Bioresource Technology 288 (2019) 121517

Contents lists available at ScienceDirect

Bioresource Technology

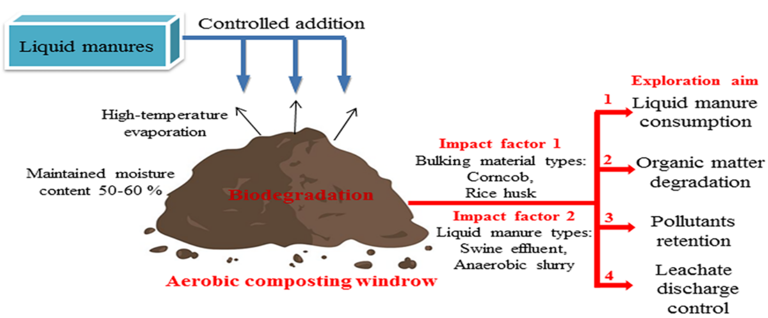
journal homepage: www.elsevier.com/locate/biortech

Effects of bulking material types on water consumption and pollutant degradation in composting process with controlled addition of different liquid manures

Hongyong Fan[a,b,c], Jie Liao[d], Olusegun K. Abass[e], Lin Liu[a,b,c], Xu Huang[a,b,c], Lili Wei[a,b,c], Wei Xie[d], Hengyang Yu[d], Chaoxiang Liu[a,b,c,*]

[a] Key Laboratory of Urban Pollutant Conversion, Institute of Urban Environment, Chinese Academy of Sciences, Xiamen 361021, China
[b] University of Chinese Academy of Sciences, Beijing 100049, China
[c] Ningbo Urban Environment Observation and Research Station-NUEORS, Chinese Academy of Sciences, Ningbo 315800, China
[d] Xiamen University of Technology, Xiamen 361021, China
[e] School of Civil and Environmental Engineering, Nanyang Technological University, Singapore 639798, Singapore

GRAPHICAL ABSTRACT



ARTICLE INFO

Keywords:
Cocomposting
Bulking materials
Water consumption
Liquid manure
Solid manure

ABSTRACT

This study was conducted to examine the effects of different bulking materials (corncob and ricehusk) on liquid manure consumption, organic matter degradation and pollutants retention in composting process under controlled addition of different types of liquid manures (LM). The results indicated that under the controlled addition of LM, bulking materials with higher content of biodegradable carbon (corncob) and LM with a higher concentration of organic pollutants (swine effluent) were more beneficial for biological heat generation and thus were more efficient for water evaporation, organic matter degradation, LM consumption and pollutants retention during the cocomposting process. Consequently, the optimization of these major influencing factors could compensate for efforts geared towards better utilization of the cocomposting process.

1. Introduction

With the rapid growth of the population, the demand for meat has steadily increased. Pork is the most consumed meat in the world, and makes up > 30% of the global meat production, while China remain the largest primary consumers of pork (Jia et al., 2018; Dhyani et al., 2018). Pig breeding now requires larger-scale farming to meet the ever-increasing demand of meat for the populace (Jia et al., 2018). However,

* Corresponding author at: Key Laboratory of Urban Pollutant Conversion, Institute of Urban Environment, Chinese Academy of Sciences, Xiamen 361021, China.
E-mail address: cxliu@iue.ac.cn (C. Liu).

https://doi.org/10.1016/j.biortech.2019.121517
Received 23 March 2019; Received in revised form 16 May 2019; Accepted 17 May 2019
Available online 19 May 2019
0960-8524/

the generation of pig manure and pig breeding wastewater, which are also known as solid manure (SM) and liquid manure (LM, swine effluent and biogas slurry), respectively, has increased largely during the intensive process of pig breeding (Fan et al., 2019). According to statistics, the total production of pig breeding wastes tripled from the 2010 to 2015, with a SM to LM ratio of 3:2 (Jia et al., 2018; Fan et al., 2019). The increasing transformation has caused considerable environmental concern about the water, air and soil, since SM and LM contain high amounts of heavy metals, pathogens, and other environmental pollutants, which cannot be disposed of adequately by the traditional technologies of SM and LM treatment (Othman et al., 2013; Ferreira et al., 2016; Dennehy et al., 2017; Wang et al., 2017a; Jia et al., 2018). Therefore, further research is required to tackle the ever increasing and pollutant intensive pig breeding wastes (SM and LM).

However, SM is rich in potential nutrients (such as nitrogen, phosphates, and potassium) that are necessary for plant growth, and in organic matters for microbiological decomposition. Thus, recycling SM as a resource material for organic fertilizer has been a common form of SM treatment in recent years (Wu et al., 2017; Othman et al., 2013). In particular, among SM-based fertilizer production technologies, aerobic composting has been considered as an efficient and economically applicable technology to ferment SM into safe and stable materials that can be used as nutrient substrates for plant growth. Basically, aerobic composting is an organic pollutant degradation process performed by microorganisms that are influenced by varying factors such as the carbon/nitrogen ratio (C/N), pH and moisture content (MC) (Wu et al., 2017). Extremely large amount of biological heat can be generated from biodegradation activities, which acts as energy to drive the composting process. The accumulation of heat increases the fermentation temperature of windrows. With the fermentation temperature evolution, the composting process usually goes through successive stages including the mesophilic, thermophilic and mature stages (Rosso et al., 1993; Dhyani et al., 2018). The higher temperature (above 50 °C) occurs at the thermophilic stage, during which microbial degradation activities are stimulated or stressed and thus, harmful organisms such as weed seeds and pathogenic bacteria in composting materials are exterminated, causing more water to evaporate (Vázquez et al., 2015). Proper moisture evaporation can remove excess heat and maintain the fermentation temperature for biodegradation activities. However, excessive moisture evaporation during composting process could result in an MC drop from the optimal range of 50–60%, thus leading to disorder in the biofermentation moisture environment, and subsequently inhibited biodegradation activity and a decelerated decomposition rate of organic pollutants (He et al., 2018; Fan et al., 2019). Therefore, controlling the moisture compensation in a composting windrow is necessary to maintain the optimum MC for biofermentation activities during the composting process (Bernal et al., 2009; Fan et al., 2019).

LM, as a kind of pig breeding wastewater is supposed to be treated biochemically due to its high concentration of organic pollutants (Li et al., 2016). Although current biotreatment technologies such as anaerobic digestion, aerobic granular sludge and other integrated technologies can transform SM into reusable water, their applications are limited to some extent by the long-treatment time, high maintenance cost and other disadvantages including engineering practices (Othman et al., 2013; Li et al., 2016; Lu et al., 2019; You et al., 2019; Wang et al., 2019). On the other hand, LM is a great candidate for moisture compensation in SM composting process. Thus, the coaddition of LM during composting is a new and optional practice for pig breeding wastes treatment and the controlled addition of LM into compost is a feasible way to maintain composting efficiency (Fan et al., 2019).

