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Abstract

Wastewater treatment facilities are major systems for managing swine manure in Korea. These facilities primarily use physical, chemical, and biological processes to remove harmful substances from manure and convert them into compost, liquid fertilizer or biogas. The 2050 carbon neutrality scenario in Korea aims to increase the proportion of manure purification treatment from 13% to 25% by 2030. As manure treatment facilities expand, it is crucial to quantify and monitor their greenhouse gas (GHG) emissions such as methane (CH_4) and nitrous oxide (N_2O) gas emissions. This study aimed to measure the GHG emissions from a swine wastewater treatment plant to develop a country-specific emissions factor for each treatment stage to determine the national GHG inventory. The facility evaluated in this study had tanks for sedimentation, manure retention, denitrification, and aeration (nitrification) and treats 121 tonnes of swine manure from approximately 24,335 pigs. Quantification of the total GHG emissions from the facility was conducted for 24h once per a month, using a CH_4/N_2O Analyzer. The emission factors for CH_4 and N_2O were estimated as follows: 0.5 kg CH_4/head/year and 0.003 kg N_2O/head/year in the sedimentation tank, 0.09 kg CH_4/head/year and 0.0008 kg N_2O/head/year in the manure retention tank, 0.0008 kg CH_4/head/year and 0.0001 kg N_2O/head/year in the denitrification tank, and 0.0002 kg CH_4/head/year and 0.00009 kg N_2O/head/year in the aeration tank. Also, field measured data showed 417 tCO_2-eq/year, whereas 2019 IPCC Tier 2 factors estimated 1,238 tCO_2-eq/year- a 66% overestimate. In conclusion, it is crucial to ensure that sedimentation and manure retention tanks are gastight to reduce the GHG emissions from a facility. Likewise, direct stage-resolved monitoring is essential to prevent overestimating GHG emissions. Therefore, this study serves as a foundation for the development of effective carbon reduction strategies in manure treatment processes.

Keywords: Wastewater purification system, Greenhouse gas, Carbon neutrality, Liquid composting, Swine manure treatment

Introduction

Swine production significantly contributes to agricultural economies worldwide, yet intensive farming leads to increased manure generation, posing substantial environmental challenges, including greenhouse gas (GHG) emissions [1,2]. Globally, livestock manure contributes approximately 10% of agricultural GHG emissions, with methane (CH_4) and nitrous oxide (N_2O) being the primary concerns due to their substantial global warming potential (GWP) [3]. The global agenda of carbon neutrality by 2050 has gained significant international momentum [4]. Following the implementation of the Paris Agreement in 2016, 121 countries joined the “Climate Ambition Alliance” at the United Nation (UN) Climate Action Summit, committing to carbon neutrality by 2050 [5]. Rapid global shifts toward addressing climate crises emphasize climate-related issues as crucial for enhancing international competitiveness. In Korea, the ministry of agricultural food and rural affairs (MAFRA) announced the ‘2050 Carbon neutral strategy in agriculture and rural communities’ which included plan to reduce the GHG emissions from manure treatment by increasing the number of facilities for livestock wastewater treatment according to the UN Framework Convention on Climate Change (UNFCCC) [6]. The proportion of livestock manure that can be processed by the purification systems, is approximately 5.3 million tonnes (10%), but will be expanded to 13.6 million tonnes (25%) by 2030 to achieve 30% mitigation from livestock sector [6]. According to the Korean Statistical Information Service and Korea Rural Economic Institute, there are approximately 11 million pigs that excrete 50 million tonnes of manure in Korea [6,7]; the average daily manure production of swine is approximately 2.63 kg/head/day (feces: 0.89 kg, urine: 1.74 kg), although this value is different at the various growth stages of the pigs [8].

There are various swine manure management technologies, including the anaerobic-anoxic-oxic (A2O), Bio Best Bacillus (B3) system, Bio-ceramic Sequencing Batch Reactor (BCS), Biosynthesis of sulfur-containing compounds (BioSUF), the Korea Institute of Science and Technology Hyundai treatment system (KHTS), Sequencing Batch Reactor (SBR), liquid corrosion (liquid-composting or oxidation ditch method), and anaerobic digestion. Among these, the liquid-composting system is the most popular owing to its operational simplicity, cost-effectiveness, and potential for reducing organic pollutants [9-11]. The liquid-composting system shares similarities with established wastewater treatment technologies, particularly A2O and Bardenpho, by recycling the activated sludge as mixed-liquid suspended solids. These technologies involve recycling wastewater internally, effectively increasing hydraulic retention time (HRT), facilitating enhanced biological nutrient removal (BNR) [10,11]. The liquid-composting method involves treating swine manure after solid-liquid separation by continuously circulating the liquid fraction while maintaining continuous exposure to air (oxygen) [11]. In comparison to other biological treatments, such as A2O and Bardenpho, liquid composting is relatively simpler and focuses on organic pollutant oxidation through circulating aerobic treatment to anoxic stages aimed at comprehensive nutrient removal [11,12,14]. As manure treatment facilities expand, accurately quantifying and monitoring GHG emissions becomes critical to meet national inventory reporting requirements and international commitments [15].

Accurately quantifying GHG emissions from the swine manure wastewater treatment facilities has become a trending topic because of the substantial variability in emissions depending on the treatment technologies, environmental conditions, and operational practices [16-18]. Previous studies on GHG emissions from swine manure management have largely relied on default emission factors (EFs) provided by the Intergovernmental Panel on Climate Change (IPCC) [2]. However, these default EFs have limitations because they do not sufficiently reflect local operational conditions and specific technological variations [2,15]. Although numerous studies have examined GHG emissions from swine manure management, most of them primarily focused on overall emissions and reductions associated with entire treatment systems, and thus, limited attention has been paid to the quantification of GHG emissions specifically at individual stages within manure treatment facilities [18,19]. Moreover, there is a significant gap in development of precise, stage-specific EFs derived from direct field measurements.

The objective of this study was to measure GHG emissions of swine manure from a public liquid-composting wastewater treatment facility in Korea, with a focus on emissions at different treatment stages such as sedimentation, manure retention, denitrification, and aeration. This study aimed to establish country-specific EFs for these tanks and, improve Korea’s national GHG inventory accuracy.

