Research Square

Preprints are preliminary reports that have not undergone peer review.
They should not be considered conclusive, used to inform clinical practice,
or referenced by the media as validated information.

Comparative study of sludge and biowaste composting dynamics: influence of substrate type and biochar addition on greenhouse gas and ammonia emissions

Florian Paillet

paillflo@gmail.com

SUEZ Environment: SUEZ SA https://orcid.org/0000-0003-1676-0728

Leila Bourdier

SUEZ SA

Elise Crestey

SUEZ SA

Research Article

Keywords: Biochar, Composting, Nitrous Oxide, Methane, Food waste, Sludge

Posted Date: November 17th, 2026

DOI: https://doi.org/10.21203/rs.3.rs-10973440/v1

License: This work is licensed under a Creative Commons Attribution 4.0 International License.
Read Full License

Comparative study of sludge and biowaste composting dynamics: influence of substrate type and biochar addition on greenhouse gas and ammonia emissions

F. PAILLET [1*], L. BOURDIER[1], E. CRESTEY [1]

[1] *Suez, CIRSEE, 38 rue du Président Wilson, 78 230 Le Pecq, France*

**Corresponding author: florian.paillet@suez.com*

Abstract

This study investigates how substrate type and biochar addition influence composting process dynamics and gas emission pathways, with the aim of improving process control. Laboratory-scale composting experiments were conducted over 32 days using sewage sludge and biowaste mixed with a bulking agent and amended with 0%, 5%, and 10% biochar (w/w). Results showed that substrate characteristics strongly governed degradation kinetics and emission profiles. Sludge composting promoted N2O emissions due to its high nitrogen content, while biowaste composting led to CH4 emissions under limited aeration conditions. Biochar addition at 10% significantly modified process conditions by enhancing aeration and microbial activity, resulting in a reduction of N2O emissions by 23% and 55% for sludge and biowaste, respectively, and a 62% decrease in CH4 emissions during biowaste composting. However, biochar also increased NH3 volatilization, likely due to higher temperatures and accelerated degradation rates. These findings demonstrate that biochar acts as a process-modifying agent rather than a universal mitigation solution, with effects depending on substrate properties and application rate. This study highlights the need for substrate-specific

strategies to optimize composting performance and provides insights for the design and control of organic waste treatment processes.

Keywords

Biochar, Composting, Nitrous Oxide, Methane, Food waste, Sludge

1. Introduction

The global increase in organic waste production, driven by population growth and industrial activities, has emerged as an environmental challenge. Sludge and biowaste are particularly prevalent in organic waste streams. The management, treatment, and valorization of this waste are essential to reducing disease risks and promoting a circular economy. In Europe, more than 6.6 million tons of sewage sludge (Eurostat, 2017) and more than 2 000 million tons of biowaste (Eurostat, 2022) are generated annually. European regulations now also require separate collection and treatment of biowaste, which further highlights the need for efficient and sustainable treatment solutions.

Composting is a widely used method to treat organic waste, transforming it into nutrient-rich compost that improves soil structure and fertility while reducing reliance on fossil-based fertilizers. However, composting also generates emissions of ammonia (NH_3) and greenhouse gases (GHGs) such as methane (CH_4), nitrous oxide (N_2O), and carbon dioxide (CO_2), which contribute to climate change. These emissions have been correlated with operational parameters such as aeration [1], composting duration [2],

and also to the initial characteristics of the medium (C/N ratio, moisture content) [3]. Because the medium's characteristics are directly influenced by the type of waste treated, gas emissions can vary significantly depending on the composted material.

For example, biowaste and especially food waste contains easily degradable organic matter, whereas sewage sludge is often characterized by lower biodegradability, higher moisture and nitrogen content. Such differences can lead to distinct microbial pathways, degradation kinetics and GHG/NH_3 emission profiles. Therefore, a single universal composting strategy for sludge and biowaste is unlikely to effectively reduce environmental impacts; process optimization must be substrate-specific to ensure operational efficiency.

To reduce these emissions, several solutions have been tested as adding chemical, physical, or microbial additives [4]. Among them, the addition of biochar, a carbon-rich material produced by the pyrolysis of biomass, has gained attention for its potential to mitigate GHG emissions during composting [5]. The porous structure and high adsorption capacity of biochar can influence microbial activity, nutrient retention, and gas emissions, making it a promising amendment for improving composting dynamics and reducing environmental impacts [6].

Although most studies show a tendency to reduce GHG and ammonia emissions during composting with biochar addition, the effectiveness of this reduction varies widely. These differences can be explained by several factors: the biochar quality (feedstock, pyrolysis technology and operating conditions, particle size), the biochar application rate, and the type of composted material. The meta-analysis performed by Yin et al., (2021) illustrates this variability comparing different biochars and composting materials

but with the same biochar application rate (10% by weight) : CH_4 emissions from manure composting decreased between 26.1 to 72.9% and N_2O between 15.1 to 81.6%, whereas on sewage sludge and green waste these reductions reached respectively about 93% and 95% for CH_4, and 96% and 14% for N_2O [7]. However, comparative studies directly evaluating the effects of biochar on sewage sludge and biowaste composting under identical experimental conditions remain limited. Further investigation is needed to fully understand the potential benefits of biochar application according to the organic waste treated.

