Methane emissions from onsite sanitation containment units in Indonesia: an empirical study at the household level

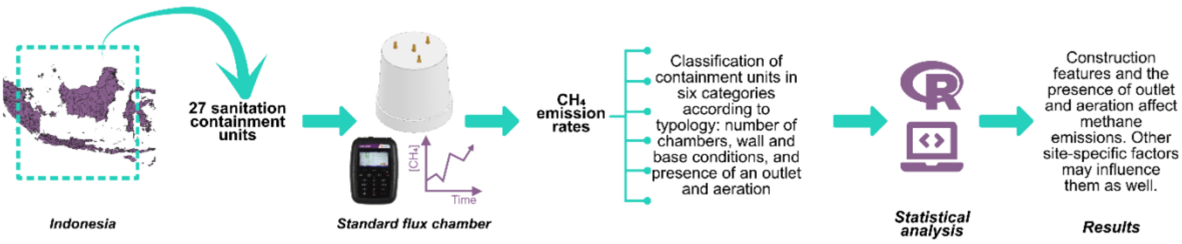
Carolina Montoya Pachongo[1,2*] & Iftita Rahmatika[3], Akna Mumtaz Ilmi[3,4], Barbara Evans[1], Ben Smeaton-Russell[1,2], Cindy Rianti Priadi[3,4,5], Dinda Fauzani[3,4], Farah Raihanah[3], Isravani Valencia[3], Jack Dalton[1,2], Jeremy Kohlitz[5], Juliet Willetts[5], Nopa Dwi Maulidiany[3], Prayatni Soewondo[6], Miller Alonso Camargo-Valero[2,7**]

[1] WASH Research Group, School of Civil Engineering, University of Leeds. Leeds LS2 9JT, UK
[2] BioResource Systems Research Group, School of Civil Engineering, University of Leeds, Leeds LS2 9JT, UK
[3] Environmental Engineering Study Program, Department of Civil and Environmental Engineering, Faculty of Engineering, Universitas Indonesia, Depok, Jawa Barat, 16424, Indonesia
[4] Collective for Climate, Environment, and Sustainability, Depok, Jawa Barat, Indonesia.
[5] Institute for Sustainable Futures, University of Technology Sydney, 15 Broadway Ultimo, NSW, 2007, Australia
[6] Environmental Engineering, Faculty of Civil and Environmental Engineering, Institut Teknologi Bandung, Jawa Barat, 40132, Indonesia
[7] Environmental Engineering Study Program, Department of Civil Engineering and Planning, Faculty of Engineering, Universitas Negeri Malang, Malang 65145, Indonesia

*Current affiliation: School of Biology, Faculty of Biological Sciences, University of Leeds, Leeds LS2 9JT, UK

**Corresponding author: M. A. Camargo-Valero (M.A.Camargo-Valero@leeds.ac.uk)

Graphic Abstract



Abstract

Sanitation contributes to greenhouse gas (GHG) emissions, but the lack of empirical data adds high uncertainty to current estimates. The Intergovernmental Panel on Climate Change (IPCC) provides expert advice on standard methodologies for estimating emissions. In the wastewater sector, limited empirical emission factors mean that national inventories often fail to reflect the diversity in onsite sanitation systems linked to design, operation and maintenance of household containment units. For that reason, we measured methane emissions from 27 containment units across five Indonesian cities during both dry and wet seasons (two to four times per season over 12- or 24-hour periods). Sealed containment units had a median emission rate of 2.22 g CH_4 cap^{-1} day^{-1}, rising to 2.43 g CH_4 cap^{-1} day^{-1} for lined and 17.27 g CH_4 cap^{-1} day^{-1} for open (unlined) units. Our findings also show that construction features such as the presence of outlets and aeration affect methane emission rates, along with other

site-specific factors (sludge age, loading rate, etc.). These support the need for expanded empirical datasets in low- and middle-income countries where sanitation technologies are predominant and highly heterogeneous. This is essential for improving the accuracy and representativeness of national GHG inventories for the sanitation sector.

Keywords: Climate Resilience, Faecal Sludge, Greenhouse Gases, Methane Emission Rates, Onsite Sanitation

Synopsis

This study presents empirical data on methane emission rates from 27 household sanitation containment units in Indonesia. It reveals how the type of containment unit and sludge-related parameters influence methane emissions.

1 INTRODUCTION

The use of empirical data to estimate national greenhouse gas (GHG) emissions from sanitation is essential for accurately assessing the sector's contribution to climate change, particularly in low- and middle-income countries (LMICs), where onsite sanitation systems dominate yet remain significantly underrepresented in existing datasets. There is no doubt that climate change is one of the most significant challenges of the 21st century, driven primarily by the emission of GHGs such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) (IPCC, 2022). Although the energy, transport and agriculture sectors are widely recognised as major sources of GHG emissions, sanitation systems have attracted increasing attention as their relative contribution is expected to grow alongside improvements in sanitation service provision (Lambiasi et al., 2024). This issue is particularly important in the Global South, where microbial decomposition of organic carbon and nitrogen compounds stored in onsite containment units and subsequently conveyed through the sanitation value chain, generates direct emissions of methane and nitrous oxide (Oshita et al., 2014).

Globally, it has been estimated that septic tanks and pit latrines contribute with 4.7% of the global anthropogenic methane emissions (Cheng et al., 2022), where a significant proportion is attributed to onsite sanitation containment units, including household pits and tanks temporarily storing human excreta under anoxic conditions (Moonkawin et al., 2026). Such estimates are primarily based on limited empirical data, leading to substantial discrepancies between reported figures. For instance, Diaz-Valbuena et al. (2011) reported that methane emissions from a conventional septic tank, when estimated using a BOD-based IPCC-type approach and BOD default assumptions, can produce between 25.5 and 27.1 g CH_4 cap^{-1} day^{-1}. In contrast, their direct measurements using a modified static flux chamber yielded emissions of 11 g CH_4 cap^{-1} day^{-1} (geometric mean) from the same type of sanitation containment units (Diaz-Valbuena et al., 2011). In contrast, Manga and Muoghalu (2024) reported the opposite pattern, from a meta-analysis of emission rates, where IPCC-based estimates were found to be seven times lower than emissions reported in the literature. This evidence stresses the need for more accurate empirical data to produce national GHG inventories that truly represent local conditions and support present and future actions to improve access to safe sanitation while mitigating climate change impacts.

These data limitations in national GHG inventories are especially significant in LMICs, where sanitation containment units are the primary mode of wastewater management. According to the World Health Organisation (WHO) and United Nations Children's Fund (UNICEF) Joint Monitoring Programme (JMP), onsite sanitation containment units now serve more people (3.8 billion) than those with sanitation facilities connected to a sewer network (3.5 billion) (WHO & UNICEF, 2025). However, the wide heterogeneity of containment units commonly found in LMICs, ranging from unlined pits to fully-sealed, multi-chamber tanks, rarely corresponds to the system characteristics assumed in global emission models.

This discrepancy is further affected by the use of local terminology that often bears little relation to the actual physical properties and performance of the systems observed in practice (Strande et al., 2023). For instance, a septic tank is conventionally characterised as a watertight, multichambered structure that allows sedimentation and partial anaerobic treatment of wastewater before effluent infiltration or discharge (EPA, 2002). However, in many LMICs, the term 'septic tank' is informally applied to any underground container that accepts domestic wastewater, irrespective of its construction quality, watertightness or treatment performance (Odagiri et al., 2021; Strande et al., 2023). This lack of uniform classification, combined with the use of different local construction materials and processes, and a lack of data on connection type and loading rates, makes it challenging to apply IPCC emission factors accurately, potentially leading to underestimation or overestimation of GHG emissions.

In Indonesia, onsite sanitation containment units predominate in domestic wastewater management. Around 85.6% of the population (approximately 239 million people) use containment units such as pits or tanks, for wastewater and faecal sludge (Indonesia Australia Infrastructure Partnership et al., 2026). Locally, the phrase *tangki septik* (septic tank) refers to all onsite systems, from unlined pits with porous bases to lined, multichambered concrete tanks, with varying degrees of watertightness and treatment efficacy (Odagiri et al., 2021). While guidelines like the Indonesian National Standard SNI 2398:2017 – *Tata Cara Perencanaan Tangki Septik dan Sistem Resapan Air Limbah Rumah Tangga* (Design Guidelines for Septic Tanks and Domestic Wastewater Infiltration Systems) guide containment design and operation, enforcement is lacking. Therefore, many sanitation containment units are built informally for cost and convenience rather than for compliance. Many containment units do not have a lined or sealed base, and few are connected to an infiltration pit to ensure appropriate operation.

For example, a field assessment in Bandung, Indonesia, discovered that over 90% of household containment units were failing due to structural leakage, poor construction and infrequent desludging (Bao et al., 2020). The high cost of emptying services and restricted accessibility and public awareness for desludging truck services further discourage regular maintenance, especially among lower-income households (Odagiri et al., 2021). These challenges not only undermine environmental and public health objectives but also increase the potential for fugitive methane emissions. Furthermore, climate hazards cause air temperatures and sea levels to rise, intense rainfall, frequent flooding and high groundwater elevations (UTS-ISF, UI and UNICEF, 2021; Reddy et al., 2023), which are anticipated to exacerbate current challenges in onsite sanitation by prolonging soil saturation times and fostering anaerobic conditions inside containment units.

While previous studies have quantified methane emissions from sanitation containment units, most field-based measurements have been conducted in high- and middle-income countries (Diaz-Valbuena et al., 2011; Knappe et al., 2022; Somlai et al., 2019; Somlai-Haase et al., 2017; Truhlar et al., 2019), and only very few studies have investigated emissions from containment units in LMICs (Camargo-Valero et al., 2026; Huynh et al., 2021; Moonkawin et al., 2023). Apart from Camargo-Valero et al. (2026), who linked types of containment units to methane emissions, these studies generally did not account for the vast diversity of containment designs commonly found in LMIC contexts. As a result, we have a limited understanding of the relationships among the design of containment units, construction quality, local management practices, and methane emissions. To improve these estimates, field-based studies that consider characteristics of containment units in relation to environmental and operational conditions with methane flux behaviour are urgently required in LMICs.

This study addressed these gaps by providing direct field measurements of methane emissions from diverse types of sanitation containment units across five major Indonesian cities: Jakarta, Bandung, Balikpapan, Manggarai, and Manggarai Barat. This study focuses on methane emissions, as previous research has shown that nitrous oxide emissions from containment units were less significant (Huynh et al., 2021) and very difficult to measure using low-cost gas analysers (Camargo-Valero et al., 2026). To our knowledge, this is the first time empirical methane emissions from a large number of household sanitation containment units in Indonesia have been reported. Methane fluxes were directly measured onsite using a static flux chamber method and low-cost gas analysers from 27 household containment units selected to represent the range of sanitation containment units commonly used in Indonesia. These units encompassed construction categories that varied in lining, sealing, outlet configuration and aeration, thereby reflecting the diversity of household sanitation containment units across urban and peri-urban settings in Indonesia. By employing a standardised static flux chamber method and measuring emissions across multiple Indonesian cities representing varied geographic and socio-economic contexts, this study aimed to provide new empirical evidence on how design of containment units influences methane emissions from sanitation containment units in LMICs.

