Bioresource Technology 275 (2019) 61–69

Contents lists available at ScienceDirect

Bioresource Technology

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

Effects of compost characteristics on nutrient retention and simultaneous pollutant immobilization and degradation during co-composting process

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ARTICLE INFO

Keywords:
Bulking material sizes
Co-composting
Liquid manure
Swine effluent
Water balance

ABSTRACT

This study was conducted to examine the effects of controlled addition of liquid (LM) to solid (SM) manure compost using a volume-model technique on the co-composting of SM and LM, and further to investigate the major effects of bulking material sizes and LM types on the co-composting process and final compost characteristics. Results indicated that this volume-model technique played a critical role in reducing leachate generation and improving the overall efficiency of the co-composting process. Specifically, the developed model enhanced the evaporation rates of windrows during the co-composting process. For improved final compost properties, small bulking materials and swine-effluent-based LM were found to be more efficient for organic matter degradation, LM consumption, hazardous metals immobilization, and essential nutrients retention than large bulking materials and biogas-based LM. Thus, process parameter optimizations represent major research options for successful co-composting applications for the future.

1. Introduction

Due to the demand for meat, eggs and milk, livestocks and poultry have skyrocketed in recent decades and the need for more professional and improved livestocks breeding at a large scale has become an inevitable trend. According to recent statistics, in China, pork production accounts for approximately 63% of total meat production, and swine breeding occupies the principal livestock market (Dhyani et al., 2018). This situation has led to an increase in the generation of livestock and poultry slurry, which mainly consists of animal manures and breeding wastewater, otherwise called solid (SM) and liquid (LM) manure, respectively. According to empirical statistics, 190 t/d of livestock breeding waste is produced by every 10,000 pigs breeding in pig farms, including approximately 40% SM and approximately 60% swine effluent (SE), which has caused significant environmental perturbations, such as water and soil pollution and atmospheric contamination (Bustamante et al., 2013; Dennehy et al., 2017). Moreover, large-scale livestock and poultry breeding farms have transitioned from formerly remote locations to suburban districts or urban and rural integration districts due to increasing human populations, which has inadvertently increased pressure on urban-pollution control facilities. Thus, the need to upgrade existing technologies to manage wastes from the livestock and poultry breeding industry is an urgent concern.

However, SM is a good resource for plant growth because of its abundant nutrients such as nitrogen, phosphate, potassium, humus, etc. Therefore, recycling SM as fertilizer, fodder and energy-producing raw material would enable a circular economy to be achieved (Dhyani et al., 2018). Livestock and poultry breeding wastewaters often have high concentrations of organic pollutants, ammonia nitrogen, and suspended materials, which could be treated biochemically (Li et al., 2016). Other bio-treatment techniques can treat wastewater and slurry by transforming them into reusable products such as aerobic granular sludge, constructed wetlands, membrane bioreactors and other integrated technologies (Zhang et al., 2011; Othman et al., 2013; Abass et al., 2015; Liu et al., 2015; Wang et al., 2017; Abass et al., 2018; Cheng et al., 2018; Wang et al., 2018). Although these technologies are attractive, the long retention time required, high cost of maintenance, high sludge production, and inconsistency in treatment performance limit their applications. Therefore, treatment of livestock manure and slurry using technologies that have low cost, are easy to operate and are

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https://doi.org/10.1016/j.biortech.2018.12.049
Received 16 October 2018; Received in revised form 14 December 2018; Accepted 15 December 2018
Available online 16 December 2018
0960-8524/

environmentally friendly will become a major research trend and composting is one such option.

Composting has been a major technology during the past few decades. It is used in converting livestock manure and slurry waste into nutrient-rich biofertilizers, thus making the waste bioavailable to plants. Unlike other treatment technologies, the energy required to drive the composting process is derived from microbial activities, which generates heat at different stages of substrate utilization depending on factors such as pH, moisture content (MC), and C/N ratios (Bernal et al., 2009). Due to different microbial activities, the composting process generally proceeds in three phases: heating, stability, and cooling (Dhyani et al., 2018). At higher temperatures (50–70 °C), microbial fermentation is accelerated and weed seeds present are easily disintegrated along with pathogenic microorganisms, resulting in water loss through evaporation (Vázquez et al., 2015). Moisture evaporation is a vital phase during composting because it can remove excess heat and moderate the temperature, both of which are necessary for successful composting (Bernal et al., 2009). Uncontrolled or excess moisture evaporation in windrows (exceeding the optimum range of 50–60% of MC) can disrupt the moisture environment for microbial fermentation, subsequently inhibit microbial activity, and reduce the decomposition rate of organic pollutants (Vázquez et al., 2015).

The practice of supplementing compost windrows with SM and mixed bulking agents (such as rice husks and corncobs) to improve C/N ratios and moderate excess MC during the composting process is well established (Li et al., 2014). However, the water adsorption rates and high porosity of bulking agents may accelerate the water consumption rate (Niccolò et al., 2017). Therefore, moisture compensation and control in windrows are required to maintain an optimum MC for microbial activity during the composting process (Bernal et al., 2009). A new technique developed to augment MC requirements in windrows delimited by added SM or bulking agents is to use wastewater from various livestock and poultry slurry wastes, generally termed LM (a form of wastewater that presents a challenge for many conventional livestock waste treatment systems) (Wang et al., 2018). This new technique is referred to as co-composting of SM with LM.

