

Greenhouse Gas Emission and Reduction in Biological Treatment of Agricultural Organic Solid Wastes Toward Resource Recycling

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Abstract

Agricultural organic solid wastes (AOSW) are often accompanied by agricultural production, with large quantities, leading to resource waste and environmental impacts. Composting and anaerobic digestion (AD) are mainstream and recommended methods of AOSW disposal in China. This research conducts a review of the greenhouse gas (GHG) emissions of AOSW composting and AD methods, with consideration of advancements, challenges, limitations, and future directions. Composting's biogenic GHG contributes 30–740 kg CO₂-eq·t⁻¹ DM, with feedstock and systems as main factors. AD's biogenic GHG comes from 0–10% methane leakage (196.5–982.4 kg CO₂-eq·t⁻¹ DM) and digestate composting (276.6–1002.3 kg CO₂-eq·t⁻¹ DM). Non-biological GHG for both is 11.3–561.8 kg CO₂-eq·t⁻¹ DM. Process regulation cuts GHG by 13.9–55.8%, exogenous addition by 6.0–97.3%. This paper highlights and discusses AOSW biological treatment's GHG emission and reduction strategy, supporting low-carbon resource recycling.

Keywords: agricultural organic solid wastes; composting; anaerobic digestion; greenhouse gas; greenhouse emission reduction; life cycle assessment; resource recovery

1. Introduction

Agricultural residues are generated during agricultural production. In particular, agricultural organic solid wastes (AOSW), such as crop straw, vegetable residues, livestock and poultry manure, and livestock bedding material [1], have become a global pollution concern. Crop straw and livestock manure account for approximately 86% of biodegradable agricultural organic solid waste (AOSW). This biodegradable waste has not only the problems of large quantity but also the problem of uneven distribution and uncoordinated treatment capacity [2]. Furthermore, these problems restrict the resource utilization of AOSW and lead to a severe waste of resources and other environmental issues. Crop straw is directly burned in fields [3], resulting in significant CO₂ emissions. Furthermore, burning produced solid particulate matter PM_{2.5} to damage air quality [4]. Meanwhile, the direct discharge of livestock and poultry manure causes eutrophication in water bodies and reduces water quality [5]. Composting and anaerobic digestion (AD) are currently the most widely used methods for AOSW treatment, as they enable resource recycling and volume reduction [6]. However, they still face the challenge of greenhouse gas (GHG) emissions.



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Composting is the establishment of an aerobic environment that converts degradable organic substances into high-value fertilizers through a series of chemical-biological reactions [7]. Organic fertilizers benefit from reducing global warming potential (GWP) by replacing chemical fertilizer production [8]. However, the composting process is microbial-driven, so greenhouse gas (GHG) emissions are unavoidable [9]. During the early stage of the composting process, abundant nutrients and organic compounds stimulate microbial growth. As oxygen is consumed, anaerobic environments form, which in turn promote the rapid reproduction of methanogens and the release of most methane (CH_4) [10]. Nitrification and denitrification lead to the release of nitrous oxide (N_2O) during composting. Although the emission amount of N_2O is considerable, the global warming potential (GWP) of N_2O is 196 times higher than that of carbon dioxide (CO_2) [11]; therefore, N_2O emission remains an important issue. In addition, composting is usually a net energy-consuming process due to the energy required for aeration or turning.

AD refers to the degradation of AOSW by microorganisms in an anaerobic environment and concurrently produces biogas and digestate, producing both renewable energy and recyclable fertilizers [12]. Biogas is used for cogeneration and grid-connected power supply. Thus, it could avoid part of the GHG emissions related to the combustion of fossil fuels [13]. This substitution resulted in emission reduction (“avoided emissions”). Digestate is used as a feedstock for composting and fertilizer to return to the field. However, high ammonia nitrogen concentration and the accumulation of volatile fatty acids can inhibit microbial activity, which has an adverse effect on methane production [14]. In addition, anaerobic digestion requires additional equipment and energy input, and the GHG emissions during digestate composting may not be avoided [15]. That means AD’s overall GHG performance depends not only on biogas capture efficiency, but also on the subsequent management of its by-products.

Thus, it is necessary to develop emission mitigation strategies to minimize the negative environmental impacts of AD and composting. The number of publications on composting has increased dramatically over the past 20 years. A review of the frontiers of research on GHG emissions from composting and AD can help guide future research and the application of both methods and their potential to improve the sustainability of AOSW treatments (see Support Information). Therefore, this review aims to (i) Comprehensively review the GHG emission of AD and composting systems, focusing on the biological GHG emission (i.e., CH_4 , CO_2) and non-biological GHG emission (i.e., CO_2), and (ii) Summarize the effective strategies for GHG emissions reduction during AD and composting process. Novel solutions are figured out to highlight the potential scale-up and address future research perspectives. This review was performed following the updated version of PRISMA guidelines [16].

2. AOSW Biological Treatment Technology

The biological treatment of AOSW is an excellent way to make it harmless and a resource. Composting and AD are the most popular biological treatment methods, which account for 75% of the total amount of treated agricultural organic waste [17]. Vermicomposting and other biological treatment methods have not been popularized due to low efficiency, complex implementation, and low market demand [18]. Composting and AD are the mainstream and recommended methods of organic waste disposal.

Microorganisms, as the main actors in biological treatment, must perform respiration, while equipment and transportation cause energy consumption, inevitably producing GHG during the process. It is essential to study and summarize the two fundamental ways of carbon reduction in the biological disposal of agricultural organic waste to achieve a carbon peak in 2030 and carbon neutrality in 2060 [19].

Composting is easier and cheaper to implement, and AD has higher environmental and economic benefits [20]. AD of waste is a process in which microorganisms decompose organic matter and release CH_4 under anaerobic conditions. The AD target product is biogas, which is often used for power generation or purification in industrial applications and has been widely used to achieve economic benefits [21] (Table 1).

Table 1. Comparison of composting systems.

System	Static Composting	Aerobic Composting		
		Windrow Composting	Tunnel Composting	Reactor Composting
Functions	None	Turning	Aeration	Aeration, temperature, stirring
Advantages	Easy and save money	Large quantity and medium efficiency		High efficiency
Biogenic GHG emissions	High	Mid	Mid	Low

2.1. Composting

Composting is considered an effective ASOW strategy in which biochemical processes convert organic matter (OM) into relatively stable humus-like material that can then be used as organic fertilizer or soil additive [22].

2.1.1. GHG Emission

The GHG directly produced by microbial degradation of organics includes carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4), and ammonia (NH_3) [23,24]. This part is also called the biogenic GHG. According to the IPCC 2006 GHG emissions inventory, the GWP of CH_4 and N_2O is 28 and 265 times the CO_2 equivalent (eq) [24]. It is well established that 36.4–56.3% carbon and 16.5–35.8% nitrogen are lost during the composting process [25–27]. Considering the total carbon loss, the maximum loss in the form of CO_2 is more than 70%, and the loss in the form of CH_4 is 0.01–0.7% [25–27]. While most nitrogen is lost in the form of NH_3 , and only 0.2–1.1% of nitrogen is lost in the form of N_2O [28]. The detailed greenhouse gas emissions during the main stages of the composting process are presented in Table 2.

During composting, organic material is degraded by microorganisms; thus, aerobic microorganisms' metabolism inevitably produces CO_2 . Most of the carbon was lost in the form of CO_2 . However, it is worth mentioning that CO_2 emissions from composting do not contribute to the greenhouse effect because CO_2 is produced from organic matter mineralization in the substrate [29].

