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A Comparison of the Carbon Footprints of Different Digested Sludge Post-Treatment Routes: A Case Study in China

Hanlin Ci [1,2,*], Ning Fang [1,2], Hang Yang [1,2], Yali Guo [1,2], Xiaojie Mei [1,2] and Xiaolei Zhao [1]

[1] Shanghai Investigation, Design & Research Institute Co., Ltd., Shanghai 200434, China; gyl@sidri.com (Y.G.); zhao_xiaolei@ctg.com.cn (X.Z.)
[2] National Engineering Research Center of Eco-Environment Protection for Yangtze River Economic Belt, China Three Gorges Corporation, Beijing 100000, China
* Correspondence: ci_hanlin@ctg.com.cn

Abstract: As the "carbon peaking and carbon neutrality" strategy advances, carbon emissions have gradually become a significant indicator in selecting and evaluating sewage and sludge treatment solutions. This study compared the carbon footprints of different digested sludge post-treatment routes, taking the Lu'an project in China as an example. Considering anaerobic digestion and digested sludge post-treatment options, the carbon footprints are as follows: 347.7 kg CO_2 (land application) < 459.7 kg CO_2 (composting-involved land application) < 858.4 kg CO_2 (brickmaking). In general, land application was superior to brickmaking from the perspective of carbon footprints. The power consumption incurred by aerating and turning and the direct N_2O and CH_4 emissions during composting increase the composting-involved land application carbon footprint. However, digested sludge that is not subject to high-temperature sterilization and compost is phytotoxic and can be fetid, which is a limitation of its applicability. And the composted sludge has a lower N ratio and water content, so the same N input means more sludge usage, which is conducive to solving the disposal problem of large amounts of sludge. Thus, if possible, composting-involved land application should be a preference, and improvements to the technique are required to minimize energy consumption and direct N_2O and CH_4 emissions.

Keywords: sludge; carbon footprint; anaerobic digestion; post-treatment

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1. Introduction

The "carbon peaking and carbon neutrality" strategy has rendered carbon emissions one of the foundational indicators for sewage sludge treatment solutions and their evaluation. The greenhouse gas (GHG) emissions from sewage treatment, which is an energy-intensive industry, account for 1–2% of the total amount [1], and sludge treatment and disposal from wastewater treatment plants (WWTPs) could account for this emission level by 40% [2], which is considered an essential priority for carbon emission reductions in the water system. The amount of sludge in China has exceeded 65 million tons per year (calculated at 80% moisture content) and is expected to exceed 90 million tons by 2025 [3]. The increase in sludge amount is bound to lead to an increase in GHG emissions. Therefore, the low-carbon management of sludge is of great significance.

Common sludge treatment processes include incineration, anaerobic digestion, composting, pyrolysis, etc. [4]. Anaerobic digestion (AD) was identified as a favorable low-carbon sludge treatment alternative for its energy recycling potential [5–7]. The proportion of AD plants in sludge treatment projects in China is 15.5% [8]. Existing studies have evaluated the carbon footprint of AD plants with negative carbon emissions (Table 1) [6,7,9–11]. However, the post-treatment method of digested sludge in the above studies involves land application. Actually, the post-treatment options of digested sludge include producing building material, land application, or incineration [12], which has a great impact on the carbon footprint of AD plants [13].

Table 1. GHG emission of AD process.

Process Description	GHG Emissions ($kgCO_2$/t Dry Solid (DS))	References
Thickening→AD→dewatering→land application	−5.36	[9]
Thickening→AD→dewatering→land application	−26.33	[10]
Thickening→AD→dewatering→land application	−178~106	[11]
Thermal hydrolysis→AD→dewatering→land application	−627~−202	[11]

There are few studies on the carbon footprint of digested sludge reuse in producing building materials, such as bricks [14]. As for application, direct use of digested sludge is not permitted in some countries [6]. It needs to be composted and sterilized to improve its agronomic performance [15]. In terms of GHG emissions, despite existing research showing that composted solid digestate has lower carbon emissions than non-composted solid digestate, land application [16,17], energy consumption, and direct GHG emissions in the composting process cannot be ignored [18]. IPCC emission factors are used in most studies to calculate the GHG emissions during composting [11,19,20]. But there is a relative uncertainty in the direct GHG emissions, which has been proven to be related to the material characteristics and composting parameters [21–23]. The studies using emission factors collected directly from field measurements are limited. Therefore, it is necessary to evaluate the carbon footprint of AD and different digested sludge post-treatment routes so as to provide low-carbon suggestions for AD plants. In addition, in view of the uncertainty of direct GHG emissions in the composting process, field monitoring is used to provide more reliable emission data.

This study compared different digested sludge post-treatment routes' carbon footprints, taking the actual engineering project in Lu'an City as an example. Our aim is to provide a decision-making reference for the city's sludge treatment and disposal from the "carbon emission management" perspective.

