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Research article

Performance evaluation of a full-scale upflow anaerobic sludge blanket reactor coupled with trickling filters for municipal wastewater treatment in a developing country

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ABSTRACT

Poor wastewater management remains a critical health and environmental challenge in most developing countries in Sub-Saharan Africa due to the lack of adequate infrastructure for collection and treatment. This study evaluated the performance and methane production of a full-scale upflow anaerobic sludge blanket (UASB) reactor of capacity 18000 m^3/d, with post-treatment unit: trickling filters followed by final settling tanks for municipal wastewater treatment in Ghana. Data was collected on operational conditions and physicochemical parameters of wastewater (influent and effluent) over a period of 35 weeks in 2021 (from January to August). The influent biochemical oxygen demand to chemical oxygen demand (BOD:COD) ratio was 0.58 $\pm$ 0.16, indicating the presence of highly biodegradable compounds in the sewage. Operational conditions for the UASB reactors were observed to be within the optimal range for anaerobic systems, with an applied organic loading rate of 1.30 $\pm$ 0.79 kgCOD/m^3/d. Generally, Plant performance was satisfactory with carbon removal at 93% for COD and 98% for BOD. Biogas yield was 0.2 m^3/kgCOD removed, culminating in an average biogas production rate of 831.6 $\pm$ 292.7 m^3/d. Average methane composition was 64.7 $\pm$ 11.9% of the biogas output, whilst an estimated 35% of the methane generated remained dissolved in the UASB effluent. The UASB reactor presents an efficient technology that can be implemented in developing countries for effective and sustainable wastewater management.

1. Introduction

Wastewater management is one of the major challenges most developing countries face in Sub-Saharan Africa (SSA) [1]. Accelerated population growth, industrialisation, and urbanisation have led to the generation of large volumes of wastewater which are often discharged indiscriminately into the environment due to the lack of adequate infrastructure for wastewater collection and treatment [1]. Meanwhile, untreated wastewater contains contaminants, including pathogens that are harmful to public health and the receiving ecosystems [2]. Notwithstanding threats from wastewater, its rich organic matter and nutrients could be harnessed as useful resources through energy recovery from biogas, plant nutrients from compost/fertilizer and water reuse for irrigation [3]. These resource recoveries are critical for sustainable wastewater treatment systems especially under modern concepts of eco-friendly technology and circular economy [4].

Conventional wastewater treatment technologies based on activated sludge process implemented in high-income countries are usually not suitable for low-income countries due to several factors including high installation and operational costs, despite their reliable treatment capacity and effluent quality [5]. Biological wastewater treatment with anaerobic digestion (AD) seem a promising alternative due to the lower or no energy consumption, operational simplicity, and ability to treat high organic load wastewater [6]. Moreover, anaerobic wastewater

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treatment (AnWT) comes with additional advantages such as suitability for warm climates, methane recovery from biogas, bio-fertiliser recovery, and less sludge production compared to aerobic processes, making AnWT an efficient and economically feasible technology that can be implemented in developing countries for effective and sustainable wastewater management [6]. AnWT technologies usually implemented include UASB reactors, anaerobic filters, fluidised bed reactors, rotating biological contactors, expanded granular sludge beds, waste stabilisation pond (WSPs), etc.

According to Mara [7], WSPs are one of the most implemented AnWT technologies in the developing world. Murray and Drechsel [8] have reported that in Ghana, in the West African subregion, WSPs make up the majority (42%) of the implemented wastewater treatment technologies. This is followed by activated sludge systems which make up 26% and anaerobic digesters make up 16%. Other technologies such as trickling filters, aeration tanks, sedimentation tanks, and granular activated carbon are rarely implemented. Moreover, in Ghana, only 4.5% of the country's population is connected to sewer networks. Onsite sanitation systems like septic tanks and pit latrines remain popular in the country. When full, they are usually emptied, conveyed and discharged at WSP facilities, or indiscriminately disposed of into the environment [9,10,11]. Despite the economic feasibility of the WSPs for low-income countries, these systems come with associated challenges as they require large land areas, have long hydraulic retention times, have odour problems, and contribute to greenhouse gas (GHG) emissions [12]. These challenges have made it imperative to adopt new sustainable technologies for low-income countries.

Among the various AnWT technologies available, the upflow anaerobic sludge blanket (UASB) reactor has gained much prominence, with several pilot and full-scale Plants in operation in countries like Brazil, India, Japan, and Columbia [13,14]. The UASB reactor technology comes with considerable advantages over other anaerobic treatment systems, which accounts for its wide acceptance in several parts of the world, despite its relatively short existence compared to other anaerobic technologies [15]. First is the UASB reactor systems' ability to handle high and fluctuating organic loadings [16]. Wolmarans and De Villiers [17] and Musa et al. [18] reported on full-scale UASB reactors attaining as high as 90% removal efficiency for high-strength influent sewage of about 30000 mg/L COD load. Hulshoff Pol et al. [19] likewise reported that the development of the biological granules in the sludge blanket is the most significant technology feature that enables UASB reactors to handle high volumetric loading compared to other anaerobic systems. UASB reactors produce less and more stabilised sludge than aerobic systems, the biogas generated from these reactors contain considerable amount of methane gas that can be harnessed for energy recovery purposes [20].

Although the UASB reactor technology comes with numerous advantages, one major setback with its application is the inability of these systems to produce high-quality effluent that meet World Health Organization (WHO) discharge guidelines for reclaimed water usage or discharge into the environment. Thus, UASB reactor effluent generally require post-treatment units to guarantee final effluent that meets regulatory standards. The main objective of post-treatment units is to eliminate residual organic matter, together with the elements that are barely affected by anaerobic treatment processes; nutrients (nitrogen and phosphorus containing compounds), and pathogenic bacteria, enhancing the quality of the reclaimed water [20,21].

According to Bressani-Ribeiro et al. [22], most anaerobic wastewater treatment systems are followed by aerobic post-treatment units, allowing the attainment of effluent that comply with discharge standards. Of the many technologies (Anaerobic filters, Polishing ponds, activated sludge, etc.) adopted as post-treatment units, the UASB/Trickling filter (TF) combination has gained dominance in many countries especially Brazil [23]. TFs are designed as non-submerging aerobic biofilm reactors [24], consisting basically of a basin filled with highly-permeable materials, on which the sewage is applied employing a distribution system. As the sewage trickles downward, bacterial growth (biofilm) occurs on the surface of the permeable materials, with upward and downward movement of natural air [25]. Other authors have opined that the stability in performance and simplicity in the operation of TFs are major reasons for their application worldwide, especially in developing countries [22,25].

Although the UASB reactor technology has been widely implemented in several parts of the world [15], its representation in the West African subregion is very limited, despite the favourable climatic conditions, and economic feasibility for low-income countries in the subregion. To date, only a few studies [26,27] have reported on the performance of a full-scale UASB reactor treating municipal wastewater in the subregion. However, these studies focused solely on the systems' performance, providing scant information on the operational conditions. Additionally, critical influent characteristics such as VFA/Alkalinity ratio in the UASBs, influent's nutrients ratio and heavy metals concentrations have not been reported. Evaluation of these parameters is essential as they can impede optimal system performance when not in right concentrations or proportions [28,29,30]. Finally, no study has reported on biogas production and composition for full-scale UASB reactors, which would permit the evaluation of the energy recovery potential from methane gas. This study therefore seeks to fill these scientific gaps in the literature regarding the application of the UASB reactor technology in the West African subregion. It investigates the performance of full-scale UASB reactors coupled with trickling filters (TFs) and final settling tanks (FSTs) as post-treatment units over 35 weeks of continuous operation for municipal wastewater treatment, with analysis of critical operational parameters. Additionally, we quantify and characterise biogas generated by UASB reactors for energy recovery purposes. Evaluation of the Plant will allow for proper comparison with previous studies conducted in different climatic conditions to assess the efficiency and biogas production potential. This study conducted under tropical climatic conditions will ascertain the proposition that the UASB technology is most favourable in the tropical climate. Additionally, this knowledge will inform policy makers on the feasibility of this technology for replication in other developing countries in SSA, to replace the predominant WSPs systems in these regions and help ameliorate the wastewater management menace in developing countries.

