ONSITE TREATMENT OF DOMESTIC GREYWATER USING CONSTRUCTED WETLAND IN GHANA

By

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Declaration

I hereby declare that this submission is my own work and that, to the best of my knowledge and belief, it contains no material previously published or written by another person nor material which to a substantial extent has been accepted for the award of any other degree or diploma at Kwame Nkrumah University of Science and Technology, Kumasi or any other educational institution, except where due acknowledgement is made in the thesis.

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Abstract

Domestic greywater is the most neglected sanitation component in Ghana, likely due to poor wastewater management in general. Sewerage coverage is less than 10%, and onsite systems mostly cater for blackwater and not greywater. Environmental and public health risks from greywater are inevitable although information on same is scanty. This study aimed at identifying and incorporating indigenous knowledge and practices of greywater disposal into a low-cost green technology like constructed wetlands (CW). First, data collection involved 451 surveyed houses in nine communities. Data was also generated through literature reviews, and laboratory analysis of greywater samples. Horizontal flow subsurface CW was indigenized by incorporating into design local vegetation – taro (*Colocasia esculenta*) and sugarcane (*Saccharum officinarum*), and local media (gravels and laterite: d_{10} = 5.5 mm & d_{3060}= 0.1 – 7 mm), and then tested. Eight experimental setups including controls were operated and monitored under residence times (HRT) of 1, 2 & 3 days, repeated for five batch runs (with 187 ml/s feeding for 1hr) between June and October 2018. Findings showed that greywater were disposed of mostly into the open (46–66%), and few (4–24%) by septic tanks/soakaways. Most respondents (84%) perceived plants usage as beneficial treatment agents in greywater disposal. Mostly used plants included sugarcane, banana/plantain, and taro among 36 plant species identified 1,259 times. Greywater characteristics showed high contaminant levels: turbidity (39.4 – 2,880 NTU), BOD_5 (64 – 700 mg/L), COD (207 – 2,308 mg/L), TSS (70 – 4,720 mg/L), TDS (420 – 2,860 mg/L), nutrients – TKN, NH_3-N, NO_3-N, NO_2-N (0 – 218.5 mg/L), TP and PO_4^{3-} (1.24 – 26.18 mg/L), anionic surfactants - AnS (2 – 10 mg-LAS/L), SO_4^{2-} (13 – 15 mg/L), SAR – 0.6 $(meq/l)^{½}$, average BOD_5/COD ratios ≥0.5, and microbial – TC, FC, and *E. coli* (2.95 – 10.4 log CFU/100ml). Greywater generation at 95% CI is 39 – 83 l/c/d with specific pollutants loads of 8 – 18 g/c/d (BOD_5) and 24 – 48 g/c/d (COD). CW performance showed the following effluent quality (mean ± standard deviation): DO (1.34±0.45 mg/L), TDS (186.5± 30.29 mg/L) and EC (380.17±42.02 µS/cm) all increased and passed discharged limits, but not NH_3-N (5.94±1.68 mg/L), P (1.56±1.10 mg/L) and Fe (4.9±3.81 mg/L). SO_4^{2-} was almost always 100% removed with few exceptions (0 – 2.8 mg/L). NO_2-N removal followed SO_4^{2-} quite closely. Effluent contaminants levels and removal efficiencies also included NO_3-N (0.2 – 1.2 mg/L, 81 – 96%) >BOD_5 (23 – 37 mg/L, 77 – 90%) >COD (45 – 81 mg/L, 69 – 86%) >TSS (12 – 27 mg/L, 59 – 81%) >AnS (1.3 – 2.1 mg/L, 42 – 75%) and >PO_4 (1.8 – 9 mg/L, 30 – 86%). Two-way MANOVA tests showed that effluent quality was significantly influenced by wetlands [Pillai's Trace = 1.790, $F(63, 658) = 3.590$, $p<0.001$], and HRT [Pillai's Trace = 0.449, $F(18, 178) = 2.859$, $p<0.001$], but not their interactions ($p=0.486$). CW features like media, vegetation and baffle have influence on performance. Prediction models fitted for main organic contaminants could explain effluent variabilities of 37% (BOD_5), 62% (COD), and 73% (AnS). Indigenous greywater disposal practices offer opportunities for low-cost technology adaptation. Ghanaian greywater is polluted, fail wastewater discharge limits, but suitable for biological treatments like CW. The designed CW is effective for treating greywater to acceptable standard by regulatory discharge limits for almost all tested parameters except effluent NH_3-N, Fe and P. Yet, further improvement and better understanding of performance of the designed CW under long-term operational conditions are needed.

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List of Abbreviations

AnS	Anionic surfactants
As	Arsenic
BOD	Biochemical oxygen demand
BOD_5	Biochemical oxygen demand at five days incubation period
Ca	Calcium
Cd	Cadmium
CFU	Colony forming units
CW(s)	Constructed wetland(s)
EC	Electrical conductivity
EPA-Gh	Environmental Protection Agency, Ghana
FAO	Food and Agriculture Organization
FC	Faecal coliform
Fe	Iron
FWS	Free water surface flow wetlands
GSS	Ghana Statistical Services
Hg	Mercury
HRT	hydraulic retention/residence time
HSSF	Horizontal subsurface flow wetlands
HWS	Hybrid wetland systems
iHFSCW(s)	Indigenized horizontal flow subsurface constructed wetland(s)
K	Potassium
K(VIP)	Kumasi (Ventilated Improved Pit) latrine
KNUST	Kwame Nkrumah University of Science and Technology
LAS	Linear akyl sulfonates
MANOVA	Multivariate analysis of variance
MDGs	Millennium Development Goals
MMDAs	Metropolitan, Municipal and District Assemblies
N	Nitrogen
Na	Sodium
NESP	National Environmental Sanitation Policy
NTU	Nephelometry unit
Ortho-P	Ortho Phosphate species
Ortho-P/P	Ortho Phosphate and Phosphorus species
OTR	Oxygen transfer rate
P	Phosphorus
Pb	Lead
PE	Population equivalent

SAR	Sodium absorption ratio
SDGs	Sustainable Development Goals
SFF	Subsurface flow wetlands
TC	Total coliform
TDS	Total dissolved solids
TKN	Total Kjeldahl nitrogen
TN	Total Nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UNICEF	United Nations Children's Fund
VSSF	Vertical subsurface flow wetlands
WC	Water closet
WHO	World Health Organization
Zn	Zinc

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Ethical Approval

The ethical clearance for the study was secured from the joint Committee on Human Research, Publication and Ethics of the Kwame Nkrumah University of Science and Technology (KNUST) and Komfo Anokye Teaching Hospital (KATH). The approval number is CHRPE/AP/240/17.

Dedication

This PhD thesis is first and foremost gracefully dedicated to the Lord and Strength of my life, my ***Saviour Jesus Christ and God Almighty*** by whose **Holy Spirit** and inspiration the entire study was successfully accomplished. (Psalm 27:1, Isaiah 9:6, Zechariah 4:6).

It is also specially dedicated to my sweetheart, lifetime partner and lover - Ophelia my wife, and to our wonderful children Jeriel, Shalom-El and Annaelle. Your prayers, love and passionate support deserve this dedication. Shalom!!

Chapter 1: General Introduction

1.1 Background of the study

Domestic greywater or sullage is wastewater from all sources except excreta or faecal matter (Al-Mamun et al., 2009; WHO, n.d-b). Greywater constitutes the largest part (50 – 80%) of all domestic wastewater flows (Kasak et al., 2011; Wurochekke et al., 2016). Greywater is increasingly receiving attention as alternative water resource especially in arid and semi-arid settings of the world (Ramprasad and Philip, 2018; Mohamed et al., 2019). While greywater management is effective and well advanced in the developed world, little or no attention is received in most developing countries (Morel and Diener, 2006; Carden et al., 2007; Hyde and Maradza, 2013; Dwumfour-Asare et al., 2017). Meanwhile, the risk associated with ineffective greywater management continues to be high including environmental degradation and pollution especially of water bodies, and public health threat issues (Morel and Diener, 2006; Carden et al., 2007; Oh et al., 2018). Pollution concerns associated with greywater disposal are mainly due to contaminants like pathogens, organics, inorganics, metals, personal care and pharmaceutical products consumption, surfactants, micropollutants and other emerging toxicants (Andersen et al., 2007; Mohamed et al., 2019).

For the sake of developing countries and the poor attitude towards greywater management, probably more focused attention on greywater is needed in global development agenda like the Sustainable Development Goals (SDGs). Greywater management although strongly implied in the Sustainable Development Goal 6 (SDG 6) (UN-Water, 2018), should be decoupled from the general wastewater Target 6.3 as it is overshadowed by blackwater.

Greywater management in Ghana is neither different nor better than most other developing countries. Management practices are abysmal partly due to poor sewerage coverage (less than 10% nationwide), and the few sewered and dominant non-sewered communities are all culprits of haphazard disposal of untreated and unsafe greywater into the environment (Dwumfour-Asare et al., 2017; Oteng-Peprah et al., 2018b). Currently, at most 5% of the populace appropriately handle greywater by disposal through sewers, septic tanks and soakaways (GSS, 2013b; DwumfourAsare et al., 2017). In addition to low sewerage coverage, poor greywater management is attributed to lack of priority for greywater management, wrong perceptions, weak enforcement of regulations, lack of appropriate and affordable local treatment technologies, and focused attention on onsite sanitation systems that cater for mainly blackwater etc (WHO, 2012; Hyde and Maradza, 2013; DwumfourAsare et al., 2017; Antwi-Agyei et al., 2019).

However, the need for permanent solutions to address poor greywater management in Ghana is imperative because of the potential risks from exposure to contaminants. These risks could be posed through pollution of water resources and the food chain, especially where vegetable farmers and household gardeners in Ghana heavily rely on informal water sources like untreated wastewater from direct sources and drains for irrigation (Keraita et al., 2003; Cronin et al., 2007; Hyde and Maradza, 2013; Azanu et al., 2016; Dwumfour-Asare et al., 2018). Already, there is evidence of groundwater pollution associated with waste disposals, including sanitation systems and wastewater (Lewis and Claasen, 2018; Lutterodt et al., 2018; Yu et al., 2018).

The prospects of greywater reuse in Ghana looks positive and rewarding in terms of resource recovery (water and biogas-energy), irrigation (water and nutrients) for biomass production, non-potable uses, cost reduction of probable sewerage network requirements, and minimising greywater carbon footprint (Hyde and Maradza, 2013; Mohammed et al., 2015; Dwumfour-Asare et al., 2018). Several greywater treatment technologies are available but mostly complex, expensive and unfit for low-income settings like Ghana (Hyde and Maradza, 2013; Mohamed et al., 2019). Constructed wetlands (CW) however, have become treatment options for greywater management including onsite and/or indoor applications (Paulo et al., 2007; Kasak et al., 2011; Ahmed and Arora, 2012; Arunbabu et al., 2015; Ramprasad and Philip, 2018). The technology can remove some major contaminants like electrical conductivity (EC), total dissolved solids (TDS), total suspended solids (TSS), biochemical oxygen demand (BOD), chemical oxygen demand (COD), anionic surfactants (AnS) etc with efficiencies as high as over 90% (Reyes-Contreras et al., 2012; Arden and Ma, 2018; Gupta and Nath, 2018; Pérez-López et al., 2018). Although the technology is highly recommended for developing countries, unfortunately it is uncommon (Erakhrumen, 2007; Gupta and Nath, 2018; Carrasco-Acosta et al., 2019).

Experts argue that constructed wetlands (CW) will be more appreciated and sustainable in developing countries if the technology were indigenized to maximize local benefits (Denny, 1997; Mullegger et al., 2014; Langergraber, 2015). This is where local knowledge and practices could play significant role in wastewater management, at least through exploratory approaches, and such efforts are strongly recommended by Ghana's National Environmental Sanitation Policy (GoG, 2010). Suggestive indigenous approaches (whether formal or not) for handling greywater exist and offer practical knowledge and experience for integration into scientifically proven technologies (Haberl, 1999; Kivaisi, 2001; Owusu-Mensah, 2016).

Application of constructed wetland in Ghana is rare and so far very limited studies can be identified with the technology. Two local studies available reported on treating heavy metal contaminated wastewater from a river (Anning et al., 2013), and the other looked at treatment of greywater from students' hall of residence (Niyonzima, 2007). The two studies demonstrated that CW is suitable for wastewater treatment in Ghana. The studies did not consider readily available lesser-known but potentially viable CW macrophytes like taro and sugarcane, and neither used laterite soil media. However, Anning et al (2013) admit that an important starting point for CW application at scale in Ghana could be identification and evaluation of potential indigenous resources especially plants. Thus, exploring indigenous practices like the use of vegetation in greywater disposal and their potential integration into CW for treating greywater in Ghana could be an asset.

Already, media and vegetation are part of the design factors like water depth, oxygen, climatic conditions (e.g. temperature), hydraulic conditions (flows, organic & surface loading, and retention time), surface area, baffle-partitions etc. that need attention (Reyes-Contreras et al., 2012; Morató et al., 2014; Papaevangelou et al., 2016; Ramprasad and Philip, 2016a). Roles of vegetation or macrophytes are diverse including uptake, phytovolatilization, antimicrobial, oxygenation in the rhizosphere, supporting biofilm growth and filtration etc. (Reyes-Contreras et al., 2012; Morató et al., 2014). Media as filter beds essentially support plants and microbes, as deposit sites for contaminants, and facilitate mechanical filtration and sedimentation (Morató et al., 2014). Hydraulic retention time (HRT) supports development and functionality of microbial biofilms (Reyes-Contreras et al., 2012; Morató et al., 2014). Also, baffle-partitions could create longer pathways to provided adequate contact for wastewater interactions with vegetation, media, microbes, and media-plants root matrix (Ramprasad and Philip, 2016a).

Adopting treatment technologies also requires credible data and information on wastewater characteristics (Kivaisi, 2001; Noutsopoulos et al., 2018; Mohamed et al., 2019). Unfortunately, such information in typical developing countries like Ghana especially for greywater is scanty if not elusive (Hyde and Maradza, 2013). This research study therefore aimed to generate data on greywater characteristics in Ghana, identify and document some indigenous greywater disposal practices especially the use of vegetation, and to assess greywater treatment performance of CW designed with local resources like vegetation and media. The study involved surveys of some selected communities, and experimentation of indigenized constructed wetlands at a residential neighbourhood within Kwame Nkrumah University of Science and Technology (KNUST) in Kumasi, Ghana.

1.2 Problem statement

Greywater treatment and management is abysmal in Ghana (Hyde and Maradza, 2013; Oteng-Peprah et al., 2018b). It remains the most deplorable component of wastewater management and rarely treated in the country. Unlike blackwater which receives some attention by the provision of onsite systems for containment and/or treatment (Amoatey and Bani, 2011), greywater is neglected even among the few sewered communities (GSS, 2013b; Awuah et al., 2014). To date, only 1-5% of generated greywater is safely handled using sewers, septic tanks, and soakaways nationwide (GSS, 2013b; Dwumfour-Asare et al., 2017). This is largely attributed to the huge sanitation infrastructure deficit in the country (Mensah and Antwi, 2013). Reasons include the following: 1) conventional wastewater technologies are considered expensive, complex and largely unsuitable and unsustainable for lowincome countries (Hyde and Maradza, 2013; Mohamed et al., 2019); 2) sanitation infrastructure gap is huge because of perceived prohibitive cost of investment without reference to the benefits associated with safe sanitation management (Agodzo et al., 2003; Dwumfour-Asare et al., 2017); 3) lack of priority for wastewater infrastructure (Amoatey and Bani, 2011); and 4) lack of readily adaptable treatment technology alternatives to safely handle wastewater including greywater.

Unfortunately, haphazard disposal of greywater in Ghana may be seriously exerting pollution risks on the environment especially water resources (Lutterodt et al., 2018; Yu et al., 2018), and contaminants could be fed into the food chain to challenge public health. Some of these contaminants may include pathogens, heavy metals, hormonal disrupters, micropollutants, and other emerging priority contaminants (De Gisi et al., 2016; Oteng-Peprah et al., 2018b; Zhao et al., 2018). For instance, in the urban environment where greywater with large volumes and complex characteristics are generated and discharged into river/streams and major drains, they become irrigation water sources for vegetable farming (Keraita et al., 2003; Cronin et al., 2007; Hyde and Maradza, 2013; Awuah et al., 2014; Azanu et al., 2016). Wastewater reuse is gaining grounds although formal discussions are ongoing at the policy formulation level (WHO, 2015).

Fortunately, greywater resource recovery potentials for water, nutrients, and biogasenergy exist but less explored in Ghana (Hyde and Maradza, 2013; Mohammed et al., 2015). One of the low-cost technologies with high potential for treating and recovering domestic greywater is constructed wetland. This phytoremediation strategy could be effective however, it is also less known and least explored in Ghana (Niyonzima, 2007; Anning et al., 2013). The two local experimental scale studies available as literature (Niyonzima, 2007; Anning et al., 2013) on the technology did not consider readily available lesser-known CW macrophytes, and neither used soil (laterite) media. Also, only one pilot scale system is installed at Tema city with barely no further details except a drawback of energy consumption due to energy-dependent units (Amoatey and Bani, 2011). Thus, the need to explore potentials of indigenous resources like plants for CW adoption as a basis for future scale-up (Anning et al., 2013) is still a valid knowledge gap requiring immediate attention. Knowledge of indigenous practices that rely on the use of vegetation in greywater disposal practices among local inhabitants will be an asset for CW adaptation studies. Such approach to

CW adaptation and/or indigenization is expected to maximize local benefits for sustainability, as widely recommended by experts (Denny, 1997; Mullegger et al., 2014; Langergraber, 2015). Currently, no available studies have identified and documented useful information from such indigenous practices, and also their integration into CW for greywater treatment in Ghana. Meanwhile, the National Environmental Sanitation Policy (NESP) of Ghana strongly recommends application of local knowledge and practices as opportunities for innovative wastewater management (GoG, 2010).

1.3 Research questions

Some research questions that have been generated as guide to the study included:

1. What is the nature (quantities and qualities) of domestic greywater in Ghana?
2. What indigenous plants are used in existing greywater disposal practices and the perceptions about plants' roles, functions and derived benefits?
3. Could some of the mostly used indigenous plants, and media (gravel and lateritic soil) be integrated into constructed wetland designs?
4. What are the performance efficiencies of CW microcosm models and do key design features influence effluent quality?
5. What effluent prediction models for key organic contaminants (BOD, COD, and anionic surfactants) could be fitted using multiple linear regression analysis?

1.4 Objectives of the study

The main research objective was to develop and assess the performance of an indigenized horizontal flow subsurface constructed wetland (iHFSCW) for onsite treatment of domestic greywater.

The specific objectives were to:

1. assess the characteristics of greywater from community drains, and other domestic and residential sources.
2. assess the use of indigenous vegetation in the disposal of greywater by inhabitants of selected peri-urban communities.
3. determine the performance of experimental scale constructed wetlands designed with two local vegetation (taro and sugarcane) and bed media

(gravel and laterite).

4. assess the influence of key design features (macrophytes, media, bafflepartitions, and HRT) on CW performance.
5. determine effluent prediction models for key organic contaminants BOD_5, COD, and anionic surfactants (AnS) using multiple linear regression.

1.5 Justification for the study

Useful information and data available on greywater characteristics in Ghana are limited and such knowledge gap needs attention in the quest to find appropriate technological solutions to poor greywater management. This study makes effort to generate some indicative data on Ghanaian greywater from both literature reviews and field studies. It also explores indigenous innovation found in local knowledge and practices associated with greywater disposal for integration and adaptation into constructed wetland designs. The study provides the basis for piloting an indigenized CW for greywater treatment in Ghana and future scale-up. The study also supports a core principle of Ghana's National Environmental Sanitation Policy (NESP) that local knowledge, practices and approaches must be explored for effective wastewater management. Therefore, this timely study does not only identify and document the worth of indigenous knowledge but also find appropriate means of application in the development of a low-cost and green technology like constructed wetlands, a rarely known domestic wastewater treatment option in Ghana. The study offers a novel localized alternative solution to the present haphazard disposal approaches to untreated and unsafe greywater in Ghana.

1.6 Scope and limitations of the study

The sampling for surveys were non-probabilistic, using snowballing technique. The sample population required referrals among minority (insiders/link-tracers) who shared and/or knew of others who had what the research sought. Again, the surveys could only cover few communities, which is comparatively nowhere near the entire country but the results could be indicative of the situation in Ghana especially the middle belt of transition and semi-deciduous ecological zones. The developed wetlands were tested on experimental scale and not as pilot systems. The good thing is that the tests were not in the lab but on the field to provide the setup with real world environmental conditions as much as possible. Meanwhile, the greywater used in the CW experimentation was generated from few available households in the residential neighbourhood largely determined by the topography and available resources. This is possible for experimental scale but not necessarily an approximation to full-scale CW systems. Also, there could have been slight differences in physical characteristics of installed wetland cells due to uncontrollable constructional factors on site although the necessary precautions were taken. The relatively short study period. It should be noted that deductions, conclusions and recommendations are made based on performance of the designed treatment systems operated under batch modes and not continuous-flow regime over the period of monitoring systems' performance.

1.7 Organization of the thesis report

This thesis report is organized into six main chapters according to the Monographbased thesis presentation format given by the KNUST School of Graduate Studies. Chapter one presents the general introduction to the study focusing on the background, problem statement, objectives, justification, and scope of the study. Chapter two is a review of literature relevant to the study. Chapter three presents the overview of the study sites, approaches and methodology. Chapter four presents the results and discussions covering all the five specific objectives. Chapter five is the general discussion section which synthesises all the findings from the sub-chapters of chapter four. Chapter six finally presents the main conclusions, relevant recommendations, and the necessary contributions to existing knowledge.

Chapter 2: Literature review

2.1 Introduction

Greywater (GW) which is simply the wastewater generated in the home from all sources except from the toilet requires attention for treatment and safe disposal because it is polluted. In most cases in the developing world, GW is not treated and disposal is haphazard. One of the wastewater treatment technology that could offer versatile treatment opportunity for greywater is constructed wetland, where configurations could include onsite/indoor options (Paulo et al., 2007; Kasak et al., 2011; Ahmed and Arora, 2012; Arunbabu et al., 2015; Ramprasad and Philip, 2018). Apart from that, GW has a lot of potential for resource recovery and it is receiving much attention (Gilboa and Friedler, 2008; Mohamed et al., 2019). This consideration has become necessary due to factors like high competing water demand, stressed water resources worsened by climate change impacts, inability to efficiently treat wastewater to avoid contamination etc (Hyde and Maradza, 2013; Qomariyah et al., 2018; Mohamed et al., 2019). Although little or no visible and aggressive effort is witnessed in most developing countries concerning productive greywater management, it is not out of place to join the advocacy for safe greywater management now. This call is effective through policy support for appropriate lowcost technologies, setting greywater treatment and reuse standards/ guidelines, promoting local research, and investment planning (Dwumfour-Asare et al., 2017).

2.2 Greywater – definitions, sources and characteristics

2.2.1 Greywater – definitions and sources

Several definitions and descriptions have been given about domestic greywater (GW) in literature. Some definitions could be very simplistic. For instance, “light wastewater”, “diluted wastewater” and “reclaimed water” (Mohamed et al., 2019). However, some representative definitions may include 1) “wastewater generated from household activities like laundry, showers, bathing, hand basins, dishwashers and kitchen sinks, except from sewer or latrine” (Morel and Diener, 2006; Rana et al., 2017); and 2) “water from the kitchen, bath and/or laundry which, generally, does not contain significant concentrations of excreta” (WHO, 2016). The two aforementioned definitions are self-explanatory and could be considered as more comprehensive without or with minimal ambiguity. Other terms also given to GW include “sullage”, and “grey wastewater” (Morel and Diener, 2006; Salifu, 2013; WHO, n.d-a). However, blackwater which primarily reflects faecal waste is also called sewage, and it could be distinguished by its composition which is mainly sullage/greywater and human excreta combined and normally produced from waterborne facilities (WHO, n.d-a).

A recent comprehensive review asserts that domestic wastewater itself could be classified into six main categories namely: brown water (wastewater with faeces), yellow water (urine), blackwater (containing both urine and faeces), greywater (containing mainly detergents), green water (contains food particles) and storm water (rainwater) (Mohamed et al., 2019). Basically, the classifications above also indicate the diversity of domestic wastewater sources. In most instances, definitions and classifications connote wastewater sources and types at the same time. Therefore, it has become almost a required practice that researchers and authors working on greywater identify the sources in all circumstances for clarity. For example, some writers consider kitchen wastewater as part of blackwater but others call such a source as “dark” greywater, and all other sources excluding toilet sources as “light” greywater (Rodríguez-Martínez et al., 2016). These concepts have led to other greywater classifications like high strength (i.e. with high pollutant loads) (Barışçı et al., 2016), or dark greywater, depending on the nutrient and chemical loads (Cook, 2016).

In some studies too, greywater categorisation is by qualitative (source) descriptions – e.g., using “dark greywater” for kitchen, laundry and dishwasher sources, and “light greywater” for bathroom, shower, bath and washbasin sources (Karabelnik et al., 2012; Barışçı et al., 2016; Cook, 2016). Meanwhile, blackwater which is wastewater from toilet or sewers, is generally distinguished from greywater by the high level of contaminants like organics, nutrients and infectious agents normally in excess of 90% in blackwater (Rodríguez-Martínez et al., 2016; Mohamed et al., 2019).

2.2.2 Characteristics of greywater

The characteristics of greywater from assessment of literature simplifies it to mean quantity and quality aspects. The quantity of greywater produced is the generation rate, while the various contaminants found in the wastewater constitute or define the quality. While some extensive data exists on greywater characterization in the developed world, very limited or almost insignificant information is available from developing countries like Ghana and such data is considered elusive (Hyde and Maradza, 2013). Because of this prevailing limitation, the review has been very limited.