2 METHODOLOGY

2.1 Study area

Fieldwork was conducted in five cities and regencies in Indonesia: Jakarta, Bandung, Balikpapan, Manggarai Barat and Manggarai (Figure 1). We selected these locations to represent different climatic conditions, geographical variations, population densities, groundwater conditions and onsite sanitation containment units at the household level. Eligibility was determined based on accessibility, safety, the household's willingness to participate in the study, and engagement with relevant local stakeholders and government counterparts.

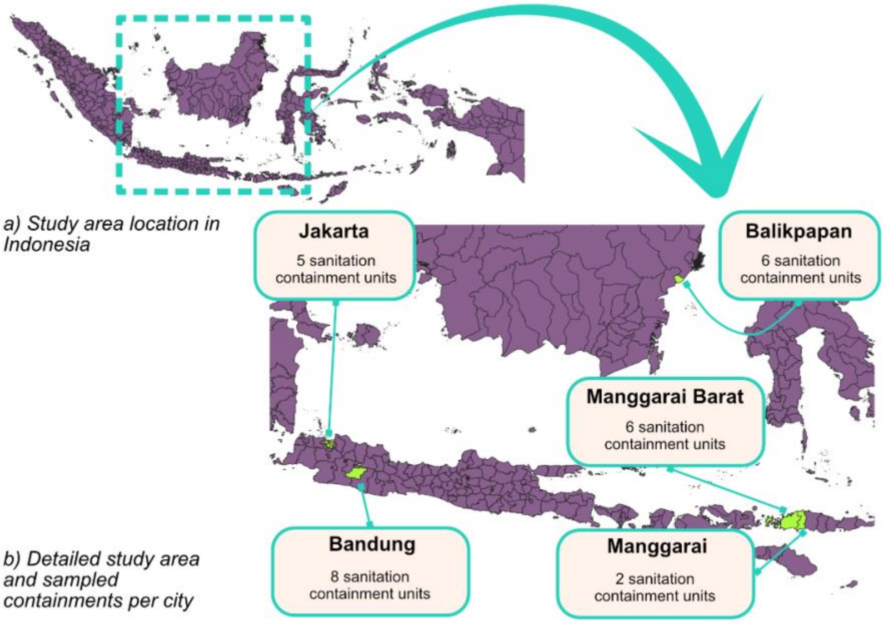


Figure 1. Study area and location of sampled sanitation containment units per city

Table 1. Summary of population, climate characteristics and onsite sanitation coverage in the five Indonesian study locations in 2024

City / Regency	Population	Climate type and key features	Percentage of the population served by onsite sanitation containment units
Jakarta	10,684,946[1]	Climate[2]: Tropical rainforest Air Temperature[1]: 22.8–36.8 °C Rainy season peak[1]: February (488.8 mm) Dry season peak[1]: August (1.5 mm)	99.7%
Bandung	2,506,600[3]	Climate[2]: Tropical monsoon Air Temperature[3]: 19.1–32.5 °C Rainy season peak[3]: November (512 mm) Dry season peak[3]: July (30 mm)	43.6%
Balikpapan	738,532[4]	Climate[2]: Tropical rainforest Air Temperature[4]: 22.9–35.6 °C Rainy season peak[4]: August (660 mm) Dry season peak[4]: April (89 mm)	93.0%
Manggarai	328,758[5]	Climate[2]: Tropical monsoon Air Temperature[5]: 8.4–29.5 °C Rainy season peak[5]: March (543 mm) Dry season peak[5]: August (79.20 mm)	98.3%
Manggarai Barat	270,917[6]	Climate[2]: Tropical rainforest Air Temperature[6]: 21.2–34.6 °C Rainy season peak[6]: April (238.2 mm) Dry season peak[6]: August (0.3 mm)	70.1%

[1] BPS-Statistics of DKI Jakarta Province (2025) | [2] Beck et al. (2018) | [3] BPS-Statistics of Bandung Municipality (2025) | [4] BPS-Statistics of Balikpapan Municipality (2025) | [5] BPS-Statistics of Manggarai Regency (2025) | [6] BPS-Statistics of Manggarai Barat Regency (2025)

General characteristics of all study areas, including population size, climate conditions and onsite sanitation services, are summarised in Table 1. Information on onsite sanitation coverage was derived from consultations with local wastewater operators and official reports from government agencies and wastewater utilities.

2.2 Overall field data collection

Informed consent was obtained from each household before sampling, in accordance with the ethical research protocol approved by the National Research and Innovation Agency Indonesia (Clearance number: 010/KE.04/SK/01/2025). The geographic coordinates of each containment unit were recorded, and physical characteristics like depth and surface area were measured directly when accessible. General information on containment units was also recorded, including the type of influent (greywater only, blackwater only, grey- and blackwater, mixed or mixed with additional inflows), presence of filter media, structural configuration (open, lined, or sealed at walls, base, and top), number and size of containment units, presence of vent pipe and outlet, number of users, and presence of aeration (See Supporting Information, Section SI-1).

Before sampling, the team briefed the household owner to obtain consent and ensure awareness of the planned activities. The description of each containment unit was also informed by household interviews, which collected contextual data on water use, sanitation practices, and containment management. The structured survey was administered face-to-face in Bahasa Indonesia with the primary respondent, typically the household head or an adult member responsible for managing sanitation facilities. All responses and location data were anonymised to ensure confidentiality and compliance with our research ethics protocol. The dataset is available in the project's open-access repository (Ilmi et al., 2026). The team followed the health and safety procedures defined in our respective standard operating procedure for field work (Smeaton-Russell et al., 2026).

2.3 Methane gas sampling campaigns

We sampled each site twice over nine months: the first campaign ran from November 2024 to March 2025, and the second from May to August 2025. The sampling design aimed to capture daily variations and the potential influence of the rainy and dry seasons on methane emissions. To account for short-term precipitation effects, we also recorded daily accumulated rainfall and ambient temperature for the three days preceding each sampling event. Rainfall data were collected from publicly available records at nearby meteorological stations (Direktorat Data, 2025). Gas emissions measurements were taken using the static flux chamber method, coupled with the use of low-cost handheld equipment for methane analysis on-site (Camargo-Valero et al., 2026). In the first sampling campaign, methane sampling surveys started approximately every six hours over 24 hours (four cycles per day), with methane gas concentrations measured and recorded from the headspace of the flux chamber every 10 minutes, over three hours per cycle. Based on results from the first sampling campaign, we reduced the survey to two cycles per day (3 hours each) over eight hours in the second sampling campaign.

2.4 Static flux chamber method and methane emission rates

Methane gas emissions from containment units were captured using the static flux chamber method, a technique previously employed by other researchers investigating greenhouse gas emissions from sanitation containment units (Diaz-Valbuena et al., 2011; Huynh et al., 2021; Moonkawin et al., 2023), but with the difference that low-cost gas analysers were used to conduct gas measurements onsite – i.e., Geotech GA5000 or Biogas 5000 (Geotech QED Environment) for methane, carbon dioxide and oxygen analysis (Camargo-Valero et al., 2026).

The flux chamber used in our study was adapted from Camargo-Valero et al. (2026) and designed to be practical and accessible by using materials commonly available in Indonesia (see Figure 2 and Supporting Information, Section SI-2).

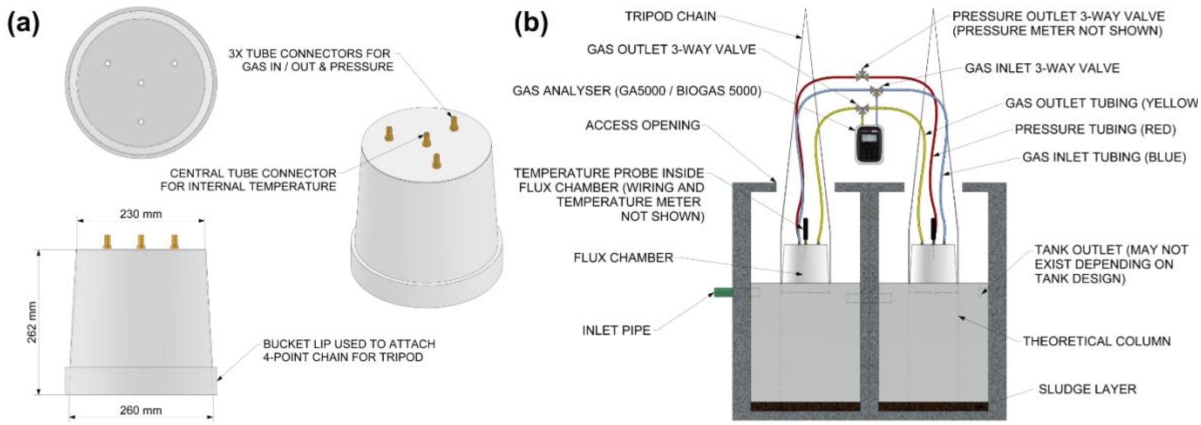


Figure 2. Schematic of the standard static flux chamber used in the field to measure methane emissions from 27 onsite sanitation containment units in Indonesia. (a) Flux chamber diagram. (b) Flux chamber and sanitation containment unit setup.

At the end of each sampling cycle, and for cross-validation of gas sample analysis, the accumulated gas in the headspace was sampled via syringe, flushed and transferred to a nitrogen-evacuated, 20mL HS-20 pressure release crimp clear vial (Shimadzu Corporation, Japan) for further laboratory GC analysis using Shimadzu Nexis™ GC-2030 (Shimadzu Corporation, Japan) for CH_4, CO_2, and N_2O. Samples were stored at 4–8°C before laboratory analysis.

Methane gas concentrations collected over time from single chamber containment units were processed to calculate normalised methane emission rates - NERs (g cap^{-1} day^{-1}) following the methods used by Camargo-Valero et al. (2026); See Supporting Information, Section SI-2. At sampling sites where onsite sanitation containment units had more than one chamber (some with up to six chambers per unit), NERs were assessed using a mixed-method approach combining empirical and modelling methods (See Supporting Information, Section SI-3).

To facilitate a wider comparison of the empirical NERs derived in this study, we calculated emissions rates using IPCC methodologies (Equations 6.1, 6.2 and 6.3c from the 2019 IPCC Guidelines - IPCC, 2019), to estimate daily methane emissions from onsite sanitation containment units (septic tanks and pit latrines) treating the nominal quantity of waste produced by a single individual. Also, a one-at-a-time (OAT) sensitivity analysis was conducted to evaluate the influence of input parameters on estimated methane emissions using the IPCC methodologies (Saltelli et al., 2008). The model was first assessed under baseline conditions for septic tanks, defined as the midpoint of the plausible range for each parameter. For more details, see Supporting Information, Section SI-4.