Bulking materials (such as agricultural wastes) used in SM composting are necessary supplements for successful composting (Wang et al., 2017b; Meng et al., 2018). They serve as porous attachment carriers of microorganisms, which are important for better biofermentation environments construction such as mixed material ventilation, C/N enhancement, and excessive MC stabilizers during the composting process (Vázquez et al., 2015; Wu et al., 2017; Fan et al., 2019). The porosity of bulking materials is closely related with the water adsorption rate, since bulking materials with a higher porosity may enhance water consumption during the composting process (Ahn et al., 2009; Vázquez et al., 2015; Niccolò et al., 2017; Fan et al., 2019). In addition, the thermal properties of composting bulking materials may vary according to the difference in the moisture content of mixed materials in windrows, which could produce considerable influence on microorganism activities, biological heat generation and organic matter degradation (Ahn et al., 2009). Thus, systematic selection of the bulking materials type plays an important role in organic pollutants degradation and water consumption in SM aerobic composting.

To date, only few works have been carried out on the effect of bulking material on the cocomposting of SM and LM. In addition, most previous studies on bulking materials addition into SM and LM cocompost were conducted under incidental conditions, and also commonly practiced in a cluttered method of the addition of LM to SM composting. For example, Vázquez et al. (2015) studied the effects of different bulking materials (agricultural wastes) and different volume ratios of manure to bulking materials on water evaporation during the SM composting process with the uncontrolled addition of LM, and observed that the composts in a lower ratio of SM to bulking materials or mixed with easily biodegradable and more energetic bulking materials showed a higher water evaporation rate. However, the most of the windrows MC was higher than 65% and the maximum MC was higher than 75% during the cocomposting process. Thus, irregular temperature stages were observed, which indicated that the high MC (caused by an uncontrolled addition of LM) hindered the aerobic biological activities and aerobic fermentation efficiency in the composting process (Ahn et al., 2009; Guidoni et al., 2018; Zhong et al., 2018).

Studies showing the effect of bulking materials on the cocomposting of SM and LM and their indirect effects on water consumption and organic matter degradation are still lacking. In a recent report, Fan et al. (2019) explored the effects of the particle sizes of rice husks (1 mm and 7 mm) on LM consumption, organic matter degradation and pollutants retention in the SM composting process with the controlled addition of LM, and found that under the controlled addition of LM, bulking materials with small particle sizes are the driving factors for cocomposting efficiency and water evaporation. However, to acquire a more elaborate understanding of the key factors driving the biological heat generation for increased microbial activities other than considering the overall water evaporation in the cocomposting process based on particle sizes only as reported earlier (Fan et al., 2019), the characteristics effect of bulking material types on the cocomposting process will be a more reliable index by which to investigate the reasons mentioned earlier. Likewise, the effect of the volume model on improvement to the aerobic composting efficiency and leachates reduction based on the bulking materials types requires further investigation. Thus, in this study, the volume model for controlled LM addition used in Fan et al. (2019) was verified in SM composting with different types of bulking materials for optimum MC (50–60%) maintenance, aerobic fermentation efficiency improvemnt and leachate reduction. Under this controlled condition of LM addition, the biological heat generation was investigated to understand the contribution of different bulking materials to water evaporation and LM consumption in the cocomposting process. Moreover, the effects of different bulking materials on organic matter degradation, pollutants retention and final composts properties with the utilization of different LMs (swine effluent and biogas slurry) in the SM composting process were evaluated and assessed.

2. Materials and methods

2.1. Experiment configuration

The raw SM and LM were collected from a pig farm with a holding inventory of 3000 pigs. Bulking materials (corncob and rice husk) were crushed to 1 mm for mixtures with SM. The main characteristics of the raw solid materials (SM and bulking materials) and LM are shown in Table 1. The composting windrows were composed of the mixtures of 15 kg SM and 3 kg bulking materials with the initial mixture ratio of 1.2:1 (SM to bulking materials, in dry weight) as shown in Table 1. The initial C/N of windrows were 23 (ricehusk-based windrows) and 24 (corncob-based windrows), which were good for starting the composting process. According to the LM and bulking material types, the compost windrows were classified as swine effluent + corncob (SE-C), swine effluent + rice husk (SE-RH), biogas slurry + corncob (BS-C), biogas slurry + rice husk (BS-RH), the MCs of which were 64.11 ± 0.87%, 64.17 ± 0.65%, 64.01 ± 0.91% and 63.72 ± 1.23%, separately. Windrow composting experiments were carried out in four Styrofoam boxes possessing great thermal insulation quality with the following inner dimensions: 46 × 36.5 × 22.5 cm. Chamber floors with 3% slope were designed for every windrow, and three holes were opened for leachate collection by gravity. Leachates were collected by proper accumulation tanks.

2.2. Sampling, monitoring and analysis

Composting materials were turned manually when the windrows were watered. The composting temperatures were monitored twice daily in the morning and in the afternoon by three thermometric sensors from the surface, middle and bottom of the windrows, respectively. The samples of mixed materials were collected every two days to measure the MC. The windrows weight was measured by an electronic scale before the windrows were watered by LMs. The leachate volumes were measured every day by graduated cylinders.

The MC, pH, electrical conductivity (EC), germination rate index (GI), total carbon (TC), total nitrogen (TN), organic matter (OM), total phosphorus (TP), total potassium (TK) and heavy metals (Cu, Zn, Cr, Pb, Ni) of the mixed materials were measured according to the methods of Bustamante et al. (2013), Wu et al. (2017) and Jiang et al. (2018). The water-soluble biochemical oxygen demand for five days ($WSBOD_5$) of mixed materials was extracted by deionized water using the method described by Sawyer et al. (1995) and detected using the BODTrak method. Additionally, all the parameters of SE and BS were measured and analyzed according to the methods used by Bustamante et al. (2013) and Vázquez et al. (2015). All analyses involved in this study were conducted in triplicate, and all the collected data were analyzed by IBM spss statistics 23.

2.3. Physical calculation and theory model

To control the LM addition to the composting windrows, the volume model used by Fan et al. (2019) was utilized in this work. In this model, the benchmark MC was set at 65% for the purpose of increasing LM consumption and ensuring effective fermentation, which was slightly higher than the optimum range 50–60% of microorganism's degradation activities during SM composting (Zang et al., 2016; Fan et al., 2019). The LM input volumes (V (L)) were governed by the actual weight (M_w (kg)) of windrows, the actual MC (MC_a (%)) of mixed materials and the densities (ρ_{LM} (kg/L)) of LMs, while the LM input time was decided by the actual MC (MC_a (%)) of the mixed materials. The volume model for LM controlled addition was as shown in Eq. (1) (Fan et al., 2019):

$$V = (65\% - MC_a) \times M_w \times 100\% / \rho_{LM} \tag{1}$$

where MC_a is the MC of windrows during composting (%); M_w is the actual weight of windrows before the windrows were watered (kg); ρ_{LM} is the density of LM, $\rho_{BS} = 1.01$ kg/L and $\rho_{SE} = 1.03$ kg/L.