2.2.3 Greywater generation rates

GW constitutes around 50 – 80% of the total domestic wastewater (RodríguezMartínez et al., 2016; Dwumfour-Asare et al., 2017). Indeed, GW is the largest fraction of domestic wastewater flows in every household setting (Hyde and Maradza, 2013), and in the absence of flush toilet systems, the fraction could hit as high as 90% (Oteng-Peprah et al., 2018a). The generation rate is normally expressed as volume per day and/or volume per capita or per household per day. The generation rates can vary from very low quantity of few litres to several hundreds of litres per person per day, all because of differences in geographical location, lifestyle, climatic conditions, water supply, culture and habits etc. (Oteng-Peprah et al., 2018a).

Table 2.1 shows some of the greywater generation rates found in literature. The records from Table 2.1 indicate that greywater generation in Ghana could be between 32 to 100 l/c/d and the minimum generation could be higher than the records given on the sub region of West African. The difference could be attributed to the factors already mentioned, but may be more linked to improved water supply in Ghana (87%) than the other Sub-Saharan Africa countries (64%) (WHO/UNICEF, 2014). For instance, in Ghana it is shown that water supply by house connection (piped on premises) could contribute to higher greywater generation rates than water source outside the house (Oteng-Peprah et al., 2018b). Again, urban settings especially with piped water are associated with more generation rates than peri-urban and likely rural areas, especially those experiencing water scarcity and receiving only basic form of water supply (Hyde and Maradza, 2013). Similarly, low greywater generation rates would be associated with water stressed Jordan and Yemen but not with water resourced counterpart developing countries like India, Nepal, and Vietnam. Developed countries with reliable water supply without water savings installations and fixtures could generate more greywater several times higher than developing countries.

Table 2.1 Some greywater generation rates in Ghana and international from literature

Study settings/site	Per capita (l/c/d)	References
Peri-urban, Ghana	32[a], 73[b]	(Oteng-Peprah et al., 2018b)
Urban Ghana	36 – 43	(Dwumfour-Asare et al., 2017)
	98 – 100	(Mohammed et al., 2015)
West Africa	18 – 25	(Hyde and Maradza, 2013)
Senegal	60	(Ghaitidak and Yadav, 2013)
South Africa	80	(Ghaitidak and Yadav, 2013)
International	14 – 59 (Jordan), 72 (Nepal), 80-110 (Vietnam)	(Al-Mughalles et al., 2012; Oteng-Peprah et al., 2018b)
	77 – 79 (India); 35 (Yemen)	(Ghaitidak and Yadav, 2013)
	200 (USA), 35–274 (Europe)[c], 72–225 (Asia)	(Al-Mughalles et al., 2012; Boyjoo et al., 2013)
	83 – 98 (Greece); 49 – 108 (Germany)	(Noutsopoulos et al., 2018; Sievers and Londong, 2018)
	117 (Australia)	(Ghaitidak and Yadav, 2013)

Note: a = water source is outside the house; b= water source is house connection; c = low European generation because of water savings mentality & installations

2.2.4 Greywater quality – contaminants and specific pollutant loads

The general quality of greywater could be endless list of water quality parameters ranging from diverse species of physical, through chemical, to biological contaminants. Countless studies worldwide have found qualities such as: turbidity, pH, EC, DO, TSS, TDS, BOD, COD, nutrients (N and P), anionic surfactants, minerals and metal species (Na, Ca, K, Fe, Pb, Hg, As, Cd etc), micropollutants or xenobiotic compounds (as physicochemical contaminants); and biological and pathogens – *E. coli*, *Salmonella spp*., faecal coliforms, total coliforms etc. (De Gisi et al., 2016; Dwumfour-Asare et al., 2017; Shi et al., 2018; Nivala et al., 2019).

The levels of contaminants in any greywater are largely depended on waste generators' demographic characteristics, living standard, lifestyles and behaviour, cultural practices, various water use options, household chemicals usage, quality of water supply, fixtures and fittings for greywater, climate, duration of containment etc.(Morel and Diener, 2006; Nghiem et al., 2006; Ghaitidak and Yadav, 2013; Oteng-Peprah et al., 2018a). The implication is that greywater may significantly differ from one place to the other, from country to country, town to town, neighbourhood to neighbourhood, and even from one household to the other. It is therefore not surprising to find developed and developing countries with contrasting greywater characteristics – both in quality and quantity, especially differences in magnitude.

Greywater qualities from international studies in ten countries are given in Table 2.2. The reports show high variability in greywater quality across the different socioeconomic and geographical settings. pH range is from slightly acidic (pH 6.35) to alkaline (pH 10). The data speaks out clearly that contaminants levels in greywater could be very high depending on the greywater sources. Certain parameters could be comparatively high as blackwater, for instance, turbidity (444 NTU), EC (3000 μS/cm), nitrate (258 mg-N/L), NH_3-N (75 mg/L), TP (19.5 mg-P/L), BOD_5 (1,056 mg-O_2/L), COD (2,568 mg-O_2/L), surfactants (118.3 mg/L) and microbial loads as much as close to 6 and 9 log CFU/100ml for faecal and total coliforms respectively. A key message that emerges from the international review is that the three Middle East countries – Israel, Jordon and Oman, are topping in terms of organic matter and nutrients (N & P) loads in greywater (Table 2.2). Such observations could be associated with household products consumption and lifestyle, but chiefly due to water scarce condition influencing less water usage to cause high contaminant loads (no dilution effects) (Boyjoo et al., 2013).

Table 2.3 is a presentation of greywater characteristics from the few available studies in Ghana. It is shown that the parameters pH, EC, TSS, turbidity, BOD_5, and COD are commonly reported. The least reported ones are TDS, TKN and phosphorus, followed by trace elements and heavy metals (except lead), and then microbiological factors – *E. coli* and total coliforms. The studies' scope included communities, schools (residential halls/hostels), and hotels, and samples taken from households (with categories composite/mixed, laundry, kitchen, and bathroom), drains, a lagoon, and a salon. The differences in scope or settings could explain the cause of high variability of the Ghanaian greywater quality presented. The studies show that greywater in Ghana could be highly polluted looking at incidence of high levels of quality characteristics like microbial loads (3 – 7 log CFU/100ml), nutrients (TP: 1 – 26 mg/L, nitrogenous species: 8 – 57 mg/L), organic contaminants (BOD_5: 540 mg/L & COD: 880 mg/L), and heavy metals (Hg: 0.4 mg/L & Pb: 0.3 mg/L).

Ghanaian greywater could be considered to be loaded with major contaminants comparable to other countries (Table 2.2 and Table 2.3). Some emerging contaminants like organic micropollutants notably benzalkonium chloride, parabens (methyl and propyl forms), sodium benzoate and hypochlorite have been identified in a study in recent times at higher levels (0 – 8 mg/L) than expected although not extensively detected in all greywater sources (Dwumfour-Asare et al., 2017). The levels of these major contaminants raise serious environmental and public health concerns about greywater in Ghana, especially since this stream of wastewater is mostly untreated and haphazardly disposed of into the environment (GSS, 2013b; Dwumfour-Asare et al., 2017).

Table 2.2 Greywater quality from international studies in ten countries

Parameters	Countries									
	Australia	**India**	**Brazil**	**Holland**	**Slovenia**	**Italy**	**Germany**	**Turkey**	**Israel**	**Jordan**	**Oman**
pH	9.3 - 10	7.3 - 8.1	-	-	9.6	7 - 9	6.9 - 8.1	7.1 - 7.2	6.7	6.35	8.3
EC (µS/cm)	190 - 1,400	489 - 550	-	-	-	1,300 - 3000	-	401 - 495	-	1,830	-
Turbidity (NTU)	50 - 210	20.6 - 38.7	254	-	-	40 - 150	-	-	-	-	444
TSS (mg/L)	88 - 250	12 - 17.6	120	-	35	90 - 200	-	48 - 54	138	168	315
NO_3-N (mg-N/L)	0.1 - 0.31	0.5 - 0.63	0.05	0.12 - 0.77	-	-	-	0.13 - 1.3	-	-	258
NH_3-N (mg-N/L)	<0.1 - 1.9	-	2.4	0.8 - 11.8	2.45	-	-	1.2 - 1.3	-	75	-
TKN (mg/L)	1 - 40	-	-	-	-	-	27.2	7.6 - 9	-	128	-
Total N (mg/L)	-	42.8 - 57.7	8.8	26.3 - 35.2	2.75	-	9.7 - 16.6	-	14	-	-
PO_4^{3-} (mg/L)	-	1.52 - 3.36	5.6	2.3 - 2.36	-	-	9.8	-	-	-	-
Total P (mg-P/L)	0.062 - 42	-	-	6.2 - 7.8	9.9	-	5.2 - 9.6	7.2 - 7.3	17.7	19.5	-
BOD_5 (mg- O_2/L)	48 - 290	56 - 100	435	-	195	-	-	90 - 116	207	1,056	179.7
COD (mg-O_2/L)	-	244 - 284	646	210 - 376	280	400 - 1000	125 - 354	177 - 277	686	2,568	2,313
Surfactants (mg/L)	-	-	-	43.5 - 54	10.1	0.01 - 25	-	-	40	-	118.3
T coliforms (Log CFU/100)	3.36 - 5.52	4.6	8.73	-	-	-	-	4.13	-	7.00	>2.30
F coliforms (Log CFU/100)	2.04 - 3.04	4.6	-	-	-	-	-	3.55 - 4.04	6	5.48	-

Excerpt from (Boyjoo et al., 2013)

Table 2.3 The quality of Ghanaian greywater from available studies

Water quality parameters	*KNUST, University campus	Kpeshie, Accra	Eastern region**
pH	7.5±0.2[a,b] , 6.83[c], 7.74[d]	7.84±0.09	7.84±0.09
EC (uS/cm)	656.42[c], 965.2[d]	17,100±~3400	628.43±57.46
Temperature (^{o}C)	29.2±0.7[a,b], 29.6[d]	29.11±0.34	29.11±0.34
DO (mg/L)	2.7±0.9[a,b]	-	1.6±0.30
TDS (mg/L)	-	-	488.85±23.01
TSS (mg/L)	212±20.8[a,b], 222.83[c], 347[d]	92.39±26.48	92.39±26.49
Turbidity (NTU)	279.89[c]	72.14±20.47	90.14±3.47
BOD_5 (mg/L)	198.3±33.3[a,b], 420.22[c], 538.5[d]	63.79±26.49	89.79±26.49
COD (mg/L)	399±108.4[a,b], 707.28[c], 874[d]	236.99±66.35	612.99±66.35
Total Coliforms Log CFU/100ml	6.4±5.8[a,b], 5.7[c]	-	-
Faecal Coliform Log CFU/100ml	4.93[d]	5.21	2.95
E. Coli Log CFU/100ml	6.2±5.3[a]	-	-
Nitrate (mg-N/L)	0.7±0.06[b], 12.91[c], 15.25[d]	2.04±0.49	-
Nitrite (mg-N/L)	0.0[b], 0.19[c], 57.5[d]	0.1±0.03	-
Ammonia (mg-N/L)	8.4±1.8[b]	-	2.88±0.48
Total Phosphate - PO_4^{3-}(mg/L)	12.43[c], 26.18[d]	1.24±0.26	-
Phosphorus (mg-P/L)	11.8±4.0[a,b]	-	-
Calcium (mg-Ca/L)	2.81±0.01[b]	-	-
Magnesium (mg-Mg/L)	6.1±0.4[b]	-	-
Cadmium (mg-Cd/L)	0.01±0.001[b], 0.015[c]	0.003±0.002	-
Copper (mg-Cu/L)	<0.01[b], 0.135[c]	-	0.001±0.0
Iron (mg-Fe/L)	0.37±0.08[b]	-	-
Lead (mg-Pb/L)	<0.01[b], 0.316[c]	0.01±0.0005	0.003±0.0
Manganese (mg-Mn/L)	0.04±0.01[b], 0.098[c]	0.61±0.13	-
Mercury (mg-Hg/L)	0.4±0.08[b]	-	-
Zinc (mg-Zn/L)	0.03±0.001[b], 0.151[c]	-	-
References Note: ± standard deviation *Kwame Nkrumah University of Science & Technology	[a](Monney et al., 2013) [b](Awuah et al., 2014) [c] (Muzola, 2007) [d](Niyonzima, 2007)	(Ansah et al., 2011)	(Anim et al., 2014) ** study with hotels and hostels in Koforidua

Apart from reporting greywater quality characteristics in the usual mg/L, it can also be defined in specific pollutant loads like g/d or g/c/d. Also, organic contaminants (mainly – BOD and COD) can be measured as organic loads expressed in terms of population equivalent (PE). PE of organic load is the amount of oxygen-demanding substances whose oxygen consumption during biodegradation equals the average oxygen demand of the wastewater produced by one person (UN, 1997; OECD, 2007). The standard measure of $1PE_{BOD}$ equals 54 g-BOD_5/d which is equivalent to 0.18 m^3/d (OECD, 2007; Henze and Comeau, 2008).

Meanwhile, some specific pollutant loads in greywater available in literature are presented in Table 2.4. The review shows that in terms of specific contaminant loads, Ghanaian greywater is neither too low nor high but somehow comparable to the international figures although the Ghanaian data could be only indicative because of the limited source of data.

Table 2.4 Some specific greywater pollutant loads found in literature

Specific pollutants	Ghana[a]	International (high- and lowincome countries)
	(g/c/d)	(g/c/d)
TSS	17 – 20	10 – 30[a]
BOD_5	8 – 15	20 – 50[a]; 28[b]
COD	24 – 48	18 – 46[a]; 49.3[b]
Ammonia	-	1 (TN)[b]
Ammonium	1 – 15	0.5[a]
Total Phosphorus - PO_4-P	0.1 – 0.2	0.2 – 6[a]; 0.38[b]

Note: g/d (gram/day); g/h/d (gram/household/day); g/c/d (gram/capita/day); TN (total nitrogen); [a] (Oteng-Peprah et al., 2018b); [b] (Sievers and Londong, 2018)

2.3 Greywater reuse and applications

There are several reuse options with greywater (both raw and treated) as documented in literature. Some of these reuse applications include the following listed below from literature (WHO, 2006b,a; Godfrey et al., 2009; Matos et al., 2012):

- Irrigation and/or agriculture application,
- Landscaping and lawn irrigation, and - Car washing and toilet flushing.

However, there are potential hazards and/or risks directly or indirectly associated with greywater reuse like all other wastewater uses and applications (Roesner et al., 2006). This applies to both treated and untreated greywater because the risks depend on the types and levels of contaminants in the water. The World Health Organization (WHO) for instance has developed guidelines on reuse of wastewater (excreta and greywater) because of the prevailing risks and hazards (WHO, 2006b,a). The prescriptions of these barriers especially for agriculture applications are to ensure protective health and safety. Figure 2.1 below shows hazard barriers for wastewater applications according to WHO.

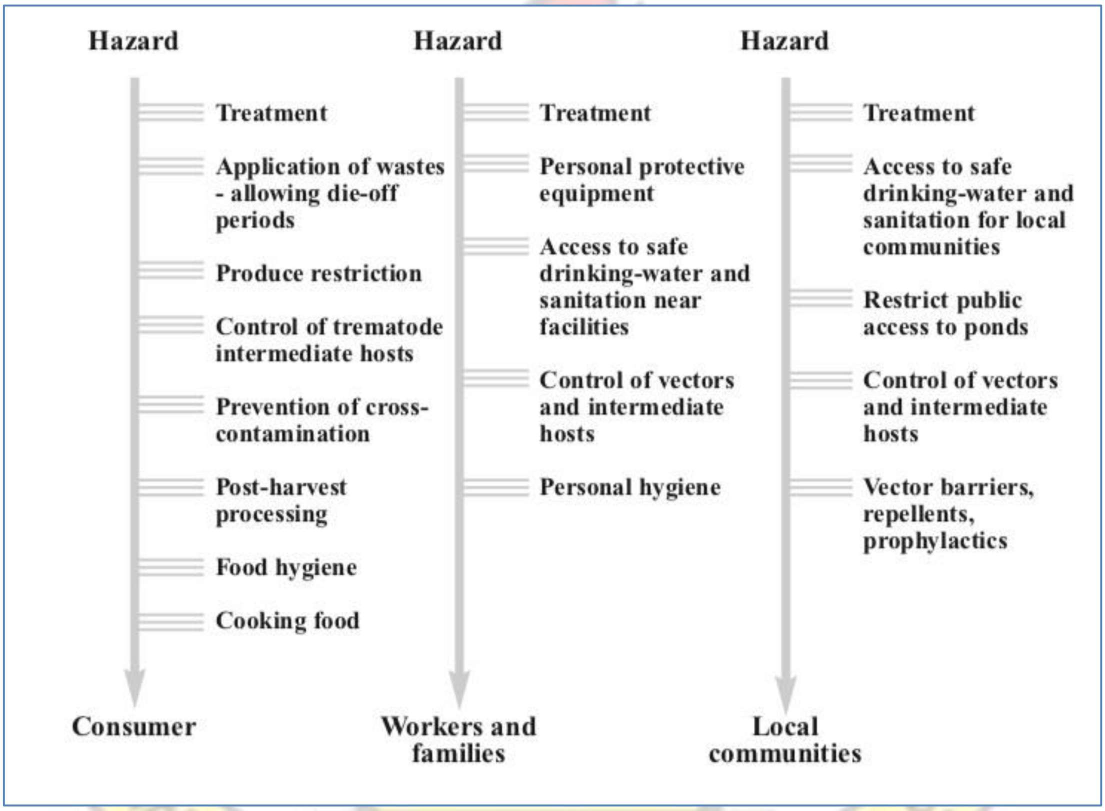


Figure 2.1 Hazard barriers for wastewater reuse
Source: (WHO, 2006a)

2.4 Wastewater discharge and reuse standards in Ghana

Ghana has wastewater quality guidelines for discharge into water bodies or courses. The guideline limits are proposed by Ghana's Environmental Protection Agency (EPA-Gh) and these standards are yet to be ratified into full regulation. EPA-Gh is the main government agency that regulates environmental protection and safety including all discharges into the environment. The document was prepared primarily to guide discharges for industrial installations and facilities (EPA-Gh, n.d.). Therefore, no explicit mention or reference is made to domestic wastewater including greywater, however greywater is implied because it is part of general wastewater. The discharge limits enshrined in the guideline are readily extensible to domestic greywater discharges because the limits could define nuisance and/or public health hazards associated with greywater. The main government agency for regulating domestic wastewater discharges is the local government authorities through byelaws and building regulations. These authorities (Metropolitan, Municipal and District Assemblies - MMDAs) are concerned with domestic wastewater that are potential sources of nuisance and/or public health hazards, and their discharges into public drains (gutters) are forbidden (Fosu, 1996; Ghana Local Government, 1998).

Table 2.5 shows available regulatory discharge and reuse guideline limits for some wastewater quality parameters. In addition to EPA-Gh discharge limits in Table 2.5, there are also non-restrictive reuse (irrigation) guideline limits for some wastewater quality parameters defined by Food and Agriculture Organization (FAO) and World Health Organization (WHO). The combined guidelines could conservatively guide any discharge and/or potential reuse of greywater sources in Ghana. The implication is that any greywater source that fails to meet the guidelines in Table 2.5 should neither be permitted for discharge into the environment nor could be used for irrigation, but receive some level of treatment before. The FAO/WHO reuse limits when available are mostly more conservative than the EPA-Gh discharge limits. While the EPA discharge limits tend to protect the environment especially water bodies, the FAO/WHO reuse limits primarily seek to protect the irrigated vegetation and soil media.

Table 2.5 Some wastewater discharge and reuse guideline limits

Parameters	**Guidelines**	
	EPA-Gh[a]	**WHO/FAO[b]**
pH	6-9	6.5 – 8.5
EC (μS/cm)	750	<700
Dissolved oxygen (mg-O_2/L)	1	-
Turbidity (NTU)	75	-
TDS (mg/L)	1000	<450
TSS (mg/L)	50	<50
Total Chlorine (mg-Cl_2/L)	250	<1

BOD_5 (mg- O_2/L)	50	-
COD (mg- O_2/L)	250	-
Total Phosphorus - PO_4-P (mg/L)	2	-
Nitrate (mg-N/L)	11.5	<5
Ammonia (mg-N/L)	1	-
Ammonium (mg-N/L)	1.5	-
Sulphate - SO_4^{2-} (mg/L)	250-300	-
Sodium (mg-Na/L)	-	<69
Cadmium (mg-Cd/L)	<0.1	0.01
Total Iron (mg-Fe/L)	-	0.01
Lead (mg-Pb/L)	0.1	5.0
Mercury (mg-Hg/L)	0.005	5.0
Zinc (mg-Zn/L)	5.0	2.0
SAR $(meq/l)^{½}$	-	0 – 3

Note: LAS = Linear Alkylbenzene sulfonate; SAR = sodium absorption ratio; a = Ghana EPA discharge limits (Owusu-Ansah et al., 2015; Oteng-Peprah et al., 2018b; EPA-Gh, n.d.); b = World Health Organization/Food and Agriculture Organization's non-restrictive wastewater reuse limits (FAO, 1992; WHO, 2006a); na = not available

2.5 Wastewater management in Ghana

Wastewater management in Ghana is part of the general sanitation, and coverage is poor because to date, only 14% of the populace is considered served with improved (and/or at least basic) service (UNICEF/WHO, 2012; Monney et al., 2015; WHO and UNICEF, 2017). There is very limited use of sewerage systems nationwide, with less than 5% coverage (Amoah et al., 2007; Agyei et al., 2011; GSS, 2013b; AppiahEffah et al., 2014). The bulk of sanitation infrastructure is therefore onsite systems for managing mainly excreta (faecal waste) (Amoatey and Bani, 2011). Such onsite technologies include flush toilets (water closet) (14%), pit latrines (19%), KVIP/VIP (12%), public toilets (WC, KVIP, pit latrine etc) (36%) and others (GSS, 2014b). Wastewater management emphasis is on excreta and the majority of excreta is collected in septic tanks and dry latrines (Murray and Drechsel, 2011).

Sewerage systems barely exist in the whole of Ghana (inclusive cities) since coverage is less than 5% (Murray and Drechsel, 2011; Dwumfour-Asare et al., 2017). From comprehensive inventory and review study of sewerage and wastewater treatment systems in Ghana, 71 facilities were identified among which 9 were operational, 21 were operating with at least a component failing, 35 were nonfunctional, and 6 had unknown operational status (Murray and Drechsel, 2011). From the same study, most sewer systems were small-scale except those of Accra and Tema; and even if all systems in Ghana were operational, only few people will still be served.

The lack of adequate wastewater management infrastructure especially sewerage systems is attributed to perceived prohibitive costs of investment without reference to the benefits derived from improved sanitation management (Agodzo et al., 2003; Dwumfour-Asare et al., 2017). All developers including private individuals (landlords/landladies or house owners) are offered the option to adopt onsite systems preferably soakaways for greywater management and well-informed systems are prescribed by local authorities' (MMDAs) bye-laws and the national building regulations (Fosu, 1996; Ghana Local Government, 1998). The authorities recognize greywater as potential sources of nuisance and/or public health hazards. However, compliance and enforcement have been weak although building codes and bye-laws have been in existence since 1948 (WHO, 2012; Antwi-Agyei et al., 2019).

Greywater management is worst probably because greywater is usually perceived as "not readily offensive but tolerable" in the environment compared to black-water (faecal matter), deemed unmistakably offensive and therefore must be attended to.. Evidence indicates that domestic greywater disposal in Ghana is haphazard, with little or no treatment (GSS, 2013b). Appropriate greywater disposal practices in Ghana are low with only 5% coverage, mainly through the use of the very few sewerage systems, septic tanks, and soakaways (Dwumfour-Asare et al., 2017). The very low to non-existence sewerage coverage together with lack of compliance with laws and weak enforcement of regulations that prescribe onsite alternatives could probably be a major contribution to why people without sewer connections dispose greywater into stormwater or public drains (Murray and Drechsel, 2011; DwumfourAsare et al., 2017). Meanwhile, it asserted that greywater in public drains threaten environmental and public health safety in Ghana (Gretsch et al., 2016).

2.6 National Environmental Sanitation Policy – Ghana

A brief review of Ghana's National Environmental Sanitation Policy (NESP) 2010 (revised) affirms the existing beliefs and understanding about greywater management in Ghana. The assessment of the policy document shows that wastewater management is all about blackwater or excreta or septage containment, transport/conveyance or haulage, treatment and disposal (GoG, 2010). Indeed, the following key summaries are deduced from the NESP:

1. Greywater as a term is not identified or found in the policy document except the use of sullage which could imply the same meaning. Also, there are prescriptions for communities' and households' responsibilities to "provide sullage conveyance drains (gutters) and soakage pits". Such responsibilities are to be ensured by the local authorities through the use of appropriate byelaws.
2. There appears to be contradictory understanding in the disposal of stormwater and greywater wastewaters in the NESP and some regulations like byelaws and building regulations. Greywater according to the national building regulations and some MMDAs' bye-laws must be discharged into soakage pits and not into public drains to avoid causing potential nuisance and public health hazards (Fosu, 1996; Ghana Local Government, 1998). Meanwhile, the NESP suggest that sullage could be discharge into drains. But the regulations are superior to the policy and their provisions must be upheld.
3. Liquid wastes are considered synonymous to industrial effluents in its use in the document, and this could be misleading. Meanwhile, the Ghana Statistical Services from the 2010 Census report is positioned to define greywater, i.e. all domestic wastewater excluding faecal or excreta related wastewaters as liquid waste (GSS, 2013b).
4. The section on the usage of stormwater drainage is silent on discharging sullage or greywater into them but explicitly forbids the disposal of faecal and solid wastes into drains.
5. On treatment and disposal systems, considerations are clear for excreta or faecal waste or septage management but not greywater/sullage.
6. An alternative approach, which appears unacceptable by all standards from environmental pollution perspective is the offer of marine disposal of sewage after primary treatment. This is because it is well established that, primary treatment ordinarily does not satisfy any high effluent standards (FAO, 1992; Vítěz et al., 2012).
7. No elaborate consideration for treatment for sullage or greywater except conveyance and disposal without treatment. This is probably because of the general poor approach to greywater management in Ghana.