Until now, very limited research has been conducted on co-composting of SM with LM. In addition, most previous studies on the addition of LM to SM composting were incidental and non-systematic and cannot fully reveal the characteristics of co-composting of SM with LM. For example, Bustamante et al. (2013) utilized liquid anaerobic digestates as moisture supplements in digested pig slurry composting and found no differences in the composting process and characteristics of the final composts both with and without LM. However, the specific effects of the addition of liquid anaerobic digestates (herein referred to as LM) on solid digested pig slurry (herein referred to as SM) composting were not elaborately studied. Therefore, exploring a systematic addition of LM to SM composting is urgently required to accelerate and maintain composting temperatures for efficient and effective aerobic composting processes. Recently, Vázquez et al. (2015) studied the co-composting of LM and SM with compact agricultural wastes at different volume ratios of LM to bulking materials and observed that the evaporation rates of windrows could be maintained at 58–75%, where the mass reduction reached nearly 90% for total Kjeldahl nitrogen, ammonium, and suspended solid removal. However, most of the windrow MC was as high as 70% during the co-composting process. This could inhibit biological activities and irregulate the temperature stage, leading to a decreased fermentation rate during the aerobic composting process (Bernal et al., 2009; Dhyani et al., 2018). Moreover, a high and uncontrolled addition of LM to the compost resulted in the generation of a considerable volume of leachates, which prevented the retention of pollutants and potential nutrients in the compost. Thus, studying the control of leachate generation by monitoring optimum MC (50–60%) will be a potential research direction for managing co-composting processes. In addition, co-composting of SM and LM with bulking materials is a feasible means of achieving LM consumption (usually by water evaporation). However, the driving factors for LM evaporation rates during the co-composting process are still largely unknown.

Thus, in this study, a volume model was developed to control the amount of LM added to the aerobic composting process to: 1) achieve optimum compost MC (50–60%), 2) regulate the temperature of the compost, and 3) reduce the amount of generated leachate. Moreover, the effects of bulking material sizes on the utilization of different LM types were evaluated to understand the underlying factors affecting evaporation rates in the co-composting process. Conditions for improved organic matter degradation and sequestration through controlled LM addition were also investigated. Finally, opportunities for nutrient retention and heavy metals immobilization using different LM types (swine effluent (SE) and biogas slurry (BS)) and their consumption amounts in the co-composting process were assessed.

2. Materials and methods

2.1. Raw materials and experimental set-up

SM (fresh pig manure) and LM (including SE and BS) were collected from a pig farm with a holding inventory of 3000 pigs. Bulking materials (rice husk) were crushed to 1 and 7 mm, before being mixed with pig manure. The main characteristics of raw SM and LM are listed in Table 1. Experiments were conducted in aerobic static composting

Table 1
Original characteristics of solid wastes and LM used in composting experiments.

Parameter	Solid wastes		LM*	
	Pig manure	Rice husk	SE	BS
Moisture (%)	74.63 ± 0.12	1.1 ± 0.1	nd	nd
pH	7.48 ± 0.03	6.83 ± 0.14	nd	nd
EC (mS/cm)	2.62 ± 0.04	1.51 ± 0.17	nd	nd
OM (%)	78.89 ± 0.18	86.5 ± 0.24	nd	nd
COD	52 ± 3.2	nd	4178 ± 4.2	1635 ± 2.7
TC (g/kg)	399.2 ± 2.41	420.9 ± 1.23	nd	nd
TN (g/kg)	30.17 ± 1.42	3.96 ± 1.1	847 ± 2.2	693.5 ± 3.1
TK (g/kg)	16.73 ± 1.09	6.24 ± 0.41	nd	nd
TP (g/kg)	35.07 ± 4.2	0.39 ± 0.01	273.5 ± 2.5	29.53 ± 1.9
Cu (mg/kg)	35.05 ± 1.9	12.23 ± 1.1	0.06 ± 0.008	0.04 ± 0.005
Zn (mg/kg)	614.2 ± 19.38	2.35 ± 1.6	0.104 ± 0.009	0.06 ± 0.008
Cr (mg/kg)	4.23 ± 0.75	42.1 ± 0.5	0.008 ± 0.001	0.006 ± 0.001
Ni (mg/kg)	51.23 ± 9.23	4.56 ± 0.4	0.003 ± 0.001	0.002 ± 0.0007
Pb (mg/kg)	44 ± 0.2	–	0.013 ± 0.005	0.009 ± 0.001
Cd (mg/kg)	0.3 ± 0.1	–	–	–

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

windrows using four Styrofoam boxes with inner dimensions 46 × 36.5 × 22.5 cm. Each windrow had a chamber floor with a 3% slope. The leachates were collected in collection tanks with dimensions of 45 × 10 × 10 cm. The initial mixing ratio of SM and bulking materials (in dry weight) was 1.2:1. The windrows comprised a mixture of 15 kg of fresh pig manure and 3 kg of bulking materials with differently sized particles (Table 1). The initial C/N ratios of the windrows were maintained at 23, which was suitable for the initial composting process. Prior to the addition of LM to the compost windrows, the LM types and bulking material particle sizes were categorized and each windrow was denoted respectively as: swine effluent + rice husk 1 mm (SE-RH1); swine effluent + rice husk 7 mm (SE-RH7); biogas slurry + rice husk 1 mm (BS-RH1); and biogas slurry + rick husk 7 mm (BS-RH7). The MC of windrows was 64.11 ± 0.87%, 63.16 ± 1.1%, 63.72 ± 1.23%, and 63.56 ± 1.35% for the windrows SE-RH1, SE-RH7, BS-RH1, and BS-RH7, respectively.

2.2. Composting procedure and model parameterization

The windrows were manually turned every two days during the thermophilic stage and every 5–10 days during other temperature stages. For improved microbial degradation activity, the optimum MC was reported to be in the range of 50–60% during composting (Zang et al., 2016). However, to increase LM consumption and ensure effective fermentation, the benchmark MC was set at 65%. The LM input frequency and volume were controlled based on the actual weight and MC of windrows. When the MC of the windrows was below 50%, the windrows were restabilized by adding LM.

The amount of LM to be added (denoted as V in liters) to the windrows was modeled using the following volume parameterization:

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

The percentage moisture content of windrows during composting (MC_a) was monitored through the weight in kilograms of windrows (M_w) before LMs were added. The densities (ρ) of the different LMs were 1.03 kg/L for SE and 1.01 kg/L for BS, respectively. Both SE and BS were maintained at an ambient temperature (30–35 °C). LM storage time could exhibit a certain effect on the concentrations of pollutants (Zang et al., 2016). Thus, pollutant concentrations in LM were detected when SE and BS were added to the various windrows. The total amounts of recovered/abstracted nutrients and pollutants from SE and BS ($T_{r/a}$, g) during the co-composting process were calculated as follows:

$$T_{r/a}(g) = C_{SE/BS} \times V_{SE/BS} \tag{2}$$

Concentrations of nutrients and pollutants in SE and BS in g/L at the time of LM addition to the composts were denoted by $C_{SE/BS}$, and LM volumes in L were denoted by $V_{SE/BS}$.