The oxygen requirement amounts and the oxygen consumption rate in composting windrows are great indicators of organic pollutant degradation strength by microbial activities during the aerobic composting process (He et al., 2018). The evolution of water-soluble chemical oxygen demand (WSCOD), volatile solids (VS), and water-soluble BOD_5 ($WSBOD_5$) of mixed materials can reflect the transformation of OM, but $WSBOD_5$ is the most persuasive to reflect the existing concentration of biodegradable degradation substances (biodegradable activities level), which is the main source for biological heat generation during the aerobic composting process (Sawyer et al., 1995; Narita et al., 2005; He et al., 2018). Therefore, the biological heat generation (q_{react} (KJ/d)) of the composting windrows during composting process can be calculated according to $WSBOD_5$ (mg/kg), as shown in Eq. (2) (Sawyer et al., 1995):

$$q_{react} = \alpha \times WSBOD_5 \times M_{OS-D} / \beta \tag{2}$$

where$\alpha$is a constant with a value of 13.8688 (kJ/g); β is a constant with a value of 5 (d); and $M_{OS\text{-}D}$ is the dry weight of organic substrates (kg).

The water evaporation rate of different windrows were embodied by the overall evaporation ratio (OER) and the specific evaporation rate (SER) during the composting process, which represented the water balance situation during the entire composting process and were influenced by the incoming water (the MC of original substrates (OMS) and added LM), the outcoming water (MC of the final substrates (FMS) and leachate) and the dry raw organic substrates (ROS-D) (Fan et al., 2019). OER (m^3/t dry weight) revealed the water consumption rate per

Table 1
Characteristics of composting materials.

Parameter	Pig manure	Rice husk	Corncob	Swine effluent*	Biogas slurry*
Moisture (%)	75.21 ± 0.12	1.2 ± 0.1	1.38 ± 0.2	nd	nd
pH	7.5 ± 0.03	6.83 ± 0.14	5.37 ± 0.12	nd	nd
EC (mS/cm)	2.59 ± 0.07	1.53 ± 0.17	1.28 ± 0.13	nd	nd
OM (%)	79.01 ± 0.18	86.6 ± 0.24	99.23 ± 0.05	nd	nd
COD (g/kg)	371 ± 3.2	nd	nd	4180 ± 4.2	1641 ± 2.7
BOD_5 (g/kg)	225 ± 1.2	nd	nd	2616 ± 5.8	837 ± 3.1
TC (g/kg)	400.02 ± 1.41	419.9 ± 1.2	473.1 ± 1.3	nd	nd
TN (g/kg)	31.07 ± 1.42	4.06 ± 0.9	2.11 ± 0.98	847.3 ± 1.9	692.4 ± 2.1
TK (g/kg)	17.23 ± 1.1	6.25 ± 0.62	5.21 ± 0.71	nd	nd
TP (g/kg)	35.17 ± 3.9	0.41 ± 0.11	0.2 ± 0.04	272.6 ± 2.7	30.01 ± 2.1
Cu (mg/kg)	34.98 ± 2.1	12.31 ± 1.2	11.42 ± 1.8	0.06 ± 0.004	0.04 ± 0.003
Zn (mg/kg)	613.9 ± 20.01	2.41 ± 0.9	0.52 ± 0.11	0.11 ± 0.008	0.06 ± 0.006
Cr (mg/kg)	4.31 ± 0.72	42.3 ± 0.5	2.7 ± 0.1	0.008 ± 0.001	0.006 ± 0.001
Ni (mg/kg)	51.27 ± 8.92	4.62 ± 0.3	–	0.003 ± 0.001	0.002 ± 0.0007
Pb (mg/kg)	43.81 ± 0.2	–	–	0.013 ± 0.005	0.009 ± 0.001
Cd (mg/kg)	0.27 ± 0.1	–	–	–	–

Note: nd: not detected.
* mg/L.

dry weight of raw organic substrates, and *SER* (L/t dry weight·d) represented the water consumption rate per dry weight of raw organic substrates per day, the calculations of which were as follows in Eqs. (3) and (4) (Fan et al., 2019):

$$OER = (M_{OES} + M_{LM} - M_{FMS} - M_{leachate})/M_{ROS-D} \quad (3)$$

$$SER = OER/t \quad (4)$$

where M_{OES} is the weight of MC of original substrates (kg); M_{LM} is the weight of added LM (kg); M_{FMS} is the weight of MC of final substrates (kg); $M_{leachate}$ is the weight of generated leachate (kg); M_{ROS-D} is the dry weight of raw organic substrates (kg); and t is the time period of composting days, which was 28 days.

However, OER_{LM} (m^3/t dry weight) and SER_{LM} (L/t dry weight·d) were used for the evaluation of the LM-only, and the consumption during the composting process was carried out as follows in Eqs. (5), (6) (Fan et al., 2019):

$$OER_{LM} = (M_{LM} - M_{leachate})/M_{ROS-D} \quad (5)$$

$$SER_{LM} = OER_{LM}/t \quad (6)$$

3. Results and discussion

3.1. Temperature evolution

The temperatures in all composting process showed similar evolution way. The first mesophilic stage for all composts was very short, and the temperature increased sharply which then reached the thermophilic value (> 50 °C) after first 2 days' composting. During the thermophilic stage, all composts were manually turned for 2 days and watered with LM almost 5 times during the total watering times (8 times) in the cocomposting process. Therefore, the thermophilic stage was the main phase for water loss and microbial degradation for all composts. Following the thermophilic stage, the average temperatures in all composts dropped to 40–50 °C, first reaching a mesophilic value and then reaching the second mesophilic stage. After the second mesophilic stage, the average temperatures in all composts dropped to 30–40 °C, which were close to ambient temperature (30–37 °C), which suggested that biological activities were limited as the mixtures stabilized (Wang et al., 2017a, Zhong et al., 2018). In the final two stages, the turning frequency was adjusted to once every 5–10 days and windrows were watered 3 times with LM. After the composts were turned and watered, the temperature in all composts showed temporary fluctuations. This shows that the addition of LM could significantly affect the temperature evolution of composts. A similar finding was reported in the treatment of compost with SM and rice husks in different particle sizes, which used LM as supplementary water during the composting process (Fan et al., 2019).

