Bioaugmentation mitigates ammonia and hydrogen sulfide emissions during the mixture compost of dewatered sewage sludge and reed straw

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Abstract

This study investigated the effectiveness of bio-augmenting aerobic cell culture to mitigate ammonia and hydrogen sulfide emission in sewage sludge composting amended with reed straw (with the weight ratio of 1:0.3–0.4). During the 20-day aerated lab-scale composting, adding 200 mL culture (56.80 NTU) reduced ammonia and hydrogen sulfide emissions by 38.00 % and 54.32 %, and conserved total nitrogen and sulfate by 39.42 % and 70.75 %, respectively. Organic matters degradation was quick started 1 day ahead. Comparing to the control, nitrate content increased 38.75 % at the end of the compost. Bioaugmentation evened the distributions of bacterial communities in the thermophilic phase. The shift was mainly due to 22.97 % of relative abundance of Proteobacteria depressed and 157.16 % of Bacteroidetes increased, which were benefit for nitrogen conservation and glycan breakdown, respectively. In summary, the results demonstrated that bioaugmentation addition could be an effective strategy for enhanced sludge composting.

Introduction

Dewatered Sewage sludge (DSS) was generated during the treatment of domestic sewage in wastewater treatment plants (WWTP)) and was identified as the biosolids. Composting was an economical and environmentally favorable technology for DSS stabilization and resource utilization, and has become more important for decomposing organic wastes than before (Doloman et al. 2020; He et al. 2018; Li et al. 2017).

The compost process involved complex physic-chemical interactions between the organic matter (OM) and decomposer. The decomposition of organic matter could be accelerated by a mixed population of microorganisms in a warm and moist environment (Kuypers et al. 2018; Zhao et al. 2019; Negi et al. 2020). Some organic matters were mineralized to carbon dioxide (CO_2), ammonia (NH_3) and water, and others were transformed to nutrient-rich, humus-like materials (Shou et al. 2019; 2017; Doloman et al. 2020; Du et al. 2019).

However, an inevitable problem in DSS compost was odor emission, especially that of NH_3 and H_2S, which correspondingly caused nitrogen and sulfur loss. Ammonia release accounted for up to 80 % of nitrogen loss during organic waste composting (Shou et al. 2019; Meng et al. 2016; Toledo et al. 2018). Sulfur deficiency in crop production was described to be a major problem in most parts of the world (Grant et al. 2012). The recovery of nitrogen and sulfur from the composting process could help to compensate these shortages and could also increase its value as a synthetic fertilizer substitute (Becarelli et al. 2019; Shou et al. 2019; Li and Li, 2015).

Odor emission was mainly resulting from the high moisture content, the low carbon-to-nitrogen (C/N) ratio of DSS and raw feedstock and low air-filled porosity along with the poor permeability (Doloman et al. 2020; Meng et al. 2016; Shou et al. 2019;). The average specific heat capacities (calorific value) of DSS in China was about 11.850 MJ/kg, 22.4–37.7 % lower than that in the developed countries. Most

agricultural and forest residues, such as annual herbage, cotton gin waste, rice straw and cornstalks, had higher calorific values from 12.18 to 18.06 MJ/kg (Cai et al. 2010; He et al. 2007). Those carbonaceous and lignocellulosic bulking agents were commonly adopted to reduce energy required to increase the pile temperature, optimize the compost condition for improved quality of the compost product as well as reduce odor emission (Shao and Zheng 2014; Shou et al. 2019; 2017).

Another dimension was the complex structures of organic matters, which was the major bottlenecks in the biodegradation and conversion in DSS compost. In China, DSS contained about 60–70 % of organic matters, including 62 % of stable sludge cells, 8–15 % of cellulose and 10.2 % humus (Dai et al. 2016; Xu et al. 2018). That signified long start-up time of sludge composting or the lag phase with low OM degradation efficiency, often relating with the nitrogen and sulfur loss and odor emission (Becarelli et al. 2019; Borowski et al. 2017; Quan et al. 2017). Therefore, bioaugmentation, the introduction of pre-grown robust consortium with specific catabolic abilities into the system, could be performed and may be advantageous.

Bioaugmentation had been established as an economical, ecological and environmental friend treatment to manipulate the microbial composition and accelerate related bio-metabolism (Borowski et al. 2017; Zhao et al. 2019; Doloman et al. 2020). It was capable of improving the removal of the contaminants in natural and environmentally systems by circumventing insufficient response time and initiating the removal with a minimal lag phase (McGenity et al. 2017; Speight 2017). Success of bioaugmentation was only possible if there was a substrate-specific niche available for the microbe to be incorporated into the already densely established consortia (Shao and Zheng, 2014; Doloman et al. 2020). Therefore, the special consortium could be isolated and enrichened from sludge, mainly to effectively improve the conditioning of wastewater sludge, such as accelerating the hydrolysis by attacking the sludge flocs, enhancing start-up of a bioreactor and shortening the reaction time (Speight, 2017; Xin et al. 2017; McGenity et al. 2017). Those microbial consortia in the system could be driven by the preponderant environmental factors to get to a relative balance level (Zhao et al. 2019; Xin et al. 2017).

In this study, aerobic indigenous consortia with less odor emission and higher OM degradation ability were isolated from DSS, and two composting batch trials were conducted with and without bioaugmentation. The objectives of this study were to (1) illustrate the changes of OM degradation and the mitigation of NH_3 and H_2S emission enhanced by bioaugmentation, and (2) dissect the shifting behaviors of bacterial community composition for the interactions of external culture with host bacteria. The outcomes were expected to put forward bioaugmentation to be a more practical alternative strategy in odor emission elimination and quick start of DSS compost.