At the start of the experiment, samples of rice husk and pig manure (otherwise referred to as "mixed materials") were collected from the prepared windrows, as were samples of SE and BS. During the composting process, a small fraction of the mixed materials was collected for MC detection every two days, whereas a larger fraction of the samples was collected before the windrows were moisturized with SE and BS for other physiochemical parameter detection. Concentrations of SE and BS were tested when the windrows were moisturized with LM, and the amount of leachate generated was quantified.

Similarly, the water balance analysis for the water loss (WL) through evaporation, overall evaporation ratio (OER), and specific evaporation rate (SER) in the composting process were computed as follows:

$$M_{WL} = M_{OMS} + M_{LM} - (M_{FMS} + M_{Leachate}) \tag{3}$$

$$M_{OER} = M_{WL}/M_{ROS-D} \tag{4}$$

$$M_{SER} = M_{OER}/t \tag{5}$$

The amount of WL (kg) through evaporation was equal to the difference in weights of the inbound water (MC of original substrates (M_{OMS}) plus added LM (M_{LM}), all in kg) and outbound water (MC of the final substrates (M_{FMS}) plus leachate ($M_{Leachate}$), all in kg) during the entire composting period (0–28 days). OER was computed as the weight ratio of WL (M_{WL}) and dry raw organic substrates ($M_{ROS\text{-}D}$), which reflected the water consumption per dry weight of raw organic substrates. The SER in kg/day was the rate of water consumed per dry weight of raw organic substrates and per day. In addition, the LM-only consumption water balance analysis in the composting process was conducted as follows:

$$M_{WL} = M_{LM} - M_{Leachate} \tag{6}$$

$$M_{OER} = M_{WL}/M_{ROS-D} \tag{7}$$

$$M_{SER} = M_{OER}/t \tag{8}$$

2.3. Monitoring and analysis

Windrow composting temperatures were monitored twice daily using three thermometric sensors, which were placed at the surface (5 cm to the surface), middle, and bottom (5 cm to the base) of the windrows, respectively. MC was measured every two days, whereas the windrow weights were measured prior to the addition of LM to windrows. The leachate volumes generated were measured daily using measuring cylinders.

The MC, pH, electrical conductivity (EC), germination rate index (GI) of mixed materials were measured using the method reported by Dhyani et al. (2018), and the total carbon (TC), total nitrogen (TN), total phosphrous (TP), total potassium (TK), and organic matter (OM) were calculated using the method described by Wang et al. (2017). Heavy metals (including Cu, Zn, Cr, Pb, and Ni) were detected by an inductively coupled plasma mass spectrometer (ICP-MS, Agilent 7500cx, Japan) (Vázquez et al., 2015). For SE and BS, all parameters were measured and analyzed using the methods described by Bustamante et al. (2013) and Vázquez et al. (2015). All analyses in this study were conducted in triplicate, and data collation was conducted using IBM SPSS Statistics 23.

3. Results and discussion

3.1. Temperature evolution

The evolution pattern of the temperature was similar in all composting processes. At the beginning of the composting process, the initial mesophilic stage for all composts was very short, and the temperature increased sharply and reached the thermophilic stage at 50 °C after the first two days of the composting, with windrows SE-RH1 and SE-RH7 reaching higher temperature values than those of windrows BS-RH1 and BS-RH7. The maximum temperatures of RH7-based composts were all above 64 ± 0.3 °C, which were higher than those in RH1-based composts (61.5 ± 0.9–63.5 ± 0.8 °C). By contrast, RH1-based windrows had a longer thermophilic stage, reaching 10 ± 0.3 days during the entire 28 days of composting, which was much longer than that in RH7-based composts, which required 8 ± 0.4–10 ± 0.2 days (Table 2). This indicates that RH1-based windrows with smaller bulking materials were more bioavailable and easily degraded by microorganisms than RH7-based windrows and thus reveals that bulking materials with smaller particles are inevitable for sustaining thermophilic conditions in the compost (Zhang and Sun, 2014). By contrast, RH7-based windrows with larger bulking materials can easily transmit oxygen, which assist in heat convection and thus achieve a quick rise in temperature (Vázquez et al., 2015). Therefore, particle sizes of bulking materials are a crucial parameter for improving the fermentation process. Furthermore, the composting and ambient temperatures for RH1-based windrows converged after 24 days from the start of operation,

Table 2
The duration of different temperature phase in different windrows.

Types	Thermophilic stage			Mesophilic stage	T_{avg} (°C)	T_{max} (°C)
	Above 60 °C (d)	Above 55 °C (d)	Above 50 °C (d)	40–50 °C (d)		
BS-RH1	5 ± 0.2	6.5 ± 0.2	10 ± 0.3	12 ± 0.3	56.4 ± 3.9	61.5 ± 0.9
SE-RH1	5.5 ± 0.4	8 ± 0.3	10 ± 0.3	10 ± 0.2	57.4 ± 4.1	63.5 ± 0.8
BS-RH7	3 ± 0.6	5.5 ± 0.4	8 ± 0.4	10 ± 0.6	57.83 ± 5.3	64 ± 0.3
SE-RH7	4 ± 0.5	7.5 ± 0.3	10 ± 0.2	8.5 ± 0.4	58.65 ± 5.1	64.5 ± 0.5