2. Materials and Methods

2.1. Sludge Testing

Collected from the Lu'an City Sludge Disposal Plant in 2023, the sludge samples were grouped into "untreated sludge", "digested sludge", and "composted sludge". The sealed samples were sent back to the laboratory and stored at 4 °C prior to testing. According to the standard method [24,25], the samples were tested for organic content, moisture content, total nitrogen, total phosphorus, and total potassium. To evaluate the phytotoxicity of digested and composted sludge, the seed germination index (GI) was tested by using Ronga's method [26].

2.2. Carbon Footprint Boundary

The accounting scope of sludge carbon footprint includes indirect emission and direct emission of CH_4 and N_2O. Because sludge is a biosolid, its CO_2 emissions are considered neutral carbon [27]. The carbon footprint boundary shown in Figure 1 indicates the scope from initial sludge transportation to digested sludge post-treatment, centering on the Lu'an Sludge Disposal Project (Phase I). As shown, the residue sludge generated by WWTPs is pumped to the AD plant, where it is gravity-concentrated to 92% moisture content. It is then blended and heated in conditioning tank. Once conditioned and preheated, the sludge is pumped into anaerobic digestion reactor, and the reaction temperature is 35 ± 1 °C. The main source of carbon emissions from the above steps is the electricity consumed by transportation, heating, and mixing. Since the reactor is fully enclosed and works well, the escape of CH_4 is not considered here [28]. Digested sludge is mechanically dewatered by a centrifuge after adding polyacrylamide (PAM). The dewatered sludge is mechanically turned and dried in the solar-heat drying shed. The carbon emission source in the process of dewatering and drying is the indirect emission produced by electricity and

PAM consumption. Biogas is collected and purified, then burned in a boiler, which is used to provide heat for the conditioning tank and the AD reactor. The combustion efficiency is set at 96%, and 4% of the CH_4 is actually released into the atmosphere [29]. The digested effluent from concentrate and dewatering system is drained to nearby WWTP, and GHG emissions during effluent treatment are also included in the boundary.

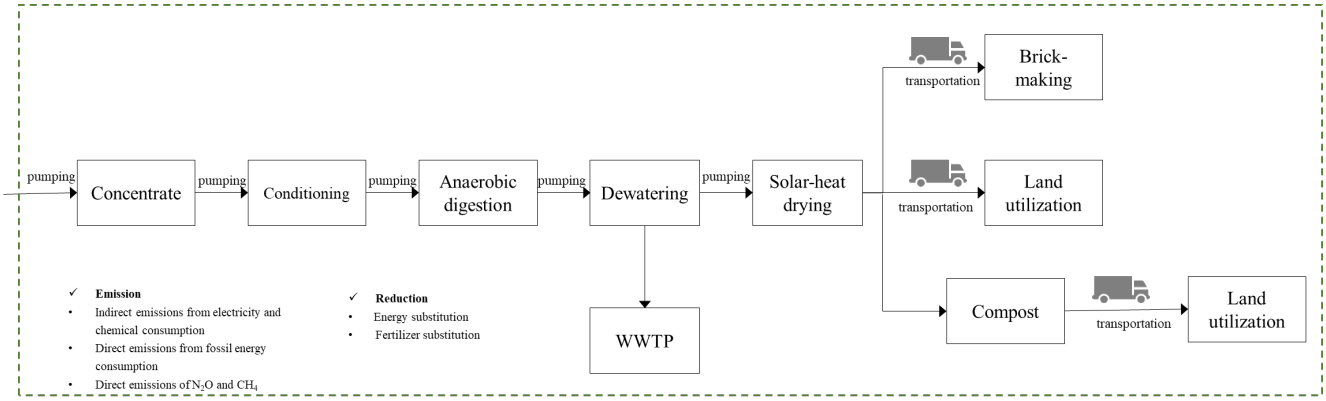

Figure 1. Carbon footprint boundary.

There are three routes of post-treatment of digested sludge: brickmaking, land application, and composting or land application. The composting equipment is in the AD plant. So, it is composted in the plant and then transported out.

Digested sludge is transported by road for brickmaking. The bricks are made by sintering a mixture of sludge, coal gangue, and clay at 1100 °C, similar to incineration. In this process, the grinding, mixing, pressing, and conveying of materials will consume electricity. Meanwhile, the waste gas produced by high-temperature sintering needs to be treated with lime and caustic soda. These are the main indirect emissions.

Digested sludge is transported by road to nearby greenbelt for artificial land application. In order to avoid the risk of environmental pollution from excessive application, the nitrogen input should be controlled within 100–300 kg N·ha^{-1} according to the agricultural fertilization process [30]. The specific application amount is calculated according to the nitrogen content in the sludge. On account of the high nitrogen content in sludge, soil microbial activities and plant activities can contribute to N_2O production through the utilization and transformation of nitrogen. Soils were generally sinks of CH_4 [31], and studies have indicated that the application of digestate did not impact the soil CH_4 emission [32]. Therefore, direct CH_4 emissions were not taken into consideration in this study.

