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What drives methane emissions from sanitation containment systems? Lessons from an empirical study in four countries

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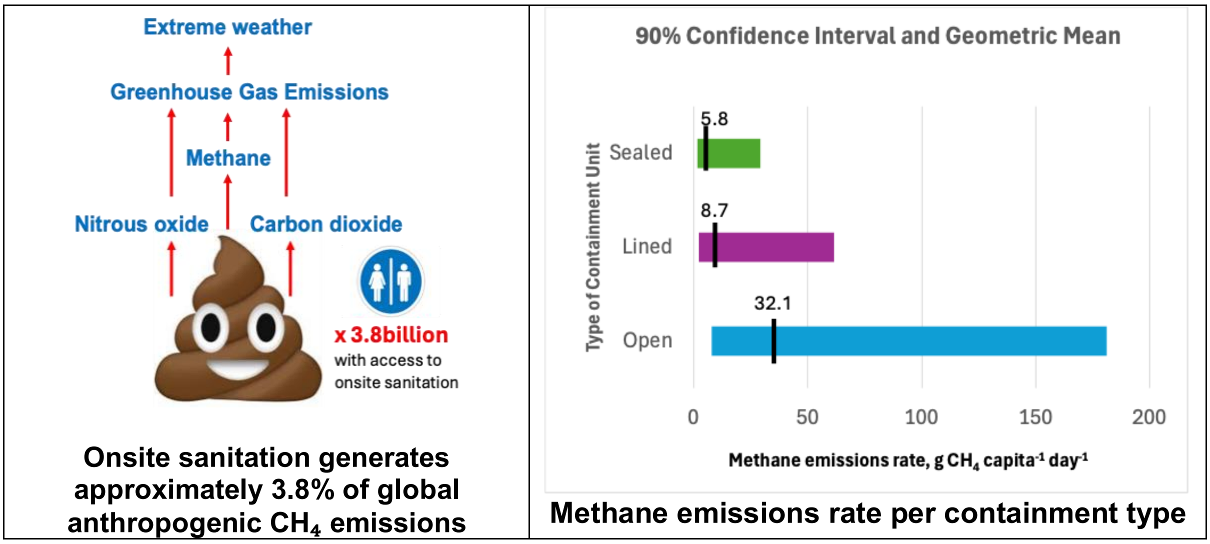
Graphic abstract



Onsite sanitation generates approximately 3.8% of global anthropogenic CH$_4$ emissions

Methane emissions rate per containment type

Abstract

Onsite sanitation coverage has expanded significantly since 2020, driven by global commitments to eliminate open defecation and the need for adaptable solutions in rapidly urbanising small- and medium-sized cities (<1 million inhabitants) in Low- and Middle-Income Countries. Currently, 53% of individuals with access to a toilet depend on onsite sanitation systems. Despite this prevalence, the contribution of greenhouse gas emissions from household-level excreta storage to climate change remains poorly quantified due to limited empirical evidence. We addressed this gap by conducting direct measurements of methane emissions from 146 onsite sanitation containment units locally referred as pit latrines, holding tanks and septic tanks, across Senegal, Ethiopia, Uganda and Nepal. Methane emission rates exhibited strong skewness with a geometric mean of 7.9 g CH_4 capita^{-1} day^{-1}, indicating that onsite sanitation containment units alone may account for approximately 3.8% of global anthropogenic CH_4 emissions.

Key words: containment unit, greenhouse gas, methane, onsite sanitation, SDG6

1 Introduction

Access to sewered sanitation systems is not the global norm. Where sewerage is not provided households and communities construct and use containment units, which collect and store excreta close to the household. This approach, often termed ‘onsite sanitation’ can be an effective method for containing human excreta, preventing users from immediate contact and controlling the spread of waterborne diseases. Onsite sanitation systems are designed to collect and store excreta (blackwater containing feaces and urine) and/or domestic wastewater (a mix of blackwater and greywater from kitchen, showers, etc.) at the point of production (usually the house), where some partial waste stabilisation occur in-situ. Storage in some continents is temporary before emptying. Solids and liquid accumulated inside containment units in the form of feacal sludge, anal cleansing material, etc., are removed as part of regular emptying practices for further treatment, disposal and reuse at a properly designed and operated treatment site, as part of a comprehensive safely managed onsite sanitation system.

The global population served by onsite sanitation is increasing rapidly, especially in low- and middle-income countries (LMICs) as a consequence of global targets to eradicate open defecation. In 2024, 47% of the global population had access to on-site sanitation and 42% to sewered sanitation; although a much smaller fraction had access to safely managed sanitation (i.e., 26% for onsite sanitation and 33% for sewered sanitation) (WHO and UNICEF, 2025). The rate of use of onsite sanitation has been growing particularly fast in small- and medium-size cities (<1 million inhabitants) in LMICs, as rapid urbanisation demands flexible and adaptive solutions that can be implemented incrementally to meet the increasing demand (Greene et al., 2021). In East and South-East Asia, one of the few regions on track to achieve universal access to sanitation by 2030, safely managed sanitation increased from 19.8% in 2020 to 64% in 2024, with significant contributions from Indonesia where access to onsite sanitation increase from 54.5% in 2000 to 94.0% in 2024 (i.e., sewer based sanitatioin accounts for less than 1% in indonesia) (WHO and UNICEF, 2025).

Along with the importance of improving coverage of safely managed onsite sanitation services, there is a growing concern for assessing their contribution to global greenhouse gas (GHG) emissions. Human excreta produce biogenic emissions of GHGs including carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) through biological processes leading to the stabilisation of organic matter, either within onsite sanitation containment units or at faecal sludge treatment facilities. Additional CO_2 emissions occur through feacal sludge transport to treatment/disposal points due to the use of fossil fuels for trucking faecal sludge and in the use of energy in treatment facilities derived from fossil fuels (Johnson et al., 2022). To date, the focus of estimating direct and indirect GHG emissions from sanitation has been mainly on sewered systems with centralised wastewater treatment plants using modelling tools, with few empirical data available (Baj et al., 2022; Wu et al., 2022). Current estimates indicate that over 1.5% of global GHG emissions can be attributed to the stabilisation of organic matter at wastewater management systems, with approximately a third of that (0.5% of global emissions) linked to non-CO_2 emissions, including the release of CH_4 and N_2O (Dickin *et al.*, 2020).

In contrast, the entire non-sewered category of sanitation is estimated to contribute 4.7% of global anthropogenic methane emissions (Cheng et al., 2022), of which nearly 20% is attributable to ‘pit latrines’ representing close to 1% of global anthropogenic methane emissions (Reid *et al.* 2014). These estimations do not use empirical emission data, instead they are based on high-resolution geospatial analysis, including data on water table depth, and combined with region-specific biochemical oxygen demand (BOD) contribution per person to calculate the corresponding methane emission factors (EFs). Such approaches follow the methodology recommended by the 2006 guidelines from the Intergovernmental Panel on Climate Change - IPCC (Doorn et al., 2006), and are supported by laboratory experiments analysing chemical oxygen demand (COD), total solids (TS), volatile solids (VS) and faecal anaerobic digestion experiments (Biometane potential (BMP) tests). In summary, the majority of reported estimates include a long list of assumptions and do not use empirical GHG emission data directly collected from pits or tanks and hence intrinsic differences linked to onsite sanitation technologies and environmental and operational conditions are not captured. This may partially explain the broad uncertainty range (0.3–12.5%) for the global estimate of anthropogenic methane emissions from non-sewered systems reported by Cheng et al (2022).