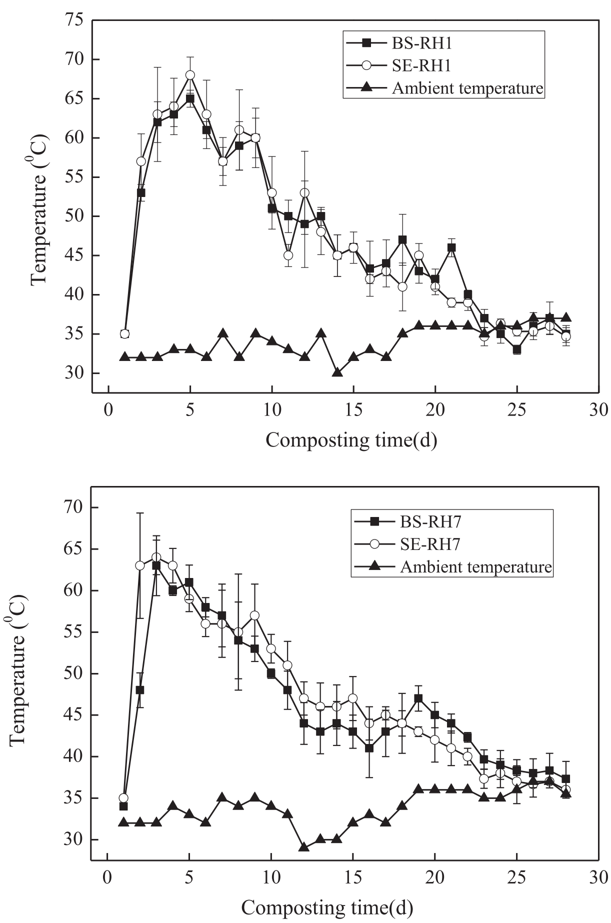

Fig. 1. Temperature evolution of different windrows with varying LM types and bulking material particle sizes during composting process.

whereas the temperature convergence for RH7-based windrows occurred on the 26th day (Fig. 1). This phenomenon reveals that the mixed materials of RH1-based windrows reached maturation earlier than those of RH7-based windrows, indicating that bulking materials with smaller particles were more beneficial to improving OM degradation efficiency.

In the thermophilic stage, all composts were manually turned over every two days and wetted with LM nearly four times during the total co-composting wetting intervals, which was conducted seven times in total. Therefore, the thermophilic stage represented the main phase for water loss and microbial degradation of OM during the composting process. After the thermophilic stage, the average temperatures in all composts dropped to 40–50 °C, thus reconditioning the composts to a mesophilic degradation stage. The duration of the mesophilic stage for all composts (8.5–12 days) was as much as 1/3 of the entire compost period (Table 2), revealing that the mesophilic stage was crucial in the degradation of OM. After the mesophilic stage, the average temperatures in all composts dropped to 30–40 °C, which approximated the ambient temperature (30–37 °C) (Fig. 1), suggesting that biological activities were reduced and the composting process was thus stabilized (Meng et al., 2018). During the last two stages, the turning frequency was adjusted to once every five to ten days, and windrows were thrice watered with LM. Temporal temperature undulations were observed in all composts after periods of turning and wetting (Fig. 1). This shows that the addition of LM could significantly affect the temperature profile of composts. A similar finding was reported in the treatment of compost with SM and wood chips, where the compost was watered with LM during the thermophilic phase (Vázquez et al., 2015).

The average and maximum temperatures for SE-based windrows during the thermophilic stage were 0.3–1 °C and 0.5–2 °C higher, respectively, than that of BS-based windrows treated with the same bulking materials (Table 2). This shows that SE, which contains comparatively higher concentrations of pollutants than BS, was more suitable to maintaining the thermophilic stage temperature. Furthermore, the addition of SE possibly functioned as an inoculator to promote microbial activity and subsequent heat generation (Meng et al., 2018). In addition, SE-based windrows had a thermophilic duration (10 ± 0.3 days), which were 0–2 days longer compared to the BS-based windrows (8 ± 0.4–10 ± 0.3 days). According to Bernal et al. (2009), when the thermophilic duration extends longer than three days during composting, pathogens and seeds will be exterminated. Thus, both the SE and BS-based composts could satisfy the biosafety requirement for final compost utilization (Jiang et al., 2017; Jiang et al., 2018).

3.2. Windrows weight and OM characteristics during composting

Windrows OM and weight for all composts showed a general decreasing trend over time (Fig. 2). Significantly positive correlations were observed between the OM and weight in respective windrows ($r > 0.8$, $p < 0.05$), indicating that reduction of windrow weight was mainly the result of OM degradation (whereas the MC was maintained at 50–60%). The highest reductions in windrow weight and OM were observed during the thermophilic stage (Fig. 2a). For instance, a 65% reduction in OM and windrow weight occurred during the thermophilic stage, and the remaining 35% occurred in the mesophilic stage (Fig. 2b). This confirms that the thermophilic and mesophilic stages were the main phases for organic pollutant degradation during the composting process when using rice-husk-based bulking materials. These major phases of pollutant degradation may have an effect on the consumption time of added LM. As shown in Fig. 2b, the final weight reduction rate was in the order of: RH1-based windrows to RH7-based windrows. In a recent report, Zhang and Sun (2014) compared the initial compost materials in a windrow with a particle size of 20 mm to another windrow with initial compost materials of 15 mm. They observed that smaller compost had a higher microbial abundance and improved organic pollutant degradation (cellulose and hemicellulose), with a reduction in the amount of TOC and TN (Zhang and Sun, 2014). Thus, in agreement with results from the previous work, the present study inferred that rice husk of smaller particles (1 mm) were more beneficial for organic pollutant degradation than rice husk of larger particles (7 mm).

However, OM degradation of SE-RH1 was 2.42% higher than that of BS-RH1, whereas in SE-RH7, the amount of OM degraded was only