The maximum temperatures in corncob-based composts were all above 65 °C, higher than that in ricehusk-based composts (61.5–64.5 °C) as indicated in Table 2. The thermophilic stage of corncob-based composts reached 15 days during the whole 28 days of composting, which was much longer than ricehusk-based composts that took 10 days (Table 2). This indicated that corncob-based windrows were better promoted to be biodegraded than RH-based windrows (BS-RH and SE-RH), which revealed that bulking materials with richer carbon maintained thermophilic conditions during the compost process (Table 1) (Zang et al., 2016; Zhong et al., 2018). The second mesophilic stage duration for rice husk-based composts (10–12 days) occupied almost 1/3 of the whole compost period (Table 2), while the corncob-based composts (2.5 days) only occupied nearly 1/12 of the whole compost period (Table 2), which proved that the mesophilic stage was another important stage for ricehusk-based composts to degrade organic polltants. However, the second mesophilic stage of corncob-based composts was shorter than that of ricehusk-based composts. This may be because the degradation of organic pollutants in the composts was nearly completed in the thermophilic stage. Furthermore, the first convergent time for the corncob-based composting temperature and the ambient temperature was the 22th day, while the first converge time for RH-based windrows was the 24th day as shown in Fig. 1 and Table 2. The difference between the first convergent times of the temperatures indicated that bulking materials with richer carbon contents were more favorable to improve organic pollutants biodegradation efficiency, which resulted in earlier maturation for corncob-based windrows than for RH-based windrows.

With different types of bulking materials mixed in composting windrows, the average thermophilic temperature and the maximum composting temperature for SE-based windrows (SE-C, SE-RH) were 0.5–1 °C and 0–2 °C higher, respectively, than that of BS-based windrows using the same type of bulking materials (Table 2). Meanwhile, the thermophilic durations (10 ± 0.4–15 ± 0.7 days) in SE-based windrows were zero to one day longer than that of the BS-based windrows (10 ± 0.3–14 ± 0.5 days). In a recent report, Fan et al. (2019) compared the composts that were watered by SE to composts that were watered by BS with the same particle size of rice husk in the controlled addition of LM. They observed that the average temperature, the maximum temperature and the thermophilic duration in windrows watered by SE were higher than windrows watered by BS. Therefore, in accord with the results from the preceding work, the present work generalized that LM with a comparatively heavier concentration of pollutants (SE) could be more suitable for thermophilic temperature maintenance, due to SE playing possible role of inoculation to promote microbial activities during the cocomposting process (Wang et al., 2017b; Sánchez et al., 2017; Fan et al., 2019). In addition, pathogens and seeds will be exterminated when the thermophilic duration is longer than 3 days (Wu et al., 2017). Thus, the final composts from SE-

Table 2
The characteristics of composting temperature in different windrows.

Type	Thermophilic stage (above 50 °C, d)	Mesophilic stage (40–50 °C, d)	T_{avg} (°C)	T_{max} (°C)	The first convergent time*
BS-C	14 ± 0.5	2.5 ± 0.3	59.7 ± 5.7	67.5 ± 0.5	22th
SE-C	15 ± 0.7	2.5 ± 0.4	60.2 ± 5.3	69.5 ± 0.6	22th
BS-RH	10 ± 0.3	12 ± 0.6	56.4 ± 3.9	61.5 ± 0.4	24th
SE-RH	10 ± 0.4	10 ± 0.4	57.4 ± 4.1	63.5 ± 0.5	24th

* The first convergent time between the composting temperature and ambient temperature for all composts.

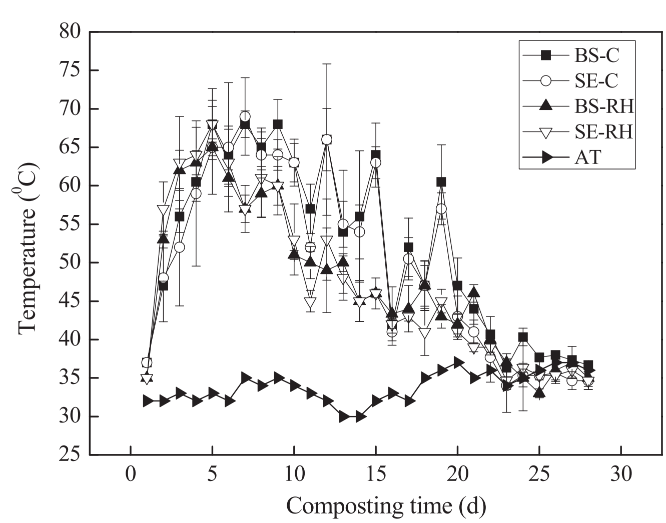


Fig. 1. Temperature evolution of windrows with different bulking materials and LM types during composting process.

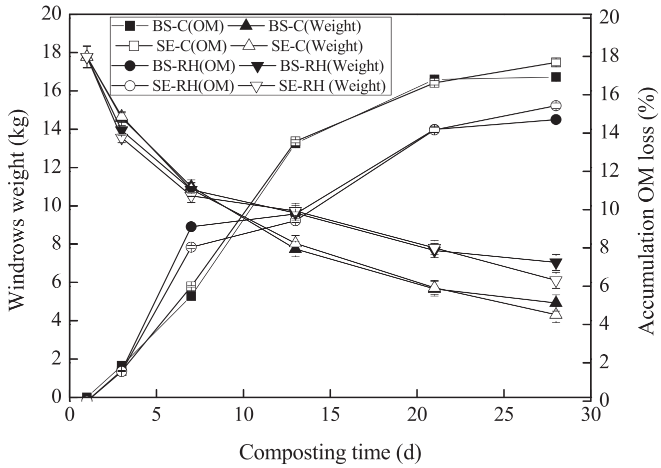


Fig. 2. Windrows weight characteristics and cumulative OM loss during composting process.

based windrows and BS-based windrows all met the sanitation requirements.

3.2. Weight and OM evolution

The evolution trends of the weight of all composts generally declined with composts proceeding as shown in Fig. 2. The OM and windrows weights showed significant positive correlations in respective windrows ($r > 0.8$, $p < 0.04$), which revealed that the reduction in windrow weight mainly resulted from OM loss with the 50–60% of MC maintenance in respective windrows. The result was similar to the finding in SM composting with different particle sizes of rice husks carried out by Fan et al. (2019), which indicated that under the controlled addition of LM, the OM loss played a major part in the reduction of windrows weight and was not restricted by bulking material types and particle sizes.

The thermophilic stage in the composting process revealed the highest reductions in OM and windrows weight for all composts (Fig. 2). However, the reduction of OM and windrows weight in corncob-based windrows (SE-C, BS-C) during the thermophilic stage accounted for 80% of the total reduction in the whole composting period, while the reduction in OM and the windrows weight in rice husk-based windrows accounted for merely 60% of the total, and the other 40% was mainly in the mesophilic stage. This depicted that the thermophilic stage was the major phase for OM degradation in the corncob-based composting process, and the thermophilic stage and the mesophilic stage were the main phases for OM degradation in ricehusk-based composting (SE-RH, BS-RH). The main phase differences of OM degradation may exert an influence on the consuming time and volumes of added LM. In addition, data from Fig. 2 suggested that the final reduction rates in the weight and OM of corncob-based windrows were higher than that of RH-based windrows. This indicated that corncob-based windrows were more favorable for OM degradation than were RH-based windrows, which may be the result of greater bioavailable carbon offered by corncob for OM degradation than that by rice husk (Table 1), and the higher content of silicon in rice husk was inhibited to some extent microorganism activities (Shen, 2017).

