

Article

The Optimal Cyanobacterial Sludge Incorporation Balances Nutrient Retention and NH₃ Emission Reduction During Composting with Chicken Manure and Wheat Straw

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Abstract

Managing eutrophic waterbodies produced large quantity of cyanobacterial sludge (CS), a biomass rich in nitrogen (N) that can be recycled through composting. However, how this management affects the compost fertility and ammonia (NH₃) volatilization is little known. This study used a chicken manure and wheat straw mixture with struvite, as the control composting treatment (CK). Subsequently, 10%, 20%, 30%, and 40% of the chicken manure was substituted with CS at the initiation of composting, which were named CS10%, CS20%, CS30%, and CS40%, respectively. The results showed that compost pH decreased by 0.2–0.5 units, while total N content significantly increased by 10.4–20.8% under all CS amended treatments compared to the CK. Furthermore, cumulative NH₃ volatilization in the CS amended treatments increased with higher CS substitution rates, showing a significant increase of 21.3–110.0%. In CS amended treatments, the initial contents of microcystin-RR and -LR were 82.0–328.0 µg kg⁻¹ and 48.0–192.0 µg kg⁻¹, respectively, which were degraded by 35.7–79.5% and 30.2–77.8%, peaking at 30% CS substitution. Notably, the CS40% treatment showed degradation rates dropping to 62.3% and 60.7%, accompanied by a significant increase in microcystin content. Meanwhile, the heavy metals (total arsenic, cadmium, chromium, mercury, and lead) contents of all composts complied with organic fertilizer standard (NY/T 525–2021) of China. Interestingly, the CS10% had significantly lower heavy metal concentrations compared to the CK, thus enhancing compost safety. In conclusion, 10% was an optimal CS incorporating ratio to improve the quality of compost derived from chicken manure, wheat straw and struvite, while reducing NH₃ emissions, which provided a feasible technical pathway for recycling the CS.

Keywords: cyanobacterial sludge; compost quality; nitrogen loss; resource recovery; ammonia volatilization



Academic Editor: Juan Antonio López González

Received: 25 August 2025

Revised: 8 October 2025

Accepted: 14 October 2025

Published: 20 October 2025

Citation: Liu, Y.; Chen, Y.; Chen, Y.; Feng, Y.; Sun, H. The Optimal Cyanobacterial Sludge Incorporation Balances Nutrient Retention and NH₃ Emission Reduction During Composting with Chicken Manure and Wheat Straw. *Agronomy* **2025**, *15*, 2429. <https://doi.org/10.3390/agronomy15102429>

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1. Introduction

Climate change and intensified human activities have led to eutrophication and cyanobacterial proliferation in lakes and rivers worldwide [1]. The Taihu Lake Basin in China has been particularly affected [2]. Mechanical salvage remains the most direct means to control widespread algal outbreaks [3]. For example, in Jiangsu province, China,

approximately 740,000 tons of cyanobacterial sludge (CS) were generated through salvage operations in 2023 [4]. The urgent challenge lies in the safe and efficient disposal of this CS. If improperly managed, CS decomposes anaerobically, releasing unpleasant odors and toxic microcystins, thereby transforming an algal problem into a secondary environmental hazard [5]. However, CS is rich in organic matter and essential nutrients, including 8–10% nitrogen (N), indicating its fertilization potential [6]. Aerobic composting is a promising method to simultaneously stabilize CS, degrade microcystins, and produce organic fertilizer [7]. Therefore, developing a reliable composting strategy for CS is of critical importance for achieving synergistic benefits between waste management and agricultural sustainability.

In conventional composting of chicken manure and straw, chicken manure serves as the primary source of N [8]. However, improper management and inadequate recycling of chicken manure can exert considerable environmental pressure [9]. Concurrently, tighter environmental regulations have led to a year-on-year increase in treatment costs [10], consequently driving up compost prices. The CS, being rich in N and carbon (C), could serve as a cost-effective alternative to partially replace chicken manure. This substitution could mitigate the environmental burden of both wastes while reducing composting costs. However, the appropriate substitution ratio of CS for chicken manure remains unclear. CS generally has a low C/N ratio, excessive substitution for chicken manure may disturb the C/N balance for compost decomposition [11], inhibit microbial activity, and increase N losses [12]. Moreover, the safe degradation of inherent microcystins must be guaranteed. Thus, the core innovation of this study is to systematically identify the optimal CS substitution ratio that maximizes resource recovery while ensuring compost quality, safety, and minimal environmental impact.

Ammonia (NH_3) volatilization is a primary pathway for N loss during composting, adversely affecting the fertility of the final compost product [13]. Released NH_3 and its subsequent deposition can degrade both water and soil quality [14]. Additionally, it may indirectly contribute to increased N_2O emissions [15]. In conventional chicken manure composting, N loss due to NH_3 volatilization can reach 47–62% [16]. CS, with its high reactive N content, may further aggravate NH_3 volatilization during composting [6]. Furthermore, in co-composting of CS and mushroom substrate, N loss due to NH_3 volatilization increased by approximately 2–4% with every 10% increase in CS proportion [4]. To mitigate NH_3 volatilization, struvite was incorporated into compost by promoting the precipitation of NH_4^+ [17]. By establishing a CS substitution gradient (10%, 20%, 30%, and 40%), this study aimed to identify a relative optimal ratio for balancing CS resource utilization with NH_3 volatilization reduction, ultimately enhancing the compost quality.

To ensure the agricultural safety, organic fertilizers must meet the Chinese Organic Fertilizer Standard (NY/T 525–2021), which sets limits for heavy metal contents. However, heavy metals such as chromium (Cr) and lead (Pb) are commonly introduced through feed additives, with more than 95% retained in animal manure, this accumulation can result in elevated heavy metal levels in the compost products [18,19]. Numerous studies have shown that algae can effectively remove heavy metals through bio-sorption [20,21]. However, the presence of endogenous microcystin residues in algal biomass poses potential ecological risks [11]. Notably, microcystin degradation predominantly takes place during the thermophilic phase of composting [5], where heat-tolerant microorganisms metabolize microcystin as a C source, achieving degradation rates of 60–77% [7]. However, this efficiency tends to decline as the proportion of CS increases [4].

In this study, chicken manure, wheat straw and struvite were used as the primary composting substrates. Then, the effects of CS substitution at varied ratios (10%, 20%, 30%, 40%) of chicken manure on maturity indicators, nutrient dynamics, safety indicators,

and NH_3 volatilization of compost were evaluated. The study aimed to find an optimal substitution ratio that balances nutrient retention and NH_3 emission reduction, offering theoretical insights and technical parameters for large scale CS composting and disposal.