This study investigates the impact of different composted substrate types (sewage sludge and biowaste) on the composting process dynamics. Then, it evaluates the effect of biochar addition on associated gas emissions (NH_3 and GHGs). By exploring these interactions, this research aims to provide insights for optimizing composting practices for enhanced waste management and climate change mitigation.

2. Material and methods

2.1. Composting feedstock

The dewatered sewage sludge, biowaste, and bulking agent (green waste and screening refusal) were collected from the same industrial composting site in France (western and northern regions). According to the European definition of biowaste, the material used in this study corresponds to the organic fraction of kitchen waste collected separately. Upon receipt, the feedstocks were stored at 4°C to prevent degradation. Due to the heterogeneity of the green waste (containing branches, leaves, grass mowing, etc.) It

was sieved at 1 cm on square meshes. The physicochemical parameters of the feedstocks are presented in Table 1.

Table 1: Main characteristics of the composted materials

Starting materials	Dry Matter (%ww)	pH	OM (%ww)	OC (%ww)	C/N	TNi (%)	TNi (gN/kgww)	N-NH_4^+ (%ww)
Sludge	20.1±1.0	7.1±0.6	15.2±0.9	7.4±0.5	5.1±0.4	1.48±0.03	0.14±0.07	14.8±0.3
Biowaste	25.9±1.8	4.7±0.1	21.6±3.8	10.6±2.3	12.0±2.4	0.90±0.02	0.07±0.01	9.0±0.2
Bulking agent	78.8±3.0	8.1±0.1	78.3±10.1	35.1±0.8	31.6±14.0	1.16±0.42	0.12±0.05	14.6
Green waste	59.2±7.0	7.4	27.6±1.3	13.5±0.1	39.5±21.4	0.41±0.21	0.03±0.01	4.04±2.02

OM: Organic Matter, OC: Organic Carbon, TNi: Total Nitrogen

The biochar has been produced from softwood residues through pyrolysis at 500–600°C in a pilot facility. The physicochemical properties of the selected biochar are presented in Table 2.

Table 2: Main characteristics of the biochar used

Dry Matter (%ww)	pH	OC (%ww)	TC (%ww)	H/C (mol/mol)	TN (%ww)	Ash content(%ww)	Bulk density	WHC (%ww)	SSA BET (m^2/g)	Pore size (nm)
50.7	7.9	39.3	39.9	0.38	0.18	5.4	207	264	90	Micro (<2nm) and meso-pores (2-50nm)

WHC: Water Holding Capacity, SSA BET: Specific Surface Area, BET N2 method

2.2. Experimental device

Experiment was performed at laboratory scale (4 L) in a closed reactor system with thermal insulation. A carrier gas (air) is injected at the top of each reactor with a flow rate of 1.5 L/min, providing oxygen and homogenizing the gas produced during the process to enable reliable measurement. The system allowed a continuous measurement of N_2O and CH_4 through a set of valves using Servomex 4200 for CH_4 and N_2O. NH_3 emission was also monitored using sulphuric acid bubbling traps (600 mL and 300 mL) containing 1.5% H_2SO_4 to capture NH_3 emissions.

NH_4^+ concentration was determined by colorimetry analysis (Gallery Sequential Analyzer™). The airflow in the ammonia trap circuit was quantified. The ratio between total airflow injected in the vessels and airflow bubbling through ammonia traps was used to estimate the total amount of emitted ammonia. Gaseous emissions kinetics (r_x) were calculated according to the equation (Eq. 1):

$$r_x = \frac{f_{DA}(C_{out} - C_{in})}{VM_{sludge}} \quad \text{Eq. 1}$$

with f_{DA} the dry airflow ($L.h^{-1}$), C_{out} the concentration of x in the outlet gas of the reactor ($gC.L^{-1}$ or $gN.L^{-1}$), C_{in} the concentration of x in the inlet gas of the reactor ($gC.L^{-1}$ or $gN.L^{-1}$) and VM_{sludge} the initial volatile matter of the sewage sludge (kg), r_x was therefore expressed in gC or $gN.h^{-1}.kg^{-1}$ VM_{sludge}. Manual mixing of the composted mixtures was performed weekly to ensure proper aeration. Finally, temperature was continuously measured and the experiment lasted 32 days.

2.3. Experimental plan

For the comparison between sludge and biowaste composting, a similar mixture of sludge or biowaste with bulking agent were prepared (1:1 w:w) in triplicate. This ratio is the one typically used in industrial composting units to ensure an optimal porosity of the composting mixture. For the bulking agent used, a mixture of screening refusal and green waste was used for sludge composting (1:1 w:w) while only green waste was used for biowaste composting. To assess the impact of biochar addition on sludge and biowaste composting, three conditions were tested: without biochar (Sludge and Biowaste), with addition of 5% biochar (Sludge-B5 and Biowaste B-5), and 10% biochar (Sludge-B10 and Biowaste-B10) by mass of the waste treated. A turning phase was performed every week for all the experiments to ensure adequate oxygen supply during composting.

2.4. Data analysis

Statistical analyses were performed using R software (version 4.3.2). Since experiments were conducted in triplicate, the reported emission values represent the mean, with the standard deviation calculated accordingly. Data processing and analysis were carried out using base R functions and, where applicable, additional packages such as dplyr for data manipulation and ggplot2 for visualization.

