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Original article

Title

Low-temperature biochars are more effective in reducing ammonia emissions through various mechanisms during manure composting

Author names and affiliations

Yunsik Shin ¹, Kazunori Iwabuchi ^{2*}, Takanori Itoh ²

¹ Graduate School of Agriculture, Hokkaido University, Kita-9, Nishi-9, Kita-ku, Sapporo, Hokkaido 060-8589, Japan

² Research Faculty of Agriculture, Hokkaido University, Kita-9, Nishi-9, Kita-ku, Sapporo, Hokkaido 060-8589, Japan

***Corresponding author**

iwabuchi@agr.hokudai.ac.jp

Abstract

Biochar is an effective additive for reducing ammonia emissions during composting due to its unique physicochemical properties. However, the properties of biochar vary according to production conditions such as the feedstock and pyrolysis temperature. This heterogeneity may affect the ammonia mitigation performance of biochar during composting, but few studies have investigated the effect of biochar properties on ammonia emissions. Here, we prepared four types of biochar from two feedstocks (dairy manure and Japanese larch) at two pyrolysis temperatures (300 °C and 800 °C) and compared their impact on ammonia emissions during composting. The composting experiments were conducted on a laboratory scale by mixing each biochar (12 g) with dairy manure (300 g) for 19 days. The results showed that biochar addition (10.7–15.3% initial total nitrogen) significantly lowered ammonia emission rates compared to dairy manure alone (18.6% initial total nitrogen), and low-temperature biochars showed better performance than high-temperature biochars. We propose that ammonia emissions are reduced by biochar due to NH_4^+ adsorption and increased nitrification. Abundant surface oxygen-containing functional groups, high cation exchange capacity, high ash content, and low pH are likely the dominant factors reducing ammonia emissions, while high specific surface area is less influential.

Keywords: Pyrolysis temperature; Feedstock; Nitrogen loss; Adsorption; Nitrification

1. Introduction

The livestock population is increasing globally to meet the growing demand for protein sources and produces large amounts of manure. Manure is typically used as an organic fertilizer since it contains many nutrients such as nitrogen (N) and phosphorus (P). However, the application of untreated manure emits gases such as ammonia (NH₃), methane (CH₄), carbon dioxide (CO₂), and nitrous oxide (N₂O) into the atmosphere, causing odor and climate change problems [1, 2]. In addition, manure contains pathogens such as *Escherichia coli* and *Salmonella spp.*, which potentially pose risks to human health [3, 4].

Composting is a biological process that converts organic materials such as manure into stable organic material known as compost. During the composting process, microbes transform the nutrients in manure into plant-available forms under aerobic conditions [5, 6]. This microbial activity generates heat that reduces the number of pathogens and weed seed viability [7, 8]. However, a large amount of ammonia is emitted during composting, reducing the nitrogen content of compost and threatening human health and the environment [9, 10].

In recent decades, significant attention has been paid to biochar addition to reduce ammonia emissions during composting. Biochar is a carbonaceous material produced by the thermochemical conversion of biomass. Biochar is commonly used to improve the composting process and the quality of compost by enhancing aeration and moisture retention, accelerating decomposition, increasing nutrient content and reducing ammonia emissions during composting [11–13]. The advantageous effects of biochar are attributed to its unique physicochemical properties, such as high specific surface area (SSA), high pore volume, abundant oxygen-containing functional groups (OFGs), and high nutrient content [14]. Due to these properties, biochar can adsorb NH₃/NH₄⁺ and promote nitrification, effectively reducing ammonia emissions during composting [14–16]. However, the properties of biochar are highly dependent on its production conditions, especially the pyrolysis temperature and feedstock [17]. Variations in biochar properties affect NH₃/NH₄⁺ adsorption capacity and microbial growth, which are involved in reducing or promoting ammonia emissions in the composting process [18, 19]. Therefore, understanding the impact of the properties of biochar in reducing ammonia emissions during composting is crucial for mitigating ammonia effectively and determining optimal

biochar production conditions, but few studies to date have investigated the influence of biochar produced under different conditions on ammonia emissions during composting [20, 21].

The objectives of this study were 1) to evaluate the effect of biochars produced from two types of feedstocks (dairy manure and Japanese larch) at low (300 °C) and high (800 °C) temperatures on NH₃ emissions during manure composting, and 2) to identify the mechanisms for reducing NH₃ emissions, depending on the different properties of biochars.

2. Materials and Methods

2.1. Materials

Four types of biochars (two feedstocks and two pyrolysis temperatures) were used as composting additives. Japanese larch (*Larix kaempferi* Sarg.) and dairy manure were collected in Hokkaido, Japan, and from the experimental farm at Hokkaido University, Sapporo, Japan, respectively. The feedstocks were dried at 105 °C for 24 h and milled to less than 2-mm particles. Each feedstock was then packed in a crucible, sealed with a lid, and pyrolyzed in a muffle furnace (FO810; Yamato Scientific, Tokyo, Japan) at 300 °C and 800 °C for 1 h at a heating rate of 10 °C min⁻¹. The prepared dairy manure (treated at 300 °C or 800 °C) and Japanese larch (treated at 300 °C or 800 °C) biochars were labeled DB300, DB800, JB300, and JB800, respectively.