Due to increased dependence and the widespread nature of onsite sanitation, there is, however, an increasing, but limited amount of GHG emission data from sanitation containment units being reported, using both theoretical and/or field-based measurements (Diaz-Valbuena *et al.* 2011; Reid *et al.* 2014; Truhlar *et al.* 2016; Ryals *et al.* 2019; Somlai et al., 2019; IPCC, 2019; Huynh *et al.* 2021, Moonkawin *et al.*, 2023). But in particular, it has been consistently reported that ‘pit latrines’ are a significant source of CH_4 emissions (Couderc *et al.* 2008; Reid *et al.* 2014; Kulak *et al.* 2017; van Eekert *et al.* 2019).

The paucity of empirical data on GHG emissions from onsite containment units is aggravated by the lack of uniform design criteria and poor construction practices along with the widespread use of vague and non-standardised terminology to describe such systems (Strande et al., 2023). This means that the limited empirical data that do exist are difficult to extrapolate. In fact, onsite sanitation is managed in a multitude of ways across different countries, and terminology is very unclear. In general, a distinction is made between basic sanitation units which are unlined (open), and those that are lined or sealed and which are respectively broadly described as ‘pit latrines’ and ‘septic tanks’ (Strande et al., 2023). Due to the uncertainty around the design of onsite sanitation technologies, developing a means of quantifying emissions from such a varied group has been difficult. To date, most research on quantifying GHG emissions from onsite sanitation in LMICs has come from a small sample of so-called ‘septic tanks’ whose specific designs and operation conditions are not always explicit or sampling is limited to only one chamber (Somlai *et al* 2019; Huynh *et al.* 2021; Moonkawin et al., 2023).

In addition, it is important to consider that the collection of empirical data to assess GHG emissions in LMICs is not straightforward. The collection of field data requires the use of (a) specialist apparatus to capture emitted gas samples (flux chambers); (b) analytical capacity to measure gas concentrations; (c) data processing expertise to calculate gas fluxes and emissions rates and (d) well trained personnel able to conduct fieldwork, sample analysis and data processing. Flux chambers (FCs) are defined as an enclosed volume over a surface area that allows the collection and sampling of GHGs that are to be measured and quantified by reliable analytical methods (Eklund, 1992). In that sense, they should be part of a simple, flexible and accurate combination of field-, lab- and desk-based methodologies to quantify GHG emission rates. To date, static (closed or passive) and dynamic (flow-through or active) FCs have predominantly been used in studies looking at GHG emissions from soils (Heinemeyer and McNamara, 2011), landfills (Reinhart, 1992), and natural and engineered aquatic ecosystems – i.e., lakes, wastewater treatment systems, etc. (Duc et al., 2013; Silva et al., 2015).

The static FC measuring technique allows the collection of gas samples from the gas mix confined within a known headspace volume placed immediately above the water or soil surface for a short period of time (i.e., typically 20–60 min) and for later analysis (Smith and Conen, 2004). In contrast, the dynamic FC measuring technique allows gases to pass through the FC in a continuous mode, for that reason it requires a gas flux meter that measures the corresponding flowrate or a pumping system delivering a constant flowrate through the chamber’s headspace. In both cases (static and dynamic FC measuring techniques), additional equipment is required to withdraw gas samples from the headspace for the analysis of GHG concentrations either in the lab or in situ.

In published literature, collected gas samples are commonly transported and processed in the lab by gas chromatography (GC) using FID (Flame ionization detector), EDC (Electron Capture Detector) or TCD (Thermal Conductivity Detectors) detectors. Alternatively for in-situ analysis, optical techniques including non-dispersive infrared spectroscopy (NDIR), Fourier-transform infrared spectroscopy (FTIR), photoacoustic spectroscopy (PAS), tunable laser absorption spectroscopy (TLAS), cavity ring-down spectroscopy (CRDS), or off-axis integrated cavity-output spectroscopy (OA-ICOS) are used for measuring GHG concentrations in the field (Zaman et al., 2021). The use of dynamic FCs coupled with in-situ gas optical analysers can produce continuous GHG emissions data and reduce equipment and staff costs, when compared with a lab-based GC, but the gas flow rate needs to be fast and stable enough so it can ensure well mixing conditions and the capacity to carry the emitted gases to the gas detector, under conditions close to continuous steady state (Lambert and Fréchette, 2005). Such conditions as are very difficul to set in a low-income environment. Instead, and although the use of FCs coupled with either lab-based or in-situ GC analysis only quantifies intermittent GHG emissions as collected data comes from discrete time intervals, the static FC mehod is more suitable for LMICs. Also, data processing for the calculation of GHG emissions relies on the actual FC's configuration and operation conditions. For the static FC, the gas flux is calculated from the rate of increase of GHG concentration over time within the chamber headspace; for the dynamic FC, gas fluxes are calculated from gas mix flow rate and GHG concentration data using a mass balance method (Lambert and Fréchette, 2005; Zaman et al., 2021).

The limited empirical GHG emissions available from onsite sanitation units are based on the use of static FCs, mainly tested on septic tanks in the USA (8 pre-fabricated septic tanks) and Vietnam (25 septic tanks) (Poudel et al., 2023). Published work using this technique also reports the collection of gas samples for lab analysis using GC to determine the concentration of GHGs within the sample (Leverenz et al., 2010; Diaz-Valbuena *et al.* 2011; Huynh *et al.* 2021; Moonkawin et al., 2023). However, while gas samples can be collected successfully, the use of expensive analytical laboratory equipment creates travel times and sample number limitations to the production of field data, as well as constraining access to researchers in resource limited and distant rural locations.

There is therefore a lack of empirical data on GHG emissions from on-site sanitation published to date, particularly from LMICs. In contrast, purely theoretical estimates of GHG emissions from onsite sanitation are widely reported but frequently appear to report values that are higher than comparable field measurements (Leverenz et al., 2010). For example, the estimated IPCC figure for CH_4 emissions from septic tanks is 25.5 g CH_4 capita^{-1} day^{-1}, compared to 10.70, 11.0 and 11.29 g CH_4 capita^{-1} day^{-1} reported from direct measurements from septic tanks receiving domestic wastewater in the USA (Leveranz et al., 2010; Diaz-Valvuena et al., 2011) and blackwater in Vietnan (Huynh *et al.* 2021; Moonkawin et al., 2023), respectively. Even though there exist some variations in the actual GHG emissions reported, based on existing literature, including both theoretical estimates and direct field measurements, it is evident that GHG emissions from onsite sanitation are not negligible and hence, the imperative need to improve the currently available data set to determine the contribution that onsite sanitation makes to changes in global climate.