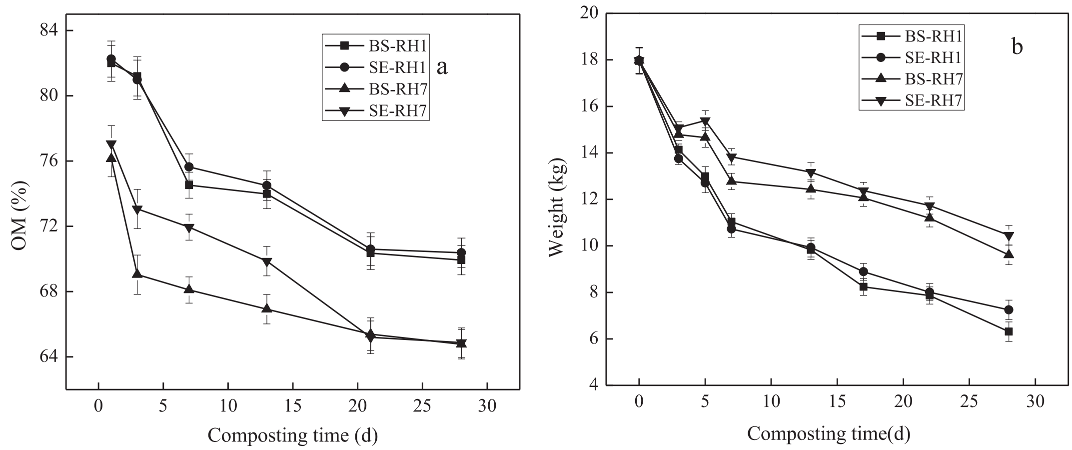

Fig. 2. Windrows weight and OM characteristics during composting process: (a) OM characteristics, (b) Weight characteristics.

0.3% higher than that of BS-RH7 (Fig. 2a). This indicates that utilizing SE as a co-composting LM fraction could improve the degradation performance of OM in windrows with bulking materials of smaller particles. Comparatively, OM degradation in BS-RH1 was 1.67% higher than that of BS-RH7, whereas the amount of OM degradation in SE-RH1 was 4.38% higher than that of SE-RH7 in the thermophilic stage. Thus, both LM types and bulking material sizes had significant effects on OM degradation.

3.3. LM utilization and leachate generation

The controlled addition of LM to all windrows resulted in a similar MC evolution, as shown in Fig. 3. The volumes of added LM in the thermophilic stage ranged from 5.7 to 7.3 L when using Eq. (1), which accounted for 80% of the total added LM to all windrows during the entire composting process (Fig. 4). The duration of the thermophilic stage in respective windrows and the added LM amounts had significant positive correlations ($r > 0.8, p < 0.05$). The high LM consumption in the thermophilic stage may have been the result of moisture evaporation at temperatures higher than 50 °C during the composting process. As the composting process neared completion, the controlled addition of LM to the windrows was unable to maintain the windrow weight due to the degradation of the organic components (bulking materials and SM) of the windrows. Thus, the decrease in added LM volumes in the windrows over time is shown in Fig. 4. In addition, using the volume model described in Eq. (1), the added LM amounts for RH1-based windrows were slightly higher than those for RH7-based windrows, whereas the added LM amounts for SE-based windrows were also slightly higher than those for BS-based windrows (Fig. 4). Thus, it was evident that particle sizes and LM types played a critical role in extending the thermophilic stage during the composting process.

Leachates generated from the compost were mixtures of LM and mixed materials produced by gravitational pull ($\rho_{leachate\ average} = 1.4 \pm 0.2$ kg/L) in the windrows when soaked with LM. Compared to the leachate generation rate (0.63–1.39 L/m^3·d) in the study conducted by Vázquez et al. (2015), a significant reduction in the leachate generation rate (0.03–0.1 L/m^3·d) was achieved in this study by modeling the volume of LM added to the compost while maintaining an optimum MC (50–60%). The LM volume modeling aided the calculated reduction of secondary pollutant discharge and nutrient recovery, as discussed in Section 3.5. Moreover, it has been reported that high MC in windrows can alter fermentation temperatures radically and reduce fermentation efficiency. In this study, the added LM volumes were conducted in accordance with the previously reported MC requirement and the weight of windrows (Zang et al., 2016). All composts successively underwent the first mesophilic stage, the thermophilic stage, the second mesophilic stage, and the later maturation stage,

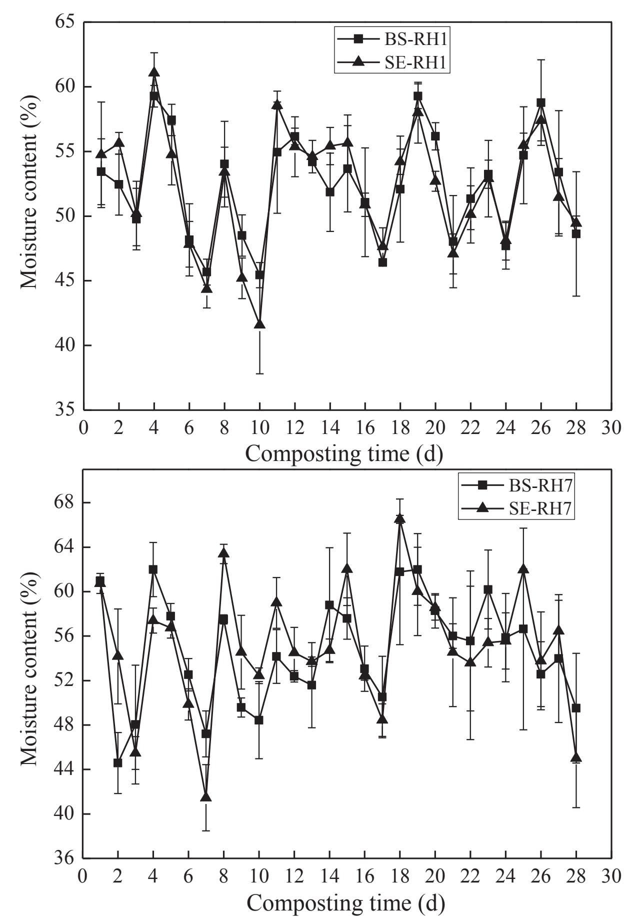

Fig. 3. Moisture contents of different windrows during composting process.

indicating that the controlled addition of LM had a considerable effect on SM aerobic composting (Bernal et al., 2009; Dhyani et al., 2018). Thus, apart from the reduction of leachate generated volumes, increased LM consumption and aerobic fermentation efficiency were achieved in the co-composting process.