Materials And Methods

Compost materials

DSS was obtained from the local municipal wastewater treatment plant, and reed straw was from the local wetland. Reed straw was sieved through a 1-mm mesh and was used as the bulking agent. Main characteristics of DSS and reed straw were list in Table 1. Aerobic indigenous consortia were isolated from the sludge using plate separation method. In pilot runs under the same condition as the following composting experiment setup (2.2), the culture with less NH_3 and H_2S emission and grown faster was adopted as bioaugmentation addition. The culture was dilated with phosphate buffering solution (pH = 6.61) and the turbidity was 56.8 NTU.

Table 1
Main characteristics of materials used in the compost

	pH	ω (%)	OM (%)	TN (%)	SO_4^- (mg/g)
DSS	6.65	83.73	66.57	57.10	3.11
Reed straw	5.77	5.35	88.36	9.78	9.46
Control	6.39	67.87	83.16	21.37	6.22
Bio-augmented	6.36	66.95	82.82	21.19	6.05

Composting experiment setup

Compost mixture consisted of DSS and reed straw with the weight ratio of 1:0.3–0.4 and the moisture content was about 65 %. Two identical lab-scale reactors were performed under the same condition throughout the experiment, one marked as “bio-augmented” with 200 mL of cell culture and another marked as “control” with 200ml of deionized water. The main characteristics of the experimental materials were also listed in Table 1. The reactors had 10 L effective volume, with height of 270 mm and inner diameter of 245 mm.

The 20-day composting process were divided into three phases based on the temperature control of the reactors: the initial calefactive at room temperature (day 0–2), the thermophilic phase with external heating from 50 ℃ hot water (day 3–12), and the maturation phase with no external heating until day 20. Continuous aeration (ZL 2019 2 2048934.3) in both runs was achieved using an air pump with air flow rate maintaining at 0.3 L/min. The exhausted gas was introduced into NaOH (2 M) and H_2SO_4 (1 M) solution for CO_2 and NH_3 absorption, respectively. There was no leachate in two systems. Pile turning was provided every other day.

Sampling and analysis

Samples were collected from the two reactors according to the quartering method and scheduled on days 0, 2, 4, 6, 8, 10, 14 and 20. Each sample was stored at 4 ℃ immediately for the determination of physical and chemical parameters. Sub-samples collected on day 2 (initial phase), day 6 (thermophilic phase) and day 20 (maturation phase) were stored at -20 ℃ for DNA extraction and sequencing.

The moisture content of composting materials (ω) was determined by drying the samples at 105 ℃ for 2 h, and organic matter (OM) by measuring the loss of dry-solid mass after ignition at 550 ℃ in a muffle furnace for 1 h. The value of pH was measured at a ratio of 1:5 (wet weight of composting sample: volume of deionized water) after shaking equilibration for approximately 4 h using a pH meter (E-201-C, Lei-ci, Shanghai, China). Water-soluble sulfate was deposited with Barium chloride and then detected with gravimetric method (HJ 635–2012). TN was carried out following the methods of digestion with alkaline Potassium persulfate and detected with UV spectrophotometry (CJ/T 221–2005). NH_4^+-N, NO_2^--N and NO_3^--N were extracted with Potassium chloride (1 mol/L) and determined by Phenol sodium-sodium hypochlorite colorimetry, Phenol disulfonyl acid colorimetry and Gerry reagent colorimetry, respectively (Belyaeva and Haynes 2009).

NH_3 emissions were quantified with titration method (Liu et al. 2011), and H_2S concentration online was detected with portable H_2S detector (range 0–20 ppm). The amount of NH_3 and H_2S production was determined every half day and the compost temperature was monitored and recorded once a day. When the emission of ammonia and H_2S were under the detection limit, 0.36 mg/d of NH_3 and 0.12 mg/d of H_2S were adopted every day.

Bacterial DNA extraction and sequencing

Bacterial genomic DNA was extracted from the sample using the MIO-BIO power soil DNA isolation kit. Fusion primer was adopted and the sequences were designed as the following:

F 5'-AATGATACGGCGACCACCGAGATCTACAC– barcode F2,

TCTTTCCCTACACGACGCTCTTCCGATCT- barcode F1- universal – 3';

R 5'-CAAGCAGAAGACGGCATACGAGAT- barcode R2,

GTGACTGGAGTTCCTTGGCACCCGAGAATTCCA- barcode R1- universal – 3'.

Universal primer (Bacterial 16S V4-V5) was adopted and the sequences was the following:

515F 5'-GTGCCAGCMGCCGCGGTAA-3',

926R 5'-CCGTCAATTCMTTTGAGTTT-3'.

AxyPrep DNA gel recovery kit recovery and real-time fluorescence quantity with the FTC-3000™ re-time PCR instrument were used to purify and quantity PCR product. The purified samples were for the sequencing of bacterial diversity using Illumina NGS (Tinygene Biotechnology Co., Ltd., Shanghai, China) (www.tinygene.com). Operational Taxonomic Units (OTU) were identified identically at a 97 % sequence homology, and the dilution curve was drawn for the optimized OTUs.

Statistical analysis

Organic matter loss and gas emission in the compost

As a microbial-mediated process, OM loss (OML) and gas emission could be described by a first-order kinetic model. The rate of OM mineralization and the production profiles of cumulative NH_3 and H_2S reflected the nature of the exponential growth curve (Negi et al. 2020). OML and the cumulative yield of gas were calculated as the following (Paredes et al. 2001; Bernal et al. 1996; Santos et al. 201b; Negi et al. 2020):

$$OML = A(1 - e^{-k_1 t})$$

$$Y = Y_{max}(1 - e^{-k_2 t})$$

where A reflected the potentially mineralizable OM, k_1 and k_2 were the rate constants (d^{-1}), Y_{max} was the maximum yield of gas (mg) and Y was cumulative yield of gas (mg) at sample time t (d). Because of the less biochemical metabolism at the start-up period in present study, ‘t’ calculated began on the 2.5th day for NH_3 and the 0.6th day for H_2S in this study.