The collected dairy manure was also used as a composting material. The moisture content of fresh dairy manure was adjusted to around 60% on a wet basis (wb) by air drying at room temperature to aid the composting process, then packed in a plastic bag and kept in a freezer at -21 °C for storage. Prior to each treatment, the manure was thawed at room temperature overnight.

2.2 Composting

This study investigated five composting treatments: dairy manure alone as a control (CK), dairy manure with DB300 (CDB300), dairy manure with DB800 (CDB800), dairy manure with JB300 (CJB300), and dairy manure with JB800 (CJB800). About 300 g of wet dairy manure (60% wb) and 12 g of dry biochar, corresponding to a mass ratio of biochar to dairy manure of 1:10 on a dry basis, were mixed and subjected to composting treatment. Each treatment was conducted in duplicate.

A schematic diagram of the laboratory-scale composting system is shown in Fig. 1 [22].

Each material was composted using a 0.5-L composting reactor with a lid and placed in an insulated chamber. Platinum resistance thermometers monitored the sample and chamber temperatures, and the chamber temperature was controlled by a microcomputer at 0.3 °C lower than the sample temperature, allowing full-scale composting to be reproduced in the laboratory. The airflow rate was maintained at 0.5 L min⁻¹ kg⁻¹ on a dry and ash-free basis by an air compressor and airflow meter. Composting was conducted for 19 days, and distilled water was added to the composting material to readjust the moisture content to about 60% on day 9 as a turning process. The emitted ammonia during the composting process was evaluated by determining the ammonium concentration in a water trap and 4% boric acid solution (H₃BO₃).

2.3 Analysis

2.3.1 Physicochemical properties

The pH values of the generated biochar and compost in suspension were measured using a pH meter (HI9912N; HANNA, Chiba, Japan). The suspensions were made by adding the samples to distilled water in a ratio of 1:5 for biochar and 1:10 (w/v) for compost, then shaking for 30 min at 120 rpm. The cation exchange capacity (CEC) of each biochar was measured using the ammonium acetate (NH₄OAc) extraction method [23]. The moisture content of each compost was determined by drying the sample at 105 °C for 24 h. The ash content of each biochar and compost was measured by heating the dry samples in a muffle furnace at 600 °C for 3 h, then calculating the amount of ash based on the residual mass. Total C, H and N of biochar and compost were determined using a CE440 Elemental Analyzer (Exeter Analytical, Chelmsford, MA, USA). The O content of biochar was determined by mass balance (O% = 100 – C% – H% – N% – ash%). Total P, K, Ca and S in the samples were analyzed by digesting with an HNO₃ and H₂O₂ system, then conducting ICP-MS (ELAN DRC-e; Perkin Elmer, Waltham, MA, USA). For NH₄⁺-N and NO₃⁻-N determination, compost samples were extracted with 2 M KCl solution [24], the NH₄⁺ and NO₃⁻ in the extracts were steam-distilled with magnesium oxide (MgO) and Devarda's alloy, and the concentrations of NH₄⁺-N and NO₃⁻-N in the distillates were determined using an Agilent 7100 CE system (Agilent Technologies, Inc., Santa Clara, CA, USA). The ammonia concentration in the water trap and boric acid solution were determined by back titration with 0.05 M or 0.005 M sulfuric acid.

2.3.2 Specific surface area

The specific surface area of the biochar was determined by the Brunauer–Emmett–Teller (BET) method. The N₂ adsorption isotherm of each sample at –196 °C was obtained using a surface area and pore size analyzer (Belsorp Mini II; MicrotracBEL, Osaka, Japan). The samples were degassed at 150 °C for 2 h before analysis to remove residual moisture using a Belprep Vac II (MicrotracBEL).

2.3.3 FTIR

Fourier transform infrared spectroscopy (FTIR) was used to determine the surface functional groups of biochar. Each biochar sample was passed through a 0.5-mm mesh sieve for analysis. Spectra from 128 scans were recorded by a JASCO FT/IR-6100 spectrometer with an attenuated total reflectance (ATR) accessory (ATR PRO450-S; Jasco International Co. Ltd., Tokyo, Japan) in the wavenumber range of 4000–550 cm^{–1} with a 4 cm^{–1} resolution. The ATR crystal and pressure tip were cleaned with isopropanol before each measurement. A background scan was subtracted from the sample scans to eliminate the effects of room conditions.

2.3.4. Phytotoxicity test

The phytotoxicity of each compost was assessed by the germination index (GI) [25] using cress (*Lepidium sativum*). Aqueous extracts were prepared by suspending solid samples in distilled water at 1:10 (w/v), shaking for 1 h, then filtered through 0.45 µm membrane filters. Ten seeds and 9 mL of extract (or distilled water for control) were added to Petri dishes containing a filter paper and incubated in the dark at 25 °C. After 3 days, the number of germinated seeds and the radicle lengths were measured. Seeds with root lengths longer than 5 mm were considered germinated. GI was calculated as follows:

$$GI (\%) = \frac{SG}{SG_0} \times \frac{RL}{RL_0} \times 100$$

where SG and SG₀ are the number of germinated seeds in extract and distilled water, and RL and RL₀ are the mean radicle lengths in extract and distilled water, respectively.

