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This study investigated the effects of different mixing ratios of crop residues and biochar with liquid digestate from anaerobically treated pig manure on CH_4, CO_2, and N_2O emissions over 84 days in a system of passive aeration composting, resembling typical Vietnamese solid manure storage conditions. Two treatments with solid manure were included for comparison. The results showed that C losses through CH_4 and CO_2 emissions accounted for 0.06–0.28% and 1.9–26.7%, respectively, of initial total C. CH_4 losses accounted for just 0.4–4.0% of total C losses. Total N losses accounted for 27.1–40% of initial total N in which N_2O emissions corresponded to 0.01–0.57% of initial total N, and hence accounted for only 0.1–1.8% of total N losses. It is assumed that the remainder was either the result of denitrification losses to N_2 or ammonia volatilization. The composting of biochar (B) or crop residue with digestate (D) showed significantly lower CH_4 and N_2O emissions compared with composting manure (M) ($p < .05$). The composting of digestate with biochar showed significantly lower CO_2 and CH_4 emissions and significantly higher N_2O emissions compared to the composting of digestate with rice straw (RS) ($p < .05$). The combined composting of digestate with biochar and rice straw (D + B + $RS_{5:0.3:1}$) showed significantly reduced N_2O emissions compared with composting digestate with biochar with alone ($p < .05$). Composting sugar cane bagasse (SC) with digestate (D + SC) significantly reduced CH_4 and N_2O emissions compared with the composting of rice straw with digestate (D + $RS_{3.5:1}$ and D + $RS_{5:1}$) ($p < .05$).

Keywords: GHG emissions; composting; biochar; digestate; crop residue

1. Introduction

The Vietnamese agricultural sector produced 26.5 million pigs in 2012.[1] A pig excretes on average 2 kg of solid manure (faeces) per day during 4 months growing time.[2,3] This means that in 2012 alone, Vietnam's annual pig production with 26.5 million pigs produced around 6.4 million tonnes of pig manure. Furthermore, 43.7 million tonnes of rice grain and a similar amount of rice straw were produced in 2012.[4] Inappropriate management of these agricultural residues not only represents the loss of a valuable resource, but also contributes to increasing environmental pollution and greenhouse gas (GHG) emissions.

Animal manure management in Vietnam is currently facing severe problems related to environmental pollution through handling and management processes such as the uncontrolled discharge of voluminous liquid digestate, a by-product of fermentation of manure in biodigesters, into watercourses.[5] This is due to logistical problems with the handling of this bulky waste product and a lack of knowledge about its real fertilizer value.[5] The digestates obtained from biodigesters will contain less organic matter

due to conversion of the easily digestible organic fraction into biogas, leaving the most recalcitrant fraction.[6] Liquid digestate application to soil will therefore contribute to reducing GHGs by having lower carbon degradability than raw manure. However, liquid digestate still contains pathogens,[7] especially as a result of mesophilic or lower process temperature, and too short retention times due to high loading rates of washing water entering the biodigester. Liquid digestate therefore needs to be treated and concentrated before being applied to crops, e.g. by composting it with rice straw to eliminate pathogens. In addition, rice straw will potentially reduce the leaching and volatilization loss of nitrogen (N) because N from digestate is immobilized by micro-organisms.

In terms of crop residue management, Hoang [8] reported that 40% of total rice straw was burnt directly on the field after harvesting while 30% of total rice straw was stored to feed animals such as cow or buffalo in the winter when grass is scare and the rest (30%) was used for animal bedding. Burning rice straw in the field is a common farmer practice, because it is not feasible

to incorporate it into the soil without having a negative biological impact on the subsequent crop and transporting rice straw from field to the home is often too labour intensive. However, burning straw results in emissions of large amounts of smoke particles and other polluting gases and furthermore depletes soil organic matter levels. Rice straw should ideally be treated in a way that ensures stable organic carbon inputs to the soil with minimal emissions and low labour costs. Handling and management processes such as pyrolysis of the straw – converting it into a biochar instead of returning it directly to the paddy field – would be a promising method for reducing CH_4 emissions and increasing the rice yield.[9] Biochars derived from rice straw have a high nutrient-holding capacity (cation exchange capacity: 40–80 cmol kg^{-1}) and a high surface area (51–900 m^{-2} g^{-1}).[10–12] Adding biochar to composting poultry manure may reduce nitrogen losses due to the enhanced NH_4^+ adsorption capacity of the biochar-amended composts.[13,14] Composting digestate with biochar may also reduce other N emissions, such as N_2O, because NH_4^+ and NO_3^- will be absorbed onto the biochar surface, preventing them from entering anaerobic microsites, and consequently reduces N_2O emissions.[15] Sommer and Møller [15] showed that deep litter manure with a high amount of barley straw provided in the pig house increased porosity and air exchange in the subsequent composting heap, resulting in reduced CH_4 and N_2O emissions. However, there has been little research to date on the potential use of rice straw or biochar derived from rice straw or their mixing ratios for treating digestate and reducing GHGs emission from the composting process in comparison with traditional manure composting.

This study was therefore conducted with the aim of quantifying how different mixing ratios of sugar cane bagasse, rice straw and/or biochar with biogas digestate and solid pig manure affect the emissions of CO_2, CH_4, and N_2O in a system of passive aeration composting for nearly 3 months, resembling typical Vietnamese solid manure storage conditions on small-scale farms. We hypothesized that (i) lower GHG emissions occur from the composting of digestate with biochar than from the composting of digestate with rice straw and from the composting of solid pig manure, and (ii) higher GHG emissions occur from composting digestate with rice straw compared to sugar cane bagasse.

2. Materials and methods

2.1. Experimental site and study materials

The experiment was conducted from 10 April to 3 July 2012 (84 days) in a greenhouse belonging to the Soils and Fertilizers Research Institute located in Tu Liem district, Hanoi city.

Pig manure (mainly faeces) was collected from a medium-sized livestock farm located in Tu village, Luong Phuong commune, Hiep Hoa District, Bac Giang Province

where pigs are fed commercial feed relatively rich in protein compared with traditional feed used by small-scale farmers. Digestate, the by-product of the anaerobic digestion of pig urine and faeces, together with water used in the animal house for floor cleaning, was collected from the digestate storage tank on a livestock farm with a biogas digester system located in Tu village, Luong Phuong commune, Hiep Hoa District, Bac Giang Province. The digestate was collected using an electric pump and slight stirring, simulating the normal outflow from the digester. Rice straw was collected from the summer rice season and kept dry for 6 months before use. One week before the experiment was initiated; rice straw was air-dried to ensure a homogeneously low moisture content (humidity 15%) and chopped into pieces of 8–12 cm in length with a knife. Biochar was produced from rice straw by pyrolysis of dry rice straw at 450°C in a closed iron reactor, externally heated by rice straw burning, with a char yield of 42% of added straw C. The biochar production reactor was made from heat-resistant bricks, 180 cm high and 120 cm in diameter (ca 2000 L) and equipped with a thermometer and a chimney. Positioned 5 cm from the bottom were four adjustable vents for temperature control. Inside the large reactor, were placed six small reactors (60 cm high, 40 cm diameter, 75 L) made from stainless steel, with a small hole (2 cm in diameter) in the lid to release vapour and smoke produced as the biochar was being made. For biochar production, the small reactors were initially filled with rice straw (9.6 kg per reactor). The six small reactors were then put upside down in the large reactor. Dry rice straw was put around the small reactors as burning material (7.5 kg per large reactor). Rice husk was then filled to a depth of 20 cm from the small reactor to the surface of the large reactor. The rice husk reduces smoke and retains heat during the biochar-producing process. Ignition took place through a small window (10 cm in diameter) at the bottom of the large reactor. Black smoke appeared during the first 15–20 min of burning. When white smoke appeared, the chimney was sealed with an iron lid put on top of the large reactor. The pyrolysis process continued for the next 35–45 min. Finally, the six small reactors were removed from the large reactor and were cooled by water, before taking out the biochar produced.

