Effect of adding biochar to suppress ammonia emission during livestock manure composting process

（家畜ふん堆肥化過程におけるアンモニア揮散抑制を目的としたバイオ炭添加の効果）

Hokkaido University　Graduate School of Agriculture

Frontiers in Production Sciences　Doctor Course

Yunsik Shin

Executive Summary

In modern agriculture, composting is crucial for managing different agricultural wastes, including livestock manure. However, ammonia emissions during manure composting pose significant challenges, such as odors, air pollution, human-health risks, and nitrogen loss. While various mitigation strategies have been developed, including physical, chemical, and biological methods, they often face limitations, particularly under high ammonia concentrations. Biochar, a by-product of biomass pyrolysis, has emerged as a promising solution due to its large surface area, high adsorption capacity, and ability to influence microbial activity, which can help reduce ammonia emissions. However, its full potential in enhancing existing ammonia removal techniques remains underexplored. Hence, this study investigates how biochar can improve the performance of two representative ammonia mitigation methods—additives during composting and biofilters—towards achieving zero ammonia emissions during composting.

Biochar application during composting (co-composting) is valued for biochar's large pore volume, oxygen-containing functional groups (OFGs), and nutrient content, which enhance NH_3 adsorption and nitrification. However, the effectiveness of biochar depends on its production conditions, such as type of feedstock used and pyrolysis temperature. To maximize its performance, we investigated how biochar properties affect ammonia reduction during composting. Biochars were produced from dairy manure and Japanese larch at 300 and 800°C. The composting experiments were conducted by mixing biochar (12 g) with dairy manure (300 g). Biochars produced at 300°C were more effective in reducing ammonia emission during composting than those made at 800°C. This was attributed to their higher content of OFGs, cation exchange capacity, ash content, and lower pH, which enhance ammonia adsorption and nitrification. However, despite the effectiveness of biochar, some ammonia emissions remain unavoidable during composting, indicating that biochar addition may not be sufficient to achieve zero ammonia emissions during composting.

Compost biofilters offer a promising solution for complete ammonia removal by combining adsorption with biological processes. However, during manure composting, high ammonia concentrations can inhibit microbial activity within biofilters, reducing their efficiency. Maintaining microbial activity is essential for continuous ammonia removal. To address this challenge, we added biochar to the compost

biofilters (co-biofiltration). We hypothesized biochar's high adsorption capacity and large surface area can help capture ammonia directly, reducing its toxicity to microbes. We explored the effect of co-biofiltration using dairy manure biochar (produced at 300°C) at different ratios (0 to 50%) under ammonia concentrations of 1000 and 2000 ppm. All biochar-mixed biofilters showed improved ammonia removal at both concentrations, with the 25% biochar-mixed biofilter achieving the highest performance. Ammonia removal in biofilters occurred through NH_3 adsorption, chemical precipitation, nitrification, and denitrification, all enhanced by biochar at 1000 ppm. However, at 2000 ppm, microbial activity was still inhibited even with biochar due to rapid ammonia accumulation. These findings suggest that compost biofilters can achieve continuous and complete ammonia removal at lower ammonia concentrations, which can be further enhanced by mixing biochar. Therefore, to ensure continuous complete removal of ammonia, it is crucial to reduce ammonia concentrations during composting. In this context, the integration of co-composting and co-biofiltration may offer a promising approach in achieving zero ammonia emissions.

The next phase of the study investigated the synergistic effect of co-composting and co-biofiltration on ammonia removal during composting. Both biofilters (with or without biochar) achieved complete ammonia removal by reducing ammonia emissions during composting through co-composting. Co-biofiltration, in particular, enhanced the long-term reuse potential by improving nitrification during co-composting.

Overall, the findings of this study demonstrate that zero ammonia emissions can be achieved during manure composting, with biochar playing a pivotal role through the combined processes of co-composting and co-biofiltration.

Acknowledgments

First and foremost, I would like to extend my heartfelt gratitude to my supervisor, Prof. Kazunori Iwabuchi, for providing me with the opportunity to study in Japan. His invaluable guidance taught me how to navigate challenges in my research and find answers on my own, ultimately empowering me to complete my work with confidence. I greatly appreciated his calm and witty approach to our conversations, an attitude I hope to carry forward in my own way. It has been an honor and privilege to work under his supervision.

I would also like to express my sincere gratitude to Asst. Prof. Takanori Itoh for his unwavering support and guidance throughout my academic journey. His insightful advice and generous dedication of time were invaluable in advancing my research. I have great respect for his integrity and sense of responsibility, and feel incredibly fortunate to have had the opportunity to learn from him.

My sincere thanks go to the other members of my dissertation committee, Prof. Shigenobu Koseki and Assoc. Prof. Naoto Shimizu, for their invaluable feedback and generous time, which significantly contributed to the refinement of this thesis.

I extend my special thanks to Dr. Dai Hanajima from the National Agriculture and Food Research Organization for granting me access to research facilities and providing insightful advice. The discussions with him greatly advanced my knowledge and played a crucial role in achieving meaningful research results.

I am deeply grateful to all the members of the Agricultural Biosystems Engineering Laboratory. Their support was instrumental to the success of my research, and the cherished memories of our five years together will remain with me always. I am especially thankful to Dr. Mahmudul Islam Piash, whose kindness and encouragement helped me overcome the challenges of my academic journey.

I would also like to thank JST SPRING (grant number JPMJSP2119) for providing financial support throughout my Ph.D. journey.

Finally, I express my deepest gratitude to my family for their unconditional love and trust, which sustained me throughout this journey. My special thanks also go to Prof. Chungsu Han and Asst. Prof. Junghyun Lee for their kind encouragement and guidance.

Table of Contents

List of Figures

List of Tables

Chapter 1 General Introduction

1.1. Background

1.1.1. Challenges for achieving sustainable food security

Due to improved living standards and medical technology, the global population continues to surge, with an estimated 10 billion people by 2050 (UN, 2024). At the same time, livestock production has increased greatly to meet the demand for animal products. These products provide essential nutrients such as protein, vitamins, and minerals, contributing to achieving food security. However, as livestock production expands, substantial quantities of crops are required for animal feed. To meet this demand, significant amounts of chemical fertilizers are typically employed in the modern agricultural field. Chemical fertilizers provide readily available nutrients for plant growth, including nitrogen, phosphorus, and potassium. The rapid nutrient release and simple application of chemical fertilizers enable rapid increases in crop yields, thereby ensuring a steady feed supply (Feinerman and Komen, 2005). However, crops often cannot uptake all the nutrients from fertilizers due to fast release, leading to multiple applications of fertilizer to provide sufficient nutrients for crop growth. These excessive applications cause serious environmental problems including eutrophication, GHG emissions, and ecological balance destruction (Snyder et al., 2009). Therefore, it is necessary to minimize the reliance on chemical fertilizers while meeting the growing demand for food production.

The increase in livestock production inevitably leads to significant manure generation. In fact, manure is rich in essential plant nutrients comparable to those in chemical fertilizer (Sharma et al., 2019), thus being considered an effective and sustainable alternative to chemical fertilizer. However, its direct use is often limited due to its high moisture content, odor nuisance, and pathogen risk (Moore et al., 2006). Thus, proper management is needed to address these problems and to make manure a useful, sustainable fertilizing option.

In summary, we face the challenge of reducing reliance on chemical fertilizers and properly managing manure to achieve sustainable food security.

1.1.2. Composting for sustainable food security and waste management

Composting can effectively address the above issues. Composting is a natural process of recycling organic waste including manure into a valuable organic fertilizer called compost. In modern agriculture, composting has become a crucial component of sustainable agriculture. Most farmers manage manure

through composting in Japan, which accounts for 72% of total manure treatment processing (MAFF, 2021) since it can be started with very little capital and operating costs and is easy to implement (Bougnom et al., 2014). During composting, the microbes convert the organic nutrients into plant-available form (ex., nitrogen in proteins into ammonium and nitrate), enhancing the quality as a fertilizer (Bernal et al., 2009). The microbial activity generates heat during composting that stabilizes the final compost by reducing the number of pathogens, and weed seed viability (Boulter et al., 2002; Eghball and Lesoing, 2000). Furthermore, unpleasant odors and moisture content are also reduced through composting, making the compost easier to store and transport. Thus, composting represents a sustainable and cost-effective solution for both food production and waste management in the agricultural sector.

1.1.3. Ammonia emissions during composting

However, despite the many merits of composting, a significant challenge must be addressed to expand its implementation and maximize its benefits. During composting, Ammonia is generated as a result of microbial decomposition of organic nitrogen present in manure (Bernal et al., 2009). Ammonia is readily volatilized under composting conditions such as high temperature and pH, leading to considerable ammonia emissions into the atmosphere. These emitted ammonia gases can contribute to air pollution, forming fine particulate matter that may pose health risks to humans and animals (Cheng et al., 2019; Wyer et al., 2022). The strong odor generated by ammonia can also be a significant nuisance, negatively impacting the quality of life for nearby residents (Kelsey and Vaserstein, 2000). Furthermore, the loss of nitrogen in the form of ammonia reduces the nutrient content of the compost, reducing its value as a fertilizer. Hence, ammonia emissions must be reduced to the greatest extent possible to maximize the environmental benefits of composting while ensuring its continued role in sustainable agriculture.

1.2. Literature review

There are various methods to mitigate ammonia emissions during composting. These methods can be categorized into internal and external types (Fig. 1.1). The former focuses on optimizing the internal environment of the composting process to minimize ammonia generation. This includes the addition of physicochemical additives and microbial inoculation. On the other hand, the latter aims to prevent the release of ammonia into the atmosphere from outside the composting pile after it has been generated. This primarily includes physical, chemical, and biological processes. Here, a brief description of each technique is described below.

1.2.1. Techniques to remove ammonia emissions during composting.

1.2.1.1. Internal methods

- Physicochemical additives

Lignocellulosic biomass such as sawdust, wood chips, and straw with rich carbon sources and low moisture content is the most commonly used additive known as a bulking agent. Application of these materials can optimize the operating parameters including C/N ratio, pH, and moisture content, reducing ammonia generation during composting (Ren et al., 2022).

Adding adsorbents including biochar, zeolite, and bentonite can effectively reduce ammonia emissions during composting (J. Chen et al., 2023). These materials have high ammonium adsorption capacity due to their large porous structure and high ion exchange capacity, which can hold the ammonia on their surface and inhibit ammonia volatilization (Shan et al., 2021).

These additives can be easily mixed into the compost material without extensively modifying existing composting systems. This simplicity and cost-effectiveness make them frequently utilized by most livestock farms. However, high dosages of materials can incur costs, increasing the overall cost of the composting process. In addition, some additives might introduce unwanted minerals or heavy metals into the compost, potentially leading to toxicity issues for plants or soil.

- Chemical additives

Ammonia readily volatilizes under high pH conditions, like those in manure. Therefore, reducing pH with acidic additives such as acetic acid, citric acid, and phosphoric acid can effectively suppress ammonia volatilization (Hussain et al., 2023; Pan et al., 2018). Additionally, lowering pH can enhance the conversion of ammonium into a less volatile form, such as nitrate, through nitrification (Hussain et al., 2023). However, some acids, such as sulfuric acid, can negatively impact microbial activity and compost maturity (R. Li et al., 2020) Furthermore, acids can be corrosive to composting equipment and infrastructure, leading to maintenance and replacement expenses.

Metal salts, including calcium, magnesium, and phosphate salts, are utilized to reduce ammonia emissions during composting. Magnesium and calcium, combined with phosphorus salts, can chemically react with ammonium ions (NH_4^+) in equimolar ratios to form crystalline compounds such as struvite ($MgNH_4PO_4$) and calcium ammonium phosphate ($CaNH_4PO_4$) (Witter and Kirchmann, 1989). This reaction can conserve more NH_4^+–N within the composting pile, minimizing ammonia volatilization. However, these salts can increase the salinity of the final compost product, potentially affecting the degradation of organic matter and limiting the application of the compost (Gondek et al., 2020).

- Microbial inoculation

Ammonia emissions during composting can be mitigated by directly introducing microorganisms to manure. This microbial inoculation typically involves either a single type of nitrifying microorganism or a complex microbial agent (J. Chen et al., 2023). These microbes help reduce ammonia production by altering enzyme activities, such as urease and protease, or by promoting nitrification or ammonia assimilation during composting (Tian et al., 2024, 2023; Zhao et al., 2020). However, ensuring consistent performance can be challenging because microbial growth and activity are highly influenced by environmental factors such as microbial composition, pH, moisture content, temperature. Additionally, the cost and complexity of developing and applying these microbial agents may pose practical challenges for large-scale composting operations (M. H. M. Zainudin et al., 2022).

1.2.1.2. External methods

- Physical method

The ammonia gas generated from compost material can be effectively removed through adsorption. Activated carbon is commonly used for this purpose, as it can adsorb odor compounds including ammonia when the odorous air stream passes through them (Tsutomu et al., 2004). This method offers advantages such as simple operation and low equipment costs. However, activated carbon has a low adsorption performance for basic gases like ammonia. As a result, it requires frequent replacement, leading to higher operating costs compared to chemical and biological methods (Miller et al., 2018).

- Chemical

Acid scrubbing presents a viable method for ammonia removal during composting. While ammonia is generally soluble in water, scrubbing with water alone is inefficient for handling large quantities. Therefore, acids, typically sulfuric acid, are often added to enhance the ammonia solubility by controlling a pH below 4 (F. Estellés et al., 2011). Catalytic oxidation of ammonia into nitrogen using catalysts such as copper, silver, and gold is another option for removing highly concentrated ammonia in industrial contexts (Lippits et al., 2008).

Nevertheless, these methods ensure sufficient ammonia removal (R. W. Melse and N. W. M. Ogink, 2005), but necessitate costly materials (acids and catalysts) and specialized infrastructure, leading to escalated expenses (Estrada et al., 2011).

- Biological

Biological methods offer eco-friendly solutions for ammonia removal through biological oxidation, primarily achieved via nitrification. Among the most prevalent biological techniques are biofilters and biotrickling filters (J. Chen et al., 2023). Biofilters utilize microorganisms attached to filter media like compost, sludge, peat, or bark to adsorb and break down pollutants, which can later be used as a fertilizer. In contrast, biotrickling filters employ inert media with a continuous trickle of nutrient solution, allowing for precise control of operational parameters. These methods typically incur lower operating costs than physical and chemical methods (Miller et al., 2018). However, like physical methods, frequent media replacement can increase overall operating costs.

1.2.1.3. Limitations of Conventional Ammonia Mitigation Methods under High Concentrations

While various technologies can effectively mitigate ammonia emissions under optimal conditions, their efficiency often diminishes in high-ammonia environments, such as those encountered during manure composting. Physical and chemical methods, such as scrubbing systems, acidification, and adsorbents, are effective but require frequent replenishment of materials and may produce harmful by-products (Mugwili et al., 2023). Biological methods, including biofilters, biotrickling filters, and microbial consortia, are considered more environmentally friendly, but their performance tends to decline under high ammonia concentrations due to the inhibitory effects on microbial activity (Pagans et al., 2005; Riis, 2007).

With the continued expansion of intensive livestock production, addressing the environmental and health impacts of ammonia emissions has become a critical concern. In response, stricter global regulations on ammonia and odor emissions are driving the need for more robust and adaptable mitigation strategies. To keep pace with these developments, it is essential to improve the effectiveness of current methods, particularly under high ammonia conditions, to support sustainable and efficient composting practices.

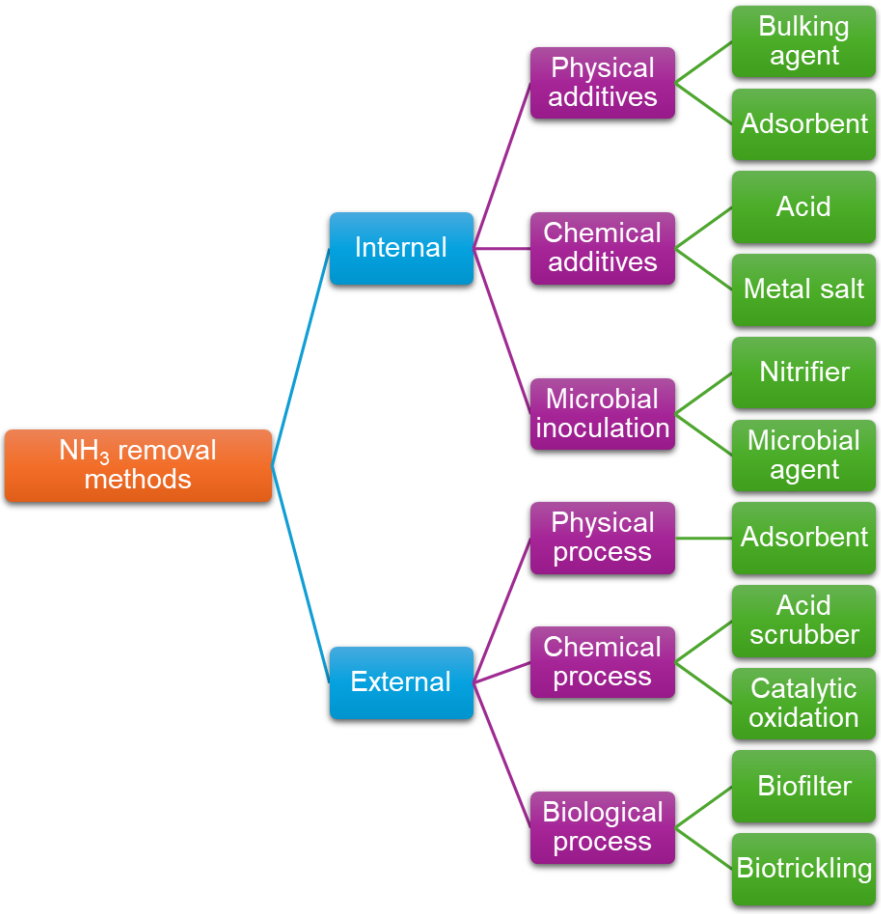

Fig. 1.1. Methods of ammonia removal from the composting process

1.2.2. Biochar application: a solution for improving the limitations of conventional methods

Biochar, a carbon-rich material derived from the thermochemical processing of organic biomass, offers a transformative approach to managing organic waste. Produced from agricultural residues, forestry by-products, and other organic materials, biochar not only reduces waste but also converts it into a valuable resource. This aligns with circular economy principles by promoting sustainable resource use and minimizing environmental impact. The high-temperature process used to create biochar decomposes organic components and forms a highly porous structure (Leng et al., 2021). This unique structure gives biochar exceptional adsorption capabilities, making it a powerful tool for soil remediation by trapping pollutants such as ammonia (Sha et al., 2019). Moreover, biochar's ability to support the growth of beneficial microorganisms enhances soil health and accelerates restoration processes (Ji et al., 2022).

Given these properties, biochar presents a promising solution to the limitations of conventional ammonia removal methods. Its porous structure and acidic functionality enable the effective adsorption of ammonia, reducing its release into the environment (Godlewska et al., 2017). Moreover, the high adsorption capacity of biochar mitigates ammonia toxicity to microbes by limiting ammonia availability (Cai et al., 2022). Beyond its physical properties, biochar supports microbial activity by serving as a habitat for beneficial microorganisms. This facilitates the microbial conversion of ammonia into less harmful compounds through processes like nitrification and denitrification (Q. Liu et al., 2024). These features allow microbial communities to thrive even under high ammonia concentrations.

By addressing these challenges in an environmentally friendly and efficient manner, biochar holds the potential to achieve zero ammonia emissions during composting while promoting sustainability.

1.3. Aims and Objectives

This study aims to explore the potential role of biochar application as a sustainable solution to achieve zero ammonia emissions during composting by addressing the limitations of conventional ammonia mitigation methods. Accordingly, the objectives of this study are to:

1. Investigate the effect of biochar addition on the ammonia removal performance of representative ammonia mitigation methods (adsorbent and biofiltration) and identify the mechanisms.

2. Determine the appropriate production conditions including pyrolysis temperature and feedstock or mixing ratio to optimize the effect of biochar on the ammonia removal performance of each method.

3. Investigate the most effective strategies for utilizing biochar to achieve sustainable, zero ammonia emission composting.

1.4. Structure of the Thesis

Chapter 1: This chapter gives the background of the study, a literature review of overall conventional and improved methods to remove ammonia emissions during composting, the research aims and objectives, and the structure of the thesis.

Chapter 2: This chapter will present the experimental study that evaluates 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 NH_3 emissions during manure composting, and identify the mechanisms for reducing NH_3 emissions, depending on the different properties of biochars.

Chapter 3: In this chapter, we will investigate the impact of mixing biochar with compost biofilter on the NH_3 removal performance, identify the mechanisms of biochar mixed compost for improving removal performance, and understand the optimal mixing ratio of biochar with compost biofilter to achieve sustainable ammonia removal.

Chapter 4: The focus of this chapter is to compare the effect of co-composting, co-biofiltration and their combined system on removing ammonia emissions during the real composting process. It will provide valuable insights into the most effective ways to utilize biochar for zero ammonia emissions.

Chapter 5: In this last chapter, we will summarize key findings, explores strategies to enhance biochar's effectiveness in ammonia removal and its broader applications, and provides recommendations for future research.

