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Application of Upflow Anaerobic Sludge Blanket Reactor coupled with Trickling Filters for Municipal Wastewater Treatment:

Technical, Environmental and Economic Assessment in the Tropical Developing Country of Ghana

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Declaration

I hereby declare that this thesis is the result of my original work and that no part of it has been presented for a degree in the institution or elsewhere. The work of others which served as a source of information for this study has been duly acknowledged in the references.

Abstract

Wastewater management remains one major challenge in most developing countries in sub-Saharan Africa (SSA) due to the lack of adequate infrastructure for wastewater disposal, collection, and treatment. This challenge has been exacerbated by rapid population growth, urbanization, and industrialization. Meanwhile, untreated wastewater contains pollutants such as toxic compounds and pathogens which put at risk public health and the receiving ecosystems. Wastewater is, however, composed of rich resources such as freshwater, nutrients and energy, which can be harnessed using modern and eco-friendly technologies that employ circular economy principles, promoting sustainable wastewater management. The Upflow Anaerobic Sludge Blanket (UASB) reactor has been identified as an efficient and low-cost wastewater treatment option which has been widely implemented in several regions of the world, especially in Latin America and India. This technology is, however, less represented in SSA despite the favourable climatic conditions. This study, therefore, evaluated the technical, environmental, and economic sustainability of a full-scale UASB reactor coupled with Trickling Filters as post-treatment units in Accra, the capital city of Ghana in the West African sub-region. The technical assessment revealed satisfactory system performance with over 70% removal efficiency for solids, organic matter, and microbial loads, whilst post-treatment further enhanced the removal of these contaminants to acceptable limits set by the Environmental Protection Agency (EPA), Ghana. However, the system could not adequately remove nutrients, with nitrogen and phosphorous compounds far exceeding discharge limits. Daily biogas production was between 101 Nm^3/d and 1673 Nm^3/d, with an average daily production of 613 ± 271 Nm^3/d, corresponding to a specific yield of 0.14 ± 0.07 m^3biogas/kgCOD removed. Biogas produced contained 65% of methane. However, 23% of the methane generated remained dissolved in the effluent, reducing biogas energy recovery potential. Environmental assessment employed the IPCC GHG inventory methodology to measure the carbon footprints of the full-scale municipal wastewater treatment plant. It was found from the study that the total greenhouse gas (GHG) emissions from the operations of the Mudor Plant were estimated at 39,619.36 tCO_2eq/yr. CO_2 emissions from energy consumption were estimated at 165.74 tCO_2eq/yr, constituting 8.5% of the total emissions. Dissolved methane in the effluent was identified as the single most significant source of GHG emissions, with over 90% contribution at 37,676.67 tCO_2eq/yr. A cost-benefit analysis was employed for the economic assessment. Cost analysis revealed that staff management was responsible for the highest percentage (37%)

of the operational cost, whilst energy consumption of the anaerobic-based wastewater treatment plant was only 7.3% of the total operational cost. Benefit analysis carried out employing resource recovery revealed that energy recovery potential (534.1 MWh/yr) from biogas and sludge generated by the Plant could completely offset the total Plant energy demand (392.7 MWh/yr). Additionally, it was found that the nutrient-rich effluent had lower heavy metals concentration with acceptable microbial load count for urban irrigation. Thus, the UASB reactor technology presents an efficient, economically feasible and sustainable wastewater treatment alternative that can be implemented in developing countries towards the attainment of sustainable wastewater management for sustainable development in this part of the world.

Keywords: Economic assessment; Environmental assessment; Ghana; Municipal wastewater treatment; Technical assessment; Trickling filter; UASB reactor.

Résumé

La gestion des eaux usées reste un défi majeur dans la plupart des pays en développement notamment ceux d'Afrique subsaharienne confrontés au manque d'infrastructures adéquates pour la collecte, l'évacuation, et le traitement de ces eaux. Ce défi est davantage exacerbé par la croissance rapide de la population, l'urbanisation galopante et l'industrialisation. Les eaux usées non traitées contiennent pourtant des polluants divers tels que des composés toxiques et des agents pathogènes qui mettent en danger la santé publique et les écosystèmes récepteurs. Elles sont cependant composées de riches ressources telles que de l'eau douce, des nutriments et de l’'énergie qui peuvent être exploitées en utilisant des technologies adaptées et respectueuses de l'environnement, en appliquant les principes de l'économie circulaire, ce qui favoriserait une gestion durable des eaux usées. Le réacteur à lit de boues anaérobie à flux ascendant (UASB) est considéré comme une option technologique efficace de traitement des eaux usées qui a été largement mise en œuvre dans plusieurs régions du monde, notamment en Europe, en Amérique latine et en Inde. Cette technologie est cependant moins représentée en Afrique subsaharienne malgré les conditions climatiques qui semblent être favorables à son fonctionnement. Cette étude de thèse a porté sur l’évaluation de la viabilité technique, environnementale et économique d'un réacteur UASB grandeur réelle couplé à des lits bactériens comme unités de post-traitement à la STEP de Mudor à Accra, capitale du Ghana dans la sous-région ouest-africaine. L'évaluation technique a révélé une performance satisfaisante du système UASB avec une efficacité d'élimination de plus de 70% pour les paramètres de pollutions considérés (matières solides, les matières organiques et les charges microbiennes), tandis que le post-traitement par lits bactériens a encore amélioré l'élimination de ces contaminants jusqu'aux limites acceptables fixées par l’Agence de Protection de l’Environnement (EPA) du Ghana. La quantité journalière de biogaz produite est comprise entre 101 Nm^3/j et 1673 Nm^3/j, avec une production moyenne journalière de à 613 ± 271 Nm^3/j, correspondant à un rendement spécifique de 0.140 ± 0.07 m^3biogaz/kgDCO traitée. La qualité du biogaz produit contient en moyenne 65% de méthane. Cependant, 23% du méthane généré est dissous dans l'effluent, ce qui réduit le potentiel de valorisation énergétique du biogaz. L'évaluation environnementale a utilisé la méthodologie du GIEC pour mesurer l'empreinte carbone de la station de traitement des eaux usées municipales de Mudor. L'étude a révélé que les émissions totales de gaz à effet de serre (GES) provenant des opérations de cette station s’élevaient à 39 619,36 tCO_2eq/an. Les émissions de CO_2 provenant de la consommation d'énergie ont été estimées à 165,74 tCO_2eq/an, soit 8,5% des émissions totales. Le méthane dissous dans les effluents a été identifié comme la source la plus importante d'émissions de GES avec une contribution de plus de 90%, soit 37 676,67 tCO_2eq/an. Une analyse coûts-avantages a été utilisée pour l'évaluation économique. Elle a révélé que la gestion du personnel était responsable du pourcentage le plus élevé (37%) des charges d'exploitation, tandis que la consommation d'énergie de la station d'épuration ne représentait que 7,3% des charges globales d'exploitation. L'analyse des bénéfices effectuée en utilisant la valorisation des ressources a révélé que le potentiel de récupération d'énergie (534,1 MWh/an) à partir du biogaz et des boues générées par la station pourrait compenser complètement la demande totale d'énergie de la station (392,7 MWh/an). Il a également été constaté que l'effluent traité riche en nutriments présentait une concentration plus faible en métaux lourds et une charge microbienne acceptable pour réutilisation en agriculture. Fort de ces résultats, la technologie du réacteur UASB présente donc une alternative de traitement efficace des eaux usées, économiquement réalisable et qui peut être mise en œuvre dans les pays en développement en vue d'une gestion des eaux usées pour le développement durable dans cette partie du monde.

Mots clés : Évaluation économique ; Évaluation environnementale ; Évaluation technique ; Ghana ; Lits bactériens ; Réacteur UASB ; Traitement des eaux usées municipales.

List of Publications

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I. Title

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- Arthur, P.M.A.; Konaté, Y.; Sawadogo, B.; Dwumfour-Asare, B.; Sagoe, G.; Ahmed, I.; Amofa-Sarkodie, E. Carbon Footprints of a Full-scale Municipal Wastewater Treatment Plant – Case of the Mudor Wastewater Treatment Plant in Accra, Ghana (Manuscript).

IV. Scientific Communications:

Oral Presentations

- Arthur, P.M.A.; Konaté, Y.; Sawadogo, B.; Sagoe, G.; Dwumfour-Asare, B.; Ahmed, I. (2022). Application of Circular Economy Concept Towards a Sustainable Wastewater Management: Case Study of a Full-Scale UASB Reactor in a Developing Country. *2022 European Wastewater Management, Conference and Exhibition*. The Hilton Birmingham Metropole. The UK.
- Arthur, P.M.A.; Konaté, Y.; Sawadogo, B. Application of Upflow Anaerobic Sludge Blanket Reactor for Wastewater Treatment: Technical, Environmental and Socio-Economic Assessment in Developing Countries. 8th Edition of 2iE Doctoriales, 2020. Ouagadougou, Burkina Faso ***(Best oral presentation)***.
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- Arthur, P.M.A.; Konaté, Y.; Sawadogo, B. Integrated Application of Upflow Anaerobic Sludge Blanket - Pond Combined Technology for Urban Wastewater Treatment in Sahelian Climate. 7th Edition of 2iE Doctoriales, « Science, Innovation et Entreprenariat pour le Développement. » 2019.

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- Arthur, P.M.A.; Konaté, Y.; Sawadogo, B. Integrated Application of Upflow Anaerobic Sludge Blanket - Pond Combined Technology for Urban Wastewater Treatment in Sahelian Climate. 7th Edition of 2iE Doctoriales, « Science, Innovation et Entreprenariat pour le Développement. » 2019.

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Résumé Substantiel en Français

Résumé substantiel en français

Couplage de réacteurs anaérobies à lit de boues à flux ascendant (UASB) et de lits bactériens pour le traitement des eaux usées municipales au Ghana :

Evaluation Technique, Environnementale et Économique

Introduction

La rareté des ressources en eau douce et la gestion des eaux usées sont deux problèmes mondiaux liés dont l'importance actuelle et future ne cesse de croître. L'épuisement des ressources en eau douce exacerbé par la démographie galopante, associé aux défis de la gestion adéquate des eaux usées, pose à l'humanité les problèmes de pénurie d'eau imminente, de contamination de l'environnement et de risques pour la santé humaine. L'extrême rareté des ressources en eau douce a conduit à l'utilisation de sources d'eau alternatives et parfois non conventionnelles pour augmenter l'approvisionnement en eau douce. Ainsi, les sources d'eau non conventionnelles telles que les eaux usées et l'eau de mer sont devenues un important approvisionnement en eau potable dans certains endroits en raison de facteurs contraignants tels que la géographie, le climat et l'augmentation de la demande en eau douce (Amy et al., 2017 ; Quentin Grafton, 2017).

Les stations d'épuration des eaux usées (WWTP) sont conçues pour éliminer la pollution de l'environnement par le rejet d'eaux usées non traitées dans les masses d'eau de surface, protégeant ainsi les sources d'eau douce et éliminant également les risques associés pour la santé humaine. En outre, la connaissance des conséquences dévastatrices du rejet d'eaux usées non traitées dans l'environnement a conduit à la mise en place de directives strictes par les gouvernements afin de contrôler la qualité des effluents rejetés. Cependant, la production d'effluents de haute qualité respectant les directives de rejet pour la plupart des pays nécessite d’énormes ressources, incluant principalement des coûts élevés d'investissement, d'énergie et d'exploitation (Shannon et al., 2008).

Si les régions développées du monde peuvent s’équiper de systèmes de traitement des eaux usées conventionnels très avancés et efficaces, généralement basés sur des procédés intensifs énergivores comme les boues activées, les pays en développement sont confrontés à la situation désastreuse du manque de systèmes de traitement des eaux usées efficaces, économiquement réalisables et durables (Martinez-Sosa et al., 2012). La gestion des eaux usées reste donc un défi majeur auquel sont confrontés la plupart des pays en développement d'Asie et d'Afrique

subsaharienne (UN Wastewater Report, 2017). La situation est aggravée par la croissance rapide de la population dans ces régions.

Les eaux usées contiennent des contaminants et des agents pathogènes qui sont nocifs pour la santé publique et les écosystèmes récepteurs s'ils sont rejetés sans traitement dans l'environnement. Elles sont néanmoins très riches en nutriments et en matières organiques qui peuvent être exploitées en ressources utiles. Les procédés de traitement anaérobie des eaux usées (AnWT) semblent être une alternative prometteuse car ils sont moins énergivores et simples à mettre en œuvre (Van Lier et al., 2008). De plus, ils sont généralement adaptés aux régions climatiques chaudes. L'avantage supplémentaire de la récupération des ressources à partir du biogaz et des boues en fait une technologie efficace et économiquement réalisable qui peut être mise en œuvre dans les pays en développement pour une gestion efficace et durable des eaux usées (Van Lier et al., 2008). Les technologies AnWT habituellement mises en œuvre comprennent les réacteurs anaérobies à lit de boues à flux ascendant (UASB), les filtres anaérobies, les réacteurs à lit fluidisé, les contacteurs biologiques rotatifs, les lits de boues granulaires expansées, les bassins de lagunage à microphytes (WSP), etc.

Parmi les technologies AnWT disponibles, le réacteur UASB est devenu plus populaire, avec plusieurs unités pilotes et à grande échelle installées dans des pays comme le Brésil, l'Inde, le Japon et la Colombie (Lettinga, 2005 ; Passos et al., 2020). La technologie du réacteur UASB présente ainsi des avantages considérables par rapport aux autres systèmes de traitement anaérobie, ce qui explique sa large acceptation dans plusieurs parties du monde, malgré son existence relativement courte par rapport aux autres technologies anaérobies (Chernicharo et al., 2015). Parmi ces avantages figure la capacité des systèmes de réacteurs UASB à supporter des fortes charges organiques et fluctuantes (Leitão, 2004). Wolmarans & De Villiers (2002) et Musa et al. (2019) ont rapporté que des réacteurs UASB en fonctionnant à pleine charge pouvaient atteindre une efficacité épuratoire allant jusqu'à 90 % pour des eaux usées brutes concentrées à 30 000 mg/L de DCO. Selon Hulshoff Pol et al. (2004), le développement de granules biologiques dans le lit de boues est la caractéristique technologique la plus importante qui permet aux réacteurs UASB de gérer des charges organiques élevées par rapport à d'autres systèmes anaérobies. Les réacteurs UASB produisent moins de boues stabilisées que les systèmes aérobies, et le biogaz généré par ces réacteurs contient une proportion considérable de gaz méthane qui peut être exploité à des fins de récupération d'énergie (Foresti et al., 2006)

La valorisation des ressources des stations d'épuration (STEP) est un facteur favorisant une gestion durable des eaux usées. En ce qui concerne la technologie des réacteurs UASB, la récupération des ressources sous la forme de biogaz riche en méthane pour la production d'énergie, de boues sous forme de biosolides pour le compostage ou à des fins agricoles et la fertilisation avec des effluents riches en nutriments, associées aux faibles coûts d'exploitation de ces systèmes, ont été les principales raisons de leur large acceptation, faisant de cette technologie une option durable pour les pays en développement (Chernicharo et al., 2015 ; Lettinga et al., 1980).

Malgré l'intérêt croissant de la technologie UASB dans plusieurs parties du monde notamment dans les régions tropicales d'Amérique latine et d'Inde (Chernicharo et al., 2015 ; Lettinga, 2005 ; Passos et al., 2020), cette technologie est assez méconnue dans la plupart des pays en développement de la sous-région ouest-africaine, seules quelques études se sont intéressées dans la sous-région (Ahmed et al., 2018 ; Awuah & Abrokwa, 2008). Par conséquent, son développement et sa mise en œuvre sont très malgré la faisabilité économique. Plusieurs facteurs justifient l'adaptation de la technologie des réacteurs UASB aux pays en développement de la sous-région. Le plus important est la condition climatique de ces régions favorable au fonctionnement efficace des systèmes anaérobies dans une plage de température mésophile (20 - 40 °C). Ainsi, sur le plan climatique, les conditions climatiques prévalant dans la région sont très propices au fonctionnement optimal des réacteurs UASB. En termes de coût d'exploitation, le réacteur UASB est considéré comme économiquement faisable par rapport aux systèmes conventionnels à boues activées, où des aérateurs sont utilisés pour l'optimisation des microorganismes aérobies, augmentant la consommation d'énergie et par conséquent le coût d'exploitation (Lettinga et al., 1980). La littérature a rapporté que ces systèmes de traitement aérobies des eaux usées produisent des volumes élevés de boues, dont la gestion pourrait être responsable d'environ 40 à 60 % du coût opérationnel total (Domini et al., 2022 ; Foladori et al., 2015). Contrairement aux systèmes aérobies, les réacteurs anaérobies produisent des volumes de boues relativement moindres, ce qui réduit les coûts de leur gestion (Chernicharo et al., 2015). De plus, un avantage reconnu des systèmes anaérobies est la production de biogaz riche en méthane. Les réacteurs UASB traitant les eaux usées domestiques pourraient produire du biogaz avec jusqu'à 80 % de méthane (Noyola et al., 2006). Ce méthane peut être récupéré pour produire de l'énergie utilisable pour compenser les besoins énergétiques du système. Ainsi, avec la technologie des réacteurs UASB, le concept «

Sanitation Financing Sanitation » peut être actualisé d’où l’interêt de cette étude de thèse de doctorat

Objectifs de l’étude

L'objectif principal de cette étude est d'évaluer la durabilité technique, environnementale et économique des systèmes UASB traitant les eaux usées municipales au Ghana dans la sous-région ouest-africaine.

Les objectifs spécifiques de l'étude sont :

• Évaluer les performances d'un réacteur UASB couplé à des lits bactérien (LB) à l’échelle réelle pour le traitement des eaux usées municipales ;

• Mesurer les empreintes carbone des opérations du système UASB/LB à grande échelle afin d’identifier les contraintes environnementales y relatives et proposer d'éventuelles mesures d'atténuation ;

• Effectuer une analyse coûts-avantages du système UASB/LB pour une utilisation durable dans les pays en développement ;

Matériel et méthodes

Site d’étude

Cette étude a été menée à la station d'épuration de Mudor à Accra, la capitale du Ghana. Cette station comprend 6 réacteurs UASB de forme modulaire, 3 lits bactériens et 2 décanteurs finaux (clarificateurs) servant d’unités de post-traitement pour les réacteurs UASB dans le traitement des eaux usées municipales de certaines banlieues de la métropole d'Accra.

Quantification et échantillonnage des eaux usées et du biogaz

Des flacons de prélèvement d’un litre ont été utilisés, pour l’échantillonnage des eaux usées avec une fréquence de deux fois par semaine pour l’analyse des paramètres de pollution organique et une fois par semaine pour l'analyse des nutriments, des métaux lourds, des indicateurs de contamination fécale et des parasites. Les échantillons ont été transportés dans une glacière avec des accumulateurs de froid dans les 24 heures pour analyse en laboratoire ou stockage dans un réfrigérateur à 4 °C, le cas échéant. Le biogaz a été collecté en utilisant des sacs Tedlar.

Les débits de biogaz et d'eaux usées ont été mesurés à l'aide d'appareils de mesure de débit automatiques installés (débitmètre Endress+Hauser et promag 53H, Suisse). Les échantillons de biogaz ont été prélevés dans les 6 réacteurs UASB pour être caractérisés, sur une période de dix (10) semaines (du 02 juillet au 15 septembre 2021).

Caractérisation des constituants du biogaz, des eaux usées et des boues

Les principaux constituants du biogaz, à savoir le méthane (CH_4), le dioxyde de carbone (CO_2), l'oxygène (O_2), l'azote (N_2) et le sulfure d'hydrogène (H_2S) ont été analysés avec un analyseur de gaz potable FM 406 (Gas Data, Royaume-Uni). Les eaux usées et les boues ont été caractérisées pour les paramètres physico-chimiques et microbiens. Les paramètres in situ (pH, température, conductivité électrique et oxygène dissous) ont été mesurés sur site avec un analyseur multi-sondes portable (HQ40D LDO10101, marque Hach - USA), tandis que le reste des paramètres a été analysé dans le Sewerage Systems Ghana Limited (SSGL) Laboratoire sur place. La demande biochimique en oxygène (DBO) a été analysée à l'aide du test DBO de 5 jours (APHA 5210). La demande chimique en oxygène (DCO) a été analysée en utilisant la digestion au dichromate de potassium avec l'instrument HACH (DR1900). Les solides totaux dissous (TDS) ont été mesurés avec l'électrode de mesure directe méthode. Les solides totaux (TS), les solides totaux en suspension (TSS) et les matières volatiles solides (MVS) ont été déterminés par séchage et combustion à 105 ^{o}C et 550 ^{o}C respectivement conformément aux méthodes APHA 2540, l'alcalinité totale a été mesurés à l'aide de pastilles de photomètre Lovibond Alka-M broyés en solution et lus avec un spectrophotomètre. Les Acides Gras Volatils (AGV) ont été analysés en utilisant la méthode de distillation telle que décrite dans APHA 5560 C.

Pour les nutriments, le phosphore total (PT) a été mesuré par la procédure de digestion au persulfate (méthode HACH 10209), tandis que l'orthophosphate (PO_4^{3-}-P) a été déterminé par la procédure d'analyse colorimétrique directe (méthode HACH 10210) et mesuré avec le spectrophotomètre DR3900. L'azote total (NT) a été mesuré à l'aide de la méthode de digestion au persulfate (méthode HACH 10208), l'azote ammoniacal (NH_3-N) a été déterminé à l'aide de la méthode Salicylate (méthode HACH 10031), l'azote nitrique (NO_3^--N) a été analysé à l'aide de la méthode de réduction au cadmium (méthode HACH 8039), le sulfate (SO_4^{2-}) a été analysé à l'aide de la méthode SulfaVer4 (méthode HACH 8051), le sulfure a été déterminé par spectrophotométrie à l'aide de la méthode au bleu de méthylène (méthode HACH 8131). Des métaux lourds sélectionnés (Zn, Cu, Cd, Pb, Ni, Hg, Mn, Cr) ont été mesurés par spectrométrie d'absorption atomique (spectromètre Perkin Elmer A Analyst 800, USA). Les coliformes fécaux (FC) dont *E. coli* et Salmonella sp. ont été analysés en utilisant la méthode par étalement sur des boites de petri coulée avec un milieu gélosé (APHA 9222), tandis que la caractérisation des œufs d'helminthes a été effectuée selon la méthodologie proposée par Moodley et al. 2008. Toutes les analyses ont été effectuées conformément aux méthodes standards (APHA, 2017).

Quantification du méthane dissous dans les effluents

Plusieurs études ont rapporté que le traitement des eaux usées domestiques avec des réacteurs UASB produit généralement du biogaz avec des concentrations élevées de méthane, cependant, une partie importante du méthane reste dissoute en solution et est rejetée avec l'effluent ou par d'autres moyens (Gupta & Goel, 2019 ; Noyola et al., 2006). Par conséquent, le méthane dissous (dCH_4) dans l'effluent de l'UASB a été estimé à l'aide de l'équation proposée par Asano et al. (2021) :

$M_d = Q * M_c * \alpha * 100$ … … … … *(Eqn. 1)*

Où M_d est le méthane dissous (L/j), Q est la production de biogaz (L/j), M_c est la composition en pourcentage de méthane dans le biogaz, α est le coefficient de solubilité de Bunsen pour le méthane. Comme l'ont rapporté Yamamoto et al. (1975), α pour l'eau non saline à 30 °C vaut 0,02898 ml CH_4.

Estimation de l'empreinte carbone d'une station d'épuration UASB/Lit bactérien

La méthodologie employée pour cette étude est basée sur les lignes directrices affinées du Groupe d'Experts Intergouvernemental sur l'Évolution du Climat (GIEC) recommandées en 2006 et 2019 pour l'évaluation des inventaires de Gaz à Effet de Serre (GES). Les émissions de CH_4 sur site ont pris en compte les émissions dues à la combustion incomplète du CH_4 lors du torchage du biogaz, les émissions dues aux fuites de CH_4 dans le système et les émissions dues à la déshydratation des boues sur les lits de séchage. Les émissions de CO_2 sur site ont été prises en compte pour la combustion de combustibles fossiles, tandis que les émissions de N_2O sur site provenant des processus d'élimination biologique des nutriments sur LB ont également été prises en compte. Les émissions hors site ont été estimées à partir de l'utilisation de l'électricité du réseau à l'usine, du méthane dissous (dCH_4) qui est émis par l'effluent dans le plan d'eau récepteur et des émissions de N_2O provenant du rejet de l'effluent dans le plan d'eau récepteur.

Emissions sur site

- *Les émissions de méthane lors du traitement des eaux usées ($E_{CH_4\text{-}WWT}$) des réacteurs anaérobies ont été estimées à l'aide de l'équation :*

$$E_{CH_4-WWT} = \left[((TOW - S_{COD}) * EF - R) * 10^{-3})) * GWP_{CH_4}\right] \ldots\ldots\ldots\ldots\ldots \quad \textit{(Eqn. 2)}$$

Où $E_{CH_4\text{-}WWT}$ correspond aux émissions de méthane provenant du traitement anaérobie au cours de l'année d'inventaire (tCO_2/an), *TOW* est la charge organique totale dans les eaux usées au cours de l'année d'inventaire (kgDCO/an), S_{DCO} est la masse de DCO convertie en boues au cours de l'année d'inventaire (kgDCOboues/an), *EF* est le facteur d'émission pour les réacteurs UASB ($kgCH_4$/kgDCO), *R* est la quantité de CH_4 récupérée ou brûlée au cours de l'année d'inventaire ($kgCH_4$/an), 10^{-3} est le facteur de conversion de kg en tonnes, GWP_{CH_4} est le potentiel de réchauffement global du méthane (IPCC = 28 CO_2eq).

- *Les émissions de méthane provenant du torchage du biogaz ont été calculées avec l'équation suivante :*

$$PE_{flare} = GWP_{CH_4} * \sum_{h=1}^{8670} MF_{CH_4}\left(1 - \eta_{flare}\right) * 10^{-3} \ldots\ldots\ldots \quad \textit{(Eqn. 3)}$$

Où PE_{flare} est les émissions provenant du torchage du biogaz résiduel au cours de l'année d'inventaire (tCO_2eq/an), GWP_{CH_4} est le potentiel de réchauffement global du méthane (GIEC = 28 tCO_2eq), $\eta flare$ est l'efficacité de la torche (valeur par défaut de la CCNUCC pour la torche ouverte = 50 %) , *8670* est le nombre d'heures dans une année, 10^{-3} est le facteur de conversion de kg en tonnes, MF_{CH_4} est le débit massique de méthane dans le gaz résiduel par heure (kg/h).

- *Les émissions dues aux fuites de méthane ont été estimées avec l'équation suivante :*

$$PE_{CH_4} = Q_{CH_4} * EF_{CH_4,default} * GWP_{CH_4} \ldots\ldots\ldots\ldots \textit{(Eqn. 4)}$$

Où PE_{CH_4} est les émissions du projet dues aux fuites de CH_4 au cours de l'année (tCO_2eq/an), Q_{CH_4} est la quantité de CH_4 générée par le réacteur anaérobie au cours de l'année (tCH_4), $EF_{CH_4,\,default}$ est le facteur d'émission par défaut pour la fraction de CH_4 produit émanant réacteur

UASB (= 0,05 tCH_4fuite/tCH_4 produit), GWP_{CH_4} est le potentiel de réchauffement global du méthane (IPCC = 28 CO_2eq)

- *Les émissions de méthane provenant du traitement des boues ont été calculées avec l'équation :*

$$E_{CH_4-sludge} = M_{sl,dry} * MCF_{sl} * DOC_{sl,dry} * DOC_F * Ƒ_{CH_4} * \frac{16}{12} * GWP_{CH_4} \dots .... (Eqn.5)$$

Où $E_{CH_4\text{-}sludge}$ est les émissions de méthane provenant du séchage des boues dans l'année (tCO_2eq/an), $M_{sl,dry}$ est la masse de boues sèches produites dans l'année (t/an), MCF_{sl} est le facteur de conversion du méthane pour les boues (IPCC par défaut = 0,5 pour les lits de séchage), $DOC_{sl,}$ est la teneur organique dégradable (DOC) dans les boues sèches, (valeur par défaut du GIEC = 0,5 pour les boues domestiques), DOC_F est la fraction de DCO dissimilée au biogaz (valeur par défaut du GIEC = 0,5), $Ƒ_{CH_4}$ est la fraction de méthane dans le biogaz (valeur par défaut du GIEC = 0,5), $\frac{16}{12}$ est le rapport des masses molaires du méthane et du carbone, GWP_{CH_4} est le potentiel de réchauffement global du CH_4 (GIEC = 28 CO_2eq).

- *Les émissions provenant de la combustion du carburant diesel ont été déterminées à l'aide de l'équation :*

$$PE_{DF} = Q_{DF} * NCV_{DF} * EF_{DF} \dots \dots .... \quad (Eqn. 6)$$

Où PE_{DF} représente les émissions de CO_2 provenant de la combustion de carburant diesel dans l'année (tCO_2eq/an), Q_{DF} est la quantité de diesel consommée pour la production d'électricité dans l'année (litres), NCV_{DF} est la valeur calorifique nette du carburant diesel (0,036 GJ/litres), EF_{DF} est le facteur d'émission de CO_2 pour le carburant diesel (0,0741 tCO_2/GJ).

- *Les émissions de N_2O de la station d'épuration ont été calculées à l'aide de l'équation :*

$$GHG_{N_2O} = \sum(U_i * T_j * EF_{N_2O}) * TN_{load} * \frac{44}{28} * 10^{-3} * GWP_{N_2O} \dots \dots \dots \quad (Eqn. 7)$$

Où GHG_{N_2O} est les émissions de N_2O provenant du traitement des eaux usées (tCO_2eq/an), EF_{N_2O} est le facteur d'émission de N_2O (valeur par défaut du GIEC 2019 = 0,016 kgN_2O-N/kg N), TN_{load} est la charge en TN présente dans les eaux usées au cours de l'année (kg N/an), $\frac{44}{28}$ est le rapport de la masse molaire de N_2O à la masse molaire de N_2, 10^{-3} est la conversion de kg en tonnes, GWP_{N_2O} est le potentiel de réchauffement global de N_2O (GIEC = 298 CO_2eq), *Ui* est la fraction de la population dans le groupe de revenu (GIEC 2019 Tableau 6.5). U_1 représente les hauts revenus urbains (0,1) et U_2 représente les faibles revenus urbains (0,38). T_j est le degré d'utilisation de la voie de traitement ou de rejet (égouts), T_1 représente les hauts revenus urbains (0,37) et T_2 représente les faibles revenus urbains (0,34) et où *3* indique l'utilisation des égouts.

Emissions hors site

- *Le méthane dissous (dCH_4) dans les effluents d'eaux usées est donné par l'équation suivante :*

$$PE_{dissolved} = Q_{ww} * CH_{4-ww} * GWP_{CH_4} \quad \text{………………} \quad \textit{(Eqn. 8)}$$

Où $PE_{dissolved}$ correspond aux émissions de dCH_4 dans les effluents d'eaux usées au cours de l'année (tCO_2eq/an), Q_{ww} est le volume d'eaux usées traitées au cours de l'année (m^3/an), CH_{4-ww} est la concentration de dCH_4 dans les effluents d'eaux usées au cours de l'année ($tonnes/m^3$), GWP_{CH_4} est le potentiel de réchauffement global du méthane (IPCC = 28 CO_2eq).

- *Les émissions de N_2O des effluents d'eaux usées domestiques ont été déterminées à l'aide de l'équation :*

$$GHG_{N_2O-Effluent} = TN_{Effluent} * EF_{N_2O-Effluent} * \frac{44}{28} * 10^{-3} * GWP_{N_2O} \dots\dots\dots \quad \textit{(Eqn. 9)}$$

Où $GHG_{N_2O\text{-}Effluent}$ correspond aux émissions de N_2O des effluents d'eaux usées rejetés dans la le milieu récepteur au cours de l'année (tCO_2eq/an), $TN_{Effluent}$ est la charge TN dans les effluents rejetés dans le milieu récepteur(kg/an), EF_{N_2O} est l'émission facteur pour le rejet d'effluents dans un environnement aquatique non eutrophe ou non impacté par des éléments nutritifs (valeur par défaut du GIEC = 0,005 kg N_2O-N/kg N), $\frac{44}{28}$ est le rapport de la masse molaire de N_2O à la masse molaire de N_2, 10^{-3} est la conversion de kg en tonnes, GWP_{N_2O} est le potentiel de réchauffement global de N_2O (IPCC = 298 CO_2eq).

- *Émissions provenant de la consommation d'énergie provenant de l'utilisation de l'électricité du réseau à l'aide de l'équation :*

$$E_{electr} = \sum Q_{electr} * EF_{electr} \dots\dots\dots\dots \quad (Eqn.\ 10)$$

Où E_{electr} représente les émissions de GES associées à la consommation d'électricité dans l'année (tCO_2eq/an), Q_{electr} est la quantité d'électricité consommée par les opérations de la station d'épuration dans l'année (MWh/an), $EF_{electric}$ est le facteur d'émission de CO_2 du réseau électrique national pour le Ghana (tCO_2eq/MWh).

Evaluation économique de la station d'épuration de Mudor

Analyse des coûts de la station d'épuration de Mudor

Les coûts d'investissement constitués des dépenses ponctuelles telles que le coût d'acquisition du terrain, le coût de construction, l'équipement mécanique, les structures de soutien et les bâtiments, ainsi que les coûts récurrents qui comprennent les réparations et l'entretien périodiques, les services publics (coûts de l'eau et de l'électricité), les dépenses administratives, les dépenses de gestion du personnel, etc. ont été évalués dans l'analyse des coûts de la station d'épuration de Mudor.

Analyse des avantages de la station d'épuration de Mudor

Pour évaluer l'analyse des avantages de la station d'épuration de Mudor, le potentiel de valorisation des ressources des différents sous-produits de la station de Mudor ont été évalués : l'eau récupérée, le biogaz et les boues produits. L'eau et les boues récupérées ont été évaluées pour la composition en éléments nutritifs, la charge microbienne et la concentration en métaux lourds afin d'évaluer leur faisabilité à des fins agricoles. Les boues ont de nouveau été analysées pour leur potentiel de récupération d'énergie grâce à l'application de procédés thermochimiques tandis que le biogaz a été évalué pour son taux de production volumétrique et sa teneur en méthane afin d'estimer le potentiel de valorisation énergétique. Pour le potentiel de valorisation énergétique des boues, la méthode calorimétrique a été utilisée pour évaluer la valeur énergétique des boues d'épuration. Les orifices de décharge des boues situés sur les côtés des réacteurs UASB ont été ouverts à l'aide de vannes désignées, ce qui a permis aux boues en excès d'être déchargées d'abord dans les épaississeurs de boues, puis sur les lits de séchage des boues. La procédure de laboratoire pour l'analyse calorimétrique pour la détermination de la teneur en énergie a suivi les méthodes décrites dans le manuel du calorimètre à bombe à oxygène de Parr (manuel Parr 1342, n° 204M), comme indiqué par la norme ASTM E711-87 (2004). Des échantillons de boues séchées à l'air ont été pesés, granulés et brûlés dans une atmosphère d'oxygène sous pression (30 atm). 1 g de granulés de boue ont été utilisés pour garantir une augmentation de la température dans la collonne d'eau, fournissant un environnement de combustion sûr, ne dépassant pas la plage optimale du thermomètre. L'acide benzoïque a été utilisé comme solution standard pour la détermination de la capacité calorifique de la bombe. Les tests ont été conduits en double. Les calculs du pouvoir calorifique supérieur (PCS) et du pouvoir calorifique inférieur (PCI) ont été effectués sur la base des directives fournies par le manuel Parr. La demande énergétique de la station par l'utilisation de l'électricité du réseau national et du carburant diesel utilisé pour alimenter les générateurs pour faire fonctionner les pompes à eaux usées pendant les interruptions de l'électricité du réseau a également été estimée. Le potentiel de valorisation énergétique du biogaz et des boues a été estimé par les équations :

$$EP_{biogas} = Q_{biogas} * C_{CH_4} * E_{CH_4} \dots\dots\dots \quad \textit{(Eqn.11)},$$

$$EP_{sludge} = P_{sludge} * NCV_s \dots\dots\dots\dots \quad \textit{(Eqn. 12)},$$

L'énergie potentielle totale (EP_{Total}) du biogaz et des boues a été déterminée avec l'équation :

$$EP_{Total} = EP_{biogas} + EP_{sludge} \dots\dots\dots\dots\dots \quad \textit{(Eqn. 13)}$$

Où EP_{Total} est le potentiel énergétique total (MJ/j), EP_{Biogas} est le potentiel énergétique du biogaz (MJ/j), Q_{Biogas} est le taux de production de biogaz (m^3/j), C_{CH_4} est la concentration de CH_4 dans le biogaz (%), E_{CH_4} est le PCI de la combustion du CH_4 (35,9 MJ/m^3), EP_{Sludge} est le potentiel énergétique des boues (MJ/j), P_{Sludge} est la production de matière sèche des boues (kg/j), *NCVs* est le pouvoir calorifique inférieur des boues (MJ/kg).

Résultats et discussion

Performances du système pour l'élimination de la matière organique et des matières solides

La concentration de DCO dans l'influent brut qui variait de 450 à 8150 mg/L a été sensiblement réduite à une plage comprise entre 226 et 1449 mg/L, atteignant une efficacité d'élimination de 45 à 88 %, avec une moyenne de 72 ± 7 % après traitement avec le réacteur UASB. Une étude précédente a rapporté une élimination maximale de la DCO de 88,9 % par les réacteurs UASB de Mudor (Ahmed et al., 2018). Le post-traitement avec les lits bactériens et la décantation ultérieure dans les clarificateurs secondaires ont encore amélioré les performances du système.

L'efficacité globale d'élimination de la DCO de la station d'épuration de Mudor a atteint 86,2 ± 2 %. Bien que satisfaisante, certaines études ont rapporté jusqu'à 99 % d'élimination globale de DCO pour les réacteurs UASB suivis de diverses unités de post-traitement. Cependant, il convient de noter qu'il s'agissait d'expériences en laboratoire et à l'échelle pilote (Banihani & Field, 2013 ; Bhatti et al., 2014 ; Gonzalez-Tineo et al., 2020). Des performances similaires ont été observées pour l'élimination de la DBO, avec 86 ± 8 % d'élimination pour les réacteurs UASB de Mudor et 97 ± 1 % après les unités de post-traitement. En ce qui concerne l'élimination des solides, les réacteurs UASB de Mudor ont éliminé respectivement 35,7 % et 16,3 % de TS et TDS. Les unités de post-traitement ont amélioré les efficacités d'élimination globales à 56,7 % et 35,5 %, respectivement. Pendant ce temps, l'élimination du TSS et du TVS était satisfaisante pour les réacteurs UASB à 73,3 % et 74 % respectivement, avec une efficacité d'élimination globale de 93 % pour les deux paramètres. Les résultats pour l'élimination des solides sont comparables à ceux des études similaires dans la littérature avec des valeurs comprises entre 41 % et 77 % pour les réacteurs UASB et des efficacités d'élimination entre 73% et 89 % pour le post-traitement LB (Chernicharo & Nascimento, 2001 ; Pontes et al., 2003).

Performance du système pour l'élimination des nutriments

Les réacteurs UASB de la station de Mudor présentent des résultats modestes dans l'élimination des composés azotés, confirmant les affirmations de la littérature. Cependant, il a été constaté que le traitement aérobie biologique aux LB était également inefficace pour éliminer les composés azotés adéquatement des eaux usées. On a constaté que la concentration d'azote total avait augmenté dans l'effluent de sortie de l'UASB. Cette augmentation dans un environnement réducteur pourrait être attribuée à l'accumulation d'azote total dans les réacteurs anaérobies en raison du temps de séjour pour lesquels les réacteurs UASB sont exploités. Le système a atteint une efficacité globale d'élimination de l'azote total de 27 %. La concentration de NH_3-N a également augmentée dans l'effluent de l'UASB, avec une efficacité d'élimination globale du système de seulement 9 %. Pour le NO_3^--N, une efficacité d'élimination globale négative a été mise en évidence, du fait d'une concentration observée dans l'effluent supérieure à la concentration dans l'influent brut admis. Les variations des concentrations de NH_3-N et de

NO_3^--N dans les réacteurs UASB peuvent être attribués à l'environnement réducteur dans les réacteurs UASB qui favorise la réduction de NO_3^--N et favorise la génération de NH_3-N ; l'inverse se produit cependant après le traitement UASB.

Des observations similaires ont été faites pour les composés du phosphore (P). TP et PO_4^{3-}-P ont également présenté une efficacité d'élimination globalement négative. Le système a présenté des teneurs moyennes résiduelles dans l'effluent traité de 28,37 mg/L et 21,15 mg/L respectivement pour TP et PO_4^{3-}-P, ce qui était supérieur aux valeurs moyennes d'effluent de 25,09 mg/L et 19,5 mg/L respectivement. De plus, ces résultats contrastent avec les découvertes de de Sousa et al. (2001) et Ahmed et al. (2018). Ces auteurs ont rapporté des efficacités d'élimination plus élevées pour le TP à 89 % et le PO_4^{3-}-P à 82 %, respectivement. En règle générale, le déséquilibre C:N:P de l'influent pourrait avoir entraîné des concentrations élevées de composés N et P dans l'effluent final et par la suite avoir influencé la faible performance de la station d'épuration de Mudor en matière d'élimination des nutriments. Les concentrations de N et de P dans les eaux usées étaient significativement élevées par rapport au carbone requis pour un rapport équilibré en éléments nutritifs pour les systèmes anaérobies optimisés (Ammary, 2004 ; Kameswari et al., 2012). Néanmoins, il est évident que les unités de post-traitement de la station d'épuration de Mudor n'ont pas été conçues pour améliorer l'élimination biologique du phosphore, d'où les faibles performances observées.

Résultats sur la performance de la station en matière d'élimination microbienne

L'identication et le dénombrement des germes indicateurs de contaminations fécales dans les eaux usées des influents ont montré des concentrations variant de $1,0.10^2$ à $1,0.10^3$, $1,0.10^1$ à $1,0.10^3$ et $1,0.10^2$ à $1,0.10^3$ (UFC/ml) respectivement pour FC, *E. coli* et Salmonella sp. Le traitement primaire avec les réacteurs UASB a contribué à des efficacités d'élimination satisfaisantes, dans l'ordre respectif : 89,3, 88,5 et 80,0 %. De plus, le post-traitement avec des TF a encore amélioré l'élimination microbienne à environ 1 unité logarithmique (94 - 95 %) pour FC, *E. coli* et Salmonella sp. Les valeurs obtenues dans cette étude concordent avec celles rapportées par Cavalcanti et al. (2001) et Lohani et al. (2020) qui ont signalé des efficacités d'élimination globales supérieures à 90 % pour FC et *E. coli* après le post-traitement de l'effluent de l'UASB avec des bassins de polissage et des filtres à sable, respectivement. Cela confirme l'hypothèse selon laquelle la combinaison de réacteurs UASB avec des unités de post-traitement peut réduire les charges de microorganismes des eaux usées municipales à des niveaux acceptables. À l'exception des éléments nutritifs, la concentration moyenne dans les effluents de tous les paramètres surveillés pour cette étude s'est avérée être dans les limites de rejet autorisées de l'Agence de protection de l'environnement (EPA) du Ghana.

Le taux de production de biogaz à varié entre 101 Nm^3/j et 1673 Nm^3/j, avec une production moyenne journalière de 613 ± 271 Nm^3/j tandis que le débit volumétrique de méthane était compris entre 65 et 1071 Nm^3/j, avec une moyenne de 392 ± 173 Nm^3/j. Le méthane dissous calculé (dCH_4) dans l'effluent de l'UASB s'est avéré être d'environ 23 % du méthane gazeux produit (21 mg/L).

L'empreinte carbone

Les émissions de GES provenant du traitement anaérobie des eaux usées ont été calculées en adoptant la méthodologie affinée du GIEC 2019. La charge organique totale dans les eaux usées (TOW) a été estimée à 2 686 131,73 kgDCO/an, tandis que la masse de DCO convertie en boues (S_{COD}) a été estimée à 1 362,35 kg de boues de DCO/an. Les émissions de méthane provenant du traitement anaérobie au cours de l'année d'inventaire ($E_{CH4\text{-}WWT}$) ont été calculées à 15 034,71 tCO_2 eq/an. Cependant, il serait erroné de rapporter cette émission comme les émissions de CH_4 du système anaérobie lorsque le biogaz de la station est torché.

Le biogaz des réacteurs UASB de la station de Mudor est brûlé ouvertement et selon la méthodologie de la CCNUCC pour les émissions provenant du torchage, une valeur par défaut de 50 % a été attribuée comme efficacité de torchage (ηflare) pour le torchage à ciel ouvert. En prenant la fraction volumétrique de CH_4 à 65 % (Arthur et al., 2022) avec une densité de 0,716 kg/m^3 (GIEC, 2006a), le débit massique de CH_4 a été déterminé à 7,61 kg/h. Les émissions provenant du projet de torchage du biogaz résiduel (PE_{flare}) ont été évaluées à 932,89 tCO_2eq/an.

Le volume total de CH_4 produit au cours de l'année d'inventaire à la température standard et à la pression a été calculé à 102,24 tonnes/an. En utilisant l'équation 4, les émissions de GES provenant des fuites de CH_4 ont été estimées à 143,14 tCO_2eq/an. En comparant ces résultats à ceux d'autres études similaires, Ashrafi et al. (2013) ont rapporté que les émissions de GES provenant des fuites de biogaz pour un réacteur anaérobie traitant les eaux usées étaient de 545 $kgCO_2eq$/j (198,93 tCO_2eq/an). Pendant ce temps, les émissions de méthane des lits de séchage ont été estimées à 305,10 tCO_2 eq/an.

Les émissions de N_2O provenant des processus de nitrification et de dénitrification sur les lits de séchage (LS) ont également été estimées. Selon la conception de la station d'épuration de Mudor, les LS agissent comme une unité de post-traitement pour l'effluent du réacteur UASB, d'où la charge de TN qui arrive aux LB où l'élimination des nutriments biologiques à lieu est la charge de TN présente dans l'effluent du réacteur UASB. Ainsi, le TN dans l'effluent du réacteur UASB a été utilisé pour estimer les émissions de N_2O lors de l'élimination des nutriments biologiques aux LB. La charge moyenne de TN sur les LB a été estimée à 165 444,7 kg/an. Les émissions totales de N_2O provenant de l'élimination des nutriments biologiques sur les LS ont été calculées à 206,02 tCO_2eq/an.

Le carburant total consommé par les générateurs pour faire fonctionner la station pendant les interruptions du réseau électrique était de 9 000,00 litres/an, transmettant 96,77 MWh/an. Les émissions de CO_2 de cette activité de projet ont été estimées à 24,00 tCO_2eq/an. Ainsi, les émissions totales sur site provenant des opérations de la station d'épuration de station de Mudor pendant la période d'étude ont été estimées à 1 608,56 tCO_2eq/an.

L'électricité totale du réseau consommée pour l'année s'est avérée être de 295,892 MWh/an. Les émissions indirectes de CO_2 provenant de la consommation d'électricité du réseau ont été déterminées à 141,7 tCO_2eq/an.

Il a été observé à partir de l'étude que 23 % du CH_4 (21 mg/L) généré restaient dissous dans l'effluent. Les émissions de GES du dCH_4 dans les effluents d'eaux usées ont été calculées à 37 676,67 tCO_2 eq/an. Cette valeur représente environ 95,1 % des émissions totales de GES de la durant cette période d'étude. La forte contribution des émissions de cette source était indéniablement la cause des fortes émissions hors site observées dans cette étude. En comparant ce résultat à des études similaires, Heffernan et al. (2012) ont signalé que 23 % du CH_4 généré restaient en solution à une concentration de 19 mg/L pour une filière de traitement des eaux usées comportant des réacteurs UASB couplés à des boues activées. Les auteurs ont en outre estimé que la perte de ce méthane représentait 78 % (20 000 tCO_2eq/an) des émissions totales de GES de leur système. Robles et al. (2020) ont également observé dans leur étude que le dCH_4 était le principal contributeur des émissions de GES du fonctionnement du bioréacteur à membrane anaérobie (AnMBR), cependant, après récupération par dégazage des membranes, les émissions de GES du système de traitement ont sensiblement diminué. Ces résultats sont comparables à ceux de notre étude.

Les émissions de protoxyde d'azote (N_2O) peuvent se produire sous forme d'émissions indirectes provenant du rejet d'effluents dans les cours d'eau (GIEC, 2019). Les émissions indirectes de N_2O provenant des rejets d'effluents ont été estimées à l'aide de l'équation 9. Avec une charge moyenne estimée de l'Azote Total (TN) de 165 444,7 kg N/an et une fraction d'élimination de TN de 0,3, la charge en TN l'effluent (TNE) a été estimé à 81 051,36 kg N/an. L'effluent GHG_{N2O} a été déterminé à 189,78 tCO_2eq/an.

Les émissions totales hors site provenant des opérations de la station d'épuration de Mudor au cours de la période d'étude ont été estimées à 38 007,82 tCO_2eq/an.

Etude technico-économique de la station d'épuration de Mudor

Evaluation des coûts

L'évaluation des coûts de réalisation de la station de Mudor a révélé qu'elle a été construite par le gouvernement du Ghana en 2000 montant d'investissement initial de 22,14 M USD. L'usine est cependant tombée en panne et n'a pas fonctionné pendant quelques années avant d'être réhabilitée entre 2012 et 2016 (Ahmed et al., 2018). Les travaux de réhabilitation et d'agrandissement ont entraîné un surcoût de 8,65 M USD. Les coûts opérationnels mensuels sont estimés à 49 209,14 USD, la gestion du personnel étant responsable du pourcentage le plus élevé (37 %) des coûts opérationnels. Les réactifs de laboratoire représentent 10,4 %, tandis que les réparations et la maintenance représentent 18,3 % des charges d'exploitation. La consommation d'énergie (électricité et carburant) de la station est très minime et ne représente que 7,3 % des charges d'exploitation. Une faible consommation d'énergie est très typique des systèmes de traitement anaérobie des eaux usées (Lettinga et al., 1980), ce qui donne un avantage sur les systèmes conventionnels à boues activées où les systèmes d'aération pourraient constituer jusqu'à 80 % de la consommation totale d'énergie de ces systèmes (Altin et al., 2020), augmentant ainsi le coût global du traitement. Plus important encore, la station d'épuration de Mudor a été conçue de telle sorte que l'écoulement gravitaire entraîne la plupart des flux de matériaux (Arthur et al., 2022), ce qui explique le coût relativement plus faible concernant la consommation d'énergie de l'usine. Le coût de la gestion des boues a été exclu de l'analyse car actuellement, le traitement des boues se fait par les lits de séchage qui n'entraînent aucun coût, à l'exception du pompage des boues des épaississeurs de boues vers les lits de séchage. Le coût unitaire par m^3 d'eaux usées traitées a varié de 0,34 à 0,45 USD/m^3

pendant toute la période d'étude, tandis que la consommation d'énergie spécifique de la station d'épuration de Mudor a varié de 0,23 kWh/m^3 à 0,31 kWh/m^3.

Evaluation des gains

Comme mentionné précédemment, les avantages ont été évalués en fonction du potentiel de récupération des ressources de la station d'épuration. L'effluent final des clarificateurs secondaires qui sont déversés dans la lagune de Korle s'est avéré avoir des concentrations moyennes d'azote total (TN) et de phosphore total (TP) respectivement de 83,61 ± 24,51 mg/L et 28,37 ± 14,17 mg/L, avec une faible concentration des métaux. Ainsi, avec une concentration moyenne de TN de 0,0836 kg/m^3, si l'on considère 90 % de récupération du volume d'eau usée entrante comme effluent, alors l'effluent riche en N récupéré de 3 746 m^3/j (en postulant que tout l'effluent est utilisé pour l'irrigation), la charge journalière de TN dans les effluents s'élèvera à 313,17 kg/j, aboutissant à une charge annuelle de 114 305 kgN/an. De même, la charge en phosphore totale (TP) de 38 790 kgP/an serait récupérée à partir de l'effluent. La FAO (2015) a déclaré que 10 kg de N peuvent cultiver 1 hectare de terre arable par an. Ainsi, la charge potentielle en TN dans l'effluent permet de cultiver environ 11 430 hectares de terres agricoles. La fertigation avec des effluents riches en nutriments provenant des stations d'épuration peut réduire la dépendance aux engrais inorganiques qui, entre autres, sont relativement coûteux, consomment beaucoup d'énergie dans leur production et ne favorisent pas le développement durable.

La caractérisation des boues d'épuration séchées obtenues à partir des réacteurs UASB de la station de Mudor s'est également avérée contenir 3,33 ± 0,33 % (33,3 kg/m^3) de TN et 2,0 ± 0,46 % (20,0 kg/m^3) de TP. De plus, la concentration moyenne de solides volatils a été estimée à 71,1 kg/m^3, soit 64 % des solides totaux. Le carbone total (TC) est évalué à 29 ± 5,3 %, avec un rapport C/N de 9,0 et une faible concentration en métaux lourds. L'épandage de biosolides sur les terres agricoles peut être une stratégie efficace pour améliorer la productivité agricole en augmentant la fertilité du sol, la matière organique du sol et les éléments nutritifs. De plus, les biosolides peuvent améliorer les propriétés physiques du sol, en particulier dans le cas de sols à texture lourde et mal structurés (Alvarenga et al., 2015 ; Castán et al., 2016). Ainsi, les concentrations élevées de nutriments et de matières organiques observées pour cette étude rendent les biosolides parfaitement adaptés à l'épandage sur les terres pour améliorer la récupération des nutriments.

En utilisant les équations 11 et 12, le biogaz EP et les boues EP s'élevaient respectivement à 1 423,96 MWh/an et 356,31 MWh/an. EP_{Total} a été estimé à 1 780,27 MWh/an. En supposant une efficacité de 30 % de la technologie de conversion de l'électricité, comme indiqué dans la littérature (Lopes et al., 2019 ; Rosa et al., 2018), l'EP_{Total} disponible sera de 534,1 MWh/an, tandis que la demande énergétique réelle de la station de Mudor a été estimée à 392,7 MWh/an. Ainsi, la valorisation énergétique des sous-produits du biogaz et des boues peut complètement compenser la demande énergétique de l'usine et rendre le système positif sur le plan énergétique.

Conclusion

Cette étude de thèse de doctorat a porté sur l'application de la technologie du réacteur UASB couplé à des lits bactériens comme une option de traitement des eaux usées économiquement réalisable et plus durable pour les pays en développement comme moyen de parvenir à une gestion durable des eaux usées. Elle a évalué les dimensions technique, environnementale et économique de la durabilité de la technologie du réacteur de ce système pour le traitement des eaux usées municipales à Accra, la capitale du Ghana.

L'évaluation technique a révélé que les réacteurs UASB fonctionnaient de manière satisfaisante avec une efficacité d'élimination d'environ 70 % pour la DCO et les MES, et une efficacité d'élimination de 86 % pour la DBO_5. Les unités de post-traitement ont encore amélioré les performances avec une élimination globale estimée à 86 %, 97 % et 91 % respectivement, pour la DCO, la DBO_5 et les MES. Les performances du système concernant l'élimination des charges microbiennes ont également révélé des performances satisfaisantes pour les réacteurs UASB avec une élimination de 80 % pour les coliformes fécaux, *E. coli* et Salmonella sp. Le post-traitement avec les LB et la décantation finale ont encore amélioré la réduction de la charge microbienne à une unité logarithmique. Cependant, le système s'est avéré incapable d'éliminer les nutriments (composés azotés et phosphorés) des eaux usées, produisant des effluents très riches en nutriments. S'agissant de la production du Biogaz, les résultats obtenus ont révélé que le débit moyen de biogaz des réacteurs anaérobies était de 613 ± 271 Nm^3/j, avec une production moyenne de 65 % de méthane. Il a cependant été constaté que 23 % du méthane produit restait dissous dans l'effluent, réduisant le potentiel de valorisation énergétique du biogaz.

La durabilité environnementale a été évaluée en mesurant l'empreinte carbone de la station d'épuration à grande échelle de l'UASB/LB. Les sources d'émissions identifiées au cours de l'étude comprenaient les émissions sur site telles que les émissions dues au torchage du biogaz (GHG_{flare}), les émissions dues aux fuites de méthane des réacteurs à travers les conduites de biogaz ($GHG_{CH_4\text{-}leakage}$), les émissions du traitement des boues avec des lits de séchage ($GHG_{sludge\text{-}CH4}$), les émissions provenant de la combustion de carburant diesel pour faire fonctionner les générateurs lors d'une interruption de l'approvisionnement en électricité du réseau électrique national (GHG_{diesel}) et les émissions de N_2O des processus d'élimination biologique de l'azote (nitrification et dénitrification) sur les filtres bactériens ($GHG_{N_2O\text{-}WWT}$). Les émissions hors site considérées au cours de l'étude étaient les émissions de méthane provenant du méthane dissous dans les effluents rejetés dans les masses d'eau réceptrices (GHG_{dCH_4}), les émissions de N_2O provenant du rejet d'effluents riches en azote dans les milieux récepteurs ($GHG_{N_2O\text{-}Effluent}$) et les émissions indirectes provenant de l'utilisation du réseau électrique national. Cette étude a utilisé la méthodologie d'inventaire des gaz à effet de serre du Groupe d'experts intergouvernemental sur l'évolution du climat (GIEC) pour estimer les émissions de GES. Il a été constaté à partir de l'étude que les émissions totales estimées des opérations de la station d'épuration à grande échelle étaient de 39 619,36 tCO_2eq/an. Le dCH_4 a été identifié comme la principale source d'émissions de méthane, représentant 95,1 % des émissions totales.

La durabilité économique utilisant une analyse coût-avantage a révélé que la gestion du personnel présentait l'élément de coût le plus élevé, responsable de 37 % du coût d'exploitation annuel total de la station. La consommation d'énergie ne représentait que 7,3 % du coût d'exploitation annuel total. La récupération intégrée des ressources utilisant les principes de l'économie circulaire a été utilisée pour l'analyse des avantages. Il a été constaté que les eaux usées traitées contiennent effluent de fortes concentrations résiduelles d'azote et de phosphore (0,0836 kg N/m^3 et 0,0284 kg P/m^3), de faibles concentrations de métaux lourds. Les charges microbiennes dans les effluents sont dans les limites fixées par l'OMS à des fins d'irrigation. Cela fait des effluents des eaux usées traitée une source d'eau considérable pour la fertilisation des sols. Il a également été constaté que les boues en excès retirées des réacteurs de l'UASB contenaient de fortes concentrations de matière organique incrustée de nutriments et convenant comme biosolides pour le conditionnement des sols afin d'améliorer la qualité des terres arables. Avec un débit moyen de biogaz de 613 Nm^3/j, avec 65% de CH_4, couplé à une production de boues sèches de 358,24 TS kg/j (130,76 tonnes/an), le potentiel énergétique brut global de la station d'épuration de Mudor sous forme d'électricité à partir de biogaz et boues est de 1 780,3 MWh/an. Avec un taux de 30 % d'efficacité de conversion énergétique, ce potentiel énergétique pourrait donner une production nette d'énergie (électricité) de 534,1 MWh/an, ce qui dépasse la demande énergétique réelle (392,7 MWh/an) de la station d'épuration de Mudor. Ainsi, la station d'épuration de Mudor a le potentiel de fournir son énergie et de se sevrer du réseau électrique national en soutien à ses opérations. Les résultats de cette étude ont révélé que l'utilisation des concepts d'économie circulaire (EC) par le biais de la récupération intégrée des ressources pourraient conduire à une gestion durable des eaux usées. En somme, la technologie du réacteur UASB a été prouvée par cette étude comme étant une technologie efficace, économiquement faisable et durable qui peut et doit être mise en œuvre dans les pays en développement pour parvenir à un développement durable dans ces régions.

Dedication

I graciously dedicate this work to my dearest husband, my king, John Kwesi Nyame ARTHUR. I am most grateful for your unconditional love, unwavering support, counsel, patience and prayers, which I always counted on. Being away from you was always stressful. Thank you for being such a wonderful husband.

To my lovely girls: Josephine Michelle Nhyira ARTHUR and Jochebed Annica Adepa ARTHUR. For my long absence from home, for all the times you wondered and asked why mummy does not return home from school like everyone else. Thank you, my Angels, for your love, understanding and patience.

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To my siblings: Mabel, Bernice, Michael, Maxwell and Peter; To family: Mr Christian MARTEY, Mr Joseph TEI, Mr Ebenezer TEI, Mr Albert TETTEH and Mr Bernard Tei MENSAH; And to friends turned family: Princess Anita Kukua NARTEY (Esquire), Pastor Dr Collins ODURO, Ing. Antoinette Ama AGBOADO, Ing. Ernest OPPONG, Mrs Anita ASANTE, and Mr and Mrs KABUTEY for your constant support, encouragement and prayers during my studies for which my mere expression of thanks likewise does not suffice.

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List of Acronyms and Abbreviations

ASIP	Accra Sewerage Improvement Project
AMA	Accra Metropolitan Assembly
ASP	Activated Sludge Process
AD	Anaerobic Digestion
AnWT	Anaerobic Wastewater Treatment
AnMBRs	Anaerobic Membrane Bioreactors
BF	Biofilter
BOD	Biochemical Oxygen Demand
CF	Carbon Footprint
CO_2	Carbon Dioxide
CAPEX	Capital Expenditure
COD	Chemical Oxygen Demand
CBA	Cost Benefit Analysis
CE	Circular Economy
CH_4	Methane
DO	Dissolved Oxygen
DT	Detention Time
DOC	Degradable Organic Content
EIA	Environmental Impact Assessment
EPA	Environmental Protection Agency
FST	Final Settling Tanks
FC	Fixed Carbon
GWP	Global Warming Potential
GHG	Greenhouse Gas
GCV	Gross Calorific Value
HRT	Hydraulic Retention Times
HFCs	Hydrofluorocarbons
H_2S	Hydrogen Sulphide

IPCC	Intergovernmental Panel for Climate Change
LCA	Life Cycle Assessment
NCV	Net Calorific Value
N_2	Nitrogen
OSSs	On-site Sanitation Systems
OLR	Organic Loading Rate
O_2	Oxygen
OPEX	Operational Expenditure
SMA	Specific Methanogenic Activity
SSA	Sub-Saharan Africa
SDGs	Sustainable Development Goals
SOR	Surface Overflow Rate
SRC	Stockholm Resilience Centre
SSGL	Sewerage Systems Ghana Limited
TF	Trickling Filter
TP	Total Phosphorous
TN	Total Nitrogen
VFAs	Volatile Fatty Acids
WSPs	Waste Stabilization Pond
WWTPs	Wastewater Treatment Plants
WRRF	Wastewater Resource Recovery Facilities
WOR	Weir Overflow Rate
WHO	World Health Organization
UN	United Nations
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change

General Introduction

Background of the Study

The scarcity of freshwater resources and wastewater management are two global issues with growing present-day and future importance. Depleting freshwater resources exacerbated by the burgeoning population and the challenges associated with safe wastewater management presents issues of looming water scarcity, environmental contamination and risks to human health. The eminent scarcity of freshwater resources has led to the employment of alternative and sometimes unconventional water sources to augment freshwater supplies. Unconventional water sources such as wastewater and seawater have become the primary potable water supply in some places due to constraining factors such as geography, climate and increasing freshwater demand (Amy et al., 2017; Quentin Grafton, 2017).

Wastewater treatment plants (WWTPs) are designed to eliminate environmental pollution caused by the discharge of untreated wastewater into surface water bodies, thereby protecting freshwater sources and eliminating associated risks to human health. Additionally, knowledge of the devastating consequences of the discharge of untreated wastewater into the environment has led to the setting of stringent guidelines by governments to control discharged effluent quality. However, producing high-quality effluent, which meets discharge guidelines for most countries, is resource intensive, associated mainly with capital, energy and operational costs (Shannon et al., 2008).

Whereas the developed regions of the world can boast of highly advanced and efficient conventional wastewater treatment systems usually based on activated sludge processes, which make them energy intensive, developing countries are faced with the dire situation of lack of efficient, economically feasible and sustainable wastewater treatment systems (Martinez-Sosa et al., 2012). Wastewater management remains one major challenge faced by most developing countries in Asia and sub-Saharan Africa (SSA) (UN Wastewater Report, 2017). The rapid population growth in these regions worsens the situation. The World Population Review (2021) reported that the African and Asian continents have the most rapid population growth rates at 2.66% and 2.49% per annum. The population growth rate in SSA is equally higher at 2.66% per annum.