Materials and Methods

Site descriptions and manure treatment process

This study was conducted at a public swine manure treatment facility in Jinan, Korea, from October 2023 (day of the year [DOY] 302) to October 2024 (DOY 297). This facility can treat approximately 115 tonnes per day of manure per day from approximately 24,335 pigs. Swine manure entering the facility initially underwent pretreatment, including solid-liquid separation and centrifugation. Subsequently, the treated manure moves to the primary treatment phase of the liquid composting method, in which the denitrification and nitrification processes occur continuously (Fig. 1). There were post-treatment stages for advanced treatment after biological treatment; however, this study focused on the primary biological treatment stages and relevant pretreatment systems to quantify the emissions of CH_4 and N_2O.

The pretreatment process removes some substances from manure, including seeds, feed residues, straw, plastics, swine hair and other debris from the manure. This stage also regulates the flow rates and ensures homogeneity and sufficient retention time before the biological treatment stages. The sedimentation tank, which receives manure after debris removal, had a total volume of approximately 108 m^3 as a liquid system without a crust cover, according to 2019 IPCC Refinement Guideline (2019-R). It can process approximately 100 m^3 of manure per day and maintain an average HRT of approximately 1.8 days. After sedimentation, the manure was transferred to a 648 m^3 retention tank, which was sufficient for retaining an average influent HRT of approximately 10.6 days. The retention tank included an aeration-type mixing system designed to ensure a homogenous manure composition, prevent solid deposition, and minimize scum formation.

The biological treatment consists of liquid composting tanks, including two denitrification tanks and three nitrification tanks. After ensuring adequate homogenization, the manure from the retention tank was transferred to the liquid composting tanks via a pump system every 2 h, running for 18 min per cycle. In the denitrification tanks (working volume of 980 m^3), two submersible mixers operated in two different cycles: one mixer operated for 260-320 min off, and the other operated for 210 min and was the switched off for 270 min. Aeration within the nitrification tanks (combined working volume of 2,024 m^3) was supplied by five blowers delivering a total air flow of 100 m^3/min, with the blower output modulated in real time by dissolved oxygen (DO) feedback control. The total HRT of liquid composting system was maintained at 47.8 days. The pH, water temperature, and mixed-liquor suspended solids (MLSS) were 8.2, 38-39 ℃, and 14,090 mg/L, respectively. The detailed characteristics of the swine wastewater are presented in Table 1.

Emission rates quantification

Emission rate measurements were conducted once a month for 24h. The measurement points were located in the air inlet (C_{in}) and air outlet (C_{out}) for four processes: sedimentation, retention, denitrification and nitrification. The measurement points were connected to a CH_4 and N_2O analyzer (Los Gatos Research, San Jose, CA, USA) and measured at 10 min each point. Prior to measurement, the analyzer was calibrated with standard CH_4 and N_2O gases at 1.01, 101, 859, and 3,000 ppm to ensure data accuracy. The flow rates of the air inlet and outlet followed the machine suction and were set at 0.8-1 L/min. The GHG measurement points are based on the manure substrate pathway. All treatment tanks (sedimentation, retention, denitrification, and nitrification) were equipped with identical rectangular frames (1.2 m x 1.2 m internal areas). Each frame was covered with a waterproof cover. A closed frame ensured an airtight seal against the tank curb, whereas the two sides of the frames allowed operation in an open dynamic chamber (steady-state chamber) to allow the gas to flow continuously. This chamber supported continuous real-time gas analysis, and was minimally affected by mixing, aeration, or manure movement, thereby yielding stage-specific emission rates with high analytical precision [20]. For the accurate measurement of GHG emissions, the ventilation rates were matched to the expected emission strength of each unit. First, because the sedimentation tank emitted the highest concentration of CH_4, Sirocco blower that emits approximately 4,000 m^3/h, was mounted. The same fan style, throttled to 1,000 m^3/h, served in the retention tank. The denitrification and nitrification tanks, which emit far lower concentrations, were fitted with compact inline duct fans operating at approximately 700 m^3/h, with fresh air entering the flexible duct hose from the inlet. All ducts were tightly by clamped, and the actual volumetric flow of each fan was verified at the start of each sampling day using with a calibrated anemometer and velometer (KIMO CTV 210-R, Mumbai, India and TSI ALNOR EBT, FLW, Inc., CA, USA). The CH_4 and N_2O emission rates were calculated using by equations (i) and (ii), respectively.

$$\Delta c = \frac{(C_{out} - C_{in}) \cdot P \cdot M}{T \cdot R} \ldots \text{(i)}$$

$$\text{Emission rate} = FR \cdot \Delta c \ldots \text{(ii)}$$

Where Δc is the concentration difference of GHG, C_{out} is concentration of GHG from the outlet (ppm) as field measurement, C_{in} is concentration of GHG from the inlet (ppm), P is the atmospheric pressure (Pa) during the experimental periods, M is molecular weight of CH_4 (16 g/mol) and N_2O (44 g/mol), T is temperature (K) (approximately 35-38 ℃) from the wastewater, R is the gas constant ($8.314 \cdot 10^3$ Pa m^3/kmol/K), FR is flow rate (m^3/s) from anemometer and velometer, and A is the unit area [21]. The analysis covered the observed emission patterns every 6 h, the predominant biochemical pathways responsible, and the key influencing factors. The reason for dividing the emission data into 6 h intervals to precisely capture and illustrate how emissions varied throughout the day in relation to specific operational activities and substrate inputs at the plant, such as morning inflow, afternoon processing, and evening stabilization.