Sugar cane bagasse, a by-product of sugar production, was collected from a Lam Son sugar Factory located in Tho Xuan District, Thanh Hoa Province, air-dried (humidity 15%) and chopped into pieces of 8–12 cm in length with a knife. For treatments with the combined composting of digestate with crop residue or biochar, these were mixed and left to soak for 24 h to ensure that all the digestate had been absorbed before commencing the compost experiment. Then 43 kg (wet weight) of the mixture of digestate and crop residue or biochar were added to each reactor. The materials from treatments with combined composting of manure with rice straw or treatments with manure composting only were also adjusted to 43 kg (wet weight) in

Table 1. Overview of the treatments, mixing ratios and quantity[a] of the different compost formulations in each of the reactors.

Treatments	Abbrev.	Ratio	Manure (kg)	Digestate (L)	Rice straw (kg)	Sugar cane bagasse (kg)	Biochar (kg)
Solid manure	M		43.0				
Manure + rice straw	M + RS	20:1	41.0		2.0		
Digestate + rice straw	D + RS	5:1		35.8	7.2		
	D + RS	3.5:1[b]		33.4	9.6		
Digestate + sugar cane	D + SC	5.5:1[c]		36.4		6.6	
Digestate + biochar	D + BC	3.5:1[d]		33.4			9.6
Digestate + rice straw	D + RS + BC	8.5:1:1		34.8	4.1		4.1
+ biochar	D + RS + BC	5:0.3:1		34.1	2.0		6.8

[a]Quantities given for manure and digestate as wet weight, whereas for crop residues as air dry weight.
[b]Max. digestate (in kg, equivalent to L) absorption capacity of 1 kg rice straw.
[c]Max. digestate (in kg, equivalent to L) absorption capacity of 1 kg sugar cane bagasse.
[d]Max. digestate (in kg, equivalent to L) absorption capacity of 1 kg biochar from rice straw.

Table 2. Properties of materials investigated before and after 84 days of passive aeration composting.

Treatments	Bulk density (kg m^{-3}, ww)	Moisture (% wb)	Dry mass (kg dm reactor^{-1})	Ash (g kg^{-1} dm)	EC (d Sm^{-1})	pH	Tot-C (g kg^{-1} dm)	Tot-N (g kg^{-1} dm)	C/N
				Before composting					
M	680 (15)	81.0 (1.0)	8.16 (0.1)	212 (1.3)	2.55 (0.06)	7.0 (0.1)	469 (4.2)	38.6 (0.3)	12.2 (0.2)
M + RS	580 (20)	74.8 (1.2)	10.82 (0.1)	180 (0.4)	1.83 (0.06)	7.1 (0.1)	488 (2.0)	31.8 (0.5)	15.3 (0.3)
D + RS (5:1)	490 (10)	77.3 (0.3)	9.36 (0.1)	165 (2.4)	1.43 (0.08)	8.7 (0.1)	482 (5.3)	21.6 (2.3)	22.3 (2.7)
D + RS (3.5:1)	440 (17)	77.0 (2.0)	9.89 (0.1)	201 (1.2)	1.70 (0.10)	8.8 (0.1)	453 (4.2)	22.5 (0.2)	20.1 (2.0)
D + SC	420 (20)	80.0 (1.0)	8.61 (0.1)	46 (4.0)	0.73 (0.06)	6.7 (0.2)	545 (3.5)	18.2 (1.1)	29.9 (2.9)
D + B	780 (10)	84.1 (1.0)	6.85 (0.1)	345 (1.6)	4.81 (0.11)	10.7 (0.2)	383 (6.5)	22.6 (0.1)	16.9 (0.3)
D + RS + B (8.5:1:1)	570 (20)	80.7 (0.2)	8.30 (0.1)	232 (0.5)	1.81 (0.08)	9.8 (0.2)	453 (4.2)	22.1 (1.1)	20.5 (1.2)
D + RS + B (5:0.3:1)	650 (20)	81.5 (0.5)	7.94 (0.1)	281 (0.3)	2.30 (0.06)	10.0 (0.1)	425 (6.2)	20.3 (3.3)	21.3 (3.7)
LSD (0.05)	26.9	1.9	0.14	3.12	0.14	0.26	6.9	2.7	3.7
				After composting					
M	634 (10)	80.6 (0.3)	6.27 (0.1)	246 (18)	2.99 (0.1)	7.2 (0.2)	440 (6.4)	29.1 (0.1)	15.1 (0.2)
M + RS	560 (20)	74.4 (0.1)	8.54 (0.6)	250 (1.5)	2.76 (0.1)	7.3 (0.2)	425 (1.3)	27.9 (1.0)	15.2 (0.1)
D + RS (5:1)	510 (17)	77.3 (0.3)	7.23 (0.6)	263 (2.5)	1.70 (0.1)	9.0 (0.1)	427 (5.5)	20.4 (2.4)	20.9 (1.9)
D + RS (3.5:1)	440 (10)	76.9 (0.1)	6.99 (0.6)	322 (5.8)	1.82 (0.1)	9.1 (0.1)	380 (13.6)	22.2 (2.0)	17.1 (0.7)
D + SC	430 (5)	80.5 (0.5)	6.10 (0.3)	107 (7.6)	0.96 (0.2)	7.1 (0.3)	500 (8.8)	17.5 (1.7)	28.6 (1.8)
D + B	650 (20)	82.4 (1.4)	5.71 (0.7)	357 (4.8)	3.58 (0.2)	10.8 (0.1)	368 (1.5)	16.8 (4.3)	22.8 (5.2)
D + RS + B (8.5:1:1)	530 (10)	80.1 (1.0)	6.62 (0.5)	339 (5.3)	2.02 (0.1)	9.9 (0.2)	385 (5.0)	19.0 (1.7)	20.4 (1.6)
D + RS + B (5:0.3:1)	600 (20)	80.8 (0.1)	6.45 (0.5)	314 (3.6)	2.53 (0.3)	10.3 (0.1)	390 (17.8)	17.2 (1.6)	22.8 (1.3)
LSD (0.05)	22.4	1.2	0.95	12.8	0.28	0.3	17	3.4	3.7

Notes: Average and (std. dev., $n = 3$). Treatments: M = Manure, RS = Rice Straw, D = Digestate, SC = Sugar cane bagasse, B = Biochar derived from rice straw, dm = dry matter.

total for each reactor. All reactors just were fed one time at the beginning of the experiment. The mixing ratio and weight of study materials from each of treatments are given in Table 1. The properties of the studied materials can be found in Table 2.