Wastewater contains contaminants and pathogens that harm public health and the receiving ecosystems if discharged untreated into the environment. Nonetheless, wastewater is rich in nutrients and organic matter, which can be harnessed into valuable resources. Anaerobic wastewater treatment (AnWT) processes seem promising as an alternative as they are less energy intensive and simple to operate (Van Lier et al., 2008). Moreover, AnWT processes are usually suitable for warm climatic regions. The added advantage of resource recovery from biogas and sludge makes them efficient and economically feasible technology that can be implemented in developing countries for effective and sustainable wastewater management (Van Lier et al., 2008). AnWT technologies usually implemented include the upflow anaerobic sludge blanket (UASB) reactors, anaerobic filters, fluidised bed reactors, rotating biological contactors, expanded granular sludge beds, waste stabilisation ponds (WSPs), etc.

Among the available AnWT technologies, the UASB reactor has become more popular, with several pilots and full-scale Plants installed in countries like Brazil, India, Japan, and Columbia (Lettinga, 2005; Passos et al., 2020). 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 (Chernicharo et al., 2015). First is the UASB reactor systems' ability to handle high and fluctuating organic loadings (Leitão, 2004). Wolmarans & De Villiers (2002) and Musa et al. (2019) reported on full-scale UASB reactors attaining as high as 90% removal efficiency for high-strength influent sewage of about 30,000 mg/L COD load. According to Hulshoff Pol et al. (2004), the development of biological granules in the sludge blanket is the most significant technology feature that enables UASB reactors to handle high organic loads compared to other anaerobic systems. UASB reactors produce less and more stabilised sludge than aerobic systems, and the biogas generated from these reactors contains a significant amount of methane gas that can be harnessed for energy recovery purposes (Foresti et al., 2006).

The recovery of resources from wastewater treatment plants (WWTPs) is one factor that promotes sustainable wastewater management. Regarding the UASB reactor technology, resource recovery in the form of methane-rich biogas for energy production, sludge as biosolids for composting or agricultural purposes and fertigation with nutrient-rich effluent, coupled with low operational costs of these systems have been the core reasons for their wide acceptance, making this technology a sustainable option for the developing world (Chernicharo et al., 2015; Lettinga et al., 1980). Despite the potential for resource recovery, which could promote sustainable wastewater management, the financial and environmental cost of energy and requisite resources to produce high-quality effluent has led to the questioning of the sustainability of WWTPs (Mo & Zhang, 2013). Highly advanced energy-intensive treatment processes usually produce high-quality effluent. Notwithstanding, less energy-intensive and equally efficient treatment technologies are expected to play a pivotal role in WWTPs, especially when the global focus has shifted to sustainable wastewater management for sustainable development.

Sustainability of Wastewater Treatment Systems

Sustainability assessment is an essential tool that directs stakeholders and policymakers towards sustainable decisions (Bond et al., 2012; Pope et al., 2004). The UN World Commission on Environment and Development defines sustainability as *"Development that meets the needs of the present generation, without compromising the ability of the future generation to provide their own needs"* (Thomsen, 2013). Social-cultural, environmental and economic factors are three major dimensions of sustainable development (Muga & Mihelcic, 2008). Other studies have, however, focused on the techno-economic and environmental sustainability of wastewater treatment systems (Sharma et al., 2021). Sustainability is a broad term with multiple dimensions, and self-definition is often much more effective (Balkema et al., 2002). In this regard, sustainability in the context of this study will be limited to the environmental and economic aspects of the UASB reactor technology. This study will additionally consider the technical aspect of the UASB reactor technology. Socio-cultural sustainability has not been considered as it falls outside the scope of this study.

- Technical Sustainability Assessment:

The technical assessment of WWTPs is a primary requirement usually associated with system operation and performance in meeting desired effluent quality. Technical evaluation of wastewater treatment systems identifies their simplicity or complexity in installation, effectiveness in pollution prevention, consistency in removing pollutants, operation and maintenance (Apau, 2017). These assessments identify the optimum technological option most suitable in each context. Other factors, such as system optimisation for resource recovery, are also considered under technical sustainability evaluation. For the UASB reactor, this will focus on resource recovery from reclaimed water, biogas and sludge by-products. Some indicators considered under resource recovery include biogas flow in anaerobic reactors, biomethanization, and sludge production (Akbulut, 2012). Wastewater treatment should be technically sustainable to meet the primary responsibility of WWTPs, with resource recovery being an added advantage. Most studies conducting technical assessments do so in conjunction with economic and environmental aspects, usually referred to as techno-economic-environmental assessment (Svanström et al., 2014).

- Economic Sustainability Assessment:

An economic assessment refers to the process of identifying, calculating and comparing the costs and benefits of an intended project or proposal to assess its advantages, either absolutely or in comparison to other alternatives. Regarding WWTPs, the economic assessment identifies these systems' operational and capital expenditures and the potential benefits and returns (Ozgun et al., 2021). In the light of sustainability, economic sustainability dwells on practices that support long-term economic growth without posing negative impacts on the socio-cultural and environmental aspects of the community. The most commonly employed parameters of economic analysis are based on capital budgeting methodologies. Other analytical tools employed include life cycle costing (LCC), cost-effectiveness analysis (CEA), and cost-benefit analysis (CBA), among others (Gupta, 2020; Pryce et al., 2022).

- Environmental Sustainability Assessment:

An environmental assessment is usually conducted to ascertain a product or system's overall environmental performance. Various environmental assessment methodologies have been applied in different studies to evaluate the environmental impacts of a WWTP. The environmental impact assessment (EIA) generally is conducted before project implementation. EIA is a methodical process that primarily evaluates the environmental consequence of a project activity (Glasson & Therivel, 2019). Every project presents positive and negative impacts directly or indirectly during the various phases; therefore, the EIA assesses whether the project is environmentally sustainable.

Life cycle assessment (LCA) is well-established and standardized. However, it is a sophisticated tool that quantifies and compares the potential impacts associated with the consumption of resources and emissions of pollutants into the environment, occurring alongside the life cycle of products, processes or services (Jensen et al., 1997). Several LCA studies have been conducted on WWTPs to assess wastewater treatment systems' environmental impacts and compare the environmental performances of different treatment technologies and control strategies (Emmersonn et al., 1995; Lundin & Morrison, 2002).

The carbon footprint (CF) is another tool used to evaluate a product or system's environmental sustainability. Unlike the LCA, CF explicitly measures the greenhouse gas (GHG) emissions from a product or system throughout its life cycle. The CF is representative of the Global Warming Potential (GWP) in LCA analysis. The significant GHGs considered in the CF assessment are Carbon dioxide (CO_2), methane (CH_4) and Nitrous oxide (N_2O) (Doorn et al., 2006). For this study, the environmental assessment will be limited to CF analysis.

The Water-Energy-Food Nexus

The world's population is likely to reach over 8 billion by 2030 and humanity will be faced with the challenge of coping with increasing demand for water, energy and food in the face of eminent resource scarcity. The United Nations estimates that humanity will need 30% more water, 45% more energy and 50% more food by 2030; climate change will aggravate this situation even further (UN, 2012). Adequate access to clean water and sanitation; clean, reliable and affordable energy services; and healthy, nutritious food are fundamental human rights and precursors for socio-economic development (Hoff, 2011). Water, energy and food are inextricably connected and referred to as the Water-Energy-Food Nexus (Figure 0.1). As the largest consumer of water, 70% of the global freshwater available is consumed in the agricultural sector with regards to food crop production, fisheries and agri-food supply chain and forestry, making this sector the primary water consumer (FAO, 2011). Large volumes of water are also required in most power-generating processes, including electricity, hydropower and cooling for thermal processes. Howbeit, food production and its supply chain consume 30% of the global energy produced (FAO, 2011; United Nations, 2014), whilst energy is utilized in the extraction, lifting, pumping, transportation and treatment for the supply of safe drinking water.

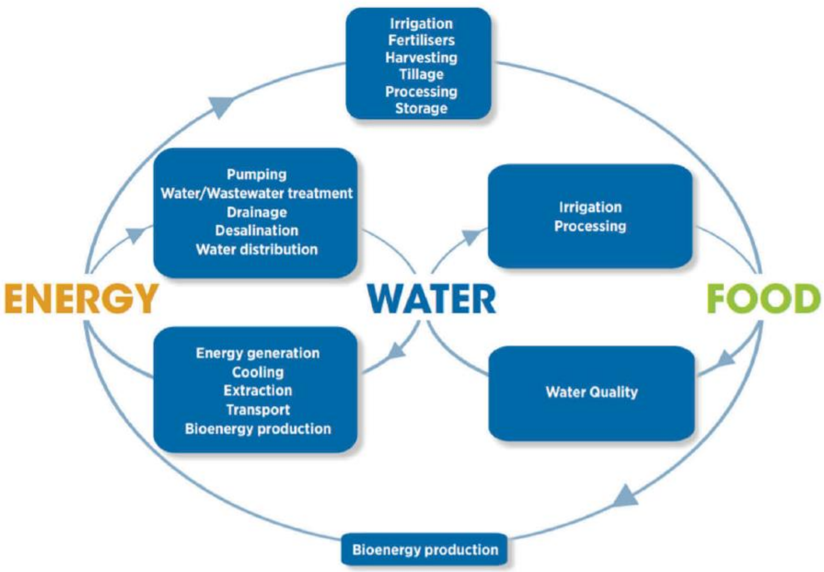


Figure 0.1: The Water-Energy-Food Nexus

(Adapted from Stephan et al., 2018)

The water-energy-food nexus is highly recognised. Continuous population growth will present competition for existing finite natural resources in the coming years. However, these resources have been traditionally independently managed. Water scarcity is currently prevalent in some countries. A United Nations report mentioned that as of 2015, 9% of the global population did not have access to drinking water sources (WHO/UNICEF JMP, 2015). It has again been projected that global water demand may rise to 55% by 2050 in order to meet the growing population demand (United Nations, 2014). Food security relates to water, land and nutrient availability. Fertilizers are needed for efficient food production to meet the rising food demand. Worldwide fertilizer demand is expected to increase between 50 and 100% by 2050 (Chanan et al., 2013). Phosphorous, for instance, is recognised as a geographically sensitive and dwindling natural resource; a secure supply of it will become essential in the near future. Similarly, energy demand is expected to increase by one-third by 2035, with a 70% rise in electricity demand (United Nations, 2014). The water-energy-food nexus is governed by complex interconnections that cannot be accounted for separately. A sudden alteration in one can result in unpredicted and unfavourable outcomes. Moreover, the water-energy-food concept strongly correlates to the realization of global sustainable development (FAO, 2014).

The forecasted depletion of natural resources establishes the need for an integrated approach to improve their security today. Such an approach effectively considers each sector's interdependence, intending to develop measures that align with sustainable development. Many opportunities exist to advance water, energy and food security. This thesis focuses on the development of a highly efficient and economically feasible wastewater technology with excellent prospects for resource recovery, and the potential to promote the continuous supply of water, energy and food for sustainable development.

Problem Statement and Justification of the Study

Despite the growing recognition of the UASB reactor technology in several parts of the world, especially in the tropical regions of Latin America and India (Chernicharo et al., 2015; Lettinga, 2005; Passos et al., 2020), this technology is relatively unknown to most developing countries in the West African sub-region. Only a few studies have reported on the UASB reactor technology in the sub-region (Ahmed et al., 2018; Arthur et al., 2022; Awuah & Abrokwa, 2008). There is a lack of adequate knowledge and sensitization on the numerous advantages that can be explored from this technology; as a result, its development and implementation is minimal despite the economic feasibility. Moreover, not many studies have reported the sustainability of this technology in the context of developing countries. Several factors are considered to make the UASB reactor technology suitable for developing countries in the sub-region. Foremost is the climate condition of these regions. A mesophilic temperature range (20 - 40 ^{o}C) is required for anaerobic systems to operate efficiently. Meanwhile, the daytime ambient temperature in Africa has been recorded to range from 18 - 42 ^{o}C with an average around 35 ^{o}C (Tusting et al., 2020). Thus, in terms of climate, the climatic conditions prevalent in the region are conducive to the optimum operation of UASB reactors.

Regarding operational cost, the UASB reactor is reported to be economically feasible compared to the conventional activated sludge systems, where blowers and aerators are utilized for aeration purposes to optimise aerobic microbes, increasing energy consumption and consequently the operational cost (Lettinga et al., 1980). Literature has reported that aerobic wastewater treatment systems produce high volumes of sludge, the management of which could be responsible for about 40 - 60% of the total operational cost (Domini et al., 2022; Foladori et al., 2015). Unlike aerobic systems, anaerobic reactors produce comparatively lesser sludge volumes, reducing sludge management costs (Chernicharo et al., 2015). Additionally, one renowned advantage of anaerobic systems is the production of methane-rich biogas. Anaerobic UASB reactors treating domestic wastewater could produce biogas with as much as 80% methane output (Noyola et al., 2006). This methane can be recovered to produce energy. This recovered energy can be used to offset the Plant's energy needs. Thus, with the UASB reactor technology, the "*Sanitation Financing Sanitation*" concept can be actualised.

Objective of this Thesis

The main objective of this study is to evaluate the three-dimensional sustainability: Technical, Environmental and Economic sustainability of the UASB reactor technology treating municipal wastewater in Ghana in the West African sub-region.

Specific Objectives:

Specifically, this study will:

- Evaluate the performance of a full-scale UASB reactor coupled with trickling filters (TF) in removing pollutants from municipal wastewater;
- Measure the carbon footprints of the operations of a full-scale UASB reactor coupled with trickling filters, identify the environmental hotspots and propose possible mitigation measures;

- Perform the cost-benefit analysis of a full-scale UASB reactor coupled with trickling filters, and ascertain its implications in developing countries;

Outline of this Thesis

This PhD thesis intends to investigate the subjects mentioned above and present results that will ascertain the sustainability of the UASB reactor technology for implementation in developing countries in sub-Saharan Africa.

This thesis begins with a general introduction, briefly explaining the study background. It also discusses the problem identified, the study justification and the objectives. Four chapters are further discussed. *Chapter 1* extensively reviews different literature on the various subjects discussed in the study. It begins with a brief literature on global and national wastewater management status. Then it discusses conventional and anaerobic wastewater treatment, the anaerobic digestion process and anaerobic treatment technologies. Next, the UASB reactor technology is discussed extensively. It touches on subjects such as the design of UASB reactors, the mechanism of operation, and the advantages and limitations of the UASB reactor technology. The chapter again briefly discusses the trickling filter, its mechanism of operation, major design components and classifications.

Additionally, the chapter discusses the carbon footprints of wastewater treatment systems and the various classifications and emission sources are discussed. Finally, the chapter closes with sustainable wastewater treatment through resource recovery under a circular economy. This section relates sustainable wastewater management to sustainable development to establish how the former can influence the latter's attainment. *Chapter 2* presents the results of the performance of a full-scale UASB and TF combined system treating municipal wastewater. This chapter also quantifies and characterizes biogas production. A COD mass balance is likewise evaluated, and finally, the chapter concludes with the specific methanogenic activity (SMA) test to evaluate sludge activity. *Chapter 3* presents primarily the environmental assessment where the carbon footprints of the operations of the full-scale UASB/TF system are measured. *Chapter 4* deals with the economic aspect of the UASB reactor technology. First, this chapter presents the Plant's cost assessment (CAPEX and OPEX). This chapter concludes by considering the benefit analysis of this technology through resource recovery from reclaimed water, biogas and sludge produced from the system. The final section presents conclusions drawn from the study and gives recommendations for future studies.

Chapter 1:

Literature Review

1.1 Sanitation and Wastewater Management – Global Status

Improved sanitation and wastewater management infrastructure are among the major challenges facing the developing world today. Adequate access to safe sanitation and wastewater management infrastructure is paramount for population health and wellbeing. Sanitation is considered safe when it ensures avoidance of human contact with excreta; it is affordable and convenient for all household members. Safely managed wastewater treatment systems ensure the proper collection, transportation, and treatment of wastewater before discharge into recipient water bodies (UN Wastewater Report, 2017).

The sustainable development goal (SDG) six set by the United Nations iterates the need for sustainable water and sanitation for all by 2030. The specific indicators in Goal Six include the provision of safe drinking water, sanitation and hygiene, wastewater treatment, improvement of water quality, water use efficiency etc. (UN Water, 2018). Undeniably, progress has been made towards the attainment of these individual targets. The WHO/UNICEF JMP (2021) report stated that many countries have rapidly progressed in access to Water, Sanitation and Hygiene (WASH) services. According to the report, between 2016 and 2020, the global population with safely managed drinking water increased from 70% to 74%, safely managed sanitation grew from 47% to 54%, and handwashing facilities with soap and water from 67% to 71%. Global wastewater treatment likewise increased from 20% (estimated 80% discharged untreated) to 56% (estimated 44% discharged untreated) (Jones et al., 2021; UN-Habitat and WHO, 2021). Notwithstanding the significant progress achieved, the report stated emphatically that meeting drinking water, sanitation and hygiene targets by 2030 requires a 4X increase in the pace of progress. The report mentions that at the current rate of progress, 1.6 billion (19%), 2.8 billion (33%) and 1.9 billion (22%) of the global population will still lack access to safely managed drinking water, safely managed sanitation and basic hygiene facilities, respectively by 2030 (United Nations, 2022).

Despite satisfactory improvement in wastewater treatment globally, SSA is still among the regions with the least proportion of wastewater treated. More than half of the domestic wastewater generated in SSA is discharged untreated, having a treatment rate of only 28% (Figure 1.1). Thus, a great deal of work is required if SSA can meet SDG 6.3, which indicates improving water quality by reducing pollution, eliminating dumping and minimizing the release of hazardous chemicals and, lastly, half the proportion of untreated wastewater, increasing recycling and safe reuse globally.

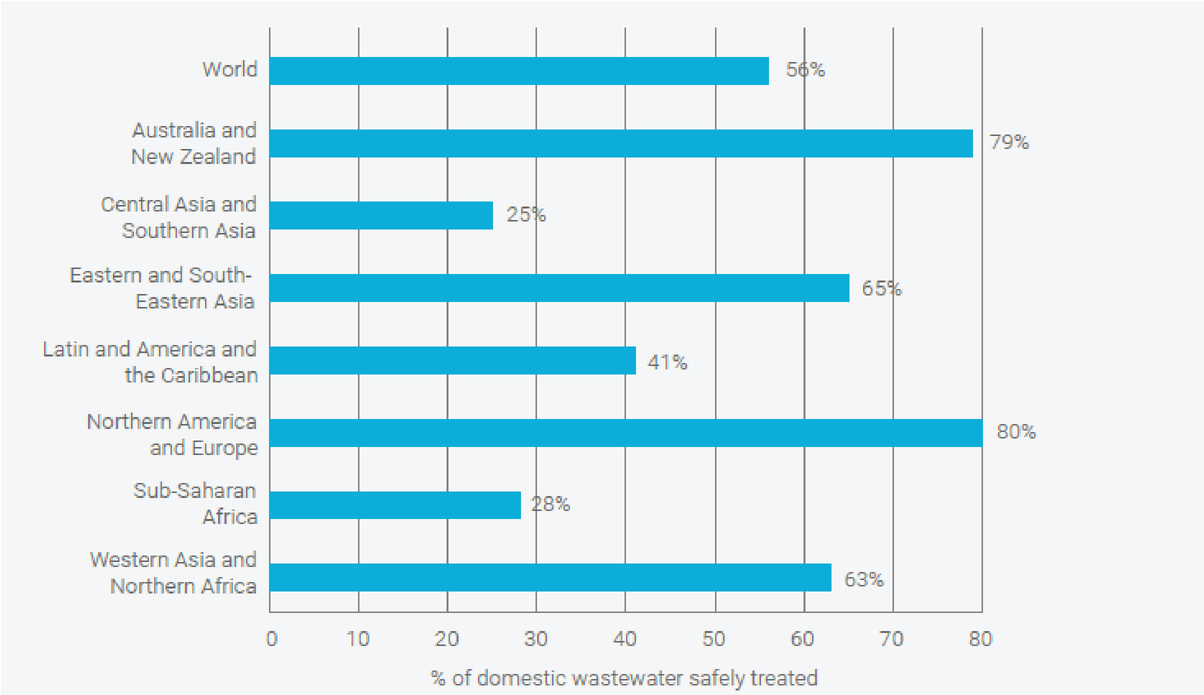


Figure 1.1: Estimated proportion of household wastewater safely treated by region

(Adapted from UN-Habitat and WHO, 2021)

1.2 Wastewater Management in Ghana

With an estimated population of 31.73 million, and a population growth rate of 2.12% per annum (GSS, 2021), Ghana is ranked 47th in terms of population and population growth rate (UN Department of Economic and Social Affairs: Population Division, 2022). Ghana is among the African countries having more than half their population living in urban areas. The acceleration in urban population growth is primarily due to migration, driven by economic, political, social, demographic and environmental factors. The rapid urbanisation brings about challenges such as pressure on social amenities and rising urban unemployment, which has made many African governments and municipal authorities to campaign against rural-urban migration (Teye, 2018). Being the capital city of Ghana, Accra is the epicentre of rural-urban migration by the youth in their quest to search for "greener pastures" and a better life (Turolla & Hoffmann, 2022). Thus, providing adequate sanitation infrastructure for such a rapidly growing population presents a significant developmental issue of concern.

Sanitation management can be by on-site, semi-centralised or centralised systems. For the centralised system, wastewater (composed of greywater, urine and faecal matter) is conveyed via sewer lines from a large catchment area to a wastewater treatment plant. The semi-centralised system generally serves neighbourhoods or a cluster of homes and institutions through shorter sewer lines. On-site sanitation systems (OSSs) are often used for wastewater and human excreta treatment, either fully or partially at the point of generation, whilst some households also dispose of their greywater into storm gutters (MLGRDE, 2008; Singh et al., 2016). The sanitation management option adopted depends chiefly on factors such as population, resources available, socio-economic disposition, legal and institutional conditions and the development planning concept of an area (Orth, 2007). Most middle to low-income countries rely heavily on OSSs (Rose et al., 2015) because they serve as a more economically sustainable option (Dubber & Gill, 2014).

Ghana depends largely on decentralised wastewater and faecal sludge treatment facilities as in other developing countries. A recent survey by the United Nations revealed that only 12.1% of the domestic wastewater generated receives some form of treatment (UN-Habitat and WHO, 2021). Wastewater sewerage coverage in the country is minimal, with only 5% of Ghana's population connected to sewer networks (MLGRDE, 2008), and just about 15% of the total land area of the central Accra business district in the Accra Metropolis is connected to a sewer network. As a result, OSSs such as pit latrines, septic tanks and ventilated improved pits are the most prevalent sanitation systems in Accra (MLGRDE, 2008).

According to Murray & Drechsel (2011), a definite inventory of WWTPs in Ghana is lacking. The authors reported that as of 2011, 62 public and private WWTPs and nine (9) faecal sludge treatment plants constructed over the last 50 years were identified based on unpublished monitoring reports by the Environmental Protection Agency (EPA) of Ghana. The authors, however, mentioned that there could be more facilities, particularly among private and public institutions such as schools and hospitals in the smaller towns that had yet not been identified. In terms of technology evaluation, waste stabilisation ponds (WSPs) constituted the majority (42%) of the implemented technologies in Ghana (Figure 1.2). This is followed by activated sludge systems which make up 26%, and anaerobic digesters make up 16%. The studies found that most WWTPs were in disrepair, under-functioning or incapable of producing effluent quality safe for environmental and public health before discharge. One remarkable conclusion the authors drew from their study was that even if all the WWTPs in Ghana were functioning, they would only serve a small percentage of the population.

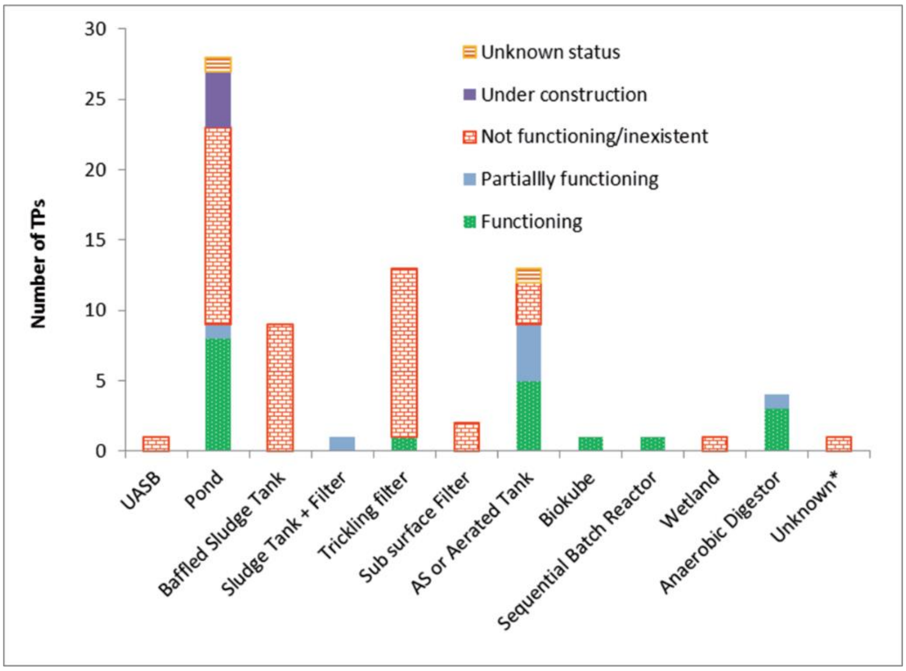


Figure 1.2: Distribution of wastewater treatment technologies in Ghana

(Adapted from Murray & Drechsel, 2011)

Through the Accra Sewerage Improvement Project (ASIP), the government of Ghana secured funding from the African Development (ADB) Bank. The project's objectives were to increase access to sanitation by providing improved and extended sewerage and sanitation systems for disposing of wastewater in Accra in an environmentally and socially acceptable manner (ASIP report, 2018). The project saw the construction of two (2) wastewater treatment plants at Densu Delta and Legon, and eight (8) pumping stations. Additionally, there was the rehabilitation of the Mudor WWTP, the extension of sewer lines and the connection of 4184 households to the Plant. As part of the project, 147 public toilets and 37 septage reception tanks were built (ASIP report, 2018). Despite these laudable initiatives, wastewater management remains an albatross for the country.

1.3 Conventional Wastewater Treatment

Globally, more than a billion people lack safe sanitation and drinking water access. The Human Development Report - United Nations Development Programme (UNDP) noted that about 80% of diseases and 30% of deaths in low-income countries are water-related (de Vries & Lopez, 2013). Agricultural and industrial activities account for a significant portion of water pollution; however, municipal wastewater containing faeces, urine, kitchen and cleaning wastes is the leading cause of water-related human health problems (Zhang, 2016). Municipal wastewater treatment should therefore be a priority to improve human health. Conventional wastewater treatment usually comprises screening and primary sedimentation, which is followed by an aerobic activated sludge process (ASP) to eliminate organic matter, nitrogen and phosphorous-containing compounds, and the effluent discharged into recipient water bodies.

A significant portion of the human population faced with the challenge of inadequate sanitation live in low-income countries where the employment of conventional wastewater treatment processes has been stalled by process dependence on electrical energy that is expensive, unreliable or even unavailable in the developing world. The few developing countries that employ conventional treatment methods may allot over 50% of their municipal budget to energy-intensive processes of wastewater collection and aerobic treatment (ASE, 2002). The high energy consumption of conventional WWTPs is ascribed to aeration activities, with an average energy intensity of 0.6 kWh/m^3 of wastewater (McCarty et al., 2011). WWTPs energy consumption accounts for about 3% of the total national electricity load in the USA (USEPA, 2006). The global concerns with climate change, increasing energy costs, and fossil fuel consumption call for new wastewater treatment technologies which are energy efficient and more sustainable from an energy-saving point of view. Therefore, wastewater treatment innovations must aim to save energy and reduce cost and environmental impact.

More extensive treatment plants are often complemented with sludge digesters to digest sludge anaerobically, and a portion of the energy in the organic waste material is recovered as biogas. Energy recovered in the form of methane from such digesters could offset up to half of the total energy consumed in conventional WWTPs (USEPA, 2006). Additionally, it has been estimated that municipal wastewater with an influent COD concentration range of 400 - 500 mg/L contains a potential chemical energy of 1.5 - 1.9 kWh/m^3 (Owen, 1982). Recovery of potential energy in wastewater for use at the Plant can make the WWTP a net energy producer instead of an energy consumer.

1.4 Anaerobic Wastewater Treatment

The challenges associated with conventional activated sludge systems can be mitigated by employing an alternative technology; anaerobic treatment. The anaerobic digestion (AD) process utilizes anaerobic microbes to break down organic matter without oxygen. Thus, anaerobic wastewater treatment does not require electrical energy for aeration purposes, thereby reducing energy consumption and the consequent operational cost of such systems (Chernicharo et al., 2015). Moreover, the biogas generated in the process is rich in methane, a renewable energy source from which energy can be recovered. Thus, the methane produced from AD processes can be used to offset the Plant's energy needs, giving anaerobic technologies an economic advantage over the aerobic treatment process.

1.4.1 The Anaerobic Digestion Process

The AD process refers to microbial degradation of organic matter in the absence of oxygen. AD is one of the clean energy technologies exploited to augment the energy supply for economic development (Abbasi & Abbasi, 2012). This process loses little energy, with the rest of the energy stored as a gas, mainly chemical bonds of CH_4, CO_2 and other trace gases such as N_2 and H_2S, which are produced along CH_4 in the form of combustible gaseous fuel (known as biogas). Biogas comprises mainly CH_4 (50 - 70%), CO_2 (30 - 45%), and nutrient-rich sludge (Herrmann et al., 2016). Romero (1999) proposed a general equation for anaerobic biological degradation, which is given by the Equation:

$$\text{organic matter} + \text{ nutrients } \xrightarrow{\text{bacteria}} \text{ new cells} + \mathrm{CH_4} + \mathrm{CO_2} \ldots\ldots\ldots\ldots \qquad \textit{(Eqn. 1.1)}$$

During the anaerobic conversion of complex substrates such as polysaccharides, lipids, and proteins, a complex microbial community consisting of many different microbial species is involved. AD comprises four basic stages: hydrolysis, acidogenesis, acetogenesis and methanogenesis (McKeown et al., 2012), schematically presented in Figure 1.3. These stages make up the biogas production process from various organic materials as it occurs in the anaerobic digester. Hydrolysis and methanogenesis are usually considered rate-limiting steps depending on such conditions as substrate type, pH, temperature, and sludge retention time (Zhang, 2016). The rate-limiting step is generally the methanogenic step in easily fermentable substrates (e.g., substrates rich in fatty acids, monomeric sugars, etc.). On the other hand, during AD of complex materials (e.g. agricultural wastes, which are mainly composed of cellulose and small amounts of proteins and lipids), the rate-limiting step of the process is often the hydrolytic step in which polymeric materials split into smaller fragments or their monomers (Soto et al., 1993). The four stages of the AD process are described in the following subsections:

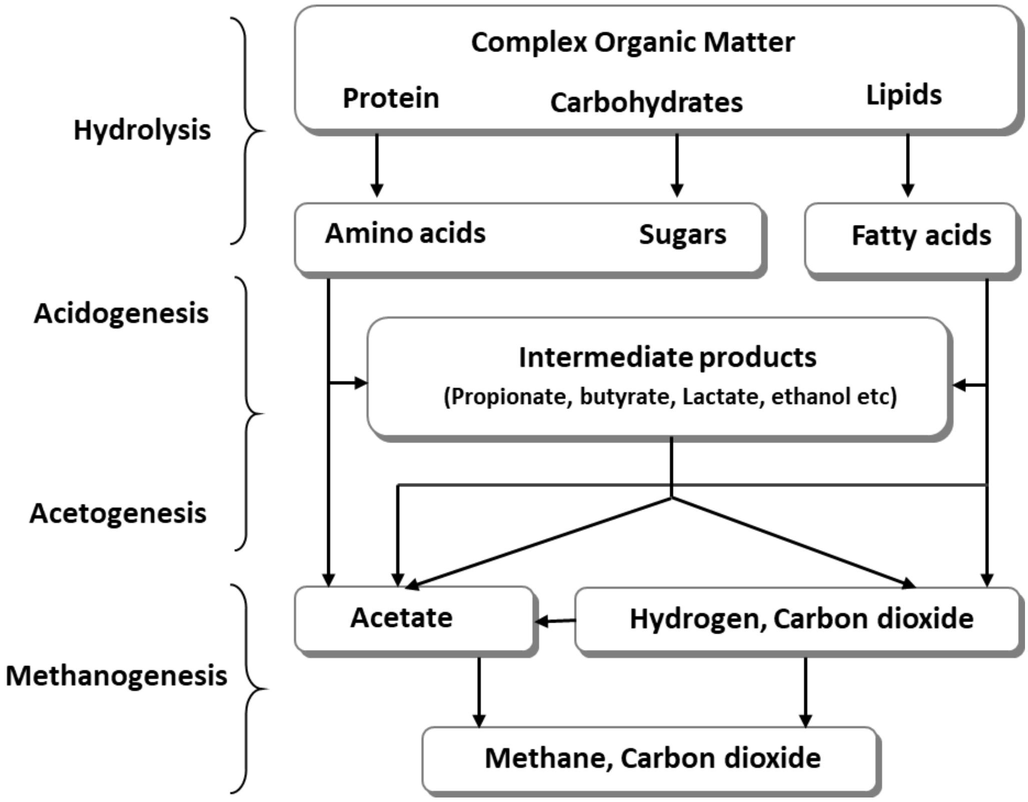


Figure 1.3: Biochemical stages of anaerobic digestion

(Adapted from Jewitt et al., 2009)

- *Hydrolysis*

Hydrolysis is the first step in the AD process. It occurs via the solubilization and degradation of biopolymer particulate organic compounds and colloidal wastes into soluble monomeric or oligomeric organic compounds (Gerardi, 2003), and is considered the rate-limiting step in the AD process (Hendriks & Zeeman, 2009). The process involves the breakdown of complex polymeric compounds such as carbohydrates, lipids and proteins into smaller water-soluble compounds such as sugars, long-chain fatty acids and amino acids. Hydrolytic microbes secrete extracellular enzymes that break down larger molecules into simpler soluble components (Eastman & Ferguson, 1981). Carbohydrates are broken down into simple sugars such as monosaccharides and disaccharides by enzymes Amylase, Xylanase, Cellulase and Cellobiase. Protease breakdown proteins into amino acids, while Lipase degrades lipids into short-chain fatty acids and glycerol (Saha & Cotta, 2007). When a substrate undergoes hydrolysis, it becomes available for cell transportation and fermentative bacteria to degrade during the acidogenesis stage. The hydrolysis rate constant can vary due to various experimental conditions such as the source of inoculum, inoculum to substrate ratio and the available surface of the substrate (Vavilin et al., 2008). A general reaction of hydrolysis is given, as depicted in Equation 1.2.

$$(C_6H_{10}O_5)x + XH_2O \xrightarrow{hydrolysis} XC_6H_{12}O_6 \ldots\ldots\ldots\ldots\ldots \qquad \textit{(Eqn. 1.2)}$$

- *Acidogenesis*

During the acidogenesis stage, the by-products from hydrolysis are broken down further by a variety of obligate and facultative fermentative microbes to produce weak acids (mostly organic acids) such as acetic, butyric acid, lactic acid, propionic acid, alcohols (Kalyuzhnyi et al., 2000). Examples of microbes involved in acid formation include *Clostridium, Lactobacillus*, *Bacteroides*, *Bifidobacterium* and *Streptococcus*. This stage involves the production of high concentrations of hydrogen by acid-producing bacteria known as Acidogens and is usually the fastest step in a balanced anaerobic process. The biodegradation of an organic substrate to produce biogas again depends on complex interactions of various groups of bacteria, the two major groups being acidogens and methanogens. Therefore, maintaining a symbiotic relationship between these two bacteria groups is critical for sustaining the successful operation of an anaerobic digester (White, 2011). The general Equation for acidogenesis is given by:

$$XC_6H_{12}O_6 \xrightarrow{acidification} 3XCH_3CO_2H \ldots\ldots\ldots\ldots\ldots \qquad \textit{(Eqn. 1.3)}$$

- *Acetogenesis*

This is the third stage in the AD process. During the acetogenesis stage, alcohols (ethanol) and volatile fatty acids (VFAs) with more than two carbon atoms are converted by acetate-forming bacteria into acetate, CO_2 and H_2. This conversion is a vital process as H_2 and CO_2 are consistently reduced to acetate by homoacetogenic microbes (Chandra et al., 2012), thereby reducing the hydrogen accumulation that may affect the functioning of acetogens (Weiland, 2010). McCarty & Smith (1986) found that to convert ethanol to methane, the H_2 partial pressure must be between 10^{-1} - 10^{-6} atm, whilst with propionate, a relatively narrow range of H_2 partial pressure (10^{-4} - 10^{-6} atm) is required. This is because acetogens can survive in a low hydrogen concentration environment. However, further increments in the concentration of H_2 partial pressure may result in these bacteria losing their ability to produce acetate. To ensure a low pressure is maintained through this stage of the AD process, a mutually symbiotic relationship must occur between acetogens and hydrogenotrophic methanogens so that acetogens can produce acetate that would be used as substrates by methanogens (Nges, 2012). This step constitutes the final step for fermentation prior to methanogenesis.

- *Methanogenesis*

Methanogenesis is the last stage in AD of organic material. It is the methane-forming step and is dominated by microbes known as methanogens. These methanogens include *Methanobacterium*, *Methanosarcina*, *Methanococcus*, and *Methanosaeta*. The methanogens use acetate, H_2, and CO_2 as substrates to produce mainly CH_4 and CO_2 through two major pathways; (i) acetoclastic methanogenesis and (ii) hydrogenotrophic methanogenesis. About 70% of CH_4 is produced by acetoclastic methanogens (*Methanosarcina* and *Methoanosaeta*), where *Methanosarcina* uses acetate, hydrogen, formate, methylamines, and methanol to form CH_4, and *Methanosaeta* employs only acetate to form CH_4 (Conrad, 1999; Ferry, 2011). Hydrogenotrophic methanogenesis converts H_2 and CO_2 to produce CH_4 and H_2O (Aiyuk et al., 2006). The hydrogenotrophic pathway, therefore, can potentially keep the H_2 pressure low in the digester through its consumption. Table 1.1 presents the major methanogenic reaction pathways indicating some of the microorganisms.

Table 1.1: Reactions related to methanogenesis

Pathway	Reaction	Microorganism
Acetoclastic methanogenesis	$CH_3COOH \rightarrow CH_4 + CO_2$	*Methanosaeta; Methanosacina*
Hydrogenotrophic methanogenesis	$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O$	*Methanobacterium; Methanobrevibacter*

(Adapted and modified from Ampomah-Benefo, 2018)

The maximum biogas yield can be estimated through the degradation efficiency of the biomass. An appropriate equation enables the theoretical estimation of the maximum yield of CH_4 when the elementary composition of biomass is known. Presented in Equation 1.4 is the modified form of Buswell's Equation, which is a stoichiometric equation of biogas production from a biopolymer.

$$C_aH_bO_cN_dS_e + \left(a - \frac{b}{4} - \frac{c}{2} + \frac{3d}{4} + \frac{e}{2}\right) H_2O$$

$$\rightarrow \left(\frac{a}{2} + \frac{b}{8} - \frac{c}{4} - \frac{3d}{8} - \frac{e}{4}\right) CH_4 + \left(\frac{a}{2} - \frac{b}{8} + \frac{c}{4} + \frac{3d}{8} + \frac{e}{4}\right) CO_2 + dNH_3 + eH_2S \ldots \quad \text{(Eqn. 1.4)}$$

1.4.2 Specific Methanogenic Activity (SMA)

As discussed above, the AD process is characterised by a sequence of metabolic processes carried out by specific microbes in four major processes. In these complex interactions amongst the anaerobic microbes, methanogens are usually regarded as the most sensitive to the conditions to which the treatment system is exposed. Thus, monitoring the methane-producing microbes is essential to determine the capability of anaerobic biomass in treating certain types of waste streams; the measurement of the maximum methane-producing rate has been a valuable tool for this. Such analysis is generally performed by the SMA test (Souto et al., 2010). The SMA is a parameter that relates to the potential of sludge in degrading certain substrates, a metabolic assay that assesses the activity or quality of a digester's microbial community. It evaluates the ability of the anaerobic sludge to convert an organic substrate into methane under certain environmental and operational conditions (Angelidaki et al., 2009; Souto et al., 2010).

The performance of the SMA test is known to provide several benefits to the successful operation of an anaerobic digester. The SMA test is used to assess the sludge activity during the various operational phases of the anaerobic system. During the start-up phase, the SMA test allows to determine the maximum organic load that should be applied to the system, and at the time of discharging excess sludge, the SMA test will inform on the minimum amount of sludge that should be maintained in the reactor for optimum performance (de Amorim et al., 2019; Jawed & Tare, 1999). Any change in SMA indicates inhibitions or toxicity in the reactor (Soto et al., 1993). Thus, the SMA test indicates the efficiency of the anaerobic treatment (Dolfing & Bloeman, 1985). The SMA is determined by the methane production rate or the substrate depletion rate and the amount of sludge (Hussain & Dubey, 2017).

1.5 Anaerobic Wastewater Treatment Technologies

Anaerobic wastewater treatment has gained recognition over aerobic technologies since the energy crises era in the 1970s, which was associated with increased demand for industrial wastewater treatment (Henze et al., 2008). Anaerobic wastewater treatment is reportedly effective in eliminating biodegradable organic substances from wastewater. There exist two main classifications for anaerobic wastewater treatment technologies. They can be classified as low-rate or high-rate systems. Figure 1.4 illustrates the classification of anaerobic wastewater treatment technologies.

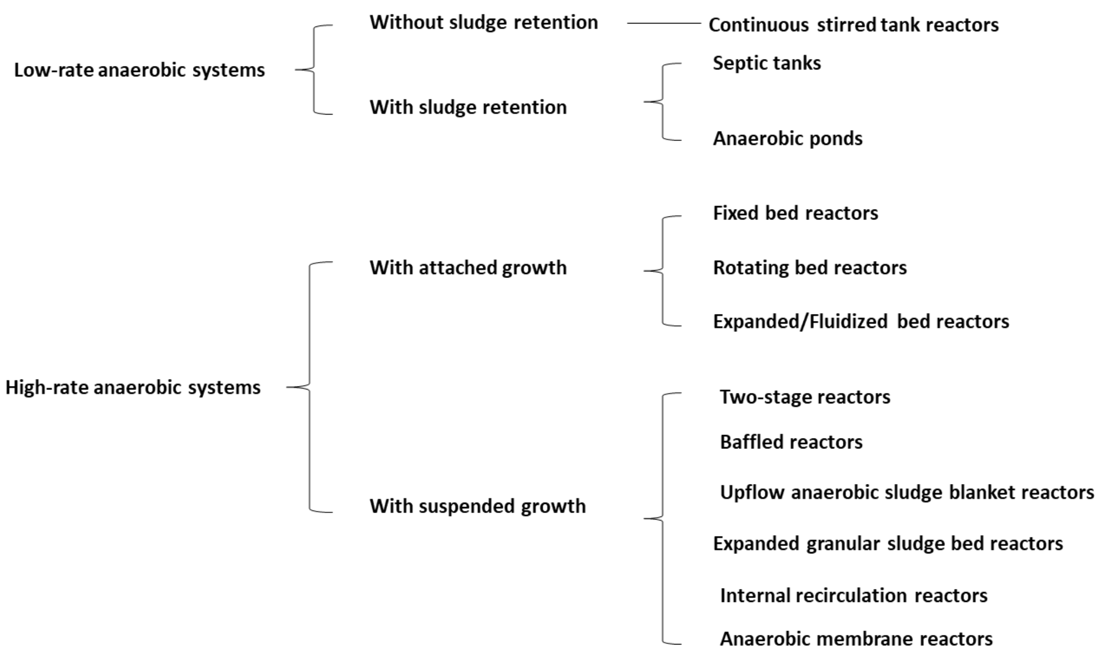


Figure 1.4: Classification of anaerobic wastewater treatment systems
(Adapted from Zhang, 2016)

1.5.1 Low-rate Anaerobic Systems

Low-rate anaerobic systems are generally considered less complicated to build, operate and maintain than high-rate systems. Moreover, they usually operate at longer hydraulic retention times (HRT). Low-rate anaerobic systems can further be classified based on their sludge retention abilities.

- *Systems With Sludge Retention*

The septic tank is classified as a low-rate anaerobic treatment system with sludge retention. The solids are retained in the system through sedimentation; therefore, the sludge retention time (SRT) is much longer than the HRT. These systems are designed to have different units. They also operate without mixing or heating properties (Lowe & Siegrist, 2008). They are one of the most implemented on-site sanitation systems in developing countries, suitable for non-sewered settlements.

Anaerobic ponds likewise fall under low-rate systems with sludge retention. They present a suitable alternative for municipal wastewater treatment in warm-climatic regions, notable for treating wastewater with high organic loads (Mara, 1987). Solids are settled and retained at the bottom of the pond. Anaerobic ponds represent an economically feasible technology option for developing countries. However, they are known to be environmentally unsustainable due to the GHG emissions from these systems (Coggins et al., 2019).

- *Systems Without Sludge Retention*

Anaerobic systems without sludge retention generally operate at low volumetric organic loads with high HRTs. The continuous-flow stirred tank reactors (CSTR) are the most commonly implemented of these systems, employed to stabilise primary and secondary sludge from wastewater treatment plants. They are also employed for industrial wastewater with high suspended solids concentrations, and HRTs are usually kept over 20 days (Hurtado et al., 2015).

1.5.2 High-rate Anaerobic Systems

High-rate anaerobic systems have the ability to support higher hydraulic loadings and, thus, shorter HRTs, and smaller tank volumes with reduced area requirements. They are applicable for both small and large scales wastewater treatment facilities treating municipal, domestic and industrial wastewater. High-rate systems are more complex in construction, operation and maintenance than low-rate systems. High-rate systems can be further classified under attached and suspended growth systems. In attached growth systems, a media grows and maintains microbial populations, creating a biofilm. With the suspended growth systems, the wastewater is mixed with free-floating microorganisms that eventually agglomerate to form biological flocs that settle out of the wastewater (Lettinga et al., 1997). Some high-rate anaerobic systems are further discussed.

- *Systems With Attached Growth*

Anaerobic filters employ the presence of non-moving packing material to which the biomass is attached and kept maintained within the interstices (Young & McCarty, 1969). The average SRT is above 20 days, which permits efficient treatment performance due to the longer SRT. However, these systems tend to be blocked by biomass accumulation or the formation of short hydraulic circuits. With the rotating bed anaerobic reactor, also referred to as anaerobic bio-discs, biomass is attached to submerged discs (Noyola et al., 1988). The applied SRT is longer. Blockages are usually minimized as the rotation of the discs creates shear forces that remove excess biomass between the discs.

An expanded bed anaerobic reactor is made up of a cylindrical structure filled with inert support materials such as gravel, sand etc. These materials account for about 10% of the total reactor volume (Switzenbaum & Jewell, 1980). The bed expansion is typically maintained between 10

- 20% and is reportedly efficient for low-strength pre-treated municipal wastewater at temperatures above 20 ^{o}C and shorter HRT. These systems can attain 60 - 70% organic removal efficiency. Compared to the expanded bed anaerobic reactors, fluidized beds anaerobic reactors have a bed expansion between 30 - 100%. They can also be operated at high OLR (20 - 30 kgCOD/m^3/d) and a higher COD removal efficiency which ranges from 70 - 90% (Şen & Demirer, 2003). However, Van Lier et al. (2015) stated that fluidised bed anaerobic reactors were unsuccessful in practice as biofilms loosen from the support material.

- *Systems With Suspended Growth*

Anaerobic baffled reactor (ABR) systems are designed with vertical baffles, which drive the materials to make sequential upflow and downflow movement, ensuring adequate contact between the biomass and wastewater (Barber & Stuckey, 1999). Applied OLR for ABR systems can be as high as 36 kgCOD/m^3/d. ABRs can be designed to have a smaller depth and built without a gas separator, saving construction costs. Nevertheless, biomass losses may occur during influent flow variations due to the absence of gas separators in ABR systems.

Anaerobic membrane bioreactors (AnMBRs) employ membrane technologies for wastewater treatment. These systems effectively eliminate solids, organics and pathogens from wastewater. However, effluent may contain considerable macronutrients (nitrogen and phosphorous) (Liao et al., 2006). Most studies reported on AnMBRs are executed on bench and laboratory scales, limiting information on operational costs and energy analysis. Other drawbacks include the low membrane flux, membrane fouling coupled with the high capital costs still hinder their applications at the full scale (Chernicharo et al., 2015).

The expanded granular sludge bed (EGSB) reactor is an extension of the UASB reactor technology. These systems employ a higher upflow velocity (Vel_{up}) between 6 - 15 m/h (Lettinga et al., 1997). Adequate mixing between biomass and substrate is enhanced due to the high Vel_{up}. Slowly settling particles in the influent do not accumulate within the reactor and are usually washed out with the effluent. The EGSB reactor is most suitable for low-strength wastewater at low temperatures but not for wastewater with a high fraction of low-density organic particles. The internal circulation (IC) reactor is designed to have two sets of phase separators. One set is located in the middle of the reactor, whilst the other is set at an upper location, as in the UASB reactor. These systems can be operated at relatively higher Vel_{up} between 20 - 30 m/h (Pereboom, 1994; Pereboom & Vereijken, 1994).

The UASB reactor is a high-rate anaerobic system with suspended microbial growth. UASB reactors employ the gas-liquid-solid separator situated at the upper part of the reactor to separate the three materials (biogas, solids and effluent) interacting within the reactor. Biogas production provides a natural mixing phenomenon that permits good contact between biomass and substrate. The UASB reactor can retain a high biomass concentration in the form of granules or well-settled flocculent sludge (Torres & Foresti, 2001). The Vel_{up} of such systems ranges between 0.5 and 1.5 m/h, with an OLR from 2 to 15 kgCOD/m^3/d, depending on the applied temperature and wastewater characteristics.

1.6 The Upflow Anaerobic Sludge Blanket Reactor

1.6.1 Design of UASB Reactors

Lettinga and his co-workers designed the UASB reactor during the mid-1980s (Lettinga et al., 1980). This reactor was initially designed to treat concentrated industrial wastewater, but its application has now been extended to treat different wastewater streams. The UASB reactor is a high-rate anaerobic system with components that are not usually movable, mainly cylindrically shaped, but there are a few with tubular designs. The critical elements of the UASB reactor design are the influent distribution system, the gas-liquid-solid (GLS) separator, also known as the three-phase separator, the gas collection dome, the effluent collection system and the sludge withdrawal system (Chong et al., 2012). The major components of a UASB reactor are presented in a schematic diagram in Figure 1.5.

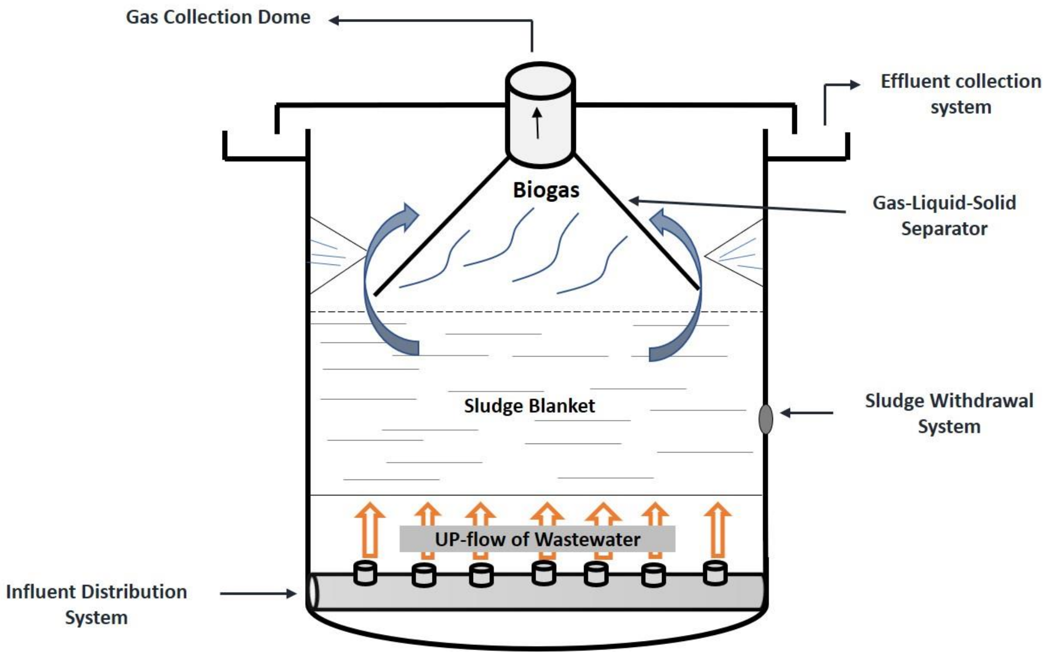


Figure 1.5: Cross-sectional view of the UASB reactor

(Adapted and modified from Arceivala & Asolekar, 2006)

The three-phase separator is the most distinctive device, situated in the upper section and divides the reactor into a lower digestion zone and an upper settling zone. This device facilitates the retention of granules to prevent the loss of solids in the effluent, thereby separating the effluent, gas and sludge solids (Lettinga & Hulshoff Pol, 1991). The digestion zone comprises a sludge bed made of a layer of densely aggregated biomass that forms at the reactor bottom. On top of this is formed a blanket made of finely suspended flocs with lower settling velocities (Chong et al., 2012), where most biochemical reactions occur (Liu et al., 2002). This suspension is held in place due to the agitation of biogas produced from the biodegradation of soluble organic compounds. Above the digestion zone is the settling zone, situated in the upper part of the reactor. This is the quiescent zone of clarified effluent, where sludge granules dislodged from the bed by the upflow movement of wastewater or biogas production with high settling velocities will settle back to the bed (Liu et al., 2002; Chong et al., 2012). Tchobanoglous et al. (2003) reported that the range of solids concentration could be from 50 - 100 g/L at the bottom section and from 5 to 40 g/L at a more diffused zone in the sludge blanket.

1.6.2 Granulation, Start-up and Process Operation of UASB Reactors

Lettinga and his team reported that the UASB reactor operates as a suspended growth system where microorganisms get attached to each other or to small particles of suspended matter to form agglomerates of highly settleable granules that form an active sludge blanket at the bottom of the reactor (Quaff et al., 2014). The same authors observed that the syntrophic relationship between various microorganisms leads to self-agglomeration within the biomass. Moreover, Liu et al. (2003) asserted that the mechanisms for the development of the biological granules are complex; they are not fully known; however, researchers suggest that the bacteria attach themselves to dense sludge particles, and the growth of the bacteria concentrates around these and form the granules. Other authors have reported that under certain conditions within an anaerobic medium and upflow hydraulics, a natural aggregation of bacteria into flocs and granules can occur, with the granular sludge particles ranging from 1.0 - 3.0 mm in diameter (Gonçalves et al., 2002; Vlyssides et al., 2008). The successful operation of UASB reactors greatly depends on the formation and maintenance of sludge granules which make up the sludge blanket. Kalyuzhnyi et al. (2006) opined that the higher settling velocities (20 - 80 m/hr) of the aggregates compared to the Vel_{up} (0.1 - 1 m/hr) permit accumulation of large biomass at the bottom. Similarly, it has been reported that the granules must have good settling abilities and be well aggregated to remain in the bed despite the Vel_{up} (Liu et al., 2003; Awuah & Abrokwa, 2008).

UASB start-up is a process that is delicate and time-consuming. It commences with the initial feeding of the reactor to the point when preferable granular sludge is attained. Hulshoff Pol et al. (2004) and Abbasi & Abbasi (2012) asserted that the successful operation of the UASB reactor is principally attributed to the formation of the anaerobic granules in the sludge bed. This critical stage is accountable for the overall stability and efficiency of the reactor, and it is influenced by various physicochemical and biological parameters as well as wastewater composition, presence and the growth of active microbial populations present in the inoculum and the operating conditions (Lew et al., 2003; Chong et al., 2012). Previous experiments have shown that the start-up process is the main drawback of using the UASB reactor. The start-up period can vary between two to several months, a major challenge in its application (Sato et al., 2006; Vlyssides et al., 2008). Inoculating with seed sludge reduces the period of acclimatization. Even though the UASB can perform efficiently without the seed sludge, applying seed sludge reduces the start-up time required, hence an added advantage (Hulshoff Pol et al., 2004).

During UASB reactor operations, influent wastewater to be treated is fed into the reactor through the influent distribution system located at the bottom of the reactor in an upflow mode (Arceivala & Asolekar, 2006). Van Lier et al. (2008) observed that feeding the reactor uniformly with influent wastewater over the bottom of the reactor ensured the maximum required contact between the wastewater and the sludge granules. The wastewater flows upward through the sludge blanket composed of biologically formed granules or particles. The biogas generated causes internal circulation and agitation, which keeps the sludge blanket thoroughly mixed, enhancing maximum contact between the wastewater and biomass. The internal circulation also helps form and maintain the biological granules. Some of the gases produced within the sludge blanket attach to the biological granules (Lettinga, 2005). Free gas and sludge particles (with the attached gas) rise to the top of the reactor. Particles that rise to the surface strike the deflectors, which release attached gas bubbles. The granules fall back to the surface of the sludge blanket. The free gases released from the granules are captured in the gas collection domes sited at the top of the reactor. Finally, the treated effluent is collected in the effluent collection system (Chong et al., 2012).

1.6.3 Advantages of UASB Reactors over other Anaerobic Technologies

The UASB reactor technology has not been in existence for a longer period compared to other anaerobic wastewater treatment technologies but has gained recognition these past few years. Several full-scale plants have been put in operation, and many more are presently under construction, especially in tropical and subtropical regions (Chernicharo et al., 2015). Some studies have also been conducted in regions with a moderate climate. In the past, wastewater from industrial sources was not treated by anaerobic technologies as they could not handle high organic loading rates (OLRs), with instability in treatment efficiency. The UASB reactor resolves these issues as it can handle high organic and hydraulic loading rates (Leitão, 2004). The addition of inoculum speeds up the biological population development, thus reducing start-up times (Van Lier, 2008). The development of dense biological granular sludge, an essential feature of the UASB process, allows for employment with high volumetric COD loadings compared to other anaerobic processes (Hulshoff Pol et al., 2004).

The UASB reactor has a simple construction, with low operation and maintenance costs, as construction materials and other required parts are locally available. Energy consumption is low as there is no need for aerators and blowers, thereby reducing operational costs (Chong et al., 2012). It also exhibits the attributes of anaerobic wastewater treatment processes, such as biogas production, with about 60 - 70% methane gas content which can be employed for energy recovery purposes (Bressani-Ribeiro et al., 2019). The UASB process generates less sludge as by-products compared to aerobic processes (Khan et al., 2011). The UASB reactor also provides high organic matter removal efficiencies, thus requiring smaller land area and reactor sizes, is very robust and can be applied to numerous wastewater streams (Leitão, 2004; Chong et al., 2012; Rizvi et al., 2015). The UASB has a short HRT (≈ 4 - 24 h) and a long SRT compared to other processes; hence can process large amounts of wastewater in a short time (Chong et al., 2012). Thus, the invention of the UASB reactor technology has permitted the expansion of sewage treatment infrastructure to an immense populace, especially for regions with inadequate financial resources, land and skilled workers (Bressani-Ribeiro et al., 2019). Several full-scale plants have been installed and operated in different regions of the world, including Asia, Latin America, and India, for the treatment of both domestic and industrial wastewater streams (Von Sperling & Chernicharo, 2002; Chernicharo et al., 2015).

1.6.4 Factors Influencing the Efficiency of UASB Reactors

The satisfactory performance of the UASB reactor is dependent on several factors. These factors influence the pollutant removal efficiencies as well as the overall performance of UASB reactors, the majority of which are peculiar to anaerobic wastewater treatment technologies. Major factors of interest are the operating conditions applied to the system and the characteristics of the substrates fed to the reactor. Prominent among the operating conditions are temperature, potential hydrogen (pH), OLR, HRT and Vel_{up}, granulation and mixing within the reactor.

1.6.4.1 Effects of UASB Reactor Operating Conditions

- *Effects of Temperature*

Temperature plays a significant role in UASB reactor performance and anaerobic wastewater treatment as a whole, as it influences the growth and survival of microorganisms (Ali & Okabe, 2015; Divya et al., 2015). Biodegradation of organic matter is enhanced within the mesophilic temperature range as microorganisms responsible for AD are mesophilic (thrive in temperatures between 20 and 45 ^{o}C) (Bodík et al., 2000). Several authors have iterated the fact that the UASB reactor is very efficient in tropical and subtropical regions, this makes the technology economically feasible for implementation in developing countries due to the favourable climatic conditions of these regions for AD processes, with no external heat application required, and the consequent reduced operational costs (Mahmoud, 2002).

According to Lettinga et al. (2001), Foresti (2002) and Kaviyarasan (2014), psychrophilic (< 20 ^{o}C) temperature range leads to reduced enzyme activity, reduction or alteration in microbial activity, slower biodegradation and lower VFA production rate, resulting in less production of methane. The authors again asserted that although generally, thermophilic (> 45 ^{o}C) temperature facilitates the AD process due to an increase in microbial activity, faster hydrolysis and enhanced substrates utilization, at these high temperatures, there is a risk of high ammonia concentrations build-up in the reactors, leading to ammonia toxicity. This can result in reduced methane yields.

- *Effects of Organic Loading Rates (OLRs)*

The volumetric and mass loadings applied to UASB reactors can influence the performance of these reactors in terms of pollutant removal and biogas production. OLRs can significantly affect microbial ecology and the characteristics of anaerobic systems (Mahmoud, 2002). Some authors have reported that the ideal organic loading that applies to a particular UASB reactor can range from 2 - 15 kgCOD/m^3/d (Sayed, 1987; Mahmoud, 2002; Leitão, 2004). The OLR applied to a reactor depends on the wastewater characteristics and strength, the HRT and the reactor volume (Mahmoud, 2002; Halalsheh et al., 2005). Cavalcanti (2003) believed that higher than optimum OLR results in biogas accumulation in the sludge, forming pockets of gases and leading to sludge flotation. Klesyk (2017) likewise asserted that too lower OLR meant the provision of little and inadequate substrates for the microorganisms.

- *Effects of Hydraulic Retention Time (HRT) and Upflow Velocity (Vel_{up})*

The HRT is one of the important parameters considered in the performance of UASB reactors and is directly related to Vel_{up}. An increment in the applied HRT necessitates a reduction in the applied Vel_{up}; thus, one parameter is a function of the other (Rajakumar et al., 2011). The optimum allowable ranges for HRT and Vel_{up} have been reported to be within 4 - 24 h and 0.5 - 1.5 m/h, respectively, mainly dependent on the reactor design and operation (Rajakumar et al., 2011). The optimum application of these two parameters ensures adequate mixing of substrates within the reactor, allows for maximum contact time between the biomass and the substrates, reduces incidents of sludge washouts, and finally increases the pollutant removal efficiency of the reactor (Foresti, 2002; Mahmoud, 2002; Rajakumar et al., 2011).

- *Effects of Mixing/Agitation*

Adequate mixing of substrates within an anaerobic reactor is obligatory as it allows effective attachment and contact between the anaerobic microbes and the organic-rich substrates for effective biodegradation and pollutant removal, enhancing the overall performance of the reactor (Habeeb et al., 2011). Good mixing within UASB reactors allows the provision of uniform environmental conditions, whilst inadequate mixing leads to the formation of pockets of substrates at the various digestion stages, which results in pH and temperature variations within the reactor (Daud et al., 2018). Adequate mixing enhances mass transfer and activates dead zones within the UASB reactor. Dead zones reduce the effective volume of the reactor (Habeeb et al., 2011). Internally generated biogas causes recirculation and agitation within the reactor, which provides adequate contact time between biomass and substrates. However, other researchers believe that mechanical mixing and slurry recirculation provides better impacts as they ensure maximum contact time compared to biogas recirculation. Nonetheless, vigorous and rapid mixing is not recommended as it adversely affects microbial activities (Karim et al., 2005). Peña et al. (2006) and Quaff & Guha (2011) have also reported that an increase in Vel_{up} and flow rate could provide adequate mixing within UASB reactors. Ward et al. (2008) asserted that the degree of mixing in a reactor could be measured by performing a hydrodynamic test using a tracer element.