Emission calculations using the 2019 Refinement IPCC GL Tier 2 approach

CH_4 and N_2O emissions were estimated using the Tier 2 method outlined in the 2019-R, which includes a manure management system [22]. Country-specific data were derived from the chemical analyses of swine manure, focusing on annual volatile solid (VS) levels and nitrogen excretion (N_{ex}) obtained from total nitrogen. The typical animal mass (TAM) for finishing swine was 116 kg/head of live weight in 2023 [23]. The amount of manure from a market swine is 4.73 kg/head, amounting to approximately 24,335 heads and 115 tonnes of manure per day [24]. To estimate the annual GHG emissions in accordance with 2019-R, three equations were used and the resulting estimates were compared with field-measured data.

$$\mathrm{EF} = (\mathrm{VS} \bullet 365) \bullet \left[\mathrm{Bo} \bullet \frac{0.67\mathrm{kg}}{\mathrm{m}^3} \bullet \sum \frac{\mathrm{MCF}}{100} \bullet AWMS\right] \ldots \text{(iii)}$$

$$CH_4 = \mathrm{EF} \bullet \mathrm{N} \ldots \text{(iv)}$$

$$N_2O = [\sum[\sum(\mathrm{N} \bullet \mathrm{Nex} \bullet \mathrm{AWMS}) + \mathrm{N}_{cdgs}] \bullet \mathrm{EF}] \bullet \frac{44}{28} \ldots \text{(v)}$$

The estimate of the annual CH_4 EF (kg CH_4/head/year) is given by Equation (iii), and the volatile solid (kg VS/head/day) are obtained from field measured data. B_0 is 0.45 m^3/kg VS as the maximum methane producing capacity as per the IPCC GL default value for finishing swine in Western Europe. Methane conversion from m^3 to kg is 0.67 kg/m^3 as an IPCC GL default value. The value of the methane conversion factor (MCF) for liquid/slurry within one month in a warm temperate moist climate was 13%, and that for the aeration treatment 0%, which was from the IPCC GL default values. As this study focuses only on finishing swine waste treatment as a single system, the value of the Animal waste management system (AWMS) was 100% in Equations (iii) and (v). The total annual CH_4 and N_2O emission estimate based on the Tier 2 approach were multiplied by the number of animals (head) to obtain EFs using Equations (iv) and (v). The EF of the direct N_2O emissions from the aerated portion of the manure treatment chain in a forced aeration system (the nitrification tanks) was 0.005 kg N_2O-N/kg N_{ex}. The IPCC default values for the annual average N excretion per head of swine (kg N/head/year) were not used, but were determined analytically from manure characteristic data of the facility, which is presented in Table 2. N_{cdgs} represents any additional nitrogen entering the system as a chemical additives or co-digested substrates, which is negligible in this study and was therefore set to zero [22].

To obtain facility-specific inputs for the inventory equations, the VS and annual N_{ex} were calculated by multiplying the daily manure mass entering each unit by the analytically determined VS and total nitrogen concentrations, as reported in Table 2. For a direct comparison with the measured emission rates, the inventory-based total CH_4 and N_2O emissions were converted to CO_2 equivalents (CO_2-eq) units using the 100-year global warming potentials recommended in the IPCC 6th Assessment Report (AR6) 27 for CH_4 and 273 for N_2O [25]. The resulting CO_2-eq values were applied to the stage-specific open-chamber data to assess the accuracy of the inventory approach for a swine manure wastewater treatment plant.

Results and discussion

Greenhouse gas emission rates from the four treatment stages

The CH_4 and N_2O emission rates were measured at four distinct stages in the swine wastewater treatment plant (SWWTP): sedimentation, retention, anoxic denitrification, and aeration nitrification. Sedimentation tanks showed the highest emissions of CH_4 and N_2O among all stages, thus, emission rates for both gases notably increased in late spring and summer, indicating strong seasonal variations possibly due to favorable temperature conditions [26]. With average CH_4 emission rates approximately 412 ± 336 mg/s throughout the study period. In Figure 2, this trend is characterized by significant fluctuations, as indicated by the magnitude of the error bars, suggesting considerable temporal variability in the emission rates. Notable peaks in CH_4 occurred on DOY 212.8 and 226.6 in the 2024 monitoring period. The observed increase in CH_4 emission rate, particularly the higher peaks in 2024, could be attributed to factors such as progressively warmer ambient temperatures influencing the manure temperature over the monitoring period, or an accumulation of more readily degradable sludge at the bottom of the tank over time [26,27]. Both warmer conditions and increased substrate availability can enhance methanogenic activity [28]. The initial lower CH_4 observed on DOY 302.8 to 303.5 in 2023 may also represent a lag phase or an adaptation period for the methanogenic microbial community during winter [18,27]. The N_2O emission rates were substantially lower than the CH_4 emission rates and, generally remained below 5 mg/s. However, distinct peaks were evident, reaching 8–15 mg/s. These peaks were not consistent with the CH_4 peaks, which suggests that different mechanisms or specific transient conditions are responsible for N_2O production [29]. The retention tank (B), serving as an equalization and balancing tank, was positioned after the initial sedimentation tank. Its primary role is to buffer variations in the flow and load to downstream biological treatment units, such as denitrification and nitrification. The GHG emission trends depend on the operational mode such as mixed, quiescent, or HRT. They exhibited moderate but highly variable GHG emissions. CH_4 emission rates generally remained below 200 mg/s during the cooler months, yet rose steadily after DOY 170 as wastewater temperatures exceeded 38-40 °C. This suggests that a significant portion of the degradable organic matter may have been converted in the upstream sedimentation tank and that the conditions in the retention tank were less conducive to methanogenesis [30]. This may also contribute to greater stabilization [31]. During mid-summer the retention tank released its highest methane pulse of 383 mg/s, and an accompanying N_2O burst of 10 mg/s. When a concentrated load of fresh slurry is received by a plant, shock-loading events of this type have been shown to elevate liquid phase volatile fatty acids [32]. In addition, the agitation of substrates occasionally leads to spikes owing to trapped CH_4 release [33]. N_2O emissions remained lower than 5 mg/s throughout the study period, suggesting limited nitrification-denitrification activity due to relatively stable aerobic and anaerobic conditions. Redox oscillations can trigger coupled nitrification, denitrification, and heterotrophic denitrification, producing an N_2O spike on DOY 270 in 2024 [34]. The anoxic denitrification tank (C) showed low GHG emissions (near zero), throughout the entire monitoring period. A single minor isolated peak of 3 mg/s was observed around DOY 137.5 in 2023. This is anticipated because anoxic conditions inhibit methanogenic archaea owing to the presence of nitrate/nitrite as electron acceptors and oxygen [35]. The isolated peak might represent an anomaly of the residual dissolved CH_4 carried from the upstream units. However, the brief emission spikes made a negligible contribution to annual emissions [36]. In addition, N_2O emission rates were consistently near zero during the monitoring period. This is a significant finding, as anoxic denitrification steps are often considered potential hotspots for N_2O emissions if denitrification is incomplete [33,37]. The observed minimal N_2O suggests highly efficient complete denitrification. This could be due to an optimal C/N ratio providing sufficient electron donors for the complete reduction of N_2O to N_2, or maintained anoxic conditions below 0.1-0.5 mg/L DO [34,35,38]. Similar to the anoxic tank, the CH_4 and N_2O emissions from the aeration nitrification (D) tank were negligible (near zero). CH_4 and N_2O emission rates were 0.2 ± 1.03 mg/s and 0.09 ± 0.17 mg/s, respectively. The aerobic conditions in this tank can promote CH_4 oxidation by methanotrophs and inhibit methanogenesis [32,39]. Nitrification processes can be significant sources of N_2O through pathways such as nitrifier-denitrification by ammonia-oxidizing bacteria (AOB) [17,40]. Furthermore, according to the plant analysis report, the DO in the nitrification tank ranged from 0.3-3.8 mg/L. Therefore, the conditions mentioned above can produce stable and optimal DO levels, and efficient nitrification with minimal accumulation of intermediate nitrite, indicating a well-balanced activity of AOB and nitrite-oxidizing bacteria (NOB), or low influent ammonia loading at this stage [40,41].