2.5 Additional physicochemical and environmental parameters

In addition to gas sample analysis, several physicochemical parameters were measured to support the interpretation of methane emission dynamics and characterise the environmental conditions within each containment unit. Those parameters included total and soluble chemical oxygen demand (COD and SCOD, respectively), oxidation-reduction potential (ORP), dissolved oxygen, temperature (air and liquid-sludge mixture), pH, sludge thickness, and total solids and volatile solids. To measure these parameters, we collected mixed liquid–sludge samples at full depth in the containment unit, capturing all available layers from each compartment at each site, using a septic checker. For laboratory analysis, samples were immediately placed on ice after collection and transported to the laboratory, where they were stored at 4 °C until analysis. For SCOD, samples were filtered using a 0.45 μm membrane filter. The corresponding analytical methods and field equipment are listed in **Table 2**.

Table 2. Analytical field and lab methods and equipment used for sample characterisation of mixed liquid-sludge samples at full depth in the containment unit

Parameter	Sample	Method / Equipment
Dissolved oxygen	Unfiltered	Multiparameter water meter (Lutron YK-2001PHA)
pH	Unfiltered	
Temperature	Unfiltered	
ORP (Ag/AgCl reference gel)	Unfiltered	
Sludge thickness	N/A	Septic checker
Chemical Oxygen Demand (COD)	Unfiltered	Closed reflux method, UV–Vis spectrophotometric analysis (HACH DR6000)
Soluble Chemical Oxygen Demand (SCOD)	Filtered	

2.6 Statistical analysis

We assessed the normality of the NER dataset using the Shapiro–Wilk test and Q–Q plots; see Supporting Information, Section SI-5.1. When data were found to be non-normally distributed ($p < 0.05$), we used non-parametric tests (Wilcoxon rank-sum, Dunn, and Kruskal-Wallis). Methane emissions are reported as median values with 95% confidence intervals, calculated via bootstrap resampling. Histograms, violin plots and box plots were used to visualise distributions and compare NER values. Spearman correlations were performed between NERs and four sludge-related parameters to evaluate monotonic relationships between sludge accumulation and methane emissions. All analyses were conducted in R software (R v. 4.4.2; RStudio v. 2025.09.1+401).

Methane emission rates were analysed using generalised linear mixed models (GLMMs) to examine the influence of system typology while accounting for the clustered data structure. GLMMs extend generalised linear models by incorporating random effects, allowing proper handling of repeated measurements collected from the same sanitation containment unit and the resulting non-independence among observations (Bolker et al., 2009). Because methane emission rates were strictly positive and strongly right-skewed, we modelled the response using a Gamma distribution, a standard approach for non-normal continuous environmental data, with a logarithmic link to ensure that the fitted values are always non-negative (Zuur et al., 2009).

Typology variables describing onsite sanitation containment units (wall condition, base condition, and presence of outlet and aeration) were specified as fixed effects simultaneously, while system identity was included as a random intercept. Random slopes were not included due to the limited number of replicates per containment unit. Model parameters were estimated using maximum likelihood and adaptive approximation methods implemented in the glmmTMB package in R, which provides robust estimation for non-normal mixed-effects models (Brooks et al., 2017). Model assumptions were evaluated using residual diagnostics and simulation-based checks, following the recommended practices for GLMMs (Zuur et al., 2009), with 250 standard simulations using the DHARMa package (Hartig, 2024). The complete tabulated and graphical results of our GLMM, along with some guidelines for result interpretation, are in Supporting Information, Section SI-5.2.

3 RESULTS AND DISCUSSION

3.1 Characteristics and classification of household sanitation containment units

The percentage of households using sanitation containment units in the five cities covered by this study is 99,7% in Jakarta, 98.3% in Manggarai, 93.0% in Balikpapan, 71.0% in Manggarai Barat, and 43.6% in Bandung, where 25% of the population is served by centralised sewerage systems connected to either communal or centralised wastewater treatment plants (Table 1). The 27 containment units included in our study were categorised into six types (i.e., Types A-F), representing standard sanitation technologies used in Indonesia (Table 3).

Table 3. Onsite sanitation unit classification considering design and construction features

Onsite sanitation unit type	**Location**					**Number of containment units**	**Replicates per containment unit**	***n***
	Jakarta	**Bandung**	**Balikpapan**	**Manggarai**	**Manggarai Barat**			
Type A: Base AND Walls sealed | One chamber | With outlet | Without aeration	1	1	1	0	0	3	6	13
Type B: Base AND Walls sealed | More than 1 chamber | With outlet | Without aeration	2	0	1	0	1	4	6	23
Type C: Base AND Walls sealed | More than 1 chamber | With outlet | With aeration	2	0	0	0	0	2	6	11
Type D: Base OR Walls lined | One chamber | With outlet | Without aeration	0	1	1	0	1	3	6	13
Type E: Base OR Walls lined | One chamber | Without outlet | Without aeration	0	6	1	2	1	10	6	55
Type F: Base OR Walls open | One chamber | With OR without outlet | Without aeration	0	0	2	0	3	5	6	28
					Total	**27**	**36**	**143**

n = number of emission rate data calculated per type of containment unit.

From the approximately 239 million people in Indonesia relying on household containment units, 18.3% (43,629,711), 14.9% (35,656,101), 15.7% (37,402,689), 14.2% (33,875,899), 16.7% (39,779,437), and 20.2% (48,278,652) use containment unit types A through F, respectively (Indonesia Australia Infrastructure Partnership) et al., 2026). Therefore, this classification framework for containment units accurately reflects the current sanitation technologies used by all households with onsite sanitation in Indonesia, enabling consistent comparisons across representative containment types. This classification distinguishes units based on design and construction characteristics that may influence performance and methane emissions, including the number of chambers per containment unit, the presence or absence of aeration, wall and base integrity, and the presence or absence of an effluent outlet. A complete description of the containment typology used in this study is presented in **Error! Reference source not found.** of the Supporting Information.

Most of the sanitation containment units included in this study receive blackwater, apart from one that receives a mix of greywater and blackwater (JA-3, Type C), leading to predominant anaerobic conditions in the containment units. The reported figures for redox potential (ORP) in liquid-sludge mixture stored in the 27 containment units ranged from −298 mV to −283 mV (95% confidence interval (CI); median = −290 mV; n = 113). Dissolved oxygen (DO) readings ranged from 0.5 mg/L to 0.7 mg/L (95% CI; median = 0.6 mg/L; n = 114). Both ORP and DO values fall close to typical ranges for anaerobic digestion. In fact, anaerobic digesters processing primary sewage sludge at wastewater treatment plants, which are designed to achieve maximum methane yields, operate with ORP values between −495 mV and −300 mV and DO concentrations from 1×10^{-8} mg/L to 0.1 mg/L (Nguyen et al., 2019). The presence of dissolved oxygen traces in tested containment units (DO probe accuracy: ± 0.4 mg/L) suggests local or transient oxygen intrusion, likely due to intermittent inflow of fresh blackwater and poor mixing conditions that induce stratification and reduce inhibitory effects on methanogens. In addition, reduced compounds demanding oxygen for their chemical/biological stabilisation are commonly present in high concentrations in onsite sanitation containment units (i.e., organic matter, hydrogen sulphide and Fe^{+2}) (Botheju & Bakke, 2011), which can rapidly consume any dissolved oxygen in the liquid and prevent toxic effects on methanogens. Additional data from the characterisation of liquid–sludge mixture samples confirms that COD (95% CI: 11,700 - 45,450 mg/L; median = 31,575 mg/L; n = 23), pH (95% CI: 7.20 - 7.43; median = 7.33; n = 111), and liquid-sludge mixture temperature (95% CI: 27 - 29 °C; median = 28 °C; n = 114) are within the ranges of conditions favouring anaerobic digestion.

Operation and environmental conditions found in all 27 onsite sanitation containment units can support anaerobic digestion and methanogenic activity as confirmed by methane emission data (Section 3.2); however, the maximum methane producing capacity will depend on the intrinsic characteristics of each containment unit. For instance, the use of water for anal cleansing is a common practice in Indonesia and this can reduce organic loading rates and residence times that directly control methane yields. Data distribution of ORP, DO, COD, pH and temperature, and their corresponding raw data, is reported in the Supporting Information, Sections SI-6 and SI-7, respectively.

3.2 Diurnal variation of methane emissions

All Methane NERs calculate per site and per sampling campaign are available at Zenodo (https://zenodo.org/records/18792536), an open-access research repository (Ilmi et al., 2026). Violin plots were used to visualise methane NERs across four sampling cycles over a 24-hour period: morning (06:00-12:00), afternoon (12:00-18:00), evening (18:00-24:00) and night

(00:00-06:00) (**Figure 3**). Results from descriptive statistics, Shapiro–Wilk tests and Q–Q plots inspection confirm that data from each cycle is not normally distributed and skewed to lower values (Supporting Information, Sections SI-5.1 and SI-8), which agrees with empirical methane emissions data reported for onsite sanitation systems (Camargo-Valero et al., 2026; Diaz-Valbuena et al., 2011; Leverenz et al., 2010). NER median values for surveys conducted in the morning, afternoon, evening and night were 2.87, 3.46, 2.23, and 2.35 g CH_4 cap^{-1} day^{-1}, respectively. Violin plots suggest variations in emission rates from morning to night, but results from central tendency variables (mean and median) do not show a consistent directional trend across the four cycles. In fact, results from a Kruskal–Wallis test, suitable for non-normally distributed data, confirmed no statistically significant differences in methane NERs between sampling cycles (p = 0.373). To our knowledge, our study is the first report of diurnal variation of methane emission rates from containment units over 24 hours in LMICs.

It is important to highlight that empirical methane emissions from onsite sanitation containment units can only be fully assessed by conducting sampling campaigns that consider inherent variability in line with influent characteristics, so representative figures can be drawn from a comprehensive data set. For instance, Leverenz et al. (2010) studied a prefabricated septic tank receiving domestic wastewater in the USA and showed a delay in peak methane emissions (afternoon and night) after the traditional two daily peaks for water consumption and domestic wastewater generation (morning and evening), which is not the case in our study with containment units mainly receiving blackwater.

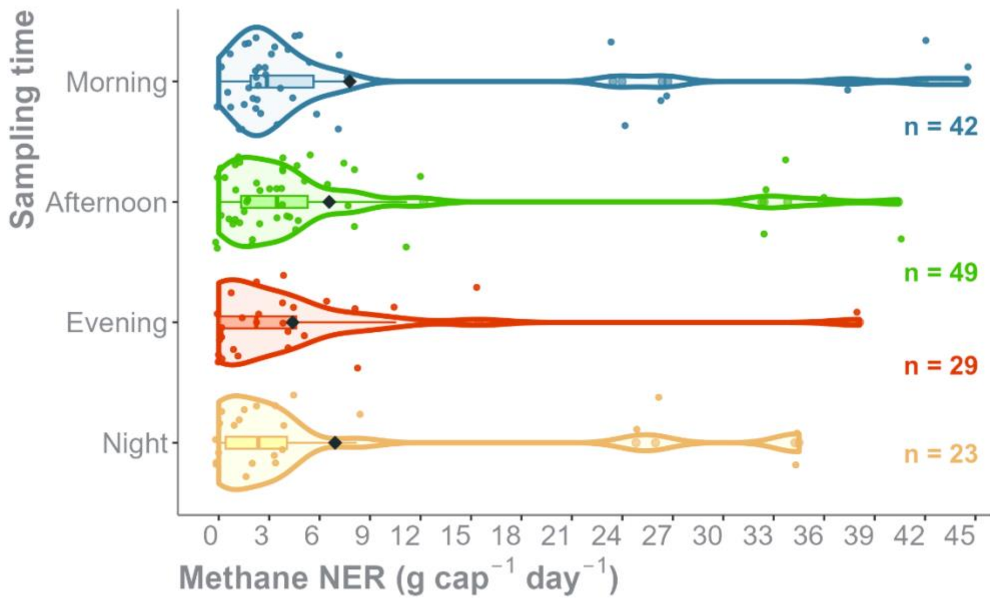


Figure 3. Methane NERs per sampling cycle for 27 sanitation containment units in Indonesia. Diamonds indicate the mean values, vertical lines inside boxes mark the median, boxes show the interquartile range, and violins display the smoothed distribution of the data. Sampling cycles: morning (06:00-12:00), afternoon (12:00-18:00), evening (18:00-24:00), night (00:00-06:00).