2. Materials and methods

2.1. Study area and system configuration

The study was conducted at the Mudor Wastewater Treatment Plant (MWWTP) in Accra (5̊36′53.3448″N, 0̊12′21.1464″W), the capital city of Ghana in West Africa. Accra has two major seasons in a year; a wet season from May to October, and a dry season from mid-November to April [11], with temperature ranging between 22.7 °C and 33.8 °C [31]. According to Ahmed et al. [27], the Plant was built in the year 2000, but operated only for a few years, after which it was shut down due to inadequate maintenance culture and financial commitment. It was rehabilitated, received some expansion works and became operational again in 2017. The Plant covers a total land area of 6.3 acres, sited within 20 m eastward from the Korle Lagoon, in James Town, Accra. It receives and treats sewage from households, offices, and commercial centres within Korle-Bu, Accra Central, parts of Dansoman, Osu-Labone, Ministries, and High-street suburbs which are all sewered communities. The Plant is currently estimated to serve approximately 100000 inhabitants, based on projections from 60000 inhabitants being served in the year 2000 at the time of construction [32], applying a population growth rate of 2.1% per annum [33]. There were no known industrial discharges received at the Plant at the time this study was conducted.

The MWWTP comprises of full-scale UASB reactors with TFs and FSTs as post-treatment units. The modular-shaped UASBs consist of 6 reactors operating in parallel, with capacity between 16000 m^3/d and 18000 m^3/d. The circular-shaped TFs and settling tanks are likewise operated in parallel. Sewage flow to the Plant ranges from 1752 m^3/d to 7534 m^3/d, with an average flow of 4330 ± 1008 m^3/d. With an estimated per capita

BOD contribution of 0.04 kg/cap/d [7] for developing countries, the average volumetric organic loading for the UASB reactors was calculated to be 0.807 ± 0.505 $kgBOD_5/m^3/d$ (157392.06 PE).Table 1 presents information on the dimensions of the various treatment units of the MWWTP.

The raw sewage received at the Plant is a typical low-strength municipal wastewater Ahmed et al. [27], which undergoes preliminary treatment through coarse screens (20 mm mesh aperture), where larger solid waste materials are trapped. The sewage then moves into a wet well and is pumped to the sand/grit removal system (vortex grit). Next, the sewage flows through a fine screen unit (5 mm mesh aperture), where further sieving occurs before entering the six (6) UASB reactors. Effluent from the UASBs flow to the TFs for further biological treatment and finally into the secondary clarifiers (FSTs) before discharge into the Korle Lagoon. Periodically, excess sludge is drawn from the UASB reactors into the sludge thickeners for physical sludge dewatering. The thickened sludge is pumped onto the sludge drying beds for air drying and further processing, whilst the supernatant flows back into the wet well and mixes with incoming sewage. Biogas generated in the UASBs is collected in the gas hoods and channelled to a biogas flaring unit where biogas is flared. The schematic layout of process flow at the MWWTP is illustrated in Figure 1. The MWWTP was designed such that gravity drives most of the material flow, minimising pumping, thereby reducing electricity consumption and associated cost.

2.2. Sampling and analytical methods

System performance was monitored over 35 weeks by analysing composite samples from the various sampling units (i.e. raw sewage after grit removal and effluent of the UASBs, TFs, and FSTs). The pH, temperature, electrical conductivity (EC), total dissolved solids (TDS) and dissolved oxygen (DO) were measured in-situ with a portable multiprobe analyser (HQ40D LDO10101, HACH), whilst the rest of the parameters were analysed at the Sewerage Systems Ghana Limited (SSGL) laboratory at the premises of the Plant. Clean sampling bottles (1 L) were utilised, with sampling diligently carried out to avoid external contamination. Wastewater samples were collected semi-weekly for organic component and on weekly basis for nutrients, microbials, and heavy metals analysis. Samples were transported from the site in iced chest box with ice cubes to the laboratory within 24 h for laboratory analysis or storage in a refrigerator at 4 °C where applicable.

Biogas and sewage flows were measured with installed automatic flow measuring devices; Prosonic Flow B (Endress + Hauser, Switzerland) and PROMAG 50 (Endress + Hauser, 50W1F-HLGA1RK5-BAAA, Switzerland) meters, respectively. Additionally, biogas samples were collected from all 6 UASB reactors and characterised, over a period of ten (10) weeks (July 02 to September 15, 2021). Biogas was sampled by connecting 1 L Tedlar sacs to the gas sampling points located on top of the reactors' gas hoods and carefully transported to the Institute of Industrial Research (IIR) laboratory. Biogas constituents namely methane (CH_4), carbon dioxide (CO_2), oxygen (O_2), nitrogen (N_2) and hydrogen sulphide (H_2S) were analysed with a potable FM 406 Gas Analyser (Gas Data, UK) at the IIR laboratory.

Analysis of biochemical oxygen demand (BOD_5) was carried out using test method APHA 5210, chemical oxygen demand (COD) was determined employing potassium dichromate digestion method. Total suspended solids (TSS), and total solids (TS) have been determined by oven drying at 105 °C, total volatile solids (TVS) by furnace combustion at 550 °C. Alkalinity and volatile fatty acids (VFA) by Lovibond and distillation methods, respectively. Nutrients; total phosphorus (TP), orthophosphate (PO_4^{3-}-P), total nitrogen (TN), ammonia nitrogen (NH_3-N), nitrate nitrogen (NO_3^--N), Sulphate (SO_4^{2-}) and Sulphide (S^{2-}) were determined using HACH DR 3900 spectrophotometer. Fecal coliform (FC), *E. coli*, and Salmonella *sp*. by pour plate method with agar medium, whilst Helminth eggs characterisation was done according to the methodology proposed by Moodley et al. [34]. Selected heavy metals (Zn, Cu, Cd, Pb, Ni, Hg, Mn, Cr) were measured using Atomic Absorption Spectrometry. All analyses were carried out in accordance with standard methods [35]. Details of the analytical methods, equipment make up, models and manufacturers have been tabulated and attached as electronic supplementary material.

2.3. Data analysis

Descriptive statistical analysis; minimum, maximum, the appropriate central tendency measurements (i.e., mean and standard deviation, median), inferential statistical analysis; One-way ANOVA followed by Tukey's posthoc pairwise test, nonlinear regression and the respective removal efficiencies were employed for data interpretation. Presentation of results were adjusted to conform to standard instrumentation scale and readings [35].

3. Results and discussions

3.1. Operational conditions of the treatment units

3.1.1. Operational conditions applied to the UASB reactors

The operational parameters applied to the UASB reactors are presented in Table 2. As reported in tropical regions, the UASB reactors operated at a typical mesophilic temperature range of 26.2 ± 1.8 °C and in a near-neutral pH at 7.2 ± 0.4 for the influent wastewater, similar to conditions reported in literature [36,37]. The upflow velocity (Vel_{up}) of system (1.0 ± 0.2 m/h) was within the optimum range which is from 0.5–1.5 m/h as reported by Tawfik et al. [38]. The average hydraulic retention time (HRT) at 45.8 ± 24.9 h was higher than the reported range of 4–14 h. However, the applied organic loading rate (OLR) at 1.30 ± 0.79 $kgCOD/m^3/d$ was found to be relatively lower than values reported in literature [39]. These results indicate that the UASBs have the capacity to treat larger volumes of sewage up to 18000 m^3/d, far more than the load (4330 ± 1008 m^3/d) currently handled by the Plant. This suggests that the Plant is not operating at its full design capacity. The average sludge concentration in the UASB reactors was found to be 65.63 ± 29.29 gVSS/L and 88.99 ± 28.93 gTSS/L. Other authors have reported lower sludge TSS concentrations in ranges 27–57 gTSS/L and 32.2–50.2 gTSS/L, respectively, when they conducted rheological studies on sewage sludge concentrations [40,41].

Table 1. Dimensions of treatment units at MWWTP.