3. Results and discussion

3.1. Comparison between sludge & biowaste composting

Temperature profiles during sludge and biowaste composting are presented in Figure 1.A. These temperature profiles correspond to common patterns observed during an aerobic degradation process in composting. That is, an initial thermophilic temperature rise (active degradation phase). During this phase, similar maximum temperature was reached for sludge and biowaste composting (55.6±2.3 °C and 57.4±9.5 °C respectively). Interestingly, the thermophilic phase lasts longer for biowaste with a period above 47°C of 7.5±3.7 days compared to sludge (1.2±0.5 days). This observation is advantageous due to the necessity to rigorously sanitize biowaste if it contains waste derived from animal by-products in order to eliminate any risk of disease. Afterwards, the temperature in sludge composting falls below 35°C before the biowaste condition (11.2±2.2 days compared to 21.1±1.3 days respectively). This observation can be explained by the difference in the degradable material between the two types of waste. In our study, the %Organic matter and %Organic carbon were 15.2±0.9 % and 7.4±0.5 % respectively for sludge and 21.6±3.8 % and 10.6±2.3 % respectively for biowaste (Table I). Biowaste is known to be a highly reactive material with a high concentration of accessible and degradable organic matter compared to sludge. These substrates were also compared in the study of Fisgativa et al.(2020) showing a significant difference in terms of COD (chemical oxygen demand) for a primary and secondary sewage sludge (79.1 and 70.4 gO_2/kgww respectively) and biowaste (287.7 kgO_2/kgww) [9]. These differences resulted in higher production of methane during anaerobic digestion (7.8 and 24.9 $NLCH_4$/kgww for primary and secondary sewage sludge respectively and 81.6 $NLCH_4$/kgww for biowaste). The temperature profiles during composting are closely linked to microbial degradation, which is exothermic during the thermophilic phase, releasing heat as organic matter is decomposed [10]. Given the differences in the

temperature profiles of the two waste types, the composting of biowaste requires a longer duration to reach a similarly low temperature commonly associated to a better stability of the organic material.

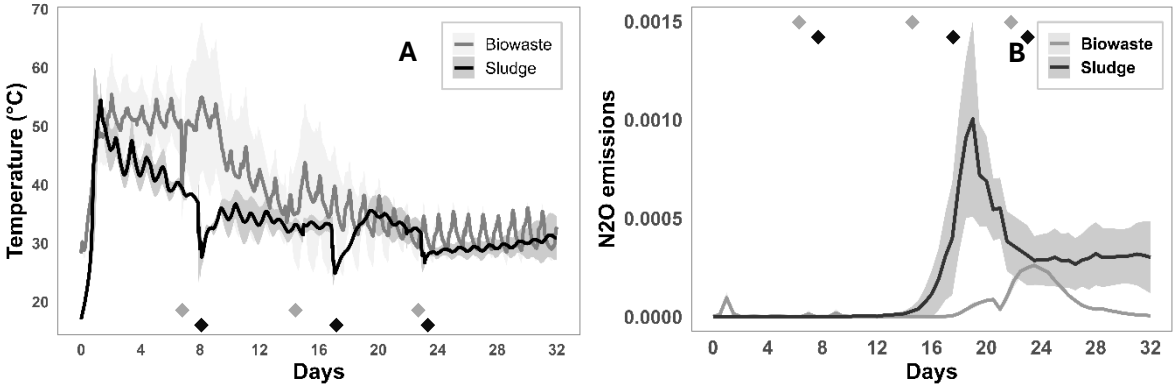


Figure 1 Temperature (1.A) and N_2O emissions (1.B) dynamics ($gN_N_2O.kgDM_{treatedwaste}$), ◆: turning phase for sludge composting, ◆: turning phase for biowaste composting

During the experiment, no CH_4 emissions were detected for sludge composting, likely due to the weekly turning process implemented to optimize oxygenation of the composting mixture and the low organic carbon content. In contrast, the oxygen supply was insufficient for biowaste composting, which exhibited cumulative CH_4 emissions of 1.25±0.16 $gC_CH_4/kgDM_{biowaste}$ (Table 3). Methane production during composting is already documented in the literature and is typically associated with the development of anaerobic conditions when oxygen supply is not sufficient [11]. In our study, CH_4 emission was mainly observed just before the first turning phase, showing a quick consumption of oxygen which required an increase in turning frequency. Therefore, with the objective of reducing GHG emissions, greater attention must be given to

oxygenation during the composting of biowaste to limit anaerobic zones compared to sludge composting.

The emissions of N_2O for both conditions are presented in Figure 1.B. Negligible N_2O emissions were measured at the start of the experiment for sludge and biowaste composting. Interestingly, for the latter phase of N_2O, the emissions started earlier for sludge composting (15.3±1.2 days) compared to biowaste composting (18.9±0.4 days). The beginning of N_2O emissions was observed in both cases following the thermophilic phase, occurring when the temperature decreased below ~35°C. The correlation between temperature and N_2O emissions has been discussed in the study of Cui et al., (2019) showing that hyperthermophilic conditions significantly decrease the emissions of N_2O due to the inhibition of nitrifier microorganisms [13]. It was also shown that the 40°C threshold could be a pivotal temperature influencing the ammonia-oxidizing bacteria responsible of N_2O emissions [14], [15] which may explain the observed difference in the initial emission duration.