3.3 Overall methane emission analysis

Methane NER distribution is shown in **Figure 4**. The original data (n = 143) are positively skewed (skewness = 2.29), with a long right tail extending to a maximum of 44.48 g CH_4 cap^{-1} day^{-1} (**Figure 4**). Ten methane NER values equal to zero were observed, all from 10 different sanitation containment units receiving blackwater only and with no aeration (Types A, B, D, E, and F). Additional replicates from same sites were below the qualification limit of the method (See Section 2.6.1), which confirmed that these 10 containment units were producing low methane emissions during this study.

36 methane NER values were above Q3 (25% data > 5.27 g CH_4 cap^{-1} day^{-1}) from 12 sanitation containment units, receiving blackwater and with no aeration chambers (Types A, B, E and F). Their high methane NERs were consistent across all sampling cycles. All Type C (multichambered, sealed walls and base, with outlet and aeration) and Type D (one chamber, lined walls or lined base, with outlet, without aeration) containment units produced methane NERs between Q1 and Q3 (75% data < 5.27 g CH_4 cap^{-1} day^{-1}). The absence of a clear pattern in high methane NERs suggests that other factors beyond design and construction characteristics may influence methane emissions in onsite containment units.

Overall, methane emission rate data from 27 household sanitation containment units included in this study had a median of 2.89 g CH_4 cap^{-1} day^{-1} with a 95% confidence interval of 2.31-3.70 g CH_4 cap^{-1} day^{-1} (n = 143). These figures are lower than those reported in previous fieldwork studies. Camargo-Valero et al. (2026) reported typical methane emission rates (geometric mean) of 7.9 g CH_4 cap^{-1} day^{-1} (90% CI range: 2.63 - 23.7 g CH_4 cap^{-1} day^{-1}; n = 387) in a study covering 146 sanitation containment units receiving blackwater and mixed water, across Ethiopia, Nepal, Senegal and Uganda. Studies covering 25 septic tanks receiving blackwater in Vietnam reported a mean methane emission rate of 10.29 g CH_4 cap^{-1} day^{-1} (2.23 – 46.38 g CH_4 cap^{-1} day^{-1}, min. to max. range; n = 25) (Huynh et al., 2021; Moonkawin et al., 2023). Studies in the USA covering 8 pre-fabricated septic tanks reported a geometric mean of 10.70 g CH_4 capita^{-1} day^{-1} (0.07 – 75.69 g CH_4 capita^{-1} day^{-1}, min. to max. range; n = 39) (Diaz-Valbuena et al., 2011; Leverenz et al., 2010). The lower methane emission values found in our study may be attributed to lower organic loading rates and shorter retention times due to the discharge of diluted blackwater containing anal cleansing wastewater.

The complete set of descriptive statistics for NER data is reported in the Supporting Information, Section SI-9.

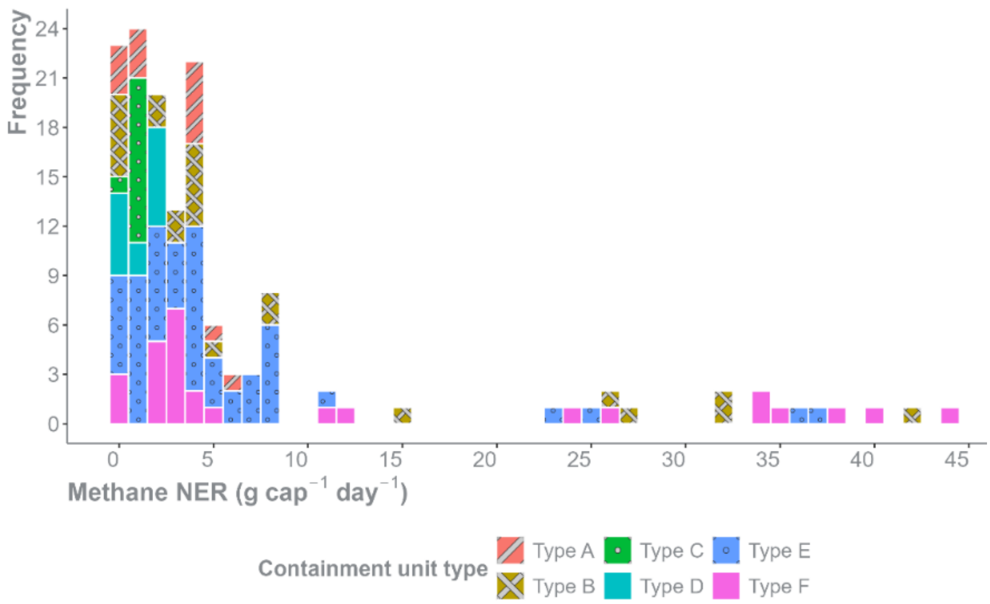


Figure 4. Histogram of methane NERs for all 27 sanitation containment units sampled in Indonesia (n=143)

3.4 NER comparison between wet and dry seasons

Indonesia has a tropical monsoon climate with high rainfall (2,500 mm to 6,000 mm annually), a consistent year-round mean air temperature ranging from 22°C to 33°C (largely unaffected by season change) and two main seasons: a dry season usually between April and October and a wet season from November to March (Indonesia Meteorology, Climatology and Geophysics Agency, 2026). However, recent shift westward of the Indian summer monsoon is altering dry/wet season patterns in Southeast Asia creating an impact on the variability of rainfall and frequency and intensity of intermittent flooding of some areas during the wet season with severe consequences to infrastructure including sanitation services (Loo et al., 2015). Analysis of historical extreme indices based on air temperature and precipitation (1983-2012) also indicate warming and wetter trends in Indonesia (Supari et al., 2017).

Given these weather variations, we assessed seasonal changes in methane emissions by comparing NER results between wet and dry seasons. Rainfall conditions during fieldwork were assessed by comparing daily cumulative rainfall over the three days preceding each sampling event for wet and dry seasons. Based on results from a Wilcoxon test (p = 0.038), we confirmed that daily rainfall was significantly higher in the wet season sampling (median = 1.7 mm; 95% CI: 1.50–3.48 mm), than in the dry season sampling (median = 0.65 mm; 95% CI: 0.30–1.80 mm).

Seasonal changes in temperature may also affect methane emissions. We confirmed that the difference between air temperature (Wet season: median = 26 °C, 95% CI: 25-27 °C, n = 95. Dry season: median = 27 °C, 95% CI: 26-28 °C, n = 56) and water column temperature inside containment units (Wet season: median = 27 °C, 95% CI: 26-28 °C, n = 33. Dry season: median = 28 °C, 95% CI: 27-29 °C, n = 81) varied modestly between sampling seasons (ΔT = 1 °C), with water column temperature inside containment units within the lower range for mesophilic anaerobic digestion in both seasons. For access to the entire air and water temperature data set, see Supporting Information, Sections SI-6.4, SI-7.2 and SI-7.3.

Despite differences in rainfall during wet and dry seasons, methane NERs were not significantly different. This is supported by results from a Wilcoxon test (p = 0.126) comparing methane emissions during the wet season (median NER = 2.57 g CH_4 cap^{-1} day^{-1}; 95% CI: 1.71 - 3.70 g CH_4 cap^{-1} day^{-1}), with those during the dry season (median NER= 3.30 g CH_4 cap^{-1} day^{-1}; 95% CI: 2.26 - 4.37 g CH_4 cap^{-1} day^{-1}) (Figure). Moonkawin et al. (2023) reported comparable methane emission rates from two septic tanks (T1 and T2) in Northen Vietnam during winter and summer despite a liquid temperature difference of 9.4 °C between seasons – i.e., the mean emission rates in winter and summer of T1 were 11.1 g CH_4 cap^{-1} day^{-1} at 21.6 °C and 11.1 g CH_4 cap^{-1} day^{-1}at 31.1 °C, and those of T2 were 15.5 g CH_4 cap^{-1} day^{-1} at 22.1 °C and 15.5 g CH_4 cap^{-1} day^{-1} at 31.4 °C, respectively. Similar results were reported by Camargo-Valero et al. (2026), who found no significant differences in methane NERs between dry (7.7 g CH_4 capita^{-1} day^{-1}, n = 171) and wet (7.9 g CH_4 capita^{-1} day^{-1}, n = 201) seasons from sanitation containment units in Ethiopia, Nepal, Senegal and Uganda.

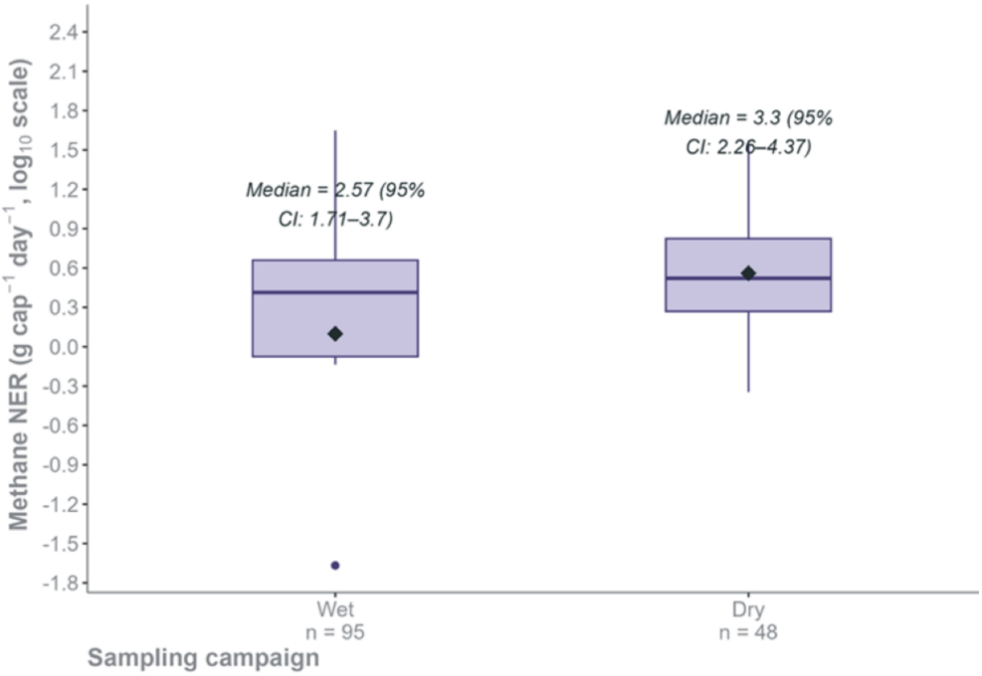


Figure 5. Methane NERs from sampling campaigns in wet and dry seasons. Diamonds indicate the mean, horizontal lines mark the median, and the box indicates the interquartile range. Box plots were produced using Log_{10} NER values, but Median and Confidence Intervals (CI) correspond to untransformed data.