Treatment Unit	Length (m)	Breadth (m)	Height (m)	Diameter (m)	No. of Units	Unit Volume (m^3)	Total Volume (m^3)
UASB Reactors	20	10	6.5	-	6	1300	7800
Sludge Thickeners	10	6	6.5	-	6	390	2340
TFs	-	-	3.0	24.5	3*	1414.3	4242.9
FSTs	-	-	4.2	24.5	2	1540.0	3080.0
Sludge Drying Beds	31	4.25	0.8	-	19	105.4	2002.6

Source: Sewerage Systems Ghana Limited (SSGL).
* One unit was non-functional at the time of the study.

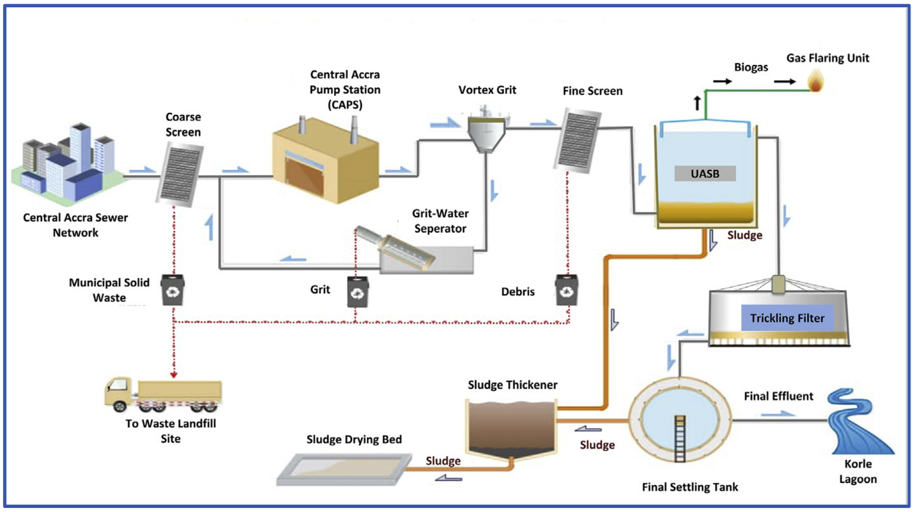

Figure 1. Process flow of MWWTP (*Source: Authors compilation*).

3.1.2. Operating conditions applied to the post-treatment units

Two TFs and two secondary clarifiers are employed in parallel as the post-treatment units of the Mudor UASBs. The dimensions of these units have been presented in Table 1.

The two TFs are filled with black plastic media (180 mm polypropylene Bio-Pac Media SF30), with 95% void ratio and 98.4 m^2/m^3 surface area as microbial carriers. Literature has reported that some important process parameters to consider during TFs operation are the loading parameters such as the hydraulic loading, organic loading and the recirculation ratio. During the study, it was found that the average flow to each TF was 90.2 ± 21.1 m^3/h. The hydraulic loading rate (HLR) and volumetric OLR were determined to be 0.19 ± 0.05 $m^3/m^2/h$ and 0.19 ± 0.14 $kgBOD/m^3/d$, respectively. The loadings on the Mudor TFs can be classified under low or standard-rate TFs [45]. Moreover, the Mudor TFs were designed without a recirculation system. The OLR range obtained in this study is comparable to the results reported by Rosa et al. [46], when they presented results on the OLR for TFs employed as post-treatment units for UASB effluent.

Critical clarifier loading parameters include the detention time (DT), surface overflow rate (SOR), weir overflow rate (WOR) and solids loading rate (SLR) [47]. The applied DT and SOR to each clarifier of the Mudor Plant were 18.1 ± 4.7 h, and 4.59 ± 1.07 $m^3/m^2/d$, respectively. The estimated WOR and SLR were 29.40 ± 6.86 $m^3/m/d$ and 4.27 ± 0.99 $kgTSS/m^2/d$, respectively.

The calculations above indicate that the solids and organic loads received by the post-treatment units were very low, and this could be attributed to the high (90%) removal efficiency of the UASB reactors for solids and organics in the influent sewage. The range of operational conditions applied to the post-treatment units have been presented in Table 3.

3.2. Sewage characteristics

3.2.1. pH and temperature profile

Figure 2 illustrates the variations in pH and temperature at the various treatment stages with error bars. Several studies reveal that anaerobic Plants as biological wastewater treatment systems require pH and temperature ranges from 6.3–7.8 and 20–40 °C, respectively for optimum performance [48,49].

pH measured in this study showed that an appropriate environment was maintained throughout the treatment process. The mean influent sewage pH was 7.2 ± 0.4 (ranging from 5.4–7.9). This falls within the reported optimum range for mesophilic anaerobic bacteria [34], hence no pH adjustment was required during the study period. We attribute the occasional acidic sewage (pH = 5.4) observed to sewage received from acid-based compounds from small and medium-scale enterprises (SME) within commercial areas. Conversely, the alkaline sewage could probably be due to the use of soapy and soapless detergents at homes and offices [27].

The pH of the sewage streams was observed to increase across the treatment units, from a value of 7.2 ± 0.4 in the influent to 8.2 ± 0.1 in the final effluent as presented in the figure. This marginal increment of pH from neutral to basic medium could be due to the nitrification and denitrification processes occurring in the aerobic post-treatment units.

Although this observation contrasts the report by Awuah and Abrokwa [26], who observed a pH drop from 8.96 ± 0.98 in the influent to 7.45 ± 0.14 in the final effluent in their studies, it is comparable to other findings [27,50].

The influent sewage temperature ranged from 22.4 °C to 30.7 °C. This falls within the temperature range required for optimum anaerobic system performance [51], and comparable to the findings by Ali and Okabe [52] and Divya et al. [53]. One interesting finding was the fact that the influent sewage temperature was within range of local ambient air temperature (22.7–33.8 °C) [31]; a typical mesophilic temperature range suitable for anaerobic reactors. This meant the system did not require heating, which comes with extra cost as applicable in the temperate regions, making the UASB reactor technology economically feasible for implementation in tropical countries in the developing world [15].

3.2.2. Volatile fatty acid (VFA) and alkalinity ratio

As can be observed in Table 4, the average VFA/Alkalinity ratio recorded in the UASB reactors' influent sewage during the study period was 0.20 ± 0.10 (ranging from 0.12–0.45). Callaghan et al. [54] and Kuglarz et al. [28] have reported that the VFA/Alkalinity ratio is a variable that can measure system performance and control the AD stability process. According to these authors, whilst the VFAs provide information on the AD intermediate steps performance, alkalinity describes the capability of the feedstock to neutralise the VFAs generated during the process, controlling pH changes. The ideal VFA/Alkalinity ratio range is 0.10–0.40 for stable anaerobic digestion; 0.40–0.80 indicate a level of system instability; >0.8 suggests gross instability which could be due to

Table 2. Operational conditions applied to the UASB reactors.

Operational Parameter	Current study		Optimum Range in Literature	Reference
	Range	Average ±SD		
OLR ($kgCOD/m^3/d$)	0.25–4.73	1.30 ± 0.79	2–14	[39,42]
HRT (h)	24.85–106.85	45.77 ± 24.85	4–14	[37,43]
Vel_{up} (m/h)	0.37–1.57	0.97 ± 0.21	0.5–1.5	[37,43]
pH	5.4–7.9	7.2 ± 0.4	6.3–7.8	[44]
T (°C)	22.4–30.7	26.2 ± 1.8	20–40	[44]

Table 3. Operational conditions applied to the post-treatment units.

Operating Parameter	Current Study		Typical Design Criteria	Reference
	Range	Average ±SD		
Tricking Filters				
Flow (m^3/h)	36.5–156.9	90.2 ± 21.1	-	-
HLR (m^3/m^2/h)	0.08–0.33	0.19 ± 0.05	1.02–4.07	[45]
OLR (kgBOD$_5$/m^3/d)	0.04–0.93	0.19 ± 0.14	80.09–400.46	[45]
Settling Tanks				
DT (h)	9.8–42.2	18.1 ± 4.7	2–3	[47]
SOR (m^3/m^2/d)	1.86–7.99	4.59 ± 1.07	12.22–32.59	[47]
WOR (m^3/m/d)	11.89–51.15	29.40 ± 6.86	~124.19	[47]
SLR (kg TSS/m^2/d)	1.73–7.43	4.27 ± 0.99	122.06–146.47	[47]

an increase in organic or hydraulic loadings to the system [55,56]. Thus the VFA/Alkalinity ratio obtained in this study falls within the optimum range ideal for anaerobic reactors as reported by several authors [30,55,56].