- *Effects of Granulation*

The successful operation of a UASB reactor is greatly dependent on the formation of the sludge granules, which make up the sludge blanket within the system (Quaff et al., 2014). The formation of quality sludge granules with desirable settling properties is influenced by factors such as the reactor's environmental conditions, which employ the temperature, pH and substrate characteristics (Lew et al., 2003; Singh & Viraraghavan, 2003). Process operating conditions, including HRT, OLR, Vel_{up} and characteristics of the seed sludge, also affect the quality of the sludge granules (Tiwari et al., 2005), and finally, chemical conditions which entail the presence of cations and polymers (Hulshoff Pol et al., 2004). Researchers have found that the formation of quality granules shortens the start-up period required for acclimatization in the event of the absence of an inoculum, thereby facilitating the biological degradation of organic matter, enhancing the overall performance of the UASB reactor (Liu et al., 2003; Aiyuk & Verstraete, 2004). In a study conducted by Rico et al. (2017) to assess the most suitable anaerobic seed sludge for the digestion of pig slurry in UASB reactors, the authors observed that the application of granular sludge provided an optimum performance which allowed a high volumetric methane production rate in comparison to the anaerobic sewage sludge which was prone to biomass wash-out. The thickened digestate-sludge, however, resulted in system failure due to sludge flotation problems.

- *Effects of pH/Buffering Capacity/VFAs/Alkalinity*

UASB reactor performance and stability are significantly influenced by pH within the system as methanogenic bacteria are pH-sensitive (Leitão, 2004). The optimum pH range for anaerobic degradation of microbes is between 6.3 and 7.8, and any pH value outside this range adversely affects the performance of UASB systems (Cavalcanti, 2003; Mao et al., 2015). Chen et al. (2017) reported a study conducted in China wherein COD removal efficiency reduced from 87.8 - 90.3% to 66.7 - 70.5% when the pH was altered from 7.3 to 9. In a similar research by Zhang et al. (2019), in a bid to evaluate the influence of pH stress on the functional bacterial and archaeal dynamics in a UASB reactor treating sugar refinery wastewater, the authors observed a significant reduction in COD removal and methane yield when pH was reduced to 5. According to the authors, the drop in pH obstructed the metabolic balance and structural community among the different trophic groups, resulting in reduced reactor performance.

One indicator for measuring AD process imbalance quicker than pH in a system is the buffering capacity (BC). This is because the BC will promptly reduce with an accumulation of fatty acids before a pH drop is even observed (Mussoline et al., 2012). The BC measures the ability of a solution to resist changes in pH when an acid or base is added to it. AD stability is dependent on the BC of the digester contents. The capacity of a solution to neutralize acids is also known as alkalinity. Alkalinity is usually measured by the concentration of bicarbonates, carbonates and hydroxide ions in a solution (Schnaars, 2012). Ward et al. (2008) asserted that an increase in BC is accomplished by mainly decreasing the influent loading rate, although the direct addition of bicarbonate is more exact as CO_2 addition would necessitate a lag phase for gas equilibrium to occur. Chernicharo (2007) likewise opined that alkalinity plays a crucial role in anaerobic digestion as it helps maintain BC and inhibits the accumulation of formed volatile fatty acids (VFAs) in the anaerobic process. Higher alkalinity values indicate a greater capacity of the system to resist pH changes.

Buffering agents capable of neutralizing pH in anaerobic reactors include soda ash, as reported by Parawira et al. (2005); lime, as reported by Satyanarayan et al. (2009); $Ca(OH)_2$ solution, as reported by Cruz-Salomón et al. (2016); $NaHCO_3$ solution, as reported by Wu et al. (2019) and Chen et al. (2020); HCl and NaOH solutions, also reported by Diamantis & Aivasidis (2007) and H_2SO_4, as reported by Verma et al. (2015). Food to microorganism (F/M) ratio can likewise be employed to maintain a high BC and constant pH within an anaerobic system. Cavalcanti (2003) and Rizvi et al. (2015) mentioned that UASB reactors employed in the treatment of sewage from tropical and subtropical countries typically have stable pH and BCs.

VFAs such as propionic, acetic and butyric acids are the most predominant indispensable intermediary products from acidogenesis and acetogenesis steps, formed during the biodegradation of organic materials in the AD process; they are also vital buffering agents and play a key role in the overall AD process (Lukitawesa et al., 2020). Their applications include biodiesel production, synthesis of complex polymers and electricity generation through microbial fuel cells (Chen et al., 2013). Additionally, they can be employed as valuable carbon sources in biological processes (Elefsiniotis et al., 2004). VFAs will likely accumulate in anaerobic systems when OLRs are relatively high or during perturbations when methanogenic bacteria cannot fully utilize VFAs and hydrogen as acidogens and acetogens produce them. Nevertheless, their accumulation in anaerobic bio-digesters (reactor acidification) indicates an imbalance between sequential AD process steps (Madsen et al., 2011). Accumulation of VFAs results in a reduction in BC and a fall in pH to levels that impede hydrolysis and acidogenesis phases, leading to eventual process failure (Yuan & Zhu, 2016). Accumulating VFAs is an easier way of detecting biochemical dysfunction of the AD process, as their accumulation causes discrepancies in various microbial metabolisms, disrupting all stages of biochemical degradation (Nielsen et al., 2007; Rathaur et al., 2017).

VFA/Alkalinity ratio is a variable that can measure system performance and control the AD stability process (Callaghan et al., 2002; Kuglarz et al., 2011). Whilst the VFAs provide information on the performance of AD intermediate steps, alkalinity describes the capability of the feed to neutralise the VFA generated during the process, controlling pH changes. Literature has reported an ideal range for stable digestion between 0.1 and 0.4. Values between 0.4 and 0.8 indicate a level of instability in the system, whilst values over 0.8 indicate gross instability, which could be due to increased organic or hydraulic loadings to the system (Hamawand & Baillie, 2015; Bakraoui et al., 2020).

1.6.4.2 Effects of Substrate Characteristics

Characteristics of substrates fed into UASB reactor systems highly influence the efficiency of these reactors. Recently, attention has been drawn to substrates rich in toxic sulphur compounds, high salinity and lipids concentrations and nutrient-rich substrates; these are conferred in this section.

The presence of toxic and inhibitory compounds in anaerobic reactors either pre-exists in the substrate; otherwise, they are formed during substrate degradation. The commonest inhibitors, such as long-chain fatty acids (LCFA), VFAs, sulphide and ammonia, are formed during substrate degradation. Other inhibitors, such as antibiotics and heavy metals, are usually already present in the substrate (Boe, 2006). According to reports presented by Boe (2006), undissociated forms of H_2S can quickly diffuse through the microbial cell membrane, resulting in the denaturation of protein and interfering with bacteria assimilatory metabolism. Higher concentrations of undissociated H_2S are fatal to sulphate-reducing and methanogenic bacteria (Hu et al., 2015). Lu et al. (2016) likewise studied the COD/SO_4^{2-} ratio effect in a UASB reactor treating starch wastewater. The reactor was observed to have exhibited satisfactory performance at $COD/SO_4^{2-} \geq 2$, resulting in excellent COD and sulphate removal rates and stable production of biogas. However, a progressive decrease in this ratio resulted in satisfactory performance of the system until at $COD/SO_4^{2-} < 2$, after which further decrement caused suppression of methanogens through inhibition of sulphide and electron competition. Khan et al. (2013) also reported that highly mineralized sulphur compounds remain as sulphides in effluent from anaerobic systems, which depends on the influent sulphate concentrations and the presence of sulphate-reducing bacteria in the reactor.

When dealing with brackish wastewater for UASB reactor treatment, the salinity issue draws attention. Liang et al. (2019) demonstrated that even under salinity levels of 10 gNaCl/L, satisfactory pollutant removal could be attained; however, it was observed that lower salt concentrations could facilitate the formation of larger granules and enhance the rate of degradation. The authors also asserted that microbial community characteristics are not considerably modified by salinity, even when methanogenesis is reduced. Similar studies by Wang et al. (2017) on a UASB reactor treating phenol highlighted that granular biomass could withstand moderate saline levels not higher than 10 gNa^+/L as levels higher than this value reduces the reactor efficiency.

Treatment of wastewater streams rich in lipids, oil and grease (O&G) concentrations needs to be given much attention due to drawbacks such as clogging, the formation of foams and odour emissions, sludge flotation and biomass washout (Miranda et al., 2005). Nakasaki et al. (2020) reported higher methane potential was observed with lipid-rich wastewater streams compared to proteins and carbohydrates. Slaughterhouse wastewater falls under this category of wastewater stream beside the presence of suspended and colloidal particles, proteins and cellulose, which makes it difficult for UASB reactors to operate efficiently at high OLRs of this wastewater stream unless a pre-treatment or post-treatment system is applied. In the study by Mannacharaju et al. (2020), the treatment of fish processing wastewater pre-treated in a moving baffled bed biofilm reactor before UASB reactor treatment showed a satisfactory reduction of COD, lipid, O&G, and proteins.

High nitrogen concentrations in wastewater streams could result in extreme accumulation of ammonia in UASB reactors; these high concentrations are known to be toxic, slowing the hydrolysis phase and biogas production rate (Geißler et al., 2019). Nevertheless, optimum nutrient levels are imperative as they serve as cellular building blocks for microbes; they also facilitate metabolic activities as they ensure cells can synthesise enzymes and co-factors responsible for that.

Kock (2015) found that macro and micronutrients, trace elements and vitamins are required in very low concentrations as their complete absence can result in detrimental effects on the growth of microorganisms. According to Choong et al. (2016), the presence of trace elements such as zinc (Zn), cobalt (Co), iron (Fe), and nickel (Ni) enhance multi-stage AD processes. The adequate concentration of trace elements supports the metabolism of anaerobic microbes. Zhang et al. (2011) opined that less than adequate concentrations of trace elements led to the inhibition of methanogenic bacteria in the system. Similarly, Ariunbaatar et al. (2016) and Wintsche et al. (2016) observed that the supply of optimum trace elements results in efficient biodegradation, higher digester stability and low fatty acids accumulation, leading to enhanced biogas production.

Martin-Ryals (2012) likewise asserted that an unbalanced carbon-to-nitrogen (C/N) ratio is one limiting factor in anaerobic digesters. The configuration of carbon to nitrogen in a biodegradable substrate is designated C:N ratio, which must be maintained in the optimum range to conserve an appropriate nutrient balance for essential microbial growth, maintaining a stable environment for efficient AD (Mata-Alvarez et al., 2014; Kainthola et al., 2019). Wang et al. (2014) and Darwin et al. (2014) reported optimum C:N values within the range of 20 - 30. Substrates with very high C:N ratios, such as the majority of crop residues and paper, will be deficient in Nitrogen required to maintain desired microbial flux in the reactor. This impedes system performance resulting in a decrease in biogas generation, whilst substrates with a meagre C:N ratio, such as animal manure, contain a relatively high concentration of nitrogenous organic matter. Ammonia formed by the biodegradation of the nitrogen-rich organic content is usually in excess for the microbe's utilisation; this may build up toxic ammonia compounds causing inhibition in the system. The excess ammonia typically accumulates and raises pH; this also is noxious to methanogenic bacteria and impedes process efficiency and biogas production (Ariunbaatar et al., 2016). In a study by Shi et al. (2016), the authors opined that high concentrations of free ammonia resulting from the substrate's low C:N ratio inhibited methanogenic activities, resulting in the accumulation of VFAs and reduction in methane yield. Co-digestion processes have proved to be effective in optimizing C:N ratios in AD (Mao et al., 2015).

1.6.5 Application of UASB Reactors in Wastewater Treatment

The vast application in the treatment of diverse wastewater streams such as municipal sewage and wastewater from industries including cannery, beverage, slaughterhouse, brewery, distillery, and pharmaceutical wastewater streams, has revealed the robustness of the UASB reactor technology. Several studies have reported the UASB reactor application at different scales: full-scale, pilot-scale and laboratory-scale in different parts of the world. Researchers have also carried out several studies on the effect of operational and design parameters such as pH, temperature, HRT, and OLR on the performance of UASB reactors. Some studies reviewed have been tabulated and presented in Table 1.2.

Table 1.2: Previous studies on the performance of UASB reactors in treating diverse wastewater streams

References	Location	Type of Exp't	Wastewater stream	HRT	OLR (kgCOD /m^3/d)	Vel$_{up}$ (m/h)	pH	Temp (ºC)	COD$_{Inf}$ (mg/L)	Rem. Efficiencies (%)		
										COD	BOD	TSS
(Slompo et al., 2019)	Brazil	Pilot-scale	Domestic	3 d	0.09 - 1.49	-	6.8 - 7.8	19 - 27	434 - 2405	73 ± 15	80 ± 11	-
(Musa et al., 2019)	Malaysia	Pilot-scale	Slaughterhouse	24 h	1.75 - 16	-	6.7 - 7	36 ± 1	32,000 ± 112	> 90	> 90	> 90
(Rizvi et al., 2015)	Pakistan	Pilot-scale	Domestic	3 - 12 h	-	-	7.39 ± 0.27	17 - 38	474 ± 36.5	57 - 82	61 - 85	41 - 73
(Takahashi et al.,2011)	Japan	Pilot-scale	Domestic	8 h	-	-	-	10.6 - 27.7	342 ± 135	63 ± 13	-	66 ± 20
(Heffernan et al.,2011)	Semi-tropical regions	10 full-scale plants	Municipal	7.3 - 10.3 h	-	0.44 - 0.62	-	-	440 - 1293	44 - 77	37 - 80	45 - 84
(Nacheva et al., 2011)	Mexico	Pilot-scale	Slaughterhouse	7.2 - 21 h	4 - 15	0.05 - 0.15	7.33 - 7.35	20.9 - 25.2	2165 ± 210	76 - 90	-	-
(Chen et al., 2011)	China	Full-scale	Pharmaceutical	16.8 - 40.3 h	12.57 - 21.02	-	5.6 - 8.3	-	4726 - 19,951	39 - 85	-	-
(Hampannavar & Shivayogimath, 2010)	India	Lab-scale	Beverage	4 - 48 h	0.5 - 24	-	-	29 - 37	-	89.5	-	-
(Atashi et al., 2010)	Iran	Pilot-scale	Beverage	4 - 6 h	-	0.3 - 1	6.8 - 7.0	35 - 38	1800 - 2600	90	-	72
(Akbarpour & Mehrdadi, 2011)	Iran	Pilot-scale	Pharmaceutical	0.96 - 10.81 h	33.7 - 46.2	-	7 - 7.5	30 - 35	1850 - 15,170	54	-	-
(Satyanarayan et al., 2009)	India	Pilot-scale	Pharmaceutical	33 - 34 h	6.26 - 10.33	-	6.7 - 7.6	Ambient	17,200 - 27,200	86.2 - 91.6	90 - 95.2	62.6 - 68
(Al-Shayah & Mahmoud, 2008)	Palestine	Pilot-scale	Domestic	2 - 4 d	0.3 - 0.6	-	7.12 - 7.49	18.2 - 29	1267 ± 158	45 - 77	41 - 72	76 - 87
(Gao et al., 2007)	China	Pilot-scale	Brewery	11 - 82 h	5 - 48.3	-	7 - 7.5	37	16,500 - 22,520	80 - 97.3	-	-
(Parawira et al., 2005)	Zimbabwe	Full-scale	Brewery	24 h	6	-	6.5 - 7.3	37 ± 2	8240 ≥ 20,000	57	-	-
(Diamantis et al.,2005)	Greece	Pilot-scale	Cannery	6 - 12 h	4 - 16	-	4 - 10	25 - 36	940 - 5080	> 75	-	-
(Álvarez et al., 2006)	Spain	Pilot-scale	Domestic	4.7 - 18.8 h	-	0.28 - 1.11	6.98 - 7.48	13 - 22	160 - 460	39 - 57	44 - 73	57 - 85
(Azimi & Zamanzadeh, 2004)	Iran	Pilot-scale	Domestic	2 - 10 h	0.95 - 6.40	-	-	20 - 26	362 - 508	46 - 63	54 - 71	53 - 65
Singh & Viraraghavan, 2003)	Canada	Pilot-scale	Domestic	3 - 48 h	-	0.25 - 0.33	-	6 - 32	350 - 600	60 - 87	75 - 88	56 - 90
(Torkian et al., 2003)	Iran	Pilot-scale	Slaughterhouse	2.3 - 45 h	6.9 - 25	0.02 - 0.06	6.8 - 7.8	31 - 35	2205 - 5973	83 - 87	-	-
(Rodríguez et al., 2001)	Colombia	Full-scale	Domestic	6.9 - 24.9 h	-	0.17 - 1.11	6.6 - 7.1	27 - 27	463 - 538	73 - 84	-	-

(Source: Compiled by author)

1.6.6 Methane Gas Production from UASB Reactors

Global concern about greenhouse gas (GHG) emissions and energy security has fostered an interest in developing and exploiting non-petroleum-based renewable energy sources (Chandra et al., 2012). Bioenergy has been recognised as the fastest-growing energy alternative amongst renewable energy resources, with incredible potential in many parts of the world today (Münster & Meibom, 2011; Banerjee & Tierney, 2011). The European strategy for renewable energy sources identifies bioenergy as the most sustainable renewable energy source. According to Wu et al. (2010), expanding bioenergy is an imperative measure that will improve energy structure, protect the environment, safeguard energy security, and promote sustainability.

Anaerobic wastewater treatment technologies promote the utilization of bioenergy as the process allows the recovery of energy in the form of methane gas (about 50 - 75% of the biogas output) which can be converted into heat energy, electricity and biofuel from the biodegradation of organic carbon content in the wastewater stream (Chandra et al., 2012; Musa et al., 2018). The report by Van Lier (2008) stated that about 77% of organics removed during anaerobic treatment are converted to methane gas; this corresponds to the production of 2.75 kWh/kg of degraded organic matter. Wellinger et al. (2013) likewise noted that CH_4 from AD biogas contains as much as 50 - 55 MJ/kg energy content that can be utilized for energy potential applications such as electricity, heat and vehicle fuel. Wastewater treatment with high-rate anaerobic reactors such as the UASB has gained prominence amongst other anaerobic technologies due to the many advantages, the most protuberant being the formation of the biological granules in the sludge blanket, which enhances biodegradation for biogas production (Hulshoff Pol et al., 2004; Quaff et al., 2014). Sawyerr et al. (2019) reviewed the factors influencing biogas production in anaerobic systems. The authors observed that different substrates yielded different biogas generation rates. It was also found that the applied operational parameters also influenced biogas and subsequent methane gas production.

Presented in Table 1.3 is a review of some studies by researchers in different parts of the world to evaluate the biogas generation rate and the percentage methane composition observed for varying wastewater streams. These studies considered different organic substrates applied at varying OLRs, HRTs, temperatures and pH.

Table 1.3: Reported literature studies on UASB reactor methane gas production

References	Substrate	OLR ($kgCOD/m^3/d$)		HRT		Temp (ºC)	pH	Maximum Biogas Production	Maximum CH_4 Production	CH_4 Content in Biogas (%)
		Applied	Optimum	Applied	Optimum					
(Bakraoui et al., 2020)	RPMWW	1 - 10 gCOD/L/d	8.3	11.8 - 106 h	15.4 h	37 ± 2	7 - 7.4	62.5 L/d	45 - 63 L/d	73
(Musa & Idrus, 2020)	CSWW	10 g/L/d	-	24 - 48 h	48 h	37 ± 1	6.9 ± 0 .8	38 L/d	33.06 L/d	87
(Gao et al., 2019)	Blackwater	0.28 - 4.87	4.7	2.08 - 36.42 d	2.56 d	35	7	-	0.68 ± 0.08 m^3/d	-
(Tassew et al., 2020)	Swine manure slurry	4.5 - 6.5 gCOD/L/d	-	3.8 d	-	25 - 35	-	4.7 L/d	-	-
(Musa et al., 2018)	Slaughterhouse WW	0.2 - 15 g/L/d	10	24 h	-	35	6.7	-	5.5 L/d	75.8
(Montoya et al., 2017)	Diluted dairy manure	6.2 - 14.2 $gCOD_t/L/d$	-	7.5 - 12 d	-	20 - 30	7 - 7.6	-	0.73 L $CH_4/L/d$	70
(Antwi et al., 2017)	Potato starch processing WW	2.7 - 13.27	3.65	36 - 72 h	-	35 ± 1	6 ± 1	4.28 L/L.d	2.97 L/L.d	63.3 - 74.8
(Verma et al., 2015)	Synthetic textile WW	0.5 - 4.48	4.48	16 - 24 h	-	22 - 27	7.4	6040 ml/d	2910 ml/d	47 - 61
(Khan et al., 2015)	Domestic sewage	0.57 - 6.35	-	8 h	-	32 ± 3	-	0.22 m^3/kg	-	-
(Urbinati et al., 2013)	Swine manure	5.5 - 40.1	20.7	3.3 - 48 h	24 h	22.3 - 25	7.1 - 7.6	-	0.812 Nm^3 $CH_4/m^3/d$	-
(España-Gamboa et al., 2012)	Hydrous ethanol vinasse	7.27 - 22.16	17.05	7.5 d	7.5 d	30 ± 5	7	-	0.263 m^3 $/kgCOD_{added}$	84
(Gotmare et al., 2011)	Dairy WW	-	-	-	-	Mesophilic	6.9 - 7.1	179.35 m^3/d (Average)	125.55 m^3/day (Average)	75
(Rico et al., 2011)	Liquid fraction cattle manure	12.3 - 72.5 gCOD/L/d	72.7 gCOD/L/d	0.22 - 1.3 d	0.22 d	35	7.5 - 7.6	-	14.06 L $CH_4/L/d$	81.1 - 85.7
(Kaparaju et al., 2010)	Pig manure and straw stillage	2.26 - 41.2 gCOD/L/d	17.1 gCOD/L/d	48 h	-	55	Neutral	-	154.8 ml $CH_4/gCOD$	-
(García et al., 2008)	Dairy manure	5 - 12.7 gCOD/L/d	12.7 g	1.5 - 3 d	1.5 d	35	6.6	-	≈ 4.4 L $CH_4/L/d$	78

RPMWW is Recycled Paper Mill Wastewater; CSWW is Cattle Slaughterhouse Wastewater;

(Source: Compiled by author)

1.6.7 UASB Reactor Operational Limitations and Constraints

Despite the prominence gained by the UASB reactor technology over the last few decades (Chernicharo et al., 2015), some limitations with the technology linger, and these have become a target for researchers in recent times. Aside from the constraints for pathogen and nutrient removal, which has called for the implementation of post-treatment units (to be discussed in the next section), other factors that can limit the comprehensive implementation and performance of UASB reactors will be discussed in this section. Critical analysis of the design and operation of UASB reactors highlights issues related to long start-up periods, temperature constraints, inability to remove emerging/micropollutants, scum formation and accumulation, resource recovery and management of biogas generated, which have called for much attention in recent times (Chernicharo et al., 2015).

- *Limitations with Long Start-up Periods*

Undeniably, the successful operation and efficiency of UASB reactors are principally dependent on the granulation within the sludge blanket where most of the treatment occurs (Hulshoff Pol et al., 2004; Chong et al., 2012). According to Sato et al. (2006), the period it takes for the biological granules to reach complete acclimatization is known as the "start-up" phase. The start-up process is delicate and can be time-consuming. It originates from the initial feeding of the reactor to the period when the preferred granular sludge is attainable. Previous research has attested that the start-up process is the most significant shortcoming of the UASB reactor implementation. The start-up period can be from two to eight months (Vlyssides et al., 2008), which could be a major challenge in its application. Studies conducted in a bid to mitigate this challenge showed that inoculating the reactor with a seed sludge could facilitate the quick adaptation and growth of the microbes, shortening the acclimatization period. This period is influenced by numerous factors, including physicochemical and biological parameters such as wastewater characteristics, the presence and evolution of active microbial community in the inoculum and the operating conditions (Singh & Viraraghavan, 2003; Lew et al., 2003; Chong et al., 2012). The inoculum could be an already digested sludge, manure, cow dung, glucose, etc. (Álvarez et al., 2006; Kumar et al., 2011; Rizvi et al., 2014).

- *Temperature Constraints*

Treatment of sewage with anaerobic systems in temperate climatic regions is considered a major challenge since the biodegradability of complex wastewater streams is prolonged, and hydrolysis of particulate matter becomes limited, almost impossible at such low temperatures (Chernicharo et al., 2015). Temperature plays a critical role in UASB reactor performance.

Organic matter biodegradation is enhanced within mesophilic temperature ranges as microbes responsible for AD are mesophilic (thrive in a temperature range from 20 - 45 ^oC) (Bodík et al., 2000; Lettinga et al., 2001). The high efficiency of UASB reactors in tropical and subtropical regions has been iterated in many studies, making this technology economically feasible for implementation in most developing countries as the climatic conditions in these regions are favourable for AD processes, with no external heat required, which increases operational costs (Von Sperling & Chernicharo, 2005; Aiyuk et al., 2006). Lew et al. (2003) reported 44% COD removal efficiency at 10 ^oC against 82% removal at 28 ^oC in their study. Similarly, Esparza-Soto et al. (2019) also observed in their research that COD removal efficiency dropped from 78 ± 12% to 39 ± 8.6% when OLR was increased from 2 to 6 $kgCOD_s/m^3/d$ at a 15 ^oC temperature whilst temperatures above 20 ^oC maintained a satisfactory COD removal efficiency of 90% even with an increase in applied OLR. Nevertheless, new technology interventions have allowed the employment of temperature-controlling units such as heated water jackets, water baths and thermostatically controlled hot water serpentine to regulate UASB reactors operating temperature in colder regions. This intervention has been reported by Musa et al. (2018) in Malaysia, Tassew et al. (2020) in Norway, Gonzalez-Tineo et al. (2020) in Canada and Chen et al. (2020) in China, howbeit this intervention has economic implications.

- *Limitations with the Removal of Emerging/Micro-pollutants*

Xenobiotic organic compounds (XOC), also referred to as micro/emerging pollutants, are chemical compounds existent as pollutants in natural environments. These can be natural products occurring in mammals and plants. They also can be anthropogenic, representing groups of pharmaceutical medicines (antibiotics, synthetic hormones, analgesics, and anti-inflammatories, etc.); they can also be obtained from personal care products (PCPs) as well as household chemicals such as surfactants, fragrances, preservatives, in compounds used in producing plastics and resins, pesticides and natural hormones together with their by-products (Brandt et al., 2013; Luo et al., 2018). These compounds have, in the last few years, gained much recognition as a result of their antagonistic effect on aquatic life as they are endocrine disrupting chemicals (EDCs) (Kim & Aga, 2007), with WWTPs considered as one of the major hotspots for the potential evolution and spread of these substances into the environment (Michael et al., 2013). Several researchers have opined that UASB reactors are inefficient in removing these emerging pollutants whose presence has become ubiquitous in wastewater streams (Brandt et al., 2013; Buarque et al., 2019). Brandt et al. (2013), however, observed that

HRT is one significant parameter that controls the removal of hydrophilic and less biodegradable compounds, such as trimethoprim and sulfamethoxazole. The same authors reported that post-treatment units (trickling filters, polishing ponds, and submerged constructed wetlands) could substantially enhance the removal of most target micro-pollutants.

Moreover, Chernicharo et al. (2010) conducted a study in which a UASB/PP combined system could attain a 99% removal of bisphenol A with an undetectable concentration of sulphide. Nevertheless, the removals of phthalates and linear alkylbenzene sulphonates (LAS) were limited. Buarque et al. (2019) likewise conducted a study to assess the efficiency of a UASB reactor integrated with a micro-aeration system to remove emerging micro-pollutants (bisphenol A; diclofenac; estrone; estradiol; ethinylestradiol; sulfamethoxazole; trimethoprim). Results showed that the UASB reactor achieved very low (< 10%) removal efficiency of these compounds, but post-treatment with micro-aeration resulted in overall removal efficiency of not less than 50% in all the compounds. The authors believed that these post-treatment units could effectively eliminate hydrophobic and hydrophilic pharmaceuticals and EDCs just like activated sludge systems, although adequate research to ascertain this fact is limited and calls for further studies. In a study to evaluate the treatment of veterinary antibiotics in swine wastewater, Qian et al. (2019) employed a UASB/SBR/Fenton-like oxidation. The system attained antibiotic reduction efficiency at 74%, which was approximately a 65% increment compared to the conventional Fenton reaction. Hou et al. (2019) also employed the integration of UASB/anoxic-oxic tank/advanced oxidation process (AOP) systems to remove 18 antibiotics and 10 antibiotic resistance genes (ARGs) in real pharmaceutical wastewater. The UASB reactor achieved 85.8 ± 16.1% for the removal of antibiotics. Mass balance results showed that degradation and sorption to sludge were also predominant removal mechanisms. The Fenton/UV combined system was the most effective AOP for ARGs removal.

- *Challenges with Scum Formation and Accumulation*

The formation and accumulation of scum in UASB reactors remain one critical challenge of the UASB operation. Scum is a layered floating material that forms at the liquid-gas interfaces of bioreactors; this substance's exact attributes mainly depend on the raw sewage composition (Souza et al., 2006; Van Lier et al., 2010). The management of scum in UASB reactors is very dire as its composition and biodegradability are not fully comprehended (Halalsheh et al., 2005; Pereira et al., 2009). Scums can be accumulated in the settler compartment, inside the GLS separator, and also in the gas hood and can lead to problems including i) causing a blockage in the effluent weirs, which results in an unbalanced withdrawal of effluent; ii) generating odour, attract flies, unwanted vermin and creates aesthetically unpleasant sight; iii) disrupting the natural flow of biogas as the unwithdrawn scum can thicken and form a scum layer which can prevent the regular flow of biogas. If this biogas finds its way into the settler compartment, it can release odorous gas and GHG emissions, reducing the energy recovery potential of biogas (Souza et al., 2006; Pereira et al., 2009). Van Lier et al. (2010) proposed that installing scum baffles and scum removal bowls in the settler compartment can help mitigate this menace. Similarly, Chernicharo et al. (2009) proposed a new system employing auto-generative gas pressure under the gas hood to mitigate the challenge of scum in the gas hood. According to Chernicharo et al. (2015), the installation of internal hydrostatic scum removal devices within the GLS separator controls the water level within the gas hood, allowing the scum to empty into a weir for collection and disposal.

- *Challenges with Resource Recovery*

One main advantage of the UASB reactor technology over other conventional wastewater treatment systems is the issue of resource recovery. Jeppsson & Hellström (2002) opined that for wastewater management to be sustainable, it has to be one that biogas can be captured for energy recovery purposes, stabilized sludge, and reclaimed water to be used for agricultural activities. These categories fit very well AD process in UASB reactor operations, giving it an upper hand over other technologies. Nonetheless, researchers in recent times have observed some limitations and constraints in the management of these resources. Reclaimed water from UASB effluent is a rich source of nutrients (N and P), which can enhance agricultural production; however, if this effluent finds its way into water bodies, the potentially detrimental effects have been discussed in the next section (sub-section 1.6.8). On the other hand, the retained sludge can be used as fertilizer or soil conditioner for agricultural purposes (Jeppsson & Hellström, 2002).

Nevertheless, using reclaimed water and sludge for agricultural purposes can result in detrimental health challenges, as Mainardis et al. (2020) asserted. These authors reported traces of micro-pollutants and heavy metals in effluent and sludge from UASB reactors. New research interest in the energy recovery potential of sludge and scum from UASB reactors is gaining recognition (Chernicharo et al., 2015). A study by Rosa et al. (2018) evaluated the potential use of combined energy obtained from sludge and biogas generated for the thermal drying of dewatered sludge.

- *Greenhouse Gas Emissions (GHGs)*

Biogas generated from AD with municipal wastewater as substrates mainly contain CH_4 (55 - 70%) and CO_2 (20 - 35), and ≈ 2% trace gas components. These gases are classified as greenhouse gases (GHGs), and they can escape into the atmosphere (GHG emissions) and contribute to global warming if not recovered or properly managed (Hartley & Lant, 2006; Souza et al., 2012). The USEPA (2012b) observed that wastewater and manure management is the second most significant contributor to global anthropogenic CH_4 emissions after landfill sites. However, energy recovery from anaerobic digesters treating sewage is still in the early stages. Many small WWTPs employing UASB reactors vent the biogas, whilst, at larger WWTPs, the biogas is flared to reduce the GHG emissions; thus, the energy recovery potential is not utilized. Chernicharo et al. (2015) mentioned that at some UASB treatment plants in India, biogas generated is used as a combustion fuel for dual-fuel biogas motors for energy generation purposes. Biogas use as a renewable energy source could enhance a decentralized generation of power, which is agreeable with sustainable development concepts.

- *Odour Nuisance from Waste Gas*

Atmospheric emissions of odorous gases from anaerobic reactors treating sewage are of enormous concern to the scientific body. This is one challenge in implementing UASB reactors, which may unquestionably thwart the dissemination of this technology in predominantly populated urban centres. The malodorous compounds are reduced amino and sulphur compounds, such as amino-sulphides, mercaptans and sulphides. H_2S, which results from the de-assimilative reduction of sulphates, is the one major compound concomitant to sewage odours, although other sulphur compounds may also contribute (Dumont, 2015). Reduced H_2S gases that dissolve in the effluent may also escape into the atmosphere causing odour problems, not neglecting the toxicity when inhaled. Another concern is their highly corrosive nature, causing considerable damage to concrete and steel at WWTPs (Souza et al., 2012; Azevedo et al., 2018). Most generated sulphide compounds occur in the sludge deposits at the bottom of pipes or the layers of biofilms fixed on the walls of pipes in the sewer networks (WEF, 2004; Sharma et al., 2008). The total effluent H_2S concentration depends on the influent's sulphate concentration and sulphate-reducing bacteria within the reactor. Khan (2012) reported sulphide concentrations between 7 and 20 mg/L in UASB reactor effluent. Several options are available for consideration during the control of malodorous emissions. Chernicharo et al. (2010) reported that in qualitative analysis, the main features of each treatment method for odorous emissions offer biochemical methods, direct combustion and biofilters as the best treatment perspectives for waste gases, taking into consideration the simplicity and cost efficiency for implementation in developing countries. Other alternatives to be considered for removing dissolved sulphide in UASB effluent include stripping in the dissipation chamber, as reported by Souza et al. (2012) and Glória et al. (2016), followed by the biological treatment step. Chuang et al. (2005) and Chen et al. (2010) likewise found that micro-aeration could be employed for the biological oxidation of sulphides to elemental sulphur, which according to the authors, provides an excellent reuse potential as the process is economically feasible.

- *Inability to Recover all CH_4 Generated*

Research has shown that a high amount of the CH_4 produced remains dissolved in the UASB reactor effluent and gets washed out with the effluent. Urban et al. (2007) reported concentrations between 20 and 25 mg/L of dissolved CH_4 observed in a pilot study. This results in a significant reduction of the energy potential of anaerobic reactors. Likewise, Chernicharo et al. (2015) asserted that in a study carried out on full-scale plants in Brazil, about 30 - 40% of the methane generated remained dissolved in the effluent. Similarly, Souza et al. (2011) stated that the loss of methane in solution is often very high and ranges from 36 - 41% of the total methane produced in a UASB reactor. Giménez et al. (2012) asserted that methane gas recovery was less at lower temperatures because the solubility of CH_4 is higher at such temperatures. However, the removal or recovery of CH_4 is essential not only due to the energy potential but mainly because CH_4 is a GHG with a higher potential for global warming than CO_2 (Gioelli et al., 2011; Sambusiti et al., 2015). Some suggested interventions to eliminate dissolved CH_4 from anaerobic reactor's effluent include micro-aeration as reported by Hartley & Lant (2006), Cookney et al. (2012) proposed the use of degassing membranes, whilst Luo et al. (2014) performed a study using membranes to remove CH_4 from the effluent, in which recovery of dissolved CH_4 was attained at 86%. Though satisfactory, the authors mentioned the technique was capital-intensive. Matsuura et al. (2010) also reported on the success of a two-stage closed down-flow hanging sponge (DHS) system to remove dissolved CH_4 in UASB reactor effluent through recovery and biological oxidation. The research conducted by Crone et al. (2016) highlighted that using non-porous membranes, micro-porous membranes, and DHS reactors recovered 92.6%, 98.9% and between 57 - 88%, respectively, of total dissolved CH_4 in UASB reactor and AnMBR effluent.

Other losses through leakages and emissions to the surface of the settling compartment could also occur. Chernicharo et al. (2015) have observed that the use of combined sewers networks for sewage and stormwater with irregular connections could result in high sewage dilutions, leading to a sharp fall in the net production of biogas during rainy seasons.

1.6.8 Post-treatment Technologies for UASB Reactor Effluent

Knowledge of UASB reactor technology has significantly expanded these past few decades (Noyola et al., 2012), and these systems have been incredibly accepted and implemented in many different parts of the world, with many full-scale Plants under construction to supplement existing facilities (Chernicharo, 2006; Von Sperling & Chernicharo, 2005). Despite the numerous advantages associated with the application of UASB reactors over other aerobic/anaerobic technologies, one major setback with the UASB technology is the inability of these systems to produce high-quality effluent that meets World Health Organization (WHO) stringent discharge guidelines for reclaimed water use or discharge into the environment as well as complying with discharge standards customary with most environmental agencies. Thus, UASB reactor effluent usually requires post-treatment to ensure the final effluent meets the requirement of the environmental legislation, protecting receiving ecosystems. The main objective of the post-treatment unit is to facilitate the removal of residual organic matter and also elements that are barely affected by anaerobic treatment processes: nutrients (nitrogen and phosphorus) and pathogens, enhancing the quality of the reclaimed water (Daud et al., 2018; Foresti et al., 2006).

Permissible limits for organic matter imposed by these environmental agencies are usually expressed in terms of BOD effluent discharge standards, thus if the receiving water body has inadequate capacity to assimilate the UASB reactor effluent, a post-treatment unit (could be aerobic or anaerobic) will be required (Chernicharo et al., 2010). Regarding limitations on nutrient removal, discharge of nitrogen (N)- and phosphorus (P)-rich effluent into surface water bodies may result in algal bloom due to eutrophication. When nutrient removal is required to meet standards for discharge into water bodies, the choice of post-treatment should be carefully analysed as anaerobic systems provide good biodegradable organic matter removal but not N and P removal efficiently (Chernicharo et al., 2010). Foresti et al. (2006) reported N and P concentrations ranging from 30 - 50 mg/L and 10 - 17 mg/L, respectively, in anaerobic effluent from municipal WWTPs. Regarding microbiological indicators, satisfactory faecal coliform (FC) reduction, usually around 1 log-unit removal, has often been reported for UASB reactors (Khan et al., 2012, 2014). With other types of microbes, such as protozoans (mainly Giardia and Cryptosporidium) and viruses, few references have been given regarding their reduction or inactivation in UASB reactors. For helminth egg removal, reports indicate gross insufficiency to produce effluent that could be used for irrigation purposes (Von Sperling et al., 2002). Thus, aerobic/anaerobic combined systems could be employed to attain desired effluent quality regarding pathogen removal. If land availability becomes an issue for the post-treatment unit for pathogen removal, compact disinfection processes such as ultraviolet (UV) radiations, ozonation and chlorination processes could be implemented as suitable post-treatment for pathogen removal. However, disinfection must be carried out carefully, considering all necessary factors, as these compounds can be toxic to human health and the aquatic environment (Von Sperling & Mascarenhas, 2005; Chernicharo et al., 2010).

Several research works have assessed the efficiency of anaerobic/aerobic integrated systems as the most suitable wastewater treatment option. However, there are instances where the combination of diverse anaerobic processes can meet less stringent requirements on the final effluent quality (Chernicharo, 2006). Although a majority of aerobic systems are well known to require aeration units and blowers, rendering them energy-intensive, implementation of these aerobic systems as post-treatment units do not consume much energy as the majority of the organic pollutants have already been degraded in the pre-treatment anaerobic systems (Chernicharo, 2006). Khan et al. (2011) reported that post-treatment systems could be classified as natural systems (polishing ponds, constructed wetlands, duckweed ponds), physicochemical methods (coagulation-flocculation, dissolved air flotation, two-stage flotation and ultraviolet), micro-aerobic methods (trickling filters, downflow hanging sponge, flash aeration), high-rate aerobic methods (activated sludge processes, sequencing batch reactors, rotating biological contactors) and final polishing techniques (membrane bioreactors, membrane filtration, slow sand filtration). Commonly implemented UASB/post-treatment systems include the UASB/activated sludge (AS) systems, UASB/sequencing-batch reactor (SBR), UASB/biofilter (BF), also known as trickling filter (TF), UASB/downflow hanging sponge (DHS), UASB/maturation (polishing ponds), UASB/rotating-biological contactor (RBC), UASB/constructed wetland (CW). Other post-treatment systems include advanced oxidative processes (AOP) and membrane bioreactors (MBR). Presented in Table 1.4 is a review of some studies conducted wherein different technologies were employed as post-treatment systems for UASB reactor effluent. The studies reviewed revealed that the post-treatment helped improve the final effluent quality, most of which met discharge limits.

Table 1.4: Reported literature studies on UASB combined with post-treatment units

References	Type of Exp't	Substrate	UASB Operating Conditions			COD Rem. Eff. in UASB (%)	Post-Treatment Unit	P.T COD Rem. Eff. (%)	Nutrient Removal (%)				Pathogen Removal (%)		
			OLR (kgCOD/m^3/d)	HRT	Temp (ºC)				NH_4^+ - N	TKN	TN	TP	TC	FC	E. coli
(Gonzalez-Tineo et al., 2020)	Pilot-scale	Swine WW	3.26 - 10.17	0.79 ± 0.20 d	37 ± 2	65 ± 13	Aerobic hybrid reactor	99 ± 0.2	57	-	-	-	-	-	-
(El-Khateeb et al., 2019)	-	Sewage	1.7654 - 1.853	5 - 6 h	-	57 - 59	P-UASB/DH NW reactor	68 - 71	56 - 57	49 - 54	-	-	-	90.5	-
(Ahmed et al., 2018)	Full-scale	Municipal sewage	-	-	< 28	88.9	TF/ST	91.2	43.6	-	-	81.7	-	99.6	-
(Von Sperling, 2015)	Small full-scale	Municipal sewage	-	-	-	57	MP/CRF	72	48	-	40	18	-	-	99.99
(Von Sperling, 2015)	Small full-scale	Municipal sewage	-	-	-	57	HSF-CW	72	7	-	6	28	-	-	98
(Raboni et al., 2014)	Full-scale	Municipal sewage	-	13.6 h	20.5 - 28.5	71.1	SSHFP	79.2	-	-	-	-	-	98.8	-
(Banihani & Field, 2013)	Lab-scale	Synthetic HS WW	1.1 - 4.4 gCOD/l/d	13.9 - 56	25	66.3 - 86.4	ASP	97.1 - 99.6	96	-	-	-	-	-	-
(Bhatti et al., 2014)	Lab-scale	MWW	0.14 gCOD/l/h	48 h	25 - 35	85	AOP	99	-	95	84.2	19.8	-	-	-
(Khan, et al., 2012)	Pilot-scale	-	-	8 h	13 - 40	≈ 65	FA	87.3	-	-	-	-	-	99	-
(Mungray & Patel, 2011)	Full-scale	Municipal sewage	-	8 - 9 h	20 - 34	41	ASP	86	-	-	-	-	99.9	99.9	-
(El-Kamah et al., 2011)	Pilot-scale	Onion dehydration WW	4.7 - 7.4	5.2 - 6 h	21 ± 6	44 - 56	DHS	85 - 92	95 - 99	65 - 72	-	-	-	-	-
(Von Sperling et al., 2010)	Full-scale	Urban WW	-	0.5 d	23	47	PP/CRF	58 - 64.5	56.3	-	56.4	-	-	-	99.97
(Von Sperling et al., 2010)	Full-scale	Urban WW	-	0.5 d	23	62.1	Planted/Unplanted CWS	81 - 89	17.7	-	16.2	-	-	-	99.98

(Bastos et al., 2010)	Pilot-scale	WW	-	-	-	73	WSP	-	90	70	-	30 - 50	-	-	99.99
(Bastos et al., 2010)	Pilot-scale	WW	-	-	-	73	HF-CW	-	20 - 80	20 - 70	-	30 - 50	-	-	-
(Moawad et al., 2009)	-	Domestic sewage	1.33 - 5.28	3 - 4h	-	54 - 57	ST/SBR	82 - 94	89 - 100	44 - 77		5.5 - 665	-	-	-
(An et al., 2008)	Lab-scale	Synthetic WW	-	3.5 - 8 h	30 - 32	-	MBR	-	98.2	-	48.1 - 82.2	-	-	-	-
(Tessele et al., 2005)	Pilot-scale	Municipal sewage	0.8	-	-	44.4	WSP	51.4	58.1	-		37.3	37.2	-	45.7
(Tessele et al., 2005)	Pilot-scale	Municipal sewage	0.8	-	-	44.4	Two-stage flotation/UV	61 - 79	40 - 58	-	-	45 - 98	35	-	77
(Von Sperling & Mascarenhas, 2005)	-	-	-	-	23 - 48	56.8	PP	61 - 72	45 -70	55 - 64	-	26 - 50	-	-	99.99
(Tandukar et al., 2005)	Pilot-scale	Municipal sewage	-	6 h	20 - 25	63	DHS	91	28	40	-	-	99.9	99.9	-
(Keller et al., 2004)	-	Domestic sewage	-	-	-	62	BF/UV	84.3	-	-	-	-	99	-	-
(Torres & Foresti, 2001)	Pilot-scale	Domestic sewage	-	6 h	21 ± 2	63 - 77	SBR	92	69 - 100	64 - 89	-	22 - 72	-	-	-
(de Sousa et al., 2001)	Demo-scale	Raw sewage	2.8 - 5.6	3 - 6 h	-	-	CWS	79 - 83	70	70	-	89	-	99.9	-
(Cavalcanti et al., 2001)	Demo-scale	Sewage	26 ± 1	3 d	-	63 - 64	PP	79	51	55	-	-	-	99	-

(Source: Compiled by author)

1.7 Trickling Filters

1.7.1 Process Mechanism

Trickling filters (TFs) are typically aerobic processes employed to eliminate organic matter and ammonia nitrogen from wastewater through an attached growth process. A TF is comprised of a fixed bed media through which wastewater trickles during treatment. Essentially, TFs are a solid-liquid-gas system in which the sewage (liquid) passes over the biofilms (solid) and interact with air (gas), and simultaneously nitrogen compounds and organic matter are absorbed and degraded subsequently by the microbes in the biofilm (Wiesmann et al., 2007; Zhu et al., 2016). Microbes responsible for the treatment are attached to a medium, which forms a slime growth on the medium referred to as the zoogleal film. During the operation of TFs, the biofilm is inclined to grow and thickens due to microbial growth. As the biofilm grows, it begins to lose its ability to cling to the media, and that portion unable to remain attached will detach from the media, a process known as "sloughing". The sloughed biofilms move together with the treated effluent into the underdrain system provided for TF effluent collection and are conveyed downstream to a solids separation unit such as a clarifier for settling and removal of sloughed biofilms before final effluent discharge (Wiesmann et al., 2007; Zhu et al., 2016). A cross-section of the solid-liquid-gas interface in a TF is schematically presented in Figure 1.6.

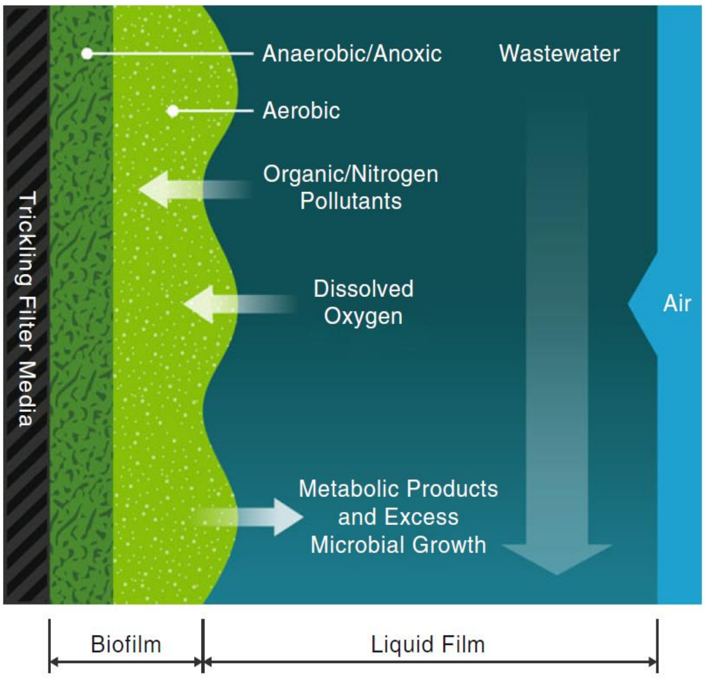


Figure 1.6: Cross-section schematic of wastewater, biofilm and media in a TF

(Adapted from Zhu et al., 2016)

1.7.2 Physical Description of Trickling Filters

Trickling filters may be circular with a rotating distribution arm or stationary, having a dosing chamber and a spray field. The major components of a TF are described below:

- *Distribution System:*

The distribution system distributes sewage over the surface of the media. A uniform hydraulic load per unit area is required for optimum treatment efficiency. Circular TFs employ rotating arms for sewage distribution, whilst stationary TFs employ fixed spray heads for wastewater distribution. The rotating arms are made of two or more horizontal pipes which are suspended over the filter media. Sewage distribution is through orifices situated along one side of the pipes. The rotary arms are usually set in motion by the force of wastewater flowing through the orifices along the side of the arm. They can also be motor-driven to control the speed of rotation. Contrarily, the stationary TFs with fixed spray heads require extensive pumping and piping systems to ensure flow distribution, but they are usually not implemented due to difficulty in access for repairs and maintenance (PDEP, 2016).

- *Filter Media:*

The media provides a surface for the zoogleal film (biological slime layer), which consists of various microbes to attach and grow. The filter media could be rock/slag, redwood or synthetic material. The rock/slag filter media was the original form of media employed for TF systems, with typical sizes between 2 - 4 inches in diameter. Uniform-size media is usually recommended to permit adequate ventilation through the void spaces. Literature has reported 35% void space between rock media. The use of redwood lumber to support biofilm growth has likewise reduced (PDEP, 2016). Recent TFs now employ synthetic materials, typically lightweight plastic materials with approximately 95% void space between the media. Synthetic media has advantages such as a larger surface area to support microbial growth, large void spaces promote adequate airflow and ventilation, and the uniform media sizes permit even load distribution coupled with ease of installation, handling and cleaning of lightweight media. Different configurations of synthetic media exist, such as random dump plastic media, modular cross flow or modular vertical flow plastic media (Figure 1.7), the foremost requirement being the large specific surface area (typically between 85 and 140 m^2/m^3) and void ratio (typically > 90%) to ensure the provision of ample space for biofilm growth and wastewater and air circulation (Burgos et al., 2015).

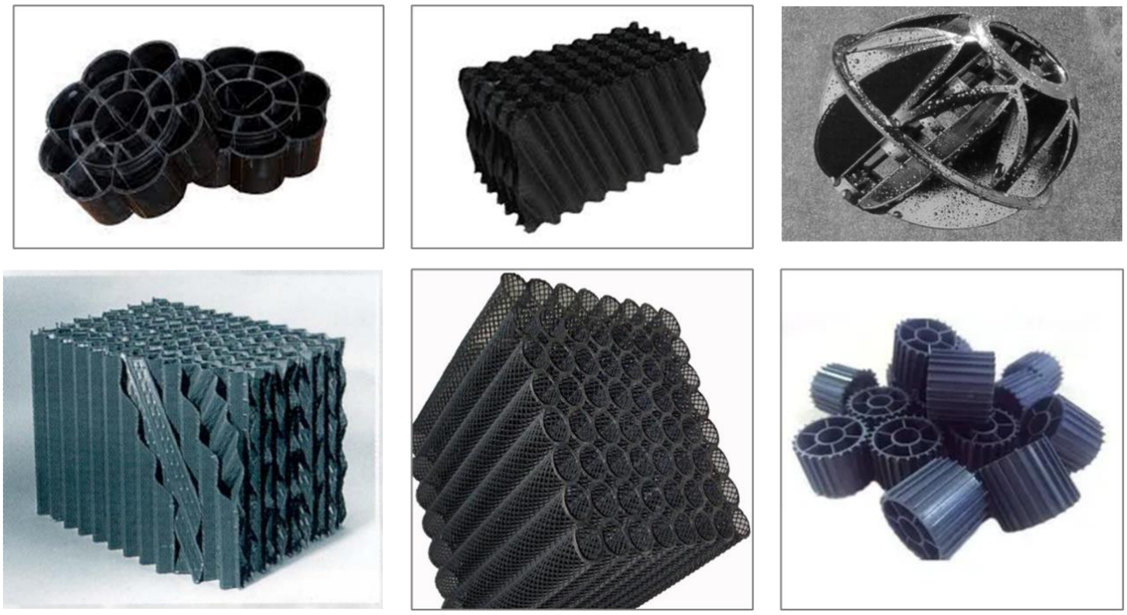


Figure 1.7: Typologies of synthetic support used as media for trickling filters

(Adapted and modified from Burgos et al., 2015)

- *Containment Structure:*

TFs containment structure, also known as retaining structure, can vary in construction and design based on the media type used. Containment structures are typically precast concrete tanks. Random dump and rock media are not self-supporting and hence require lateral structural support to contain the filter media, whilst modular plastic media are self-supporting and do not require lateral structural support from the containment. Self-supporting media might require using materials such as fibreglass, wood, welded bolt and steel as containment structures (USEPA, 2000; Zhu et al., 2016).

- *Underdrain System:*

TFs are primarily designed to have an underdrain system which serves the following purposes: i) convey the treated effluent to a solids separation unit such as a clarifier; ii) provision of support to media; and iii) create a plenum that permits air circulation through the media bed. Several designs for underdrain systems have been employed over the years. Rock media are usually designed with concrete underdrain blocks, whilst various support structures, including concrete columns with beams and reinforced fibreglass grating, have been used for plastic and wood media. Recently, field-adjustable plastic stanchions have been used as alternatives to conventional underdrains (PDEP, 2016; Zhu et al., 2016).

- *Recirculation System*

Recirculation involves recycling the TF effluent and bringing it into contact with the biofilm more than once. Thus, returning a portion of the treated effluent to the TF treatment unit is recommended for enhancing BOD and nitrogen removal efficiencies of high-rate TF processes (PDEP, 2016; Pearce, 2004). Some benefits of recirculation include: i) it helps improve flushing, keeping biofilms refreshed, thereby reducing odour; ii) recirculation of treated effluent dilutes incoming influent load concentration, which helps with even distribution of load through the entire depth of the filter media. Dilution of influent flow through recirculation acts as a buffer for toxic or shock loads; iii) recirculation recycles dissolved oxygen from the TF effluent and helps to improve system performance; iv) research has shown that recirculation enhances the possibility of maintaining a consistent flow to the TF during diurnal flow variations, providing a steady rotation speed for hydraulically driven rotating arms, and maintaining adequate wetting rates. Biofilm not continually wetted and supplied with substrates from sewage becomes ineffective (Farmer, 2013), but a single forward flow cannot provide adequate wetting efficiency in most cases. The recirculation rate usually ranges from 0.5 - 4 times the average influent flow. The ratio of sewage volume circulated to the influent sewage volume is called the recirculation ratio (Zhu et al., 2016). A cross-sectional view of a typical TF has been presented in Figure 1.8.

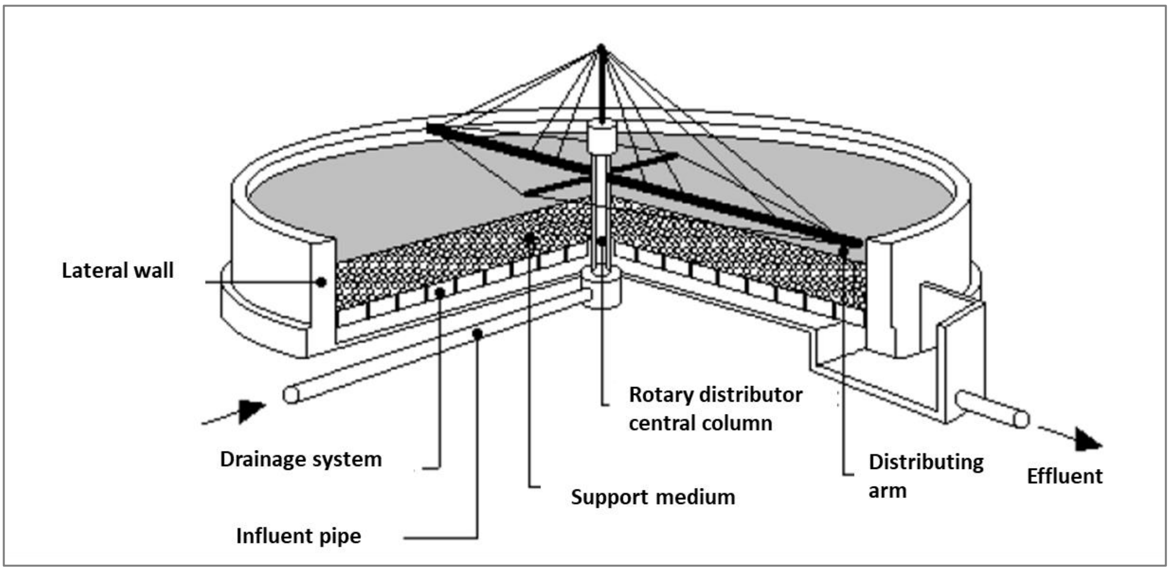


Figure 1.8: Cross-sectional view of a Trickling Filter

(Adapted from Jordao & Pessoa, 2009)

1.7.3 Classification of Trickling Filters

Trickling filters are generally classified according to their design hydraulic and organic loading rates. The loading rate is an essential design factor regardless of the mode of sewage application, whether intermittent or continuous flows, at a constant or varying rate. These classifications are Low-rate, Intermediate or Standard-rate, High-rate and Super high-rate, also known as Roughing filters (Guyer, 2014).

Low-rate filters are usually employed for loadings below 40 $kgBOD_5/100m^3/d$ (25 $lbBOD_5/1000/ft^3/d$). These types of TFs have fewer challenges regards to odours, filter flies, and medium plugging due to the lower applied loads. Most low-rate filters are circular with rotating arms. The sloughed solids from these filters are generally well-digested; hence, they have lower solids yields than higher-rate filters. High effluent quality is readily attainable if the TF design incorporates filter media with bio-flocculation capabilities or an excellent secondary clarification (USEPA, 2000).

Intermediate-rate filters have applied loads up to 64 $kgBOD_5/100m^3/d$ (40 $lbBOD_5/1000ft^3/d$). These systems require recirculation to ensure good distribution and thorough blending of the filter and secondary effluent. Sloughed biological solids from an intermediate-rate TF are not well digested compared to those from a low-rate TF (USEPA, 2000).

High-rate filters are those TFs that have their applied loads at the maximum organic loading capacities of the TFs and receive loads ranging from 64 to 160 $kgBOD_5/100m^3/d$ (40 to 100 $lbBOD_5/1000ft^3/d$). These systems are less likely to attain desired secondary effluent quality without a second-stage process. Thus, high-rate TFs are mostly used with combined processes (USEPA, 2000).

Super high-rate filters, as the name implies, have very high design loading, ranging from 160 - 480 $kgBOD_5/100m^3/d$ (100 to 300 $lbBOD_5/1000ft^3/d$). These systems are usually employed to decrease organic loading preceding discharge to subsequent oxidation processes, or they can act as a pre-treatment for high organic load wastewater for another biological treatment process. They apply to high-strength industrial wastewater (Guyer, 2014; USEPA, 2000). Table 1.5 presents the typical design and operational parameters for the various classifications of TFs.

Table 1.5: Design and operation data for trickling filters

Parameter	Unit	Classification of trickling filters			
		Low-rate	**Intermediate-rate**	**High-rate**	**Super high-rate**
Support material	-	Rock, Slag	Rock, Slag	Rock	Plastic
Specific surface area	m^2/m^3	40 - 70	40 - 70	40 - 70	80 - 200
Porosity	m^3/m^3	0.4 - 0.6	0.4 - 0.6	0.4 - 0.6	0.90 - 0.97
Density of support material	kg/m^3	800 - 1500	800 - 1500	800 - 1500	30 - 100
Hydraulic loading	$m^3/m^2/d$	0.5 - 3.0	3 - 10	8 - 40	10 - 70
Loading per volume	$g/m^3/d$ BOD_5	100 - 400	200 - 500	500 - 1000	500 - 1000
Height	m	1.0 - 2.5	1.0 - 2.5	1.0 - 2.5	1.0 - 3.0
Recirculation ratio	-	0	0 - 1	1 - 2	1 - 2
Removal efficiency	% BOD_5	80 - 90	50 - 70	65 - 80	65 - 80

(Adapted from Wiesmann et al., 2007)

1.7.4 Advantages and Disadvantages of Trickling Filters

The use of TFs for wastewater treatment are prevalent in most developing countries due to their high efficiencies in eliminating organic matter. TFs are popular due to their simplicity and reliability. They are also suitable for places where large land areas are not available for wastewater treatment or where land costs are high such as urban centres. Additionally, TFs are usually implemented as post-treatment systems in anaerobic wastewater treatment plants as they can eliminate residual organic matter and remove nutrients from wastewater; one major weakness of anaerobic systems (Lemji & Eckstädt, 2014; USEPA, 2000). Most often, treatment configuration for high-strength organic wastewater involves anaerobic systems followed by aerobic processes. In these instances, TFs are preferable to activated sludge processes (ASPs) due to the aeration method. For TFs, air flows passively employing natural air drift or forced ventilation systems with low-capacity blowers. In contrast, the ASP requires actively driven diffused aeration, using high-capacity blowers or aerators, making the ASP a more energy-intensive process than the TF (Chernicharo, 2006; Foresti et al., 2006). Another major advantage of the TF over ASP is their ability to handle shock loads without any extreme upset to the system's performance. TFs are also considered to be less affected by toxic compounds compared to other biological systems (WEF, 2000). TFs are efficient nitrification units for nitrogen removal. The process is durable and requires moderate technical expertise and skills compared to other aerobic systems (PDEP, 2016; USEPA, 2000).

Despite the numerous advantages of TFs, they have some limitations, just like any other wastewater treatment plant. Some of which include the possibility of needing additional treatment, such as disinfection, to meet more stringent standards of effluent discharge. They may also be inclined to vectors, odour and snail problems. TFs can accumulate excess biomass that cannot retain the aerobic conditions required for their optimum performance, thereby impairing TF performance. Biomass thickness is influenced by hydraulic dosage rate, media type, temperature, organic substrate and the nature of biological growth. Depending on the type of media used, TFs may be more prone to clogging and ponding problems due to excessive sloughing, resulting in inefficient pollutant removal and poor effluent quality. Again, they may require low loadings in order to prevent clogging problems. Thus, regular operator attention may be required to control these issues. Another challenge TF operation faces in colder climates is the tendency to freeze. Low hydraulic loading onto the filters may cause icing of the rotating arm orifices or the spray nozzles during winter months (PDEP, 2016; USEPA, 2006).

1.7.5 Application of Trickling Filters in Wastewater Treatment

Several studies have been reported wherein the TF was employed either as the major biological treatment unit preceded by primary settlers or as a post-treatment unit for a preceding treatment system. Zhang et al. (2016) studied the biofilm characteristics in natural ventilation trickling filters for municipal wastewater. The authors compared the performance of three biofilm carriers; sponge, zeolite and ceramsite. The authors found that in terms of COD removal and ammonium nitrogen removal, the sponge presented the best performance compared to zeolite and ceramsite media, which was attributed to the large specific surface area of the sponge, which allowed for better biomass attachment compared to the rest.

