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Organic loading on biochemical fractions degradation pattern during food waste bioevaporation

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Manuscript Number:	WM-21-548
Article Type:	Full Length Article
Section/Category:	Biological treatment
Keywords:	Organic loading; Lipids; Protein; Carbohydrates; Bioevaporation.
Abstract:	The degradation pattern of biochemical fractions during food waste (FW) bioevaporation was significantly influenced by the organic loading (OL).Lower OL facilitated the lipids degradation, while higher OL favored the proteindegradation. It was the more porous structure and abundant oxygen accelerated the lipids degradation, and the rapid proliferation of aerobic microorganisms offset the degraded protein in lower OL. Detailly, 76.8% of the lipids was degraded in the trial with OL of 1.22 kg VS FW /kg TS BS (Trial A), but in the trial with OL of 3.66 kg VS FW /kg TS BS (Trial C) it was only 0.5%. For protein, the degradation was different that 17.5% of the protein was degraded in Trial A, whereas 69.1% was degraded in Trial C. Lipids degradation contributed 63.0% to the metabolic heat in Trial A, but its contribution in Trial C was only 0.5%. For protein, it contributed 4.1% to the metabolic heat in Trial A, but in Trial C it accounted for 53.6%. In addition, the degradation of carbohydrates (61.2−68.6%) and their contribution to metabolic heat (32.8−45.9%) was comparable in all trials, thus OL had little effect on carbohydrates degradation.

Highlights

Lower organic loading facilitated the degradation of lipids

Higher organic loading favored the degradation of protein

Organic loading had little effect on the degradation of carbohydrates

63.0% of metabolic heat was from the degradation of lipids in $OL_{1.22}$

Degradation of protein contributed 53.6% to metabolic heat in $OL_{3.66}$

Organic loading on biochemical fractions degradation pattern during food waste bioevaporation

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Organic loading on biochemical fractions degradation pattern during food waste bioevaporation

Abstract

The degradation pattern of biochemical fractions during food waste (FW) bioevaporation was significantly influenced by the organic loading (OL). Lower OL facilitated the lipids degradation, while higher OL favored the protein degradation. It was the more porous structure and abundant oxygen accelerated the lipids degradation, and the rapid proliferation of aerobic microorganisms offset the degraded protein in lower OL. Detailly, 76.8% of the lipids was degraded in the trial with OL of 1.22 kg VS_{FW}/kg TS_{BS} (Trial A), but in the trial with OL of 3.66 kg VS_{FW}/kg TS_{BS} (Trial C) it was only 0.5%. For protein, the degradation was different that 17.5% of the protein was degraded in Trial A, whereas 69.1% was degraded in Trial C. Lipids degradation contributed 63.0% to the metabolic heat in Trial A, but its contribution in Trial C was only 0.5%. For protein, it contributed 4.1% to the metabolic heat in Trial A, but in Trial C it accounted for 53.6%. In addition, the degradation of carbohydrates (61.2−68.6%) and their contribution to metabolic heat (32.8−45.9%) were comparable in all trials, thus OL had little effect on carbohydrates degradation.

Keywords: Organic loading; Lipids; Protein; Carbohydrates; Bioevaporation.

1. Introduction

A tremendous amount of food waste (FW) has been produced every year with the rapid development of the foodservice industry as well as the severe phenomenon of wasting food (Onwosi et al., 2017). It is essential to take positive measures to dispose of the FW for a cleaner environment. Bioevaporation, proposed in 2013, is an aerobic process that can dispose of the highly concentrated organic wastewater including FW by microbial generated metabolic heat (Yang et al., 2013). Simultaneous removal of volatile solids (VS) and water could be achieved in this process through the degradation of organic matters and the accompanying metabolic heat (Yang & Jahng, 2015).

Organic loading (OL) significantly influenced the performance of bioevaporation process (Yang & Jahng, 2014). A high OL was preferred as more FW could be treated in a certain period, while too high OL led to a low VS and water removal since the added FW VS could not be successfully degraded by the microorganism (Yang & Jahng, 2015). Based on the literature, the moisture content (MC) of FW is around 80%, namely, FW contains a lot of water. When too much FW was added, water in the FW will penetrate the voids of the bulking agent and be absorbed, then a thick water film will be formed on the surface of the bulking agent. Based on the double-film theory, the resistance of molecular oxygen transfer essentially all lies on the thickness of the liquid film as the gas phase diffusion coefficients are typically greater than these in the liquid phase by a factor of 1×10^{4}–10^{5} (Hirschfelder et al., 1955; Mccabe, 1976). Thus, the thick water film with high OL would hinder the oxygen transfer to the microorganisms, and the aerobic degradation of FW becomes difficult. As a consequence, lots of anaerobic zones will be

formed in the matrix and the anaerobic microorganisms might proliferate, then anaerobic degradation might occur. Because the degradation of organic matters in the anaerobic condition is much slower, the VS degradation and water removal in bioevaporation were impeded as the metabolic heat is the driven force of water evaporation (Yang et al., 2013).

As for specific biochemical fractions, they might present different degradation patterns in different OL. Ge et al. (2014) reported that the aliphatic compounds could be easily oxidized into aromatic compounds in the aerobic layer of pig manure, indicating the lipids degradation might be affected by the OL. Moreover, the content of protein was the balance of its degradation (Liu et al., 2020) and extracellular polymeric substances formation (Tchobanoglous et al., 2003). Thus, the different growth of aerobic and anaerobic microorganisms in different OL might also cause a distinct degradation of the protein as well as other biochemical fractions like water-extractable organic carbon (WEOC), amylums, and lignocellulose. Because the distinct degradation pattern of biochemical fractions during different temperature stages of bioevaporation process was related to the corresponded enzymatic activity and would further affect the metabolic heat generation to the process. Thus, it was critical to investigate the degradation pattern of biochemical fractions in different OL.

Therefore, in this study, the degradation pattern of biochemical fractions under different OL and their contribution to metabolic heat during FW bioevaporation were investigated. First, bioevaporation of FW under different OL was conducted using biofilm-developed sponge (BS) as a bulking agent and microbial carrier, and the bioevaporation performance was evaluated. Next, the degradation of biochemical

fractions and the functional enzymatic activities were analyzed to help understand the dynamics of organics degradation and the kinetics of enzymes under different OL. Finally, the metabolic heat generated by the degradation of biochemical fractions under different OL was calculated. Results from this study could provide critical insights for the practical application of bioevaporation technology.

2. Materials and methods

2.1 Materials

The activated sludge was obtained from a sewage-treatment plant in Kunming, Yunnan. The FW was obtained from the cafeteria in the Kunming University of Science and Technology, then it was ground to about 1 mm-diameter-smooth paste by a meat grinder (Jinhuiyuan Food Machinery Factory Co., Jiang Su, China) and an electrical blender (MJ-BL80Y21, Midea Electrical Co., China). The polyurethane sponge was purchased in the market and crashed to about 1 mm diameter. The parameters of these materials were shown in Table 1.

Table 1. Characteristics of the materials [a]

Materials	MC (%)	VS (%)	pH	EC (×1000 mS cm^{-1})
FW	71.95	94.09	4.3	2.3
BS	74.57	89.32	6.6	0.7

[a] FW: food waste; BS: biofilm-developed sponge

2.2 Experimental setup

Firstly, the BS was cultivated as the microbial carrier and the bulking agent for the bioevaporation process. The cultivation was conducted as recommended by Yang and Jahng (2015). The cultivated BS was squeezed, and the resulted MC was 74.57% and VS

was 89.32%.

Three OLs for FW bioevaporation were designed. 2 kg BS was separately mixed well with 2.01 kg (Trial A: OL of 1.22 kg VS_{FW}/kg TS_{BS}), 4.04kg (Trial B: OL of 2.44 kg VS_{FW}/kg TS_{BS}), 6.09 kg (Trial C: OL of 3.66 kg VS_{FW}/kg TS_{BS}) FW and then added into 28.3 L cylindrical reactors, of which length in 275 mm, width in 225 mm and height in 325 mm (wall thickness is 50 mm). The aeration ball was set at the bottom and the airflow rate was 3 $L \cdot min^{-1}$. The gas analyzer (AO2000, ABB Co., Germany) was connected to the reactors for the CO_2 and O_2 analysis in the exit gas. The thermometer was inserted in the middle of the reactor to record the pile temperature.

2.3 Sample collection and analysis

Each sample was divided into three parts. The first part was immediately used for MC, VS, pH, EC, and WEOC analysis. The second part was stored at -40°C and then freeze-dried for biochemical fractions (amylums, lipids, protein, hemicellulose, cellulose and lignin) analysis. The third part was freezed at -80°C for the analysis of the enzymatic activity, including amylase, lipase, protease, xylanase, cellulase, and dehydrogenase. Each analysis of above parameters was in triplicate.

The MC was measured by dried from a drying oven (DHG-9036A, Shanghai Jinghong Experimental Equipment Co., China) at 105 °C for 24 h. The VS was measured by burned in a muffle furnace (SXL-1208, Shanghai Jinghong Experimental Equipment Co., China) at 550°C for 7 h, as well as, the removal ratio of water and VS were calculated. The variation of free air space (FAS) with different OL was calculated (Yang & Jahng, 2015). The pH and EC were monitored using pH Meter (UB-7, Denver, America)

(Gabhane et al., 2012) and Conductivity Meter (Multi 350i, WTW, Germany) (Jain et al., 2018).

WEOC was determined by TOC Analyzer (Elementar Vario TOC cube, Germany) in a 1:10 (w/v, wet basis) water-soluble extract. The protein was determined according to Kieldahl methods, lipids was detected according to Soxhlet extraction (Castro & Ayuso, 2020), and amylums were determined according to acid hydrolysis (Liu et al., 2020). The hemicellulose, cellulose, lignin were detected as recommended by Zhang et al. (2018). The enzymatic activity was assayed using an indirect ELISA method via a commercial kit (MLBIO, Shanghai Enzyme-linked Biotechnology Co., Ltd.).

3. Results and discussion

3.1 OL on physic-chemical properties of the bioevaporation process

The temperature of Trial A, B, and C raised about 45 °C within 2 days during the bioevaporation process (Fig. 1A). The highest temperature of Trial A reached 59 °C on the 2.65th day, then after decreased, the temperature increased again and reached the third peak of 58.4 °C on the 6.73th day. Until the 8th day, the temperature dropped to room temperature. For Trial B, the highest temperature was 69 °C on the 4th day. After dropping to 45 °C until the 5.87th day, the temperature maintained above 45 °C for about 3.5 days. For Trial C, the temperature reached the highest of 67.7 °C on the 5.14th day. The high temperature (above 45 °C) lasted for 7 days, probably due to the more provided organic substances, which prolonged the biodegradation time.

With the consumption of O_2 and production of CO_2 during the FW bioevaporation, the O_2 concentration in the exit gas decreased and CO_2 concentration increased (Fig. 1B).

About the 0.94th day, the first peaks of the lowest O_2 and the highest CO_2 concentration were observed in the three trials, where the first small temperature peaks of around 45 °C appeared. Due to the abundant bioavailable nutrients in the initial matrix and the fast proliferation of mesophilic microorganisms, rapid biodegradation of organic matters occurred, thus the peak value of respiratory activity was observed during this stage (Miyatake & Iwabuchi, 2006). The second peaks of lowest O_2 and highest CO_2 appeared on the 3.49th, 4.25th, 4.97th day, respectively, for Trial A, B, and C, which were corresponded to the time of the second temperature peaks, respectively. The second gas peaks of these periods were thought to be attributed to the thermophilic microorganisms since the temperature was above 45 °C. Therefore, it might be the transform of mesophiles to thermophiles that caused the two distinguished peaks both in gas and temperature (Miyatake & Iwabuchi, 2006). Those correspondences between gas data and temperature implied that high microbial respiratory activity led to the fast release of metabolic heat, whereas the high temperature, in turn, affected the microbial metabolic activity. With the degradation of the recalcitrant organics, the third temperature peak appeared in Trial A, and the corresponded gas peaks were less obvious than those appeared in the first two temperature peaks.