2. Materials and Methods

2.1. Experimental Materials

The composting raw materials consisted of wheat straw sourced from Huifeng Straw Agricultural Products Deep Processing Co., Ltd., and chicken manure from Musen Fertilizer Co., Ltd., both located in Nanjing, China. CS collected from the Taihu Lake Basin. The struvite was synthesized using the aqueous phase (AP) derived from the hydrothermal carbonization process [22], which was obtained from the Jiangsu Academy of Agricultural Sciences [23]. Mg^{2+} and PO_4^{3-} sources were added to the AP as precipitants, and the pH of the mixture was adjusted to 10 using a NaOH solution to promote struvite crystallization. After the precipitation reaction, the supernatant was collected, and its pH was adjusted to 7.0 with HCl to obtain the final struvite product used in this study. Prior to composting, chicken manure was crushed into 2–3 cm pieces, wheat straw cut into 1–2 cm segments, and struvite ground into powder. The basic properties of these materials are summarized in Table 1.

Table 1. Basic properties of composting materials.

Tested Materials	Moisture	Organic Matter	Total N	Total P	Total K
	%		(g kg ^{−1})		
CS	23.4	410.2	74.1	109.8	17.5
Chicken manure	10.5	345.9	76.0	32.7	8.8
Wheat Straw	3.7	647.7	7.2	8.7	178.5

Note: Organic matter, Total N, Total phosphorus (P), and Total Potassium (K) are all expressed on a dry matter basis.

2.2. Experimental Design

The composting experiment began on 23 September 2022, at the greenhouse of the Jiangsu Academy of Agricultural Sciences and lasted for 43 days. Five treatments were evaluated: a chicken manure and wheat straw mixture with struvite as the control composting treatment (CK), and four treatments in which 10%, 20%, 30%, and 40% of the chicken manure was replaced by CS on a fresh weight basis (CS10%, CS20%, CS30%, and CS40%). The total initial mass for each composting pile was therefore 9.27 kg, comprising 9.0 kg of organic materials and a fixed addition of 0.27 kg of struvite. The struvite was added uniformly in all treatments at a rate of 3% of the organic material mass as a common conditioner to mitigate NH_3 volatilization [24], ensuring that the CS substitution rate was the sole variable under investigation. The specific material proportions for each treatment are shown in Table 2. At the beginning, raw materials were thoroughly blended and loaded into customized insulated compost bins, with internal and external dimensions of 53 × 40 × 29 cm and 64 × 48 × 36 cm, respectively, and other parameters adapted from Sun et al. (2023) [23]. The moisture content was maintained at 55–60% throughout the process.

Table 2. Material Proportions of Different Composting Treatments (Unit: kg).

Treatments	Chicken Manure	Wheat Straw	Struvite	CS
CK	5.4	3.6	0.27	–
CS10%	4.86	3.6	0.27	0.54
CS20%	4.32	3.6	0.27	1.08
CS30%	3.78	3.6	0.27	1.62
CS40%	3.24	3.6	0.27	2.16

Note: In the table, the “–” indicates that no CS was added to the corresponding treatment, respectively. The struvite was added at a rate of 3%, calculated based on the total mass of the organic materials (chicken manure + wheat straw).

2.3. Sample Collection and Measurement

2.3.1. Sample Collection

The compost was turned, and samples were collected on days 0, 7, 17, 24, 32, 39, and 42. Each treatment was set up with three replicates (i.e., three independent piles). On each sampling day, approximately 100 g of sub-sample was collected from each replicate pile using the five-point sampling method (from the four corners and the center of the pile). These sub-samples were immediately thoroughly mixed to form a composite sample weighing approximately 500 g. Subsequently, about 100 g was taken from each composite sample for subsequent analysis, while the remaining material was returned to the original pile to maintain the total mass. The 100 g analytical sample was further processed: one fresh portion was stored at 4 °C for analysis of pH, electrical conductivity (EC), germination index (GI), ammonium N ($\text{NH}_4^+\text{-N}$), and nitrate N ($\text{NO}_3^-\text{-N}$); the other was air-dried, ground, and sieved for determination of total N, total P, total K, and cation exchange capacity (CEC).

2.3.2. Composting Temperature, pH, EC, GI and CEC

During composting, an RC-4 temperature logger probe was inserted 30 cm into the compost heap to record temperatures every 30 min. The 24 h monitoring data were averaged to derive the daily temperature, while the ambient temperature was simultaneously logged using the same logger. Accumulated temperature is defined as the sum of the daily differences between the composting pile temperature and a benchmark temperature. This metric integrates both “temperature” and “time” into a single quantitative parameter, serving to measure the total thermal effect throughout the entire composting process. It is a crucial basis for assessing whether the compost has met the required sanitation standards. Accumulated temperature was calculated using the following Formula (1):

$$T = \sum_{i=1}^{n=43} (T_i - T_0) \quad (1)$$

In the formula, T denotes the cumulative temperature; T_i represents the daily average temperature, and T_0 refers to the biological zero temperature (15 °C) [25].

Fresh compost (2 g) was extracted with deionized water at a 1:10 (w/v) ratio. The mixture was then stirred at 180 rpm for 30 min and allowed to stand for 10 min. The extract was filtered through a 0.45 μm membrane, and pH and EC were measured using a pH/EC analyzer (Sanxin, SX723, Shanghai, China). The GI was determined using radish seeds according to the method described by Wei et al. (2022) [26].

CEC was determined using air-dried compost (sieved to 0.25 mm). Samples were repeatedly extracted with sodium acetate and centrifuged, followed by ethanol washing and a final extraction with ammonium acetate. The supernatant was collected, and sodium (Na^+) concentration was measured by flame photometry to calculate CEC.

2.3.3. Compost Nutrient Content

Approximately 0.1 g of air-dried compost was digested with a $\text{H}_2\text{SO}_4\text{-H}_2\text{O}_2$ mixture. Total N was determined by the Kjeldahl method [27], while total P and total K were measured by inductively coupled plasma optical emission spectrometry (ICP-OES) [28]. Total nutrient content was calculated as the sum of N, P, and K.

Fresh compost was extracted with KCl solution, followed by shaking, centrifugation, and filtration. $\text{NH}_4^+\text{-N}$ was determined by indophenol blue colorimetry, and $\text{NO}_3^-\text{-N}$ was measured using UV–visible spectrophotometry.

2.3.4. Microcystin and Heavy Metal Content of Compost Products

Samples were extracted with methanol using ultrasonic treatment, followed by centrifugation and evaporation. microcystin content was determined by high-performance liquid chromatography (HPLC) [29].

Air-dried samples were digested with nitric acid and hydrogen peroxide. After filtration, heavy metal concentrations were determined using inductively coupled plasma mass spectrometry (ICP-MS) [30].

2.3.5. NH_3 Volatilization from Composting

The continuous power pumping and confined chamber method was used, with a 2% boric acid solution as the NH_3 absorbent and a methyl red-bromocresol green mixture as the indicator. Daily pumping occurred from 8:00 to 10:00 am and from 2:00 to 4:00 pm. After each pumping, the compost was titrated with 0.02 M H_2SO_4 to calculate daily NH_3 volatilization. Cumulative NH_3 volatilization was calculated by summing the daily values using the following Formula (2):

$$C_n = \sum_{i=1}^n A_i \quad (2)$$

In the formula, C_n represents the cumulative NH_3 volatilization on day n , where A_i denotes the daily NH_3 volatilization on day i . The calculation is defined by the following equation.