CH_4 is produced mainly by methanogens under suitable organic substrates and anaerobic conditions [30]. Therefore, a local anaerobic environment in the compost is the source of CH_4 production [31]. Especially in the early stage of composting, due to the abundance of organic matter and nutrients in the early stage, aerobic bacteria consume a lot of oxygen, leading to the intensification of the anaerobic environment [32].

Studies have shown that the mechanism of N_2O production is complex. Nitrification is one of the essential microbial processes responsible for the emission of N_2O during composting. The initial transformation of ammonia to nitrite by different genera of ammonia-oxidizing bacteria (AOB) is the primary process involved in aerobic nitrification [9]. Subsequently, under the action of denitrifying bacteria, the $\text{NO}_3^-/\text{NO}_2^-$ is converted into N_2O . And if an aerobic environment (O_2) is absent, NO_3^- will be reduced

to N_2 without appreciable N_2O production [33]. So the N_2O is produced in an aerobic environment by the nitrification and denitrification of microorganisms [34].

The mineralization of microorganisms releases the N-containing compound, which undergoes ammoniation to produce NH_3/NH_4^+ . The NH_3 from compost also contributes indirectly to GHGs. Because the NH_3 is redeposited and volatilized, a proportion of N_2O was released during denitrification and nitrification [10,34].

2.1.2. Factors Influencing Biogenic GHG Emissions

Indigenous Factors

- Composting process

Static composting and active composting were the most common forms of composting [35]. Static composting, or anaerobic composting, called “ou fei” in Chinese, is a very old and traditional compost that does not require active contact with air and turning of the pile. And thus, in some studies, it is also treated as waste storage [36,37]. The static compost accounted for a large proportion of agricultural organic solid waste treatment. The pig, chicken, cattle, horse, and sheep manure treated by static composting accounted for 36.6%, 42.0%, 51.3%, 38.1%, and 46.9% of the volume of each livestock, respectively [17]. The advantages of static composting were low investment and easy-to-control operation. However, the inefficient and longer time for agricultural organic solid waste to be converted into non-stabilized fertilizer were the main disadvantages of decision-making. It was reported that it would take at least three months to a year for one pile to turn out to be manure fertilizer during static composting. Furthermore, the GHG from static composting was $174\text{--}790\text{ kg CO}_2\text{-eq}\cdot\text{t}^{-1}$ AOSW [38]. The CH_4 emission contributes to 76% of GHG because static compost does not carry out active aeration and produces an anaerobic environment inside the pile [38].

Composting requires machinery for active aeration. The advantages of composting include high efficiency and a controllable process [39]. Fatty manure with vigorous aeration can mature within two to three months [40]. Windrow composting, static forced-aerated composting (also called aerated trough composting or tunnel composting), and reactor composting [10] were the primary composting types. The windrow composting by turning provides sufficient oxygen concentration for aerobic biodegradation [41]. The GHG emission from windrow composting ranges from $169\text{--}392\text{ kg CO}_2\text{-eq}\cdot\text{t}^{-1}$ dry matter (DM) of AOSW [42,43]. Bai [42] reported that the emissions of N_2O from windrow composting were significantly higher than from static composting, with windrow composting accounting for 70% of the GHG of N_2O , and CH_4 no longer dominating. Since active pile turning aeration reduced CH_4 production while promoting N nitrification, the static forced-aerated composting is achieved by arranging ventilation holes in the channel and using a blower for active aeration, thus performing efficient aerobics. Studies on the total GHG emissions from static forced-aerated composting (or aerated trough composting) are relatively rare, and it is hard to summarize the range of GHG emissions. In He's [44] study on aerated trough composting, the contribution of N_2O was higher than that of CH_4 . It also confirmed that CH_4 accounted for 83% of the total emissions when not aerated.

The reactor composting is even more advanced, with active aeration and turning of the pile in the reactor, even adjusting the temperature and step-by-step processing of raw materials [41]. The biogenic GHG emission from reactor composting was $12\text{--}240\text{ kg CO}_2\text{-eq}\cdot\text{t}^{-1}$ DM of AOSW. Reactor composting can reduce GHG emissions by regulating the appropriate aeration conditions and turning frequency, thus avoiding an anaerobic environment or accelerating nitrification.

These composting techniques promote maturation and reduce CH₄ production due to active aeration to achieve appropriate oxygen levels. Therefore, the biological composting process produces less GHG, and the composting efficiency is high. But currently, they all have their application scenarios, and no one can completely replace the other.

- Feedstocks

Scholars have conducted much research on feedstock affecting GHG emissions during composting. The primary feedstock studied by the researchers is livestock manure and AD. Figure 1 shows the GHG emission ranges of composting with different raw materials. Typically, composting produces about 30–740 kg CO₂-eq·t^{−1} DM GHG emissions from all types of AOSW.

Feedstock properties, including ultimate and proximate composition (C, N, H, ash content), moisture content, particle size, and C/N ratio, strongly affect the composting process. These characteristics influence oxygen transfer, microbial community succession, aerobic/anaerobic microenvironments, and nutrient availability, thereby affecting CH₄ and N₂O emissions. For example, pig manure has high protein and fat content compared to other livestock manures, resulting in elevated NH₃ and N₂O emissions [28]. N₂O was the predominant emission from pig manure composting, accounting for more than 40% of the total GHG emissions. Furthermore, composting size significantly affects GHG emissions during pig manure composting. Small piles (60–200 L) had higher GHG emissions than large composting pile sizes (5–35 m³) because the gases produced in the center of the pile could easily reach the surface to escape in the small-sized pile [45].

Chicken manure composting resulted in the lowest GHG emissions of all livestock manure. In the chicken manure compost studied by Zhang [46], its GHG emissions reached a minimum of 5.49 kg CO₂-eq·t^{−1} DM AOSW due to his choice of tobacco waste, mushroom substrate, and chicken manure for co-composting while adjusting the right aeration intensity and adding 10% biochar. However, chicken manure composting does not always maintain low GHG emissions, and in other studies, chicken manure composting had a GHG between 15.05–147.88 kg CO₂-eq·t^{−1} DM [47].

The GHG emission range of cattle manure composting is about 51.2–207 kg CO₂-eq·t^{−1} DM [48,49]. And in Pattey's [49] study, the GHG of beef cattle manure compost was less than that of dairy cattle manure. Sheep manure composting also produced less GHG emissions than other manure composts, and the lowest emissions reached 21.27 kg CO₂-eq·t^{−1} DM. But the GHG emissions from sheep manure composting have been as high as 300 kg CO₂-eq·t^{−1} DM in recent studies [50].

The manure digestate composting produced the most GHG in the 276.6–1002.3 kg CO₂-eq·t^{−1} DM, which is attributed to the excess ammonia in the digestate, which leads to high N₂O production from the composting process [10]. In general, even composts using the same feedstock can vary substantially in emissions, highlighting that feedstock ultimate/proximate composition, particle size, and process conditions jointly determine microbial activity, GHG formation, and energetic potential. These insights emphasize the importance of reporting feedstock characteristics when evaluating composting and AD emissions.