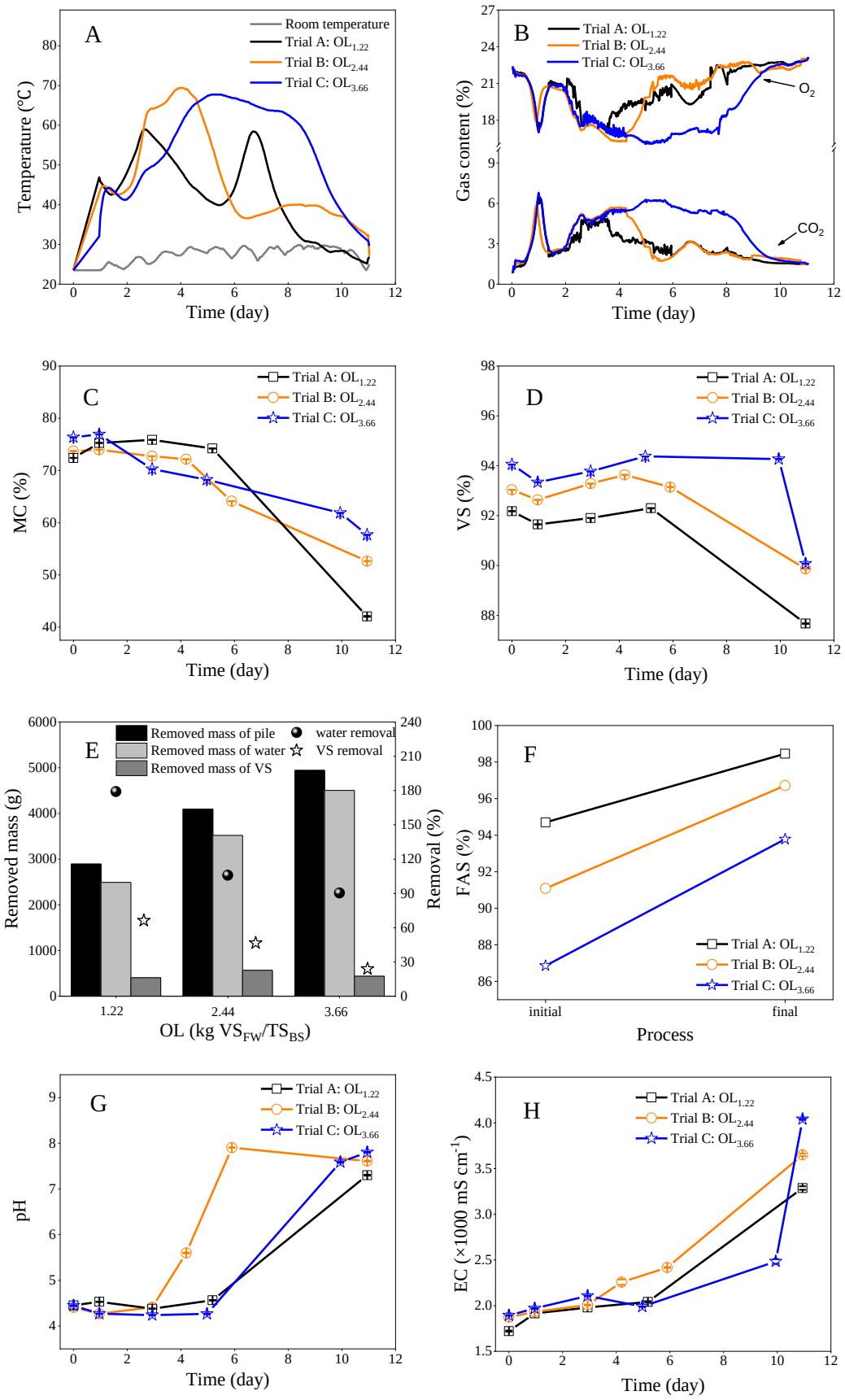
As shown in Fig. 1C and 1D, MC of the three trials decreased from the initial values of 72.39, 73.72, 76.36% to the finals of 42.05, 52.62, 57.62%, respectively, and VS also declined from 92.18, 93.03, 92.15% to 87.67, 89.87, 90.07%, respectively. The resulted water removal in these three trials were 179.05, 105.92, and 90.34%, respectively, and the VS removal were 66.25, 46.63, and 23.88%, respectively (Fig. 1E). Moreover, the

MC of Trial A on the 0.76th day was 75.27%, which was much higher than its initial value of 72.39%, while that in Trial B and C were 73.96 and 76.92%, respectively, and were only a little bit higher than their initial values of 73.72 and 76.36%. The mass difference in those three trials might be the reason. The mass of Trial B and C were 6.48 and 8.80 kg, respectively, while that of Trial A was only 4.20 kg. Thus, for Trial A, a considerable MC increase might be caused when the same amount of water was taken away from the bottom layer into the middle (rewet effect) (Yang et al., 2021), but for Trial B and C, the resulted MC increase might not be so evident since their mass was much larger than that in Trial A.

For conducting the FW bioevaporation process, FW needs to be added to the BS. The added FW would fill the voids of the airspace, so a higher OL means more FW was added and then a lower initial FAS was obtained (Fig. 1F). With the evaporation of water and degradation of VS, channels of air transfer were created, resulting in an increase of FAS in all trials. The FAS of the three trials increased from the initial values of 94.70, 91.09, and 86.86% to the finals of 98.46, 96.72, and 93.78%, respectively. Since the removed mass of water and VS in Trial B and C were higher, their increased FAS were relatively higher than that in Trial A.

pH variations were comparable in the three trials (Fig. 1G). Due to the production of acid substances, the pH of the three trials all decreased during the thermophilic phase, and with the consumption of the acid substances and production of ammonia, it increased again and finally became slightly alkaline (Liu et al., 2020). EC is an indicator of water purity, and a higher EC representing more ions existed (Waqas et al., 2018). As shown in

175 Fig. 1H, as the mineralization of nitrogen, transforming of complex organic substances to
176 lower molecular weight compounds proceeded, the EC increased significantly in different
177 OL ($p < 0.01$) (Haug, 1993). Moreover, the increase of EC in higher OL trials was higher,
178 indicating more ions were produced in them.



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180 Fig. 1. Bioevaporation performance under different OL: (A) temperature profile; (B) O_2

and CO_2 concentration in the exit gas; (C) MC variation; (D) VS variation; (E) removed mass and removal of water and VS; (F) FAS; (G) pH variation; (H) EC

3.2 OL on biochemical fractions degradation and enzymatic activities

Fig. 2A and 2B show the degradation ratio and degraded mass of biochemical fractions at the end of the bioevaporation process and Fig. 2C is their detailed temporal variation. As can be seen, the degradation ratio of WEOC in Trial A, B and C were 70.9, 15.7, and 33.0%, respectively, with 37, 16, and 27 g of the WEOC were degraded. This indicated that Trial A favored the degradation of WEOC compared to the other two trials. Because the variation of WEOC was the balance between its production rate and degradation rate, it seems that a lower OL led to faster degradation of WEOC to some extent (Charest et al., 2004; Huang et al., 2006). As to the detailed variation, WEOC in Trial A increased from the initial value of 44.6 $mg \cdot g^{-1}$ DM to 50.4 $mg \cdot g^{-1}$ DM during the first 2.93 days, and then decreased to 19.9 $mg \cdot g^{-1}$ DM at the end of the process. For Trial B and C, WEOC gradually decreased from the initial values of 59.5 and 75.6 $mg \cdot g^{-1}$ DM to the finals of 39 and 33.3 $mg \cdot g^{-1}$ DM, respectively. By comparing the overall hydrolytic enzymatic activity in Fig. 3, it was interesting that the activities of lipase and protease in Trial A increased from the beginning to the peak on day 0.96 (46.9 °C). Those of amylase, xylanase, and cellulase peaked on day 2.93 (57.9 °C), then decreased until the end of the process. Thus, increased hydrolase activity led to the increase of WEOC in Trial A during the first 2.93 days since the hydrolytic reactions there were quite strong. However, for Trial B and C, although the initial enzymatic activity were higher than those in Trial A due to the higher initial biochemical fraction's concentration, the hydrolase activity

mostly all had a decreasing trend.

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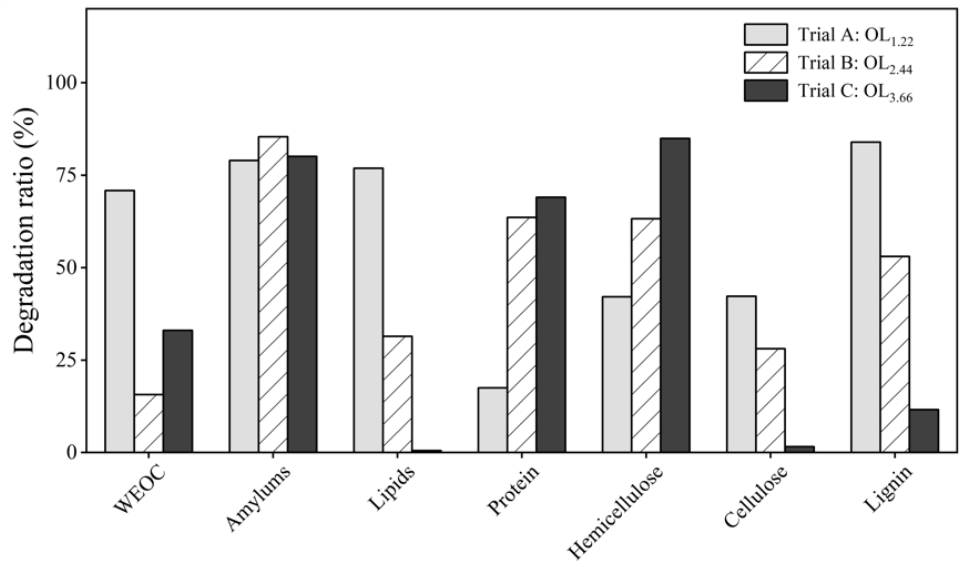

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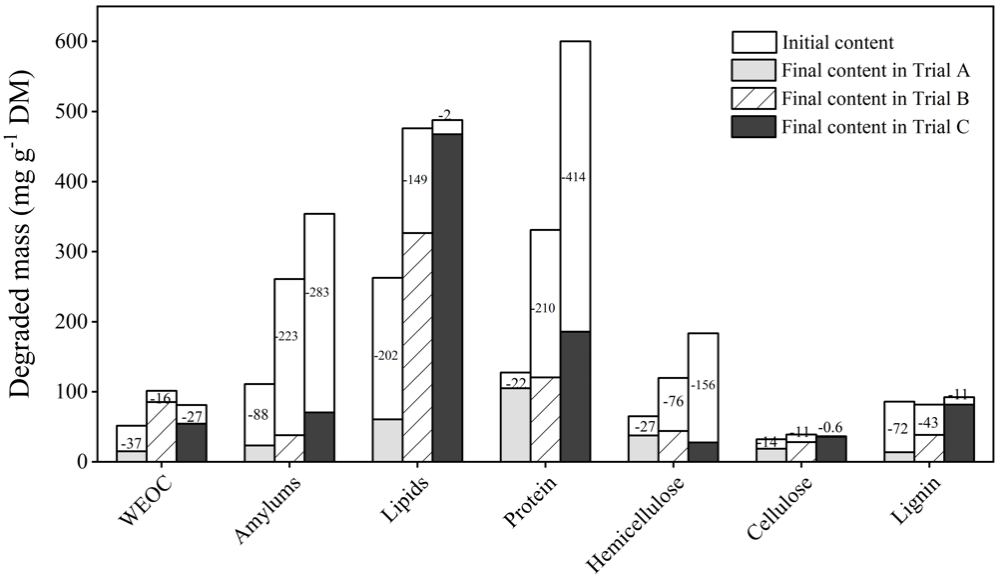

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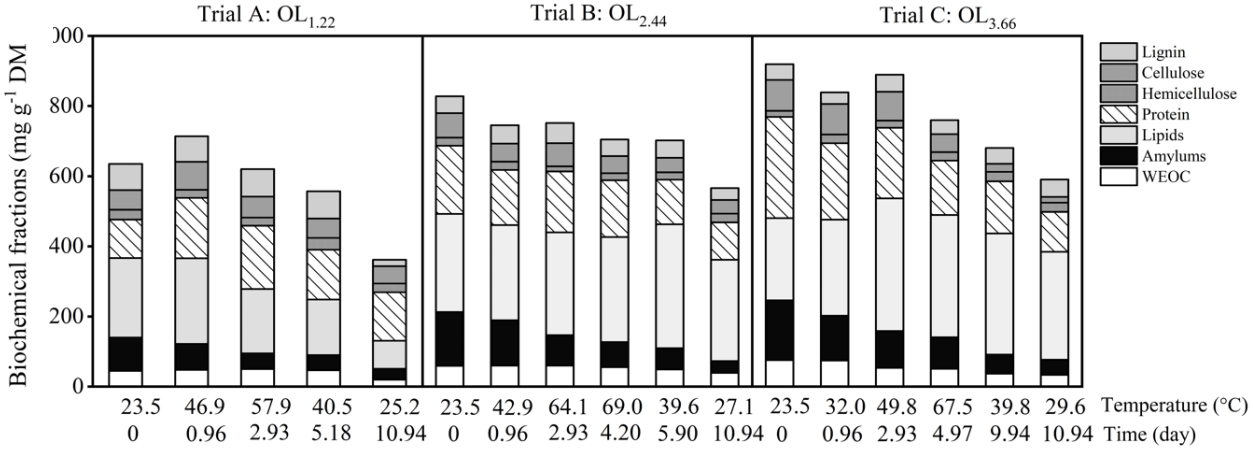

207 Fig. 2. Biochemical fractions degradation under different OL: (A) degradation ratio; (B)

208 degraded mass; (C) detailed temporal variations.