As for composting, several 220 L composting buckets are used in Lu'an AD plants. The compression fans are used to blow air into the buckets, and the electric stirring paddles are installed to stir regularly to ensure that the material is mixed and exposed to air. Stirring and ventilation consume electricity and produce indirect emissions. Moreover, the composting process also produces direct emissions of N_2O and CH_4. Composted sludge is transported by road to nearby greenbelt for artificial land application similar to digested sludge with N_2O emissions.

2.3. Accounting Method of Indirect Emissions

The IPCC emission factor approach was used for indirect emissions. The equation is as follows:

$$C = EF \times AD \quad (1)$$

where *EF* is the emission factor (Table 2), and *AD* is the activity data (Table 3). The whole-year operating data of the Lu'an Sludge AD Plant, the operating data of the brickmaking factory (160,000 bricks; d^{-1}), and the operating data of the composting equipment were taken as the activity data shown in Table 3. N_2O and CH_4 are converted to CO_2 equivalent by global warming potential (GWP). The GWP of N_2O is 298, and that of CH_4 is 21.

Table 2. Carbon emission factors.

	Emission Factor
PAM	2.62 kg $CO_2 \cdot kg^{-1}$ [33]
Electricity	0.5703 kg $CO_2 \cdot kg^{-1}$
Diesel Fuel	1.45 kg$\cdot kg^{-1}$ Standard coal
Coal Transportation	0.665 kg CO_2/km [34]
Coal Gangue	0.807 kgCO_2/kg [35]
Limestone	0.43 kgCO_2/kg [36]
Caustic Soda	1.912 kgCO_2/kg [37]
Wastewater Treatment	3.3 kgCO_2/t [38]

Table 3. Activity data from operational data in the actual project.

		Activity Data
Anaerobic Digestion	Power Consumption—Thickening	10 kWh$\cdot tDS^{-1}$
	Power Consumption—Pumping	70.71 kWh$\cdot tDS^{-1}$
	Power Consumption—Conditioning	37.71 kWh$\cdot tDS^{-1}$
	Power Consumption—Anaerobic Digestion	106.29 kWh$\cdot tDS^{-1}$
	Power Consumption—Centrifugal Dewatering	77.14 kWh$\cdot tDS^{-1}$
	PAM Consumption—Centrifugal Dewatering	2 kg$\cdot tDS^{-1}$
	Power Consumption—Solar Drying	49.37$\cdot tDS^{-1}$
	Biogas Slurry Treatment Capacity	226.8 t$\cdot d^{-1}$
	Power Consumption—Biogas Boilers	21.85 kWh$\cdot tDS^{-1}$
	Direction Emissions—Biogas Production	1200 $m^3 \cdot d^{-1}$
	Direction Emissions—Methane Ratio	60%
	Amount of Digested Effluent Produced	8.10 t water$\cdot tDS^{-1}$
Brickmaking	Power Consumption—Material Grinding and Transport	233.31 kWh$\cdot d^{-1}$
	Power Consumption—Material Mixing and Transport	837 kWh$\cdot d^{-1}$
	Power Consumption—Compression Molding	1756.8 kWh$\cdot d^{-1}$
	Power Consumption—Baking Bricks	2320 kWh$\cdot d^{-1}$
	Limestone Consumption—Waste Gas Treatment	119.53 kg$\cdot d^{-1}$
	Caustic Soda Consumption—Waste Gas Treatment	24.15 kg$\cdot d^{-1}$
Composting	Power Consumption—Compost	112.8 kWh$\cdot tDS^{-1}$
	Transport Distance	50 km

2.4. Monitoring Method for Direct N_2O and CH_4 Emissions

2.4.1. N_2O and CH_4 Monitoring While Composting Digested Sludge

This research was based on a practical project to monitor N_2O and CH_4 emissions during the process of digested sludge composting, which was subject to forced ventilation, with straw being the assistant material at a proportion of 10%. The N_2O and CH_4 emission fluxes were monitored using the portable soil–atmosphere interface carbon–water and nitrogen–water flux measurement system (the LI-7810 $CH_4/CO_2/H_2O$ Trace Gas Analyzer and the LI-7820 N_2O/H_2O Trace Gas Analyzer, LI-COR Biosciences, Lincoln, NE, USA) supported by the smart gas flux survey chamber (the 8200-01S Smart Chamber, LI-COR Biosciences, Lincoln, NE, USA). The monitoring time was January, April, and June 2023.

Specific monitoring steps are as follows: (1) Fix the PVC soil measuring ring in the piles of compost, and the top of ring is about 5 cm above the piles. (2) Place the chamber on the ring and initially open the equipped vent to harmonize the internal pressure of the chamber with the external atmospheric pressure for about 45 s, then close the vents and prepare for monitoring. (3) The period from the closure of the chamber vents until its reopening marks the monitoring length. To minimize the effect of the presence of the chamber on gas gradients within the piles of compost, the monitoring length was set to 120 s [39]. Gas

increase rates, dX/dt (nmol·m^{-2}·s^{-1}), are automatically calculated using SoilFluxPro software 5.0. Gas fluxes are calculated according to the size of the soil measuring ring S (m^2).

$$F_X = \frac{1}{S} \cdot \frac{dX}{dt} \quad (2)$$

The monitoring period of the compost is 11 days, based on the temperature of the compost. Gas fluxes were measured between 9:00 and 11:00 a.m. every day during the period.