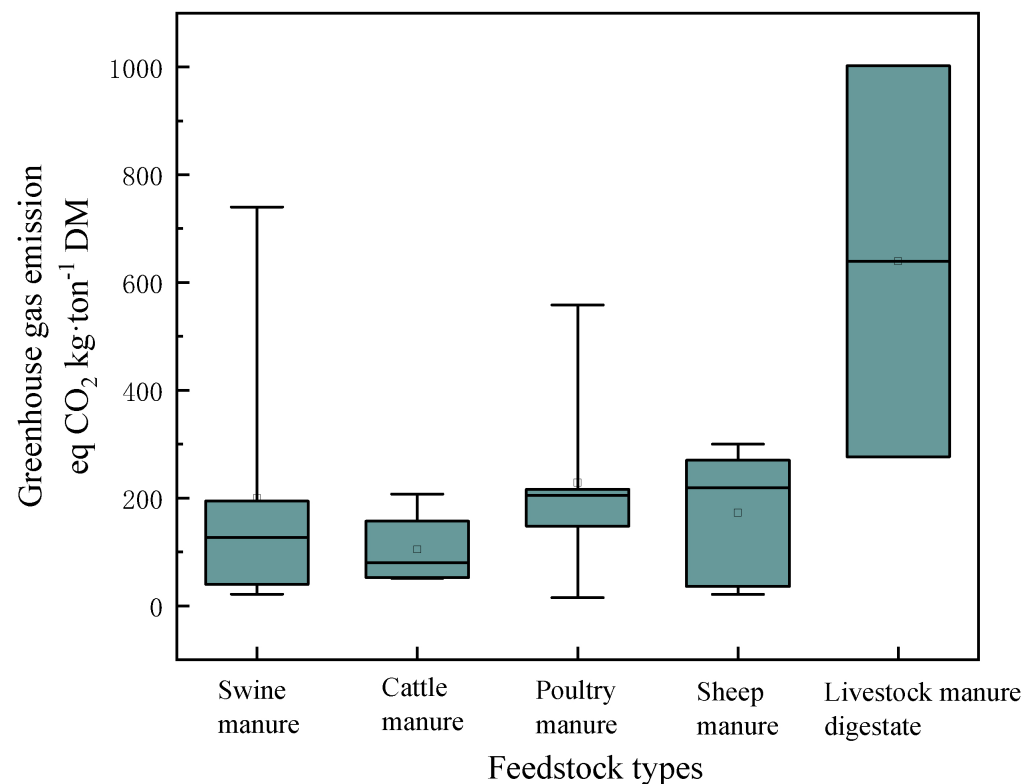


Figure 1. Biogenic GHG emission of different compost materials. (reference: Swine manure [51–53], Cattle manure [48,49], Poultry manure [46,47], Sheep manure [50,54,55], Livestock manure digestate [56]).

- **Transportation**

In addition to producing GHG through biological processes, other processes related to composting indirectly have GHG emissions. The indirect GHG emissions contributed to 25–70% of the total GHG emissions during the whole composting process [56]. Many studies on life cycle assessment (LCA) of composting and environmental impact assessment studies have helped us understand the abiotic GHG emissions of the composting process. In life cycle assessment investigations, the system boundary of composting is generally from raw materials collection and transportation to the final utilization stage [57]. The system boundary includes transportation, pretreatment, composting, and compost used as fertilizer. The energy consumption by the equipment causes indirect GHG emissions (Table 2). The main contributor to GHG emissions during transportation is the combustion of diesel fuel. The GHG produced by the explosion of diesel fuel in engines includes CO₂, CO, NO_x, and SO₂, and the primary GHG emissions are CO₂ and N₂O. The GHG emissions of the transport process depend mainly on the truck's fuel consumption, transport distance, and efficiency. Previous studies have indicated that average transport distances were 10–200 km [58], with fuel consumption by transport trucks being 0.23–0.36 L·km^{−1}. In Li et al.'s study [59], transportation contributed to 38% of environmental burdens based on GWP when the distance was 30 km. The transport of fuels is considered to be 0.4–0.5 kg CO₂ L^{−1} diesel [60]. At the same time, the straw collection and baling will significantly improve transport efficiency; carrying as much straw as possible without overloading will dramatically improve efficiency and reduce fuel consumption.

Table 2. GHG emissions in the composting process.

Parameter	Transportation	Pretreatment and Mixing	Aeration and Turning	Composting Biogenic	Deodorization	Postprocessing
GHG emission kg CO ₂ -eq·t ⁻¹ ww	7.33–300	1.08–2.34	1.14–20	14.02–468	1.92–24.06	3.57–4.12
GHG emission kg CO ₂ -eq·t ⁻¹ DM	11.27–461	1.66–3.6	1.75–30.77	21.57–720	2.95–37.02	5.49–6.33
References	[34,52]	[52,54]	[56,59,60]	[34,52]	[56,61]	[59]

Note: The references without GWP are calculated according to the current electricity GWP coefficient of 600 g CO₂·KWh⁻¹. And in the literature, labeled as wet weight, we convert dry weight by the average moisture content of the feedstocks as 65%.

- Feedstock pre-treatment

Pretreatment is a crucial step in the composting process. Non-biological GHG emissions of pretreatment differ in pretreatment methods and machine applications. The standard pretreatment machines used in composting are shredders, magnetic separators, air separators, and dewaterers, all designed to better screen compost materials and build good aeration pores. After the straw is crushed in the chopping machine, the mixing process of raw materials will be carried out. Composting often consists of two or more kinds of raw materials for co-composting. Thus, it needs to be mixed before starting to make the material homogeneous. The GHG emissions of the pretreatment process vary with the chopping and mixing machine. Li et al. [59] reported the chopping machine with a capacity of 10 m³·h⁻¹ and rated power of 7.5 kW, and the mixer with a rated power of 4 kW. The GWP for shredding varies between 12.21 and 16.37 kg CO₂-eq per m³ of compostable materials. Studies [61] have shown that the energy consumption of pretreatment and mixing is approximately 1.8–3.9 kWh·t⁻¹ wet weight (WW) with a GWP of 1.66–3.6 kg CO₂-eq·t⁻¹ DM according to the corresponding electricity GHG emission factor.

- Equipment and energy consumption

After composting fermentation, the energy consumption of the process mainly includes turning and aeration devices. Different composting systems are supplied with oxygen in different ways. Windrow composting relies largely on turners as well as shovels for turning. Static tunnel composting relies on blowers for aeration. And the reactor composting the reactor compost has both turning blades and blower aeration inside the reactor. In-plant scale composting, the electricity consumption of the turner is 1.3–1.9 kWh·d⁻¹·t⁻¹, and the electricity consumption of the blower is 9.1–14.1 kWh·d⁻¹·t⁻¹ [61]. However, some scholars have studied several major composting plants in Beijing [62] and concluded that the total electricity consumption for large-scale composting is in the range of 5.5–9.2 kWh·d⁻¹·t⁻¹. The corresponding resulting GHG emissions range from 3.3–5.52 kg CO₂-eq·t⁻¹. Each kilowatt-hour of electricity will produce 600 g of CO₂ emissions [63]. The more frequently the pile is turned, the greater the ventilation intensity, and the more GHG is produced. In general, the closed composting systems caused more GHG emissions than the open composting systems [43]. A large amount of odor is generated in composting production, and large-scale composting is often equipped with deodorization units. The Deodorizing devices include air centrifuges, pumps, sprayers, etc., and their energy consumption ranges from 3.2 to 40.1 kWh·t⁻¹ WW [64]. Its corresponding GWP is in the range of 2.95–37.02 kg CO₂-eq·t⁻¹ DM. After composting, the product often has to be productized if it needs to be sold. This includes steps such as granulation, tumbling, and sieving. Making organic fertilizer requires a sifter, twin-shaft grinder, granulator, packaging machine, etc.,

and its total energy consumption reaches 5.95–6.87 kWh·t^{−1} WW. The resulting GWP is about 5.49–6.33 kg CO₂-eq·t^{−1} DM [65]. Composting, returning to the soil, or making a substrate for cultivation can also cause energy consumption. The GHG emission values of the utilization process are between 1.6 and 29.23 kg CO₂-eq·t^{−1} DM. However, as the nutrient return to the field replaces the production and application of chemical fertilizers, the substitution effect GWP contribution is −4 to −223.07 kg CO₂-eq·t^{−1} DM [43]. And the use of alternative peat soil as a seedling substrate will reduce GWP [43]. The GHG emissions are also mainly affected by transportation distance and the power consumption of fertilizer manufacturing equipment.