Indexes of bacterial community diversity, evenness and similarity

In order to evaluate the bacterial community shifts in the piles, shannon index (H) (Shannon, 1948; Eichner et al. 1999) and equitability index (E) (Stamper et al. 2003) were applied to demonstrate the diversity and the evenness of bacterial community (Xin et al. 2015).

$$H = \sum p_i ln p_i$$

$$E = \frac{H}{\ln S}$$

where pi was the relative abundance of the species i and S was the number of the OTUs;

Data analysis

Experimental data were compared using analysis of paired Student's t test (T test) at a 95 % confidence level ($p < 0.05$). Spearman correlation with 2-tailed test were carried out to identify the relationships of the parameters. All the above analyses were performed in SPSS 13.0 software package. Redundancy analysis (RDA) was implemented to identify the effect of environmental variables on the bacterial community composition. The likelihood phylogenetic tree was constructed based on the sequence representative for each OTU as determined with Jaccard similarity coefficient. Multiple samples similarity tree was generated with MEGA 5.

Results And Discussion

Temperature, pH and degradation of organic matters profiles

In both piles, parameters of temperature, pH and degradation of organic matters showed a similar pattern along the process (Fig. 1).

Temperature variations during composting were the result of the thermal balance between the heat generated from the bio-degradation of organic matters and the heat loss through convection, evaporation and radiation (Becarelli et al. 2019; McGenity et al. 2017; Toledo et al. 2018). The temperature of the bio-augmented pile was higher than the control (Fig. 1a). Energy (kJ) emitted from the bioaugmentation mass was the product of the mass of compost (kg), the heat capacity of the compost (kJ/kg/K) and the increase of temperature of the composting mass (K) (Santos et al. 2016a, 2016b). The mean of the energy produced in the bio-augmented mass was 9908.60 MJ/d more than that in the control. Marks due to the addition of cell culture to the bio-augmented reactor made sense (Santos et al. 201b).

PH was a common and vital data to mediate the microbial multi-functional metabolization (Negi et al. 2020; Shou et al. 2019). The mean of pH value of the control pile was 7.26 ± 0.78, higher than 6.87 ± 0.46 of the bio-augmented ($p < 0.05$) (Fig. 1b). Both pH values fluctuated as the following trend: slowly decreased at the first 2 days, probably due to the mineralization of the highly biodegradable OM, leading to the acidification of the compost piles, and NH_3 volatilization; then increased sharply until the 6th day, due to the degradation of the biodegradable OM, such as the nitrogen- contain complex, and the enhancement of the ammonification; with a quick decrease after this day until the end of the composting process, due to nitrification, an acidification process (Negi et al. 2020; Zhang et al. 2020).

The OM available in the substrate was used for evaluating the different phases of composting, and the extent of organics removal by microbial degradation was evaluated by OML during aerobic sludge composting (Paredes et al. 2001; Santos et al. 201b; Negi et al. 2020). The experimental data of OM degradation in both piles fitted the first order reaction (Fig. 1c). Though parameter A and k_1 only varied slightly between two piles, the final OMLs of the control and the bio-augmented piles were 64.89% and 68.10%, respectively. This was due to the recalcitrant organic compounds of the compost mixture which hindered the bio-degradation process (Santos et al. 201b). However, bioaugmentation moved the degradation date 1 day ahead (Fig. 1c). Bioaugmentation was a suitable way to shorten the compost time and accelerates the compost efficiency (Negi et al. 2020). Further study on bioaugmentation with high hydrolyzing activity of poorly-degraded material should be noted.

Nitrogen- and sulfur- containing compounds profiles

In this study, the trends of the contents of TN, NH_4^+, $NO_3^-$and NO_2^- were similar during the compost process and the difference of the data between the two reactors were great ($p < 0.05$), especially that of TN and NO_3^- (Fig. 2a and Fig. 2b). The contents of TN, NO_3^- and NH_4^+ were significantly higher than that

of NO_2^- ($p < 0.05$). Because of the coupling of assimilatory nitrate and nitrite reduction, the release of nitrite from NO_3^- reduction was low in the earth (Kuypers et al. 2018).

The contents of TN increased smoothly along the compost process and widened by 28.96 % and 39.42 % at the end of the process in the control and bio-augmented reactors, respectively. The contributions were mainly from the reduction of total mass caused by OM degradation and the rising of moisture content (Shou et al. 2019; Li et al. 2017; Zhang et al. 2020). TN content in the bio-augmented reactor was 29.84 mg/g DW at the end of maturation phase, about 12.35% higher than that in the control (26.56 mg/g DW) ($p < 0.01$). That signified the effectiveness of bioaugmentation to conserve nitrogen and mitigate ammonia emission in the compost.

In the control and bio-augmented reactors, the contents of NH_4^+ increased rapidly during the initial phase, with the maximum of 2.87 and 2.23 mg/g DW on day 6, and then decreased slowly until the end of the maturation process (Fig. 2). During the present process, the evolution of OM and TN decreased the C/N ratio, which was higher in the control than the bio-augmented pile. Higher C/N ratio may favor the dissimilatory nitrate reduction to ammonium (Zhang et al. 2020; Kuypers et al. 2018), leading to more NH_4^+ and less NO_3^- accumulations in the control. Low concentration of NH_4^+ in the bioaugmentation may improve some enzyme activities in a certain degree, such as glutamate synthase, which had a relatively higher affinity for NH_4^+-N to consume low concentration of NH_4^+ as N sources for cell mass synthesis to form organic-N (Du et al. 2019; Shou et al. 2019).