The data clearly demonstrated a significant reduction in CH_4 emissions as manure progressed through the SWWTP. The sedimentation tank was the primary CH_4 source. The emissions were substantially lower in the retention tank, and negligible in the anoxic and aeration tanks. This progressive reduction is consistent with the transition from anaerobic conditions favoring methanogenesis, and aerobic conditions preventing methanogenesis and promoting CH_4 oxidation [42]. In the case of N_2O, while the sedimentation tank showed some irregular spikes, potentially linked to influent events and temperature, the dedicated nitrogen removal stage, which is often cited as a potential N_2O hotspots in wastewater treatment, exhibited minimal emissions in this system during the monitoring period. This suggests that the operational conditions may be conducive to minimizing N_2O formation. In addition, this suggests well-managed nitrification, avoiding common triggers for N_2O production, such as low DO or high nitrite levels [35,43]. The peaks observed in the sedimentation and retention tanks support the idea that influent loading events for working hours and associated daily temperature increases are significant drivers of emission variability in the initial, less controlled stages of the treatment process [28,29].

Seasonal patterns differed among the tanks and gases (Fig. 3). According to Korean Meteorological Administration, meteorological spring, summer, autumn, and winter are defined as March to May, June to August, September to November, and December to February, respectively [44]. Unfortunately, emission rate monitoring was suspended during winter because the facility was placed under access restrictions following an African swine fever (ASF) outbreak, so no data are available for that season. In the sedimentation tank (a-1 and a-2), the CH_4 emission rates in summer, with a median value of approximately 724 mg/s, exceeded those in autumn (median 80 mg/s), spring (median 109 mg/s). Notably, data variability, as indicated by the interquartile range (IQR), was significantly higher in summer and autumn than in the other seasons. This confirmed that elevated temperature and organic loading enhanced both methanogenesis and nitrifier-denitrification during the primary settling stage [27,37]. Unlike CH_4, N_2O emission rates did not exhibit a clear seasonal trend and were emitted at low levels throughout the study period. The retention tanks (b-1 and b-2) showed a median of 107 mg/s in summer, which is double that observed in autumn. Seasonal N_2O differences were approximately 0 mg/s year-round, with occasional outliers in summer and autumn. In the anoxic denitrification tanks (c-1 and c-2), seasonal variability was minimal as CH_4 and N_2O remained below 1 mg/s year-round. The efficient maintenance of reducing conditions suppressed methanogens and supported the conversion of nitrate to N_2, irrespective of temperature. The aeration nitrification tanks (d-1 and d-2) recorded near 0.1 mg/s in CH_4 and N_2O across all seasons, reflecting strict aerobic inhibition of methanogenesis and limiting nitrification and denitrification even in summer.

Comparison emissions between field measured and calculated using 2019 Refinement IPCC Guideline

Using the integrated CH_4 and N_2O emission rates and an average herd size of 24,335 that continuously supplied the plant, annual gas releases were converted to EFs. Direct field measurements from sedimentation, retention, anoxic denitrification, and aeration nitrification tanks were used to calculate the total annual GHG emissions. Table 3 summarizes the resulting stage-specific EFs based on the field measurements. These EFs demonstrated a substantial reduction in CH_4 and N_2O of approximately 99% and 96%, respectively, from the sedimentation to aeration tank, as manure progressed through the multi-stage treatment system. Following the EFs in 2019-R for liquid/slurry without natural crust cover, representing the sedimentation tank, by Tier 2, CH_4 was 1.34 kg CH_4/head/year. The plant specific EF value observed in this study was approximately 63% lower than that of the Tier 2 value [22]. The 2019-R calculation yielded approximately 3 times higher annual emission than the field measurements. This discrepancy suggests that country-specific conditions, such as actual VS loading and, temperature profiles may differ from the default parameter or generic Tier 2 calculations applied [22,45]. VanderZaag et al. and Petersen et al. mentioned that IPCC values may overestimate methane emissions for certain liquid manure management systems [47,48]. IPCC default values have also been reported to exceed field measured emissions, particularly in systems with shorter storage durations or periodic aeration [45]. Regarding N_2O emissions, the 2019 R specified a default direct EF_3 of 0 kgN_2O-N/kg N in manure management system for “Liquid systems” [23]. This resulted in a calculated IPCC emission of 0 kg N_2O/year for this stage. However, the field measurements recorded 72 kg N_2O/year. This finding emphasizes that even in predominantly anaerobic storage, N_2O is likely generated at oxic-anoxic interfaces [48]. The IPCC methodology acknowledges that direct N_2O emissions from anaerobic systems are generally low but does not capture these site-specific, intermittent emissions. For the retention and anoxic denitrification tank, 2019-R did not provide differentiated default values or definitions of manure management systems, leading to uncertainty in the classification. These units are specifically engineered components of a multi-stage treatment rather than being standalone. If the retention tank is considered a continuation of anaerobic liquid storage, its CH_4 emissions would be calculated similarly to those of the sedimentation tank, but likely with a different MCF owing to pre-stabilization. Additionally, an anoxic denitrification tank was designed for active biological nitrogen removal [11]. Although it operates under anoxic conditions, it is difficult to store. The 2019-R does not provide specific definitions or default values for the CH_4 and N_2O EFs. The estimation of such units typically requires Tier 2 country-specific EFs based on measurement data or Tier 3 for modeling approaches [49]. Therefore, field measurements provided crucial site-specific emission data for the intermediate tanks: 2,270 kg CH_4/year, 20 kg N_2O/year, 21 kg CH_4/year, and 2.4 kg N_2O/year. For the aeration nitrification tank, an aerobic treatment with a forced aeration system, the 2019-R showed a value of 0.005 kg N_2O-N/kg N, which produces an estimated 1,311 kg N_2O/year. In contrast, field measured EF was substantially lower at approximately 0.00009 kg N_2O/head/year, resulting in approximately 2.3 kg N_2O/year. This is a highly significant difference, with the IPCC values being over 570 times higher. This large difference suggests that N_2O emissions can be influenced by actual tank operational conditions, including actual N load treatment, DO control, C/N ratio of the influent, temperature, pH, and the balance and efficiency of AOB and NOB in minimizing nitrite accumulation and N_2O byproduct pathways [39,40]. CH_4 emissions from aerobic stage were minimal, which is consistent with the IPCC assumption of negligible CH_4 emissions from well-aerated systems [22].