2.4. Statistical Analysis

Data processing was performed using Microsoft Excel version 2021, and graphical outputs were generated in Origin version 2024. Significant differences among treatments ($p < 0.05$) were identified by one-way ANOVA in SPSS version 27. Results are expressed as mean $\pm$ SD ($n = 3$). Different letters denote statistically significant differences between treatments at $p < 0.05$.

3. Results

3.1. Dynamics in Maturation Parameters During Composting

The temperature in all treatments was with a typical composting trend, initially rising and then gradually declining (Figure 1a). By the 2nd day, temperatures increased rapidly, reaching the thermophilic phase, and peaked at 65.8–70.1 °C within the first week. Thereafter, temperatures gradually declined and began to stabilize on the 33rd day, maintaining a range of 22.0–27.0 °C. Compared with the CK treatment, the thermophilic phases in the CS amended treatments were shortened by approximately 1 day.

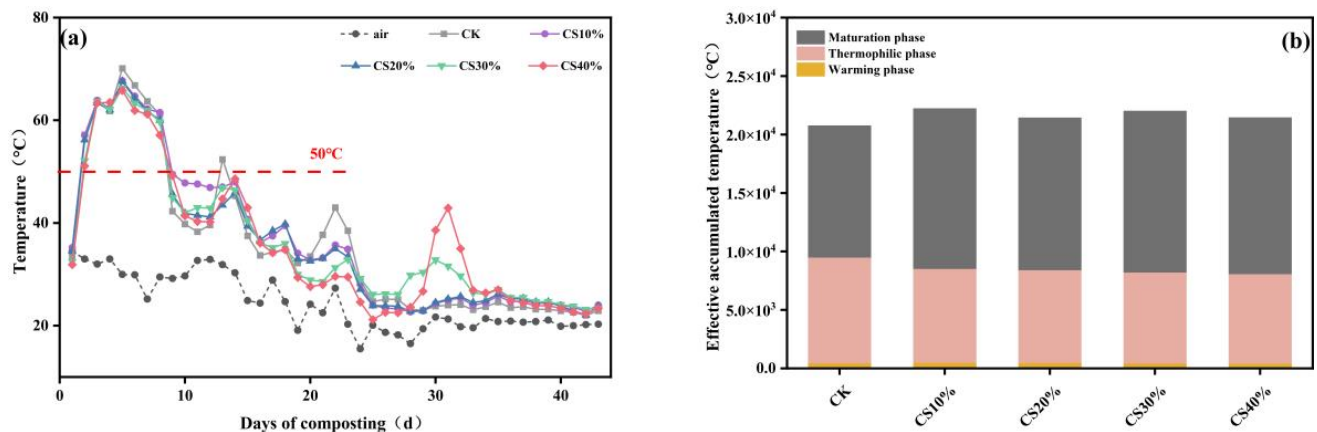


Figure 1. The dynamic changes in temperature (a) and effective accumulated temperature (b) during composting.

During the warming phase, the effective accumulated temperature in the CS amended treatments decreased with increasing CS substitution ratios, compared with the CK treatment, CS amended treatments showed a stepwise reduction of 12.7–18.2% in the thermophilic phase and an increase of 13.4–22.6% in the maturation phase (Figure 1b). By the end of composting, total effective accumulated temperatures ranged from 20,779.9 to 22,246.8 °C, while those in the CS amended treatments increased by 3.2–7.1% compared with the CK treatment.

The initial pH of all compost treatments ranged from 6.5 to 7.2. During the first week of composting, the pH increased rapidly and then gradually reached peaks of 9.2–9.4 between the 7th and 24th days, followed by a decline between the 25th and 42nd days (Figure 2a). At the end of composting, the pH of all the CS amended treatments (8.2–8.5) remained below 8.5, meeting compost maturity standards, and was 0.2–0.5 units lower than that of the CK treatment.

During the first week of composting, the EC values of all treatments increased rapidly, then decreased and subsequently rose again (Figure 2b). By the end of composting, EC of each treatment ranged from 4.5 to 5.3 mS cm^{−1}, with the CS30% treatment exhibiting the lowest value. Compared with the CK treatment, compost EC decreased significantly by 6.7–16.7% in the CS added treatments.

The CEC values at the end of composting ranged from 29.4–37.1 cmol kg^{−1} (Figure 2c), with the CS30% treatment exhibiting the highest value. The CEC values in the CS10% and CS30% treatments were 10.6% and 20.0% higher than those of the CK treatment, respectively.

GI increased gradually throughout composting in all compost treatments (Figure 2d). By the 24th day, GI exceeded 50% in all treatments. At the final stage, GI were 93.0–123.2%, with the CK treatment showing the lowest value.