3.4. Water balance analysis

Water balance analyses were conducted during all stages of the composting process until the windrows reached the maturation stage, and the analyses were based on the mass balances between the inbound and outbound water for the entire composting period as described by

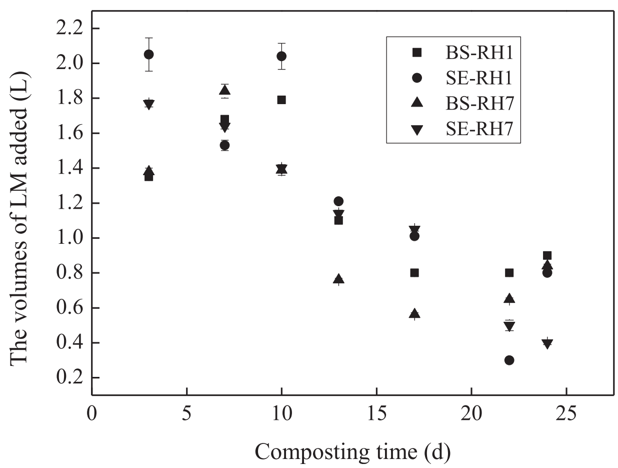

Fig. 4. Required volumes of LM added to all windrows to reach an optimum MC of 65% during the whole composting process.

Eq. (3) and as presented in Table 3. The original moisture in each windrow was calculated from the in-bound water and is also presented in Table 3. Following the water balance analysis in Eq. (4), the OER of RH1-based windrows ranged between 2.58 ± 0.03 and 2.64 ± 0.01 m^3/t·dry weight, which was 12.4–16.67% higher than that of RH7-based windrows; whereas the SER obtained using Eq. (5) ranged between 92.17 ± 0.1 and 94.20 ± 0.32 L/t·dry weight·d, which was 1.24–1.30% higher than that of RH7-based composts (Table 3). Smaller bulking materials possess a larger surface area, which increases their bioavailability and thus raises the temperature, leading to an enhanced evaporation process (Sánchez et al., 2017). In a previous study, the OER and SER observed in the aerobic composting process, which used SM and SE as substrates, were 2.59 m^3/t dry weight and 76.3 L/t dry weight·d, respectively, for 26 days (Vázquez et al. 2015). The present study obtained an OER and SER of 2.64 ± 0.01 m^3/t dry weight and 94.2 ± 0.32 L/t dry weight·d, respectively, for 28 days. However, as earlier proposed by Vázquez et al. (2015), the effects of operational conditions on compost evaporation rates are crucial factors yet to be investigated. Thus, the LM volume model developed in this study was distinctly advantageous for improving the evaporation rates of windrows during the composting process.

Moreover, for non-water usage collection of SM, the MC of pig manure is usually between 70% and 80%. Thus, following the water balance calculation with respect to the original manure, the starting MC will be non-reflective of the consumption rate of added LM during the co-composting process, as described in Table 3. Therefore, a more accurate and precise calculation for the overall LM consumption and rate of consumption during the composting process was computed. This was necessary to remove the original MC of the windrows from the LM. The water balance calculation is presented in Table 4. Thus, the OER and SER of LM in the windrows were individually evaluated based on the effects of the sizes of bulking materials and LM types. In RH1-based windrows, an OER of 1.25 ± 0.006–1.37 ± 0.007 m^3/t dry weight was obtained, which was 10.77–13.22% higher than that of RH7-based windrows; whereas the SER of LM in RH1-based windrows (44.78 ± 0.21–48.89 ± 0.25 L/t dry weight·d) was 10.8–3.3% higher than that of RH7-based windrows (Table 4). In the SE-based windrows, an OER of 1.19 ± 0.003–1.37 ± 0.007 m^3/t dry weight was 5.81–8.4% higher than that of BS-based windrows. Similarly, the SER of LM in SE-based windrows (42.41 ± 0.11–48.89 ± 0.25 L/t dry weight·d) was 5.8–8.4% higher than that of BS-based windrows (Table 4). Therefore, the evaporation rates of added LM in each windrow were significantly influenced by the sizes of bulking materials and LM types, even though the sizes of the bulking materials had greater effects on LM consumption than did LM types. This was because bulking materials with smaller particles (rice husk of 1 mm) were more

Table 3
Water balance characteristics in the windrows during composting process.

Windrows[a]	Raw organic substrates[b]		In-bound water		Final compost		Out-bound water		Water loss (kg)	Evaporation	
	Original weight (kg)	Dry weight (kg)	Original moisture weight (kg)[c]	Added LM (kg)	Final windrows weight (kg)	Final moisture content (%)	Final moisture weight (kg)	Leachates (kg)		Overall ratio (m^3/t dry weight)	Specific rate (L/t dry weight·d)
BS-RH1	18 ± 0.01	6.45 ± 0.01	11.55 ± 0.2	8.16 ± 0.03	6.31 ± 0.02	44.55 ± 0.8	2.81 ± 0.08	0.073 ± 0.001	16.83 ± 0.21	2.58 ± 0.03	92.17 ± 0.1
SE-RH1	18 ± 0.01	6.48 ± 0.01	11.52 ± 0.1	8.94 ± 0.04	7.25 ± 0.03	45.51 ± 0.5	3.30 ± 0.07	0.07 ± 0.002	17.09 ± 0.1	2.64 ± 0.01	94.20 ± 0.32
BS-RH7	18 ± 0.01	6.53 ± 0.01	11.47 ± 0.1	7.42 ± 0.03	9.61 ± 0.01	42.22 ± 0.7	4.05 ± 0.1	0.114 ± 0.003	14.73 ± 0.04	2.26 ± 0.001	80.54 ± 0.03
SE-RH7	18 ± 0.01	6.56 ± 0.01	11.44 ± 0.1	7.9 ± 0.01	10.45 ± 0.03	44.73 ± 0.5	4.67 ± 0.09	0.11 ± 0.004	14.56 ± 0.02	2.20 ± 0.001	79.27 ± 0.05

Note: [a]: Composting period (0–28 d).
[b]:Solid fraction of manure (SM) 3.8 kg (dry weight) and bulking materials 2.9 kg (dry weight), dry weight in this SM lot was only 25.37%.
[c]:The original moisture content of mixed materials were calculated in the analysis.

Table 4
Water balance for LM consumption in the windrows during composting process.