The maximum N_2O emissions were found at day 18.7±0.3 for a value of $2.2±0.1x10^{-3}$ $gN_N_2O.kgDM_{treatedwaste}$ for sludge composting, higher than the peak found for biowaste composting (day 23.8±0.6, $2.7±0.2x10^{-4}$ $gN_N_2O.kgDM_{treatedwaste}$). Cumulative N_2O emissions showed that sludge composting emitted significantly more than biowaste composting (14.0±1.6 $gN_N_2O/kgDM_{treatedwaste}$ compared to 1.15±1.03 $gN_N_2O/kgDM_{treatedwaste}$ respectively). The difference can be attributed to the higher nitrogen content present in sewage sludge (TKN: 1.48±0.03 $\%_{ww}$, C/N: 5.1±0.4, Total nitrogen: 0.14±0.07 gN/kg_{ww}) compared to biowaste (TKN: 0.90±0.02 $\%_{ww}$, C/N: 12.0±2.4 and Total nitrogen: 0.07±0.01 gN/kg_{ww}). Previous study showed that the C/N ratio

impacts the emissions of N_2O, with low C/N favoring the emissions of N_2O due to excess of nitrogen availability and limited carbon for microbial assimilation [16]. However, this observation is not clear since other studies didn't find the correlation [2], [17]. A major limitation of the C/N ratio is that it does not account for the proportions of biodegradable and readily available carbon and nitrogen, factors that would significantly improve its practical relevance and predictive accuracy. For example, the study made by Bareha et al., (2018) demonstrates that the fractions of biodegradable and accessible nitrogen are strongly substrate dependent ranging from 15% to 82% of organic nitrogen, and from 36% to 74% for organic carbon, across different categories of waste (agricultural residues, livestock waste and wastes from agroindustry and municipalities) [19]. These observations highlight the strong influence of substrate composition on nitrogen dynamics during composting.

Table 3: Cumulated emissions of N_2O, CH_4 and NH_3 during sludge and biowaste composting ($g/kgDM_{treatedwaste}$ and $g/kgVS_{treatedwaste}$)

	$gN_N_2O/kgDM$	**$gN_N_2O/kgVS$**	**gC_CH_4/ kgDM**	**gC_CH_4/kgVS**	**gN_NH_3/ kgDM**
Sludge	14.0±1.6	18.8±2.1	-	-	7.5±0.3
Biowaste	2.17±0.18	2.52±0.21	1.25±0.16	1.46±0.19	1.1±0.7

Finaly, the cumulative emissions of NH_3 also showed a significant difference with higher emissions for sludge composting compared to biowaste (7.5±0.3 compared to 1.1±0.7 $gN_NH_3/kgDM_{treated\ waste}$ respectively). NH_3 emissions could be directly linked with the initial ammonia content of each substrate due to the phenomenon of volatilization accentuated by the temperature rise during the thermophilic phase [20]. In our study, ammonia nitrogen content of sludge and biowaste was 0.22 and 0.06 $\%_{ww}$ corroborating

this hypothesis. In terms of total loss relative to the initial nitrogen content of biowaste and sludge (Figure 2), the results show that a higher proportion of nitrogen was lost during sludge composting (17.3±1.2 %N-loss$_{tot}$, 11.2±1.3 %N-lossN_2O and 5.7±0.3 %N-lossNH_3) compared to biowaste composting (2.7±1.2 %N-loss$_{tot}$, 0.5±0.1 %N-lossN_2O and 2.1±1.1 %N-lossNH_3).

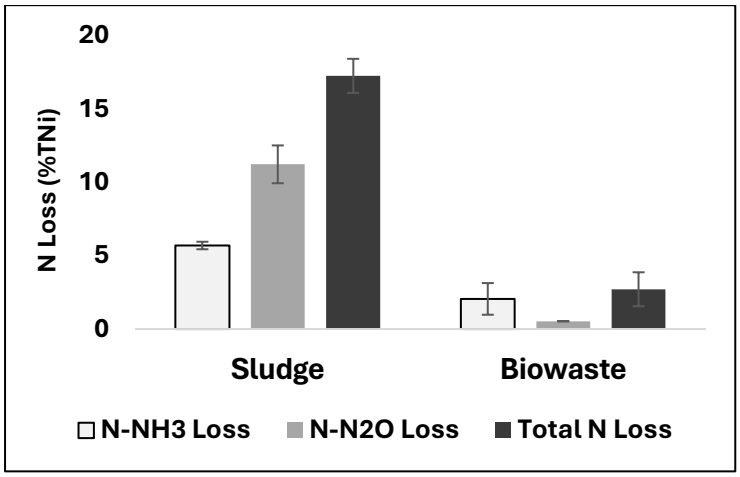


Figure 2 Nitrogen loss in the form of NH_3 and N_2O (%TNi)

3.2. Impact of biochar addition during sludge and biowaste composting

The previous section highlighted notable differences in nitrogen and methane emissions between sludge and biowaste composting, suggesting distinct dynamics for these substrates. Given that biochar is known to influence nitrogen transformations during composting through various mechanisms such as adsorption, alteration of microbial activity, and changes in oxygen diffusion it is reasonable to expect that biochar's impacts may differ depending on the waste type. Therefore, investigating how biochar addition affects sludge and biowaste composting individually will provide

252 insights into substrate-dependent interactions and their influence on greenhouse gas
253 emissions and nitrogen losses.