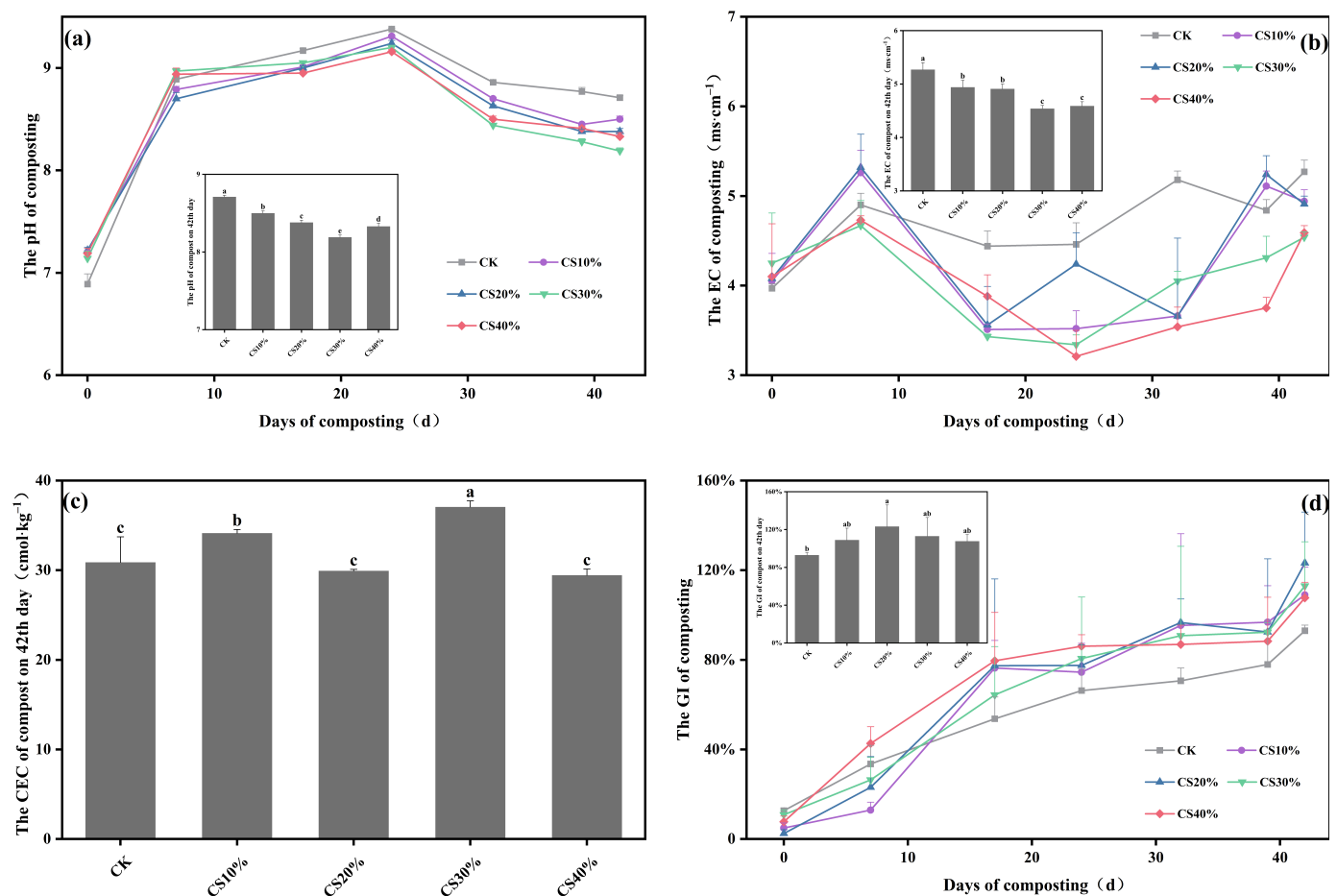


Figure 2. The dynamic changes in pH (a), EC (b), CEC (c), and GI (d) during composting. Data are expressed as mean $\pm$ SD ($n = 3$). Different letters denote statistically significant differences among treatments ($p < 0.05$).

3.2. Dynamics in N, P, and K Content During Composting

During composting, total N content in all treatments increased from 17.1–18.0 g kg⁻¹ at the initial stage to 21.7–27.4 g kg⁻¹ (Figure 3a). By the end of composting, total N content in the CS amended treatments was significantly 10.4–20.8% higher than that in the CK treatment, and increased with the CS substitution ratio.

All treatments exhibited an initial decline in total P content, followed by a gradual increase during composting (Figure 3b). By the end of composting, the total P contents were 15.3–17.1 g kg⁻¹, with the highest value observed in the CS10% treatment. Furthermore, the total P content in the CS40% treatment was 5.2–6.5% lower than that in other CS amended treatments.

During composting, total K content in all treatments exhibited an overall increasing trend (Figure 3c), reaching 24.0–24.9 g kg⁻¹ by the end. Although significant differences were observed among treatments, the overall variation in total K content remained relatively limited.

Total nutrient content in all treatments declined during the first week, followed by a gradual increase from 7.0–7.2% on the 7th day to 8.6–9.4% at the end of composting (Figure 3d). Compared with the CK treatment, total nutrient content in the CS amended treatments increased significantly by 9.1–9.4%.

At the end of composting, NH₄⁺-N content in all treatments ranged from 96.8–129.5 mg kg⁻¹ (Figure 4a). Compared to the CK treatment, NH₄⁺-N content in treatments with 10–30% CS substitution decreased significantly by 7.1–13.4%. Remarkable, the CS40%

treatment exhibited a significant increase of 15.9%. Furthermore, NO_3^- -N content ranged from 29.0–59.5 mg kg^{-1} , with the CS amended treatments exhibiting a significant reduction of 8.3–51.1% compared to the CK treatment (Figure 4b).

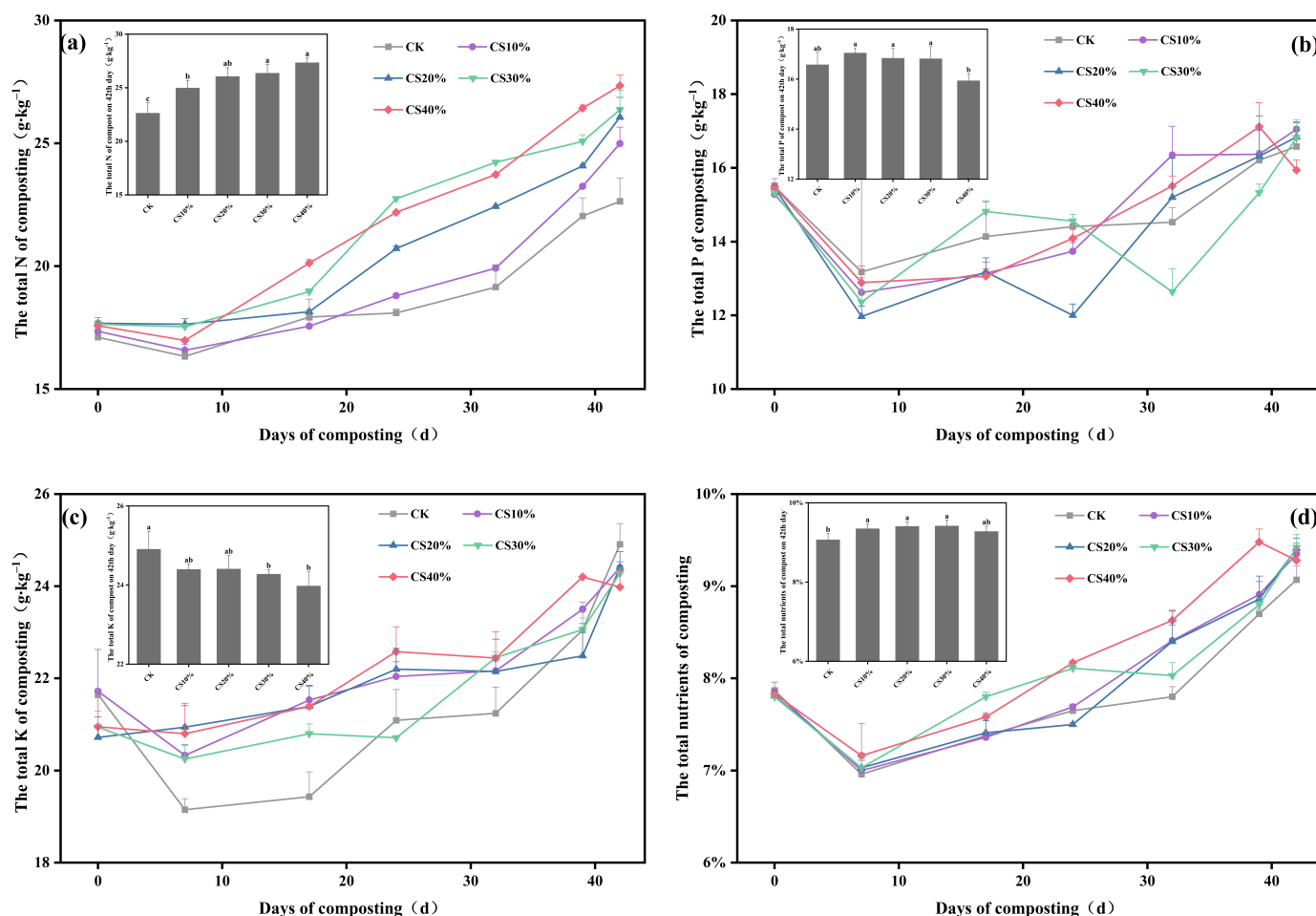


Figure 3. The dynamic changes in the contents of total N (a), total P (b), total K (c), and total nutrients (N and P plus K, (d)) in the compost samples collected during the composting. Data are presented as mean $\pm$ SD ($n = 3$). Different letters indicate significant differences among treatments ($p < 0.05$).