Overcoming the current research gaps requires strengthening empirical data collection through reliable and practical field methodologies that are affordable and reproducible for communities in low- and middle-income countries (LMICs). In this context, this study presents empirical data on methane emissions from a wide range of onsite containment units in Senegal, Nepal, Uganda and Ethiopia, using field methods co-developed, tested, and cross-validated by research groups in each country under local conditions. These methods were designed to provide a comprehensive understanding of the complex interactions between sanitation and climate change by generating robust, site-specific data and using low-cost equipment. The ultimate aim is to reduce the high uncertainty in existing literature on GHG emissions from onsite sanitation. Data produced using these approaches can support countries in more accurately accounting for emissions from onsite sanitation systems in their nationally determined contributions (NDCs)

2. Results and Discussion

2.1. Sampling site description and containment unit typology

In Senegal, sanitation coverage reaches 79.7%, with onsite systems accounting for 70.4% of services; containment units described as 'septic tanks' (41.4%) and improved 'pit latrines' (29.0%) are the predominant options (WHO and UNICEF, 2025). Although national standards for the design and management of onsite containment units exist (Standard NS 17-074, Association Sénégalaise de Normalisation – ASN, 2021), implementation is inconsistent. For example, "septic tanks" vary widely in design, featuring one to three chambers and, in some cases, lacking effluent outlets. Sampling sites in Tivaouane, Thiès, and Kaolack were selected to capture the diversity of sanitation practices, population mobility influenced by cultural and religious dynamics, hydrogeological vulnerability, and exposure to climate variability, particularly flood risk. Two wastewater flow types were identified in the sampled containment units: (a) blackwater (56%) from toilet discharges and (b) mixed water (44%), combining blackwater with greywater from kitchens, showers and other sources. No units managed greywater exclusively. This distribution underscores the strong reliance on toilet-connected waste streams, which typically exhibit high organic and microbial loads, increasing the likelihood of anaerobic conditions and GHG production. The significant proportion of mixed wastewater also reflects limited segregation practices, which can compromise treatment efficiency by increasing dilution, flow rates and reducing retention times, while limiting opportunities for greywater reuse, which is particularly critical in arid and semi-arid regions. Except for one site, all sampled containment units had lined walls, indicating user efforts to improve structural stability and longevity. However, none were fully sealed, raising concerns about infiltration and environmental contamination, especially in flood-prone areas and regions with high groundwater tables.

In Uganda, sanitation services provide at least improved sanitation to 42.1% of the population, mainly by the delivery of onsite sanitation services (41.3%), with containment units described as improved 'pit latrines' (39.0%) and 'septic tanks' (2.3%) as the preferred options (WHO and UNICEF, 2025). National sanitation and hygiene guidelines (Ministry of Health, 2017) and minimum standards for onsite sanitation in Kampala (KCCA, 2020), present a comprehensive description of local onsite sanitation technologies and good practices, but they do not provide standardised designed criteria for onsite containment units, which explains the wide variation of technical specifications found in the containment units selected in Uganda. Sampling sites for GHG emission measurements were selected from low-income informal settlements of two urban areas, Kampala and Gulu, experiencing slightly differently climate scenarios. In Kampala, the parishes of Banda and Mbuya were selected and are located partly along the Kinawaka wetland that drains off the city into Lake Victoria at Luzira with high groundwater table, and partly on the lower sides of Mbuya and Kyambogo hills respectively. In Gulu, the parishes of Kirombe and Kasubi were selected and are both largely flat with patches of wetland and streams flowing through the settlements. The areas closer to the streams are prone to flooding, more especially in Kasubi. All containment units were randomly selected, in proportion to the number of households in the parishes in each city.

In Ethiopia, 18.9% of the population has access to at least improved sanitation services, mainly provided by onsite sanitation (18.2%), with containment units described as improved ‘pit latrines’ (16.1%) and ‘septic tanks’ (2.1%) as the preferred options (WHO and UNICEF, 2025). The design, operation and maintenance of onsite sanitation services are governed by the Ethiopian Building Code Standard for Plumbing Services of Buildings – EBCS-9 (Ministry of Urban and Development and construction, 2013), but the compliance with such building codes is limited to the planning permission stage. For instance, removal of septage and faecal sludge is recommended annually but we found that some containment units have never been emptied after many years of operation. Onsite containment units described as ‘pit latrines’ in Ethiopia from Harar (15) and Dire Dawa (4), showed no difference in terms of their construction methods and materials used. They were unlined and unsealed, allowing liquid waste to seep into the surrounding soil. The slabs covering these ‘pit latrines’ were made of either concrete cement or wood coated with mud or cement, and the superstructures were constructed from bricks, steel or other locally available materials. These contrasted with so called ‘holding tanks’ (5 from Dire Dawa and 4 from Harar), which were permanent onsite sanitation facilities made up of durable, water-tight reinforced concrete, which are fully lined or sealed to prevent the infiltration of the liquid into the surrounding soil or groundwater into the tanks. But the main issues in the study area are related to frequent filling rate due to groundwater infiltration and long emptying frequency in others due to poor maintenance, despite national building codes include design criteria considering relevant waste production rates and maximum sludge accumulation before emptying.

In Nepal, 98.0% of the population has access to at least improved sanitation services, mainly provided by onsite sanitation (94%), with containment units described as ‘septic tanks’ (55.4%) and improved ‘pit latrines’ (38.6%) as the preferred options (WHO and UNICEF, 2025). There are national efforts focusing on standardising design criteria for onsite containment units, including septic tanks and (twin) pit latrines (Ministry of Water Supply, 2021). Selected sampling sites represent the urban/rural and topographic characteristics in Nepal including (a) lowlands – Ratnanagar Municipality (12 containment units), (b) midlands – Dhulikhel Municipality (12), and (c) highlands – Bethanchowk Rural Municipality (6). The municipalities of Bethanchowk and Dhulikhel are semi urban areas but are not densely populated. In the lowland regions, most containment units are ring ‘pit latrines’ and ‘holding tanks’; these containment units are often inundated by groundwater, resulting in more diluted faecal sludge. In contrast, containment units in the midland and highland regions are typically termed as ‘pit latrines’ and are made of rock and mud or rings. Groundwater inundation is lower in these areas and have more limited effect on the condition of the faecal sludge inside containment units. Out of the total 30 containment units selected in Nepal, 3 were sealed with outlets (referred to as ‘septic tanks’ in Nepal), 18 were not sealed (termed ‘pit latrines’) and 9 had sealed walls but were open at the bottom and had no outlet (usually termed ‘holding tanks’).

In general, the onsite containment units included in this study often deviate from standard designs primarily due to a combination of financial constraints, lack of awareness and enforcement of regulations, use of untrained personnel and site-specific environmental and operational challenges. Observations at our sampling sites suggest a weak correlation between what a structure is locally termed and its design and performance, which is a critical lack of technical depth needed to differentiate between safe and unsafe containment units without standardised indicators. For that reason we produced a generic coding system to better identify onsite sanitation containment units (Table 1).