The long-term sampling analysis helped achieve long time-series gas fluxes under different conditions. Specifically, the accumulation of time spans allowed for the calculation of the emission of gas X within a long time series.

$$q_X = M_X \cdot \int_0^T F_X dt \quad (3)$$

where q_X is the emission of gas X within time T from the pile (g), M_X is the molar mass of gas X (g·mol^{-1}), and F_X is the soil emission flux of gas X (nmol·s^{-1}). In this case, the 11-day composting cycle was considered the monitoring span. With the q_X value obtained and the mass of the sludge dry basis considered together, the GHG emission per unit of dry basis (i.e., the emission factor; kg·tDS^{-1}) can be calculated.

2.4.2. N_2O Emissions from the Application of Digested and Composted Sludge on Land

Digested sludge and composted sludge are high in nitrogen and may cause an increase in soil N_2O emissions when applied on land. It has been proved that N_2O emissions in soil are related to N input [40,41]. Using the monitoring method described in Section 2.4.1, our team [27] conducted long-term monitoring of N_2O emission fluxes in the case of applying digested sludge and composted sludge as soil amendments in landscaping. By monitoring the N_2O emission fluxes of control groups with different N inputs, a mathematical relationship between N input and N_2O emissions was established. The logarithmic model of N input and N_2O emissions presented a higher coefficient of determination (R^2) than linear and quadratic models due to the special characteristics of sludge [27,40]. Specifically, the models for digested sludge and composted sludge are, respectively, $y = 3.84\ln(x + 371.9) - 22.73$ and $y = 4.49\ln(x + 875.9) - 30.51$ [27]. In the previous formula, x refers to the amount of nitrogen (kg N) input, and y refers to the N_2O emission (kg N_2O-N). Therefore, in the context of the application of sludge on land, the carbon emissions can be calculated based on the actual nitrogen inputs.

2.5. Accounting of Carbon Emission Reduction

Sludge as organic fertilizer could be an ideal alternative to some chemical fertilizers for its richness in nitrogen (N), phosphorus (P), and potassium (K), which is beneficial to improving soil fertility and promoting plant growth. But excessive application of organic fertilizers will also increase the risk of nitrogen loss. In this research, it is assumed that the chemical fertilizer replacement rate of sludge organic fertilizer (N, P, and K) is 70% [42,43]. The carbon emission reduction upon replacing chemical fertilizers with sludge can be calculated according to the carbon emissions from fertilizer production, which are drawn from the existing literature (Table 4).

$$E_N = A_N \cdot EF_N \quad (4)$$

$$E_P = A_P \cdot \frac{142}{62} \cdot EF_P \quad (5)$$

$$E_K = A_K \cdot \frac{94}{78} \cdot EF_K \quad (6)$$

Table 4. Emission factors of chemical fertilizers.

	Emission Factor
Synthetic Nitrogen Fertilizer	4.42 kg CO_2·kg N^{-1} [44]
Phosphorus Fertilizer	0.636 kg CO_2·kg $P_2O_5{}^{-1}$ [45]
Potassium Fertilizer	0.180 kg CO_2·kg K_2O^{-1} [45]

The sludge-based carbon substitution in land application refers to Equations (4)–(6), where E_N, E_P, and E_K are the carbon emission reductions after replacing N, P, and K fertilizers with sludge; A_N, A_P, and A_K are the contents of N, P, and K in sludge; and EF_N, EF_P, and EF_K are the general emission factors of N, P, and K fertilizers.

3. Results and Discussion

3.1. Sludge Characteristics

Sludge characteristics are shown in Table 5. Organic matter, total nitrogen, total phosphorus, and total potassium are all percentages based on a dry basis. The raw sludge was pumped to the AD plant. After concentration, the sludge had a moisture content of 92.0%, an organic content of 42%, and a C/N ratio of 10. After digestion and dewatering, the digested sludge had a moisture content of approximately 67.1%, with a decreased organic content and total organic carbon content since a part of the organic substances was degraded and converted to methane, and the C/N ratio fell to 8.7. In the composting procedure, straw (10%) was input as the assistant material to increase the C/N ratio, and the moisture content further decreased to 45.6% because of biological drying.

Table 5. Sludge characteristics.

	Organic Matter	Moisture	Total Organic Carbon	Total Nitrogen	Total Phosphorus	Total Potassium
Raw Sludge	42.0%	92.0%	32.5%	3.2%	1.9%	0.5%
Digested Sludge	30.6%	67.1%	29.8%	3.4%	2.8%	0.5%
Composted Sludge	38.3%	45.6%	22.2%	2.1%	2.1%	0.06%

3.2. Carbon Emission Analysis of Anaerobic Digestion

The scale of the AD plant in Lu'an City is 140 t·d^{-1} (80% moisture content). Within the study timeframe, it operated at almost full capacity. The residual sludge from the WWTPs was pumped to the AD plant through pipelines, where it underwent concentration and conditioning to enter the AD reactors. The hydraulic retention time is 28 days. After centrifugal dewatering, the digested sludge went into the solar-heat drying shed; meanwhile, the digested effluent flowed back to the WWTP. At the same time, the biogas burned in the biogas boilers to provide heat to maintain the temperature inside the AD reactors. The carbon emissions are provided in Figure 2.