Windrows[a]	Raw organic substrates (kg, dry weight)[b]	In-bound water (added LM[c])	Out-bound water (leachates)	Water loss (LM consumption, kg)	Evaporation	
					Overall ratio (m^3/t dry weight)	Specific rate (L/t dry weight·d)
BS-RH1	6.45 ± 0.01	8.16 ± 0.03	0.073 ± 0.001	8.087 ± 0.04	1.25 ± 0.006	44.78 ± 0.21
SE-RH1	6.48 ± 0.01	8.94 ± 0.04	0.07 ± 0.002	8.87 ± 0.05	1.37 ± 0.007	48.89 ± 0.25
BS-RH7	6.53 ± 0.01	7.42 ± 0.03	0.114 ± 0.003	7.306 ± 0.04	1.12 ± 0.005	39.96 ± 0.19
SE-RH7	6.56 ± 0.01	7.9 ± 0.01	0.11 ± 0.004	7.79 ± 0.01	1.19 ± 0.003	42.41 ± 0.11

Note: [a]: Composting period (0–28 d)
[b]: Solid fraction of manure (SM) 3.8 kg (dry weight) and bulking materials 2.9 kg (dry weight), dry weight in this SM lot was only 25.37%.
[c]: The original moisture content of mixed materials were not calculated in the analysis.

capable of inducing fermentation temperature and maintaining the temperature at thermophilic conditions, as described previously in Section 3.1. SE was more easily consumed during aerobic composting than was BS during the co-composting process.

Comparison of the moisture loss and actual LM consumed in the composting windrows were evaluated and are presented in Table 4. The amount of water loss due to LM consumption (7.3–9.6 kg) and the specific water consumption rate (40–49 L/t dry weight·d) in the windrows represented only half of the overall water consumption analysis. Thus, it can be deduced that the MC of the original mixed materials was a critical factor affecting LM consumption. Empirical statistics reveal that approximately 190 t/d of swine breeding waste is produced for every 10,000 pigs breeding in a pig farm, generating approximately 75 t of SM (in wet weight, average MC 75%) and approximately 115 t of LM. Therefore, given the specific water consumption rate (40–49 L/t dry weight·d) for an SM compost mixed with bulking materials at the mixing ratio of 1.2:1 (in dry weight) and a C/N ratio of 24, approximately 1.8 t/d of LM could be consumed in this type of farm. This result reveals that high volumes of LM can be consumed in the co-composting process. However, compared with the total amounts of LM generated in pig farms, the SER of water in the composting windrows does not meet LM consumption requirements. Therefore, improving the LM consumption rate in the co-composting process will be a major research topic in the near future.

3.5. Final compost characteristics

Nutrient retention characteristics of composting windrows were derived from the concentrations of added LM to the compost. Amounts in weight of retained and immobilized nutrients and pollutants are shown in Fig. 5. The percentage increase in total weighted amounts of NO_3^--N, NH_4^+-N, TN, PO_4^{3-}-P, and TP in SE-based windrows were 5–16%, 8–19%, 10–23%, 52–69%, and 200–300% higher, respectively, than that of BS-based composts (Fig. 5a). In general, SE contained higher contents of nutrients and heavy metals than in BS (Fig. 5a and b). Thus, the added LM type is a major factor to consider in terms of nutrient availability, while not compromising the pollutant immobilization characteristics of composts. Moreover, the total retained amounts of NO_3^--N, NH_4^+-N, TN, PO_4^{3-}-P, and TP in RH1-based windrows were 16.58–16.88%, 10.63–22.15%, 11.47–21.49%, 11.71–20.73%, and 11.49–21.94% higher, respectively, than in RH7-based windrows. A possible reason for the disparity in nutrient retention characteristics of the two composts could be the existence of a longer thermophilic stage, which favored greater LM consumption in RH1-based windrows than in RH7-based windrows. This was evident from the significantly positive correlations observed between the total volumes of added LM and total amounts of potential nutrients retained in both compost windrows ($r > 0.7$, $p < 0.05$).

Similarly, the amount of retained nutrients (derived from the concentrations of LM nutrients added to the composts) in the respective windrows had significantly positive correlations ($r > 0.75$, $p < 0.05$) with the concentrations of TP and TK in the final composts (Fig. 5c). The concentrations of TP and TK in SE-based final composts were 1.1–6.6% and 0.03–0.05% higher, respectively, than in corresponding BS-based final composts with the same sizes of bulking materials (Fig. 5a). Therefore, the retention of nutrients derived from LM in the SM compost was favorable for nutrient enrichment in the final composts. Likewise, the maximum amount of immobilized heavy metals (Cu, Zn, Cr, Ni, and Pb) by weight for SE-based windrows were 50–75%, 65–90%, 16–20%, 20–50%, and 28–43% higher, respectively, than in BS-based composts (Fig. 5b), thus increasing the concentrations of immobilized heavy metals in the SE-based final composts. The maximum concentrations of Cu, Zn, Cr, Ni, and Pb found in the SE-RH1 final compost were 1040.36 ± 25, 53.11 ± 8, 2.67 ± 0.04, 1.59 ± 0.26, and 3.07 ± 0.61 mg/kg dry weight, respectively, which were 25.24–96.63%, 4.7–14.75%, 19.07–19.73%, 25.20–32.43%, and 6.23–6.34% higher, respectively, than in the corresponding BS-based final composts using similar sizes of bulking materials (Fig. 5c).

At the end of the co-composting process, all composts showed final pH values that were within the range of 6.0–8.5 and thus close to the neutrality (Fig. 6a), suggesting that the final composts were suitable for agricultural use (Bustamante et al., 2013). The EC values of all windrows were 2–3 mS/m, which were far less than the values (5–9 mS/m) of final composts reported by Bustamante et al. (2013). This may have been partly due to the different salinity contents of original manure, bulking materials, and LM types. Correspondingly, the GI of all composts in this study were over 80% (Fig. 6b), indicating that phytotoxicity was eliminated (Bernal et al., 2009). Thus, the nutrients and immobilized heavy metal concentrations in the final composts all met the requirements of the organic fertilizer standard in China (NY525-2012) for final compost application and were suitable for improved plant growth. Although, the GI of SE-based windrows was 1.01–2.03% lower than the corresponding BS-based composts using the same sizes of bulking materials, the phytotoxicity of the SE-based final composts were not inhibited (Fig. 6b). This implied that application of the co-composting process with the selected LM type could enhance nutrient retention and was beneficial as a soil-fertilizing agent.