Exogenous Factors

Seasonal variation is an important uncontrollable factor affecting GHG emissions during composting, mainly because ambient temperature, humidity, and ventilation conditions jointly regulate microbial activity, oxygen availability, and the dominance of methanogens, nitrifying bacteria, and denitrifying bacteria. In general, warmer conditions favor faster organic matter degradation and a longer thermophilic phase, whereas colder conditions suppress microbial activity and prolong stabilization. For example, in a composting experiment on hatchery waste from a Brazilian layer farm, the thermophilic phase in summer composting lasted 11.3% longer than that in winter composting, and summer composting showed greater substrate degradation with higher nitrogen and carbon losses than winter composting [66,67]. Similar seasonal effects have also been reported in full-scale composting systems: the maximum N₂O flux in cold-season animal manure windrow composting was 23 times higher than that in warm-season composting, while methane emissions from composting plants were about 50% higher in spring and summer than in autumn and winter. Consistent with these observations, an industrial-scale study on semipermeable membrane-covered composting showed that cumulative N₂O emissions under cold ambient conditions were 10.16–30.05% lower than those under hot conditions, and the membrane-plus-optimization treatment reduced N₂O by 47.41–52.71% in cold conditions compared with 21.50–32.04% in hot conditions [68].

Temperature is one of the key environmental factors regulating composting emissions, because it affects microbial activity, moisture loss, and the balance between aerobic and anaerobic microenvironments. Controlled composting at 40, 55, and 67 °C showed that CH₄ and N₂O emissions were generally low, with CH₄ at 67 °C decreasing to 0.001% of the mineralized carbon under 16% O₂ [69].

Moisture also interacts strongly with temperature: in food- and garden-waste composting, increasing moisture from 44% to 66% significantly changed CH₄, N₂O, and CO₂ emissions, indicating that high moisture can amplify the temperature effect [70]. Seasonal field measurements further showed that in full-scale animal manure windrow composting, the maximum N₂O flux during the cold season was 23 times higher than during the warm season, and the direct N₂O emission factors were 0.0046 and 0.0002 kg N₂O-N/kg TN, respectively [71]. Methane emissions from composting plants were also about 50% higher in spring and summer than in autumn and winter, confirming that ambient season has a clear influence on CH₄ release [72].

2.1.3. Summary of GHG Emission Characteristics in Composting

As a mainstream biological treatment technology for AOSW, composting exhibits significant contradictions in GHG emissions, although it can achieve partial GHG offset by replacing chemical fertilizers, its emissions are still dominated by biogenic GHGs such as CH₄ and N₂O, with substantial variations across different process modes. Static composting is widely used due to its low cost and simple operation; however, the lack of

active aeration creates an anaerobic environment inside the compost pile, leading to massive CH_4 production and a long maturation period. In contrast, active aeration processes like windrow composting and reactor composting can effectively inhibit CH_4 generation by increasing oxygen supply. Nevertheless, windrow composting causes elevated N_2O emissions due to enhanced nitrification-denitrification, while reactor composting, despite its advantages, results in higher GHG emissions due to increased equipment investment and energy consumption. Meanwhile, non-biogenic emissions from transportation and pretreatment equipment energy consumption cannot be ignored. Additionally, the characteristics of composting feedstocks further exacerbate emission fluctuations. For instance, pig manure, which is rich in protein and fats, has significantly higher N_2O emissions than manure from other livestock and poultry. Therefore, the AOSW composting process faces the trade-off dilemma between biogenic emissions and non-biogenic energy consumption.

2.2. Anaerobic Digestion

AD is a process that uses anaerobic microorganisms to decompose organic waste and produce biogas [73]. The biogas can be burned to generate electricity or collected for industrial use. The by-product of AD is biogas residue (i.e., biogas slurry and digestate), which can be returned to the field after composting and compound treatment. Therefore, AD is an effective way to realize waste resource utilization.

2.2.1. Indigenous Factors

Biogas Leakage

Anaerobic fermentation takes place in a closed reactor. Theoretically, there are no GHG emissions unless there are leaks. However, some leakage is inevitable due to no completely ideal state. In the AD process, biogas leakage is classified as a biogenic greenhouse gas emission. CH_4 and CO_2 are the main components of biogas, accounting for 50–70% and 30–50%, respectively [74]. NH_3 , H_2S , and other odor gases account for about 5%. Biogas leaks from digesters, biogas treatment facilities, and unburned CH_4 in exhaust gases can seriously increase GWP [75]. The gas leakage from the AD process accounts for about 1–5% of the total biogas content [76], which is responsible for the GHG emission of 196.5–982.4 $\text{kg CO}_2\text{-eq}\cdot\text{t}^{-1}$ DM.

Digestate Treatment

The GHG emissions from the disposal of digestate should not be neglected. In the past, it was often burned or landfilled as a by-product, ignoring the resource value of the digestate [77]. With further research, the fertilization of digestate is now attainable [20]. Usually, the digestate is dewatered in a centrifuge, and the liquid part, the biogas slurry, is made into a liquid organic fertilizer through a membrane concentration process and elemental compounding. The solid fraction can be composted by adding carbon source material [77]. The method of digestate treatment contributes to substantial GHG emissions. This contribution comes from the biogenic GHG emissions from digestate composting [78]. The GHG emission of biogas residue compost is significantly higher than that of straw and livestock manure compost, which is about 276.6 to 1002.3 $\text{kg CO}_2\text{-eq}\cdot\text{t}^{-1}$ DM [55].