Chapter 2 Low-temperature biochars are more effective in reducing ammonia emissions through various mechanisms during manure composting

2.1. Introduction

Among existing ammonia mitigation methods, the use of adsorbents is relatively easy to implement and cost-effective, making it a popular choice for most livestock farms. In recent years, significant attention has been given to the use of biochar as an adsorbent to reduce ammonia emissions during composting. Biochar improves 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 (Qian et al., 2023; Ravindran et al., 2022; Shan et al., 2021). 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 (Sanchez-Monedero et al., 2018). Due to these properties, biochar can adsorb NH_3/NH_4^+ and promote nitrification, effectively reducing ammonia emissions during composting (Chowdhury et al., 2014; Sanchez-Monedero et al., 2018; Steiner et al., 2010). However, the properties of biochar are highly dependent on its production conditions, especially the pyrolysis temperature and feedstock (Tomczyk et al., 2020). The variation in biochar properties can affect its NH_3/NH_4^+ adsorption capacity and microbial growth, which, in turn, will affect ammonia mitigation by biochar (Mandal et al., 2018). Furthermore, biochar can stimulate ammonia emissions depending on its properties (Feng et al., 2022). 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.

2.2. Material and Methods

2.2.1. Material

Four types of biochars (two feedstocks and two pyrolysis temperatures) were used as composting additives (Fig. 2.1). 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.

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 (Fig. 2.2). Each treatment was conducted in duplicate.

Fig. 2.1. Materials for the biochar preparation. (a) dairy manure, (b) japanese larch, and (c) biochar

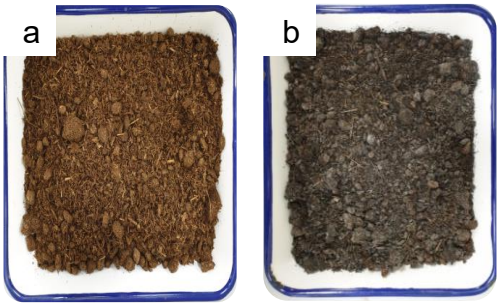

Fig. 2.2. Materials for composting. (a) dairy manure alone and (b) dairy manure with biochar

2.2.2. Composting system

A schematic diagram of the laboratory-scale composting system is shown in Fig. 2.3 (Saludes et al., 2007). 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 (H_3BO_3).

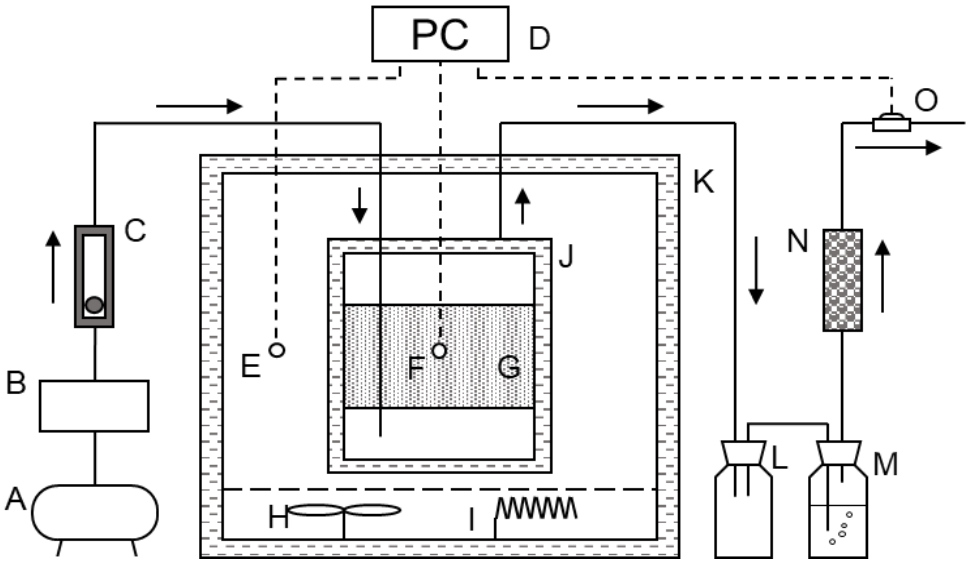

Fig. 2.3. Schematic diagram of composting system. A: air compressor; B: air receiver tank; C: air flowmeter; D: microcomputer; E: chamber temperature; F: sample temperature; G: sample; H: fan; I: heater; J: reactor; K: insulation chamber; L: water trap; M: boric acid solution for ammonia trap; N: silica gel; O: oxygen sensor.

2.2.3. Analysis

2.2.3.1. 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 (NH_4OAc) extraction method (W. Song & Guo, 2012). 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 HNO_3 and H_2O_2 system, then conducting ICP-MS (ELAN DRC-e; Perkin Elmer, Waltham, MA, USA). For NH_4^+–N and NO_3^-–N determination, compost samples were extracted with 2 M KCl solution (Keeney and Nelson., 1982), the NH_4^+ and NO_3^- in the extracts were steam-distilled with magnesium oxide (MgO) and Devarda's alloy, and the concentrations of NH_4^+–N and NO_3^-–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.

2.2.3.2. Specific surface area

The specific surface area of the biochar was determined by the Brunauer–Emmett–Teller (BET) method. The N_2 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).

2.2.3.3. 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.

2.2.3.4. Phytotoxicity test

The phytotoxicity of each compost was assessed by the germination index (GI) (Kebibeche et al., 2019) 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:

$$\text{GI}\ (\%) = \frac{\text{SG}}{\text{SG}_0} \times \frac{\text{RL}}{\text{RL}_0} \times 100$$

where SG and SG_0 are the number of germinated seeds in extract and distilled water, and RL and RL_0 are the mean radicle lengths in extract and distilled water, respectively.

2.2.3.5. 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 NH_3 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).

2.3. Results

2.3.1. Physicochemical properties of biochars

The physicochemical properties of biochars produced using different feedstocks and temperatures are presented in Table 2.1. 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.

Table. 2.1. The physicochemical properties of biochars

Parameters	DB300	DB800	JB300	JB800
pH (H_2O)	9.45	11.38	6.66	8.52
C (%)	51.7	46.7	65.14	90.57
H (%)	3.56	0.34	4.34	0.57
N (%)	2.61	1.98	0.08	0.09
O (%)	16.98	3.38	30.42	8.52
P (%)	0.8	1.11	0.01	0.01
K (%)	2.61	4.12	0.03	0.13
S (%)	0.3	0.16	0.06	0.09
Ash (%)	25.15	47.6	0.02	0.25
C/N ratio	19.81	23.59	814.25	1006.33
CEC ($cmol_+ kg^{-1}$)	73.8	38.7	40.3	31.7
SSA ($m^2 g^{-1}$)	3.30	174.55	2.38	658

FTIR-ATR spectra of biochars produced using different conditions are shown in Fig. 2.4. The absorbance peaks between 2935 and 2885 cm^{-1} represent the asymmetric and symmetric C-H stretching of aliphatic functional groups (Cantrell et al., 2012; Domingues et al., 2017). The peaks at 1700 and 1600 cm^{-1} were assigned to C=O stretching for carboxyl groups and aromatic C=C bonds, respectively (Cole et al., 2019; X. Liu & He, 2017). The peak at 1430 cm^{-1} was due to the in-plane bending of carbonyl (-COH) groups (Yuan et al., 2011). The peak at 1270 cm^{-1} was attributed to phenolic -OH stretching (Domingues et al., 2017). Given the higher ash content of DB300 compared to JB300 (Table 2.1), the broad peak at 1048

cm^{-1} could be attributed to the S=O bond of sulfoxide (Huang et al., 2019). In JB300, the peak at 1030 cm^{-1} was assigned to the symmetric C-O stretching of cellulose, hemicellulose, and lignin (Cantrell et al., 2012; Domingues et al., 2017).

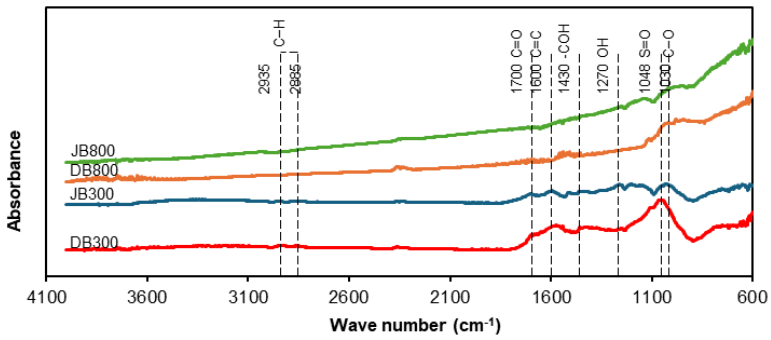

Fig. 2.4. FTIR spectra of different biochars

2.3.2. Composting process

2.3.2.1. Temperature

All treatments resulted in a sharp increase in temperature and the maximum temperature was reached within 2 days (Fig. 2.5), 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.

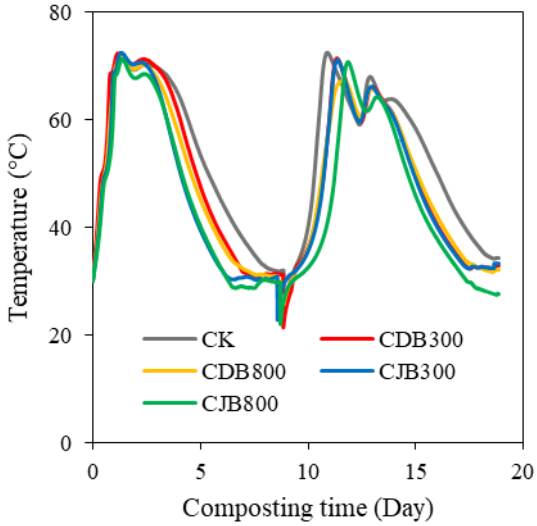

Fig. 2.5. Temperature changes of composting materials during composting

2.3.2.2. Properties of the composting materials

The properties of the composting materials before and after composting are presented in Table 2.2. 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 those 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.

Table. 2.2. The properties of 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		C (%)		N (%)		Ash (%)	
	Before	After	Before	After	Before	After	Before	After	Before	After	Before	After
CK	60.19 ± 1.46a	61.54 ± 0.63a	10.04 ± 0.05a	8.76 ± 0.04bc	18.19 ± 2.72a	14.82 ± 1.08a	40.12 ± 0.21a	38.15 ± 1.38b	2.22 ± 0.21a	2.58 ± 0.09a	12.33 ± 0.67a	15.83 ± 0.17bc
CDB300	58.60 ± 1.75a	59.84 ± 1.65a	9.99 ± 0.02ab	8.85 ± 0.00b	20.02 ± 0.46a	14.73 ± 0.98a	42.97 ± 0.05a	40.50 ± 1.70ab	2.15 ± 0.05a	2.76 ± 0.07a	12.67 ± 1.00a	17.24 ± 0.43b
CDB800	58.42 ± 0.00a	59.71 ± 0.42a	10.01 ± 0.01a	9.35 ± 0.04a	19.58 ± 1.08a	15.54 ± 0.09a	43.07 ± 3.71a	40.45 ± 1.23ab	2.20 ± 0.07a	2.34 ± 0.12a	16.67 ± 0.01a	21.33 ± 0.01a
CJB300	58.40 ± 0.44a	58.58 ± 0.70a	9.88 ± 0.04b	8.54 ± 0.14c	22.94 ± 0.03a	18.12 ± 1.63a	43.85 ± 0.10a	42.32 ± 1.55ab	1.91 ± 0.01a	2.60 ± 0.09a	11.33 ± 0.33a	14.00 ± 0.00c
CJB800	58.88 ± 0.58a	60.32 ± 0.76a	9.87 ± 0.01b	8.94 ± 0.08b	24.08 ± 6.53a	19.15 ± 2.62a	46.78 ± 6.61a	45.66 ± 2.04a	1.98 ± 0.26a	2.40 ± 0.22a	11.71 ± 0.40a	15.67 ± 1.00bc

2.3.3. Nitrogen dynamics

2.3.3.1. Ammonia emissions

The highest NH_3 emissions were found on day 2 and gradually decreased for all composting materials (Fig. 2.6a). As shown in Fig. 2.6b, 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 emissi ons during composting (p < 0.05), and low-temperature biochar-amended materials showed lower cumulative NH_3 emissions rates than high-temperature biochar-amended materials.

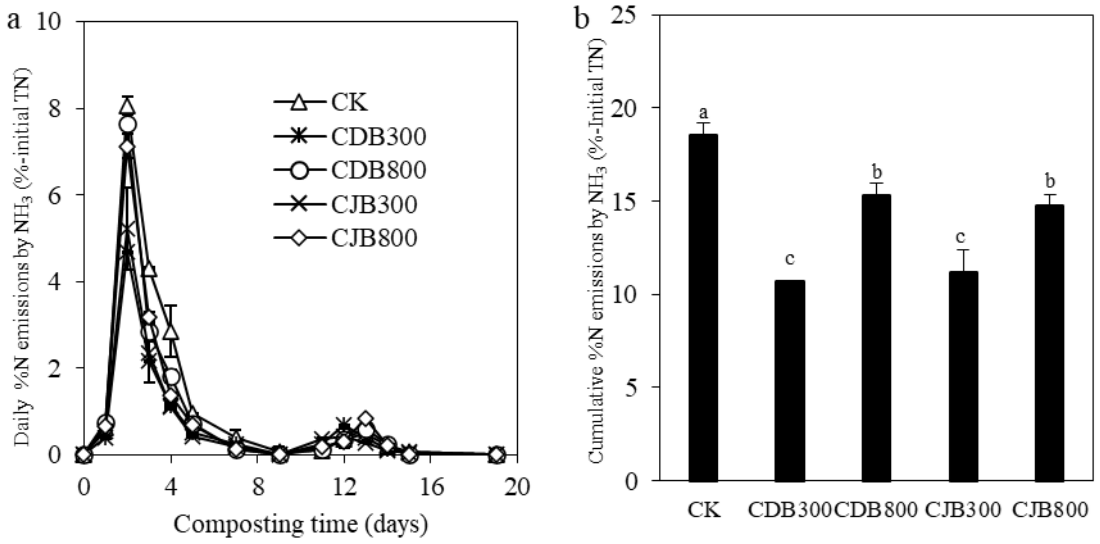

Fig. 2.6. (a) Daily and (b) cumulative %N emissions as NH_3 during composting. The results represent the mean of two replicates and error bars indicate the standard deviation. Different letters indicate significant differences according to the Tukey test ($p < 0.05$).

2.3.3.2. NH_4^+−N and NO_3^-−N content

Figure 2.7 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.

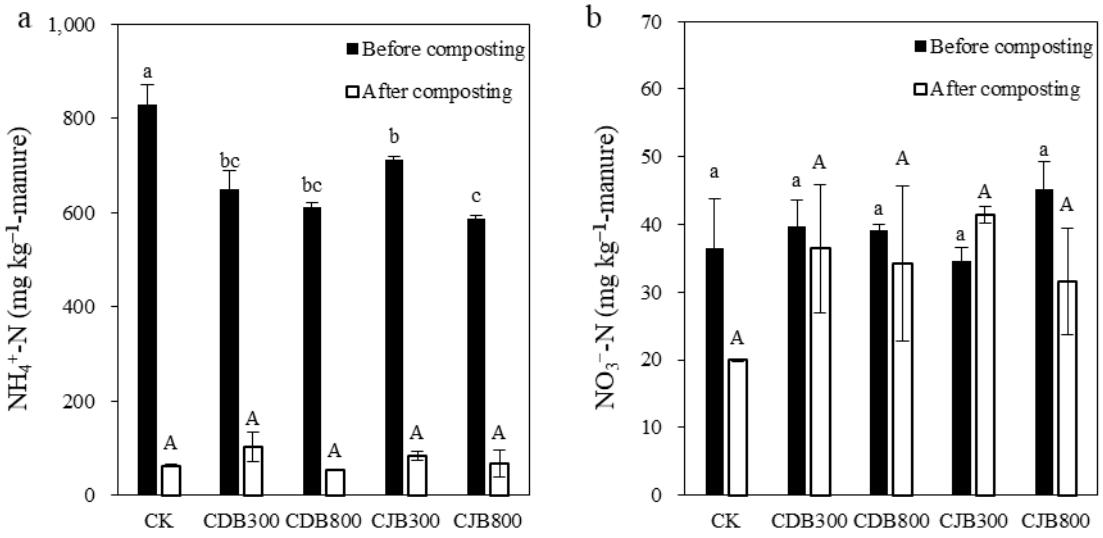

Fig. 2.7. (a) NH_4^+−N and (b) NO_3^-−N content of composting materials before and after composting. The results represent the mean of two replicates and error bars indicate the standard deviation. Different letters indicate significant differences according to the Tukey test ($p < 0.05$)

2.3.3.3. TN loss

The highest TN loss was observed in CK (20.82% of initial TN content; Fig. 2.8), 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.

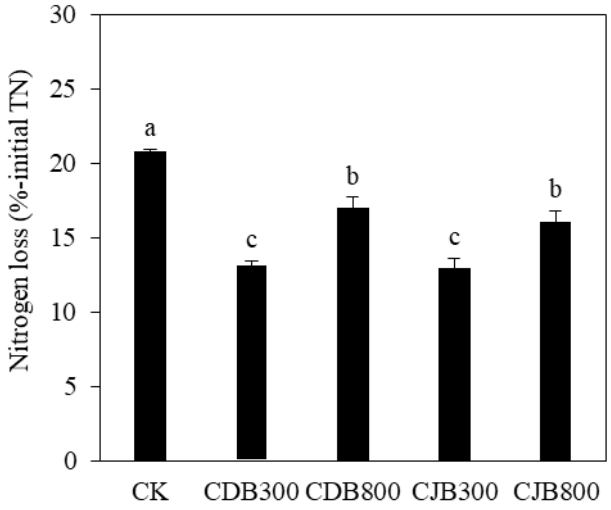

Fig. 2.8. Total nitrogen loss of composting materials during composting. The results represent the mean of two replicates and error bars indicate the standard deviation. Different letters indicate significant differences according to the Tukey test ($p < 0.05$). Total nitrogen loss (%) =100 × (initial TN − final TN)/initial TN

2.3.4. Phytotoxicity test

As shown in Fig. 2.9, 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.

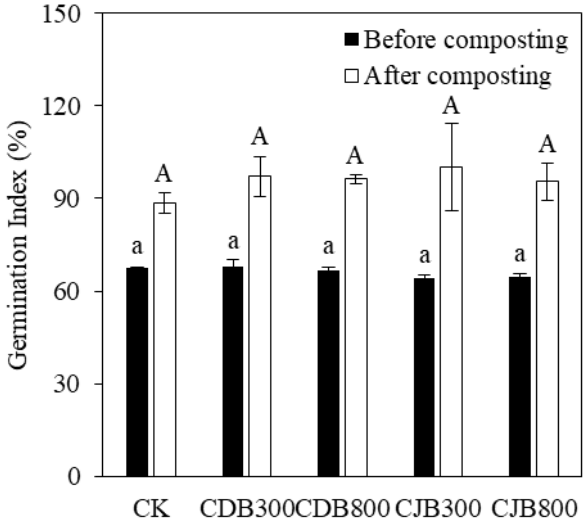

Fig. 2.9. Germination index of composting materials before and after composting. The result represents the mean of two replicates and error bars indicate the standard deviation. Different letters indicate significant differences according to the Tukey test ($p < 0.05$)

2.3.5. Two-way ANOVA

Two-way ANOVA for the effect of biochar pyrolysis temperature and feedstock on NH_3 emissions and TN loss during composting is summarized in Table 2.3. Only the biochar pyrolysis temperature significantly affected the NH_3 emissions and TN loss during composting. This study found no significant interaction effect on NH_3 emissions between pyrolysis temperature and feedstock.

Table. 2.3. Two-way ANOVA for the effect of biochar production temperature and feedstock on NH_3 emissions and TN loss during composting

Factor	Response variables	
	NH_3 emissions	TN loss
Temperature	$p = 0.0016$	$p < 0.001$
Feedstock	$p = 0.928$	$p = 0.231$
Temperature × Feedstock	$p = 0.388$	$p = 0.541$

2.4. Discussion

2.4.1. 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 2.1). Cation exchange occurs through electrostatic attraction by OFGs on the surface of biochar (Huff et al., 2018) and thus CEC is positively correlated with the abundance of OFGs on biochar (Lago et al., 2021). 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 (Sumaraj et al., 2020), 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 (Domingues et al., 2020; Mészáros et al., 2007).

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 (Mukome et al., 2013).

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 (Tan et al., 2018). 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 (Lee et al., 2010).

2.4.2. Effect of different biochars on the composting process and compost maturity

All materials showed a typical temperature change pattern during composting (Fig. 2.5). 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 (Bernal et al., 2009), 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 (Chowdhury et al., 2014; Steiner et al., 2010) 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 (Vandecasteele et al., 2016; Zhan et al., 2021), although the difference was not significant (Table 2.2).

The effect on the initial and final pH of the composting material depended on the type of biochar (Table 2.2). 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 (Liang et al., 2006) (This will be discussed in more detail in section 2.4.3). After composting, the pH decreased for all composts, due to NH_3 volatilization and nitrification (Gao et al., 2010; Wong et al., 2017). The addition of DB800 resulted in a compost with the highest pH, consistent with the report by Duan et al. (2019) 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 (Neina, 2019), 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 (Zhan et al., 2021), and a value lower than 21 is considered indicative of matured compost (Khan et al., 2014). The C/N ratio for all composts decreased to less than 21 in this study (Table 2.2), 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 (Hase and Kawamura, 2012). A GI value above 80% implies that negligible phytotoxic compounds are present in the compost (Tiquia et al., 1996), 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 (Carpıcı et al., 2009). 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.