In a related study, Pearce (2004) found that sensitivity analysis revealed mechanisms that enhanced the performance of TFs. The authors reported that for conventional stone media TFs, reducing media size led to a reduction in the effective porosity of the media, resulting in low settling rates and accumulation of trapped solids in the media voids. Contrary to stone media, the oxygen transfer capacity of the plastic media exceeded the biofilm oxygen demand, allowing nitrifying organisms to develop for enhanced nitrification. Other studies have been reported on full and pilot scales wherein the TF was employed as a post-treatment unit for anaerobic effluent, most commonly UASB reactor effluent. Chernicharo (2006) asserted that using TFs as post-treatment to UASB reactors showed remarkable advantages compared to other post-treatment options for anaerobically treated wastewater. According to the authors, besides the reduced operating cost, this combination ensures low sludge production coupled with relative operational and maintenance simplicity. In a different study, Bressani-Ribeiro et al. (2017) researched the resource recovery potential of a UASB reactor coupled with sponge-bed TF for domestic sewage treatment in developing countries. The authors found that the UASB reactor biogas presented a potential energy recovery source due to its rich methane content, whilst the sponge-bed TF effluent presented a nitrogen-rich resource for fertigation purposes. A survey in Brazil revealed that amongst the 333 sewage treatment plants employing UASB reactors coupled with post-treatment units, TFs accounted for 25% of the post-treatment systems employed (Bressani-Ribeiro et al., 2018). The authors again revealed from their study that the replacement of primary settlers by UASB reactors as treatment units preceding TFs had remarkable advantages in terms of construction simplicity and operational requirements associated with sludge handling. They also found that sponge-bed support media had better performance associated with more excellent retention of biomass and longer hydraulic retention times compared to the conventional rock and plastic-bed media TFs. Thus, using TFs as the main biological treatment unit or as a post-treatment unit presents a simple, efficient and cost-effective system for domestic wastewater treatment.

1.8 Carbon Footprints and Sustainability of Wastewater Treatment Plants

1.8.1 Carbon Footprints of Wastewater Treatment Plants

1.8.1.1 Definition of Carbon Footprints

Over the past few decades, the concept of carbon footprint (CF) has gained popularity and is now widely used across the media, governments and the business world. It is used in public debates on responsibility and abatement action against the global threat of climate change (Pandey et al., 2011). Scientists have given various definitions (Table 1.6) to explain carbon footprints. Whilst all these definitions are accurate and similar, the Intergovernmental Panel for Climate Change (IPCC) defines CF as the sum of individual GHGs emitted by an entity. In which methane (CH_4), carbon dioxide (CO_2), and nitrous oxide (N_2O) are expressed as carbon dioxide equivalents (CO_2eq) (IPCC, 2006a). It is a tool to quantify GHG emissions and identify opportunities to reduce climate change impacts. Anthropogenic GHGs have, in recent decades, shown a correlation with the effect of global temperature rise, posing hazards for human and natural systems (IPCC, 2014). Climate change poses numerous consequences, including changes in rainfall patterns, rising average earth's surface temperature, rising sea levels and extreme weather (IPCC, 2013). The risks associated with global warming and climate change led industrial nations to agree to reduce their GHG emissions by signing the Kyoto Protocol (UNFCCC, 1997).

Table 1.6: Reported definitions of carbon footprints in literature

Source	Definition
USEPA (2014)	The total amount of GHGs emitted into the atmosphere each year by a person or organization. A person's CF includes GHG emissions from the fuel they burn directly, such as heating a home or riding a car. It also includes GHGs emitted from producing goods and services, including emissions from power plants, factories and landfills (USEPA, 2014).
Time for Change (2014)	The total amount of GHGs produced to directly or indirectly support human activities. It is usually expressed in ton CO_2eq (Rohrer, 2014).
The Guardian (2010)	The term CF describes the most accurate estimate that can be obtained for the total climate impact of something. Where that "something" could mean anything – an item, activity, lifestyle, organization or even the whole world (Berners-Lee et al., 2010).
Wiedmann (2009)	A "footprint" indicator should encompass all "traces" that an activity leaves behind. For the CF, all GHG emissions associated directly or indirectly with this activity are considered (Wiedmann, 2009).
Grubb and Ellis (2007)	A CF measures the amount of CO_2 emitted by burning fossil fuels. For a business organization, it is the measure of CO_2 emitted directly or indirectly from their everyday activities. It might also reflect the fossil energy in a product reaching the market (Grubb and Ellis, 2007).
Carbon Trust (2007)	A CF measures the total GHG emissions caused directly by a person, organization, product or event (Carbon Trust, 2007).

(Adapted from Nejad, 2020)

1.8.1.2 The Global GHG Emissions and Global Warming Potential

The major GHGs emitted are CH_4, CO_2 and N_2O. Other gases considered by the IPCC include Hydrofluorocarbons (HFCs), Perfluorocarbons (PFCs), Sulphur Hexafluoride (SF_6) and Nitrogen Trifluoride (NF_3). The percentage distribution of GHGs and their contribution per economic sector has been illustrated in Figure 1.9 (a and b). The economic sectors considered are industry, transportation, buildings, agriculture, forestry and other land use, electricity and heat production, and others (IPCC, 2014).

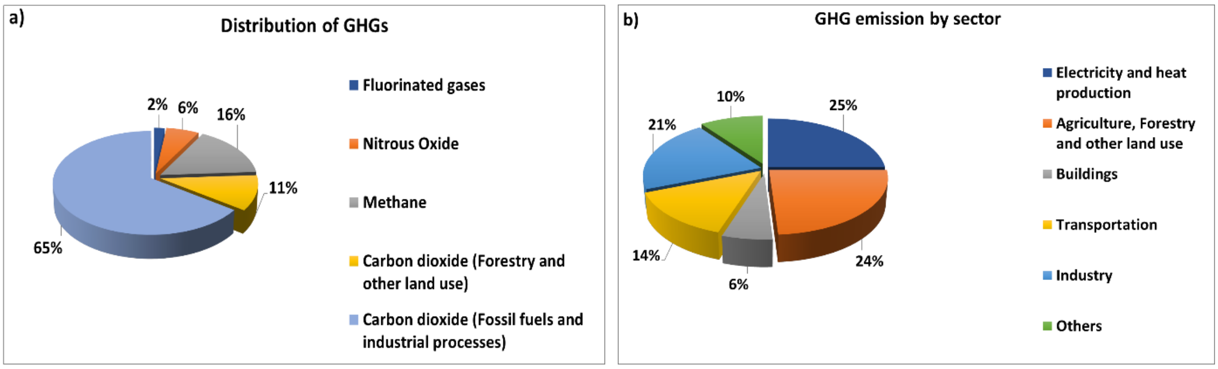


Figure 1.9: GHG emissions percentage distribution by type and source (IPCC, 2014)

The Global Warming Potential (GWP) is the ratio of the radiative forcing of an instantaneous release of 1 kg of a trace substance relative to that of 1 kg of a reference gas (IPCC, 2001). The IPCC has selected CO_2 as the reference gas, having carbon dioxide equivalent unit (CO_2eq). Different gases are reported to have different residence times in the atmosphere. The GWP is generally reported over a 100-year horizon. GWP of various GHGs allows for comparing the impact of the various emissions. The IPCC's fifth assessment report stated that being the reference gas, CO_2 has a GWP of 1 CO_2eq, CH_4 has a GWP of 28 CO_2eq, and N_2O has a GWP of 298 CO_2eq (IPCC, 2006a, 2014).

1.8.1.3 GHG Emissions from WWTPs

Whilst the focus of many studies has been on the GHG emissions from various sectors (Figure 1.9b), WWTPs have been identified to also play a significant role in emitting GHGs. Bogner et al. (2008) stated that globally, the waste sector is responsible for approximately 3% of the total GHG emissions, 90% of which is ascribed to CH_4 emissions from landfills and WWTPs, and equal to approximately 18% of the global anthropogenic CH_4 emissions. In a related study, the USEPA (2012b) reported that the waste sector was responsible for the third most significant contribution of non-CO_2 GHG emissions in 2005, which accounted for 13% of total emissions. A typical WWTP comprises a series of unit processes, including primary treatment, secondary biological treatment, occasional tertiary treatment and sludge treatment units. These units emit some GHGs during their operations. From the least to the most dominant, the three most significant GHGs emitted from WWTPs during the treatment processes are CO_2, CH_4 and N_2O (Chen, 2019). Emissions from WWTPs may be classified as direct or indirect emissions and can be estimated in a CF assessment (IPCC, 2014).

- *Direct Emissions*

Direct GHG emissions include non-biogenic CO_2, CH_4 and N_2O emitted within a system boundary. These are usually on-site emissions that occur directly or are controlled by WWTP processes, such as the biological treatment phase and the combustion of fossil fuels on-site. CO_2 emissions from secondary biological treatment are usually not considered in the direct emissions estimation as they are of a biogenic source. In other words, the produced CO_2 is equivalent to the CO_2 extracted from the atmosphere during photosynthesis; therefore, they form part of the natural carbon cycle (IPCC, 2006a). However, it has been argued that up to 20% of the carbon in wastewater is of fossil origin (Griffiths-Sattenspiel & Wilson, 2009), and fossil CO_2 emissions from wastewater treatment were underestimated, which can be natural or anthropogenic (Law et al., 2013).

- *Indirect Emissions*

Indirect GHG emissions are defined as emissions from the effect of activities within the system boundary but occur outside the defined boundary. These include emissions from effluent, chemical production, external energy production, transport, composting of sludge and landfills. Studies have proven that indirect emissions make up about 60% of the GHG emissions from WWTPs. The final product from sludge treatment; biosolids, can likewise be a source of indirect emissions, just as the emissions from the transportation of sludge. Additionally, indirect N_2O emissions can occur from the discharge of effluent into recipient water bodies (IPCC, 2006a).

1.8.1.4 Methane and Nitrous Oxide Emissions

CH_4 emissions are released in systems where anaerobic conditions exist. Most CH_4 emissions originate from anaerobic reactors and ponds when organic matter is eliminated from wastewater, and sludge handling processes (Chen, 2019). The amount of CH_4 generated depends on the biodegradable substance, temperature, and treatment processes (IPCC, 2006a). Similar to the wastewater treatment process, GHGs are emitted during the composting process. CH_4 is formed in the anaerobic zones of the composting sludge piles (Brown et al., 2008), and N_2O in different degradation steps of the nitrogen compounds (Sánchez-García et al., 2014). Some amount of CH_4 may also be emitted from poorly managed aerobic systems. Practically, most anaerobic WWTPs flare their biogas to neutralise the CH_4 by combustion; however, depending on the flare efficiency, some residual CH_4 may remain in the exhaust gas and get emitted into the atmosphere. Thus, inefficient CH_4 collection systems with leakages and incomplete combustion can still lead to CH_4 emissions (IPCC, 2006a).

Degradation of nitrogenous compounds like urea, proteins and nitrate are major contributors to N_2O emissions. Direct N_2O emissions may be produced in both nitrification and denitrification processes in centralized wastewater treatment systems and indirectly in the recipient water body. Nitrification is an aerobic process that converts NH_3 and other nitrogen compounds into NO_3^-, whereas denitrification occurs without free oxygen and involves the biological conversion of NO_3^- into dinitrogen gas (N_2). N_2O can be an intermediary product from both processes but is often associated with the denitrification process (IPCC, 2006a). Some operating parameters such as pH, dissolved oxygen (DO), nitrite and ammonium concentrations, and environmental conditions such as temperature impact N_2O production in WWTPs. Additionally, the absence of adequate organic substrate (low COD:N ratio) could result in N_2O emissions (Chen, 2019; Massara et al., 2017).

If biological nutrient removal is not prioritised and enforced, excess nitrogenous compounds will continue to pollute the waterways due to the discharge of high-concentration N effluent, but there would be fewer N_2O emissions into the atmosphere. In this way, another environmental impact on recipient water bodies would be inevitable: Eutrophication, which would result in algal blooms and depletion of aquatic oxygen. In addition to this, depending on the type of recipient water body, nutrient-impacted, hypoxic and stagnant water bodies have higher emissions factors for N_2O emissions. Thus, the discharge of N-rich effluent into waterways could either result in eutrophication or off-site N_2O emissions. On the other hand, enhancement of biological nutrient removal would lead to higher N_2O emissions from the wastewater treatment process on-site, but the waterways would be safeguarded against eutrophication and off-site N_2O emissions. The trade-off between N_2O emissions and eutrophication is clear. The challenge is inevitable, how to eliminate GHG emissions and simultaneously minimize ecological effects caused by eutrophication (IPCC, 2019; Nejad, 2020; Xu, 2013).

1.8.2 Sustainability Consideration in Wastewater Treatment Systems

WWTPs play a vital role in returning safe and clean water to the environment; hence they are regarded as part of the broader nexus between energy and water (Gremillion & Avellan, 2016). Each step in the wastewater treatment process requires energy, from the point of collection to the final discharge. To produce high-quality effluent, wastewater treatment processes consume large amounts of energy, primarily electricity. This energy consumption will undoubtedly increase due to rapid population growth and stricter discharge regulations. The higher energy demand will have a negative impact on the global water industry, which is inextricably related to climate change as the electrical energy consumed by the WWTPs is sourced from fossil fuels. This has become a concern for the wastewater industry in its quest to progress towards economic viability and environmental sustainability simultaneously (Stillwell et al., 2010).

Another issue of concern in the wastewater industry is the management of sludge produced during wastewater treatment processes. Several challenges are associated with sludge management, such as the increase in sludge production with associated sludge treatment costs and the risks to human and environmental health. This is because contaminants, including pathogens in wastewater, are concentrated in the sludge. The awareness of sludge-related hazards to human and environmental health has increasingly affected sludge application as fertilizers in the agricultural sector. This has led governments and agencies to outline regulations and policies for safe sludge handling and management practices for sustainable development.

Despite the seeming challenges faced by WWTP managers in their quest to attain sustainability of these systems, biogas, sludge and reclaimed water by-products from WWTPs have the potential to transform the wastewater industry into a more sustainable venture through resource recovery with the employment of circular economy concepts, which would undoubtedly improve the sustainability of these systems.

1.8.2.1 Sustainable Wastewater Management through Resource Recovery under Circular Economy

The growing consensus among global leaders and policymakers on the importance of sustainable development has shifted focus to low-carbon societies and circular economy (CE) as primary economic and environmental ideologies. The CE is *"a regenerative system which minimises resource input and wastage, energy usage, emissions and leakages by slowly narrowing and closing the energy and material loop"* (Geissdoerfer et al., 2017). In simple terms, this implies the focus on reducing waste to a minimum and recovering valuable resources from waste streams generated by social and industrial activities. Recovered resources can further be used in the economy, creating more value for stakeholders. The CE concept has been recognized globally, with governing agencies integrating it into local, national and international policies. The extensive European Circular Economy Action Plan is evident of the increasing importance of this concept (European Commission, 2020).

The function of modern WWTPs as just end-of-life and disposal facilities is being reconsidered by industry experts, with the focus shifting to WWTPs being seen as avenues to recover valuable resources to become centres for a bio-based CE (Andersson et al., 2016; Rodriquez et al., 2020). WWTPs need to be energy-efficient and economically viable (Bachmann, 2015). Maktabifard et al. (2018) mentioned that due to the increasing cost of energy and diminishing available resources, decision-makers and consumers must take greater recognition of the socioeconomic and environmental impacts of their activities. The paradigm shift in wastewater management indicates that WWTPs should be designed and operated with resource optimization and recovery as major objectives besides the primary goal of protecting human and environmental health.

Several concepts have been introduced that consider WWTPs less as end-of-cycle processing facilities and more as centres where water, energy, fuels and nutrients can be recovered from urban wastewater sources. Agudelo-Vera et al. (2012) stated that the “Urban Harvesting Concept” pertaining to urban centres has become more sustainable by closing urban cycles and harvesting resources from their waste streams, thereby reducing their energy and resource consumption. In the Netherlands, the “NEW Factory” concept has been introduced, which suggests WWTPs can become factories for recovering Nutrients, Energy and Water, thereby providing a pictorial view of how sustainable WWTPs will operate in the future (Roeleveld et al., 2010). The biorefinery concept envisages WWTPs as factories (refineries) where the raw materials (wastewater and sludge) are refined to extract and recover several beneficial products, such as energy and chemicals, whilst wastewater treatment remains the primary objective (Amulya et al., 2016; Bertanza et al., 2018). Moreover, there is a growing consensus for WWTPs to be regarded as Wastewater Resource Recovery Facilities (WRRF), where resource recovery will be a primary function of the facility, along with wastewater treatment (Lebrero et al., 2017). The value of recoverable resources varies based on their end uses, with potable water being the most valuable resource that can be recovered during wastewater treatment. Figure 1.10 presents a ladder diagram which expounds the value proposition of various resources based on the cost of recovering the resource.

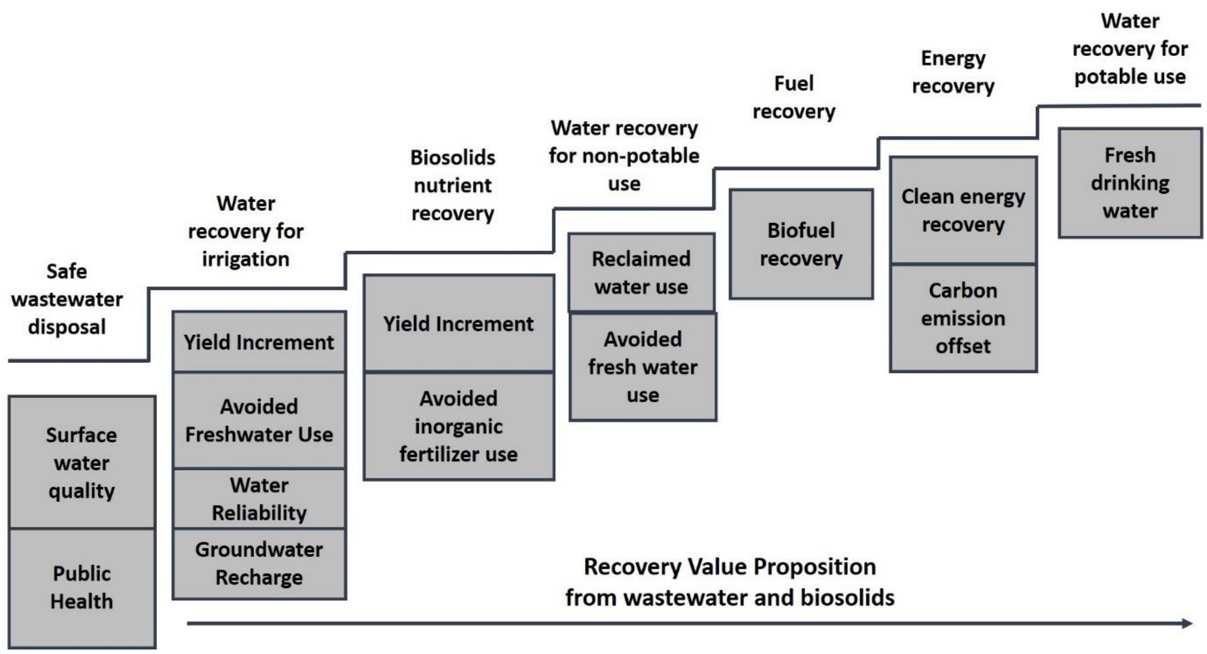


Figure 1.10: Ladder diagram illustrating the increasing value of recoverable resources in WWTPs with increased capital investment and cost recovery potential.

(Adapted and modified from Drechsel et al., 2015)

1.8.2.2 Reclaimed Water Recovery and Use

Wastewater use is gaining recognition in the sustainable agenda globally due to the fact that: i) Foreseen freshwater scarcity resulting from pollution of water bodies and rapid global population growth, which has placed substantial pressure on the limited available freshwater sources (Bhaduri et al., 2016; UN Wastewater Report, 2017); ii) Tarpani & Azapagic (2018) noted that there exist significant gains when reclaimed water usage is prioritized over the traditional means of boosting water supply such as the building of dams or transferring water from one basin to another. Both approaches have significant environmental and economic implications unsuitable for sustainable development in the 21st century. Wastewater treatment and use employ less energy than treatment processes such as water desalination and are hence much more profitable; iii) Governments are beginning to appreciate the "double value proposition" in wastewater use. Recovery and reuse of water, nutrient and energy from wastewater treatment come with environmental and economic benefits, making a double value proposition (Drechsel et al., 2015).

Sustainable wastewater management can help to increase the level of wastewater treatment globally, providing greater volumes of treated wastewater that can replace freshwater withdrawals for agricultural, industrial and sanitation purposes, increasing water use efficiency (Mo & Zhang, 2013; Rodriquez et al., 2020). Several water-stressed cities across the world have realised the value of wastewater use and are utilizing wastewater as water resources; In Namibia, Windhoek city has a pioneer record in recycling wastewater to potable levels, which is expected to offset up to 60% of the water demand (UN Wastewater Report, 2017). Likewise, Lima in Peru employed treated wastewater from Huascar WWTP to create a large urban park in the city centre. This provides essential recreational and social benefits to the residents. As a city with unfertile soil conditions, this has enhanced the soil quality, enabling vegetation growth in a notably dry city (WHO, 2016). Thus, the recovery and use of treated wastewater employ CE principles that close the urban water cycle loop, minimizing waste and promoting sustainable development.

1.8.2.3 Nutrients in Wastewater

It has been estimated that globally about 380 billion m^3 (380 trillion litres) of wastewater is generated annually (Qadir et al., 2019). The average nitrogen concentration in the urban wastewater sample is estimated to be 43.7 mg/L, whilst average wastewater phosphorus and potassium concentrations are estimated at 7.8 mg/L and 16.5 mg/L, respectively. This postulates that on the global scale, wastewater produced annually is embedded with about 16.6 Tg of nitrogen, 3.0 Tg of phosphorus and 6.3 Tg of potassium, collectively summing up to 25.9 Tg of nutrients (IFA, 2017). Meanwhile, globally, the current demand for nitrogen fertilizers stands at 115.5 Tg; 43.8 Tg and 33.6 Tg, respectively, for phosphorus and potassium. Thus, with full recovery, the total nutrients embedded in wastewater can offset 14.4%, 6.8% and 18.6% of the global demand for nitrogen, phosphorus and potassium, respectively, and about 13.4% of the global fertilizer nutrient demand, which stands at 192.9 Tg (IFA, 2017). Considering the potential gains economically on the assumption that nutrients recovered are of the same quality and market acceptance as inorganic industrial fertilizer, recovery of these nutrients from wastewater could generate revenue of USD 13.6 billion globally, of which USD 9.0 billion would be from nitrogen recovery, and USD 2.3 billion each from recovery of phosphorus and potassium (Figure 1.11).

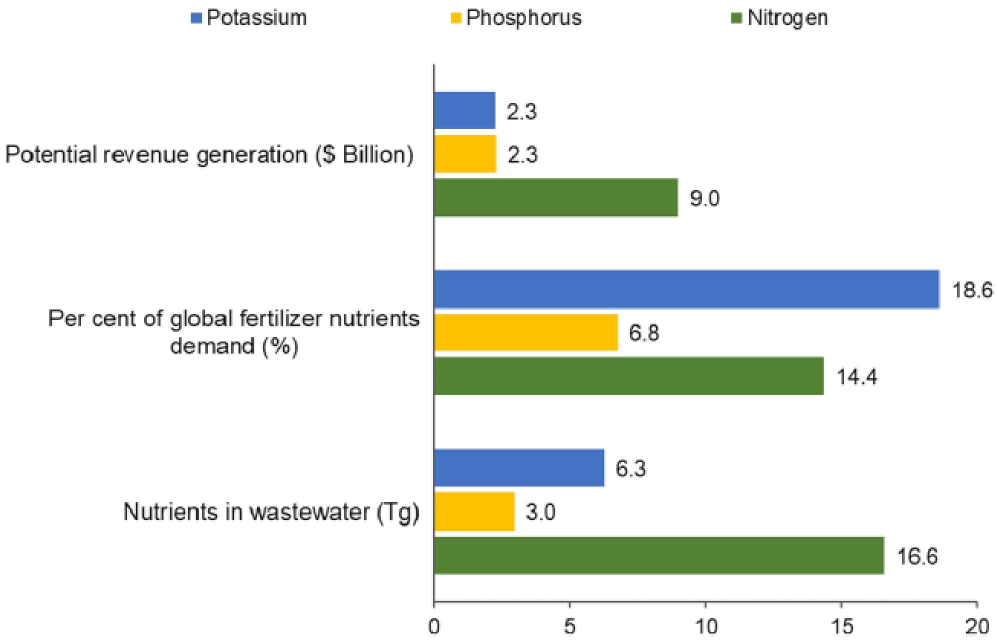


Figure 1.11: Estimation of the global status of nutrients in wastewater

(Adapted from Qadir et al., 2019).

Although these estimations are made on assumption that the maximum theoretical amounts of nutrients present in wastewater generated globally are fully recovered, the current wastewater nutrient recovery technologies are nowhere near reaching 100% efficiency levels. Despite the lower overall nutrient recovery efficiency, significant progress has been made in phosphorus recovery, with recovery rates ranging between 25 and 90%, depending on the recovery processes applied (Egle et al., 2016).

1.8.2.4 Phosphorous Recovery

Recovering phosphorous from wastewater is becoming more of a necessity than an option because phosphorous is a vital nutrient obtained from finite deposits. It has been estimated that the demand for phosphorous will begin to exceed its supply by 2035, creating a global challenge for food production as there exists no substitute for the nutritional value of phosphorous (Cordell et al., 2011). Kroiss (2004) stated that wastewater treatment could provide a viable opportunity to recover phosphorus from waste streams, and this has the potential to offset 15% of the global phosphorus demand. Recovery of phosphorous from WWTPs can be implemented in diverse streams: possible recovery from the untreated wastewater directly, from the sludge generated, effluent from sludge dewatering or the incineration of sludge at the end of treatment (Cordell et al., 2011; Kalavrouziotis, 2017).

Phosphorous recovery as struvite (magnesium ammonium phosphate) by adding magnesium chloride to wastewater streams with high phosphorus concentration is a typical recovery technique in large-scale WWTPs, demonstrated in several full-scale Plants across the world (Otoo et al., 2015). Countries such as Switzerland and Sweden have mandated phosphorous recovery from wastewater treatment. These countries are pioneering what a regulatory framework essential to unlock the global potential for phosphorous recovery could look like (Andersson et al., 2016; Bachmann, 2015).

1.8.2.5 Nutrient Recovery as Biosolids

Wastewater sludge has high concentrations of nutrients and organic matter, which makes it highly efficient for energy and nutrient recovery. Sludge properly treated is referred to as biosolids and can be used for landscaping and agricultural purposes (Mateo-Sagasta et al., 2015). Sludge contains nutrients such as nitrogen and phosphorus; these are essential for plant growth and can be employed as a source of organic fertilizer. Additionally, sludge is rich in organic carbon, which can be utilized to improve soil structure for plant growth when stabilised. Biosolids can also be applied to help improve the physical and chemical qualities of existing soil or to create new soil. The beneficial use of sludge differs from one country to another depending on local regulations and development priorities. Countries with limited arable land prefer mostly to use sludge for agricultural purposes, as seen in Spain, where nearly all biosolids are used for agricultural purposes (Mateo-Sagasta et al., 2015). A survey by the United Nations revealed likewise that in most developing countries, sewage sludge was employed directly on agricultural lands without any adherence to regulations regarding its use (UN-Habitat, 2008a).

Notwithstanding the nutrients in sludge, toxic compounds such as heavy metals and emerging contaminants in biosolids have been a concern. This has led several developed countries to outline strict guidelines and policies regarding sludge usage on agricultural lands. Due to these developments, in countries such as Japan, Germany, the Netherlands and Austria, the use of biosolids for agricultural purposes is limited due to concerns regarding groundwater pollution. Other countries like Brazil, Turkey and Mexico modestly use biosolids for agricultural purposes (UN-Habitat, 2008a).

1.8.2.6 Energy in Wastewater

It has been reported that wastewater contains approximately five times the energy required for the wastewater treatment process (Tarallo, 2015). Wastewater is richly embedded with organic matter, making it a carrier of chemical energy. These organic compounds can be converted into methane-rich biogas through anaerobic digestion. Wastewater also contains thermal energy, which is mostly untapped (Nowak et al., 2015). This thermal energy reserve depends on factors such as flow rate, temperature, the specific heat capacity of the water and heat transfer efficiency (Zhao et al., 2010). Estimations based on the anaerobic conversion of wastewater organic carbon to methane, the calorific value of methane and an assumption that there is full wastewater energy recovery, the energy value in the projected 380 billion m^3 of global annual wastewater generated will be estimated at 53.2 billion m^3 of methane with a calorific value of 1908 billion MJ. Considering that a household's average electrical energy needs are 3350 kWh (World Energy Council, 2016), the 1908 billion MJ of energy embedded in wastewater will be sufficient to provide electricity to 158 million households. These projections, however, are based on the maximum theoretical levels of energy recovery, excluding considerations from technical and economic limitations of wastewater energy processes or the existing energy coverage systems (Qadir et al., 2015).

- *Energy Recovery Potential from Biogas*

Energy recovery from biogas is a growing technology mostly implemented in developed countries due to the high methane component (Chernicharo et al., 2015). Energy production from biogas technologies is predominant in Europe and the United States, although Asian and Latin American countries have also made steady progress in deploying this technology in recent times (Kummamuru, 2017). In the review performed by Scarlat et al. (2018), the authors reported that of about 15,000 WWTPs in the United States as of 2017, 1240 of the Plants operated biodigesters which produced biogas. These biogas Plants were usually installed at large-scale facilities, treating several hundred million gallons of wastewater daily. Some installed biogas Plants employed livestock manure as feedstock. The overall energy potential of biogas in the United States was evaluated to be 18.5 billion m^3 of biogas/yr, of which 8.0 billion m^3 is from the landfill's gas recovery (43.24%), 7.3 billion m^3 from livestock manure (39.46%) and 3.2 billion m^3 from WWTPs (17.29%), producing in all about 41.2 TWh of electricity per year. As reported for the United States, European Union member states are likewise highly advanced in biogas technology, which is attributed to the favourable support schemes set up in these regions, with their major contributors being anaerobic digesters, landfill gas recovery and sewage biogas from wastewater treatment systems (Lisowyj & Wright, 2020; Scarlat et al., 2018).

Despite the growing recognition of sewage biogas energy recovery in the developed world, this phenomenon is not well advanced in developing countries. Patinvoh & Taherzadeh (2019) asserted that biogas energy recovery from wastewater treatment systems had been insufficiently explored in developing countries; hence, this technology needs progression. The same authors reported that inadequate infrastructure and technical services, capital and implementation of appropriate policies are some factors that hinder the successful exploitation of wastewater biogas energy recovery. Lopes et al. (2019) have likewise reported that there exist WWTPs in Brazil that implement biogas energy recovery from sewage treatment; however, the full potentials of these facilities are not sufficiently explored. Like in other developing countries, biogas energy recovery technology is also incipient in Ghana. Several studies; Arthur et al. (2020); Kemausuor et al. (2014), and Präger et al. (2019) have researched the potential of generating renewable energy from biomass (woody biomass, food waste, crop residue, animal manure, municipal waste, etc.) in Ghana, but research on energy recovery from biogas generated from anaerobic wastewater treatment systems is lacking. Thus, this technology has not yet been explored in Ghana.

- *Energy Recovery Potential from Sludge Generated by WWTPS*

Sewage sludge produced by WWTPs comprises a mixture of complex organic and inorganic materials, including microorganisms (Muter et al., 2022). Sludge is rich in nutrients and organic matter, which are valuable for agricultural usage, and also has considerable energy embedded, making it relevant for waste-to-energy technologies (Peccia & Westerhoff, 2015; Syed-Hassan et al., 2017). The huge volumes of sewage sludge and their significant contaminant levels makes it imperative to adopt appropriate handling measures. Conventional methods for nutrient and energy recovery include land application, composting, landfilling, incineration and anaerobic digestion (AD); however, these methods are sometimes hampered by the presence of harmful contaminants such as heavy metals, pathogens, antibiotics and persistent organic pollutants (Clarke & Smith, 2011; Udayanga et al., 2018).

Peccia & Westerhoff (2015) asserted that out of the 7.2 million dry tons of sludge produced within the United States in 2015, 15% was incinerated, 30% was sent to landfills, and 55% was directly applied to land. Similarly, it has been reported that the final destinations of sewage sludge generated from the 27 European Union member countries were: agricultural usage (42%), incineration (27%), sanitary landfills (14%) and for other uses (17%) (Rosa et al., 2018). Whilst studies recounting sewage sludge management in most developing countries are grossly lacking, the majority of prevailing conditions have not been documented. A survey report presented by the UN-Habitat (2008a) on sewage sludge management revealed that for the few developing countries (in sub-Saharan Africa) included in the survey, sewage sludge was spread mainly on agricultural lands.

In recent times, the application of thermochemical conversion technologies such as gasification, pyrolysis, combustion, hydrothermal liquefaction, and other supercritical methods for sludge management is gaining momentum in several developed countries (Chun et al., 2011; Manara & Zabaniotou, 2012). Although these technologies involve complex processes and equipment compared to conventional methods, they exhibit higher efficiency and economic performance (Mulchandani & Westerhoff, 2016). Moreover, they provide some added advantages, such as effective pathogen destruction and a significant reduction in sludge volume, besides their effectiveness for energy recovery (Jiang et al., 2016; Syed-Hassan et al., 2017).

Several studies have reported sewage sludge's energy recovery potential through anaerobic digestion or thermochemical processes in the literature (Gu et al., 2017; Lopes et al., 2019; Singh et al., 2020). Singh et al. (2020) for instance reported that in India, the average energy recovery potential of sewage sludge via incineration ranged from 555 - 1068 kWh/tonne of dry sludge, whilst that for an anaerobically digested sludge was between 315 and 608 kWh/tonne of dry sludge for various wastewater treatment technologies such as sequencing batch reactors (SBR), activated sludge (AS) processes, moving bed biofilm reactor (MBBR), and UASB reactors. A comparative study found that dry sludge could generate 1400 - 1700 kWh/tonne via incineration and approximately 1400 kWh/tonne for AD (Karagiannidis et al., 2012). Silvestre et al. (2015) likewise opined that about 67% of wastewater energy consumption is transferred to sludge management, and subsequently, 52% of this energy could be recovered through AD of the sludge. The recovered energy, the authors reported, could be used for combined heat and power generation.

- *Biogas and Sludge Energy Recovery towards Energy Self-sufficiency of WWTPs*

Several studies have reported on the energy self-sufficiency of WWTPs through energy recovery from biogas and sludge, which among other things, promote the sustainability of these systems, with the majority of such Plants situated in European and North American countries such as the USA, Austria and Germany etc. (Gu et al., 2017; Maktabifard et al., 2018; Nowak et al., 2011; Shen et al., 2015). Sarpong & Gude (2020) mentioned that aerobic systems such as the conventional AS systems could progressively be transformed into energy-self-sufficient WWTPs in the near future through the integration of schemes such as co-digestion, carbon capture and the gradual replacement of AS process with less energy-intensive biological treatment units.

Energy recovery from biogas and sludge in developing countries has been progressing recently, with the ultimate goal for these WWTPs to become energy self-sufficient (Gu et al., 2017). In a study conducted in India, wherein energy was recovered from sewage sludge through incineration and AD, the authors reported that for an AS system, the theoretical energy recovery from sewage sludge through AD was estimated to be ≈185 kWh/MLD (estimation from 2 Plants); meanwhile the energy demand of such Plants is ≈170 kWh/MLD (estimation from 5 Plants). Thus, there is a potential to offset the total energy demand for these Plants to become energy-neutral (Singh et al., 2020). In a similar study, Guo et al. (2019) opined that green-energy technology integration could help to attain 84 - 100% energy self-efficiency of municipal WWTPs. Jangid & Gupta (2014) likewise stated in their study that 75 - 80% of the Plant's energy demand could be generated by AD of the sludge. Again, Tran et al. (2021) researched the feasibility of recovering energy from sewage sludge through co-digestion in Vietnam. Lopes et al. (2019) also evaluated the energy recovery potential of biogas and sludge for 239 WWTPs in Brazil, among which 182 were UASB reactors. These studies all provided positive results regarding energy recovery from biogas and sludge from WWTPs, looking forward to attaining energy-self-sufficient WWTPs to achieve sustainable wastewater management. According to Bernal et al. (2017), energy recovery from biogas and sludge by-products allows diversification of a country's energy matrix whilst reducing fossil fuel consumption and GHG emissions. Besides the prospective economic gains, biogas and sludge for energy recovery could enhance the social and environmental benefits of WWTPs (Larsson et al., 2016; Lindkvist & Karlsson, 2018; Rosa et al., 2018).

Although recovery of energy from biogas and sludge produced by WWTPs has the potential to contribute to the energy self-sufficiency of the Plant, it is recommended that further studies be conducted to evaluate the OPEX, CAPEX, and revenue flows for the energy recovery systems. This is to assess better the economic feasibility of these systems, which may vary depending on several factors such as the location of the WWTP, distance from the resource recovery facility, the Plant size and the population being served etc. (Rosa et al., 2018).

1.9 Sustainable Wastewater Management for Sustainable Development

The SDGs established by the United Nations enabled the coordination of global efforts towards ensuring the long-term development and well-being of the planet and its people, whilst providing a set of goals that can certify that this development is socially, economically and environmentally sustainable. Resource recovery through sustainable wastewater management thus provides an exceptional opportunity within the water-energy-food nexus to contribute to the progress towards global sustainable development (Mo & Zhang, 2013; UN-Water, 2015). The Stockholm Resilience Centre (SRC) has proposed a framework structure in which the 17 SDGs established by the United Nations have been classified under three main sections of sustainability: Economy, Society and Biosphere (natural environment), where Economy and Society are embedded in the Biosphere (Figure 1.12). The structure reveals the interdependence and hierarchy amongst the various development sectors, with the biosphere forming the base for human societal and economic activities (Stockholm Resilience Centre, 2016). Giannetti et al. (2020) and Gupta (2020) deftly illustrated how cleaner energy production significantly contributed to the attainment of the SDGs employing the SRC model. As wastewater forms part of the natural resource cycle in the biosphere, sustainable wastewater treatment with resource recovery while closing energy and nutrient loops has a cascading effect on almost all levels of development. Therefore, understanding the correlation and contribution of sustainable wastewater management with resource recovery towards the SDGs is imperative.

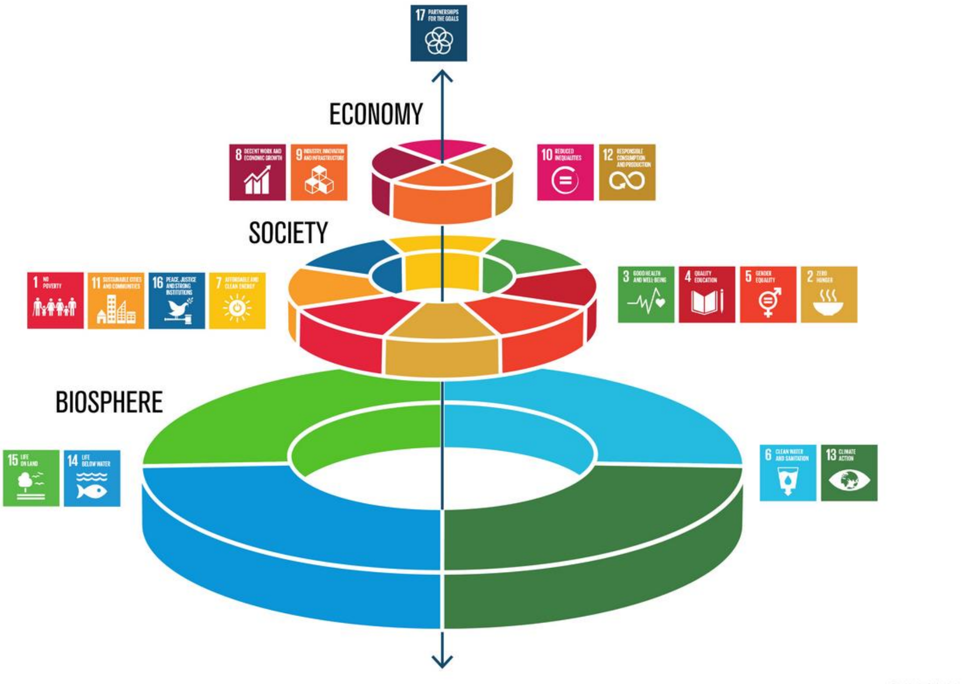


Figure 1.12: Grouping of SDGs presented by Stockholm Resilience Centre

(Adapted from Giannetti et al., 2020)

1.9.1 Contributions to Biosphere-Level SDGs

At the Biosphere level, the SDGs' primary concern is to ensure the preservation of the natural environment and its resources and safeguard the availability and fair access to clean water and land resources globally. To support these goals, sustainable wastewater management with resource recovery can contribute to achieving all 4 SDGs (6, 13, 14, and 15) at this level. The employment of advanced technologies such as anaerobic digestion, membrane filtration and nutrient removal in wastewater treatment contributes directly to clean water and sanitation – SDG 6, and life below water – SDG 14 goals as untreated wastewater is no more discharged into water bodies to threaten water purity and endanger aquatic lives, whilst improving sanitation (UNEP, 2015). The availability of reclaimed water for irrigation of agricultural lands will likewise assuage the need for freshwater withdrawals, relieving pressure on the already stressed freshwater resources. Some studies have also reported that effective nutrient removal during wastewater treatment helps to eliminate eutrophication in recipient water bodies; thus, adopting these technologies will significantly enhance the quality of waterbodies receiving wastewater effluent (Garrido-Baserba et al., 2016). Thus, even if the reclaimed water is not utilized for agricultural purposes, effective pollutant removal during wastewater treatment will safeguard recipient water bodies.

Furthermore, advanced sludge treatment, such as anaerobic digestion and thermochemical processes, prevents the discharge of sludge to landfills, water bodies etc., thereby eliminating methane emissions into the atmosphere. This reduces GHG effects, contributing to the climate action goal of SDG 13 (Demirbas et al., 2016; Hultman & Levlin, 1998). Towards the attainment of the life on land goal – SDG 15, the replacement of inorganic mineral fertilizers with organic biosolids produced as by-products from sludge treatment during the wastewater treatment process has benefits such as improving soil quality by providing an organic and nutrient-rich medium and structure to support plant growth (UN-Habitat, 2008a). Additionally, phosphorous recovery from wastewater can be a viable source of fertilizer, reducing the amount of minerals to be mined to obtain this resource (Andersson et al., 2016; Drechsel et al., 2015). Thus, a strong relationship exists between sustainable wastewater management with resource recovery and the Biosphere-level SDGs.

1.9.2 Contributions to Society-Level SDGs

At the Society level, the SDGs relate to the sustainable growth of developing countries with concerns of achieving universally acceptable levels of education – SDG 4, good health and well-being – SDG 3, peace, justice and strong institutions – SDG 16. The goals are diversified at the Society level, making it challenging to link sustainable wastewater management and these SDGs directly. Notwithstanding, many indirect and cascading benefits can be observed. The goals on zero hunger, no poverty, good health, quality education and gender equality are associated with creating a peaceful society where everyone benefits equally from sustainable development. Sustainable wastewater management with resource recovery provides direct and indirect benefits that can contribute towards attaining these goals.

The goal on no poverty – SDG 1, is indirectly supported by building wastewater treatment plants and resource recovery facilities across countries, especially in the developing world. The new treatment plants and recovery facilities would require human resources to operate, thereby providing employment. Biosolids recovery improves agricultural yields for farmers, thereby improving their livelihood sources, which will help eliminate poverty (Andersson et al., 2016; Hagman & Eklund, 2016). The zero-hunger goal – SDG 2 is equally influenced by reclaimed water and biosolids usage as nutrients for agricultural activities, thereby boosting food production (Hernández-Sancho et al., 2010; Mateo-Sagasta et al., 2015). The target on good health and well-being – SDG 3 is indirectly linked to several factors relating to sustainable wastewater management: Adequate provision of wastewater treatment facilities eliminates indiscriminate wastewater disposal practices which leads to the spread of water-borne diseases; improved wastewater treatment eliminates pathogens in effluent, improving downstream water quality (IWA, 2018). The use of organic biosolids helps to replace toxic mineral fertilizers (Otoo et al., 2015). Additionally, the use of clean recovered energy instead of fossil-based energy reduces GHG emissions, positively influencing climate and the negative cascading impacts of climate change on humanity (Molinos-Senante et al., 2014). The goal on good health and wellbeing (SDG 3) extends to SDG 4 – quality education, in that the quality of education is improved when teachers and students have good health. Moreover, improved livelihoods permit the populace to afford quality health care and education (UN Wastewater Report, 2017).

The goal on affordable and clean energy – SDG 7 is directly related to sustainable wastewater management in the following ways: the recovery of energy in the form of electricity from biogas and sewage sludge can be used on-site to offset the Plant's energy needs, thereby eliminating the use of fossil-based electrical energy sources. When produced in large quantities by energy-positive WWTPs, this energy can be used to augment national grid energy, and be supplied to households, communities and organisations (Dos Santos et al., 2016). Furthermore, methane from biogas can be used as domestic fuel to replace charcoal (whose production leads to uncontrolled cutting down of trees), firewood and kerosine. Moreover, biochar produced from sewage sludge has a high heating value, and could reduce the dependence on unsustainable fuel sources for domestic use, providing access to clean energy in energy-stressed communities, improving quality of life and providing economic opportunities (Racek et al., 2019). As a sustainable alternative for cooking fuel, clean energy usage comes along with several benefits, such as the improvement of indoor air quality due to cleaner combustion, whilst helping women and children save a considerable amount of time in searching and collecting firewood, this time savings can be used for other productive activities (Tilley et al., 2014). Lastly, an indirect contribution to SDG 7 is the replacement of energy-intensive inorganic mineral fertilizers with high-quality organic biosolids from WWTPs (Robles et al., 2020).

The direct and indirect benefits cited above will eventually lead to the attainment of the target on sustainable cities and communities – SDG 11. The application of sustainable and integrated water resource management with resource recovery, use of clean recovered energy, agricultural use of biosolids and the overall impact of the wide deployment of sustainable WWTPs in urban and rural centres will ultimately help in creating resilient and sustainable communities

(Agudelo-Vera et al., 2012; Weitz et al., 2014). The final target at the society level; peace, justice and strong institutions – SDG 16 is supported by factors such as energy independence, better transnational water relations and rural-urban area development opportunities. Most of the world's conflicts today are related to resource scarcity; meanwhile, sustainable wastewater management with resource recovery can help alleviate these scarcities by providing solutions to water, energy and food challenges within the water-energy-food nexus (UN Wastewater Report, 2017).

1.9.3 Contributions to Economy-Level SDGs

The SDGs contributing to the economic level include ensuring decent work and economic growth – SDG 8; industrial innovations and resilient infrastructure – SDG 9; reduced inequalities – SDG 10; and responsible production and consumption practices – SDG 12 (Stockholm Resilience Centre, 2016). Wastewater treatment and resource recovery are relevant industrial processes and part of the urban economy; it is, therefore, vital to understand the relationship between sustainable wastewater management with resource recovery and the attainment of the Economy-level SDGs (Gupta, 2020).

Sustainable wastewater management has several indirect benefits towards attaining the economy-level SDGs. The readily available, cheap, high-quality organic digestate is essential to farmers. Using organic fertilizers can improve the quality and quantity of their produce, reducing dependence on increasingly expensive and harmful inorganic fertilizers, and use the economic savings to boost their income sustainably (Mateo-Sagasta et al., 2015). The production of organic biosolids also presents an important revenue stream if sold on commercial scales, thereby improving the cost-benefit ratio for the producers. For urban water management, it is imperative that the growing urban economy is governed by responsible use of natural resources. Thus, as water demand grows, it will become increasingly important that a fair share of that demand is recovered from wastewater treatment and use (IWA, 2018; Kiselev et al., 2019). Building WWTPs across urban centres will create new industries and jobs and promote economic growth by providing essential utility services with favourable environmental benefits (UN World Water Assessment Programme, 2015). The establishment of all these interventions will help reduce inequalities, promoting equal opportunities and partnerships for sustainable growth. Thus, sustainable wastewater management is of special interest in this regard as WWTPs will take up new roles as producers of materials, resources, energy, and not just centres for wastewater treatment, promoting sustainable development, making the planet a better place for humanity.

Chapter 2:

Performance Evaluation of a Full-scale UASB Reactor coupled with Trickling Filters Treating Municipal Wastewater in Accra, Ghana

Abstract: Poor wastewater management in most emerging economies in sub-Saharan Africa remains a critical health and environmental challenge due to inadequate infrastructure for the collection and treatment of wastewater. This study assessed the performance of a full-scale upflow anaerobic sludge blanket (UASB) reactor having a capacity of 18,000 m^3/d, with trickling filters and clarifiers as post-treatment units, treating municipal wastewater in Accra, Ghana, as well as the methane production rate. Data was collected on Plant operational conditions and the physical and chemical parameters of wastewater (influent and effluent) over 52 weeks in 2021 (from January to December). The ratio of biochemical oxygen demand to chemical oxygen demand (BOD:COD) of influent water was found to be 0.6 ± 0.2, which indicated the presence of readily biodegradable compounds in the sewage. UASB reactors' conditions of operation were observed to be within the optimal range reported for anaerobic systems, with an applied organic loading rate of 1.22 ± 0.71 kgCOD/m^3/d. System performance was generally satisfactory, with the removal of organics at 86% for COD and 97% for BOD. Biogas yield was 0.14 ± 0.07 m^3/kgCOD removed, resulting in an average biogas production rate of 613 ± 271 Nm^3/d. The average methane proportion was 65% of the biogas output, with an estimated 23% of the methane generated dissolved in the effluent. SMA test likewise revealed that an ISR of 1:1 resulted in the highest biogas yield and corresponding SMA value. The UASB reactor presents an efficient technology that can be applied for effective and sustainable wastewater management in developing countries.

Keywords: Anaerobic digestion; Biogas production; Municipal wastewater; Post-treatment units; Removal efficiency; Trickling filter; UASB reactor.

2.1 Introduction

Wastewater management remains one significant challenge faced by most countries in sub-Saharan Africa (SSA) (UN Wastewater Report, 2017). Burgeoning population growth, coupled with urbanisation and industrialisation has led to the production of huge volumes of wastewater, which is often discharged haphazardly into the environment due to the lack of adequate infrastructure for wastewater collection and treatment (UN Wastewater Report, 2017). Untreated wastewater, however, contains pathogens and contaminants that can harm the health of humans and the receiving ecosystems (UN-Habitat, 2008b). Notwithstanding threats from untreated wastewater, it is richly embedded with organic matter and nutrients that could be harnessed as valuable resources for energy recovery in biogas, plant nutrients in compost/fertilizer and water that can be reused for irrigation (Jeppsson & Hellström, 2002). The recovery of these resources is essential for sustainable wastewater treatment, especially under modern concepts of circular economy and eco-friendly technologies (Dionisi et al., 2018).

Despite their reliable treatment capacity and effluent quality, conventional wastewater treatment systems based on activated sludge processes that are typically used in developed countries are usually unsuitable for developing countries due to such factors as the high cost of installation and operation (Martinez-Sosa et al., 2012). Biological wastewater treatment with anaerobic digestion (AD) may seem an appropriate alternative due to the reduced or zero energy consumption, simplicity in operation, and ability to treat high-strength organic wastewater (Van Lier et al., 2008). In addition, anaerobic wastewater treatment (AnWT) comes with additional advantages such as suitability for warm tropical climates, energy recovery from methane, recovery of biofertilizers, and less sludge production compared to aerobic processes. These advantages make AnWT systems efficient and economically viable technologies that can be implemented in emerging economies for efficient and sustainable management of wastewater (Van Lier et al., 2008). AnWT technologies frequently implemented include UASB reactors, rotating biological contactors, anaerobic filters, expanded granular sludge beds, fluidised bed reactors, and waste stabilisation ponds (WSPs).

Mara (2003) found that WSPs are among the AnWT technologies most implemented in the developing world. This finding is consistent with the study by Murray & Drechsel (2011) who reported that in Ghana, WSPs are the most adopted wastewater treatment technology (42%), followed by activated sludge systems which make up 26%, whereas anaerobic digesters make up 16%. Other technologies, such as aeration tanks, trickling filters, aeration tanks, granular activated carbon and sedimentation tanks are rarely implemented. Furthermore, it has been reported that only 5% of Ghana's population is connected to sewer networks. On-site sanitation systems such as pit latrines and septic tanks remain common in Ghana. When full, the vaults are emptied and conveyed to WSPs facilities for discharge or disposed of indiscriminately into the environment (DCE-KNUST, 2016; Iwugo, 1981; Sagoe et al., 2019). Notwithstanding the economic viability of WSPs for developing countries, there are challenges associated with these systems, including large land area requirements, long HRTs, problems with odour, and contributing to greenhouse gas (GHG) emissions (Coggins et al., 2019). These challenges have made the adoption of new sustainable technologies imperative for emerging economies.

Among the many existing AnWT technologies, the upflow anaerobic sludge blanket (UASB) reactor has received much recognition, with several pilots- and full-scale Plants in operation in countries like India, Brazil, Columbia and Japan (Lettinga, 2005; Passos et al., 2020). The UASB reactor technology has a number of advantages over other AnWT technologies, accounting for its popularity in several regions of the world, despite being in existence for a relatively short period compared to other anaerobic technologies (Chernicharo et al., 2015). First is the systems' capability to handle high and fluctuating organic loadings (Leitão, 2004). In their studies, Wolmarans & De Villiers (2002) and Musa et al. (2019) found that full-scale UASB reactors attained 90% removal efficiency for high-organic load sewage of ≈30,000 mg/L COD. Hulshoff Pol et al. (2004) similarly observed that the most significant technology feature that enabled UASB reactors to handle higher volumetric loading than other anaerobic systems was the development of the biological "granules" in the sludge blanket. UASB reactors additionally produce lesser and more stabilised sludge compared to aerobic systems. Biogas generated from these reactors contains a significant portion of methane gas that can be harnessed for energy recovery (Foresti et al., 2006).

Despite the numerous advantages associated with UASB reactors, one key setback with their application is their inability to produce high-quality effluent that meets World Health Organization (WHO) standards for wastewater reuse or discharge. Thus, UASB reactor-treated effluent generally requires post-treatment to guarantee the final effluent meets regulatory standards. Post-treatment units have the primary objective of eliminating residual organic matter, along with the elements that are rarely affected by anaerobic treatment processes:

phosphorus- and nitrogen-containing compounds (plant nutrients), and pathogens, enhancing reclaimed water quality (Daud et al., 2018; Foresti et al., 2006).

As was observed by Bressani-Ribeiro et al. (2018), aerobic post-treatment units usually follow AnWT systems, which allows the attainment of effluent complying with discharge limits. Among the various technologies (activated sludge processes, polishing ponds, and anaerobic filters etc.) commonly adopted as post-treatment units, the UASB/Trickling filter (TF) configuration is dominating in many countries, especially in Brazil (Chernicharo et al., 2018). TFs are intended as non-submerged aerobic biofilm reactors (Rittman & McCarty, 2001), which consist primarily of a basin filled with very-permeable materials, on which the sewage is applied using a distribution system. As the sewage trickles downward, bacteria (biofilm) grow on the surface of the permeable materials (Chernicharo, 2006). Other authors believe that the performance stability and operational simplicity of TFs are major reasons for their worldwide application, especially in developing countries (Bressani-Ribeiro et al., 2018; Chernicharo, 2006).

Notwithstanding the wide application of the UASB reactor technology in several parts of the world (Chernicharo et al., 2015), its application in the West African sub-region is minimal, despite the favourable climatic conditions and economic viability for developing countries in the sub-region. Only a few studies (Ahmed et al., 2018; Awuah & Abrokwa, 2008) have been conducted on full-scale UASB-based WWTPs in the sub-region. However, these studies only reported systems' performance, with information on operating conditions scantly provided. Moreover, critical characteristics of the influent such as the ratio of VFA/Alkalinity in the UASB reactors, the influent's nutrients ratio, and concentrations of heavy metals have not been reported. Assessment of these parameters is crucial as they can obstruct optimum system performance when in inappropriate concentrations or proportions (Chen et al., 2008; Martin-Ryals, 2012). Finally, studies on biogas production and composition for full-scale UASB-based Plants within the sub-region have not yet been reported to permit the valuation of the energy recovery potential from methane gas for such anaerobic systems. This study seeks, therefore, to fill these scientific gaps regarding the application of the UASB reactor technology in the West African sub-region. It investigates the performance of full-scale UASB reactors in combination with trickling filters (TFs) and final settling tanks (FSTs) as post-treatment units over 52 weeks of continuous operation for municipal wastewater treatment. Analysis of critical operational parameters was also undertaken. The study, additionally, quantifies and characterises biogas generated by the UASB reactors for their use in energy recovery. Assessing the Plant will allow for proper comparison with previous studies conducted in different climatic regions to evaluate the efficiency and biogas production potential. The plant under study operates under tropical climatic conditions and will establish the proposition that the UASB reactor is most favourable in the tropical climate. Moreover, this knowledge will inform policymakers on the viability of this technology for replication in other developing countries in SSA as a substitute for the WSPs systems predominant in these regions and ameliorate the wastewater management hazard in developing countries.

2.2 Materials and Methods

2.2.1 Description of Study Area and Climatic Conditions

The study was carried out at the Mudor Wastewater Treatment Plant (Mudor WWTP) in Jamestown, Accra (5˚36′53.3448″N, 0˚12′21.1464″W), the capital city of Ghana in the West African sub-region. The Accra Metropolitan District is one of the Districts within the Greater Accra Region of Ghana. Accra is believed to be one of the most populous cities in Africa and is now the most populated city in Ghana, with an estimated population of 5,446,237 (GSS, 2021). Situated on the southern coast of the Gulf of Guinea and spans over a land area of approximately 60 km^2 (Figure 2.1).

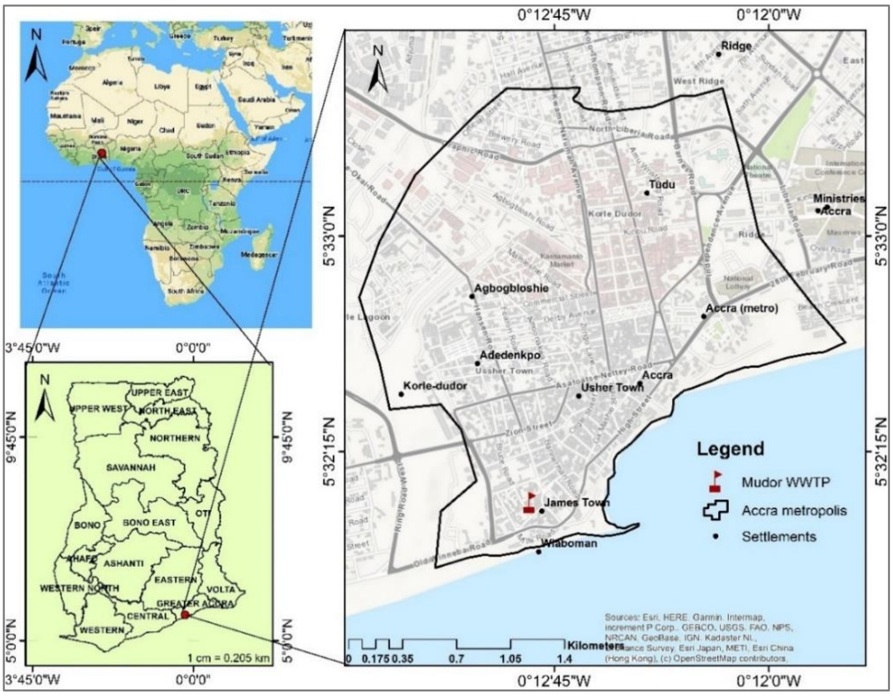


Figure 2.1: Map indicating the location of the Mudor WWTP

Accra has two major seasons each year; May to October is a wet season, whilst from mid-November to April is a dry season (Sagoe et al., 2019). Daily ambient temperature generally ranges between 20.8 ^{o}C and 35.6 ^{o}C, with a mean annual temperature of 29 ^{o}C (GMA, 2021). Figure 2.2 presents the climatic conditions of Accra during the study period. Jamestown, located eastward of the Korle Lagoon, is one of the oldest towns in Accra, and where the Mudor WWTP is situated.

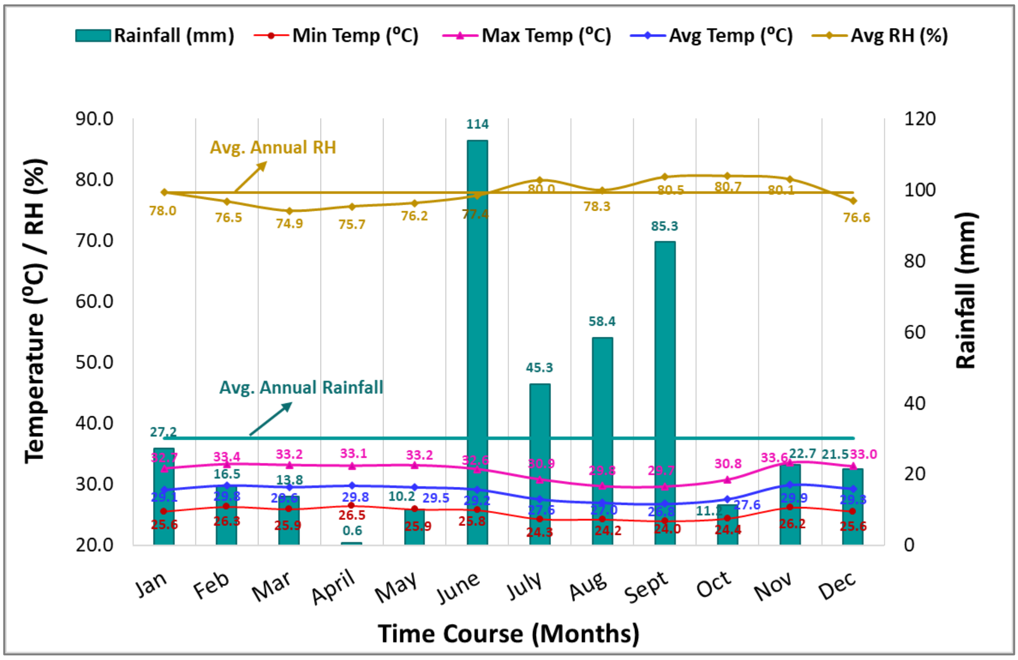


Figure 2.2: Climate condition of Accra during the study period

2.2.2 Description of Wastewater Treatment Plant

Ahmed et al. (2018) reported that the Plant was constructed in the year 2000 and operated for a few years, after which it was shut down due to poor maintenance practices and financial commitment. It, however, became operational after major rehabilitation and expansion works were carried out in 2017. Covering a total area of 6.3 acres of land, the Plant which is sited 20 m east of the Korle Lagoon in James Town, receives and treats sewage from commercial centres, offices and households, within Accra Central, Korle-Bu, Osu-Labone, Ministries, parts of Dansoman, and High-street suburbs; all sewered communities. Currently, the Mudor WWTP is estimated to serve approximately 100,000 inhabitants, based on projections from 60,000 inhabitants served at the time of construction (ASIP, 2005), and a population growth rate of

2.1% per annum (GSS, 2021). The Plant was not known to receive industrial discharges at the time of this study.

The Mudor WWTP comprises six (6) full-scale UASB reactors, with three (3) TFs and two (2) clarifiers as post-treatment units. The modular-shaped UASB reactors operate parallelly, with 16,000 - 18,000 m^3/d capacity. The circular-shaped TFs and clarifiers equally operate in parallel. The synoptic view of the Plant is presented in Figure 2.3. Sewage flow to the WWTP ranges from 1572 to 6054 m^3/d, with a mean flow of 4096 ± 837 m^3/d. The average volumetric OLR for the UASB reactors was found to be 0.77 ± 0.49 kgBOD$_5$/m^3/d (157,392.06 PE), based on the estimation of per capita BOD contribution of 0.04 kg/cap/d (Mara, 2003) for developing countries. Table 2.1 describes the dimensions of the various treatment units of the Mudor WWTP.