3.2.3. Carbon, nutrients and trace elements

The BOD:COD ratio which indicates the biodegradability index of the influent sewage was observed to range from 0.3–0.8 (Table 4). According to Manyuchi et al. [58], BOD:COD ratio measures the presence of highly biodegradable compounds in the sewage. The observed BOD:COD ratio for this study was found to be within the optimum range indicated in literature [57,63].

The composition of macronutrients present in the influent sewage has likewise been presented in Table 4. Balanced amount of the required nutrients, coupled with ideal growth conditions are vital for optimised performance of anaerobic systems [64,65]. From the study, it was found that the carbon: nitrogen (C:N) ratio was between 2.4:1 and 37:1, with an average of 11 ± 8.5:1. The average value obtained was found to be less than the values reported by other authors. Romano and Zhang [66] reported an optimum C:N ratio of 15:1, whilst Cerón-Vivas et al. [67] reported that they observed increased methane production and COD removal rate at C:N ratio of 14.2:1. Several authors have equally reported optimum C:N ratio values for maximum methanogenesis to be within the range 20–30:1, depending on the nature of substrate fed into the reactor [48,59,68]. The finding in this study is however comparable to the assertion by Kwietniewska and Tys [69], who reported that municipal wastewater usually has a low C:N ratio (<8.0) The lower average C:N ratio of influent sewage obtained for this study could also be attributed to the high N content in the influent sewage. Martin-Ryals [29] asserted that an unbalanced C:N ratio is one limiting factor in anaerobic digesters. The configuration of C:N should be maintained in an optimum range to

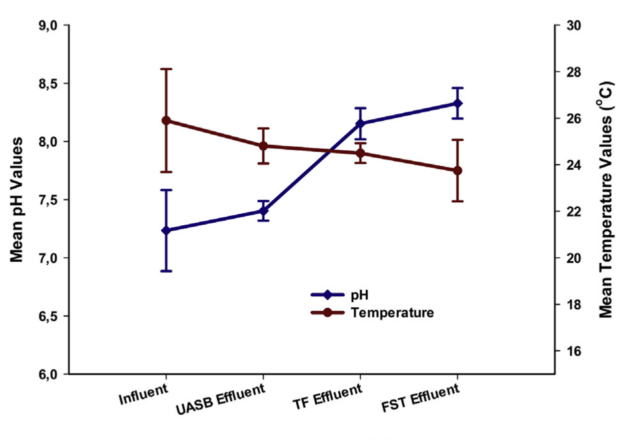

Figure 2. pH and temperature variations at the various stages of the treatment process.

Table 4. Sewage nutrients and heavy metals concentrations.

Parameter	Current study		Optimum Range in Literature	Reference
	Range	Average ±SD		
VFA: Alk Ratio	0.12–0.45	0.20 ± 0.10	0.1–0.4	[56]
BOD: COD Ratio	0.3–0.8	0.6 ± 0.2	0.3–0.8	[57,58]
C: N Ratio	2.4–36.9	11.0 ± 8.3	20–30	[48,59]
C: N: P Ratio	-	85:4.8:1	250–500:5:1	[60,61]
Cr (mg/L)	0.080–2.270	0.830 ± 0.550	-	-
Ni (mg/L)	0.050–0.050	0.050 ± 0.000	0.8–50	[62]
Zn (mg/L)	0.007–0.036	0.009 ± 0.005	0–5	[62]
Cd (mg/L)	0.002–2.020	0.157 ± 0.535	0.1–0.3	[62]
Mn (mg/L)	0.005–0.040	0.009 ± 0.008	-	-
Pb (mg/L)	0.005–0.005	0.005 ± 0.000	-	-
Cu (mg/L)	0.035–0.675	0.190 ± 0.160	0–100	[62]
Hg (μg/L)	0.309–1.597	0.742 ± 0.385	-	-

conserve an appropriate nutrient balance for essential microbial growth, maintaining a stable environment for efficient AD [68,70].

The C:N:P ratio for the influent sewage in this study was assessed to be 80:4.8:1. According to USEPA [60] and Ammary [61], during start-up of an anaerobic reactor, the optimum range should be within 250:5:1 and 500:5:1. The C:N:P ratio obtained in the study indicates that the influent sewage had high N and P contents.

Table 4 also presents the heavy metals concentrations of MWWTP's influent sewage and the optimum ranges reported in the literature. Chen et al. [44]; Chen et al. [71] and Şengör et al. [72] have reported that some heavy metals are vital trace elements that are essential for microbial growth and development, promoting biogas and methane production. However, in excess, these elements exert toxicity, causing inhibitions in the microbial community activity and destabilising the system. Guo et al. [62] reported on optimum range for Ni^{2+} (0.8–50 mg/L), Zn^{2+} (0–5 mg/L), Cd^{2+} (0.1–0.3 mg/L), Cu^{2+} (0–100 mg/L), and Fe^{2+} (50–5000 mg/L) as concentrations which promote biogas production. As presented in the table, the average concentrations of heavy metals were in descending order: Cr > Cu > Cd > Ni > Zn > Mn > Pb, with recorded values 0.830 mg/L, 0.190 mg/L 0.157 mg/L, 0.050 mg/L, 0.009 mg/L, 0.009 mg/L, 0.005 mg/L respectively.

Generally, the levels of the elements tested were within the reported range which did not obstruct optimum system performance. Hg also had an average concentration of 0.742 μg/L. Heavy metals like Pb and Hg are not biologically essential, and have only toxic impacts. According to Appels et al. [73], domestic sewage heavy metals sources are associated with the use of detergents, washing powders, and body care products. Other sources may include leachates from plumbing materials, roofs, and gutters.

3.3. System performance

Table 5 presents the UASB influent, effluent of the treatment units of the MWWTP and the regulatory body; Environmental Protection Agency (EPA) Ghana standards for sewage effluent discharge.

3.3.1. Organic matter and solids removal

The UASB reactors are the primary treatment units that receive influent sewage after the preliminary treatment for solids and coarse materials removal. From this study, influent COD concentration which ranged from 450–8150 mg/L were appreciably reduced to a range between 226 and 1449 mg/L (Table 5), achieving 45–88% removal efficiency, with an average of 77% after treatment with the UASB reactors. Similar to our findings, an earlier study reported a maximum COD removal of 88.9% by the MWWTP UASBs [27]. Lettinga [13] and Slompo et al. [74] in their studies have proven the exceptional ability of UASB reactors to remove organic matter from domestic and municipal sewage.

Table 5. MWWTP efficiency in pollutant removal.