2.4.3. Effect of different biochars on the composting process and compost maturity

Our results demonstrated that biochar addition can reduce NH_3 emissions during composting (Fig. 2.6b). Low-temperature biochars mitigated NH_3 emissions better than high-temperature biochars, possibly due to higher CEC (Table 2.1). CEC reflects the ability of biochar to adsorb positively charged ions such as NH_4^+ (Sumaraj et al., 2020). Thus, as mentioned in section 2.4.1, 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 (Mandal et al., 2018). However, although JB300 (40.3 $cmol_+ kg^{-1}$) had about 1.8 times lower CEC than DB300 (73.8 $cmol_+ kg^{-1}$), 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. (K. Zhang et al., 2017) 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. 2.7), suggesting that the nitrification process is promoted by adding JB300 (Bello et al., 2020). As mentioned in section 2.4.2, JB300 decreased the pH of the compost material (Table 2.2), which may increase the solubility of ammonia, and promote the nitrification and transformation of NH_4^+ to NO_3^- (X. Li et al., 2020). Zainudin et al. (M. H. Zainudin et al., 2020) 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 (López-Cano et al., 2016; Zhang and Sun, 2014) 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 (Agyarko-Mintah, Cowie, Singh, et al., 2017; W. Chen et al., 2017; Jassal et al., 2015). High-temperature biochars with high SSA reduced NH_3 emissions during composting (Table 2.1) but their performance was lower than that of low-temperature biochars despite their larger SSA. Previous studies (Cai et al., 2016; H. Zhang et al., 2017) 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 (Kah et al., 2018). Pore blockage may interfere with the penetration of adsorbents and microorganisms into the biochar interior (Quilliam et al., 2013; J. Zhang et al., 2018), consequently limiting the potential of high SSA to mitigate ammonia emissions during composting. Furthermore, as the pyrolysis temperature increases, biochar becomes more aromatic (Wiedemeier et al., 2015), possibly decreasing the amount of carbon available to microbes and increasing toxicity (PAHs) (G. Zhang et al., 2018). These factors may decrease microbial activity during composting (Vijayanand et al., 2023).

Most nitrogen losses were associated with NH_3 emissions, which accounted for 81.23–91.57% of the nitrogen loss (Figs 2.6b and 8), 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 2.3) because CEC, functionality, pH and aromaticity are more sensitive to pyrolysis temperature than to the feedstock (Mukome et al., 2013).

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.

2.5. 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.

Chapter 3 Mixing biochar with compost biofilters to improve removal of ammonia

3.1. Introduction

In the previous chapter, we demonstrated that adding biochar during composting can effectively reduce ammonia emissions. However, this method primarily suppresses ammonia emissions within the compost pile; consequently, some emissions to the atmosphere remain unavoidable. This is a common limitation of internal methods. To address this, external methods are often employed. Among these, biotechnologies are considered more sustainable than physicochemical methods due to their lower energy consumption and reduced environmental impact, as their removal processes rely on microbial activity (Wang et al., 2021).

Especially, compost biofilters have been used for the past several decades to sustainably and cost-effectively remove undesired gaseous compounds such as ammonia from the air (Barbusiński et al., 2020; Zhong et al., 2020). Compost comprises a varied microbial population, abundant nutrients, and has a high adsorption capacity, allowing the effective removal of ammonia through biological oxidation (nitrification) and physicochemical adsorption (Pagans et al., 2007). Many studies have reported the high efficiency of compost biofilters for removing ammonia gas (Y. X. Chen et al., 2005; Hort et al., 2009; Pagans et al., 2005). The removed ammonia can act as a valuable nitrogen source in compost, improving its quality as an organic fertilizer. However, biofilters are typically used to remove only low-concentration odorous gases due to their low removal efficiency at higher malodorous gas concentrations (Barbusiński et al., 2020). For example, Pagans et al. (2005) reported essentially 100% ammonia removal at ammonia concentrations of up to 650 mg-NH_3 m^{-3}, but a drastic decrease at concentrations above 2000 mg-NH_3 m^{-3}. Some nitrification is required to remove ammonia continuously through a biofilter, but high levels of toxic ammonia often inhibit the growth of nitrifying bacteria, limiting the performance of compost biofilters (Maia et al., 2012), thus impacting their utility, especially in composting facilities where ammonia emissions are highest at the beginning of the composting process. We therefore need to better understand the changes in microbial dynamics, and particularly of nitrifying bacteria, in biofilters as a function of ammonia concentration. Improved strategies to ensure stable operation at high ammonia concentrations are also necessary. Baquerizo et al. (2009) previously suggested that moistening the top of the biofilter during filtration might improve nitrification by washing out the ammonia, but this approach entails additional operating costs and the potential risk of pollution, such as by leaching. There is therefore a need for a more sustainable and efficient solution.

Biochar is produced by the pyrolysis of various types of biomasses under limited oxygen conditions and has recently found use as an additive in composting (Agyarko-Mintah, Cowie, Van Zwieten, et al., 2017), wastewater treatment (Yu et al., 2020) and anaerobic digestion (Cai et al., 2022) to mitigate ammonia toxicity. Biochar can alleviate the inhibition of biological reactions by ammonia because it can improve the resistance of the microbial community to ammonia by providing a porous habitat for the growth and survival of microorganisms (Cai et al., 2022), and it has a high capacity for NH_4^+ adsorption, which can reduce the level of ammonia toxicity to microbes (Zhu et al., 2017). Moreover, biochar can directly lower ammonia emissions by adsorbing ammonia during composting (Shin et al., 2024). However, no study to date has examined the effect of biochar on the efficiency of ammonia removal by compost biofilters at various ammonia concentrations, although Das et al. (2019) investigated the effect of biochar addition on the removal efficiency of H_2S (a toxic and odorous gas similar to NH_3) by a compost biofilter at a high loading of H_2S. They suggested that biochar can improve the H_2S removal performance of biofilters due to its porous structure, thereby enhancing the surface retention of H_2S, mitigating the toxicity of H_2S to microbes and promoting microbial growth and overall removal efficiency. Similarly, mixing biochar into compost can alleviate the inhibitory effect of highly concentrated ammonia by adsorbing more ammonia and improving the growth of nitrifying bacteria.

We anticipated that mixing biochar with compost would improve the ammonia removal performance of a compost biofilter by enhancing its adsorption capacity and nitrification. The objectives of this study were (1) to investigate the impact of mixing biochar on the NH_3 removal performance of compost biofilters, (2) to identify the mechanisms by which biochar mixed compost improves removal performance, and (3) to determine the optimum mixing ratio of biochar with compost biofilter to achieve sustainable ammonia removal. We mixed dairy manure-derived biochar with dairy manure compost at different ratios (0, 10, 25, and 50%, dry basis) and evaluated their NH_3 removal capacity at two ammonia concentrations (1000 and 2000 ppm). Additionally, we analyzed changes in the properties, microbial communities, and the abundance of functional genes in the compost biofilters during biofiltration.

3.2. Materials & Methods

3.2.1. Preparation of compost and biochar

Matured dairy manure compost, obtained from a dairy farm in Hokkaido, Japan, was used as the filter medium. The compost was sieved to a particle size of less than 9.55 mm to ensure proper packing in the reactor. Raw dairy manure was collected from the Hokkaido University dairy farm for biochar production. The manure was oven-dried at 105 °C for 24 hours and then sieved to a particle size of less than 2 mm. The dried manure was placed in alumina crucibles with lids and pyrolyzed in a muffle furnace at 300 °C for 1 hour using a heating rate of 10 °C per minute. The properties of the compost and the biochar are summarized in Table 3.1.

Table. 3.1. Properties of compost and biochar

	Compost	Biochar
pH	8.40	9.19
C (%)	41.48	52.36
N (%)	2.68	2.76
P (%)	1.26	1.45
K (%)	2.41	2.68
Ca (%)	2.12	2.36
Mg (%)	0.92	1.03
Ash (%)	22.67	24.00
SSA ($m^2\ g^{-1}$)	2.42	4.13
CEC ($cmol_c\ kg^{-1}$)	55.61	70.53

3.2.2. Experimental setup

The experimental setup is illustrated in Fig. 3.1. The laboratory-scale biofilter system consisted of an air compressor, airflow meter, ammonia solution, biofilter, and boric acid. The compost (400 g) or compost and biochar mixture (400 g) was enclosed in a cylindrical acrylic column (Ø100 x 200 mm). Biochar was mixed with the compost in ratios of 1:9, 1:4, and 1:1 (dry basis), corresponding to 18, 45, and 90 g of biochar, respectively. The mixtures were labeled B10, B25, and B50, with compost alone labeled as CB. The moisture content of the filter medium was adjusted to about 60% to standardize the effect of water content on ammonia removal. A perforated plate was placed at the bottom of the filter medium to support

the packing and ensure that the vapor stream containing ammonia was distributed uniformly. Air was provided to the biofilter using an air compressor and airflow meter, with an airflow rate of 124.1 mL/min. Ammonia gas was supplied by bubbling air through an ammonia solution. The target ammonia gas concentrations were 2000 and 1000 ppm and were controlled by adjusting the concentration of the ammonia solution. The empty bed resistance time (EBRT) and concentration range for the supplied ammonia are summarized in Table 3.2. The ammonia solution was replaced every day. The process was stopped when the outlet concentration of ammonia exceeded 20% of the inlet ammonia concentration.

Each biofiltration experiment was conducted twice. The first experiment, performed in triplicate, was carried out without sampling to evaluate the ammonia removal performance of the biofilter. The second experiment, with sampling, was conducted in duplicate to understand the mechanism of ammonia removal by the biofilter and the effect of biochar on this mechanism. Samples were collected 5 cm above the bottom of the biofilter medium on days 0, 3, and 7 for the 2000 ppm system, and on days 0, 3, 7, 14, and 21 for the 1000 ppm system. The samples were immediately stored at –20 °C until subsequent analysis of their physicochemical properties and DNA extraction.

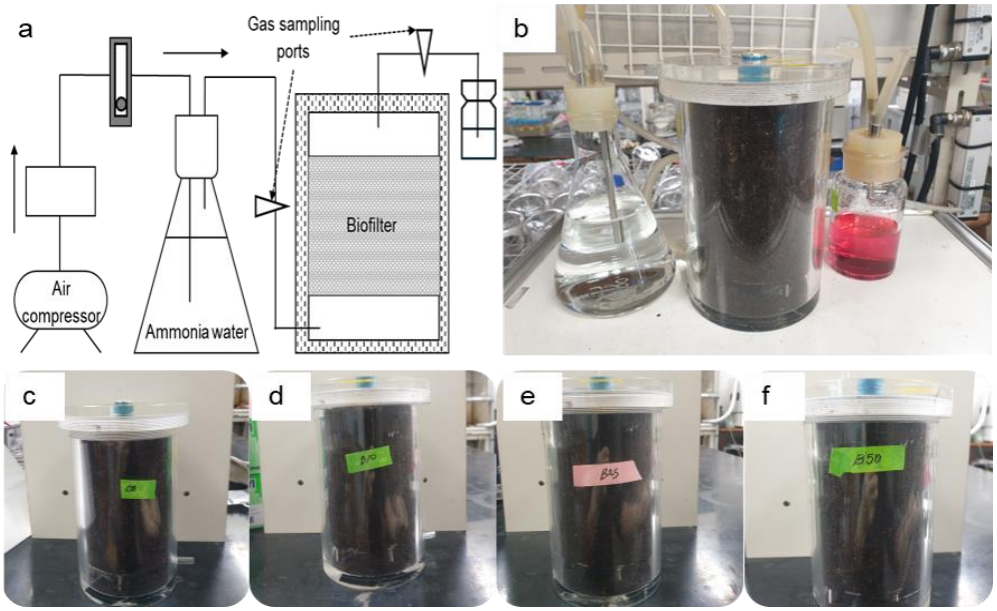

Fig. 3.1. Experimental set up of biofiltration system. (a) schematic diagram of the biofiltration system, (b) the external view of the system, and (c-e) filter medium: (c) CB, (d) B10, (e) B25, and (f) B50

Table. 3.2. Summary of biofiltration conditions

Treatments	Inlet NH_3 concentration range, mg m^{-3} (ppm)	EBRT (min)	Mass loading rate (g-NH_3 m^{-3} h^{-1})	Average biofiltration time* (day)
CB_2000	1044.7–1463.5 (1500–2100)	10.1	6.2–8.7	7
B10_2000	1044.7–1742.3 (1500–2100)	9.8	6.4–9.0	8
B25_2000	1044.7–1427.8 (1500–2050)	9.1	6.9–9.5	9
B50_2000	1044.7–1463.5 (1500–2100)	9.6	6.6–9.2	8
CB_1000	418.2–766.6 (600–1100)	10.1	2.5–4.5	24
B10_1000	418.2–696.9 (600–1000)	9.8	2.6–4.3	27
B25_1000	418.2–766.6 (600–1100)	9.1	2.8–5.1	28
B50_1000	418.2–696.9 (600–1000)	9.6	2.6–4.4	27

* Removal efficiency under 80%

3.2.3. Gas sampling and assessment of biofilter performance

The ammonia gas from the inlet and outlet ports of the biofilter system was collected daily using Tedlar bags, and the ammonia concentration was determined using a gas detector tube (Kitagawa, Japan) with a gas aspiration pump (Kitagawa, Japan).

The performance of the biofilter was evaluated in terms of the inlet loading rate, the removal efficiency, total removal capacity of the biofilter, and the EBRT, estimated using the following equations:

$$Inlet\ loading\ rate\ (\mathrm{mg\ m^{-3}\ h^{-1}}) = \frac{Q}{V} \times C_{in}$$

$$Removal\ efficiency\ (\%) = \frac{(C_{in} - C_{out})}{C_{in}} \times 100$$

$$Total\ removal\ capacity\ (\mathrm{g\ m^{-3}}) = \int \frac{Q(C_{in} - C_{out})}{1000V} \mathrm{d}t$$

$$EBRT\ (\mathrm{h}) = \frac{V}{Q}$$

where Q is the gas flow rate (m^3 h^{-1}), V is the volume of the filter medium (m^3) and C_{in} and C_{out} are the average inlet and outlet ammonia concentrations (mg m^{-3}), respectively. The ammonia concentration is converted from ppm to mg m^{-3} using the equation provided by the manufacturer:

$$(\mathrm{mg\ m^{-3}}) = (ppm) \times\ 10^{-6}\ \times \frac{17}{22.4} \times \frac{273.15}{(273.15+\mathrm{t})}$$

where t is the ambient temperature (°C).

3.2.4. Microbial analysis

3.2.4.1. DNA extraction

DNA extraction was performed in duplicate for each sample using Quick-DNA Fecal/Soil Microbe Kits (Zymo Research, USA) following the manufacturer's instructions. The DNA concentration and quality were determined using a Nanodrop ND-1000 Spectrophotometer (Thermo Scientific, USA). Equimolar amounts of the duplicate DNA samples were combined and stored at − 20 °C until quantitative polymerase chain reaction (qPCR) analysis and high-throughput sequencing.

3.2.4.2. Real-time quantitative PCR (qPCR)

The nitrification-related genes (amoA, encoding ammonia monooxygenase; nxrA, encoding nitrite oxidoreductase) were quantified in triplicate using a QuantStudio 3 Real-Time PCR system (Thermo Fisher Scientific, USA). The primers used for qPCR analysis are shown in Table 3.3. The reaction mixture consisted of 10 µL of 2 × GoTaq® Green Master Mix (Promega, USA), 1.0 µL of forward primers (4 pmol), 1.0 µL of reverse primers (4 pmol), 4 µL sample DNA, and distilled water added to 20 µL. The qPCR reaction conditions were as follows: (1) 94 °C, 3 min; (2) 94 °C, 30 s, (3) annealing temperature, 45 s, and (4) 72 °C for 60 s. Specificity was confirmed by analyzing the melting curves. External standard curves were prepared using serial dilutions of a known copy number of pGEM-T Easy vector (Promega, USA) containing the *amoA* gene of *Nitrosomonas europaea* (NBRC 14298) and the nxrA gene of *Nitrobacter winogradskyi* (NBRC 14297).

Table. 3.3. Primers for the *amoA* and *nxrA* genes

Target genes	Primer sequence (5'-3')	Amplicon size (bp)	Annealing temp (°C)	Reference
*amo*A	amoA-1F: GGGGTTTCTACTGGTGGT amoA-2R: CCCCTCKGSAAAGCCTTCTTC	491	61	Rotthauwe et al., 1997
*nxr*A	nxrA-1F: CAGACCGACGTGTGCGAAAG nxrA-2R: TCYACAAGGAACGGAAGGTC	322	58	Poly et al., 2008

3.2.4.3. High-throughput sequencing

The 16S rRNA library gene was prepared using the 2-step tailed PCR method. The 16S rRNA primer sets used were 515F/806R and 2ndF/2ndR for the 1st and 2nd PCR respectively. After the 1st PCR reaction, the product was purified using VAHTS DNA Clean Beads (Vazyme, China), which were then used for the 2nd PCR reaction. The concentration of the 1st and 2nd PCR products was determined using Synergy H1 (BioTek, USA) with a QuantiFluor ds DNA system (Promega, USA). Equimolar amounts of the PCR

products were pooled and sequenced using the Illumina MiSeq platform with 2 × 300 base paired reads by Bioengineering Lab Co., Ltd., Japan.

The barcodes and primers of the raw sequence data were trimmed, quality controlled, and then imported into QIIME2 (ver. 2023.7). Reads were de-noised, and representative sequences and amplicon sequence variants tables were generated using the DADA2 plug-in. Taxonomy assignment of representative sequences was performed using the feature-classifier plugin against SILVA (ver. 138) with a 99% similarity.

3.2.5. Physicochemical analysis

The moisture content of the compost was measured 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. The pH was measured using a pH meter (HM-40P; DKK-TOA, Japan) using an aqueous suspension of compost sample (1:10 w/v). The total C and N of the biochar and compost were determined using a CE440 Elemental Analyzer (Exeter Analytical, USA). NH_4^+–N were extracted with 2M KCl (1:10 w/v) and quantified using an Agilent 7100 CE system after steam distillation. NO_2^-–N, NO_3^-–N and water-extractable PO_4^-–P were extracted in distilled water (1:10 w/v) and determined using ion chromatography. Precipitated N was calculated by subtracting the NH_4^+ extracted using 2 M KCl from the nitrogen extracted using 2 M KCl with 0.5 M HCl (Jiang et al., 2016). Organic N was calculated by subtracting the NH_4^+–N and precipitated N from the total nitrogen determined by the Kjeldahl method. Total P, K, Ca, Mg, and Na were determined using an ICP-OES-5900 (Agilent, USA) after microwave digestion. The cation exchange capacity (CEC) was measured using the ammonium acetate (NH_4OAc) extraction method. The specific surface area of each material was determined using a surface area and pore size analyzer (Belsorp Mini II; MicrotracBEL, Japan).

3.2.6. Statistical analysis

A one-way ANOVA was performed to assess overall group differences, followed by the Dunnett test to compare the compost-only biofilter and biochar-mixed biofilters.

3.3. Results

3.3.1. Ammonia removal performance of compost biofilters

The performance of the biofilters to remove different concentrations of ammonia is shown in Figs. 3.2 and 3.3. All biofilters exhibited 100% removal efficiency, even at an NH_3 concentration of 2000 ppm, for the first 5 days. However, at 2000 ppm, the removal efficiency gradually decreased, with the compost-only biofilter dropping below 80% by day 7, and the compost-biochar mixtures by days 8 or 9 (Fig. 3.2). At 1000 ppm, all biofilters maintained 100% removal efficiency for a longer period compared to at 2000 ppm (Fig. 3.3). The average duration of biofiltration was 24 days for the compost-only biofilter, and 27, 28, and 27 days for B10, B25, and B50, respectively (Table 3.1).

Figure 3.4 shows the capacity of the biofilters for removing ammonia. At 1000 ppm it ranged from 2.07 to 2.67 g m^{-3}-biofilter, which is higher compared to at 2000 ppm, where the capacity ranged from 1.13 to 1.70 g m^{-3}-biofilter over 7 to 9 days. All biochar-mixed biofilters exhibited a higher removal capacity than the compost-only biofilter, with the 25% biochar-mixed biofilter (B25) showing the highest removal capacity at both concentrations of ammonia.

