physical and chemical properties (Table 1). Soil and compost were air-dried, sieved through a 2-mm sieve and analyzed

for particle size distribution (Gee and Bauder 1986), electrical conductivity and pH (Thomas 1996), available phosphorus (P) (Olsen and Sommers 1982), total N, total C and inorganic C (Sherrod et al. 2002). Total C and total N were determined by dry combustion using elemental analyzer (NC2100 Carlo Erba Instruments, Italy). Total organic C (TOC) was calculated by subtracting inorganic C from total C (Schumacher 2002).

Part 1: soil organic matter mineralization

Three 40-ml plastic centrifuge tubes containing 30 g of soil mixed with the required amount of compost and DI water were placed in each of 15 1-l glass jars held. 0.68 g, 0.45 g and 0.23 g of compost representing 45, 30 and 15 Mg ha⁻¹ were added to the soil. Ten ml of DI water was added to the bottom of each jar to minimize soil evaporation during the incubation. Jar tops were covered with semi-permeable wax film paper (Parafilm 'M') and placed in an incubator. One centrifuge tube was sampled from each jar weekly while the rest of the centrifuge tubes continued incubation. Soil was analyzed for gravimetric moisture content (Gardner 1986). Ten grams of soil was added to 25 ml of two molar potassium chloride (2 M KCl), placed on a shaker for 30 min and filtered through ashless filter paper (Q5 Fischer Scientific, USA) (Bremner and Kenney 1966). Obtained extracts were analyzed on spectrophotometer microplate reader (UV–Vis Biotek Instruments, Highland park, USA) for ammonium (NH₄⁺) using sodium salicylate (Sims et al. 1995) and nitrate (NO₃⁻) using vanadium chloride (Doane and Horwath 2003). Both parameters were then combined for a single estimate of inorganic N. Ten grams of the soil was also added to 25 ml of one-half molar potassium sulfate (0.5 M K₂SO₄), processed in the same way as the 2 M KCl extracts and used for the determination of dissolved organic carbon (DOC) on a total organic carbon analyzer (TOC-VCSH, Shimadzu, Japan).

Part 2: greenhouse gas fluxes

Twenty-seven grams of soil was placed in a 120-ml plastic specimen cup and 0.2 g, 0.4 g and 0.6 g of compost representing 15, 30 and 45 Mg ha⁻¹ were added to soil and mixed. A non-amended soil was treated as a control. Specimen cups were placed in 1000 ml mason jars. Five milliliters of DI water were placed at the bottom of each jar. Two jars with specimen cups and DI water only were treated as controls. Jars were sealed with air-tight lids secured with rubber septa and placed in a low-temperature incubator (BOD low-temperature incubator, VWR International, LLC) at 5 °C.

Headspace sampling for GHG fluxes occurred at days 1, 4, 7 and 14 for each temperature regime resulting in a total

of 16 sampling campaigns. At each sampling, a needle connected to a 60-ml syringe was plunged through rubber septum and a 50-ml of headspace air was withdrawn. A 30-ml of gas was transferred to a pre-evacuated 12-ml Labco Exetainer® glass vial secured with borosilicate air-tight septum. Following gas sampling, jars were opened, the headspace evacuated using house vacuum, kept open to equilibrate with the lab air pressure, sealed with lids and returned to the incubator. Gas samples were stored in a cool dry place until analysis.

The gas samples were analyzed for carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) on an automated gas chromatograph (Varian 38001) equipped with thermo-conductivity detector for CO₂, flame ionization detector for CH₄ and electron capture detector for N₂O analyses (Mosier et al. 1991).

Carbon dioxide, CH₄ and N₂O fluxes 1, 4, 7 and 14 days for every temperature were averaged and calculated using the equation from Hutchinson and Mosier (1981):

$$F_x = \frac{b * V_{ch} * MW * 60 * 10^6}{A_{ch} * MV_{corr} * 10^9}, \quad (3)$$

where F_x flux of greenhouse gas \times ($\mu\text{g m}^{-2} \text{h}^{-1}$), b change in concentration of greenhouse gas \times over time (ppb min^{-1}), V_{ch} volume of jar (m^3), MW molecular weight of greenhouse gas \times (g mol^{-1}), 60 conversion factor from minute to hour, 10^6 conversion factor from g to μg , A_{ch} jar basal area (m^2), MV_{corr} temperature correction for change in volume of greenhouse gas \times ($\text{m}^3 \text{mol}^{-1}$)