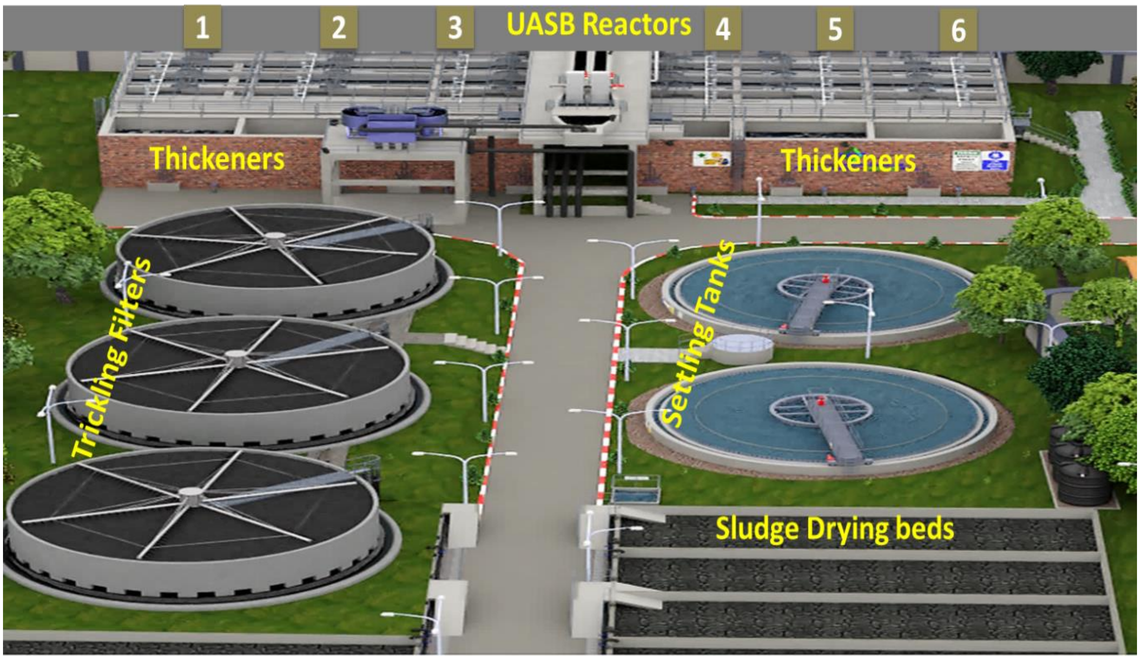


Figure 2.3: Synoptic view of the Mudor WWTP

Table 2.1: Dimensions of treatment units at Mudor WWTP

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

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

The sewage received at the Plant is typically low-strength municipal sewage (Ahmed et al., 2018), which first undergoes preliminary treatment by passing through coarse screens (20 mm mesh apertures), where more extensive solid waste materials are trapped. The sewage then moves into a wet well and is pumped to the sand/grit removal system (vortex grit). Subsequently, the sewage flows through a fine screen unit with 5 mm-mesh apertures, where further sieving takes place before entering the reactors. The UASB reactors' effluent flows to the TFs for further biological treatment and the last step before being discharged into the Korle Lagoon is the secondary clarifiers (FSTs). Excess sludge is periodically withdrawn from the UASB reactors into the thickeners for physical sludge dewatering. The thickened sludge is now pumped onto the sludge drying beds to be air-dried and processed further whilst the supernatant is redirected into the wet well to mix with incoming sewage. Biogas generated in the UASB reactors is trapped in the gas hoods and directed to a flaring unit. Process flow at the Mudor WWTP is illustrated in Figure 2.4. The Mudor Plant was designed such that most material flow is gravity-driven, thereby minimising pumping and reducing electricity consumption with the associated costs.

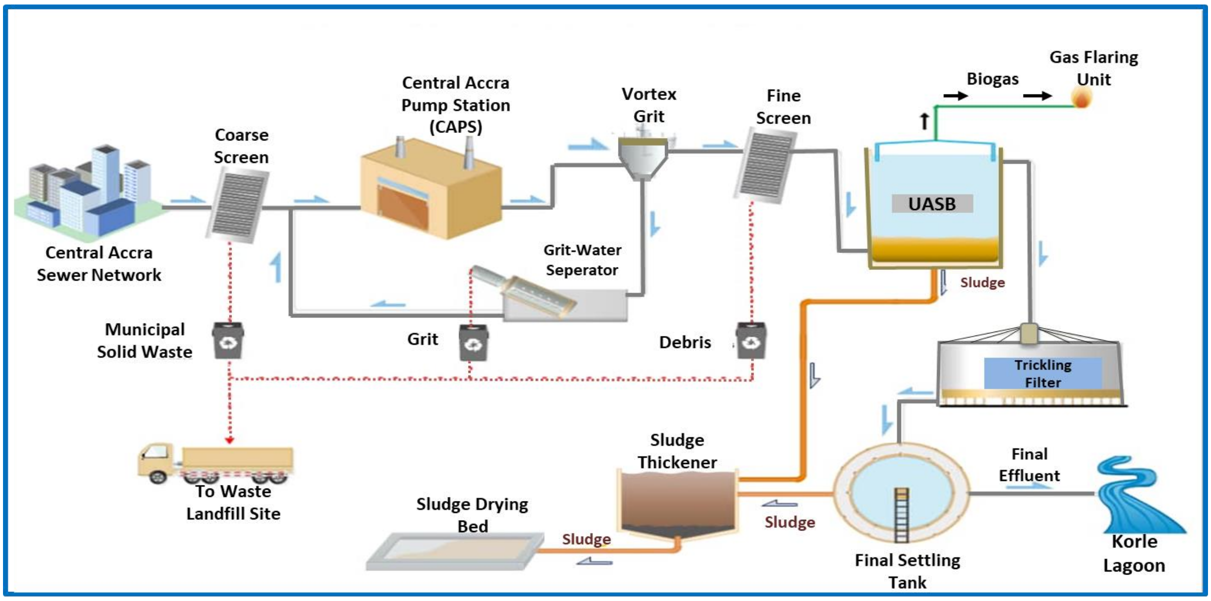


Figure 2.4: Process flow of the Mudor WWTP

2.2.3 Sampling and Analytical Methods

System efficiency was monitored over 52 weeks by analysing composite samples from the various sampling units: these include raw sewage after grit removal and effluent from the UASB reactors, TFs, and FSTs. The temperature, pH, electrical conductivity (EC), dissolved oxygen (DO) and total dissolved solids (TDS) were measured in situ with a portable multi-probe analyser (HQ40D LDO10101, HACH), while the remaining parameters were analysed at the laboratory of Sewerage Systems Ghana Limited (SSGL), situated at the Plant's premises. Clean sampling bottles (1 litre) were used, with sampling carried out diligently to avoid external contamination. Wastewater was sampled semi-weekly for organic components and weekly for nutrient, microbial, and heavy metal analyses. Samples were conveyed from the site in an ice-filled insulated chest box to the laboratory within 24 hours for analysis or stored in a refrigerator at 4 ^oC where applicable.

The flows of biogas and sewage were measured with automatic flow measuring devices installed in the Plant; Prosonic Flow B (Endress+Hauser, Switzerland) and PROMAG 50 (Endress+Hauser, 50W1F-HLGA1RK5BAAA, Switzerland) meters, respectively. Biogas was measured at STP employing the method proposed by Strömberg et al. (2014). In addition, biogas was sampled from each of the UASB reactors and characterised over ten (10) weeks (July 02 to September 15, 2021). Biogas sampling was conducted by connecting Tedlar sacs (1 litre) to the gas sampling points sited on top of the reactors' gas hoods and transported carefully to the Institute of Industrial Research (IIR) laboratory. Biogas constituents, namely carbon dioxide (CO_2), methane (CH_4), nitrogen (N_2), oxygen (O_2), and hydrogen sulphide (H_2S), were analysed with a potable FM 406 Gas Analyser (Gas Data, UK) at the IIR laboratory. Biochemical oxygen demand (BOD_5) analysis was by test method APHA 5210, whilst chemical oxygen demand (COD) was measured employing the potassium dichromate digestion method. Total solids (TS) and total suspended solids (TSS) were determined by oven drying at 105 ^oC, whereas total volatile solids (TVS) were determined by furnace ignition at 550 ^oC. Volatile fatty acids (VFA) and alkalinity were by distillation and Lovibond methods, respectively. Nutrients: total nitrogen (TN), nitrate nitrogen (NO_3^--N), ammonia nitrogen (NH_3-N), orthophosphate (PO_4^{3-}-P), total phosphorus (TP), sulphide (S^{2-}) and sulphate (SO_4^{2-}) were measured with HACH DR 3900 spectrophotometer. For microbial analysis, faecal coliform (FC), *E. coli*, and Salmonella *sp*. employed the pour plate method with agar medium, whereas Helminth eggs were determined following the method proposed by Moodley et al. (2008). Selected heavy metals (Cu, Zn, Pb, Cd, Ni, Cr, Mn, Hg) were measured employing Atomic Absorption Spectrometry. All analyses were conducted in accordance with standard methods (APHA, 2017). Details of the analytical methods, equipment make-up, models, and manufacturers have been tabulated in appendices.

2.2.4 Quantification of Dissolved Methane in Effluent

Some studies have reported that domestic wastewater treatment with UASB reactors usually produces methane-rich biogas; however, a significant portion of the methane remains dissolved in solution and gets discharged together with the effluent or by other means (Gupta & Goel, 2019; Noyola et al., 2006). Therefore, the dissolved methane (dCH_4) in the UASB effluent was estimated using the Equation proposed by Asano et al. (2021):

$M_d = Q * M_c * \alpha * 100$ ... ... *(Eqn. 2.1)*

Where:

M_d = Dissolved methane (L/d)

Q = Biogas production (L/d)

M_c = Percentage methane composition in biogas (%)

α = Bunsen solubility coefficient for methane. As was reported by Yamamoto et al. (1975), α for non-saline water at 30 ^oC = 0.02898 ml CH_4

2.2.5 COD Mass Balance

The model developed by Lobato et al. (2012) was employed to determine the mass balance of COD in the UASB reactors. The COD mass balance considers the various transformation pathways of the influent COD (Figure 2.5). Influent COD load represents the average COD load applied ($COD_{applied}$) to each UASB reactor, which undergoes a series of conversion routes. Some portions of the COD are converted to CH_4 present in biogas ($COD_{CH4-biogas}$); some portions are converted to sludge (COD_{sludge}); some are used in the reduction of sulphate by sulphate-reducing bacteria ($COD_{sulphate}$), some COD are not converted to methane but are lost in the effluent ($COD_{effluent}$), whilst others are converted to CH_4 and are lost dissolved in the effluent (COD_{dCH4}) or lost with the waste gas or into the atmosphere ($COD_{CH4-lost}$) (Noyola et al., 2006; Souza et al., 2011). Equations used to calculate the various portions of the COD mass balance are presented in Table 2.2.

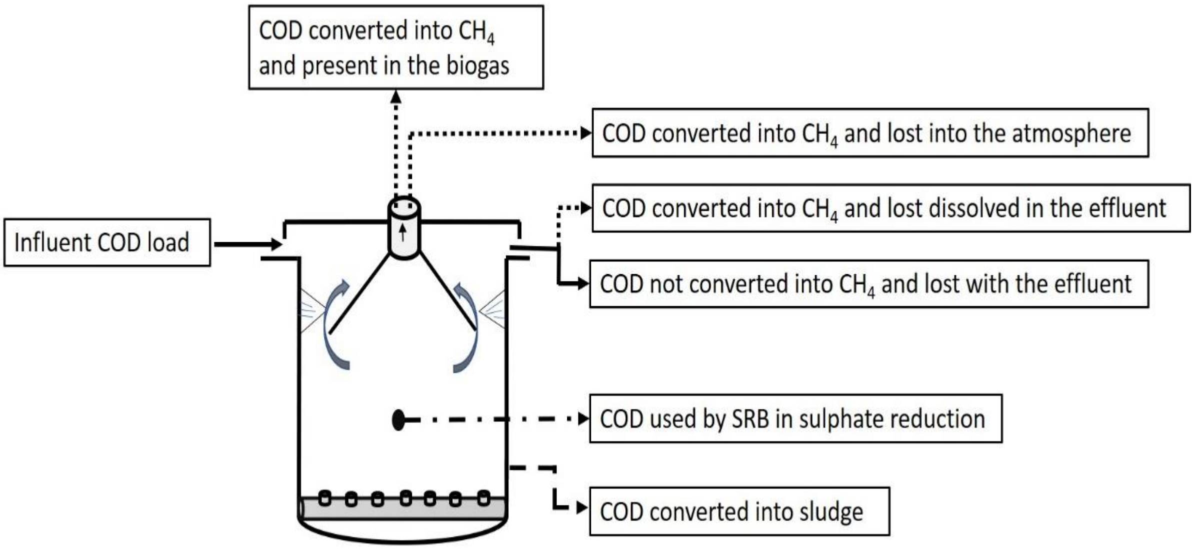


Figure 2.5: COD conversion routes in UASB reactors

Table 2.2: Equations for calculating the portions of the mass balance of COD

Parameters	Equations (Eqn. 2.2 - 2.8)	Variables
Estimate of mean influent flow rate	F_{mean} = Pop * QPC *(Eqn. 2.2)*	F_{mean} = mean influent flowrate (m^3/d); Pop = population (inhab.); QPC= per-capita wastewater contribution ($m^3/cap/d$)
Estimate of daily COD mass removed from the system	$COD_{removed}$ = Pop * QPC_{COD} * E_{COD} *(Eqn. 2.3)*	$COD_{removed}$ = daily COD mass removed from the system (kgCOD/d); Pop = population (inhab.); QPC_{COD} = per-capita COD contribution (kgCOD/inhab/d); E_{COD} = efficiency of COD removal (%)
Estimate of daily COD mass converted to sludge	$COD_{sludge} = COD_{removed}$ * Y_{COD} *(Eqn. 2.4)* Y_{COD} = Y * $K_{TVS\text{-}COD}$	COD_{sludge} = daily COD mass converted into biomass ($kgCOD_{sludge}/d$); Y_{COD} = coefficient of solids production ($kgCOD_{sludge}/kgCOD_{removed}$); Y = sludge yield as TVS (0.15 $kgTVS/kgCOD_{removed}$); $K_{TVS\text{-}COD}$ = conversion factor (1 kgTVS = 1.42 $kgCOD_{sludge}$); TVS = total volatile solids
Estimate of sulphate load converted into sulphide	COD_{SO_4} converted = F_{mean} * C_{SO_4} * E_{SO_4} *(Eqn. 2.5)*	COD_{SO_4} converted = load of SO_4 converted into sulphide ($kgSO_4/d$); C_{SO_4} = average influent SO_4 concentration ($kgSO_4/m^3$); E_{SO_4} = efficiency of sulphate reduction (%)
Estimate of daily COD mass used in sulphate reduction	$COD_{SO_4} = COD_{SO_4}$ converted * $K_{COD\text{-}SO_4\_}$ *(Eqn. 2.6)*	COD_{SO_4} = COD used by sulphate reducing bacteria for sulphate reduction ($kgCOD_{SO_4}/d$); $K_{COD\text{-}SO_4\_}$ = COD consumed in sulphate reduction (0.667 $kgCOD_{SO_4}/kgSO_4$)
Estimate of daily COD mass converted into methane	$COD_{CH_4} = COD_{removed} - COD_{sludge} - COD_{SO_4}$ $Q_{CH_4} = \left(\frac{COD_{CH_4} * R * (273+T)}{P * K_{COD} * 1000}\right)$	COD_{CH_4} = daily COD mass converted into methane ($kgCOD_{CH_4}/d$); Q_{CH_4} = theoretical volumetric production of methane (m^3/d); R = gas constant (0.08206 atm L/mol/ K); T = operational temperature of the reactor (ºC); P = atmospheric pressure (1 atm); K_{COD} = COD of one mole of CH_4 (0.064 $kgCOD_{CH_4}/mol$)
Estimate of methane loss	$Q_{W\text{-}CH_4} = Q_{CH_4} * p_w$ $Q_{O\text{-}CH_4} = Q_{CH_4} * p_o$ $Q_{L\text{-}CH_4} = F_{mean} * p_L * f_{CH_4} * \left(\frac{R * (273+T)}{P * K_{COD} * 1000}\right)$	$Q_{W\text{-}CH_4}$ = methane loss as waste gas (m^3/d); p_w = percentage of methane in the gaseous phase lost as waste gas (%); $Q_{O\text{-}CH_4}$ = other methane losses in the gaseous phase (m^3/d); p_o = percentage of methane in the gaseous phase considered as other losses (%); $Q_{L\text{-}CH_4}$ = loss of dissolved methane in the liquid effluent (m^3/d); p_L = concentration of dissolved methane in the liquid effluent (kg/m^3); f_{CH_4} = conversion factor of methane mass into COD mass (4 $kgCOD/kgCH_4$)

(Adapted from Aragão et al., 2021; Lobato et al., 2012; Rosa et al., 2020)

2.2.6 Specific Methanogenic Activity Test

2.2.6.1 Source and Characteristics of Inoculum Sludge

The inoculum sludge used for the SMA tests was sourced from each of the six modular-shaped UASB reactors at the Mudor WWTP. The Mudor UASB reactors treat municipal wastewater from various suburbs within the Accra central business district (A detailed description of the UASB reactors is given in Section 2.2.2). Sludge was sampled at sludge discharge ports sited approximately 2 meters from the bottom of the reactors. Composite samples were made from the grab samples taken from each reactor and used for the experiment. Sludge preserved at 4 ^{o}C for a maximum period of 7 days was used as the inoculum for the experiment. The inoculum was tested for pH, alkalinity, TS, VSS, COD and BOD prior to the experimentation.

2.2.6.2 Experimental Procedure

The tests were performed in 500 ml Mariotte glass bottles closed with rubber seals and highly resistant adhesive tapes connected to a syringe hose to transport the biogas. The bottles have been purged with CO_2 gas to allow an anaerobic environment. The experiments were carried out in triplicates with a 20% headspace (de Amorim et al., 2019). Analytical reagent grade anhydrous sodium acetate and glucose were employed as standard substrates at a ratio of 2:1. These substrates have been reported to be easily digestible and usually well assimilated by methanogenic bacteria. It is worth noting that the use of nutrients has not been considered in this study, as some studies have reported that the addition of nutrients and trace elements to their substrate (glucose) had no positive influence on methane production (Liu et al., 2016). Different inoculum-to-substrate ratios (ISRs) at 2:1, 1:1, 1:2, 1:4, 1:6, and 1:8 were applied in this study to assess the impact of ISR on methane production and SMA. The ISR was modified by varying the concentration of the inoculum whilst keeping the concentration of the substrate constant. The ISR was calculated according to the formula proposed by Kreuk et al. (2012), presented in Equation 2.9:

$$\frac{I}{S}\ ratio = \frac{V_{sludge} * VS_{sludge}}{V_{substrate} * COD_{substrate}} \dots\dots\dots\dots\dots \textit{(Eqn. 2.9)}$$

Where;

V_{sludge} = Volume of inoculum sludge (litres)

VS_{sludge} = Volatile solids concentration of sludge (mg/L)

$V_{substrate}$ = Volume of the substrate (litres)

$COD_{substrate}$ = COD of the substrate (mg/L)

The reaction and control (inoculated without a substrate) bottles were incubated in a water bath at a controlled temperature of 37 °C. The bottles were subjected to manual agitation twice daily except on weekends. This was done to ensure adequate contact between the substrate and biomass.

2.2.6.3 Methane Quantification

Biogas measurement was carried out by the volumetric method of direct measurement of methane volume by passing the biogas from the reaction bottles through an alkaline solution of 3.0% (m/v) sodium hydroxide (NaOH) for the absorption of CO_2 (de Amorim et al., 2019). The volume of displaced water was estimated to be the CH_4 generated. The experimental design for the SMA test is illustrated in Figure 2.6.

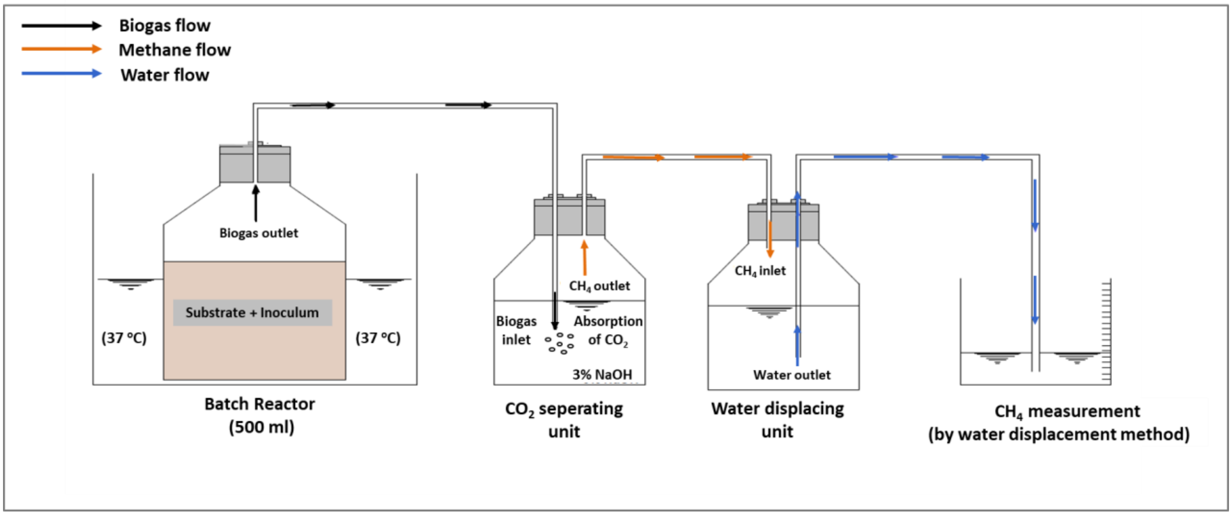


Figure 2.6: Experimental design for SMA test analysis

The SMA was calculated using the maximum methane production in 24 hours, according to the Equation proposed by de Amorim et al. (2019):

$$SMA = \frac{V_{CH_4}/t}{f * VSS * V_u} \ldots\ldots\ldots\ldots\ldots (Eqn.\ 2.10)$$

Where:

V_{CH4} = The maximum volume of methane produced in the considered time interval (ml)

t = The considered time interval in days

f = The stoichiometric conversion factor (350 mlCH_4/gCOD)

(Germany VDI 4630, 2016)

VSS = Volatile suspended solids concentration of biomass (g/l)

V_u = The active volume of the reaction bottle.

2.2.6.4 Application of SMA Test

To avoid the unwanted loss of solids in the final effluent, which would result in the reduction of the UASB reactor effluent quality, excess sludge is withdrawn from the reactors. Excess sludge build-up in the reactor was defined by monitoring results of the reactor's effluent; deterioration of effluent quality for COD, BOD and TSS concentrations was indicative of excess sludge build-up, which needed to be withdrawn. During sludge withdrawal, the SMA test is employed to determine the minimum amount of biomass that should be retained in the reactor in order to ensure optimum system performance (Hussain & Dubey, 2017). The minimum mass to be retained was calculated by Equation 2.11, proposed by Chernicharo (2007):

$$M_{min} = \frac{L_{COD}}{SMA} \ldots \ldots \ldots \ldots \ldots \qquad (Eqn\ 2.11)$$

Where;

M_{min} = Minimum mass of sludge (kgTVS)

L_{COD} = Influent organic load (kgCOD/d)

SMA = Specific methanogenic activity.

Equation 2.12 was employed to estimate the minimum volume of sludge that should remain in each reactor during sludge discharge:

$$V_{sludge} = \frac{M_{min}}{C_{sludge}} \ldots \ldots \ldots \ldots \ (Eqn.\ 2.12)$$

Where;

V_{sludge} = Minimum volume of sludge to be retained in the reactor (m^3)

C_{sludge} = Concentration of microorganisms in the reactor ($kgTVS/m^3$)

2.2.7 Data Analysis

Descriptive statistical analysis: maximum, minimum, and the appropriate central tendency measurements (i.e. mean and standard deviation); inferential statistical analysis: One-way ANOVA followed by Tukey's posthoc pairwise test, Pearson and Spearman correlations; and the respective removal efficiencies were used to interpret data.

2.3 Results and Discussions

2.3.1 Conditions of Operation of the Treatment Units

2.3.1.1 Operational Conditions Applied to the UASB Reactors

Table 2.3 presents the operational parameters applied to the UASB reactors. As reported for tropical regions, the UASB reactors were operated at a typical mesophilic temperature of 26.2 ± 1.8 ^{o}C and a near-neutral pH of 7.2 ± 0.4 for the influent sewage, comparable to reports in the literature (Casserly & Erijman, 2003; Halalsheh et al., 2005). The upflow velocity (Vel_{up}) of the system (1.0 ± 0.2 m/hr) was within the reported optimum range (0.5 - 1.5 m/hr) (Tawfik & Klapwijk, 2010). The observed mean hydraulic retention time (HRT) at 47.9 ± 11.8 hours far exceeded the reported range of 4 - 14 hours. The applied organic loading rate (OLR) of 1.22 ± 0.71 kgCOD/m^3/d was, however, found to be relatively lower than the values reported in the literature (Kaviyarasan, 2014). These findings indicate that the Mudor UASB reactors have a higher capacity than the load (4096 ± 837 m^3/d) currently handled by the Plant, signifying the Plant is operating under capacity. Sludge concentration in the UASB reactors was 65.63 ± 29.29 gVSS/L and 88.99 ± 28.93 gTSS/L. Other authors had reported lower sludge concentrations in ranges 27 - 57 gTSS/L and 32.2 - 50.2 gTSS/L, respectively, when they conducted rheological studies on sewage sludge concentrations (Abu-jdayil et al., 2010; Mori et al., 2006).

Table 2.3: 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	1.22 ± 0.71	2 - 14	(Bokhary et al., 2021; Kaviyarasan, 2014)
HRT (h)	30.9 - 119.1	47.9 ± 11.8	4 - 14	(Halalsheh et al., 2005; Rajakumar et al., 2011)
Vel_{up} (m/h)	0.33 - 1.26	1.0 ± 0.2	0.5 - 1.5	(Halalsheh et al., 2005; Rajakumar et al., 2011)
pH	5.4 - 7.9	7.2 ± 0.4	6.3 - 7.8	(Leitão et al., 2005)
T (^{o}C)	22.4 - 30.7	26.2 ± 1.8	20 - 40	(Leitão et al., 2005)

2.3.1.2 Operating Conditions Applied to the Post-Treatment Units

Two TFs and two secondary clarifiers, each operating in parallel are employed as the post-treatment units of the UASB reactors. Table 2.1 presents the dimensions of these units. The TFs are filled with black plastic media (180 mm polypropylene Bio-Pac Media SF30), with a 95% void ratio and 98.4 m^2/m^3 specific surface area as microbial carriers. Literature has reported that some critical process parameters to consider during TFs operation are the loading parameters such as hydraulic and organic loading, and recirculation ratio. The study found that the average flow to each TF was 90.2 ± 21.1 m^3/h. The volumetric OLR and hydraulic loading rate (HLR) were determined to be 0.19 ± 0.14 kgBOD/m^3/d and 0.19 ± 0.05 $m^3/m^2/h$, respectively. The loads on the Mudor TFs can be classified under low or standard-rate TFs (PDEP, 2016). Moreover, the Mudor TFs were designed without a recirculation system. The range of OLR observed in this study tied in with the findings by Rosa et al. (2018) when they presented results on OLR for TFs employed as post-treatment units for effluent from a full-scale UASB reactor.

Significant clarifier operating parameters include the detention time (DT), weir overflow rate (WOR), surface overflow rate (SOR) and solids loading rate (SLR) (Michigan Department of Environmental Quality Operator Training and Certification Unit). The DT and SOR applied to each clarifier were 18.1 ± 4.7 hours, and 4.59 ± 1.07 $m^3/m^2/d$, respectively. The estimated WOR and SLR were respectively, 29.40 ± 6.86 $m^3/m/d$ and 4.27 ± 0.99 kgTSS/m^2/d. The loading parameters obtained indicate that the organic and solids loads received by the post-treatment units were very low, and this could be ascribed to the high (> 70%) removal efficiency of the UASB reactors for solids and organics in the influent sewage. The operational conditions applied to the post-treatment units have been presented in Table 2.4.

Table 2.4: Operational conditions applied to the post-treatment units

Operating Parameter	Current Study		Typical Design Criteria
	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
OLR ($kgBOD_5/m^3/d$)	0.04 - 0.93	0.19 ± 0.14	80.09 - 400.46
Settling Tanks			
DT (h)	9.8 - 42.2	18.1 ± 4.7	2 - 3
SOR ($m^3/m^2/d$)	1.86 - 7.99	4.59 ± 1.07	12.22 - 32.59
WOR ($m^3/m/d$)	11.89 - 51.15	29.40 ± 6.86	≈ 124.19
SLR (kg TSS/m^2/d)	1.73 - 7.43	4.27 ± 0.99	122.06 - 146.47

2.3.2 Sewage Characteristics

2.3.2.1 pH and Temperature Profile

The variations in pH and temperature at the various treatment stages have been illustrated in Figure 2.7. Several studies have revealed that wastewater treatment with anaerobic systems requires a pH range of 6.3 - 7.8, and a temperature between 20 and 40 °C, for optimum performance (Chen et al., 2017; Mao et al., 2015).

pH measured in this study revealed that an appropriate environment was maintained throughout the treatment process. The influent sewage pH ranged from 5.4 - 7.9, with an average of 7.2 ± 0.4. This falls within the optimum range reported for mesophilic anaerobic bacteria; hence pH adjustment was not required during the study. The acidic sewage (pH = 5.4) observed occasionally could be ascribed to inflows of acid-based compounds from small and medium-scale enterprises (SMEs) within commercial areas. Conversely, the alkaline sewage probably could be due to soapy and soapless detergents used at offices and homes (Ahmed et al., 2018). The pH of the sewage streams increased across the treatment units, from a mean 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 attributed to the nitrification and denitrification processes which occur at the aerobic post-treatment units.

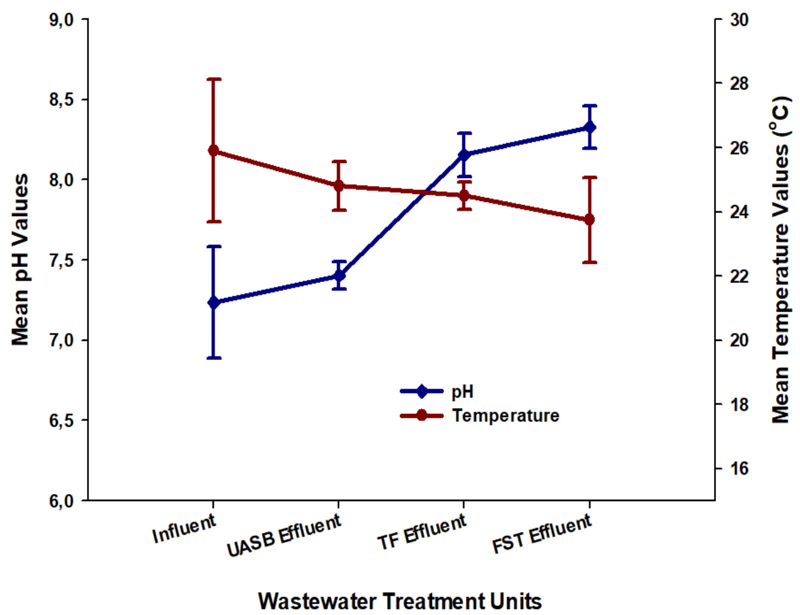


Figure 2.7: pH and temperature variations at the various stages of the treatment process.

While this observation contrasts the report by Awuah & Abrokwa (2008), who observed a pH drop from 8.96 ± 0.98 to 7.45 ± 0.14 for the influent and final effluent, respectively, it agrees with other findings (Ahmed et al., 2018; Belhaj et al., 2014). The influent sewage temperature ranged from 22.4 - 30.7 ^{o}C. This falls within the required temperature range for optimum anaerobic system performance (Bodík et al., 2000), and agrees with the findings by Ali & Okabe (2015) and Divya et al. (2015). One interesting finding was the fact that the temperature of the influent sewage fell within the range of local ambient air temperature (20.8 - 35.6 ^{o}C) observed during the study period (GMA, 2021), 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 temperate regions, making the UASB reactor technology economically viable to be implemented in tropical countries regions in the developing world (Chernicharo et al., 2015).

2.3.2.2 Volatile Fatty Acid (VFA) and Alkalinity Ratio

As can be observed in Table 2.5, an average VFA/Alkalinity ratio of 0.20 ± 0.10 (ranging from 0.12 - 0.45) was recorded for the UASB reactors' influent sewage during the study. Studies such as the ones by Callaghan et al. (2002) and Kuglarz et al. (2011) mentioned that VFA/Alkalinity ratio is a variable that measures system performance and controls the stability of the AD process. According to these authors, VFAs provide information on the AD intermediate steps performance, whilst alkalinity describes the capability of the feedstock to neutralize the VFAs generated during the process, controlling pH changes. VFA/Alkalinity ratio range from 0.10 - 0.40 is ideal for stable anaerobic digestion; 0.40 - 0.80 is indicative of system instability; and > 0.8 suggests gross instability, which could be ascribed to an increase in hydraulic or organic loadings to the system (Bakraoui et al., 2020; Hamawand & Baillie, 2015). Thus, the VFA/Alkalinity ratio obtained for this study falls within the optimum range for anaerobic reactors, as reported by several authors (Bakraoui et al., 2020; Chen et al., 2008; Hamawand & Baillie, 2015).

Table 2.5: 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	(Bakraoui et al., 2020)
BOD:COD Ratio	0.3 - 0.8	0.6 ± 0.2	0.3 - 0.8	(Aboulhassan et al., 2008; Manyuchi et al., 2018)
C:N Ratio	2.4 - 36.9	11.0 ± 8.3	20 - 30	(Li et al., 2011; Mao et al., 2015)
C:N:P Ratio	-	90:5:1	250 - 500:5:1	(Ammary, 2004; USEPA, 1995)
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	(Guo et al., 2019)
Zn (mg/L)	0.007 - 0.036	0.009 ± 0.005	0 - 5	(Guo et al., 2019)
Cd (mg/L)	0.002 - 2.020	0.157 ± 0.535	0.1 - 0.3	(Guo et al., 2019)
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	(Guo et al., 2019)
Hg (μg/L)	0.309 - 1.597	0.742 ± 0.385	-	-

2.3.2.3 Carbon, Nutrients, and Trace Elements

The BOD:COD ratio indicating the biodegradability index of influent sewage was observed to range from 0.3 - 0.8 (Table 2.5). According to Manyuchi et al. (2018), BOD:COD ratio measures the presence of biodegradable compounds in sewage. The BOD:COD ratio observed for this study was within the optimum range observed in the literature (Aboulhassan et al., 2008; Lee & Nikraz, 2014).

Influent sewage micronutrient composition has similarly been presented in Table 2.5. A balanced proportion of required nutrients, in combination with ideal growth conditions, are essential for the optimised performance of anaerobic systems (Li et al., 2015; Mata-Alvarez et al., 2014). It was observed in this study that the carbon:nitrogen (C:N) ratio was between 2.4:1 and 37:1, with a mean of 11 ± 8.5:1. The mean value obtained was comparatively lower than the values reported by other authors. Romano & Zhang (2008) reported an optimum C:N ratio of 15:1, whilst Cerón-Vivas et al. (2019) observed an increase in methane production and COD removal rate at C:N ratio of 14.2:1. Extensive studies by other authors have likewise reported an optimum C:N ratio for the highest rates of methanogenesis to be within 20:1 - 30:1, depending on the substrate fed into the reactor (Li et al., 2011; Mao et al., 2015; Zhang et al., 2014). However, this study's finding is comparable to the assertion by Kwietniewska & Tys (2014), who noted that municipal wastewater typically has a lower C:N ratio (< 8.0). The higher N concentration in the influent sewage could account for the low mean C:N ratio of influent sewage observed in this study. Martin-Ryals (2012) asserted that an unbalanced C:N ratio is one restraining factor in anaerobic digesters. The C:N ratio should be maintained within the optimum range to conserve the appropriate nutrient balance essential for microbial growth, and maintain a stable environment for an efficient AD (Cai et al., 2017; Zhang et al., 2014).

The C:N:P ratio of the influent sewage in this study was determined to be 90:5:1. USEPA (1995) and Ammary (2004) observed that a range between 250:5:1 and 500:5:1 was ideal during the start-up of an anaerobic reactor. The C:N:P ratio observed for this study indicates the influent sewage had high N and P contents.

Table 2.5 again presents the concentrations of heavy metals in the Mudor WWTP's influent sewage and the optimum ranges reported in the literature. Chen et al. (2014) and Şengör et al. (2009) have mentioned that some heavy metals are essential trace elements for microbial growth and development, promoting biogas and methane production. However, these elements in excess exert toxicity, inhibiting microbial community activity and destabilising the system. Guo et al. (2019) reported an optimum range for Zn^{2+} (0 - 5 mg/L), Ni^{2+} (0.8 - 50 mg/L), Cu^{2+} (0 - 100 mg/L), Cd^{2+} (0.1 - 0.3 mg/L), and Fe^{2+} (50 - 5000 mg/L) as concentrations which promote biogas production. As presented in the table, the average influent sewage heavy metals concentrations 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, and 0.005 mg/L, respectively.

Generally, the levels of the elements that were tested were within the reported range which did not impede optimum system performance. Moreover, Hg was recorded at an average concentration of 0.742 µg/L. Heavy metals like Pb and Hg are biologically not essential, having only toxic impacts. Appels et al. (2008) demonstrated that the use of both soapy and soapless detergents, and body care products are major pathways for heavy metals in domestic sewage. Other sources may include leaching from roofs, plumbing materials and gutters.

2.3.3 System Performance

Table 2.6 presents the general characteristics of the influent sewage received at the Mudor WWTP. The characteristics of the UASB effluent and TF effluent settled in the final settling tanks are also presented. The final effluent quality has been compared to effluent discharge guidelines of the Ghanaian Environmental Protection Agency (EPA); the regulatory body for sewage effluent discharge in Ghana.

2.3.3.1 Organic Matter and Solids Removal

Influent sewage, after preliminary treatment for the removal of solids and coarse materials, is directed to the UASB reactors which are the primary treatment units. From the study, influent COD concentration, which ranged from 450 - 8150 mg/L appreciably reduced to 590 ± 221 mg/L (Table 2.6), attaining an average removal efficiency of 72 ± 7% after treatment with the UASB reactors. Similar to this finding, a previous study recorded a maximum COD removal efficiency of 88.9% by the Mudor UASB reactors (Ahmed et al., 2018).

In their studies, Lettinga (2005) and Slompo et al. (2019) have proven the UASB reactor's exceptional ability to eliminate organic load from municipal and domestic sewage. These assertions have been confirmed by the robust performance of the UASB reactors at Mudor WWTP in removing COD from raw sewage. In a related study, Heffernan et al. (2011) found that the COD removal efficiencies for ten (10) full-scale UASB reactors, seven (7) of which were located in Brazil, two (2) in India and one (1) in the Middle East treating municipal sewage were between 44 and 77%. Other extensive studies have over the years reported the exceptional ability of UASB reactors in removing organic pollutants in diverse wastewater streams, attaining COD removal efficiencies as high as 90% (Musa et al., 2019; Satyanarayan et al., 2009; Verma et al., 2015).

Post-treatment with the TFs and settling in the secondary clarifier further improved system performance. The overall COD removal efficiency of the Mudor WWTP was 86.2 ± 2%. Although satisfactory, some studies have reported as high as 99% overall COD removal for UASB reactors followed by various post-treatment units; however, it is worthy to note that these were laboratory and pilot-scale experiments (Banihani & Field, 2013; Bhatti et al., 2014; Gonzalez-Tineo et al., 2020). Pilot-scale studies of UASB reactors followed by TFs as post-treatment attained UASB removal efficiencies between 65 and 84% and overall efficiencies were observed between 74 and 88% (Chernicharo & Nascimento, 2001; Pontes et al., 2003; Rodríguez et al., 2001).

Table 2.6: General wastewater characteristics and performance of Mudor WWTP

Wastewater Parameter (unit)	*n*	Influent Sewage (Range)	Influent Sewage (Average ± SD)	UASB Effluent (Average ± SD)	FST Effluent (Average ± SD)	Overall Plant* Efficiency (%)	EPA Guidelines
Physico-chemical							
pH	300	5.4 - 7.9	7.2 ± 0.4	7.2 ± 0.9	8.2 ± 0.1	-	6 - 9
Temperature	300	22.4 - 30.7	26.2 ± 1.8	26.0 ± 1.6	24.1 ± 2.3	-	< 30
EC (µS/cm)	300	1233 - 31,000	3097 ± 2922	3221 ± 340	2977 ± 371	-	1500
DO	300	0.00 - 1.00	0.29 ± 0.23	0.54 ± 0.45	3.56 ± 1.72	-	-
COD (mg/L)	300	450 - 8150	2127 ± 1251 **A**	590 ± 221 **B**	152 ± 115 **C**	92.8	250
BOD (mg/L)	300	308 - 5134	1384 ± 887 **A**	120 ± 73 **B**	33 ± 31 **C**	97.6	50
TS (mg/L)	300	1181 - 6450	2439 ± 661 **A**	1569 ± 301 **B**	1056 ± 188 **C**	56.7	50
TSS (mg/L)	300	24 - 2330	979 ± 410 **A**	262 ± 129 **B**	72 ± 18 **C**	92.6	50
TDS (mg/L)	300	893 - 6110	1480 ± 562 **A**	1241 ± 178 **B**	955 ± 182 **C**	35.5	1000
TVS (mg/L)	300	16.8 - 1504.0	682.0 ± 293.0 **A**	177.4 ± 99.8 **B**	48.3 ± 14 **C**	92.9	75
Nutrients							
TN (mg/L)	40	35.10 - 360.00	114.46 ± 59.20 **A**	121.01 ± 48.34 **A**	83.61 ± 24.51 **B**	27.0	-
NH_3 -N (mg/L)	40	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)	40	0.60 - 30.00	7.94 ± 6.44 **A**	5.92 ± 6.54 **A**	10.93 ± 7.94 **A**	-37.7	50
PO_4^{3-}-P (mg/L)	40	13.35 - 26.26	19.50 ± 3.70 **A**	12.48 ± 5.93 **A**	21.15 ± 4.28 **A**	-8.4	-
TP (mg/L)	40	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)	40	11.00 - 620.00	146.46 ± 106.20 **A**	45.08 ± 32.49 **B**	82.45 ± 23.99 **B**	43.7	-
Sulphide (mg/L)	40	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)	40	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	-
E. coli (CFU/100mL)	40	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	-
Salmonella (CFU/100mL)	40	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	-

*****_Pollutant concentration of final effluent from Mudor WWTP discharged into the environment relative to the raw influent received at the Plant._

One-way ANOVA Results; **A-C:** Columns that do not share the same letter indicate significant statistical difference ($p < 0.05$, Tukey's posthoc pairwise test) between means of treatment units' effluent. *n* = number of samples

A similar observation was made for BOD removal, with 86 ± 8% removal efficiency for the Mudor UASB reactors and 97 ± 1% after post-treatment. Previous studies on the same Plant reported BOD removal efficiency for the UASB reactors and post-treatment units at 93% and 98%, respectively (Awuah & Abrokwa, 2008). Ahmed et al. (2018) likewise attained 96% and 99% removal efficiencies, respectively, for the UASB reactors and post-treatment units. The efficiency of the Mudor WWTP reported by the two previous studies is higher than the performance observed for this study for BOD removal. The full-scale Plant studies by Heffernan et al. (2011) attained BOD removal efficiencies between 37 and 80%. Other pilot-scale experiments had BOD removal efficiencies ranging from 54 - 88% (Azimi & Zamanzadeh, 2004; Rizvi et al., 2015; Singh & Viraraghavan, 2003), whilst some pilot-scale studies reported on the combined system attaining BOD removal up to 94% (Chernicharo & Nascimento, 2001; Pontes et al., 2003). Regarding solids removal, the Mudor UASB reactors removed 35.7% and 16.3% of TS and TDS, respectively. The post-treatment units further enhanced the removal efficiencies respectively, to 56.7% and 35.5%. The poor TDS removal could be ascribed to the inability of the anaerobic reactor to remove ions from the sewage, which negatively influenced TS removal as well. Meanwhile, TSS and TVS removal were satisfactory for the UASB reactors at 73.3% and 74%, respectively, with 93% overall removal efficiency for both parameters. The findings for solids removal are comparable to similar studies in literature, with values ranging between 41 and 77% for UASB reactors and between 73 and 89% for TF post-treatment (Chernicharo & Nascimento, 2001; Pontes et al., 2003). Generally, the variations in organics and solids removal could be attributed to factors such as the Plant operating conditions. Figure 2.8 presents the system performance for COD, BOD and TSS during the study period.

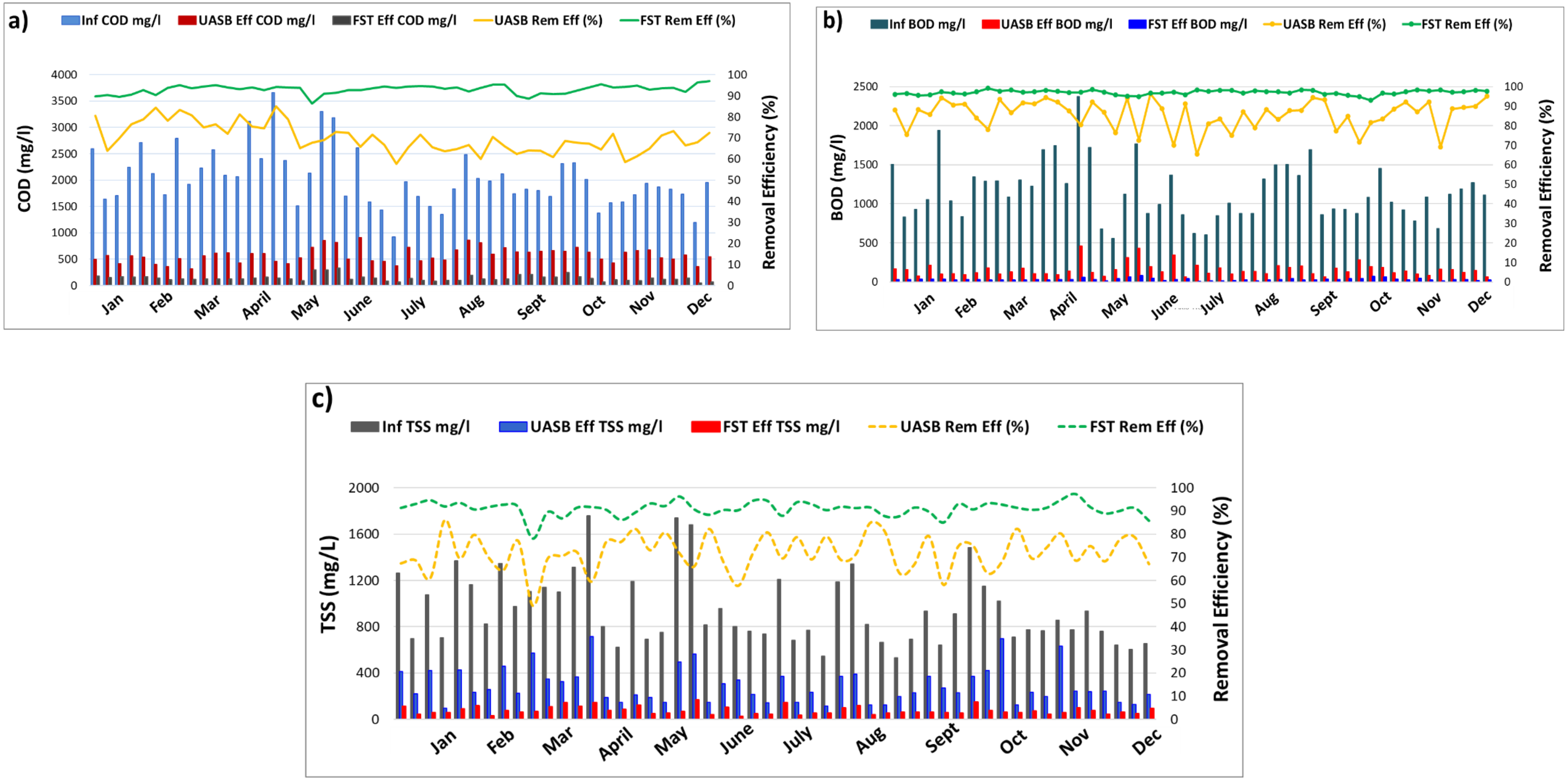


Figure 2.8: Plant performance in a) COD removal, b) BOD removal, c) TSS removal

2.3.3.2 Nutrients Removal

The relative ineffectiveness of UASB reactors in eliminating nutrients from wastewater has been recorded by a number of studies, necessitating the employment of post-treatment units (Daud et al., 2018; Foresti et al., 2006). As observed in Table 2.6, the Mudor UASB reactors performed poorly in removing N compounds, confirming assertions in the literature. However, it was found that the biological aerobic treatment at the TFs was also inefficient in removing adequate N compounds from the sewage. TN concentration was found to have increased slightly in the UASB effluent. This increment could be explained by the phenomenon of sludge flotation leading to the re-emergence of nitrogen it contains. The system attained an overall TN removal efficiency of only 27%. The NH_3-N concentration was likewise found to have increased in UASB effluent. This could be attributed to the processes of ammonification of organic nitrogen to ammonia under anaerobic conditions. Similarly, NO_3^--N attained an overall negative removal efficiency, with observed effluent concentration higher than the influent concentration. The alternating levels of NH_3-N and NO_3^--N concentrations in the UASB reactors can be attributed to the reducing environment in the UASB reactors that favour NO_3^--N reduction by denitrification and promote NH_3-N generation; however, the reverse occurs after the UASB treatment in the trickling filters. Subjecting the results for nitrogenous compounds to one-way ANOVA analysis, no statistically significant difference was found to exist between the means of treatment units' effluent for NO_3^--N ($p = 0.521$), but effluent means for TN, and NH_3-N observed a statistically significant difference ($p < 0.05$).

Several studies have demonstrated satisfactory nitrogen removal after coupling UASB reactors with other post-treatment units. For instance, Bhatti et al. (2014) reported on the efficiency of a UASB reactor coupled with hydrogen peroxide (H_2O_2) treating municipal wastewater. The configuration attained N removal efficiency between 51.7 and 87.5%. Other studies attained similar satisfactory results (> 60%) when a constructed wetland system and a sequential batch reactor were employed as post-treatment units for UASB reactor effluent (de Sousa et al., 2001; Torres & Foresti, 2001). The Mudor WWTP's abysmal performance regarding nitrogen removal could be ascribed to several factors. Firstly, since nitrification is an oxygen (O_2) demanding process, aeration at the TF might be inadequate. Unlike the activated sludge systems where O_2 is vigorously injected to enhance treatment, the TF employs a natural flow of atmospheric O_2 for the aerobic microorganisms, which might be ineffective under the prevailing Mudor WWTP operating conditions. Moreover, the Mudor WWTP's TFs did not have a system to recirculate effluent. Pearce (2004) opined that complete nitrification could be achieved in single filtration Plants if effluent recirculation is considered. Additionally, effluent recirculation increases the wetted surface area and has proven to advance nitrification processes (Pearce & Foster, 1999). Thus, the absence of the recirculation system could have reduced the contact time, resulting in the wash-out of nitrifiers and leading to incomplete nitrification (Li & Wu, 2014; Song et al., 2020).

Again, it has been reported that recirculation introduces nitrate onto the top of the filter media, where heterotrophic activity and supposedly denitrification activities are highest (Pearce & Foster, 1999). Furthermore, denitrifying bacteria require an amount of organic carbon (BOD) for complete denitrification. However, the UASB reactors eliminate about 86% of BOD from the influent sewage. The residual BOD available to the denitrifiers might be insufficient for the denitrification process to eliminate nitrogen. Thus, under the prevailing conditions, nitrification and denitrification processes might be incomplete (Curtin et al., 2011; Wisconsin Department of Natural Resources, 2018), negatively affecting nitrogen removal. Figure 2.9 illustrates the concentrations of N compounds at the various treatment units.

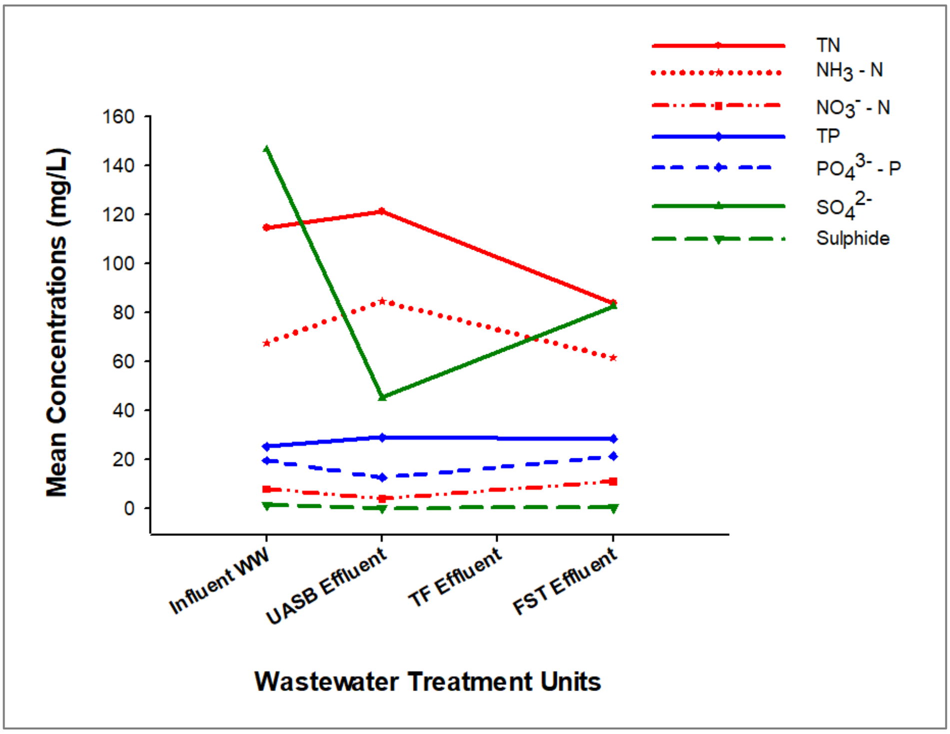


Figure 2.9: Sewage nutrient concentrations at the various treatment units.

Similar observations were made for phosphorus (P) compounds. TP and PO_4^{3-}-P equally attained an overall negative removal efficiency (Figure 2.9). The system achieved average effluent concentrations of 28.37 ± 14.17 mg/L and 21.15 ± 4.28 mg/L, respectively, for TP and PO_4^{3-}-P, higher than the average influent concentrations of 25.09 ± 4.90 mg/L and 19.50 ± 3.70 mg/L for the respective parameters. This observation could be ascribed to the phenomenon of phosphorus release and sludge flotation. Nevertheless, 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 EPA discharge limits. Moreover, this finding contrasts the findings by de Sousa et al. (2001) and Ahmed et al. (2018). These authors observed a satisfactory performance with over 80% removal efficiency for TP and PO_4^{3-}-P. Generally, the influent C:N:P imbalance could account for the high concentrations of N and P compounds in the final effluent, subsequently influencing the Mudor Plant's poor performance regarding nutrient removal. The N and P concentrations in the influent sewage significantly exceeded the carbon required for a balanced nutrient ratio for optimised anaerobic systems (Ammary, 2004; Kameswari et al., 2012). Some studies have suggested the addition of carbon supplements to augment the carbon content (co-digestion) to attain a balanced C:N:P ratio for optimum system performance (Lin et al., 2011; Kameswari et al., 2012). Nonetheless, it is evident that the post-treatment units at the Mudor WWTP have not been designed to effectively enhance biological phosphorus removal.

This study has revealed the inability of the post-treatment units at the Mudor WWTP to eliminate nitrogenous and phosphorous compounds from the UASB effluent, hence it will be prudent for a more efficient technology to be incorporated into the unit processes to enhance nutrient removal. Some technologies reported in the literature for effective nitrogen removal include ammonia precipitation as struvite, ammonia striping and distillation, ion exchange for ammonia and nitrate removal, chemical oxidation of ammonia processes and di-electrophoresis-enhanced adsorption, etc. (Asada et al., 2006; He et al., 2015; Rozic et al., 2000; Tchobanoglous et al., 2003; Zhang et al., 2009). Physico-chemical phosphorus elimination technologies include ion exchange, precipitation, and sorption mechanisms (Bunce et al., 2018). There exists likewise well-enhanced biological nutrient removal processes (Muduli et al., 2021; Winkler & Levi, 2019). Dębowski et al. (2022) identified anaerobic reactor filling as a modern, economically viable and effective method of phosphorus removal by metal dissolution. The availability of various nutrient removal technologies calls for further assessment to select the most viable alternative in terms of cost, efficiency, and sustainability to be implemented in developing countries.

The Mudor WWTP's performance was satisfactory for sulphate (SO_4^{2-}) and sulphide (S^{2-}) removal, with overall removal efficiencies of 43.7 and 85.9%, and mean effluent concentrations of 82.45 and 0.19 mg/L, respectively (Table 2.6). As illustrated in Figure 2.9, SO_4^{2-} and S^{2-} concentrations were reduced considerably in the UASB effluent but increased substantially in the final effluent. The considerable decrease in the UASB effluent may result from sulphate reduction to H_2S under anaerobic conditions (Deng et al., 2018). Aerobic post-treatment after the UASB treatment triggered oxidation of the dissolved sulphur species, resulting in increased sulphate levels in the final effluent (Rao et al., 2003). 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. (2003) recorded sulphide removal efficiency between 60 and 70% for an anaerobic-aerobic treated industrial wastewater employing the stripper technique. Yun et al. (2019) in a similar study recorded sulphate removal efficiency of 84 ± 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) for synthetic wastewater treatment. Comparably, Oliveira et al. (2020) examined biochar usage for sulphate-rich wastewater treatment and obtained 98% H_2S, 89% unionised sulphide, and 94% dissolved sulphide removal efficiencies.

2.3.3.3 Microbial Loads Reduction

Mean levels of microbial loads at the various treatment units have been illustrated in Figure 2.10. Results on Plant performance regarding microbial loads removal as presented in Table 2.6 revealed influent sewage levels of bacterial 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 faecal coliform (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 further increased microbial elimination to approximately 1 log unit (94 - 95%) for FC, *E. coli*, and Salmonella *sp.* The findings of this study agree with those reported by Cavalcanti et al. (2001) and Lohani et al. (2020), who recorded overall removal efficiencies > 90% for FC, and *E. coli* after post-treating UASB effluent with polishing ponds and sand filters, respectively. This finding affirms that configuration of UASB reactors with other post-treatment systems can reduce municipal sewage indicator organisms loads to acceptable levels. However, it is worth noting that although the post-treatment units improved pathogen load reduction, inferential statistics with one-way ANOVA revealed no statistically significant difference between the UASB and FST effluent means. With the exception of nutrients, the average effluent concentrations for most parameters monitored during this study were within the allowable discharge limits of EPA-Ghana.

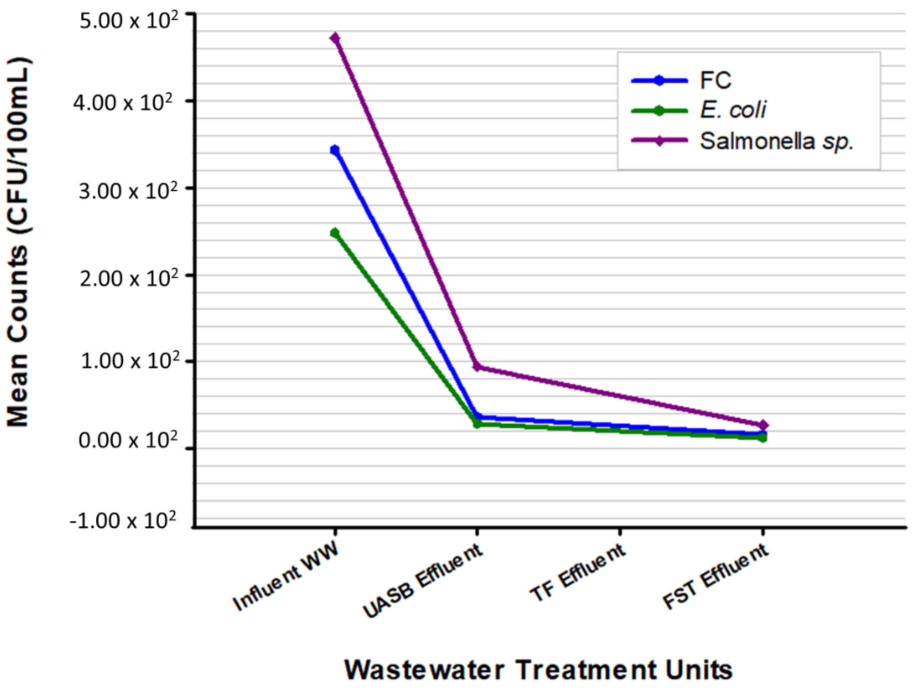


Figure 2.10: Mean levels of microbial loads at the various treatment units.

2.3.4 Biogas Production

2.3.4.1 Biogas Production Rate

Biogas production ranged from 101 - 1673 Nm^3/d, with an average daily production rate of 613 ± 271 Nm^3/d, and a specific biogas yield of 0.14 ± 0.07 $m^3/kgCOD$ removed. Presented in Figure 2.11 is the biogas flow variation compared with the COD load removed, sewage flow, and ambient temperature observed during the study period. Several studies have reported that OLR correlates with biogas production in anaerobic systems such as the UASB reactor (Daud et al., 2018; Ince et al., 2005; Klesyk, 2017; Musa et al., 2018). This fact is not disputed in this study; it is, however, argued that the biodegradable organic load removed from the system strongly correlates with biogas production compared to OLR. This is because OLR will only correlate significantly with biogas production depending on the biodegradability of the substrate. Easily biodegradable substrates would undergo hydrolysis at a faster rate during the AD process, producing biogas, as compared to substrates that are not easily biodegradable. Spearman correlation between COD flux removed from the system and biogas production rate revealed a strong, statistically significant linear relationship (R = 0.928, p < 0.0001) existed between the two variables (Figure 2.11a).

A similar analysis was performed for sewage and biogas flows (Figure 2.11b). A moderate correlation coefficient (R = 0.689, p < 0.0001) was attained between these two variables. This observation could be explained that as much as some form of correlation exists between the two variables, highly diluted or low-strength sewage would invariably reduce biogas flow. This does not preclude the fact that for a Plant connected via sewer networks to a specific number of households and treats relatively constant sewage in terms of load and volume, any significant drop in sewage flow volume would invariably result in a consequent drop in biogas flow. A critical observation of the graph for sewage flow revealed a significant reduction from June to December. The period from June to September experienced the highest precipitation in the year (Figure 2.2). During high rainfall, vast volumes of stormwater (combined sewer systems) are received at the Plant, compelling operators to divert the highly diluted influent sewage from entering the UASB reactors through a bypass into the Korle Lagoon (Arthur et al., 2022). The bypass eliminates the cost of pumping enormous volumes of highly diluted sewage into the UASB reactors. Thus, the sewage received at the treatment plant reduces during these periods. The low flows observed from September through December could be attributed to water shortages. Usually, in the months preceding the harmattan period, Ghana Water Company Limited (GWCL) rations tap water flow in several parts of Accra, leading to recurrent water shortages and consequent reduction in wastewater flow.

Moreover, just about the same period, a media house reported that GWCL was to cut the water supply to some parts of Accra, which included Korle Bu and Dansoman suburbs for maintenance activities (Pulse.com.gh, 2021). These two suburbs are part of those whose sewage is received by the Mudor WWTP, explaining the significant drop in sewage flow observed during those periods.