Table 1. Container characteristic code to describe onsite sanitation units*

Element	Component	Code	Description	Notes
Nature of Influent		G	Greywater only	No excreta enter the containment
		B	Blackwater only	Excreta and anal cleansing water only enter containment
		M	Mixed black and greywater	Excreta, plus anal cleansing water, domestic wastewater from washing, cleaning and cooking all enter the containment
		F	Mixed black and greywater plus significant other flows	Other flows could include waste from domestic manufacturing/ farming etc
		X	Unknown	
Structure	Walls	O	Open	No lining
		L	Lined	Lining is present but allows liquid ingress/ exit, for example semipermeable membrane, bamboo, stones, honeycomb brick.
		S	Sealed	The lining is of concrete or plastered masonry/ similar
		X	Unknown	
	Bottom	O	Open	No lining
		L	Lined	Lining is present but allows liquid ingress/ exit, for example semipermeable membrane, bamboo, stones, honeycomb brick.
		S	Sealed	The lining is of concrete or plastered masonry/ similar
		X	Unknown	
	Top	O	Open	For example, where manholes are broken or absent
		C	Closed	For example, manhole covers in place but not mortared or otherwise sealed
		S	Sealed	Good seal around all joints on the top. Can be the case even when a vent pipe is in place (see below)
Physical Features	Number of Chambers in Series	Integer	Total	Total number of chambers (T) in series as influent flows through the containment, and number of chambers with aeration (A) = T/A.
		integer	Aerated	
	Vent Pipe	V	A vent pipe is present	A vent pipe needs to be capable of carrying gases from one of the chambers into the atmosphere. To use 'y' here the vent pipe must be present in at least one chamber of the containment, the lower end is open and located either above or just under the surface of the contents of the containment. Do not count vent pipes if you cannot locate the lower end (use X) or if the lower end is not inside a chamber or is closed, or if the upper end is closed (use N).
		N	No vent pipe	
		X	Unknown	
	Outlet	O	Outlet	Here an outlet enables *outflow* of contents from the containment to a pipe, soakaway, open ground or open waterbody. It will usually comprise a short pipe. Outlets do not include points where liquid can infiltrate out through walls or bottoms.
		N	No outlet	
		X	Unknown	
Scale	Number of Users	Integer	Estimated total daily users	A rough estimate is needed here – note if more than one household use the containment or people from outside the household *regularly* have access they should be counted. Count each person only once.
	Volume	Three-digit number	Internal volume in m^3	Calculate from internal measurements to nearest integer.

*Full site descriptions available from Reddy et al. (2025) (https://doi.org/10.5281/zenodo.16531507)

This coding system helps to define key characteristics of containment units and describes every single individual site as fully as possible. For instance, an onsite continment unit with the code M-SOS-N4/1NO-(010-008) refers to a unit that receives a mix of blackwater and grey water (M); has a sealed top (S), an open bottom (O) and sealed walls (S), but has no filter media (N); has four chambers, one of them aerated (4/1); has no ventilation pipe (N); has an outlet (O); serves ten users (010); and has a volume of 8m^3 (008).

This coding system was used to identify the containment units included in this study and the full description has already been made available through an open access repository at Zenodo (https://zenodo.org) (Reddy et al., 2025).

2.2. Environmental characteristics inside containment units

Lab results from samples collected from all containment units confirmed environmental conditions suitable to support anaerobic digestion (see Table 2). In particular, redox potential (ORP), pH and temperature were within reported ranges suitable for methane production under mesophilic conditions (Nguyen et al., 2019). All containment units received either blackwater or a mix of blackwater and greywater that provide balanced nutrients (COD/N/P) to support a series of biological processes occurring inside containment units under anaerobic conditions (hydrolysis, acetogenesis and methanogenesis).

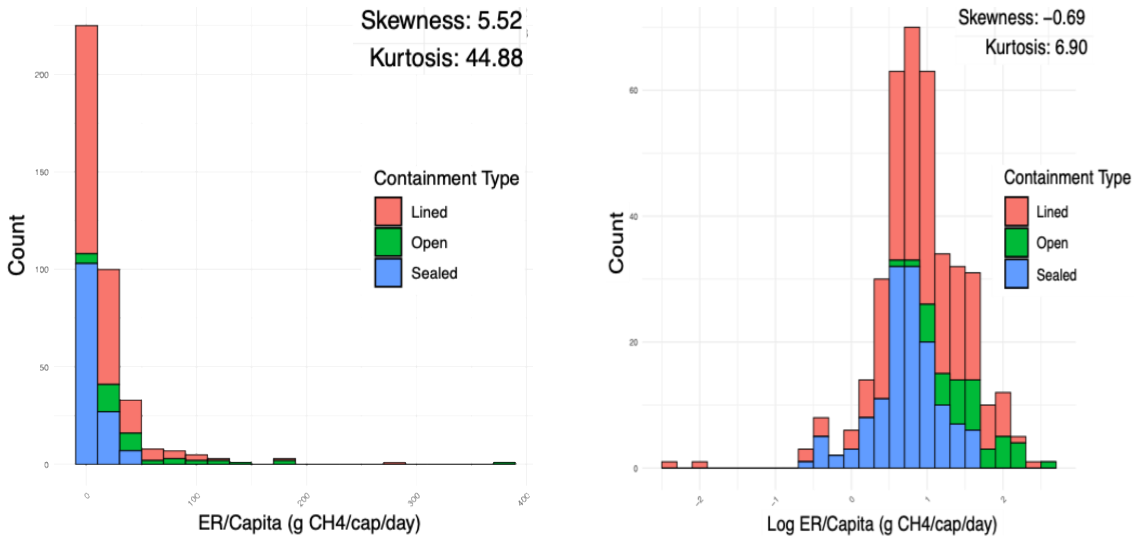
Table 2. Environmental characteristics inside containment units*

Country	**ORP (mV)**	**T (ºC)**	**pH**	**Sludge volume** (%)**	**COD (g L^{-1})**
Nepal	-217 ± 105	21.7 ± 5.4	7.5 ± 0.8	52%	30.89 ± 21.92
Ethiopia	86 ± 162	23.1 ± 3.1	7.0 ± 0.5	17%	0.375 ± 0.21
Senegal	-27 ± 27	29.4 ± 3.2	7.5 ± 0.5	23%	2.77 ± 9.13
Uganda	557 ± 518	27.7 ± 11.8	7.0 ± 0.4	10%	-----

*Mean values
**Volume of sludge as a percentage of the entire volume of the containment units

2.3. Methane emission rates from onsite sanitation containment units

Methane emissions rates (ER) were calculated from direct measurements in the field and reported in grams of methane emitted per day from each onsite sanitation containment unit included in this work and normalised against the nominal number of users per household (ER = g CH_4 capita^{-1} day^{-1}). Methane emissions rates were initially processed to assess whether our dataset was likely to be drawn from a normal distribution by using the Kolmogorov-Smirnov Test (*K-S*) and the Shapiro-Wilk (*S-W*) test. The value of the resulting *K-S* statistic *D* was 0.2943, which indicates that the difference between the sample data and the normal distribution is large (p = 0). This was confirmed by a *S-W* statistic *W* equal to 0.4697, which is not in the 95% region of acceptance (p = 0). Based on that, there is enough evidence to conclude that the original dataset deviates significantly from a normal distribution. In addition, the corresponding histogram (Figure 1a) and results from the *K-S* test for skewness (5.52), confirms asymmetry with data positively skewed indicating that there are more values clustered towards the lower end of the ER data range. A similar trend was reported by Leverenz *et al.* (2010) and Diaz-Valbuena *et al.* (2011) from GHG emission data collected at septic tanks in the USA, which defines the most suitable set of statistical tools to be used for data processing and analysis. Unfortunately, this initial step is often ignored as it is the case of results reported by Moonkawin *et al.* (2023). Indeed, we re-processed their original data and found that reported methane emissions are not normally distributed (Moonkawin *et al.,* 2023; n = 15, $p_{S\text{-}W}$ = 0.00300; $p_{K\text{-}S}$ = 0.00448), but despite that statistical tools assuming a normal distribution were used, which affects reported mean gas emission figures.