When measured CH_4 and N_2O emissions were converted into total CO_2-equivalents, significant differences emerged between the IPCC-based calculations and field measurements. Using the 2019 R, the total annual GHG emissions amounted to approximately 1,238,055 kg CO_2-eq/year, whereas field measurements resulted in a 66% lower value of 417,308 kg CO_2-eq/year. This variance was driven by the lower field measurement from CH_4 of the sedimentation tank and N_2O of the aeration nitrification tank. The discrepancies identified between the IPCC estimates and actual measurements indicate that the default EFs provided by the IPCC may not accurately reflect certain operational practices and local conditions.

Conclusion

This study provides a comprehensive analysis of the CH_4 and N_2O emissions from public swine manure treatment system that uses liquid composting. Based on the analysis of the GHG emission rate data from the four-stage SWWTP, the overall multi-stage treatment system appears to be effective in reducing CH_4 emissions. N_2O emissions from the dedicated nitrogen removal stage were unexpectedly low during the monitoring period. It is crucial to consistently monitor key parameters in each tank, including VS, temperature (from 35-40 ℃), pH, ORP, DO, nitrogen species (ammonia, nitrites, and nitrates), BOD, and COD. To minimize the GHG emissions from each stage, the organic loading rate was to balance the treatment capacity with CH_4 production. In addition, solid-liquid separation from pre-storage should be considered to reduce the organic load entering the liquid storage. In addition, adequate HRT for complete denitrification needs to ensure to prevent nitrite accumulation, and maintain stable DO levels in the range of 0.3-4 mg/L in the aeration nitrification tank. Discrepancies between field measurements and IPCC-estimated total GHG emissions emphasize the necessity for developing and adopting refined country and technologically specific EFs. Consequently, enhanced monitoring protocols, including continuous emissions measurements, are necessary to improve the accuracy and reliability of national GHG accounting practices. This approach not only supports more precise emission reporting but also aids policymakers and stakeholders in implementing targeted mitigation strategies, and advances efforts for achieving international carbon neutrality goals.

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Table 1. Typical characteristics of influent/effluent in the swine wastewater treatment plant

Parameter		**BOD (mg/L)**	**COD (mg/L)**	**SS (mg/L)**	**T-N (mg/L)**	**T-P (mg/L)**	***E. coli* (MPN/mL)**
Water quality	Influent	21,600	8,954.6	11,425.0	3,838.3	262.5	-
		(20,100~23,100)	(8,495~9,414)	(9,800~13,050)	3,647~4,030)	(197~328)	
	Effluent	2.3	28.9	1.5	25.503	0.113	< 30
		(1.1~3.5)	(24.8~32.9)	(1.2~1.8)	(24.2~26.8)	(0.03~0.2)	

Parameters include biochemical oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SS), total nitrogen (T-N), total phosphorus (T-P), and *Escherichia coli.* concentration.

Table 2. Average volatile solids (VS) and total nitrogen (T-N) concentrations in each treatment unit (mean ± Standard deviation) from Oct 2023 to Oct 2024.

Treatment unit	Volatile solid (mg/L)	Total nitrogen (T-N) (mg/L)
Sedimentation tank	19,792 ± 8,746	3,975 ± 857
Retention tank	11,712 ± 2,962	3,428 ± 420
Denitrification tank	10,597 ± 2,246	295 ± 173
Nitrification tank	17,565 ± 8,403	324 ± 218

Table 3. CH_4 and N_2O emission factors (kg/head/year) from different stage of the swine wastewater treatment plant.

Greenhouse gas	Sedimentation tank	Retention tank	Anoxic denitrification tank	Aeration nitrification tank
CH_4 (kg/head/year)	0.5	0.09	0.0008	0.0002
N_2O (kg/head/year)	0.003	0.0008	0.0001	0.00009

Table 4. The comparison of CH_4 and N_2O emissions from the plant between the calculation of the 2019 Refinement IPCC GL and field measurement.

Treatment stage / Category	Parameter	2019 IPCC GL	Field measurement
1. Liquid/Slurry (Without crust cover) (Sedimentation tank)	CH_4 (kg/year)	32,591	12,171
	N_2O (kg/year)	-	72
2. Retention tank	CH_4 (kg/year)	-	2,270
	N_2O (kg/year)	-	20
3. Anoxic denitrification tank	CH_4 (kg/year)	-	21
	N_2O (kg/year)	-	2.4
4. Aeration nitrification tank (Forced aeration system)	CH_4 (kg/year)	0	6
	N_2O (kg/year)	1,311	2.3
Total emissions	CH_4 (kg/year)	32,591	14,468
	kg CH_4/kg VS	0.04	0.02
	CH_4 (kg/head/year)	1.34	0.6
	N_2O (kg/year)	1,311	97.7
	kg N_2O/kg N	0.008	0.0009
	N_2O (kg/head/year)	0.05	0.004
Total emission (kg CO_2-eq)		1,238,055[a]	417,308[b]

[a] : Total emissions for kg CO_2-eq has to sum the 2019 R CH_4 and N_2O
[b] : Total emissions for kg CO_2-eq has to sum the field measurement CH_4 and N_2O
Global Warming Potential (GWP) for CH_4 and N_2O are 27 and 273, respectively, in the 100-year time horizon of the CO_2 equivalent followed by AR6 [24].