8. No new and/or emerging sanitation technologies are identified for any exploitation. However, it is good that the NESP recognizes the need for committed research and development in finding appropriate technologies and technical approaches.
9. Moreover, the policy adopts the principle of recognizing indigenous knowledge and practices. This provision is great but the policy at the same time failed to connect this principle to the research and development theme.
10. The use of Information, Education and Communication strategy for promoting waste reduction, reuse, recycling and recovery is welcoming. Although, no explicit mention is made of which waste streams but it can be assumed that all kinds of waste including solid waste, blackwater, and greywater are implied. However, since greywater reuse, recycling and recovery may not necessarily involve expensive technologies compared to others like excreta, more emphasis could have been given to that component.

2.7 Sustainable Development Goals and wastewater management

The Sustainable Development Goals (SDGs) are 17 universal, integrated and transformational global set goals with 169 targets and 232 monitoring indicators achievable by the year 2030 (ICSU and ISSC, 2015; Hák et al., 2016). This is the current and main global development direction called Agenda 2030 following the post-2015 Millennium Development Goals (MDGs) adopted by all member states of the United Nations in September 2015 (Omisore, 2018; pS-Eau, 2018). The SDGs are significant improvement over the MDGs because previous lessons from MDGs are incorporated in addition to the recognition of "key systemic barriers to sustainable development such as inequality, unsustainable consumption patterns, weak institutional capacity, and environmental degradation" (ICSU and ISSC, 2015; Ait-Kadi, 2016; pS-Eau, 2018).

The umbrella goal for wastewater (of all types) is the "Water Goal", SDG 6 formulated as "Ensure availability and sustainable management of water and sanitation for all" (ICSU and ISSC, 2015; Ait-Kadi, 2016). The SDG 6 is directly or indirectly linked to the others and there are strong links with majority of them (ICSU and ISSC, 2015; Mugagga and Nabaasa, 2016).

The targets under goal 6 that specifically address wastewater issues are (Ait-Kadi, 2016; UN-Water, 2018):

- Target 6.3: “By 2030, improve water quality by reducing pollution, eliminating dumping and minimizing release of hazardous chemicals and materials, halving the proportion of population with untreated wastewater sources, and at least doubling recycling and safe reuse globally.”
- Target 6.6a: “By 2030, expand international co-operation and capacitybuilding support to developing countries in water and sanitation-related activities and programmes, including water harvesting, desalination, water efficiency, wastewater treatment, recycling and reuse technologies.”

The sanitation aspect of the SDG 6 is on the dimension of service-based approach, which simply means that “service improvements are no longer merely installing a toilet or latrine, but embodiment of the entire sanitation chain including toilet user interface, wastewater collection, transport and discharge, treatment, disposal and/or potential reuse.” (pS-Eau, 2018). Apart from the toilet user interface, the rest of the sanitation chain constitute wastewater management. This strictly implies that without effective wastewater management, the SDG 6 is not achievable and the consequence on other SDGs would also be inevitable.

Already, the SDG 6 is considered as overambitious because more efforts and resources are required from political governments within a short time of 15 years to achieve the needed costly interventions (Hutton, 2016; Mara, 2016). It is on record that 1.5 billion sewerage users have no wastewater treatment, and also 80% of wastewater generated worldwide is untreated (Mara, 2016). However, the SDG 6 is a positive aspiration and all professionals must strive to contribute real improvements to better the lives of millions even if targets were not fully achieved by the set time, and after all it may not be a big deal to realize full achievements by 2050 (Mara, 2016). While there appears to be some pessimism about meeting the targets of SDG 6, the optimist calls for focused attention to be able to make some significant achievements if not 100% success.

The situation of wastewater management in the developing world appears somehow complex with high uncertainties, and probably calls for attention to a more disaggregated SDG target(s) for effective management of the two main wastewater streams, greywater and blackwater. For instance, in Ghana there has been some good attention on the management of blackwater or faecal matter component of domestic wastewater (Amoatey and Bani, 2011), but this has happened at the detriment of greywater over the years (Dwumfour-Asare et al., 2017). In the developed world, grey- and black- water streams are well managed, and receive almost equal attention through the use of sewerage systems (Karabelnik et al., 2012). This is not observed in developing countries, an example is Ghana.

In Ghana, onsite technologies are mostly provided for blackwater, although improved sanitation coverage is yet low (14%) (UNICEF/WHO, 2012; Monney et al., 2015). Little and/or almost nothing is done for greywater because only 5% improved disposal practices exist with the populace from 2010 census data (GSS, 2013b; Dwumfour-Asare et al., 2017). Stormwater drains are mostly used for disposing of greywater and this predisposes residents and the larger environment to potential risks including pathogens and chemical contaminants (Gretsch et al., 2016; Dwumfour-Asare et al., 2017). This situation in the developing world probably calls for focused global attention on greywater management. The attention may be attained by disaggregating the target 6.3 of SDG 6 to reflect the two main wastewater streams – blackwater (excreta or faecal waste) and greywater. This also means that, indicators for tracking, monitoring and evaluating progress of achieving targets must have explicit coverage for both safely managed blackwater and greywater. Without refocusing some urgent attention on greywater management, all efforts from the developing world will continue to be biased in favour of blackwater in the quest to meet SDG 6 and targets 6.3 and 6.6a without recourse to safeguarding the environment against imminent threat from haphazard disposal of untreated greywater. The current study is a contribution towards efforts in meeting SDGs on wastewater management.

2.8 Available greywater treatment technologies

Several greywater treatment technologies are currently available. These technologies include activated sludge systems, trickling filters, waste stabilization ponds, rotating biological contactors, filtration and ultrafiltration membrane bioreactors, constructed wetlands, upflow anaerobic sludge blankets, ultraviolet disinfection, tower gardens, slanted soil systems, ion exchange resin processes, phytoremediation, phycoremediation etc (Fenner and Komvuschara, 2005; Boyjoo et al., 2013; OtengPeprah et al., 2018a; Mohamed et al., 2019). Some are simple physical filtration systems such as membranes and sand filters and others are highly automated and energy-intensive systems including biological, chemical and physical treatment mechanisms (Arden and Ma, 2018).

However, most of these systems are complex, expensive and unfit for low-income settings (Hyde and Maradza, 2013; Mohamed et al., 2019). Suitable treatment systems for developing economies should be sustainable: by being simple and effective; demanding little and/or no skilled labour; requiring little and/or no energydependent systems; and with low cost for investment, operations and maintenance (Dallas et al., 2004; Arunbabu et al., 2015). One of the most recommended technologies for greywater management is constructed wetland and it is also receiving attention for onsite and/or indoor applications (Paulo et al., 2007; Kasak et al., 2011; Ahmed and Arora, 2012; Arunbabu et al., 2015; Ramprasad and Philip, 2018).

Membrane bioreactors and/or sequencing batch reactors produce high quality treated effluent but they are associated with high cost and relatively low public acceptance among others that limit their application in developing countries (Masi et al., 2010).

2.9 Constructed wetlands – a phytoremediation technology

Constructed wetlands (CWs) are natural treatment systems based on the principle of exploiting "natural materials (e.g. gravel, sand, soils, plants, associated microbial assemblage) and naturally occurring processes including physical, chemical and biological forms, for treatment purposes" (Vymazal, 2007; Stefanakis, 2016).

Basically constructed wetlands employ phytoremediation strategy in removing various contaminants (Worku et al., 2018b). In a classical definition, phytoremediation is "a technology that is based on the combined action of plants and their associated microbial communities to degrade, remove, transform, or immobilize toxic compounds located in soils, sediments, polluted groundwater and wastewater in treatment wetlands" (Truu et al., 2015). Phytoremediation, also called phytotechnology (Conesa et al., 2012) in itself is not new in the world because it existed before civilization but it was first documented about 300 years ago (Sharma and Pandey, 2014). Meanwhile, dated back in 1952 and 1974, the technology received significant attention and improvement for full scale operations including the development of horizontal flow CW in Germany by the 1960s and 1970s (Vymazal, 2009; Mthembu et al., 2013).

It is therefore understandable why constructed wetlands and phytoremediation are sometimes used interchangeably in most literature, because the concepts and principles of removing contaminants are based on phytoremediation – using plantfacilitated processes including support for microbial activities. However, constructed wetland may be considered as using bioremediation strategy when the emphasis is on both the utilization of plants and other biological entities like microbes and enzymes (Bijalwan and Bijalwan, 2016). Thus, bioremediation is a natural remediation process involving only biological entities like plants and microbes (Bijalwan and Bijalwan, 2016; Worku et al., 2018a). CW technology basically operates on 'green liver' theory, which states that "plants have many of the same metabolic enzymes as mammals, and that the entire plant has the potential to detoxify contaminants the same way as a mammalian liver with its associated enzymes". The activities of the plants synchronize with bacteria called endophytes which colonize plant vascular tissues to augment biodegradation processes (Kulakow and Pidlisnyuk, 2007).

2.9.1 Types and options for constructed wetland technology

The types of constructed wetlands are fundamentally grouped according to the water flow regime: free water surface flow (FWS) and subsurface flow (SSF), and the type of macrophytic growth (Vymazal, 2007; Stefanakis et al., 2014). However, there is classification by the nature of macrophytes – free floating plants, floating leaved plants, emergent plants and submerged plants (Vymazal, 2007). A combination of different types of constructed wetlands gives a class called hybrid constructed wetland systems (Hoffmann and Platzer, 2010; Mthembu et al., 2013). The hybrid types are also called multi-designed wetland systems (Mthembu et al., 2013). An additional type called floating treatment wetland is also mentioned in literature (Zhang et al., 2014).

From the available literature (Vymazal, 2007; Hoffmann and Platzer, 2010; Zhang et al., 2014; Ilyas and Masih, 2017), brief descriptions and explanations of the fundamental working principles of major classes of constructed wetlands are summarised in the next four subsections.

2.9.1.1 Free water surface flow (FWS) wetlands

The brief description of the FWS (Figure 2.2) is that:

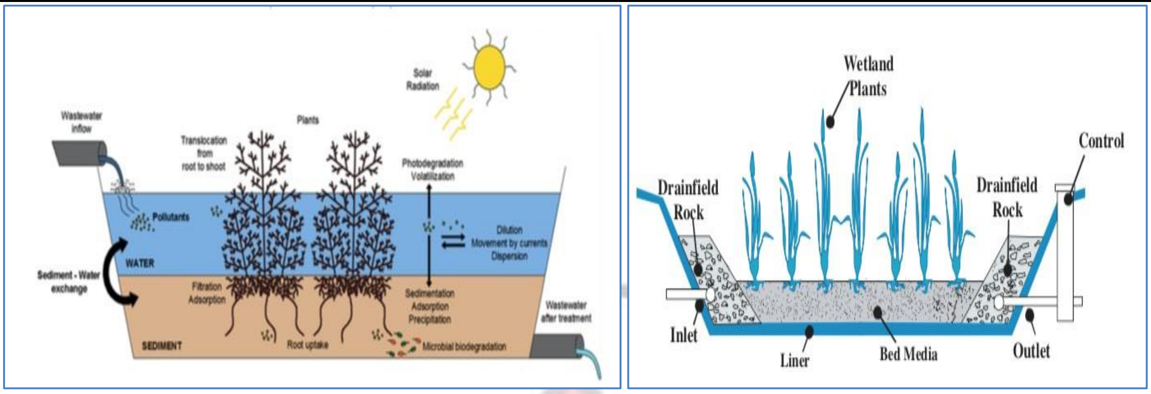
1. They are shallow basins with water on the surface.
2. The treatment processes occur through complex interactions between vegetation and associated biofilms in the water phase.
3. They behave like natural marshes with a broad spectrum of biological characteristics capable of removing wastewater contaminants.
4. The near-surface layer of water is aerobic and the deeper waters and the substrate are usually anaerobic.
5. FWS systems typically have water depths less than 0.4 m.

2.9.1.2 Subsurface flow (SSF) wetlands

The brief description of the SSF (Figure 2.2) is that:

1. Designs are either horizontal (HSSF) or vertical subsurface flow (VSSF) through a permeable medium (e.g. sand, gravel, crushed rock etc)
2. The most common forms are horizontal SSF configurations.
3. In HSSF configurations, wastewater flows horizontally through the media and comes into contact with a network of aerobic, anoxic, and anaerobic zones in the subsurface
4. The aerobic zones are found around plant roots and rhizomes matrix in the media by oxygen diffusion or exchanges.
5. For VSSF systems, the feed is by the whole surface area via a distribution system and passes through the media vertically down.
6. Typically, bed depth for SSF wetlands is less than 0.6 m.

7. Various studies point that VSSF systems are more efficient than HSSF especially for contaminants like NH_3-N, and NO_3-N.



(a) (b)

Figure 2.2 Free surface and subsurface constructed wetlands
General illustration: A) free water surface flow (FWS), and B) subsurface water flow – horizontal (HSSF). Sources: [a](Gorito et al., 2017) [b](Tanaka et al., 2011)

2.9.1.3 Hybrid wetland systems (HWS)

The brief description of the HWS is that:

1. These wetland systems consist of different forms of constructed wetlands staged in series.
2. Most hybrid systems compose of VSSF and HSSF systems arranged in a staged manner.
3. While VSSF units are intended for removal of organics and suspended solids and to support nitrification, HSSF units facilitate denitrification and further removal of organics and suspend solids.
4. The main purpose of hybrid systems is to exploit maximum benefits from combined different wetland types to leverage their advantages. For instance, it is not readily simple to simultaneously create both aerobic and anaerobic conditions in a single-stage system to aid high TN removal but by hybrid wetland configurations, such targets are possible.

2.9.1.4 Floating treatment wetlands (FTWs)

The brief description of the FTWs (Figure 2.3) is that:

1. They are designed based on the principle of using rooted, emergent macrophytes grown on a floating mat rather than rooted in the sediments.
2. These systems are robust to the application of stormwater treatment especially under high flow velocities, and have ability to cope with variable water depth associated with stormwater events.
3. They are endeared systems suitable for extended detention basins.
4. FTW systems have better performance efficiencies than the FWS systems

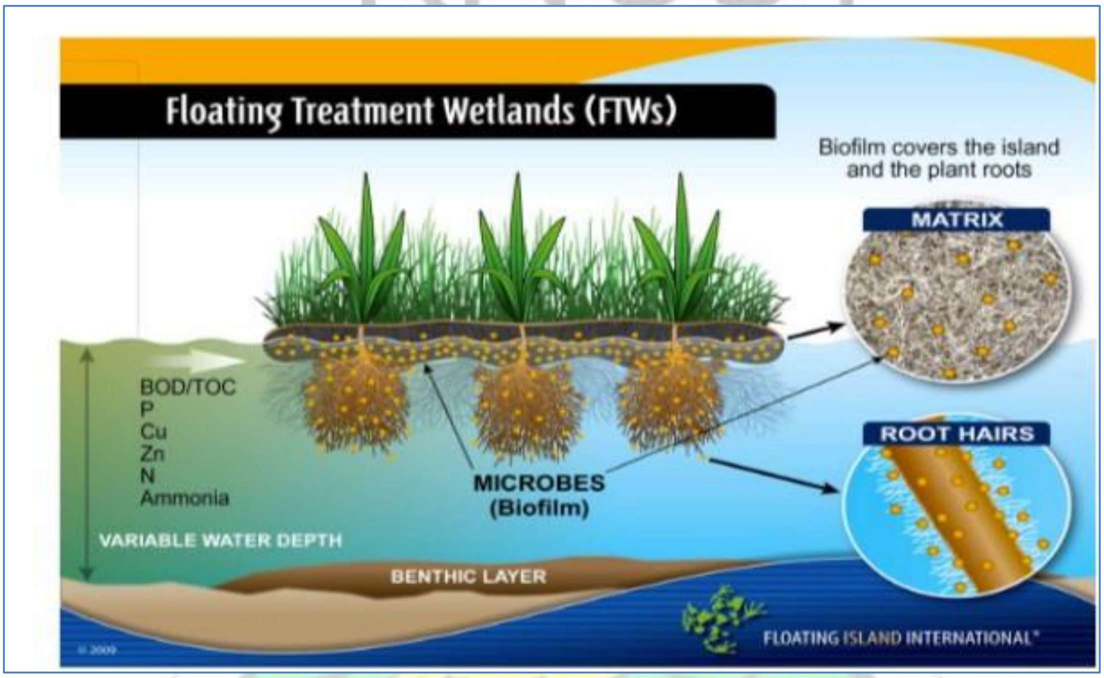


Figure 2.3 An illustration of floating treatment wetland

Source: (CH2MHill, 2014).

The new and emerging forms of constructed wetlands are the hybrid and the floating treatment wetlands. While the hybrid combines two or more of the traditional wetland versions (FWS, HSSF and VSSF), the floating wetlands are a type of FWS (see Figure 2.4) with application of the principles of hydroponic systems. Hydroponic systems, also called floating gardens (Anonymous, 1948), utilize natural processes facilitated by plants and microbes grown in a nutrient-rich solution without any soil or sediment but on top of a floating platform that allows free development of plant roots into the flowing solution/water (Worku et al., 2018a). Figure 2.4 clearly shows the main classes or types of constructed wetlands, and their configurational mechanisms as generally applied in wastewater treatment.

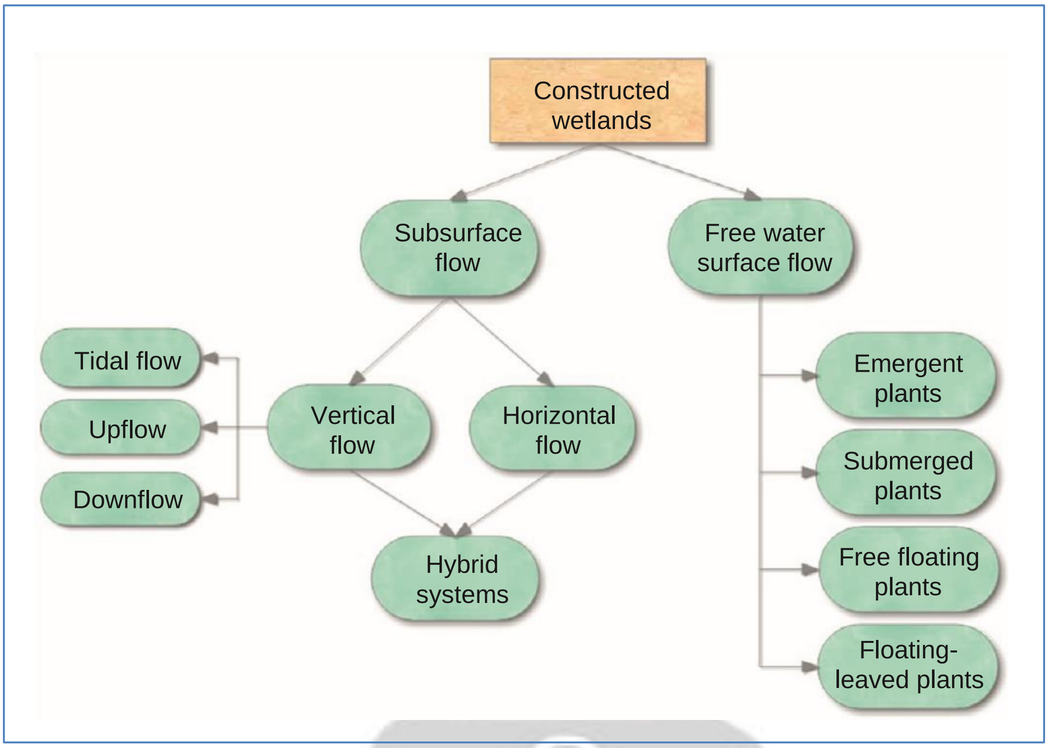


FIGURE 2.1 Classification of constructed wetlands for wastewater treatment.

Figure 2.4 Classification of CWs based on water flow regime
Source: (Stefanakis et al., 2014)

2.9.2 Advantages of constructed wetlands

The technology offers several advantages to users and some of these are listed from literature as follows (Masi, 2004; Ilyas and Masih, 2017):

1. Less expensive over other alternative treatment options.
2. Simple construction, operation and maintenance.
3. Low operation and maintenance costs.
4. High ability to tolerate fluctuations in flow and inlet quality (i.e. high buffer capacity for hydraulic and organic load fluctuations).
5. High process stability and robustness.
6. Sludge produced only by the primary treatment stage (i.e. low sludge production).
7. High pathogen removal and/or inactivation – offers good water reuse and recycling options.
8. Optimal aesthetic appearance (provides green areas and improve environmental quality).

2.9.3 Performance of constructed wetland systems (CWs)

CW has the credibility of treating greywater especially with the removal of major contaminants like turbidity (47 – 97%), TDS (13 – 59%), TSS (25 – 98%), BOD_5 (63 – 99%), COD (81 – 82%), TP (24 – 74%), TN (44 – 59%), Anionic surfactants – AnS (23 – 90%), pharmaceutical and personal care products –PPCP (16 – 96%), and microbial (0 – 3 Log_{10} CFU/100ml) (Ling et al., 2009; Reyes-Contreras et al., 2012; Arden and Ma, 2018; Gupta and Nath, 2018; Oteng-Peprah et al., 2018a; PérezLópez et al., 2018). System performance is always linked to key design features, operational and environmental factors such as bed media, water depth, pH, oxygen, seasonal and climatic conditions including temperature, vegetation, hydraulic conditions (flows, organic & surface loading, and retention time), surface area etc. (Reyes-Contreras et al., 2012; Saeed and Sun, 2012; Morató et al., 2014; Papaevangelou et al., 2016).

2.10 Design considerations for constructed wetland systems

2.10.1 Design approaches

There are several design approaches commonly used for constructed wetlands ranging from "simple but dirty" to more complex forms. The main design approaches that have been reviewed from literature are the rule of thumb, loading charts, and reactor model approaches.

2.10.1.1 Rule of Thumb design approach

This is also called empirical method or black-box model or conservative or scaling factors approach (Masi, 2004; CWRS, 2016; Nagabhatla and Metcalfe, 2018). The approach is called a black-box model because it does not represent internal wetland hydraulics according to Centre for Water Resources Studies (CWRS, 2016). This way of design is considered too simplistic and also used as an alternative and/ or exploratory design option (Rousseau et al., 2004). The approach uses areal coefficients, influent and effluent data relationship, and hydraulic loading rates determined from observations of a wide range of existing systems (Masi, 2004; Rousseau et al., 2004). The areal coefficients normally used include area per population equivalent (surface area/pe), and area per gram of organic load (i.e. area/gBOD or area/gCOD) (Masi, 2004; CWRS, 2016). These are quick and rough approximation models that offer simple mathematical expression for design (Rousseau et al., 2004; Kadlec and Wallace, 2009).

This conservative approach of designing works well and could ensure reliable and functional system designs when appropriate rates are generated from the same or similar settings under consideration (Kadlec and Wallace, 2009; CWRS, 2016). Guidelines for design sizing are developed from extensive experiences and enough data sets on performing treatment systems with different construction practices and materials that are applicable in specific and similar settings like in-country or particular region and geographical context (Masi, 2004; Kadlec and Wallace, 2009). Meanwhile, data is commonly available for developed regions like Europe, North America, United Kingdom, and New Zealand (Kadlec and Wallace, 2009).

Generally, the design approach can be used to check on other design calculations (Rousseau et al., 2004) especially when data is not site specific and effluent compliance with standards cannot be verified (CWRS, 2016). Experts therefore normally caution against reliance on Rule of Thumb approaches because of the following: oversimplification of design process to a single prescriptive scaling factor as a design parameter; over extrapolation beyond local regulations; and use of obsolete prescriptive rules (Kadlec and Wallace, 2009). One can advisably begin CW designs with Rules of Thumb in the preliminary stage, but final decisions must be predicated by performance requirements or regulatory prescriptions (Rousseau et al., 2004; Kadlec and Wallace, 2009).

Some guideline values that could be used for sizing decisions under the Rule of thumb design approach are presented in Table 2.6.

Table 2.6 Guideline values under Rule of Thumb design approach

Criteria	Germany	Austria	USA/Canada	Denmark	Cold climate/ *EU guidelines	Warm climate
Surface area (m^2/pe)	[a] 5, minimum size 20 m^2 [c] 1.7 [e] 4 (VF)	[a] 6 [c] 1.7 [e] 4 (VF)	[e] 5 – 10 (HF)	[a] 1 – 2 [d] 3.2 for CVF with 60 g BOD/pe; 12 g NH_4-N/pe; and 200 l/pe per day [e] 3 (VF)	[c] 4 – 8 [e] 3 – 10 (HF) [e] 1.2 – 5 (VF) [e] 2 – 5 (FVF) [a*] 5 for BOD <300 mg/L; otherwise 10 m^2/pe	[c] 1.2 – 3 [e] 3 – 10 (HF) [e] 1.2 – 5 (VF) [e] 2 – 5 (FVF)
Areal organic load	[e] 20g COD/ m^2d (VF)	[a] 11.2g BOD/m^2d [e] 20g COD/ m^2d for VF	[a] 6g BOD/m^2d 4 – 8g BOD/m^2d (HF) [b] 8kg BOD/ha-d or 0.8g BOD/m^2d	[e] 27g COD/ m^2d (VF)		

Note: 2° - secondary treatment; 3° - tertiary treatment; pe – population equivalent; VF – vertical flow wetland; FVF – French vertical flow wetland; HF- horizontal flow wetland; (Masi, 2004)[a]; (CWRS, 2016)[b], (Hoffmann et al., 2011)[c]; (Randerson, 2006)[d], (Dotro et al., 2017)[e]

2.10.1.2 Loading Charts design approach

The loading chart is a presentation of plots of contaminant effluent concentrations versus mass loadings that serve as a guide to choose a particular mass-loading rate for a specific desired effluent quality or vice versa in constructed wetland design (Kadlec and Wallace, 2009; CWRS, 2016). Thus, loading chart is the positive relationship between contaminant loading rate and effluent concentration (Tao et al., 2017). For instance, an influent loading rate is chosen to give a targeted effluent concentration, and then wetland area is calculated from the mass loading rate (see Figure 2.5) (Kadlec and Wallace, 2009). The loading charts are used as design tools to aid sizing of constructed wetlands (Kadlec and Wallace, 2009; Tao et al., 2017).