Wastewater Parameter (unit)	#auto; Influent Sewage (Range)	Influent Sewage (Average ±SD)	UASB Effluent (Average ±SD)	TF Effluent (Average ±SD)	FST Effluent (Average ±SD)	Plant* Efficiency (%)	EPA Guidelines
#auto; *Physico-chemical*							
pH	5.4–7.9	7.2 ± 0.4	7.2 ± 0.9	8.1 ± 0.1	8.2 ± 0.1	-	6–9
Temperature	22.4–30.7	26.2 ± 1.8	26.0 ± 1.6	25.7 ± 1.4	24.1 ± 2.3	-	<30
EC (μS/cm)	1233–31000	3097 ± 2922	3221 ± 340	3077 ± 339	2977 ± 371	-	1500
DO	0.00–1.00	0.29 ± 0.23	0.54 ± 0.45	1.82 ± 1.14	3.56 ± 1.72	-	-
COD (mg/L)	450–8150	2122 ± 1251 ^{A}	496 ± 221 ^{B}	476 ± 228 ^{B}	152 ± 115 ^{C}	92.8	250
BOD (mg/L)	308–5134	1384 ± 887 ^{A}	120 ± 73 ^{B}	160 ± 79 ^{B}	33 ± 31 ^{C}	97.6	50
TS (mg/L)	1181–6450	2439 ± 661 ^{A}	1569 ± 301 ^{B}	1599 ± 259 ^{B}	1056 ± 188 ^{C}	56.7	50
TSS (mg/L)	24–2330	979 ± 410 ^{A}	262 ± 129 ^{B}	313 ± 162 ^{B}	72 ± 18 ^{C}	92.6	50
TDS (mg/L)	893–6110	1480 ± 562 ^{A}	1241 ± 178 ^{B}	1188 ± 180 ^{B}	955 ± 182 ^{C}	35.5	1000
TVS (mg/L)	16.8–1504.0	682.0 ± 293.0 ^{A}	177.4 ± 99.8 ^{B}	190.4 ± 55.3 ^{B}	48.3 ± 14 ^{C}	92.9	75
Nutrients							
TN (mg/L)	35.10-360.00	114.46 ± 59.20 ^{A}	121.01 ± 48.34 ^{A}	-	83.61 ± 24.51 ^{B}	27.0	2
NH_3 -N (mg/L)	31.20–141.90	67.51 ± 24.30 ^{A}	84.60 ± 22.60 ^{B}	-	61.41 ± 15.17 ^{A}	9.0	1
NO_3^--N (mg/L)	0.60–30.00	7.94 ± 6.44 ^{A}	7.93 ± 6.54 ^{A}	-	10.93 ± 7.94 ^{A}	-37.7	50
PO_4^{3-}-P (mg/L)	13.35–26.26	19.50 ± 3.70 ^{A}	22.48 ± 5.93 ^{A}	-	21.15 ± 4.28 ^{A}	-8.4	2
TP (mg/L)	16.32–34.69	25.09 ± 4.90 ^{A}	29.56 ± 6.38 ^{A}	-	28.37 ± 14.17 ^{A}	-13.1	2
SO_4^{2-} (mg/L)	11.00–620.00	146.46 ± 106.20 ^{A}	45.08 ± 32.49 ^{B}	-	82.45 ± 23.99 ^{B}	43.7	-
Sulphide (mg/L)	0.16–1.62	1.32 ± 0.36 ^{A}	0.07 ± 0.03 ^{B}	-	0.19 ± 0.44 ^{B}	85.9	1.5
Microbials							
FC (CFU/100mL)	1.0×10^2–1.0×10^3	$3.4 \times 10^2 \pm 3.3 \times 10^2$ ^{A}	$3.7 \times 10^1 \pm 4.6 \times 10^1$ ^{B}	-	$1.7 \times 10^1 \pm 1.6 \times 10^1$ ^{B}	95.2	1.0×10^2
E. coli (CFU/100mL)	1.0×10^1–1.0×10^3	$2.5 \times 10^2 \pm 3.7 \times 10^2$ ^{A}	$2.8 \times 10^1 \pm 4.9 \times 10^1$ ^{B}	-	$1.2 \times 10^1 \pm 1.7 \times 10^1$ ^{B}	95.0	1.0×10^2
Salmonella (CFU/100mL)	1.0×10^2–1.0×10^3	$4.7 \times 10^2 \pm 3.2 \times 10^2$ ^{A}	$9.4 \times 10^1 \pm 1.5 \times 10^2$ ^{B}	-	$2.7 \times 10^1 \pm 2.9 \times 10^1$ ^{B}	94.3	-
Helminth Eggs	Not detected	Not detected	Not detected	-	-	-	-

* *Pollutant concentration of final effluent from MWWTP being discharged into the environment relative to the raw influent received at the Plant.* One-way ANOVA Results; $^{A-C}$: Columns which do not share the same letter indicate significant statistical difference ($p < 0.05$, Tukey's posthoc pairwise test) between means of treatment units' effluent.

These assertions have been confirmed by the robust performance of the UASB reactors at MWWTP in removing COD from raw sewage. Many other studies reported the performance of UASB reactors in removing organic pollutants in diverse wastewater streams, attaining COD removal rates as high as 90% [18,75,76].

Post-treatment with the TFs attained a menial 4% removal efficiency whilst the FSTs which acts as secondary clarifiers for TF effluent attained an additional 70% removal. The overall COD removal efficiency of the MWWTP was 92.8%. Although satisfactory, other studies have reported as high as 99% overall COD removal on both pilot and full-scale Plant experiments [77,78,79]. Similar performance was observed for BOD, with 91.3% removal for the UASBs and 97.6% after the post-treatment units. These results are comparable to other findings [80,81].

Regarding solids removal, the UASBs removed 35.7% and 16.3% of TS and TDS, respectively; but the post-treatment units enhanced the overall efficiencies to 56.7% and 35.5%, respectively. Meanwhile, TSS and TVS removal were satisfactory for the UASBs at 73.3% and 74% respectively, with 93% overall removal efficiency for both parameters.

Mean concentrations for solids and organic load pollutants at the various treatment units have been graphically represented in Figure 3. It is worth noting that the reductions observed for most of the organics and solids compounds between the UASB and TF effluent were marginal (no statistically significant difference at $p = 0.05$). The observed marginal difference in performance between the two units for the residual loads, with some parameters (BOD_5, TS, TSS and TVS) recording even higher concentrations in the TF effluent compared to the UASB effluent could be attributed to biofilm sloughing from the TFs' filter media. The sloughed biofilms move with the effluent into the underdrain system provided for TF effluent collection. Hence, the TF effluent usually contains substances that could increase the solids and organic concentration of the effluent. This explains why in Plant designs, clarifiers usually follow TFs to ensure the sloughed biofilms, and other solids are settled for clearer and cleaner final effluent [45,82]. It was therefore unsurprising that after settling in the FSTs, the residual loads reduced significantly in the FST effluent. Thus, the UASB-TF/FST combination significantly contributes to the overall organic matter and solids removal, ensuring that the effluent quality meets acceptable levels.

3.3.2. Nutrients removal

Different studies have reported on the relative ineffectiveness of UASB reactors in removing nutrients from wastewater, which necessitates the employment of post-treatment units [20,21]. As observed in Table 5, the percentage errors relative to the mean values of the nutrients, especially nitrogen and sulphur indicators, were generally large (73–82%) for the UASB effluent, suggesting significant variability in the concentrations. Hence, the median values were considered a more appropriate central tendency measure. Figure 4 presents box and whisker plots of the nutrients. The median TN level in the sewage remained almost constant (from 108.00 mg/L to 108.30 mg/L) in the UASB effluent before dropping to 80.70 mg/L in the final effluent. However, NH_3-N concentration increased from 63.60 mg/L to 79.70 mg/L and finally dropped to 61.00 mg/L. The median NO_3^--N concentration in the sewage (5.90 mg/L) dropped in the UASBs (3.40 mg/L) and then increased to 9.30 mg/L in the FST effluent, even above the influent concentration. Hence the overall efficiencies of the Plant for TN, NH_3-N, and NO_3^--N removals were 25.3%, 4. 1%, and −57.6%, respectively. We attribute the

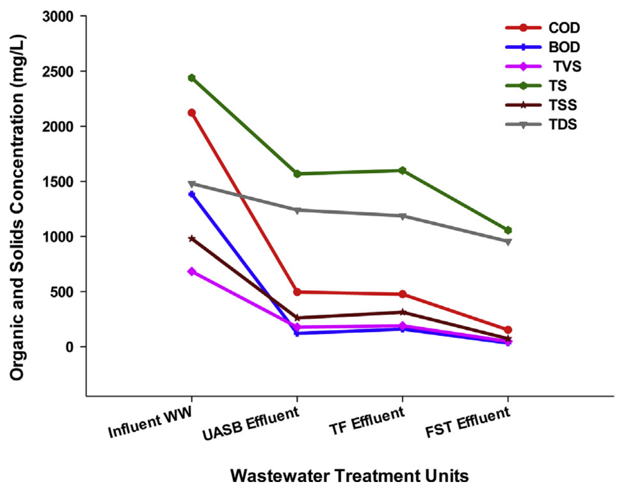

Figure 3. Mean concentrations of solids and organic loads at the various treatment units.

alternating levels of NH_3-N and NO_3^--N concentrations in the UASBs to the reducing environment in the UASB reactors that favour NO_3^--N reduction and promote NH_3-N generation; the reverse occurs after the UASB. Subjecting the data on nitrogenous compounds to one-way ANOVA analysis, it was revealed that no statistically significant difference existed in the means of treatment units' effluent for NO_3^--N ($p = 0.521$), but effluent means for TN and NH_3-N observed significant statistical difference ($p < 0.05$).