(a) (b)

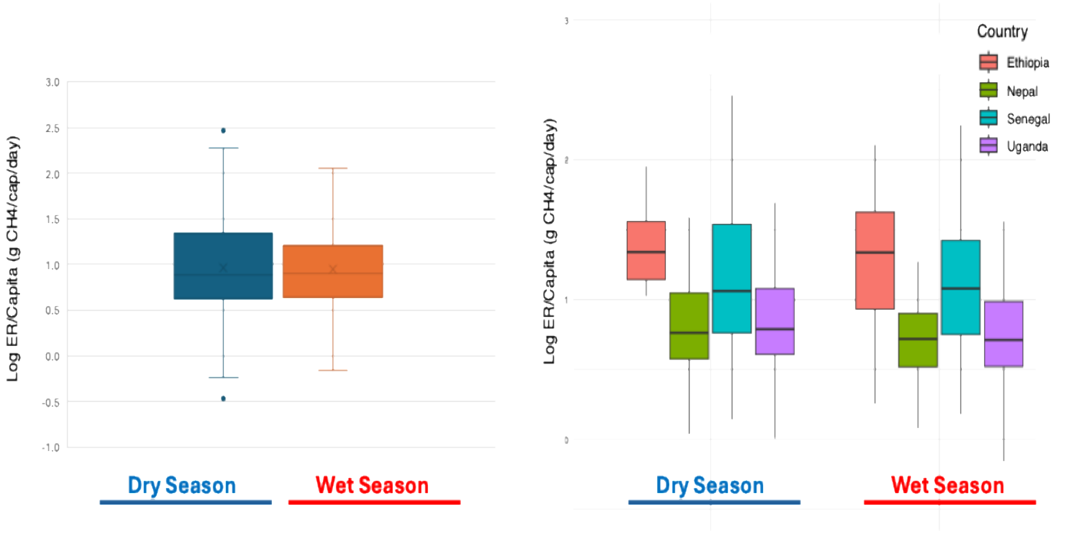
Figure 1. Methane emissions rate (ER) in grams of methane per person per day. Data include lined (red, n = 209), open (green, n = 41) and sealed (blue, n = 137) onsite sanitation containment units. All collected data is plotted using (a) arithmetic and (b) logarithmic scales to illustrate issues with skewness and normality.

In order to conduct a more robust statistical analysis, a log-transformation technique was used by applying a based 10 logarithm to each data point to help to address skewness and other deviations of the dataset from a normal distribution – i.e., this technique makes non-normal data more normally distributed. The corresponding histogram of all $\log_{10}$-transformed data (n = 387) confirmed a reduction in asymmetry and hence, a better alignment with a normal distribution (Figure 1b). Based on that, $\log_{10}$-transformed data was processed to remove outliers by using the Tukey's method based on the interquartile rage (IQR) – i.e., data points that fall outside of 1.5 times the IQR below Quartile 1 (Q1) or above Quartile 3 (Q3) are considered as outliers. The remaining data after removing outliers (n = 374) was reprocessed using the *K-S* and *S-W* tests for normality and for descriptive statistics.
The $\log_{10}$-transformed data for methane emissions rate values follows a normal-like distribution (*K-S* test; $p = 0.13799$). The corresponding histogram and Kernel density analysis suggested a unimodal distribution, which was confirmed by running a Hartigan's dip test (Dip statistic = 0.0; $p = 1.0$). This means we can assume that all methane emissions data belongs to the same dataset and can confidently represent methane emission rates from selected onsite sanitation containment units in the four countries.

Overall methane emission rates from onsite sanitation containment units were with a geometric mean of 7.9 g CH_4 capita^{-1} day^{-1} (80 kgCO_2 equivalent (e) capita^{-1} year^{-1}; based on a methane global warming potential of 28, over a 100-year period) and geometric standard deviation (*GSD*) of 3.0. The range equivalent to one *GSD* around the geometric mean value (68% of the data) falls between 2.6 and 23.7 g CH_4 capita^{-1} day^{-1} (30 – 240 kgCO_2e capita^{-1} year^{-1}). The middle 90% of emissions data (between the 5th and 95th percentiles) ranges from 1.7 to 65.7 g CH_4 capita^{-1} day^{-1} (20 – 670 kgCO_2e capita^{-1} year^{-1}). This information can be used to model and reproduce our data set for further independent analysis. The ranges of methane emissions reported here from onsite sanitation containment units acknowledge the wide range of technologies used in practice and the lack of standarised criteria for design, construction and operation of such units. For that reason, the following sections will analyse the impact of environmental conditions and containment unit typology on methane emisisions.

2.4. Impact of weather conditions on methane emissions per country.

Local weather conditions have a marked effect on the operation of onsite sanitation containments particularly in areas affected by a high-water table during periods of heavy rain. In Bangladesh, rainfall driven by the Southwest monsoon leads to a rise in the water table that disturbs the operation of onsite sanitation units causing pits and tanks to fill with groundwater, leading to overflow and service disruption (Evans et al., 2015). Based on those conditions, the IPCC methodology suggests that increased water content can enhance hydrolysis inside pit latrines leading to higher methane production and hence, based on expert judgment by the Lead Authors, it makes distinction between methane correction factors (MCFs) for pit latrines depending on the climate conditions (dry and wet) and groundwater table levels (higher or lower than water level inside the pit) (IPCC, 2019).



(a) (b)

Figure 2. Box plots of log-transformed methane emission rate (ER) data by season and country. This includes: (a) data from all onsite sanitation containment units tested during dry (*n* = 171) and wet (*n* = 201) seasons and (b) a comparison per season by country.

Taking that into consideration, we decided to assess the impact of dry and wet weather seasons on methane emissions. Log_{10}-transformed data from all surveys conducted during dry and wet seasons were independently processed to remove outliers by using the Tukey's method. Resulting data were used to compared dry (*n* = 171) and wet (*n* = 201) seasons for statistical significance using the Student's *t* test. By conventional criteria (resulting two-tailed *p* = 0.7415), there was no significant statistical difference between methane emission rates from onsite containment units surveyed during dry and wet seasons (Figure 2a). Same conclusion was reached when comparing in-country dry and wet season data (See Figure 2b).

Deeper statistical analysis of in-country data revealed that the prevalent typology of the onsite sanitation containment units in a country is a more influential factor when assessing their impact on methane emission rates (Figure 2b). For instance, data from Uganda and Nepal (with no open containments) were lower and significantly different from data collected in Ethiopia, the country with the largest data from open containments, both for dry and wet seasons. In fact, a direct comparison between dry and wet seasons for log10-transformed methane emissions rates from open containment units in Ethiopia (*t*-test) showed no significant statistical difference between mean values (p = 0.9466). Also, the other group of containment units in Ethiopia, locally refered as 'holding tanks', are commonly used to serve block of flats and handle mix water in large volume tanks (29.9 to 346.7 m^3) and higher numbers of nominal users (60 to 232 people).