Global warming potential (GWP) was estimated from equations based on Xu et al. (2015) using CO₂, CH₄ and N₂O averages from each consecutive 14-day period:

$$\text{GWP} = \frac{\text{CO}_2}{12} \times 44 + \frac{\text{N}_2\text{O}}{28} \times 44 \times 264.5 + \frac{\text{CH}_4}{12} \times 16 \times 28.5 (\text{g CO}_2\text{eq. ha}^{-1}), \quad (4)$$

where GWP Global Warming Potential ($\text{g ha}^{-1} \text{day}^{-1}$), CO₂ Carbon dioxide flux ($\text{g ha}^{-1} \text{day}^{-1}$), 12 the atomic mass of C in CO₂ molecule, 44 the atomic mass of CO₂, N₂O Nitrous oxide flux ($\text{g ha}^{-1} \text{day}^{-1}$), 28 the atomic mass of N in N₂O molecule, 44 the atomic mass of N₂O, 264.5 atmospheric warming potential of N₂O compared to CO₂ in a 100-year time frame, CH₄ Methane flux ($\text{g ha}^{-1} \text{day}^{-1}$), 12 the atomic mass of C in CH₄ molecule, 16 the atomic mass of CH₄, 28.5 atmospheric warming potential of CH₄ compared to CO₂ in a 100-year time frame.

Statistical analyses

The assumptions of normality and equality of variance were met after checking the measured parameters (Soil C and N parameters, GHG parameters and global warming potential

calculations). All parameters were subjected to analysis of variance using Statistical Analysis System (SAS) (SAS Enterprise, Cary, North Carolina, USA). The model tested the effect of compost, moisture and compost * moisture interaction on the measured parameters separately for each temperature regime. Fischer's protected least significant difference (LSD) was used to separate means at 5% probability.

Results and discussion

Effect of the applied treatments on soil DOC dynamics and CO₂ emission

Soil % WFPS was an important driver of many biochemical transformations during the stepwise soil warm-up simulating the transition between seasons. Even at the lowest temperature (5 °C, late winter) (Table 2), increasing the soil % WFPS by 2 (from 5% WFPS to 7% WFPS), appeared to have the greatest impact on CO₂ emissions; either due to the liberation of CO₂ trapped in soil macropores (Bista et al. 2017) or chemical transformations of inorganic carbonates (Smart and Penuelas 2005).

The observance of no significant differences in DOC concentration with water increment under only late spring and early summer temperatures (Table 3) but not the late winter and early spring temperatures indicate the predominance of the above- mentioned processes than increased microbial activity and CO₂ fluxes with water pulses in the semi-arid soil as reported by Austin et al. (2004) and Norton et al. (2008).

Though temperature increment has demonstrated accelerated SOM decomposition and respiration (Filep and Rekasi 2011), doubling soil temperature from late winter to early spring temperature (Fig. 1a) did not raise the magnitude of CO₂ emissions far above that of the late winter (Table 2). This was because the temperatures were too low for significant increases in soil respiration and microbial activity as temperature ranging from 30 to 35 °C have been reported to be optimum for significant CO₂ emission in such soils (Guntinas et al. 2013; Dilekoglu and Sakin 2017). Higher CO₂ fluxes with increment in soil moisture in late spring (Fig. 1b) was the effect of the increased microbial activity with water pulses (Austin et al. 2004) especially under the relatively higher temperature. The increased microbial activity further made more soil DOC available (Table 3), which served as substrate (Filep and Rekasi 2011) for higher CO₂ emission.

Increase in temperature from late spring (15 °C) to early summer (25 °C) (Table 3) caused a more than Q₁₀ effect (Guntinas et al. 2013) on soil DOC concentration from all % WFPS levels. (Q₁₀ refers to the doubling of the rate of a bio/chemical reaction per every 10° rise in temperature, Sterratt 2014). However, contrary to the general concept that,

Table 2 Effect of soil moisture (5%, 7% and 15% of WFPS) on CO₂ emissions in late winter (5 °C)

Soil moisture (% WFPS)	CO ₂ (μg g ⁻¹ day ⁻¹)
5	0.07 ± 0.13 (c)
7	0.17 ± 0.15 (b)
14	0.20 ± 0.14 (a)