2.3.5 Statistical analysis

One-way analysis of variance (ANOVA) was performed to assess differences among the treatments. Two-way ANOVA was used to evaluate the effect of production conditions (feedstocks and pyrolysis temperature) of biochars on reducing NH₃ emission and total nitrogen loss during composting. Where significant differences were observed, the Tukey HSD test was conducted at a 5% level ($\alpha < 0.05$). All statistical analysis was

performed using RStudio v. 1.4.1106 (RStudio team, 2021).

3. Results

3.1 Physicochemical properties of biochars

The physicochemical properties of biochars produced using different feedstocks and temperatures are presented in Table 1. The pH values of biochars derived from dairy manure were about 1.9 units higher than those of Japanese larch biochars generated using the same pyrolysis temperature. Dairy manure biochars had higher N, P, K, S and ash content compared to Japanese larch biochars, whereas Japanese larch biochars had higher C, H and O content and C/N ratios than dairy manure biochars. Low-temperature biochars showed higher CEC values than high-temperature biochars, with dairy manure biochar produced at 300 °C showing the highest value. The SSA of dairy manure and Japanese larch biochars significantly increased with increasing pyrolysis temperature.

FTIR-ATR spectra of biochars produced using different conditions are shown in Fig. 2. The absorbance peaks between 2935 and 2885 cm^{-1} represent the asymmetric and symmetric C-H stretching of aliphatic functional groups [26, 27]. The peaks at 1700 and 1600 cm^{-1} were assigned to C=O stretching for carboxyl groups and aromatic C=C bonds, respectively [28, 29]. The peak at 1430 cm^{-1} was due to the in-plane bending of carbonyl (-COH) groups [30]. The peak at 1270 cm^{-1} was attributed to phenolic -OH stretching [27]. Given the higher ash content of DB300 compared to JB300 (Table 1), the broad peak at 1048 cm^{-1} could be attributed to the S=O bond of sulfoxide [31]. In JB300, the peak at 1030 cm^{-1} was assigned to the symmetric C-O stretching of cellulose, hemicellulose, and lignin [26, 27].

3.2 Composting process

3.2.1 Temperature

All treatments resulted in a sharp increase in temperature and the maximum temperature was reached within 2 days (Fig. 3), then the temperature gradually decreased and stabilized. After the turning process on day 9, the temperatures of all samples increased to over 70 °C within 4 days and then decreased and became constant.

3.2.2 Properties of the composting materials

The properties of the composting materials before and after composting are presented in Table 2. The initial moisture content of all materials was 58.40–60.19% and increased to 58.58–61.54% after composting. All composting materials initially were

alkaline. The initial pH values of Japanese larch biochar-amended materials were significantly lower than that of the control ($p < 0.05$). Subsequently, the pH of all treatments decreased after composting. The initial C content was higher in biochar-amended materials than in the control and decreased after composting for all composting materials, from 40.12–46.78% to 38.15–45.66%. In contrast, the initial total nitrogen (TN) content was lower in biochar-amended materials and increased after composting for all composting materials, from 1.91–2.22% to 2.34–2.76%. Biochar addition increased the initial C/N ratio of the composting material. However, there was no significant difference between the composting materials and the control ($p > 0.05$). As the composting process advanced, the C/N ratio decreased to a range of 14.73–19.15 for all composting mixtures. The ash content increased for all composting materials from 11.33–16.67% to 14.00–21.33% after composting.

3.3 Nitrogen dynamics

3.3.1 Ammonia emissions

The highest NH_3 emissions were found on day 2 and gradually decreased for all composting materials (Fig. 4a). As shown in Fig. 4b, the cumulative NH_3 emissions during composting in CK, CDB300, CDB800, CJB300 and CJB800 were 18.55, 10.73, 15.32, 11.19 and 14.74% initial TN, respectively. Biochar addition significantly reduced NH_3 emissions during composting ($p < 0.05$), and low-temperature biochar-amended materials showed lower cumulative NH_3 emissions rates than high-temperature biochar-amended materials.

3.3.2 NH_4^+ -N and NO_3^- -N content

Figure 5 shows that all biochar-amended composting materials had a lower initial NH_4^+ -N content than CK ($p < 0.05$) due to the low NH_4^+ -N content of biochars. The NH_4^+ -N content of composting materials decreased greatly during composting. Compost with biochars produced at 300 °C showed a relatively higher NH_4^+ -N content than other composts but the difference was not significant ($p > 0.05$). Biochar addition did not significantly affect the initial and final NO_3^- -N content of the composting materials except for CJB300, which showed an increase in NO_3^- -N from 34.5 to 41.5 mg/kg-manure after composting.

3.3.3 TN loss

The highest TN loss was observed in CK (20.82% of initial TN content; Fig. 6),

whereas all biochar-amended materials showed lower TN losses with 13.23, 16.97, 12.90 and 16.05% of initial TN content for CDB300, CDB800, CJB300 and CJB800, respectively.

3.4 Phytotoxicity test

As shown in Fig. 7, the GI of all composting materials significantly increased, from 64.3–67.7% to 88.5–100.3% after composting. Biochar-amended composting materials showed higher GI values than CK, but the difference was not significant.

3.5 Two-way ANOVA

Two-way ANOVA for the effect of biochar pyrolysis temperature and feedstock on NH₃ emissions and TN loss during composting is summarized in Table 3. Only the biochar pyrolysis temperature significantly affected the NH₃ emissions and TN loss during composting. This study found no significant interaction effect on NH₃ emissions between pyrolysis temperature and feedstock.