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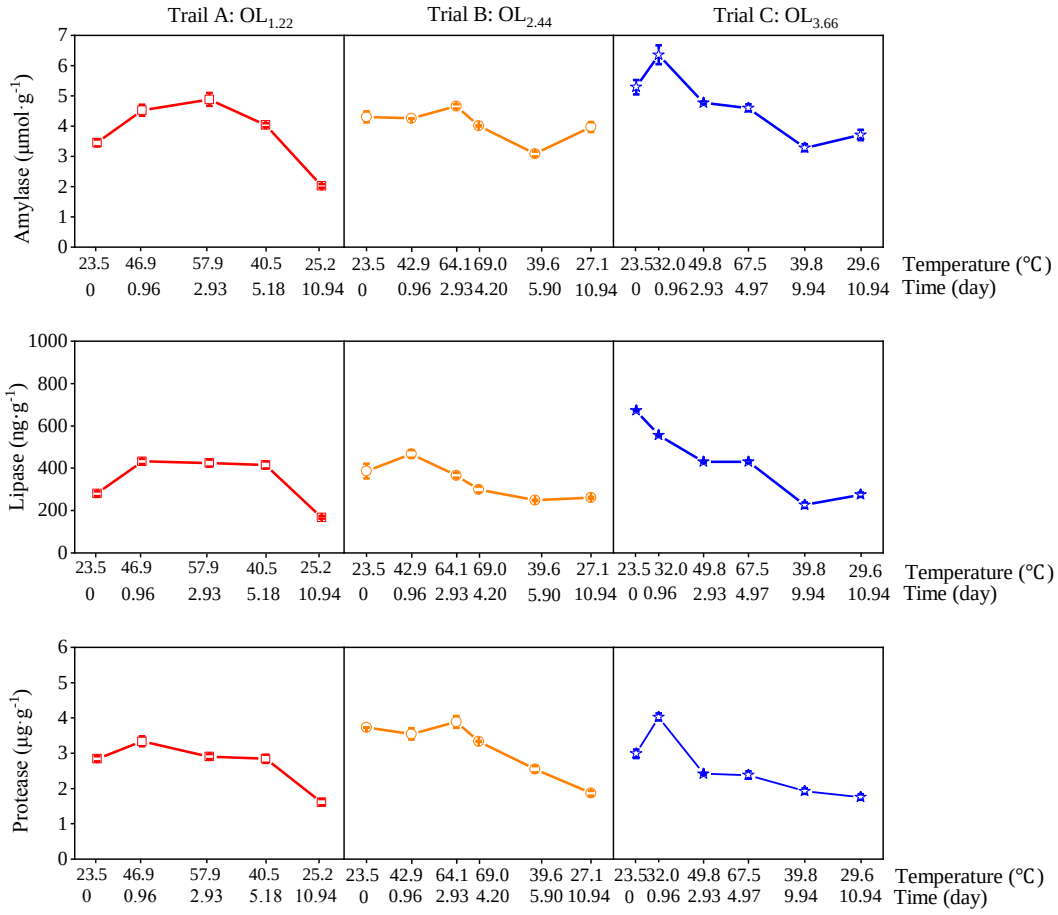

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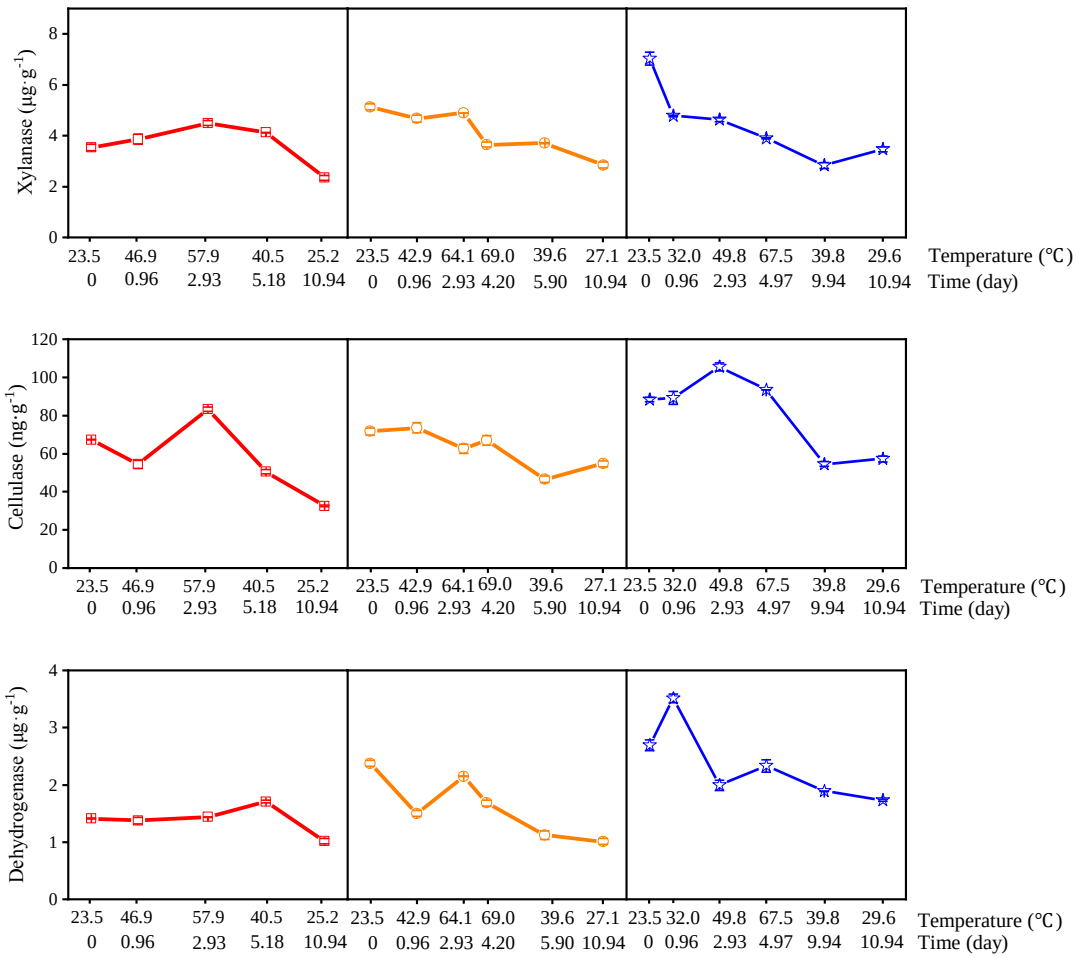

Fig. 3. Variations of enzymatic activity under different OL.

As the readily biodegradable organic matters, amylums had a comparable degradation ratio in those three trials and around 78.97−85.38% of the amylums were degraded during the bioevaporation process. The degraded amylums in Trial A, B, and C were 88, 223, and 283 g, respectively, and the fast degradation of amylums all occurred during the warming period of the process (first 2.93 days) regardless of the OL. Especially for Trial A, due to the increased amylase activity in the first 2.93 days, the degradation of amylums almost occurred during that period and nearly ceased after that. Thus, for amylums, the OL had little effect on its degradation.

The degradation ratio of lipids in Trial A, B, and C were 76.84, 31.39, and 0.49%,

222 respectively, and the degraded mass were 202, 149, and 2 g, respectively. Lipids content
223 in Trial A decreased from 226.5 mg·g^{-1} DM to 80.2 mg·g^{-1} DM, and those in Trial B and
224 C increased instantly, and the increase was probably due to the fast degradation of other
225 biochemical fractions like amylums. Those results showed that the high OL severely
226 inhibited lipids degradation. Given that a decrease in the aliphatic compounds caused by
227 oxidation has widely reported (Baddi et al., 2003; Kim & Yu, 2005; Smidt & Parravicini,
228 2009), indicating the aliphatic component was barely decomposed in the anaerobic
229 condition, thus it seems that it was the insufficient oxygen in the high OL inhibited the
230 lipids degradation. In the pig manure composting, Ge et al. (2014) also reported that the
231 aliphatic compounds could be easily oxidized into aromatic compounds in the aerobic
232 layer of pig manure, but in the anaerobic layer the aliphatic compounds were barely
233 decomposed. Moreover, the lipase activity in Trial A and B increased to peak on day 0.96
234 (46.9 °C), and maintained high until day 2.93 and 5.18, while that in Trial C steadily
235 decreased. This further showed that sufficient oxygen stimulated the lipase activity and
236 caused the degradation of lipids in Trial A, while the more added FW impeded the oxygen
237 transfer severely and the anaerobic condition created by high OL inactivated the lipase
238 activity and thus hindered the lipids degradation, especially in Trial C.

239 For protein, the degradation ratio in Trial A, B, and C were 17.48, 63.58, and 69.05%,
240 respectively, with 22, 210, and 414 g protein degraded. The initial increase in protein
241 content in Trial A ascribed to the degradation of other biochemical fractions like lipids
242 and amylums. It seems that the protein degraded in lower OL was much less than those
243 in higher OL. For protein is the main component of extracellular polymeric substances

and cell membranes, the observed level of protein was a balance of its formation and consumption. Because the biomass yield in aerobic metabolism is around 0.4 g volatile suspended solids (VSS)/g COD and much higher than that in anaerobic condition (0.06 g VSS/g COD) (Tchobanoglous et al., 2003), the much less degraded protein in Trial A might be owing to the fast proliferation of microorganisms in aerobic condition, while the observed higher amount of degraded protein in Trial B and C especially in Trial C was the result of the slow growth of anaerobic microorganisms. Moreover, our recently published study also observed that during the initial period of co-bioevaporation process in which microorganisms proliferated faster, the degraded protein in top layer of the reactor was less than that in middle and bottom layers as the higher temperature in top layer favored the growth of microorganisms (Yang et al., 2021). Also, from the protease variation, it suggested that the protease activity in Trial A increased and maintained high until day 5.18, indicating the degradation of protein actually proceeded well. This further verified that it was the fast growth of aerobes in Trial A that led to the less observed degradation of protein.

As to the lignocellulose including hemicellulose, cellulose, and lignin, the degradation of cellulose and lignin in Trial A were 42.3 and 84.8%, respectively. Considering their degradation in Trial B were 28.1 and 53.0%, and in Trial C were as low as 1.6 and 11.6%, respectively, so the degradation of cellulose and lignin in Trial A were much higher than those in Trial B and C. Referring to the degraded mass, the degraded cellulose in Trial A, B, and C were 14, 11, and 0.6 g respectively, and the degraded lignin were 72, 43, and 11 g respectively. From their detailed content variation, it seems that the

degradation of cellulose and lignin in Trial A occurred from day 5.18 to day 10.94, which was nearly the end of the process, and during which the temperature decreased from 40.5 to 25.5 °C. The reason was owing to the inhabitation of their main decomposer, fungi. Because most of the fungi are aerobic and multiplies at the cooling period of biodegradation process especially when the temperature is below 50 °C (Gu et al., 2017; Yang et al., 2021; Yang et al., 2008), the relatively abundant oxygen and suitable temperature in the later period of Trial A caused the proliferation of fungi and the degradation of cellulose and lignin. For Trial B and C, as the oxygen transfer was interfered, thus the growth of fungi was inhibited, which impeded the cellulase activity and cellulose degradation especially in Trial C. Different from cellulose and lignin, the degradation of hemicellulose posed a different pattern that its degradation in Trial C (156 g) was much higher than that in Trial A (27 g) and B (76 g), showing that a higher OL preferred the hemicellulose degradation.