Lower-case letters indicate significant differences between means at minimum $p \leq 0.05$

Numbers following “±” are standard errors of the mean

Table 3 Effect of soil moisture (5%, 7% and 15% of Water Filled Pore Space, WFPS) on soil dissolved organic C concentrations in late spring and early summer

Soil moisture (% WFPS)	Dissolved organic C (μg g ⁻¹ day ⁻¹)	
	Late spring	Early summer
5	22.03 ± 0.35(b)	131.24 ± 1.76 (a)
7	33.67 ± 0.36 (a)	90.24 ± 1.74 (b)
14	34.12 ± 0.69 (a)	90.39 ± 2.24 (b)

Lower-case letters within each row indicate significant differences between means at $p \leq 0.05$

Numbers following “±” are standard errors of the mean

temperature has greater effect on soil activity in wetter soils (Guntinas et al. 2013; Cable et al. 2011, Craine and Gelderman 2011), soil DOC concentration from 5% WFPS (Table 3) was significantly higher than the wetter soil moistures (7 and 14% WFPS) under the early summer temperature (Table 3). Craine and Gelderman (2011) had similar findings and concluded that more research was required to clarify the situation. The decreased soil DOC concentration with increase in moisture in the early summer (Table 3), did not have a direct effect on CO₂ emission under the same temperature (Fig. 1c), similar to the findings of Guntinas et al. (2013).

Guntinas et al. (2013) also concluded from similar findings that variations in soil moisture alter the temperature sensitivity of the decomposition process and may either amplify or reduce the direct effects of temperature on soil respiration. The general increase in soil DOC concentration under the summer temperature (Table 3) compared to the late spring temperature, created more substrate availability (Filep and Rekasi 2011) for higher soil respiration and consequently higher CO₂ emission in the early summer temperature (Fig. 1c).

Effect of the applied treatments on soil inorganic N dynamics and N₂O emission

The increasing emissions of N₂O with increasing moisture from the control and 45 Mg ha⁻¹ compost treatments under late winter temperature (Fig. 2a) may be due to enhanced

reduction processes with the creation of anaerobic microsites by moisture addition (Bergsma et al. 2011).