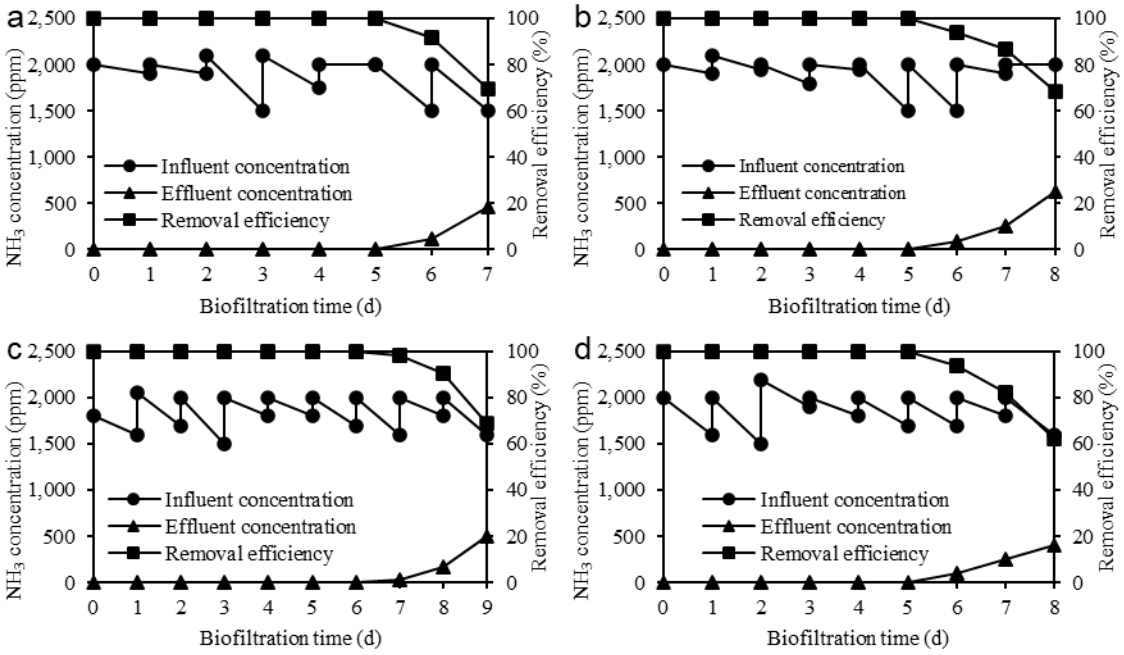

Fig. 3.2. NH_3 removal performance of (a) compost-only biofilter and (b–d) biochar-mixed biofilters: (b) 10% biochar, (c) 25% biochar and (d) 50% biochar at 2000 ppm (Trial 2 of 1st experiment)

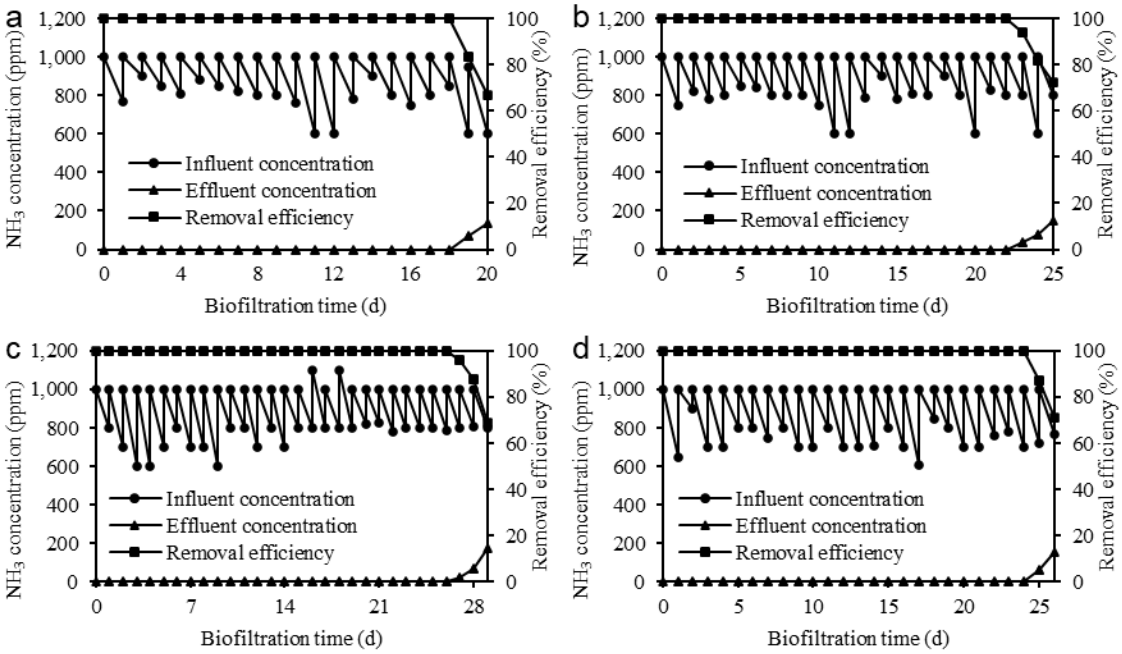

Fig. 3.3. NH_3 removal performance of (a) compost-only biofilter and (b–d) biochar-mixed biofilters: (b) 10% biochar, (c) 25% biochar and (d) 50% biochar at 1000 ppm (Trial 2 of 1st experiment)

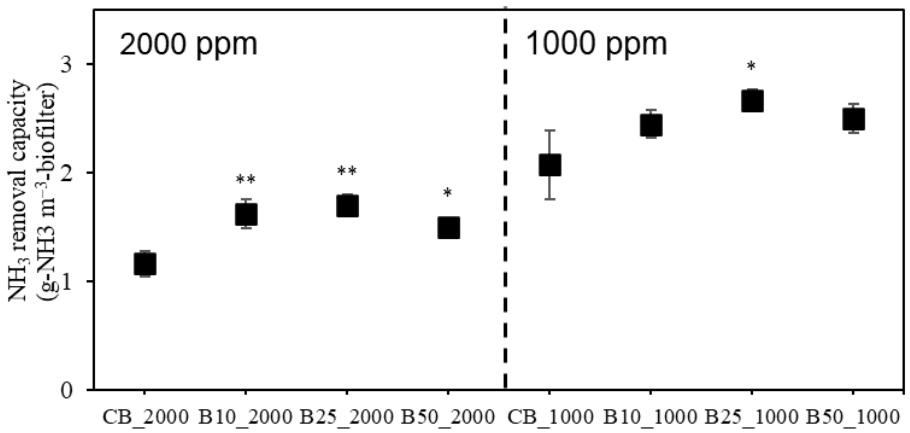

Fig. 3.4. NH_3 removal capacity of compost-only biofilter (CB) and biochar-mixed biofilters: B10 (10% biochar), B25 (25% biochar) and B50 (50% biochar) at 2000 and 1000 ppm (1st experiment). The results represent the mean of three replicates and error bars indicate the standard deviation. Asterisks indicate significant differences by the Dunnett test (*=$p < 0.05$, **=$p < 0.01$).

3.3.2. Nutrient dynamics during biofiltration

Figure 3.5 shows the change in the properties during biofiltration of the compost-only biofilters and of the biofilters mixed with 25% biochar. All the biofilters exhibited a huge increase in NH_4^+ content regardless of the influent NH_3 concentration during biofiltration (Fig. 3.5a). The biochar-mixed biofilter showed a significantly higher final NH_4^+ content than the compost-only biofilter at 2000 ppm ($p<0.05$), while no significant difference was observed at 1000 ppm. Under the 2000 ppm condition, the NH_4^+ content of the biochar-mixed biofilter was higher than that of the compost-only biofilter on day 7, whereas under the 1000 ppm condition, even on day 21 the biochar-mixed and compost-only biofilters had similar NH_4^+ levels. The NO_2^-–N content of both biofilters increased negligibly at 2000 ppm (Fig. 3.5b), whereas the NO_3^-–N content had slightly increased by the end of biofiltration (Fig. 3.5c). In contrast, at 1000 ppm, all biofilters showed a significant increase in NO_2^-–N and NO_3^-–N content, rising from 0.02 to 1.69 g-N kg^{-1}-dry biofilter and from 0.63 to 1.45 g-N kg^{-1}-dry biofilter in the compost-only biofilter, and from 0.03 to 0.79 g-N kg^{-1}-dry biofilter and from 0.36 to 2.07 g-N kg^{-1}-dry biofilter in the biochar-mixed biofilter, respectively, during biofiltration. All the biofilters showed a substantial increase in both NO_2^-–N and NO_3^-–N content by day 7. At 2000 ppm, there were no significant differences in NO_2^-–N and NO_3^-–N content between the compost-only and biochar-mixed biofilter, whereas at 1000 ppm the biochar-mixed biofilter exhibited a significantly lower NO_2^-–N content from day 14 onwards while showing a higher final NO_3^-–

N content on day 21 ($p<0.05$). The initial water-extractable phosphorus (WEP) content of the biofilters was significantly higher for the biochar-mixed biofilter than the compost-only biofilter ($p<0.05$) (Fig. 3.5d). All the biofilters showed a decrease in WEP after NH_3 filtration, ranging from 638.25 to 409.18 g-P kg^{-1}-dry biofilter for the compost-only biofilter and from 902.39 to 566.50 g-P kg^{-1}-dry biofilter for the biochar-mixed biofilter. The WEP in all biofilters slightly decreased by day 3, followed by an increase by day 7 under the 2000 ppm conditions, with a similar trend being observed at 1000 ppm, with the WEP decreasing gradually after day 7.

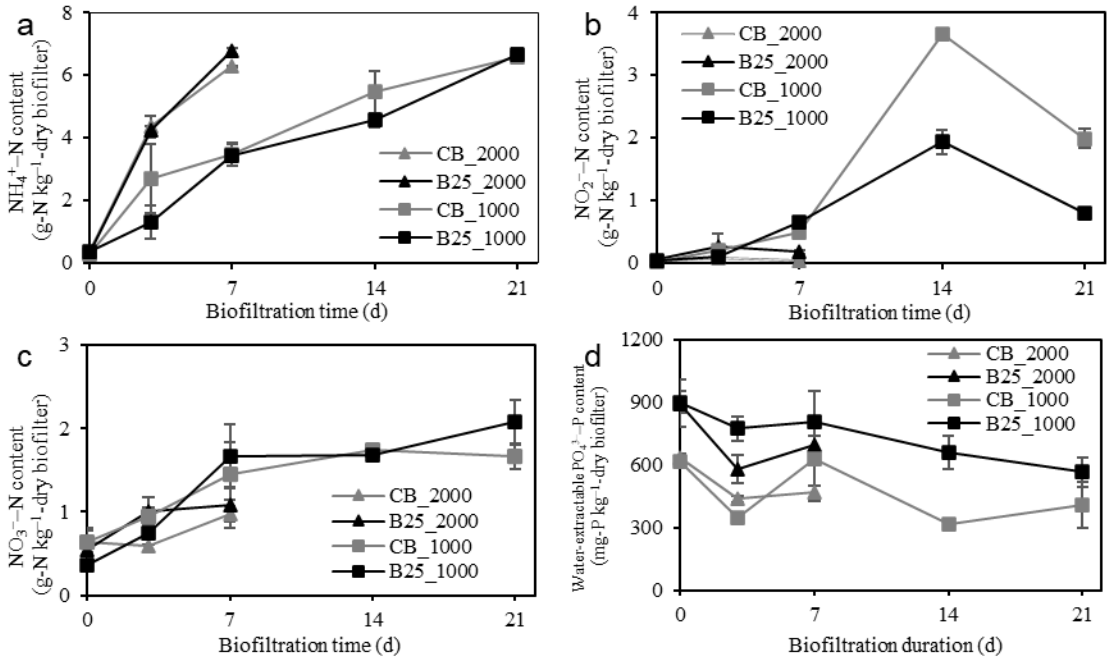

Fig. 3.5. Dynamics of (a) NH_4^+–N, (b) NO_2^-–N, (c) NO_3^-–N, (d) water-extractable PO_3^{4-}–P content in the compost-only biofilter (CB) and 25% biochar-mixed biofilter (B25) during biofiltration at 2000 and 1000 ppm (2nd experiment). The results represent the mean of two replicates and error bars indicate the standard deviation

3.3.3. Copy numbers of nitrifying genes

The changes in copy number of nitrification genes targeting the *amoA* and *nxrA* genes in the biofilters are presented in Figure 6. During the filtration period, the *amoA* and *nxrA* copy numbers for all the biofilters ranged from 2.0×10^8 to 2.6×10^9 copies g^{-1} dry biofilter and 8.3×10^6 to 5.2×10^7 copies g^{-1} dry biofilter, respectively. Under the 2000 ppm condition, the copy number of *amoA* did not significantly change during filtration for all biofilters, whereas the copy number of *nxrA* increased slightly but not in a statistically significant manner. In contrast, at 1000 ppm, both biofilters showed a notable increase in *amoA* and *nxrA* levels during biofiltration, peaking on day 7 and then gradually decreasing, except that *nxrA* levels in the biochar-mixed biofilter remained high up to day 21. Biochar did not have a significant effect on the levels of either gene under the 2000 ppm condition, whereas at 1000 ppm the copy numbers increased by nearly 61% and 57.3% for *amoA* and *nxrA*, respectively, compared to the compost-only biofilter on day 7.

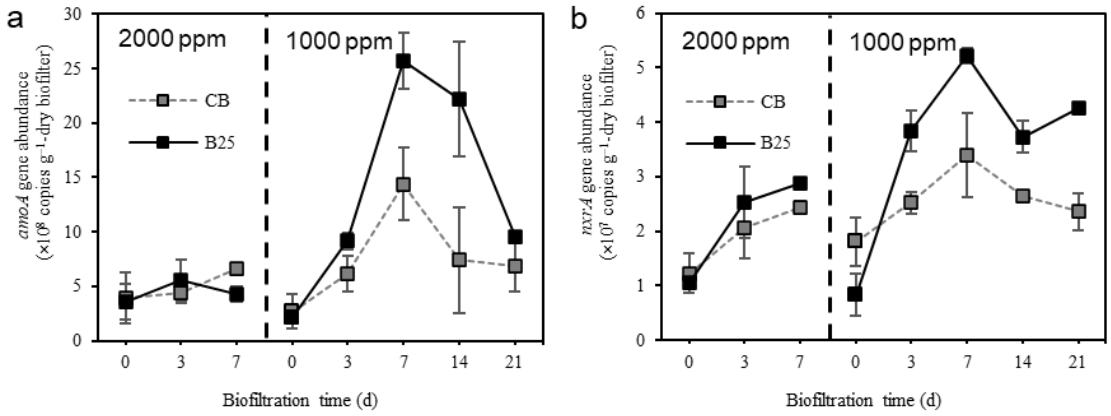

Fig. 3.6. Dynamics of (a) *amoA* and (b) *nxrA* gene copies in the compost-only biofilter (CB) and 25% biochar-mixed biofilter (B25) during biofiltration at 2000 and 1000 ppm (2nd experiment). The results represent the mean of two replicates and error bars indicate the standard deviation

3.3.4. Nitrogen mass balance after biofiltration

Figure 3.7 shows the nitrogen mass balance in the composite biofilters as a function of the supplied NH_3–N. Organic N comprised 14.2% of the total nitrogen in the compost-only biofilter and 11.9% of the total nitrogen in the 25% biochar-mixed biofilter, under the 2000 ppm condition. These percentages increased respectively to 17.2% and 15.4% under the 1000 ppm condition. NH_4^+–N was the dominant form of removed nitrogen in both biofilters, particularly at 2000 ppm, where it accounted for 59.8% of the total nitrogen in the compost-only biofilter and 66.5% in the biochar-mixed biofilter. This proportion decreased to 42.0% and 40.9%, respectively, under the 1000 ppm condition. The proportion of NO_2^-–N was negligible in both biofilters at 2000 ppm, but increased to 13.1% and 5.0% for the compost-only biofilter and biochar-mixed biofilter, respectively, at 1000 ppm. Similarly, both biofilters showed increased NO_3^-–N at 1000 ppm, reaching 11.2% and 6.9% for the compost-only and biochar-mixed biofilters, respectively, compared to 3.2% and 5.4% at 2000 ppm. In the compost-only and biochar-mixed biofilters at 2000 ppm, the proportion of precipitated nitrogen from the supplied nitrogen was similar, at 8.7% and 6.9%, respectively. In contrast, at 1000 ppm, the biochar-mixed biofilter showed a significantly higher proportion of precipitated nitrogen, 17.0%, compared to 9.2% in the compost-only biofilter. The fraction of unknown N (Unknown-N) was calculated by subtracting all the determined nitrogen forms from the supplied nitrogen. The unknown N form accounted for 7.6% and 7.5% at 2000 ppm in the compost-only and biochar-mixed biofilters, respectively, and increased slightly to 8.6% and 10.6% under the 1000 ppm condition.

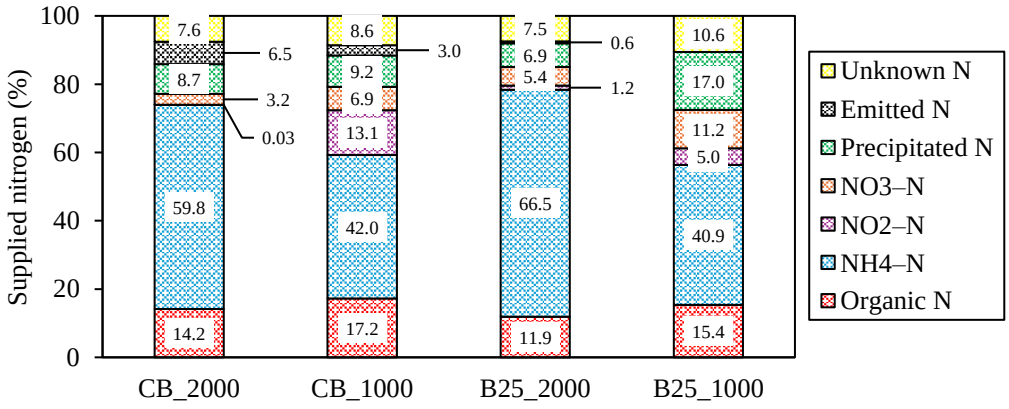

Fig. 3.7. Nitrogen mass balance in the compost-only biofilter (CB) and 25% biochar-mixed biofilter (B25) (2nd experiment). The values represent the mean of two replicates.

3.4. Discussion

Biochar improved the ammonia removal performance of compost biofilters provided with a feedstock containing 2000 or 1000 ppm ammonia, regardless of the mixing ratio of biochar to compost (Fig. 3.3). Here, we discuss how biochar improves the performance of compost biofilters for removing ammonia, focusing on the underlying mechanism.

3.4.1. NH_4^+ adsorption

We confirmed that all compost biofilters initially removed ammonia with 100% efficiency (Fig. 3.2 and 3), likely due to NH_4^+ adsorption onto the filter medium (Jun & Wenfeng, 2009). This is a major advantage of biofilters over other biological removal methods, such as bio-trickling filters. This adsorption increased the amount of NH_4^+ in the biofilter (Fig. 3.5a), potentially inhibiting ammonia emission from the biofilter and limiting the availability of ammonia to the nitrifying bacteria (Y. Song et al., 2014). NH_4^+ adsorption was the dominant mechanism by which the compost-only and biochar-mixed biofilters removed ammonia, regardless of the ammonia concentration (Fig. 3.7). The importance of adsorption is emphasized under the 2000 ppm condition because other ammonia transformation mechanisms are absent. Biochar could thus be effective in improving the removal of NH_3 by biofilters due to its high adsorption capacity. At the end of the biofiltration experiment, the biochar-mixed biofilter contained significantly more NH_4^+ than the compost-only biofilter at 2000 ppm (Fig. 3.5a), indicating that more ammonia was removed through adsorption when compost was mixed with biochar, perhaps because the biochar used in this study has a higher CEC compared to compost (Table 3.1). Oxygen-containing functional groups form on the biochar surface during biochar production (Banik et al., 2018), resulting in a high CEC, which enhances the ability of biochar to adsorb cations such as ammonium (Sumaraj et al., 2020). However, in contrast to the 2000 ppm condition, the biochar-mixed biofilter did not contain a higher level of NH_4^+ than the compost-only biofilter at 1000 ppm. Ammonia is simultaneously microbially oxidized and adsorbed onto the biofilter (Pagans et al., 2007), which can prevent ammonia from accumulating. More nitrification occurred under the 1000 ppm condition than the 2000 ppm condition, and nitrification was enhanced by biochar (Fig. 3.6; detailed in Section 3.4.2), suggesting that the biochar-mixed biofilter was not saturated with ammonia, even on day 21 at 1000 ppm.

3.4.2. Biological processes

Biological processes such as nitrification are essential for the sustained removal of ammonia by a biofilter. As we initially expected, the ammonia removal capacity of a biofilter was significantly lower at 2000 ppm than at 1000 ppm (Fig. 3.4), perhaps due to inhibition caused by ammonia toxicity towards nitrifying bacteria. The 2000 ppm condition resulted in no notable changes in the expression levels of nitrification-related genes (Fig. 3.6) or in the by-product of nitrification during biofiltration, even in biochar-mixed biofilters (Fig. 3.5b and 3.5c), indicating negligible nitrification during biofiltration. Under the 2000 ppm condition used in this study, the ammonia loading rate was 6.2–9.5 g-NH_3 m^{-3} h^{-1} (Table 3.2), suggesting this range is adequate to inhibit nitrification, even in the presence of biochar. In contrast, Jun & Wenfeng (2009) reported that compost biofilters could remove ammonia through nitrification with nearly 99% efficiency at a maximum loading rate of 12.1 g-NH_3 m^{-3} h^{-1}, which is higher than the loading rate observed in the present study at 2000 ppm. These conflicting results may be due to the gradual increase in ammonia concentration in their study, from 20 to 60 ppm for 60 days, corresponding to 1.0 to 12.1 g-NH_3 m^{-3} h^{-1}, whereas we consistently supplied ammonia at a high loading rate. Their gradual supply of ammonia might have functioned similar to that of inoculation (Y. X. Chen et al., 2005), ensuring the growth of nitrifying bacteria and thus of a stable removal process (Villaverde, 2000). Gradually increasing the ammonia concentration or pre-supplying a low concentration might be necessary to further maximize the effect of biochar on the removal capacity of a biofilter at high ammonia concentrations.