Specific biochemical fractions presented different degradation patterns in different OL bioevaporation processes, except for amylums. The variation of WEOC was more significant in higher OL. The higher aerobic layer was beneficial for the degradation of lipids, while the rapid biomass yield in lower OL made protein degraded less. Higher OL impeded the degradation of cellulose and lignin for it limited the growth of fungi, while preferred the degradation of hemicellulose.

The WEOC, amylums, hemicellulose, cellulose, and lignin were all belonged to carbohydrates, so the sum of their degraded mass could show the degradation of carbohydrates. By summation, the degraded mass of carbohydrates in Trial A, B, and C

were 238, 369, and 477 g, respectively, with the degradation ratio of 68.6, 61.2, and 63.9%. Thus, it seems that even though the degradation ratio of WEOC, cellulose, and lignin was higher in lower OL, as a whole, OL had little effect on carbohydrates degradation ratio.

In summary, although the initial enzymatic activity in Trial B and C were higher than those in Trial A due to the higher level of biochemical fractions in them, the subsequential enzymatic activity mostly had a decreasing trend during the bioevaporation process. Unlikely, the enzymatic activity in Trial A increased during the process, which accelerated the biochemical fractions degradation. Except for protein and hemicellulose, the WEOC, lipids, cellulose, and lignin in Trial A degraded more and the degradation ratio was higher than those in Trial B and C. For high biomass yield in Trial A, the resulted protein degradation was lower than that in Trial B and C. The aerobic condition in Trial A facilitated the lipids degradation and the insufficient oxygen in Trial B and C impeded their degradation.

3.3 OL on metabolic heat generation

The heat generated from biochemical fractions degradation and their contribution to metabolic heat (Q) was calculated based on Eq. (1). It was assumed to be the product of combustion heat and degraded mass (Zhao et al., 2010).

$$Q = Hc \cdot m \quad (1)$$

where, Hc was the combustion heat of biochemical fractions and m was the mass of degraded biochemical fractions. Haug (1993) gave the value of Hc for carbohydrates as 17.4 kJ/g, for lipids as 39.3 kJ/g, and for protein as 23.4 kJ/g. WEOC, amylums,

hemicellulose, cellulose, and lignin were all summed up in carbohydrates for convenience in computation.

The calculated metabolic heat by the degradation of biochemical fractions was given in Table 2 and Fig. 4. As shown in Table 2, the total metabolic heat generated from Trial A, B, and C were 12592, 17215, and 18090 kJ, respectively. Among them, the heat generated from lipids degradation were 7935, 5872, and 93 kJ, respectively, accounting for 63.0, 34.1, and 0.5% of the total metabolic heat (Fig. 4). As for protein, because its degradation was quite different from lipids in different OL, the heat generated from protein degradation in Trial A, B, and C were 521, 4925, and 9695 kJ, respectively, accounting for 4.1, 28.6, and 53.6% of the total metabolic heat. For carbohydrates which included WEOC, amylums, hemicellulose, cellulose, and lignin, there were 4136, 6418, and 8302 kJ of metabolic heat came from their degradation in Trial A, B, and C, respectively, which contributed 32.8, 37.3, and 45.9% to the total metabolic heat.

From the above data, the lipids degradation contributed 63.0% to metabolic heat in lower OL, but only contributed 0.5% in higher OL. Conversely, the heat from protein degradation reached as high as 53.6% of the total metabolic heat in higher OL, but it was only accounted for 4.1% in the lower OL.

Table 2. Metabolic heat generated from biochemical fractions degradation in different OL

Biochemical fraction	Degradation mass (g)			∑Q (kJ)			Contribution (%)		
	$OL_{1.22}$	$OL_{2.44}$	$OL_{3.66}$	$OL_{1.22}$	$OL_{2.44}$	$OL_{3.66}$	$OL_{1.22}$	$OL_{2.44}$	$OL_{3.66}$
Lipids	201.9	149.4	2.4	7935	5872	93	43.7	20.5	6.3
Protein	22.3	210.5	414.3	521	4925	9695	4.8	28.9	46.4
Carbohydrates	237.7	368.8	477.1	4136	6418	8302	51.5	50.6	53.4
Total	461.9	728.7	893.8	12592	17215	18090			

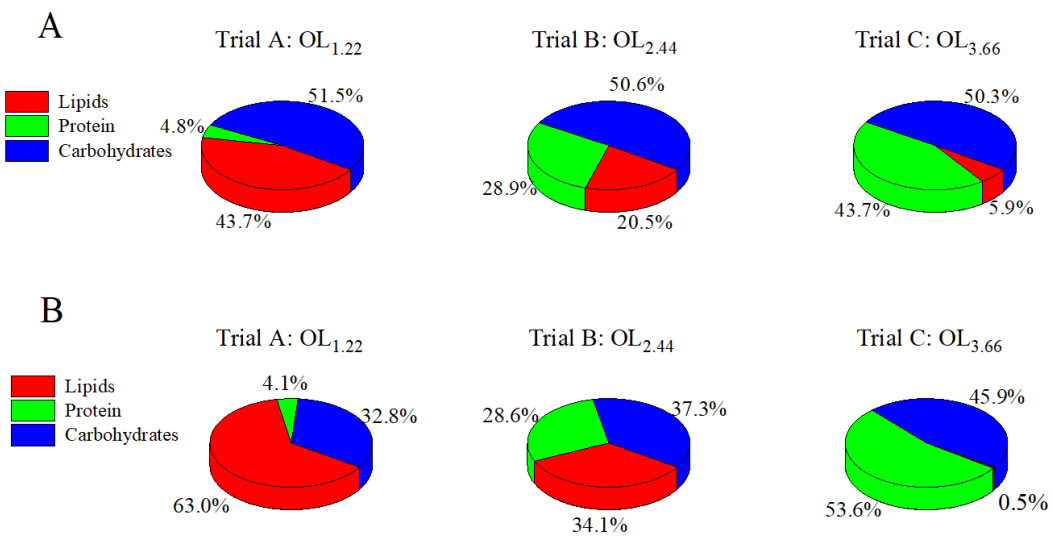

Fig.4. OL on biochemical fractions degradation (A); their contribution to metabolic heat (B).

4. Conclusion

The degradation pattern of biochemical fractions under different OL and their contribution to metabolic heat during FW bioevaporation were investigated in this study. OL significantly influenced the degradation pattern of biochemical fractions. Because the more added FW in higher OL hindered the oxygen transfer, the degradation of lipids in the higher OL was impeded. While for lower OL, the more porous structure and abundant oxygen accelerated the lipids degradation, and the rapid proliferation of aerobes offset the degraded protein. Thus, 76.8, 31.4, and 0.5% of the lipids were degraded in Trial A, B, and C, whereas the corresponded degradation ratio of protein were 17.5, 63.4, and 69.1%, respectively. Moreover, in Trial A, 63.0% of the metabolic heat was from lipids degradation, but only 4.1% was from protein. However, in Trial C, lipids degradation contributed 0.5% to the metabolic heat, while protein degradation contributed 53.6%. For carbohydrates, 61.2−68.6% of the carbohydrates were degraded, contributing 32.8−45.9%

to metabolic heat. Thus, the degradation ratio of carbohydrates was not affected by the OL.

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Availability of data and materials Not Applicable.

Author Contribution Statement

Benqin Yang: Resources, Methodology, Investigation, Writing-Original Draft, Review & Editing, Data analysis.

Die Hu: Experiment, Data Curation, Review & Editing.

Yanmei Liu: Methodology, Review & Editing.

Zhiqiang Lin: Writing, Review & Editing.

Xiandong Zhou: Review & Editing.

Qian Pan: Review & Editing.

Hongxu Zhu: Methodology, Writing- Reviewing and Editing.

Xuejun Pan: Review & Editing, Resources, Supervision.

All authors read, revised and approved the manuscript.

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Compliance with ethical standards

Declaration of interests The authors declare that they have no potential conflicts of interest with respect to the research, authorship or publication of this article.

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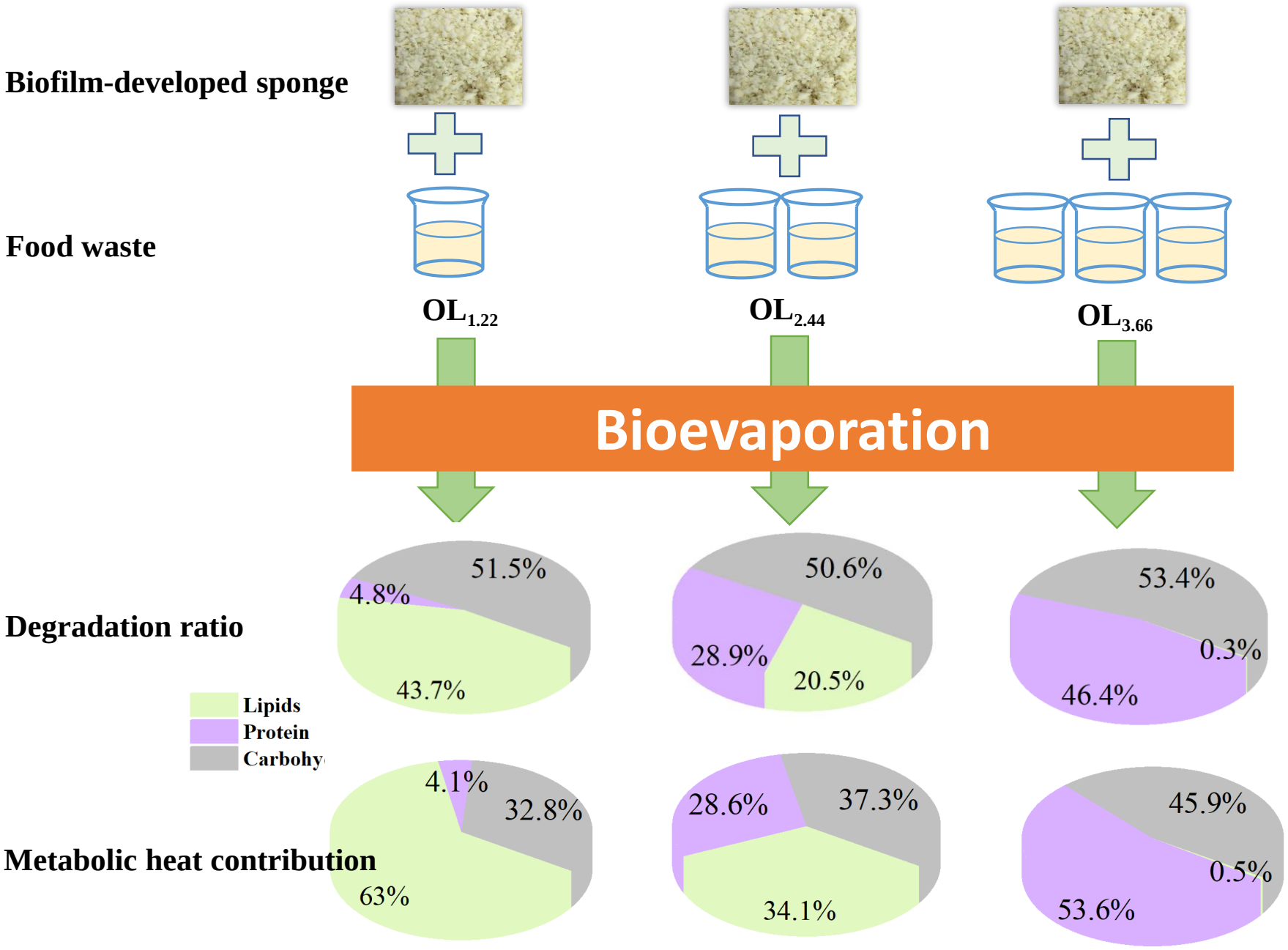

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Waste Management

A comprehensive evaluation of the biodrying of organic solid waste using a recyclable amendment, insights of energy saving and emission reduction

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Manuscript Number:	WM-22-2254
Article Type:	Full Length Article
Section/Category:	Biological treatment
Keywords:	Biodrying; amendment; organic solid waste; carbon emissions; Life cycle assessment
Abstract:	This study developed a recyclable amendment (RA) for the biodrying of organic solid waste (OSW) and assessed the effect of RA on biodrying. Carbon emissions analysis and life cycle assessment (LCA) were conducted for different treatments of OSW (landfilling, incineration, composting, and biodrying). The results showed that the biodrying of OSW using RA significantly reduced the abundance of pathogenic microorganisms and improved the net energy of the substrate system. The carbon emissions analysis and LCA showed that biodrying using RA resulted in the lowest carbon emissions among the four treatments. Biodrying of OSW using RA had low potential environmental impact, effectively alleviating the need to add a conventional organic amendment, especially the kitchen waste represented the lowest total environmental impact potential. The results above showed that biodrying using RA effectively biodried wastes and achieved stabilization. The RA could be reused to save resources, which was consistent with the goals of achieving a peak carbon dioxide emissions and carbon neutrality.