Nitrate contents peaked 2.40 and 2.84 mg/g DW on day 4, and the mean was 2.01 mg/g DW in the bio-augmented piles, significantly higher than 1.45 mg/g DW in the control ($p < 0.01$) (Fig. 2). Bioaugmentation addition advanced and enhanced the nitrification of NH_4^+-N to form NO_3^-. Nitrification in this study mainly occurred in the mesophilic and the early thermophilic phase, earlier than that reported in the maturation phase (Santos et al. 201b; Shou et al. 2019). Higher nitrate accumulation also could keep pH lower and eliminate ammonia emission (Meng et al. 2016). NO_3^- coupled with ammonium to enhance the assimilation of nitrogen could prolong N retention and conserve more nitrogen in the pile (Kuypers et al. 2018; Zhang et al. 2020).

The Sulphur cycle was closely related to the main odorous substances produced during the composting. In the aerobic condition, many enzymes, such as arylsulfatase (ARS), could convert organic Sulphur to sulfate. The ARS activity was significantly higher during the maturing phase than other phases (Toledo et al. 2018), which was related with the gradual release of organic Sulphur from the compost mixture with the degradation of refractory organic matter (Shou et al. 2019; Santos et al. 201b; Negi et al. 2020). Therefore, the contents of SO_4^{2-} were rising gradually and continuously and reached the maxima of 11.67 mg/g DW on day 10 and 12.58 mg/g DW on day 14 in the control and the bio-augmented piles, respectively.

NH_3 and H_2S emission during the composting process

During the thermophilic phase with alkaline pH in both reactors, most ammonium was probably converted to ammonia for the shift of NH_4^+/NH_3 equilibrium (Chan et al. 2016; Li et al. 2017; Shou et al. 2019). Ammonia emission amount sharply increased to the maxima of 1200 and 1039.8 mg/d on day 4.5 in the control and the bio-augmented piles, respectively (Fig. 3a). Temperature and anaerobic condition were the main variables to effect H_2S produce and emission, especially in the initial phase of the compost (Zang et al. 2016; He et al. 2018; Toledo et al. 2018). In this study, H_2S emission occurred on day 0–4, and peaked 17.52 and 12.84 mg/d in control and experiment piles, respectively (Fig. 3b).

After day 14 and day 4, ammonia and H_2S emission were under the detection limits, respectively. 3707.52 mg of NH_3 and 37.44 mg of H_2S were generated in the bio-augmented reactor, reduced 38.00 % of 5997.84 mg of NH_3 and 54.32 % of 81.96 mg of H_2S measured in the control, respectively ($p < 0.001$). The 'k_2' values of NH_3 and H_2S obtained were found to be 0.30 d^{-1} and 0.70 d^{-1} in the bio-augmented pile, and 0.43 d^{-1} and 0.71 d^{-1} in the control, respectively (Fig. 3). Bioaugmentation could greatly reduce the amounts of NH_3 and H_2S emission and the emission rate of NH_3, but effect less on the emission rate of H_2S.

Nitrate reduction and sulfate reduction were two important pathways involved in the ammonia and H_2S emission over large scales and potentially widespread in the world (Zhang et al. 2020; Toledo et al. 2018; Negi et al. 2020). However, both reductions compete the electron donor, such as OM, to reduce NO_3^- and SO_4^{2-} (Zhang et al. 2020; Kuypers et al. 2018). With the less content of bio-degradable OM in the mature phase, these reductions were lowered. Furthermore, higher aerobic biomass of bioaugmentation inhibited the anaerobic cell activity in the compost, and mitigated the reductions to form NH_3 and H_2S (He et al. 2018; McGenity et al. 2017). As a result, more NO_3^- and SO_4^{2-} were accumulated in the bio-augmented reactor (Fig. 2a and Fig. 2b) and less NH_3 and H_2S emission than the control (Fig. 3a and Fig. 3b). Bioaugmentation in present study could inhibit the denitrification and sulfate reduction pathways in a certain degree (Fig. 4).

Bacterial community composition and dynamics

Diversity, evenness and similarity of bacterial community

In this study, the activity and composition of bacterial communities in the 16S rRNA gene annotation results were used to explore the relationship between the truly important functional implementers, and functional changes in the compost system (Zhao et al. 2019; Shou et al. 2019; Du et al. 2019). The rarefaction curve of sample was constructed and tended to flatten out at 97 % similarity level, and the Shannon-Wiener curve of each sample tended to be flat when the sequencing value was nearly 20000. Samples nearly attained the saturated stage and the sequencing covered more than 99.5 % of bacilli. Reads and OTU were listed in Table 2.

Table 2
Evenness and diversity indexes of bacterial communities from samples

	Coverage	Valid sequence number	OTU[a]	H	E
Control-2	0.997423	49145	1024	5.22	0.75
Control-6	0.996760	48593	551	2.91	0.46
Control-20	0.996452	51046	826	4.63	0.69
Bio-augmented-2	0.996941	54362	1049	5.11	0.73
Bio-augmented-6	0.996441	50854	576	3.48	0.55
Bio-augmented-20	0.996461	50779	737	4.54	0.69
a: The similarity level was 0.97.					