It is important to note that the temperature inside containment units governs not only bacterial metabolism (methane yields), but also the solubility of methane in water. At 20 °C, the solubility of methane in water is 23 g CH_4 m^{-3}, and it drops to 19 g CH_4 m^{-3} at 30 °C (Yamamoto et al., 1976). In our study, water column temperature inside containment units ranged from 21 °C to 32 °C (min-max), indicating that methane gas is solubilised in unsaturated fresh influent within the water column and/or wash out of the containment unit through the effluent outlet. Gómez-Borraz (2025) found that septic tanks operated at high temperatures (30 °C) produce more methane gas emissions than at lower temperatures, with >80% of methane remaining dissolved at ≤20 °C, which is often discharged with the effluent.

Although another factor influencing seasonal variation in methane NERs from sanitation containment units may be dilution of blackwater with rainwater, relevant information from the sampling campaigns conducted in this study, such as runoff flows and groundwater levels in each sampled containment unit, was not available or insufficient to conduct a quantitative analysis to establish clear correlations. Therefore, we could not verify in the field whether rainwater directly entered the sanitation containment units, whether this altered containment unit biochemistry, or whether sealed tanks effectively prevented rainwater (or groundwater) from entering the chambers at all.

Indeed, IPCC guidelines for GHG inventories differentiate methane conversion factors (MCF) and emission factors (EF) in latrines by considering weather conditions and groundwater table levels, setting higher factor values in wet climates and when the groundwater table is higher than water latrine levels – i.e., no evidence is provided to support those factors, as they are recommended based on author's expert judgement (IPCC, 2019). However, we found that rainfall did not affect methane emissions. Future studies on GHG emissions from sanitation containment units could include monitoring groundwater levels and identifying infiltration of runoff/groundwater during wet seasons to better assess the influence of rainwater on methane emissions.

3.5 Methane emissions and structural characteristics of containment units

3.5.1 Wall and base characteristics

Three types of wall and base categories were used to identify the impact of construction materials on methane emissions from containment units: a) open (some of type F), b) lined (Types D and E), and c) sealed (Types A, B and some of type F). Type C containment units were excluded from this analysis because they differ significantly from the other sealed containment units, as they are aerated intermittently (2 hours per day). In containment units with open (meaning unlined) walls or bases and in most lined systems, there is potential for liquid to infiltrate into the soil, while in sealed containment units the liquid can only exit the unit via an outlet or when the unit is emptied. Methane NERs considering wall types are reported in **Figure** – i.e., NER data have been Log_{10}-transformed to facilitate comparison. Median methane emissions were substantially higher in containment units with open walls (17.27 g CH_4 cap^{-1} day^{-1}) compared with containment units with lined (2.43 g CH_4 cap^{-1} day^{-1}) or sealed walls (3.62 g CH_4 cap^{-1} day^{-1}). Further statistical analysis (Dunn's test) confirmed that there is no significant difference between NER values from sealed and lined containment units (p = 0.658), but a clear difference between open and sealed (p = 0.013) and open and lined (p = 0.005). Our results are comparable with those of Camargo-Valero et al., (2026), who reported that open containment units produced higher methane emissions (32.1 g CH_4 $capita^{-1}$ day^{-1}) than sealed or lined units (5.8 and 8.7 g CH4 $capita^{-1}$ day^{-1}, respectively), with no significant difference between sealed and lined units sampled in Ethiopia, Nepal, Senegal, and Uganda. There are no statistically significant differences among sealed, lined, and open base alone in containment units (p = 0.052, Supporting Information, Section SI-10).

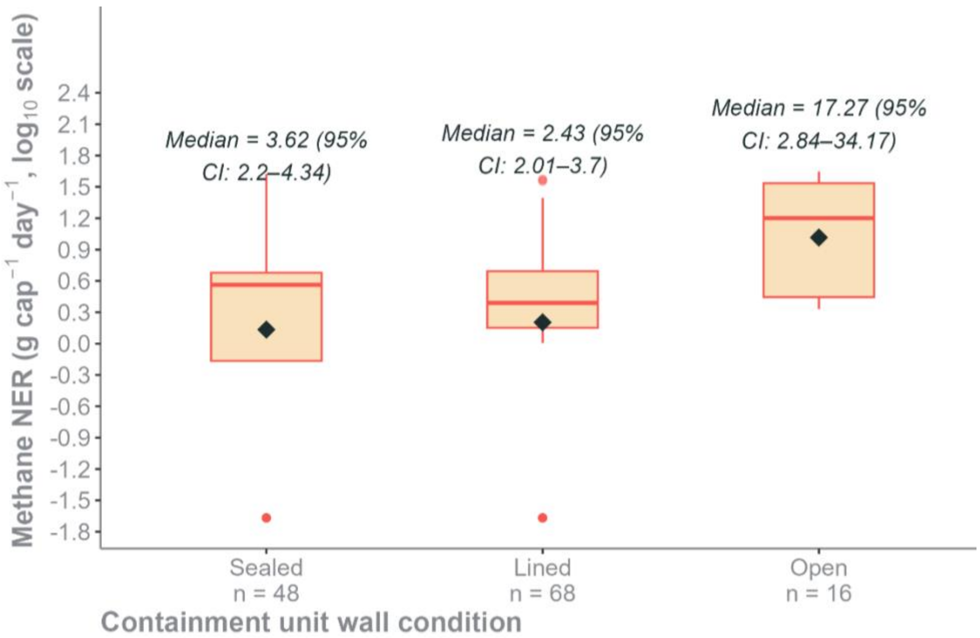


Figure 6. Methane NERs for containment units with sealed (Types A, B & some of type F), lined (Types D & E) and open walls (some of type F). Diamonds indicate the mean, horizontal lines mark the median, and the box indicates the interquartile range. Box plots were produced using Log10 NER values, but Median and Confidence Intervals (CI) correspond to untransformed data.

3.5.2 Effluent outlet

The presence (Types A, B, D, and some of type F) or absence (Types E and some of type F) of an effluent outlet determines operational conditions and performance. We excluded the Type C containment units from this analysis for the same reason mentioned in the previous section. Containment units with an effluent outlet are expected to operate as intermittent-flow reactors, whilst a containment unit without an effluent outlet may operate as a sequencing batch reactor; however, infiltration is more commonly through an open base or walls. In this study, median methane NERs were significantly higher in sanitation containment units without an outlet (3.70 g CH_4 cap^{-1} day^{-1}) than in those with an outlet (2.35 g CH_4 cap^{-1} day^{-1}) (Wilcoxon test, p = 0.011), see Figure – i.e., NERs are plotted as Log_{10}-transformed data to facilitate comparison.

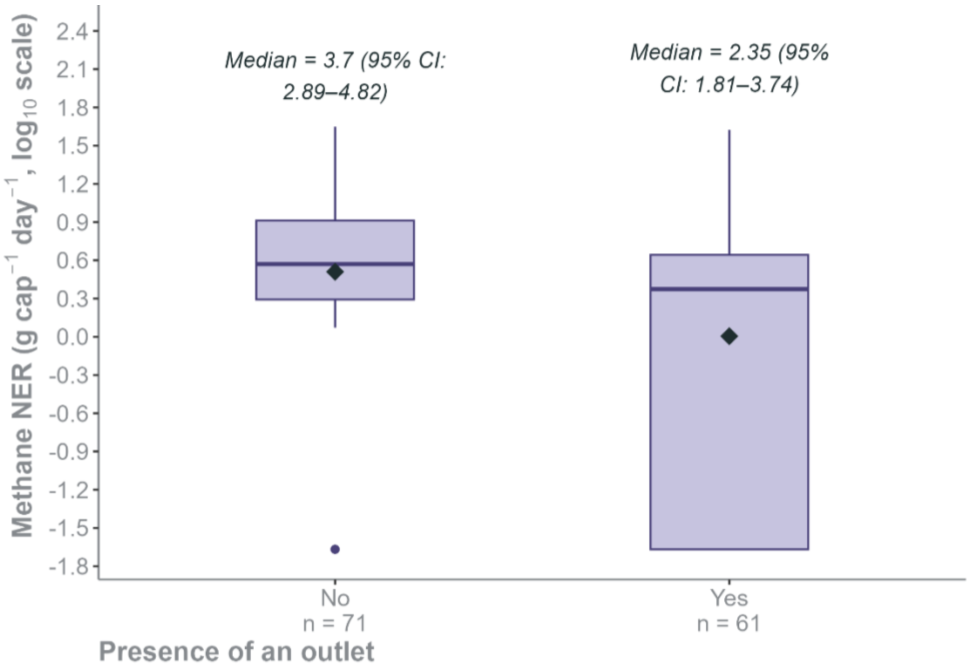


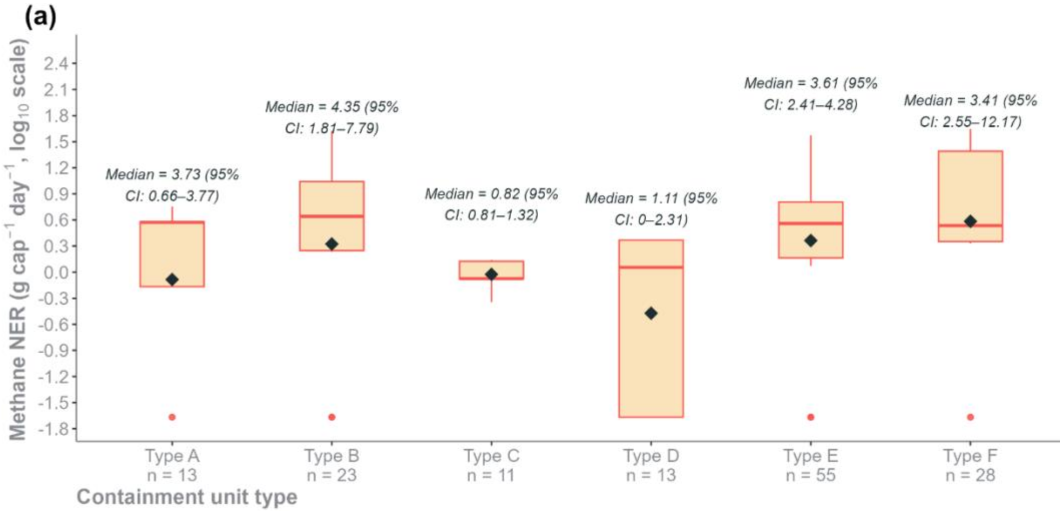
Figure 7. Methane NERs for containment units, with (types A, B, D & some of type F)) and without outlets (types E and some of type F). Diamonds indicate the mean, horizontal lines mark the median, and the box indicates the interquartile range. *n* = number of replicates. Box plots were produced using Log_{10} NER values, but Median and Confidence Intervals (CI) correspond to untransformed data.