Meanwhile, some general classifications put Loading charts and Regression equation approaches together as similar sizing tools used in constructed wetland design (Nagabhatla and Metcalfe, 2018). The two are also considered as scientific approaches, for instance, Areal loading charts are developed for contaminant loads (e.g. biochemical oxygen demand, total suspended solids, total Kjeldahl nitrogen, and total phosphorus) based on first-order pollutant removal rates and non-zero background concentrations (Wallace and Knigh, 2006). Also loading charts are usually developed as design tools based on performance data of existing full-scale and scientifically built constructed wetlands (Tao et al., 2017). Meanwhile, loading chart design approach could also be as sophisticated as the P-k-C^* method (Dotro et al., 2017).

Figure 2.5 shows an example of a typical loading for influent BOD loadings against effluent BOD concentration with illustrations of target effluent concentration in relation to given instances of data set distributions.

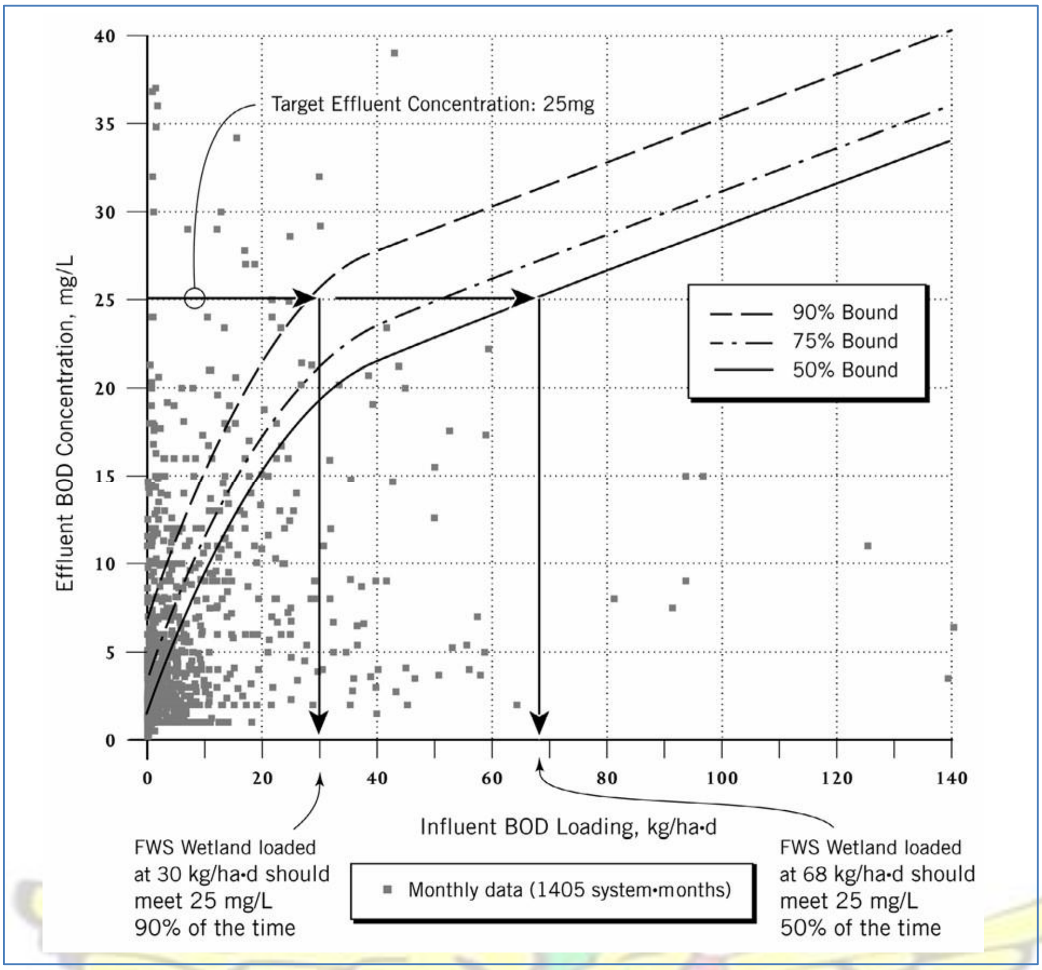


Figure 2.5 Example of a loading chart
Note – the areal influent BOD loading is plotted on the x-axis. Effluent BOD concentration is plotted on the y-axis. The solid and dashed lines bound 50%, 75%, and 90% of the data points in the set. Source: (Wallace and Knigh, 2006)

2.10.1.3 Process or chemical reactor model approaches

These approaches are also given several different names depending on authors and/or designers that describe them. The approaches are mainly identified as listed below from literature (Dotro et al., 2017; Nagabhatla and Metcalfe, 2018):

- Regression equations/models,
- Plug-flow k model, an ideal chemical reactor model with or without background concentration (C^*), and
- P-k-C^*, termed as non-ideal chemical reactor models

Apart from the P-k-C^*, the remaining approaches are considered non mechanistic and based on the equation of first-order reaction in an ideal plug flow reactor (Mena et al., 2008). While kinetic models are used to describe pollutants removal in constructed wetlands, the Darcy law describes the hydraulic aspects of wetlands especially mimicking flows through a porous medium (Masi, 2004; Rousseau et al., 2004).

2.10.1.4 Regression equation/model design approach

This approach extensively relies on the use of Regression models which are generated from a large data sets on performance of existing treatment constructed wetlands (Rousseau et al., 2004; Dotro et al., 2017). The approach requires input and output data points, and these are generally one or two input values (inlet concentration or mass load, and possibly hydraulic loading rate, HLR), to produce expected effluent or outlet concentration (Rousseau et al., 2004; Dotro et al., 2017). The design approach is also labelled as a black box 'model' that oversimplifies a complex system like wetlands into just two to three design parameters and neglects critical design factors like climate, bed material, bed design (length, width, depth) (Rousseau et al., 2004). Very few regression equations exist for both influent concentrations and hydraulic loading rates as inputs for predicting effluent concentrations (Rousseau et al., 2004).

Regression models are associated with large uncertainties and therefore fit best as a tool for analyzing and interpreting input and out data of wetlands systems (Rousseau et al., 2004).

Some examples of regression models for wetland designs are shared in Box 2.1.

Box 2.1 **ole models for Regression equation desi approach**

Parameter	*Model*	*Input range*	*Output range*	*R^2*	*References*
BOD_5	$M_0 = 0.13 M_i + 0.27$	$6 < M_i < 76$	$0.32 < M_o < 21.7$	0.85	(Dotro et al., 2017)
	$C_0 = 0.11C_i + 1.87$	$1 < C_i < 330$	$1 < C_o < 50$	0.74	(Dotro et al., 2017)
	$L_{rev} = 0.653L_i + 0.292$	$4 < L_i < 145$	$4 < L_{rev} < 88$	0.97	(Rousseau et al., 2004)
	$C_o = 0.009C_i + 3.24$	$5.8 < C_i < 328$	$1.3 < C_o < 51$	0.33	(Rousseau et al., 2004)
COD	$M_o = 0.17M_i + 5.78$	$15 < M_i < 180$	$3 < M_o < 41$	0.79	(Dotro et al., 2017)
	$L_o = 0.17L_o + 5.78$	$15 < Li < 180$	$3 < Lo < 41$	0.73	(Rousseau et al., 2004)
TSS	$M_o = 0.048M_i + 4.7$	$3 < M_i < 78$	$0.9 < M_o < 6.3$	0.42	(Dotro et al., 2017)
	$C_o = 0.09C_i + 0.27$	$0 < C_i < 330$	$0 < C_o < 60$	0.67	(Dotro et al., 2017)
	$C_o = C_i \text{ x } (0.1058 + 0.0011 \text{ x } q)$	$22 < C_i < 118$	$3 < C_o < 23$	ng	(Rousseau et al., 2004)

Where:

C_o and C_i = outlet and inlet concentrations, mg/L

M_o and M_i = Mass loads out and into the wetland system, kg/ha.d

$L_{rev,}$ L_o and L_{in} = loads removed, effluent, and influent loads, kg-BOD/ha.d

q = hydraulic loading rate (HLR), cm/d

ng = not given/stated

Note: Models are valid for the specified input and output ranges

R^2 = regression coefficient

2.10.1.5 Plug-flow design approaches (plug-flow k or k-C*)

Almost all the state-of-the-art modelling in constructed wetlands are built around the first-order kinetic equations, including enhancements that incorporate additional design parameters like precipitation, evapotranspiration, temperature, timedependent characteristics etc. (Rousseau et al., 2004). By the use of the Arrhenius temperature equation, water temperature effect can be used to correct areal (k_A) and volumetric (k_V) rate coefficients during design of CW (Dotro et al., 2017).

Although plug-flow model has limitations because actual wetlands do not behave like ideal plug-flow reactors, the model is still widely used and most modern designs still use this approach (Patel and Dharaiya, 2013; Stefanakis, 2016; Dotro et al., 2017; Yuan et al., 2017). The challenge is that predictions of outlet concentrations may suffer deviations from reality due to inability to account for non-ideal and real world hydraulic flows in wetlands (Dotro et al., 2017). In this approach, a model expression may or may not consider background concentration of contaminants (Box 2.2). The model without a background concentration is based on the simplest firstorder model. The background concentration in constructed wetlands is attributed to processes such as autochthonous production and/or sediment releases (Rousseau et al., 2004).

Box 2.2 shows the models used in the First-order plug flow k-C* design approaches.

Box 2.2 First – order equations for Plug-flow (k-C) design approaches

C$_o$=!"#$kτ…………………………….[Eqn 1] (Dotro et al., 2017)

(%& / $ %*)$%* = #$kτ = #$(,-/.)……….....[Eqn 2] (Rousseau et al., 2004)

%) ( )

0$_1$ = 2$_{34}$ 1.06 9$34×;< ……… ....[Eqn 3] (Patel and Dharaiya, 2013)

Where:

C_o = outlet concentration, mg/L

C_i = inlet concentration, mg/L

C^* = background concentration, representing the remaining non-biodegradable pollutant k = first – order reaction coefficient, d^{-1} τ = nominal (theoretical) hydraulic retention time, d k_A = areal first – order reaction coefficient, m/d q = hydraulic loading rate (HLR), =/> (m/d) T = operational temperature of the system (°C) K_{20} = rate constant at 20°C (d^{-1})

d = water depth of the wetland medium/substrate (m) n = porosity of the bed medium/substrate (in fraction or %) e = the base of the natural logarithm

Source: After (Patel and Dharaiya, 2013; Stefanakis, 2016; Dotro et al., 2017)

2.10.1.6 P-k-C* design approach

This design approach relies on the kinetic model based on modified first-order equation with a non-zero background of pollutant concentrations (Dotro et al., 2017). The C^* stands for the background concentration of a contaminant, and k is the first order reaction constant, and P is the apparent number of tank-in-series. The model for this design approach is presented in Box 2.3. This model better describes real world constructed wetland because it is able to account for non-ideal reactor hydraulics, however it is complex and demand generation of several parameters (Dotro et al., 2017).

Box 2.3 **Modified first – order equation for P-k-C* design approach**

$$\frac{C_o - C^*}{C_i - C^*} = \frac{1}{\left(1 + \frac{k\tau}{P}\right)^P} \text{……………………………..[Eqn 1] (Dotro et al., 2017).}$$

Where:

C_o = outlet concentration, mg/L

C^* = background concentration, mg/L

C_i = inlet concentration, mg/L k =

first- order reaction coefficient, d^{-1}

τ = nominal (theoretical) hydraulic retention time, d

P = apparent number of tanks-in-series (TIS), dimensionless

2.10.2 Selection of design approaches – a brief commentary

All the design approaches have their merits and demerits. While some approaches may have comparatively more or less demerits, their selection depends on the wetland designer taking into consideration the approach's feasibility, applicability, and the risk tolerance level and designer's experience (Rousseau et al., 2004; Stefanakis, 2016; Dotro et al., 2017; Nagabhatla and Metcalfe, 2018). A brief commentary on choosing an appropriate design approach after reviewing the just mentioned references is presented in the next four paragraphs.

The most applicable design approach fit for most developing settings like Ghana could be the Plug-flow k-C ideal kinetic model based on a balance between design requirements and the pros and cons. Although too simplistic and inadequate compared to the P-k-C*, it is still valid, widely used and has served as the design basis for many existing and well performing CW systems (Stefanakis, 2016). Also by this same approach, a large number of existing wetland systems have been designed, built and operated to generate credible data to inform other design approaches like the rule of thumb, loading charts, and regression models.

The P-k-C* is proven to be the most suitable design approach for modern constructed wetlands. However, it is too complex and requires too many parameters some of which are not easily measured or determined, a disadvantage that makes it unfeasible in most developing countries settings especially where little or no practical design experience, data, and logistics exist.

The Loading chart and Regression equation approaches are far better than the Rule of thumb, although with weaknesses which make it ineffective compared to the k-C and P-k-C design approaches. This is primarily because in most developing countries like Ghana, there is non-existence of any local data capable of generating standard charts and regression models fit for application in new designs. The two approaches (loading charts and regression equations) require the existence of enough local data on the performance of functional designed constructed wetlands for the generation of appropriate and standard loading charts and regression models.

Rule of thumb approach is too simplistic, may require local data or information as well to start with, and on top of that its use could generate a lot of arguments and scepticism. Simply put, the approach does not necessarily account for any pertaining local conditions and settings relevant for constructed wetland design and performance.

2.10.3 Constructed wetland design features

Some of the key design and operational features associated with constructed wetlands include vegetation, media, hydraulic retention time and baffle-partitions in wetland cells/basins.

2.10.3.1 Macrophytes or vegetation in constructed wetlands

The functions of vegetation in CWs are diverse including uptake, phytovolatilization, release of bactericidal or antimicrobial and other functional exudates, oxygen pumping to the rhizosphere, providing surface for biofilm growth, enhancing filtration effects, stabilizing bed surface etc. (Reyes-Contreras et al., 2012; Morató et al., 2014). However, depending on macrophyte types and other prevailing factors, treatment performance efficiencies may differ (Morató et al., 2014). Some of the popular macrophytes include *Phragmites sp.*, *Ipomoea aquatica*, *Canna indica*, *Gynerum sagittatum*, *Heliconia psittacorum*, elephant grass, Tifton 85 bermudagrass, etc. (Mateus et al., 2014; Madera-Parra et al., 2015; Ramprasad and Philip, 2016b; Gupta and Nath, 2018; Saraiva et al., 2018).

Meanwhile, vegetation in constructed wetlands is also classified based on their growth in relation to wetland water surface (Figure 2.6). By this criterion, there are about four classes namely: (a) submergent macrophytes, grows completely under water (e.g. *Hydrilla verticillata*), (b) emergent macrophyte, roots and part of stem under water but leaves & flowers exist above water level (e.g. *Typha angustifolia*); (c) floating-leaves macrophytes with leaves and flowers/fruites floating on the water surface and roots anchored in the bottom sediments (e.g. *Nelumbo lutea*); and (d) floating macrophytes with leaves and flowers/fruits floating on the water surface while roots just not not in sediments but exist below the surface of water (e.g.

Eichhornia crassipes) (Tanaka and Weragoda, 2011).

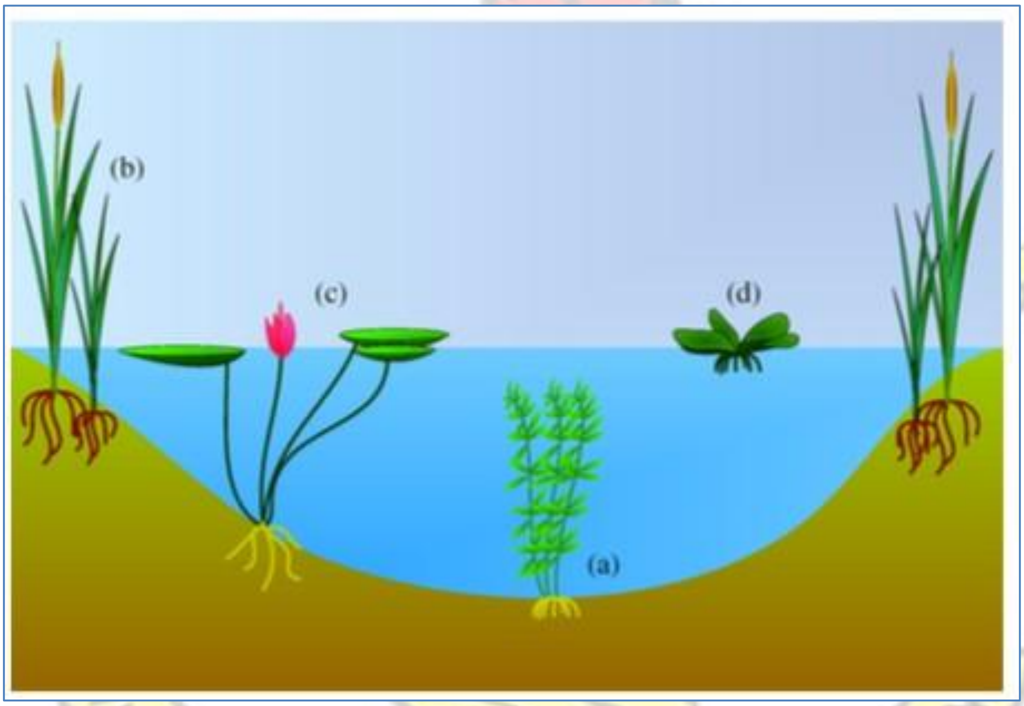


Figure 2.6 Types of macrophytes in constructed wetlands
(a) submergent macrophytes, (b) emergent macrophytes, (c) floating-leaves macrophytes, and (d) floating macrophytes.
Source: (Tanaka and Weragoda, 2011).

2.10.3.2 Specific roles of macrophytes in constructed wetlands

In constructed wetlands (CWs), plants have the potential to become well-established 2 - 3 months but may not begin to reach maturity and equilibrium until late in the second growing season. Also, plants are subject to gradual year-to-year change with some tendencies of species die out and replacements (Davis, 1994).

Vegetation or macrophytes have special specific roles in the treatment processes and they are believed to be critical for performance efficiencies. Two tables (Table 2.7 and Table 2.8) generated from literature (Masi, 2004; Mthembu et al., 2013; Wu et al., 2016) present the main removal mechanisms and the critical roles of vegetation in constructed wetlands. The Table 2.7 shows that the main contaminants from suspended solids through organics, metals, nutrients to pathogens are removed by mechanisms like sedimentation, biochemical degradation, sorption and adsorption, nitrification and denitrification, redox reactions, predation, uptake, filtration, exudates inhibition and toxicity, and irradiation (Mthembu et al., 2013; Wu et al., 2016). From Table 2.8, the tree main parts of the macrophytes – aerial/shoot plant tissue, plant tissue in water, and roots and rhizomes in sediments contribute significantly to most of the mechanisms already mentioned.

Moreover, Figure 2.7 shows how contaminants, plant roots, soil and microorganism interact act the rhizosphere. The most active reaction zone of constructed wetlands is the rhizosphere where physicochemical and biological processes occur under the influence of the interaction of plants, microorganisms, the soil and pollutants (Stottmeister et al., 2003).

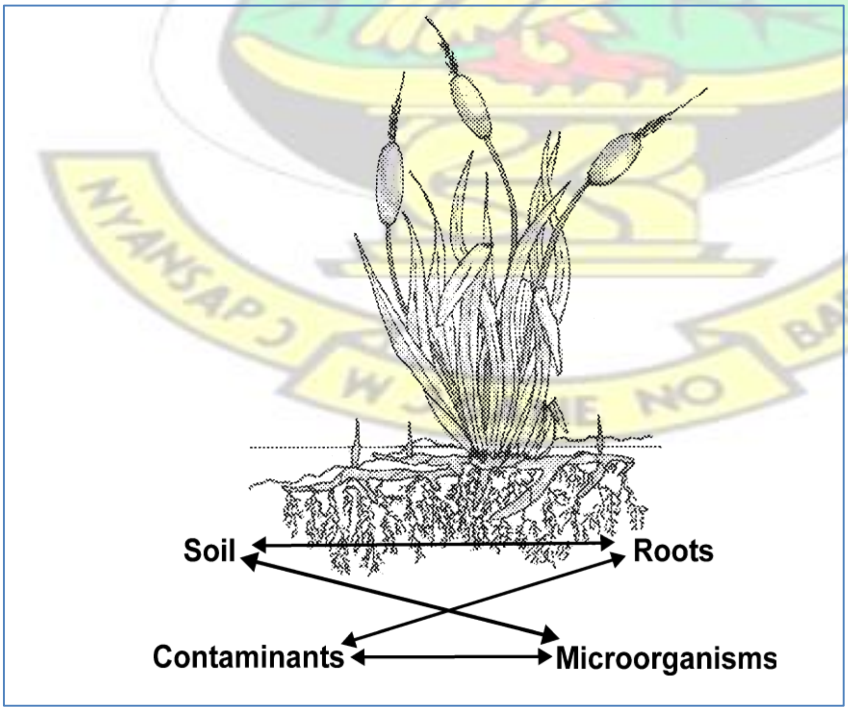


Figure 2.7 Rhizosphere, the active most active reaction in CW
Source: (Stottmeister et al., 2003)

Table
2.7 Contaminants and removal mechanisms in constructed wetlands

Major wastewater contaminants	Removal mechanisms
Suspended solids	Sedimentation (settling by gravity) Filtration (particulate pollutants are filtered mechanically by media/substrate, root masses, etc
Soluble organics/Organics	Aerobic microbial degradation (Bacterial metabolism of colloidal solids and soluble organics by suspended, benthic, and plant-supported bacteria) Anaerobic microbial degradation
Nutrient - Phosphorus	Matrix /media sorption (pollutants are adsorbed onto substrate, plants, roots and other surfaces) Plant uptake (Plant metabolism including absorption or uptake of nutrients - nitrogen and phosphorus)
Nutrient - Nitrogen	Ammonification & microbial nitrification (Bacterial metabolism of colloidal solids and soluble organics by suspended, benthic, and plant-supported bacteria) Denitrification (Bacterial metabolism of colloidal solids and soluble organics by suspended, benthic, and plant-supported bacteria) Plant uptake (Plant metabolism including absorption or uptake of nutrients - nitrogen and phosphorus) Matrix /media sorption (pollutants are adsorbed onto substrate, plants, roots and other surfaces) Ammonia volatilization
Metals	Adsorption and cation exchange (inter-particle attractive forces like the van der Waals) Complexation Plant uptake (Plant metabolism by absorption or uptake of heavy metals and refractory organics as contaminants in wastewater) Precipitation (formation and/or co-precipitation with insoluble compounds) Microbial oxidation/reduction (redox reactions) (decay and/or alteration of less stable compounds by events like UV irradiation, redox reactions)
Pathogens/microbes	Sedimentation (settling by gravity) Filtration (particulate pollutants are filtered mechanically by media/substrate, root masses, etc Natural die-off (natural biological decay of organisms, especially pathogens, due to unfavourable environment created by the wetland system and processes including exposure to UV irradiation; and also, starvation or predation, sedimentation and filtration, and adsorption) UV irradiation (decay and/or alteration of pathogens/microbes by events of UV irradiation) Plant exudates/excretion of inhibitors/antibiotics (Plant metabolism including root excretions which may be toxic to organisms of enteric origin like pathogens)

Source: Adapted from (Masi, 2004; Mthembu et al., 2013; Wu et al., 2016)

Table

2.8 Macrophytes parts and roles in CW treatment processes

Vegetation property	**Key role in treatment process**
Aerial plant tissue	Light attenuation to reduced growth of photosynthesis
	Influence of microclimate of wetland bed by insulation
	Reduced wind velocity to reduce risk of re-suspension of settled contaminants
	Aesthetic pleasing appearance of the treatment system
	Storage of nutrients
Plant tissue in water	Filtering effect-filter out large debris
	Reduced current velocity to aid increased rate of sedimentation, and reduced risk of re-suspension
	Excretion of photosynthesis oxygen - increased aerobic degradation
	Uptake of nutrients
	Provision of surfaces for periphyton attachment
Roots and rhizomes in the sediment	Stabilizing the sediment surface to induce less erosion
	Prevention of the medium clogging in vertical flow systems
	Provision of surface for bacterial growth
	Release of oxygen - increases degradation (and nitrification)
	Uptake of nutrients
	Releases of antibiotics, inhibitors, toxins - to increase die-off in pathogens etc

Source: (Masi, 2004; Mthembu et al., 2013; Wu et al., 2016)

2.10.3.3 Media used in constructed wetlands

Media is used as filter beds to perform essential roles especially in subsurface CWs as support for plants and microbes and deposit sites for removed contaminants. Media also facilitates processes like mechanical filtration and sedimentation (Morató et al., 2014). Media usually cited as substrates in CWs include natural materials – sand, gravel, clay, limestone, zeolite, laterite, shale etc; industrial by-products – fly ash, coal cinder, alum sludge, oil palm shell etc., and artificial ones – activated carbon, lightweight aggregates, compost, calcium silicate hydrate, and ceramics (Ge et al., 2018; Qomariyah et al., 2018). Some of these media with their performance efficiencies are shown in (Table 2.9).

2.9 Some constructed wetland media and performance efficiencies

Substrate	**Removal efficiencies (%)**						
	BOD_5	COD	TSS	NH_4-N	TN	TP	Anionic surfactant

Table

Alum sludge	57-84	36-84		49-93	11-78	75-94	
Biochar	83±8	72±15		83±122	47±5	83±4	
Coal slag	59	64	79	51		38	
Tyre chips	92		69	87	56	65	97
Rice straw				81	78		
Zeolite	82	49		28		20	98
Gravel	63-71	88-93	82-91		30-50	20-27	
PET bottles (polyethylene terephthalate)	60-85	86-92	91-96		21-43	23-29	

Source: Adapted from (James and Ifelebuegu, 2018; Saraiva et al., 2018; Yang et al., 2018)

2.10.3.4 Hydraulic retention/residence time in constructed wetlands

Also critical hydraulic condition that supports development and functionality of microbial biofilms in wetlands is the hydraulic retention time (HRT) (Morató et al., 2014). HRT mainly aids contaminant elimination mechanisms that exist as physical, chemical and biological processes (Reyes-Contreras et al., 2012). It is also found that long HRT helps to improve wetland efficiencies and some studies have shown that a wide range of HRTs, between <1 and 17 days, could be applied to effect treatment (Papaevangelou et al., 2016; Ramprasad and Philip, 2016a; Ramprasad and Philip, 2016b; Gupta and Nath, 2018; Oteng-Peprah et al., 2018a). Table 2.10 shows removal efficiencies in constructed wetlands along different hydraulic retention times for major contaminants like BOD, COD, TSS and nutrients (P and N).