Higher biomass for the external addition by bioaugmentation indicated the strong ability of bioaugmentation cultures could survive and function over time in the complex interaction of multiple variables (Table 2). The depressing of the biodiversity and evenness of bacterial community in the thermophilic phase in both piles was a response of the bacterial community to resist the thermophilic condition (Doloman et al. 2020; McGenity et al. 2017; Xin et al. 2017).

Bacterial diversity and abundance directly affect the composting process, and the prevalent physic-chemical conditions also modulate the dynamics of bacteria community succession (Kuypers et al. 2018; Li et al. 2017; Meng et al. 2016; Du et al. 2019). The result of RDA showed that component 1 (PC1) and component 2 (PC2) explained 88.09 % and 8.94 % of the data variance, respectively (Fig. 4). The bacterial community in the bio-augmented pile could significantly influent pH value and NH_3 and H_2S emission. Table 2 and Fig. 4 also presented a remarkable difference and dissimilarity of bacterial community between the thermophilic phase and the other two phases.

Bioaugmentation could interfere the bacterial community succession and augmented the populations' evenness (E index) and diversity (H index) in the thermophilic phase (Table 2). As a result, the bacterial community in the bio-augmented pile tended to be stable to resist outer fluctuations and achieve the stability of system performance (Shao and Zheng 2014; Xin et al. 2017). Bioaugmentation could be used as a design tool to enhance bacterial diversity and evenness in the compost (Doloman et al. 2020; McGenity et al. 2017).

Bacterial community shifts in the thermophilic phase

The bacterial community in DSS composting was dominated by the phyla: Proteobacteria, Bacteroidetes, Actinobacteria, Firmicutes and Chloroflexi phyla (Shou et al. 2019; Du et al. 2019). The enhancement of populations' evenness (E index) and diversity (H index) in the thermophilic phase in the bio-augmented pile was mainly from the decreasing of Relative abundance (RA) of Proteobacteria and the rising of RA of Bacteroidetes (Fig. 5a).

The RA value of Proteobacteria was absolutely the peak in the whole process, especially in the thermophilic phase with 77.95 % and 60.04 % of RA in the control and the bio-augmented reactors, respectively. Bioaugmentation greatly depressed 26.69% of OTU number in the thermophilic phase, mainly for the decreased of RA of *Pseudoxanthomonas* from 50.60 % to 36.60 % (Fig. 5b). *Pseudoxanthomonas* could reduce nitrite but not nitrate with the only production of nitrous oxide (N_2O) (Finkmann et al. 2000; Thierry et al. 2004). This maybe one of the reasons to preserve nitrogen as nitrate in the bio-augmented pile. Certainly, the bioaugmentation in present study also mitigated the emission of greenhouse gas.

Bioaugmentation stabilized the RA of Bacteroidetes throughout the composting process. Its RA values in the bio-augmented pile were 23.38 %, 24.21 % and 28.75 % in the initial, thermophilic and mature phases, and 24.38 %, 9.42 % and 27.96 % in the control, respectively (Fig. 5b). Bacteria of the Bacteroidetes phylum were considered primary degraders of polysaccharides and were found in many ecosystems (Zhao et al. 2019). Collectively Bacteroidetes had elaborated a few thousand enzyme combinations for glycan breakdown (Lapébie et al. 2019). The fact that some bacteria with a lignocellulose-degrading ability become dominant microorganisms made sense, indicating accelerated composting for enhanced humification as well as compost maturity (Shou et al. 2019; Zhao et al. 2019). Bioaugmentation improved the distribution of bacterial community and enhanced the degradation of refractory organic matter.

Conclusion

Aerobic bioaugmentation could keep pH 6.87 ± 0.46. Comparing to the control, the bioaugmentation produced 9908.60 MJ/d energy, increased 38.75 % of NO_3^- and mitigated 38.00 % and 54.32 % of NH_3 and H_2S emissions, respectively. TN and sulfate contents increased 39.42% and 70.75% at the end, respectively. Bioaugmentation reduced 'k_2' of NH_3 from 0.43 d^{-1} to 0.30 d^{-1}, but effect less on that of H_2S and OM. OM degradation date was moved 1 day ahead. E and H indexes in the thermophilic phase was enhanced, mainly for the reduction of 22.97 % of RA of Proteobacteria and the rising of 157.16 % of Bacteroidetes.

Declarations

Ethical Approval: Not applicable.

Consent to Participate: Not applicable.

Consent to Publish: Not applicable.

Authors Contributions: Cheng Qingli, Zhang Longlong and Wang Dawei contributed to the conception and design of this study. All authors took part in the whole compost process. Parameters were measured by Zhang Longlong, Wang Dawei and Niu Bochao and analyzed by Cheng Qingli and Zhang Longlong. The

first draft of the manuscript was written by Cheng Qingli and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