254 The evolution of the temperature during the composting of sludge and biowaste with the
255 addition of biochar (0, 5 and 10% ww) is presented in Fig. 3 A & B. While biochar
256 addition did not alter the overall temperature profiles (thermophilic and mesophilic
257 phases) of sludge and biowaste composting, it significantly increased the maximum
258 temperature and temperature rise rate at the 10% addition level. The condition Sludge-
259 B10 reached a maximum temperature of 67±3°C, which was 18% higher than the Sludge
260 and Sludge-B5 conditions (55±2 and 55±4°C respectively). Interestingly, the maximum
261 temperature reached for biowaste was similar for the biowaste-B10 (77±1°C) compared
262 to 77±2°C and 75±1°C for Biowaste-B5 and Biowaste conditions respectively,
263 suggesting that addition of biochar to highly degradable organic matter does not
264 promote the increase in temperature. However, the temperature rise kinetics increased
265 with the addition of biochar during biowaste composting to 27.7°C/day and 22.9°C/day
266 for Biowaste-B10 and Biowaste-B5 respectively compared to 14.8 °C/day without
267 biochar. These observations clearly highlight the impact of biochar on the composting
268 temperature. These results are consistent with the review conducted by Godlewska et
269 al., (2017), who compiled data from 16 laboratory trials investigating the effects of
270 biochar addition on composting temperature, showing that biochar addition led to
271 higher maximum temperatures [22]. For example, a 6% biochar addition led to an 8.1%
272 increase in peak temperature [23]. At 10 %, increases of 4.8%, 4.2%, and 17.3 % were
273 reported [23], [24]. Zhang et al., (2016) also obtained a 22.1% increase with 15 %
274 biochar addition. Two main mechanisms have been proposed to explain these effects:
275 the biochar particles may fill the interstitial spaces between compost pieces, reducing

heat loss, while the porous structure of biochar supports microbial proliferation and activity, thereby increasing heat generation [21]. Despite the diversity of biochars used in the literature (feedstock, production parameters), consistent trends were observed regarding peak temperature kinetics, even though a clear dose-response relationship remains difficult to establish. This suggests that a wide range of biochars can enhance composting temperature dynamics. However, biochars vary in performance, suggesting that optimizing biochar properties could enhance composting outcomes.

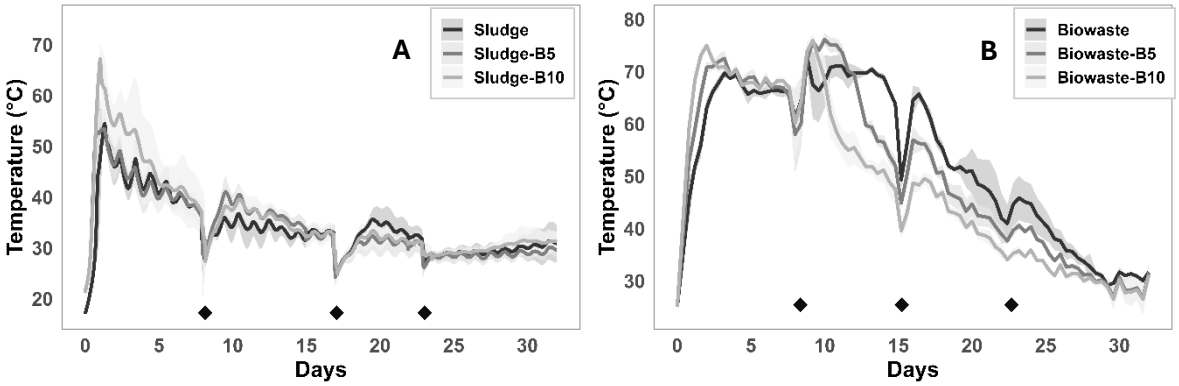


Figure 3 Temperature dynamics during composting of sludge and biowaste with addition of biochar (0, 5% and 10%), ◆: turning phase

As already mentioned in the previous section, CH_4 emissions were negligible for sludge composting while CH_4 was measured on the biowaste composting experiment. The cumulative CH_4 emissions showed significant differences between the control and the conditions with biochar addition with 0.87±0.28, 0.36±0.19 and 0.33±0.12 $gC_CH_4/kgDM_{treatedwaste}$ for the control, Biowaste-B5 and Biowaste-B10 respectively (Table 3). Interestingly, the CH_4 emissions profile showed a significant difference during the thermophilic phase between the control and the other conditions with biochar addition just before the first turning phase (Supplementary Material). Thus, the addition of biochar even at 5% seems to significantly improve the oxygen supply which limits CH_4

296 emissions during the thermophilic phase. Therefore, in addition to the phenomenon
297 already discussed above, biochar seems to increase the aeration of the composting
298 mixture by increasing the oxygen circulation which could improve the degradation of
299 organic matter and be an additional explanation for difference observed in the
300 temperature dynamics.

301

302

303 The emissions of N_2O during sludge and biowaste composting with the addition of
304 biochar are presented in Table 3. As observed, the 5 % biochar addition didn't show
305 significant differences with the control for sludge composting (14.6±1.9 compared to
306 14.0±1.6 $gN_N_2O/kgDM_{treatedwaste}$ respectively) and biowaste composting (1.94±0.71 and
307 1.39±0.10 $gN_N_2O/kgDM_{treatedwaste}$ respectively). These results indicate that minimal
308 biochar additions (5%) are insufficient to significantly affect N_2O emissions. Meanwhile,
309 the addition of 10 % of biochar systematically showed a reduction on N_2O emissions
310 which decreased over 23 % and 55 % for sludge and biowaste composting respectively
311 compared to controls. Literature already suggests that biochar impacts the emissions
312 of N_2O during composting with an average reduction of 59.8% (n=27) [4]. The large
313 surface area is often mentioned to be one of the parameters affecting the mitigation of
314 N_2O emissions by improving its ion exchange and limiting nitrogen availability (NO_3^- or
315 NO_2^-) for N_2O production [25]. Meanwhile, it is important to highlight that in our study, a
316 significant difference in the global performance of N_2O emissions reduction by biochar
317 was observed and depending on the type of waste treated: biochar shows lower
318 mitigation performance when the nitrogen content is high, as it is the case with sludge.