HIGHLIGHTS:

- The experiment developed a recyclable amendment to condition the biodrying.
- Adding the recyclable amendment resulted in good heating and satisfactory drying.
- Biodrying achieved low carbon emissions and potential environmental impact.
- Biodrying of kitchen waste performed well in life cycle assessment.

A comprehensive evaluation of the biodrying of organic solid waste using a recyclable amendment, insights of energy saving and emission reduction

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Abstract

This study developed a recyclable amendment (RA) for the biodrying of organic solid waste (OSW) and assessed the effect of RA on biodrying. Carbon emissions analysis and life cycle assessment (LCA) were conducted for different treatments of OSW (landfilling, incineration, composting, and biodrying). The results showed that the biodrying of OSW using RA significantly reduced the abundance of pathogenic microorganisms and improved the net energy of the substrate system. The carbon emissions analysis and LCA showed that biodrying using RA resulted in the lowest carbon emissions among the four treatments. Biodrying of OSW using RA had low potential environmental impact, effectively alleviating the need to add a conventional organic amendment, especially the kitchen waste represented the lowest total environmental impact potential. The results above showed that biodrying using RA effectively biodried wastes and achieved stabilization. The RA could be reused to save resources, which was consistent with the goals of achieving a peak carbon dioxide emissions and carbon neutrality.

Keywords: Biodrying; amendment; organic solid waste; carbon emissions; life cycle assessment

1. Introduction

Organic solid waste (OSW) mainly includes but not limited to sewage sludge, kitchen waste, and agricultural waste. It is characterized by high organic content, decay

easily, and breeds pathogenic microorganisms. Proper treatment is needed to avoid secondary pollution (Campuzano and Gonzalez-Martinez, 2016). Disposing of OSW consumes energy and emits greenhouse gases, both of which are detrimental to the sustainability of the solid waste treatment industry and meeting the challenge of climate changes (Moghadam et al., 2021). In view that the global warming problem is now one of the hottest issues in the world, reducing carbon emissions is key for achieving carbon peak and carbon neutrality in the solid waste industry (Wang et al., 2020). The Life Cycle Assessment (LCA) method is widely used as an analytical tool in the field of solid waste management (Mulya et al., 2022). It is frequently used to assess the full life cycle of a product or service system, quantifying its energy inputs and outputs, and potential environmental impacts (Zhou et al., 2018a). The research based on LCA shows that incineration had better environmental impacts than landfilling with respect to acidification and eutrophication potential. Composting reduces the impacts of global warming, ecotoxicity, eutrophication, and fossil fuel consumption (Zhou et al., 2018b; Keng et al., 2020).

Biodrying is an economical and energy-saving method to treat biomass waste, and uses the energy generated by microorganisms to degrade organic matter and maximize water removal from the material (Guerra-Gorostegi et al., 2021). Biodrying product has low moisture, high heating value, and stable composition, allowing its use as a biofuel and soil amendment (Ma et al., 2022). As such, biodrying is an important approach for realizing the resource utilization of OSW in a way that saves energy and reduces

emissions (González et al., 2019).

During biodrying, amendments are often added to improve the physical and chemical indexes of the initial material and adjust the pile structure to achieve better biodrying performance (Zhang et al., 2018; Yang et al., 2014). Conventional amendments, such as straw and sawdust, have good conditioning performance, but require large amounts of consumption which are difficult to recover, costly, and mostly disposable. The development of synthetic amendments has received increasing attention. Biochar has been utilized to enhance humification by lowering nitrogen loss, accelerating organic matter degradation, and increasing microbial activity (Zainudin et al., 2020). The addition of ceramsite, recycled ceramsite and activated alumina balls accelerated the composting process and had good recycling potential (Wang et al., 2019; Li et al., 2019). These amendments are easy to screen and can be reused. However, these synthetic amendments mainly play the role of composting bulking agents. There are few systematic studies on synthetic amendments in the field of biodrying of OSW. As a result, developing a biodrying amendment that can be reused and facilitates dewatering has the potential to improve biodrying efficiency while lowering costs.

In this study, a silicate-based recyclable amendment (RA) developed in a laboratory was added to separate pile samples of sewage sludge, kitchen waste, and agricultural waste, to investigate its effect of biodrying and environmental impact. To compare the biodrying performance of RA on OSW, first, biodrying of sewage sludge, kitchen waste, and agricultural waste using RA was conducted. Second, the bacterial

and fungal communities were analyzed for the OSW during biodrying. Third, the carbon emissions of four sewage sludge treatments (landfilling, incineration, aerobic composting, and biodrying) were compared. Finally, the environmental performance of the four sewage sludge treatments and the biodrying of OSW with RA was evaluated using the LCA method. Based on energy saving and emission reduction, the study provides information for the selection of biodrying amendments and environmental management of the OSW treatment process.

2. Materials and methods

2.1 Materials and processes

The test sewage sludge was collected from a municipal sewage treatment plant; kitchen waste (1 cm particle sizes), was acquired from the university canteen; pig manure and biogas residue were collected from the rural biogas station. Sawdust was collected from a furniture factory (0.2 cm particle sizes), and straw was acquired from a farm (1 cm particle sizes). The experimental RA was made using silicate, volcanic rock, sand, and metakaolin (RA is a cube with a side length of 3cm). The parameters of the biodrying materials are shown in Table S1.

< Table S1 >

2.2 Analysis methods

2.2.1 Recyclable amendment performance test

The first step was to mix the RA amendment raw materials with water. The mortar

mixture was then poured into the mold to ensure that the mold was internally filled with a smooth surface. The mold was removed after the mortar was formed to retrieve the amendment and test its density.

The mechanical properties of the RA were tested by placing the RA in a 200 kN microcomputer-controlled SANS pressure testing machine, ensuring flat upper and lower surfaces during the test.

2.2.2 Key parameter measurements

To investigate biodrying performance, the temperature of each pile was measured daily using a temperature sensor, and then the average value of the upper, middle and lower layers was calculated. The moisture content (MC) and volatile solids (VS) of the pile samples were determined using the gravimetric method.

The pile samples were mixed with deionized water at a ratio of 1:10 (weight: volume), and centrifuged at 4000 r/min for 10 minutes. The pH and the electrical conductivity (EC) were measured in the supernatant using a pH and EC meter, respectively.

The seed germination index (GI) is a test used to assess the phytotoxicity of biodrying products (Kebibeche et al., 2018). A rapeseed germination experiment was conducted to GI as follows (Cai et al., 2021): 5 mL of the supernatant was placed in a petri dish lined with filter paper, and 20 rapeseeds were evenly distributed in each petri dish. A separate dish with deionized water instead of the extract was set as the control. For each sample, the seeds were incubated at 25 ℃ for 48 h. The root length emerging

from the seeds was measured with a ruler, and the GI was calculated using the following formula:

$$GI = \frac{\text{Seed germination of treatment (\%)} \times \text{Root length of treatment}}{\text{Seed germination of control (\%)} \times \text{Root length of control}} \quad (1)$$

2.2.3 Relative abundance of microorganisms

DNA was extracted using DNA extraction kit for the corresponding sample. The integrity, concentration and purity were measured using the NanoDrop One (Thermo Fisher Scientific, MA, USA). Genomic DNA was used as template for PCR amplification using specific primers with barcode and TaKaRa Premix Taq® Version 2.0 (TaKaRa Biotechnology Co., Dalian, China) according to the selection of sequencing regions. To study the microbial species composition and diversity information of the samples, the clean tags of all samples were clustered after splicing and filtering (97% similarity was selected by default), and OTU (Operational Taxonomic Unit) was generated, while singleton OTU and chimeras were removed. The sequence information in the OTU_table was extracted at the genus level, the relative abundance of species was calculated, and the top three species were selected to plot the relative abundance of species.

2.2.4 Carbon emissions analysis

In this study, the emission factor method was used to account for carbon emissions generated by the different disposal routes of sewage sludge at the plant scale. The emission factor is the result of long and repeated practical experience (Liu et al., 2014).

In this study, the following formula was used in the calculation (The nomenclature of terms involved in carbon emissions formula is shown in Table S2):

$$PE_{TD,y} = PE_{elec,y} + PE_{fuel,on-site,y} + PE_{d,y} - BE_{compost,y} \quad (2)$$

$$PE_{elec,y} = EG_{PJ,FF,y} \times CEF_{elec} \times (1 + TDL_y) \quad (3)$$

$$PE_{fuel,on-site,y} = F_{consys} \times NCV_{fuel} \times EF_{fuel} \quad (4)$$

$$PE_{tran,y} = \sum_{i}^{n} NO_{vehicles,i,y} \times DT_{i,y} \times VF_{cons,i} \times NCV_{fuel} \times D_{fuel} \times EF_{fuel} \quad (5)$$

< Table S2 >

The carbon emissions inventory in this study was generated for CO_2 and CH_4, and the final result was converted to CO_2 emissions. Expressed as tons of CO_2 equivalent per ton (tCO2e), which is the number of tons of a gas multiplied by the index of its greenhouse effect production.

2.2.5 LCA analysis

LCA is an analytical method for aggregating and assessing the potential environmental impacts of all inputs and outputs of a product or service system throughout its life cycle (ISO, 2006). GaBi9 software was used to evaluate the life cycle of OSW. The potential environmental impact value was calculated using IBM SPSS Statistics 26.0. The data were standardized and weighted.

3. Results and discussion

3.1 Recyclable amendment characteristics

The density of the RA used was 2.10 $g \cdot cm^{-3}$, which was a lower density compared

to ordinary concrete (2.5 g·cm^{-3}) and which was close to the soil density range. The ultimate compressive strength of the RBA was 46.32 MPa and 40.76 MPa before and after biodrying, respectively (Fig. 1.). The recovery rates of the RBA at the end of sewage sludge biodrying, kitchen waste biodrying and agricultural waste biodrying were 98%.

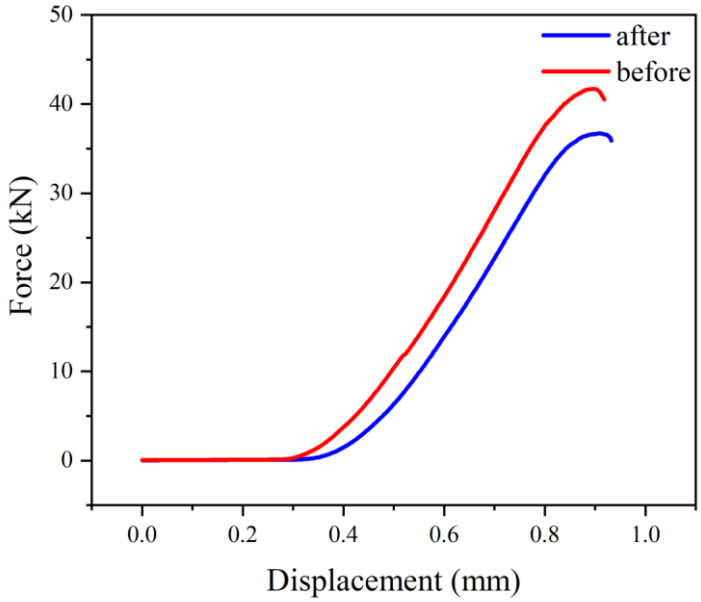

Figure 1. The compressive strength before and after the use of recycle amendment.