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Figures

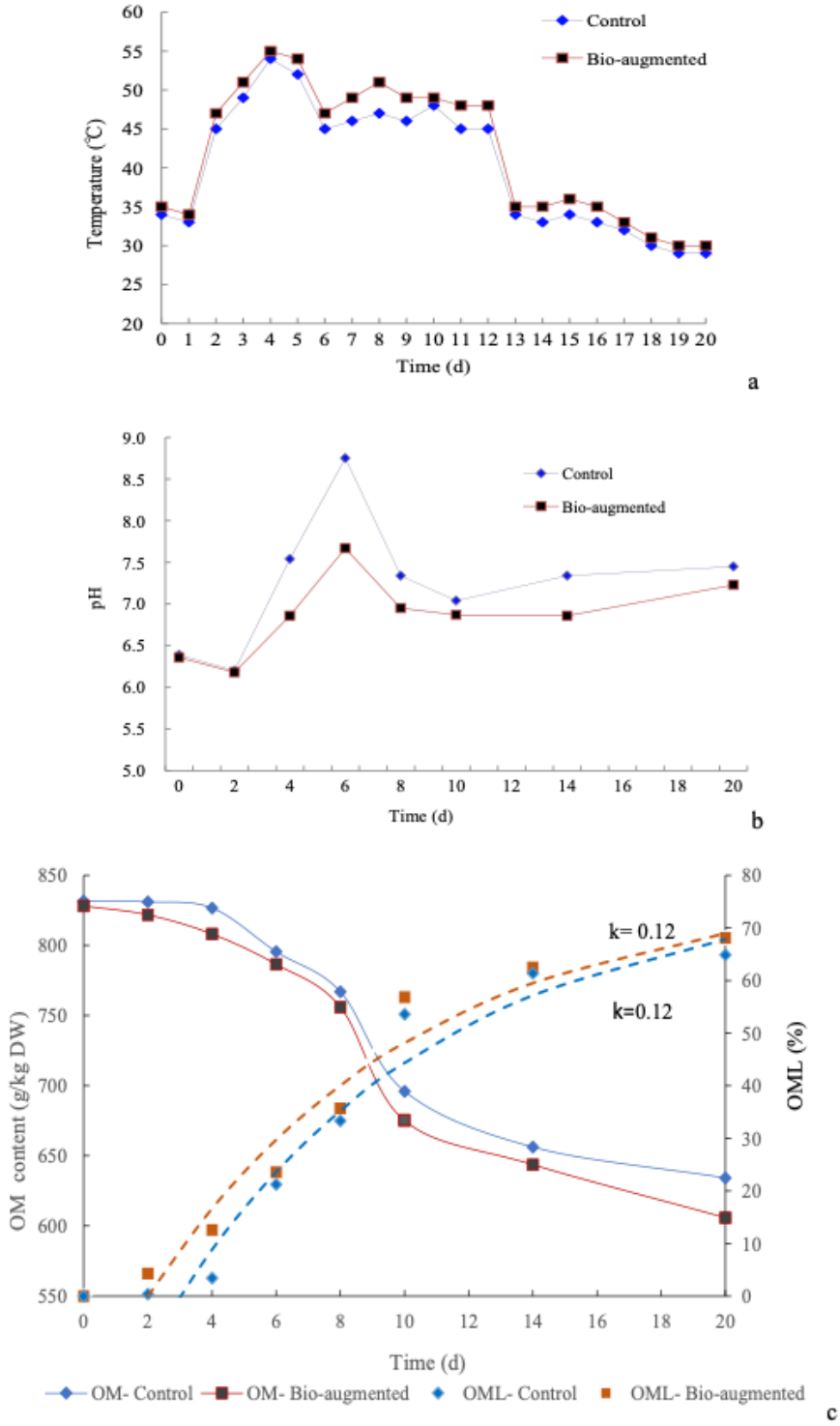

Figure 1

Time-course profiles of temperature (a), pH (b), OM and OML (c) of the compost mixture.