However, the mitigating capacity of biochar was measured at 0.463 $gN_N_2O/g_{biochar}$ for sludge and 0.330 $gN_N_2O/g_{biochar}$ for biowaste, indicating relatively similar mitigation capacity across both substrates (for 10% biochar addition). This suggests that the effectiveness of biochar is comparable regardless of the substrate type, and that the nitrogen content of each substrate influences overall performance in terms of reducing nitrous oxide emissions. Given the importance of surface area and adsorption capacity, higher nitrogen content in the waste requires greater biochar surface area to achieve effective mitigation.

Overall, the differential impact of biochar amendment rates on N_2O emissions during composting seems to be dose and substrate dependent phenomenon. Regarding the dosage, a 5% biochar addition does not demonstrate an impact regardless of the waste treated, while higher rates showed significant effects. This phenomenon has been reported by Nguyen et al., (2022) and may be due to the enzyme's activity of the bacterial community which was strengthened (at 10 and 20% of biochar addition) [26]. Similarly, López-Cano et al., (2016) reported that low dosage had negligible influence on N_2O emissions (4% of biochar addition) [28]. Also, Yin et al., (2021) mentioned that the addition of 10% dosage has been reported to be the most effective while a low amount of biochar does not have a significant effect on GHG emissions [29]. Therefore, a threshold limit of 10 % biochar application should be recommended to use the biochar as a mitigating solution for GHG emissions. However, the difference of performance regarding the substrate is not well referenced and seems to be due to the initial nitrogen content of each waste.

342

343 The results obtained for the reduction of cumulative CH_4 and N_2O emissions are
344 consistent with those reported in the previously mentioned studies. However, the
345 ranges of these results are very wide; while the overall tendencies are similar, the
346 percentages of emission reductions are not directly comparable with the literature.
347 Indeed, the magnitude of biochar effects seems strongly dependent on the type of
348 waste treated but also on the biochar used, which inevitably differs from one study to
349 another. The quality of biochar, defined by physicochemical properties such as carbon
350 content, specific surface area, porosity, pH, cation exchange capacity (CEC), and
351 electrical conductivity, varies according to three main parameters: the type of biomass
352 used as feedstock, the production conditions (especially the pyrolysis temperature),
353 and post-production treatments, including sieving and/or grinding, which affect particle
354 size and size distribution. Because biochar quality is intrinsically complex, comparing
355 the magnitude of its effects on gas emissions across studies remains of limited
356 reliability. To better understand the influence of biochar quality, new comparative trials
357 would be required to compare under the same conditions biochars differing in specific
358 surface area, pore size, CEC, and pH.

359
360

	N_N_2O	C_CH_4	N_NH_3	N-NH_3 Loss	N-N_2O Loss	Total N Loss
Sludge	14.0±1.6	-	7.5±0.3	5.7±0.3	11.2±1.3	17.3±1.2
Sludge-B5	14.6±1.9	-	7.6±1.3	5.7±1.0	11.7±1.5	17.8±1.8
Sludge-B10	10.8±0.7	-	7.3±0.6	5.5±0.5	8.7±0.5	14.6±0.1

Biowaste	1.94±0.71	0.87±0.28	1.7±0.2	3.1±0.3	3.7±2.7	7.0±3.0
Biowaste-B5	1.39±0.10	0.36±0.19	5.2±0.7	9.3±1.2	2.6±0.2	12.5±1.5
Biowaste-B10	0.88±0.23	0.33±0.12	4.6±0.4	8.3±0.8	1.7±0.4	10.4±1.2

Table 4: Cumulative greenhouse gas emissions (g/kg$DM_{treatedwaste}$) and nitrogen losses (%TNi) during the sludge and biowaste composting.

Table 4 presents the effect of biochar addition on nitrogen losses as ammonia and nitrous oxide during the composting of sludge and biowaste. For sludge, the addition of 10% biochar (Sludge-B10) enabled to reach a total nitrogen loss of 14.6 ± 0.1%TNi, compared to 17.3±1.2 %TNi in the control without biochar. This reduction was mainly driven by a significant decrease in nitrogen loss through N_2O emissions (8.7 ± 0.5 %Ni versus 11.2 ± 1.3 %Ni), while NH_3 losses remained relatively stable around 5.5–5.7 %TNi. In the case of biowaste, biochar addition increased NH_3 losses from 3.1 ± 0.3 %TNi in the control to 8.3 ± 0.8%TNi for B-10 and 9.3 ± 1.2%TNi for B-5, while nitrogen losses from N_2O decreased from 3.7%TNi to 1.7%TNi compared to the control and B-10. This partial shift in nitrogen loss pathways suggests that biochar may enhance NH_3 emissions, likely due to elevated temperature and accelerated temperature rise favoring ammonia volatilization [30]. Several studies have shown that the impact of biochar on NH_3 mitigation is mainly due to the adsorption capacity of the biochar used [5]. This adsorption capacity is related to the porosity of biochar and its ion exchange capacity. However, in our case, the increase in temperature appears to have a greater impact on NH_3 emissions than the adsorption capacity of biochar during the initial phase of emission. Although ammonia emissions increased, biochar addition during biowaste

composting reduced total nitrogen losses by 10.4 ± 1.2% and 12.5 ± 1.5% for the 10% and 5% biochar addition, respectively.