Other studies have indicated higher nitrogen removal efficiencies after coupling UASB reactors with other post-treatment units. Bhatti et al. [83] reported on the performance of a UASB reactor coupled with hydrogen peroxide (H_2O_2) for municipal wastewater treatment. Their combined system attained 51.7–87.5% N removal efficiency. Other researchers attained similarly satisfactory results (>60%) in their studies on UASB reactors coupled with constructed wetlands system and sequential batch reactor as post-treatment units, respectively [84,85]. The MWWTP's poor performance regarding N removal could be attributed to several factors. First of all, since nitrification is an oxygen (O_2) demanding process, aeration at the TF might be insufficient. Unlike the activated sludge systems where O_2 is actively injected to enhance biological nutrient removal (BNR), the TF employs a natural flow of atmospheric O_2 for the aerobes, which is probably ineffective under the prevailing MWWTP operational conditions. Also, the MWWTP's TFs did not have a recirculation system. Pearce [86] opined that single filtration Plants could achieve complete nitrification if effluent recirculation is considered. Effluent recirculation additionally increases the wetted surface area, and has been proven to advance nitrification processes [87]. Thus, the absence of the recirculation system could have reduced the contact time, and also caused nitrifiers to be washed out, resulting in incomplete nitrification [88,89]. Again, it has been reported that recirculation introduces nitrate onto the top of the filter media, where heterotrophic activity, and hence, supposed denitrification activities are highest [87]. Moreover, denitrifying bacteria require certain amount of organic carbon (in the form of BOD) to complete the denitrification process. However, the UASB reactors removed about 90% BOD from the influent sewage, the residual BOD available to these denitrifiers might be insufficient for denitrification process to eliminate N. Thus, under the prevailing conditions, nitrification and denitrification processes would be incomplete [90,91], negatively affecting nitrogen removal.

Similar observations were made for phosphorus (P) compounds. TP and PO_4^{3-}–P also attained an overall negative removal efficiency. The system attained mean effluent values of 28.38 and 21.15 mg/L respectively, which were higher than the mean influent values of 25.09 and 19.5 mg/L respectively. However, no statistically significant difference existed between the mean effluent values ($p = 0.215$) and ($p = 0.08$) for TP and PO_4^{3-}–P, respectively. The P levels exceeded the standard limits. Moreover, these results contrast the findings of De Sousa et al. [84] and Ahmed et al. [27], who reported higher removal efficiencies for TP at 89% and PO_4^{3-}- P at 81.7%, respectively. Generally, the influent C:N:P imbalance could have resulted in the high concentrations of N and P compounds in the final effluent and subsequently in the MWWTP's poor performance in nutrient removal. The N and P concentrations in the sewage were significantly high compared to the carbon required for a balanced nutrient ratio for optimised anaerobic systems [61,92]. Some studies have suggested adding other carbon supplements (co-digestion) to augment the carbon content to achieve a balanced C:N:P ratio for optimum system performance [92,93]. Nevertheless, it is evident that the post-treatments units at the MWWTP have not been designed to enhance biological phosphorus removal.

Since the current post-treatment units at the MWWTP have been ineffective in removing nitrogenous and phosphorous compounds from the UASB effluent, more effective technologies should be incorporated among the unit processes to enhance nutrient removal. Some technologies reported in literature for effective nitrogen removal include ammonia striping and distillation, ammonia precipitation as struvite, ion exchange for ammonia and nitrate removal, di-electrophoresis-enhanced adsorption, chemical oxidation of ammonia processes, etc. [94,95,96,97,98]. Physico-chemical technologies for phosphorus removal include processes such as precipitation, sorption and ion exchange mechanisms [99]. Well-enhanced biological nutrient removal processes also exist [100,101]. Debowski et al. [102] recently reported on the anaerobic reactor filling as a modern economically feasible and effective phosphorus removal method by metal dissolution. With several nutrient removal technologies available, further assessment is required to select the most feasible option for implementation in terms of cost, efficiency and sustainability for developing countries.

The MWWTP performed satisfactorily for sulphate (SO_4^{2-}) and sulphide (S^{2-}) removal, with overall removal efficiencies of 43.7 and 85.9%, and mean influent concentrations of 146.46 and 1.32 mg/L, respectively (see Table 5). As presented in Figure 4 (f, g), SO_4^{2-} and S^{2-} concentrations reduced considerably in the UASB effluent but increased in the final effluent. The considerable decrease in the UASB effluent is possibly a result of sulphate and sulphide reduction to H_2S under anaerobic conditions [103]. Aerobic post-treatment after the UASB triggered oxidation of the dissolved sulphur species, increasing the sulphate levels in the final effluent [104]. Although SO_4^{2-} and S^{2-} both observed a significant statistical difference ($p < 0.05$) between the influent and UASB effluent, no significant difference was observed between the UASB effluent and FST effluent. Rao et al. [104] however reported on sulphide removal efficiency between 60–70% for an anaerobic-aerobic treated industrial wastewater using the stripper technique. Yun et al. [105] reported sulphate removal efficiency of $84 \pm 0.4\%$ when they designed a sulphate-reducing bacteria (SBR)-based wastewater treatment system (SWTS) for a UASB reactor integrated with sulphide fuel cell (SFC) to treat synthetic wastewater. Comparably, Oliveira et al. [106] examined the use of biochar in treating sulphate-rich wastewater and obtained over 98% H_2S, 89% unionised sulphide, and 94% dissolved sulphide removal efficiencies.

3.3.3. Microbial loads reduction

Mean levels of microbial loads at the various treatment units have been presented in Figure 5. Results on Plant performance regarding microbial removal: fecal coliform (FC), *E. coli*, and Salmonella *sp.* as presented in Table 5 revealed influent sewage levels of microbial count ranged from 1.0×10^2–1.0×10^3, 1.0×10^1–1.0×10^3 and 1.0×10^2–1.0×10^3 (CFU/100mL) respectively for FC, *E. coli* and Salmonella sp. Primary treatment with UASB reactors significantly contributed to satisfactory removal efficiencies, in their respective order: 89.3, 88.5 and 80.0%. Moreover, post-treatment with TFs and FSTs further enhanced the microbial removal to approximately 1 log unit (94–95%) for FC, *E. coli*, and Salmonella *sp.* The values obtained in this study agree with