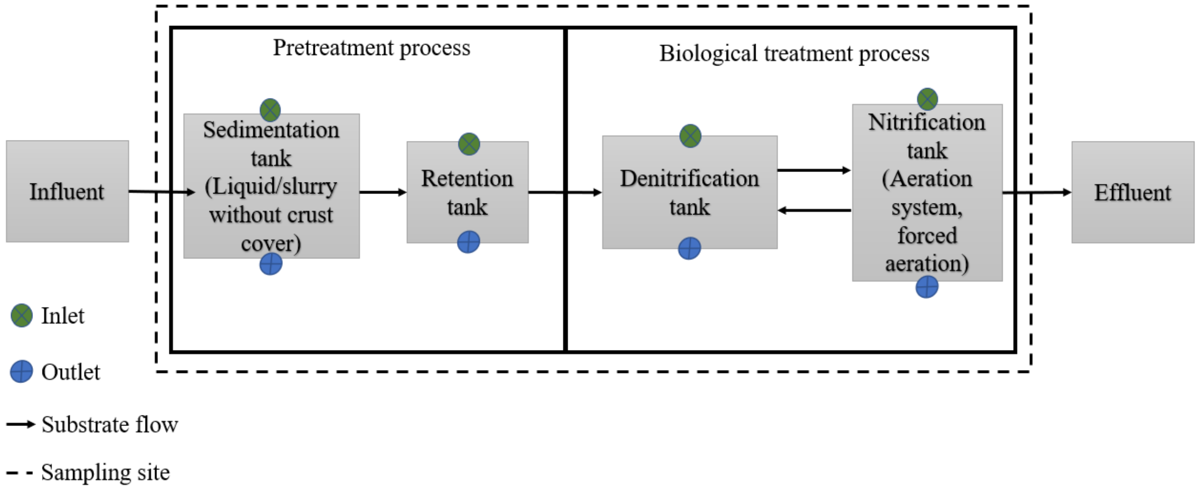
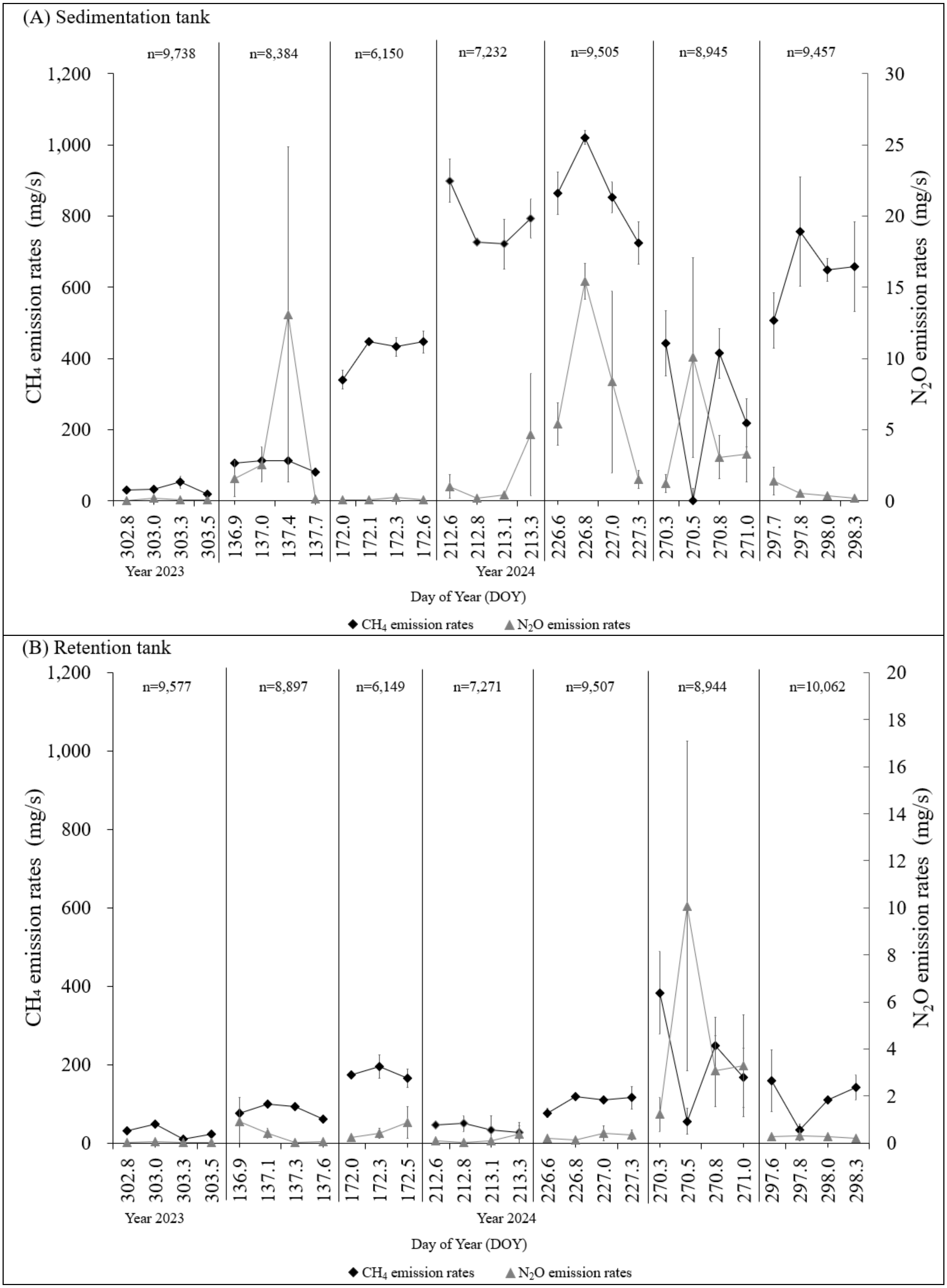


Fig. 1. Schematic of wastewater treatment system and gas sampling location.