Equipment and Energy Consumption

Similar to composting, the system boundary of AD is usually delimited from the feedstock collection and transportation to the final stage of biogas and biogas residue utilization [79]. Transportation, pretreatment, mixing, anaerobic device power consumption, biogas purification, and biogas residue treatment and return to the field of biogas residues can all lead to non-biological GHG emissions [80]. The whole process of the AD GHG effect is shown in Figure 2. The transportation distance still determines the carbon emissions

generated by the transportation process. The GWP of the pretreatment process of AD was $11.22 \text{ kg CO}_2\text{-eq}\cdot\text{t}^{-1} \text{ DM}$ in the study of Yang [81]. And $250 \text{ kg CO}_2\text{-eq}\cdot\text{t}^{-1} \text{ DM}$ in the study of Li [59]. The difference is that the pretreatment methods differ according to the raw materials. The GWP of anaerobic equipment varies, ranging from 280 to $456.84 \text{ kg CO}_2\text{-eq}\cdot\text{t}^{-1} \text{ DM}$. The main influencing factors are the temperature of the anaerobic system, the stirring frequency, and the movement mode of the feedstock [82]. The GWP caused by biogas purification treatment ranges between 19.3 and $424.3 \text{ kg CO}_2\text{-eq}\cdot\text{t}^{-1} \text{ DM}$. Biogas purification is mainly to remove sulfur-containing gases, such as H_2S . The difference is caused by the big difference in the desulfurization process. Meanwhile, some projects are used for industrial purposes after biogas purification. Then, the purified biogas compression canning process will cause more carbon emissions [81]. During the soil application process of biogas residue and biogas slurry, GHG emissions were mainly caused by transportation and fertilizer machines [83]. The GWP of this process was reported as $170\text{--}220 \text{ kg CO}_2\text{-eq}\cdot\text{t}^{-1} \text{ DM}$.

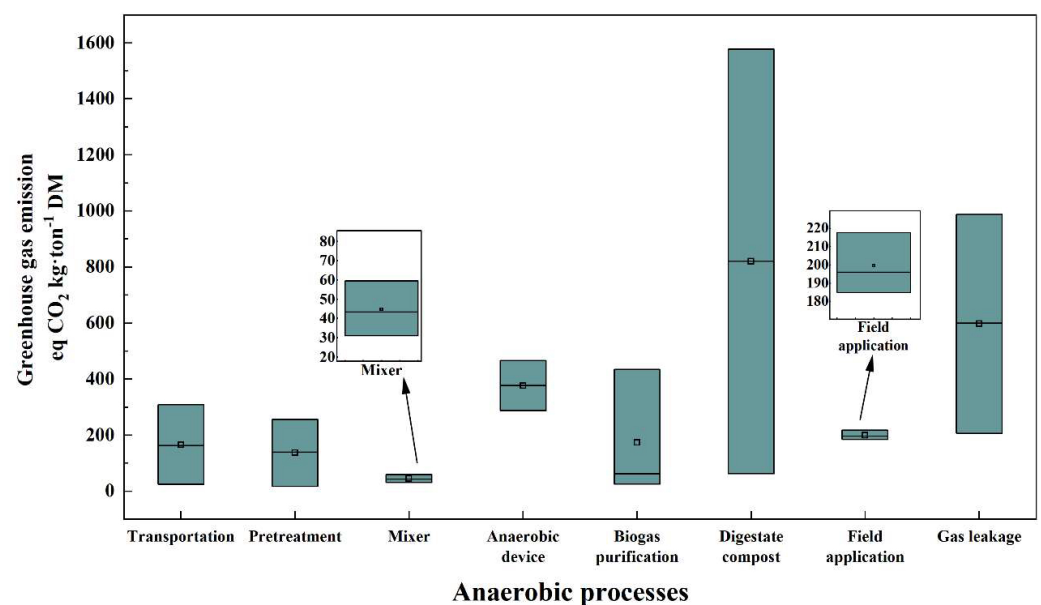


Figure 2. GHG emission in the whole AD process (collected data from [59,81,83]).

2.2.2. Exogenous Factors

Different types of anaerobic digestion (AD) reactors, due to differences in substrate feeding methods and mixing degrees, affect greenhouse gas emissions during the AD process [84]. However, reactor types usually cannot be changed in a short period during operation, and are regarded as uncontrollable external factors. The continuously stirred tank reactor (CSTR) stands as one of the most extensively utilized AD systems [85]. It is equipped with an agitator for mixing purposes, along with inlets and outlets designed for substrates and digestate, respectively. By means of continuous agitation, the CSTR sustains the homogeneity of the mixture, thereby guaranteeing comprehensive contact between the feedstock and microorganisms [86]. This system provides consistent reactions and stable conditions for gas production. Up-flow anaerobic sludge beds (UASBs) can provide high sludge concentration and long residence time, promoting good contact between microorganisms and organic matter [87]. A sequencing batch reactor (SBR) is an intermittent reactor that operates in multiple stages: feed, reaction, precipitation, discharge, and idle [88]. SBR is heavily dependent on process control and automation technology, so the design of the SBR reactor itself is a key factor in the energy consumption and greenhouse gas emissions of the AD process [89]. The scale of the reactor is also an uncontrollable factor affecting greenhouse gas (GHG) emissions during the anaerobic digestion (AD) process.

Under the same type of continuously stirred tank reactor (CSTR), the GHG production of small-scale reactors (2.6 L) is much higher than that of large-scale ones (500 L), with an increase of approximately 54.7% [90].

2.2.3. Summary of GHG Emission Characteristics in AD

Although AD can realize renewable energy utilization through digestate and biogas recovery, the GHG emission reduction advantages are constrained by two major bottlenecks: digestate disposal and CH₄ leakage and treatment. Theoretically, AD in closed reactors should achieve low emissions; however, incomplete sealing of digesters and biogas treatment facilities leads to partial biogas leakage. Furthermore, small-scale AD reactors have higher CH₄ leakage than large-scale ones. Regarding digestate disposal, due to the high ammonia nitrogen content of digestate, the GHG emissions during its composting process are even higher than those from the composting of raw manure. In addition, the energy consumption of pretreatment processes such as centrifugal dewatering of digestate further increases non-biogenic emissions. Currently, life cycle assessment studies on AD are mostly limited to the digestion process itself, omitting the emission accounting of links such as biogas purification and biogas slurry field application, which tends to overestimate the actual emission reduction benefits.

Digestate management is an important but often underestimated component of the overall GHG burden of AD systems. In this review, digestate handling is mainly reflected in digestate composting and soil application. The solid fraction of digestate can be composted by adding carbon source material, but digestate composting may generate substantial GHG emissions, with reported values of 276.6–1002.3 kg CO₂-eq·t⁻¹ DM [54]. In addition, the soil application of biogas residue and biogas slurry can also contribute notable emissions, with a reported GWP of 170–220 kg CO₂-eq·t⁻¹ DM [77]. These results indicate that digestate treatment is not a secondary issue but a key determinant of the total carbon footprint of AD systems. Therefore, future mitigation efforts should integrate digestate handling into the system boundary of AD rather than evaluating digestion alone.

3. Strategies for GHG Reduction

GHG emissions vary significantly between composting and AD. Even when using the same treatment technology, GHG emissions could differ due to the immense diversity of ASOW components and operational parameters. Thus, there is a great potential to bring out GHG reduction advantages when applied to composting and AD.

3.1. GHG Reduction Strategies for AD

3.1.1. Biogas Energy Utilization

AD is an effective technology for capturing the CH₄ and generating and recovering energy or fuels from low-cost biogas. The produced biogas is often purified and cleaned, used as fuel for cogeneration and generator systems, generating heat and electricity for internal consumption in AD plants, and remitting excess power to the local grid for energy gain. Alternatively, it is purified to remove the non-methane component and upgraded to biomethane for enrichment applications, primarily for cooking, home heating, and fuel for the shipping and road transportation sectors and chemical feedstocks. The most prevalent technology for the utilization segment downstream of AD is the generation of heat and electricity from the biogas/biomethane produced. There is a need to increase energy storage capacity and reduce energy losses due to grid entry.