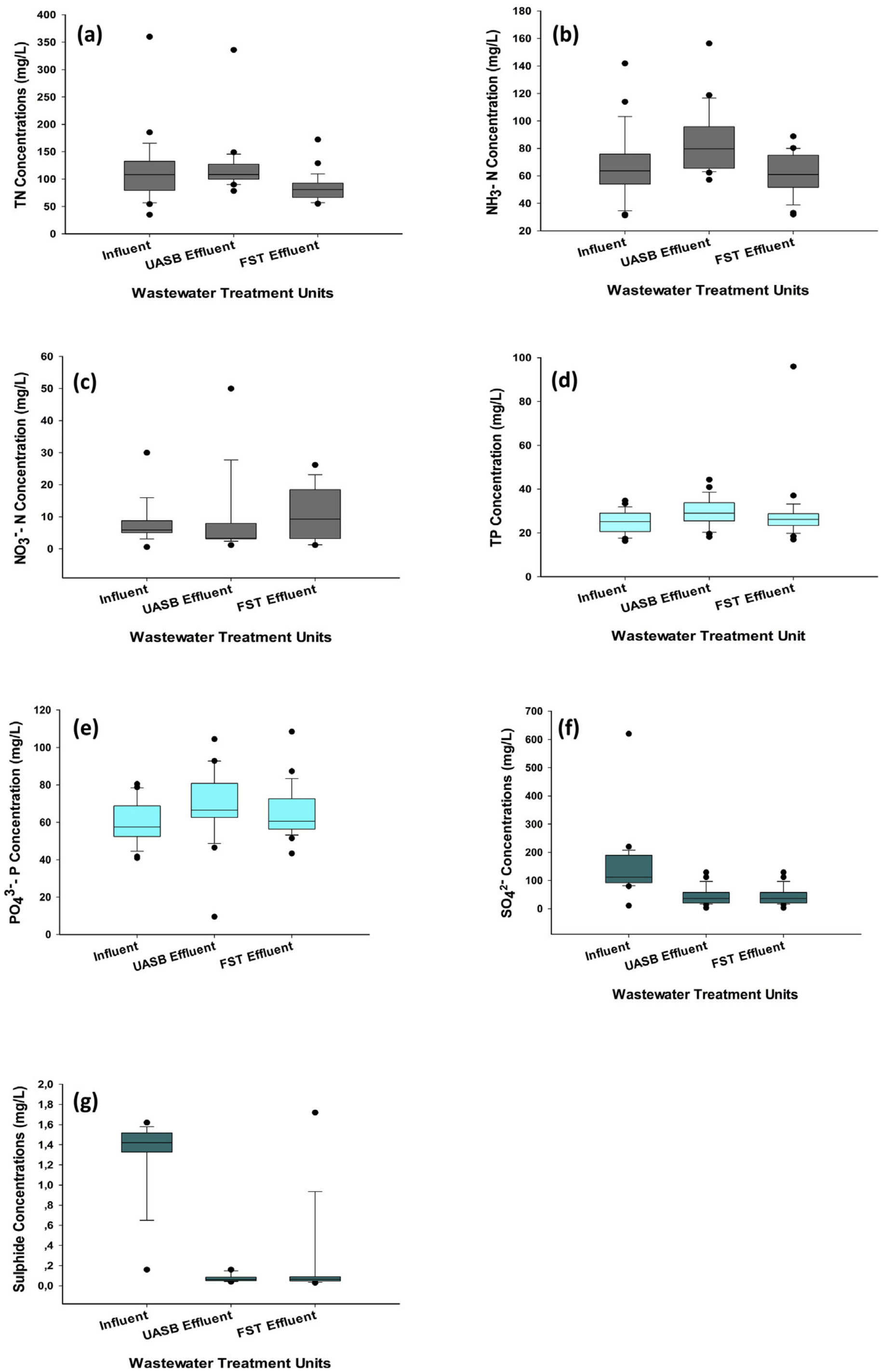

Figure 4. Sewage nutrient concentrations: (a) TN, (b) NH_4-N, (c) NO_3^--N, (d) TP, (e) PO_4, (f) Sulphide, and (g) SO_4^{2-} at the various treatment units.

those reported by Cavalcanti et al. [12] and Lohani et al. [107] who reported overall removal efficiencies >90% for FC, and *E. coli* after post-treating UASB effluent with polishing ponds and sand filters, respectively. This affirms the premise that the combination of UASB reactors with post-treatment units can reduce municipal wastewater indicator organism loads to acceptable levels. It is however worth mentioning that notwithstanding the fact the post-treatment units improved pathogen load reduction, inferential statistics with one-way ANOVA revealed there was no statistically significant difference between the UASB and FST effluent means. No helminth eggs were detected in all the sewage streams. With the exception of the nutrients, the average effluent concentration of all parameters monitored for this study were found to be within the permissible discharge limits of EPA-Ghana. Detailed information on inferential statistics results and performance of individual treatment units have been presented in supplementary material.

3.4. Biogas production

3.4.1. Biogas production rate, composition, and methane yield

Figure 6 presents monthly variation in sewage flow, OLR and biogas production. The minimum and maximum biogas production rates were 217.5 m^3/d and 2060.8 m^3/d, respectively, with an average daily production rate of 831.6 ± 292.7 m^3. Non-linear regression analysis indicated a strong and statistically significant relationship between the

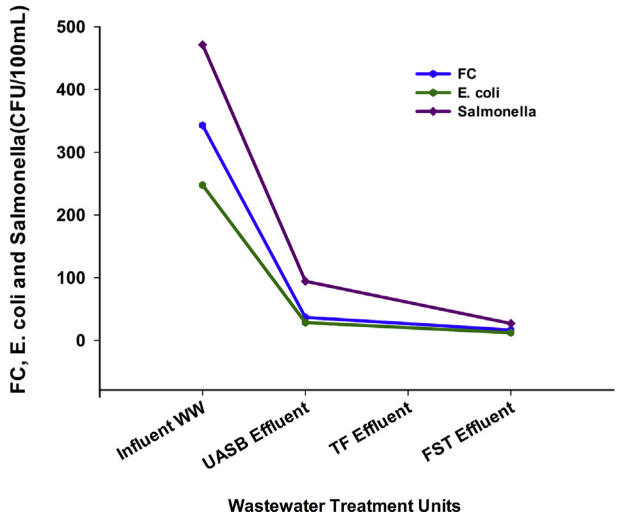

Figure 5. Mean levels of microbial loads at the various treatment units.

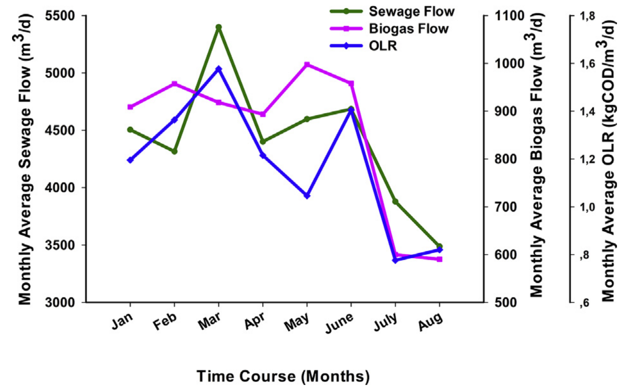

Figure 6. Variations in OLR, sewage and biogas flows.

sewage flow and biogas generation rates ($r^2 = 0.95, p < 0.001$). However, no significant relationship between OLR and biogas generation rate could be established ($r^2 = 0.24$, $p = 0.255$). This finding could be ascribed to the fact that the Plant operates under capacity as stated early on. Additionally, it could be also attributed to the fact that the applied OLR to the Mudor UASB reactors (1.30 $\pm$ 0.79 kg COD/m^3/d) is much lessor than the optimum range of 2–14 kg COD/m^3/d reported in literature [31]. Contrary to this finding, several authors; Ince et al. [108] and Musa et al. [109] have reported studies they conducted wherein there existed a direct correlation between applied OLR and biogas production.

According to these authors, initial increment in OLR resulted in increase in biogas production, until after a certain concentration, after which any further OLR increment led to subsequent decrease in biogas production. The observations made by these authors could be attributed to the fact that the ideal OLR range for optimum biogas production by anaerobic digesters had been exceeded, hence the decline in biogas production [39,110]. It was observed that both sewage flow and OLR peaked in March, but the corresponding biogas flow rate was relatively low. This could result from a lower readily biodegradable COD fraction in the sewage; hence, the system could not produce equivalent biogas yield that correlates with increased OLR [111]. The lowest sewage flow with corresponding low organic loading and biogas flow was observed between July and August. This could be explained as a result of seasonal impacts, where during maximum precipitation (rainfall), a bypass is employed so that urban stormwater is not received at the Plant.

Methane production ranged from 188–1783 m^3/d, with an average of 719.2 $\pm$ 253 m^3/d. The Specific methane yield was observed to range from 0.03–0.37 m^3CH_4/kgCOD removed. This range is comparable to observations made by other authors [109,112,113], although these studies were conducted on laboratory and pilot scales.

Characterisation of biogas samples revealed an average methane composition of 65%, ranging from 54–77% in the biogas output. CO_2, O_2, and N_2 compositions were respectively in ranges 3.2–9.1%, 1.4–14.6% and 19.9–28.2%, as presented in Figure 7. H_2S gas concentration was detected to be between 78 and 314 ppm. Besides the relatively lower CH_4 fraction, other biogas compositions observed from this study were comparable to the study by Noyola et al. [114], who reported biogas composition of 70–80% CH_4, 5–10% CO_2, and 10–25% N_2 from domestic wastewater treatment. According to the authors, dissolved N_2 in influent wastewater probably caused the high nitrogen content in biogas generated in the UASB reactors. Similarly, Konaté et al. [115] reported on biogas composition for an anaerobic pond treating domestic wastewater to be 80.5%, 11.8%, 5%, 2.5%, and 0.2% for CH_4, N_2, O_2, CO_2, and other gases, respectively. This study's observed methane composition (54–77%) was lower than values reported (70–85%) by some authors for UASB reactors treating sewage [15,116]. The relatively lower methane composition in biogas regarding this study could result from many factors such as sludge activity, Plant loading, etc.