4. Conclusion

In this study, the controlled addition of LM to SM compost greatly improved the compost MC, reduced leachate generation, and efficiently regulated the thermophilic stage. The specific evaporation rates of LM in each of windrows were significantly influenced by bulking material sizes and LM types, although the former had greater impact on LM consumption. Moreover, phytotoxicity of the final compost was eliminated regardless of LM types or bulking material sizes used. Notwithstanding, the selected LM type improved nutrient retention potential of the final compost. Future efforts on co-composting of LM with SM should be focused on key process parameters optimization.

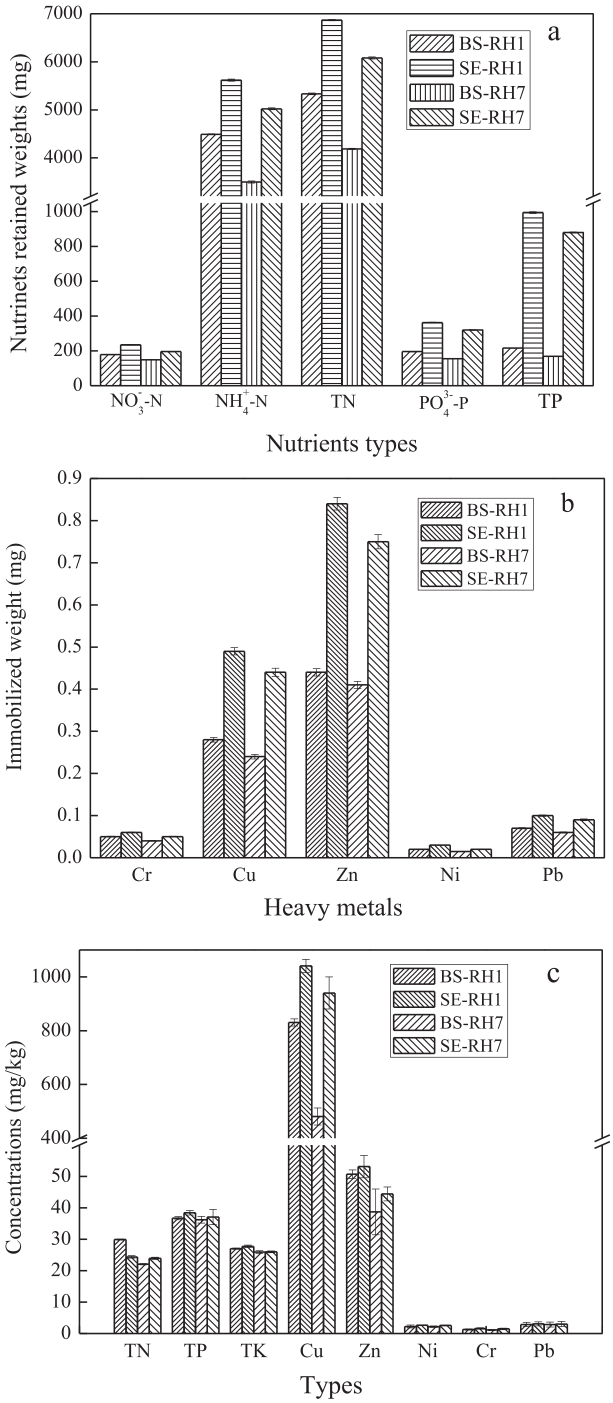

Fig. 5. Weighted amounts of (a) nutrients retained and (b) heavy metals immobilized in the SM composts after controlled addition of LM and (c) concentrations of retained nutrients and immobilized heavy metals in the final composts.

Acknowledgments

This research was supported with funds from the STS project of Science and Technology Department in Fujian [2016T3006], Key Research and Development Program Social Development Project in Zhejiang [2015C03009], and Key Laboratory of Environmental Biotechnology, Fujian Province University [EBL2018008].

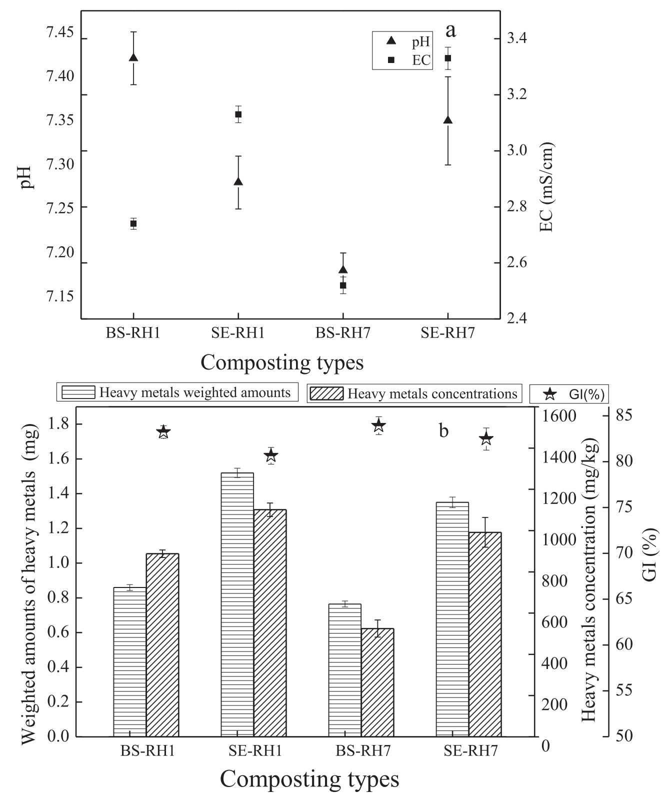

Fig. 6. EC, pH and GI of the final composts: (a) EC and pH of the final composts; (b) Relationship between weighted amounts and concentrations of immobilized heavy metals (Cu, Zn, Ni, Cr, Pb) in final composts and GI.

Declaration of Interest

There is no conflict of interest.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biortech.2018.12.049.

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