4. Discussion

4.1 Different properties of biochar depending on pyrolysis temperature and feedstocks

Low-temperature biochars showed higher CEC than high-temperature biochars, likely due to abundant OFGs on the surface of low-temperature biochars (Table 1). Cation exchange occurs through electrostatic attraction by OFGs on the surface of biochar [32] and thus CEC is positively correlated with the abundance of OFGs on biochar [33]. FTIR analysis showed that most OFGs such as carboxyl, phenolic, carbonyl and sulfinyl groups disappeared from high-temperature biochars due to the release of surface oxygen [34], resulting in the lower CEC of high-temperature biochars. Dairy manure biochars showed higher CEC than Japanese larch biochars produced at the same temperature, possibly due to the abundance of minerals such as P, K and S that could contribute to the formation of OFGs such as carboxylic and sulfinyl groups [35, 36].

The pH of dairy manure-derived and high-temperature biochars was higher than that of Japanese larch-derived and low-temperature biochars. The high pH of biochar is associated with a high ash content due to large amounts of carbonate in manure-based biochar compared with wood-based biochar, and carbonate levels are enhanced by high pyrolysis temperature [37].

A higher pyrolysis temperature greatly increased the SSA of biochar. The

increased SSA of biochar is mainly due to the release of volatile matter and the formation of porous structures with increasing pyrolysis temperature [38]. The increase in SSA was lower in dairy manure-derived biochar than in Japanese larch-derived biochar, possibly due to the higher ash content in manure biomass. The pores might be filled or blocked by ash during the pyrolysis process, which can inhibit the increase in SSA [39].

4.2 Effect of different biochars on the composting process and compost maturity

All materials showed a typical temperature change pattern during composting (Fig. 3). The rapid increase in temperature at the beginning of composting is due to the consumption of easily decomposable compounds by microbes. All treatments resulted in a temperature increase above 55 °C for 1 to 2 days, which is necessary to effectively eliminate pathogens and toxic substances in manure [40], indicating that all produced composts underwent a stabilization process. In this study, all composting materials showed similar temperature changes regardless of biochar addition. This result contrasts with previous studies [15, 16] which reported that biochar increased the temperature and maintained higher temperatures than the control without biochar. These previous studies suggested that biochar could decrease the initial moisture content of the composting material, leading to improved aerobic conditions and higher temperatures. However, in this study, the moisture content of the dairy manure was adjusted to around 60%, which is the optimal moisture content for composting, potentially reducing the impact of biochar on moisture content and subsequent temperature changes. Furthermore, adding biochar decreased the temperature more rapidly, perhaps due to the more intensive decomposition associated with increasing the initial C/N ratio of the composting material by biochar [41, 42], although the difference was not significant (Table 2).

The effect on the initial and final pH of the composting material depended on the type of biochar (Table 2). Japanese larch biochars significantly decreased the initial pH of the composting material, while, as mentioned above, dairy manure biochars did not decrease the initial pH due to their high ash content. These differences in the initial pH of the composting mixture could affect ammonia volatilization during composting [43] (This will be discussed in more detail in section 4.3). After composting, the pH decreased for all composts, due to NH₃ volatilization and nitrification [44, 45]. The addition of DB800 resulted in a compost with the highest pH, consistent with the report by Duan et al. [46] that high alkaline biochar resulted in compost with a higher pH than compost without

biochar at the end of the composting process. The amendment of alkaline compost is one effective option for improving plant growth and heavy metal stabilization in acidic soil [47], suggesting that the addition of dairy manure biochar is a promising strategy to address soil acidification.

The C/N ratio is an indicator of compost maturity. In general, the C/N ratio of compost decreases during composting due to the different degradation rates between nitrogen and organic carbon [42], and lower than 21 of the C/N ratio is considered indicative of matured compost [48]. The C/N ratio for all composts decreased to less than 21 in this study (Table 2), indicating that all composts were stabilized even after adding biochar.

The germination index (GI) is a broadly accepted indicator for evaluating the stability and maturity of compost. The GI of all composting materials significantly increased and reached over 80% after composting (Fig. 7), reflecting the reduction of phytotoxic compounds such as organic acid [49]. A GI value above 80% implies that negligible phytotoxic compounds are present in the compost [50], indicating that all composts produced in this study were matured. The phytotoxic test revealed that biochar does not have any adverse effect on seed germination. High ash content in compost is generally undesirable for plant growth because it can lead to high salinity that can suppress seed germination [51]. However, in this study, dairy manure-based biochars did not negatively affect seed germination despite their high ash content, indicating that adding a small quantity of biochar produced from high ash biomass containing manure may not be detrimental to seed germination.