2.10 Contaminant removal efficiencies along hydraulic retention time

Main contaminant	% Removal efficiencies per HRT		
	1 day	3 days	5 days
BOD	87.3	85.9	88.2
COD	87.1	88	87.6
TSS	83	76	57
Nitrogen	94.7	97.5	91.3
Phosphorus	13 – 99.4	99.2	97.6

Table
Source: Adapted from (Gupta and Nath, 2018; Winanti et al., 2018)

2.10.3.5 Baffle-partitions in constructed wetlands

In addition to vegetation, media and HRT, an emerging design feature is the installation of baffle-partitions in wetlands. Baffle-partitions in wetlands could create longer pathways to provided more contact for adequate wastewater interactions with vegetation, media, microbes, and the media-plants root matrix (Ramprasad and Philip, 2016a; Ramprasad et al., 2017). Baffle-partitions could enhance removal efficiencies of contaminants in constructed wetlands to ranges of 84-92% (BOD_5),
86-94% (COD), 88-95% (NO_3-N), 92-98% (TSS), 85-99% (faecal coliforms), 8698% (anionic surfactants) (Ramprasad and Philip, 2016a).

2.10.3.6 Hydraulic regime or feeding modes in constructed wetlands

There are two main feeding modes: batch-load and continuous-flow (Burgoon et al., 1995; Zhang et al., 2012a). The batch-load mode involves alternate draining and flooding or periodic draining and filling, while the continuous-flow mode is constantly feeding or filling the constructed wetland as long as it is operational (Burgoon et al., 1995; Zhang et al., 2012a). Thus, continuous-flow wetlands are continuously supplied with the feed to maintain flooded wetland bed conditions while batch wetlands are made to experience successive flooded – drained cycles (Elsayed et al., 2014).

Although there appears to be dissenting views about the best hydraulic regime (continuous versus batch) for CW especially subsurface flow systems, batch flows appear to be highly recommended. Batch-fed mode of alternate draining and flooding encourages entrainment of air within micro-pores of the media to augment both carbon and nitrogen oxidation (Burgoon et al., 1995; Stein and Hook, 2005; Zhang et al., 2012b; Ni et al., 2013). Table 2.11 shows some removal efficiencies associated with continuous- and batch-flow feeding regimes in constructed wetlands.

Table 2.11 Removal efficiencies of constructed wetlands by batch and continuous feed

Contaminant	**Performance efficiencies (%)**	
	Batch-feed	**Continuous -feed**
Pharmaceuticals (Diclofenac, Ibuprofen, Naproxen etc)	28 - 90	27 - 93
BOD	72 – 100	95 - 99
COD	92 – 96	91 – 95
Faecal coliforms	99	99
NH_4-N	93-95	70 – 81
TP	60 – 67	31 – 43

Source: Adapted from (Zhang et al., 2012a; Zhang et al., 2012b; Yu et al., 2015)

2.10.3.7 Application of CW in developing countries

Constructed wetlands (CWs) are among the few preferred wastewater treatment technologies for low-income settings simply because of low cost, easy maintenance, high treatment efficiency, visual appeal, environmental friendliness, provides ecosystem services like flood control, carbon sequestration and wildlife habitats, diversified treatment alternative for wastewater, and cost-effective for removing a broad range of contaminants (Machado et al., 2017; Gupta and Nath, 2018; CarrascoAcosta et al., 2019). Upon all the numerous advantages, unfortunately however, this green and eco-friendly technology is uncommon and rarely commercially available in developing countries (Denny, 1997; Erakhrumen, 2007). It is not clear the very reasons for the low uptake of the technology in the developing world although these countries are found with favourable warm temperatures and conducive climate to offer greater efficiency (Denny, 1997).

The depressingly low uptake of the technology in the developing world including Africa (Mohan and Hosetti, 2002; Erakhrumen, 2007; Conesa et al., 2012; Mekonnen et al., 2015) could be partly attributed to several factors such as inability to appreciate the resourcefulness of the technology, lack of awareness and/or better understanding of design principles, lack of local skilled manpower or in-house professionals to effectively design and install such technologies, donor driven aid programmes that tend to favour overt and western commercial technologies, and others (Denny, 1997; Kivaisi, 2001). However, some few African countries that are noted in literature for using constructed wetlands at the levels of pilot- and/or fullscale include Uganda, South Africa, Tanzania, Egypt, Morocco, Tunisia, and Kenya (Denny, 1997; Sarneckis, 2000; Masi et al., 2010; Mthembu et al., 2013; Mekonnen et al., 2015). Meanwhile, a report on the review of performance of wetlands adopted in Africa concluded that the technology is highly efficient in removing organic matter (biochemical and chemical oxygen demand) and suspended solids but low on nutrient removal efficiency (Mekonnen et al., 2015).

2.10.4 Potential use of indigenous vegetation for CW in Ghana

The widely used macrophyte in constructed wetlands across the world is *Typha* spp. while *Scirpus (Schoenoplectus)* spp., *P. australis*, *Juncus* spp. and *Eleocharis* spp. are the other frequently used options (Vymazal, 2013).

Generally, there is lack of adequate research in Ghana in areas of phytoremediation as a treatment technology for domestic wastewater management (Anning et al., 2013). Application of constructed wetland in Ghana is rare and so far, very limited studies can be identified with the technology (Amoatey and Bani, 2011; Anning et al., 2013). Two local studies available reported on treating heavy metal contaminated wastewater from a river (Anning et al., 2013) and the other looked at treatment of greywater from students' hall of residence (Niyonzima, 2007). The two studies reported that CW was suitable for domestic wastewater treatment in Ghana. However, the studies did not consider readily available and lesser-known CW macrophytes, and neither used laterite soil media and these present opportunities for further studies as well.

In their studies, Anning et al (2013) assessed the suitability of three locally available macrophytes namely – *Limnocharis flava* L. Buchenau, *Thalia geniculata* L. and *Typha latifolia* L., for constructed wetland technology in treating metal contaminated wastewater. It also means that a well-known and preferred constructed wetland macrophyte like cattail (*Typha* spp.) (Denny, 1997; Vymazal, 2013) is readily available in Ghana because another study explored its use in treating greywater from students' halls of residence (Niyonzima, 2007).

Meanwhile, other studies have also identified some local vegetation in natural wetlands that are extensively receiving wastewater flows especially greywater. Already, natural wetlands are used as convenient sites for wastewater discharge for centuries (Haberl, 1999) and therefore such local vegetation could be potential choice for CW designs. Findings from review of local studies on macrophytes in natural wetlands are highlighted below:

- Studies of natural wetland vegetation distribution in Kumasi city found 48 – 112 plant species in over 10 sites and the vegetation included known macrophytes like *Typha australis,* and *Ipomoea* spp., and less known *Colocasia. esculentus.* Almost all were native to the forest region of Ghana except *Limnocharis flava* and *Ceratophyllum demersum* (Campion and Venzke, 2011; Campion and Odametey, 2012).
- Additional two studies assessed the dominant macrophytes found in two local natural wetlands namely Kpeshie lagoon in Accra and Wiwi wetlands of KNUST campus. The plant species identified to be involved in treating greywater/wastewater were - *Sesevium portulacastum, Avicennia germinans, Paspalum polystachyum, Eleis guineensis, Colocasia esculenta* (taro), *Xanthosoma* spp, *Saccharum officinarum* (sugarcane), and Coix lacryma-jobi, *Aspilia africana* (nfufu), *Nymphaea nouchali* (water lily), *Justicia flava*, *Nephrolepis biserrata* (fern), *Arundinaria gigantea* (giant cane), and *Panicum maximum* (guinea grass) (Muzola, 2007; Ansah et al., 2011).
- Sugarcane is also grown in natural wetlands in Ghana on farms that receive polluted wastewater discharges (Oppong et al., 2018).

Some of these native macrophytes from natural wetlands are aquatic and semiaquatic plants which are potential candidates for selection as vegetation for constructed wetland technology for reasons such as 1) have few environmental and public health risks, 2) require less maintenance, and 3) tolerant to soils, climatic conditions, and harsh seasonal variations (Bindu et al., 2008).

Some indigenous vegetation used or involved in greywater disposal practices among local residents should be explored. For instance, literature shows that plants like, taro and sugarcane are found in very few exploratory studies (Bindu et al., 2008). These plants are common in Ghana as indicated earlier. Review shows that the two plants have strong potential for adoption in constructed wetland technologies (Mateus et al., 2014; Madera-Parra et al., 2015; Rana and Maiti, 2018).

2.10.4.1 Taro (*Colocasia esculenta*) – candidate macrophyte in Ghana

Taro a common semi-aquatic or amphibious herbaceous perennial macrophyte with the scientific name *Colocasia esculenta* and other common names like 'dasheen', 'elephant ears' and 'potato of the tropics'. It is dominantly found in the tropics and subtropics with good growth rate to spread very fast and colonize its environment like weeds (Bindu et al., 2008; Men and Ghazi, 2018; Rana and Maiti, 2018). It is arguably one of the oldest crops on earth which has been grown dating back over 10,000 years (Greenwell, 1947; Gouveia et al., 2018).

The plant can grow up to 1–1.5 m with thick shoots arising from a large corm and has strong metal uptake ability (Bindu et al., 2008; Rana and Maiti, 2018). At about 20 weeks, the leaves and corms develop synchronously to a maximum canopy formation (Sivan, 1980). Taro corms can be harvested at maturity from 9 to 18 months depending on the variety and environmental conditions (Greenwell, 1947; Sivan, 1980).

The plant has high oxalate exudates as non-absorbable salts with unavailable mineral species of Ca, Fe, and Mg, and also the oxalates causes acridity, causing lips, mouth and throat tissues swelling if consumed fresh, but cooking breaks down the raphide renders it harmless in the edible tissues (Greenwell, 1947; Gouveia et al., 2018; Hang et al., 2018). Meanwhile taro is grown as a staple food (from the corms, see Figure 2.8) among tropical and developing countries (Gouveia et al., 2018; Hang et al., 2018). Taro is not only grown for food but also for medical applications (Prajapati et al., 2011). For instance, taro-lactin could be regularly prescribed in infant food and newborn babies can be fed immediately too (Greenwell, 1947). Taro for its rich phytochemicals, has been used since ancient times for curative purposes in the treatment of diseases like asthma, arthritis, diarrhea, internal hemorrhage, neurological disorders, and skin disorders (Prajapati et al., 2011; Krishnapriya and Suganthi, 2017).

Taro possesses the ability to endure or survive high COD loadings in wetland environments and strongly flourishes in abundance than all other wetland plants at most wastewater discharge points (Bindu et al., 2008). Probably this inherent characteristic makes it a strong candidate macrophyte for wetland application.

Figure 2.8 shows pictures of a planted taro vegetation and harvested edible corms.

Figure 2.8 Pictures of taro plant and edible corms
Left hand side) and taro corms (right hand side). Source: (Prajapati et al., 2011)

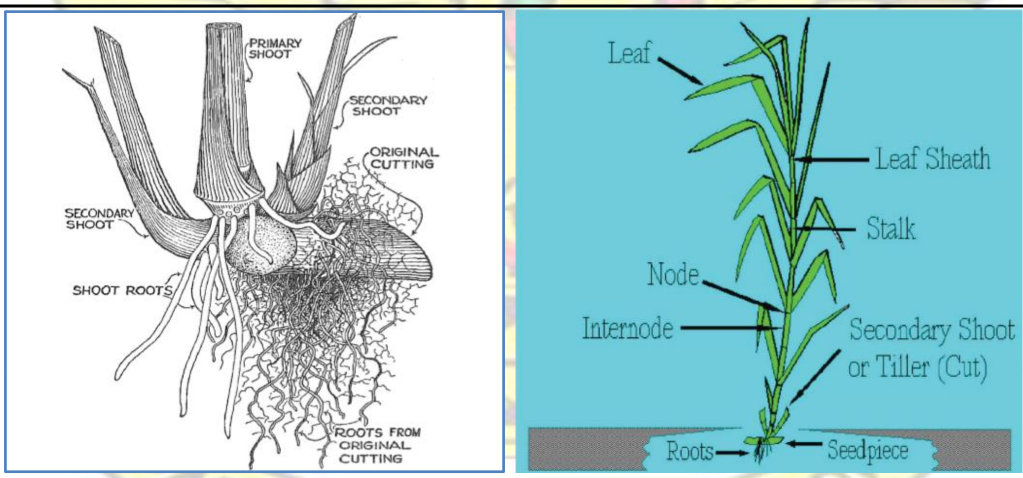
2.10.4.2 Sugarcane (*Saccharum officinarum*)

Sugarcane is known with the scientific name *Saccharum officinarum*. The country Guinea is accorded as the origin of sugarcane since about 6000 BC. It is a tall perennial tropical grass (also called giant grass) with unbranched stems of about 2 – 8 m height and around 5 cm in diameter (TNAU, 2019c).

Sugarcane plant consists of roots, leaves, stem and inflorescences or tassels (TNAU, 2019c). Propagation is mainly asexual, by cuttings (sets, seed cane) with one or more buds which grow into shoots and stems (Figure 2.9) (Dillewijn, 1952). Growth in sugarcane includes increase in dry matter, size and weight. Growth development pattern is termed grand period, starts very slowly during bud germination and increases gradually till it reaches maximum. Growing period could range between 10 - 20 months largely depending on variety and external conditions (Dillewijn, 1952; TNAU, 2019c,b). Seasonal and climatic fluctuations especially temperature and/or rainfall may alter the grand period growth and this could push growing period over one year (Dillewijn, 1952).

Sugarcane water requirement is dependent on the growth stage. Maximum water may be required during tillering and elongation or grand growth phases. Under waterlogged conditions the root respiration becomes poor, nutrients are leached down, activities of useful micro-organisms are reduced and the crop quality becomes poor to give low yield. However, supporting soil or medium must have sufficient moisture to support growth and development (TNAU, 2019a). Sugarcane is a commercial crop for the production of sugar, sugarcane juices, molasses for animal feed and industrial uses, ethanol for automotive fuel or gasoline additives (bioenergy), syrup for preparing commercial foods, producing candies and confectioneries, distilled beverage like rum, production of baker's yeast etc (Mateus et al., 2016; Mateus et al., 2017; TNAU, 2019b).

Figure 2.9 illustrates sugarcane shoots and roots development systems.



(a) (b)

Figure 2.9 Young sugarcane shoots with root primordia of the cutting and shoot roots
Sources: (Dillewijn, 1952)[a] (Sandhu et al., 2016)[b]

2.10.4.3 Risk associated biomass produced from constructed wetlands

There is some degree of risk associated with application of constructed wetlands (USEPA, 1999). Potential hazards from constructed wetlands are a concern because of ecological reality that "everything must go somewhere" (Davis, 1994). Potential long-term ecological risk concerns have been raised with constructed wetlands use because of retention and accumulation of contaminants in the system and plants (Budd et al., 2011). However, less research work has been done in this area of ecological risk assessment of constructed wetlands, such that bioaccumulation and biotoxicity, contaminant detainment and releases are neither well documented nor understood (Davis, 1994; USEPA, 1999). The main concerns are that the processes of bioaccumulation, biomagnification, translocation, bioconcentration and others associated with toxic contaminants are sequestered in plant tissues and substrate, and through the aquatic food web (Helfield and Diamond, 1997).

Details of key terms regarding phytoaccumulation are given by the book "Treatise on Geochemistry" (Adriaens et al., 2003). From the book, bioconcentration is "the accumulation of a chemical in an organism resulting from an equilibrium distribution of the chemical between the organism's tissue and its environment", while biomagnification is "the accumulation of a chemical by an organism from water and food exposure that results in a concentration that is greater than would have resulted from water exposure only and thus greater than expected from equilibrium". Thus, "compounds that biomagnify have greater concentrations in higher trophic levels of food webs". Meanwhile, bioaccumulation is a generic term referring to both processes (Adriaens et al., 2003). Other relevant terms also include: 1) Bioconcentration, expressed as a factor that indicates the accumulation of contaminants such as metals in plants growing in contaminated medium; 2) Translocation, expressed as the factor that indicates the potential of a plant to translocate a metal from roots to shoots; 3) Biotransformation, the biochemical transformation of contaminants in living organisms especially using enzymatic activities – by microbes and plant (phytotransformation) (Arthur et al., 2005; Rana and Maiti, 2018; Xu and Mills, 2018; Tang et al., 2019).

In optimal phytoremediation, preference is not only for plant uptake of contaminants but should further degrade contaminants in tissues and cells to non-toxic metabolites and/or carbon dioxide for atmospheric releases and/or reuse in photosynthetic pathways (Kulakow and Pidlisnyuk, 2007). CWs are able to degrade, transform, or assimilate many contaminants but they are also sinks for some persistent toxic contaminants in the long-term, likely to endanger flora and fauna including humans (Davis, 1994; Lemly and Ohlendorf, 2002; Beharrell, 2004; Wong, 2004). In some cases, plants may not be able to fully degrade contaminants within their tissues or cells, and such contaminants with the associated risks could be released into the environment directly or indirectly by processes like transpiration (phytovolatilization), volatilization, biomass consumption by man and animals etc (Kulakow and Pidlisnyuk, 2007).

Typical example could be illustrated with treatment of selenium-laden wastewater. Selenium is known to strongly bioaccumulate in wetland ecological organisms creating an important dichotomy of removing selenium from water and posturing wetlands as effective treatment tools, while on the other hand wetlands become unsafe by exposing wildlife to toxic levels of selenium (Lemly and Ohlendorf, 2002).

Therefore, some recommended application of biomass produced from constructed wetlands and other phytoremediation systems include the following listed below from literature (Williams, 2002; Bindu et al., 2008; Mateus et al., 2017; Worku et al., 2018a).

1. Used as animal fodder especially when contaminant levels are safe and meet regulatory standards.
2. Raw materials for creative arts and handicrafts
3. Valuable materials for organic farming, e.g. for co-compositing
4. Source of energy – bioenergy, bioethanol fuel etc
5. There is an approach like phytomining, which is the recovery of accumulated trace metals from plant biomass after employing timely, and careful harvesting techniques.

Thus, recommended that biomass should be timely and safely harvested and disposed of using appropriate methods. At least three technologies identified as effective in plant biomass disposal after phytoaccumulation of contaminants specially like lead are: 1) co-firing plant material resulting in 90% concentration to fly ash, 2) 26% reduction by composting, and 3) extraction by chelating agents which could remove over 98% of accumulated lead. Meanwhile, risk issues surrounding fly ash disposal is challenging the utility of co-firing method (Williams, 2002). But this could be safely handled through application of biomass fly ash in mortar and concrete production in the construction industry (Basak et al., 2004; Teixeira et al., 2019).

2.10.4.4 Constructed wetlands versus other traditional treatment alternatives

Constructed wetlands (and combinations of pond/wetland systems) are the best low cost, low-energy, and low-maintenance alternative to traditional wastewater treatment, with wider applications (Langergraber, 2015; Avellán and Gremillion, 2019). "… wetlands can be used in developing countries and provide higher effluent quality and more stable treatment than other technologies (without electrical power input)…" (Langergraber, 2015). In any case, the far less energy requirement is only about 6.8% of the energy demand of a traditional activated sludge plant, and in addition, it has great potential to be net supplier of bioenergy while offsetting greenhouse gas (GHG) emissions (Avellán and Gremillion, 2019). Again CWs' full potential of treatment wetlands abilities is yet to be realised especially with intensification strategies (Langergraber, 2015). One of the advantages of constructed wetlands as phytoremediation strategy probably over other alternatives is that communities may support maintenance of vegetation by taking pride in caring for living systems that are cleansing their environment (Kulakow and Pidlisnyuk, 2007).

Notwithstanding all the positives so far, physical space requirement for CW technology is larger than that for other technologies but comparable to other more traditional low-technology alternatives. CWs will need space of about 2–10 m^2/person as against 0.2–0.5 m^2/person for an aerated lagoon, and even an area for CW will still be larger than lagoon/pond systems but comes with enhanced ecosystem services like aesthetics, biodiversity, wild-life refugia, and nutrient capture for reuse (Hoffmann et al., 2011; Avellán and Gremillion, 2019). The space requirement will also be a problem for adopting the technology in urban context in Ghana where the premium on space is high. However, CW can still be applied as a single-home (onsite) management and medium-density cluster system approaches in urban settings (Wallace and Knigh, 2006).

Chapter 3: Study sites, Approach and Methodology

3.1 Study sites

The sites chosen for the study were in three levels: first is the selection of the sites for household surveys on indigenous knowledge and practices in greywater disposal among peri-urban residents; second is a selected sewered community in the second largest city of Ghana, Kumasi, for characterization of greywater in public drains in addition to literature review; and the third is the site for developing and testing an indigenized constructed wetland technology.

3.1.1 Description of peri-urban areas selected for household surveys

Peri-urban areas also called transition or interaction zones are immediately adjoining urban areas, localized outside formal urban boundaries and urban jurisdictions (Appiah-Effah et al., 2014). Peri-urban communities are difficult to define and at times they are considered as abstract delineations because of diffused and/ or mostly imaginary boundaries in the real world. These areas are generally communities with less infrastructure development and planning, and are characterised by backyard open spaces planted with vegetation and crops normally watered by residents. Some residents of these areas also use planted vegetation in the disposal of their domestic greywater.

The household surveys which aimed at assessing the use of indigenous plants in greywater disposal among peri-urban residents were carried out in the selected areas of Asante-Mampong, Kokoben, and Apromase all in the Ashanti region; Nyankumasi in the Central region; and five (5) communities – Kato, Biadan, Kyiribaa, Kutre, and Senase all within the Berekum Municipality in the Brong Ahafo region (Figure 3.1). These study sites were chosen based on prior knowledge of availability of households or homes practicing irrigation of indigenous plants or vegetation with greywater as means of greywater disposal. These areas are within the middle belt of Ghana largely made of the transition and semi-deciduous ecological zones with similar major soil class (Awadzi et al., 2004; Nuhu et al., 2012).). Besides, it is common to find similar plants species native or indigenous to the forest region of Ghana (Campion and Venzke, 2011).

3.1.2 Description of selected sewered community

The sewered community selected for the study of greywater flowing through public drains is Asafo in the city of Kumasi (Figure 3.1). Asafo is one of the few sewered suburbs in Kumasi. Unlike a similarly sewered community of Kwame Nkrumah University of Science and Technology (KNUST), there is little or no documentation on Asafo's greywater. The suburb has several stormwater drains that are not supposed to carry domestic wastewater, especially black-water, and the worst quality discharges should be flows of light residential greywater from showers and baths. The sewerage system was modified to allow discharge of both grey- and black- water into sewers to minimize blockages especially during periods of unreliable water supply (Awuah et al., 2014). Sampling points in Asafo were chosen following discussion with the Subin sub-metro Environmental Health Officer for places noted for greywater nuisance.

3.1.3 Description of site for testing indigenized constructed wetland

The site selected for developing and testing of indigenized constructed wetland technology for onsite greywater treatment is the campus Kwame Nkrumah University of Science and Technology (KNUST) in Kumasi, Ghana (see Figure 3.1). The campus community is considered as a suburban area of Kumasi, the Ashanti regional capital and second largest city of Ghana. The KNUST campus which is located at 06°41′5.67″N and 01°34′13.87″W hosts a student population around 42,590 (2016 year records) (Wikipedia, 2018). A residential neighbourhood called G-line of Hall 6 area within KNUST campus was chosen because of the following: 1) availability of enough space for experimental setup; 2) good slope for gravitybased sewer line installations; 3) convenient terrain that allows connection of at least four households; 4) availability of at least four households with family sizes not less than 3 people per dwelling unit; and 5) willingness of households to voluntarily participate in the study.

The schematic layout of the experimental site and arrangement of the treatment setup are shown in Figure 3.2 and Figure 3.3.