Conclusion

This study demonstrates that compost feedstock characteristics strongly influence temperature dynamics and gas emissions (N_2O, CH_4, and NH_3) during composting. Sewage sludge and biowaste exhibited distinct emission profiles, with methane emissions absent in sludge composting due to adequate aeration. The addition of biochar at 10% effectively reduced N_2O emissions in both substrates and significantly lowered CH_4 emissions during biowaste composting. However, biochar slightly increased ammonia volatilization, particularly in biowaste composting. These findings underscore the importance of tailoring mitigation strategies to both substrate type and biochar dosage to maximize emission reductions. Further research is needed to elucidate the mechanisms behind biochar's effects and to optimize its application for sustainable waste management.

References

[1] J. Zeng *et al.*, « Effect of aeration interval on oxygen consumption and GHG emission during pig manure composting », *Bioresource Technology*, vol. 250, p. 214-220, févr. 2018, doi: 10.1016/j.biortech.2017.11.010.

[2] E. Crestey, F. Paillet, P. Camacho, et A. Tremier, « Correlations between sewage sludge composting physicochemical parameters and emissions of greenhouse gases and ammonia: A statistical analysis », *Journal of Environmental Management*, vol. 368, p. 122167, sept. 2024, doi: 10.1016/j.jenvman.2024.122167.

[3] J. Zhang *et al.*, « Operational parameters regulate in-situ emissions of volatile organic compounds (VOCs) and greenhouse gases (GHGs) during kitchen waste composting: Linking functional microbial dynamics to emission mitigation », *Waste Management*, vol. 204, p. 114935, août 2025, doi: 10.1016/j.wasman.2025.114935.

[4] Y. Cao *et al.*, « Mitigation of ammonia, nitrous oxide and methane emissions during solid waste composting with different additives: A meta-analysis », *Journal of Cleaner Production*, vol. 235, p. 626-635, oct. 2019, doi: 10.1016/j.jclepro.2019.06.288.

[5] M. K. Nguyen *et al.*, « Evaluate the role of biochar during the organic waste composting process: A critical review », *Chemosphere*, vol. 299, p. 134488, juill. 2022, doi: 10.1016/j.chemosphere.2022.134488.

[6] X. Guo, H. Liu, et J. Zhang, « The role of biochar in organic waste composting and soil improvement: A review », *Waste Management*, vol. 102, p. 884-899, févr. 2020, doi: 10.1016/j.wasman.2019.12.003.

[7] Y. Yin *et al.*, « Research progress and prospects for using biochar to mitigate greenhouse gas emissions during composting: A review », *Science of The Total Environment*, vol. 798, p. 149294, déc. 2021, doi: 10.1016/j.scitotenv.2021.149294.

[8] H. Fisgativa, B. Zennaro, C. Charnier, C. Richard, G. Accarion, et F. Béline, « Comprehensive determination of input state variables dataset required for anaerobic digestion modelling (ADM1) based on characterisation of organic substrates », *Data in Brief*, vol. 29, p. 105212, avr. 2020, doi: 10.1016/j.dib.2020.105212.

[9] H. Fisgativa, A. Tremier, et P. Dabert, « Characterizing the variability of food waste quality: A need for efficient valorisation through anaerobic digestion », vol. 50, p. 264-274, 2016, doi: https://doi.org/10.1016/j.wasman.2016.01.041.

[10] J. Su *et al.*, « Enhanced organic degradation and microbial community cooperation by inoculating Bacillus licheniformis in low temperature composting », *Journal of Environmental Sciences*, vol. 143, p. 189-200, sept. 2024, doi: 10.1016/j.jes.2023.08.037.