2.5 Methane emission rates and containment typology

Preliminary ER data analysis suggested that the main factors influencing methane emissions were related to walls construction characteristics and hence, onsite sanitation containment units from all countries were grouped according to wall containment typology: open, lined and sealed (Table 3).

Table 3. Number of containment units depending on walls construction characteristics

Containment typology	**Country**				**Total**
	Ethiopia	**Uganda**	**Senegal**	**Nepal**	
Open	19		1		20
Lined		17	36	21	74
Sealed	9	44		9	62

Consequently, methane emissions rates ($\log_{10}$-transformed data) from the 387 sampling surveys were divided in the same three categories: lined (n = 209), open (n = 41) and sealed (n = 137) and compared using a one-way ANOVA test (Figure 3). As a result, there was a significant statistical difference between geometric mean values when comparing open vs. lined (p = 0), open vs. sealed (p = 0) and sealed vs. lined (p = 0.000193), which confirms that the actual design and construction of the onsite sanitation containment units, which affect operation and maintenance practices (e.g., emptying frequency), are highly influential factors with open containment units producing higher methane emissions than lined containers and sealed containment units producing the lowest methane emissions measured as part of this study.

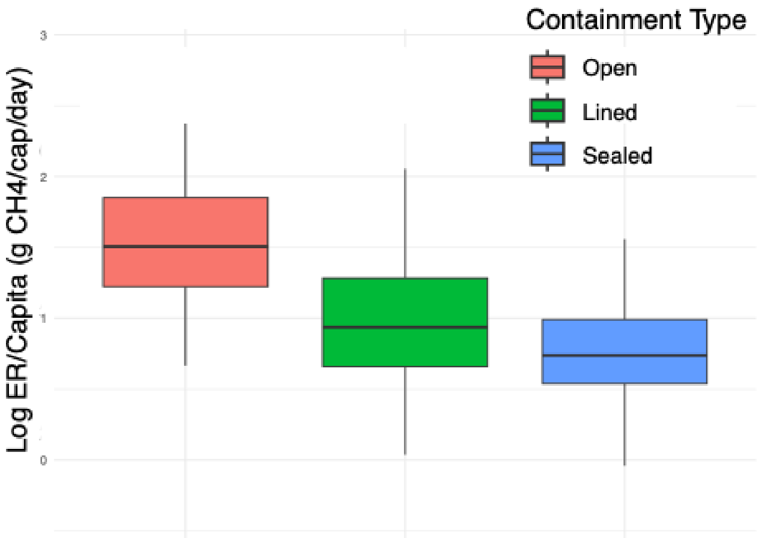


Figure 3. Log-transformed methane emission rate (ER) data by containment type. This figure includes data from all onsite sanitation containment units including open (orange, n = 41), lined (green, n = 200), and sealed (blue, n = 128) onsite sanitation systems.

Methane emission rates from open containment units (n = 41) are reported with a geometric mean of 32.1 g CH_4 capita^{-1} day^{-1} (898.8 g CO_2 e capita^{-1} day^{-1}) and a geometric standard deviation (*GSD*) of 2.8. The middle 90% of emissions data (between the 5th and 95th percentiles) ranges from 8.0 to 181.2 g CH_4 capita^{-1} day^{-1} (224.0 – 5,073.6 g CO_2 equivalent capita^{-1} day^{-1}). The range equivalent to one *GSD* around the geometric mean value (68% of the data) falls between 11.5 and 89.3 g CH_4 capita^{-1} day^{-1} (322.0 – 2,500.4 g CO_2 equivalent capita^{-1} day^{-1}). For sealed and lined containment units, the geometric mean was 5.8 and 8.7 g CH_4 capita^{-1} day^{-1} (243.6 and 162.4 g CO_2 equivalent capita^{-1} day^{-1}), respectively (Table 4).

Table 4. Methane emissions rates per containment typology

Variable	**Methane emissions rate, g CH_4 $capita^{-1}$ day^{-1}**		
	Open	**Lined**	**Sealed**
Geometric Mean (*X*)	32.1	8.7	5.8
GSD	2.8	1.5	2.2
X + *GSD*	89.3	13.5	13.0
X - *GSD*	11.5	5.6	2.6
X_{95}	181.2	61.4	29.2
X_5	8.0	2.2	1.6
n	41	200	128

GSD = geometric standard deviation; *X* = geometric mean; X_{95} = 95th percentile; X_5 = 5th percentile; *n* = sample size.

2.6 Comparisson with reported empirical methane emissions

The very few empirical data currently available prevent a comprehensive analysis of all our data. However, an initial analysis can be drawn from sealed containment units with an effluent in this study (48 sampling sites) and septic tanks (published literature). The range of minimum and maximum methane emissions rates reported for septic tanks in the USA by Diaz-Valbuena *et al*. (2011) (0.07 – 75.69 g CH_4 $capita^{-1}$ day^{-1}, *n* = 39), and in Vietnam by Huynh *et al.* (2021) (4.42 – 18.79 g CH_4 $capita^{-1}$ day^{-1}, *n* = 10) and Moonkawin *et al.* (2023) (2.23 – 46.38 g CH_4 $capita^{-1}$ day^{-1}, *n* = 15;) are within the range of emission rates found in this study for sealed containment units receiving blackwater and with an effluent (0.30 – 49.26 g CH_4 $capita^{-1}$ day^{-1}, *n* = 95). However, after running a non-parametric test for mean comparison (Kurstall-Wallis H test), our data (mean = 5.33 g CH_4 $capita^{-1}$ day^{-1}) are significantly different from figures reported from the USA (mean = 10.70 g CH_4 $capita^{-1}$ day^{-1}, *p* = 0.01345; Diaz-Valbuena et al., 2011) and from Vietnam (mean = 11.29 g CH_4 $capita^{-1}$ day^{-1}, *p* = 0.00609; Huynh *et al.,* 2021; Moonkawin *et al.,* 2023), which may be influenced by differences in organic matter discharge at the household level and dilution. Based on the IPCC methodology, the methane emission rate for a septic tank is 25 g CH_4 $capita^{-1}$ day^{-1}.

Anaerobic environmental conditions inside onsite sanitation containment units are triggered by high content of bioderadable organic matter leading to low redox potential, near neutral pH and limited oxygen transfer (Nakagiry et al., 2017; Wanda et al., 2021). These anaerobic microenvironments favour the production of methane and carbon dioxide as gaseous products from anaerobic biological degradation, which is related to the actual volumentric organic loading rate at which the containment unit operates – i.e., a diluted influent will produce less methane and a shorter hydraulic retention time. In such settings, although methane generation is expected, yet what determines atmospheric release is not net methane conversion alone but how methane gas partitions among sludge, liquid, scum and container's headspace. This partitioning is dynamic and governed mainly by changes in organic loading and temperature influencing net methane gas production, solubility and mass transfer from the liquid to the atmosphere. Broader wastewater studies and inventory guidance emphasise precisely these controls, highlighting that process understanding must couple biogenic production with phase behaviour when interpreting emissions from onsite sanitation containment units (IPCC, 2019).