In contrast, at 1000 ppm the ammonia loading rate was 2.5–5.1 g-NH_3 m^{-3} h^{-1}, which may represent less challenging conditions for nitrifying bacteria, allowing for more efficient ammonia removal than at 2000 ppm. The biochar-mixed biofilter showed higher *amoA* and *nxrA* gene levels than the compost-only biofilter at 1000 ppm, suggesting that both nitritation and nitratation were enhanced by the addition of biochar, perhaps due to the higher specific surface area (SSA) of biochar compared to compost (Table 3.1), providing favorable sites for nitrifying bacteria to establish colonies (Y. Song et al., 2014). Consistent with the higher *nxrA* gene levels in biochar-mixed biofilter at 1000 ppm compared to the compost-only biofilter, the biochar-mixed biofilter removed more ammonia in the form of NO_3^- (Fig. 3.7). On the other hand, the compost-only biofilter removed more ammonia as NO_2^- through nitritation, probably because nitrite accumulated in the compost-only biofilter due to the selective inhibition of nitrite-oxidizing bacteria (NOB)

by free ammonia (Z. Wang et al., 2020) which was mitigated by biochar (Meng et al., 2022). We conducted a next-generation sequencing analysis to examine the changes in the microbial communities in the biofilters during biofiltration (Fig. 3.2). Both biofilters contained representative ammonia-oxidizers, including *Nitrocosmicus*, *Nitrososphaera*, *Nitrosospira* and *Nitrosomonas* (Soliman & Eldyasti, 2018), with more noticeable changes in the relative abundance of ammonia-oxidizers in the biofilters during biofiltration at 1000 ppm compared to at 2000 ppm. The rate of increase of *Nitrosomonas* was higher than that of other ammonia-oxidizing bacteria in both biofilters, indicating that *Nitrosomonas* likely plays a key role in ammonia oxidization during biofiltration. A similar result was reported by Posmanik et al. (2014), who observed that *Nitrosomonas* was the dominant ammonia-oxidizing bacteria (AOB) in compost biofilters exposed to high NH_3 loads. They suggested that *Nitrosomonas* outcompetes other AOB for survival under high ammonia concentrations (Lin et al., 2017), a trend that might be enhanced by biochar, which provides additional surface area for colonization. In contrast, NOB was not observed in either biofilter, despite nitratation occurring during biofiltration, perhaps because heterotrophic NOB were more dominant than autotrophic NOB. Tsang et al. (2015) reported that the abundance of NOB significantly decreased in their bio-trickling filter when the ammonia loading was increased from 78 g m^{-3} h^{-1} to 140 g m^{-3} h^{-1}, while the heterotrophic bacteria were less affected. Well-known NOB are typically autotrophic and are therefore highly sensitive to ammonia toxicity (Kouba et al., 2014), whereas heterotrophic bacteria are more resistant (Bae et al., 2015). Accordingly, heterotrophic NOB might dominate biofilters under high ammonia conditions. However, further research is required to better understand NOB, and especially heterotrophic NOB.

We observed no leaching from the biofilters, suggesting that some of the unknown N in the nitrogen mass balance may be due to denitrification (Fig. 3.7). Previous studies indicated that compost biofilters release some of the removed ammonia as N_2O, ranging from a minimum of 1.9% to a maximum of 19% (Maia et al., 2012; Shang et al., 2022; Yasuda et al., 2022). This is consistent with the range of 7.5 to 10.6% observed in the present study. Our results indicated that denitrification could have occurred in biofilters both at 1000 ppm and at 2000 ppm (Fig. 3.7). Denitrification is closely linked to nitrification, since nitrification increases the availability of nitrate, the primary substrate for denitrification. According to Kong et al. (2020), nitrification is the dominant pathway for N_2O generation in ammonia removal biofilters. In

the present study, a higher amount of unknown N was present at 1000 ppm (1284.6 and 1632.6 mg-N kg^{-1}-dry biofilter in the compost-only and biochar-mixed biofilter) compared to 761.2 and 725.3 mg-N kg^{-1}-dry biofilter at 2000 ppm, respectively, possibly due to enhanced nitrification. For the same reason, the biochar-mixed biofilter sustained a significantly higher amount of unknown N than the compost-only biofilter at 1000 ppm ($p<0.05$). In addition, biochar could further support heterotrophic denitrification by promoting organic matter decomposition and accelerating electron transfer processes within the biofilter (Guo et al., 2022).

3.4.3. Chemical precipitation

We observed that the WEP for all biofilters decreased during biofiltration (Fig. 3.5d), possibly due to chemical precipitation of NH_4^+ with PO_4^{3-}. Generally, PO_4^{3-} can combine with NH_4^+ and Mg^{2+} or Ca^+ in equimolar ratios to generate $MgNH_4PO_4$ or $CaNH_4PO_4$ complexes under high pH conditions (Witter & Kirchmann, 1989), potentially preserving the NH_4^+ in the biofilter in crystalline form. We also confirmed that the amount of NH_4^+ in all the biofilters increased when the ammonia was extracted with both KCl and HCl compared to extraction using KCl alone, regardless of the ammonia concentration (Table S1), again indicating that ammonia removal by the biofilters occurs at least in part through precipitation (X. Wang et al., 2016). Ammonia precipitation in the compost biofilter is not surprising, given that this system is highly conducive to ammonia precipitation due to the high pH and abundance of nutrient ions such as PO_4^{3-}, K^+, Ca^{2+}, and Mg^{2+} (T. Zhang et al., 2009). Similarly, the biochar-mixed biofilter showed more precipitated N than the compost-only biofilter at 1000 ppm due to its higher ion content and pH compared to compost (Table 3.1), aiding the ammonia removal performance of the biofilter. However, biochar did not improve chemical precipitation at 2000 ppm, probably because the nutrient ions are released over time from the compost or biochar and then combine with ammonia. At 2000 ppm, unlike at 1000 ppm, there was probably insufficient time for this process to occur.

3.4.4. Limitation and future work

The typical EBRT ranges from 60 to 180 s (Kennes & Veiga, 2001), which is noticeably lower than the conditions used in this study. A lower EBRT is generally recommended to prevent the buildup of pollutants in biofilters. Practical limitations prevented us from using a typical EBRT in this study. In the future, it will be important to investigate the effect of biochar at lower EBRT values because the ammonia removal

efficiency of a biofilter is influenced by the EBRT, even at the same ammonia loading rate (Taghipour et al., 2008).

Biochar improved the ammonia removal performance of the biofilter under the 2000 ppm ammonia condition but its effect on nitrification during biofiltration remained limited in this study. Further research is thus required to ensure effective nitrification for sustained ammonia removal at high ammonia concentrations. As discussed in section 3.4.2, the stepwise introduction of low levels of ammonia could help increase the resistance of nitrifying bacteria to ammonia toxicity. Reducing ammonia emissions from the composting process would also enhance the impact of biochar at high ammonia concentrations.

3.5. Conclusion

This study demonstrated that biochar can significantly enhance the capacity of compost biofilters to remove ammonia. We observed optimum performance at a 25% mixing ratio of biochar to compost. Biochar's high adsorption capacity, large specific surface area, and nutrient content contribute to improved NH_4^+ adsorption, which aids biological processes and enhances chemical precipitation during biofiltration. However, the effectiveness of biochar decreased at higher ammonia concentrations. These findings suggest that mixing biochar with compost is a promising strategy for improving the ammonia-removal performance of compost biofilters. Further research is needed to develop strategies to maintain the biological processes that occur in biofilters, and especially at high ammonia concentrations, to support sustained ammonia removal from compost biofilters.

Chapter 4 Combining system of co-composting and co-biofiltration with biochar for continuous zero ammonia emission during composting

4.1. Introduction

In Chapter 3, the following key findings were confirmed 1) Compost biofilters achieved stable and complete ammonia removal when exposed to lower NH_3 concentrations (≤ 1000ppm). 2) The addition of biochar (co-biofiltration) further enhanced the ammonia removal performance of the biofilter. 3) However, nitrification in the biofilter was inhibited even with biochar addition when NH_3 concentrations were elevated (~2000 ppm). These findings emphasize that fundamentally reducing the ammonia concentration entering the biofilter is a critical factor for ensuring its long-term sustainability and effective ammonia removal performance

Regarding ammonia reduction during composting, we demonstrated in Chapter 2 that biochar application in the composting process (co-composting) can significantly reduce ammonia emissions. Based on these results, we propose that integrating co-composting with co-biofiltration has the potential to achieve sustainable ammonia management.

Therefore, we hypothesize that 1) biochar application in the composting process (co-composting) will enable biofilters to completely remove ammonia emissions during composting. and 2) mixing biochar with biofilter (co-biofiltration) will further enhance the ammonia removal efficiency of biofilter.

4.2. Material and Methods

4.2.1. Material

Fresh dairy manure and compost were sourced from the Hosozawa dairy farm in Hokkaido, Japan. The dairy manure was used both as the composting material and as the feedstock for biochar production. The biochar was produced under the same conditions detailed in Chapter 3. Dairy manure compost served as the filter medium in the biofilter system. The properties of each material are summarized in Table 4.1.

Table. 4.1. The properties of dairy manure, compost and biochar

	Manure	Compost	Biochar
pH	8.4	8.4	9.2
C (%)	41.8	41.5	52.4
N (%)	1.9	2.68	2.76
Ash (%)	18.0	22.7	24.0

4.2.2. Experimental setup

The experimental setup for the combined system is illustrated in Figure 4.1. Approximately 3 kg of wet dairy manure (80% wet basis) was mixed with 150 g of biochar for the composting process, corresponding to a biochar-to-dairy manure mass ratio of 1:4 on a dry basis. The composting process lasted for 14 days. Further details on the composting process are described in Chapter 2.

For the biofiltration, 300 g of filter media consisting of either compost or a compost-biochar mixture was used. The compost-biochar mixture was prepared by mixing compost with biochar in a 3:1 ratio (dry basis), equivalent to 33.75 g of biochar. Further details on the biofiltration system are described in Chapter 3.

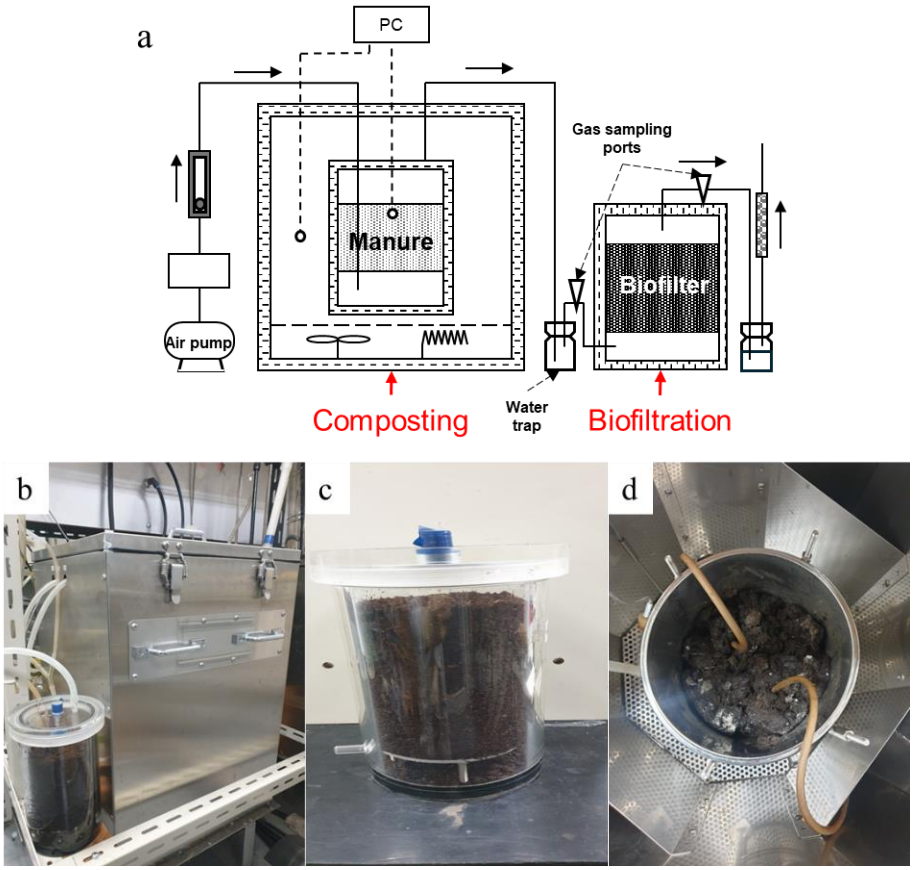

Fig. 4.1. Experimental setup of the combined system. (a) schematic diagram of the combined system, (b) the external view of the system, (c) the biofilter reactor containing compost, and (d) the interior of the composting reactor with dairy manure

4.2.3. Determination of emitted NH_3 concentrations during composting

The concentration of NH_3 in the liquid phase was determined using a water trap by back titration with 0.05 M or 0.005 M sulfuric acid. The concentration of NH_3 in the gaseous phase was measured at the sampling port using a gas detector tube (Kitagawa, Japan) with a gas aspiration pump (Kitagawa, Japan).

4.2.4. Assessment of biofilter performance

The performance of the biofilter, including the inlet loading rate, removal efficiency, and total removal capacity, was evaluated as described in Chapter 3.

4.2.5. Physicochemical analysis

The moisture content of the manure and compost was determined by drying the samples at 105 °C for 24 h. The ash content of each material 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. The pH was measured using a pH meter (HM-40P; DKK-TOA, Japan). The total C and N were determined using a CE440 Elemental Analyzer (Exeter Analytical, USA). The NH_4^+ and NO_3^- contents were determined by extracting manure or compost with 2M KCl and distilled water, respectively. The KCl-extracted samples were further distilled to eliminate interference from K before NH_4^+ determination. NH_4^+ and NO_3^- contents were then quantified using an Agilent 7100 capillary electrophoresis system.

4.3. Result

4.3.1. Composting process

4.3.1.1. Variation in temperature and pH in the composting pile during composting

Figure 4.2 shows the temperature, pH, and moisture content changes during composting. All composting piles exhibited a rapid increase in temperature, reaching their peak within two days, regardless of biochar addition (Fig. 4.2a). The initial pH in the biochar-amended pile was 8.96, higher than the manure-only pile at 8.44 (Fig. 4.2b). The pH of the manure-only pile showed a slight increase by the end of composting, while that of the biochar-amended pile remained nearly unchanged. The initial and final moisture content of the manure-only pile was 79.8% and 64.4%, respectively, while in the biochar-amended pile, it was reduced from 69.1% to 53.4%.

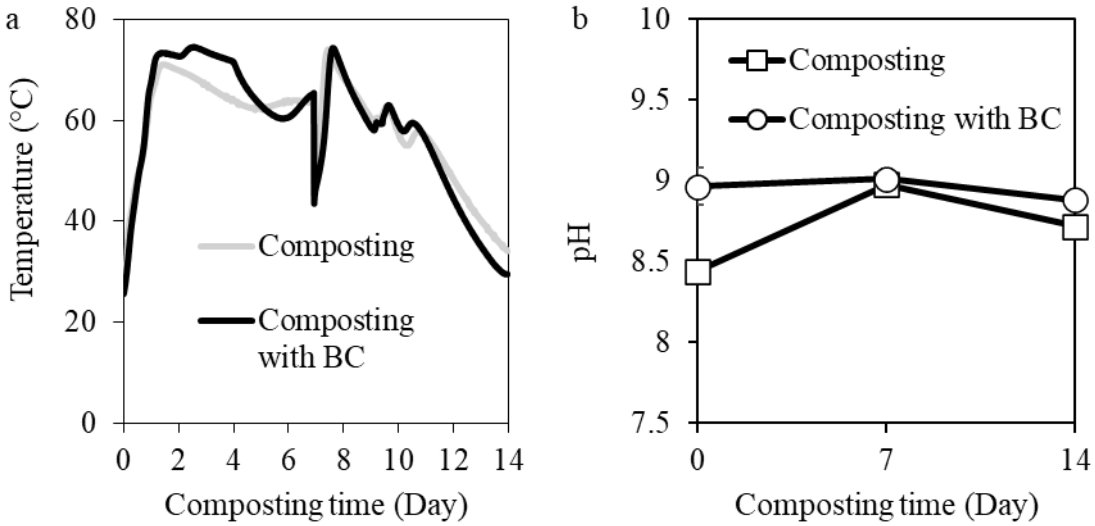

Fig. 4.2. Variation in (a) temperature and (b) pH of the composting piles during composting

4.3.1.2. Variation in NH_4^+ and NO_3^- content in the composting pile during composting

The initial NH_4^+ content in the biochar-amended composting pile was significantly lower than in the manure-only pile (Fig. 4.3a). However, both piles showed substantial declines in $NH4_4^+$ content during composting, resulting in negligible differences in final NH4+ content between the manure-only and biochar-amended piles. Conversely, NO_3^- content in both composting piles increased during composting (Fig. 4.3b). The manure-only pile exhibited a more pronounced increase in NO_3^- content by day 7, although this difference became negligible by the end of the process.

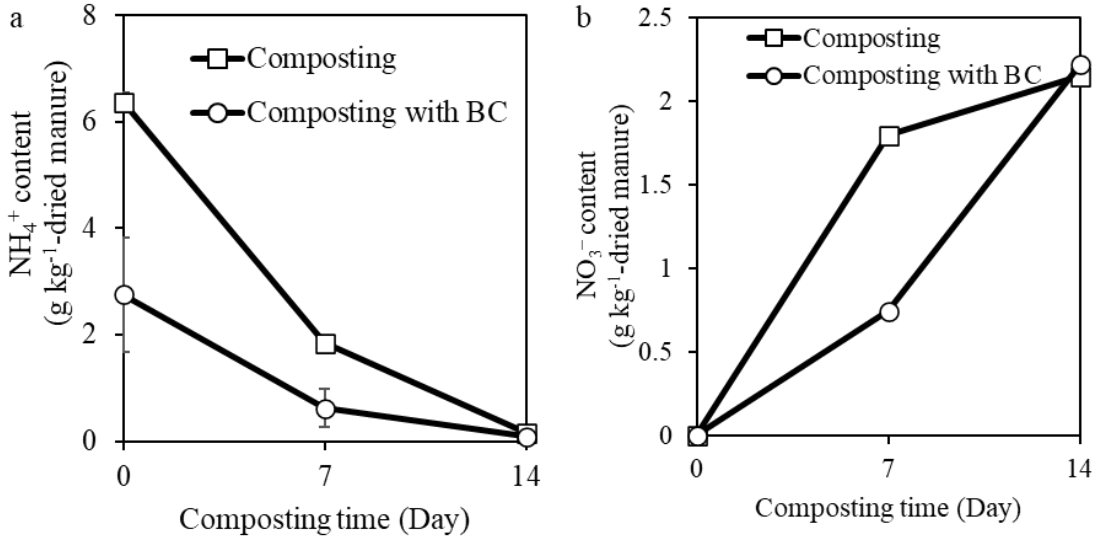

Fig. 4.3. Variation in (a) NH_4^+ and (b) NO_3^- contents in the composting piles during composting

4.3.2.3. The daily and cumulative ammonia NH_3 emissions during composting

For manure-only composting, the highest NH_3 emissions were observed on day 3 (Fig. 4.4a), while for composting with biochar, the peak occurred on day 2 (Fig. 4.4b). In both conditions, emissions gradually decreased over time after reaching the peak, with a secondary peak observed after the turning process on day 7. The cumulative ammonia emissions during manure-only composting were 6353.5 mg kg^{-1} $manure^{-1}$-AFS in the liquid phase and 922.8 mg kg^{-1} $manure^{-1}$-AFS in the gaseous phase (Fig. 4.4c). Adding biochar significantly reduced these emissions to 2741.8 mg kg^{-1} $manure^{-1}$-AFS in the liquid phase and 361.5 mg kg^{-1} $manure^{-1}$-AFS in the gaseous phase, representing reductions of 56.8 and 60.8 %, respectively. Ammonia emissions in the liquid phase were consistently higher than those in the gaseous phase throughout the composting process.

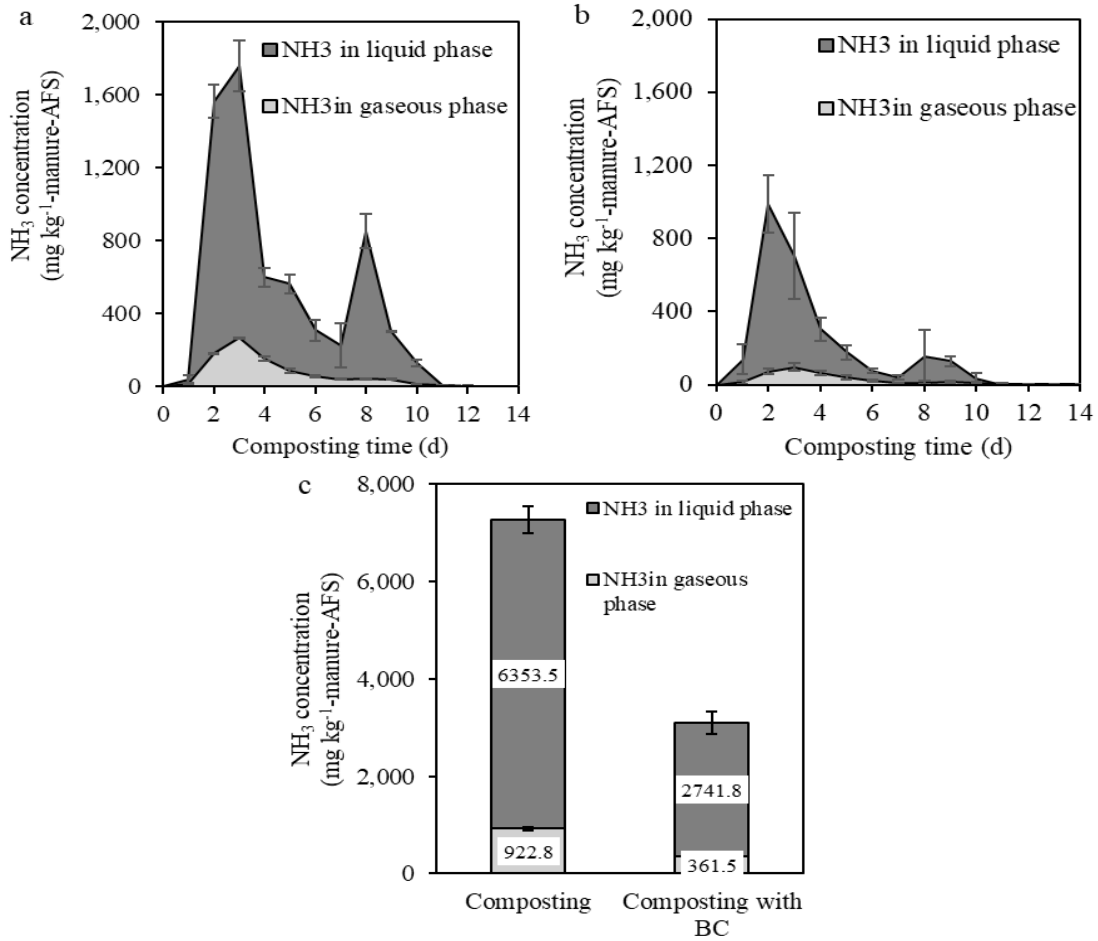

Fig. 4.4. Daily ammonia emissions during (a) manure-only composting, (b) composting with biochar, and (c) cumulative ammonia emissions during both composting processes

4.3.2. Biofiltration

4.3.2.1. NH_3 removal performance of biofilters

Figure 4.5 illustrates the NH_3 removal performance of the biofilters during composting. During manure-only composting, both biofilters achieved 100% removal efficiency in the early stages (Fig. 4.5a). However, on day 6, the efficiency started to decline and eventually fell below zero. An efficiency of 0 or lower means that more ammonia is being emitted from the filter than is being introduced. In contrast, during biochar-amended composting, both biofilters consistently maintained 100% removal efficiency for the first 14 days (Fig. 4.5b). To evaluate the reuse potential of the biofilters, the biofiltration experiment with biochar-amended composting was repeated multiple times using the same biofilters. During the second reuse, both biofilters nearly completely removed the supplied ammonia. However, during the third reuse, the removal efficiency of the compost-only biofilter started to decline on day 4, whereas the biochar-mixed biofilter sustained a stable 100% removal efficiency until the end of the composting process.