According to the statistics, from sludge pumping to AD and solar-heat drying, the total carbon emission was 9351.48 kg CO_2·d^{-1}, equivalent to 333.98 kg CO_2·tDS^{-1} on a dry basis. Specifically, the solar-heat drying procedure had the highest carbon emissions and involved sludge turning and air blowing, which can consume massive amounts of electricity. Compared with other studies, in which carbon emissions of AD are negative [9,11], there is no carbon reduction in this study. There is no cogeneration because of the direct combustion of biogas.

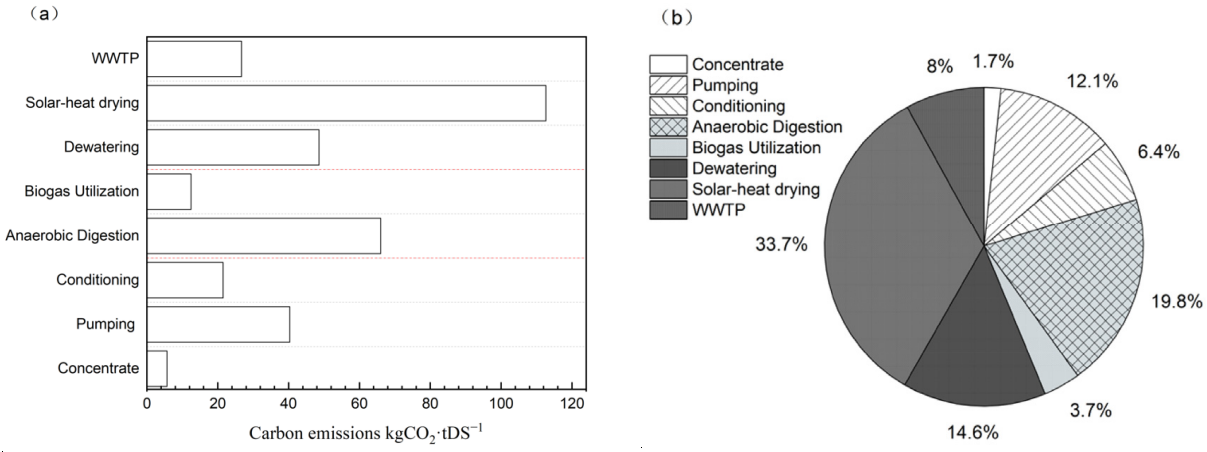

Figure 2. (**a**) Carbon emissions from different procedures in anaerobic digestion. (**b**) Carbon emission proportions from different procedures in anaerobic digestion.

3.3. Carbon Emission Analysis of Digested Sludge Disposal

3.3.1. Brickmaking

Digested sludge that has been through solar drying is sent by road to brick factories for brickmaking and is mixed with coal gangue and clay. In scenarios where sludge is not considered, coal gangue and clay, the two raw materials in sintered bricks, are mixed at a ratio of 1:4. However, due to the requirements in terms of heat value and ignition loss, the additive amount of sludge for each brick should not exceed 4%. Also, digested sludge has a limited contribution to the heat value because of its low organic content, but it necessitates more coal gangue due to the extra moisture from sludge. In this case, the heat value of coal gangue is 8380 kJ·kg^{-1}, and each kilogram of water needs 2260 kJ to evaporate, which means that an additional 0.6 t of coal gangue is required per 1 tDS of sludge added. The additional consumption of coal gangue should be regarded as the carbon emissions resulting from sludge-involved brickmaking. Other emissions can be calculated based on the factory's indirect carbon emissions per unit of raw material consumed. Carbon emissions from different procedures in digested sludge-involved brickmaking are shown in Figure 3. Thus, the total carbon emissions from digested sludge-involved brickmaking reach 600 kgCO_2·tDS^{-1}, of which 95% is incurred by the necessary increase in coal gangue.

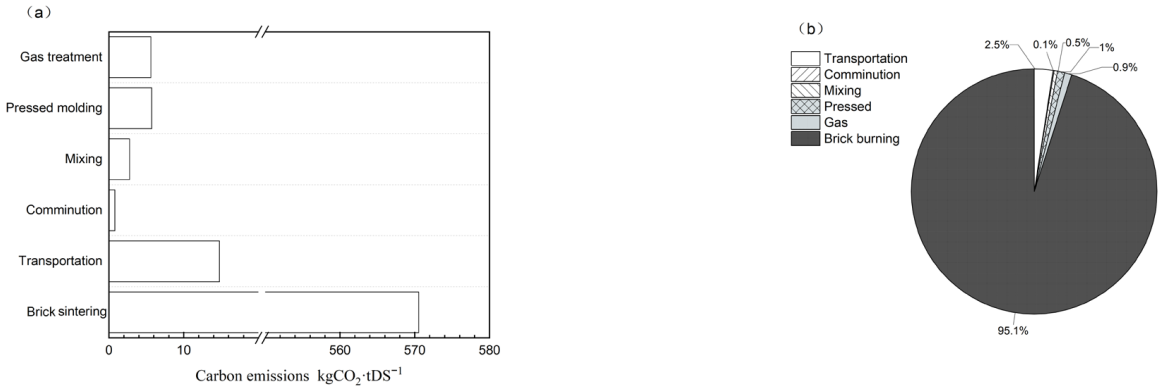

Figure 3. (**a**) Carbon emissions from different procedures in digested sludge-involved brickmaking. (**b**) Proportions of carbon emissions from different procedures in digested sludge-involved brickmaking.