3.2 Biodrying performance

3.2.1 Fundamental parameters

The duration of the thermophilic phase (> 50℃) for the Ts, Tk and Ta was as follows: 9d, 11d, and 8d, respectively. The peak pile temperatures of Ts, Tk, and Ta were 74.1 ℃, 61.5 ℃, and 61.9 ℃, respectively (Fig. 2a). Each pile was kept above 50℃ for more than 7d and above 55℃ for more than 5d, reaching the stabilization standard (Sakarika et al.,

2019), and meeting the compost sanitation requirements specified in the Chinese national standard (GB7989-87). After 21-days biodrying, the MC of the Ts, Tk, and Ta decreased significantly from 59.6%, 68.79%, and 64.55% to 43.7%, 23.31%, and 37.33% ($P < 0.05$), respectively; this reflects decreases of 26.7%, 66.1%, and 42.2%, respectively (Fig. 2b). Compared with the conventional amendment, biodrying with RA resulted in higher heating and cooling rates in the piles, longer durations of thermophilic phase, and a higher water removal rate (Yuan et al., 2019; Yao et al., 2021). Biodrying with RA appears to provide benefits over conventional amendments for lowering the moisture content of piles, according to the above findings. It is due to the high water absorption capacity of RA, which absorbs water from sludge, and its cubic shape improves the volume porosity and promotes the evaporation of water.

VS reflects the changes of total organic matter during biodrying (Fig. 2c). The highest and lowest level of organic matter was found in Tk and Ts. The VS of Tk, Ta, and Ts samples degraded from 96.41%, 80.98%, and 42% on day 0 to 93.43%, 65.02%, and 33.6% at the end of biodrying, respectively. The maximum degradation rate of organic matter was 19.7% in Ta.

The pH of the three OSW piles increased overall, as shown in Fig. 2d. The initial pH values in Ts, Tk, and Ta were 6.38, 4.36, and 6.21, respectively. The pH values in Ts, Tk, and Ta increased rapidly to their peaks of 7.23, 6.96, and 9.18 on day 6, day 6, and day 18, respectively, and then stabilized at 6.68, 6.88 and 8.75, respectively. The EC of Ts and Tk increased from 1.37 $mS \cdot cm^{-1}$ and 0.46 $mS \cdot cm^{-1}$ on day 0 to 1.67 $mS \cdot cm^{-1}$ and 0.78

208 $mS \cdot cm^{-1}$ on day 21, is represented in Fig. 2e. The EC of Ta was 5.08 $mS \cdot cm^{-1}$ on day 0,
209 decreased significantly to 2.8 $mS \cdot cm^{-1}$ ($P < 0.05$) on day 10, and then increased to 3.9
210 $mS \cdot cm^{-1}$ at the end of biodrying. The EC of all biodrying products was less than 4
211 $mS \cdot cm^{-1}$, with no significant toxic effects on plants as fertilizers（Zhao et al., 2018).

212 3.2.2 Seed germination index

213 The GI of the Ts, Tk, and Ta were 145.67%, 150.03%, and 156.10%, respectively
214 (Fig. 2f). After biodrying, all exceeded 80%, which indicated that the products were
215 highly stabilized (Jakubus et al., 2018). Using natural amendments (leaf litter, sawdust,
216 wheat straw) to biodrying kitchen waste, the GI value of biodrying products was
217 32.4%~114.7% (Wang et al., 2016). The addition of the RA resulted in a higher degree
218 of stabilization of the biodrying products.

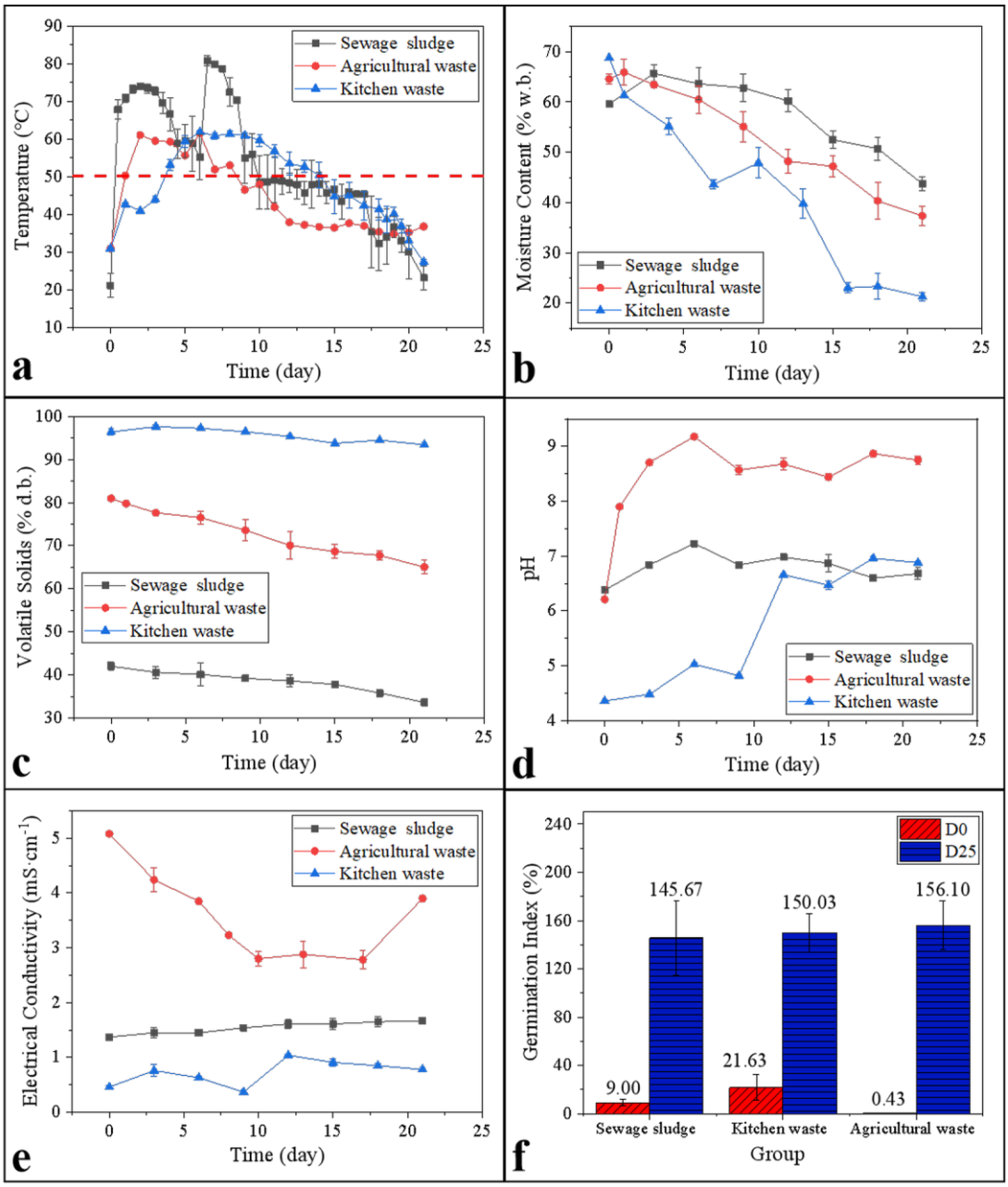

Figure 2. Variations in the indicators during biodrying in Ts, Tk, and Ta. (a: Temperature, the red dashed line at 50℃ in the figure is the boundary of the thermophilic phase; b: Moisture content (MC); c: Volatile solids (VS); d: pH; e: Electrical conductivity (EC); f: Germination index (GI)).

3.3 Thermal balance analysis

Before biodrying, the moisture content of Ts and Ta was 59.6% and 64.55%, respectively, and the net energy of the system was 2979.17 $kJ \cdot kg^{-1}$ and 2613.90 $kJ \cdot kg^{-1}$, respectively. After biodrying, their MC decreased to 43.7% and 37.33%, respectively, and the net energy of the system increased to 4337.41 $kJ \cdot kg^{-1}$ and 6053.28 $kJ \cdot kg^{-1}$, respectively (Table 1). It met the requirement of self-sustained combustion, meaning that the net energy of the system was > 3346 $kJ \cdot kg^{-1}$ (Hao et al., 2017). Due to the high level of organic matter content and combustible components of kitchen waste, the net energy of the system of Tk exceeded the threshold before biodrying, which increased by 230% after biodrying. Efficient dewatering increased net system energy, indicating biodrying has a good prospect for heating value resource recovery.

Table 1 The Energy required for biodrying of organic solid waste and energy generated from dry basis of biodrying materials with different moisture contents

Group		Heating value in dry basis /$kJ \cdot kg^{-1}$	Moisture content /%	Energy consumption of drying /$kJ \cdot kg^{-1}$	Energy generated from dry basis incineration /$kJ \cdot kg^{-1}$	Net energy of the system /$kJ \cdot kg^{-1}$
Ts	Before biodrying	12790.57	59.60	2188.22	5167.39	2979.17
	After biodrying	10553.93	43.70	1604.45	5941.86	4337.41
Tk	Before biodrying	19377.70	68.79	2525.64	6047.78	3522.14
	After biodrying	15500.21	23.31	741.28	12370.72	11031.28
Ta	Before biodrying	14058.88	64.55	2369.97	4983.87	2613.90
	After biodrying	11845.96	37.33	1370.58	7423.86	6053.28

3.4 Microbial succession

3.4.1 Analysis of sewage sludge biodrying microbial communities

Fig. 3 shows the relative abundance distribution of fungi and bacteria at the genus level during biodrying. During sewage sludge biodrying, the dominant fungi during the mesophilic phase of biodrying were *Kazachstania* and *Candida,* and the relative aboundance of *Kazachstania* was 26.4%; the dominant bacteria were *Weissella* and *Leuconostoc*, of which *Weissella* accounted for 18.1%. *Apiotrichum* and *Candida* as the dominant fungi of the thermophilic phase, the relative abundance were 36.0% and 18.8%, respectively; the dominant bacteria during the thermophilic phase were *Escherichia-shigella* and *Sphingobacterium*, with the relative abundance of 19.5% and 14.1%, respectively. In the cooling phase, the dominant fungi were *Pseudallescheria* and *Apiotrichum*, of which *Pseudallescheria* accounted for 20.4%; the dominant bacteria were *Enterobacter* and *Escherichia-shigella*, of which *Enterobacter* accounted for 20.0%. As the dominant genus, *Kazachstania* was able to ferment glucose and galactose (Ke et al., 2019). *Weissella*, as lactic acid bacteria, degraded carbohydrates and reduced the cycle of biodrying (Li et al., 2020).

The dominant genera in the biodrying of the sewage sludge were mostly functional microorganisms that degrade carbohydrates and facilitated the biodrying. Pathogenic bacteria identified during sewage sludge biodrying included *Candida* and *Pseudallescheria. Candida* is the most common conditionally pathogenic fungi, and *Pseudallescheria* is a medically important fungus that can cause several diseases in

humans (Rodrigo et al., 2017). The relative abundance of *Candida* before biodrying was 3.4%, and the percentage decreased to 1.66% after biodrying. The removal rate of *Pseudallescheria* reached 99%. Thus, biodrying using RA inactivated pathogenic bacteria to a certain extent.

3.4.2 Analysis of kitchen waste biodrying microbial communities

During the biodrying of the kitchen waste, the dominant fungi during the mesophilic phase were *Trichoderma* and *Candida*, and the relative abundance of *Trichoderma* was 30.2%; *Weissela* accounted for up to 77.2% of the bacteria during the mesophilic phase. *Thermomyces* and *Aspergillus* accounted for nearly 100% of the fungi in the thermophilic phase; *Bacillus* and *Aeribacillus* were the dominant bacteria in the thermophilic and cooling phase. The dominant fungi in the cooling phase were *Aspergillus* and *Thermomyces*, of which *Aspergillus* accounted for 46.9%. As the dominant genus, *Aspergillus* exhibited significant inhibitory activity in the growth process and has shown to have a high capacity for cellulase production (Ma et al., 2021). *Thermomyces* belongs to *Ascomycota*, which can produce heat-stable enzymes, such as heat-stable xylanase and cellulases, and *Ascomycota* has a strong ability to decompose lignocellulose in thermophilic environments, compared to normothermic fungi (Zhang et al., 2021).