These results suggest that containment units without outlets retain biodegradable organic matter for longer under anaerobic conditions, allowing solubilised organic matter to be transferred from the sludge layer to the liquid above it, remaining in the containment unit until exiting via overflows, potentially by infiltration in lined or open units or when the containment unit is emptied. This would favour methane production and subsequent emissions. In containment units with an effluent outlet, the remaining biodegradable organic matter in the liquid is transported outside the sanitation containment unit, along with any solubilised methane gas, reducing emissions at the containment unit itself. It is known that dissolved methane in the liquid fraction and can reach supersaturation concentrations within septic tanks (Gómez-Borraz et al., 2025). This dissolved methane gas is transported outside the containment unit in the effluent and can be potentially released, depending on downstream conditions along the sanitation chain, and should be accounted for in GHG inventories (Short et al., 2017; Camargo-Valero et al., 2026). For example, the USEPA added the category 'discharge' to account for dissolved methane in the effluent of wastewater treatment plans in their national inventory 1990-2020 (EPA, 2022).

3.6 NER comparison among sanitation containment unit types

Methane NER data distribution, median and confidence interval (95%) for the six household sanitation containment types (A to F) are shown in Figure 5. Methane NERs differed significantly among system types (Kruskal–Wallis test, $p < 3.73 \times 10^{-5}$). Pairwise comparisons using Dunn's test (Figure 5b) showed that Type B containment units had significantly higher methane NERs than Types C and D. This may be related to the higher capacity of multichambered sanitation containment units to remove more organic matter as black water flows through the system. The Wilcox test indicated that there are no significant differences in methane NER between multichambered and single-chambered containment units ($p = 0.403$). Still, the median methane NER is slightly lower in multichambered containment units than in single-chambered ones (1.75 g cap^{-1} day^{-1} vs 3.13 g cap^{-1} day^{-1}). In the presence of aeration devices (Type C), dissolved oxygen increases, favouring aerobic bacterial metabolism, which in turn decreases methane production. The complete set of descriptive statistics for the six typologies is in **Error! Reference source not found.**, Supporting Information, Section SI-11.

(a)



(b)

p-values for Dunn's test						
	Type A	Type B	Type C	Type D	Type E	Type F
Type A						
Type B	0.719					
Type C	0.790	0.010 *				
Type D	0.909	0.015 *	0.759			
Type E	0.809	1.000	0.009 **	0.014 *		
Type F	0.310	1.000	0.002 **	0.003 **	1.000	

* $p < 0.05$ | ** $p < 0.01$ | *** $p < 0.001$

Figure 5. Log10-transformed methane NERs for every sanitation containment unit type (Type A to F – see Table 3). (A) Diamonds indicate the mean, horizontal lines mark the median, and the box indicates the interquartile range and *n* = number of replicates per unit type. (B) The table corresponds to the *p*-values for Dunn's test. Box plots were produced using Log$_{10}$ NER values, but Median and Confidence Intervals (CI) correspond to untransformed data.

In contrast, methane NERs in type E and F containment units emitted significantly more methane than those in types C and D. Higher methane emissions from sanitation containment units without an outlet and with open or lined walls or a base might be related to the accumulation of particulate organic matter in the sludge layer, with a lower liquid fraction due to potential infiltration into the soil, which could favour higher methane yields and less accumulation of dissolved methane gas in the liquid fraction.

For containment units of Type C, low methane emissions may be influenced by intermittent aeration; however, current data are insufficient to draw definitive conclusions. A preliminary analysis (Supporting Information, Section SI-12) suggests a potential effect of short periods of aeration (2 hours per day) as median methane NERs were significantly higher ($p = 9\times10^{-4}$) in containment units without aeration (3.40 g CH_4 cap^{-1} day^{-1}) compared to those with intermittent aeration in intermediate chambers (0.82 g CH_4 cap^{-1} day^{-1}) (see **Error! Reference source not found.** of the Supporting Information). Aeration creates localised oxidising zones, potentially lowering methane generation or promoting methane oxidation on the surface, but may also simply strip dissolved methane (Wang et al., 2011). Nevertheless, this finding should be interpreted with caution, as the limited number of aerated containment units ($n = 11$) reduces statistical power and constrains the robustness of the comparison. We also found that there were no statistically significant differences between single- and multichambered containment units (Wilcoxon test, $p = 0.402$, **Error! Reference source not found.**), but the sample size differed notably between the two data groups (34 vs 109). Further research based on larger sample sizes is therefore required to validate these observed trends.

3.7 Correlation between NERs and stored sludge volume

Four sludge-related parameters were compared with methane NERs in containment units with a single chamber and reported those results in Section SI-14, Supporting Information, including: sludge thickness (**Error! Reference source not found.**a), sludge volume (**Error! Reference source not found.**b), sludge thickness relative to total used depth (**Error! Reference source not found.**c), and sludge volume relative to total used volume (**Error! Reference source not found.**d). We observed a moderate correlation for sludge volume ($\rho = 0.481$, $p = 1.18\text{x}10^{-7}$) and weak and significant correlations for the other parameters (Pearson correlation coefficient: $0.295 < \rho < 0.339$). Despite the high variability in methane emission rates, higher sludge accumulation was generally associated with slightly increased methane emissions. Moonkawin and co-authors reported a linear correlation between methane emissions and sludge depth ($R = 0.596$, $p = 0.002$) with data from 25 septic tanks in Hanoi, Vietnam (Moonkawin et al., 2023). These results suggest that, in addition to sludge quantity, other containment characteristics and operational or maintenance factors likely contribute to variation in methane emissions in Indonesia.

3.8 Multivariate analysis

A Gamma GLMM with a log link was fitted to methane emission rates, with walls, base, outlet, and aeration as fixed effects, and the unique system name as a random intercept. The detailed results of this analysis are in the Supporting Information, Section SI-5.2. None of the typology variables used for this analysis was a statistically significant predictor of emissions. The most important practical effects on methane emissions were associated with the presence of an outlet and aeration; however, neither was statistically significant. Containment units with an outlet showed 73% lower emissions than those without an outlet (rate ratio = 0.27, 95% CI: 0.07–1.08), while containment units with aeration showed 70% lower emissions (rate ratio =

0.30, 95% CI: 0.07–1.36). The wall and base configurations showed weak, inconsistent associations. The random intercept exhibited substantial variability (SD = 0.82, 95% CI: 0.98–1.23 on the log scale), indicating that methane emissions differed markedly between containment units beyond the measured typology categories. This suggests that system-level factors not captured by simple typological classifications may have a more substantial influence on emissions, supporting caution in applying uniform emission factors across diverse on-site sanitation designs.

The intra-cluster correlation (ICC) indicated substantial clustering by system, with approximately 46% of the variance in methane emission rates attributable to differences between containment units (ICC ≈ 0.46). This large system-level contribution supports the inclusion of a random intercept for the system. It suggests that system-level factors not captured by the measured typology explain a substantial proportion of emissions.

These findings further indicate that containment units exhibit distinct behaviours that are not fully captured by simple typological descriptors, highlighting the need to be cautious against generalisation and the need for using more detailed, process- and household-level variables to improve predictive accuracy, such as desludging frequency, hydraulic retention time, sludge age, and organic matter loading rates, among others.

3.9 IPCC-guided emission rates

3.9.1 IPCC-guided vs empirical emission rates

IPCC-guided methane NERs from septic tanks and pit latrines were calculated and reported in Table 5, along with figures from this study for lined or sealed systems, with an outlet, and non-aerated (Types A, B, D & E) and for open pits (Type F), which can be used for direct comparison. Results from applying the IPCC methodology confirm that the methane NERs calculated by this approach align with our findings, since emission rates for Types A, B, D & E (comparable to septic tanks) are lower than for Type F (comparable to pit latrines). However, this approach overestimates methane emissions by a factor between 3.2 and 4.0 in containment units Type A, B, D & E (septic tanks) and by a factor between 1.8 and 5.3 for containment units Type F (pit latrines).

Table 3. Methane NERs for equivalent containment units in IPCC guidelines (IPCC, 2019)

IPCC guidelines		This work	
Type of unit	**Methane NERs* (g CH_4 cap^{-1} day^{-1})**	**Type of unit**	**Methane NERs** (g CH_4 cap^{-1} day^{-1})**
Septic tank (alone or with land dispersal field)***	7.35 – 15.13	Sealed and lined walls with outlet	2.32 – 3.75
Latrine (wet climate, high water table)	13.48 – 22.00	Open walls no outlet	2.83 – 34.26

* NER range values calculated based on recommended *MCF* and *EF* factors, see Table SI-3, supporting Information

** 95% confidence interval from empirical data

***The IPCC methodology assumes no CH_4 emissions from land dispersion fields

The overestimation of methane emission rates from the IPCC methodology (Tier 1 approach) could come from assuming that all the biodegradable organic matter (*TOW*) produced per capita per day goes to one single onsite sanitation containment unit. This is even less representative when considering people movement (from home to work or to study) and that most onsite containment units in our study only receive blackwater. This approach is perhaps more relevant in centralised sewage systems, where domestic wastewater is collected and transported, together with any commercial, institutional, and non-trade wastewater, to a single point for treatment. We also argue that septic tank emission rates can be affected by design and construction criteria; for instance, not all so-called ‘septic tanks’ in LMICs are comparable to prefabricated septic tanks used in the USA. The evidence for greenhouse gas (GHG) emissions from septic tanks in the IPCC Guidelines comes primarily from a limited set of experimental studies and field measurements, largely conducted in industrialised countries (Diaz-Valbuena et al., 2011; Leverenz et al., 2010; Truhlar et al., 2016).

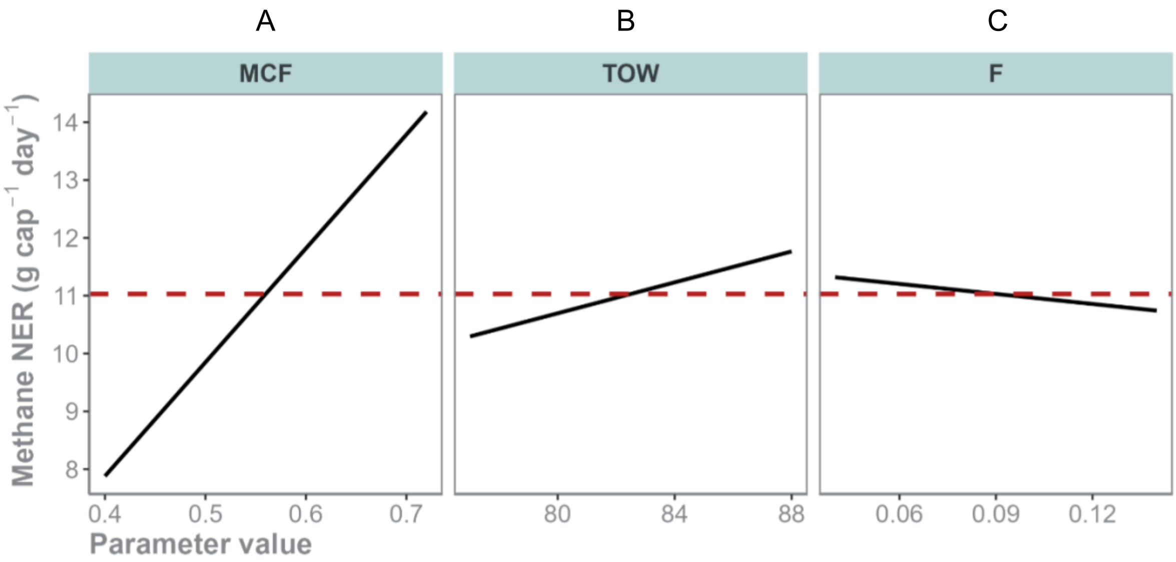
For pit latrines in particular, the IPCC methodology assumes that little or no water is used to flush excreta into the pit and that methane emissions occur when the water table is high, and the organic waste in the pit is submerged. This assumption leads to the use of higher methane conversion factors (0.7 < *MCFs* < *1.0*) for latrines in wet climates or where the groundwater table is higher than the latrine water level. However, that assumption does not account for the lower organic loading rates and higher dilution in the latrines when groundwater infiltrates the pit or when water is used for anal cleansing (i.e., a common practice in many LMICs, including Indonesia), leading to lower net methane emissions, as reported in this work. We also found no difference in NERs between the wet and dry seasons.