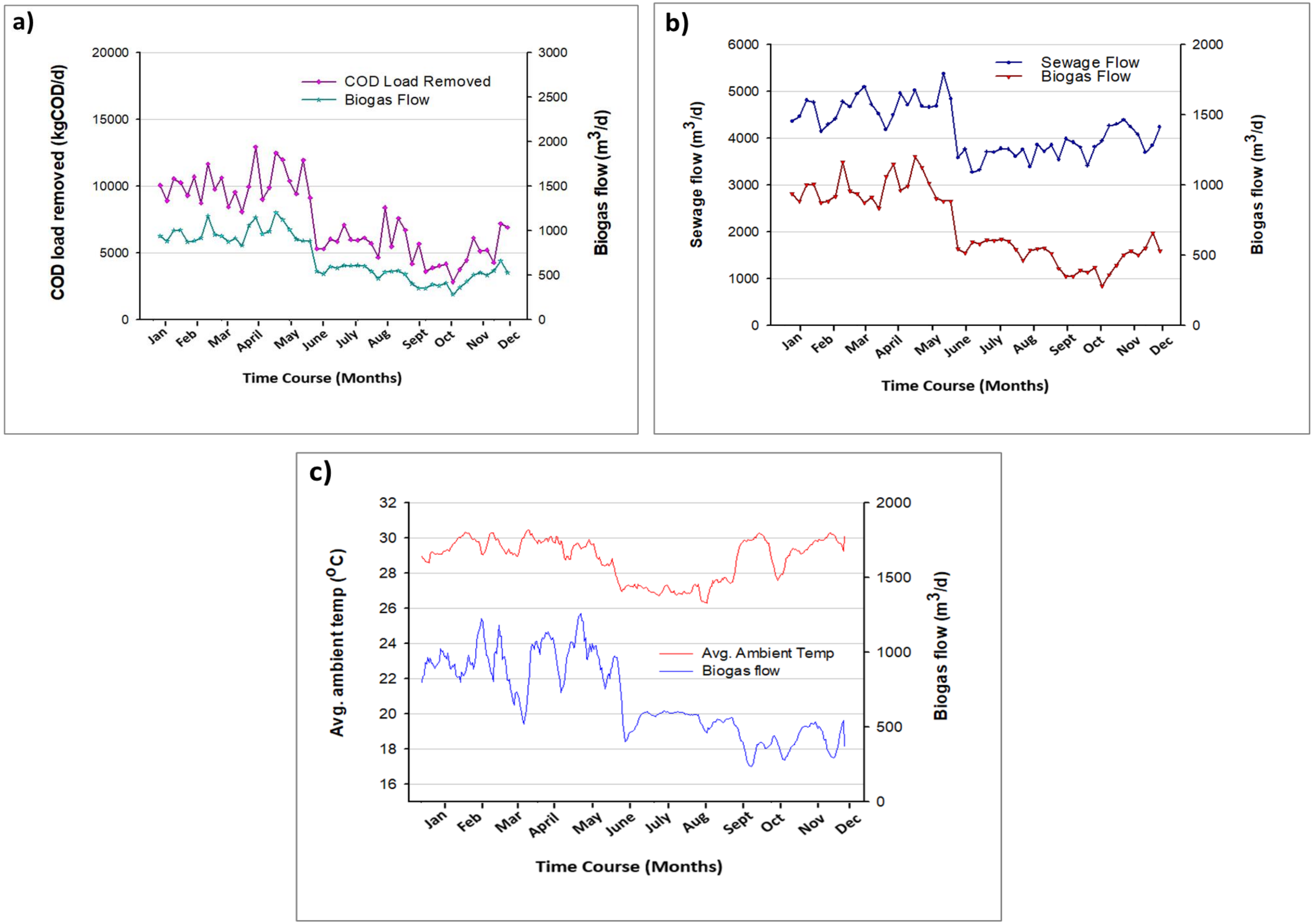


Figure 2.11: Correlation between biogas production, and a) COD load removed; b) Sewage flow; c) Ambient temperature

Figure 2.11c illustrates the variation between mean ambient temperature and biogas flow during the study period. The mean daily ambient temperature ranged from 26.3 - 30.4 ^oC, with a mean annual temperature of 29 ^oC. As seen in the graph, a consistent drop in mean ambient temperature was observed during the period from June to September. This period also experienced the highest precipitation within the year (Figure 2.2). Thus, the drop in ambient temperature could be ascribed to the rains, which generally lower ambient temperature during these periods. Spearman correlation nonetheless revealed a weak relationship between biogas flow and ambient temperature ($R = 0.195$, $p = 0.003$).

One prominent factor that influences performance and subsequent biogas production in anaerobic reactors is temperature. Several studies: Lew et al. (2003), Rizvi et al. (2015), Singh & Viraraghavan (2003), and Takahashi et al. (2011) have been carried out wherein was evaluated the influence of temperature on organic load removal and biogas production in UASB reactors. In all these reports, one common trend was observed; system performance and biogas production improved at temperatures above 20 ^oC and reduced at temperatures below 20 ^oC. Thus, a mesophilic temperature range is required for the optimum performance of UASB reactors (Bodík et al., 2000; Foresti, 2002; Lettinga et al., 2001). The weak correlation between ambient temperature and biogas production observed in this study could be attributed to the following reasons: First, the mean daily ambient temperature observed during the study ranged between 26.3 and 30.4 ^oC, as stated earlier. This range falls within the mesophilic temperature range required by anaerobic microbes. Thus, the system ought to function satisfactorily within this temperature range, explaining the satisfactory performance of the UASB reactors in BOD and COD removal. Additionally, the variation between the two temperatures is only 4 ^oC, not wide enough to cause any significant and easily recognisable change in the biogas production rate. These reasons could explain this study's weak correlation between ambient temperature and biogas flow.

2.3.4.2 Biogas Characterization

Biogas characterisation revealed methane ranged from 54 - 77% of the biogas output, with an average of 65%. The compositions of CO_2, O_2, and N_2 respectively, were in ranges 3.2 - 9.1%, 1.4 - 14.6% and 19.9 - 28.2%, as illustrated in Figure 2.12. H_2S gas concentration was detected to be between 78 and 314 ppm. Besides the relatively lower CH_4 fraction, the other biogas constituents observed for this study were comparable to the finding by Noyola et al. (2006), who recorded biogas composition of 70 - 80% CH_4, 5 - 10% CO_2, and 10 - 25% N_2 from a

UASB reactor treating domestic wastewater. The authors stated that dissolved N_2 in influent wastewater probably accounted for the high nitrogen content in the biogas generated. Konaté et al. (2013) similarly found 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. The observed methane composition (54 - 77%) in this study was lower than values reported (70 - 85%) by some authors for UASB reactors treating domestic sewage (Chernicharo et al., 2015; Souza et al., 2011). This study's relatively lower methane composition could result from many factors, such as sludge activity and Plant loading.

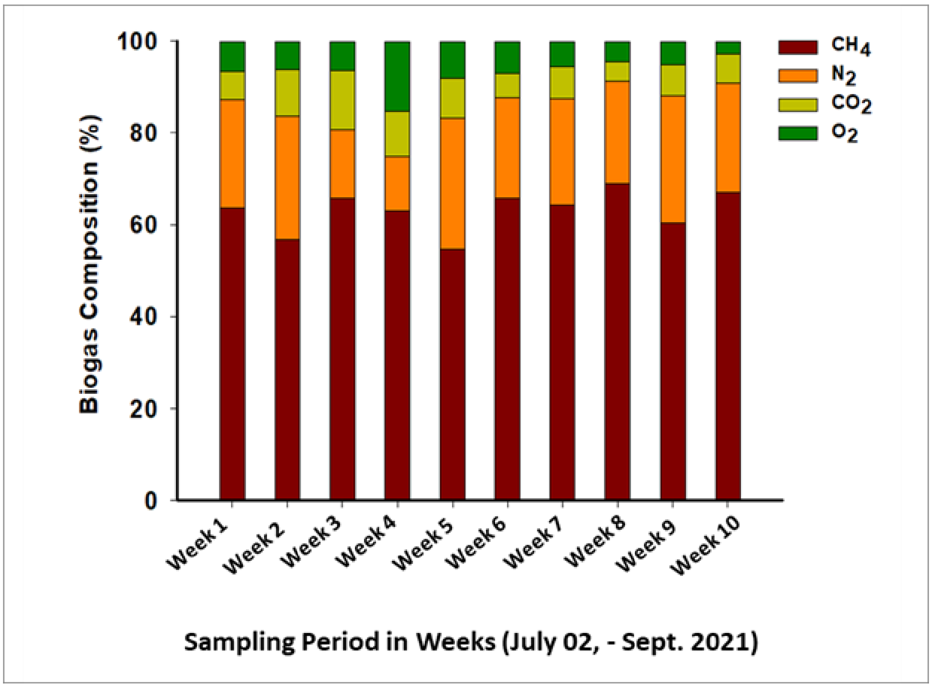


Figure 2.12: Biogas composition

Methane production ranged from 65 - 1071 Nm^3/d, with an average of 392 ± 173 Nm^3/d, and an average specific methane yield of 0.10 ± 0.05 $m^3CH_4/kgCOD$. Lobato et al. (2012) simulated a model to predict the specific methane yield of full-scale UASB reactors operating at optimum conditions. The authors attained methane yields between 0.11 and 0.19 $m^3CH_4/kgCOD$. The findings of this study were found to agree with their findings. Additionally, this study's findings are comparable to observations made by other authors (España-Gamboa et al., 2012; Ince et al., 2001; Musa et al., 2018), however, these studies were conducted on laboratory and pilot scales.

2.3.4.3 Methane Dissolution in UASB Effluent

As illustrated in Figure 2.13, the calculated dissolved methane (dCH_4) in the UASB effluent was approximately 23% of the gaseous methane produced (21 mg/L). Masuda et al. (2018) and Kong et al. (2021) reported a lower range from 19.8 - 22.3% for dissolved methane. However, Keller & Hartley (2003) opined that methane losses due to dissolution in the effluent of anaerobic systems could range from 20 - 60%. Souza et al. (2011) likewise recorded a range from 36 - 41% whilst Cookney et al. (2016) observed a range between 45 and 88%. According to these authors, the wide variations could be due to several factors including temperature, loading and the type of reactor.

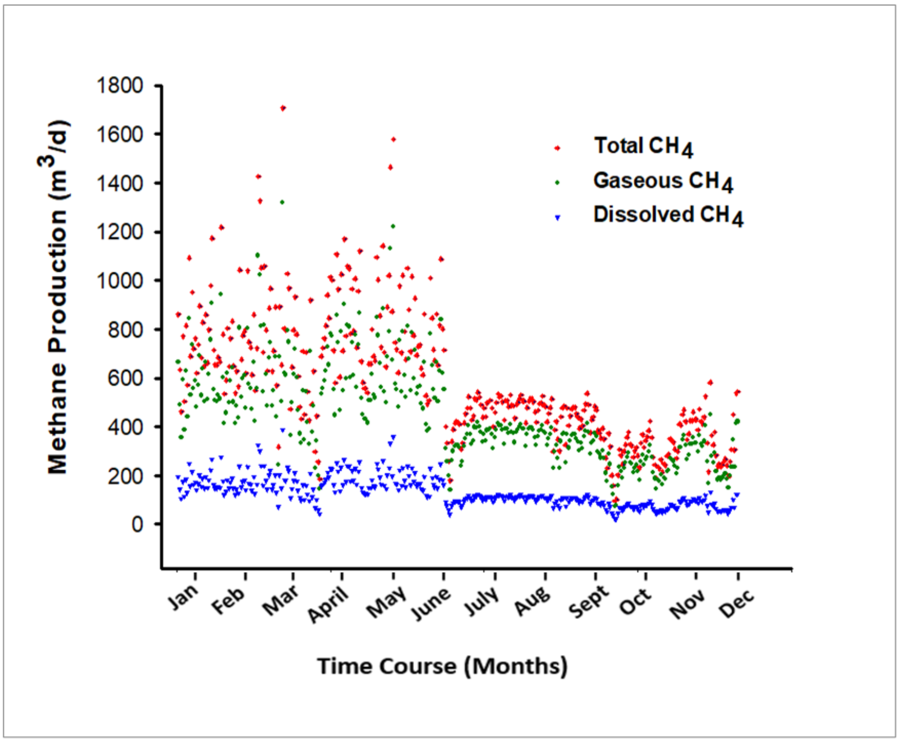


Figure 2.13: Scatter plot presenting Total, Gaseous and Dissolved CH_4

The solubility of methane in an aqueous medium is higher for temperate climatic zones than in tropical zones. Thus, temperate regions observed higher concentrations of dCH_4 in the effluent. The significant methane loss in wastewater effluent reduces the biogas energy recovery potential of wastewater treatment by anaerobic systems. Moreover, it presents an environmental challenge as methane is a potent greenhouse gas (GHG), with a global warming potential (GWP) about 28 times higher than carbon dioxide (IPCC, 2014). Henares et al. (2016) mentioned methane emissions could generate an explosive environment when effluent is discharged into drains or enclosed spaces. Several studies have been conducted to strip the

dCH_4 from anaerobically treated wastewater effluent (Centeno-mora et al., 2020; Cookney et al., 2016; da Silva Ramos et al., 2021).

2.3.5 COD Mass Balance

Table 2.7 presents the average COD load applied to each UASB reactor and the portions for the various conversion routes. The average applied COD load was estimated to be 1475.5 kgCOD/d, and the portions for the various conversion routes have been determined using the model developed by Lobato et al. (2012). The findings revealed that 33.5% of the applied COD load was converted into methane in the biogas, which is available for use by energy recovery. 15.4% of the COD load applied was converted into sludge, whilst the sulphate-reducing bacteria used 4.4% for sulphate reduction. 27.7% of the applied COD load was retained as residual COD in the effluent, whilst 13.7% was converted as methane but remained dissolved in the effluent. 5.3% was estimated to be converted to methane but lost with waste gas or leaked into the atmosphere. Comparing this finding to the literature, the values obtained are comparable to that presented by Lobato et al. (2012). The model developed by these authors presented similar percentage fractions, as shown in the table. In the same regard, the study by Souza (2010) likewise presented similar results.

Table 2.7: Conversion route of influent COD load applied to each reactor

Variable	COD load (kgCOD/d)	Percentage Distribution (%)		
		This Study	(Lobato et al., 2012)	(Souza, 2010)
$COD_{applied}$	1475.5	-	-	-
$COD_{CH_4\text{-lost}}$	77.5	5.3	2-3	-
COD_{dCH_4}	202.04	13.7	11 - 17	16 - 18
$COD_{Effluent}$	409.31	27.7	30 - 40	14 - 24
COD_{Sludge}	227.06	15.4	13 - 15	8 - 10
$COD_{Sulphate}$	64.99	4.4	3 - 7	4.5 - 5
$COD_{CH_4\text{-biogas}}$	494.60	33.5	20 - 39	24 - 30

2.3.6 Specific Methanogenic Activity Test

The characteristics of the sludge used for the SMA test have been presented in Table 2.8. The COD concentration of the substrate was found to be 65.4 g/L. Daily methane production observed during the experiment has been presented in Figure 2.14a. The various ISRs applied in this study performed differently regarding methane production. It was found in the experiment which lasted for barely ten (10) days, that an ISR of 1:1 resulted in the highest methane production on day 1, after which methane production began to drop systematically. ISR of 1:8 was found to result in the least methane production on day 1. Generally, it was observed that methane production decreased with a decrease in ISR. Figure 2.14b presents the cumulative methane production during the experiment. ISR 1:1 had the highest value in terms of methane production, with a cumulative value of 471 ml. This was followed by ISR 2:1, with a cumulative production of 365 ml. ISR 1:2, 1:4, 1:6 and 1:8 had cumulative values, respectively, 298 ml, 224 ml, 174 ml and 146 ml.

Table 2.8: Characteristics of sludge used for the SMA test

Parameter	Range	Average
pH	7.0 - 8.5	7.7 ± 0.4
TS (g/L)	24.7 - 156.7	113.0 ± 32.9
TVS (g/L)	6.7 - 118.7	71.1 ± 22.8
TSS (g/L)	11.7 - 102.0	53.4 ± 21.1
$^{VS}/_{TS}$ (%)	50.5 - 80.9	64.2 ± 4.9
BOD (g/L)	2.9 - 3.2	3.01 ± 0.09
COD (g/L)	56.6 - 82.5	69.3 ± 9.4
Alkalinity (g$CaCO_3$/L)	2.8 - 10.4	7.2 ± 2.1
VFA (mg/L)	72.3 - 433.8	144.6 ± 112.6
EC (mS/cm)	3.1 - 5.7	4.2 ± 0.6

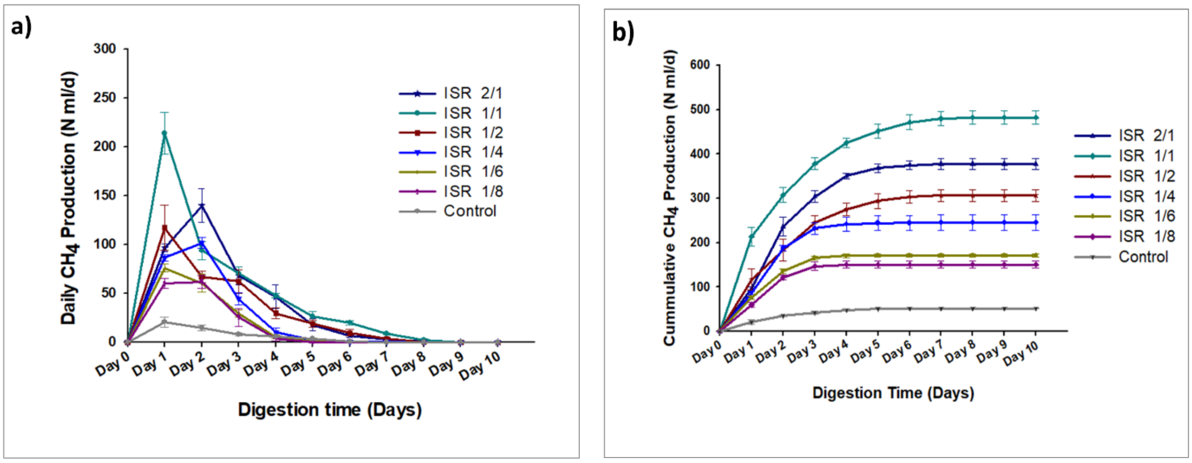


Figure 2.14: a) Daily methane production; b) Cumulative methane production

Thus, an ISR of 1:1, which presents an equal concentration of biomass to substrate, resulted in the highest methane production, whilst ISR 1:8, which presented much higher food relative to biomass, resulted in the most negligible methane production. Some studies have been reported wherein it was found that ISR of 1:1 yielded the best results in terms of methane yield, compared to providing too little or much food for the biomass. Córdoba et al. (2018) studied the effect of ISR on the kinetics of methane production. They found that ISR of 1:1 yielded the highest methane production compared to ISR of 3:1 and 6:1. In a related study, Yoon et al. (2014) found that ISR of 10:1, which implied low substrate concentration compared to the concentration of biomass resulted in the least cumulative methane yield, whilst the ISR of 2:3 presented the highest cumulative methane yield. This observation has been made by many other authors: Moset et al. (2015), Pellera & Gidarakos (2016), and Rouches et al. (2019), for the various substrates they employed. These findings could be attributed to the accumulation of inhibitors. Holliger et al. (2016) found that a higher substrate fraction than the inoculum for readily biodegradable substrates would result in the rapid accumulation of fermentation intermediates such as VFAs, which could lead to the inhibition of AD. Moreover, very low ISR may cause system overloading, leading to a pH drop due to VFA and ammonia accumulation, causing inhibition and a subsequent drop in methane yields (Angelidaki et al., 2009).

Figure 2.15 presents a graph of the cumulative methane production plotted against SMA values obtained for the various ISRs. It was found for this study that the ISR with the highest cumulative methane production correspondingly had the most significant SMA values. Thus, the ISR of 1:1 resulted in the highest SMA value of 0.039 $gCOD_{CH_4}$/gVSS.d, whilst the ISR of 1:8 had the most negligible SMA value of 0.006 $gCOD_{CH_4}$/gVSS.d. It can be inferred that methane production directly correlates with SMA values. Different studies have reported varying SMA values. A review by Hussain & Dubey (2017) reported a wide range of SMA values obtained under varying conditions. Substrates employed in the literature include Acetate, glucose, sucrose, phenol, and various types of biomass (Fang et al., 1994; Gali et al., 2006; Tay et al., 2001). However, the authors mentioned that acetate is the most common substrate for SMA tests. SMA values obtained from these studies range from as low as 0.01 $gCOD_{CH_4}$/gVSS.d, as reported by Kalogo et al. (2001), to as high as 1.10 $gCOD_{CH_4}$/gVSS.d as was reported by Shin et al. (2001). Souto et al. (2010) likewise found that different sodium acetate concentrations resulted in different SMA values, ranging from 0.00 $gCOD_{CH_4}$/gVSS.d for a concentration of 10.0 g/L to 0.0783 $gCOD_{CH_4}$/gVSS.d for a concentration of 2 g/L. Thus, SMA values are influenced by several factors like the choice of substrates, ISR, type of inoculum, inoculum storage conditions, environmental conditions and test procedure applied (Astals et al., 2020; de Amorim et al., 2019; Liu et al., 2016; Soto et al., 1993). Van Haandel & Lettinga (1994) found that sludge from anaerobic digesters mostly had SMA values between

0.01 - 0.04 $gCOD_{CH_4}$/gVSS.d. This could explain the smaller SMA values obtained for this study, using sludge from the anaerobic UASB reactors as inoculum.

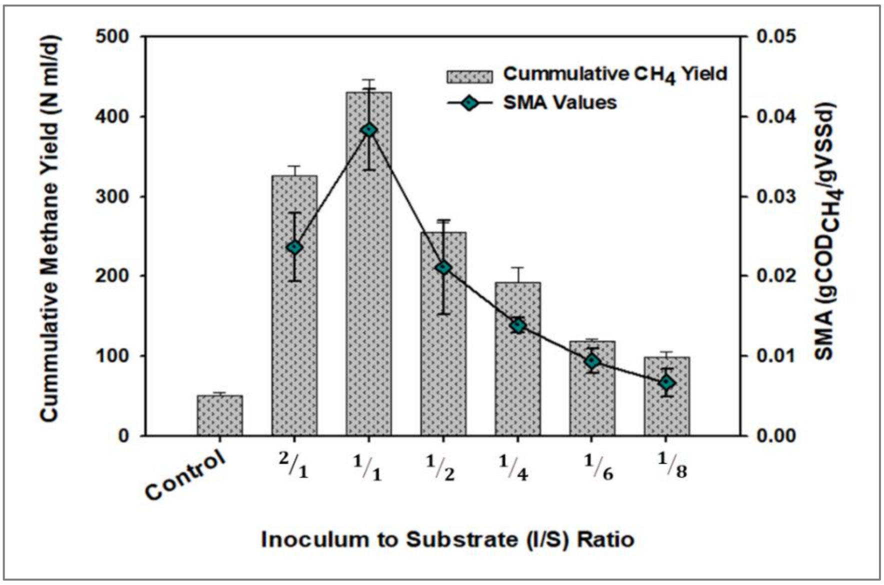


Figure 2.15: A Plot of SMA against cumulative methane yield

2.3.7 Application of SMA Test

Employing Equation 2.11, the maximum SMA value attained for the study (0.039 $gCOD_{CH_4}$/gVSS.d) and the COD load applied to the system (1475.5 kgCOD/d - Table 2.7), the minimum mass of sludge to be retained in the reactor was estimated at 37,833.3 kgTVS. Employing Equation 2.12, the minimum volume to be occupied by sludge during sludge discharge was estimated at 532.34 m^3. Meanwhile, each reactor has a unit volume of 1300 m^3 (Table 2.1). Thus, at least 41% (Figure 2.16) of the UASB reactor volume should be filled with sludge in order not to underfeed the biomass and enhance system performance during the withdrawal of excess sludge.

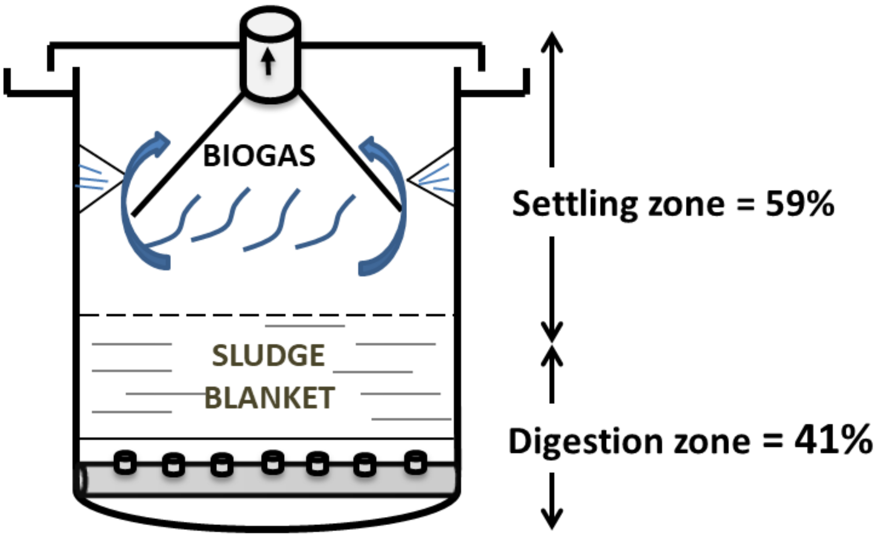


Figure 2.16: Minimum sludge volume to be retained in the digestion zone during sludge discharge

2.4 Partial Conclusion

This research evaluated the performance of 6 full-scale UASB reactors coupled with trickling filters and secondary clarifiers to treat municipal sewage in Accra, Ghana's capital city. The study revealed that the UASB reactors operated at a mesophilic temperature ideal for anaerobic systems. Operating conditions were favourable with HRT of 47.9 ± 11.8 hr, Vel_{up} of 1.0 ± 0.2 m/h, and OLR of 1.22 ± 0.71 kgCOD/m^3/d. Plant performance was satisfactory for organic and solids removal, which were further enhanced by the post-treatment units, ensuring the effluent quality met the regulatory body (EPA Ghana) discharge guidelines for municipal sewage. The Plant, however, failed to remove adequate nutrients (N and P) from the wastewater, with the final effluent containing significant concentrations of N and P compounds. The Poor nutrient removal by the system is ascribed to the absence of a recirculation system at the TFs to enhance the nitrification process, the inadequate 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 average biogas flow was 613 ± 271 Nm^3/d, with an average 65% methane output. SMA test revealed that an ISR of 1:1 resulted in the highest methane production.

Chapter 3:

Carbon Footprints of a Full-scale UASB Reactor coupled with Trickling Filters Treating Municipal Wastewater in Accra, Ghana

Abstract: Wastewater treatment plants (WWTPs) are primarily designed to eliminate water pollution to meet water quality guidelines for environmental and public health protection. However, in recent years, WWTPs have been identified as major sources of anthropogenic GHG emissions. The carbon footprint (CF) is a globally accepted sustainability measure in the wastewater sector for estimating the GHG emissions from WWTPs. This study measured the CFs of a full-scale municipal WWTP based on a UASB reactor configuration with Trickling Filters as post-treatment units. The study employed the IPCC GHG inventory protocol to estimate these emissions. It was found from the study that the GHG emissions from the operations of the Mudor WWTP were totalled at 39,619.36 tCO_2eq/yr. CO_2 emissions from energy consumption were estimated to be 165.74 tCO_2eq/yr, constituting 8.5% of the total emissions. Dissolved methane in the effluent was identified as the single most significant source of GHG emissions with over 90% contribution at 37,676.67 tCO_2eq/yr. Total specific emission intensity was determined to be 26.49 $kgCO_2eq/m^3$ when dissolved methane was considered and 1.29 $kgCO_2eq/m^3$ when this factor was excluded. Resource recovery from wastewater effluent, biogas and biosolids proved to be the surest way by which carbon offsets could be attained.

Keywords: Carbon footprints; Climate change; GHG emissions; Global warming; IPCC GHG inventories; Trickling filter; WWTPs; UASB reactor

3.1 Introduction

Global warming has become a crucial challenge for the human society, which may result in a rise in average earth temperature if no urgent mitigation measures are considered. This could result in irreparable damage to the most vulnerable societies and fragile ecosystems (IPCC, 2018). The Intergovernmental Panel for Climate Change (IPCC) consistently publicise the dangers associated with the rising mean earth temperature and has forecasted future impacts of this menace (IPCC, 2018). Greenhouse gas (GHG) emissions from anthropogenic sources are major contributors to global warming, leading to climate change (IPCC, 2014). Meanwhile, climate change brings about devasting effects such as changes in rainfall patterns, rising seawater levels and extreme weather conditions (IPCC, 2013).

Growing concerns about the causes and impacts of climate change have led governments worldwide to launch policies and strategies which aim to control GHG emissions. A primary step towards effective GHG emission reduction is identifying and quantifying emission sources (Ascui & Lovell, 2012; Stechemesser & Guenther, 2012). Due to this, many establishments consistently take measures towards this direction as part of their corporate sustainability objectives, often referred to as an organisation's carbon accountability.

Primarily, wastewater treatment plants (WWTPs) are designed to eliminate wastewater contaminants to mitigate water pollution, ensuring public health and protecting the environment. In the past, utmost efforts were focused on improving WWTPs to enhance effluent quality (Zhang, 2016). However, nowadays, novel approaches are being developed towards the realisation of sustainability in WWTPs. Sustainability is a comprehensive concept in which an entity's socio-economic and environmental facets are considered. The wastewater treatment industry has in recent times been identified as a potential source of GHG emissions, responsible for an estimated 3% of the global anthropogenic GHG emissions, and second in position after landfills in the waste sector (Bogner et al., 2008; Delre et al., 2019; Maktabifard et al., 2020). Although GHG emissions from WWTPs are not high compared to the other emission sources, evaluation of these emissions is of interest due to environmental impact concerns. WWTPs GHG emissions can be classified as direct or indirect emissions. Direct emissions include carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) emissions on-site, whilst indirect emissions refer to emissions from grid electricity, transport and chemical usage at the WWTPs. Direct CO_2 emissions are considered biogenic; thus, they form part of the natural carbon cycle and are excluded from the GHG emissions inventory (IPCC, 2019). A WWTP's contribution to climate change is determined by evaluating the Plant's carbon footprint (CF). Thus, reducing GHG emissions and, subsequently, the CF could be one key factor in attaining sustainability in WWTPs, leading to better management of the Plant (Delre et al., 2019; Sweetapple et al., 2015).

Evaluation of CF is beneficial at two stages of the footprint reduction policy: at the initial stages, it indicates the sources with the most significant environmental impacts, which informs the selection of an appropriate CF reduction strategy. During the final stages, CF evaluation authenticates the success in CF reduction (EEA, 2014; Szatyłowicz et al., 2021). Moreover, the determination of CF will enable stakeholders of WWTPs to know the contribution of these facilities' emissions to global anthropogenic GHGs emissions (EEA, 2014).

Nowadays, there has been high motivation to reduce energy consumption and improve the efficiency of WWTPs. Subsequently, much attention has been drawn to CF reduction (Hertwich & Peters, 2009). The relevant energy sources (such as electricity, chemicals, heat and fossil-based fuels) and GHG emissions; CO_2, CH_4 and N_2O, are usually considered in CF evaluation. These gases form part of the GHGs targeted to be mitigated under the Kyoto Protocol and have been reported in the GHG inventory (Yoshida et al., 2014). A GHG inventory considers emissions from both direct and indirect sources under the GHG protocol classification (IPCC, 2006a). A comprehensive GHG inventory preparation is required to measure the CF of a WWTP.

Two main methods exist for assessing GHG emissions in WWTPs: the dynamic/static and emission factors approach (Koutsou et al., 2018). Dynamic/static approach has been reported to include complex reaction mechanisms making model applicability challenging (Mannina et al., 2019). The emission factors approach is reportedly widely employed due to operational simplicity. Various organisations have reported emission factors, including IPCC (IPCC, 2006a), Briddle (Briddle Consulting, 2007), USEPA (USEPA, 2014), and Danish Centre for Environment and Energy (Thomsen, 2016), among others. However, the IPCC emission factor method is the most implemented among these methods. The IPCC methodologies are based on the United Nations Framework Convention on Climate Change (UNFCCC) guidelines for implementing sustainable development mechanisms. The 2006 IPCC guidelines for national GHG inventories have been refined to the recent 2019 version (IPCC, 2006a, 2019).

As part of its commitment to the Paris Agreement, Ghana has committed to unconditionally reduce its GHG emissions by 15% by 2030. Being the capital city, the Accra Metropolitan Assembly (AMA) has developed its first five-year plan from 2020 to 2025–the Accra Climate

Action Plan (CAP), which builds on national climate policies; a pathway for lower emissions, carbon neutrality and climate-resilience by 2050 towards the attainment of Ghana's SDG on climate change (Accra Climate Action Plan, 2020). According to the report, in 2015, GHGs emitted in Accra by sector were estimated at 26% for Stationary Energy, and 30% for Transportation, with Waste leading by 44%. Wastewater treatment and discharge contributed 14% of the total emissions, and 32% within the waste sector in the capital city (Accra Climate Action Plan, 2020).

On the national scale, Ghana's fourth national GHG inventory report to the UNFCCC mentioned that the waste sector was the second largest source of methane emissions, contributing 38% of the national methane emissions as of 2016. The waste sector was again identified to be the third leading source of GHG emissions, constituting 8% of the overall national GHG emissions. The report again stated that wastewater treatment and discharge was the highest source of emissions within the waste sector, with a 58% contribution (EPA, 2021). In order to estimate the wastewater emissions in the report, data was obtained from national and international databases such as the World Bank, Ghana Statistical Service, and National Environmental Sanitation Strategy and Action Plan, coupled with interpolations and extrapolations, which could result in some levels of uncertainties. Another research gap observed is that only a few studies have reported on the CFs of full-scale WWTPs in Ghana; thus, achieving environmental sustainability has not been prioritised in the Ghanaian wastewater treatment industry. At a time when the government of Ghana aims to expand the population served by sewage treatment plants, which has led to the construction of full-scale WWTPs in some regions across the country, it will be prudent for such studies to be carried out to identify the primary sources of emissions. This would direct the formulation of measures to help mitigate emissions from these sources towards the attainment of sustainable wastewater management in the country.

This study, therefore, seeks to employ facility-specific data available to assess the GHG emissions of a full-scale UASB reactor coupled with trickling filters as post-treatment units treating municipal wastewater in some suburbs of Accra, the capital city of Ghana. Results obtained from this study will help mitigate the current limitations by i) Estimating the total GHG emissions from the operations of a full-scale UASB-based WWTP; ii) Identifying the major sources of emissions from the UASB-based WWTP; iii) Identifying potential measures to mitigate GHG emissions to improve the environmental performance of UASB-based wastewater treatment systems. Additionally, this study should be useful to policymakers, stakeholders, environmental professionals and researchers who seek to manage GHG emissions from WWTPs in a sustainable manner.

3.2 Materials and Methods

3.2.1 Description of the Wastewater Treatment Plant

This study was conducted at the Mudor WWTP in Accra, Ghana's capital. Ahmed et al. (2018) reported that the Plant was built in 2000, and after operating for a few years, it was shut down due to poor maintenance culture and a lack of financial commitment. It was, however, rehabilitated, expanded and became operational again in the year 2017. The Plant receives and treats municipal sewage from offices, households and business centres within the Accra central business district (CBD) and its environs connected to a sewer network, and is projected to serve roughly 100,000 inhabitants. The Mudor WWTP consists of six (6) modular-shaped UASB reactors, with three (3) trickling filters (TFs) and two (2) clarifiers which act as post-treatment units to the UASB reactor effluent. A detailed description of the treatment plant has been discussed in Chapter 2 (sub-section 2.2.2). Figure 3.1 presents the synoptic view of the Mudor WWTP.

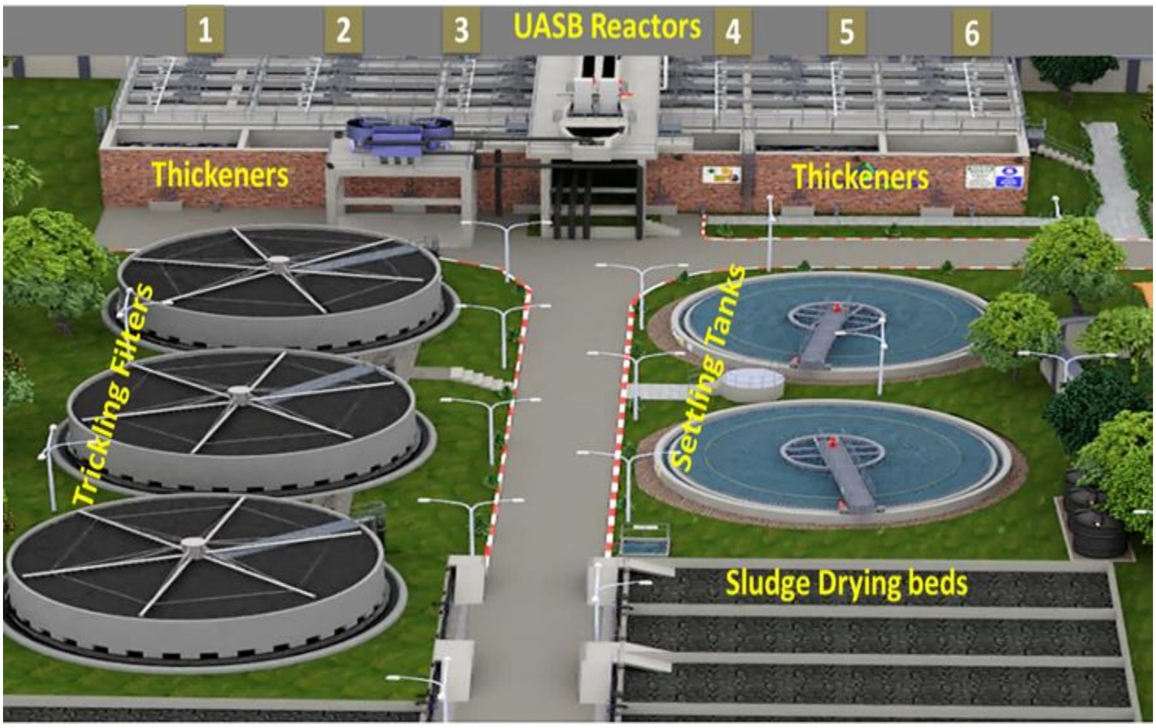


Figure 3.1: Synoptic view of the Mudor WWTP

3.2.2 Inventory Methodology, GHG Protocol and System Boundaries

Tiers 1 and 2 IPCC methodologies have been used to estimate GHG emissions depending on data availability. Tier 1 was employed when only IPCC default parameters (emission factors and activity data) were available. Tier 2 was used when country-specific or facility-specific parameters existed or could be attained (IPCC, 2019). The GHG protocol has established three emission scopes to execute GHG inventories. Scope 1 emissions apply to the direct emissions from an organisation's process; scope 2 emissions apply to the indirect emissions as a result of the consumption of energy, whilst scope 3 refers to emissions from activities such as transportation, chemical usage and other activities over which the organisation has no control over the source. Based on the GHG protocol, a system boundary was defined for this study (IPCC, 2019).

A precise system boundary is essential for accurate CF calculations. This study considered on-site emissions (scope 1), such as direct CH_4 emissions from the anaerobic wastewater treatment without energy recovery. Emissions from this scenario were compared to emissions from biogas flaring to evaluate how much emissions are avoided due to the flaring of biogas at the Mudor WWTP. The study again considered emissions due to CH_4 leakages from the anaerobic reactors, CH_4 emissions from sludge treatment by the drying beds, N_2O emissions from nitrification and denitrification processes at the TFs and CO_2 emissions from the combustion of diesel fuel for power supply during interruptions in national grid electricity under direct on-site emissions. Off-site emissions such as CO_2 emissions from national grid electricity use, emissions from dCH_4 in wastewater effluent discharge and N_2O emissions from recipient water bodies were likewise considered. The study has not considered life cycle analysis, hence GHG emissions during the construction phase of the WWTP, construction of upstream infrastructure such as sewer lines, wastewater methane emissions from sewer lines, acquisition of equipment, operation and management services, and scope 3 emissions such as the use of chemicals and all forms of transportation have not been considered due to data unavailability.

Directives given by the IPCC guidelines indicate that CO_2 generated during the biological treatment process (non-fossil CO_2 emissions) from the anaerobic wastewater treatment and the aerobic biodegradation at the TFs, and CO_2 from the combustion of CH_4 are not considered as they are biogenic, forming part of the natural carbon cycle. Hence these were not considered for this study. N_2O emissions from the anaerobic treatment have likewise not been considered in this study as they are in a minute, hence negligible concentrations (IPCC, 2019). Emissions from residual organic matter in wastewater effluent discharged into water bodies have also not been considered, as the organic effluent concentration falls within the acceptable limits given by EPA Ghana for discharging effluent into water bodies (Chapter 2; Table 2.6). Figure 3.2 presents the system boundary for the scope of emissions considered in the study.

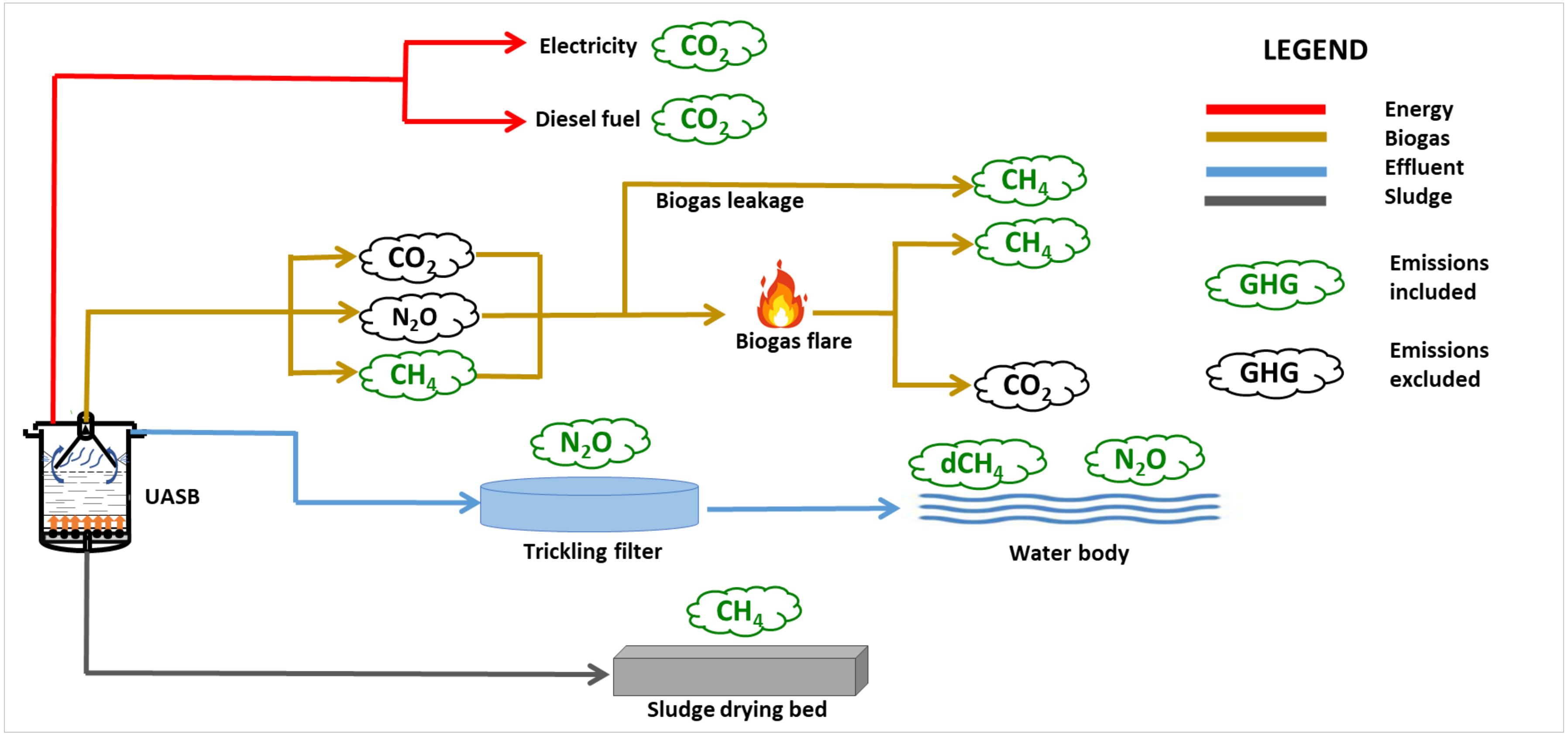


Figure 3.2: System boundary considered for the study

3.2.3 Estimation of GHG Emissions from WWTPs

3.2.3.1 On-site Emissions

The methodology employed for this study is based on the recommended 2006 IPCC guidelines, and where relevant, the updated 2019 version was used. Calculations for corresponding CO_2eq were performed employing a GWP of 1 CO_2eq for CO_2, 28 CO_2eq for CH_4 and 298 CO_2eq for N_2O over a 100-year time horizon as reported in the Fifth IPCC Assessment Report (IPCC, 2014). On-site CH_4 emissions considered emissions from incomplete combustion of CH_4 during biogas flaring, emissions from leakages of CH_4 in the system and emissions from sludge dehydration on drying beds. On-site CO_2 emissions were considered for the fossil fuel combustion, whilst on-site N_2O emissions from biological nutrient removal processes at the TFs were also considered. The total on-site emissions were calculated as follows:

$$GHG_{on-site} = GHG_{flare} + GHG_{CH_4-leakage} + GHG_{sludge-CH_4} + GHG_{diesel} + GHG_{N_2O-WWT} \ldots\ldots\ldots\ldots \quad \textit{(Eqn. 3.1)}$$

- *Estimation of Methane Emissions from UASB Reactors without Energy Recovery or Biogas Flaring*

Detailed descriptions of the quantification and characterization of biogas produced by the Mudor UASB reactors have been provided in Chapter 2 (sub-section 2.2.3) to estimate the CH_4 emissions from biological anaerobic wastewater treatment. Raw sewage and UASB effluent were analysed for COD concentrations, and the removal efficiency was estimated using the Equation:

$$E\ (\%) = \frac{C_i - C_e}{C_i} * 100 \ldots\ldots\ldots\ldots\ldots \quad \textit{(Eqn. 3.2)}$$

Where:

E = Removal efficiency (%)

C_i = Influent COD concentration (mg/L)

C_e = Effluent COD concentration (mg/L)

The refined IPCC (2019) methodology was employed to estimate the methane emissions during wastewater treatment *($E_{CH_4\text{-}WWT}$)* from the anaerobic reactors using the Equation:

$$E_{CH_4-WWT} = \left[((TOW - S_{COD}) *\ EF - R) *\ 10^{-3})) * GWP_{CH_4}\right] \ldots\ldots\ldots\ldots\ldots \quad \textit{(Eqn. 3.3)}$$

Where:

$E_{CH_4\text{-}WWT}$ = Methane emissions from anaerobic treatment in the inventory year (tCO_2eq/yr)
TOW = Total organic load in wastewater in the inventory year (kgCOD/yr)
S_{COD} = COD mass converted into sludge in the inventory year ($kgCOD_{sludge}/yr$)
EF = Emission factor for UASB reactors ($kgCH_4/kgCOD$)
R = Amount of CH_4 recovered or flared in the inventory year ($kgCH_4/yr$)
10^{-3} = Conversion factor from kg to tonnes
GWP_{CH_4} = The global warming potential of methane (IPCC = 28 CO_2eq)

In order to estimate the *TOW*, Equation 3.4 was applied:

$TOW = Q_{ww} * COD_{inf}$ ……………… *(Eqn. 3.4)*

Where:

Q_{ww} = Volume of wastewater treated in the inventory year (m^3/yr)

COD_{inf} = Total influent COD in the inventory year (kg/m^3)

S_{COD} was estimated by the Equation proposed by Lobato et al. (2012):

$S_{COD} = COD_{rem} * Y_s * K_{TVS-COD}$…………… *(Eqn. 3.5)*

Where:

S_{COD} = COD mass converted to sludge in the inventory year ($kgCOD_{sludge}/yr$)
COD_{rem} = COD mass removed from the system in the inventory year (kg/yr)
Y_s = Sludge yield (0.15 $kgTVS/kgCOD_{rem}$)
$K_{TVS\text{-}COD}$ = Conversion factor (1kg TVS = 1.42 $kgCOD_{sludge}$)

Moreover, the emission factor *(EF)* is given by the Equation:

$EF = B_o * MCF$ ………….. *(Eqn. 3.6)*

Where:

B_o = Maximum methane producing capacity (IPCC 2019 = 0.25 $kgCH_4/kgCOD$)
MCF = Methane correction factor for UASB reactors (IPCC 2019 = 0.8)

R was determined by the Equation (UNFCCC, 2022):

$R = V_{CH_4} * Ƒ_{CH_4} * \rho_{CH_4}$ ………… *(Eqn. 3.7)*

Where:

R	= Amount of CH_4 recovered or flared in the inventory year ($kgCH_4$/yr)
V_{CH_4}	= Volumetric flow of CH_4 in the inventory year (Nm^3/yr)
$Ƒ_{CH_4}$	= Volumetric fraction of methane in biogas (0.65)
ρ_{CH_4}	= Density of CH_4 at STP (0.716 kg/m^3)

Two scenarios were compared, the instance when biogas is not recovered or flared but just emitted into the atmosphere and the instance when biogas is flared, as currently practised at the Mudor WWTP, in order to estimate the avoided emissions due to biogas flaring. Thus, the *R* component in Equation 3.3 will go to zero (0) for the estimation of CH_4 emissions, assuming the biogas was emitted directly into the environment.

- *Estimation of Methane Emissions from UASB Reactors' Biogas Flaring*

The UNFCCC (2006) methodology for project emissions from flaring was employed for the estimation of CH_4 emissions as a result of incomplete combustion of CH_4 gas during biogas flaring as practised currently at the Mudor Plant. The emissions from the open flare were calculated with the following Equation:

$PE_{flare} = GWP_{CH_4} * \sum_{h=1}^{8670} MF_{CH_4} \left(1 - \eta_{flare}\right) * 10^{-3}$ ……… *(Eqn. 3.8)*

Where:

PE_{flare}	= Emissions from the flaring of residual biogas in the inventory year (tCO_2eq/yr)
GWP_{CH_4}	= The global warming potential of methane (IPCC = 28 CO_2eq)
$\eta flare$	= Flare efficiency (UNFCCC default value for open flare = 50%)
8670	= Number of hours in a year
10^{-3}	= Conversion factor from kg to tonnes
MF_{CH_4}	= Mass flow of methane in the residual gas per hour (kg/hr)

The MF_{CH_4} was estimated with the Equation (UNFCCC, 2022);

$MF_{CH_4} = Flow_{CH_4} * Ƒ_{CH_4} * \rho_{CH_4}$ …………… *(Eqn. 3.9)*

Where:

$Flow_{CH_4}$ = Volumetric flow of CH_4 (m^3/hr)

F_{CH_4} = Volumetric fraction of CH_4 in biogas (0.65)

ρ_{CH_4} = Density of CH_4 (0.716 kg/m^3)

- *Estimation of Emissions due to Methane Leakages*

The UNFCCC indicates that project emissions from anaerobic reactors such as the UASB could be attributed to physical leakages of CH_4 through the side walls and roofs, gas hoods of the reactor, biogas lines during maintenance activities and release from safety valves as a result of excessive pressure build-up in the reactors. Hence the UNFCCC proposed methodology (UNFCCC, 2017b) was employed to estimate the GHG emissions from CH_4 leakages from the anaerobic reactors. The emissions are calculated by the following Equation:

$$PE_{CH_4} = Q_{CH_4} * EF_{CH_4,default} * GWP_{CH_4} \quad \ldots\ldots\ldots\ldots \textit{(Eqn. 3.10)}$$

Where:

PE_{CH_4} = Project emissions from CH_4 leakages in the year (tCO_2eq/yr)

Q_{CH_4} = Quantity of CH_4 generated by the anaerobic reactor in the year (tCH_4)

$EF_{CH_4,default}$ = Default emission factor for the fraction of CH_4 produced that leaks from the UASB reactor (= 0.05 tCH_4leaked/tCH_4produced)

GWP_{CH_4} = The global warming potential of methane (IPCC = 28 CO_2eq)

- *Estimation of Methane Emissions from Sludge Treatment*

To estimate the methane emissions from sludge treatment, the amount of excess sludge withdrawn from the UASB reactors was quantified. Plant operators monitored the system for excess sludge production during the study period. Excess build-up of sludge was defined by observing effluent concentrations for TSS, BOD and COD. Effluent deterioration of these parameters indicated excess sludge build-up in the reactors (Rosa et al., 2012). Sludge discharge ports sited at the sides of the UASB reactors were opened using designated valves, which allowed excess sludge to be discharged first into the sludge thickeners and subsequently onto the sludge drying beds. Plant operators revealed that sludge withdrawal was conducted once every two weeks, with the discharged sludge volume approximately 30% of the volume of the sludge thickener. Based on these projections, the volume of discharged sludge was estimated. Sludge moisture content was analysed, and the dry sludge matter generated for the year was estimated.

Sludge produced from biological wastewater treatment processes contains a significant amount of organic content, which results in methane emissions during decomposition under anaerobic conditions (Begak et al., 2013). The IPCC (2006) methodology indicates that methane emissions from sludge treatment by drying beds for a year can be calculated with the following Equation:

$$E_{CH_4-sludge} = M_{sl,dry} * MCF_{sl} * DOC_{sl,dry} * DOC_F * F_{CH_4} * \frac{16}{12} * GWP_{CH_4} \ldots \ldots \textit{(Eqn. 3.11)}$$

Where:

$E_{CH_4\text{-}sludge}$ = Methane emissions from sludge drying in the inventory year (tCO_2eq/yr)

$M_{sl,dry}$ = Mass of dry sludge matter produced in the inventory year (t/yr)

MCF_{sl} = Methane conversion factor for sludge (IPCC default = 0.5 for drying beds)

$DOC_{sl,dry}$ = Degradable organic content (DOC) in the dry sludge (IPCC default value = 0.5 for domestic sludge)

DOC_F = Fraction of DOC dissimilated to biogas (IPCC default value = 0.5)

F_{CH_4} = Fraction of methane in biogas (IPCC default value = 0.5)

$\frac{16}{12}$ = Ratio of the molar mass of methane to carbon

GWP_{CH_4} = The global warming potential of CH_4 (IPCC = 28 CO_2eq)

- *Emissions from the Combustion of Diesel Fuel*

GHG emissions associated with the combustion of diesel fuel to power generators during interruptions in the national electricity grid are considered on-site emissions per the IPCC guidelines. The CO_2 emission factor for diesel fuel - 0.0741 tCO_2/GJ (IPCC, 2006b), was employed to calculate GHG emissions from diesel combustion at the Plant. The UNFCCC (2017c) methodological tool version 03 was used to calculate emissions from the combustion of diesel fuel using Equation 3.12:

$$PE_{DF} = Q_{DF} * NCV_{DF} * EF_{DF} \ldots \ldots \ldots \quad \textit{(Eqn. 3.12)}$$

Where:

PE_{DF} = CO_2 emissions from the combustion of diesel fuel in the year (tCO_2eq/yr)

Q_{DF} = Quantity of diesel consumed for electricity generation in the year (litres)

NCV_{DF} = Net calorific value of diesel fuel (0.036 GJ/litres)

EF_{DF} = CO_2 emission factor for diesel fuel (0.0741 tCO_2/GJ)

- *N_2O Emissions from Wastewater Treatment*

The IPCC has reported that biological nutrient removal processes which involve nitrification and denitrification can be one primary source of N_2O emissions from WWTPs. The 2019 refined guidelines (IPCC, 2019) for national GHG inventories methodology for N_2O emissions from domestic WWTP was employed using the Equation:

$$GHG_{N_2O} = \sum(U_i * T_j * EF_{N_2O}) * TN_{load} * \frac{44}{28} * 10^{-3} * GWP_{N_2O} \dots\dots\dots \qquad \textit{(Eqn. 3.13)}$$

Where:

GHG_{N_2O} = N_2O emissions from wastewater treatment (tCO_2eq/yr)

EF_{N_2O} = Emission factor for N_2O (IPCC 2019 default value = 0.016 kgN_2O-N/kg N)

TN_{load} = TN load present in the wastewater in the year (kgN/yr)

$\frac{44}{28}$ = Ratio of the molar mass of N_2O to molar mass of N_2

10^{-3} = Conversion from kg to tonnes

GWP_{N_2O} = The global warming potential of N_2O (IPCC = 298 CO_2eq)

U_i = Fraction of population in income group (IPCC 2019, Table 6.5). U_1 represents urban high-income (0.1), and U_2 represents urban low-income (0.38).

T_j = Degree of utilization of the treatment or discharge pathway (sewers), T_1 represents urban high-income (0.37), T_2 represents urban low-income (0.34) and where 3 denotes use of sewers.

Thus, $(U_1*T_{13} + U_2*T_{23}) * EF_{N_2O} = (0.1 * 0.37 + 0.38 * 0.34) * 0.016 = 2.6590 \times 10^{-3}$

3.2.3.2 Off-site Emissions

Off-site emissions were estimated from grid electricity use at the Plant, dCH_4 and N_2O emissions from the discharge of effluent into the recipient water body. Off-site emissions were estimated by the Equation:

$$GHG_{off-site} = GHG_{dCH_4} + GHG_{N_2O-Effluent} + GHG_{electr} \dots\dots\dots \qquad \textit{(Eqn. 3.14)}$$

- *Estimation of GHG Emissions from Dissolved Methane (dCH_4) in Effluent*

Equation 2.1 (Chapter 2, sub-section 2.2.4) was employed to estimate the dCH_4 in wastewater effluent. This study quantified the emissions from dCH_4 employing UNFCCC (2008) methodology. The project activity emissions from dCH_4 in wastewater effluent are given by the following Equation:

$$PE_{dissolved} = Q_{ww} * CH_{4-ww} * GWP_{CH_4} \quad \ldots\ldots\ldots\ldots \quad \textit{(Eqn. 3.15)}$$

Where:

$PE_{dissolved}$ = Emissions from dCH_4 in wastewater effluent in the year (tCO_2eq/yr)

Q_{ww} = Volume of wastewater treated in the year (m^3/yr)

CH_{4-ww} = dCH_4 concentration in wastewater effluent in the year ($tonnes/m^3$)

GWP_{CH_4} = The global warming potential of methane (IPCC = 28 CO_2eq)

- *Estimation of N_2O Emissions from Discharge of Effluent into Water Body*

The IPCC has reported that the discharge of high-nitrogen concentration effluent into recipient water bodies could result in N_2O emissions. However, the quantum of this emission is dependent on the nature of the recipient. Nutrient-impacted water bodies such as eutrophic lakes and rivers and hypoxic or stagnant water bodies tend to have higher N_2O emission factors, whilst non-hypoxic, non-nutrient-impacted or flowing water bodies have lesser N_2O emission factors. Equation 6.7 (updated) in the refined IPCC 2019 guidelines for N_2O emissions from domestic wastewater effluent was employed as follows:

$$GHG_{N_2O-Effluent} = TN_{Effluent} * EF_{N_2O-Effluent} * \frac{44}{28} * 10^{-3} * GWP_{N_2O} \ldots\ldots\ldots \quad \textit{(Eqn. 3.16)}$$

Where:

$GHG_{N_2O-Effluent}$ = N_2O emissions from wastewater effluent discharged into the recipient water body in the year (tCO_2eq/yr)

$TN_{Effluent}$ = TN load in effluent discharged into the recipient water body (kg/yr)

EF_{N_2O} = Emission factor for effluent discharge into non-eutrophic or non-nutrient-impacted aquatic environment (IPCC default value = 0.005 kg N_2O-N/kg N)

$\frac{44}{28}$ = Ratio of the molar mass of N_2O to molar mass of N_2

10^{-3} = Conversion from kg to tonnes

GWP_{N_2O} = The global warming potential of N_2O (IPCC = 298 CO_2eq)

The $TN_{Effluent}$ per the updated IPCC 2019 guidelines is given by the Equation:

$$TN_{Effluent} = \left[\left(TN_{load} * \sum T_j\right) * (1 - N_{Rem})\right] \ldots\ldots\ldots\ldots \qquad \textit{(Eqn. 3.17)}$$

Where:

$TN_{Effluent}$ = Total nitrogen in the effluent wastewater discharged into the recipient water body in the inventory year (kgN/yr)

TN_{load} = TN load present in the wastewater in the inventory year (kgN/yr)

$\sum T_j$ = Degree of utilization of the treatment system, T_1 represents urban high-income (0.37), T_2 represents urban low-income (0.34) and where 3 denotes use of sewers. $\sum T_j = T_{13} + T_{23} = 0.37 + 0.34 = 0.71$

N_{Rem} = Fraction of TN removed during wastewater treatment = 0.3 (Table 2.6)

- *Estimation of GHG Emissions Associated with Grid Electricity Usage*

GHG emissions associated with grid electricity consumption by the Mudor WWTP were estimated based on the electricity consumption and grid electricity emission factor. The country-specific emission factor of 0.479 tCO_2eq/MWh was employed (IGES, 2022). The UNFCCC (2017a) methodology was used to calculate emissions from energy consumption from grid electricity usage using Equation 3.18:

$$E_{electr} = \sum Q_{electr} * EF_{electr} \ldots\ldots\ldots\ldots \qquad \textit{(Eqn. 3.18)}$$

Where:

E_{electr} = GHG emissions associated with electricity consumption in the year (tCO_2eq/yr)

Q_{electr} = Quantity of electricity consumed from the operations of the WWTP in the year (MWh/yr)

EF_{electr} = National grid electricity CO_2 emission factor for Ghana (tCO_2eq/MWh)

3.2.3.3 Avoided Emissions

Avoided emissions resulting from the production of electricity on-site through biogas and sludge energy recovery, thereby eliminating the use of grid electricity for Plant operations, and avoided emissions from the non-production and non-usage of inorganic fertilizers (N and P), which is replaced by use of nutrient-rich wastewater effluent for fertigation purposes have been estimated. The methodology proposed by Rosa et al. (2018) was employed to evaluate the electrical energy recovery potential of sewage sludge and biogas, whilst the method proposed by Robles et al. (2020) was employed to estimate the energy savings from non-use of inorganic N–P-based fertilizers. Avoided emissions were calculated with the Equation proposed by Heffernan et al. (2012):

$$A_e = E_s * EF_e \ldots\ldots\ldots\ldots \qquad \textit{(Eqn. 3.19)}$$

Where:

A_e = The avoided emissions (tCO_2eq/yr)

E_S = Electricity savings from the use of on-site produced electricity or non-use of inorganic fertilizer (MWh/yr)

EF_e = National grid electricity CO_2 emission factor for Ghana (0.479 tCO_2eq/MWh)

3.3 Results and Discussion

3.3.1 On-site Emissions

- *Emissions from Anaerobic Wastewater Treatment without Biogas Flaring*

GHG emissions from anaerobic wastewater treatment were calculated adopting the 2019 IPCC methodology presented in Equation 3.3. Only methane emissions from the biodegradation of organic substrates were considered, as mentioned earlier. COD removal efficiency for the UASB reactors was calculated to be 72% (Chapter 2, Table 2.6). The monthly volumes of wastewater treated and influent COD load received by the UASB reactors have been presented in Table 3.1. *TOW* was estimated to be 2,686,131.73 kgCOD/yr, whilst S_{COD} was estimated to be 1362.35 $kgCOD_{sludge}$/yr. $E_{CH4\text{-}WWT}$ were calculated to be 15,034.71 tCO_2eq/yr. However, it would be erroneous to report this as CH_4 emissions from the anaerobic system as biogas generated at the Plant is flared.

Table 3.1: Monthly wastewater and methane flows

Month	WW flow (m^3/month)	Inf_{COD} (kgCOD/month)	Gaseous CH_4 (Nm^3/month)	dCH_4 (m^3/month)
January	138,155	250,422.60	16,011.17	5367.67
February	121,539	245,759.95	15,027.92	5038.04
March	144,526	259,858.05	15,388.29	5158.85
April	130,734	271,245.38	16,573.76	5556.27
May	140,323	295,542.59	17,554.09	5884.93
June	134,878	198,072.76	13,368.90	4481.86
July	108,236	134,297.52	9809.34	3288.54
August	109,847	201,232.58	10,193.91	3417.46
September	110,253	192,201.10	8855.29	2968.69
October	114,394	237,270.38	6153.15	2062.82
November	124,068	219,797.37	6635.57	2224.54
December	118,153	180,431.46	7607.00	2550.21
Total	1,495,106.00	2,686,131.73	143,178.40	47,999.92

- *Emissions from Anaerobic Wastewater Treatment with Biogas Flaring*

Equation 3.8 was employed to estimate the CH_4 emissions from the UASB reactors when biogas is flared. Biogas from the Mudor UASB reactors is openly flared, and per the UNFCCC methodology for emissions from flaring, a default of 50% was adopted as flare efficiency (ηflare) for open flaring. With the estimated volumetric fraction of CH_4 as 65% (Arthur et al., 2022) and CH_4 density of 0.716 kg/m^3 (IPCC, 2006a), the mass flow of CH_4 was determined to be 7.61 kg/hr. Project emissions from the flaring of residual biogas ***(PE_{flare})*** were calculated to be 932.89 tCO_2eq/yr.