Biogas is typically composed of CH₄ and CO₂, with smaller amounts of H₂S, NH₃, and water vapor, and its heating value depends mainly on methane concentration. Although biogas can be used for heat and electricity generation or upgraded to biomethane, practical application is constrained by leakage control, gas purification efficiency, corrosion risk from impurities, and the energy demand of upgrading and compression. Therefore, the net environmental benefit of AD depends not only on methane capture but also on the technical performance of biogas utilization and purification systems [91,92].

3.1.2. Biomethane Utilization

The biogas industry is currently undergoing transformation and upgrading, with biomethane emerging as a low-carbon, clean, safe, and efficient renewable energy source [93]. Also referred to as renewable natural gas (RNG), biomethane is produced via biogas upgrading, which is a technically mature process [94]. Compared with biogas, biomethane offers expanded application advantages, such as serving as vehicle fuel or being injected into natural gas grids, which broadens its value beyond traditional biogas uses. Globally, biomethane production through biogas upgrading has been widely promoted and applied in countries including Germany, the United Kingdom, Sweden, Brazil, and Canada [95]. In contrast, this technology is still in its initial development stage in China, despite significant potential. Specifically, the total potential biomethane yield from available crop residues in China is estimated at 82.2×10^9 m³ per year [96], and this volume could meet 29.2% of China's annual natural gas consumption in 2018 while achieving an annual greenhouse gas emission reduction of 197×10^6 t CO₂ equivalent. Therefore, developing biomethane represents a promising supplementary energy supply approach for China's energy system.

3.1.3. AD Process Optimization

Although the AD process is a mature technology, new approaches are still needed to improve the process specifications of biogas production facilities. Anaerobic co-digestion, enzyme addition, and micro-oxygen treatment are innovative methods to enhance substrate biodegradability, promote bioconversion, and improve biogas yield [97]. For the digestion segment, biological processes to remove impurities such as CO₂ and H₂S from biogas are receiving increasing attention due to increasingly stringent environmental regulations and power generation efficiency improvement [98].

3.1.4. Biogas Residue Treatment

The biogas residue treatment significantly contributes to GHG emissions during the whole AD process, urgently needing to be reduced [55]. In the previous section, we mentioned that carbon emissions from digestate composting are mainly generated from biogenic carbon emissions due to the large production of CH₄ and N₂O in compost, which contributes a large amount of GWP. After all, digestate composting is a process that reduces carbon in the same way as composting.

3.2. GHG Reduction Strategies for Composting

The GHG emission reduction potential of composting mainly focuses on the emissions from biological sources, and it is also essential to control composting energy consumption. Currently, the research on GHG emissions from natural sources of composting can be divided into two categories: the regulation of the composting process and the regulation of exogenous additives. The greenhouse gas emission reduction potential of different strategies is detailed in Table 3.

Table 3. GHG reductions of different parameters regulation method.

Feedstock	Bulking Agent	Control Parameter	The Conditions Before Control	The Conditions After Control	GHG Reduction Ratio	Reference
Sheep manure	Maize stover	moisture content	75%	65%	49%	[46]
donkey dung	none	moisture content	60%	55%	13.9%	[95]
Solids swine wastes digestate	cornstalks	C/N	24	18	55.80%	[83]
Pig manure	mushroom residue	turning frequency	once a week	Twice a week	−200.00%	[84]
Pig faeces	cornstalks	Aeration intensity	0.54	0.18	31.06%	[35]
sewage sludge	mushroom residue	ambient temperature	spring	winter	55%	[96]

3.2.1. Parameters Control

The technological parameters of composting process have been proven to affect the emission of GHG. Process parameter control is the most commonly used approach. The bulking agent, C/N, moisture content, aeration intensity, and the parameters are usually regulated. Table 3 is obtained through literature research and used to introduce the role and effect of different parameter adjustments.

Co-Composting

Co-composting uses two or more different types of raw materials mixed in a specific ratio to increase free air space (FAS) and achieve the adjustment of C/N and moisture content, to build a suitable environment for the growth of microorganisms [99].

The bulking agent is essential for manure composting. The characteristics (i.e., FAS, density, and C/N ratio) vary with a different bulking agent. Therefore, co-composting with another bulking agent has other effects on GHG emissions. Yang [100] used cornstalks, sawdust, and spent mushrooms as bulking agents and carbon sources co-composting with kitchen waste, and the results showed that the lowest GHG emission was 33 kg CO₂-eq·t^{−1} DM for sawdust treatment, followed by 55 kg CO₂-eq·t^{−1} DM for cornstalks. The highest GHG emission was 85 kg CO₂-eq·t^{−1} DM for the spent mushroom substrate. The result indicated that composting with sawdust as a bulking agent has better for GHG reduction.

During co-composting, the mixing ratio of the substrate is a crucial factor, which will change the moisture content, C/N, and further affect the GHG emission. Yang [100] investigated the effect of different addition ratios of the spent mushroom substrate on GHG emission of compost, the addition ratios of spent mushroom substrate were 15%, 25%, and 35% of wet weight. The results showed that the most negligible GHG emission was 193.5 kg CO₂-eq·t^{−1} DM when the addition ratio was 15%, and the GHG emission reached 406.4 kg CO₂-eq·t^{−1} DM when the addition ratio was 35%. The 35% treatment has significantly higher N₂O emissions than the 15% treatment, resulting in this result. The underlying reason is the difference in C/N, with lower C/N producing more N₂O emissions than higher C/N ratios [101]. The research of Xu [102] illustrates the same pattern when the initial material C/N ratio is 18, and the GHG emission is reduced by 55.8%, reaching 210.3 kg CO₂-eq·t^{−1} DM, which is compared with C/N 24. In the study of Li [53], moisture content can significantly affect the generation of GHG. Compared with 75% moisture content, treatment with 65% reduces GHG emissions by 48.6%, reaching 21.27 kg CO₂-eq·t^{−1} DM. High moisture content affects FAS, air-filled porosity, and matrix porosity, all of which can limit oxygen transport into the composting zone where cause higher CH₄ emission form pile.

Aeration Strategy

Aeration is the critical parameter in composting, which determines the oxygen concentration in a pile and thus influences microbial activity. Aeration strategies include the intensity and mode of ventilation, continuous or intermittent. Jiang [45] reported that the aeration intensity affected the level of oxygen supply to the pile and finally affected the production of GHG. During pig manure co-composting with corn stalks, the ventilation volume increased from $0.18 \text{ L} \cdot \text{kg}^{-1} \text{ DM} \cdot \text{min}^{-1}$ to $0.54 \text{ L} \cdot \text{kg}^{-1} \text{ DM} \cdot \text{min}^{-1}$, compared with a ventilation rate of $0.54 \text{ L} \cdot \text{kg}^{-1} \text{ DM} \cdot \text{min}^{-1}$, a 31% reduction of GHG emissions was achieved in the compost pile with ventilation volume of $0.36 \text{ L} \cdot \text{kg}^{-1} \text{ DM} \cdot \text{min}^{-1}$. Higher CH_4 production from compost pile with volume increased of $0.18 \text{ L} \cdot \text{kg}^{-1} \text{ DM} \cdot \text{min}^{-1}$, while higher N_2O production from too high aeration intensity. Similar findings were obtained by Zhao [103]. In their study, the effect of three gradient oxygen supply levels (0.1 , 0.2 , $0.3 \text{ L} \cdot \text{kg}^{-1} \text{ DM} \cdot \text{min}^{-1}$) on pollutant gas emissions during dewatered sludge compost, and the results showed that low oxygen supply levels exacerbated CH_4 and N_2O emissions.