2.2. Composting and gas sampling

2.2.1. Compost reactor design

The composting reactors were 120 L cylindrical (slightly conical) plastic containers (diameter: 52 (top)/41 (bottom)

cm, depth 63 cm) with airtight lids (Figure 1). To maintain heat energy generated during composting, the reactors were insulated in a square box made of polystyrene (wall thickness 5 cm, with the void between the box and the reactor filled with Styrofoam chip packaging material, yielding an average insulation thickness of 10 cm). A rubber septum and two mini fans (12 V) were installed at the top of the head space of each chamber to ensure a homogeneous air mixture during sampling time (fans not running in between). A pressure control (plastic tube: 7.6 m length and 1.5 mm diameter) was also installed to maintain an
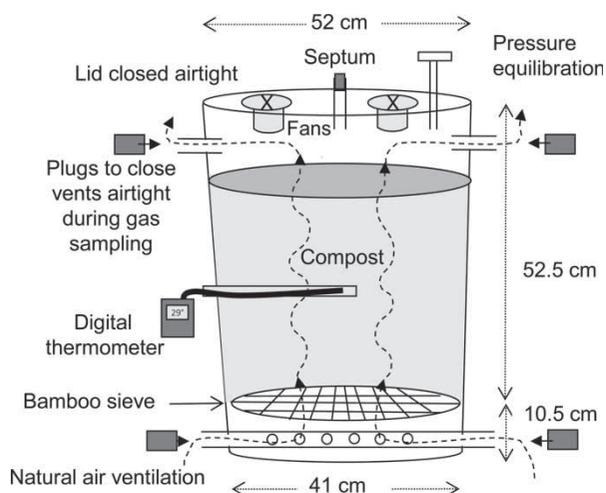
Figure 1. Chamber design for passive aeration composting and gas sampling.

equilibrium gas pressure between the inside and outside of the chamber and to minimize mixing of the internal chamber gases with the exterior atmosphere during closure of the reactor for gas measurements.[16] A bamboo sieve was positioned 10.5 cm above from the bottom of the reactor to ensure aeration of the composting materials at the bottom of the reactor. Two plastic tubes (3 cm in diameter) were connected with the bottom of the reactor to allow natural ventilation and two other plastic tubes were connected with the head space of the reactor to ensure natural ventilation and the circulation of gas in each reactor. Ventilation tubes were closed airtight with rubber plugs during measurements of gas flux. One small plastic tube was placed in the centre of the compost heap and an electronic thermometer inserted through to the middle of the composting reactor for the heap's daily temperature measurement at 10:00 am. The leachate was collected from the bottom of reactor through the bottom vent tube and was poured to the surface of the composting heap through by the top vent tube every week. The composting materials were not mixed during the composting process, as this is the farmer practice in the study site.

2.2.2. Gas sampling

GHG fluxes were determined using static flux chamber and gas chromatography techniques.

Gas samples were collected 13 times (days 1, 4, 7, 11, 15, 22, 29, 36, 43, 49, 59, 67, and 84) during the composting trial, and gas concentration accumulation was measured between 8:00 am and 11:45 am on each sampling day. Four gas samples were taken at 0, 20, 40, and 60 min (or at slightly longer intervals, based on flux rates) after closing the reactor. Gas samples were removed through the rubber septum using a 60 mL gas-tight syringe and a stainless steel hypodermic needle. The collected gas samples were immediately transferred into pre-evacuated vacuum glass containers (3 mL soda glass vial) and sent to the laboratory for analysis.

The gas samples were analysed by gas chromatography (Bruker 450-GC 2011). CO_2 was determined using a thermal conductivity detector at a temperature of 200°C, CH_4 was determined using a flame ionization detector at a temperature of 300°C and N_2O was determined using an electron capture detector at a temperature of 350°C. The oven temperature was set to 50°C. The carrier gas for CO_2 and CH_4 was He (99.999%), and for N_2O it was argon with 5% methane at a flow rate of 60 mL min^{-1}.

The emission rate in mg tonne^{-1} h^{-1} from compost initial dry matter (dm) was calculated by Equation (1) [15]

$$F = \left(\frac{\Delta C}{\Delta t}\right) * \left(\frac{v}{A}\right) * \left(\frac{M}{V}\right) * \left(\frac{P}{P_0}\right) * \left(\frac{273}{T}\right) * 60(\text{min})$$
$$* \left(\frac{A}{W}\right) * 1000(\text{mg}), \tag{1}$$

where F is the flux rate of the gas studied (mg tonne^{-1} compost in initial dm h^{-1}), ΔC the change in concentration of gas of interest in time interval Δt (20 min), v (m^3) the volume of the headspace in the reactor, M the molecular weight of the gas of interest (16, 44 and 44 for CH_4, CO_2, and N_2O, respectively), V the volume occupied by 1 mol of the gas at standard temperature and pressure (22.4 L), and T the temperature in Kelvin (°K). W is the total initial weight of compost material in dm (kg). The accumulated gas emission from each compost heap was calculated using the trapezoidal rule.[17]

The Global Warming Potential (GWP) over a 100-year period was calculated by multiplying the CH_4 emission rate (in mg CH_4) by a factor 25 and by a factor of 298 for N_2O to convert them into CO_2 equivalents.[18]

2.3. Analytical procedure

Samples were taken before and after the composting trial only, with three replications for each of the treatments. These were analysed for dry matter, ash, pH, EC, total N, and OC.

Dry matter was determined by drying at 105 °C for 24 h (Oven, Wiseven WOF, Daihan, Korea), and ash content by combustion at 600 °C for 5 h (Oven, Nabertherm, 214831, Germany). The pH was measured in a 1:5 manure:water suspension by pH meter (Hanna Hi 8424, Italy). Total N was measured by the Kjeldahl method (automatic Kjeldahl digestion Velp DKL and the semi-automatic steam distilling unit, UDK132, Velp Scientifica, Italy). EC was measured by EC portable meter (Hanna Hi 9033, Multi-range EC Portable Meter). OC was measured by the Walkley–Black method.[19]

2.4. Statistical analysis

The experimental layout was a completely randomized block design with eight treatments and three replications of

each treatment, resulting in 24 sub-reactors. The treatments are given in Table 1.

Statistical analyses of the results were performed by SAS 9.1 (SAS Institute, Inc., Cary, NC, USA). The effects of different organic mixtures on GHG emissions were examined by one-way ANOVA (PROC GLM). When the differences among treatments were significant ($p < .05$), the differences in means then were compared using the Duncan ($\alpha = 0.05$) post-hoc test for multiple comparisons.