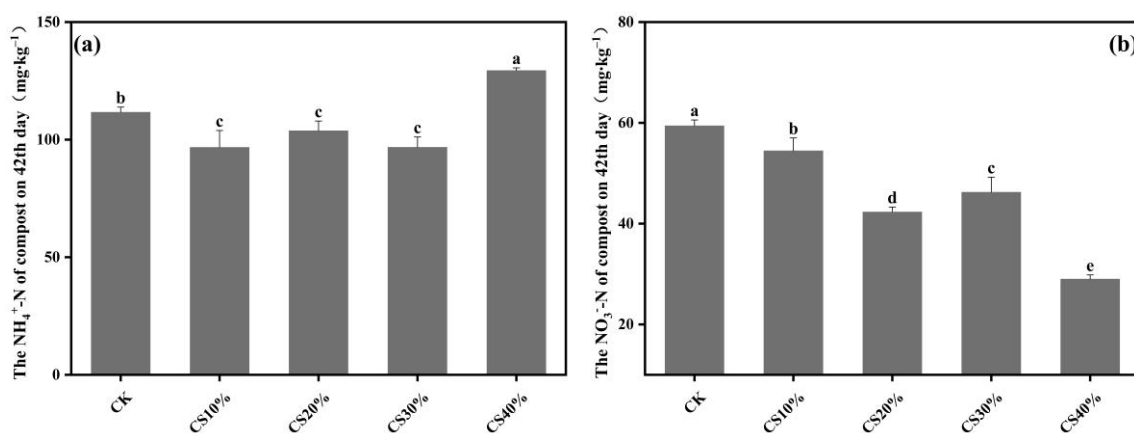


Figure 4. The contents of NH_4^+ -N (a) and NO_3^- -N (b) in the compost samples collected after the composting. Data are presented as mean $\pm$ SD ($n = 3$). Different letters indicate significant differences among treatments ($p < 0.05$).

3.3. NH_3 Volatilization During Composting

NH_3 volatilization in all treatments initially increased during composting, followed by a gradual decline, and eventually approaching 0 g d^{-1} (Figure 5). Peak NH_3 emissions occurred on the 12th day in the CK treatment and the CS40% treatment, and on the 8th day in the other CS amended treatments. Peak NH_3 volatilization ranged from 15.3 g day^{-1} to 37.5 g day^{-1} , with the CK treatment exhibiting the lowest value. Compared to the CK treatment, peak NH_3 volatilization in the CS amended treatments increased significantly by 59.6–144.9%.

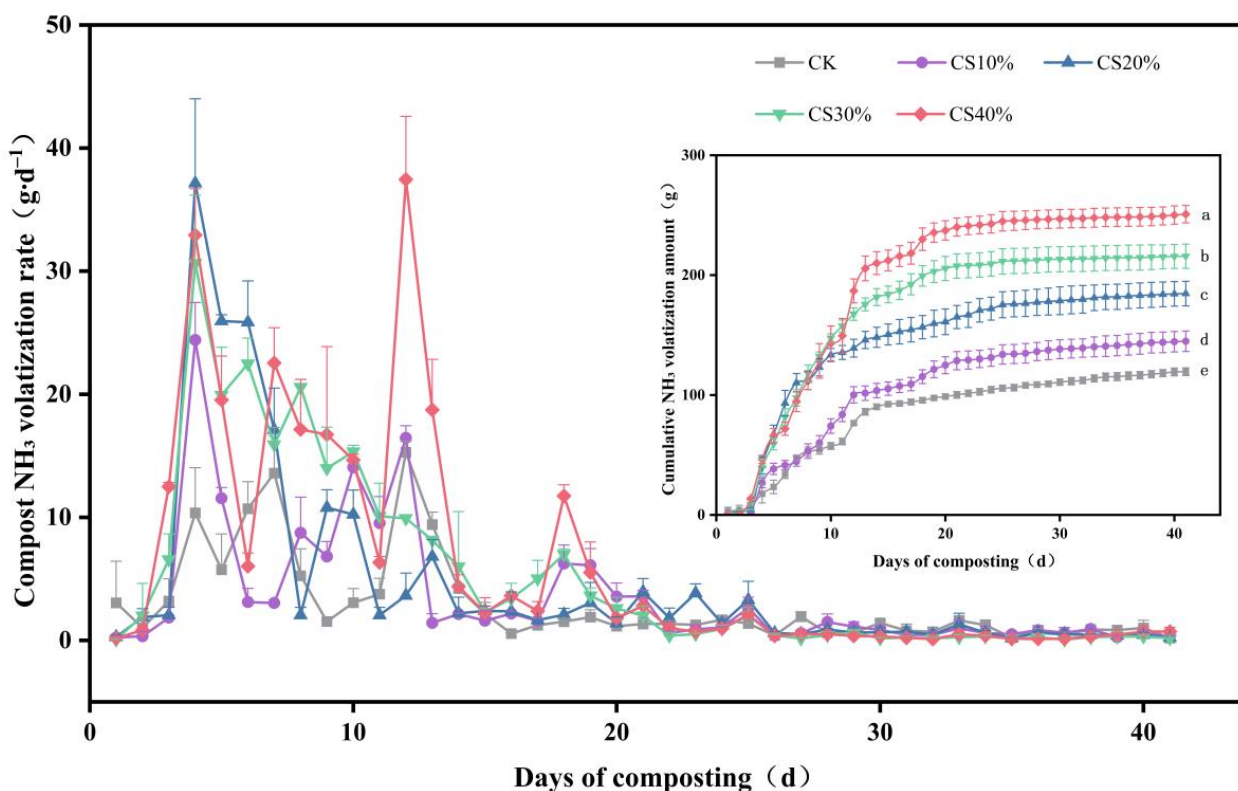


Figure 5. The dynamic changes in NH_3 volatilization and the cumulative total during the composting. Data are presented as mean $\pm$ SD ($n = 3$). Different letters indicate significant differences among treatments ($p < 0.05$).

During composting, cumulative NH_3 volatilization in all treatments increased continuously until stabilizing at a plateau (Figure 5). By the end of composting, cumulative NH_3 losses were 119.5–250.8 g, with the CK treatment exhibiting the lowest value. Compared to the CK treatment, cumulative NH_3 volatilization in the CS amended treatments increased significantly by 21.3–110.0% with rising CS substitution ratios.