A mass balance approach explains why even containers receiving organic waste with similar biological methane potentials can yield different net methane gas emissions. In containment units with low water content, methane saturation in the liquid phase progresses much faster, promoting the formation of metane bubbles and vertical methane transport through the sludge-water column, resulting in higher direct atmospheric emissions (e.g., containers receiving blackwater alone with no anal cleansing wastewater). By contrast, when influent is more diluted (e.g., blackwater mixed with greywater or blackwater mixed with anal cleansing wastewater), the larger liquid volume offers dilution of organic matter concentration and greater capacity to dissolve methane in the liquid fraction, moderating the headspace burden and reducing immediate gaseous release. This liquid phase buffering is consistent with observations in anaerobic wastewater treatments, where dissolved methane can reach concentrations of up to 25 mg L^{-1} (at 15°C), underscoring that solubility can temporarily sequester a significant fraction of produced methane (Stazi and Tomei, 2021). This may explain why the IPCC methodology ofter overestimate methanane emissions, considering the great emphasis on the use of higher methane conversion factors for containment units in wet climates or in locations with a high water table.

Onsite sanitation containment units with an effluent (often termed ‘septic tanks’), add a further element in the mass balance as soluble methane is exported outside the containment unit within the effluent discharge, reducing net atmospheric emissions. In fact, it has been reported that 30 to 40% of the produced methane are kept dissolved in the liquid effluent of anaerobic digesters in tropical countries like Brazil (Chernicharo et al., 2015). This highlights the important need to better understand the impact of the downstream fate of effluent, including transport in open drainage, or direct discharge to surface waters, when compared, for example, to further treatment in polishing units and final disposal through land infiltration.

Additional factors like turbulence, aeration and temperature will govern methane desorption from the liquid and bio-oxidation by methanotrophs will reduce the methene budget and net emissions. For instance, effluents from prefabricated septic tanks will produce very low/negligible methane emissions if disposed in infiltration fields (Leverenz et al., 2010; Diaz-Valbuena, et al., 2021; Truhlar et al., 2016), while discharges into lentic water bodies will promote localised hypoxia and eutrophication due to further degradation of the remaining organic matter and increased nutrient concentration, leading to additional GHG emissions. Accounting frameworks therefore need to track both gaseous and dissolved pathways to avoid under or over attributing methane emissions to the containment unit itself. Process level studies at centralised plants reach similar conclusions, showing substantial dissolved methane sinks and transformations that can either add to or abate net flux depending on design and operation.

New controlled experiments with full scale replicate septic tanks in temperate climate countries sharpen this analysis by resolving temperature dependent partitioning. Under cooler conditions (≤ 20°C) in Scotland, often more than 80% of the methane generated in septic tanks remained in solution and left the tank within the effluent, while at 30°C desorption was favoured and roughly 70% accumulated in the headspace (Gomez-Borraz, et al., 2025). These results demonstrate that soluble losses through effluent can dominate the methane balance in containment units with an effluent discharge for significant portions of the year depending on prevalent climate conditions, thereby producing lower direct emissions at the containment unit itself even when methanogenesis is active. Therefore, as temperature rises, the balance flips toward greater headspace accumulation and higher on container methane fluxes, making seasonal and geographic context critical to both emissions' measurement and mitigation.

Sanitation is critical for reducing public health risks associated with pathogens in human excreta; mitigating environmental impacts from untreated wastewater and faecal sludge storage and disposal; and enabling resource recovery and reuse. Open defecation poses the greatest health risk, and progress along the sanitation value chain inevitably increases GHG emissions from the sector (Johnson et al., 2022). Our data demonstrates that storage of blackwater and greywater in onsite sanitation containment units fosters anaerobic conditions, leading to methane production at an estimated average rate of 7.9 g CH_4 capita^{-1} day^{-1}. Considering the numbers of people using such systems, this could account for approximately 3.8% of all global anthropogenic CH_4 emissions, before emissions from effluent are considered.

Based on our findings, mitigation strategies should prioritise upgrading existing facilities by transitioning from open containment units (32.1 g CH_4 capita^{-1} day^{-1}) to lined (8.7 g CH_4 capita^{-1} day^{-1}) or sealed units (5.8 g CH_4 capita^{-1} day^{-1}). Improved operation and maintenance of faecal sludge management, including frequent emptying, can further reduce emissions, as methane generation in septic tanks correlates positively with emptying intervals and sludge depth (Moonkawin et al., 2023). Promoting frequent emptying will, however, require increased treatment capacity at existing faecal sludge management facilities but also opens up opportunities for capture and reuse of biogas. Also, there are further economic considerations that will need to be addressed if more frequent emptying is promoted.

Additional improvements towards net-zero emissions in the sanitation sector requires a holistic approach that integrates climate-resilient sanitation services with resource recovery strategies, such as: (a) in-situ conversion of methane to biogenic CO_2 through methane capture and flaring or energy recovery for cooking, cooling or heating; (b) implementing nature-based solutions for faecal sludge and septage management that enhance carbon capture (e.g., constructed wetlands, wastewater ponds and algal systems); and (c) recycling nutrients into food and energy crops by safely reusing treated wastewater and stabilised dry faecal sludge, thereby reducing fossil fuel use in industrial fertiliser production and contributing to global net-zero targets.

4. Methods

4.1. Sampling sites selection and characterisation

This study was conducted in selected sites in Senegal (Tivaouane, Thiès, and Kaolack), Ethiopia (Harar and Dire Dawa), Uganda (Kampala and Gulu) and Nepal (Ratnanagar, Dhulikhel and Bethanchowk). Our dataset contains 146 sampling sites in total. Within the project teams 58 of these onsite systems were referred to as ‘septic tanks’, 50 referred to as ‘holding tanks’ and 38 referred to as ‘pit latrines’ (Table 5). This multi-country approach allowed for comparative analysis of emission patterns across different geographical and climatic contexts.

Table 5. Classification of containment units based on local definitions.

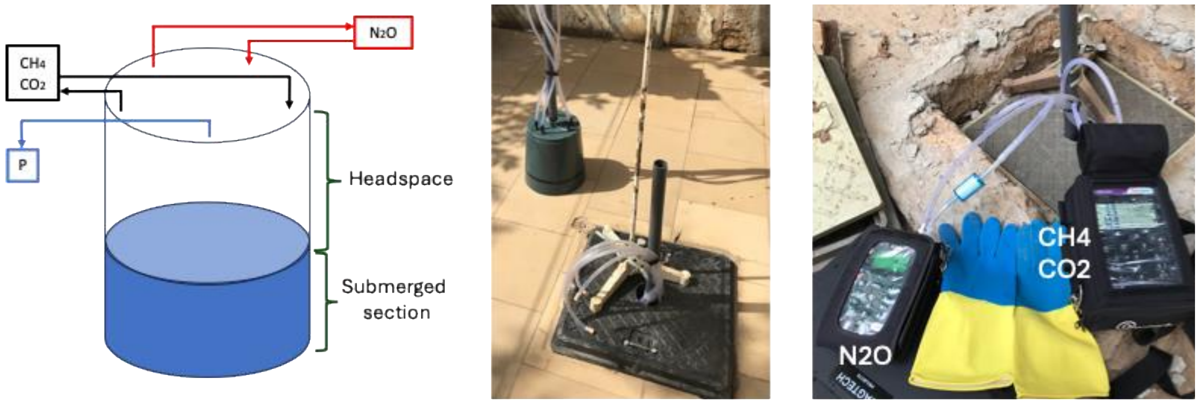
Local definition	Senegal	Uganda	Ethiopia	Nepal	Total
Pit latrine	1	---	19	18	38
Holding tank	16	16	9	9	50
Septic tank	20	35	---	3	58
Total	37	51	28	30	146