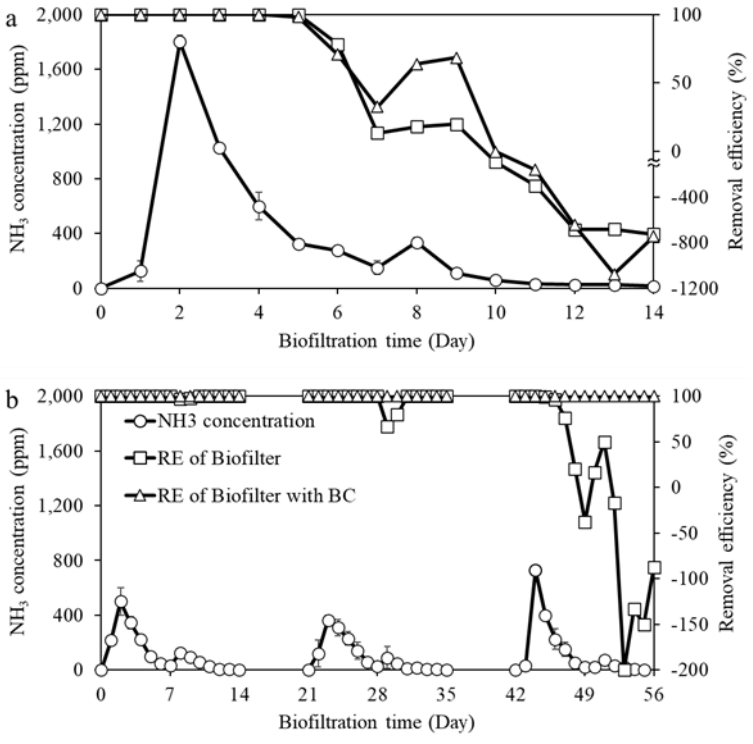

Fig. 4.5. Ammonia removal performance of biofilters during (a) manure-only composting and (b) composting with biochar.

4.3.2.2. Ammonia removal capacity of biofilters

The total ammonia removal capacity of the biofilters during manure-only composting was 1.12 g-NH_3 m^{-3}-biofilter for the compost-only biofilter and 1.42 g-NH_3 m^{-3}-biofilter for the biochar-mixed biofilter (Fig. 4.6). In biochar-amended composting, the removal capacities increased to 2.27 g-NH_3 m^{-3}-biofilter and 2.96 g-NH_3 m^{-3}-biofilter for the compost-only and biochar-mixed biofilters, respectively.

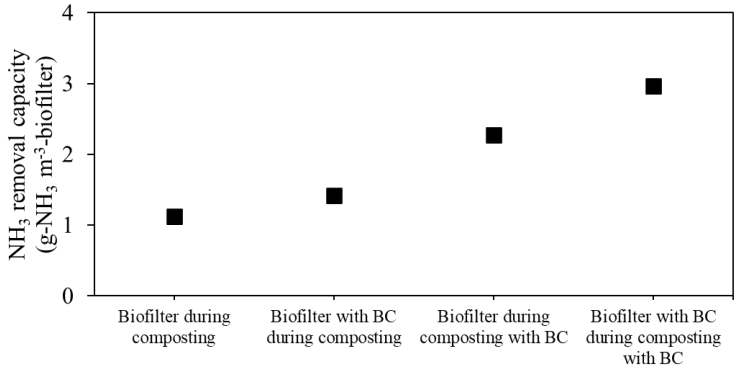

Fig. 4.6. Ammonia removal capacity of biofilters during composting with and without biochar (BC).

4.3.2.3. Variation in NH_4^+ and NO_3^- content in biofilters

Figure 4.7 illustrates the variation in NH_4^+ and NO_3^- content in biofilters during the biofiltration process. Both biofilters showed an increase in NH_4^+ content during manure-only composting. Similarly, both biofilters showed increased NH_4^+ content during the first 14 days of biochar-amended composting. However, biochar-mixed biofilter showed significantly lower NH_4^+ content than compost-only biofilter, which gradually increased throughout the biofiltration. The NO_3^- content in both biofilters increased significantly during the first co-composting phase and continued to rise throughout the co-composting process. In contrast, the increase in NO_3^- content was negligible during manure-only composting.

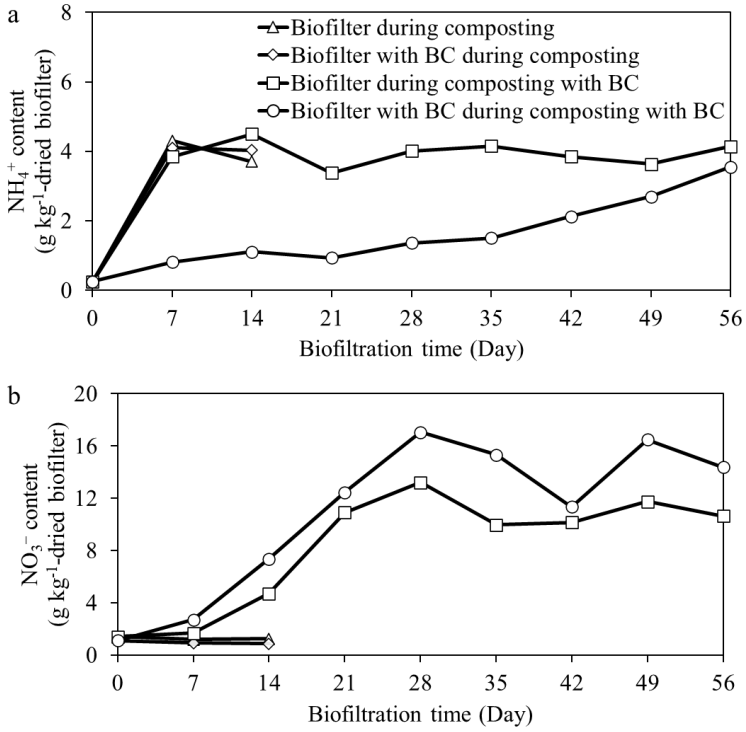

Fig. 4.7. Variation in (a) NH_4^+ and (b) NO_3^- content in the biofilters during composting.

4.4. Discussion

4.4.1. NH_3 emissions during composting

Biochar reduced ammonia emissions during composting by 56.9% in the aqueous phase and 60.8% in the gaseous phase (Fig. 4.4c). Previous studies have demonstrated that incorporating biochar into compost piles can reduce ammonia emissions by improving ventilation and providing microbial habitat, which enhances overall microbial activity including nitrification (J. Liu et al., 2024; W. Liu et al., 2017; López-Cano et al., 2016; Zhou et al., 2021). In this study, the initial moisture content of raw dairy manure was about 80%, which decreased to around 70% with the addition of biochar. This reduction in moisture likely improved micro-aeration, supporting microbial activity, including nitrification, during composting (M. H. Zainudin et al., 2020). However, despite the significant reduction in ammonia emissions, the biochar-amended compost pile did not exhibit higher NO_3^- levels compared to the manure-only pile throughout the composting period (Fig. 4.3b). This may be due to the concurrent enhancement of denitrification, which converts nitrate to nitrogen gas, thereby reducing NO_3^- accumulation (Agyarko-Mintah, Cowie, Van Zwieten, et al., 2017). Recent papers reported that biochar can increase the abundance of both nitrifying and denitrifying genes during composting (J. Liu et al., 2024; Zhou et al., 2021). Furthermore, Li et al. (2022) demonstrated biochar addition could improve the relative abundance of aerobic denitrifying bacteria such as *Pusillimonas* during composting. Accordingly, biochar may have promoted denitrification, further contributing to the ammonia reduction during composting. However, this study does not provide direct data on denitrification, highlighting the need for further analysis.

4.4.2. NH_3 removal performance of biofilters

The biofiltration system failed prematurely during manure-only composting (Fig. 4.5a), consistent with a previous study that reported a decline in biofilter efficiency from 89.5% to 46.7% by day 4 during animal by-product composting under a peak ammonia loading rate of 67.1 g m^{-3} h^{-1} (Pagans et al., 2005). This premature failure is likely due to the inhibitory effects of high ammonia toxicity on nitrification (Niu et al., 2023). In our previous study, nitrification in compost biofilters was inhibited at ammonia loading rates of 6.2 to 9.2 g m^{-3} h^{-1}, leading to early biofilter failure. Similarly, in this study, ammonia loading rates peaked at 21.16 g m^{-3} h^{-1} by day 3 of manure-only composting—well above the expected threshold for nitrification inhibition. The absence of a measurable increase in NO_3^- content further confirms that nitrification was

suppressed in the biofilter during manure-only composting. However, when biochar was added during the composting, ammonia loading rates were significantly reduced to a range of 1.7 to 6.4 g m^{-3} h^{-1}, this reduction enabled substantial NH_3 conversion to NO_3^- via nitrification in both biofilters, achieving near-complete ammonia removal even during their second reuse (Fig. 4.5b). By the third reuse, the biochar-mixed biofilter maintained complete ammonia removal, whereas the compost-only biofilter began emitting ammonia, highlighting biochar's role in enhancing biofilter reusability. In our previous study, the biochar-amended biofilter did not show higher NH_4^+ levels, even though biochar can improve NH_4^+ adsorption when nitrification is active in the biofilter. This suggests that active nitrification prevented NH_4^+ accumulation in the biofilter. This trend became even more apparent in this study, with lower NH_4^+ accumulation and consistently higher NO_3^- levels in the biochar-amended biofilter compared to the compost-only biofilter (Fig. 4.7a and b). These findings suggest that biochar improves nitrification during biochar-amended composting. In addition to nitrification, the fluctuating trend of NO_3^- content throughout the biofiltration period indicates that denitrification, as observed in our previous study, may have contributed to ammonia removal. Since biochar could influence denitrification, further research is needed to clarify its role and evaluate its potential to enhance overall ammonia removal efficiency.

4.5. Conclusion

This study demonstrates the synergistic effects of co-composting and co-biofiltration with biochar in achieving sustainable ammonia management during composting. Adding biochar to the composting process effectively reduced ammonia emissions, enabling the biofilter to remove ammonia continuously. Furthermore, incorporating biochar into the biofilter enhanced its reusability by sustaining high removal efficiency over multiple uses. These findings highlight biochar's dual role in reducing ammonia emissions during composting and improving biofilter performance, offering a practical and sustainable solution for ammonia management in agricultural waste treatment systems.

Chapter 5 General discussion and Future research

5.1. General Discussion

In this study, we successfully achieved zero ammonia emissions during manure composting by integrating biochar with conventional ammonia removal methods. This section focuses on strategies to enhance the effectiveness of biochar application and explores its potential uses in agricultural practices.

5.1.1. Improving the impact of biochar

We demonstrated in Chapter 2 that the impact of biochar on ammonia removal varies significantly depending on its physicochemical properties. This highlights the potential for enhancing ammonia removal efficiency by producing biochar with tailored properties for specific applications. In this regard, biochar modification represents a promising strategy for enhancing ammonia removal efficiency in composting and biofiltration systems. A variety of modification techniques have been developed to selectively improve biochar properties, such as specific surface area, functionality, and pH, thereby boosting its ammonia adsorption capacity (Chen et al., 2021; Suliman et al., 2016; Xie et al., 2022). A review by Zhang et al. (2020) reported that the average ammonium adsorption capacity of modified biochar was 22.79 mg N g^{-1}, significantly higher than the 11.19 mg N g^{-1} observed in unmodified biochar. These advancements highlight the potential for incorporating modified biochar into composting systems and biofilters to achieve greater efficiency in ammonia management. In practical application, Hestrin et al. (2020) demonstrated that oxidized biochar reduced ammonia emissions by 64.4% compared to unmodified biochar, primarily due to its enhanced adsorption capacity. However, not all modification techniques yield positive results. For example, Cao et al. (2024) reported that NaOH-modified biochar did not improve even though the content of oxygen-containing functional groups was increased. This discrepancy may be due to alkali modification raising the biochar's pH, which can inhibit the activity of nitrifying and denitrifying bacteria, thereby hindering further ammonia oxidation and reduction. These findings highlight the significant potential of biochar modification to enhance its effectiveness, while also emphasizing the need to optimize modification techniques to maximize its impact.

Achieving zero ammonia emissions during composting starts with reducing ammonia emissions, and modifying biochar presents a promising strategy for this. However, ammonia emissions can also be minimized by adjusting composting conditions such as pH, C/N ratio, and aeration rate (Bernal et al., 2009), which can, in turn, influence biochar’s effectiveness. For example, Chowdhury et al. (2014) found that

higher aeration rates resulted in increased ammonia emissions during composting, but the impact of biochar on reducing ammonia was more pronounced under a higher aeration rate than a lower aeration rate. Similarly, the initial moisture content of composting material plays a critical role in microbial activity, which in turn can influence biochar's performance during composting (Li et al., 2021). Therefore, to maximize biochar's potential for ammonia mitigation, it is also crucial to comprehensively optimize both the production of biochar and the composting conditions.

5.1.2. Potential applications of biochar

Ammonia is a crucial compound in various industrial and environmental processes, but its management presents significant challenges. Depending on how it is treated, ammonia can either act as a harmful pollutant with adverse effects on ecosystems and human health or as a valuable resource with diverse applications. Therefore, developing sustainable and efficient methods for ammonia management has become important across multiple fields (Han et al., 2021). In this context, biochar may prove to be a promising solution. In wastewater treatment, for example, conventional ammonia removal techniques often rely heavily on chemical agents to achieve high removal efficiencies (Ulu and Kobya, 2020). However, the extensive use of chemicals not only raises operational costs but also introduces secondary environmental burdens, such as chemical waste and unintended ecological consequences (Farghali et al., 2024). In contrast, biochar can offer significant advantages in two ways: 1) Direct application: The direct application of biochar to wastewater enables the removal of ammonia, primarily through adsorption (Sun et al., 2021). Given the similarities between the removal mechanisms of biochar in wastewater and those in composting, it can be anticipated that our findings on the optimal biochar production conditions for ammonia reduction in the composting process will also contribute to optimizing its effectiveness in wastewater treatment. Furthermore, in addition to ammonia, biochar offers the added benefit of removing other pollutants, such as phosphorus, heavy metals, and biological contaminants (Xiang et al., 2020), making it a valuable tool for improving overall water quality. 2) Biochar-mixed biofilters: As demonstrated in our studies, biochar-mixed biofilters not only achieve high ammonia removal efficiency but also offer long-term sustainability. The effectiveness of biochar-mixed biofilters can reduce the need for frequent replacements or maintenance, a common limitation of conventional systems. This durability and reduced reliance on chemical agents can lower operational costs and minimize environmental impacts. However, to remove ammonia from

wastewater using biofilters, it is necessary to volatilize ammonia through heat and pH adjustments (Luqmani et al., 2024). This step could introduce additional costs compared to the direct application of biochar. Despite this, the combination of high removal performance, environmental safety, and economic viability can make biochar-mixed biofilters a practical and sustainable option for ammonia management in wastewater treatment.

Given the potential applications of compost in biofilters, it is also important to consider its role as a fertilizer. Although compost is widely valued for its fertilizing properties, the low availability of nutrients is a challenge (E. et al., 2024). This can lead to large application rates, which in turn might cause the accumulation of excessive salt in the soil and an increase in costs (Sikora, 1998). In this context, compost biofilters can offer a significant advantage by capturing and utilizing ammonia as a nitrogen source, thereby enhancing the fertilizer value of the compost. In Chapter 4, we confirmed a notable increase in total nitrogen levels in the compost after biofiltration. The increased nitrogen was mainly in the form of ammonium and nitrate, which are more readily available to plants (Messiga et al., 2020). This indicates the possibility of more efficient nutrient release, which could enhance the effectiveness of the biofilter as a fertilizer. However, excessive ammonia retained in the biofilter can be desorbed, potentially harming soil biota, or released into the atmosphere (Savci, 2012). Interestingly, adding biochar into the biofilter could help address some of these concerns. We found that biochar-mixed biofilters accumulated less ammonia than those with compost-only biofilters while significantly boosting the nitrate content. This shift makes nitrogen more plant-available and less toxic (Kim et al., 2019), offering a safer and more efficient fertilizer. Additionally, previous studies demonstrated that co-application of compost with biochar is more beneficial for soil fertility, soil microbial diversity, and plant growth than compost or biochar alone (Mikajlo et al., 2024; Wu et al., 2016). Accordingly, biochar addition in composting and biofilter can not only reduce ammonia sustainably but also improve the value of compost as a soil amendment. However, to date, no studies have fully explored the efficiency or safety of these systems in soil applications. Further research is needed to evaluate their full potential, particularly in balancing their benefits with any potential environmental risks.

5.2. Suggestions for future studies

Although we have achieved zero ammonia emissions during composting, we are still in the early stages of exploring how these findings can be applied to sustainable ammonia management in practice. Further

studies are necessary to optimize the system's impact and assess potential environmental risks before it can be recommended for field implementation. Key areas for future research include:

- Optimization of conditions: Comprehensive studies to determine the most optimum conditions for biochar production including feedstock, pyrolysis temperature and modification, and composting process for minimizing ammonia emissions should be conducted to achieve zero ammonia emissions efficiently.
- Denitrification Mechanisms: The role of denitrification in the ammonia removal process within both composting and biofilter systems, along with the impact of biochar on denitrification, should be investigated to understand the specific ammonia removal mechanisms and evaluate the potential environmental risks associated with the use of biochar in these systems.
- Broader applications: The potential of biochar and biofilters for ammonia removal in various environmental contexts, such as wastewater treatment and agricultural fertilization should be explored to expand their applicability and environmental benefits.

References

Agyarko-Mintah, E., Cowie, A., Van Zwieten, L., Singh, B.P., Smillie, R., Harden, S., Fornasier, F., 2017a. Biochar lowers ammonia emission and improves nitrogen retention in poultry litter composting. Waste Management 61, 129–137. https://doi.org/10.1016/j.wasman.2016.12.009

Agyarko-Mintah, E., Cowie, Annette, Singh, B.P., Joseph, S., Van Zwieten, L., Cowie, Alan, Harden, S., Smillie, R., 2017b. Biochar increases nitrogen retention and lowers greenhouse gas emissions when added to composting poultry litter. Waste Management 61, 138–149. https://doi.org/10.1016/j.wasman.2016.11.027

Bae, H., Chung, Y.-C., Yang, H., Lee, C., Aryapratama, R., Yoo, Y.J., Lee, S., 2015. Assessment of bacterial community structure in nitrifying biofilm under inorganic carbon-sufficient and -limited conditions. Journal of Environmental Science and Health, Part A 50, 201–212. https://doi.org/10.1080/10934529.2014.975550

Banik, C., Lawrinenko, M., Bakshi, S., Laird, D.A., 2018. Impact of Pyrolysis Temperature and Feedstock on Surface Charge and Functional Group Chemistry of Biochars. J Environ Qual 47, 452–461. https://doi.org/10.2134/jeq2017.11.0432

Baquerizo, G., Maestre, J.P., Machado, V.C., Gamisans, X., Gabriel, D., 2009. Long-term ammonia removal in a coconut fiber-packed biofilter: Analysis of N fractionation and reactor performance under steady-state and transient conditions. Water Res 43, 2293–2301. https://doi.org/10.1016/j.watres.2009.02.031

Barbusiński, K., Urbaniec, K., Kasperczyk, D., Thomas, M., 2020. Biofilters versus bioscrubbers and biotrickling filters: state-of-the-art biological air treatment, in: Soreanu, G., Dumont, E. (Eds.), From Biofiltration to Promising Options in Gaseous Fluxes Biotreatment. Elsevier, pp. 29–51. https://doi.org/10.1016/B978-0-12-819064-7.00002-9

Bello, A., Deng, L., Sheng, S., Jiang, X., Yang, W., Meng, Q., Wu, X., Han, Y., Zhu, H., Xu, X., 2020. Biochar reduces nutrient loss and improves microbial biomass of composted cattle manure and maize straw. Biotechnol Appl Biochem 67, 799–811. https://doi.org/10.1002/bab.1862

Bernal, M.P., Alburquerque, J.A., Moral, R., 2009. Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresour Technol 100, 5444–5453. https://doi.org/10.1016/j.biortech.2008.11.027

Bougnom, B.P., Boyomo, O., Nwaga, D., Essia Ngang, J.J., Etoa, F.X., 2014. Compost: A Tool to Sustainable Urban and Peri-Urban Agriculture in Sub-Saharan Africa? pp. 269–283. https://doi.org/10.1007/978-3-319-08004-8_13

Boulter, J.I., Trevors, J.T., Boland, G.J., 2002. Microbial studies of compost: Bacterial identification, and their potential for turfgrass pathogen suppression. World J Microbiol Biotechnol 18, 661–671. https://doi.org/10.1023/A:1016827929432

Cai, Y., Qi, H., Liu, Y., He, X., 2016. Sorption/Desorption Behavior and Mechanism of NH4+ by Biochar as a Nitrogen Fertilizer Sustained-Release Material. J Agric Food Chem 64, 4958–4964. https://doi.org/10.1021/acs.jafc.6b00109

Cai, Y., Zhu, M., Meng, X., Zhou, J.L., Zhang, H., Shen, X., 2022. The role of biochar on alleviating ammonia toxicity in anaerobic digestion of nitrogen-rich wastes: A review. Bioresour Technol 351, 126924. https://doi.org/10.1016/j.biortech.2022.126924

Cantrell, K.B., Hunt, P.G., Uchimiya, M., Novak, J.M., Ro, K.S., 2012. Impact of pyrolysis temperature and manure source on physicochemical characteristics of biochar. Bioresour Technol 107, 419–428. https://doi.org/10.1016/j.biortech.2011.11.084

Cao, Z., Zhu, R., Li, Y., Kakade, A., Zhang, S., Yuan, Y., Wu, Y., Mi, J., 2024. Mitigation of ammonia and hydrogen sulfide emissions during aerobic composting of laying hen waste through NaOH-modified biochar. J Environ Manage 365, 121634. https://doi.org/10.1016/j.jenvman.2024.121634

Carpıcı, E.B., Celık, N., Bayram, G., 2009. Effects of salt stress on germination of some maize (Zea mays L.) cultivars. Afr J Biotechnol 8, 4918–4922.