In windrow composting, one of the functions of turning the pile is to aerate the pile actively. Zhao [103] investigated the effect of turning frequency on pollutant gas emissions in pig manure and mushroom residue composting and showed that twice as much GHG emissions were produced in the first phase of twice-weekly turning compared to once-weekly turning. In Jiang's [104] study on turning frequency, the non-turning treatment had higher GHG. Compared with the turning pile, higher GHG emission ($147.13 \text{ kg CO}_2\text{-eq} \cdot \text{t}^{-1} \text{ DM}$) was produced in a pile without turning. Since the material of pile without turning has an uneven distribution, it leads to more nitrification and denitrification, resulting in more N_2O emission.

The GHG emission from continuous and intermittent ventilation on composting was remarkably different [45]. The intermittent ventilation set to 40 min of aeration per hour with a 20-min stoppage produces lower GHG than that in continuous ventilation at $0.36 \text{ L} \cdot \text{kg}^{-1} \text{ DM} \cdot \text{min}^{-1}$ for one hour, which was $50 \text{ kg CO}_2\text{-eq} \cdot \text{t}^{-1} \text{ DM}$ [45]. The reason for this is appropriate aeration intermittently can provide the proper oxygen content and reduce the production of N_2O . Ma [105] studied two types of intermittent ventilation. A 9.7% reduction of GHG was found in treamens with 10 min on and 10 min off compared to treatment with 10 min on and 30 min off.

3.2.2. Exogenous Additives

According to the properties of additives, exogenous additives can be divided into physical additives (i.e., biochar and bentonite), chemical additives (i.e., superphosphate, calcium magnesium phosphate fertilizer, and dicyandiamide), and biological additives (i.e., exogenous microbial agents [106]).

Physical Additives

Physical additives are characterized by porous structures, large specific surface areas, and good water-holding capacity, which can improve air diffusion within composting piles [87]. Their surfaces also contain abundant adsorption sites, making them effective for immobilizing inorganic ions such as NH_4^+ and reducing the loss of reactive nitrogen species. Common physical additives include biochar, zeolite, calcium bentonite, activated carbon, and fly ash. Among them, biochar is the most widely used additive [88]. In a sewage sludge and wheat straw composting system, Awasthi et al. [54] reported that adding biochar at 8%, 12%, and 18% DW reduced CH_4 , N_2O , and NH_3 emissions by 92.8–95.3%, 95.1–97.3%, and 58.0–65.2%, respectively. Zeolite is typically applied at 5–15% DW. Wang et al. [107] found that a 10% DW combination of zeolite, biochar, and wood vinegar reduced NH_3 loss by 64.5–74.3% and lowered CH_4 and N_2O emissions by 50.4–61.2% and 79.5–81.1%,

respectively. Cao et al. [34] further showed through meta-analysis that zeolite alone reduced NH_3 volatilization by 32.7% and N_2O emissions by 28.4% on average. Calcium bentonite is commonly added at 3–10% DW. Its layered structure can adsorb and retain NH_4^+ , and a 5% DW addition has been reported to reduce total GHG emissions by 26.9%, with the strongest mitigation effect observed for N_2O (37.2%) [34]. Fly ash, as an industrial by-product, has the advantage of low cost and is usually applied at 5–20% DW. It can improve pile porosity and adsorb gaseous pollutants; a 10% DW addition has been reported to reduce CH_4 and N_2O emissions by 41.6% and 35.8%, respectively [54]. Overall, these materials regulate the nitrification-denitrification process mainly by adsorbing and immobilizing NH_4^+ , thereby achieving GHG mitigation.

Chemical Additives

Chemical additives include lime, phosphogypsum, calcium superphosphate, calcium-magnesium phosphate fertilizer, and other materials containing SO_4^{2-} , $\text{PO}_4^{3-}/\text{H}_2\text{PO}_4^-$, Ca^{2+} , and Mg^{2+} . These additives can reduce GHG emissions through several mechanisms. For example, phosphogypsum and superphosphate can promote the formation of phosphate-ammonium complexes, while Ca^{2+} can exchange with NH_4^+ and other cations to reduce nitrogen loss and improve the nutrient content of the final compost product. Calcium-magnesium phosphate fertilizer can also reduce NH_3 volatilization by promoting the formation of struvite through the reaction between H_2PO_4^- and NH_4^+ . In addition, sulfate ions may inhibit methanogenic activity and thus suppress CH_4 formation. For chemical additives, the recommended dosages are also generally expressed on a dry-weight basis (DW). Superphosphate is commonly applied at 2–5% DW. In the study by Zhang et al. [51], superphosphate addition during pig manure composting achieved a 35% reduction in total GHG emissions relative to the control, mainly through nitrogen fixation and reduced N_2O formation. Calcium-magnesium phosphate fertilizer is usually applied at 3–6% DW, and its slow-release characteristics allow sustained nitrogen retention throughout the composting process. Phosphogypsum is typically added at 5–10% DW; due to its high Ca^{2+} and SO_4^{2-} content, it has been reported to reduce CH_4 emissions by 38.5% and NH_3 volatilization by 29.1% on average [34]. Lime (CaO or $\text{Ca}(\text{OH})_2$) is generally used at 1–3% DW to adjust pile pH. It is most effective when the initial pH is below 6.0, because it can enhance organic matter degradation; however, excessive addition (>5% DW) may sharply increase NH_3 volatilization [34]. Dicyandiamide, a nitrification inhibitor, is commonly applied at 0.1–0.5% DW. Yang et al. [108] showed that combining dicyandiamide with hydroquinone and phosphorus fertilizer in pig manure composting reduced GHG emissions by 24.3% to 75.84 kg $\text{CO}_2\text{-eq}\cdot\text{t}^{-1}$ DM. Hydroquinone, which acts as a urease inhibitor, is usually applied at 0.05–0.2% DW. When used together with dicyandiamide at 0.1% DW, nitrogen loss was reduced by 42.7%, and total GHG reduction increased to 68.3% [108].

These additives are generally mixed uniformly into the feedstock during the pre-treatment stage to ensure sufficient contact with the materials and to avoid local over-concentration that may inhibit microbial activity.

Biological Additives

Biological additives mainly include multifunctional microbial inoculants, specialized functional strains (e.g., ammonia-oxidizing bacteria, nitrogen-fixing bacteria, and methane-oxidizing bacteria), and enzyme preparations. By establishing a more favorable microbial community or strengthening specific metabolic pathways, these additives can reduce GHG emissions during composting. Mature compost itself contains a rich microbial community and can therefore serve as an effective biological additive. In Yang et al. [109], adding 10% WW mature compost reduced total GHG emissions by 69.2%, corresponding to

158.2 kg CO₂-eq·t⁻¹ DM, mainly because mature compost improved aeration and weakened denitrification, thereby significantly reducing N₂O emissions.