3.4.2. Methane dissolution in UASB effluent

Several studies have reported that domestic wastewater treatment with UASB reactors usually produces biogas with high methane concentrations, however, a significant portion of the methane remains dissolved in solution and gets discharged together with the effluent or by other means [114,117]. Therefore, the dissolved methane (M_d in l/d) in the UASB effluent was estimated using the equation, $M_d = Q \times M_c \times \alpha \times 100$ [118], where Q is the biogas production (l/d), M_c is the methane gas concentration (%), and α is the Bunsen's absorption coefficient; 0.03469 ml CH_4 [119]. Figure 8 illustrates the total methane, gaseous and estimated dissolved CH_4 generated during the study period.

As presented in Figure 8, the calculated dissolved methane in the UASB effluent was found to be approximately 35% of the gaseous methane produced. Masuda et al. [120] and Kong et al. [121] reported a lower range of 19.8–22.3% of the total methane yield being dissolved. However, Keller and Hartley [122] opined that methane losses due to dissolution in the effluent of anaerobic systems could range from 20–60%. Souza et al. [116] had reported a range from 36–41%, and Cookney et al. [119] reported a range between 45 and 88%. According to these authors, the wide variations could be due to several factors such as temperature, loading, reactor type etc. The highly significant volume of methane lost in wastewater effluent reduces the energy recovery potentials of anaerobic wastewater treatment systems. Moreover, it presents an issue of environmental concern as methane is a potent greenhouse gas

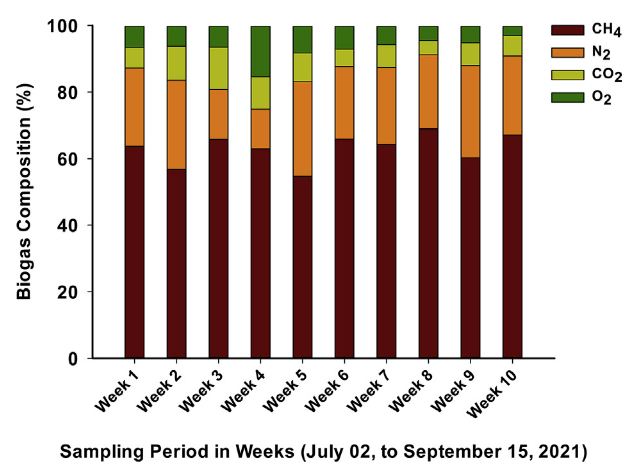

Figure 7. Biogas composition.

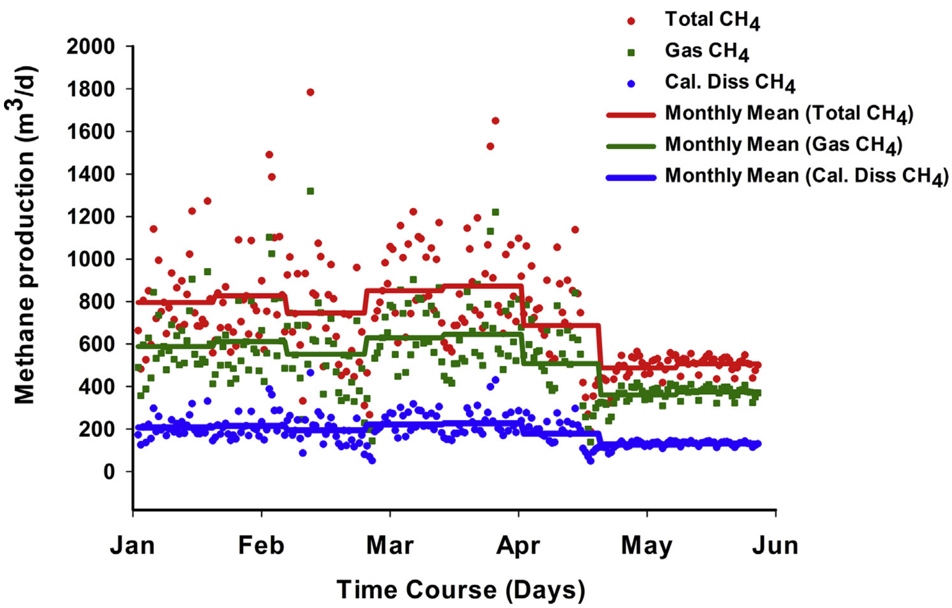

Figure 8. Comparison of total, gaseous and calculated dissolved methane

(GHG), having a global warming potential (GWP) about 28 times higher than carbon dioxide [123]. Henares et al. [124] similarly reported that methane emissions could generate explosive environment when effluent are discharged into drains or enclosed space. Several studies have been conducted in the quest to strip the dissolved methane from wastewater effluent [119,125,126].

4. Conclusion and recommendations

This study assessed the performance of the full-scale MWWTP in Accra, the capital city of Ghana, for municipal wastewater treatment. The Plant combines UASB reactors with trickling filters and final settling tanks as post-treatment units. The authors found that the UASB reactors operated at mesophilic temperature and under generally favourable operational conditions including HRT of 45.8 ± 24.9 h, Vel_{up} of 1.0 ± 0.2 m/h, and OLR of 1.3 ± 0.8 kgCOD/m^3/d. The Plant showed high removal efficiencies for COD (93%), BOD (98%), TSS (93%), and microbial loads (94–95%). The post-treatment units subsequently enhanced the Plant's performance for solids, organic matter and microbial loads removal, ensuring the effluent quality met the regulatory (EPA Ghana) discharge guidelines for municipal sewage. Nevertheless, the Plant failed in removing adequate nutrients (N and P) from the sewage as the final effluent contained significant concentrations; 84 mg/L of TN and 29 mg/L of TP, exceeding regulatory discharge guidelines for nutrients. Poor nutrient removal performance is attributed to the absence of a recirculation system at the TFs to enhance nitrification process, insufficient residual organic carbon in the UASB effluent to facilitate denitrification at the TFs, and the overall C:N:P nutrient imbalance in the influent sewage. The UASB reactors attained biogas yield of 0.2 m^3 per kg COD removed, with an average daily biogas production of 831.6 ± 292.7 m^3. The methane component of the biogas produced from the treatment system was estimated at 65% averagely.

With the results obtained from this study, the following recommendations have been made:

- Recovery of methane-rich biogas and dissolved methane in effluent sewage into energy could ensure energy self-sufficiency of the Plant, promoting a sustainable wastewater management.
- The Mudor WWTP has been ineffective in removing nutrients (nitrogen and phosphorus compounds). Hence, modern technologies which are effective in nutrient removal should be incorporated in the Plant's treatment unit configuration to enhance nutrient removal. Moreover, as urban vegetable agriculture is a common practice in Accra; the study area, the authors recommend conveyance systems be installed which would permit the delivery of the pathogen-free and nutrient-rich effluent from the Plant to urban farmers to be employed for urban irrigation. This will prevent eutrophication in receiving water bodies, which causes degradation of water reservoirs whilst boosting food production in the urban city of Ghana.

Declarations

Author contribution statement

Philomina M. A. Arthur: Conceived and designed the experiments; Performed the experiments; Analyzed and interpreted the data; Wrote the paper.

Yacouba Konaté; Boukary Sawadogo: Conceived and designed the experiments; Contributed reagents, materials, analysis tools or data; Wrote the paper.

Gideon Sagoe; Bismark Dwumfour-Asare; Issahaku Ahmed: Analyzed and interpreted the data; Contributed reagents, materials, analysis tools or data; Wrote the paper.

Myron N. V. Williams: Contributed reagents, materials, analysis tools or data; Wrote the paper.

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Data availability statement

Data will be made available on request.

Declaration of interest statement

The authors declare no conflict of interest.

Additional information

Supplementary content related to this article has been published online at https://doi.org/10.1016/j.heliyon.2022.e10129.

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