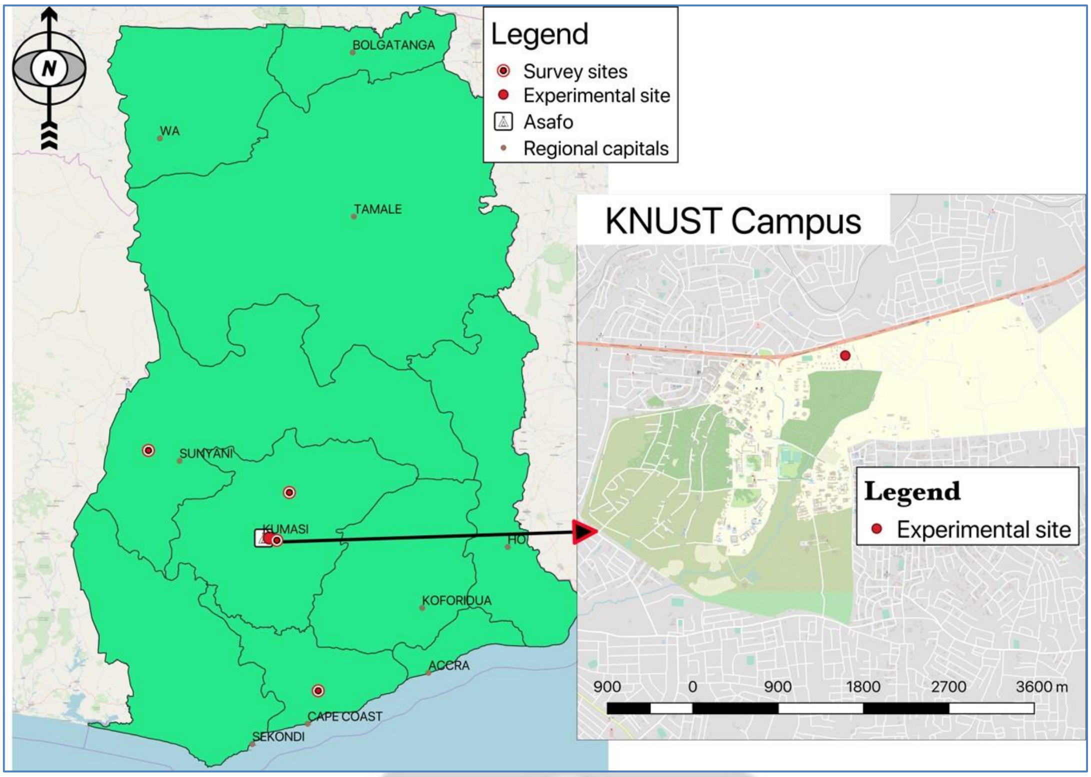


Figure 3.1 A map of all study areas – survey and experimental sites

Note: Survey sites are on the Ghana map, and experimental site on KNUST Campus map

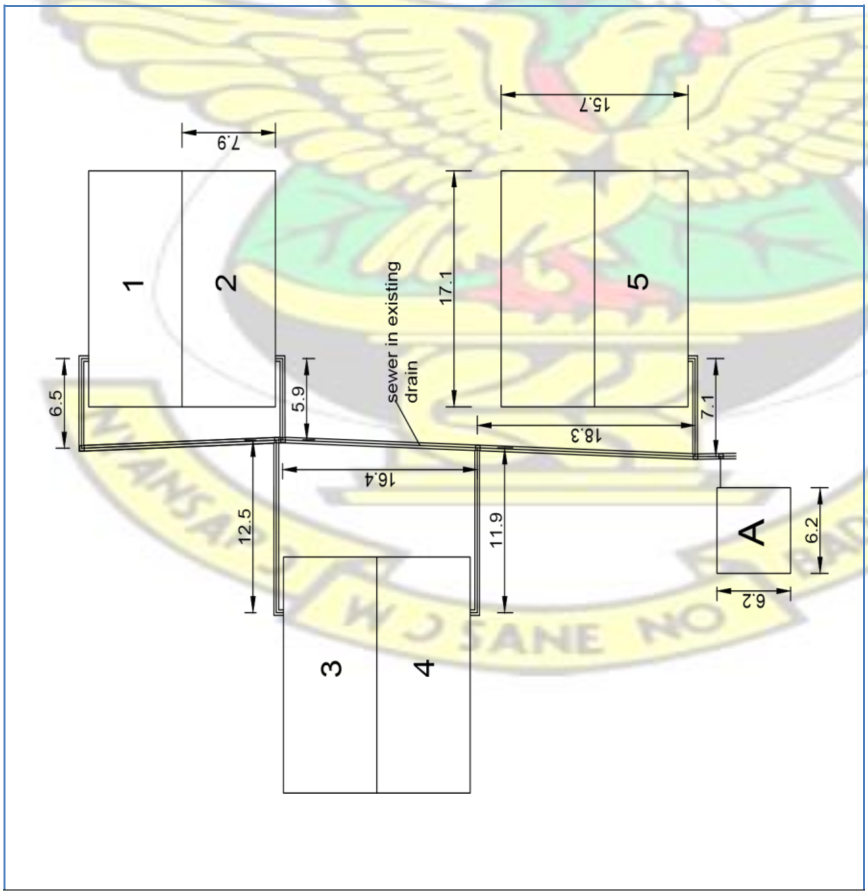


Figure 3.2 Schematic layout of the experimental site

Note: 1-5 are household dwelling units; A=An enclosed space for wetland experimental setup; all distance measurements are in meters.

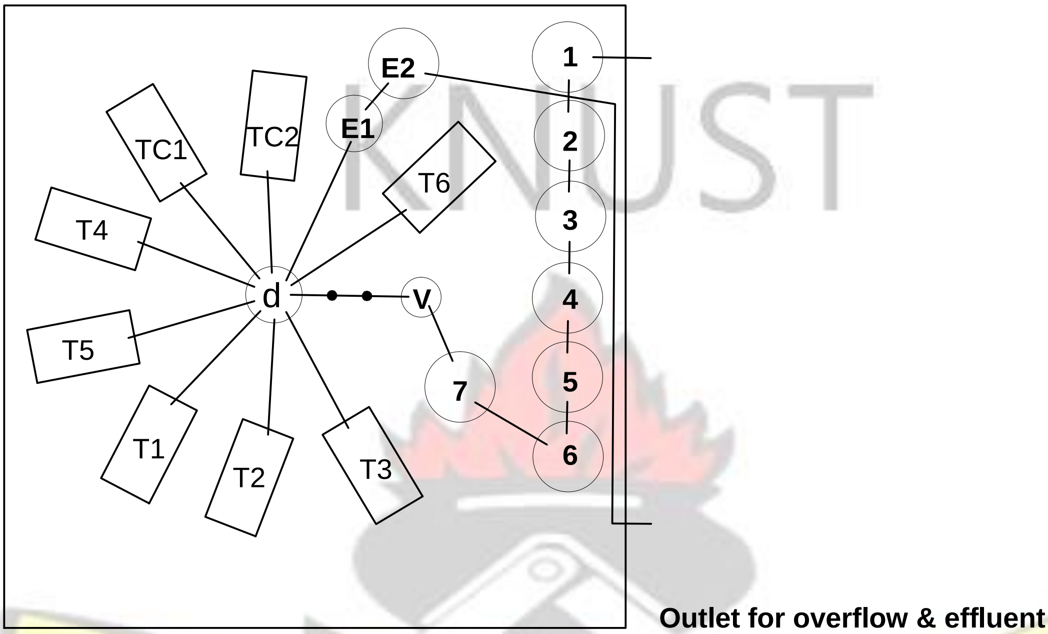


Figure 3.3 Schematic arrangement of the treatment system setup

Note: 1-7 are sedimentation vessels; v= control valve, d=distribution unit; E1&2 = distribution unit extensions; TC1=gravel bed control; TC2=laterite-gravel mix bed control; T1-6 are planted treatment beds

3.2 Approach and Methodology

This section presents the approaches and methods used for data collection and analyses for all specific objectives covered in the dissertation. First of all, the study received ethical clearance from the joint Committee on Human Research, Publication and Ethics of KNUST and the Komfo Anokye Teaching Hospital (KATH) in Kumasi, Ghana.

3.2.1 Characterising greywater from drains and other domestic sources

The sampling sites were chosen following discussion with an environmental health officer (EHO) who indicated greywater odour troubled sites in the Asafo community. A transect walk enabled identification of six sampling sites in the study area. Greywater grab samples were taken at the sampling points within 2 hours on 17 May 2017. Some physicochemical parameters were determined in the field – e.g., temperature, pH, dissolved oxygen (DO), electrical conductivity (EC), and total dissolved solids (TDS) – using Milwaukee Portable pH/EC/TDS (MW-802) and HACH (HQ30d flexi) meters with an Intellical™ LBOD101 optical DO probe. Samples were stored in an icebox and taken to the laboratory within 30 minutes of taking the last sample. The parameters determined in the laboratory included elemental species (cadmium, calcium, copper, iron, lead, magnesium, manganese, mercury, and zinc), nutrients (nitrogen and phosphorus), biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), and microbial identification and enumeration. All laboratory analyses followed the standard methods of wastewater analysis according to the manual of American Public Health Association (APHA), American Water Works Association (AWWA) and Water Environment Federation (WEF) (Clesceri et al., 1999).

Greywater contaminant levels were compared with the characteristics of greywater reported by other studies that were reviewed, and the wastewater discharge guidelines published by Environmental Protection Agency of Ghana (EPA-Gh). The EPA-Gh is the main government agency that regulates all discharges (also domestic wastewater) into the environment, including watercourses. They also enabled greywater samples to be identified as potential sources of nuisance and/or hazard to public health. Any material including wastewater (e.g. "water tainted with impurities", i.e. contaminated or polluted water) with potential to cause nuisance and/or public health hazard should not be discharged into public drains (gutters) according to the local authority (KMA) by-laws and the national building regulation (Fosu, 1996; Ghana Local Government, 1998). The literature review involved assessment of all available literature both peer-reviewed and greywater literature sources by searching various databases including PubMed, Science Direct, Africa Online Journals (AOJ), Google Scholar, and Google Search. The search involved use of key terms like "greywater", "wastewater", "Ghana", "sullage", "light wastewater", "gray water" and their combinations using "AND", "OR". All data were processed and analysed using Microsoft Excel.

3.2.2 Use of indigenous plants in greywater disposal in Ghana

A household survey was conducted with 451 respondents from peri-urban homes in the 5 study communities. The data collection exercise took place between April 2016 and April 2017. The surveys involved interviews and observations centred on greywater disposal practices, use of local plants in the greywater disposal (by irrigation or subsurface infiltration) and other key related themes. The households were purposively selected using the snowball sampling technique that uses "referrals among people who share or know of others who possess some characteristics that are of research interest" (Biernacki and Waldorf, 1981). This is because the sampling required insiders (link-tracers) who had knowledge of houses that use native plants in greywater disposal. The approach allowed the identification and location of our special population (respondents) (Biernacki and Waldorf, 1981; Faugier and Sargeant, 1997; Handcock and Gile, 2016).

The sampling approach was also supported by asking around vicinities for potential respondents, and/ or link-tracers, in places where a respondent is not able to identify the next potential respondent. All data processing and analysis were carried out using Microsoft Excel and SPSS (IBM Mac version 21). The analyses of vegetation involved plant identification and nomenclature (at times using local names that were captured during field surveys), by relying on appropriate databases and literature sources (published and grey) (Amisah et al., 2002; Bonsu, 2011; Amagloh and Nyarko, 2012; Berhow et al., 2012; Lim, 2013; Gadegbeku et al., 2014; CSIR-G, 2016; WOW Magazine, 2016) and Google search engine (with images). The analysis carried out on the data included descriptive statistics using cross tabulations and pivot tables, distributions and trends, 95% confidence intervals, Tukey's Hinges percentile analysis, Chi-square tests at 5% significance level with effect size measures of Cramer's Phi and V.

3.2.3 Characterising greywater from the experimental site

The approach involved a household survey, greywater quantification, greywater sampling and laboratory analyses, and data analysis. The household survey involved use of semi-structured questionnaires for interviews with households. The questionnaires captured information on basic demographics, water consumption, cost of water, greywater sources and disposal, use of personal care and household cleansing agents etc. Five (5) household dwelling units in separate apartments participated in the study. The primary respondents were female heads (adults) because women are largely responsible for managing water, sanitation, hygiene and related issues at the household level (Hyde and Maradza, 2013; Dwumfour-Asare et al., 2017).

An experimental site of about 8m x 8m size was prepared as experimental installations site. The experimental site was connected to the household via 3 and 4 inches PVC pipes installed as greywater sewer lines with some inspection chambers. In some instances, greywater discharge outlets (for bathroom, kitchen and toilet sinks) were refurbished with masonry and/or PVC pipes installation. The greywater from the households were collected through the sewers by gravity without any electro-mechanical pumps. All installations were checked for leakages for two weeks before greywater quantification started. Regular leakage checks were carried out during the study period.

Greywater was quantified using volumetric flow measurement technique of Bucket and Stop Watch, which is simple and useful for small wastewater flows (Palmquist and Hanæus, 2005; USEPA, 2015). Calibrated containers (seven 230 L capacity buckets), and a stop watch were used to determine greywater quantity over every hour (Price, 1991; USEPA, 2015). The installed storage capacity allowed for the containment of greywater for at least 15 hours for composite sampling. Hourly greywater flows were read from the graduation marks in the storage vessels in real time for the first 15 hrs (5am – 8pm) while overnight flows (8pm – 5am) were accumulated and read off early in the morning (5am) to finish up a 24-hr quantification. The quantification was done for two weeks (15 days between November 22 and December 23, 2017). Real time hourly flows over the night could not be determined mainly because of safety and security concerns at the site, and also overnight flows especially from a small population are always low and fall outside notable peak flow periods (Palmquist and Hanæus, 2005; Boyjoo et al., 2013; Awuah et al., 2014; Shankhwar et al., 2015).

Greywater samples were taken during the quantification periods as indicated. Samples were taken each day at the end of 15 hrs (5am to 8pm) from cumulatively stored greywater. After adequate stirring of accumulated greywater, a 1000 ml sample was taken and divided into two aliquots of 500 ml, and an aliquot was used for onsite and main laboratory analyses respectively. Samples for laboratory were wrapped in aluminium foil (to provide dark condition) and also stored under ice in an icebox and immediately taken to the laboratory for analysis within 20-25 minutes after sampling including samples that were taken to third-party laboratories.

The physicochemical parameters that were determined onsite were temperature, pH, dissolved oxygen (DO), electrical conductivity (EC), turbidity and total dissolved solids (TDS) using the instruments Palintest multiparameter pH meter (Micro 800 Multi), Palintest Turbidimeter (v5.12), and Intellical™ LBOD101 DO probe with Hach HQ30d portable meter. The free and total chlorine were analysed using HANNA DPD Pillow method (HANNA, 2003).

Laboratory analyses involved the use of standard methods for wastewater assessment (Clesceri et al., 1999) that were mostly Hach methods (HACH, 2013). These were methods for sulphate (using Sulfaver 4 Hach reagents), nutrients (nitrogen and phosphorus – NitraVer 5, NitriVer 3 and total phosphorus Hach reagents), biochemical oxygen demand (BOD_5) using Dilution method (Hach BOD pillows) with Intellical™ LBOD101 DO probe, and chemical oxygen demand (COD) using the Reactor digestion method (Hach COD digestion vials). Anionic surfactants (Linear Alkylbenzene Sulfonates - LAS) were determined using standard Methylene blue active substances (MBAS) Test kit (TNTplus) from Hach (Shafran et al., 2005; HACH, 2018). The elemental species Ca, Mg, K, Na, Cd, Fe, Pb, Hg, and Zn were analysed using Atomic Absorption Spectroscopy (AAS) at the Soil Research Institute of the Council for Scientific Industrial Research (CSIR), a third-party laboratory.

The data analyses included descriptive statistics, paired-sample t-test at 5% significance level, per capita greywater generation calculations, specific pollutant discharge loads, biodegradability ratio, and sodium absorption ratio (SAR). The expressions for specific pollutant loads and SAR are presented below as:

- Specific pollutant load

  $P_{avc} = C_{avs} \times Q_{avc}$ (Katukiza et al., 2015); where (P_{avc}) is specific pollutant load, (C_{avs}) average concentration of the specific pollutant *S* in greywater, and (Q_{av}) is the average greywater generation in a day.

- The SAR is the ratio of sodium concentration to the square root of one-half of the sum of the concentrations of calcium and magnesium.

  $SAR = [Na]/\sqrt{½([Ca]+[Mg])}$ (FAO, 1992; USDA, 2017); where [Na], [Ca], and [Mg] are sodium, calcium and magnesium concentrations in meq/l.

Wastewater quality characteristics are intrinsically natural positive values with skewed distributions (van Buren et al., 1997; Tchobanoglous et al., 2004; Jian et al., 2011; Oliveira et al., 2012; Wolter, 2018). The most recommended Shapiro-Wilk (SW) test for normality (especially for small samples $3 \leq n \leq 5000$) (Razali and Wah, 2011; Ghasemi and Zahediasl, 2012) confirmed that most parameters (69%, n=22) were statistically normally distributed ($p>0.05$). Therefore, parametric statistical analysis of Paired-Samples T-Test was used to test statistical significance difference between mean values of week 1 and 2. The contaminants' levels were also compared with wastewater discharge limits of Ghana's Environmental Protection Agency (EPA-Gh) and wastewater non-restrictive reuse of World Health Organization (WHO) and Food Agriculture Organization (FAO). All data were processed and analysed using Microsoft Excel and IBM SPSS Statistics (version 23).

3.2.4 Performance of experimental scale constructed wetland for greywater treatment

The main approach adopted for the constructed wetland design was the Plug-flow kC^* kinetic model by considering a balance between design requirements, and the pros and cons. However, the background concentration (C^*) was ignored because of the principle that C^* is considered in design when wetland influent concentrations (C_i) are low (i.e. $C_i \leq 3C^*$, where C^* for HFCW is 10 mg-BOD_5/L) (Dotro et al., 2017). Greywater from the study site has influent load of at least 250 mg-BOD_5/L even after sedimentation pre-treatment. The influent BOD_5 is far more than 30 mg-BOD_5/L, the limit below which background concentration must be considered in wetland designs.

The empirical design equations (Eqn.) used in the study are presented in Box 3.1.

Box 3.1 Key formulas used in the design of horizontal flow CW (HFSCW)

$\tau = \frac{(—)}{\$@A\,BBC)}$,……………………....Eq. (1) (Burgoon et al., 1995; Yocum, 2007)

$k = K_{20}\,(1.06)^{T-20}$ ……………..Eq. (2) (Burgoon et al., 1995)

$\tau = \frac{D*A}{E\&}$ ………………………….Eq. (3) (Buchberger and Shaw, 1995)

$HLR = \frac{E\&}{1I}$ ……………………….Eq. (4) (Tilak et al., 2016)

$Q_o = Q_i - [Q_L + Q_E]$ …………...…Eq. (5) (Dotro et al., 2017)

Where:

C_e = outlet/effluent concentration, mg/L C_i = inlet concentration, mg/L k = first – order reaction coefficient, d^{-1} τ = nominal (theoretical) hydraulic retention time, d L_s = areal contaminant loading rate, HLR x C_i, g/m^2d

Q_i = average daily inflow received by wetland, m^3/d

Q_o = average daily outflow, also design inflow rate, m^3/d from water balance equation

Q_L = loss due to potential leakage, m^3/d

Q_E = loss due to evapotranspiration, m^3/d; Q_E = ET×As

V = volume of wetland, m^3

HLR = hydraulic loading rate, m/d

As = surface area of the wetland bed, m^2

T = operational temperature of the system, ^oC

K_{20} = rate constant at 20^oC, d^{-1}

n = porosity of the bed medium/substrate (in fraction or %)

V*n = effective or available volume of wetland bed to receive influent flows, m^3

In = natural log

ET = evapotranspiration, mm/yr or mm/day

Mostly, mean monthly temperatures of the coldest month are used for safe design (Dotro et al., 2017). From a 5-year data (2012 to 2016) available from KNUST Meteorological Station, the average minimum temperature adopted for design was

19.4 ºC. The reaction rate constant K_{20} assumed for the design was 1.104 d^{-1} for our tropical conditions since no site-specific value was available (Buchberger and Shaw, 1995; Tanaka et al., 2011; Tayler, 2018). From Eqn. 2 (see Box 3.1), first-order kinetic reaction coefficient (design) was calculated as 1.062 d^{-1} for BOD_5, the mostly used wetland design parameter (Stefanakis, 2016; Ramprasad and Philip, 2018).

Again from Eq. (1) in Box 3.1, the nominal hydraulic retention time (nHRT) determined was 1.5 days based on the expected influent and effluent of 245 and 50 mg-BOD_5 L^{-1} respectively (Table 3.1). Influent BOD_5 was based on the conservative assumption that at least 30% of BOD_5 in greywater could be removed after sedimentation pre-treatment (ESF and SG, 2007; Ghunmi, 2009; Hoffmann and Platzer, 2010; Abdel-Shafy et al., 2014). Also by standard performance assessment, wetlands are expected to maintain at most 30 mg-BOD_5/L in effluent (Kincanon and McAnally, 2004), and this level of effluent achievement by estimation may require 2 days nHRT. The two effluent target scenarios meant monitoring wetland performance between 1 and 2 days HRT. However, the study considered three-time point HRT of 1, 2, and 3 days for the experimentation. The obvious preference would be high performance from a shorter HRT to allow for potential gains such as reduction in land size requirement, minimizing odour and aesthetic concerns, and also reducing potential health risk issues associated with long containment durations (Toet et al., 2005; Oh et al., 2018; Mohamed et al., 2019).

The actual void and effective volumes of the wetland were determined after filling the wetland basin (horizontal HPDE plastic barrels) with bed media (Paulo et al., 2009). The determined figures were: effective bed volume of 64 L, and average water depths of 28 cm (0.28 m) for gravel beds, and 30 cm (0.3 m) for gravel-laterite beds, also corresponds to the height of the baffle-partitions. The baffle partitions were 3 installed walls of 28 – 30 cm height that divided the CW basin into three chambers. The first 2 were installed 12cm from the inlet and outlet walls of the basin and the 3rd wall installed in the 40cm midway from the first two on both inlet and outlet sections. The actual porosities of wetland bed filter, which is the void fraction available for water, was 51% for gravel beds and 44% for gravel-laterite beds. The characteristics of the media used for the wetland beds are presented in Table 3.1. Although field conditions allowed for running batch-load and not continuous-flow conditions, the former comes with positive results of air entrainment within bed media to aid oxidation of C and N constituents of wastewater (Burgoon et al., 1995; Stein and Hook, 2005).

Already stated, the choice of vegetation was informed by the findings from specific objective two which assessed greywater disposal practices among some local households. The most commonly used plants included taro and sugarcane which have been selected for the constructed wetland design. The two plants have been selected because it is proven that they have the potential to be adopted as constructed wetland macrophytes as extensively discussed in Chapter 2. The detailed descriptions of the designed wetland microcosm models are presented in Table 3.1, and a generic schematic drawing is shown in Figure 3.4. Any slight differences in physical characteristics of the treatment wetland cells were purely due to constructional factors beyond researchers' control (Aguirre et al., 2005).

Table 3.1 Description of designed HFSCW microcosm models used for the study

Specification	Details

Vessel/basin material	high density polyethylene (HDPE) barrel, 250 L type
Dimensions of a basin	Length =104 cm, radius =28 cm, surface width = 55 cm, surface area = 0.57 m^2
Water depth	28 – 30 cm
Media types	Granitic gravel; gravel-laterite mix
Vegetation	Taro (*Colocasia esculenta)*, Sugarcane (*Saccharum officinarum*)
Planting density	33.52 to 35.29 plants m^{-2}
Granitic gravel media	d_{10} = 5.5 mm, d_{30}=7 mm, d_{60}=8 mm, porosity (51%)
Gravel-laterite mix media (1:1 v/v)	gravel = 43%, sand =29%, Silt and clay = 28%, $d_{30,40,50\&60}$ = 0.1, 3, 6 & 7 mm, porosity (44%)
Inlet and outlet section media	19 – 20 mm size gravels
Nominal hydraulic retention time (nHRT or τ)	1.5 days
Experimental HRT (HRT)	1, 2, & 3 days
Designed effluent BOD_5 level	50 mg-O_2 L^{-1}, according to Ghana EPA wastewater discharge limit
Designed influent BOD_5 level (mg-O_2/L)	245 (approximately 70% of BOD_5 in raw greywater after sedimentation)
Evapotranspiration (ET)	1400 – 1450 mm yr^{-1} (Amisigo et al., 2015; Abubakari et al., 2017) 1.99 – 2.06 L d^{-1}, approximately 2 L d^{-1}
Water loss by leakage, assumed	50% of ET, 1 L d^{-1}
Potential water loss (ET and leakage)	3 L d^{-1}
Effective/available volume of packed bed	0.064 m^3 (64 L)
Design average daily inflow rate (Q_o)	0.043 m^3 d^{-1} (43L d^{-1})
Daily batch-load into wetland (Q_i) incl. ET & leakage	67 L d^{-1}, by throttle valve release of about 187 ml per 10 seconds for 1 h
Hydraulic loading rate (HLR)	0.12 m d^{-1} (12 cm d^{-1})
Operational mode	Batch loading (fill and draw)
Rate constant at 20ºC, K_{20}	1.104 d^{-1}
Design operational temperature	19.4ºC
Slope of bed	≤4%

Specification	Details
Experimental units (8 wetland cells)	
2 Control cells	Controls: unplanted gravel bed (CT1) & gravel-laterite bed (CT2)
6 Treatments cells	Treatments: 2 media and plants, 3 baffle-partitions (28 cm height)
	TT1 – gravel-laterite bed with taro & sugarcane plants
	TT2 – gravel-laterite bed with taro plants only
	TT3 – gravel-laterite bed with sugarcane plants only
	TT4 – gravel bed with taro & sugarcane plants
	TT5 – gravel bed with baffles and planted with taro & sugarcane
	TT6 – gravel-laterite bed with baffles and planted with taro & sugarcane
Pre-treatment option	Sedimentation with HRT of 0.9 – 1.5 days
Screen bucket	Mesh bucket lined with mesh net of <2 mm openings

Note: granitic gravel beds = gravel beds; Laterite-gravel mix beds = laterite based beds or laterite beds

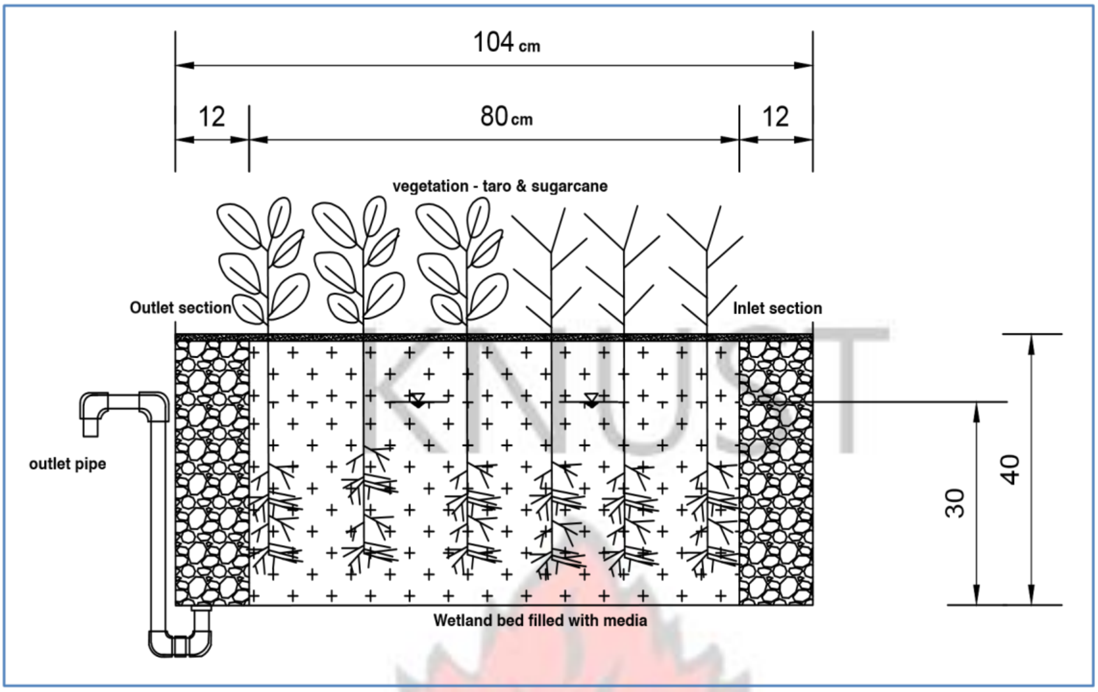


Figure 3.4 A generic schematic drawing of CW microcosm models
Note: all dimensions are in cm

The experimental setup involved eight cells (2 controls and 6 treatments), a pretreatment stage of storage and sedimentation unit with screening, and distribution unit (Table 3.1). The experimental design was factorial that looked at the 8 treatment cells, operated at 3 different hydraulic retention times (1, 2, and 3 days HRT) and each repeated five times, and monitored for system performance using thirteen water quality parameters as outcome variables (pH, EC, DO, TDS, TSS, BOD_5, COD, AnS, NH_3-N, Ortho-P, P, NO_3-N, NO_2-N and SO_4^{2-}). The experimental design allowed direct comparison of the influence of wetland media options, vegetation types, hydraulic retention time, and other key features because all treatment cells received same greywater flows and quality. The first planting of vegetation was done in October 2017, growing and grooming was allowed for 6 months (up to March 2018), and the beds were then exposed to the greywater gradually by dilution with tap water (50%, 30% and 10%) for 2 months for gradual adaptation to greywater (up to May 2018). The beds received influent greywater every 1 – 2 days by filling at 187ml per 10s for 1hr batch-loads up to June 2018. The systems were then fully operated for performance monitoring for four months, thus over a period of at least 100 days (Paulo et al., 2009; Laaffat et al., 2015). Any slight differences in physical characteristics of microcosm wetland cells, the relatively short period of the study, and other practical field challenges like inability to run system in continuous flow regime are considered as limitations.