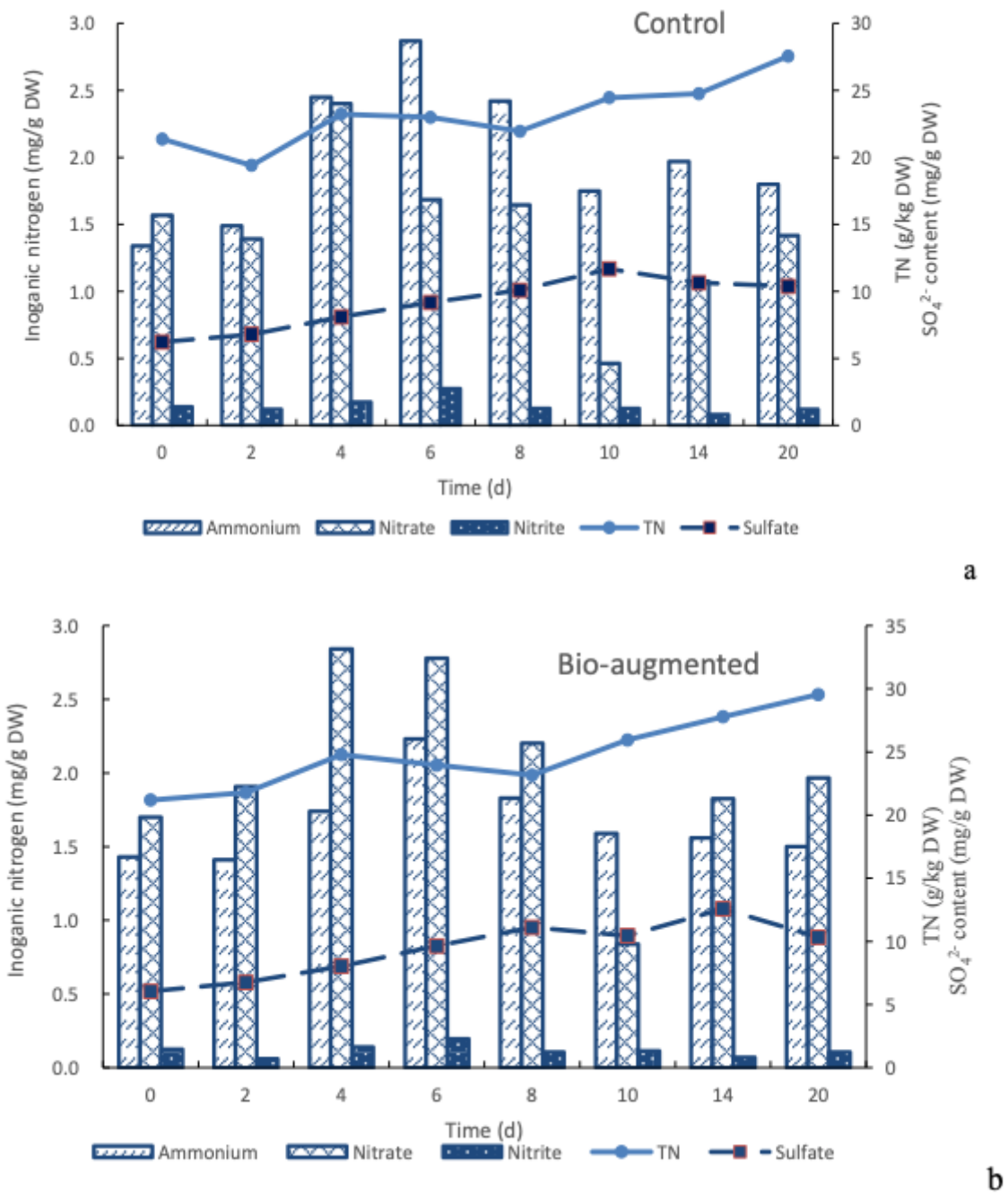

Figure 2

Time-course profiles of the contents of TN, inorganic nitrogen mass and sulfate in the control reactor (a) and the bio-augmented reactor (b)

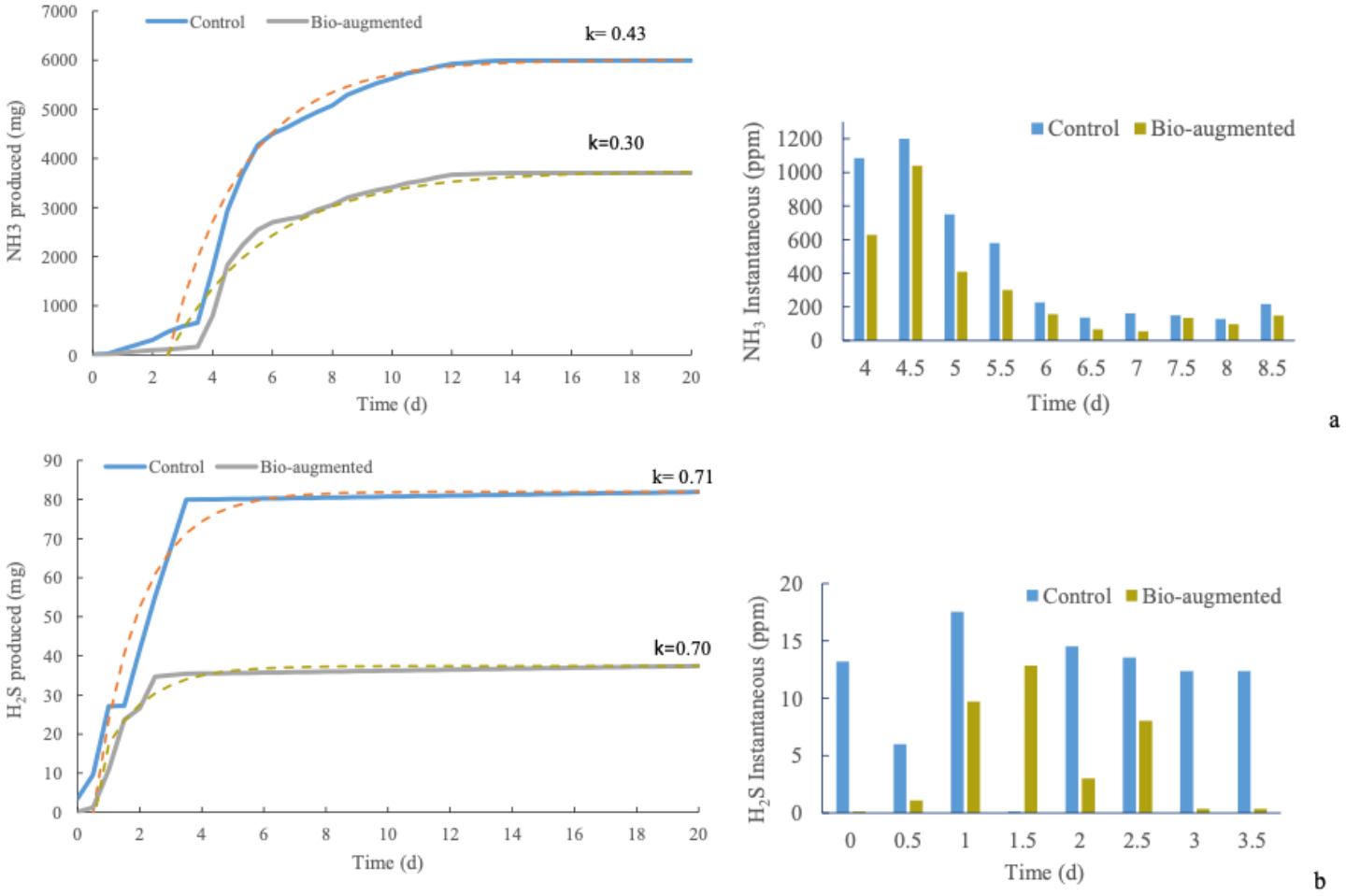

Figure 3

Time-course profiles of NH3 (a) and H2S (b) emission.