Discrepancies between IPCC-guided methane emissions highlight the need to increase the availability of empirical data from different typologies of sanitation containment units in LMICs to improve estimates of national GHG inventories.

3.9.2 Sensitivity analysis on IPCC-guided NER values

We conducted a one-at-a-time (OAT) sensitivity analysis to assess changes in IPCC-guided methane emission rates for septic tanks in response to variation in key independent variables: Methane Conversion Factor (*MCF*), Total Organic Waste contribution per person per day (*TOW*) and sludge removal fraction (*F*). The corresponding OAT response curves together with the range-based impact (ΔE) and relative percentage changes for *MCF*, *TOW*, and *F* are reported in Figure 6.

Across the evaluated parameter space, methane emissions were most sensitive to changes in *MCF* values, which induced the most considerable variation in model output, followed by *TOW*. In contrast, changes in *F* values made a comparatively minor influence. The dominance of *MCF* reflects its direct proportional role in the emission formulation, amplifying uncertainty linearly. *TOW* also scales emissions proportionally, but within a narrower relative range, resulting in a minor overall impact. By comparison, *F* values only modify the anaerobically biodegradable fraction of organic matter and therefore produces limited variation within the tested interval. However, Moonkawin et al. (2023) reported statistically significant linear correlations between methane NERs and desludging frequency in Vietnam, with the maximum rate approximately 11 times higher than the minimum, over a range of 4-23 years. In countries with large populations, such as Indonesia, small per capita differences may translate into significant differences in national methane inventories.



Rank	Parameter	ΔE	Percentage of change
1	MCF	6.30	57.14
2	TOW	1.47	13.33
3	F	-0.58	-5.24

Figure 6. Response curves for the OAT sensitivity analysis of three factors affecting methane NERs from septic tanks, according to the IPCC guidelines. Each panel shows the response to varying key variables: A) Methane conversion Factor (*MCF*), B) Total organic load (*TOW*), and C) Sludge removal fraction (F). The dashed line represents the central baseline methane emission rate.

The magnitude of *ΔE* indicates that uncertainty in *MCF* contributes disproportionately to overall model uncertainty. Consequently, improving empirical estimates of *MCF* values in onsite sanitation containment units would yield the most significant reduction in uncertainty in methane inventories developed under IPCC guidelines. Refining organic load estimates would provide secondary improvements.

4. Study limitations and future research

This study has several limitations that should be considered when interpreting the results. Sample sizes were small for some containment unit types, particularly the small number of multi-chamber aerated containment units, which may limit statistical power and the generalisability of the findings. Systems with difficult access points (e.g., sealed beneath thick concrete) or those with small openings (e.g., a 4-inch pipe opening, which is common in densely populated areas) were not sampled due to the requirement for a relatively large surface area for flux chamber deployment. However, if the system owner agreed, modifications were made to create a sufficiently large opening (~30 cm diameter), except for prefabricated systems, which could not be modified. Due to these same access challenges, we had to make assumptions in multichambered containment units to estimate emission rates from unsampled chambers. Also, we measured methane emissions only from chambers, not from infiltration units, which are a required element of lined or sealed containment units if they are to conform to the Indonesian standard SNI 2398:2017. The standard requires that 'septic tanks' be sealed units with two chambers and soil infiltration.

Finally, methane production in sanitation containment units is inherently heterogeneous and temporally variable; therefore, longer-term monitoring would be necessary to better capture the influence of environmental conditions, system characteristics, and operational factors. Future research should prioritise expanded and more balanced sampling across containment unit types and infiltration fields, account for dissolved methane in the liquid phase, incorporate targeted sampling under contrasting hydrological conditions, monitor groundwater levels, and extend observation periods to improve understanding of methane emission dynamics.

Acknowledgements

This work was conducted as part of the EMISI project, an initiative of the Ministry of Public Works of the Government of Indonesia supported by the Department of Foreign Affairs and Trade (DFAT) through KIAT, the Indonesia Australia Infrastructure Partnership. The research is delivered by a consortium of sanitation experts from the University of Technology Sydney (UTS), Universitas Indonesia (UI), Centre for Regulation, Policy and Governance (CRPG), University of Leeds (UoL), and the University of New South Wales (UNSW). We gratefully acknowledge Mullah Barokah (UI) for his essential contributions during field sampling, which were crucial to the successful execution of the project. We also thank the Indonesian Endowment Fund for Education (LPDP) on behalf of the Indonesian Ministry of Higher Education, Science and Technology and managed under the EQUITY Program (Contract No. 4302/B3/DT.03.08/2025 and 573/PKS/R/UI/2025) that supported the time and resources of the UI team. The wider research team is acknowledged and listed in Section SI-15 of the Supporting Information. The EMISI team is grateful for the opportunity to contribute to improved sanitation outcomes in Indonesia.

CRediT author statement

Conceptualization: BE, CRP, JK, JW, MACV; Data Curation: CMP, IR, AMI, DF, FR, IV, NDM, MACV; Formal Analysis: CMP, MACV; Funding Acquisition: BE, CRP, DF, JW, MACV; Investigation: CMP, IR, AMI, BSR, FR, IV, JD, MACV; Methodology: CMP, IR, AMI, BE, BSR, CRP, DF, JD, JW, NDM, MACV; Project Administration: BSR, DF, JD, JK; Resources: IR, AMI, DF, FR, IV, NDM; Supervision: BE, CRP, JW, MACV; Visualization: CMP, BSR, MACV; Writing – Original Draft: CMP, IR, AMI, BSR, JD, MACV; Writing – Review & Editing: All

Author's name abbreviation: Carolina Montoya Pachongo (CMP), Iftita Rahmatika (IR), Akna Mumtaz Ilmi (AMI), Barbara Evans (BE), Ben Smeaton-Russell (BSR), Cindy Rianti Priadi (CRP), Dinda Fauzani (DF), Farah Raihanah (FR), Isravani Valencia (IV), Jack Dalton (JD), Jeremy Kohlitz (JK), Juliet Willetts (JW), Nopa Dwi Maulidiany (NDM), Prayatni Soewondo (PS), Miller Alonso Camargo-Valero (MACV).

5 REFERENCES

Bao, P. N., Abfertiawan, M. S., Kumar, P., & Hakim, M. F. (2020). Challenges and Opportunities for Septage Management in the Urban Areas of Indonesia—Case Study in Bandung City. *Journal of*

Engineering and Technological Sciences, *52*(4), 481–500. https://doi.org/10.5614/j.eng.technol.sci.2020.52.4.3
Beck, H. E., Zimmermann, N. E., McVicar, T. R., Vergopolan, N., Berg, A., & Wood, E. F. (2018). Present and future Köppen-Geiger climate classification maps at 1-km resolution. *Scientific Data*, *5*(1), 180214. https://doi.org/10.1038/sdata.2018.214
Bolker, B. M., Brooks, M. E., Clark, C. J., Geange, S. W., Poulsen, J. R., Stevens, M. H. H., & White, J.-S. S. (2009). Generalized linear mixed models: A practical guide for ecology and evolution. *Trends in Ecology & Evolution*, *24*(3), 127–135. https://doi.org/10.1016/j.tree.2008.10.008
Botheju, D., & Bakke, R. (2011). Oxygen Effects in Anaerobic Digestion – A Review. *The Open Waste Management Journal*, *4*(1), 1–19. https://doi.org/10.2174/1876400201104010001
BPS-Statistics of Balikpapan Municipality. (2025). *Balikpapan Municipality in Figures 2025*. BPS-Statistics Balikpapan Municipality.
BPS-Statistics of Bandung Municipality. (2025). *Bandung Municipality in Figures 2025*. BPS-Statistics Bandung Municipality.
BPS-Statistics of DKI Jakarta Province. (2025). *DKI Jakarta Province in Figures 2025*. BPS-Statistics of DKI Jakarta Province.
BPS-Statistics of Manggarai Barat Regency. (2025). *Manggarai Barat Regency in Figures 2025*. BPS-Statistics of Manggarai Barat Regency.
BPS-Statistics of Manggarai Regency. (2025). *Manggarai in Figures 2025*. BPS-Statistics of Manggarai Regency.
Brooks, M. E., Kristensen, K., Benthem, K. J. van, Magnusson, A., Berg, C. W., Nielsen, A., Skaug, H. J., Mächler, M., & Bolker, B. M. (2017). glmmTMB Balances Speed and Flexibility Among Packages for Zero-inflated Generalized Linear Mixed Modeling. *The R Journal*, *9*(2), 378–400. https://doi.org/10.32614/RJ-2017-066
Camargo-Valero, M. A., Dalton, J., Faye, A., Ghimire, A., Geremew, A., Ngom, B., Okurut, K., Poudel, P., Reddy, O., Smeaton-Russell, B., Evans, B., & Howard, G. (2026). *What drives methane emissions from onsite sanitation containment units? Lessons from an empirical study in four countries*. Engineering Archive. https://doi.org/10.31224/6369
Cheng, S., Long, J., Evans, B., Zhan, Z., Li, T., Chen, C., Mang, H.-P., & Li, Z. (2022). Non-negligible greenhouse gas emissions from non-sewered sanitation systems: A meta-analysis. *Environmental Research*, *212*, 113468. https://doi.org/10.1016/j.envres.2022.113468
Diaz-Valbuena, L. R., Leverenz, H. L., Cappa, C. D., Tchobanoglous, G., Horwath, W. R., & Darby, J. L. (2011). Methane, Carbon Dioxide, and Nitrous Oxide Emissions from Septic Tank Systems. *Environmental Science & Technology*, *45*(7), 2741–2747. https://doi.org/10.1021/es1036095
Direktorat Data, D. K. (2025, December 1). *Data Online* [Online database]. Data Online Parameter Tersedia. https://dataonline.bmkg.go.id/dataonline-home
EPA. (2002). *Onsite Wastewater Treatment Systems Manual*. United States Environmental Protection Agency (US EPA).
EPA. (2022). *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2020* (EPA 430-R-22-003). https://www.epa.gov/system/files/documents/2022-04/us-ghg-inventory-2022-main-text.pdf