Figure 3.5 shows an illustration of the experimental setup with the key components.

Figure 3.5 An illustration of the experimental setup with main components
Sedimentation unit, distribution unit, controls and treatment beds, with inserts – sheltered working space (left upper corner).

Grab samples of raw, influent (after pre-treatment), and effluent flows were taken during every batch-load test (Worku et al., 2018b). Effluent from treatment cells were taken at the end of each hydraulic retention period for all five (5) repeated experimental runs. Influent samples were taken from the distribution tank during filling of wetland cells. The raw greywater samples were taken from the first of seven vessels connected in series as a sedimentation unit. During sampling, about 3L of initial effluent from wetland cells was discarded to promote representative sampling of effluent (Aguirre et al., 2005). In all, 152 samples were taken – raw (16), pre-treatment (16) and 120 (control and treatment cells). Water quality analyses involved onsite- and laboratory-based forms immediately after sampling. Almost all physical water quality parameters were analysed onsite and the rest in the laboratory within KNUST campus. Water quality parameters that were assessed included pH, EC, DO, TDS, turbidity and TSS (all physical); organic contaminants – BOD_5 and COD; nutrients – Ortho phosphate, phosphorus, total inorganic nitrogen (nitrate, nitrite, and ammonia); sulphate; anionic surfactants, AnS (as linear alkylbenzene sulfonates – LAS); and dissolved iron. Standard methods of wastewater analyses were used to analyse all parameters by following the approaches and protocols as indicated earlier.

Data analyses included descriptive statistics, contaminant removal efficiency (Rev %), estimation of oxygen transfer rate (OTR) in the wetlands, and inferential statistics of paired T-test and pairwise comparison tests from MANOVA analysis at 5% significant level. The approach and formulae used in some already stated analyses included:

1. Contaminant removal efficiency of wetlands,

   %Removal = $((C_{inf} - C_{eff})/C_{inf}))*100$ (Ling et al., 2009; Abdelhakeem et al., 2016).

   where C_{eff} and C_{inf} are the concentrations of a contaminant in the effluent and influent flows of the wetland respectively

2. Oxygen transfer rate,

   OTR = $[Q*(BOD_{inf} – BOD_{eff}) + 4.3*(NH_4\text{-}N_{inf} – NH_4\text{-}N_{eff})]/A$ (Cooper, 2005; Randerson, 2006).

   where Q is the flow ($m^3\ d^{-1}$); BOD is the biochemical oxygen demand of influent (BOD_{in}) and effluent (BOD_{eff}) flows in mg L^{-1}; NH_4-N is the ammonia of influent ($NH_4\text{-}N_{inf}$) and effluent ($NH_4\text{-}N_{eff}$) flows in mg L^{-1}; and A is the surface area of the wetland (m^2).

Normality test for dependent variables (TDS, TSS, BOD_5, COD, Ortho-P, P, NO_3-N, NO_2-N, NH_3-N, SO_4^{2-} and AnS) with the Shapiro-Wilk (SW) test (Razali and Wah, 2011; Ghasemi and Zahediasl, 2012) showed few fitted the normal distribution function. When the data was log-transformed there was minimal improvement. Thus, data does not usually conform to normal distribution even when log-transformed and such transformation may somewhat reduce but not fully remove heteroscedasticity (Erceg-Hurn and Mirosevich, 2008; Harrar and Bathke, 2012; Feng et al., 2014), especially where multivariate normality can be rare occurrence with real data (Zylstra, 1994; Gupta et al., 2008; Friedrich et al., 2018; Friedrich and Pauly, 2018).

This situation could be observed likely because wastewater characteristics have intrinsic natural positive values although postulated to fit the lognormal distribution function (van Buren et al., 1997; Tchobanoglous et al., 2004; Jian et al., 2011; Oliveira et al., 2012; Wolter, 2018). Because multivariate normality rarely exists in nature and especially with wastewater characteristics, parametric and semiparametric statistical tools were first used to analyse the data for inference convergence of results from both tools. Use of both the classical parametric and semi-parametric MANOVA tools from SPSS (IBM version 23) and Rstudio (Version 1.1.456) statistical software respectively served as a quality assurance check to confirm validity of using parametric analysis for the data. Again, a nonparametric alternative Kruskal-Wallis ANOVA test was performed to validate the two earlier tests on between group effects, and all results converged and gave same conclusions (inference convergence). Full details of the inference convergence of outputs (results leading to the same or common conclusion) as quality control assurance approach for statistical tool selection are presented with the results.

Moreover, a classical parametric statistical tool of MANOVA has power and robustness against some violations to multivariate normality to validate inferential statistics (Olson, 1974; Ito, 1980; Field, 2013). Also, semi-parametric MANOVA tool known as Analysis of Multivariate Data and Repeated Measures Designs (MANOVA.RM) from R (version used was R 3.5.1, 2018-07-02) is robust and does not rely on multivariate normality or specific covariance assumptions (Friedrich et al., 2018; Friedrich and Pauly, 2018). The MANOVA.RM tool is able to analyse non-normal data and therefore allows inferring hypotheses on main and interaction effects in general factorial MANOVA designs using parametric bootstrap resampling technique based on Modified ANOVA-type statistic (MATS) to control Type 1 error, if necessary using the Bonferroni-adjustment method (Yu, 2013; Friedrich et al., 2018; Friedrich and Pauly, 2018). Again, the bootstrapping is “a computationally intensive statistical technique that allows inferences from data without making strong distributional assumptions about the data or the statistic being calculated” (Haukoos and Lewis, 2005).

Inference convergence found with all the three statistical tools (parametric, semiparametric and nonparametric) on the multivariate analysis connoted that normality deviations (if any) were minimal and not extreme to undermine the use of classical MANOVA test in the SPSS environment, and this was confirmed with high observed power for test statistics which assured that Type II error may not occur (Ito, 1980; Erceg-Hurn and Mirosevich, 2008; Nimon, 2012). The SPSS which offered more flexible user interface was then used for all Multiple Comparison Test including pairwise and univariate tests. The significant p-values were based on Fisher's Least Significant Difference (LSD) test at $p<0.05$. Preference for LSD test, a single-step procedure that assumes independence of every comparison, is chosen because of the following:

- the current study explores detecting real possible effects from the individual comparisons,
- the Type II error should be avoided and all possible effects must be identified,
- although LSD test has less control over familywise errors, it is less conservative than Bonferroni adjustment and has more power to detect any real least differences,
- in this exploratory context, identification of any real effect will inform further studies (theory), hence a strict conservative correction would be inappropriate, and finally
- less relevant is the universal null hypothesis that "all null hypotheses are true simultaneously", a requirement for conservative methods like Bonferroni. (Perneger, 1998; Williams and Abdi, 2010; Armstrong, 2014; Lee and Lee, 2018).

Table 3.2 shows the arrangement of wetland setups based on their design features. Also, the table shows list of main wetland systems that were compared for identification of potential effects from key design features on performance. These systems are chosen preferentially because they shared some similar features that allowed more focused comparisons adequate enough to speculate about their effects. The HRT was considered as a co-predictor of effluent water quality, rather than part of the wetlands that were deliberately fitted designs with defined features.

Table 3.2 Arrangement of CW microcosm models based on design considerations

Treatment Systems	**Design factors**			**Design factors**
	Media	**Vegetation type**	**Baffles**	**HRT**
CT1 – control	Gravel (control)	Unplanted (control)	No baffles (control)	

CT2 – control	Laterite soilgravel mix (control)	Unplanted (control)	No baffles (control)	
TT1 – treatment	Laterite soilgravel mix	Mixed – Taro & Sugarcane	No baffles	
TT2 – treatment	Laterite soilgravel mix	Taro	No baffles	
TT3 – treatment	Laterite soilgravel mix	Sugarcane	No baffles	1 day 2 days 3 days
TT4 – treatment	Gravel	Mixed – Taro & Sugarcane	No baffles	
TT5 – treatment	Gravel	Mixed – Taro & Sugarcane	Baffle-partitions	
TT6 – treatment	Laterite soilgravel mix	Mixed – Taro & Sugarcane	Baffle-partitions	
Main systems compared	**CT1, CT2, TT1 & TT4**	**CT2, TT1, TT2 & TT3**	**CT1, CT2, TT1, TT4, TT5 & TT6**	**All systems**

Note: henceforth, laterite soil-gravel mix bed is termed laterite or laterite-based bed in this paper

3.2.5 Prediction models for effluent BOD_5, COD and AnS

The regression method of predicting wetland effluent water quality has been a design approach labelled as a "black box model" because it oversimplifies a complex system (Rousseau et al., 2004). Such regression models are normally generated from a large data sets on the performance of existing constructed wetlands (Rousseau et al., 2004; Dotro et al., 2017), and this gives them some level of credibility as potential design tools and source of useful information. The contaminants listed above were selected based on the following conditions and assumptions:

- First, the effluent levels of contaminants to be predicted should not be widely different (not statistically significant) among treatment wetlands and along the different HRT used in the study. This allowed pulling together enough data points required for robust regression analysis. This condition appeared to be generally well satisfied by the main organic contaminants – BOD, COD and AnS.
- BOD_5: Showed no significant differences in effluent levels along all three HRTs, and also across planted wetlands, only that controls (unplanted beds) differed from the treatment (planted) beds. The data points used for fitting the model only excluded records on control beds.
- COD: No significant differences in effluent levels were recorded among wetlands, and also along the HRTs except 3 days HRT that differed from 1 & 2 days HRT. Only data sets on controls and 3 days HRT were excluded.

- AnS: Only one planted wetland (TT1) differed from about 50% of other planted beds (TT3, TT5 & TT6) in performance. Also, 1 day HRT differed from both 2 & 3 days in influencing AnS removal. Therefore, data points on controls, TT1, and 1 day HRT were all excluded.
- Three separate databases were generated for the contaminants (one for fitting each model), according to the conditions stated earlier.
- All nine main influent contaminants that have been analysed throughout this report and the HRT were added into the model to explore the data for variable selection. Also, an input variable of BOD/COD ratio was added to improve variable spectrum, since biodegradability of these major organic contaminants are interdependent on their ratios.

The multiple regression models were fitted using Stepwise selection method because it combines both forward selection and backward elimination methods (Ghani and Ahmad, 2010; Denis, 2019).

The general linear model for the effluent contaminant levels is defined as the sum of weighted variables:

! = #$ + #& (&+. . +#*(* + + ……Eqn. (1) (Ristinmaa et al., 2013; Yurtsever et al., 2017; Haque et al., 2018).

Where, Y= a dependent variable (any of the effluent contaminant BOD_5, COD & AnS); β = coefficients estimated by the least squares methods (β_0 = intercept point, β_k = regression coefficients); X = independent variables (influent contaminants – TDSi, TSSi, BODi, CODi, Ortho-Pi, NO_3-Ni, NH_3-Ni, AnSi, BODi:CODi, and HRT), k = number of independent variables; and ε = the errors associated with observations. Thus, each effluent parameter was considered as a function of the influent concentrations of the major contaminants, especially those influent parameters that correlated with the effluent.

Chapter 4: Results and discussions

4.1 Characteristics of greywater from drains of a sewered community and other studies in Ghana[1]

4.1.1 Asafo and its sanitation systems

Asafo is part of Kumasi's main business district, within the Subin sub-metro (Subin) under the Kumasi Metropolitan Assembly (KMA). Subin was one of nine sub-metros within the KMA as at the time of study. Kumasi is the second largest city in Ghana and has a population of 2 to 2.7 million people. With an area of about 250 km^2, its population density is about 5,400/km^2 (Mensah, 2006). The extent of the city's sewerage is thought to be similar to that in other Ghanaian cities, including Accra – i.e., well below 10%. Typically, on-site household facilities (water closets, aqua privies, VIPs and other pit latrines, etc.) serve about 47% of the population, public/communal toilets serve about 38%. Open defecation is thought to be practiced by around 3 to 5% (Mensah, 2006; Furlong and Mensah, 2015). Kumasi's sewerage coverage extends to six communities including Asafo (Maoulidi, 2010; Furlong and Mensah, 2015).

Key characteristics of Asafo include: 1) it is a community of tenements and business entities, distinguished by 2 to 3 storey buildings interspersed with single storey buildings; 2) its population density is up to 600 persons/hectare; 3) most houses have 20 to 30 rooms and are shared by up to 20 families (40 to 100 people); 4) the main water source is the Ghana Water Company Limited (GWCL), but there is growing demand for groundwater (mainly for self-supply) due to unreliable service; 5) private flush toilets are common in houses but there are also public toilets (mostly water closets) (Apau, 2017).

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[1] This chapter has been published as a paper with the following details: **Dwumfour-Asare, B.**, Nyarko, K. B., Essandoh, H. M. K., Awuah, E., Anim, K. K. A. & Quaye, A. (2018). Greywater in the drains of a sewered community in Ghana. *Water Practice & Technology*, 13, 4**,** 965-979. doi:10.2166/wpt.2018.103

4.1.2 Asafo's simplified sewerage system

Asafo's simplified sewerage system (ASSS) was a pilot scheme launched in the mid1990s and is one of six functional systems in Kumasi (Maoulidi, 2010; Furlong and Mensah, 2015). It is over 20 years old (Salifu, 2013) and owned by KMA, but operated and maintained by a private franchisee. ASSS was initially designed for 20,000 users (Salifu, 2013) but now serves about 50,000 from households, four schools, public toilets, a tertiary institution (Kumasi Technical University), and the Golden Tulip Hotel (Furlong and Mensah, 2015).

Initially, ASSS was reported to be operating below its intended capacity by serving 60% of the target population due to issues such as unreliable water supply for toilet flushing, user inability to pay connection fees, and access difficulties due to heavily built-up surroundings (Keraita et al., 2003). This probably contributed to the low initial subscription of 255 houses instead of the 320 target (Awuah et al., 2014). The subscription trend was:

- 1997 initial wave of house connections fees (30% in the first three years);
- 2004/2005 end of slow build up to achieve 50%; and,
- 2008/2009 final achievement of 100% connection (Salifu, 2013).

The unreliable and inadequate water supply for toilet flushing became an initial operational issue, causing frequent sewer and manhole blockages, largely because the system was only designed to handle black-water (faecal waste) flushed from toilets. It was later corrected by adding domestic greywater flows to the sewers (Awuah et al., 2014)

There is no current indication of potential expansion of the 900 m^3/day waste stabilization pond treatment facility in Asafo (Salifu, 2013). While the capacity has remained the same, the user population has doubled, and the sewerage system could be overstretched and stressed. Asafo's demography has changed since the sewerage system was designed, because of rapid population growth and urbanisation. The population was 20,000 in 1997, with 63 inhabitants on average in each of the 320 houses – roughly 5 persons per household, – with moderate water consumption of 68 l/c/d (Salifu, 2013). The 2010 Census reports a population of 28,100 comprising 8,162 households inhabiting 1,913 houses (GSS, 2014a). Clearly, some houses might not be connected – other studies indicate that between 60 and 90% of houses are connected to the sewerage system (IWAWW, 2013; Awuah et al., 2014; Greenland et al., 2016). The number of houses has increased six-fold over almost two decades with no evidence of commensurate system expansions. A recent report on wastewater flows in Kumasi asserted that ASSS serves 50,000 people, not 20,000 (Furlong and Mensah, 2015). It is probable that houses not connected to the sewer dispose of greywater into lanes, drains, ditches, open urban spaces, streets, etc. although faecal matter may be contained onsite, a common practice in urban Ghana (Vodounhessi and Münch, 2006; Kuffour, 2010; Gretsch et al., 2016; Nkansah et al., 2016; Dwumfour-Asare et al., 2017; Oteng-Peprah et al., 2018b).

4.1.3 Greywater studies in Ghana

Few greywater studies have been done in Ghana, although greywater research appears to be increasing and more data may become available soon. Greywater includes wastewater from showers/baths, wash basins, laundries, kitchen sinks and dishwashers, but has no feeds or joint flows from toilets, or black-water (Shi et al., 2018).

A brief review of greywater (quality) studies in Ghana is presented in Table 4.1. Eight studies from 5 different Ghanaian locations that were available are reported. The common parameters are pH, EC, TSS, turbidity, BOD_5, and COD. The least reported parameters are TDS, TKN and phosphorus, followed by trace elements and heavy metals (except lead), and then microbiological factors – *E. coli* and total coliforms. The studies covered communities, schools (residential halls/hostels), and hotels, and samples were taken from households (with categories composite/mixed, laundry, kitchen, and bathroom), drains, a lagoon, and a salon.

Almost all studies (Table 4.1) reveal greywater diversity, probably because greywater characteristics depend on a variety of factors including water supply quality and type, household activities (lifestyle, custom, personal care product use, etc), greywater origin (kitchen sink, bathroom, etc), geographic location and demographic characteristics (De Gisi et al., 2016).

The pH of greywater was generally within the range 6.3 to 10, a slightly acidic to alkaline range (El-Fadl, 2007; Mohamed et al., 2019). The pH values were also within the EPA-Gh pH discharge limits – 6 to 9 (EPA-Gh, n.d.). On EC, some studies showed failed discharge limits – 1500 µS/cm (EPA-Gh, n.d.) – but other greywater passed. Two studies reported DO levels well below 4 mg-O/L, which could impact biota negatively especially in water environments, according to National Academies of Sciences, Engineering, and Medicine (NASEM, 2015). The TSS and turbidity levels varied widely across the studies (Table 4.1). One study passed the EPA-Gh discharge limit for turbidity (75 NTU), but the rest failed in addition to the EPA-Gh's TSS discharge limit of 50 mg/L (EPA-Gh, n.d.). High levels of greywater TSS and turbidity, perhaps due to the presence of solids, fabric softeners, and detergent residues (Mohamed et al., 2019), caused failures against EPA-Gh discharge limits. TDS was recorded in only one of the greywater studies, and the level complied with the discharge limit of 1000 mg/L.

The BOD_5 and COD contents exhibited the high variability commonly associated with greywater (Sievers and Londong, 2018). The lowest values reported in the studies were around 60 mg-BOD_5/L and 230 mg-COD/L, and the highest around 540 mg-BOD_5/L and 2,200 mg-COD/L (Table 4.1). The BOD_5 and COD values of the greywater were high and exceeded the EPA-Gh discharge limits (50 mg-BOD_5/L and 250 mg-COD/L). The biodegradability ratios (BOD_5:COD) also showed a wide range, from 0.12 to 0.62. Based on the biodegradability ratio reference level of 0.5 (Kulabako et al., 2011), only greywater from the two Ghanaian locations – KNUST Campus (Kumasi), and Accra Metropolis – can be described as potentially biodegradable, the others having ratios below 0.5. Greywater with low BOD_5:COD ratios are generally rich in chemical contaminants like non-biodegradable surfactants, detergents, etc, and generated by people with low water consumption behaviour (El-Fadl, 2007; Boyjoo et al., 2013; Mohamed et al., 2019). It is noted in this context that EPA-Gh's BOD_5 and COD discharge thresholds do not favour potential environmental biodegradability. The ratio for EPA-Gh limits is around only

0.20, far below the minimum 0.5 biodegradability reference point.

The microbial contaminants found in the greywater sources studied included total coliforms, faecal coliforms and *E. coli*. The microbial loads were high at around 1 to 8 log CFU/100ml (Table 4.1), similar to the pathogenic loads reported in the literature (De Gisi et al., 2016; Shi et al., 2018). However, some greywater sources exceeded the EPA-Gh limits for *E. coli* (1 log CFU/100ml) and total coliforms (2.6 log CFU/100ml), while others did not. Greywater may contain faecal coliforms from bathrooms, laundries, kitchen sinks and dishwashers – e.g., washed off clothing, hands, diapers, childcare items, etc (Ottoson and Stenstrom, 2003; Gilboa and Friedler, 2008). The findings indicate that greywater flows in Ghana contain similar infectious agents to those in other studies, the diversity and concentrations depending on the greywater sources, health status and number (diversity) of waste generators, and the geographic location and its seasonality (Mohamed et al., 2019).

The greywater also contained nutrients (nitrogen and phosphorus), with appreciable levels of nitrate, nitrite, ammonia, and phosphate (Table 4.1). Of the other studies reviewed, those that reported on nitrate showed levels far below EPA-Gh's 75 mgN/L limit, but failed in ammoniacal nitrogen (limit 1 mg/L) and phosphorus (2 mg/L) by between 3 and 8 times. High levels of ammoniacal nitrogen are mostly associated with fresh greywater, suggesting that little or no nitrification has occurred (Mohamed et al., 2019). The phosphorus and phosphate in greywater are connected to the use of household detergents, etc, while the nitrogenous components come mainly from cationic surfactants – e.g., fabric softeners and laundry disinfectants (El-Fadl, 2007; Li et al., 2008; Widiastuti et al., 2008; Mohamed et al., 2019).

Greywater also contains some cationic species – e.g., calcium and magnesium – and heavy metals (copper, cadmium, iron, lead, mercury, manganese and zinc) at varying concentrations (Table 4.1). However, for those macro elements with EPA-Gh specified discharge limits, the levels were generally below the EPA-Gh thresholds – e.g., 2.5 mg-Cu/L, <0.1 mg-Cd/L, 0.1 mg-Pb/L, and 5 mg-Zn/L. Only one study showed failure to meet the EPA-Gh discharge limits for lead and mercury – with lead, around 0.30 mg/L, exceeding by about three times and mercury eighty-fold. Similar macro-element concentrations are found in greywater world-wide (Mohamed et al., 2019). Metal sources are likely to include household plumbing materials, as well as jewellery, cutlery, coins, etc, which can be absorbed onto the skin and washed off, as well as general wear and tear of metal containing household products (Eriksson and Donner, 2009).