(A) Sedimentation tank
n=9,738
n=8,384
n=6,150
n=7,232
n=9,505
n=8,945
n=9,457
CH4 emission rates (mg/s)
N2O emission rates (mg/s)
Year 2023
Year 2024
Day of Year (DOY)
CH4 emission rates
N2O emission rates
(B) Retention tank
n=9,577
n=8,897
n=6,149
n=7,271
n=9,507
n=8,944
n=10,062
CH4 emission rates (mg/s)
N2O emission rates (mg/s)
Year 2023
Year 2024
Day of Year (DOY)
CH4 emission rates
N2O emission rates

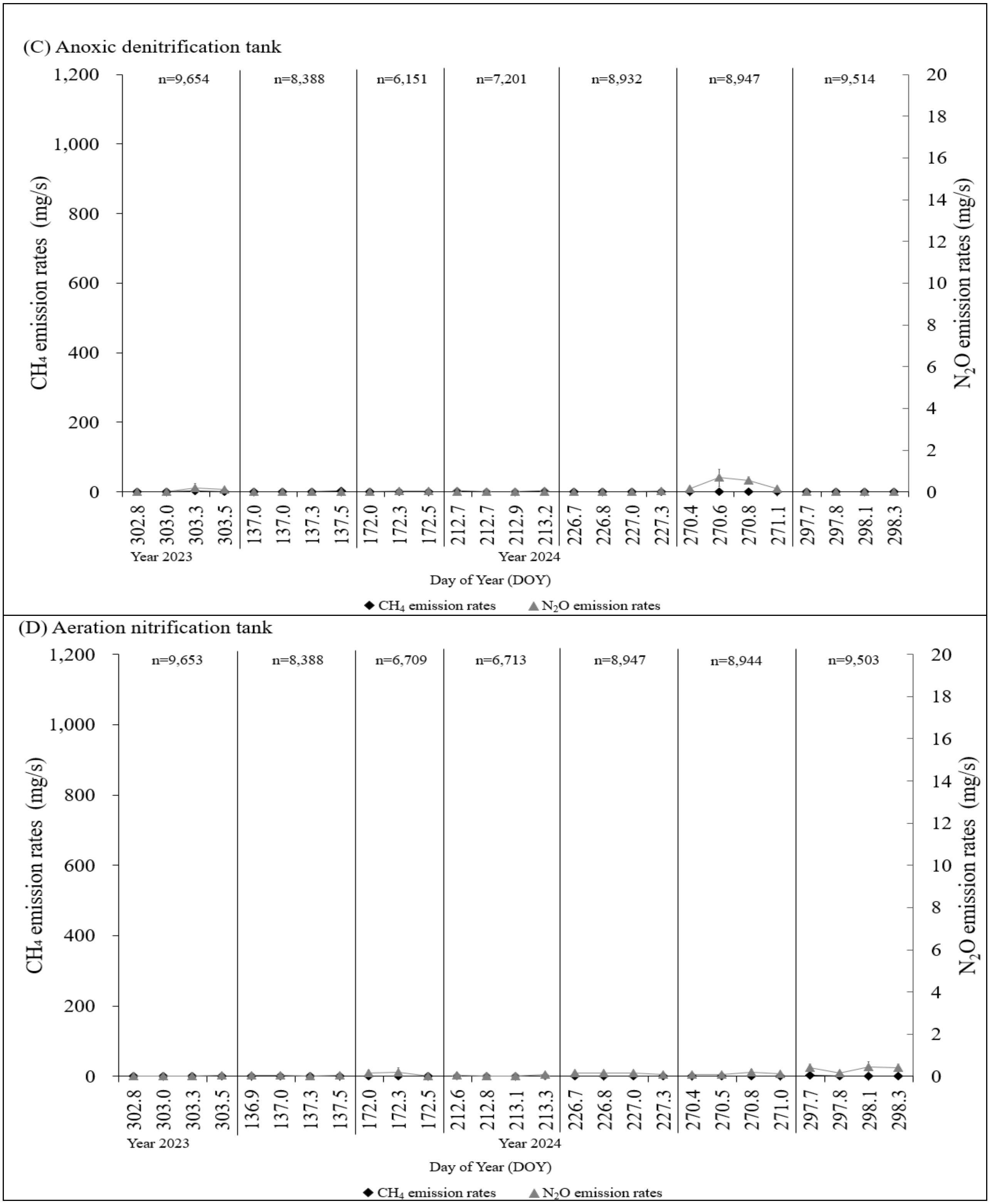


Fig. 2. Emission rates of CH_4 (♦) and N_2O (▲) at four process units of the swine wastewater treatment plant; means of GHG emission rates at 6h intervals, measured with open-dynamic chambers from October 2023 (DOY 302) to October 2024 (DOY 298). (A) Sedimentation, (B) retention, (C) anoxic denitrification, and (D) aeration nitrification tanks. Y-axis (left) shows CH_4 emission rate (mg/s) and y-axis (right) shows N_2O emission rate (mg/s). Error bars represent the standard error of mean (SEM). The secondary x-axis marked year boundaries. In each segment of figure, the value shown as (n) indicates the total number of emission rate measurements acquired during that 24 h period.