Gómez-Borraz, T. L., Cuthill, C., Herzyk, T., Connelly, S., & Sloan, W. (2025). *Temperature effect on performance and methane emissions of highly controlled replicate septic tanks (Preprint)*. EarthArXiv. https://eartharxiv.org/repository/view/11305/

Hartig, F. (2024). *DHARMa: Residual diagnostics for hierarchical (multi-level/mixed) regression models* [Computer software]. University of Regensburg. https://cran.r-project.org/web/packages/DHARMa/vignettes/DHARMa.html

Huynh, L. T., Harada, H., Fujii, S., Nguyen, L. P. H., Hoang, T.-H. T., & Huynh, H. T. (2021). Greenhouse Gas Emissions from Blackwater Septic Systems. *Environmental Science & Technology*, *55*(2), 1209–1217. https://doi.org/10.1021/acs.est.0c03418

Ilmi, A. M., Evans, B., Smeaton-Russell, B., Montoya-Pachongo, C., Priadi, C. R., Fauzani, D., Raihanah, F., Rahmatika, I., Valencia, I., Dalton, J., Kohlitz, J., Willetts, J., Camargo-Valero, M. A., Maulidiany, N. D., & Soewondo, P. (2026). *EMISI project—32 Sanitation containment units—Two seasons Dataset* [Dataset]. Zenodo. https://doi.org/10.5281/zenodo.18792536

(Indonesia Australia Infrastructure Partnership), K., Ardani, F., Auliarahma, F., Barokah, M., Camargo Valero, M., Casey, L., Chenny, W., Dalton, J., Danisha, D., Dwi Maulidiany, N., Duan, H., Eukarin, C., Evans, B., Fathia, R., Fauzani, D., Immanuela, K., Kohlitz, J., Medina Valenzuela, A., Mills, F., … Zhai, M. (2026). *Achieving safely managed services and climate mitigation in the sanitation sector in Indonesia* (p. 5364008 Bytes). University of Technology Sydney. https://doi.org/10.71741/4PYXMBNJAQ.31895149

Indonesia Meteorology, Climatology and Geophysics Agency. (2026). *Cuaca BMKG*. https://weather.bmkg.go.id/

IPCC. (2019). Chapter 6. Wastewater Treatment and Discharge. In *Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*. https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/5_Volume5/19R_V5_6_Ch06_Wastewater.pdf

IPCC. (2022). *Working Group III contribution to the Sixth Assessment report*. Intergovernmental Panel on Climate Change (IPCC).

Knappe, J., Somlai, C., & Gill, L. W. (2022). Assessing the spatial and temporal variability of greenhouse gas emissions from different configurations of on-site wastewater treatment system using discrete and continuous gas flux measurement. *Biogeosciences*, *19*(4), 1067–1085. https://doi.org/10.5194/bg-19-1067-2022

Lambiasi, L., Ddiba, D., Andersson, K., Parvage, M., & Dickin, S. (2024). Greenhouse gas emissions from sanitation and wastewater management systems: A review. *Journal of Water and Climate Change*, *15*(4), 1797–1819. https://doi.org/10.2166/wcc.2024.603

Leverenz, H. L., Tchobanoglous, G., & Darby, J. L. (2010). *Evaluation of Greenhouse Gas Emissions from Septic Systems. Final report* (DEC1R09). IWA Publishing. https://doi.org/https://doi.org/10.2166/9781780403359

Loo, Y. Y., Billa, L., & Singh, A. (2015). Effect of climate change on seasonal monsoon in Asia and its impact on the variability of monsoon rainfall in Southeast Asia. *Geoscience Frontiers*, *6*(6), 817–823. https://doi.org/10.1016/j.gsf.2014.02.009

Manga, M., & Muoghalu, C. C. (2024). Greenhouse gas emissions from on-site sanitation systems: A systematic review and meta-analysis of emission rates, formation pathways and influencing factors.

Journal of Environmental Management, *357*, 120736. https://doi.org/10.1016/j.jenvman.2024.120736

Ministry of Environment and Forestry (Indonesia), General Directorate of Climate Change Control, & Directorate of GHG Inventory and MVP. (2024). *Laporan Inventarisasi Gas Rumah Kaca (Grk) Dan Monitoring, Pelaporan, Verifikasi (MPV)* (Volume 9, p. 152). https://kemenlh.go.id/dashboard/public/storage/reports/XmPwBrjHiWx6I9Lw6MBvXE3psmGrS3DwAq9PY8uL.pdf

Moonkawin, J., Huynh, L. T., Schneider, M. Y., Fujii, S., Echigo, S., Nguyen, L. P. H., Hoang, T.-H. T., Huynh, H. T., & Harada, H. (2023). Challenges to Accurate Estimation of Methane Emission from Septic Tanks with Long Emptying Intervals. *Environmental Science & Technology*, *57*(43), 16575–16584. https://doi.org/10.1021/acs.est.3c05724

Moonkawin, J., Schneider, M. Y., Fujii, S., Yasui, H., Nguyen, V.-A., Pham, A. N., Echigo, S., & Harada, H. (2026). Impact of GHG mitigation measures in sanitation service chains: A focus on septic tanks and sewers. *Water Research*, *288*, 124618. https://doi.org/10.1016/j.watres.2025.124618

Nguyen, D., Wu, Z., Shrestha, S., Lee, P.-H., Raskin, L., & Khanal, S. K. (2019). Intermittent micro-aeration: New strategy to control volatile fatty acid accumulation in high organic loading anaerobic digestion. *Water Research*, *166*, 115080. https://doi.org/10.1016/j.watres.2019.115080

Odagiri, M., Thomas, A., Listyasari, M., Mills, F., Bain, R. E. S., Muhammad, Z., Slaymaker, T., Mardikanto, A., Gultom, A., Indiyani, A., Rangkuti, H., & Willetts, J. (2021). Safely Managed On-Site Sanitation: A National Assessment of Sanitation Services and Potential Fecal Exposure in Indonesia. *International Journal of Environmental Research and Public Health*, *18*(15), 8204. https://doi.org/10.3390/ijerph18158204

Oshita, K., Okumura, T., Takaoka, M., Fujimori, T., Appels, L., & Dewil, R. (2014). Methane and nitrous oxide emissions following anaerobic digestion of sludge in Japanese sewage treatment facilities. *Bioresource Technology*, *171*, 175–181. https://doi.org/10.1016/j.biortech.2014.08.081

Poudel, P., Ghimire, A., Howard, G., Evans, B., Camargo-Valero, M. A., Mills, F., Reddy, O., Sharma, S., Tuladhar, S., Geremew, A., Okurut, K., Ngom, B., Baidya, M., & Dangol, S. (2023). Field-based methods for measuring greenhouse gases emissions from on-site sanitation systems: A systematic review of published literature. *Heliyon*, *9*(9), e19947. https://doi.org/10.1016/j.heliyon.2023.e19947

Saltelli, A., Ratto, M., Andres, T., Campolongo, F., Cariboni, J., Gatelli, D., Saisana, M., & Tarantola, S. (2008). *Global Sensitivity Analysis: The Primer* (1st ed.). Wiley-Interscience. https://www.perlego.com/book/2768780/global-sensitivity-analysis-the-primer

Short, M. D., Daikeler, A., Wallis, K., Peirson, W. L., & Peters, G. M. (2017). Dissolved methane in the influent of three Australian wastewater treatment plants fed by gravity sewers. *Science of The Total Environment*, *599–600*, 85–93. https://doi.org/10.1016/j.scitotenv.2017.04.152

Smeaton-Russell, B., Dalton, J., Montoya Pachongo, C., Evans, B., Priadi, C. R., Willetts, J., & Camargo-Valero, M. A. (2026). *Sampling and Health, Safety and Environment Standard Operating Procedures for GHG emissions measurements from On-Site and Small-scale Sewered Sanitation Systems*. https://doi.org/10.31224/6728

Somlai, C., Knappe, J., & Gill, L. (2019). Spatial and temporal variation of CO2 and CH4 emissions from a septic tank soakaway. *Science of The Total Environment*, *679*, 185–195. https://doi.org/10.1016/j.scitotenv.2019.04.449

Somlai-Haase, C., Knappe, J., & Gill, L. (2017). Carbon dioxide emissions from a septic tank soakaway in a northern maritime climate. *Science of The Total Environment*, *586*, 485–491. https://doi.org/10.1016/j.scitotenv.2017.01.206

Strande, L., Evans, B., von Sperling, M., Bartram, J., Harada, H., Nakagiri, A., & Nguyen, V.-A. (2023). Urban Sanitation: New Terminology for Globally Relevant Solutions? *Environmental Science & Technology*, *57*(42), 15771–15779. https://doi.org/10.1021/acs.est.3c04431

Supari, Tangang, F., Juneng, L., & Aldrian, E. (2017). Observed changes in extreme temperature and precipitation over Indonesia. *International Journal of Climatology*, *37*(4), 1979–1997. https://doi.org/10.1002/joc.4829

Truhlar, A. M., Ortega, K. L., & Walter, M. T. (2019). Seasonal and diel variation in greenhouse gas emissions from septic system leach fields. *International Journal of Environmental Science and Technology*, *16*(10), 6043–6052. https://doi.org/10.1007/s13762-019-02314-6

Truhlar, A. M., Rahm, B. G., Brooks, R. A., Nadeau, S. A., Makarsky, E. T., & Walter, M. T. (2016). Greenhouse Gas Emissions from Septic Systems in New York State. *Journal of Environmental Quality*, *45*(4), 1153–1160. https://doi.org/10.2134/jeq2015.09.0478

UTS-ISF, UI and UNICEF. (2021). *Climate Resilient Urban Sanitation in Indonesia: Hazards, impacts and responses in four cities*. Institute for Sustainable Futures, University of Technology Sydney.

Wang, J., Zhang, J., Xie, H., Qi, P., Ren, Y., & Hu, Z. (2011). Methane emissions from a full-scale A/A/O wastewater treatment plant. *Bioresource Technology, Special Issue on Challenges in Environmental Science and Engineering, CESE-2010: Technological Advances in Waste Treatment for a Sustainable Future*, *102*(9), 5479–5485. https://doi.org/10.1016/j.biortech.2010.10.090

WHO & UNICEF. (2025). *Progress on household drinking water, sanitation and hygiene 2000–2024: Special focus on inequalities*. World Health Organization (WHO) and the United Nations Children's Fund (UNICEF).

Yamamoto, S., Alcauskas, J. B., & Crozier, T. E. (1976). Solubility of methane in distilled water and seawater. *Journal of Chemical & Engineering Data*, *21*(1), 78–80. https://doi.org/10.1021/je60068a029

Zuur, A. F., Leno, E. N., Walker, N., Saveliev, A. A., & Smith, G. M. (2009). *Mixed effects models and extensions in ecology with R* (1st ed.). Springer. https://doi.org/10.1007/978-0-387-87458-6