Chen, J., Jin, C., Sun, S., Yang, D., He, Y., Gan, P., Nalume, W.G., Ma, Y., He, W., Li, G., 2023. Recognizing the challenges of composting: Critical strategies for control, recycling, and valorization of nitrogen loss. Resour Conserv Recycl 198, 107172. https://doi.org/10.1016/j.resconrec.2023.107172

Chen, M., Wang, F., Zhang, D., Yi, W., Liu, Y., 2021. Effects of acid modification on the structure and adsorption NH4+-N properties of biochar. Renew Energy 169, 1343–1350. https://doi.org/10.1016/j.renene.2021.01.098

Chen, W., Liao, X., Wu, Yinbao, Liang, J.B., Mi, J., Huang, J., Zhang, H., Wu, Yu, Qiao, Z., Li, X., Wang, Y., 2017. Effects of different types of biochar on methane and ammonia mitigation during layer manure composting. Waste Management 61, 506–515. https://doi.org/10.1016/j.wasman.2017.01.014

Chen, Y.X., Yin, J., Wang, K.X., 2005. Long-term operation of biofilters for biological removal of ammonia. Chemosphere 58, 1023–1030. https://doi.org/10.1016/j.chemosphere.2004.09.052

Cheng, Z., Sun, Z., Zhu, S., Lou, Z., Zhu, N., Feng, L., 2019. The identification and health risk assessment of odor emissions from waste landfilling and composting. Science of the Total Environment 649, 1038–1044. https://doi.org/10.1016/j.scitotenv.2018.08.230

Chowdhury, M.A., de Neergaard, A., Jensen, L.S., 2014. Potential of aeration flow rate and bio-char addition to reduce greenhouse gas and ammonia emissions during manure composting. Chemosphere 97, 16–25. https://doi.org/10.1016/j.chemosphere.2013.10.030

Cole, E.J., Zandvakili, O.R., Xing, B., Hashemi, M., Herbert, S., Mashayekhi, H.H., 2019. Dataset on the effect of hardwood biochar on soil gravimetric moisture content and nitrate dynamics at different soil depths with FTIR analysis of fresh and aged biochar. Data Brief 25, 104073. https://doi.org/10.1016/j.dib.2019.104073

Das, J., Rene, E.R., Dupont, C., Dufourny, A., Blin, J., van Hullebusch, E.D., 2019. Performance of a compost and biochar packed biofilter for gas-phase hydrogen sulfide removal. Bioresour Technol 273, 581–591. https://doi.org/10.1016/j.biortech.2018.11.052

Domingues, R.R., Sánchez-Monedero, M.A., Spokas, K.A., Melo, L.C.A., Trugilho, P.F., Valenciano, M.N., Silva, C.A., 2020. Enhancing Cation Exchange Capacity of Weathered Soils Using Biochar: Feedstock, Pyrolysis Conditions and Addition Rate. Agronomy 10, 824. https://doi.org/10.3390/agronomy10060824

Domingues, R.R., Trugilho, P.F., Silva, C.A., De Melo, I.C.N.A., Melo, L.C.A., Magriotis, Z.M., Sánchez-Monedero, M.A., 2017. Properties of biochar derived from wood and high-nutrient biomasses with the aim of agronomic and environmental benefits. PLoS One 12, 1–19. https://doi.org/10.1371/journal.pone.0176884

Duan, Y., Awasthi, S.K., Liu, T., Zhang, Z., Awasthi, M.K., 2019. Response of bamboo biochar amendment on volatile fatty acids accumulation reduction and humification during chicken manure composting. Bioresour Technol 291, 121845. https://doi.org/10.1016/j.biortech.2019.121845

E., P., Sarkar, S., Maji, P.K., 2024. A review on slow-release fertilizer: Nutrient release mechanism and agricultural sustainability. J Environ Chem Eng 12, 113211. https://doi.org/10.1016/j.jece.2024.113211

Eghball, B., Lesoing, G.W., 2000. Viability of Weed Seeds Following Manure Windrow Composting. Compost Sci Util 8, 46–53. https://doi.org/10.1080/1065657X.2000.10701749

Estrada, J.M., Kraakman, N.J.R.B., Muñoz, R., Lebrero, R., 2011. A comparative analysis of odour treatment technologies in wastewater treatment plants. Environ Sci Technol 45, 1100–1106. https://doi.org/10.1021/es103478j

F. Estellés, R. W. Melse, N. W. M. Ogink, S. Calvet, 2011. Evaluation of the NH3 Removal Efficiency of an Acid Packed Bed Scrubber Using Two Methods: A Case Study in a Pig Facility. Trans ASABE 54, 1905–1912. https://doi.org/10.13031/2013.39831

Farghali, M., Chen, Z., Osman, A.I., Ali, I.M., Hassan, D., Ihara, I., Rooney, D.W., Yap, P.-S., 2024. Strategies for ammonia recovery from wastewater: a review. Environ Chem Lett 22, 2699–2751. https://doi.org/10.1007/s10311-024-01768-6

Feinerman, E., Komen, M.H.C., 2005. The Use of Organic vs. Chemical Fertilizer with a Mineral Losses Tax: The Case of Dutch Arable Farmers. Environ Resour Econ (Dordr) 32, 367–388. https://doi.org/10.1007/s10640-005-6647-5

Feng, Yuanyuan, Feng, Yanfang, Liu, Q., Chen, S., Hou, P., Poinern, G., Jiang, Z., Fawcett, D., Xue, L., Lam, S.S., Xia, C., 2022. How does biochar aging affect NH3 volatilization and GHGs emissions from agricultural soils? Environmental Pollution 294, 118598. https://doi.org/10.1016/j.envpol.2021.118598

Gao, M., Liang, F., Yu, A., Li, B., Yang, L., 2010. Evaluation of stability and maturity during forced-aeration composting of chicken manure and sawdust at different C/N ratios. Chemosphere 78, 614–619. https://doi.org/10.1016/j.chemosphere.2009.10.056

Godlewska, P., Schmidt, H.P., Ok, Y.S., Oleszczuk, P., 2017. Biochar for composting improvement and contaminants reduction. A review. Bioresour Technol 246, 193–202. https://doi.org/10.1016/j.biortech.2017.07.095

Gondek, M., Weindorf, D.C., Thiel, C., Kleinheinz, G., 2020. Soluble Salts in Compost and Their Effects on Soil and Plants: A Review. Compost Sci Util. https://doi.org/10.1080/1065657X.2020.1772906

Guo, F., Xu, F., Cai, R., Li, D., Xu, Q., Yang, X., Wu, Z., Wang, Y., He, Q., Ao, L., Vymazal, J., Chen, Y., 2022. Enhancement of denitrification in biofilters by immobilized biochar under low-temperature stress. Bioresour Technol 347, 126664. https://doi.org/10.1016/j.biortech.2021.126664

Han, B., Butterly, C., Zhang, W., He, J. zheng, Chen, D., 2021. Adsorbent materials for ammonium and ammonia removal: A review. J Clean Prod. https://doi.org/10.1016/j.jclepro.2020.124611

Hase, T., Kawamura, K., 2012. Evaluating compost maturity with a newly proposed index based on a germination test using Komatsuna (Brassica rapa var. peruviridis) seeds. J Mater Cycles Waste Manag 14, 220–227. https://doi.org/10.1007/s10163-012-0063-z

Hestrin, R., Enders, A., Lehmann, J., 2020. Ammonia volatilization from composting with oxidized biochar. J Environ Qual 49, 1690–1702. https://doi.org/10.1002/jeq2.20154

Hort, C., Gracy, S., Platel, V., Moynault, L., 2009. Evaluation of sewage sludge and yard waste compost as a biofilter media for the removal of ammonia and volatile organic sulfur compounds (VOSCs). Chemical Engineering Journal 152, 44–53. https://doi.org/10.1016/j.cej.2009.03.026

Huang, S., Liang, Q., Geng, J., Luo, H., Wei, Q., 2019. Sulfurized biochar prepared by simplified technic with superior adsorption property towards aqueous Hg(II) and adsorption mechanisms. Mater Chem Phys 238, 121919. https://doi.org/10.1016/j.matchemphys.2019.121919

Huff, M.D., Marshall, S., Saeed, H.A., Lee, J.W., 2018. Surface oxygenation of biochar through ozonization for dramatically enhancing cation exchange capacity. Bioresour Bioprocess 5, 18. https://doi.org/10.1186/s40643-018-0205-9

Hussain, A., Huang, W.Y., Lin, C.Y., Ahsan, W.A., Lin, C., 2023. Mitigation of ammonia emissions during food waste composting through acetic acid addition: A promising strategy for sustainable waste management. Sustain Chem Pharm 36. https://doi.org/10.1016/j.scp.2023.101324

Jassal, R.S., Johnson, M.S., Molodovskaya, M., Black, T.A., Jollymore, A., Sveinson, K., 2015. Nitrogen enrichment potential of biochar in relation to pyrolysis temperature and feedstock quality. J Environ Manage 152, 140–144. https://doi.org/10.1016/j.jenvman.2015.01.021

Ji, M., Wang, X., Usman, M., Liu, F., Dan, Y., Zhou, L., Campanaro, S., Luo, G., Sang, W., 2022. Effects of different feedstocks-based biochar on soil remediation: A review. Environmental Pollution. https://doi.org/10.1016/j.envpol.2021.118655

Jiang, T., Ma, X., Tang, Q., Yang, J., Li, G., Schuchardt, F., 2016. Combined use of nitrification inhibitor and struvite crystallization to reduce the NH3 and N2O emissions during composting. Bioresour Technol 217, 210–218. https://doi.org/10.1016/j.biortech.2016.01.089

Jun, Y., Wenfeng, X., 2009. Ammonia biofiltration and community analysis of ammonia-oxidizing bacteria in biofilters. Bioresour Technol 100, 3869–3876. https://doi.org/10.1016/j.biortech.2009.03.021

Kah, M., Sigmund, G., Chavez, P.L.M., Bielská, L., Hofmann, T., 2018. Sorption to soil, biochar and compost: Is prediction to multicomponent mixtures possible based on single sorbent measurements? PeerJ 2018. https://doi.org/10.7717/peerj.4996

Kebibeche, H., Khelil, O., Kacem, M., Kaid Harche, M., 2019. Addition of wood sawdust during the co-composting of sewage sludge and wheat straw influences seeds germination. Ecotoxicol Environ Saf 168, 423–430. https://doi.org/10.1016/j.ecoenv.2018.10.075

Keeney, D.R., Nelson., D.W., 1982. Nitrogen—Inorganic Forms. Methods of soil analysis. Part 2. Chemical and microbiological properties 5, 643–698.

Kelsey, T.W., Vaserstein, G., 2000. Farming and non farming neighbors: Conflict, coexistence, and communication. J Soil Water Conserv 55, 462–466.

Kennes, C., Veiga, M.C. (Eds.), 2001. Bioreactors for Waste Gas Treatment, Environmental Pollution. Springer Netherlands, Dordrecht. https://doi.org/10.1007/978-94-017-0930-9

Khan, N., Clark, I., Sánchez-Monedero, M.A., Shea, S., Meier, S., Bolan, N., 2014. Maturity indices in co-composting of chicken manure and sawdust with biochar. Bioresour Technol 168, 245–251. https://doi.org/10.1016/j.biortech.2014.02.123

Kim, Jun-Hwan, Kang, Y.J., Kim, K. Il, Kim, S.K., Kim, Jong-Hyun, 2019. Toxic effects of nitrogenous compounds (ammonia, nitrite, and nitrate) on acute toxicity and antioxidant responses of juvenile olive flounder, Paralichthys olivaceus. Environ Toxicol Pharmacol 67, 73–78. https://doi.org/10.1016/j.etap.2019.02.001

Kong, X., Ying, S., Yang, L., Xin, Y., Cai, Z., Zhu, S., Liu, D., 2020. Microbial and isotopomer analysis of N2O generation pathways in ammonia removal biofilters. Chemosphere 251, 126357. https://doi.org/10.1016/j.chemosphere.2020.126357

Kouba, V., Catrysse, M., Stryjova, H., Jonatova, I., Volcke, E.I.P., Svehla, P., Bartacek, J., 2014. The impact of influent total ammonium nitrogen concentration on nitrite-oxidizing bacteria inhibition in moving bed biofilm reactor. Water Science and Technology 69, 1227–1233. https://doi.org/10.2166/wst.2013.757

Lago, B.C., Silva, C.A., Melo, L.C.A., Morais, E.G. de, 2021. Predicting biochar cation exchange capacity using Fourier transform infrared spectroscopy combined with partial least square regression. Science of The Total Environment 794, 148762. https://doi.org/10.1016/j.scitotenv.2021.148762

Lee, J.W., Kidder, M., Evans, B.R., Paik, S., Buchanan III, A.C., Garten, C.T., Brown, R.C., 2010. Characterization of Biochars Produced from Cornstovers for Soil Amendment. Environ Sci Technol 44, 7970–7974. https://doi.org/10.1021/es101337x

Leng, L., Xiong, Q., Yang, L., Li, Hui, Zhou, Y., Zhang, W., Jiang, S., Li, Hailong, Huang, H., 2021. An overview on engineering the surface area and porosity of biochar. Science of the Total Environment. https://doi.org/10.1016/j.scitotenv.2020.144204

Li, M.-X., He, X.-S., Tang, J., Li, X., Zhao, R., Tao, Y.-Q., Wang, C., Qiu, Z.-P., 2021. Influence of moisture content on chicken manure stabilization during microbial agent-enhanced composting. Chemosphere 264, 128549. https://doi.org/10.1016/j.chemosphere.2020.128549

Li, R., Xu, K., Ali, A., Deng, H., Cai, H., Wang, Q., Pan, J., Chang, C.C., Liu, H., Zhang, Z., 2020. Sulfur-aided composting facilitates ammonia release mitigation, endocrine disrupting chemicals degradation and biosolids stabilization. Bioresour Technol 312. https://doi.org/10.1016/j.biortech.2020.123653

Li, X., Shi, X.S., Lu, M.Y., Zhao, Y.Z., Guo, R.B., Peng, H., 2020. Improved nitrogen conservation capacity during composting of dairy manure amended with oil shale semi-coke as the porous bulking agent. J Hazard Mater 388. https://doi.org/10.1016/j.jhazmat.2019.121742

Li, Y., Ma, J., Yong, X., Luo, L., Wong, J.W.C., Zhang, Y., Wu, H., Zhou, J., 2022. Effect of biochar combined with a biotrickling filter on deodorization, nitrogen retention, and microbial community succession during chicken manure composting. Bioresour Technol 343. https://doi.org/10.1016/j.biortech.2021.126137

Liang, Y., Leonard, J.J., Feddes, J.J.R., McGill, W.B., 2006. Influence of carbon and buffer amendment on ammonia volatilization in composting. Bioresour Technol 97, 748–761. https://doi.org/10.1016/j.biortech.2005.03.041

Lin, Y., Ding, W., Liu, D., He, T., Yoo, G., Yuan, J., Chen, Z., Fan, J., 2017. Wheat straw-derived biochar amendment stimulated N2O emissions from rice paddy soils by regulating the amoA genes of ammonia-oxidizing bacteria. Soil Biol Biochem 113, 89–98. https://doi.org/10.1016/j.soilbio.2017.06.001

Lippits, M.J., Gluhoi, A.C., Nieuwenhuys, B.E., 2008. A comparative study of the selective oxidation of NH3 to N2 over gold, silver and copper catalysts and the effect of addition of Li2O and CeOx. Catal Today 137, 446–452. https://doi.org/10.1016/j.cattod.2007.11.021

Liu, J., Hu, Y., Gu, S., Li, X., Ji, Z., Qin, H., Zhang, L., Zhang, J., Huang, H., Yan, B., Luo, L., 2024. Insight into mitigation mechanisms of N2O emission by biochar during agricultural waste composting. Bioresour Technol 406, 130970. https://doi.org/10.1016/j.biortech.2024.130970

Liu, Q., Wu, Y., Ma, J., Jiang, J., You, X., Lv, R., Zhou, S., Pan, C., Liu, B., Xu, Q., Xie, Z., 2024. How does biochar influence soil nitrification and nitrification-induced N2O emissions? Science of The Total Environment 908, 168530. https://doi.org/10.1016/j.scitotenv.2023.168530

Liu, W., Huo, R., Xu, J., Liang, S., Li, J., Zhao, T., Wang, S., 2017. Effects of biochar on nitrogen transformation and heavy metals in sludge composting. Bioresour Technol 235, 43–49. https://doi.org/10.1016/j.biortech.2017.03.052

Liu, X., He, X., 2017. Effect of pore characteristics on coalbed methane adsorption in middle-high rank coals. Adsorption 23, 3–12. https://doi.org/10.1007/s10450-016-9811-z

López-Cano, I., Roig, A., Cayuela, M.L., Alburquerque, J.A., Sánchez-Monedero, M.A., 2016. Biochar improves N cycling during composting of olive mill wastes and sheep manure. Waste Management 49, 553–559. https://doi.org/10.1016/j.wasman.2015.12.031

Luqmani, B., Brookes, A., Moore, A., Vale, P., Pidou, M., McAdam, E.J., 2024. Transitioning through the vapour-liquid equilibrium for low energy thermal stripping of ammonia from wastewater: Enabling transformation of NH3 into a zero-carbon fuel. Water Res 248, 120856. https://doi.org/10.1016/j.watres.2023.120856

Maia, G.D.N., Day V, G.B., Gates, R.S., Taraba, J.L., 2012. Ammonia biofiltration and nitrous oxide generation during the start-up of gas-phase compost biofilters. Atmos Environ 46, 659–664. https://doi.org/10.1016/j.atmosenv.2011.10.019

Mandal, S., Donner, E., Vasileiadis, S., Skinner, W., Smith, E., Lombi, E., 2018. The effect of biochar feedstock, pyrolysis temperature, and application rate on the reduction of ammonia volatilisation

from biochar-amended soil. Science of The Total Environment 627, 942–950. https://doi.org/10.1016/j.scitotenv.2018.01.312

Meng, J., Hu, Z., Wang, Z., Hu, S., Liu, Y., Guo, H., Li, J., Yuan, Z., Zheng, M., 2022. Determining Factors for Nitrite Accumulation in an Acidic Nitrifying System: Influent Ammonium Concentration, Operational pH, and Ammonia-Oxidizing Community. Environ Sci Technol 56, 11578–11588. https://doi.org/10.1021/acs.est.1c07522

Messiga, A.J., Dyck, K., Ronda, K., van Baar, K., Haak, D., Yu, S., Dorais, M., 2020. Nutrients Leaching in Response to Long-Term Fertigation and Broadcast Nitrogen in Blueberry Production. Plants 9, 1530. https://doi.org/10.3390/plants9111530

Mészáros, E., Jakab, E., Várhegyi, G., Bourke, J., Manley-Harris, M., Nunoura, T., Antal, M.J., 2007. Do all carbonized charcoals have the same chemical structure? 1. Implications of thermogravimetry-mass spectrometry measurements. Ind Eng Chem Res 46, 5943–5953. https://doi.org/10.1021/ie0615842

Mikajlo, I., Lerch, T.Z., Louvel, B., Hynšt, J., Záhora, J., Pourrut, B., 2024. Composted biochar versus compost with biochar: effects on soil properties and plant growth. Biochar 6, 85. https://doi.org/10.1007/s42773-024-00379-2

Miller, U., Sówka, I., Grzelka, A., Pawnuk, M., 2018. Application of biological deodorization methods in the aspect of sustainable development. SHS Web of Conferences 57, 02006. https://doi.org/10.1051/shsconf/20185702006

Ministry of Agriculture Forestry and Fisheries, 2021. 家畜排せつ物の発生と管理の状況.