3. Results

3.1. Evolution of physico-chemical characteristics of compost mixture

For the compost mixtures, the lowest initial bulk density was found in $D + RS_{3.5:1}$, $D + RS_{5:1}$ and $D + SC$ treatments and the highest initial bulk density were observed in $D + B$ and M treatments.

After 84 days of composting, dm losses varied from 17% to 29% of initial dm, depending on the treatments.

The dry mass was sharply reduced in treatments mixing digestate with high ratios of crop residues ($D + RS_{3.5:1}$, $D + RS_{5:1}$ and $D + SC$), whereas the opposite trends were observed in $D + B$ treatments. Total C losses were therefore the highest in $D + RS_{3.5:1}$ and the lowest in $D + B$ treatments, with 16% and 4% of initial Total C, respectively (Table 2).

At the beginning of the composting trial, the lowest C/N ratios were found in M, M + RS and D + B treatments and the highest pH value was found in treatments containing biochar (around pH 10), indicating the potential risk of nitrogen losses during the composting process in M, M + RS and D + B treatments with low C/N ratio.

3.2. Changes in temperature during the composting process

The peak temperature in all treatments was relatively low and ranged from 34 to 41°C during the composting process. The temperature in the two solid manure (M and
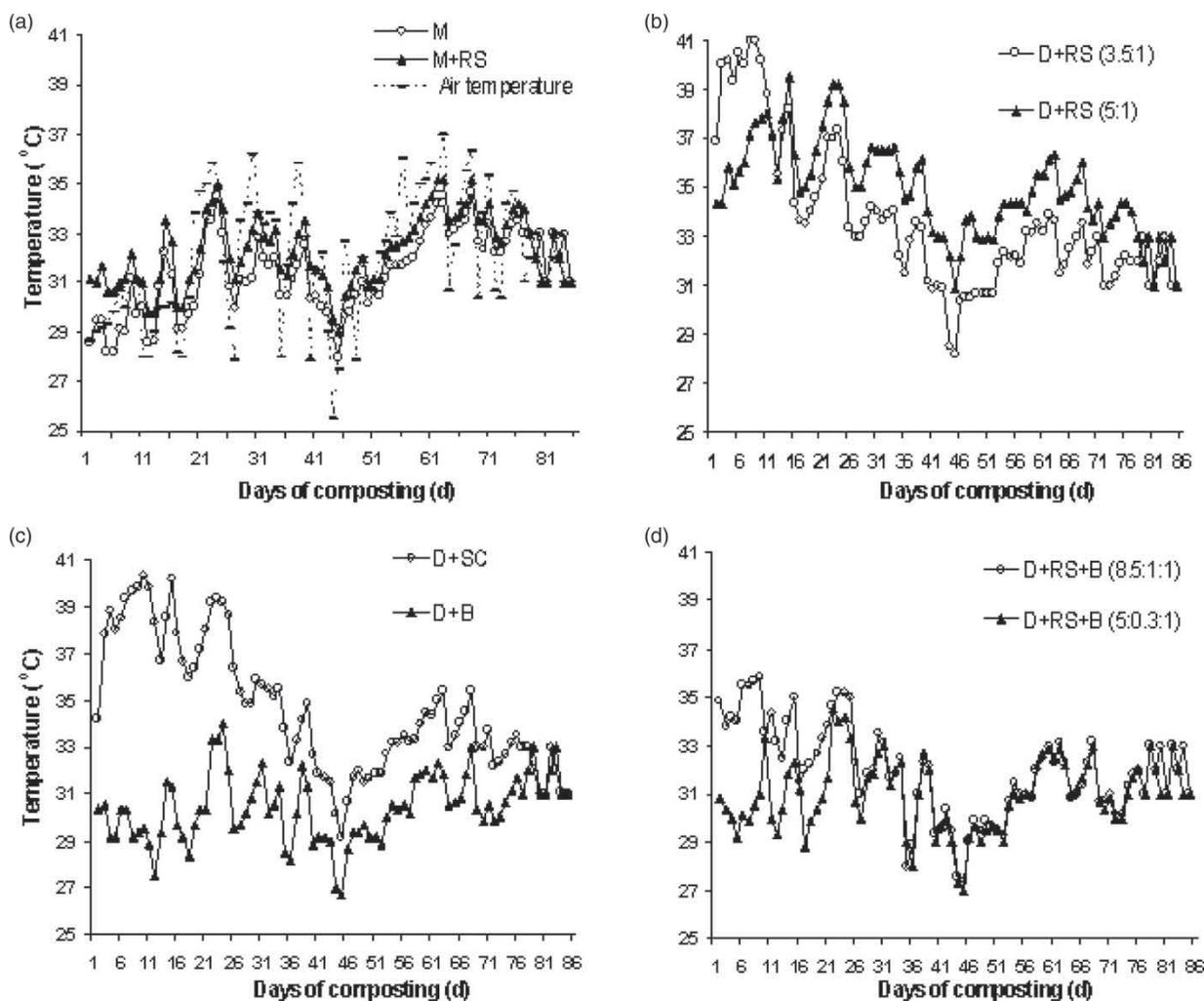
Figure 2. Evolution of temperature during 84 days composting process: (a) manure, manure + rice straw and ambient temperature, (b) digestate + rice straw, (c) digestate + sugar cane bagasse and digestate + biochar, (d) digestate + rice straw + biochar.
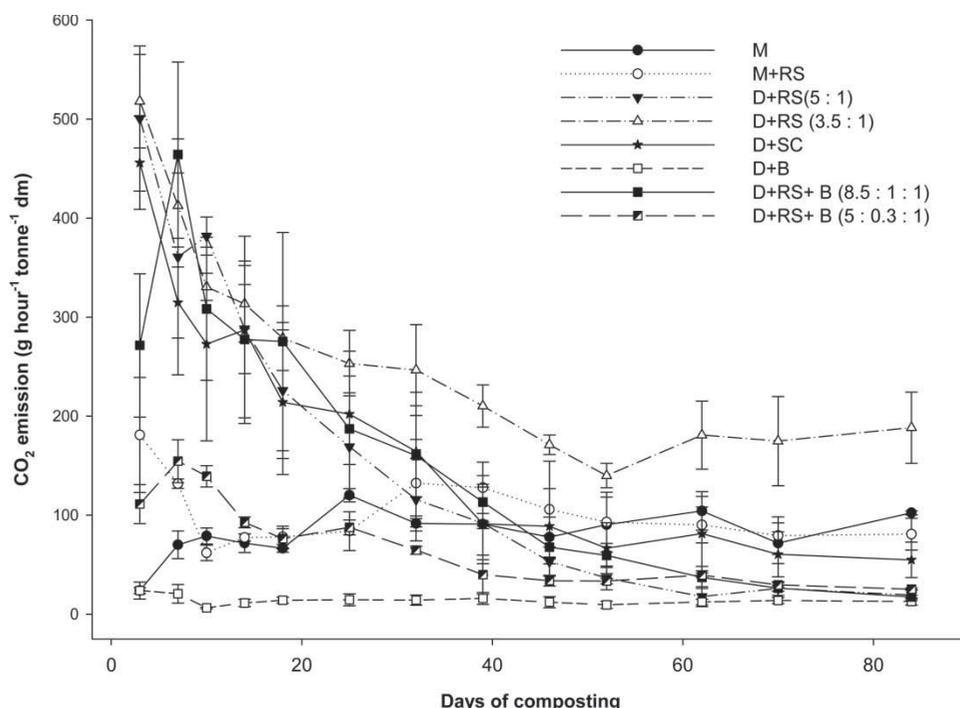
Figure 3. The variation in CO_2 emission for different composting treatments during 84 days of composting period. M = solid manure, RS = rice straw, D = digestate, SC = sugar cane bagasse, B = biochar.
Note: Bars indicate one standard error ($n = 3$).