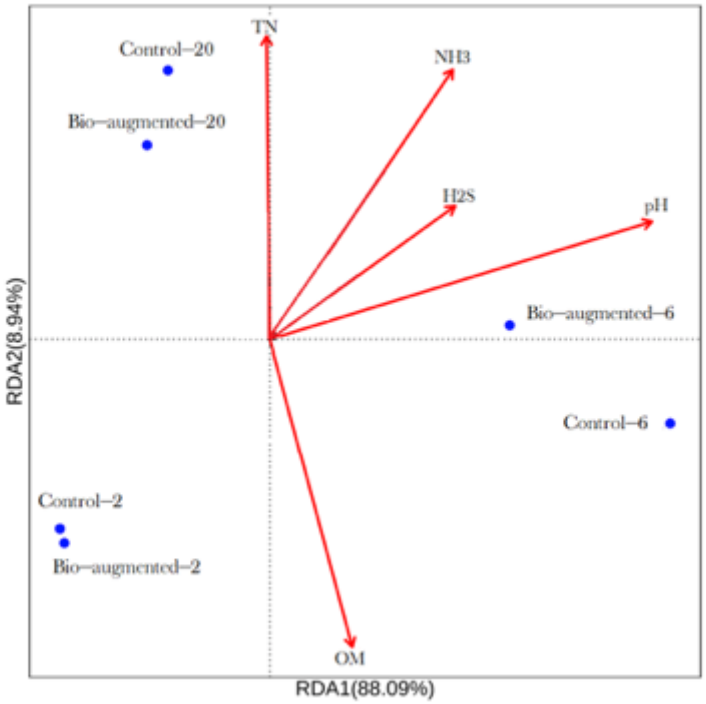

Figure 4

RDA of the relative abundance of bacterial communities and some environmental factors

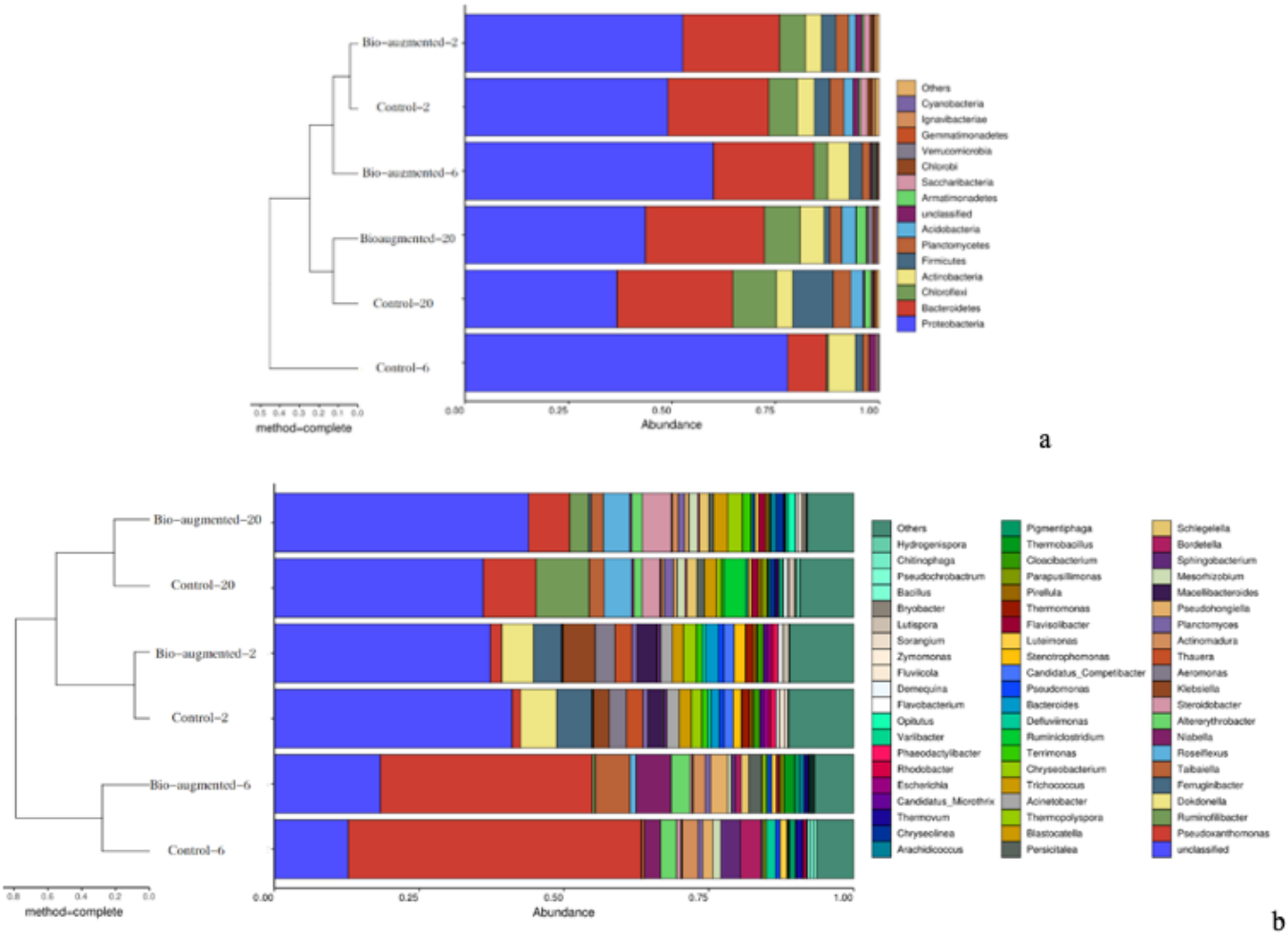

Figure 5

The changes of relative abundance of operational taxonomic units (OTU) of bacterial phyla (a) and genus (b) in different compost phase.