During the thermophilic stage, the OM reduction rate in SE-based windrows were 1.32 ± 0.04–2.42 ± 0.07% higher than that of the corresponding BS-based windrows using the same type of bulking materials (Fig. 2). Hence, LM types exerted an influence on OM degradation during the composting process. However, the reduction rate of OM in corncob-based windrows was 13.24 ± 0.32–14.34 ± 0.27% higher than that of the corresponding RH-based windrows using the same type of LM (Fig. 2), which presented a larger improvement in OM degradation than that between SE-based windrows and the corresponding BS-based windrows. Therefore, the differences in the degradation efficiencies indicated that bulking material types had a greater influence on OM degradation than that of LM types.

3.3. Addition of LM and leachate generation

Under the controlled addition of LM, MCs of all composts showed fluctuating evolution during the composting process, as shown in Fig. 3. However, the thermophilic stage was the main phase for LM consumption, in which the volumes of LM consumption ranged from 5.9 to 7.4 L, as was described by Eq. (1), thus occupying over 80% of the total LM consumption amounts for all windrows during the whole composting process. Furthermore, significant positive correlations ($r > 0.8$, $p < 0.04$) were shown between the volumes of LM utilized and the thermophilic duration for respective windrows, which may be due to the more severe moisture evaporation loss that is caused by thermophilic temperature (> 50 °C) during the composting process. With the degradation of organic pollutants, the weight of the windrows greatly reduced in the thermophilic stage and became stable in the second mesophilic stage and in the mature stage. A decrease in LM in the composting windrows was observed when LM was added based on the windrows weights and MCs (Fig. 3). In addition, under the controlled addition of LM following Eq. (1), the total added LM volumes for corncob-based windrows were higher than ricehusk-based windrows,

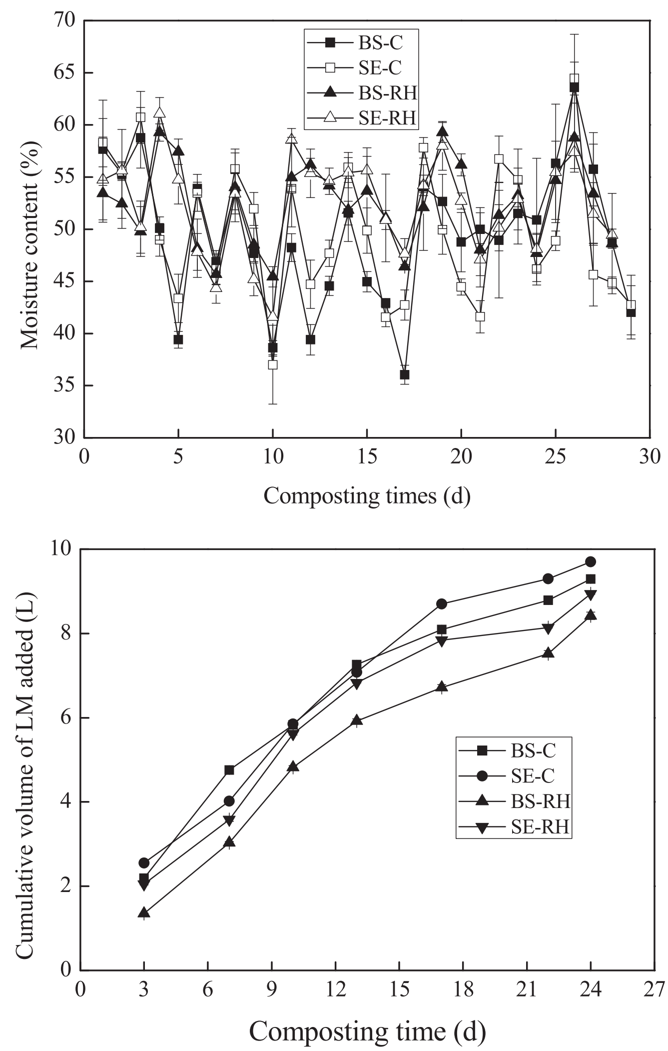


Fig. 3. Moisture content of different windrows and cumulative volumes of LM added during composting process.

and the total added LM volumes for SE-based windrows were also higher than that of BS-based windrows (Fig. 3). This was so because of the longer thermophilic durations in corncob-based windrows and SE-based windrows compared to those of ricehusk-based windrows and BS-based windrows, respectively (Table 2). Hence, bulking material types and LM types showed greater function in thermophilic stage maintenance and LM consumption during the composting process.

However, numerous studies have shown that the optimal moisture range for aerobic composting is 50–60%, which is good for the biodegradation of organic pollutants (Bustamante et al., 2013; Zhong et al., 2018; Fan et al., 2019). In this study, most of the moisture contents during the composting process were not > 65% and were maintained in the optimal moisture content range with the controlled addition of LMs (Fig. 3). The density increase caused by the added LM in controlled settings is negligible and will not produce a significant influence on microorganism degradation activities. Therefore, the control addition of LM in the different windrows will give rise to the adequate consumption of LM, thus minimizing density increment in the windrows.

The leachates generated were the mixtures of LM and solid mixed materials, which were produced by the gravity after windrows were watered, and the characteristics were similar to the SE in this research ($\rho_{\text{leachate average}} = 1.38 \pm 0.2$ kg/L). Under the controlled addition of LM using Eq. (1), the maximum volume of generated leachate among four windrows was < 0.12 L, and the leachate generation rate was 0.03–0.13 L/m^3·d in this work, which was greatly lower than that of 0.63–1.39 L/m^3·d in the research conducted by Vázquez et al. (2015), but was similar to that of 0.03–0.1 L/m·d in the work of Fan et al. (2019). This is based on the fact that so much LM was added into the windrows without being controlled in the work of Vázquez et al. (2015), which has caused too much leachates generation and aerobic fermentation efficiency inhibition due to the high MC of windrows (58–75%); at the same time, the LM addition was controlled according to the actual MC and the windrows weight with the benchmarked MC 65% (the MC of windrows needed to be adjusted when the MC was lower than 50%) to maintain the MC of windrows in the range of 50–60% in the present study and in the work of Fan et al. (2019), which could be consumed during the composting process. All composts successively went through the mesophilic stage, the thermophilic stage, the second mesophilic stage and the mature stage during the whole process, which indicated that the calculated control of LM addition had not produced a negative influence on the SM aerobic composting efficiency. Thus, not only was a reduction in the volume of leachates generated achieved, but the LM consumption and aerobic fermentation efficiency were also improved in the cocomposting process using different bulking materials.