M + RS) treatments varied from 29 to 33°C and was more or less equal to the ambient temperature throughout the 84 days (Figure 2(a)). During the first 21 days, temperatures in the digestate treatments with decomposable bulking materials, D + RS (Figure 2(b)) and D + SC treatments (Figure 2(c)), were 7–10°C higher than the ambient air temperature, indicating that the heat production from aerobic decomposition was greater than the heat loss, but after 21 days the temperature gradually declined and only stayed slightly above ambient for the rest of composting period. The digestate treatments containing biochar (Figure 2(d)) had a similar temperature development to the solid manures, except for the D + RS + $B_{8.5:1:1}$, containing a higher rice straw proportion, which showed a slightly elevated temperature for the first 12 days.

3.3. GHG emissions during the composting process

CO_2 emissions were generally high in treatments with a high mixing ratio of crop residue (D + $RS_{3.5:1}$ and D + SC), whereas the lowest CO_2 emissions were observed in the treatment without straw (D + B treatment) or with a low mixing ratio of crop residue (D + RS + $B_{5:0.3:1}$) (Figure 3).

CO_2 emissions increased sharply and reached a peak at day 3 from the M + RS, D + SC, D + $RS_{5:1}$, and D + $RS_{3.5:1}$ treatments, then gradually decreased until the last day of composting (day 84), except the M + RS treatment where CO_2 emissions reached a second peak at day 32 before decreasing to day 84 of the composting

process. CO_2 emissions also increased moderately and reached a peak at day 7 from the D + RS + $B_{5:0.3:1}$ and D + RS + $B_{8.5:1:1}$ treatments (Figure 3).

In general, CH_4 emissions peaked later than CO_2 in manure or digestate treatments mixed with rice straw (M + RS, D + $RS_{5:1}$ and D + $RS_{3.5:1}$), with CO_2 emissions peaking in those treatments from days 4 to 7. However, CH_4 emissions sharply increased and peaked at days 7–10 from the M + RS, D + $RS_{5:1}$, D + $RS_{3.5:1}$, and D + RS + $B_{5:0.3:1}$ treatments. CH_4 emissions from these treatments then generally decreased sharply from day 14 and remained low until the end of composting (Figure 4).

In contrast, CH_4 emissions slowly increased and reached a peak at days 25 and 39 from the D + SC treatment and D + B treatment, at 0.5 and 0.7 g h^{-1} $tonne^{-1}$ initial dm, respectively, and then decreased until day 84 of composting.

N_2O emissions were very low from the M + RS treatment during first 25 days of composting, and then sharply increased and reached a maximum of 472 mg h^{-1} $tonne^{-1}$ initial dm at day 32 of composting, before decreasing but being maintained at levels between 100 and 200 mg h^{-1} $tonne^{-1}$ initial dm (Figure 5). The same pattern was found for the M treatment which peaked a little later at day 62 of composting with 567 mg h^{-1} $tonne^{-1}$ initial dm.

For the D + $RS_{5:1}$, D + $RS_{3.5:1}$, D + B and D + RS + $B_{5:0.3:1}$ treatments, small peaks of N_2O emissions (60–70 mg h^{-1} $tonne^{-1}$ initial dm) occurred in the initial 18 days. In all other treatments, N_2O emissions were negligible throughout the 84 days of composting.
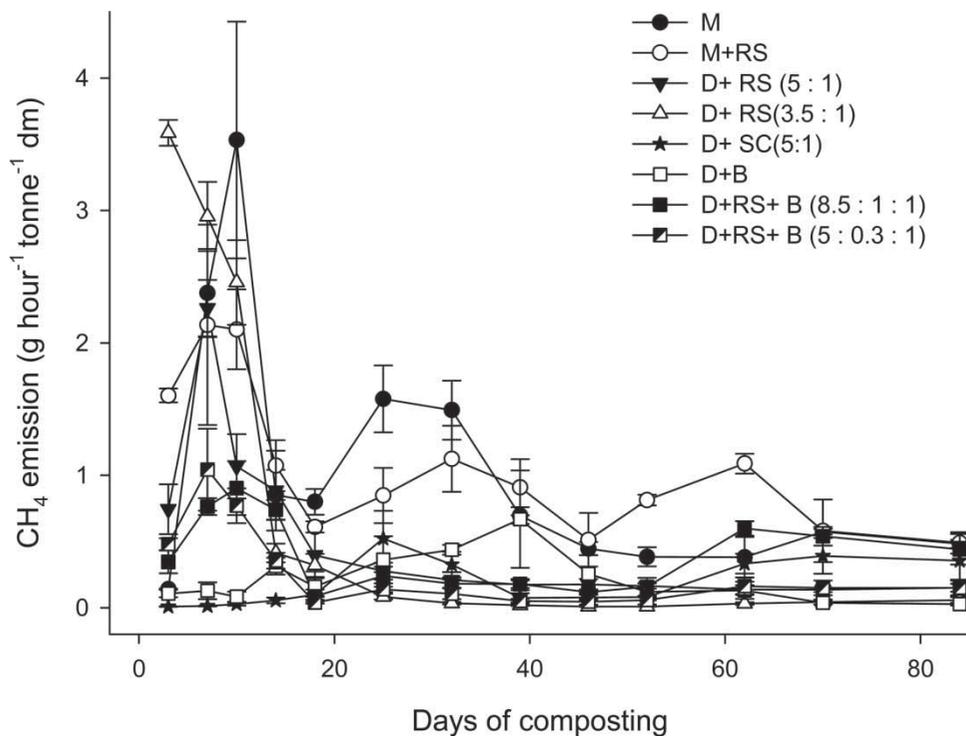
Figure 4. The variation in CH_4 emission for different composting treatments during 84 days of composting. M = solid manure, RS = rice straw, D = digestate, SC = sugar cane bagasse, B = biochar.
Note: Bars indicate one standard error (n = 3).
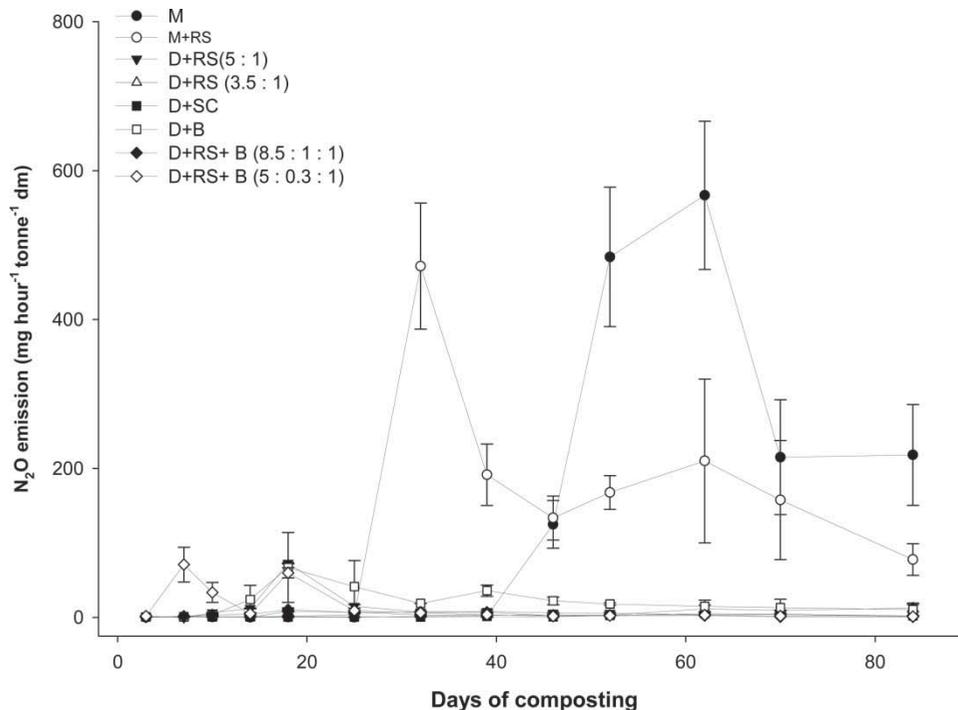
Figure 5. The variation in N_2O emission for different composting treatments during 84 days of composting. M = solid manure, RS = rice straw, D = digestate, SC = sugar cane bagasse, B = biochar.
Note: Bars indicate one standard error (n = 3).