The dominant strains throughout the biodrying process were mostly functional microorganisms that degrade organic matter, with significant levels of lignocellulose-degrading microorganisms in the thermophilic and cooling phases. And significant levels of protein and fat degrading microorganisms in the cooling phase. The

pathogenic bacteria detected during kitchen waste biodrying mainly included *Candida*, *Klebsiella*, *Enterobacter*, and *Acinetobacter*. Of these, *Klebsiella* is a conditional pathogen and can cause human pneumonia (Boonsarngsuk et al., 2015). The relative abundance of *Klebsiella* before biodrying was 3.9%. This level dropped to 0.1% after biodrying. The relative abundance of *Candida* before biodrying was 12.6%. This level dropped to 0.05% after biodrying, reflecting a removal rate close to 100%. The removal rates of *Enterobacter* and *Acinetobacter* reached 98.5% and 95.6%, respectively. These results indicated that the number of pathogenic bacteria was significantly reduced after the biodrying of kitchen waste with RA.

3.4.3 Analysis of agricultural waste biodrying microbial communities

During the biodrying of the agricultural waste, trace amounts of identifiable fungi were identified, 99% of which were unknown. The significant bacteria in the mesophilic phase were *Clostridium* and *Anaerococcus*, with a relative abundance of 13.6% and 15.2%, respectively. In the thermophilic phase, *Brevundimonas* and *Thermobifida* are the dominant bacteria, and their relative abundance is 13.4% and 11.7%, respectively. *Flavobacterium* and *Idiomarina* were the dominant bacteria in the cooling phase. *Clostridium*, belonging to the thick-walled phylum, other studies have noted that it can regulate the function of the micro-ecological balance of flora, degrade protein, fat, and carbohydrate, and shorten the biodrying cycle (Luo et al., 2014; Jang et al., 2014). Cellulase produced by *Thermobifida* can decompose cellulose, and *Flavobacterium* can promote the degradation of carbohydrates (Liao et al., 2021; Kraut-Cohen et al., 2021).

The dominant strains detected during the biodrying were mostly functional microorganisms that degrade lignocellulose, carbohydrates, proteins, and fats. It promoted the stabilization and reduction of biodrying substrates. The pathogenic bacteria detected during the biodrying of agricultural waste included *Anaerococcus*, *Terrisporobacter*, and *Helcococcus*. Of these, *Anaerococcus* lacks a complete metabolic enzyme system and performs energy metabolism through anaerobic fermentation, which can cause human diseases (Cobo et al., 2021). The relative abundance of *Anaerococcus* was reduced from 13.6% to 0.01% in the end of biodrying with a removal rate of 99.9%. *Terrisporobacter* was reduced from 6.4% to 0.8%, and *Helcococcus* was removed after biodrying. In conclusion, biodrying of agricultural waste with RA effectively deactivated pathogenic bacteria.

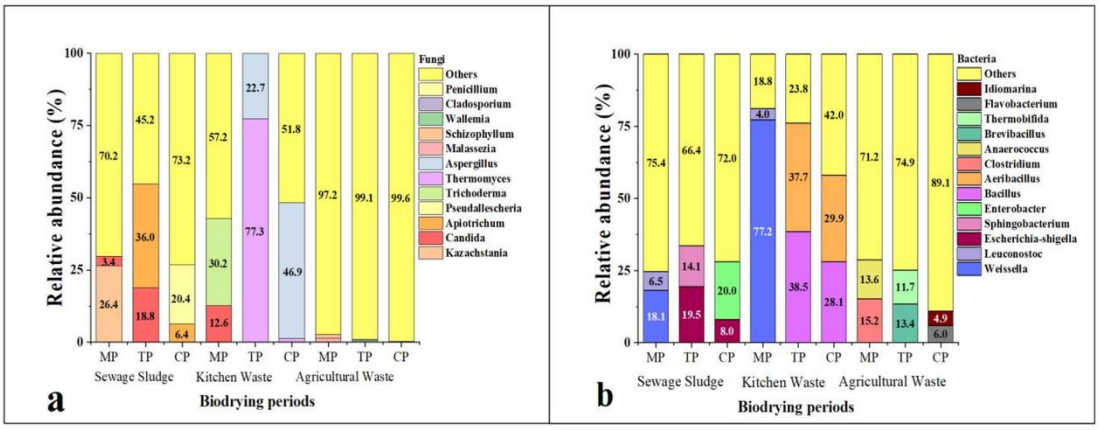

Figure 3. Relative abundance distribution of species at the genus level during biodrying a: fungi b: bacteria (MP: Mesophilic phase, TP: Thermophilic phase, DP: Cooling phase).

In the thermophilic phase of OSW biodrying with RA, the dominant bacteria such as Thermomyce can decompose lignocellulose and promote the degradation of difficult to

degrade substances. Bacillus presents in most composts as core microbial taxa and could largely consume organic matter. Compared with OSW biodrying using conventional amendments, there are significant differences in microbial community structure (Koyama et al., 2018; Xu et al., 2021).

3.5 Carbon emissions

3.5.1 Carbon emissions calculation results and analysis

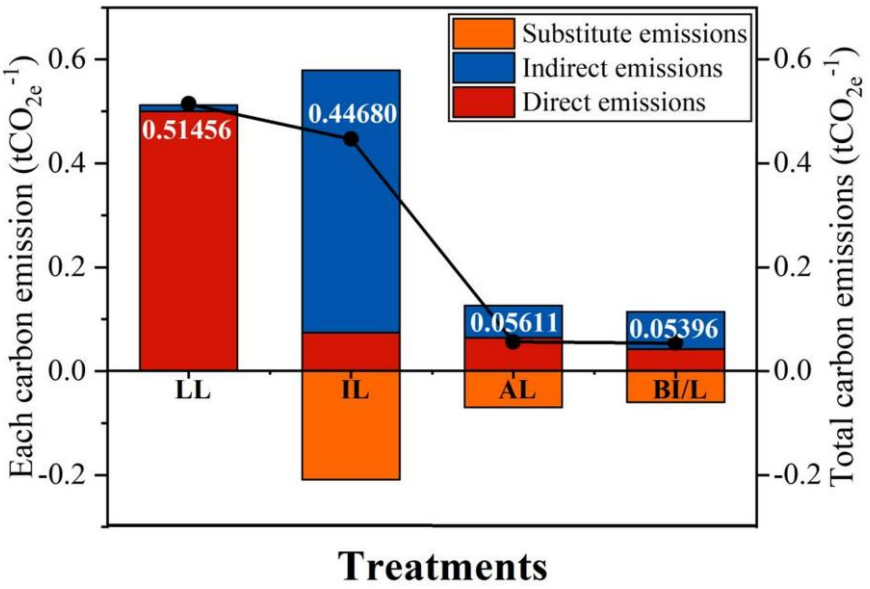

Figure 4. Calculation results of carbon emissions of each treatment.

The carbon emissions of four plant-scale sludge treatment methods were investigated in this study: lime curing-landfilling (LL), incineration-landfilling (IL), aerobic composting- land use (AL), and biodrying with RA - incineration/land use (BI/L). LL had the highest carbon emission, at 0.51456 tCO_{2e}^{-1}, of which direct emission was the main form, accounting for 97.2%. LL carbon emissions was followed

by 0.44680 tCO_{2e}^{-1}, and its indirect emissions account for 87.2% of the total emissions. It was worth noting that the substitute emissions of IL was the highest among the four treatments, which was 0.20860 tCO_{2e}^{-1}. AL and BI/L reduce direct and indirect emissions, so their total carbon emissions were significantly lower than LL and IL, which were 0.05611 tCO_{2e}^{-1} and 0.05396 tCO_{2e}^{-1}, respectively (Fig. 4).

3.5.2 Analysis of influencing factors of carbon emissions

Land resource was occupied and a large amount of CH_4 was produced after landfilling, and CO_2 was produced by fuel consumption and electricity consumption in landfill process. And the greenhouse effect of CH_4 was greater than that of CO_2 (Huang et al., 2022), Therefore, LL had high direct carbon emissions. IL resulted in adequate sewage sludge disposal and a high degree of sewage sludge reduction. However, the high MC in the material generated a large amount of water vapor, resulting in flue gas emission and increased energy consumption. Therefore, the indirect carbon emissions of IL was high. Due to the recovery of heat energy generated by incineration, the substitute carbon emissions was higher. AL and BI/L were biological treatments and were considered resource-saving treatments, with a low carbon emissions level, and enhanced soil carbon sink level. BI/L achieved the lowest carbon emissions level, and the use of RA reduced the reliance on the conventional amendments, which reduced transportation and operation costs of the biodrying process. Lower water content and sustainable development have realized the reduction, harmlessness, and effective resource utilization of sewage sludge.

3.6 Life cycle assessment

3.6.1 Research objectives and scoping

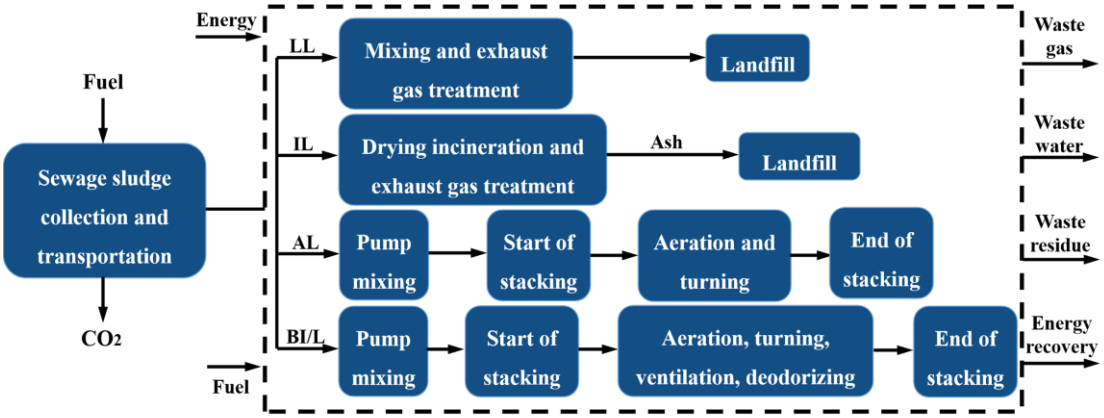

Figure 5. The system boundary of sewage sludge treatments (The treatment objects in BI/L include sewage sludge, kitchen waste and agricultural waste).

The system boundary of each treatment scheme was shown in the Fig. 5. In this study, raw materials and energy were tracked in each procedure, using 1t OSW as a functional unit. Based on functional units, all resources, emissions and potential environmental impact values were analyzed. The process included analyzing the four sewage sludge disposal routes of LL, IL, AL, BI/L, and the energy consumption and environmental impact of biodrying with RA for the sewage sludge, kitchen waste, and agricultural waste. LCA was used to evaluate and quantify the environmental impact. And using different data sources to evaluate their energy and mass balance. The evaluation starting point was sewage sludge collection, and included sewage sludge transportation and the product treatment process. LL included mixing and landfill gas treatment; IL included tail gas treatment and ash disposal; AL and BI/L did not include

product utilization processes. BI/L was implemented with the addition of RA in place of 50% of the conventional amendment.

3.6.2 Environmental impact assessment

Environmental impact equivalent values were calculated for the different sewage sludge treatment and disposal processes associated with LL, IL, AL, BI/L, as shown in Table 2. Eight environmental indicators were considered in assessing the impact: global warming potential (GWP), acidification potential (AP), eutrophication potential (EP), ozone depletion potential (ODP), freshwater aquatic eco-toxicity potential (FAETP), human toxicity potential (HTP), photochemical oxidant creation potential (POCP), and terrestrial eco-toxicity potential (TETP).

Table 2 Environmental impact equivalent values for each treatment scenario

Type of environmental impact	LL	IL	AL	BI/L
GWP [kgCO_2eq]	821	348	84.9	76.1
AP [kgSO_2eq]	2	1.37	0.377	0.338
EP [kgPeq]	2.92	0.177	0.0311	0.0283
ODP [kgCFC-11eq]	2.8E-9	5.87E-10	2.43E-11	2.17E-11
FAETP [kg1,4-DBeq]	21.6	1.14	1.34	1.2
HTP [kg1,4-DBeq]	22.3	21.2	28	25
POCP [kgNMVOC]	-1.33	1.129	0.0246	0.0209
TETP [kg1,4-DBeq]	0.634	0.429	0.627	0.559

The environmental impact of LL was significant with respect to GWP, AP, EP, ODP, and FAETP, at 821 kgCO_2eq, 2 kgSO_2eq, 2.92 kgPeq, 2.8 E-9kgCFC-11eq, and 21.6 kg1,4-DBeq, respectively. IL was better than LL concerning these variables but did not

perform as well as LL with respect to POCP, and incineration produced ash. Because heavy metals and other hazardous substances were generally highly concentrated in ash, the material needed to be carefully controlled to avoid other serious environmental impacts.