- *Emissions from CH_4 Leakages*

The total volume of CH_4 produced during the inventory year at STP was calculated to be 102.24 tonnes/yr. The GHG emissions from CH_4 leakages were estimated at 143.14 tCO_2eq/yr. Comparing this finding to previous studies, Ashrafi et al. (2013) reported GHG emissions from biogas leakage for an anaerobic reactor treating wastewater to be 545 $kgCO_2$eq/d (198.93 tCO_2eq/yr).

- *Emissions from Sludge Drying Beds*

Methane emissions from drying beds were estimated by Equation 3.11. The total methane emissions from sludge drying were estimated to be 305.10 tCO_2eq/yr.

- *N_2O Emissions from Wastewater Treatment*

The N_2O emissions from nitrification and denitrification processes at the TFs have been estimated. Per the design of the Mudor WWTP, the TFs act as the post-treatment unit for the

UASB reactor effluent; hence the TN load which arrives at the TFs where biological nutrient removal takes place is the TN load present in the UASB reactor effluent. Thus, the TN concentration in the UASB reactor effluent was used to estimate the N_2O emissions during biological nutrient removal at the TFs. The total TN load to the TFs were estimated to be 165,444.70 kg/yr, using an average TN concentration of 0.121 kg/m^3 (Chapter 2, Table 2.6) and assuming approximately 90% of the influent UASB wastewater volume is discharged as effluent to the post-treatment unit. The total N_2O emissions from biological nutrient removal at the TFs were calculated to be 206.02 tCO_2eq/yr.

- *Emissions from Diesel Combustion*

Total diesel fuel consumed by generators to run the Plant during interruptions in grid electricity supply was 9000.00 litres/yr, transmitting into 96.77 MWh/yr. CO_2 emissions from this project activity were estimated to be 24.00 tCO_2eq/yr.

Total on-site emissions from the operations of the Mudor WWTP during the study period were estimated with Equation 3.1 to be 1608.56 tCO_2eq/yr. The monthly variations have been presented in Figure 3.3. The month of May reported the highest on-site emissions. This could be attributed to the highest emissions from flaring (111.92 tCO_2eq/yr) observed during that period. This observation is buttressed by the fact that the same month recorded the highest CH_4 flow (Table 3.1); hence, the high emissions from methane flaring could have resulted in this observation. The least on-site emissions were, however, observed in October.

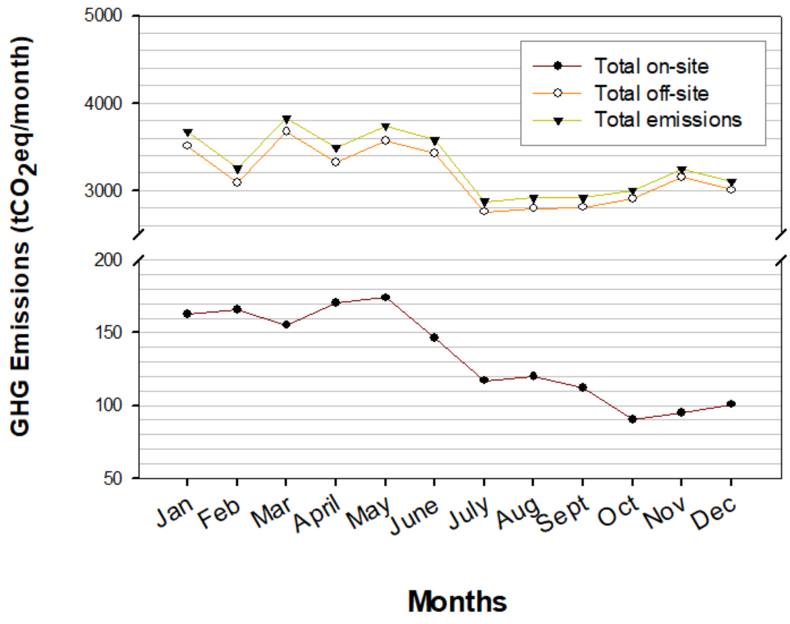


Figure 3.3: Monthly on-site and off-site GHG emissions

3.3.2 Off-site Emissions

- *Emissions from Grid Electricity Usage*

The total grid electricity consumed for the inventory year was 295.892 MWh/year. Indirect CO_2 emissions from grid electricity consumption were determined to be 141.7 tCO_2eq/yr, employing Equation 3.18.

- *Emissions from dCH_4 in Effluent*

It was observed from the study that 23% of CH_4 (21 mg/L) generated remained dissolved in the effluent (Chapter 2, sub-section 2.3.4.3). Souza et al. (2011) reported values between 18 to 23 mg/L for UASB reactors treating domestic wastewater, comparable to the findings of this study. From Equation 3.15, GHG emissions from dCH_4 in wastewater effluent were calculated to be 37,676.67 tCO_2eq/yr. This value constituted 95.1% of the total emissions from the Plant for the year under study (Figure 3.5). The large emissions from this source were undeniably the cause of the high off-site emissions observed in this study. Comparing this finding to previous studies, Heffernan et al. (2012) reported that 23% of CH_4 generated remained in solution at a concentration of 19 mg/L for a UASB-Activated sludge configuration used in their study. The authors further opined that the loss of this methane accounted for 78% (20,000 tCO_2eq/yr) of the total GHG emissions from their system. Robles et al. (2020) likewise observed in their study that dCH_4 was the main contributor to GHG emissions from the operation of the anaerobic membrane bioreactor (AnMBR). However, after recovery by degassing membranes, the GHG emissions from the treatment system were noticeably reduced. These observations from previous studies are comparable to the findings of this research.

- *N_2O Emissions from Effluent Nitrogen Discharged into Recipient Water Body*

Nitrous oxide (N_2O) emissions can be indirect emissions from effluent discharge into waterways (IPCC, 2019). Indirect N_2O emissions from effluent discharge were estimated using Equation 3.16. At an estimated average TN load of 165,444.70 kgN/yr and TN removal fraction of 0.3, $TN_{Effluent}$ was estimated to be 81,051.36 kgN/yr, and $GHG_{N_2O\text{-}Effluent}$ was determined to be 189.78 tCO_2eq/yr.

Total off-site emissions from the operations of the Mudor WWTP during the study period were estimated with Equation 3.14 to be 38,007.82 tCO_2eq/yr. The monthly variations have been presented in Figure 3.3. Due to the highly significant contribution of dCH_4 to the off-site emissions, the month of March, which recorded the highest sewage flow (Table 3.1), correspondingly recorded the highest off-site emissions for the inventory year at 3672.27 tCO_2eq/yr. The least off-site emissions were, however, observed in July.

3.3.3 Avoided Emissions

Avoided emissions from on-site potential electricity production were estimated from the yearly methane production, the net calorific value of methane, yearly dry sludge matter production, and the net calorific value of sludge. The estimations were 427.19 MWh/yr and 106.89 MWh/yr for biogas and sludge energy recovery potentials, respectively. The avoided emissions were determined to be 204.62 tCO_2eq/yr and 51.20 tCO_2eq/yr, respectively, for biogas and sludge. Maktabifard et al. (2020) stated that the production of biogas had both positive and negative impacts on CF. The positive effect is ascribed to the energy recovery from biogas, which reduces indirect emissions related to electricity supplied from the grid. The authors attributed the negative effect to biogas leakages from the reactor and incomplete combustion, which results in fugitive emissions and causes a positive greenhouse effect.

The high concentration of nutrients (TN = 0.0836 kg/m^3, TP = 0.02838 kg/m^3; Chapter 2, Table 2.6), coupled with low heavy metals concentrations (Chapter 4, Table 4.3) in the final effluent permits the use of effluent for fertigation purposes (water and nutrient simultaneous reuse), an attractive approach for resource recovery. Fertigation promotes the conservation of freshwater resources whilst reducing energy consumption for ammonia-based fertilizer production (19.3 kWh/kg of N produced by Haber-Bosh process, McCarty et al. 2011), and extraction of

Phosphorus (2.1 kWh/kg of P, Gellings & Parmenter, 2004). Employing the effluent concentrations of TN and TP, the energy savings consequent from using wastewater effluent instead of inorganic fertilizer will be 1.6137 kWh/m^3 for TN and 0.05959 kWh/m^3 for TP, culminating in 1.6733 kWh/m^3. Considering the total volume of wastewater treated per year, a total of 2251.45 MWh of energy will be saved from non-usage of inorganic fertilizer. This corresponds to an avoided GHG emissions of 1078.44 tCO_2eq/yr. Total avoided emissions from biogas and sludge energy, and nutrients recovery were estimated to be 1334.26 tCO_2eq/yr.

Figure 3.4a presents the GHG emissions from the respective sources considered in this study and the avoided emissions (GHG emissions offset) applicable under resource recovery for sustainable wastewater management. As presented in Figure 3.4b, the total emissions were estimated to be 39,619.36 tCO_2eq/yr, considering offset emissions of 1334.26 tCO_2eq/yr, the net GHG emissions were determined to be 38,285.10 tCO_2eq/yr.

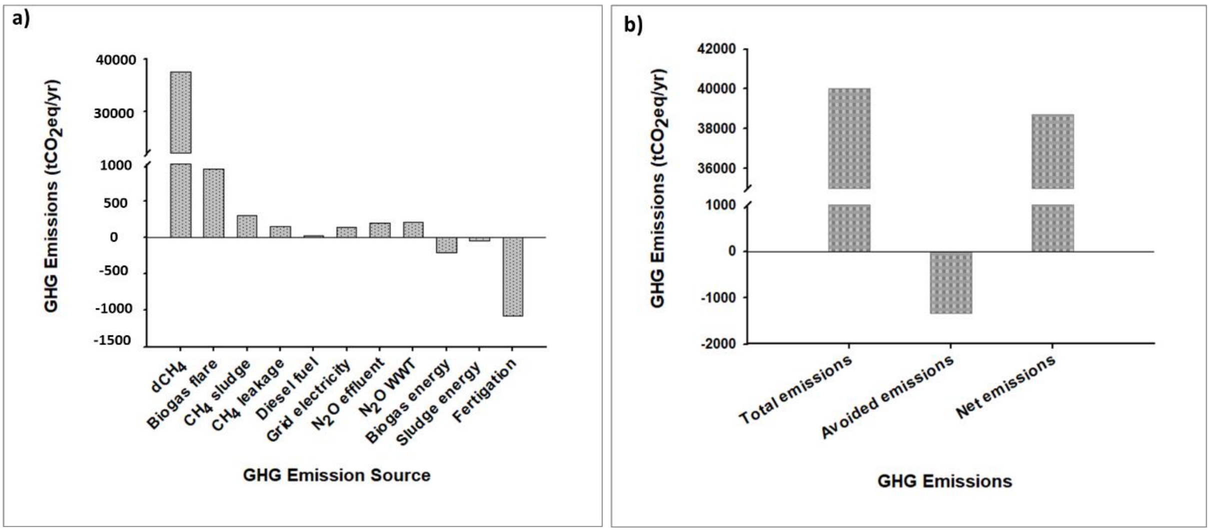


Figure 3.4: GHG emissions and avoided emissions

3.3.4 Overview of GHG Emissions from the Operations of the Mudor WWTP

Regarding CH_4 emissions due to anaerobic wastewater treatment by the UASB reactors, it was found that the total emissions for a scenario devoid of energy recovery or biogas flaring, with biogas released directly into the atmosphere, amounted to 15,034.71 tCO_2eq/yr. This value, compared to the 932.89 tCO_2eq/yr obtained due to biogas flaring at the Plant shows that biogas flaring reduced the CH_4 emissions by ≈ 94%, emitting only 6% of the actual. Thus, it can be concluded that in the absence of a biogas energy recovery system, biogas flaring presents the most sustainable alternative to manage biogas generated by anaerobic WWTPs.

Additionally, it was observed from the study that for the various on-site and off-site emissions considered, dCH_4 was responsible for 95.1% of the total emissions (Figure 3.5), whilst the remaining sources were highly insignificant to be compared. CH_4 is 28 times more potent than CO_2, coupled with the climate change crises the world is currently facing, this observation is disquieting and requires immediate intervention. Besides, the loss of this CH_4 significantly reduces the biogas energy recovery potential of the Mudor UASB reactors.

Excluding dCH_4, the percentage distributions of the remaining GHG emission sources have been illustrated in Figure 3.5. CH_4 emissions from biogas flaring contributed 48%; the highest source of emissions. The CH_4 emissions amounted to 1381.13 tCO_2eq/yr, constituting 71% of the total emissions. In a parallel study, Santos et al. (2015) asserted that for a full-scale WWTP in the Bahia state of Brazil, CH_4 emissions were estimated at 865,462.92 tCO_2eq, constituting about 90% of the total emissions considered. The same authors reported that BOD removal by anaerobic wastewater treatment was responsible for 448,858.84 tCO_2eq (≈ 52%) of the reported CH_4 emissions, whilst emissions from residual BOD discharged into the environment were estimated at 46,749.70 tCO_2eq (≈ 5.4%). The present study did not consider CH_4 emissions from residual organic matter in wastewater effluent; this is ascribed to the fact that the final effluent discharged into the receiving water body meets discharge guidelines for organic effluent concentration set by EPA Ghana (Chapter 2, Table 2.6).

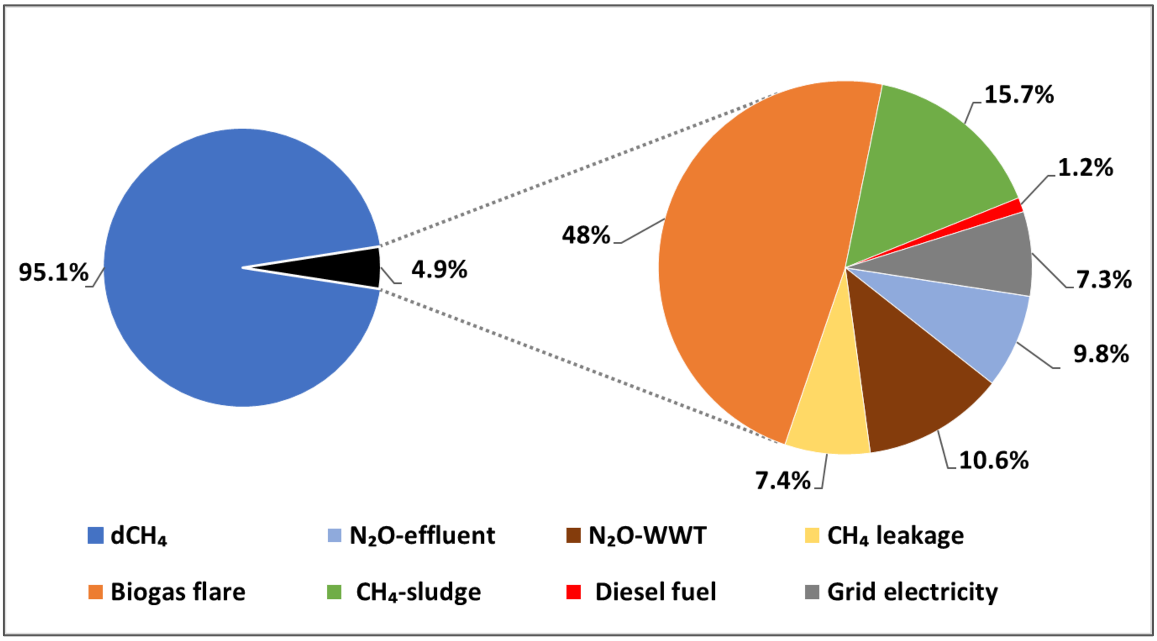


Figure 3.5: Percentage distribution of GHG emission sources

The most negligible emissions were observed for diesel fuel combustion, estimated at 24 tCO_2eq/yr, constituting only 1.2% of the total emissions. The combined energy emissions were found to be 8.5% of the total emissions, which unarguably is on the lower side. Some studies have reported that energy consumption was among the highest emission sources. For instance, Bao et al. (2016) recorded in their study that 38 - 50% of the GHG emissions in WWTPs were attributed to the carbon emissions from energy consumption. These observations are coherent with aerobic treatment systems notable for high energy consumption. Contrarily, anaerobic wastewater treatment systems consume lesser energy; hence it comes as no surprise that GHG emissions from energy consumption for this study were lower. Additionally, the very low emissions from energy consumption by the Mudor Plant, consequent of the lower energy demand of the Plant (Chapter 4, Figure 4.3), is because the Plant is designed such that gravity drives most material flow, hence energy consumption by pumps is very minimal (Arthur et al., 2022).

Maktabifard et al. (2020) investigated the CFs of six different full-scale WWTPs and mentioned that direct emissions from wastewater treatment constituted the largest share of the total GHG emissions (62 - 74%), this was followed by energy consumption (1 - 23%) and biogas production (8 - 30%). In a similar study, Gustavsson & Tumlin (2013) reported that energy consumption and the use of chemicals were among the significant contributors to GHG emissions for WWTPs. However, a different observation was made for the Mudor UASB reactors. First, the Mudor Plant does not use chemicals such as polymers for dewatering and chlorine to disinfect treated sewage. The only instance of chemical usage is the use of detergents for cleaning purposes, which falls outside the system boundary of this study.

N_2O emissions from biological nutrient removal (BNR) have been reported to be a major emission source for WWTPs and could be responsible for up to half of WWTPs emissions (Maktabifard et al., 2019). The findings from this study, however, contradict this assertion. N_2O emissions from BNR at the TFs of the Mudor WWTP accounted for only 10.6% of the total emissions, which could be attributed to the poor N removal at the TFs. N_2O emissions from effluent discharge in the recipient water body likewise accounted for just 9.8% of the total emissions. The lower N_2O emissions estimated from the discharge of a high-nitrogen concentration effluent can be ascribed to the fact that the effluent is discharged into the Korle Lagoon, a non-eutrophic, shallow and flowing water body, which meets the criteria for water bodies with lower emission factors according to the IPCC refined 2019 guidelines. Law et al. (2012) mentioned that N_2O emissions from WWTPs are dependent on Plant size and the deployed treatment process. According to the authors, the mechanisms surrounding N_2O

formation and emissions in WWTPs are a combined effect of physical and biological processes which are complex and intrinsically variable. The actual process emission rates for N_2O are more likely to vary, depending on a variety of physical (temporal and spatial) factors related to the size, design, loading and operation of a WWTP. These variabilities tend to confound N_2O emissions estimates based typically on a single variable, such as the influent TN load (Daelman et al., 2015; Pan et al., 2016; Valkova et al., 2021).

3.3.5 Specific Carbon Footprint Indicators

Carbon footprint indicators, also known as emission intensities (EIs), allow for comparing the CFs of WWTPs for different studies using a standard unit (Maktabifard et al., 2020). The most popular specific CF indicators used are CF based on the volume of wastewater treated (CF_V) and CF based on population equivalent (CF_{PE}). The population equivalent (PE) for the Mudor WWTP was estimated to be 157,392.06 PE (Chapter 2, sub-section 2.2.2). The study revealed that CF_V ranged from 0.016 kgCO_2eq/m^3, which was the least observed for diesel fuel, to 25.20 kgCO_2eq/m^3 observed for dCH_4. The total EIs were estimated at 26.49 kgCO_2eq/m^3. EI for all emission sources excluding dCH_4 was found to be 1.29 kgCO_2eq/m^3, whilst CH_4 emissions amounted to 0.92 kgCO_2eq/m^3 (Figure 3.6). EI for N_2O emissions was calculated to be 0.26 kgCO_2eq/m^3, whilst that for CO_2 emissions was estimated at 0.11 kgCO_2eq/m^3. Considering the EIs in terms of PE, it was found that the total EIs were 251.71 kgCO_2eq/PE, ranging from 0.153 kgCO_2eq/PE for diesel fuel to 239.38 kgCO_2eq/PE for dCH_4. Excluding emissions from dCH_4, it totalled at 12.32 kgCO_2eq/PE, for which CH_4, CO_2 and N_2O emissions were 8.76, 1.05 and 2.51 kgCO_2eq/PE, respectively.

Comparing these findings to the literature, Maktabifard et al. (2020) reported CF_V and CF_{PE} ranging from 0.6 - 1.7 kgCO_2eq/m^3 and 25.8 - 99.7 kgCO_2eq/PE, respectively, in their studies. In other related studies, Gustavsson & Tumlin (2013) and Mamais et al. (2015) reported ranges between 7 and 161 kgCO_2eq/PE. Other studies have reported CFv to range from 0.1 - 2.4 kgCO_2eq/m^3 (Li et al., 2017; Wang et al., 2016). Wang et al. (2016) studied the EIs for wastewater facilities in different countries and explained that CF_V was influenced by discharge limits or desired effluent quality and the variations in emission factors employed.

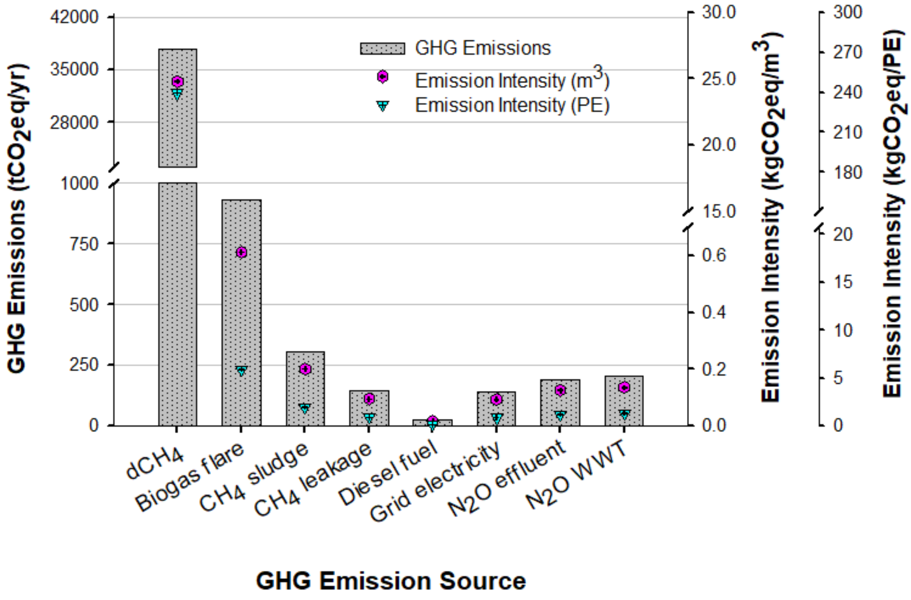


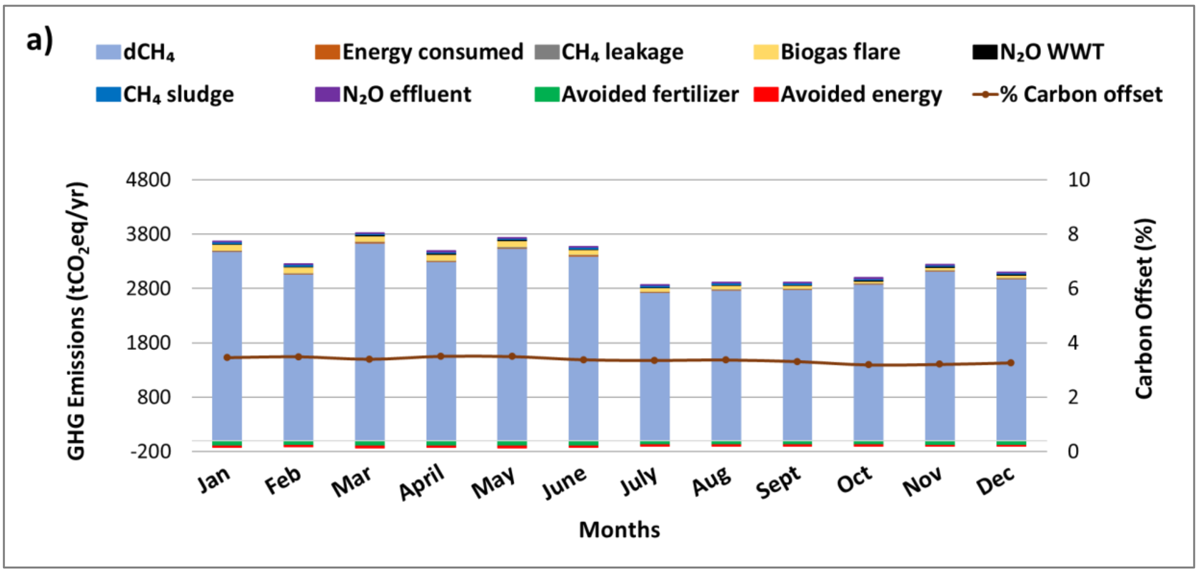
Figure 3.6: Emission intensity

In the study by Mannina et al. (2019), the authors developed a model to estimate CFs in WWTPs. They reported specific CF_V for direct and indirect emissions between 0.18 - 1.18 $kgCO_2eq/m^3$. Xi et al. (2021) likewise reported that EI ranged from 0.268 - 0.738 $kgCO_2eq/m^3$ for 50 municipal WWTPs in Shanghai, China. According to these authors, several factors influenced the EIs of WWTPs, such as the treatment processes employed. Among the various treatment technologies they considered, they found that the membrane bioreactor (MBR) obtained the highest EI at 0.738 $kgCO_2eq/m^3$ compared to other processes, which they attributed to the high energy consumption of MBRs. The authors again found that other factors such as desired influent quality, discharge limits, Plant scale and loading rates all influenced the EIs of WWTPs. It is indisputable that the emissions from dCH_4 accounted for in this study resulted in the extremely high EIs obtained; however, when this factor was excluded, values attained were within the range of values reported by other authors.

3.3.6 Carbon Offsets

It is worth noting that this study did not consider energy recovery and avoided emissions from the recovery of dCH_4. Undeniably the single significant source of GHG emissions, as revealed from the study, whose recovery would likely neutralize carbon emissions from the Plant. Recovery of dCH_4 in anaerobic wastewater effluent is a novel technology which has not yet been thoroughly studied and understood in most developing countries. However, biogas and sludge energy recovery are gradually advancing in most developing countries (Lopes et al., 2019; Patinvoh & Taherzadeh, 2019). Additionally, using reclaimed water for urban irrigation is a prevalent practice in most developing countries, including Ghana (Ait-Mouheb et al., 2018; Amponsah et al., 2016; Helmecke et al., 2020). Hence these have been considered in the avoided emissions.

Notwithstanding, Jiménez-Benítez et al. (2020) found that energy production from biogas and dCH_4 decreased their system's carbon footprints by reducing the energy demand. Figure 3.7a illustrates the monthly emissions considering dCH_4. It was observed that the high emissions resulting from dCH_4 caused the avoided emissions to be insignificant. The percentage of carbon offset was found to range from 3.1 - 3.5%. However, presented in Figure 3.7b is the emissions and avoided emissions for the various months excluding dCH_4. It was found that avoided emissions from the non-use of inorganic fertilizer and energy recovery from biogas and sludge could result in a 59.2 - 86.0% carbon offset. Thus, for an anaerobic/aerobic system such as the Mudor WWTP, resource recovery is the surest way such a Plant can reduce or balance its carbon emissions to promote sustainable wastewater management.



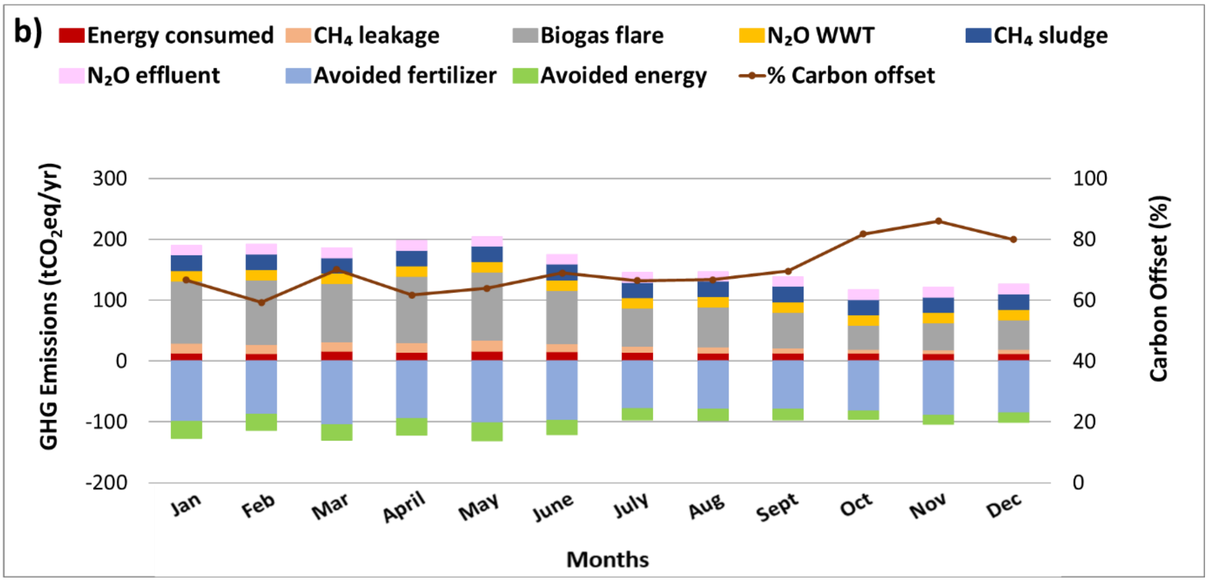


Figure 3.7: Potential carbon offsets considering avoided emissions. a) Carbon offsets considering dCH_4; b) Carbon offsets excluding dCH_4

3.3.7 Carbon Neutrality

Carbon neutrality in WWTPs has been defined as achieving net-zero GHG emissions over the lifetime of a WWTP. Carbon neutrality can be attained by zero GHG emissions, clean energy production, employment of less energy-intensive technologies, and implementation of energy-saving equipment at the WWTP (Markov et al., 2013; Xu et al., 2018). Thus, the embodied energy and associated CF need to be offsetted for carbon neutrality to be attained. Embodied energy is comprised of energy consumption during the construction and operational (including maintenance) phase of the WWTP (Mo & Zhang, 2012). Several studies have highlighted the possibility of attaining carbon neutrality by offsetting embodied energy and CFs through integrated resource recovery: energy; nutrients; and water recovery from wastewater (Hao et al., 2015; Mo & Zhang, 2012; Wett et al., 2007). However, based on the scope of this study defined in the system boundary, a holistic carbon neutrality evaluation will be impossible, and such an assessment would be erroneous in this context.

3.3.8 Measures to Mitigate Carbon Footprints of the Mudor WWTP

The carbon neutrality of WWTPs has been thoroughly discussed in literature. Environmental scientists believe that the energy embedded in wastewater is far beyond the energy required by WWTPs, hence carbon neutrality should be attainable (Zhou et al., 2013). Several other authors have iterated the possibility of reducing CFs by improving the energy balance from wastewater (Chen et al., 2019; Gu et al., 2016; Mamais et al., 2015; Sweetapple et al., 2015). Maktabifard et al. (2020) found that co-digestion of sewage sludge with external substrates has become a common phenomenon in the bid to improve energy balance in municipal WWTPs. However, the authors mentioned that adding an external carbon source to improve the AD process would increase the direct emissions from the Plant. Thus, a trade-off exists between direct emissions from wastewater treatment due to biogas leakages and CH_4 emissions due to incomplete combustion, on the one hand, and indirect emissions from fossil fuel-based energy consumption. This must be carefully evaluated during decision-making to attain a sustainable strategy for meeting energy neutrality and CF reduction targets. As mentioned earlier, this study was conducted in an outlined system boundary without considering the embodied energy of the Mudor WWTP. The following have been proposed as possible mitigation measures for the emission sources considered in the system boundary employed for this study.

- *Emissions from dCH_4*

Identified as the sole significant source of emissions from the Mudor Plant, measures to eliminate dCH_4 are an essential sustainable wastewater management practice. As previously stated, other authors have mentioned this emission source as the single largest source of GHG emissions from WWTPs they investigated (Heffernan et al., 2012; Robles et al., 2020). For instance, Robles et al. (2020) mentioned that recovery of dCH_4 by degassing membranes notably reduced the total GHG emissions from the WWTP they investigated. Thus, recovery of dCH_4 from the Mudor WWTP effluent remains the surest way this emission can be mitigated. Other proposed technologies for recovery of dCH_4 from effluent include the micro-aeration technique (Hartley & Lant, 2006), closed down-flow hanging sponge (Matsuura et al., 2010), and membrane technologies (Cookney et al., 2012; Crone et al., 2016; Luo et al., 2014).

- *Emissions from Biogas Flaring*

Flaring of biogas from anaerobic reactors remains the best alternative to direct biogas release into the atmosphere. As found early on in the study, emissions reduced from 15,034.71 tCO_2eq/yr (in the event of no flaring) to 932.89 tCO_2eq/yr when biogas was flared (sub-section 3.3.4). Notwithstanding, near complete elimination of GHG emissions from anaerobic systems, can be attained by resource recovery from biogas produced. This way, near-zero emissions would be accounted for flaring as there will be no flaring.

- *Emissions from Sludge Drying*

Comparable to resource recovery from biogas, resource recovery from sewage sludge could help mitigate emissions from this source. The IPCC has, however, reported N_2O emissions from the use of sewage sludge as fertilizer/compost on agricultural lands (IPCC, 2006a). Undeniably, the use of sewage sludge for agricultural purposes is one way to valorise this resource; nevertheless, sludge can also be used for energy recovery purposes through the application of thermochemical technologies or anaerobic co-digestion (Chun et al., 2011; Gu et al., 2017; Singh et al., 2020).

- *Emissions from Methane Leakages*

Fugitive emissions due to methane leakages from gas hoods and biogas lines, besides their contribution to GHG emissions, generate an explosive environment at the WWTP, as methane gas is a highly flammable gas (Henares et al., 2016). Thus, the risks of fire explosion and atmospheric emissions can be controlled by regular maintenance practices to ensure gas hoods, biogas lines, and any other probable sources of biogas leakage are well sealed.

- *CO_2 Emissions from Diesel Fuel Combustion and Grid Electricity Usage*

A number of studies have reported that WWTP's energy consumption can range from 3 - 6% of the total share of electricity consumption attributed to the water industry (Daw et al., 2012; Simon-Várhelyi et al., 2020). However, fossil fuel-based electrical energy emits significant GHGs into the atmosphere. Nonetheless, the recovery of clean, renewable energy from wastewater biogas and sludge by-products can provide a clean energy source that can offset the Plant’s energy requirement. In this way, CO_2 emissions from fossil fuel and grid electricity can be mitigated by using clean, recovered energy.

- *N_2O Emissions from Wastewater Treatment and Effluent Discharge*

N_2O emissions may occur directly in nitrification and denitrification processes, as a by-product, intermediary product or indirectly in the recipient water body (IPCC, 2006a). Literature has reported the eminent trade-off between the two N_2O emission sources: good biological nutrient removal, eliminated eutrophication in the recipient water body with consequent high N_2O emissions from nitrification and denitrification processes, otherwise poor nitrogen removal, lower N_2O emissions at the Plant, and discharge of high-nitrogen concentration effluent, resulting in eutrophication and N_2O emissions downstream from the recipient water body (Nejad, 2020; Xu, 2013). Notwithstanding, Accra being a city with rampant peri-urban vegetable farming activities, it will be proposed from a sustainability perspective that poor nitrogen removal with lower N_2O emissions; thereafter, nutrient-rich effluent be conveyed to farming sites for peri-urban irrigation to boost food crop production. Thus, for the course of sustainability, it would be prudent to recover nutrients in the effluent instead of eliminating them.

3.4 Partial Conclusion

Due to the progressive development in Ghana's wastewater industry, which has seen to the construction of WWTPs in some regions across the country, and the Ghana government's commitment to reduce GHG emissions by 15% by 2030 towards the Paris Agreement, the contribution of GHGs from WWTPs cannot be disregarded. This study employed the globally recognized IPCC GHG inventory methodology to estimate the GHG emissions from a full-scale UASB-based WWTP treating municipal sewage in Accra, the capital city of Ghana. It was found from the study that dCH_4 in effluent discharged into the Korle Lagoon was the most significant source of GHG emissions from the Plant. This emission source accounted for about 95% of the total emissions. CO_2 emissions from energy consumption (grid electricity and diesel fuel) contributed the least emissions, responsible for only 8.5% of the Plant's emissions. Recovery of readily available resources such as nutrients and energy from wastewater treatment by-products remains the surest way carbon offsets for the Plant can be attained. Again, the recovery of dCH_4 in effluent would significantly reduce emissions from the operations of the Mudor WWTP.

Chapter 4:

Economic Evaluation of a Full-scale UASB Reactor coupled with Trickling Filters Treating Municipal Wastewater in Accra, Ghana

Abstract: Basic sanitation in emerging economies like Ghana currently require large investments to meet the demands of a massively growing population. The study of technologies that can enable the deployment of sewage treatment systems becomes important. The UASB reactor technology has been recognised as one of the most efficient and economically feasible wastewater treatment technologies that can be deployed in emerging economies towards attaining sustainable wastewater management in these parts of the world. However, it is reported that funding and economic challenges are two major factors that could hinder the implementation of wastewater treatment systems in developing countries. This chapter employs Cost-Benefit Analysis to perform an economic evaluation of the Mudor wastewater treatment plant which employs the UASB reactor as the main biological treatment unit coupled with Trickling Filters to treat municipal wastewater in some suburbs of Accra, the capital city of Ghana. The study employed resource recovery under circular economy to evaluate the resources that can be explored from this technology. It was found from the study that wastewater effluent was rich in nitrogen and phosphorus with average concentrations of 0.0836 kgN/m^3 and 0.0284 kgP/m^3, respectively, coupled with low heavy metals concentrations which were within acceptable limits of the World Health Organization for reclaimed water use in agriculture made water and nutrient recovery a viable option. Sewage sludge was likewise found to be rich in nutrients and high organic matter making it suitable as a soil conditioner to support plant growth. Dried sewage sludge was found to have an average calorific value of 9.81 MJ/kg, which makes it viable for energy recovery from thermochemical processes. The high methane content in biogas additionally makes it possible for energy recovery from biogas. The net energy recovery potential from biogas and sludge was estimated at 534.1 MWh/yr, meanwhile, the actual energy demand of the Mudor Plant was determined to be 392.7 MWh/yr. Thus, with energy recovery, the Mudor Plant can be energy positive. Integrated resource recovery from wastewater is one surest way by which sustainable wastewater management can be attained in Ghana.

Keywords: Biogas production; Economic assessment; Energy recovery; Nutrient recovery; Resource recovery; Sludge production; Trickling filter; UASB reactor

4.1 Introduction

Most traditional wastewater treatment technologies based on activated sludge processes have been widely implemented in recent decades worldwide, especially in developed countries (Gavasci et al., 2010). Nevertheless, the growing global concerns over environmental degradation and rising energy costs have led to the advocacy and subsequent development of innovative technologies that are less energy intensive with fewer environmental impacts. Improving energy efficiency is a subject that should be considered in the construction of new WWTPs, during the renovation of old Plants and the general operation of these facilities. The development and implementation of innovative technologies for energy-efficient systems involve costs and benefits that should be evaluated. Economic feasibility studies are an essential tool employed in the decision-making process for implementing new technology alternatives in the wastewater treatment sector (Molinos-Senante et al., 2012).

One of the most popular tools to evaluate a project's economic feasibility is the cost-benefit analysis (CBA). This tool substantiates the economic rationality of investment testing, whether the benefits outweigh the costs. The CBA methodology is a generally accepted economic evaluation method in the wastewater industry. It allows the assessment of financial costs and revenues to build and operate wastewater treatment facilities and the evaluation of monetary and non-monetary benefits that can be explored from these systems during their lifetime (Molinos-Senante et al., 2012). Treating and using wastewater has numerous significant environmental, social and health benefits. However, these benefits are often not monetized or calculated due to the absence of baseline or market value (Drechsel et al., 2015). Valuation of these benefits is nonetheless necessary to justify appropriate investments and financing mechanisms to sustain wastewater management.

The UNEP report on *"Economic evaluation of wastewater; the cost of action and the cost of no action"* states that discharge of untreated wastewater into the environment can lead to negative impacts, which are grouped into three: impacts on human health, impacts on the environment and impact on economies, and these three are interrelated (UNEP, 2015). Therefore, an economic evaluation will allow a better understanding of wastewater economics requisite for appropriate technology selection and the non-monetized benefits of sustainable wastewater treatment systems with resource recovery.

Some studies have been reported wherein the authors found that the application of eco-friendly technologies and modern circular economy (CE) concepts could permit the recovery of resources from wastewater, to attain sustainable wastewater management (Lopes et al., 2019; Metcalf & Eddy, 2014; UN Wastewater Report, 2017). Three major by-products are generated during anaerobic wastewater treatment: reclaimed water, biogas and sludge. Under the CE perspective, these by-products can be transformed into valuable resources that would impact humanity, the environment and the economy healthily. The United Nations World Water Report mentioned that wastewater is embedded with rich resources which have not yet been tapped (UN Wastewater Report, 2017). Treated wastewater effluent, devoid of pollutants and pathogenic contamination, is a reliable source of fresh water, which can be used for potable and non-potable purposes, depending on the level of purification. Reclaimed water and sludge generated from wastewater treatment processes are also rich in nutrients required for plant growth. Improved agricultural activities will in turn boost food production (UN Wastewater Report, 2017; UNEP, 2015). Many studies have recounted the possibility of recovering nutrients (nitrogen and phosphorus) from wastewater treatment systems with the employment of CE concepts. Nutrient recovery will lessen the dependence on and production of inorganic fertilizers, whilst the application of biosolids on agricultural lands will help improve soil quality (Beckinghausen et al., 2020).

Additionally, the biogas generated contains a significant portion of methane gas that can be harnessed for energy recovery. Methane-rich biogas harnessed from anaerobic wastewater treatment systems and the high energy-embedded biosolids could likewise be employed for energy recovery through the application of modern-day technologies. Energy recovered from WWTPs could be used to offset the Plant's energy requirement, promoting sustainable wastewater treatment with the concept of "*sanitation financing sanitation*". Depending on the degree of energy recovery, WWTPs can be energy self-sufficient or energy positive, eliminating the reliance on fossil fuel-based energy sources (Gu et al., 2017). It is reported that anaerobic co-digestion, algal technology, anammox technology, thermochemical processes, and microbial fuel cells (MFCs) are some technologies successfully employed for energy recovery from wastewater treatment systems (Daverey et al., 2019). Capodaglio & Callegari (2020) opined that energy recovery from wastewater treatment helps to improve energy efficiency and reduce the environmental impacts associated with conventional wastewater treatment systems.

With the recent persistent increase in global stressors such as freshwater pollution and scarcity, increasing energy crises, deadly impacts and cascading effects of climate change, all coupled with ever-growing global populations (UNEP, 2021), resource recovery from wastewater treatment will indeed promote sustainable development. Based on the classifications developed by the Stockholm Resilience Centre, resource recovery from wastewater under CE can be linked to the SDGs set by the United Nations. For direct consideration, whilst efficient wastewater treatment will eliminate associated diseases, the sanity of water bodies will also be preserved, thereby meeting SDG targets 3, 6 and 14, respectively, on good health and wellbeing, clean water and sanitation and life below water. Nutrients recovery will improve food production and eliminate hunger and poverty as livelihoods are improved (SDGs 1 and 2). Finally, recovery of clean, renewable energy from wastewater biogas and sludge would promote the attainment of the SDGs on affordable and clean energy (SDG 7) and climate change (SDG 13), which aim to lessen the impacts of climate change by regulating and promoting the utilization of clean, renewable energies (Griggs et al., 2017). The United Nations' climate-change policy advocates the use of renewable energy in place of traditional fossil-based energy. Gupta (2020) likewise deftly revealed the direct and indirect links between wastewater resource recovery and the SDGs. Therefore, sustainable wastewater management with resource recovery can contribute to the realization of these targets (Quaschning, 2019).

Wastewater has vast potential as a resource: water, nutrients and energy, which remains underexploited. Integrating on-site energy recovery technologies from biogas and sludge treatment can transition WWTPs from major energy consumers to energy-neutral or even net energy producers. Besides minimizing operational costs, energy recovery facilitates the reduction of CFs of WWTPs, enabling increased revenue streams through carbon credits and carbon trading programmes. Additionally, developing technologies for nitrogen and phosphorus recovery from sewage effluent and sludge, besides boosting food production and improving livelihoods, will mitigate reliance on inorganic fertilizers, reducing energy and emissions associated with their production (UN Wastewater Report, 2017).

This chapter evaluates the economic implications of the UASB reactor technology in an emerging economy like Ghana through the performance of cost analysis by evaluating the capital and operational costs of such a system and also the benefits assessment under resource recovery employing CE concepts to ascertain the sustainability of these systems for the developing world. The benefits assessment will be carried out by evaluating the various resources that can be recovered from the operations of the Mudor WWTP. Integrated resource recovery from wastewater can be classified under the following: i) Water recovery: industrial reuse, groundwater recharge, recreation and non-potable use; ii) Material recovery: biosolids use in agriculture, bioplastics, volatile fatty acids recovery; iii) Heat and energy recovery: biogas from anaerobic digestion of wastewater and sludge, thermochemical energy recovery from sludge; iv)

Nutrient recovery: nitrogen recovery, phosphorous recovery through struvite crystallization and trace elements such as zinc and iron (Montwedi et al., 2021). However, for this thesis, resources to be considered are electrical energy recovery from biogas and sludge, nitrogen and phosphorus recovery from reclaimed water and biosolids, and biofuel (biochar) recovery from sludge.

4.2 Materials and Methods

4.2.1 Description of the Wastewater Treatment Plant

This research was carried out at the Mudor WWTP in Accra, Ghana's capital. Ahmed et al. (2018) reported that the Plant was built in 2000, and after operating for a few years, it was shut down due to poor maintenance culture and a lack of financial commitment. It was, however, rehabilitated, expanded and resumed operations in 2017. The Plant receives and treats municipal sewage from offices, households and business centres within the Accra central business district (CBD) and its surroundings connected to sewer networks, and is projected to serve roughly 100,000 inhabitants. The Mudor WWTP consists of six (6) modular-shaped UASB reactors, with three (3) trickling filters (TFs) and two (2) clarifiers which act as post-treatment units to the UASB reactor effluent. A detailed description of the treatment plant has been discussed in Chapter 2 (sub-section 2.2.2). Figure 4.1 presents the synoptic view of the Mudor WWTP.

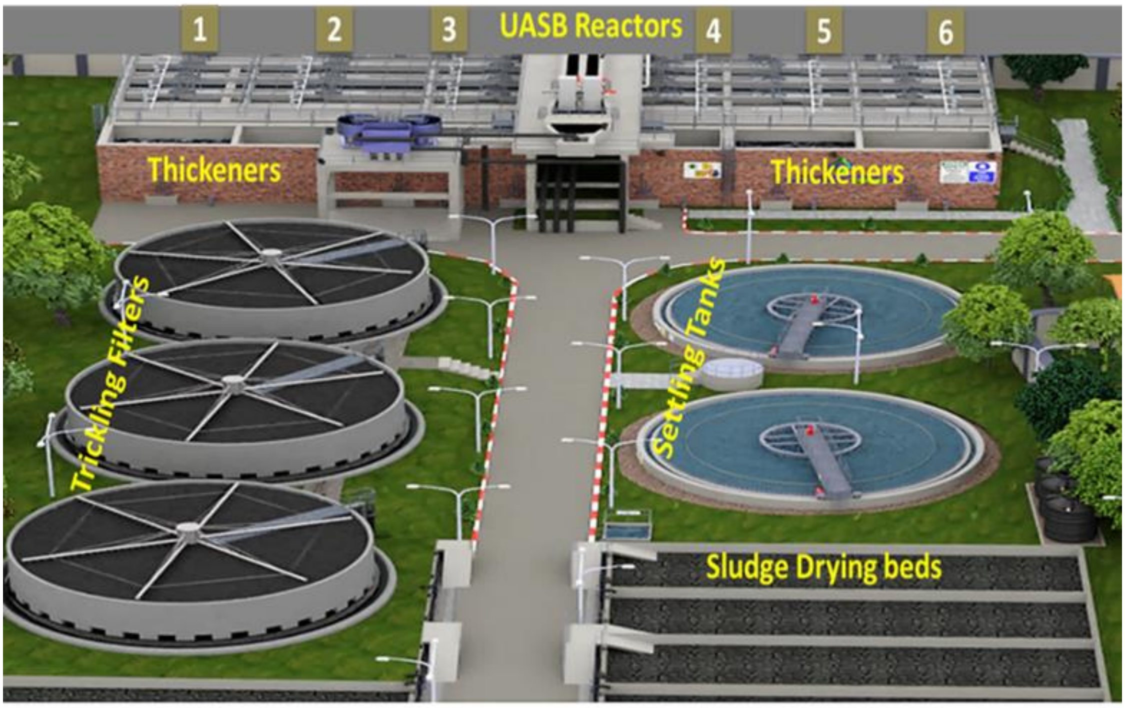


Figure 4.1: Synoptic view of the Mudor WWTP

4.2.2 Cost Analysis of the Mudor WWTP

To perform the cost analysis of the Mudor WWTP, the operational and capital expenditures were determined as reported in the literature.

4.2.2.1 Capital Expenditure (CAPEX)

Capital expenditure constitutes the one-time expenses incurred during the implementation of a project. They usually comprise the initial investment costs, including land acquisition, construction of WWTP, mechanical equipment, support structures and buildings. These have been considered in the cost analysis for the Mudor WWTP.

4.2.2.2 Operational Expenditure (OPEX)

Operational expenditure constitutes recurring expenses that are incurred throughout the project's life span. Such expenses include periodic repairs and maintenance, utilities (water and electricity costs), administrative expenses, and staff management expenses. These have also been considered in the cost analysis for the Mudor WWTP.

4.2.3 Benefit Analysis of the Mudor WWTP

4.2.3.1 Reclaimed Water Use in Agriculture (Water and Nutrients Recovery Potential)

In order to evaluate the potential of wastewater use in irrigation, the quality and nutrient concentrations were evaluated. For the quality analysis, wastewater effluent was analysed for the presence of pathogenic microorganisms, parasites and heavy metals. The presence of pathogens in effluent wastewater poses health risks to farmers, whilst depending on the type of farm produce cultivated, produce ingested without or with little cooking, such as vegetables, pose great risks if cultivated with contaminated water (Alcalde-Sanz & Gawlik, 2017). Moreover, concentrations of heavy metals in irrigation water build up in the soil after years of irrigation till it reaches levels where plants take up these toxic substances and get transmitted into food crops. Though in minute concentrations, accumulation over a period of time leads to toxicity to humans. Due to this, the world health organization (WHO) has set some guidelines regarding the allowable concentrations of heavy metals permissible for urban irrigation. The analytical methods, equipment models and standard references employed for the wastewater nutrients and heavy metals characterization have been presented in Appendices.

4.2.3.2 Nutrient Recovery from Sludge (Use as Biosolids)

The excess sewage sludge discharged was sampled for nutrient (nitrogen and phosphorus) characterisation to evaluate the recovery of nutrients in sludge and potential use as biosolids. Sludge samples were again characterized for pathogenic organisms, parasites and heavy metals to ascertain their quality for safe agricultural use.

- *Determination of Physico-chemical Parameters*

Excess sludge samples withdrawn from UASB reactors were analysed for relevant physicochemical parameters, traces of heavy metals and microbial loads. Analytical techniques were strictly guided by the standard methods for examining sludge characteristics (APHA, 2017). Preceding the laboratory analysis, sludge samples were air dried and finely grounded in a crucible and sieved with a 0.5mm sieve mesh to fine texture. pH and EC were determined in sludge-water suspension (1:10 w/v) with a portable multi-probe analyser (HQ40D LDO10101, HACH). Total phosphorous was measured by digesting 0.1 g of fine sludge samples with 25 ml of a solution of concentrated HNO_3^- and 60% $HCLO_4$ at a ratio of 1:1.5. The mixture was heated until it became viscous. De-ionized water was added to the mixture, filtered into a 100 ml volumetric flask, and filled to the 100 ml mark with distilled water. Phosphorus concentration in the solution was analysed using Murphy & Riley (1962) method with the Cole Parmer UV Spectrophotometer.

- *Determination of Sludge Heavy Metals Concentration*

The concentrations of heavy metals in sludge samples were measured using the methodology proposed by Chapman & Pratt (1961). 0.2 g of the finely grounded samples have been weighed into digestion tubes. 10 ml of a ternary mixture (20 ml $HClO_4$, 50 ml H_2SO_4 and 500 ml HNO_3^-) was added to the sample under a fume hood. The well-mixed solution was gently heated (about 60 ^oC) on a hot plate. The heating continued until a viscous white substance (sulphuric acid fumes) formed. The content was made to cool, and 50 ml of distilled water was added, after which the solution was filtered with the Whatman filter paper into a volumetric flask. The solution was topped-up to the 100 ml mark, and heavy metals were analysed using Atomic Absorption Spectrometry (Perkin Elmer A Analyst 800).

- *Determination of Microbial Quality*

The microbial quality of wastewater sludge was monitored for pathogenic organisms (faecal coliform, *E. coli*, Salmonella *sp.)* and parasitic pathogens (helminth eggs). Faecal coliform (FC), *E. coli*, and Salmonella *sp*. were determined by the pour plate method with agar medium and colony count technique according to APHA (2005) standard methods for the examination of wastewater and sludge. Coliforms and *E. coli* were inoculated for 10 g (w/v) of biosolid samples using Chromocult Coliform Agar. 1 ml of homogenised raw sample was diluted into 9 ml of ringer solution for the first dilution (1:10) and repeated until the desired dilution for the possible coliform count was obtained. The bacterial load count was expressed as the number of Coliform Forming Units per one gram of dry sludge (CFU/g). Helminth egg characterisation was according to the methodology proposed by Moodley et al. (2008) for helminth eggs and cysts in sludge and expressed as the number of egg counts per gram of dry sludge (No./g DM).

4.2.3.3 Biogas Quantification and Characterization

Detailed methodology for the quantification and characterization of the biogas produced by the Mudor UASB reactors has been provided in Chapter 2 (sub-section 2.2.3) to evaluate the energy recovery potential of biogas from the Plant.

4.2.3.4 Sludge Quantification and Characterization

The system was monitored for excess sludge production during the study. The observation of effluent concentrations for TSS, BOD and COD defined sludge build-up. Deterioration of these parameters indicated excess build-up of sludge (Rosa et al., 2012). Sludge withdrawal valves were opened for desludging excess sludge into the sludge thickeners and next onto the drying beds. Sludge was withdrawn biweekly, and the discharged sludge was approximately 30% of the volume of the sludge thickener. Based on these estimates, the discharged sludge was quantified for this study.

In evaluating the energy recovery potential of sewage sludge by thermochemical processes, sampled sludge was characterised to determine the chemical and physical composition. Proximate analysis was conducted to measure immediate components: volatile matter (VM), fixed carbon (FC), moisture (MC) and ash contents. Regarding ultimate analysis: elemental analysis for carbon (C), nitrogen (N), hydrogen (H) and sulphur (S) contents were determined. These analyses followed standard procedures (APHA, 2017).

- *Proximate Analysis*

Proximate analysis was conducted to quantify the solids (total, fixed and volatile solids) and MC. MC was measured by estimating the observed weight loss in the substrate after evaporation. 5 g of sludge aliquot was placed in a porcelain crucible, preheated in an oven to 105 ^oC for 1 hr, cooled in a desiccator and weighed to determine the empty crucible's weight. The sludge sample was oven dried for 24 hrs. The dried sample was allowed to cool in the desiccator. MC was then determined using Equation 4.1, based on the percentage weight (wt%).

$$MC\ (\%) = \frac{W_{wet} - W_{dried}}{W_{wet}} * \ 100 \dots \dots \dots \dots \qquad \textit{(Eqn. 4.1)}$$

Where:

MC = Moisture content (%)

W_{wet} = Weight of initial wet sample

W_{dried} = Weight of oven-dried sample.

The dried residue is expressed as the total solids (TS) or dry matter (DM), Expressed by Equation 4.2.

$$TS\ (\%) = 100 - MC \dots \dots \dots \dots \qquad \textit{(Eqn. 4.2)}$$

The VM signifies the organics in the sludge. After recording the TS concentration, the same specimen was placed in a furnace preheated to 550 ^{o}C and ignited for 2 hrs. After a reasonable temperature drop in the furnace, the ignited sample was placed in a desiccator to cool. Afterwards, VM was determined using Equation 4.3:

$$VM\ (\%) = \frac{M_{dried} - M_{ignited}}{M_{dried}} \times \ 100 \dots \dots \dots \dots \qquad \textit{(Eqn. 4.3)}$$

Where:

VM = Volatile matter (%)

M_{dried} = Sample mass after oven drying

$M_{ignited}$ = Sample mass after ignition.

The Ash content was then calculated with $M_{ignited}$ in *Eqn. 4.3.*; the residue left in the dish after ignition and presented by the Equation:

$$Ash\ (\%) = \frac{M_{ignited}}{M_{dried}} \times \ 100 \dots \dots \dots \dots \qquad \textit{(Eqn. 4.4)}$$

Fixed carbon (FC) was computed as the mass difference.

$$FC\ (\%) = 100 - VM\ (\%) - \ Ash\ (\%) \dots \dots \dots \dots \qquad \textit{(Eqn. 4.5)}$$

- *Ultimate Analysis*

For elemental analysis, C was measured by Walkley-black wet oxidation method as described by Nelson & Sommers (1982). H was determined by the titrimetric method proposed by Mclean (1965), using phenolphthalein indicator and sodium hydroxide as reagents. N was determined with the Kheldahl method, as described by Bremner & Mulvaney (1982), using ammonia-free-grade concentrated H_2SO_4, Boric acid solution, NaOH and selenium as reagents. S was measured turbidimetrically using the spectrophotometry method reported by Singh et al. (1999), with di-acid (HNO_3-$HClO_4$) for digestion. Oxygen (O) was then estimated as the difference between CHNS and ash values (Petrovič et al., 2021; Rosa et al., 2018).

- *Calorific Value (CV) Analysis*

The laboratory methods for calorimetry analysis to determine the energy value of sludge followed the procedure described in the Parr oxygen-bomb calorimeter manual (Parr 1342 manual, No. 204M), as described in ASTM E711-87 (2004). Air-dried sludge samples previously weighed and pelletised were combusted in a pressurised (30.0 atm) oxygen atmosphere. 1.0 g of pelletised sludge was used to certify the rising temperature in the water jacket provided a safe combustion environment, which did not exceed the optimum thermometer range. Benzoic acid was used as the standard solution to determine the heat capacity of the bomb. Experiments were conducted in duplicates. Figure 4.2 presents the apparatus employed for the bomb calorimetry experimentation. The gross calorific value (GCV) and net calorific value (NCV) calculations were done following directives provided by the Parr manual.

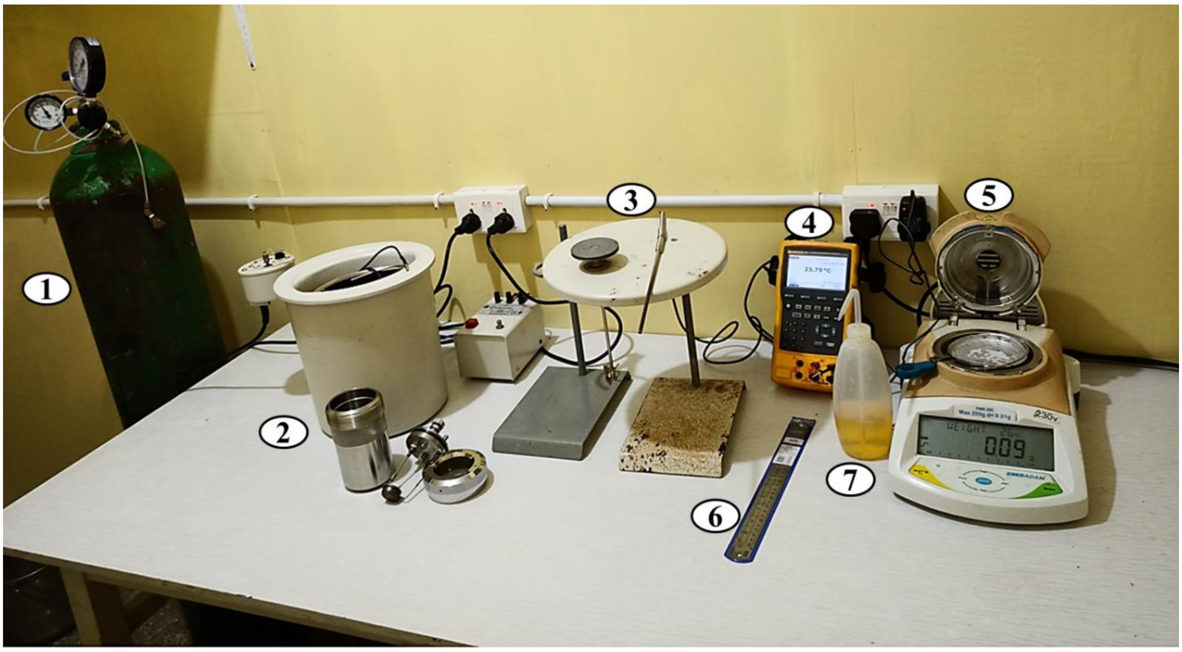


Figure 4.2: Bomb calorimetry apparatus (1) O_2 filled gas cylinder; (2) Bomb calorimeter; (3) Lid of the bomb; (4) Thermometer; (5) Weighing scale; (6) Ruler; (7) Methyl orange

The theoretical energy values of sludge were measured based on the proposed model by Galhano dos Santos & Bordado (2018) and have been reported in similar studies (Lopes et al., 2019; Rosa et al., 2018) to estimate sewage sludge heating values theoretically. This approach allows the comparison of actual and theoretical values, permitting the energy value of a substrate to be determined based on its elemental composition in the absence of a bomb calorimeter (Rosa et al., 2018). The theoretical GCV was estimated by the Equation:

$$GCV = \frac{[337.3 * C + 1418.9 * (H - {}^{O}/_{8}) + 93.1 * S + 23.3 * N]}{1000} \ldots\ldots\ldots\ldots \qquad \textit{(Eqn. 4.6)}$$

The theoretical NCV was estimated by the Equation:

$$NCV = \left[(GCV - \lambda) * (r + 0.09 * H) * \left(\frac{100 - W_t}{100}\right)\right] \ldots\ldots\ldots. \qquad \textit{(Eqn. 4.7)}$$

And $r = \left(\frac{W_t}{100 - W_t}\right) \ldots\ldots\ldots\ldots\ldots..$ *(Eqn. 4.8)*

Where:

GCV = Gross calorific value (MJ/kg), dry basis

C = % carbon in the sludge, dry basis

H	= % hydrogen in the sludge, dry basis
O	= % oxygen in the sludge, dry basis
S	= % sulphur in the sludge, dry basis
N	= % Nitrogen in the sludge, dry basis
NCV	= Net calorific value (MJ/kg), dry basis
r	= Solids content and dehydrated sludge moisture ratio
Wt	= Solids content in dehydrated sludge (%), moist basis
λ	= latent heat of water (2.31 MJ/kg at STP)

4.2.3.5 Evaluation of Energy Balance at the Mudor WWTP

The energy balance calculations correspond to the difference between the actual energy demand of the Plant and the energy recovery potential from biogas and sludge by-products (Rosa et al., 2016). The Mudor Plant's energy demand is the level of power consumption by the Plant, principally, power consumption by the pumping stations (PS). The energy potential relates to the all-out energy that is recoverable from by-products from the Plant operation.

4.2.3.6 Energy Demand at the Mudor WWTP

Data on the monthly energy consumption of the Plant was used to estimate the energy demands of the facility (in kWh/month) during the study period. Energy consumption data considered all activities, units' operations and processes that relied on energy to operate i.e., laboratory, lighting, administration and pumps, including diesel fuel used to power standby generators when power from the national grid was interrupted. Management of the facility mentioned that the four pumping stations (PS-1, PS-2, PS-3, and PS-4) were the most significant energy consumers, responsible for approximately 95% of the Plant's energy consumption. PS-1 constitutes the lifting pumps at the CAPS that pump untreated sewage into the UASB reactors. PS-2 pumps sludge from the clarifiers into the thickeners, whereas PS-3 pumps sludge from the thickeners onto the drying beds. PS-4 distributes clean water for daily cleaning activities. Table 4.1 presents a summary of the main characteristics of the pumping stations which operate at the Plant.