3.3.2. Land Application

Digested sludge can be used as a soil amendment in landscaping after solar drying and natural aging. The main sources of carbon emissions from digested sludge-involved

land application are fuel consumption and direct N_2O emissions from soil; the latter is proven to be correlated with water management, crop type, soil property, and nitrogen input, among others [46]. The application amount of digested sludge is 2.94 t~8.82 tDS·ha^{-1} according to the nitrogen input within 100–300 kgN·ha^{-1}, as described in Section 2.2. The range of the nitrogen emission factor is 0.6–1%, which was obtained from the references, as described in Section 2.4.2 [27], whereas the 2019 IPCC Guidelines for National Greenhouse Gas Inventories—Volume 4: Agriculture, Forestry and Other Land Use suggests that the default value of the direct nitrogen emission factor should be 1%, slightly greater than the measured values.

Based on the total nitrogen content in digested sludge, the direct N_2O emission from land application is 0.32~0.53 kg N_2O·tDS^{-1}, equivalent to 95.53~159.22 kg CO_2·tDS^{-1} in CO_2 equivalence. With a nitrogen application level of 200 kgN·ha^{-1} considered, the average carbon emission resulting from direct N_2O emissions is 0.40 kg N_2O·tDS^{-1}, equivalent to 119.2 kg CO_2·tDS^{-1} in CO_2 equivalence.

In addition, digested sludge is an ideal alternative to some chemical fertilizers since it is high in N, P, and K, for which the carbon emission reduction should be 118.29 kg CO_2·tDS^{-1}. The carbon emissions from land application are shown in Figure 4.

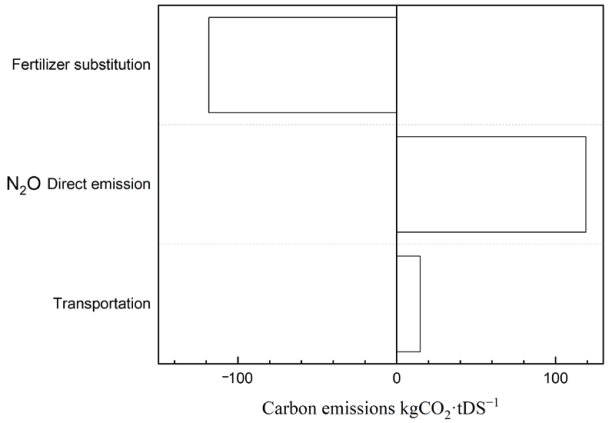

Figure 4. Carbon emissions from digested sludge-involved land application.

3.3.3. Composting

Straw can be mixed with digested sludge as an assistant material for composting, where the indirect carbon emission source is the power consumption of spreaders and centrifugal blowers. The direct emissions from composting and its products can be obtained by virtue of GHG fluxes. In the 11-day composting cycle, we monitored the cumulative N_2O and CH_4 emissions under different ventilation volumes and calculated the emission factors. The monitoring results are summarized in Figure 5.

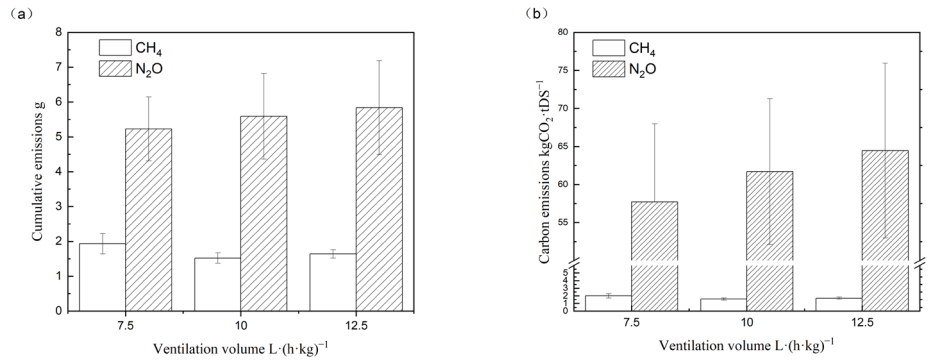

Figure 5. (**a**) Cumulative N_2O and CH_4 emissions under different ventilation volumes. (**b**) CO_2 equivalences of direct N_2O and CH_4 emissions under different ventilation volumes. The data shown in the graph are means and standard deviations.