3.4. Biological heat generation and water consumption

The evolutions of biological heat in all composts during the composting process were similar to those of temperature, as shown in Fig. 4a. Using the biological heat model described in Eq. (2), the amounts of the biological heat that were generated showed significant positive correlations with the temperature ($r > 0.85$, $p < 0.05$). The maximum biological heat of all composts was > 600 KJ/d and appeared in the thermophilic stages. The biological heat that was generated in corncob-based windrows were 11 ± 0.3–40 ± 0.5% higher than that in ricehusk-based windrows in the first mesophilic stage and in the thermophilic stage, but 10 ± 0.6–50 ± 0.8% was lower than that in ricehusk-based windrows in the second mesophilic stage and in the mature stage. This pointed out that mixed materials in windrows using corncob as bulking material better benefited for microbiological activities for the degradation of OM than was windrows using ricehusk as a bulking material. Simultaneously, it provided more support for the reason for the main phase differences in OM degradation between corncob-based windrows and ricehusk-based windrows, as was discussed previously in Section 3.2, which may produce great influences

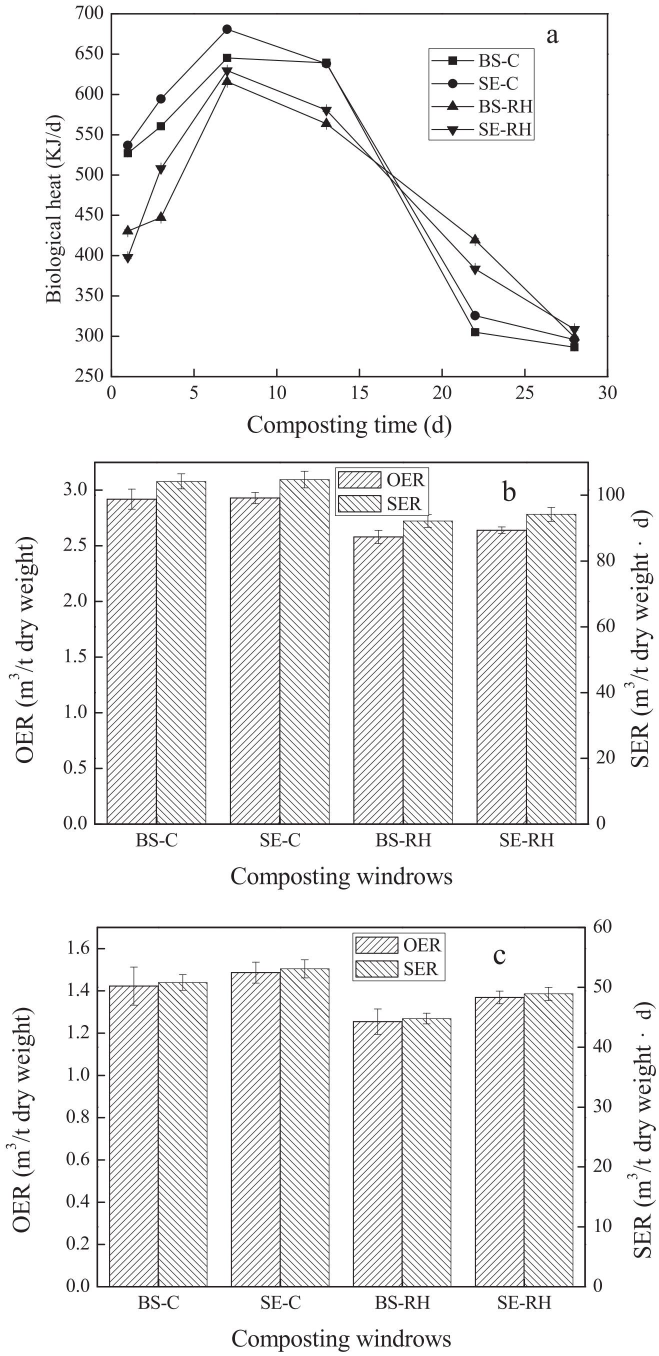


Fig. 4. The characteristics of (a) biological heat generation, (b) OER and SER for water balance during the whole composting process and (c) OER and SER for LM-only consumption during the composting process.

on water evaporation and LM consumption. However, the biological heat that was generated in SE-based windrows was 0.2 ± 0.004–25 ± 0.7% higher than that of BS-based windrows during the composting process, which may be because of the addition of SE which transferred a higher concentration of bioavailable organic pollutants and thus more $WSBOD_5$ was produced during the composting process than BS, which led to the improvement in biodegradation activities.

The calculation of *OER* and *SER* were carried out according to the weight balance of incoming water, outgoing water and dry raw organic substrates during the whole composting process, as described by Eqs. (4) and (5), which fully presented the water balance fully for the entire composting period. Significant positive correlations ($r > 0.85$,

$p < 0.05$) were shown between the amounts of biological heat that were generated and the water loss rate (*OER* and *SER*) for windrows in this study. According to the description of Eqs. (4) and (5), the *OER* of corncob-based windrows ranged from 2.82 ± 0.02 m^3/t dry weight to 2.93 ± 0.04 m^3/t dry weight, which was 9.3 ± 0.2–12.4 ± 0.1% higher than ricehusk-based windrows, while the *SER* ranged from 103.23 ± 0.5 t/dry weight·d to 104.77 ± 0.6 L/t dry weight·d, which was 11.24 ± 0.65–12.11 ± 0.45% higher than that of ricehusk-based composts (Fig. 4). This was basically because corncob contains higher content of carbon, which makes it more biodegradable and favors an increase in heat generation compared to ricehusk, and can maintain a longer thermophilic duration that leads to improvement in water evaporation (Blanes et al., 2009). Under the controlled addition of LM, the maximum *OER* and *SER* reached 2.93 ± 0.04 m^3/t dry weight and 104.77 ± 0.6 L/t dry weight·d, which appeared in corncob (1 mm)-based windrows during the whole composting period of 28 days in this study, respectively (Fig. 4b). However, under a similar controlled addition of LM, Fan et al. (2019) observed that the maximum *OER* and *SER* reached 2.64 ± 0.01 m^3/t dry weight and 94.2 ± 0.32 L/t dry weight·d, which appeared in ricehusk (1 mm)-based windrows during the whole composting time of 28 days, respectively. As was suggested earlier by Fan et al. (2019), under the controlled addition of LM, the effect of the process of influencing factors (including bulking material types) on water evaporation rates was crucial content yet to be explored. Thus, in the controlled addition of LM, bulking material types with higher content of bioavailable carbon (such as corncob) were noticeably advantageous to improve the evaporation rates of windrows during the cocomposting process of SM and LM.