3.4. Safety of Compost Products

In CS amended treatments, the initial contents of microcystin-RR and -LR were $82.0\text{--}328.0 \mu\text{g kg}^{-1}$ and $48.0\text{--}192.0 \mu\text{g kg}^{-1}$, respectively. By the end of composting, the degradation rates of microcystins exhibited a trend of first increasing and then decreasing with the increase in CS substitution ratio. The degradation rates of microcystin-RR and -LR were 35.7–79.5% and 30.2–77.8%, respectively, both peaking at a substitution ratio of 30% (Table 3). In the CS40% treatment, the degradation rates of microcystin-RR and -LR were 62.3% and 60.7%, respectively. Additionally, microcystin content increased significantly with the increase in CS substitution ratio. Compared with other CS amended treatments,

the contents of microcystin-RR and -LR in the CS40% treatment group were 114.3–145.0% and 92.8–136.4% higher, respectively.

Table 3. The content of microcystins and degradation rates in the final compost samples.

Treatments	Microcystins Content ($\mu\text{g kg}^{-1}$)		Degradation Rate (%)	
	Microcystin-RR	Microcystin-LR	Microcystin-RR	Microcystin-LR
CS10%	52.7 $\pm$ 27.6 b	33.5 $\pm$ 13.6 b	35.7%	30.2%
CS20%	57.7 $\pm$ 13.2 b	39.1 $\pm$ 0.2 b	65.9%	59.3%
CS30%	50.5 $\pm$ 7.9 b	31.9 $\pm$ 3.3 b	79.5%	77.9%
CS40%	123.7 $\pm$ 2.4 a	75.4 $\pm$ 2.9 a	62.3%	60.7%

Note: Data are presented as mean $\pm$ standard deviation ($n = 3$). Different lowercase letters indicate significant differences among treatments at $p < 0.05$.

By the end of composting, the contents of heavy metals in all compost treatments comply with China's organic fertilizer standard (NY/T 525–2021) for agricultural use safety (Table 4). The concentrations of total arsenic (As), cadmium (Cd), mercury (Hg), and Pb in all treatments generally exhibited a decreasing trend with increasing CS substitution rates. Compared with the CK treatment, the As, Cd, Hg, and Pb concentrations in the CS amended treatments were significantly 46.4–53.0%, 47.3–78.1%, 33.3%, and 29.3–36.4% lower in the CS amended composts. Notably, the CS10% treatment exhibited the lowest total Cr content (29.4 mg kg^{-1}), representing a significant reduction of 10.5–25.2% compared with the other CS amended treatments.

Table 4. The content of selected heavy metals in the final compost samples.

Treatments	As	Cd	Cr	Hg	Pb
	(mg kg^{-1})				
NY/T	≤ 15	≤ 3	≤ 150	≤ 2	≤ 50
CK	1.53 $\pm$ 0.01 a	1.14 $\pm$ 0.03 a	35.00 $\pm$ 0.70 c	0.03 $\pm$ 0.00 a	6.48 $\pm$ 0.01 a
CS10%	1.00 $\pm$ 0.02 b	0.60 $\pm$ 0.02 b	29.40 $\pm$ 0.30 e	0.02 $\pm$ 0.00 b	4.58 $\pm$ 0.04 b
CS20%	1.09 $\pm$ 0.15 b	0.41 $\pm$ 0.02 c	39.30 $\pm$ 0.80 a	0.02 $\pm$ 0.00 b	4.26 $\pm$ 0.12 c
CS30%	0.91 $\pm$ 0.07 cd	0.24 $\pm$ 0.01 d	37.85 $\pm$ 0.65 b	0.02 $\pm$ 0.00 b	4.12 $\pm$ 0.00 d
CS40%	0.82 $\pm$ 0.04 d	0.25 $\pm$ 0.00 d	32.85 $\pm$ 0.35 d	0.02 $\pm$ 0.00 b	4.26 $\pm$ 0.12 c

Note: Data are presented as mean $\pm$ SD ($n = 3$). Different letters indicate significant differences among treatments ($p < 0.05$). Standard: Agricultural Industry Standard of the People's Republic of China NY/T 525–2021.

4. Discussion

4.1. Effects of CS Addition on Maturation Parameters During Composting

Temperature is a key parameter for defining composting phases and assessing compost maturity [31]. In the present study, the rapid temperature increase observed on the 2nd day of composting may be attributed to the activity of thermophilic bacteria in the pile, which are associated with the accelerated breakdown of small molecules and soluble compounds in the compost material [32]. However, the low C/N ratio of the CS material likely constrained oxygen supply [33], which consequently suppressed microbial activity, leading to a shortened thermophilic phase in the CS amended treatments [34]. During composting, the temperature gradually declined and stabilized in the later stages, attributable to the depletion of readily degradable organic matter, consistent with previous studies [23]. Although the thermophilic phase in CS amended treatments was shortened to 7 days, this duration was still sufficient for effective aerobic composting. Moreover, the total effective accumulated temperature exceeded the spoilage-prevention threshold ($\geq 10,000$ °C) (Figure 1b). This threshold is widely recognized as a key indicator for effective sanitization, as it signifies that the composting material has absorbed sufficient thermal

energy over time to ensure the inactivation of pathogenic microorganisms, parasite eggs, and weed seeds [35]. In addition, biodegradation is considered the most effective method for the transformation of microcystins [36], with thermophilic bacteria in the compost pile playing a crucial role in this degradation process, suggesting that CS addition enhanced composting efficiency without impairing thermophilic performance.

The initial pH of all compost treatments were 6.5–7.2, providing a near-neutral environment conducive to the initiation of microbial activity [37]. The rapid increase in pH during the first week (Figure 2a), reflects intensive mineralization of N-rich organic matter from both chicken manure and CS, which released substantial NH_4^+ and degraded organic acids, collectively elevating pH [38,39]. Simultaneously, EC increased sharply in the first week (Figure 2b), consistent with the accumulation of soluble ions from decomposition [40]. After reaching a peak of 9.2–9.4, pH gradually declined. The decrease can be attributed to NH_3 volatilization and microbial nitrification, which consumes NH_4^+ and releases H^+ ions [23,37]. Furthermore, after the initial rise in the first week, EC decreased rapidly. This decline was mainly attributed to the reduction in ammonium ion concentration due to enhanced NH_3 volatilization under high temperature and alkaline conditions, as well as a further decrease in soluble ion content caused by the precipitation of mineral salts such as phosphates and carbonates [41]. Interestingly, during the phase of pH decline, EC exhibited a gradual increasing trend. This can be attributed to the stabilization of temperature in the later stage of composting, which enhanced nitrification activity, promoting the oxidation of NH_4^+ to NO_3^- and consequently increasing the concentration of soluble ions in the compost matrix [42]. At the end of composting, both pH and EC in the CS amended treatments were significantly lower than those in the CK treatment. The pH values in all CS treatments were ≤ 8.5 , which can be attributed to enhanced NH_3 volatilization coupled with nitrification dominated by ammonia-oxidizing bacteria (AOB) [43]. Notably, compost standards specify no absolute upper pH limit. However, this factor is highly correlated with the control of N loss through NH_3 volatilization [44]. Furthermore, as an organic fertilizer, the compost product must comply with the Chinese agricultural industry standard NY/T 525-2021, which sets a pH requirement of 5.5–8.5 for mature compost [45]. At the end of composting, the EC values of all treatments remained above the ideal threshold of 4 mS cm^{-1} [44]. Elevated EC levels can inhibit plant growth by reducing seed germination rates, causing wilting, and even inducing phytotoxicity [46]. Nevertheless, all treatments achieved a GI of $\geq 80\%$, meeting the maturity standard for compost. It should be noted that high EC may also suppress microbial metabolic activity by altering cellular osmotic pressure [47]. Notably, the addition of CS significantly reduced EC, indicating mitigated salinity stress. This reduction may be attributed to the role of CS in facilitating the fixation or precipitation of mineral salts, thereby decreasing the soluble ion concentration in the final compost [48].