In the composting cycle, the cumulative emissions of N_2O and CH_4 are 1.52~1.93 g and 5.23~5.84 g, respectively. Taking the treatment capacity and the global warming potential into consideration, the emission factors (dry basis) are 1.58~2.01 kg $CO_2 \cdot tDS^{-1}$ and 57.72–64.46 kg $CO_2 \cdot tDS^{-1}$, respectively. Increasing ventilation can reduce CH_4 emissions to a certain extent but has little effect on N_2O emissions. The digested sludge itself contains CH_4 [47]. Composting is strictly an aerobic process; however, it is unavoidable that anaerobic conditions prevail in a few zones of the compost piles. These zones lead to the formation of CH_4 due to the insufficient diffusion of oxygen [48]. Therefore, improved ventilation can reduce methane emissions. As for N_2O, nitrification and denitrification take place simultaneously and produce N_2O, which is a complicated process [49]. It may also be related to temperature microbial activity rather than just ventilation.

The 2019 IPCC Guidelines for National Greenhouse Gas Inventories—Volume 5: Waste suggests that the N_2O and CH_4 emission factors of composting are 59.6~476.8 kg $CO_2 \cdot tDS^{-1}$ and 1.68~420 kg $CO_2 \cdot tDS^{-1}$, respectively. The IPCC emission factors are derived from different solid compost data from around the world, which may give a relatively broad range of results. Therefore, monitoring data based on actual projects is more accurate. In this study, the direct emission from composting is 59.3~66.47 kg $CO_2 \cdot tDS^{-1}$. With a ventilation volume of 10 $L \cdot (h \cdot kg)^{-1}$ considered, the average direct emission of N_2O and CH_4 is equivalent to 63.28 kg $CO_2 \cdot tDS^{-1}$.

The N_2O emissions from composting-involved land application are shown in Figure 6, which is lower compared with the direct use of digested sludge, and a possibility is that composted sludge tends to have a lower moisture content and a higher C/N ratio. Similarly, the nitrogen input should be controlled within 100~300 kg $N \cdot ha^{-1}$, and the corresponding compost sludge application amount is 4.61~14.2 $t \cdot ha^{-1}$. The range of the nitrogen emission factor is 0.25~0.5% [27]. According to the total nitrogen content in composted sludge, the direct N_2O emission in land application is 0.0825~0.165 kg $N_2O \cdot tDS^{-1}$, equivalent to 24.59~49.17 kg $CO_2 \cdot tDS^{-1}$. With a nitrogen application level of 200 kg $N \cdot ha^{-1}$ considered, the average carbon emission resulting from direct N_2O emissions is 0.12 kg $N_2O \cdot tDS^{-1}$, equivalent to 35.76 kg $CO_2 \cdot tDS^{-1}$.

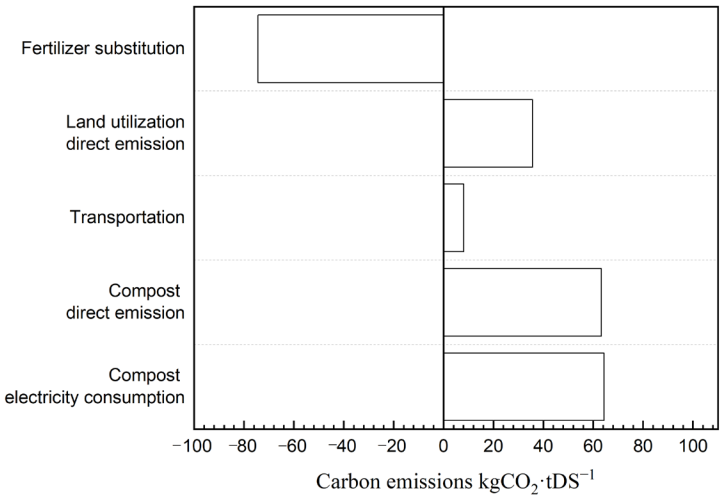

Figure 6. Carbon emissions from composting-involved land application.

3.3.4. Comparison of Different Digested Sludge Post-Treatment Routes

Different disposal solutions for digested sludge may have significantly different influences on Lu'an City's carbon footprint. After AD, the organic content can decrease to approximately 30% due to degradation. Figure 7 illustrates the carbon footprint of 1 tDS of sludge (dry matter excluded). The brickmaking solution can result in a carbon footprint of 858.4 $kgCO_2$, the land application solution has a carbon footprint of 347.7 $kgCO_2$, and the composting-involved land application solution is subject to 459.7 $kgCO_2$.