However, the *OER* and *SER* regarding the water balance analysis contained the original moisture of SM (MC of 70–80%), as incoming water accurately cannot reflect the consumption rate of added LM during the cocomposting process (Fan et al., 2019). Hence, the analyses of OER_{LM} and SER_{LM} for the precise evaluation of added LM consumption during the composting process needs to be carried out by Eqs. (5) and (6). As described in Eqs. (5) and (6), the OER_{LM} in corncob-based composts reached 1.42 ± 0.01–1.49 ± 0.03 m^3/t dry weight, which was 9.3 ± 0.2–11.1 ± 0.1% higher than that in ricehusk-based composting; and the *SER* reached 50.8 ± 0.3–53.9 ± 0.2 L/t dry weight·d, which was 10.25 ± 0.7–13.45 ± 0.3% higher than that of ricehusk-based composts (Fig. 4c). Meanwhile, the range of *OER* in SE-based windrows was 1.37 ± 0.02–1.49 ± 0.03 m^3/t dry weight, which was 2.3 ± 0.3–3.9 ± 0.1% higher than BS-based windrows, and the *SER* of SE-based windrows were 48.89 ± 0.42–53.9 ± 0.2 L/t dry weight·d, which was 6.1 ± 0.3–9.3 ± 0.1% higher than that of BS-based windrows (Fig. 4). Consequently, the bulking material types and the added LM types had significant influence on LM consumption during the composting process. However, compared with the extent of influence on LM consumption, bulking material types showed greater function in LM consumption than did LM types with the controlled addition of LM.

In accordance with empirical statistics, approximately 115 t of LM and approximately 75 t of SM with an average MC of 75% could be produced in a pig farm for every ten thousand pigs that were bred (Fan et al., 2019). Based on the maximum SER_{LM} (53.9 ± 0.2 L/t dry weight·d) in this study, a compost mixed with SM and corncob in a ratio of 1.2:1 (in dry weight) and carbon and nitrogen ratio of 24 could consume approximately 1.91 t/d of LM for a ten thousand pigs breeding farm. Although compared with the total volumes of LM produced, the consumption rate cannot meet the requirements of LM consumption, the consumption rate rose 5% on the basis of LM consumption in the work of Fan et al. (2019), which used ricehusk as bulking material. Hence, great amounts of LM could be consumed by aerobic composting, and the improvement of LM consumption could be achieved by the optimization of significant influencing factors, which would be an interesting trend for further research on cocomposting.

3.5. Pollutants retention and final composts properties

The total amounts of pollutants (potential nutrients and heavy metals) retention (mg) from LM (SE and BS) during the composting process were the product of the LM added volumes (L) and the pollutant concentrations (mg/L) of LM detected when LM was added to the composting windrows (Fan et al., 2019). In this study, the total retaining weights of TN, $NH_4{}^+$-N, $NO_3{}^-$-N, TP and $PO_4{}^{3-}$-P from LM in corncob-based windrows were 19 ± 0.31–25 ± 0.43%, 19 ± 0.23–28 ± 0.37%, 14 ± 0.3–30 ± 0.45%, 23 ± 0.91–70 ± 0.69% and 20 ± 0.7–26 ± 0.62%, respectively, which were higher than those in ricehusk-based windrows (Fig. 5a). After fitting analysis, the total retaining weights of potential nutrients from LM showed significant positive correlations with the amounts of biological heat generation ($r > 0.85$, $p < 0.04$) and with the duration of the thermophilic stage ($r > 0.8$, $p < 0.05$). Therefore, the dissimilarity of the potential nutrient retention between both corncob-based composts and ricehusk-based composts maybe the result of the generation of greater biological heat and the maintenance of a longer thermophilic stage, which was more favorable for LM consumption in corncob-based windrows than ricehusk-based windrows. Moreover, the total retaining weights of TN, $NH_4{}^+$-N, $NO_3{}^-$-N, TP and $PO_4{}^{3-}$-P in SE-based windrows were 8 ± 0.43–27 ± 1.31%, 5 ± 0.38–20 ± 1.1%,10 ± 0.3–32 ± 0.62%, 82 ± 1.33–110 ± 2.47% and 52 ± 2.45–74 ± 1.72%, respectively, which was higher than that in BS-based composts (Fig. 5a), which verified one conclusion that was observed by Fan

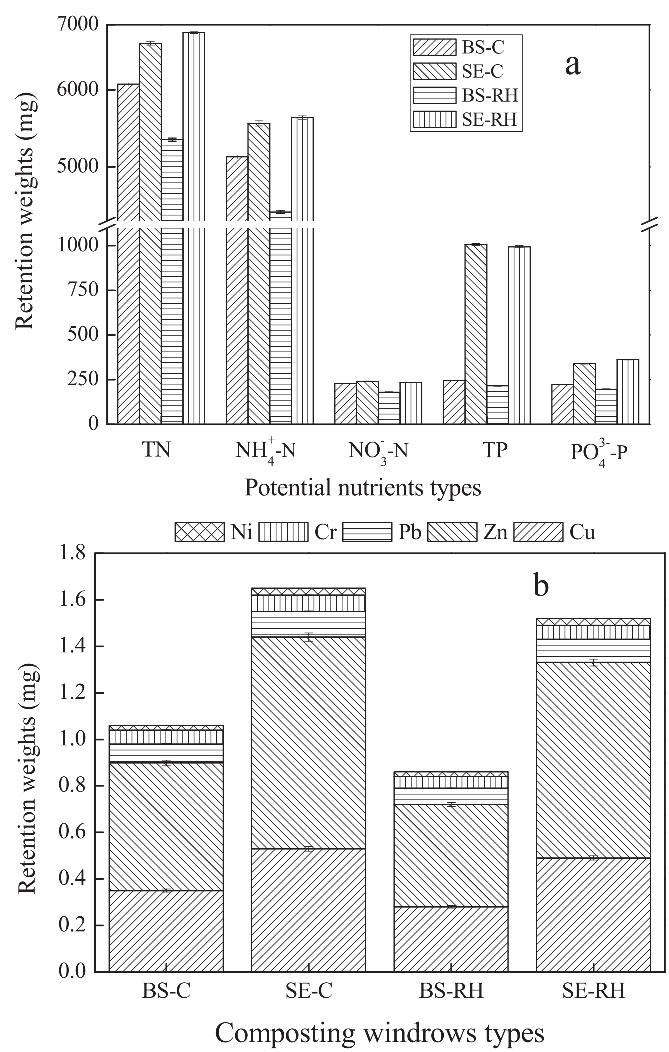


Fig. 5. The retention weights of (a) potential nutrients and (b) heavy metals.

et al. (2019), in that the added LM type was an important factor of the influence of nutrient retention and SE was the more advantageous LM type for potential nutrients retention than was BS.