At the end of composting, although the differences in CEC among treatments were relatively minor (Figure 2c), the CS30% treatment exhibited the highest CEC value. The interrelationship between pH, EC, and CEC in this treatment provides important insights into the composting process. The lower pH value observed in CS30% was likely due to enhanced microbial activity, which promoted nitrification and the formation of organic acids, resulting in a milder alkaline environment ($\text{pH} = 8.2$). The decrease in pH further influenced EC by reducing the solubility and promoting the precipitation of certain mineral salts, thereby decreasing the ionic concentration in the compost. Simultaneously, the increase in CEC suggests greater formation of stable organic matter and humic substances, which provide additional cation exchange sites [49]. Although the buffering effect of the mildly alkaline environment may have reduced the overall differences in CEC among treatments, the higher CEC in CS30% nevertheless indicates a greater nutrient retention

capacity, potentially reducing nutrient leaching and enhancing the agricultural value of the compost.

A GI exceeding 80% indicates that the compost meets the phytotoxicity threshold and is safe and suitable for use as a plant fertilizer [25]. This is consistent with the significantly lower EC values observed in these treatments compared to the control. The reduction in EC, indicating decreased salinity and lower concentration of soluble ions, likely contributed to the alleviation of salt stress on germinating seeds, thereby promoting higher germination rates and root elongation. Furthermore, the decomposition of organic matter and the formation of humic substances, accelerated by the elevated temperatures during composting, also played a crucial role in enhancing GI by providing beneficial compounds for root development [50]. Although elevated pH levels can sometimes reduce nutrient availability and impair seed vigor [51], the lower pH in CS amended treatments compared to CK likely mitigated these adverse effects. The combined effects of reduced salinity (as reflected by lower EC) and the presence of growth-promoting humic substances ultimately led to the superior GI performance in CS treatments, demonstrating an improvement in overall compost quality.

4.2. Effect of CS Addition on Nutrient Retention During Composting

During composting, total N content in all treatments exhibited a gradual increase (Figure 3a). The increase was relatively slow in the first week, mainly due to microbial assimilation and NH_3 volatilization [17,52]. Furthermore, as the rate of organic matter degradation exceeded the loss of N, the TN concentration increased slowly during this stage [53]. Throughout composting, the degradation of N-containing organic compounds reduced dry matter content, thereby increasing total N concentration [35,54]. Furthermore, in the later stages of composting, the decrease in pH facilitated the retention of NH_4^+ -N through the formation of crystalline compounds such as struvite and calcite [17]. By the end of composting, the total N content in the CS-amended treatments significantly increased ($p < 0.05$) by 10.4–20.8% compared to the CK treatment, with the increase corresponding to higher substitution ratios. Additionally, the total N content in the CS30% and CS40% treatments was significantly higher ($p < 0.05$) by 5.5–9.4% compared to the CS10% treatment. This enhancement can be primarily attributed to the high intrinsic N content of CS, which served as an exogenous N source and improved the overall N balance in the composting system.

In the initial stage of composting, total P content decreased in all treatments (Figure 3b), likely due to enhanced microbial activity facilitating P uptake for cellular synthesis [55]. As the composting process progressed, the total P content gradually increased, likely due to the concentration effect resulting from the mass loss during organic matter decomposition and transformation [40]. It is noteworthy that although no significant differences were observed between the CS amended treatments and the CK, the TP content in the CS40% treatment was significantly lower ($p < 0.05$) than that in the other CS-treated groups. This phenomenon may be attributed to the combined effects of its elevated pH and higher CS incorporation ratio. The high CS substitution rate likely enhanced microbial assimilation and the retention of P within humic substances [56]. Furthermore, the relatively high pH in CS40% may have promoted the precipitation of P into sparingly soluble forms, such as struvite or calcium phosphates [57]. It is important to note that P retained in humic substances or as amorphous phosphate precipitates can be partially released through the action of low-molecular-weight organic acids exuded by plant roots, this slow-release mechanism of P availability may ultimately support plant growth [58].

The dynamics of total K content exhibited a distinct pattern compared to N and P. Since K is not subject to volatilization losses and no leachate was generated during composting,

its concentration in all treatments showed a consistent increasing trend throughout the process (Figure 3c). At the end of composting, the TK contents in the CS30% and CS40% treatments were significantly lower ($p < 0.05$) than that in the CK. This phenomenon can be attributed to the temporary fixation of K into microbial biomass. Specifically, soluble K^+ is widely utilized by microorganisms for cellular functions and growth [59]. In treatments with higher CS substitution, this K retention process may be more pronounced, leading to a further reduction in measurable TK content.

Total nutrient content ($N + P_2O_5 + K_2O$) followed a decreasing trend followed by an increase (Figure 3d). According to the Chinese agricultural industry standard NY/T 525–2021, the total nutrient mass fraction should be $\geq 4.0\%$. All treatments had total nutrient contents significantly greater than 8.0%, meeting the requirements for organic fertilizer composting. Overall, the addition of CS has to some extent enhanced the retention of N, P, and K in the compost product. This has positive implications for improving the agronomic value of compost and reducing nutrient loss. However, practical application requires more detailed evaluation through subsequent field trials, soil quality assessments, and product analysis.

By the end of composting, NH_4^+-N ($< 0.4 \text{ g kg}^{-1}$) and NO_3^--N ($< 0.16 \text{ g kg}^{-1}$) contents in all treatments complied with the maturity standards for compost products [44]. Compared with the CK treatment, NH_4^+-N content was significantly higher in the CS40% treatment (Figure 4a), attributable to CS addition lowering the compost C/N ratio, which facilitated microbial degradation of organic N compounds such as proteins [11,13]. Furthermore, NO_3^--N content in the CS amended treatments exhibited a gradual decreasing trend with increasing CS substitution ratios (Figure 4b). Previous studies have demonstrated that cyanobacteria, the main component of CS, suppress the activity of nitrifying bacteria, consequently impacting NO_3^--N production [60].