Moore, P.A., Joern, B.C., Edwards, D.R., Wood, W., Daniel, T.C., 2006. Effects of manure amendments on environmental and production problems, in: Animal Agriculture and the Environment, National Center for Manure & Animal Waste Management White Papers

 American Society of Agricultural and Biological Engineers, St. Joseph, MI. https://doi.org/10.13031/2013.20271

Mugwili, M.E., Waanders, F.B., Masindi, V., Fosso-Kankeu, E., 2023. An update on sustainabilities and challenges on the removal of ammonia from aqueous solutions: A state-of-the-art review. J Environ Manage 347, 119172. https://doi.org/10.1016/j.jenvman.2023.119172

Mukome, F.N.D., Zhang, X., Silva, L.C.R., Six, J., Parikh, S.J., 2013. Use of chemical and physical characteristics to investigate trends in biochar feedstocks. J Agric Food Chem 61, 2196–2204. https://doi.org/10.1021/jf3049142

Neina, D., 2019. The Role of Soil pH in Plant Nutrition and Soil Remediation. Appl Environ Soil Sci 2019, 1–9. https://doi.org/10.1155/2019/5794869

Niu, L., Tariq Baig, Z., Yeung, M., Soomro, A.F., Lu, L., Xi, J., 2023. Low-concentration organics mitigate the inhibition of free nitrous acid on nitrification in biofilters for gaseous ammonia removal. Chemical Engineering Journal 476, 146757. https://doi.org/10.1016/j.cej.2023.146757

Pagans, E., Font, X., Sánchez, A., 2007. Adsorption, absorption, and biological degradation of ammonia in different biofilter organic media. Biotechnol Bioeng 97, 515–525. https://doi.org/10.1002/bit.21246

Pagans, Estel. la, Font, X., Sánchez, A., 2005. Biofiltration for ammonia removal from composting exhaust gases. Chemical Engineering Journal 113, 105–110. https://doi.org/10.1016/j.cej.2005.03.004

Pan, J., Cai, H., Zhang, Z., Liu, H., Li, R., Mao, H., Awasthi, M.K., Wang, Q., Zhai, L., 2018. Comparative evaluation of the use of acidic additives on sewage sludge composting quality improvement, nitrogen conservation, and greenhouse gas reduction. Bioresour Technol 270, 467–475. https://doi.org/10.1016/j.biortech.2018.09.050

Poly, F., Wertz, S., Brothier, E., Degrange, V., 2008. First exploration of Nitrobacter diversity in soils by a PCR cloning-sequencing approach targeting functional gene nxrA. FEMS Microbiol Ecol 63, 132–140. https://doi.org/10.1111/j.1574-6941.2007.00404.x

Posmanik, R., Gross, A., Nejidat, A., 2014. Effect of high ammonia loads emitted from poultry-manure digestion on nitrification activity and nitrifier-community structure in a compost biofilter. Ecol Eng 62, 140–147. https://doi.org/10.1016/j.ecoleng.2013.10.033

Qian, S., Zhou, X., Fu, Y., Song, B., Yan, H., Chen, Z., Sun, Q., Ye, H., Qin, L., Lai, C., 2023. Biochar-compost as a new option for soil improvement: Application in various problem soils. Science of The Total Environment 870, 162024. https://doi.org/10.1016/j.scitotenv.2023.162024

Quilliam, R.S., Glanville, H.C., Wade, S.C., Jones, D.L., 2013. Life in the 'charosphere' – Does biochar in agricultural soil provide a significant habitat for microorganisms? Soil Biol Biochem 65, 287–293. https://doi.org/10.1016/j.soilbio.2013.06.004

R. W. Melse, N. W. M. Ogink, 2005. AIR SCRUBBING TECHNIQUES FOR AMMONIA AND ODOR REDUCTION AT LIVESTOCK OPERATIONS: REVIEW OF ON-FARM RESEARCH IN THE NETHERLANDS. Transactions of the ASAE 48, 2303–2313. https://doi.org/10.13031/2013.20094

Ravindran, B., Awasthi, M.K., Karmegam, N., Chang, S.W., Chaudhary, D.K., Selvam, A., Nguyen, D.D., Rahman Milon, A., Munuswamy-Ramanujam, G., 2022. Co-composting of food waste and swine manure augmenting biochar and salts: Nutrient dynamics, gaseous emissions and microbial activity. Bioresour Technol 344, 126300. https://doi.org/10.1016/j.biortech.2021.126300

Ren, X., Jiao, M., Chen, X., Liu, T., Zhang, Y., Zhang, Z., 2022. Role of bulking agents and additive on composting, in: Current Developments in Biotechnology and Bioengineering: Advances in Composting and Vermicomposting Technology. Elsevier, pp. 127–142. https://doi.org/10.1016/B978-0-323-91874-9.00015-2

Riis, A.L., 2007. Removal of odour and ammonia in ventilation air from growing-finishing pig units using vertical biofilters.

Rotthauwe, J.H., Witzel, K.P., Liesack, W., 1997. The ammonia monooxygenase structural gene amoA as a functional marker: molecular fine-scale analysis of natural ammonia-oxidizing populations. Appl Environ Microbiol 63, 4704–4712. https://doi.org/10.1128/aem.63.12.4704-4712.1997

Saludes, R.B., Iwabuchi, K., Kayanuma, A., Shiga, T., 2007. Composting of dairy cattle manure using a thermophilic–mesophilic sequence. Biosyst Eng 98, 198–205. https://doi.org/10.1016/j.biosystemseng.2007.07.003

Sanchez-Monedero, M.A., Cayuela, M.L., Roig, A., Jindo, K., Mondini, C., Bolan, N., 2018. Role of biochar as an additive in organic waste composting. Bioresour Technol 247, 1155–1164. https://doi.org/10.1016/j.biortech.2017.09.193

Savci, S., 2012. Investigation of Effect of Chemical Fertilizers on Environment. APCBEE Procedia 1, 287–292. https://doi.org/10.1016/j.apcbee.2012.03.047

Sha, Z., Li, Q., Lv, T., Misselbrook, T., Liu, X., 2019. Response of ammonia volatilization to biochar addition: A meta-analysis. Science of the Total Environment 655, 1387–1396. https://doi.org/10.1016/j.scitotenv.2018.11.316

Shan, G., Li, W., Gao, Y., Tan, W., Xi, B., 2021. Additives for reducing nitrogen loss during composting: A review. J Clean Prod 307, 127308. https://doi.org/10.1016/j.jclepro.2021.127308

Shang, B., Zhou, T., Tao, X., Chen, Y., 2022. Greenhouse Gas Emissions From Biofilters for Composting Exhaust Ammonia Removal. Front Bioeng Biotechnol 10. https://doi.org/10.3389/fbioe.2022.918365

Sharma, B., Vaish, B., Monika, Singh, U.K., Singh, P., Singh, R.P., 2019. Recycling of Organic Wastes in Agriculture: An Environmental Perspective. Int J Environ Res. https://doi.org/10.1007/s41742-019-00175-y

Shin, Y., Iwabuchi, K., Itoh, T., 2024. Low-temperature biochars are more effective in reducing ammonia emissions through various mechanisms during manure composting. J Mater Cycles Waste Manag 26, 138–148. https://doi.org/10.1007/s10163-023-01808-3

Sikora, L.J., 1998. Benefits and Drawbacks to Composting Organic By-Products, in: Beneficial Co-Utilization of Agricultural, Municipal and Industrial by-Products. Springer Netherlands, Dordrecht, pp. 69–77. https://doi.org/10.1007/978-94-011-5068-2_6

Snyder, C.S., Bruulsema, T.W., Jensen, T.L., Fixen, P.E., 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agric Ecosyst Environ. https://doi.org/10.1016/j.agee.2009.04.021

Soliman, M., Eldyasti, A., 2018. Ammonia-Oxidizing Bacteria (AOB): opportunities and applications—a review. Rev Environ Sci Biotechnol 17, 285–321. https://doi.org/10.1007/s11157-018-9463-4

Song, W., Guo, M., 2012. Quality variations of poultry litter biochar generated at different pyrolysis temperatures. J Anal Appl Pyrolysis 94, 138–145. https://doi.org/10.1016/j.jaap.2011.11.018

Song, Y., Zhang, X., Ma, B., Chang, S.X., Gong, J., 2014. Biochar addition affected the dynamics of ammonia oxidizers and nitrification in microcosms of a coastal alkaline soil. Biol Fertil Soils 50, 321–332. https://doi.org/10.1007/s00374-013-0857-8

Steiner, C., Das, K.C., Melear, N., Lakly, D., 2010. Reducing Nitrogen Loss during Poultry Litter Composting Using Biochar. J Environ Qual 39, 1236–1242. https://doi.org/10.2134/jeq2009.0337

Suliman, W., Harsh, J.B., Abu-Lail, N.I., Fortuna, A.M., Dallmeyer, I., Garcia-Perez, M., 2016. Modification of biochar surface by air oxidation: Role of pyrolysis temperature. Biomass Bioenergy 85, 1–11. https://doi.org/10.1016/j.biombioe.2015.11.030

Sumaraj, Xiong, Z., Sarmah, A.K., Padhye, L.P., 2020. Acidic surface functional groups control chemisorption of ammonium onto carbon materials in aqueous media. Science of the Total Environment 698, 134193. https://doi.org/10.1016/j.scitotenv.2019.134193

Sun, Y., Wang, T., Sun, X., Bai, L., Han, C., Zhang, P., 2021. The potential of biochar and lignin-based adsorbents for wastewater treatment: Comparison, mechanism, and application—A review. Ind Crops Prod 166, 113473. https://doi.org/10.1016/j.indcrop.2021.113473

Taghipour, H., Shahmansoury, M.R., Bina, B., Movahdian, H., 2008. Operational parameters in biofiltration of ammonia-contaminated air streams using compost–pieces of hard plastics filter media. Chemical Engineering Journal 137, 198–204. https://doi.org/10.1016/j.cej.2007.04.015

Tan, Z., Zou, J., Zhang, L., Huang, Q., 2018. Morphology, pore size distribution, and nutrient characteristics in biochars under different pyrolysis temperatures and atmospheres. J Mater Cycles Waste Manag 20, 1036–1049. https://doi.org/10.1007/s10163-017-0666-5

Tian, X., Gao, R., Li, Y., Liu, Y., Zhang, X., Pan, J., Tang, K.H.D., Scriber, K.E., Amoah, I.D., Zhang, Z., Li, R., 2023. Enhancing nitrogen conversion and microbial dynamics in swine manure composting process through inoculation with a microbial consortium. J Clean Prod 423. https://doi.org/10.1016/j.jclepro.2023.138819

Tian, X., Qin, W., Zhang, Y., Liu, Y., Lyu, Q., Chen, G., Feng, Z., Ji, G., Yan, Z., 2024. The inoculation of thermophilic heterotrophic nitrifiers improved the efficiency and reduced ammonia emission during sewage sludge composting. Chemical Engineering Journal 479. https://doi.org/10.1016/j.cej.2023.147237

Tiquia, S.M., Tam, N.F.Y., Hodgkiss, I.J., 1996. Effects of composting on phytotoxicity of spent pig-manure sawdust litter. Environmental Pollution 93, 249–256. https://doi.org/10.1016/S0269-7491(96)00052-8

Tomczyk, A., Sokołowska, Z., Boguta, P., 2020. Biochar physicochemical properties: pyrolysis temperature and feedstock kind effects. Rev Environ Sci Biotechnol 19, 191–215. https://doi.org/10.1007/s11157-020-09523-3

Tsang, Y.F., Wang, L., Chua, H., 2015. Effects of high ammonia loads on nitrogen mass balance and treatment performance of a biotrickling filter. Process Safety and Environmental Protection 98, 253–260. https://doi.org/10.1016/j.psep.2015.08.008

Tsutomu, I., Takashi, A., Kuniaki, K., Kikuo, O., 2004. Comparison of Removal Efficiencies for Ammonia and Amine Gases between Woody Charcoal and Activated Carbon. Journal of Health Science 50, 148–153. https://doi.org/10.1248/jhs.50.148

Ulu, F., Kobya, M., 2020. Ammonia removal from wastewater by air stripping and recovery struvite and calcium sulphate precipitations from anesthetic gases manufacturing wastewater. Journal of Water Process Engineering 38, 101641. https://doi.org/10.1016/j.jwpe.2020.101641

United Nations, 2024. World Population Prospects 2024 [WWW Document]. https://population.un.org/wpp/. URL https://population.un.org/wpp/ (accessed 10.31.24).

Vandecasteele, B., Sinicco, T., D'Hose, T., vanden Nest, T., Mondini, C., 2016. Biochar amendment before or after composting affects compost quality and N losses, but not P plant uptake. J Environ Manage 168, 200–209. https://doi.org/10.1016/j.jenvman.2015.11.045

Vijayanand, M., Ramakrishnan, A., Subramanian, R., Issac, P.K., Nasr, M., Khoo, K.S., Rajagopal, R., Greff, B., Wan Azelee, N.I., Jeon, B.-H., Chang, S.W., Ravindran, B., 2023. Polyaromatic hydrocarbons (PAHs) in the water environment: A review on toxicity, microbial biodegradation, systematic biological advancements, and environmental fate. Environ Res 227, 115716. https://doi.org/10.1016/j.envres.2023.115716

Villaverde, S., 2000. Nitrifying biofilm acclimation to free ammonia in submerged biofilters. Start-up influence. Water Res 34, 602–610. https://doi.org/10.1016/S0043-1354(99)00175-X

Wang, X., Selvam, A., Wong, J.W.C., 2016. Influence of lime on struvite formation and nitrogen conservation during food waste composting. Bioresour Technol 217, 227–232. https://doi.org/10.1016/j.biortech.2016.02.117

Wang, Z., Zheng, M., Xue, Y., Xia, J., Zhong, H., Ni, G., Liu, Y., Yuan, Z., Hu, S., 2020. Free ammonia shock treatment eliminates nitrite-oxidizing bacterial activity for mainstream biofilm nitritation process. Chemical Engineering Journal 393, 124682. https://doi.org/10.1016/j.cej.2020.124682

Wiedemeier, D.B., Abiven, S., Hockaday, W.C., Keiluweit, M., Kleber, M., Masiello, C.A., McBeath, A. v., Nico, P.S., Pyle, L.A., Schneider, M.P.W., Smernik, R.J., Wiesenberg, G.L.B., Schmidt, M.W.I., 2015. Aromaticity and degree of aromatic condensation of char. Org Geochem 78, 135–143. https://doi.org/10.1016/j.orggeochem.2014.10.002

Witter, E., Kirchmann, H., 1989. Effects of addition of calcium and magnesium salts on ammonia volatilization during manure decomposition, Plant and Soil. Kluwer Academic Publishers.

Wong, J.W.C., Karthikeyan, O.P., Selvam, A., 2017. Biological nutrient transformation during composting of pig manure and paper waste. Environ Technol 38, 754–761. https://doi.org/10.1080/09593330.2016.1211747

Wu, H., Zeng, G., Liang, J., Chen, J., Xu, J., Dai, J., Li, X., Chen, M., Xu, P., Zhou, Y., Li, F., Hu, L., Wan, J., 2016. Responses of bacterial community and functional marker genes of nitrogen cycling to biochar, compost and combined amendments in soil. Appl Microbiol Biotechnol 100, 8583–8591. https://doi.org/10.1007/s00253-016-7614-5

Wyer, K.E., Kelleghan, D.B., Blanes-Vidal, V., Schauberger, G., Curran, T.P., 2022. Ammonia emissions from agriculture and their contribution to fine particulate matter: A review of implications for human health. J Environ Manage. https://doi.org/10.1016/j.jenvman.2022.116285

Xiang, W., Zhang, X., Chen, J., Zou, W., He, F., Hu, X., Tsang, D.C.W., Ok, Y.S., Gao, B., 2020. Biochar technology in wastewater treatment: A critical review. Chemosphere 252, 126539. https://doi.org/10.1016/j.chemosphere.2020.126539

Xie, Y., Wang, L., Li, H., Westholm, L.J., Carvalho, L., Thorin, E., Yu, Z., Yu, X., Skreiberg, Ø., 2022. A critical review on production, modification and utilization of biochar. J Anal Appl Pyrolysis 161, 105405. https://doi.org/10.1016/j.jaap.2021.105405

Yasuda, T., Fukumoto, Y., Kuroda, K., Hanajima, D., Waki, M., Suzuki, K., 2022. Nitrogen Fate and Adaptation of the Microbial Community Responsible for Ammonia Removal in a Biofilter Treating Waste Gas from Livestock Manure Composting. Japan Agricultural Research Quarterly: JARQ 56, 25–32. https://doi.org/10.6090/jarq.56.25

Yu, J., Hu, H., Wu, X., Zhou, T., Liu, Y., Ruan, R., Zheng, H., 2020. Coupling of biochar-mediated absorption and algal-bacterial system to enhance nutrients recovery from swine wastewater. Science of the Total Environment 701. https://doi.org/10.1016/j.scitotenv.2019.134935

Yuan, J.-H., Xu, R.-K., Zhang, H., 2011. The forms of alkalis in the biochar produced from crop residues at different temperatures. Bioresour Technol 102, 3488–3497. https://doi.org/10.1016/j.biortech.2010.11.018

Zainudin, M.H., Mustapha, N.A., Maeda, T., Ramli, N., Sakai, K., Hassan, M., 2020. Biochar enhanced the nitrifying and denitrifying bacterial communities during the composting of poultry manure and rice straw. Waste Management 106, 240–249. https://doi.org/10.1016/j.wasman.2020.03.029

Zainudin, M.H.M., Zulkarnain, A., Azmi, A.S., Muniandy, S., Sakai, K., Shirai, Y., Hassan, M.A., 2022. Enhancement of Agro-Industrial Waste Composting Process via the Microbial Inoculation: A Brief Review. Agronomy. https://doi.org/10.3390/agronomy12010198

Zhan, Y., Wei, Y., Zhang, Z., Zhang, A., Li, Y., Li, J., 2021. Effects of different C/N ratios on the maturity and microbial quantity of composting with sesame meal and rice straw biochar. Biochar 3, 557–564. https://doi.org/10.1007/s42773-021-00110-5

Zhang, G., Guo, X., Zhu, Y., Liu, X., Han, Z., Sun, K., Ji, L., He, Q., Han, L., 2018. The effects of different biochars on microbial quantity, microbial community shift, enzyme activity, and biodegradation of polycyclic aromatic hydrocarbons in soil. Geoderma 328, 100–108. https://doi.org/10.1016/j.geoderma.2018.05.009

Zhang, H., Voroney, R.P., Price, G.W., 2017. Effects of Temperature and Activation on Biochar Chemical Properties and Their Impact on Ammonium, Nitrate, and Phosphate Sorption. J Environ Qual 46, 889–896. https://doi.org/10.2134/jeq2017.02.0043

Zhang, J., Lu, M., Wan, J., Sun, Y., Lan, H., Deng, X., 2018. Effects of pH, dissolved humic acid and Cu2+ on the adsorption of norfloxacin on montmorillonite-biochar composite derived from wheat straw. Biochem Eng J 130, 104–112. https://doi.org/10.1016/j.bej.2017.11.018

Zhang, K., Chen, L., Li, Y., Brookes, P.C., Xu, J., Luo, Y., 2017. The effects of combinations of biochar, lime, and organic fertilizer on nitrification and nitrifiers. Biol Fertil Soils 53, 77–87. https://doi.org/10.1007/s00374-016-1154-0

Zhang, L., Sun, X., 2014. Changes in physical, chemical, and microbiological properties during the two-stage co-composting of green waste with spent mushroom compost and biochar. Bioresour Technol 171, 274–284. https://doi.org/10.1016/j.biortech.2014.08.079

Zhang, M., Song, G., Gelardi, D.L., Huang, L., Khan, E., Mašek, O., Parikh, S.J., Ok, Y.S., 2020. Evaluating biochar and its modifications for the removal of ammonium, nitrate, and phosphate in water. Water Res 186, 116303. https://doi.org/10.1016/j.watres.2020.116303

Zhang, T., Ding, L., Ren, H., 2009. Pretreatment of ammonium removal from landfill leachate by chemical precipitation. J Hazard Mater 166, 911–915. https://doi.org/10.1016/j.jhazmat.2008.11.101

Zhao, Y., Li, W., Chen, L., Meng, L., Zheng, Z., 2020. Effect of enriched thermotolerant nitrifying bacteria inoculation on reducing nitrogen loss during sewage sludge composting. Bioresour Technol 311. https://doi.org/10.1016/j.biortech.2020.123461

Zhong, X.-Z., Sun, Z.-Y., Wang, S.-P., Tang, Y.-Q., Kida, K., Tanaka, A., 2020. Minimizing ammonia emissions from dairy manure composting by biofiltration using a pre-composted material as the packing media. Waste Management 102, 569–578. https://doi.org/10.1016/j.wasman.2019.11.022

Zhou, S., Wen, X., Cao, Z., Cheng, R., Qian, Y., Mi, J., Wang, Y., Liao, X., Ma, B., Zou, Y., Wu, Y., 2021. Modified cornstalk biochar can reduce ammonia emissions from compost by increasing the number of ammonia-oxidizing bacteria and decreasing urease activity. Bioresour Technol 319. https://doi.org/10.1016/j.biortech.2020.124120

Zhu, X., Chen, B., Zhu, L., Xing, B., 2017. Effects and mechanisms of biochar-microbe interactions in soil improvement and pollution remediation: A review. Environmental Pollution. https://doi.org/10.1016/j.envpol.2017.04.032

Appendix

Table S1.Change in amount of precipitated NH_4^+–N in the compost-only biofilter (CB) and 25% biochar-mixed biofilter (B25) before and after biofiltration at 2000 and 1000 ppm. The values represent the mean of two replicates.

	Day 0		Day 7 or 21		Precipitated NH_4^+–N (mg/kg-dry biofilter)
	KCl	KCl+HCl	KCl	KCl+HCl	
CB_2000	337.2	452.4	6284.6	7271.9	872.1
CB_1000	265.1	510.7	6575.3	8201.1	1380.2
B25_2000	306.3	518.2	6768.7	7653.7	673.0
B25_1000	365.8	549.6	6641.7	9428.8	2603.4