4.3 Effect of different biochars on N transformation during composting

Our results demonstrated that biochar addition can reduce NH_3 emissions during composting (Fig. 4b). Low-temperature biochars mitigated NH_3 emissions better than high-temperature biochars, possibly due to higher CEC (Table 1). CEC reflects the ability of biochar to adsorb positively charged ions such as NH_4^+ [34]. Thus, as mentioned in section 4.1, the higher CEC of low-temperature biochars might allow the adsorption of more NH_4^+ than high-temperature biochars and, as a result, could reduce NH_3 emissions more during composting [18]. However, although JB300 ($40.3 \text{ cmol}_+ \text{ kg}^{-1}$) had about 1.8 times lower CEC than DB300 ($73.8 \text{ cmol}_+ \text{ kg}^{-1}$), the addition of JB300 mitigated NH_3 emissions similar to DB300, indicating that NH_4^+ adsorption is not the only mechanism

by which biochar reduces NH_3 emissions during composting. Zhang et al. [52] confirmed that the application of biochar can promote nitrification in soil, reducing NH_3 emissions. We found that JB300 addition increased the NO_3^- -N content during composting (Fig. 5), suggesting that the nitrification process is promoted by adding JB300 [53]. As mentioned in section 4.2, JB300 decreased the pH of the compost material (Table 2), which may increase the solubility of ammonia, and promote the nitrification and transformation of NH_4^+ to NO_3^- [54]. Zainudin et al. [55] observed that the abundance of nitrifying bacteria was negatively correlated with the pH of composting materials in the range of 8.5 to 9.5. Previous studies [56, 57] also suggested that biochar might improve the nitrification process by creating a favorable microenvironment, such as high SSA and optimal pH, for nitrifying bacteria.

A high SSA is reportedly required for biochar to be an effective composting amendment because high SSA may enhance adsorption capacity and microbial activity [20, 58, 59]. High-temperature biochars with high SSA reduced NH_3 emissions during composting (Table 1) but their performance was lower than that of low-temperature biochars despite their larger SSA. Previous studies [60, 61] reported that the surface area of biochar is less influential than CEC in determining NH_4^+ adsorption capacity. Accordingly, in this study, the NH_4^+ adsorption capacity of high-temperature biochar might be lower than that of low-temperature biochar due to its lower CEC and thus have a lower mitigation effect on NH_3 emissions during composting. In addition, the pores on the biochar surface can become partially blocked during composting [62]. Pore blockage may interfere with the penetration of adsorbents and microorganisms into the biochar interior [63, 64], consequently limiting the potential of high SSA to mitigate ammonia emissions during composting. Furthermore, as the pyrolysis temperature increases, biochar becomes more aromatic [65], possibly decreasing the amount of carbon available to microbes and increasing toxicity (PAHs) [66]. These factors may decrease microbial activity during composting [67].

Most nitrogen losses were associated with NH_3 emissions, which accounted for 81.23–91.57% of the nitrogen loss (Figs 4b and 6), indicating that biochar helps reduce TN losses by inhibiting NH_3 volatilization during composting. The effect of biochar addition in reducing NH_3 emissions and TN losses during composting was influenced mainly by pyrolysis temperature rather than the feedstock (Table 3) because CEC,

functionality, pH and aromaticity are more sensitive to pyrolysis temperature than to the feedstock [37].

In this study, we found that the effect of biochar on ammonia emissions during composting depends on the production conditions, and have suggested different possible mitigation mechanisms due to the biochar properties. We recognize that the experimental design employed in this study, which involved two feedstock types and pyrolysis temperatures, may have provided a partial overview of the impacts of different biochars and their properties. However, we believe that this study addressed the influences of most of the key properties related to the reduction of ammonia emissions and provided valuable information on the optimal production conditions of biochar for reducing ammonia emissions during composting.

5. Conclusion

This study investigated the effects of biochar production conditions (feedstock type and pyrolysis temperature) on reducing ammonia emissions during dairy manure composting. Abundant OFGs on the biochar surface contribute to high CEC, suppressing ammonia volatilization through NH_4^+ adsorption. The high ash content of biochar also influences ammonia mitigation by forming OFGs such as sulfonyl groups, and low pH biochar suppresses ammonia volatilization by promoting nitrification. In contrast, high SSA was less effective in reducing ammonia emissions than the above biochar properties, possibly due to the removal of OFGs by a high pyrolysis temperature, pore-blockage during the composting process, and high aromaticity. These factors are more sensitive to pyrolysis temperature than to the feedstock and thus the pyrolysis temperature is more influential than the feedstock for the ammonia mitigation effect of biochar. Our findings suggest that the use of biochar produced at low temperatures as an additive is effective for reducing ammonia emissions during composting.

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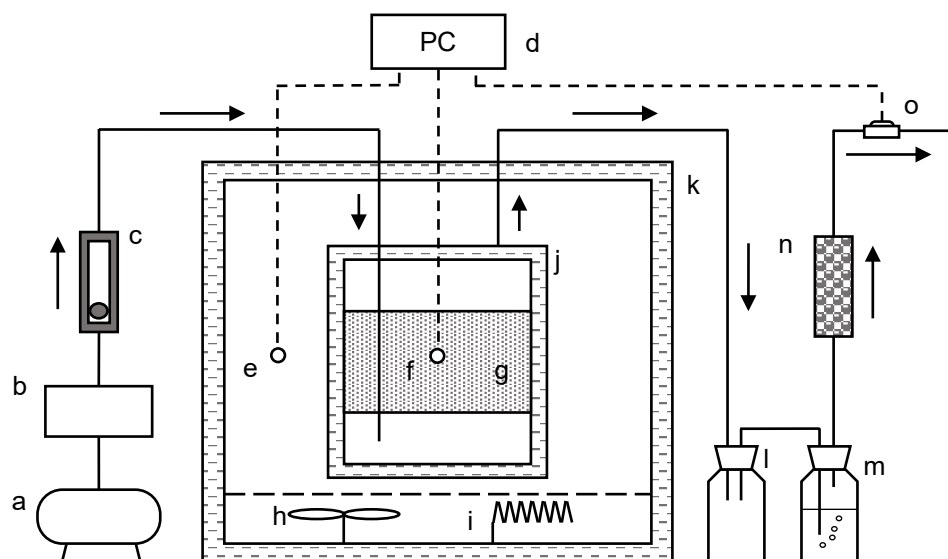