4.2. Faecal sludge sampling and characterisation

Representative samples from all containment units (water column and sludge layer) were collected to determine prevalent onsite environmental conditions to support anaerobic processes. Temperature, pH, redox potential (ORP), electric conductivity (EC) and salinity (reported as mg of total dissolved solids per litre – TDS) were measured in situ using a multi-parameter probe (Hydro Check HC1000, UK). Collected sludge samples were transported to the lab and processed for chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN) and total ammoniacal nitrogen (NH_4^+) following standard protocols for the characterisation of sludge samples (APHA, 2017).

4.2. Gas emission rate measurements

The in-situ measurement of GHG emission rates was conducted using a static flux chamber method adapted from Díaz-Valbuena et al. (2011) (see Figure 4). The flux chamber consists of a rigid body constructed from inert materials, such as high-density polyethylene (HDPE), polypropylene (PP), polyvinyl chloride (PVC) or fiberglass, to prevent gas leakage. Chamber dimensions varied by site (internal diameter: 150–300 mm; headspace volume: 5 – 44 L), depending on the size of the inspection manhole for tanks or slab opening and pit depth for latrines. The chamber is equipped with five ports on the top, to connect monitoring equipment via PTFE tubing: one for a pressure gauge and two for each gas analyser, enabling sample collection and recirculation into the headspace using the analysers' internal gas pumps (total flow rate: 600 mL·min^{-1} during sampling). This configuration eliminates the need for battery-powered fans to ensure internal mixing (as employed by Díaz-Valbuena et al., 2011 and Huynh et al., 2021) and avoids transporting gas samples to a laboratory for analysis (Figure 4a).



(a) Static flux chamber set up
(b) Static flux chamber onsite
(c) Gas analysers

Figure 8. Static flux chamber used for assessing GHG emissions from onsite sanitation containment units. (a) Static chamber diagram with ports for gas sample collection and pressure measurements; (b) Static flux chamber used on site and its deployment in a "holding tank" through an inspection manhole in Senegal; (c) Portable field gas analysers.

In addition, the flux chamber was equipped with a PVC pipe (1.5-inch diameter, variable length) to ensure secure placement within the onsite sanitation containment unit. This was achieved by using either a tripod with a rope pulley or, when the inspection manhole could be covered, a clamp stand (Figure 4b). In terms of monitoring equipment, this method repurposes portable landfill gas analysers (GeoTech GA5000, QED Environmental Systems Ltd., UK) for measuring CH_4 (0-70% v/v; ± 0.5%) and CO_2 (0-60% v/v; ± 0.5%) concentrations in the headspace, along with a portable indoor N_2O analysers (0-100ppm; ± 5ppm; GeoTech G200, QED Environmental Systems Ltd., UK) – See Figure 4c. A digital manometer was also installed to monitor changes in headspace pressure (300–1200 mbar; ± 3.0 mbar; Testo 511; Testo, UK) and I-button sensors were attached inside the flux chamber to record temperature in the headspace (DS1922L Thermochron Logger, Measurement Systems Ltd, UK).

The flux chamber was placed inside each containment unit and CH_4, CO_2, N_2O, temperature, and absolute pressure readings were recorded every 10-15 min during each sampling test; Sampling tests were conducted in triplicate and lasted for 2.5 to 3.0 hours, both during dry and wet seasons per sampling location.

4.3. Data processing and analysis

It is worth mentioning at this stage that N_2O data analysis confirmed that the gas analyser selected for this study (GeoTech G200) did not provide reliable data due to high uncertainty at low ppm readings and hence, N_2O data is not reported. CO_2 data was used to monitor the integrity of the static chamber method, as the biological degradation of organic matter will always result in CO_2 emissions, regardless of methane production. Gas concentration data collected in the field was analysed by linear plot methods that included three basic calculation steps. Firstly, gas concentration readings in percentage concentration were converted to milligrams per cubic meter (mg/m^3) parts per million (ppm) units. Here, the concentration of CH_4 was initially converted into ppm units by simply multiplying by 10,000. The concentration of the gases is converted into $mg \cdot m^{-3}$ concentrations by using the following Equation 1.

$$Gas\ concentration\left(\frac{mg}{m3}\right) = (Cppm/10^{6})(MW)(1000mg/g)/(\frac{RT}{P}) \qquad \text{Equation 1}$$

where:

C_{ppm} = concentration of gas in ppm
MW = molecular weight of the gas under consideration (g mol^{-1})
R = universal gas constant (0.000082057 atm m^{3} mol^{-1} K^{-1})
T = absolute sampling temperature (K)
P = absolute sampling pressure (atm)

Gas concentration (mg m^{-3}) values were plotted against time (min). The slope m (mg m^{-3} min^{-1}) is the gas accumulation rate in the headspace derived from a linear fit of field data. This is used to compute the flux gas emission rate per nominal user using the following Equation 2:

$$ER = \frac{m*1.44*10^{6}*V_{FC}*A_{Comp}}{A_{FC}*N} \qquad \text{Equation 2}$$

where,

ER = gas emission rate per nominal user (g CH_4 capita^{-1} day^{-1})
m = gas production rate (mg m^{-3} min^{-1})
1.44 x 10^{6} = factor to convert minutes into days and mg into g (min g mg^{-1} day^{-1})
V_{FC} = chamber's headspace volume (m^{3})
A_{Comp} = surface area of the compartment in the containment unit (m^{2})
A_{FC} = surface area covered by the floating flux chamber (m^{2})
N = nominal number of users per containment unit

Statistical analysis

The Kolmogorov-Smirnov test (*K-S*) and the Shapiro-Wilk (*S-W*) test were employed to assess the normality of the data. If the datasets did not follow a normal distribution, a log transformation was applied to normalise the emissions data. After transformation, parametric (*t*-test, ANOVA, and Pearson correlation) and non-parametric (Kruskal Wallis Test) tests were conducted normalised and non-normalised data, respectively. For descriptive statistics, outliers were removed by using the Tukey's method based on the interquartile rage (IQR), before assessing mean values, standard deviation, confidence intervals, etc., to help with data analysis and interpretation.

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CRediT authorship contribution statement

Conceptualisation: GH, BE, MACV; Experimental design: MACV, BE, AGh, AGe, BN, KO, OR; Lab and fieldwork: AF, AGh, Age, BN, KO, PP, OR, MACV; Supervision: GH, BEE, AGh, AGe, BN, KO, MACV; Data curation and processing: MACV, OR, PP, AGe, BN, KO, JD, BSR, BE; Writing – original draft: MACV, BE, JD, OR; Writing – review and editing: All; Funding acquisition: GH, BEE, AGh, AGe, BN, KO.

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Competing interests

The authors declare no competing interests.

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