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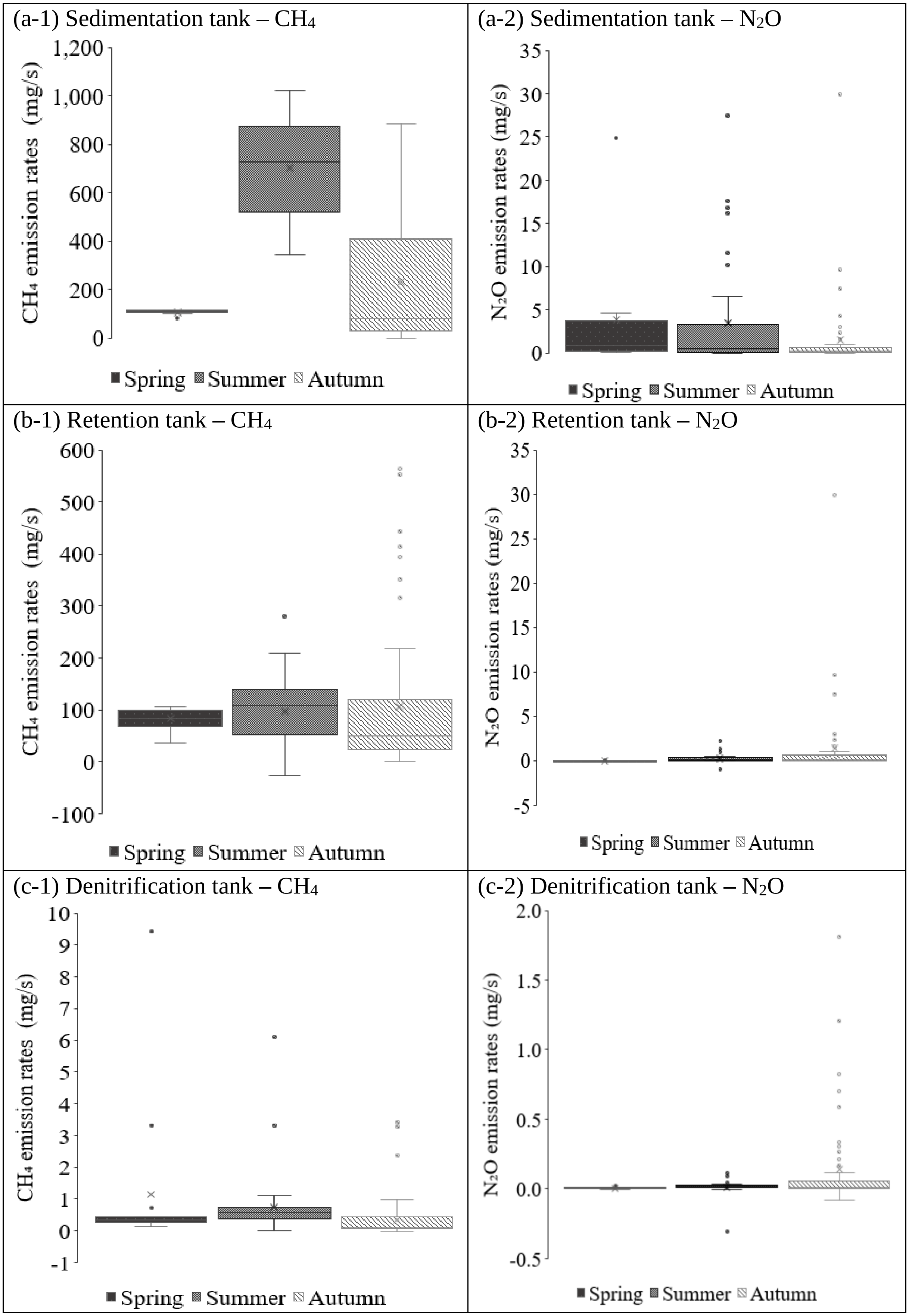

(a-1) Sedimentation tank – CH_4
CH_4 emission rates (mg/s)
1,200
1,000
800
600
400
200
0
Spring
Summer
Autumn
(a-2) Sedimentation tank – N_2O
N_2O emission rates (mg/s)
35
30
25
20
15
10
5
0
Spring
Summer
Autumn
(b-1) Retention tank – CH_4
CH_4 emission rates (mg/s)
600
500
400
300
200
100
0
-100
Spring
Summer
Autumn
(b-2) Retention tank – N_2O
N_2O emission rates (mg/s)
35
30
25
20
15
10
5
0
-5
Spring
Summer
Autumn
(c-1) Denitrification tank – CH_4
CH_4 emission rates (mg/s)
10
9
8
7
6
5
4
3
2
1
0
-1
Spring
Summer
Autumn
(c-2) Denitrification tank – N_2O
N_2O emission rates (mg/s)
2.0
1.5
1.0
0.5
0.0
-0.5
Spring
Summer
Autumn

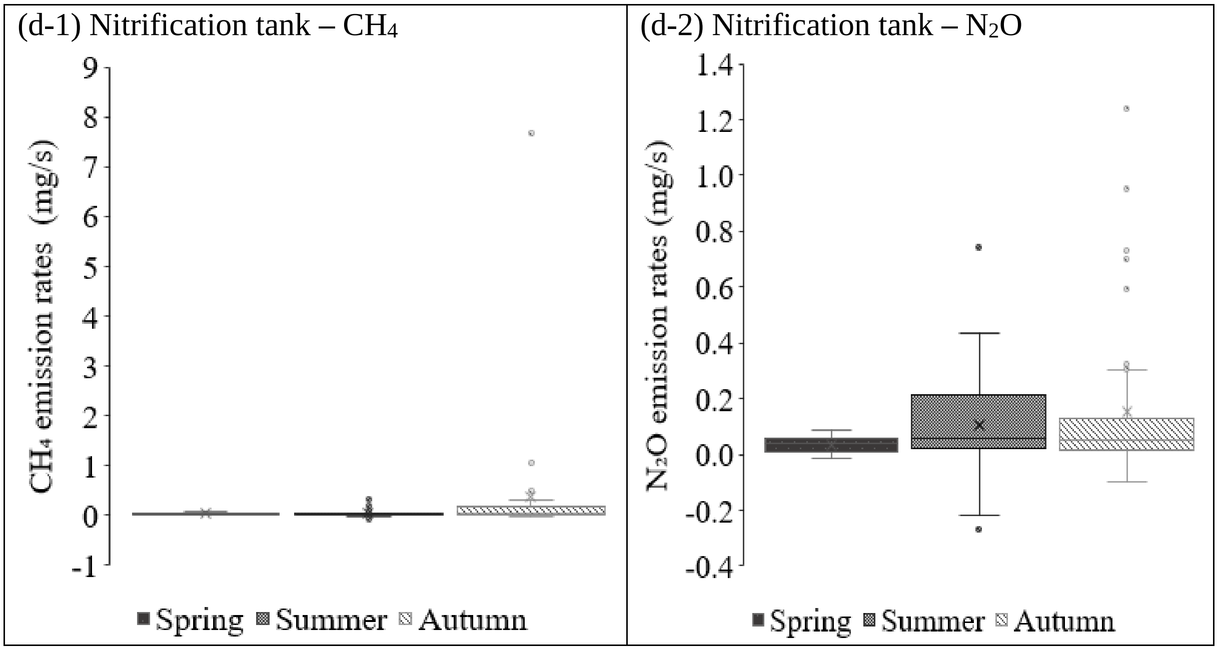

Fig. 3. Seasonal distribution of CH_4 and N_2O emission rates at each process stage of the swine-wastewater treatment plant; Box and whisker plots showing CH_4 (left) and N_2O (right) emission rates in the sedimentation, retention, anoxic denitrification, and aeration nitrification tanks. Boxes show the inter-quartile range and median, and the whiskers show the 10-90 percentiles; crosses denote the seasonal mean.