4.3. Effects of CS Addition on NH_3 Volatilization During Composting

NH_3 volatilization is a major pathway of N loss during composting, which contributes to reduced compost quality and odor issues in the final product [13]. NH_3 volatilization occurred predominantly during the thermophilic phase (Figure 5), and its dynamics were closely related to pile temperature, pH, and the transformation of N forms. In the initial stage of composting, the rapid temperature increase promoted the ammonification of nitrogenous organic matter, leading to a rise in NH_4^+-N concentration, which provided the substrate for NH_3 volatilization [61]. Simultaneously, the increase in pH further drove the conversion of NH_4^+ to NH_3 [62]. Subsequently, as the composting temperature declined, enhanced nitrification converted NH_4^+ into NO_3^- , thereby reducing NH_3 emissions [63]. Although all treatments exhibited this general trend, significant differences in volatilization intensity and temporal dynamics were observed among them, mainly reflected in the following aspects: The CK treatment exhibited the lowest peak and cumulative emissions of NH_3 . Notably, its emission peak occurred later, on the 12th day. This delayed peak is primarily linked to the treatment's lower initial pH, which moderated the rate of ammonification [64], thereby limiting the rapid accumulation of NH_4^+-N and suppressing a concentrated release of NH_3 . In contrast, the NH_3 emission peaks for the CS10% and CS20% treatments appeared earlier, on the 4th day; however, their peak intensities were lower than those in the high-CS treatments. The addition of an appropriate amount of CS slightly adjusted the C/N ratio of the composting mixture, leading to a shift in the microbial community that included a higher relative abundance of fungal biomass [65]. Consequently, this resulted in an earlier but less intense release of NH_3 . Conversely, the CS30% treatment demonstrated a characteristically high and early emission peak (on the 4th day), coupled with high cumulative emissions. The core reason for this pattern is that

the CS addition significantly reduced the overall C/N ratio, creating a relative surplus of substrates for ammonification [13]. Furthermore, this treatment maintained a relatively higher pH level during the thermophilic phase, and these two factors synergistically promoted substantial NH_3 generation and volatilization [66]. Finally, The CS40% treatment exhibited a delayed peak (on the 12th day) yet the highest cumulative emissions, which may have been caused by the low initial C/N ratio resulting from the high substitution ratio of CS. Initially, the ammonification process was potentially delayed due to factors such as carbon source structure or microbial inhibition [67]. However, as composting progressed, the excessive N substrates were continuously mineralized during the thermophilic phase and converted into NH_3 under the prevailing high-pH conditions, leading to the highest N loss. Although NH_3 emissions generally declined over time, certain treatments exhibited a transient increase followed by a subsequent decrease. This phenomenon may be attributed to increased oxygen availability during turning, which stimulated microbial activity and accelerated the decomposition process [62].

Based on previous reports, the addition of struvite can effectively reduce the cumulative NH_3 volatilization during composting. This reduction is primarily attributed to the adsorption capacity and crystalline structure of struvite, which enables it to capture NH_4^+ generated in the compost, thereby mitigating NH_3 emissions [57]. The formation of struvite crystals significantly minimizes gaseous N loss, leading to a substantial increase in N retention within the final compost product. Furthermore, the elevated pH associated with struvite addition can influence the microbial community structure and the abundance of functional genes involved in the N cycle [68]. In the present study, struvite was directly added to the composting mixture. Theoretically, struvite itself is a valuable compound fertilizer as it simultaneously supplies P, N, and magnesium. Future research should focus on optimizing the addition strategy of struvite (e.g., timing and rate) to maximize its benefits in N conservation and compost quality, and to further elucidate its long-term agronomic value.

4.4. Effects of CS Addition on the Safety of Compost Products

The degradation of microcystin primarily relies on microbial activity during the thermophilic phase. According to previous studies, most thermophilic microorganisms can utilize microcystin as a carbon source, thereby facilitating its effective degradation during composting [5]. However, when the CS substitution ratio exceeded 40%, microcystin concentrations increased significantly by the end of composting (Table 3), likely because higher initial concentrations reduce removal efficiency due to microbial inhibition or saturation [69]. Moreover, an excessively high CS substitution ratio lowers the compost C/N ratio, which markedly inhibits the degradation efficiency of microcystin [4].

The potential toxicity of compost for agricultural applications is typically assessed based on the total concentrations of heavy metals [70]. The transformation of heavy metal forms during composting primarily occurs alongside the degradation of organic matter [71]. First, organic matter is degraded by microorganisms and transformed into humic substances [72]. Subsequently, in an oxidative environment, metal ions react with these newly formed humic substances, leading to the formation of stable organic-metal complexes facilitated by microbial activity [73]. Although this complexation process is accompanied by a concentrating effect due to the reduction in compost mass from mineralization [41], it ultimately significantly reduces the solubility of heavy metals, resulting in a decrease in their detected concentration. Inter-group comparative analysis revealed that the concentrations of most heavy metals (e.g., As, Cd, Hg, and Pb) were significantly lower ($p < 0.05$) in CS-amended treatments compared to the CK treatment, showing a general declining trend with increasing CS substitution ratios (Table 4). This reduction can be

attributed to the algal biomass in CS, which can immobilize heavy metals through cell wall adsorption and chelation, thereby contributing to their decreased concentrations [74]. This significant difference underscores the positive effect of CS addition on the immobilization of heavy metals. Notably, Cr concentrations did not consistently decrease, possibly because chromium primarily existed in a stable form that was resistant to degradation [75].

5. Conclusions

This study demonstrates that CS incorporation comprehensively enhanced the composting process across multiple quality dimensions. All CS-amended treatments successfully met maturity standards and agricultural safety requirements, confirming the feasibility of CS utilization. The principal finding identified 10% CS substitution as the optimal ratio, achieving the most favorable balance among key parameters: it significantly enhanced nutrient conservation (increasing TN by 10.4% and maintaining high TP content), effectively regulated pH dynamics, and minimized environmental impact through the lowest cumulative NH₃ volatilization. Furthermore, CS addition consistently improved product safety by facilitating heavy metal immobilization and microcystin degradation. The 10% substitution ratio thus represents the optimal strategy for harmonizing nutrient enhancement, environmental protection, and product safety. These findings provide both theoretical insights and technical parameters for large-scale agricultural application of CS, offering a sustainable approach for agroforestry waste management while addressing eutrophication challenges through circular economy principles.

Author Contributions: Y.L.: Investigation, Data curation, Writing—original draft. Y.C. (Yudong Chen): Investigation, Writing—review and editing. Y.C. (Ying Chen): Investigation. Y.F.: Writing—review and editing, Conceptualization. H.S.: Writing—review and editing, Supervision, Funding acquisition, Conceptualization. All authors have read and agreed to the published version of the manuscript.

Funding: This work was financially supported by the Science and Technology Innovation Program of Jiangsu Province, China for “Carbon Dioxide Emission Peaking and Carbon Neutrality” (BE2022305), the Priority Academic Program Development of Jiangsu Higher Education Institutions (PAPD), and the National Key Research and Development Program of China (2017YFD0801300; 2016YFD0801100).

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: Data supporting the findings of this study are available from the corresponding author upon reasonable request.

Acknowledgments: We would like to thank Wang Huang for helping us to produce the composting.

Conflicts of Interest: The authors declare no conflicts of interest.

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