In the study by Gao et al. [110], bacterial agents were applied at 0.3%, 0.6%, 0.9%, and 1.2% of wet weight to investigate the microbial enhancement mechanism during kitchen waste and garden waste composting. The results showed that GHG emissions were reduced by 11–54% depending on the dosage and composting conditions. Importantly, a higher inoculum dose did not necessarily correspond to a proportionally greater reduction in GHG emissions, indicating that the mitigation effect depends on both additive dosage and process conditions. After inoculating microorganisms, bacterial community structure, particularly Thermobifida, Bacillus, and Saccharomonospora, was enriched [110]. All of which promote organic polymerization and aromatization to form humus. However, parthenogenic anaerobic bacteria (i.e., *Weissella*) and denitrifying bacteria (i.e., *Pusillimonas*) were decreased. Therefore, organic fermentation and nitrate respiration was reduced and led to GHGs emissions reduction. Further analysis indicates that microbial inoculation could diversify and enrich the bacterial community, particularly the Bacillus, Thermobifida, and Saccharomonospora, all of which could accelerate organic aromatization and polymerization for humus formation. Furthermore, the proliferation of facultative anaerobe bacteria (i.e., *Weissella*) and denitrifying bacteria (i.e., *Pusillimonas*) was inhibited with microbial inoculation to reduce organic fermentation and nitrate respiration and resulting in a decrease in GHG emission.

3.3. Critical Synthesis of GHG Emission Characteristics

The wide variation in reported GHG emission ranges can be attributed to multiple interacting factors, including feedstock composition, process configuration, seasonal conditions, and assessment boundaries. Static composting generally exhibits higher CH₄ emissions because of the limited oxygen supply inside the pile, whereas windrow and reactor composting can suppress CH₄ formation by improving aeration, but they may increase N₂O emissions or energy-related emissions depending on the system design [37,40,41]. Feedstock type is another major driver of emission variability: pig manure usually shows a wider emission range than chicken or cattle manure because of its higher protein and fat contents, while digestate composting tends to generate even higher GHG emissions because of its excess ammonia load [27,45–49,54,55]. In addition, non-biogenic emissions caused by transportation, pretreatment, deodorization, and postprocessing are not negligible and may contribute substantially to the total GWP [55–64]. Seasonal temperature differences further amplify this variability by changing microbial activity, oxygen availability, and the balance between nitrification and denitrification [65–72]. Taken together, these findings indicate that emission performance should be interpreted in the context of both feedstock properties and process boundaries rather than as a single fixed value.

3.4. Multi-Dimensional Evaluation of Mitigation Strategies

Mitigation strategies should be evaluated not only by emission reduction efficiency, but also by their practicality under specific treatment conditions. Process regulation, including co-composting, moisture control, aeration intensity, turning frequency, and intermittent ventilation, is the most direct and widely applicable approach because it can reduce emissions without requiring additional material input [44,91–97]. Physical additives such as biochar and zeolite can effectively reduce CH₄ and N₂O emissions through adsorption and porosity improvement, while their performance depends strongly on dosage and substrate characteristics [53,54,99,103]. Chemical additives such as superphosphate, calcium-magnesium phosphate fertilizer, phosphogypsum, dicyandiamide, and hydroquinone can further suppress nitrogen losses and GHG emissions, but their mitigation effects remain

dose-dependent [50,51,100,108]. Biological additives such as mature compost and microbial inoculants may improve humification and microbial succession, but higher inoculum doses do not necessarily produce proportionally greater emission reductions [101,102,109,110]. Therefore, the most effective mitigation strategy is not universal; it should be selected according to the specific feedstock, process conditions, and operational goal.

4. Future Perspective

The future trajectory of AD and composting is unequivocally centered on achieving quantifiable and cost-effective GHG mitigation. To move beyond incremental gains, research and implementation must pivot towards integrated systems that leverage digital, economic, and policy tools to maximize GHG reduction impact. A primary focus will be the development of sophisticated digital monitoring, reporting, and verification (MRV) platforms to accurately track the carbon footprint across the entire waste-to-value chain, from feedstock logistics to final compost application. This data integrity is the critical enabler for integrating these projects into carbon markets, transforming avoided emissions into tradable assets that directly improve financial viability and incentivize further GHG reduction technological upgrades. Concurrently, policy frameworks must evolve beyond generic support to include robust carbon pricing and targeted subsidies that explicitly reward verifiable emission reductions and the production of high-quality, compost to displace carbon-intensive fertilizers. Techno-economic assessments will increasingly need to evaluate the cost-benefit trade-offs of such integrated systems—where the synergies between AD (for optimized renewable energy production) and composting (for carbon sequestration and nutrient recycling) are fully realized—not merely on energy output but on their total carbon balance. Ultimately, the convergence of precise MRV, carbon market finance, and supportive policy will be the cornerstone for advancing AD and composting from waste treatment processes into indispensable, economically sustainable pillars of a circular bio-economy dedicated to deep and accountable decarbonization. In addition to biological treatment routes, thermochemical pathways such as pyrolysis and gasification may also contribute to waste valorization, while transportation-related emissions should be considered when evaluating the overall environmental performance of AOSW management.

5. Conclusions

This study presents a systematic quantitative synthesis of GHG emissions and reduction strategies in the biological treatment of agricultural organic solid wastes (AOSW), based on 105 high-quality studies published between 2010 and 2024. The main findings of this review are as follows:

Biogenic GHG emissions dominate the total global warming potential (GWP) of both composting and anaerobic digestion (AD) systems. For composting, biogenic emissions (primarily CH₄ and N₂O) account for 65–85% of total GWP, with emission ranges of 30–740 kg CO₂-eq·t^{−1} DM. Static composting has the highest emissions due to extensive anaerobic conditions, while reactor composting with controlled aeration achieves the lowest emissions. For AD, biogenic emissions mainly stem from 1–5% CH₄ leakage (196.5–982.4 kg CO₂-eq·t^{−1} DM) and digestate composting (276.6–1002.3 kg CO₂-eq·t^{−1} DM). Non-biogenic emissions, dominated by transportation (11.27–461 kg CO₂-eq·t^{−1} DM), are also significant and cannot be ignored, especially for small-scale facilities with long transport distances.

Among GHG reduction measures, exogenous additives achieve the highest N₂O reduction efficiency (58–97.3%) by adsorbing NH₄⁺ and inhibiting denitrifying enzymes, with biochar showing the most promising results. Process optimization strategies, such as aeration rate adjustment and C/N ratio control, are more cost-effective for CH₄ control,

reducing emissions by 13.9–55.8% without additional material input. However, no single mitigation strategy is universally optimal, and their effectiveness depends strongly on local conditions, feedstock characteristics, and operational parameters. Pollution swapping is a common trade-off: for example, increased aeration reduces CH₄ emissions but may increase N₂O emissions, while chemical additives can reduce nitrogen loss but may introduce other environmental risks.

Digestate management is a critical bottleneck for improving the overall GHG performance of AD systems. Traditional digestate composting often results in higher emissions than raw manure composting due to its high ammonia nitrogen content. Advanced technologies such as covered storage, solid-liquid separation, and ammonia recovery can significantly reduce emissions while improving nutrient recovery efficiency. The integration of AD with optimized digestate management represents a promising approach for achieving both renewable energy production and low-carbon waste treatment.

Supplementary Materials: The following supporting information can be downloaded at: <https://www.mdpi.com/article/10.3390/agronomy16131260/s1>, Figure S1. The PRISMA Flow Diagram. “n” refers to the number of relevant studies; PRISMA_2020_checklist.

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Abbreviations

The following abbreviations are used in this manuscript:

AD	Anaerobic digestion
GHG	Greenhouse gas
GWP	Global warming potential
DM	Dry matter
AOSW	Agricultural organic solid wastes
OM	Organic matter
AOB	Ammonia-oxidizing bacteria
LCA	Life cycle assessment
WW	Wet weight

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