3.4. *Global warming potential*

The CO_2-equivalent emissions of CH_4 and N_2O from the M and M + RS treatments were similar and significantly higher than those in the other treatments composting crop residues or biochar with digestate. By far the lowest cumulative CO_2-equivalent emissions were found in D + SC
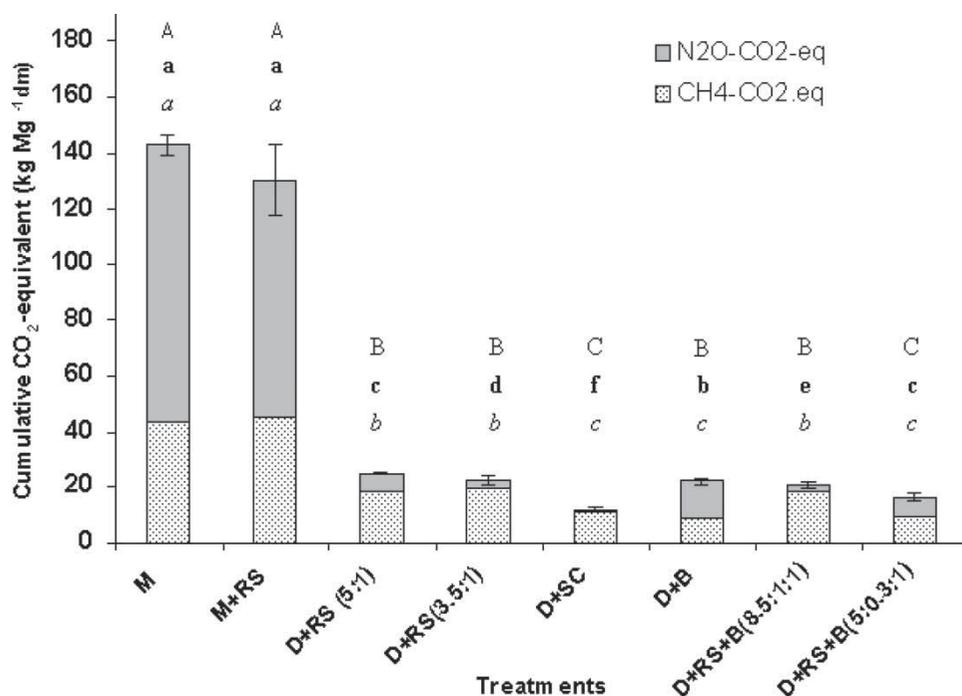
Figure 6.   The total cumulative CO_2 equivalent for different composting treatments from the 84-day composting trial. M = solid manure, RS = rice straw, D = digestate, SC = sugar cane bagasse, B = biochar.
Notes: The italic small letter indicates significance on CH_4-CO_2 equivalent among treatments ($p < .05$), The bold small letter indicates significance on N_2O-CO_2 equivalent among treatments ($p < .05$), capital letter indicates significance on total CO_2 equivalent among treatments in spring rice ($p < .05$).

and D + RS + $B_{5:0.3:1}$, with 12 and 16 kg CO_2 tonne^{-1} initial dm, respectively (Figure 6).

The treatment mixing a high ratio of biochar with digestate (D + B) showed the lowest CH_4-CO_2 equivalent emissions by far compared with the rice straw treatment (D + $RS_{5:1}$, and D + $RS_{3.5:1}$), whereas N_2O-CO_2 equivalent emissions were significantly higher in the D + B treatment compared with the rice straw treatments.

3.5.   Carbon (C) and nitrogen (N) mass balance

Based on the chemical analysis of the compost materials, C and N mass balances were estimated from the amounts of

C and N present in the composts before composting, minus the measured losses of C (as CO_2 and CH_4) and N (as N_2O) via gaseous emissions, and the measured C and N amounts present in the composts at the end of the composting trial (Table 3).

Cumulative C losses from CH_4 emissions accounted for 0.06–0.28% of initial total C at D + SC and M or M + RS treatments, respectively while C losses from CO_2 emission accounted for 1.9–26.7% of initial total C at D + B and D + $RS_{3.5:1}$ treatments, so CH_4 accounted for a very small proportion of C losses (0.4–4.0% of C loss). Between 59% and 85% of initial total C was retained in the final compost, whereas the C unaccounted

Table 3.   Carbon (C) and nitrogen (N) mass balance for different compost treatments from 84 days of passive aeration composting.