Fig. 1 Schematic diagram of a composting system. **a:** air compressor; **b:** air receiver tank; **c:** airflow meter; **d:** microcomputer; **e:** chamber temperature; **f:** sample temperature; **g:** sample; **h:** fan; **i:** heater; **j:** reactor; **k:** insulation chamber; **l:** water trap; **m:** boric acid solution for ammonia trap; **n:** silica gel; **o:** oxygen sensor

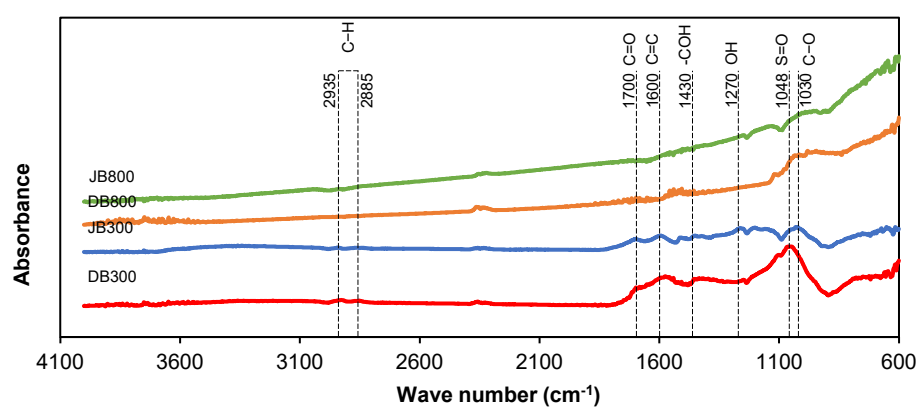


Fig. 2 FTIR spectra of different biochars

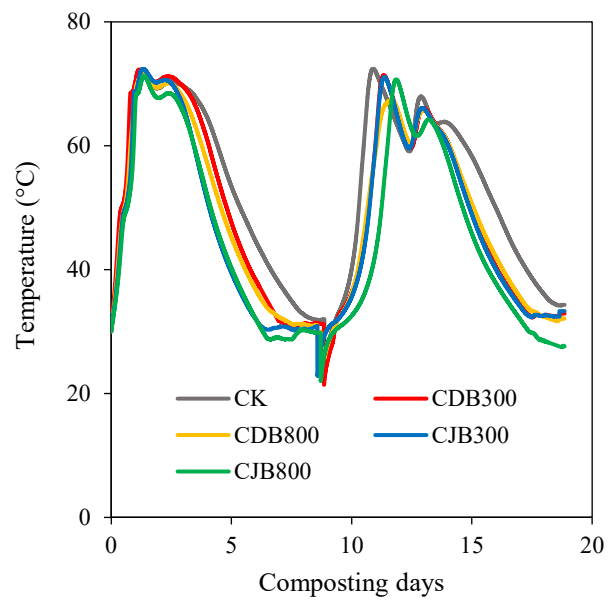


Fig. 3 Temperature changes of composting materials during composting

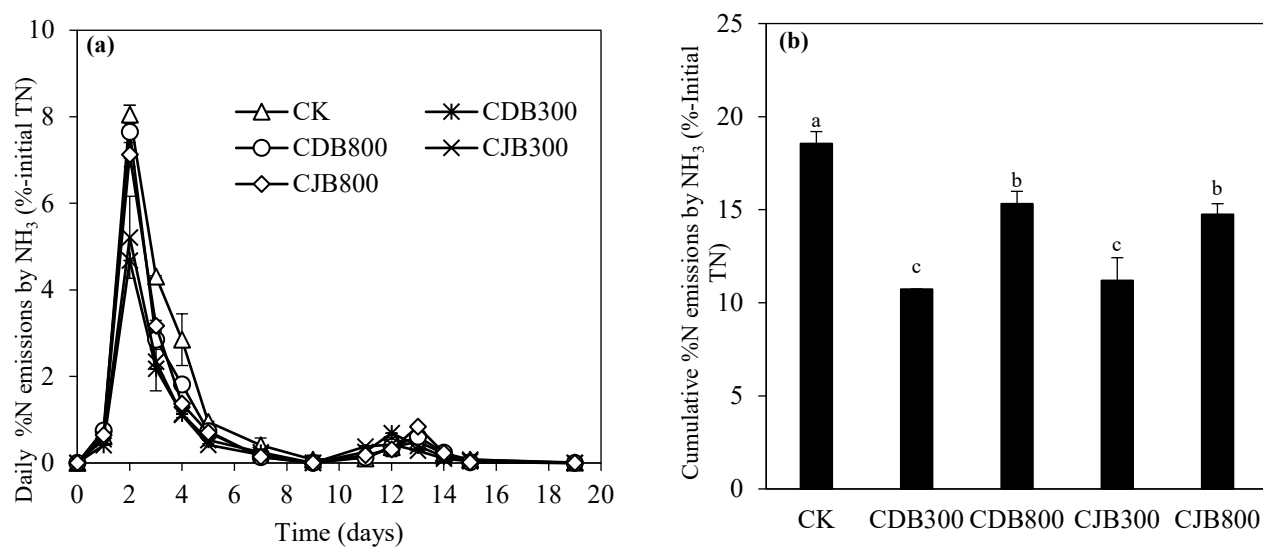


Fig. 4 Daily (a) and cumulative (b) %N emissions as NH₃ during composting. The results represent the mean of two replicates and error bars indicate the standard deviation. Different letters indicate significant differences according to the Tukey test ($p < 0.05$)