Table 4.1: Characteristics of pumping stations operating at the Mudor WWTP

Characteristics	PS - 1	PS - 2	PS - 3	PS - 4
No. of pumps	3	2	2	2
Model	Wilo	Wilo	Netzsch	Netzsch
Type	Submersible	Submersible	Surface pump	Surface pump
Flowrate (m^3/h)	1800.0	200.0	25.0	48.0
Power ratings (KW)	90.0	15.0	5.5	7.5

4.2.3.7 Energy Recovery Potential of the Mudor WWTP Biogas and Sludge By-products

As mentioned early on, the Mudor Plant's energy recovery potential corresponded to the sum of the energy potential of the system's by-products: biogas and sludge. Biogas energy recovery potential (EP_{biogas}) was determined using the Equation:

$$EP_{biogas} = Q_{biogas} * C_{CH_4} * E_{CH_4} \dots\dots\dots \quad \textit{(Eqn. 4.8)}$$

The sludge energy recovery potential (EP_{sludge}) was estimated with the Equation:

$$EP_{sludge} = P_{sludge} * NCV_s \dots\dots\dots\dots \quad \textit{(Eqn. 4.9)}$$

Moreover, the total energy potential (EP_{Total}) from biogas and sludge was determined with the Equation:

$$EP_{Total} = EP_{biogas} + EP_{sludge} \dots\dots\dots\dots\dots \quad \textit{(Eqn. 4.10)}$$

Where:

EP_{Total} = Total energy potential (MJ/d)

EP_{Biogas} = Biogas energy potential (MJ/d)

Q_{Biogas} = Biogas production rate (m^3/d)

C_{CH_4} = Concentration of CH_4 in biogas (%)

E_{CH_4} = NCV of CH_4 combustion (35.9 MJ/m^3)

EP_{Sludge} = Sludge energy potential (MJ/d)

P_{Sludge} = Production of dry sludge matter (kg/d)

NCVs = Net calorific value of sludge (MJ/kg)

4.2.4 Data Analysis

Microsoft Excel 2019 software was used for the descriptive statistical analysis of data obtained for the study. Pearson correlation was employed for inferential statistical analysis, and Sigma Plot Version 12 software was used to present the graphical results.

4.3 Results and Discussion

4.3.1 Cost Analysis of the Mudor WWTP

4.3.1.1 Capital and Operational Expenditure of the Mudor WWTP

Table 4.2 presents a breakdown of the expenditure of the Mudor WWTP. The Plant built by the government of Ghana incurred an initial investment cost of 22.14 M USD as of the year 2000. The Plant, however, broke down and was non-functional for some years before being rehabilitated between 2012 and 2016 (Ahmed et al., 2018). The rehabilitation and expansion work incurred an extra cost of 8.65 M USD. Monthly operational costs are estimated at 49,209.14 USD.

Table 4.2: Capital and operational expenditures of the Mudor WWTP

Cost entity	Amount (GHS)	Amount (USD)
Initial investment cost (in the year 2000)	37,400,000	22.14 M
Renovation and expansion cost (2012 - 2016)	18,000,000	8.65 M
Total initial cost	55,400,000	30.79 M
Monthly operational cost	700,000	49,209.14

Presented in Figure 4.3 is the percentage distribution of the monthly operational costs recorded at the Mudor WWTP. From the figure, it is found that staff management constitutes the highest portion of the operational costs at 37%. Laboratory reagents were responsible for 10.4%, whilst repairs and maintenance were responsible for 18.3% of the OPEX. Energy consumption (electricity and fuel) at the Plant was minimal and made up only 7.3% of the OPEX. Lower energy consumption is typical with anaerobic wastewater treatment systems (Lettinga et al., 1980), and this gives an advantage over the conventional activated sludge systems where air blowing could constitute as high as 80% of the total energy costs of these systems (Altin et al., 2020), thereby increasing the overall cost of treatment. Most importantly, the Mudor WWTP has been designed such that gravity drives most material flow (Arthur et al., 2022). This explains the relatively lower energy consumption costs by the Plant. Cost on sludge management had been excluded from the analysis as currently, sludge treatment is by the drying beds, which incurs no cost, except the pumping of sludge from the sludge thickeners onto the drying beds.

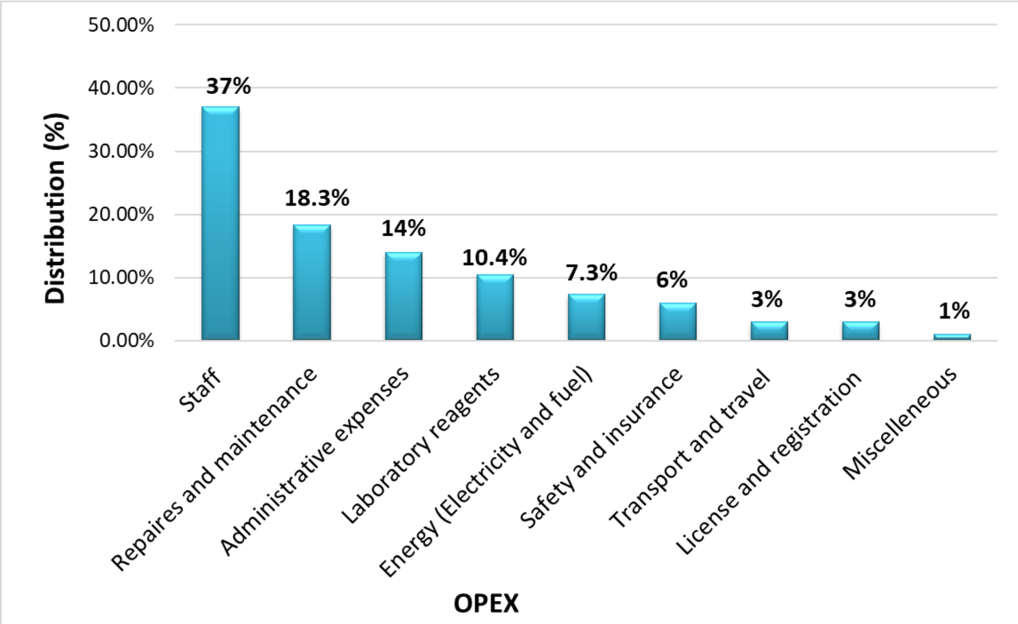


Figure 4.3: Percentage distribution of monthly operational costs

4.3.1.2 Unit Cost of Wastewater Treatment

One indicator that measures the cost of wastewater treatment is the unit cost assessment. Several studies have been reported wherein authors employed various functional units to express the unit cost of wastewater treatment as USD/m^3, USD/inhabitant and USD/PE. For this study, the unit volume of wastewater was employed to express the unit cost of wastewater treatment at the Mudor WWTP. Figure 4.4 presents a plot of the unit cost against wastewater flow and monthly OPEX. A constant figure was employed for the monthly OPEX as provided by the facility's management for 2021. Unit cost per m^3 of wastewater treated ranged from 0.34 - 0.45 USD/m^3 during the entire study period. It can be inferred from the figure that the unit cost of treatment was relatively lower for large volumes of sewage compared to smaller volumes of sewage treated. The month of march recorded the highest sewage flow, but it also reported the least unit cost of treatment. This observation can be attributed to two reasons. First is the fixed cost of OPEX for the various months, which implied that a constant OPEX was applied regardless of the influent volume treated within the month. This approach can increase the error margins in the values reported. The second is ascribed to the concept of economies of scale.

It has been reported in the literature that larger WWTPs generally have lesser unit treatment costs compared to smaller capacity Plants. Metcalf & Eddy (2014) mentioned that economies of scale were more applicable to energy-intensive technologies. The authors stated that pump efficiency was linked to the size of the treatment plant in that increased flow rate with increased pipe diameters led to decreased frictional losses, which allowed energy economies of scale to be attained. In a related study, McNamara (2018), after performing an economic assessment of wastewater treatment systems employing a life cycle perspective, stated that economies of scale were most applicable to electro-mechanical wastewater treatment systems compared to natural systems. In the same regard, Sato et al. (2007), in their study, which performed an economic evaluation of wastewater treatment systems in India, found that for UASB reactors, the unit capital cost decreased by half with an increase in the treatment volume. Additionally, they found that the annual operation and maintenance (O & M) cost for UASB operations likewise decreased with an increase in sewage volume, with the unit cost of treatment for the various treatment technologies ranging between 0.03 and 3.85 USD/m^3. They concluded that such factors as the Plant capacity, the desired effluent quality, and the treatment technology applied all influenced the unit cost of treatment.

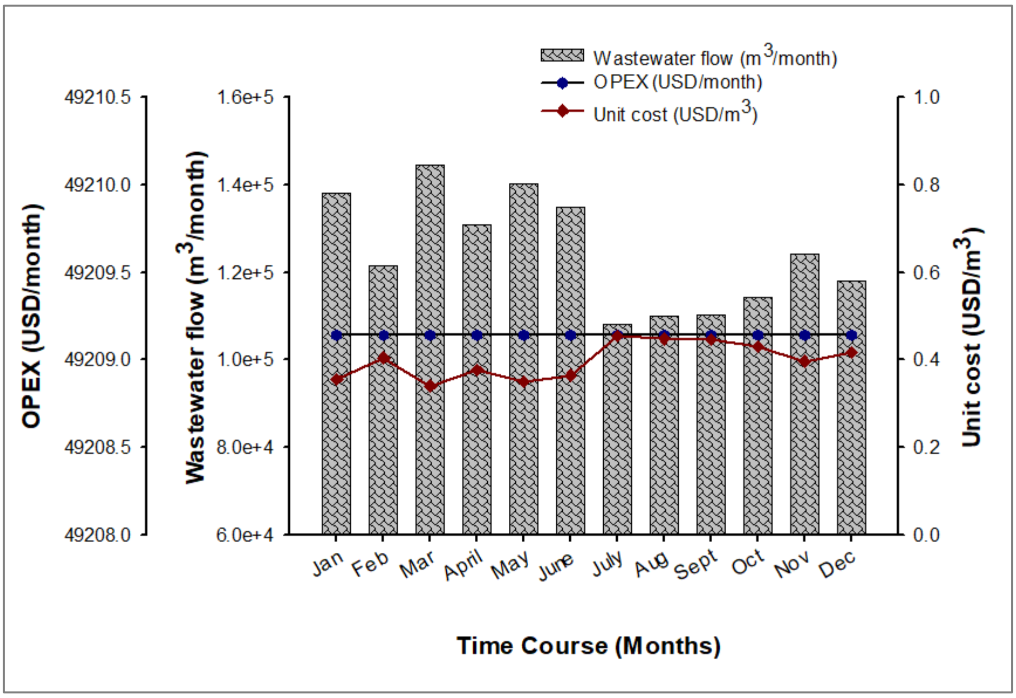


Figure 4.4: Unit cost of wastewater treatment

4.3.1.3 Energy Cost of Mudor WWTP

The energy demand of the Mudor WWTP was estimated based on the Plant's energy consumption (electricity and fuel). As stated earlier, the four (4) sets of pumps (PS-1, PS-2, PS-3, and PS-4) are responsible for about 95% of the total energy consumption at the Plant. Administrative activities, lighting and laboratory consumption accounted for barely 5% of the energy costs. This finding agrees with a parallel study conducted in Brazil (Rosa et al., 2016), wherein was reported that the energy consumption of a full-scale UASB WWTP was entirely ascribed to the pumping stations of the Plant. Energy cost as a result of sludge treatment was reported to be zero due to the sludge treatment method employed at the facility. From Figure 4.5, the actual Plant energy demand, comprising electricity use and diesel fuel consumed ranged from 28,836 kWh/month for November 2021 to 38,442 kWh/month; the highest consumption observed for May 2021 during the study period. The average energy consumption was 32,722 kWh/month, with an overall consumption of 392.7 MWh/yr. Wastewater flow likewise ranged between 108,236 and 144,526 m^3/month.

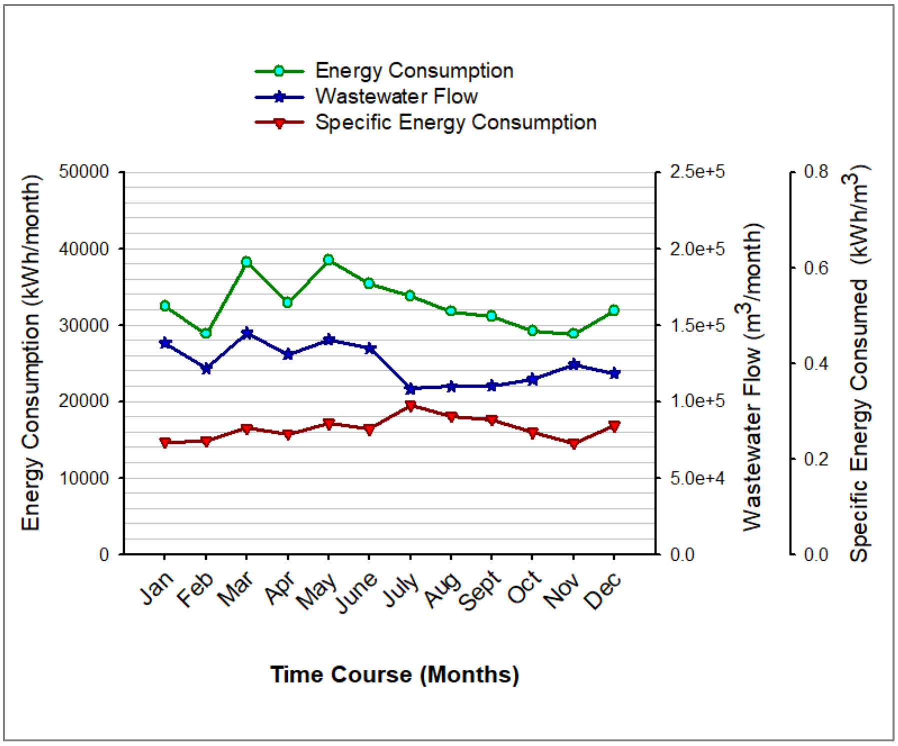


Figure 4.5: Variation in wastewater flow and energy consumption

A correlation analysis was carried out to assess the relationship between wastewater flow and energy consumption. A moderate correlation was observed between the two variables (Pearson correlation coefficient R = 0.647). As presented in the figure, the lowest sewage flow recorded in July did not result in a corresponding least energy consumption, that month rather recorded the highest specific energy consumption during the study period. This finding could be attributed to the concept of economy of scale as reported by Metcalf & Eddy (2014), and consistent with the assertion by Vaccari et al. (2018) that higher energy efficiency of WWTPs is associated with Plants having large capacities.

Maktabifard et al. (2018) mentioned in their study that the specific energy consumption (energy intensity) in kWh/m^3 is the most frequently used key performance indicator (KPI) to evaluate WWTPs' energy performance. The specific energy consumption of the Mudor Plant ranged from 0.23 - 0.31 kWh/m^3, presented in the figure. Gu et al. (2017) reported a wide range of values from 0.02 - 3.75 kWh/m^3 for WWTPs in several European, North American and Asian countries. These authors, in a review, opined that energy consumption of WWTPs varied across regions, with that of the developed countries far higher because of the energy-intensive technologies implemented in these regions, whilst consumption for developing countries is relatively lower. For instance, it was reported that in the USA, the energy intensity of WWTPs was approximately 0.52 kWh/m^3. However, that of South Africa was found to be between 0.079 and 0.41 kWh/m^3 due to the low energy-intensive technologies such as TFs and lagoons widely implemented. In a related study, it was found that the energy consumption in three Asian countries: Korea, Japan and China were 0.243, 0.304 and 0.310 kWh/m^3, respectively. These were likewise lower compared to those of the highly developed regions (Chae & Kang, 2013; Wang et al., 2016; Yang et al., 2010). Burton (1996) found that generally, the specific energy consumption of WWTPs decreased with increased sewage flow; thus, the economy of scale concept is applicable and is comparable to the finding of this study. A number of authors have mentioned that a Plant's energy consumption is influenced by some factors, including Plant size (population equivalent, organic and hydraulic load), type of treatment process, location, desired effluent quality, Plant age and operators' experience. Moreover, aerobic-based systems, particularly the conventional activated sludge processes, and also treatment technologies for advanced nutrient removal are associated with higher energy consumption (Chae & Kang, 2013; Mamais et al., 2015; Panepinto et al., 2016). Thus, the lower energy intensity observed for the Mudor Plant attests to the claim that anaerobic WWTPs consumed less energy than aerobic processes.

4.3.2 Benefit Analysis of the Mudor WWTP

4.3.2.1 Potential of Reclaimed Water Use in Agriculture

- *Nutrients Concentration*

As mentioned early on (Chapter 2, sub-section 2.3.3.2), nutrient (N and P compounds) removal is one known deficiency of anaerobic wastewater treatment systems, which warrants the need for a post-treatment unit which could be an aerobic or chemical process. However, post-treatment with the Tricking Filters also failed to eliminate adequate nutrients from the effluent. Thus, the Mudor WWTP effluent contains high concentrations of N and P compounds. The final effluent from the secondary clarifiers, which is discharged into the Korle Lagoon, was found to have average total nitrogen (TN) and total phosphorous (TP) concentrations of 83.61 ± 24.51 mg/L and 28.37 ± 14.17 mg/L, respectively (Chapter 2, Table 2.6). Thus, with an average TN concentration of 0.0836 kg/m^3, should we consider 90% recovery of influent wastewater volume as effluent, then the recovered N-rich effluent will be 3,746 m^3/d (postulating that all effluent will be used for irrigation). The average daily load of TN in the effluent will amount to 313.17 kg/d, culminating in a yearly load of 114,305 kgN/yr (114.3 tN/yr). Likewise, a TP of 38,790 kgP/yr (38.8 tP/yr) would be recovered from the effluent. According to Bressani-Ribeiro et al. (2017), the Food and Agriculture; FAO (2015) has stated that 10 kg of N can cultivate 1 hectare of arable land per year. Thus, the TN in the effluent can cultivate an estimated 11,430 hectares of farmland. Fertigation with nutrient-rich effluent from WWTPs can reduce dependence on inorganic fertilizers, which among other things, are costly, consume high energy in their production (Chapter 3, sub-section 3.3.3) and do not promote sustainable development.

- *Heavy Metals and Pathogens Concentration*

The concentration of selected heavy metals was measured to evaluate effluent suitability for food crop irrigation. Mean concentrations have been presented in Table 4.3. Heavy metal concentrations were found to be in increasing order Hg < Ni < Pb < Zn < Cd < Cu < Mn < Cr. These concentrations have been compared to international standards. Shoushtarian & Negahban-Azar, (2020) critically reviewed worldwide regulations and guidelines for agricultural water reuse. The extensive review considered about 70 regulations from several countries and international organizations. These guidelines were based on criteria such as restricted and unrestricted irrigation, food and non-food crops, and processed food crops, with varying monitoring periods ranging from daily to monthly. However, the findings from this study have been compared to only three regulatory bodies, as presented in the table. The table shows that Ni, Zn and Pb were recorded in concentrations lesser than the limits provided by these organisations. The remaining heavy metals were in concentrations higher than the standard limits. Urban irrigation with treated and untreated domestic wastewater is a common phenomenon in Ghana. Thus, with the findings from this study, it will be imperative that appropriate regulatory bodies be tasked with developing policies and guidelines to ensure the safe practice of urban and peri-urban irrigation with reclaimed water. Notwithstanding, for microbial loads in sewage effluent, FC was found to be within the range for safe use of reclaimed water in agriculture.

Table 4.3: Permissible heavy metals and pathogens concentrations for the use of effluent in irrigation

Contaminant	Current study	(USEPA, 2012a)	(FAO)*	(WHO, 2006)
Heavy metals				
Cr (mg/L)	0.872 ± 0.520	0.1	0.1	0.1
Ni (mg/L)	0.005 ± 0.000	0.2	0.2	0.2
Zn (mg/L)	0.008 ± 0.012	2	2	2
Cd (mg/L)	0.066 ± 0.133	0.01	0.01	0.01
Mn (mg/L)	0.386 ± 0.259	0.2	0.2	0.2
Pb (mg/L)	0.005 ± 0.000	5	5	-
Cu (mg/L)	0.262 ± 0.246	0.2	0.2	0.2
Hg (µg/L)	0.358 ± 0.044	-	-	-
Pathogens				
FC (CFU/100ml)	$1.7\times10^1 \pm 1.6\times10^1$	2.0×10^2 - 8.0×10^2	2.0×10^2 - 1.0×10^3	1.0×10^3 - 1.0×10^4
E. coli (CFU/100 ml)	$1.2\times10^1 \pm 1.7\times10^1$	-	-	-
Salmonella (CFU/100ml)	$2.7\times10^1 \pm 2.9\times10^1$	-	-	-
Intestinal nematodes (No./L)	-	-	1	1

* Accessed online

4.3.2.2 Potential of Biosolids Use in Agriculture

- *Nutrient Concentration*

Dried sewage sludge obtained from the Mudor UASB reactors was characterised and found to contain 3.33 ± 0.33% (33.3 kg/m^3) of TN and 2.0 ± 0.46% (20.0 kg/m^3) of TP. With an estimated 8.4 m^3 excess sludge discharged daily (projection based on the volume and frequency of excess sludge discharged into the thickeners), the daily average load of TN in excess sludge will amount to 279.7 kg/d, culminating in a yearly load of 102,098 kgN/yr (102.1 tN/yr). Similarly, daily TP load of 168 kg/d will culminate in 61,320 kgP/yr (61.3 tP/yr). Additionally, the average volatile solids concentration was 71.1 kg/m^3, 64% of the TS. TC was found to be 29 ± 5.3%, with C:N ratio of 9.0. In a related study, Kumar et al. (2017) reported that sewage sludge generally comprises 30 - 40% TC, 3% TN, 1.5% TP, and C:N ratio between 10 and 20%, which agrees with the findings of this study. The high volumes of sludge generated from WWTPs necessitate the search for sustainable management practices to handle this waste stream. The application of biosolids on agricultural lands can be an effective strategy to improve agricultural productivity by increasing soil fertility, soil organic matter and nutrients. Additionally, biosolids can improve soil physical properties, especially in heavy textured and poorly structured soils (Alvarenga et al., 2015; Castán et al., 2016). Again, utilising sewage sludge for agriculture and soil amelioration helps to eliminate unhealthy management practices such as landfilling or sea dumping. Amorim Júnior et al. (2021) found from their study that the application of biosolids highly improved the quality of infertile soils. Biosolids use in agriculture under CE promotes sustainable agriculture. Thus, the high nutrient and organic matter concentrations observed for this study make the biosolids highly suitable to be applied on arable lands, enhancing nutrient recovery.

- *Heavy Metals and Pathogens Concentration*

Sewage sludge is rich in nutrients and organic matter, but this waste stream can also be embedded with toxic substances and pathogens, which may require careful handling practices. Without appropriate treatment processes, the direct application of sludge on agricultural lands may pose a greater risk to human and environmental health (Clarke & Smith, 2011; Islam et al., 2013). Table 4.4 presents the concentrations of heavy metals and pathogens in the sludge obtained from the Mudor UASB reactors to evaluate their suitability for agricultural use. Results from the study indicate that sewage sludge from the Mudor UASB reactors generally contains low concentrations of heavy metals in the ascending order: $Hg<Ni<Cd<Zn<Mn<Cr<Pb<Cu$, ranging from 1.046 ± 0.1891 µg/kg for Hg to 1.923 ± 0.958 mg/kg for Cu. This observation could be attributed to the fact that sewage flow to the Plant is basically from domestic and commercial centres, without industrial discharges. Currently, there are no known legislations regarding the agricultural use of sewage sludge in Ghana; hence the EU Directive 86/278/EEC will be employed as a reference guide for this study. The norms regarding the application of sewage sludge on agricultural lands at the European level are regulated by Directive 86/278/EEC (European Council Directive, 1986). However, EU member countries have the liberty to adopt their applicable guidelines, which has permitted some member states to adopt more stringent limits. Comparing the measured heavy metals concentration to Directive 86/275/EEC, it was found that the concentrations observed in this study are insignificant compared to the EU Directives, which makes them highly suitable for agricultural usage.

Despite the lower heavy metal concentrations, the microbiological analysis indicated the presence of high levels of pathogens (FC, *E. coli*, Salmonella *sp.*) and parasites (helminth eggs). Although Directive 86/278/EEC does not include limits for pathogen levels in biosolids, most EU member states have set limits for some pathogens, two of which have been randomly selected and presented in the table. Comparing the pathogen loads in the Mudor UASB reactor's sludge to the limits set by the two selected EU member states, the findings of this study far exceed their limits, making the sludge inappropriate for direct land application without adequate treatment. Thus, to valorise the nutrient resource embedded in sludge, it is proposed that sludge composting will be most appropriate, as this process will stabilize the sludge and eliminate pathogens while retaining the nutrients for improved agricultural activities.

Table 4.4: Permissible heavy metals and pathogens concentrations for the use of biosolids for Agricultural purposes

Contaminant	Current study	Directive 86/278/EEC [1]	Selected EU Member states	
			Austria [2]	Bulgaria [3]
Heavy metals				
Cr (mg/kg)	1.044 ± 1.255	-	50 - 500	500
Ni (mg/kg)	0.072 ± 0.074	300 - 400	25 - 100	350
Zn (mg/kg)	0.216 ± 0.138	2500 - 4000	200 - 2000	3000
Cd (mg/kg)	0.152 ± 0.128	20 - 40	2 - 10	30
Mn (mg/kg)	1.002 ± 2.372	-	-	-
Pb (mg/kg)	1.413 ± 0.951	750 - 1200	45 - 500	800
Cu (mg/kg)	1.923 ± 0.958	1000 - 1750	70 - 500	1600
Hg (µg/kg)	1.046 ± 0.189	16,000 - 25,000	400 - 10,000	16,000
Pathogens				
FC (CFU/gram)	$1.5\times10^3 \pm 1.4\times10^3$	-	-	-
E. coli (CFU/gram)	$1.5\times10^3 \pm 1.4\times10^3$	-	100	100
Salmonella (CFU/gram)	$5.9\times10^3 \pm 1.2\times10^4$	-	Absent in 1g	Absent
Helminth eggs (No./gram)	68.0 ± 36.0	-	Absent	1

[1] EUR-Lex Council Directive 86/278/EEC (1986); [2] Lander of Lower Austria (2019); [3] Government of Bulgaria (2016)

(Adapted from Collivignarelli et al., 2019)

4.3.2.3 Biogas Energy Recovery Potential

The characteristics of the raw sewage, biogas flow rate and composition during the study period have been presented in Table 4.5.

Table 4.5: Descriptive statistics of some major parameters obtained at the Mudor WWTP

Descriptive statistics	Raw sewage (m^3/d)	COD_{inf} (mg/L)	COD_{rem} (kg/d)	Biogas flow (Nm^3/d)	CH_4 (%)	N_2 (%)	CO_2 (%)	O_2 (%)
Maximum	6054	8150	34,194	1673	76.5	28.2	9.1	14.6
Minimum	1572	450	889	101	54.0	19.9	3.2	1.4
Average	4096	2007	6304	613	65.0	24.6	4.7	5.7
SD	837	1061	4826	271	9.0	3.1	2.2	4.6

It was found from the study that the raw sewage inflows ranged from 1572 - 6054 m^3/d, with an average of 4096 ± 837 m^3/d (47.41 ± 9.69 L/s). This flow constitutes one-fourth of the Plant's design capacity (18,000 m^3/d), indicating the Plant currently operates under capacity (Ahmed et al., 2018). Regarding the flow to the Plant, the Mudor Plant can be classified as a small-scale WWTP based on reports in the literature (Lopes et al., 2019). Volumetric biogas production ranged from 101 to 1673 Nm^3/d, with a 613 ± 271 Nm^3/d average flow. Methane gas constituted 65% of the biogas output, with an average flow of 392 ± 173 Nm^3/d. Other biogas constituents; N_2, O_2 and CO_2 were reported at mean portions of 24.6%, 5.7% and 4.7%, respectively. H_2S was in minute concentrations between 78 and 314 ppm. Despite the fraction of CH_4 being relatively lower than was reported in a previous study by Noyola et al. (2006) with 70 - 80% of CH_4, the other biogas constituents were found to agree with the findings by the same authors: 10 - 25% N_2, and 5 - 10% CO_2, for a UASB reactor treating domestic wastewater. As was reported by the authors, dissolved nitrogen in influent sewage explains the high content of N_2 in the biogas. Chernicharo et al. (2015) and Souza et al. (2011) similarly observed higher percentages (70 - 85%) of CH_4 in biogas for UASB reactors treating domestic wastewater. This study's seemingly lower CH_4 could be ascribed to such factors as sludge activity and Plant loading.

The biogas energy recovery potential (EP_{biogas}) was estimated at 14,044.59 MJ/d, which culminates into 3901.27 kWh/d (1423.96 MWh/yr). When compared to the finding by Lopes et al. (2019), the observed biogas energy potential (14.04 GJ/d) in this study was found to be insignificant to that reported by the authors (380 GJ/d) for a typical scenario. This observation could be attributed to the fact that the Mudor Plant operates under capacity presently, as stated earlier. Nevertheless, with projected population growth and consequent increase in sewage flows, the energy potential of biogas produced by the Plant would increase. Additionally, this observation could be attributed to the low OLR applied to the Plant as reported earlier (Chapter 2; Table 2.3), strengthening the assertion by Ahmed et al. (2018) that the Mudor WWTP treats typically low-strength sewage.

4.3.2.4 Characteristics of Sewage Sludge for Energy Recovery Potential Evaluation

Findings on parameters essential to evaluate sludge's applicability for thermal conversion processes for energy recovery have been presented in Table 4.6. As Syed-Hassan et al. (2017) reported, proximate and ultimate analyses are useful for the evaluation of the thermochemical conversion characteristics of fuel. Similarly, Chiang et al. (2012) mentioned that proximate and ultimate analyses provide an estimate of feedstock efficiency for power generation as well as the yield of fuel by-products used in thermal conversion systems. For the proximate analysis, MC was found to range from 63 - 82%, with VM between 50.5 and 80.9%. FC and ash contents ranged from 2.4 - 5.2% and 19.0 - 49.5%, respectively. These findings are comparable to the results reported in similar studies (Syed-Hassan et al., 2017; Tic et al., 2018). Furthermore, the elemental constituents were found to be in average percentages of 28.5 ± 5.27% for C, 11.8 ± 0.64% for H. N, S, and O were found to be 3.33 ± 0.33%, 1.14 ± 0.32% and 27.5 ± 6.5%, respectively (see Table 4.6). The obtained results for elemental constituents of sewage sludge from the Mudor Plant were found comparable to the range of values reported in the literature (Soria-Verdugo et al., 2013; Syed-Hassan et al., 2017; Tic et al., 2018) for various studies conducted to evaluate sewage sludge's applicability for energy recovery through the adoption of thermochemical conversion processes.

Table 4.6: Proximate and ultimate analysis of dewatered sludge at Mudor WWTP

Parameter	Current Study		Reported ranges in the literature	References
	Range	Average		
Proximate Analysis				
Moisture Content (wt %)	63.00 - 82.00	75.00 ± 2.60	73.40 - 86.40	(Chan & Wang, 2016)
Volatile matter (wt %)[a]	50.50 - 80.90	62.90 ± 5.50	21.70 - 82.30	(Syed-Hassan et al., 2017; Tic et al., 2018)
Ash content (wt %)[a]	19.00 - 49.50	36.60 ± 5.10	10.80 - 76.80	(Syed-Hassan et al., 2017)
Fixed carbon (wt %)[a]	2.40 - 5.20	3.10 ± 1.20	1.81 - 21.80	(Syed-Hassan et al., 2017)
Ultimate Analysis				
Carbon (wt %)	22.30 - 32.80	28.5 ± 5.27	32.1 - 69.3	(Syed-Hassan et al., 2017; Tic et al., 2018)
Hydrogen (wt %)	11.02 - 12.69	11.8 ± 0.64	3.85 - 8.60	(Syed-Hassan et al., 2017; Tic et al., 2018)
Nitrogen (wt %)	2.68 - 3.82	3.33 ± 0.33	2.25 - 9.08	(Syed-Hassan et al., 2017)
Sulphur (wt %)	0.31 - 1.56	1.14 ± 0.32	0.60 - 2.05	(Syed-Hassan et al., 2017)
Oxygen (wt %)	15.90 - 36.20	27.50 ± 6.50	18.20 - 56.30	(Syed-Hassan et al., 2017; Tic et al., 2018)

[a] *Dry basis; wt = weight*

Analysis of the calorific value of biomass fuel provides information on biomass energy content (Chiang et al., 2012). The results on sludge heating values (Figure 4.6), revealed that the actual gross calorific value GCV(a) and actual net calorific value NCV(a) obtained from the bomb calorimetry experiment were lower than the theoretical values; GCV(t) and NCV(t). The average GCV(a) was found to be 14.6 ± 1.1 MJ/kg, against the average GCV(t) of 16.3 ± 1.4 MJ/kg. A similar pattern was observed for the net heating values, with an average NCV(a) of 9.8 ± 1.1 MJ/kg, whilst the average NCV(t) was 13.8 ± 1.3 MJ/kg, as shown in the figure. This study's GCV and NCV values were relatively higher compared to those recorded in a parallel study in Brazil (Rosa et al., 2018). The Brazilian study reported average values of 8.7 ± 1.2 MJ/kg and 7.4 ± 1.4 MJ/kg for the GCV actual and theoretical, respectively. They also reported 2.0 ± 0.8 MJ/kg and 1.7 ± 1.8 MJ/kg, for the actual and theoretical NCVs, respectively.

Singh et al. (2020) asserted that higher heating values of sludge are associated with higher VM contents. This assertion was evident in this study, where VM ranged from 50 - 80%, resulting in higher heating values. The actual GCV of sewage sludge has been reported in several studies to range from 11 - 22 MJ/kg (García et al., 2013; Vamvuka et al., 2015; Yu et al., 2015). The observed wide variation in the calorific values (CV) could be ascribed to factors such as the applied treatment processes. Different sludge types, such as activated sludge, raw primary sludge and anaerobically digested sludge were found to have different heating values ranging from 8.9 to 23 MJ/kg (Gezer Gorgec et al., 2016). Galhano dos Santos & Bordado (2018) studied the correlation between GCV and ultimate analysis parameters. The authors concluded that high C and H contents were the most significant elemental constituents, representing higher energy content. When compared to the literature, the H values observed in this study (11.02 - 12.69%) were higher than the range (3.85 - 8.60%) reported. Despite the C values (22.30 - 32.80%) not very high as reported (32.1 - 69.3%), the obtained actual GCV was within the reported range.

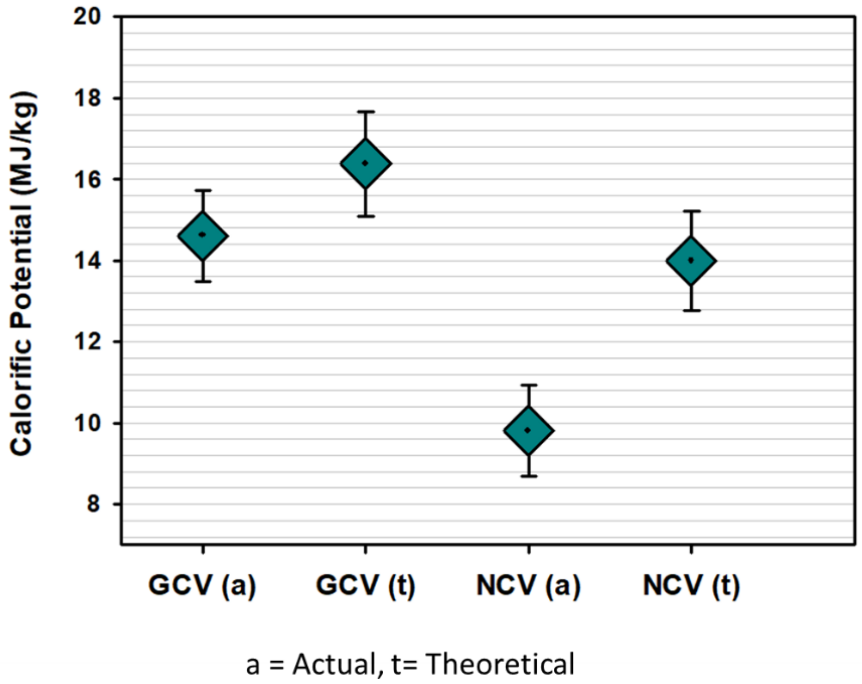


Figure 4.6: Actual and theoretical GCVs and NCVs for the dehydrated sludge of the Mudor WWTP

4.3.2.5 Sewage Sludge Energy Recovery Potential

Based on the projected volume of dewatered sludge transported to the drying beds biweekly during the study period, the mass of dry sludge produced was estimated to be 358.24 TS kg/d (130.76 tonnes/yr). The average NCV of 9.81 MJ/kg (Figure 4.6), resulted in an estimated sewage sludge's energy recovery potential of 3514 MJ/d (3.5 GJ/d), translating into 356.31 MWh/yr for thermochemical process energy recovery. Compared to the study by Rosa et al. (2018), they found that a filter-pressed dehydrated sludge mass of 3759 TS kg/d, translated into an energy recovery potential of 7518 MJ/d, much higher than was found in this study. In a related study by Lopes et al. (2019), it was revealed that for typical scenarios, the energy recovery potential of full-scale UASB reactors sewage sludge dehydrated with drying beds ranged from 15 GJ/d to over 100 GJ/d. The same study reported the worst scenarios for small-scale WWTPs with energy recovery potentials as low as 0.2 GJ/d. Several factors, including the population served, characteristics of influent sewage, applied sludge retention times (SRTs), and sludge drying methods can influence dry sludge production volume, and subsequently, the energy recovery potential of sewage sludge (Gezer Gorgec et al., 2016).

It is worth noting that although the energy recovery potential of sludge generated by the Mudor UASB reactors may seem promising, an undisputable fact remains that the low sludge volumes generated from the Plant resulted in low energy recovery potentials. Nevertheless, an alternative to maximising the sludge energy recovery potential is via AD. Different authors have iterated the prospects of recovering energy from sludge through this process. Qi (2013) mentioned that the AD of sewage sludge was a more profitable venture for WWTPs with capacities larger than 22,000 m^3/d, which produce large volumes of sludge for energy recovery. Bachmann (2015) found that sewage sludge energy potential from AD ranges from 42 to 3050 GWh/yr for some WWTPs in Europe, North America and some Asian countries. Other authors equally recommended co-digestion of sewage sludge with organic feedstock such as food waste, livestock manure and intestines, plant biomass etc., ensuring the generation of methane-rich biogas for energy recovery purposes (Hallaji et al., 2019; Vinardell et al., 2021). Thus, the readily available organic feedstock in Ghana (Arthur et al., 2011, 2020; Präger et al., 2019) makes co-digestion a sustainable option to recover energy from the sludge generated by the Mudor UASB reactors.

4.3.2.6 Energy Self-sufficiency of the Mudor WWTP through Biogas and Sludge Energy Recovery

Biogas and sewage sludge energy recovery remain the surest way by which the Mudor WWTP can be energy self-sufficient. EP_{biogas}, EP_{sludge} and EP_{Total} estimations were made based on the monthly CH_4 and dry sludge matter production. EP_{biogas} and EP_{sludge} amounted to 1423.96 MWh/yr and 356.31 MWh/yr, respectively. EP_{Total} was estimated at 1780.27 MWh/yr. Assuming a 30% efficiency of the electricity conversion technology as reported in the literature (Lopes et al., 2019; Rosa et al., 2018), the resulting EP_{Total} will be 534.1 MWh/yr.

Figure 4.7 presents a graph of EP_{biogas} and EP_{sludge} plotted against the actual energy demand of the Plant during the study period. The EP_{biogas} ranged from 18,408 kWh/month to 52,515 kWh/month. Based on the biweekly excess sludge withdrawal from the UASB reactors, a constant EP_{sludge} of 8770 kWh/month was estimated for the entire study period. It was found that the majority (80%) of total by-products' energy recovery potential was from biogas, whilst sludge energy potential accounted for only 20%. Similarly, Lopes et al. (2019) reported values between 65 and 74% of biogas energy recovery potential for small, medium and large-scale WWTPs, as against sludge energy recovery potential. Moreover, they found that treatment plants which employed sludge drying beds recorded an average biogas energy recovery potential of 64% of the total by-product's energy recovery potential. In their study, Rosa et al. (2016) observed similar trends, where sludge energy recovery potential of 41% was obtained, against biogas energy recovery potential of 59.3%. The study by Bachmann (2015) likewise revealed that for WWTPs in some European countries, 7 - 49% of the total biogas produced was from sewage sludge. The authors concluded that other biogas sources dominated the overall energy recovery balance. These assertions made by previous authors agree with the findings of this study.

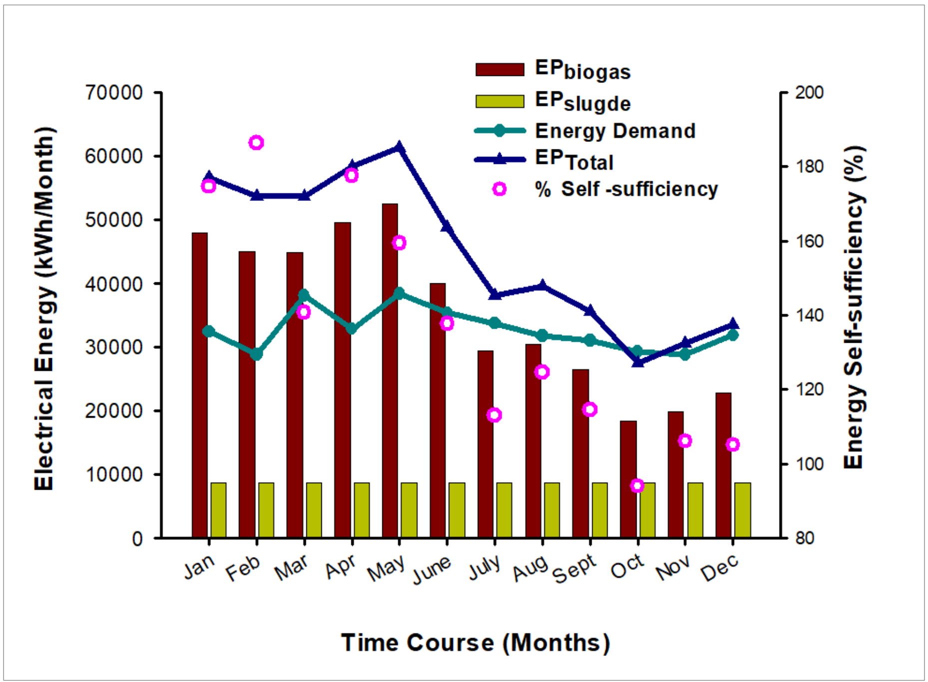


Figure 4.7: Energy recovery potential from biogas and sludge

Figure 4.7 again presents the total estimated energy recovery potential (EP_{Total}) from both biogas and sludge, plotted against the actual Plant energy demand. It is seen from the figure, that for each month, except for October, energy recovery from both by-products could offset the total energy demand of the Plant. The percentage of energy self-sufficiency ranged from 94% for October to 186% for February for energy recovery from the by-products. The study also revealed that EP_{biogas} estimated for January through June could completely offset the energy demand for those periods. Additionally, a steady drop was observed for EP_{biogas} from June through to December, highly attributable to the drop in biogas production during those periods (detailed explanation in section 2.3.4). Notwithstanding the observed reduced EP_{Total} for some of the months, the overall EP_{Total} recorded during the study period (534.1 MWh/yr) could completely offset the Plant energy demand (392.7 MWh/yr). Thus, with biogas and sludge energy recovery, the Mudor WWTP can move from energy neutral to energy positive. This finding provides evidence of the plausibility of "*sanitation financing sanitation*" through sustainable wastewater management practices in an emerging economy such as Ghana.

As discovered in this study, the potential of the Mudor WWTP to be energy self-sufficient is comparable to other reports from different regions of the world. Several studies have been reported wherein complete energy self-sufficiency of WWTPs was attained, especially in North American and European countries such as Austria, the USA and Germany through biogas and sludge energy recovery (Gu et al., 2017; Maktabifard et al., 2018; Nowak et al., 2011). Similarly, studies in developing countries such as India and Brazil (Guo et al., 2019; Jangid & Gupta, 2014; Singh et al., 2020) have reported the possibility of offsetting at least 75% of a treatment Plant's energy demand through biogas and sludge energy recovery. It should be noted that if the Mudor WWTP, which operates currently at approximately one-fourth of its design capacity can completely offset its energy needs, then should the Plant be operated at full capacity (anticipated soon), the Mudor WWTP can provide significant energy to augment the national electricity grid, which could mitigate the persistent energy crises in the country.

4.3.2.7 Sewage Sludge Biofuel (Biochar) Recovery Potential

The circular economy strategy for wastewater management postulates the search for more sustainable alternatives for managing sewage sludge. Thermochemical treatment processes seem to be among the most promising approaches. Excess sewage sludge produced at the Mudor WWTP is treated thermochemically through pyrolysis to produce biofuel (biochar). This development is still at the pilot stage. When biofuel recovery potential from the excess sludge was determined, it was found that the production of 358.24 TS kg/d culminated in 130.76 tonnes/yr of dry sludge. Meanwhile, Singh et al. (2020) found that 2.17 kg of dry sewage sludge could produce 1 kg of biochar. Thus, the daily dry sewage sludge generated can produce an equivalent of 165.1 kg of biochar per day, with a yearly production of 60,261.5 kg of biochar.

In line with sustainable development, biochar production could reduce dependence on wood fuels and charcoal. Over 2 billion people living in developing countries, especially SSA, rely on wood fuels for their primary energy needs, such as cooking and heating (UN Department of Economic and Social Affairs, 2019). Again it has been reported that an estimated 80% of the rural population in Ghana depends on wood fuel, whilst 50% of the urban population relies on charcoal as their primary fuel source (UNDP, 2016). Moreover, the Ghana Energy Commission has found that wood fuels provide the bulk of the energy needs for most informal enterprises in Ghana

(Ghana Energy Commission, 2006). Meanwhile, the high usage of charcoal has led to the uncontrolled cutting of trees, resulting in deforestation. Deforestation is one major contributor to climate change. Due to this, several international and local organizations are calling for an end to the uncontrolled cutting of trees. In this regard, a switch from reliance on wood fuels and charcoal to biochar produced from sludge which is readily available and in excess would be a move towards sustainable development.

4.3.2.8 Proposed Resource Recovery Scenarios for the Mudor WWTP

Municipal wastewater treatment has, in recent times, become a subject of interest within the Water-Energy-Food nexus, given that it allows the recovery of all three resources; water, energy and nutrients. Thus, sustainable wastewater management provides an additional value proposition besides the protection of the aquatic environment (Drechsel et al., 2015). Currently, there is a paradigm shift underway from a notion that wastewater is another waste stream to be treated and disposed of to one that stirs interest in recovering valuable resources in support of a circular economy that can offer environmental and economic benefits (Otoo & Drechsel, 2019; Rao et al., 2017).

The current study has evaluated the resource recovery potential of a full-scale UASB/TF system treating municipal wastewater in Accra. Although several resources can be recovered from wastewater, this study focused on three major streams; water, nutrients and energy recovery (Figure 4.8). The study revealed the possibility of recovering reclaimed water, which can be employed for non-potable usages or irrigation activities due to the high concentration of nutrients embedded. The study again revealed that excess sludge produced from wastewater treatment processes is rich in organic matter and nutrients, making the biosolids highly suitable for soil reparation, especially on arable lands, to improve food production. Additionally, it has been found that air-dried sewage sludge produced from municipal wastewater treatment has a high calorific value, making it suitable for energy recovery through pyrolysis for biofuel (biochar) production. Finally, methane-rich biogas is also viable for energy recovery.

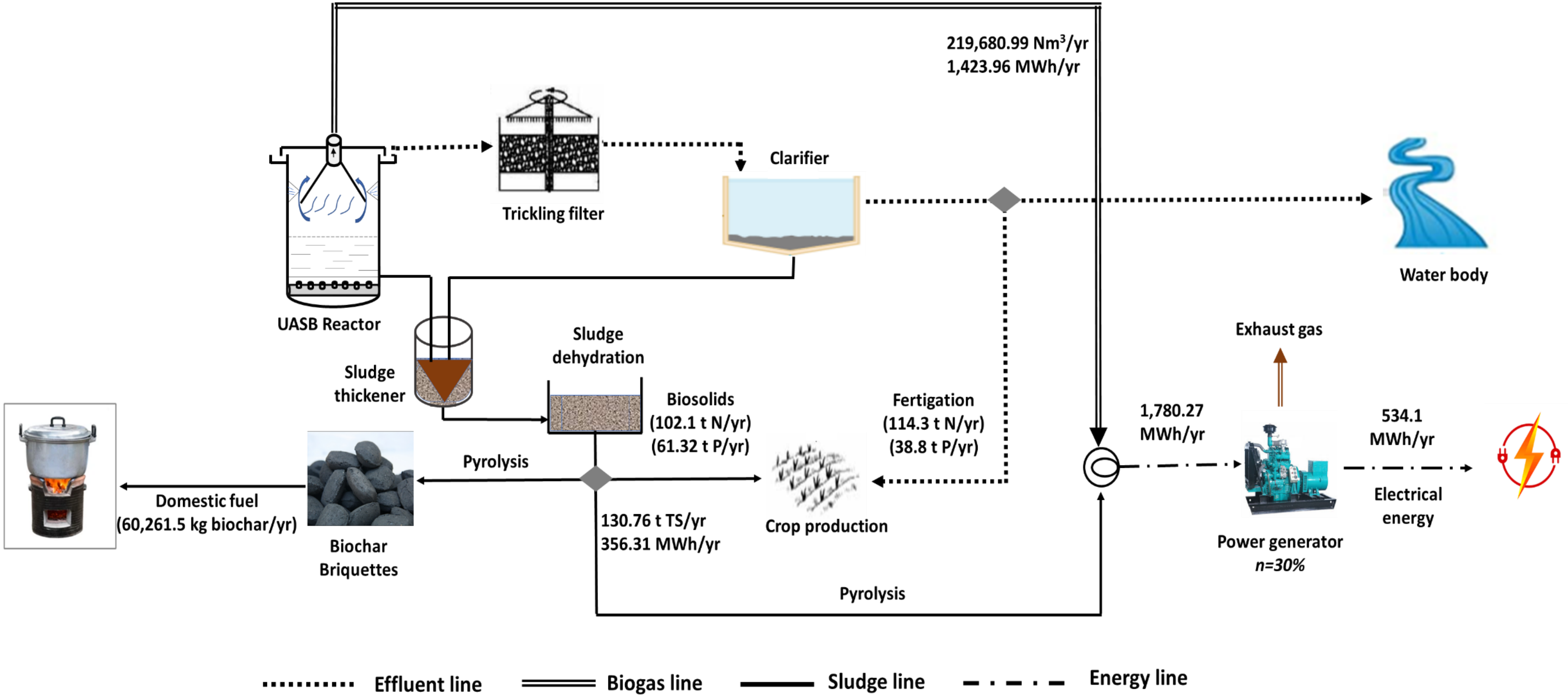


Figure 4.8: Potential resource recovery scenarios for the Mudor WWTP

4.4 Partial Conclusion

This study evaluated the economic implications of a UASB-based WWTP. The study employed the cost-benefit analysis approach to assess the Mudor WWTP economically. Cost analysis revealed that the Plant built in the year 2000 has a monthly operational cost of GHS 700,000, of which the highest component – 37% – is apportioned to staff management. The anaerobic-based WWTP's energy consumption was on the lower side at a meagre 7.3% of the total operational cost. Integrated resource recovery under circular economy was employed for the benefit analysis. It was found that the wastewater effluent was embedded with high concentrations of nitrogen, phosphorus and low heavy metals concentrations. Microbial loads in effluent were likewise revealed to be within limits set by WHO for irrigation purposes, making effluent wastewater a reliable source of fresh water for fertigation. Excess sludge withdrawn from the UASB reactors was also found to contain high concentrations of organic matter, embedded with nutrients and suitable as biosolids for soil conditioning to improve the quality of arable lands. The average biogas flow for the Plant is 613 ± 271 Nm^3/d, with 65% CH_4 of the biogas output. Dry sludge matter production is estimated at 358.24 TS kg/d (130.76 tonnes/yr), with an average NCV of 9.81 MJ/kg. The Mudor WWTP's overall gross energy potential in the form of electricity from biogas and sludge is 1780.3 MWh/yr. A conservative rating of 30% energy conversion efficiency could give a net energy (electricity) production of 534.1 MWh/yr, which is 36% in excess of the actual energy demand of the Mudor WWTP. Thus, the Mudor WWTP has the potential to supply its energy and wean itself from the national electricity grid in support of its operations. Therefore, integrated resource recovery from wastewater treatment by-products remains the surest way sustainable wastewater management can be attained for sustainable development.

General Conclusions and Perspectives

General Conclusions

This study has expounded the UASB reactor technology as an economically feasible and more sustainable wastewater treatment option for developing countries to attain sustainable wastewater management. The study evaluated the techno-enviro-economic dimensions of sustainability of the UASB reactor technology. The study focused on a full-scale UASB reactor coupled with TFs employed for municipal wastewater treatment in Accra, the capital of Ghana. The technical sustainability assessment revealed that the UASB reactors performed satisfactorily, with approximately 70% removal efficiency for COD and TSS, and 86% for BOD. Post-treatment units further enhanced performance, with overall removal estimated at 86%, 97% and 91%, respectively, for COD, BOD and TSS. System performance regarding microbial loads elimination revealed satisfactory performance for the UASB reactors with 80% removal efficiency for Faecal Coliforms, *E. coli* and Salmonella *sp.* Post-treatment with the TFs and final settling further improved microbial load reduction to one log unit. The system, however, failed to remove adequate nutrients (nitrogen and phosphorus compounds) from the sewage, producing nutrient-rich effluent. The study again revealed that the average biogas flow from the anaerobic reactors was 613 ± 271 Nm^3/d, with 65% methane output. However, 23% of the methane produced remained dissolved in the effluent, reducing the energy recovery potential of the biogas. Specific methanogenic activity (SMA) test revealed that inoculum to substrate ratio (ISR) of 1:1 resulted in the highest methane production. Further study on the material conversion route in the UASB reactors employing a COD mass balance assessment revealed that only 33.5% of the influent COD load applied was converted to methane gas which is available for use. Other conversion routes included COD which remained in the effluent (27.7%), COD converted to sludge (15.4%), COD used in sulphate reduction (4.4%), COD converted to methane and remained dissolved in the effluent (13.7%), and lastly, COD converted to methane and lost through leakages and waste gas (5.3%). Environmental sustainability was evaluated by measuring the carbon footprints of the full-scale UASB/TF wastewater treatment plant. Emission sources identified during the study included on-site emissions such as emissions from biogas flaring (GHG_{flare}), emissions from methane leakages from the reactors and through the biogas lines ($GHG_{CH_4\text{-leakage}}$), emissions from sludge treatment with drying beds ($GHG_{sludge\text{-}CH_4}$), emissions from the combustion of diesel fuel to run generators during interruptions in national grid electricity supply (GHG_{diesel}) and N_2O emissions from biological nitrogen removal processes (nitrification and denitrification) at the Trickling Filters ($GHG_{N_2O\text{-}WWT}$). Off-site emissions considered during the study were methane emissions from dissolved methane in effluent discharged into the recipient water body (GHG_{dCH_4}), N_2O

emissions from the discharge of nitrogen-rich effluent into recipient water bodies ($GHG_{N_2O\text{-}Effluent}$) and indirect emissions from the use of national grid electricity (GHG_{Electr}). This study employed the Intergovernmental Panel for Climate Change (IPCC) greenhouse gas inventory methodology to estimate GHG emissions. It was found from the study that the total estimated emissions from the operations of the full-scale WWTP were 39,619.36 tCO_2eq/yr. dCH_4 was identified as the single significant source of methane emissions, contributing 95.1% of the total emissions. Avoided emissions estimated from the energy recovery from biogas and sludge produced by UASB reactors and the use of nutrient-rich effluent to offset the use of inorganic fertilizers for agricultural purposes, could result in negative emissions of 1334.26 tCO_2eq/yr, resulting in net emissions of 38,285.10 tCO_2eq/yr. Resource recovery remained the surest way emissions from the Plant could be abated. Economic sustainability assessment employing cost-benefit analysis revealed that staff management presented the highest cost element, responsible for 37% of the total annual operating costs of the Plant. Energy consumption represented just 7.3% of the total operating cost. Integrated resource recovery under circular economy was employed for the benefit analysis. It was revealed that wastewater effluent is embedded with high concentrations of nitrogen and phosphorus (0.0836 kgN/m^3 and 0.0284 kgP/m^3, respectively), with low heavy metals concentrations. Microbial loads in effluent were likewise found to be within limits set by WHO for irrigation purposes. This makes effluent wastewater a reliable source of fresh water for fertigation purposes. Withdrawn excess sludge from the UASB reactors was also found to contain high concentrations of organic matter embedded with nutrients and suitable as biosolids for soil conditioning to improve the quality of arable lands. With an average biogas flow of 613 ± 271 Nm^3/d, comprising 65% CH_4, and dry sludge matter production of 358.24 TS kg/d (130.76 tonnes/yr), the Mudor WWTP's overall gross energy recovery potential in the form of electricity from biogas and sludge by-products is 1780.3 MWh/yr. A conservative rating of 30% energy conversion efficiency could give a net energy (electricity) production of 534.1 MWh/yr, which exceeds the actual energy demand (392.7 MWh/yr) of the Mudor WWTP. Thus, the Mudor WWTP has the potential to supply its energy and wean itself from the national electricity grid in support of its operations. Findings from this study have revealed that the employment of CE concepts through integrated resource recovery could lead to sustainable wastewater management. The study again found that resource recovery under CE has direct and indirect connections towards the attainment of the SDGs set by the United Nations for sustainable development, especially for emerging economies. In conclusion, the UASB reactor technology has been proven by this study to be an efficient, economically feasible and sustainable technology that can and should be implemented in developing countries in order to achieve sustainable development in these regions.

Perspectives

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

- Urban vegetable farming is a common practice in Accra, the study site. It is therefore recommended that feasibility studies on the conveyance of nutrient-rich effluent to the farmlands be conducted to allow urban farmers access to this resource, which would cut down operational expenses and boost food production whilst minimizing N_2O emissions associated with wastewater treatment.
- Further studies on economically feasible technologies to recover dissolved methane in effluent should be conducted. 23% of methane dissolved in effluent significantly reduces the energy recovery potential of biogas. Simultaneously, this is the single significant source of GHG emissions from the Plant, threatening environmental sustainability.
- Energy recovery facilities should be incorporated at the Plant to facilitate energy recovery from biogas and sludge by-products. This will promote the "*Sanitation financing Sanitation*" concept and make the UASB reactor technology economically sustainable for developing countries.
- This study focused solely on the carbon footprints of the Mudor WWTP. Therefore, a comprehensive life cycle assessment is recommended to evaluate the overall environmental implications of the full-scale UASB-based municipal WWTP.
- Future studies should be conducted with a laboratory pilot-scale UASB reactor for controlled operational parameters. This would permit the modification of the various parameters to evaluate their influence on biogas production, methane yield, effluent and sludge quality for system optimization.

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Appendices

Appendix 1: Peer-Reviewed Articles

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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 ± 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 ± 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 ± 292.7 m^3/d. Average methane composition was 64.7 ± 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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Appendix 2: Analytical methods for wastewater parameters, equipment and models

PARAMETER	LABORATORY METHOD	EQUIPMENT, MODEL AND COUNTRY	METHOD REFERENCE	RL
pH	Direct measurement	Multi- probe analyser, HQ40D (HACH-USA)	APHA	0.1
Temperature	Direct measurement	Multi- probe analyser, HQ40D (HACH-USA)	APHA 2550	0.1
DO	Direct measurement	Multi- probe analyser, HQ40D (HACH-USA)	HACH HQ40D manual	0.01
EC	Direct measurement	Multi- probe analyser, HQ40D (HACH-USA)	APHA 2520	10
TDS	Direct measurement	Multi- probe analyser, HQ40D (HACH-USA)	HACH HQ40D manual	2
BOD_5	Dissolved oxygen method	Binder BD-53 Incubator, (Germany)	APHA 5210	5
COD	Potassium dichromate digestion method	DR 1900 Spectrophotometer, (HACH – USA)	APHA 5220	5
TS	Drying at 105 ºC	Oven – Faithful, WHL- 45B, (China)	APHA 2540	2
TSS	Drying at 105 ºC	Oven – Faithful, WHL- 45B, (China)	APHA 2540	2
TVS	Ignition at 550 ºC	Daiham Scientific Furnace, FX - 03 (South Korea)	APHA 2540	0.1
Total Alkalinity	Lovibond method	Spectrophotometer XD 7500 (UV-VIS) (UK)	Lovibond M35 Manual	5
VFAs	Distillation method	Simple distillation set-up	APHA 5560C	5
TP	Persulfate digestion	DR 3900 Spectrophotometer (USA)	HACH method 10209	0.03
PO_4^{3-}-P	Colorimetric analysis	DR 3900 Spectrophotometer (USA)	HACH method 10210	0.02
TN	Persulphate digestion	DR 3900 Spectrophotometer (USA)	HACH method 10208	0.05
NH_3-N	Salicylate method	DR 3900 Spectrophotometer (USA)	HACH method 10031	0.02
NO_3^--N	Cadmium reduction method	DR 3900 Spectrophotometer (USA)	HACH method 8039	0.1
SO_4^{2-}	SulfaVer4 method	DR 3900 Spectrophotometer (USA)	HACH method 8051	1
Sulphide	Methylene Blue Method	DR 3900 Spectrophotometer (USA)	HACH method 8131	1
FC	Pour plate method	Memmert Oven Model UFB 500 (Germany)	SGM, (2006)	1
E. coli	Pour plate method	Memmert Oven Model UFB 500 (Germany)	SGM, (2006)	1
Salmonella sp.	Pour plate method	Memmert Oven Model UFB 500 (Germany)	SGM, (2006)	1
Helminth eggs	AmBic/$ZnSO_4$ method	OPTIKA, B-380 binocular microscope (Italy)	WRC Report No. TT322/08	1
Zn, Cu, Cd, Ni, Hg, Mn, Cr	Atomic Absorption Spectrometry	Perkin Elmer AAnalyst 800 spectrometer, (USA)	NIOSH, (1994)	0.001
Pb	Atomic Absorption Spectrometry	Perkin Elmer AAnalyst 800 spectrometer, (USA)	FAAS Method 7082	0.001

RL = Reporting Limit

Appendix 3: Pictures taken during the study

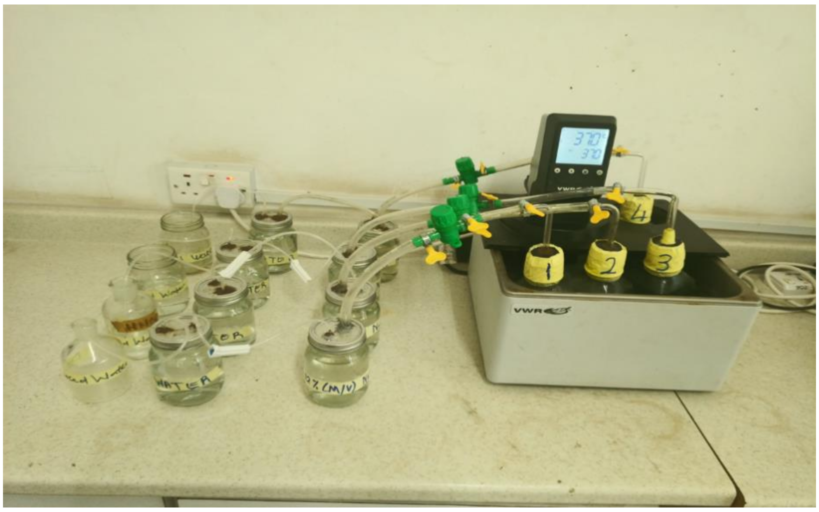

Figure 3-A: The SMA Test Experimental Setup

Figure 3-B: The Penstock

Figure 3-C: The Coarse Screens

Figure 3-D: The Vortex Grits

Figure 3-E: The Fine Screens

Figure 3-F: The UASB Reactors and Sludge Thickeners

Figure 3-G: The Trickling Filters

Figure 3-H: The Clarifiers

Figure 3-I: The Sludge Drying Beds

Figure 3-J: Biogas Flaring Unit