[11] R. Chen, W. Yiming, W. Shiping, W. Wei, et Lin Xiangui, « Windrow composting mitigated CH4 emissions: characterization of methanogenic and methanotrophic communities in manure management », *FEMS Microbiology Ecology*, vol. 90, n° 3, p. 575-586, 2014, doi: DOI: 10.1111/1574-6941.12417.
[12] P. Cui *et al.*, « Hyperthermophilic composting significantly decreases N2O emissions by regulating N2O-related functional genes », *Bioresource Technology*, vol. 272, p. 433-441, janv. 2019, doi: 10.1016/j.biortech.2018.10.044.
[13] P. Cui *et al.*, « Hyperthermophilic composting significantly decreases N2O emissions by regulating N2O-related functional genes », *Bioresource Technology*, vol. 272, p. 433-441, janv. 2019, doi: 10.1016/j.biortech.2018.10.044.
[14] C. Grunditz et G. Dalhammar, « Development of nitrification inhibition assays using pure cultures of nitrosomonas and nitrobacter », *Water Research*, vol. 35, n° 2, p. 433-440, févr. 2001, doi: 10.1016/s0043-1354(00)00312-2.
[15] X. Lin *et al.*, « Relative contribution of ammonia-oxidizing bacteria and denitrifying fungi to N2O production during rice straw composting with biochar and biogas residue amendments », *Bioresource Technology*, vol. 390, p. 129891, déc. 2023, doi: 10.1016/j.biortech.2023.129891.
[16] Z. Xu, G. Li, N. Huda, B. Zhang, M. Wang, et W. Luo, « Effects of moisture and carbon/nitrogen ratio on gaseous emissions and maturity during direct composting of cornstalks used for filtration of anaerobically digested manure centrate », *Bioresource Technology*, vol. 298, p. 122503, févr. 2020, doi: 10.1016/j.biortech.2019.122503.
[17] T. Jiang, F. Schuchardt, G. Li, R. Guo, et Y. Zhao, « Effect of C/N ratio, aeration rate and moisture content on ammonia and greenhouse gas emission during the composting », *Journal of Environmental Sciences*, vol. 23, n° 10, Art. n° 10, oct. 2011, doi: 10.1016/S1001-0742(10)60591-8.
[18] Y. Bareha, R. Girault, J. Jimenez, et A. Trémier, « Characterization and prediction of organic nitrogen biodegradability during anaerobic digestion: A bioaccessibility approach », *Bioresource Technology*, vol. 263, p. 425-436, sept. 2018, doi: 10.1016/j.biortech.2018.04.085.
[19] Y. Bareha, R. Girault, J. Jimenez, et A. Trémier, « Characterization and prediction of organic nitrogen biodegradability during anaerobic digestion: A bioaccessibility approach », *Bioresource Technology*, vol. 263, p. 425-436, sept. 2018, doi: 10.1016/j.biortech.2018.04.085.
[20] Y. Wang *et al.*, « NH3, N2O, and NO emissions from digested pig slurry stored under different temperatures: Characteristics and microbial mechanisms », *Journal of Cleaner Production*, vol. 319, p. 128560, oct. 2021, doi: 10.1016/j.jclepro.2021.128560.
[21] P. Godlewska, H. P. Schmidt, Y. S. Ok, et P. Oleszczuk, « Biochar for composting improvement and contaminants reduction. A review », *Bioresource Technology*, vol. 246, p. 193-202, déc. 2017, doi: 10.1016/j.biortech.2017.07.095.
[22] P. Godlewska, H. P. Schmidt, Y. S. Ok, et P. Oleszczuk, « Biochar for composting improvement and contaminants reduction. A review », *Bioresource Technology*, vol. 246, p. 193-202, déc. 2017, doi: 10.1016/j.biortech.2017.07.095.
[23] Y.-X. Chen *et al.*, « Effects of bamboo charcoal and bamboo vinegar on nitrogen conservation and heavy metals immobility during pig manure composting », *Chemosphere*, vol. 78, n° 9, p. 1177-1181, févr. 2010, doi: 10.1016/j.chemosphere.2009.12.029.
[24] J. Zhang, G. Chen, H. Sun, S. Zhou, et G. Zou, « Straw biochar hastens organic matter degradation and produces nutrient-rich compost », *Bioresource Technology*, vol. 200, p. 876-883, janv. 2016, doi: 10.1016/j.biortech.2015.11.016.
[25] Y. Yuan, H. Chen, W. Yuan, D. Williams, J. T. Walker, et W. Shi, « Is biochar-manure co-compost a better solution for soil health improvement and N2O emissions mitigation? »,

Soil Biology and Biochemistry, vol. 113, p. 14-25, oct. 2017, doi: 10.1016/j.soilbio.2017.05.025.
[26] M. K. Nguyen *et al.*, « Evaluate the role of biochar during the organic waste composting process: A critical review », *Chemosphere*, vol. 299, p. 134488, juill. 2022, doi: 10.1016/j.chemosphere.2022.134488.
[27] I. López-Cano, A. Roig, M. L. Cayuela, J. A. Alburquerque, et M. A. Sánchez-Monedero, « Biochar improves N cycling during composting of olive mill wastes and sheep manure », *Waste Management*, vol. 49, p. 553-559, mars 2016, doi: 10.1016/j.wasman.2015.12.031.
[28] I. López-Cano, A. Roig, M. L. Cayuela, J. A. Alburquerque, et M. A. Sánchez-Monedero, « Biochar improves N cycling during composting of olive mill wastes and sheep manure », *Waste Management*, vol. 49, p. 553-559, mars 2016, doi: 10.1016/j.wasman.2015.12.031.
[29] Y. Yin *et al.*, « Research progress and prospects for using biochar to mitigate greenhouse gas emissions during composting: A review », *Science of The Total Environment*, vol. 798, p. 149294, déc. 2021, doi: 10.1016/j.scitotenv.2021.149294.
[30] E. Pagans, R. Barrena, X. Font, et A. Sánchez, « Ammonia emissions from the composting of different organic wastes. Dependency on process temperature », *Chemosphere*, vol. 62, n° 9, p. 1534-1542, mars 2006, doi: 10.1016/j.chemosphere.2005.06.044.

Statements and Declarations

The authors declare that no funds, grants, or other support were received during the preparation of this manuscript. The authors have no relevant financial or non-financial interests to disclose. All authors contributed to the study conception and design. Material preparation, data collection and analysis were performed by L. Bourdier and F. Paillet. The first draft of the manuscript was written by F. Paillet and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Data Availability Statement

The datasets generated and/or analyzed during the current study are not publicly available due to the confidential nature of the industrial data involved but are available from the corresponding author on reasonable request.

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- Highlights.docx