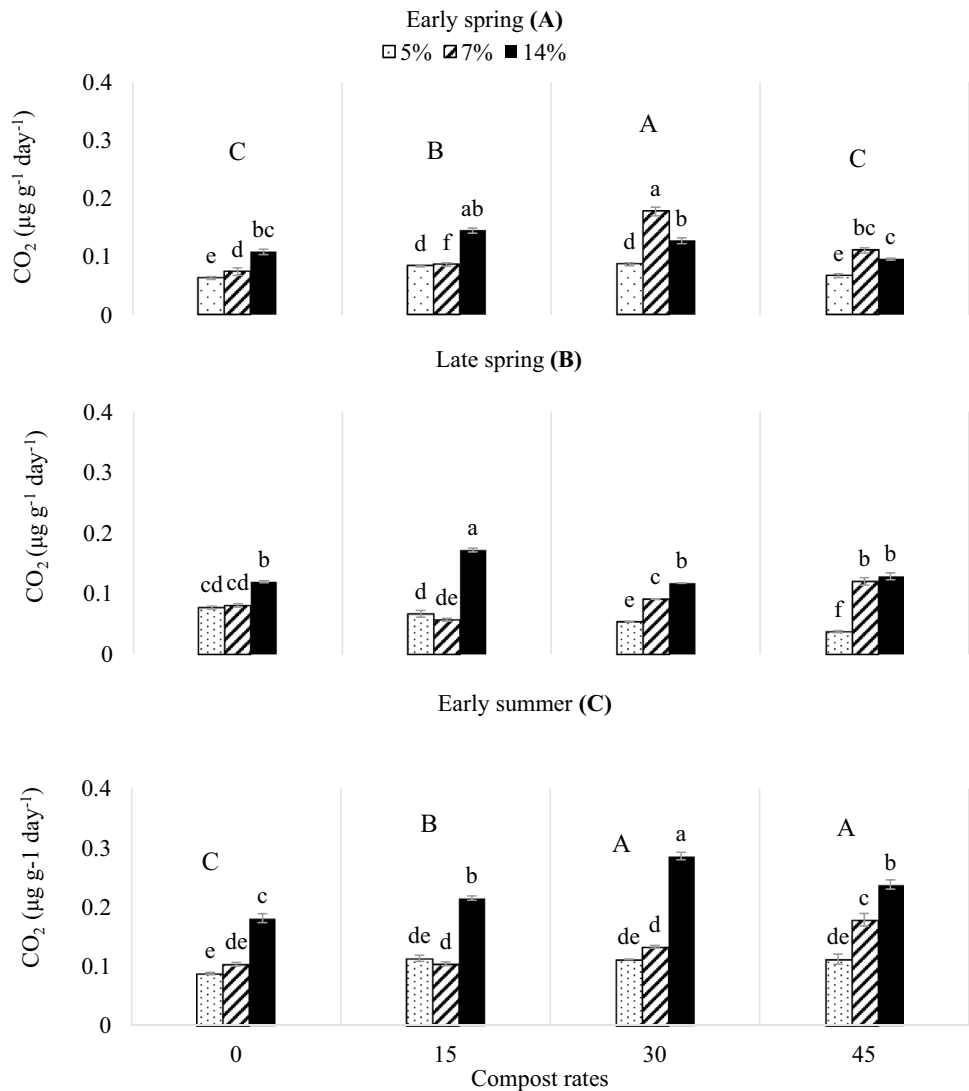
This is consistent with the findings of Weier et al. (1993), who observed an increase in N₂O emissions with increase in WFPS of soil and attributed it to denitrification. However, since the soils incubated in our study was not saturated (less than 80%; Schaufler et al. 2010), the predominant process affecting N₂O emissions must have been the process of nitrification. Nitrification has been reported to account for 61–98% of N₂O produced in moist soils of a semi-desert (Mummey et al. 1994). Moreover, the small magnitude of denitrification in the anaerobic microsites must still depend on potential nitrification rates, using NO₃⁻ as substrate (Schlesinger and Bernhardt 2013). Doubling soil moisture from 7% WFPS to 14% WFPS was likely associated with increases in anaerobic, oxygen-deprived microsites. Such loci host microorganisms that use alternative sources of oxygen and hence, carry out reduction processes (Bergsma et al. 2011). The burst out in N₂O emission with the control at 14% WFPS under the late winter temperature simulation (Fig. 2a) was probably an artefact of the disturbance of the soil during the mixing of compost and soil right before the incubation. The disturbance caused temporary destruction or discontinuum of soil macro and micro pores and microbial mortality (Lehrsch et al. 2016). This confirms reports that soils need to be preincubated (Guntinas et al. 2013) before the start of an incubation experiment.

Contrary to the findings that supply of nitrogen through organic amendments increases N₂O emissions (Deng et al. 2017), the highest compost rate (45 Mg ha⁻¹) did not emit the highest amount of N₂O during the early spring temperature simulation (Fig. 2b) like the 15 Mg ha⁻¹ compost. This suggests the ability of the highest compost rate to effectively improve the nutrient holding capacity of the soil and retain N, preventing its loss as N₂O.

Response of inorganic N to organic N additions from compost during the summer (Table 4) is the effect of major microbially mediated processes which are enhanced by increased temperature in the summer season (Lehrsch et al. 2016). The addition of the highest compost rate (45 Mg ha⁻¹) during the summer (Table 4) enhanced SOM and the high temperature boosted microbial activity which fueled the highest N mineralization also observed by Masunga et al. 2016.

Other studies have shown similar results where N mineralization increased with SOM addition (Campbell and Souster 1982) and temperature (Stanford et al. 1973). Under the same summer temperature simulation, the 45 Mg ha⁻¹ compost coupled with 14% WFPS resulted in the highest N₂O emissions (Fig. 2D) mainly because of N availability from the compost rate (Table 4) (Deng et al. 2017) and the sufficient supply of moisture and temperature for active

Fig. 1 Effect of the interaction between compost rates and soil moisture (5, 7 or 14% WFPS) on CO₂ fluxes from soil during **a** early spring (10 °C), **b** late spring (15 °C) and **c** early summer (25 °C). Vertical axes represent CO₂ emissions (μg g⁻¹ day⁻¹). Horizontal axes represent compost rates at 0, 15, 30 and 45 Mg ha⁻¹. Lower-case letters indicate significant differences at $p \leq 0.05$. Capital letters indicate significant differences between main compost treatments at $p \leq 0.05$



nitrification and microsite denitrification (Kurganova and de Gerenyu 2010).