Treatments	C-CO$_2$ losses	C-CH$_4$ losses	Total gaseous C emission	C retained	C un-accounted	N-N$_2$O lost	N retained	N unaccounted for (lost as NH3, N2, and NOx)
	(% of initial total C)					(% of initial Total N)		
M	9.8 (1.5)	0.28 (0.01)	10.1 (1.5)	75.1 (2.3)	14.9 (2.7)	0.55 (0.03)	60.1 (0.5)	39.4 (0.6)
M + RS	10.5 (3.2)	0.28 (0.04)	10.8 (3.2)	68.6 (2.8)	20.6 (3.0)	0.57 (0.13)	69.1 (6.9)	30.3 (6.9)
D + RS (5:1)	13.4 (4.7)	0.12 (0.01)	13.5 (4.7)	68.4 (5.0)	18.1 (9.4)	0.06 (0.004)	72.9 (5.2)	27.0 (5.2)
D + RS (3.5:1)	26.7 (4.6)	0.13 (0.02)	26.8 (4.6)	59.2 (2.6)	14.0 (7.2)	0.03 (0.005)	69.7 (8.1)	30.2 (8.1)
D + SC	14.7 (5.7)	0.06 (0.01)	14.8 (5.7)	65.0 (1.4)	20.2 (7.0)	0.01 (0.003)	68.2 (6.7)	31.8 (6.7)
D + B	1.9 (0.8)	0.07 (0.02)	2.0 (0.8)	85.1 (8.5)	12.9 (9.0)	0.12 (0.02)	60.7 (7.2)	39.2 (7.2)
D + RS + B (8.5:1:1)	13.1 (2.5)	0.12 (0.01)	13.3 (2.5)	67.7 (4.6)	19.1 (6.3)	0.02 (0.002)	68.8 (11.5)	31.2 (11.5)
D + RS + B (5:0.3:1)	7.3 (0.8)	0.07 (0.01)	7.3 (0.8)	74.7 (8.5)	17.9 (7.9)	0.07 (0.003)	70.6 (17.9)	29.4 (17.9)

Note: Treatments: M = solid manure, RS = rice straw, D = digestate, SC = sugar cane bagasse, B = biochar.

for varied from 14% to 21% of the initial total C (Table 3).

Cumulative N losses through N_2O emissions accounted for 0.01–0.57% of initial total N at D + SC and M + RS treatments, respectively. The N retained in the final compost varied from 60% to 73% of initial total N, meaning that the N unaccounted for (lost as NH_3, N_2, or NO_x) varied from 27% to 40% of initial total N. Both the highest CH_4-C and N_2O-N losses were found in the M and M + RS treatments and were markedly higher than those in the other treatments using mixed crop residue or biochar with digestate.

4. Discussion

4.1. Changes in physico-chemical parameters during the composting process

In the present study, dry mass and C losses were the lowest in the D + B treatment (15% of initial C), probably as a result of the high recalcitrance of the biochar bulking agent, and the highest in D + $RS_{3.5:1}$ and D + SC (41% and 35% of initial C, respectively), due to the higher decomposability of RS and SC bulking agents. These results were consistent with the highest CO_2 losses in D + $RS_{3.5:1}$ and D + SC, and the lowest CO_2 losses in the D + B treatment. Tran et al. [20] reported dry mass losses between 31 and 62% of initial dry mass over 80 days of composting of solid pig manure with straw. Under laboratory composting conditions, dry mass reduced from 18% to 22% after 28 days of composting of sow manure.[21] There was little change in moisture content over time across treatments. This result was attributed to the fact that metabolic water produced by microbial activity during aerobic decomposition probably exceeded the moisture losses via evaporation.[9] The high moisture content (75–84%) could be one of reasons for the low or moderate temperature developments during the composting process in the present study.

The pH value was highest in treatments containing biochar (pH from 9.8 to 10.7). This could be one of the reasons for low CH_4 emissions from the biochar treatment. The high ammonium content of the manure can inhibit the growth of methanogenic bacteria,[22] particularly at pH values above 9.0 [23] where a significant proportion of free ammonia is present, thereby reducing CH_4 losses, but promoting NH_3 losses during composting.

Bulk density varied from 420 to 789 kg m^{-3}. Bulk density was significantly lower in D + RS treatments compared with manure treatments ($p < .05$). This may be the reason why CH_4 emissions were lower in rice straw treatments than manure treatments, where the latter, even if dominantly aerobic, may contain partly anaerobic pockets where methanogenesis can take place. This result is in agreement with Tamura and Osada [24] who reported that CH_4 losses were 53.7 g CH_4-C kg^{-1} OM with high bulk

density (810 kg m^{-3}), whereas just 3.1 g CH_4-C kg^{-1} OM with low bulk density (300 kg m^{-3}).

However, CH_4 emissions from the biochar treatment were significantly ($p < .05$) lower than that in the rice straw treatment, although bulk density was significantly ($p < .05$) higher in the biochar treatment than in the rice straw treatment, yielding a much lower aeration status and potential for higher degree of anaerobic conditions. This could have several reasons; (i) the very low degradability of biochar compared to rice straw, as it is a well-established fact that carbon characteristics play a role with respect to CH_4 production or (ii) the effect of high pH and consequently high free NH_3 concentration inhibiting methanogenic bacteria, as discussed earlier.

4.2. Temperature dynamics during the composting process

In the present study, although each of the reactors was initially filled with 43 kg of fresh materials and the reactors were well insulated on all sides by a 10 cm layer of polystyrene, the peak temperature in all treatments only reached from 34°C to 41°C during the composting process, at most 10°C above ambient at most, and certainly not within the thermophilic range normally prescribed for effective sanitizing effect of composting against pathogens and weed seeds. The peak temperature was similar to the 32–38°C observed by Tran et al. [20] and 20–30°C observed by Wang et al.,[25] and was lower than the 60–70°C reported by others.[21,26,27] Low temperatures in composting are typically related to low microbial activity within the composting mass.[28] The low temperature observed in the composting treatments in the present study could primarily be due to the oxygen supply from the reactor's natural ventilation system being relatively low, and high moisture content of the piles. However, this degree of aeration is very typical of real manure storage composting piles on small-scale Vietnamese farms, as observed by Tran et al.[20]

The present study showed that during the first 21 days, the temperature from digestate treatments mixed with rice straw or sugar cane bagasse (D + $RS_{5:1}$, D + $RS_{3.5:1}$ and D + SC) was 7–10°C higher than that from treatments containing manure or biochar (M, M + RS, D + B, D + RS + $B_{5:1:1}$ and D + RS + $B_{3.5:0.3:1}$). It is likely that this was due to the higher content of readily available carbon as well as lower bulk density in treatments of mixed rice straw or sugar cane bagasse with digestate compared with treatments containing manure or biochar. Niwagaba et al. [29] demonstrated that a compost reactor with a larger quantity of volatile mass generates more heat and is better at retaining it.

4.3. GHG emissions during the composting process

In the present study, CO_2 emissions peaked within the first 3–7 days, while CH_4 emissions peaked later after

7–10 days of composting. During the initial heating phase of the composting, high oxygen consumption and CO_2 production rates coincide with moisture release and self-compaction of the compost mixture, which in combination may decrease oxygen availability. When occurring in a heterogeneous composting matrix, it creates a mosaic of aerobic and anaerobic microsites, with conducive conditions for CH_4 production and emissions.[14] However, previous studies reported that CO_2 and CH_4 emissions occur simultaneously, shortly (2–4 days) after the start of composting.[14,30–32] Other composting studies also found CH_4 emission peaks within 2–5 days of composting.[33,34] This difference to the results from the present study could be due to low ventilation and hence low oxygen supply rate in the composting reactor system used in this study compared with the other studies. This is in agreement with Wang et al.,[25] who reported that CH_4 emissions were observed to peak at 10 days after the start of composting, probably caused by low temperatures (20–30°C) during the 49-day cattle manure composting trial.