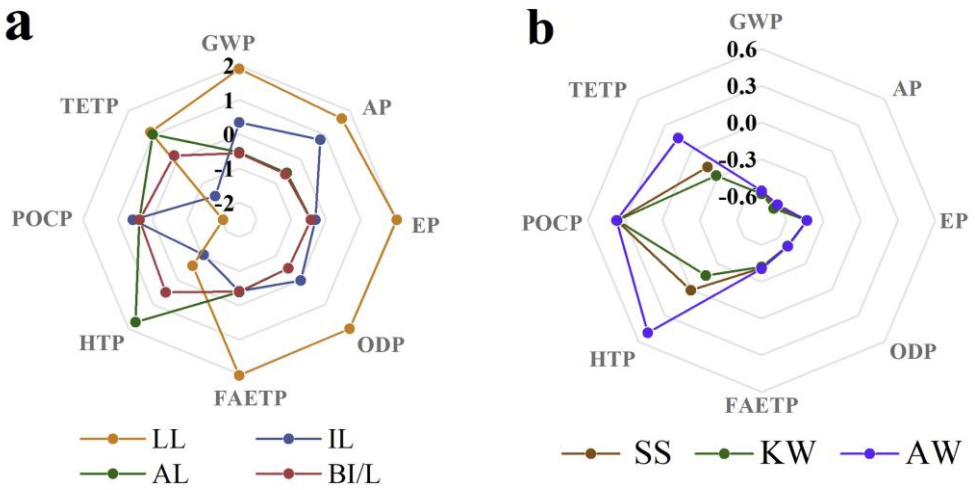

Figure 6. Potential environmental impact of four disposal methods of sewage sludge (a) and biodrying with RA of three organic solid waste (b) (SS: Sewage Sludge, KW: Kicthen Waste, AW: Agricultural Waste)

In comparison, the total environmental impact equivalent values of AL and BI/L were significantly better compared to LL and IL. BI/L was less hazardous than AL with respect to GWP, AP, and ODP in terms of environmental impact, which were 76.1 $kgCO_2eq$, 0.338 kgSO2eq, 2.17E-11 kgCFC-11eq, respectively (Fig. 6a). This might be because using RA reduced the reliance of biodrying on an organic amendment, thereby reducing the energy consumption during transport and disposal of material.

The analysis indicated that biodrying using RA might be an optimal way to treat

sewage sludge. This paper further analyzed the differences of biodrying with RA on different types of OSW. Fig. 6b shows that biodrying had a significant environmental impact in the treatment of agricultural waste, especially in terms of HTP and TETP, reaching 25 kg1,4-DBeq and 0.559 kg1,4-DBeq, respectively. It might be due to agricultural waste such as biogas residue contained heavy metals, and there was a risk of metal accumulation, which had a certain impact on HTP and TETP (Chang et al., 2022). This was followed by the treatment of sewage sludge. In comparison, the environmental impact of kitchen waste was the lowest.

4. Conclusions

Biodrying of OSW adding RA reduced moisture content levels effectively, significantly removed pathogenic bacteria and improved the net energy of the substrate system. In terms of environmental impact, biodrying with RA led to the lowest carbon emissions at the plant scale. And the total environmental impact potential value of aerobic composting and biodrying were significantly better than landfilling and incineration. Biodrying was found to be less hazardous than aerobic composting with respect to GWP, AP, and ODP. In the horizontal comparison of OSW biodrying, the environmental performance of biodrying of kitchen waste was best. The result provided an important reference for selecting OSW management strategy with the lowest environmental impact under carbon peaking and carbon neutrality policy.

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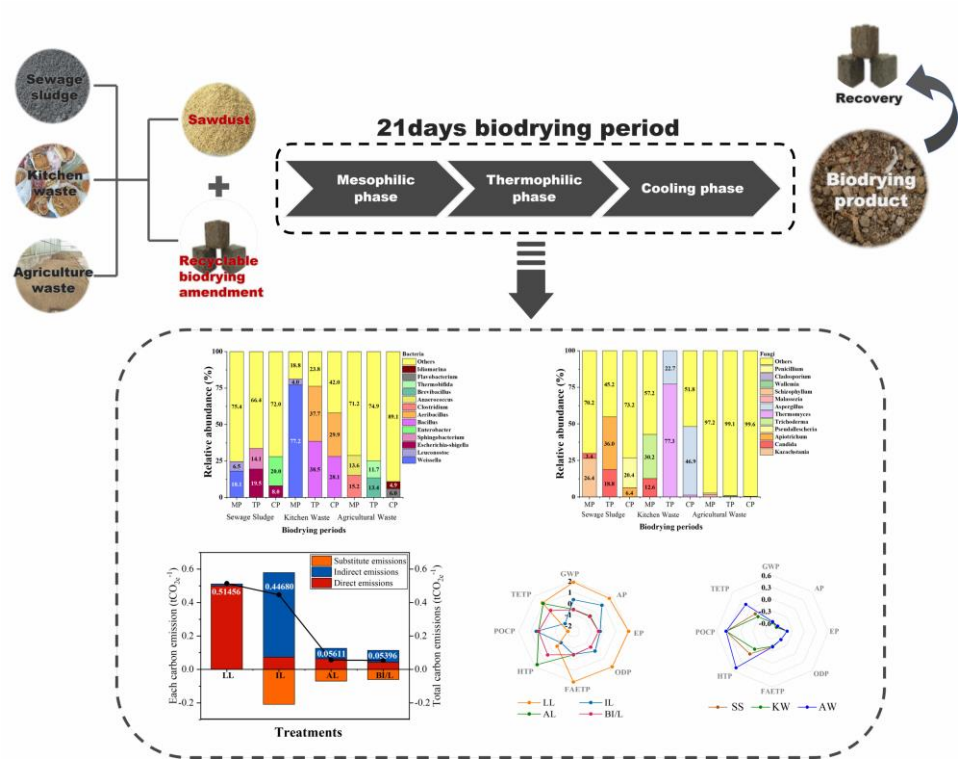
Graphical Abstract



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Ms. Ref. No.: WM-22-2254

Title: A comprehensive evaluation of the biodrying of organic solid waste using a recyclable amendment, insights of energy saving and emissior

Waste Management

Dear Badrus,

Thank you for agreeing to review this manuscript for Waste Management. We greatly appreciate your assistance.

Please use the review submittal page for your comments. We ask that you rate a few aspects of the manuscript, offer specific suggestions for imp

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Thank you for the review of WM-22-2254

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Dear Badrus,

This is to confirm reception of your review. We greatly appreciate your willingness to serve as a referee for Waste Management.

Thank you again for sharing your time and expertise.

As a token of appreciation, we would like to provide you with a review recognition certificate on Elsevier Reviewer Hub (reviewerhub.elsevier.com) Hub, you can also keep track of all your reviewing activities for this and other Elsevier journals on Editorial Manager.

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Attachments for Manuscript Number WM-22-2254
"A comprehensive evaluation of the biodrying of organic solid waste using a recyclable amendment, insights of energy saving and emission reduction"

Action	Uploaded By	Description	File Name	File Size
Original Submission				
Download	Reviewer 2	WM-22-2254	review WM-22-2254.docx	14.3 KB

Main Menu

Abstract

1. In the abstract need to improve with novelty statement clearly

Introduction:

1. What are the references of the statement at line 50-51?
2. The word ” three species were selected to plot the relative abundance of species (line 148-149)” . Its any reference for this method?
3. The gap and novelty of this research was needed to be clearly in this manuscript

Result and Discussion

1. Improve the newest references for the discuss (some references was more than 5 years ago)
2. Should be discussed (Line 172) why the ultimate compressive strength of the RBA before and after biodrying was decrease? (line 175-176) and the recovery rates of the RBA at the end of sewage sludge biodrying was 98%, added with the references about this phenomena.
3. How the relation between biodrying scale and peak pile temperatures (line 184)?. Especially Ts need to discussed were it very high thermophilic temperature condition
4. What mean the conventional amendments?, please picture it with clearly compare with research in this manuscript (line 191, table S1)
5. Need to discuss the variation of oraganic degradation rate was happen (line 201)
6. Is any graph for pH variation along process in Ts, Tk,Ta especially in Ta with trend to base condition? I can’t find it
7. In the analysis should not to use word “conclusion” (line 310)

Close

WM-22-2254
"A comprehensive evaluation of the biodrying of organic solid waste using a recyclable amendment, insights of energy saving and emission reduction"
Original Submission

Badrus Zaman, Ph.D (Reviewer 2)

Reviewer Recommendation Term:	Significant Revision
Transfer Authorization	**Response**
If this submission is transferred to another journal, do we have consent to share your identity with the receiving journal Editor(s)?	Yes
If this submission is transferred to another journal, do we have your consent to share your full review with the receiving journal Editor(s)?	Yes

Comments to Editor:

Please complete the review by rating the manuscript and providing comments in the areas above and below.
****

RATING: Rate the following aspects of the paper from 1 to 5, with 1 being 'poor' and 5 being 'excellent'.

Level of interest to readers of Waste Management: 5

Scientific significance: 4

Quality and completeness of abstract: 3

Adequacy of experimental approach and techniques:3

Adequate support of conclusion by the data: 3

Quality and importance of tables and figures: 4

Thoroughness of literature citations: 3

Clarity of presentation: 4

****
Please note that your recommendation and reviewer report are expected to cover the Highlights and Graphical Abstract if submitted with the manuscript.

COMMENTS: need to added and improve newer references especially part of result and discussion

Comments to Author:

Close

Reviewer Notification of Editor Decision

1 message

Waste Management <em@editorialmanager.com> Mon, Dec 5, 2022 at 1:00 PM
Reply-To: Waste Management <support@elsevier.com>
To: Badrus Zaman <badruszaman2@gmail.com>

Ref: WM-22-2254
Title: A comprehensive evaluation of the biodrying of organic solid waste using a recyclable amendment, insights of energy saving and emission reduction
Article Type: Full Length Article

Dear Badrus,

Thank you once again for reviewing the above-referenced paper. With your help the following final decision has now been reached:

Reject

We appreciate your time and effort in reviewing this paper and greatly value your assistance as a reviewer for Waste Management.

Yours sincerely,

Pinjing HE, Ph.D.
Editor-in-Chief
Waste Management

Comments from the editors and reviewers to author:
Associate Editor: Reject decision is due to weakness of experimental design (contrils are missing) and to lack of novelty

Reviewer #1: Evaluation of the biodrying on energy saving and emission reduction could be an important topic. This MS reported some data from different aspects. However, the authors failed to organize them with more focus. I suggested the authors to re-organize their MS. Some other suggestions are listed as follows:

Remarks:
1. The anaerobic digestion could be a good step for food waste treatment. How about its energy saving and carbon emission reduction compared with others?
2. Did the authors compare the biodrying using RA with conventional biodrying?
3. Could the authors give some reasons why the carbon emission of incineration is higher than biodrying?
4. Lines 83-85, bulking agents for composting could be same as one in biodrying.
5. Is the silicate-based recyclable amendment (RA) first reported? Please be honest.
6. The experimental setup is not clearly. How many reactors? What's the substrate, single component or mixture of some or all?
7. Lines 150-164, what's the emission of CH4 and NOx?

8. Lines 112-115, it is not analytical method!
9. Section of 3.1 is too weak.
10. Line 212, it is not related to the main topic of this study.
11. Lines 217-218, why the biodrying with RA showed a better GI value?

Reviewer #4: This paper presents some results of organic waste biodrying using recyclable amendements and compares this process with conventional processes for organic waste treatment.
Some significant shortcomings prevent its publication in waste management:

- Control experiments without renewable amendments are missing
- the system boundary definition (Figure 5) does not allow a proper comparison of the four processes. Indeed, the first two ones lead to complete disposal of waste and the 3rd and 4th ones to uncompleted disposal. Land application and incineration should be included in these processes
- Why anaerobic digestion which is a classical process for organic waste treatment has not been included in the comparison?
-

Other remarks:
Tables S1 and S2 contain information that is necessary to understand the paper and should be included in the manuscript
Double check the unit of carbon emissions.
Even if amendment used is recyclable, the environmental impact of its production should be presented.
Extensive use of abbreviations makes the paper very difficult to read. Please replace most of them by full names.

Full description of abbreviations should be given in figure captions (ex Figure 2 cannot be understood by itself)