Effect of the applied treatments on CH₄ flux

The emission of CH₄ rather than the usual assimilation by dryland soils (Bista et al. 2017) during the late winter temperature simulation (Fig. 3a) was an artefact of soil and compost mixing, similar to the effect caused by land preparation and tilling compost in field conditions (Grant 1997). The disturbance of the soil caused temporary destruction or discontinuum of soil macro and micro pores and microbial mortality (Lehrsch et al. 2016), resulting in higher CH₄ emissions.

Generally, upland soils act as sinks for atmospheric CH₄ (Pachauri et al. 2014). The highest CH₄ assimilation rate by the normal moisture level (7% WFPS) under early

spring, late spring and early summer temperature simulations (Fig. 3b–d) demonstrates the positive impact relatively warmer temperatures and adequate moisture supply have on the activity of methanotrophs (Potter et al. 1996) to assimilate CH₄. Doubling WFPS to from 7 to 14% created an environment at which a possible transition from methanotrophy to methanogenesis might have occurred, leading to the reduction of CH₄ assimilation rate with moisture addition (Fig. 3b–d).

Global warming potential of the applied treatments

Contributions from different GHGs to GWP differed seasonally and in response to the compost and soil moisture treatments (Table 5). Pulses in soil moisture generated a burst in CO₂ and N₂O emissions which in turn, caused increased GWP and cumulative GWP (Thangarajan et al. 2013). The highest CH₄ assimilation at 7% WFPS partially



Fig. 2 Interaction between compost (0,15, 30 and 45 Mg ha⁻¹) and soil moisture (WFPS of 5, 7 and 14%) on N₂O fluxes during **a** late winter (5 °C), **b** early spring (10 °C), **c** late spring (15 °C) and early summer (25 °C). Vertical axes represent N₂O emissions (μg g⁻¹ day⁻¹). Horizontal axes represent compost rates at 0, 15, 30 and 45 Mg ha⁻¹. Lower-case letters indicate significant differences at *p* ≤ 0.05