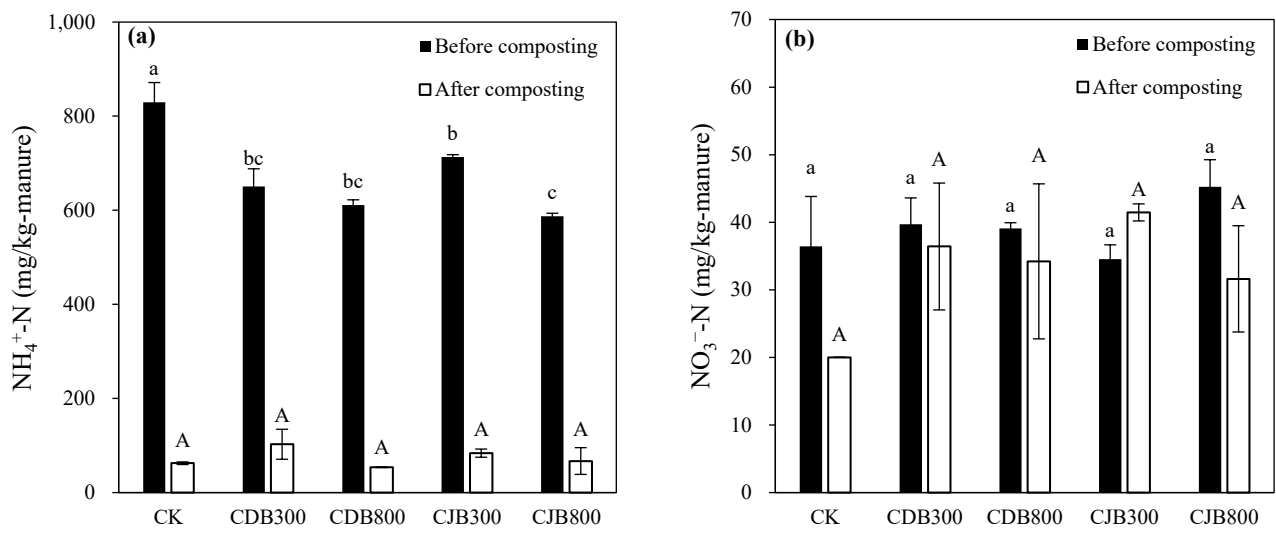


Fig. 5 $\text{NH}_4^+\text{-N}$ content (a) and $\text{NO}_3^-\text{-N}$ content (b) of composting materials before and after composting.

The results represent the mean of two replicates and error bars indicate the standard deviation. Different letters indicate significant differences according to the Tukey test ($p < 0.05$)

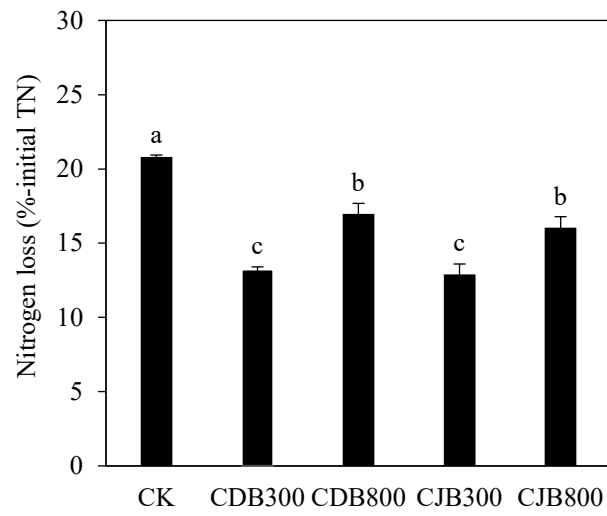


Fig. 6 Total nitrogen loss of composting materials during composting. The results represent the mean of two replicates and error bars indicate the standard deviation. Different letters indicate significant differences according to the Tukey test ($p < 0.05$). Total nitrogen loss (%) = $100 \times (\text{initial TN} - \text{final TN}) / \text{initial TN}$

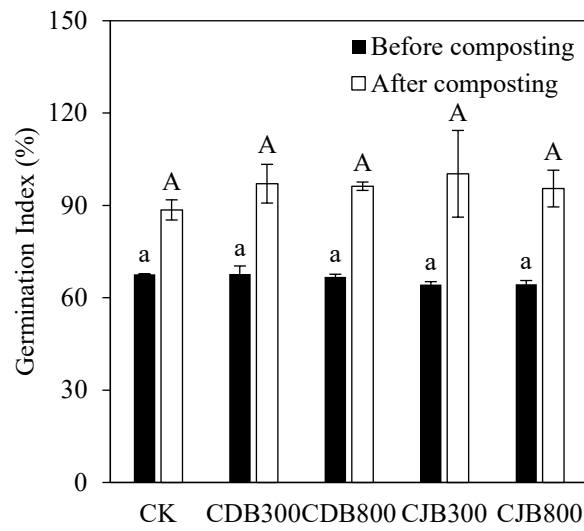


Fig. 7 Germination index of composting materials before and after composting. The result represents the mean of two replicates and error bars indicate the standard deviation. Different letters indicate significant differences according to the Tukey test ($p < 0.05$)

Table 1 The physicochemical properties of the biochars. Results represent the mean of three replicates and error bars indicate the standard deviation.