Overall, in the present study, the high N_2O emissions from pig fresh manure treatments (M and M + RS) occurred after day 25 of the composting process. This is probably due to the fact that fresh manure usually contains little or no NO_3^- for denitrification reactions to take place at the start of composting.[14] Mahimairaja et al. [35] reported that N_2O losses occur when the nitrate levels are high in poultry and animal manures. Møller et al. [36] and Sommer and Møller [15] also reported that rising N_2O emissions were observed after the cooling of composting deep litter. Production of N_2O was negligible during the thermophilic phase of composting, since nitrifying and denitrifying micro-organisms are generally not thermophilic.[30,37,38] Fukumoto et al. [33] reported that N_2O emissions occurred at day 28 of the composting process after the temperature in the compost pile and NH_3 emissions decreased.

The main part of N_2O emissions from the D + B and D + RS + $B_{5:0.3:1}$ treatments initially occurred after 25 days of the composting process. Both the available NH_4 and NO_3 content in digestate and the high bulk density of these treatments could be the reason for that result. Several studies reported that most N_2O emissions occur during the thermophilic phase.[22,32,39,40]

4.4. *The effect of mixing ratios on emissions during composting*

The CO_2 emissions were also high in treatments mixing a high ratio of rice straw or sugar cane bagasse with digestate (D + $RS_{5:1}$, and D + $RS_{3.5:1}$ and D + SC), because they are assumed to increase oxygen availability in the compost heap, resulting in increased turnover and CO_2 emissions. The higher oxygen availability in these compost treatments can be inferred from their low bulk density value (Table 2).

Methane emissions were significantly higher in manure treatments (M & M + RS) than in rice straw treatments (D + $RS_{5:1}$, and D + $RS_{3.5:1}$) ($p < .05$). The lower bulk density in rice straw treatments (Table 2) leading to improved aeration conditions probably resulted in reduced CH_4 emissions in the rice straw treatments compared to the manure treatments. Sommer and Møller [15] also observed methane emissions during the first 25 days of composting swine manure and straw when the density of the material was high (440 kg m^{-3}), compared with no emissions when the density was low (230 kg m^{-3}).

The lowest CO_2 and CH_4 emissions were found in the biochar treatments with a high mixing ratio of biochar (D + B and D + RS + $B_{5:0.3:1}$). The low degradability of biochar (recalcitrant material) could be the main reason for these results. In addition, the biochar treatments received a lower total C input (kg tonne^{-1} dm) in comparison with the other treatments (Table 2), and this could also be a reason for the result. Chowdhury et al. [14] studied the effect of mixing solid digestate with different bulking agents such as biochar, wood chip, and barley straw on GHG emissions during a 28-day composting trial and reported that the biochar composts had the lowest cumulative CH_4-C losses compared to wood chip and barley straw composts.

A higher amount of rice straw significantly reduced N_2O emissions in D + $RS_{3.5:1}$ compared with the D + $RS_{5:1}$ treatment ($p < .05$). Both increasing the amount of rice straw and reducing the amount of biochar resulted in a significant reduction of N_2O emissions: in order D + RS + $B_{8.5:1:1}$ > D + RS + $B_{5:0.3:1}$ > D + B treatments ($p < .05$). Previous studies showed that the addition of straw during manure composting and storage reduced N_2O emissions due to improved aerobic conditions.[30,39,41] Some studies have reported a decrease in N_2O emissions with an increasing aeration rate.[31,34] However, in other studies the increased aeration rate is reported to increase N_2O emissions [42,43] because it causes high NO_3^- production via nitrification which may then lead to N_2O emissions through denitrification.

4.5. *Carbon and nitrogen mass balance*

In the present study, unaccounted C (not retained in the compost or emitted as CH_4-C and CO_2-C) ranged from 14% to 21%. This result was lower than the previously reported value of 25–36% of initial Tot-C [27]; whereas Chowdhury et al. [14] reported only 10% of C being unaccounted for in a 28-day composting trial. It is likely that the unaccounted for C in the present study was caused by the moderate frequency of air sampling used (13 sampling times during the 84-day composting trial).

The unaccounted N across the compost treatments varied from 27% to 39% of initial Total N. This relatively high value can be explained by NH_3 and N_2 losses which were not measured. Chowdhury et al. [14] demonstrated that

when NH_3 losses were also measured in a 28-day composting trial, 95% of initial total N was accounted for, leading to just 5% N unaccounted for (potentially lost as N_2). It is therefore assumed that most of the 27–39% of N not accounted for was lost as NH_3.

5. Conclusions

The present study showed that gaseous C emissions from the various compost treatments peaked relatively early in the composting process – within the first 3–7 days for CO_2 and 7–10 days for CH_4 – after which emission rates gradually declined. For N_2O emissions, emission peaks were more variable and occurred over the entire composting period, depending on the individual treatments.

Cumulative C losses from CH_4 emissions accounted for 0.06–0.28% of initial total C at D + SC and M or M + RS treatments, respectively, while C losses from CO_2 emission accounted 1.9–26.7% of initial total C at D + B and D + $RS_{3.5:1}$ treatments, respectively. Total C losses therefore predominantly occurred as CO_2, with CH_4 only accounting for 0.4–4% of total C losses. Cumulative losses of N through N_2O emissions accounted for just 0.01–0.57% of initial total N; unaccounted N losses ranged from 27% to 39% of initial compost N and were assumed to be mainly in the form of NH_3.

The composting of digestate with biochar or crop residue showed significantly lower CH_4 and N_2O emissions compared with the composting of manure ($p < .05$). However, composting of digestate with biochar (D + B) showed significantly lower CO_2 and CH_4 emissions ($p < .05$), but significantly higher N_2O emissions ($p < .05$) compared with the composting of digestate rice with straw (D + $RS_{3.5:1}$; D + $RS_{5:1}$), so hypothesis (i) was only partly rejected. N_2O emissions were reduced significantly ($p < .05$), however, by the combined composting of digestate with both biochar and rice straw (D + $RS_{5:0.3:1}$) compared with biochar only (D + B). Overall, the global warming impact (in CO_2 equivalents) of all the digestate composting strategies are in the same low range, with the combined biochar and rice straw composting having by far the lowest impact. Similarly, composting digestate with sugar cane bagasse (D + SC) significantly reduced CH_4 and N_2O emissions ($p < .05$) compared with the composting of rice straw with digestate (D + $RS_{3.5:1}$ and D + $RS_{5:1}$), thus confirming hypothesis (ii).

Overall, it can be concluded that passive aeration composting of biogas digestate with various bulking agents is feasible with relatively low GHG emissions, with a combined use of biochar and straw residues yielding the lowest GHG and ammonia emissions.

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