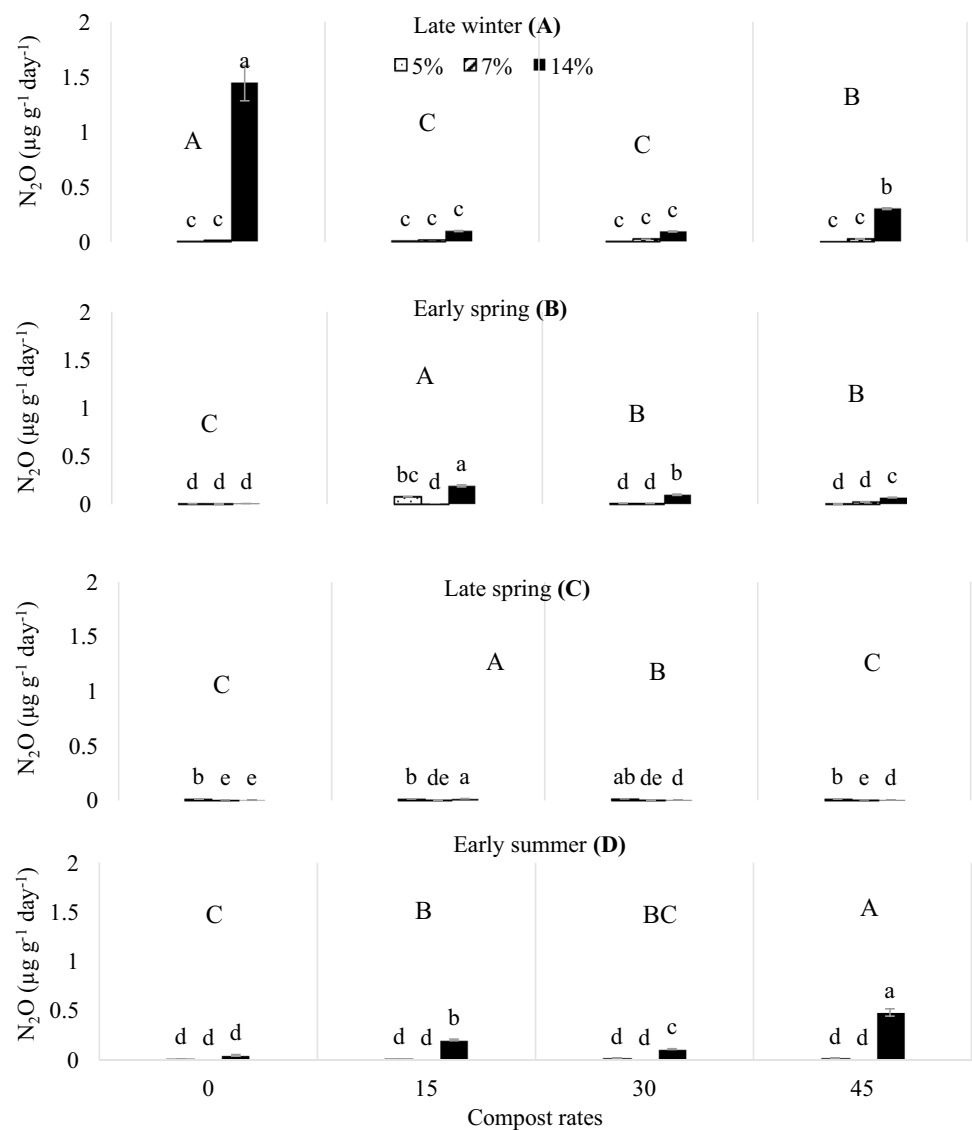


Table 4 Effect of compost (0, 15, 30 and 45 Mg ha⁻¹) on soil inorganic nitrogen (inorganic N) concentrations in early summer (25 °C)

Compost (Mg ha ⁻¹)	Inorganic N (μg g ⁻¹ day ⁻¹)
15	6.71 ± 0.04 (b)
30	7.49 ± 0.05 (ab)
45	7.72 ± 0.04 (a)

Lower-case letters indicate significant differences between means at minimum *p* ≤ 0.05

Numbers following “±” are standard errors of the mean

offsets negative effects of CO₂ and N₂O on GWP. Interestingly, cumulative GWP was only affected by soil moisture, and especially by 14% WFPS. Since soils in Wyoming are rarely at 14% WFPS, soils amended with high rates of

compost do not make significant contributions to global warming and instead, are able to improve soil health.

Conclusions

One-time application of compost at 30 and 45 Mg ha⁻¹ are not a significant source of GHGs provided soils do not experience extended episodes of wetting in a particular season of the year. They rather release high amounts of soil N beneficial to plants’ growth during warmer periods of the year. In addition, GWP depends more on soil moisture than compost.

Contributions of individual gas species to GWP vary CO₂ increases with soil moisture regardless of the temperature. Nitrous oxide and CH₄ respond to disturbance even during the cold months and increase with increasing temperature