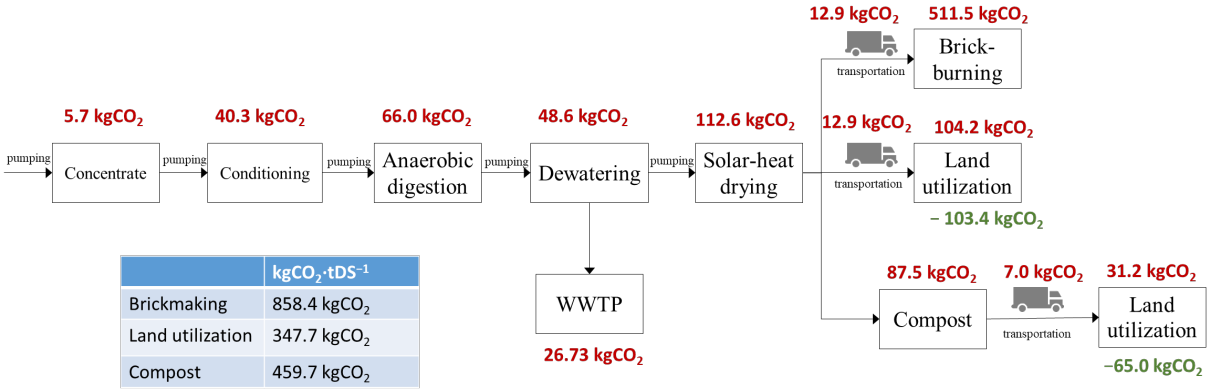

Figure 7. Sludge carbon footprint in Lu'an city.

Here, the carbon emissions of brickmaking solutions are highest. But, in a past study, the carbon emissions of brickmaking by sludge were 36.5 $kgCO_2 \cdot t^{-1}$ [50], much lower than this study. That is because, in Chen's study, only carbon emissions from electricity were calculated. However, in this study, the addition of digested sludge is equivalent to increasing the water content of the brick. In order to evaporate the water, more coal gangue is consumed, thus releasing fossil carbon, which is the main source of carbon emissions. Studies have shown that, similar to incineration, the carbon emissions of brickmaking are higher, mainly because of higher sludge drying requirements [3]. But brickmaking offers an alternative to the disposal of digested sludge in landfills and avoids the generation of incineration ash. So, it remains a sustainable way of sludge management [20].

Overall, land application is superior to brickmaking in terms of carbon emission reductions, whether composted or not. Land application of composted sludge has a lower carbon emission than the direct use of digested sludge. However, the power consumption of aerating and turning, as well as the direct N_2O and CH_4 emissions in composting, increases its carbon emission reduction, making it higher than that of the direct use of digested sludge. In practice, nevertheless, digested sludge that is not subject to high-temperature sterilization and fermentation is phytotoxic and can be fetid [51], which limits its universality. The GI represented phytotoxic is shown in Table 6. Composting can effectively improve the GI of digested sludge. In addition, due to the higher C/N ratio of composted sludge (Table 5), the usage of composted sludge is about 1.6 times that of digested sludge for the same N input, which is conducive to solving the disposal problem of large amounts of sludge. Therefore, if possible, composting-involved land application should be a preference; yet, technique improvements are required to minimize energy consumption and direct N_2O and CH_4 emissions.

Table 6. Comparison of GI and sludge usage.

	GI	Sludge Usage (Dry Basis) 200 kg N ha^{-1} Input	Sludge Usage (80% Moisture Content) 200 kg N ha^{-1} Input
Digested Sludge	42~53%	5.88 $t \cdot ha^{-1}$	29.40 $t \cdot ha^{-1}$
Composted Sludge	81~92%	9.52 $t \cdot ha^{-1}$	47.60 $t \cdot ha^{-1}$

In this study, we monitored the cumulative GHG emissions in only the first composting cycle (11 days) since the sludge subject was applied on land immediately after the 11-day composting process. In practice, most composting projects involve both primary fermentation and secondary fermentation, plus the possible GHG emissions during natural aging, which this research did not cover. Moreover, composting parameters also influence GHG emissions, which provides an opportunity for future research to reduce energy consumption and minimize GHG emissions by optimizing or adjusting these parameters.

4. Conclusions

This study compared the carbon footprints of different digested sludge post-treatment routes, taking Lu'an City in China as an example, and the results showed the following: (1) solar drying was the highest carbon emissions procedure in AD plant; (2) the carbon footprint of digested sludge-involved brickmaking was 858.4 $kgCO_2 \cdot tDS^{-1}$; (3) the carbon footprint of digested sludge-involved land application was 347.7 $kgCO_2 \cdot tDS^{-1}$; and (4) the carbon footprint of composted sludge-involved land application was 459.7 $kgCO_2 \cdot tDS^{-1}$. In general, land application was superior to brickmaking from the perspective of carbon footprint. As for land application, specifically, the composting-involved land application should be superior to the direct use of digested sludge on land (i.e., digested sludge-involved land application). However, the power consumption incurred by aerating and turning and the direct N_2O and CH_4 emissions during composting increase the former's overall carbon footprint, which exceeds that of the latter. However, digested sludge that is not subject to high-temperature sterilization and fermentation is phytotoxic and can be fetid, which is a limitation of its applicability. Since the composted sludge has a lower N ratio and water content, the same N input means more sludge usage, which is conducive to solving the disposal problem of large amounts of sludge. Thus, if possible, composting-involved land application should be a preference, and improvements to the technique are required to minimize energy consumption and direct N_2O and CH_4 emissions.

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