Parameter	DB300	DB800	JB300	JB800
pH (H ₂ O)	9.45 ± 0.01	11.38 ± 0.02	6.66 ± 0.01	8.52 ± 0.01
C (%)	51.7 ± 0.24	46.7 ± 0.48	65.14 ± 0.24	90.57 ± 0.25
N (%)	2.61 ± 0.02	1.98 ± 0.02	0.08 ± 0.00	0.09 ± 0.00
O (%)	16.98 ± 0.19	3.38 ± 1.75	30.42 ± 0.23	8.52 ± 0.32
P (%)	0.8 ± 0.00	1.11 ± 0.01	0.01 ± 0.00	0.01 ± 0.01
K (%)	2.61 ± 0.03	4.12 ± 0.04	0.03 ± 0.00	0.13 ± 0.01
Ash (%)	25.15 ± 0.04	47.6 ± 1.22	0.02 ± 0.00	0.25 ± 0.06
C/N ratio	16.35 ± 0.15	23.58 ± 0.04	853.1 ± 53.55	939.19 ± 42.91
CEC (cmol _c kg ⁻¹)	73.79 ± 0.27	38.68 ± 0.01	40.33 ± 0.18	31.67 ± 0.09
SSA (m ² g ⁻¹)	3.30 ± 0.17	174.55 ± 21.45	2.38 ± 0.32	658.00 ± 39.12

Table 2 The properties of the composting materials before and after composting. Results represent the mean of two replicates and error bars indicate the standard deviation. Different letters indicate significant differences according to the Tukey test ($p < 0.05$).

Treatment	Moisture content (%)		pH		C/N		TC (%)		TN (%)		Ash (%)	
	Before	After	Before	After	Before	After	Before	After	Before	After	Before	After
CK	60.19 ± 1.46 ^a	61.54 ± 0.63 ^a	10.04 ± 0.05 ^a	8.76 ± 0.04 ^{bc}	18.19 ± 2.72 ^a	14.82 ± 1.08 ^a	40.12 ± 0.21 ^a	38.15 ± 1.38 ^b	2.22 ± 0.21 ^a	2.58 ± 0.09 ^a	12.33 ± 0.67 ^b	15.83 ± 0.17 ^{bc}
CDB300	58.60 ± 1.75 ^a	59.84 ± 1.65 ^a	9.99 ± 0.02 ^{ab}	8.85 ± 0.00 ^b	20.02 ± 0.46 ^a	14.73 ± 0.98 ^a	42.97 ± 0.05 ^a	40.50 ± 1.70 ^{ab}	2.15 ± 0.05 ^a	2.76 ± 0.07 ^a	12.67 ± 1.00 ^b	17.24 ± 0.43 ^b
CDB800	58.42 ± 0.00 ^a	59.71 ± 0.42 ^a	10.01 ± 0.01 ^a	9.35 ± 0.04 ^a	19.58 ± 1.08 ^a	15.54 ± 0.09 ^a	43.07 ± 3.71 ^a	40.45 ± 1.23 ^{ab}	2.20 ± 0.07 ^a	2.34 ± 0.12 ^a	16.67 ± 0.67 ^a	21.33 ± 0.01 ^a
CJB300	58.40 ± 0.44 ^a	58.58 ± 0.70 ^a	9.88 ± 0.04 ^b	8.54 ± 0.14 ^c	22.94 ± 0.03 ^a	18.12 ± 1.63 ^a	43.85 ± 0.10 ^a	42.32 ± 1.55 ^{ab}	1.91 ± 0.01 ^a	2.60 ± 0.09 ^a	11.33 ± 0.33 ^b	14.00 ± 0.00 ^c
CJB800	58.88 ± 0.58 ^a	60.32 ± 0.76 ^a	9.87 ± 0.01 ^b	8.94 ± 0.08 ^b	24.08 ± 6.53 ^a	19.15 ± 2.62 ^a	46.78 ± 6.61 ^a	45.66 ± 2.04 ^a	1.98 ± 0.26 ^a	2.40 ± 0.22 ^a	11.71 ± 0.40 ^b	15.67 ± 1.00 ^{bc}

1 **Table 3** Two-way ANOVA for the effect of biochar production temperature and feedstock on NH₃ emissions
2 and TN loss during composting

3

Factor	Response variables	
	NH ₃ emissions	TN loss
Temperature	$p = 0.0016$	$p < 0.001$
Feedstock	$p = 0.928$	$p = 0.231$
Temperature $\times$ Feedstock	$p = 0.388$	$p = 0.541$