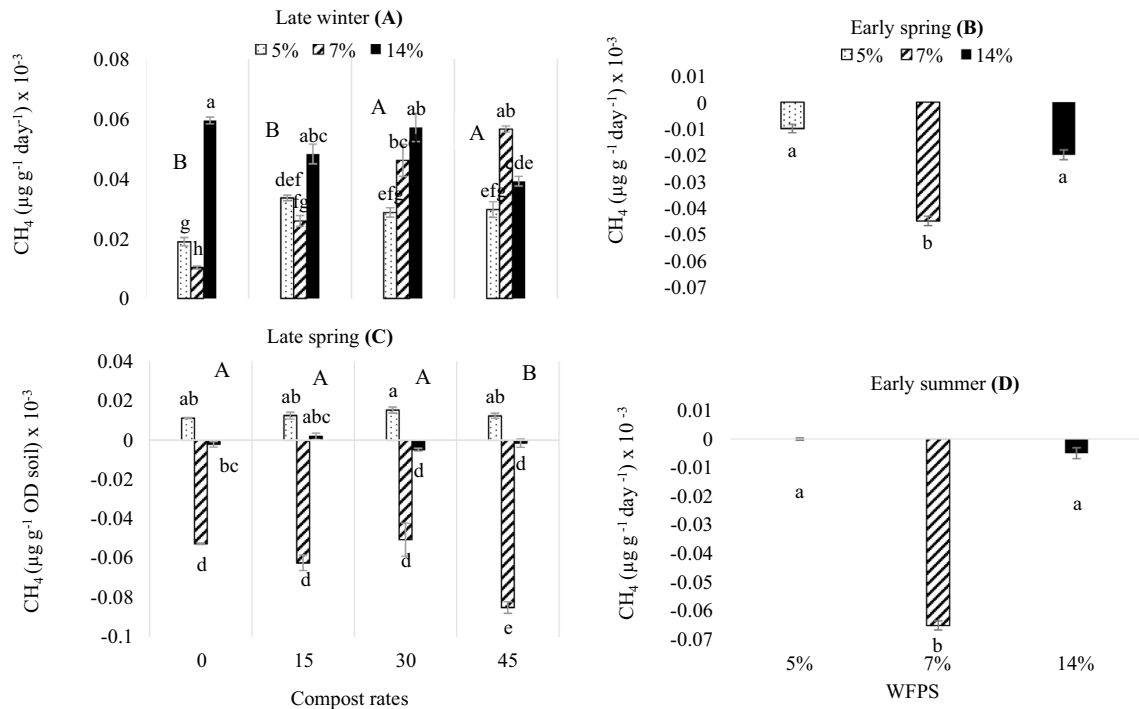


Fig. 3 Effect of the interaction between compost rates (0,15, 30 and 45 Mg ha⁻¹) and soil moisture (5%, 7% and 14% of WFPS) on CH₄ fluxes during **a** late winter and **b** late spring and soil moisture alone on CH₄ fluxes in early spring and early summer. Lower-case letters

indicate significant differences at $p \leq 0.05$. Vertical axes represent CH₄ emissions ($\mu\text{g g}^{-1} \text{day}^{-1}$). Horizontal axes represent compost rates at 0, 15, 30 and 45 Mg ha⁻¹. Capital letters indicate significant differences in the compost and control treatments at $p \leq 0.05$

Table 5 Effect of compost rates (0, 15, 30 and 45 Mg ha⁻¹) and soil moisture (5%, 7% and 14% of WFPS) interaction on the global warming potential of the soil during late winter, early spring, late spring and early summer

% WFPS	Global warming potential ($\mu\text{g CO}_2 \text{ eq. g}^{-1} \text{ soil} \times 10^3$)			
	Late winter (5 °C)	Early spring (10 °C)	Late spring (15 °C)	Early summer (25 °C)
Control (0)				
5	1497 ± 80 (k)	2114 ± 97 (k)	5089 ± 169 (hk)	3206 ± 58 (k)
7	5983 ± 116 (h)	1769 ± 165 (k)	1527 ± 44 (k)	1425 ± 30 (k)
14	600,668 ± 32,501(a)	3247 ± 450 (hk)	6765 ± 552 (hk)	19,271 ± 3472 (g)
15 Mg ha ⁻¹				
5	3026 ± 87 (k)	32,740 ± 416 (f)	5685 ± 400 (hk)	3858 ± 100 (hk)
7	7581 ± 219 (hk)	1186 ± 14 (k)	1634 ± 42 (k)	1429 ± 65(k)
14	41,409 ± 1897 (e)	80,192 ± 4831 (d)	2103 ± 72 (k)	84,236 ± 3743 (d)
30 Mg ha ⁻¹				
5	1222 ± 66 (k)	33,788 ± 530 (f)	4773 ± 175 (hk)	6067 ± 94 (hk)
7	10,441 ± 317 (h)	3570 ± 427 (hk)	1537 ± 108 (fg)	1233 ± 33 (k)
14	40,085 ± 2014 (e)	41,702 ± 3080 (e)	2703 ± 76 (c)	45,751 ± 1738 (e)
45 Mg ha ⁻¹				
5	973 ± 43 (k)	1714 ± 136 (k)	4935 ± 136 (hk)	5920 ± 327 (hk)
7	10,605 ± 123 (h)	10,395 ± 94 (h)	1307 ± 62 (k)	1153 ± 81 (k)
14	126,086 ± 3404 (c)	10,395 ± 1680 (h)	1527 ± 14 (k)	200,246 ± 15,351 (b)

Lower-case letters indicate significant differences between means at minimum $p \leq 0.05$
 Numbers following “±” are standard errors of the mean

and soil moisture. Dryland soil at normal moisture levels is the most effective CH₄ sink.

Hence, since dryland soil of the NHP are often dry (5% WFPS) or normal (7% WFPS) for most parts of the year, the one-time application of the high compost rates will not pose significant danger to the environment through greenhouse gas emission and global warming.

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