Subsequently, the amounts of retained nutrients from the LM added into the composts showed significant positive correlations with the concentrations of nutrients in the final composts for respective windrows (r > 0.8, p < 0.05). As was vividly observed in Fig. 5, the maximum contributed concentrations of TP and PO_4^{3-} for the final composts were obtained from SE-C windrows, thus reaching 327.31 ± 3.9 mg/kg dry weight and 110.53 ± 2.3 mg/kg dry weight, respectively, which was 43.1 ± 1.31–261.5 ± 3.22% higher than that in other composts (Fig. 6a). SE-C windrow reached the maximum amount of nutrient retention before other windrows did due to the favorable factors (as discussed in Sections 3.2 and 3.3) that were assocaited with the windrow, especially the LM consumption rate during the composting process. Thus, the nutrients retention from the added LM made a contribution to the improvement of nutrient concentration in the final composts. Similarly, the contributed concentrations of Cu, Zn, Cr, Ni and Pb for the final composts were also obtained from SE-C windrows, thus reaching 0.17 ± 0.021, 0.3 ± 0.02, 0.023 ± 0.006, 0.01 ± 0.002, and 0.04 ± 0.001 mg/kg dry weight, respectively (Fig.s 5b, 6b). In addition, heavy metals could be concentrated after composting due to their refractory nature (Wang et al., 2016, Fan et al., 2019). According to the data shown in Fig. 2 of the manuscript, the

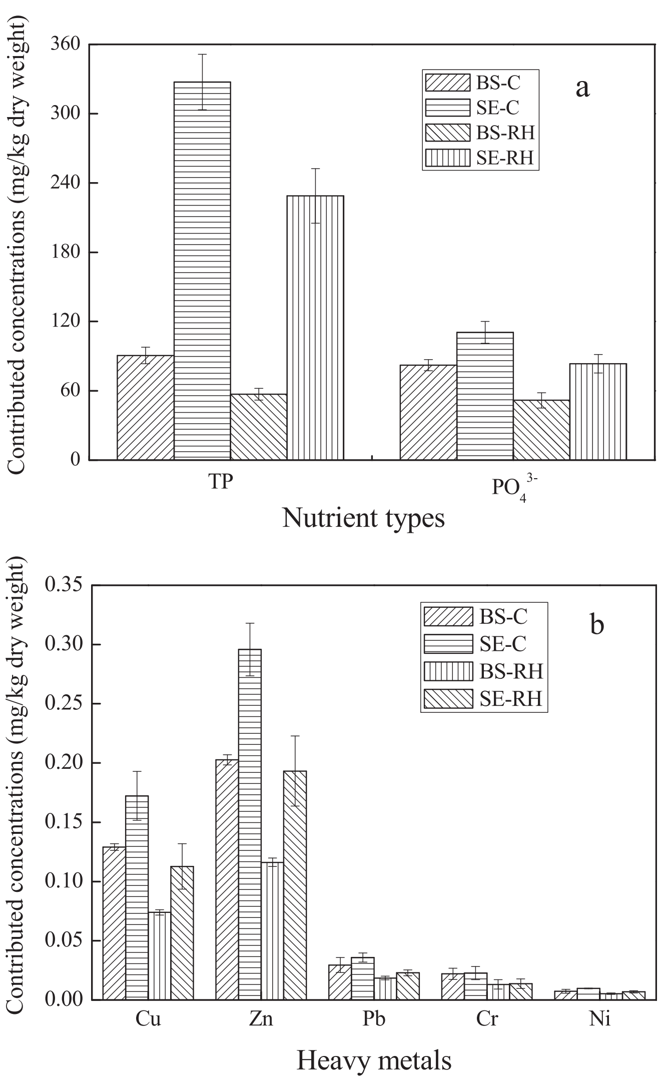


Fig. 6. The contributed concentrations of (a) potential nutrients retention about TP, PO_4^{3-} and (b) heavy metals immobilized for final composts.

Table 3
Characteristics of final composts.

Windrows	pH	EC (mS/cm)	TN (g/kg)	TP (g/kg)	TK (g/kg)	Cu (mg/kg)	Zn (mg/kg)	Cr (mg/kg)	Pb (mg/kg)	Ni (mg/kg)	GI (%)
BS-C	8.05 ± 0.04	2.27 ± 0.01	37.18 ± 1.2	27.16 ± 1.1	19.35 ± 0.5	55.34 ± 2.41	1010.4 ± 27	2.54 ± 0.05	3.19 ± 0.02	2.34 ± 0.13	82.19 ± 0.45
SE-C	7.96 ± 0.03	2.71 ± 0.02	36.51 ± 1.6	29.75 ± 2.1	20.12 ± 1.4	56.22 ± 6.3	1080.1 ± 21	2.61 ± 0.23	3.38 ± 0.13	2.69 ± 0.11	81.38 ± 0.73
BS-RH	7.42 ± 0.03	2.74 ± 0.01	29.9 ± 0.7	26.71 ± 0.9	18.79 ± 0.75	50.72 ± 1.35	970.3 ± 13.3	1.27 ± 0.17	2.89 ± 0.08	2.23 ± 0.04	83.39 ± 0.68
SE-RH	7.28 ± 0.05	3.13 ± 0.03	24.37 ± 0.4	28.21 ± 1.4	19.78 ± 0.37	53.11 ± 3.52	1040.2 ± 30.1	1.59 ± 0.26	3.07 ± 0.06	2.67 ± 0.01	82.64 ± 0.91

final reduction rate of the weight and OM in corncob-based windrows (BS-C, SE-C) were higher than that of RH-based windrows (BS-RH, SE-RH), which indicated that the concentrations of heavy metals in the final corncob-based windrows were more concentrated compared to that of rice husk-based windrows. Therefore, although the concentration of heavy metals in raw rice husk is higher than that of raw corncob, the concentration of heavy metals in corncob-based windrows (BS-C and SE-C) is higher than rice husk-based windrows (BS-RH and SE-RH) in the final composts. However, although heavy metal concentrations in final composts could be increased through the pollutants retention, the risk could be reduced or avoided by the preparation work for the original materials, such as heavy metal passivation and adsorption removal from SM and LM (Wang et al., 2016; Zeng et al., 2018). Furthermore, the GI of the final compost from SE-C was 0.95 ± 0.003–2.03 ± 0.014% less than that of other composts, but the GI of all final composts was over 80% in this work, which indicated that the phytotoxicity of the final composts was obviated (Table 3) (Meng et al., 2018). According to the requirements of the organic fertilizer standard in China (NY525-2012) (Ministry of Agriculture of the People's Republic of China, 2012), the final composts from different windrows could be utilized as organic fertilizer for plant growth. Therefore, under the controlled addition of LM, the selected bulking materials and LM type were critical for better application of the co-composting process with nutrient retention enhancement and better soil-fertilizing agent production.

4. Conclusion

The study finds that regardless of the different bulking materials, the controlled addition of LM is key to improving the efficiency of composting and aided the reduction of leachate, which was commonly generated during the cocomposting process. Under the controlled addition of LM to SM composting, biodegradable bulking materials with richer bioavailable carbon content and SE-based LM types were significant factors for improving organic pollutants degradation, LM consumption, and pollutants retention compared to that of the other bulking materials and LM types that were investigated during the co-composting process. Thus, this work showed that the optimization of factors that influences the cocomposting process is vital for its practical field application.

Acknowledgements

This research has received funding from the STS project from Science and Technology Department in Fujian [2016T3006], Key Research and Development Program Social Development Project in Zhejiang [2015C03009], Science and Technology Planning Project of Fujian Province [2019T3023].

Declaration of Competing Interest

There is no conflict of interest.

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