Table 4.1 Greywater characteristics (quality) from some studies in Ghana

Water quality parameters	KNUST, University campus	Kpeshie, Accra	Eastern region	Accra Metropolis	Three suburbs, Kumasi
pH	7.5±0.2[a,b] , 6.83[c], 7.74[d]	7.84±0.09	7.84±0.09	6.3±0.47	6.4-9.7
EC (uS/cm)	656.42[c], 965.2[d]	17,100±~3400	628.43±57.46	2985±755.6	351-3530
Temperature (°C)	29.2±0.7[a,b], 29.6[d]	29.11±0.34	29.11±0.34	-	-
DO (mg/L)	2.7±0.9[a,b]	-	1.6±0.30	-	-
TDS (mg/L)	-	-	488.85±23.01	-	-
TSS (mg/L)	212±20.8[a,b], 222.83[c], 347[d]	92.39±26.48	92.39±26.49	11,866±1603[e]	372-4720
Turbidity (NTU)	279.89[c]	72.14±20.47	90.14±3.47	-	204-729
BOD_5 (mg/L)	198.3±33.3[a,b], 420.22[c], 538.5[d]	63.79±26.49	89.79±26.49	309±82	132.5-269
COD (mg/L)	399±108.4[a,b], 707.28[c], 874[d]	236.99±66.35	612.99±66.35	555±119	400-2210
Total Coliforms Log CFU/100ml	6.4±5.8[a,b], 5.7[c]	-	-	-	2.6-8.3
Faecal Coliform Log CFU/100ml	4.93[d]	5.21	2.95	-	-
E. Coli Log CFU/100ml	6.2±5.3[a]	-	-	-	1.0-7.0
Nitrate (mg-N/L)	0.7±0.06[b], 12.91[c], 15.25[d]	2.04±0.49	-	5.2±1.4[f]	-
Nitrite (mg-N/L)	0.0[b], 0.19[c], 57.5[d]	0.1±0.03	-	-	
Ammonia (mg-N/L)	8.4±1.8[b]	-	2.88±0.48	101.3±23.3[f]	
TKN (mg-N/L)	-	-	-	-	7.7-29.5
Total Phosphate - PO_4^{3-}(mg/L)	12.43[c], 26.18[d]	1.24±0.26	-		11.3-23.2
Phosphorus (mg-P/L)	11.8±4.0[a,b]	-	-		-
Calcium (mg-Ca/L)	2.81±0.01[b]	-	-	-	-
Magnesium (mg-Mg/L)	6.1±0.4[b]	-	-	-	0.33-5.67
Cadmium (mg-Cd/L)	0.01±0.001[b], 0.015[c]	0.003±0.002	-	-	
Copper (mg-Cu/L)	<0.01[b], 0.135[c]	-	0.001±0.0	-	
Iron (mg-Fe/L)	0.37±0.08[b]	-	-	-	0.129-0.469
Lead (mg-Pb/L)	<0.01[b], 0.316[c]	0.01±0.0005	0.003±0.0	-	
Manganese (mg-Mn/L)	0.04±0.01[b], 0.098[c]	0.61±0.13	-	-	
Mercury (mg-Hg/L)	0.4±0.08[b]	-	-	-	-
Zinc (mg-Zn/L)	0.03±0.001[b], 0.151[c]	-	-	-	
Settings & sampling sources	School halls/hostels & drains	Community, drains and lagoon	Hotel & hostels	Households (composite, kitchen, bathing & washing)	Households (composite, laundry, kitchen, bathroom & salon
References Note: ± standard deviation	[a](Monney et al., 2013) [b](Awuah et al., 2014) [c](Muzola, 2007) [d](Niyonzima, 2007)	(Ansah et al., 2011)	(Anim et al., 2014)	(Mohammed et al., 2015) [e]Total solids [f]Unit is mg/kg	(Dwumfour-Asare et al., 2017)

4.1.4 Observations and characterization of Asafo greywater

4.1.4.1 Physical observations from sampling sites

The physical observations centred on qualitative information, especially the hygienic conditions around the sites. Figure 4.1 shows two sampling points in the study area, while Table 4.2 provides information from observations made during visits.

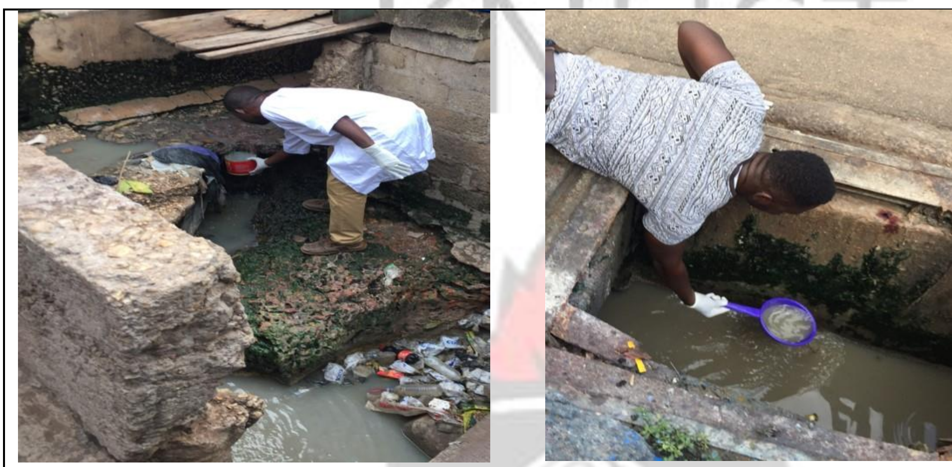

Plate A Plate B

Figure 4.1 Asafo sampling site 1 (Plate A), and site 3 (Plate B)

All the drains where samples were taken are along tarred roads in the community and are engineered (in concrete). The greywater sampled came from residential buildings and other local activities (Table 4.2). The “other activities” include those of vendors, petty traders and squatters, and are similar to household chores – e.g., cooking, washing and cleaning.

At 2 of the 6 sampling sites there were no petty traders, food vendors and/or squatters, just mainly residential buildings. There were no visible signs of blackwater or faecal matter discharges at any sampling sites, including the two occupied by squatters. This is in contrast to what is reported from some urban slum neighbourhoods (Owusu and Afutu-Kotey, 2010; Monney et al., 2013).

People discharged domestic greywater from kitchens, laundries, bathrooms, etc, at almost all sites and only 1 of the 6 was not associated with greywater malodour.
Typically, the smell included strong fresh urine-like odours, and ranged upward from

mild. It is thought that the source could be the ammonia in urine and/or septic conditions arising from the decomposition of organic contaminants, probably aided by the greywater's slow flow and stagnation. There were also instances of solid waste and silt contributing to drain blockage and stagnant greywater flow, conditions that are not uncommon in urban Ghana (Labite et al., 2010; Gretsch et al., 2016).

Table 4.2 Key physical observations from sampling sites

Observations	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Sampling point	Minor and major drain confluence	About 100 to 120 m from, and below, Site 1	About 20 to 35 m from and above site 2	Highest elevation of all sites. Confluence of drains serving residential buildings.	About 200 to 250 m downstream of Site 4	About 100 to 120 m downstream of Site 5
Key features/ landmarks	Asafo market, bus terminal	Main drain in Asafo market	Stormwater drain inspection chambers with cover ripped off	Business enclave in old Asafo township.	Residential area with private kindergarten and pre-school agency.	Residential area opposite a basic school (primary and JSS)
Greywater sources	Residential buildings upstream	Residential buildings and drains uphill, squatters' greywater	Nearby houses and main drains upstream	Residential buildings within the drain catchments	Residential buildings and other drains upstream	Residential buildings and neighbourhood drains
Vendors/ squatters	Vendors/squatters living in area	Vendors/squatters living in area	Store owners, petty traders, and food vendors nearby	No squatters or evidence of them, but some food vendors	No food vendors or squatters, or evidence of them	No food vendors, petty traders or squatters
Faecal matter and/or solid waste	No faecal matter. Some solid waste – e.g., plastics, food peel, leaves, old clothes, etc – Figure 2A	No faecal matter, but some solid waste identified – e.g. food and kitchen debris.	No faecal matter either inside or around the spot/chamber. Possibility of solid waste below the silt deposits	Only kitchen wastewater from food vendors disposed casually into drains	None. No significant silt, either.	No faecal matter, but drain choked with silt and solid waste. Two people seen pouring soapy water into the drain
Stench/odour /malodour	Minimal	Strong, urine-like odour	Presence of malodour	Strong stench. Very turbid and dark greywater flowing through	No odour, even from the sample(s)	Some odour, and there was a complaint of the same in addition to mosquito nuisance.

Greywater stagnation	Minimal. (The concrete drain ends in a runoff created earthen drain)	Slow flow, almost stagnant. Informants confirmed that stagnation always occurs during peak flows	Chamber silted with stagnant greywater (Figure 2B) and wastewater up to 14 cm deep.	Greywater relatively free flowing without significant stagnation	No stagnation, freest flow of all drains visited. Flowing greywater depth was just 5 cm.	Silted/choked drain caused greywater stagnation. Flow minimal.
Other remarks	Solid waste appeared to have been transported and gathered at the concrete drain end	New stores have been built on the drain. People seen urinating and pouring kitchen wastewater into drain	Traces of recurrent kitchen wastewater discharge seen (kitchen oil, food, etc) by the chamber openings	Comparatively small drains – less than 35 cm deep.	Greywater free-flowing and relatively clear.	Least turbid greywater of all sampled, because of stagnation and settling. Water was 25 cm deep because drain was choked.

4.1.4.2 Characteristics of greywater samples from Asafo drains

The characteristics of the greywater samples collected in Asafo are presented in Table 4.3. The greywater pH, which showed slightly acidic to alkaline quality, all fell within the EPA-Gh range acceptable for discharge. The ECs of the greywater at only two sites (5 and 6) were below the 1,500 µS/cm EPA-Gh limit, the lowest being recorded at site 6. High EC levels indicate high loads of dissolved salts and inorganic materials (Prieto et al., 2001). No site's greywater met EPA-Gh's TSS discharge limit (50 mg/L), and only that from site 6 met the turbidity limit (75 NTU). The greywater at site 6 was stagnant and had settled (Table 4.2). The DO concentrations reported were low between 0.3 and 1.6 mg/L (Table 4.3), which probably explains the malodourous environment at some sampling sites, especially sites 3 and 4.

The BOD_5 and COD concentrations at five sites exceeded EPA-Gh's discharge limits (BOD_5 – 50 mg/L, COD – 250 mg/L) significantly. Site 6, however, reported a COD concentration of 207 mg/L (Table 4.3). Generally, the BOD_5 and COD levels are similar to those established in literature – see above.

Nitrogen and phosphorus were analysed for this study as TKN, total phosphate and phosphorus. Phosphorus concentrations were in the range 5 to 23.3 mg/L, even the lowest exceeding the EPA-Gh discharge limit (2 mg-P/L) by a factor of more than two (Table 4.3). Similarly, nitrogen levels were high, with TKN measured in the range 28 to 218.5 mg-N/L. The high nutrient levels found in Asafo's greywater reflect the findings reported by others, confirming that Ghanaian greywater pose a potential eutrophication threat to urban water bodies.

Apart from discharge limits, knowledge of greywater contaminant and nutrient levels is critical for decisions on treatment options. The desirable COD:N:P ratio for biological treatment is 100:20:1 (Boyjoo et al., 2013). Those for the sampled greywater were between 7:1:1 and 55:5:1, however (Table 4.3). They are too low and none of the greywater is considered biologically treatable, as there is no biochemical balance between the biodegradable organics and nutrient levels. However, almost all samples (83% or 5 out of 6) were biodegradable according to the BOD_5:COD ratio.

Greywater with a BOD_5:COD ≥ 0.5 is potentially biodegradable (Kulabako et al., 2011), and five sampled sites gave ratios between 0.58 and 0.65 (Table 4.3). The ratio at Site 2 is substantially below 0.5. The findings are similar to those found in Accra – BOD_5:COD ratios of 0.29 to 0.86 (Mohammed et al., 2015). The BOD_5:COD ratio of 0.26 from Site 2 indicates low potential for biodegradability, similar to some other greywater studied in Kumasi (Dwumfour-Asare et al., 2017). The characteristics of greywater vary widely but those with low potential biodegradability could probably be improved by combination with waters from other sources.

The elemental species determined in the greywater samples were calcium (7.2 – 50.3 mg/L), magnesium (0.1 – 3.3 mg/L), sodium (0.5 – 8.7 mg/L), iron (0 – 0.3 mg/L), arsenic (49 – 191.5 µg/L) and mercury (221 – 1,673.5 µg/L) – see Table 4.3. According to the EPA-Gh discharge limits no sample failed on arsenic (500 µg/L) but all failed on mercury (5 µg/L). This is not positive for the environment or public health since greywater contaminated with heavy metals is currently discharged untreated.

All samples reported the presence of high total and faecal coliform loads, at around 7.2 to 10.4 log CFU/100ml (Table 4.3). There is no discharge limit for faecal contaminants, but the levels (7.2 to 9 log CFU/100 ml) match the total coliforms loads, which exceed the limit between three- and four- fold. The presence of faecal coliforms indicates that the greywater sources were contaminated with faecal matter (from humans and/or animals). Also, solid waste was seen at some sites as well as kitchen greywater discharges (Table 4.2).

Assessment of Table 4.3 shows that four sampling sites (1 to 4) failed all discharge limits for which the relevant species were determined. Two were within the discharge limits for some parameters, however – thus Site 5 was acceptable in relation to TDS and EC, and Site 6 to TDS, EC, turbidity and BOD_5. The threat to the environment, especially water resources but also public health, from urban greywater in Ghana, including this sewered community, is real because of widespread failure in pollutant discharge limits.

Table 4.3 Asafo greywater characteristics of samples from major public drains

Parameters	[a]EPA-Gh guidelines	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Mean	Std. Dev.
pH	6-9	8	6.7	6.6	7.7	7.6	8.2	7.5	0.7
EC (µS/cm)	1,500	1,830	2,950	1,840	4,210	1,280	620	2,121.7	1,278.7
DO mg/L	na	1.5	1.6	0.3	0.5	1.5	1.4	1.1	0.6
TDS (mg/L)	1,000	1,220	2,020	1,250	2,860	870	420	1,440.0	872.1
TSS mg/L	50	248	1,490	447	2,550	272.5	70	846.3	976.3
Turbidity (NTU)	75	360	484	442	2,880	301	39.4	751.1	1,054.6
BOD_5 (mg-O/L)	50	610	600	580	700	480	121	515.2	205.5
COD (mg-O/L)	250	952	2,308	967	1,167	744	207	1,057.5	695.2
BOD_5/COD ratio	na	0.64	0.26	0.60	0.60	0.65	0.58	0.56	0.15
TKN (mg-N/L)	na	117.7	198.9	117.7	218.5	67.2	28	124.7	73.6
Total phosphate (mg-P/L)	na	27.4	41.8	29.6	71.5	15.4	30.4	36.0	19.3
Phosphorus (mg-P/L)	2	9	13.8	9.6	23.3	5	9.8	11.7	6.3
COD:N:P ratio	na	35:4:1	55:5:1	33:4:1	16:3:1	48:4:1	7:1:1	29:3:1	-
Calcium (mg-Ca/L)	na	15.7	7.2	50.3	21.7	12.4	15.9	20.5	15.3
Magnesium (mg-Mg/L)	na	1.2	0.5	3.0	3.1	3.3	0.1	1.9	1.4
Sodium (mg-Na/L)	na	7.3	0.5	2.3	3.4	2.9	8.7	4.2	3.1
Iron (mg-Fe/L)	na	0.1	0	0.3	0	0.1	0	0.1	0.1
Arsenic (µg-As/L)	500	125.8	102	191.8	159.1	149	49	129.4	49.8
Mercury (µg-Hg/L)	5	540	221	264.0	1,673.5	874	1,366	823.1	595.9
Total coliforms Log CFU/100ml	2.6[b]	9	9.6	9.4	10.4	8.3	9.2	9.3	0.7
Faecal coliform Log CFU/100ml	na	7.6	8	7.3	9	7.2	7.9	7.8	0.6

[a](EPA-Gh, n.d.); [b]LogMPN/100ml; na – not available

Note: 1 CFU is equivalent to 1 MPN (Noble et al., 2004; Chen et al., 2017; AWQC, 2018)

4.1.4.3 Strength of sampled greywater

Greywater can be of high (dark) or low (light) strength depending on its pollutant load. In some studies categorisation is by qualitative (source) descriptions – e.g., using "dark greywater" for kitchen, laundry and dishwasher sources, and light for bathroom, shower, bath and washbasin sources (Karabelnik et al., 2012; Barışçı et al., 2016; Cook, 2016). The greywater sources in this study are categorised quantitatively using key chemical and nutrient contaminant concentrations. The framework adopted (Table 4.4) is based on a comprehensive global review of greywater characteristics (Boyjoo et al., 2013). The classification in this study is simplified for logical and practical application purpose, cognisant of greywater' inherent high variability regardless of its strength. The simplification means that any failure in concentration of a single parameter at the low strength limit pushes the greywater to a higher strength class.

Table 4.4 Classification framework for Asafo greywater

Parameters (unit)	**High pollutant-load/ strength**	**Low pollutant-load/ strength**
BOD_5 (mg/L)	>300	≤ 300
COD (mg/L)	>630	≤ 630
Nutrient – phosphorus (mg/L)	>2	≤1.8
Nutrient – nitrogen (mg-N/L)	>17	≤16.4

Source: Adapted from (Boyjoo et al., 2013)

The greywater classification assigned to each site using the framework in Table 4.4 is presented in Table 4.5. All six sites have high (dark) greywater (Table 4.5), suggesting that the greywater came predominantly from kitchens and/or laundries (Karabelnik et al., 2012; Boyjoo et al., 2013; Barışçı et al., 2016; Cook, 2016). This would explain why samples consistently failed to meet most discharge limits, as well as the strong smell associated with sampling sites (Boyjoo et al., 2013).

Table 4.5 Asafo greywater classifications

Parameter (units)	**Site 1**	**Site 2**	**Site 3**	**Site 4**	**Site 5**	**Site 6**
TKN (mg-N/L)	117.7	198.9	117.7	218.5	67.2	28
Total phosphate (mg-PO_4^{3-}/L)	27.4	41.8	29.6	71.5	15.4	30.4
Phosphorus (mg-P/L)	9	13.8	9.6	23.3	5	9.8
BOD_5 (mg/L)	610	600	580	700	480	121
COD (mg/L)	952	2,308	967	1,167	744	207
Classification	High	High	High	High	High	High

Note: Nutrient nitrogen was measured as Total Kjeldahl Nitrogen (TKN) instead of total nitrogen

4.2 Use of indigenous plants in greywater disposal in Ghana[2]

4.2.1 Profile of study households

The gender distribution of respondents is 66% females and 34% males (N=451). The survey purposely targeted female respondents because they are the ones largely responsible for sanitation issues in the home (Dwumfour-Asare et al., 2017). The average age of respondents is 43 years with 95% confidence interval (CI) 41.5 – 44 years (Table 4.6). The majority of the respondents were married 67% whilst the rest (33%) were single, separated, divorced or widowed.

The majority of the respondents were self-employed (56%, N=451), followed by unemployed 25% (n=111) and the balance in paid employment (17%) or retired (2%). The average household size 4.5 is close to the national figure of 4.4 (2010 Census data) and typical of household sizes of rural areas in Ghana (GSS, 2014b). The average number of households in a house or dwelling unit is around 3 (CI 3.1 – 3.5) and this translates into about 14 persons per house, supporting the view that most Ghanaian households live in compound or tenement houses (GSS, 2014b).

Table 4.6 Some biodata of respondents

Parameters	Mean ± standard deviation	95% CI
Age of respondents (in years)	42.8±13.7	41.5–44.0
Number of households in a house	3.3±2.0	3.1–3.5
Number of people in the house	9.8±5.4	9.3–10.3
Household size of respondent	4.5±1.8	4.3–4.6

CI, confidence interval.

The majority of households (61%, n=276) have access to toilet facilities for use in their houses. Moreover, majority of these toilets were improved sanitation facilities comprising 38% (n=107) Ventilated Improved Pit Latrine (VIP) and 18% (n=50) Flush toilets. The low number of wet (flush) toilet facilities is comparable to the national

[2] This Chapter has been published as a paper with details as: **Dwumfour-Asare, B**., Nyarko, K. B., Awuah, E., Essandoh, H. M. K., Gyan, B. A. & Ofori-Addo, H. (2018). Indigenous plants for informal greywater treatment and reuse by some households in Ghana. *Journal of Water Reuse and Desalination*, 8, 4, 553-565. doi:10.2166/wrd.2018.061

figure, which is around 15% (GSS, 2012, 2014b). Availability of wet toilet systems suggest a potential opportunity exists for households to reuse greywater to flush toilets, as was found in similar studies (Kabange and Nkansah, 2015; Dwumfour-Asare et al., 2017).

4.2.2 Greywater handling and disposal practices

All the major greywater streams, namely kitchen, bathroom and laundry, are disposed off in similar manner, as reported in the 2010 census report and another study (GSS, 2013b; Dwumfour-Asare et al., 2017). This includes disposal by septic tanks, soak pits (soak away or catch pits), stormwater gutters or drains, and open discharge onto streets and communal areas in compounds (GSS, 2013b; DwumfourAsare et al., 2017). Almost all houses (99.1%, N= 451) have their greywater sources separated, with minority having both no source separation (0.2%, n=1), and partial separation of sources (0.7%, n=3). The main disposal practices for greywater sources are: discharge into open spaces within compound (46 – 66%) and use of septic tanks and soakaway systems (4 – 24%). The disposal into open is mainly from kitchen (46%, n=208) and laundry (66%, n=296) sources. The septic tanks and soakaway systems were used for greywater from bathroom sources. When all open space disposals (streets, compounds, bushes, and open drains) are aggregated, about threequarters of households use unimproved disposal practices.

The only improved or appropriate final disposal options identified in this study were the use of soakage (soakaway) pits (0.2 – 23%) and septic tanks (0.2 – 2.4%) systems. In Ghana, the minimum requirement for greywater disposal is by the use of soakage pits as defined in local authorities sanitation bylaws and the national building regulations (GoG, 2012; GWMA, 2014). The building regulation recommends seepage pits when the soil and subsoil conditions are favourable (GoG, 2012), but does not give details for their construction/installation, although such details are found in a World Health Organization publications (Fagan, 2015).

4.2.2.1 Greywater use besides indigenous plants irrigation

All the households interviewed used greywater for watering at least an indigenous plant, as this was the criterion for their participation in the current study. However, a relatively large number of households (12.4–63.4%, *n*=56–284) reuse greywater for other domestic end uses, but without any treatment (Table 2). The greywater source used most for alternative end uses was the laundry stream (63.4%, *n*=284), whilst the least was the bath stream (12.4%, *n*=56). This behaviour is likely attributed to the ease of collection and/or the useful volumes. Laundry sources are generated in vessels like buckets, whilst bath sources are discharged directly from the bathroom floors, the point of generation. The greywater from kitchens is usually in low quantities, and contains more contaminants (oil & grease etc.) and therefore has low appeal for reuse. Common end uses for alternative greywater uses are watering down dust, followed by cleaning/scrubbing, and "others" (Table 4.7). Of the greywater from baths that was allocated to alternative uses, most was generated from bathing toddlers and children in basins and vessels. The findings confirm that some households find greywater (untreated and treated) useful for end uses such as toilet flushing, watering lawns, car washing, and fire extinguishing, especially in places where water supply could be scarce and/or erratic (Kulabako et al., 2011).

The Ghana National Building Regulation promotes reuse of treated wastewater, including greywater, for non-domestic end uses such as water for cooling, toilet flushing, lawns, parks, fire-fighting and certain industrial purposes (GoG, 2012). However, the current regulations do not encourage domestic reuse practices observed in this study, and this should be food for thought for policy makers, and other influential stakeholders.

Table 4.7 Main alternative end uses of greywater flows

Greywater source	Main specific end use
Kitchen	16% (*n*=74) – watering compound against dust
	2% (*n*=7) – others (quenching fire – firewood and charcoal)
	82% (*n*=370) – none
Bath	11% (*n*=50) – watering compound against dust
	1% (*n*=6) – cleaning/scrubbing floor and flushing toilet
	88% (*n*=395) – none

Laundry	41% (*n*=186) – cleaning/scrubbing floor 19% (*n*=84) – watering compound against dust 3% (*n*=14) – others (washing bike and flushing toilet) 37% (n=167) – none

For those households that did not reuse greywater (36%, *n*=161) other than watering native plants, there are three main reasons for avoiding greywater use (Figure 4.2). The majority perceived greywater to be unsafe or not efficacious (73%, *n*=117), followed by "seeing no reason for reuse" (24%, *n*=39), and then lack of awareness that greywater could be reused (3%, *n*=5) (Figure 2). These results largely confirm similar health risk concerns and perceptions from other greywater reuse surveys, especially if the greywater is untreated (Kabange and Nkansah, 2015; DwumfourAsare et al., 2017).

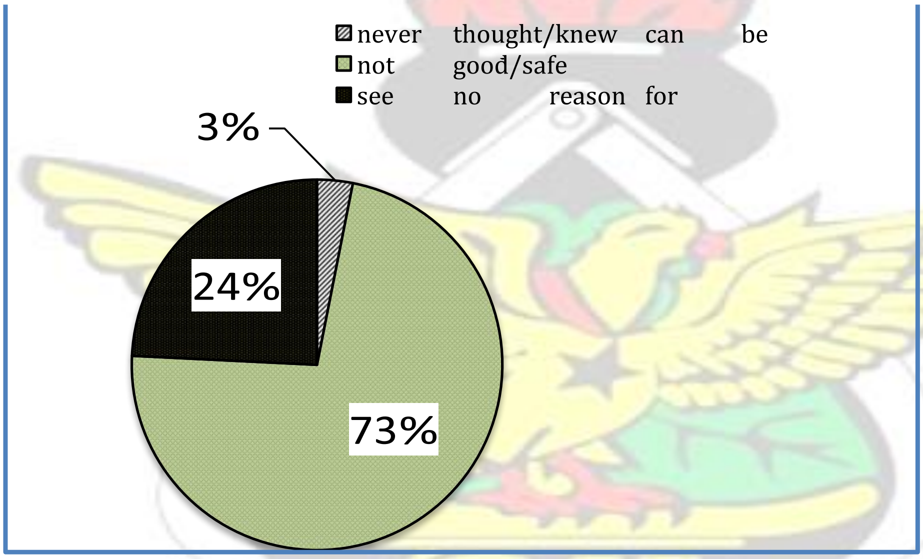

Figure 4.2 Main reasons for non-reuse of greywater, other than for watering plants

4.2.2.2 Indigenous plants use in greywater disposal

The main greywater source for watering indigenous plants is bath water. This was well supported by the visual evidence of planted vegetation along the disposal courses for bathroom greywater, in all 451 houses visited in the study. The respondents' level of awareness for any specific roles or functions played by plants in the greywater disposal was assessed (Table 4.8).

The majority of respondents (84%, n=378) claimed they knew what plants were doing to the "dirty" water (greywater) discharged from their residence. The average number of beneficial functions performed by plants, according to respondents, is 1.6 (CI 1.48–1.72). Thus, each participant could mention approximately two functions performed by the plants in the greywater disposal. Also, the Tukey's Hinges analysis, which collaborated well with the weighted average percentiles, was found to be 1, 1, and 2 stated functions for the lower hinge (25th percentile), mid-hinge (50th percentile), and upper hinge (75th percentile) respectively (Table 4.9). Thus, most respondents (75%) were able to mention 1–2 roles played by native plants in their current greywater disposal practices. Moreover, the top 25% of respondents could list as many as four beneficial functions of the plants (Table 4.9). The findings are interesting and support the existence of informal indigenous knowledge, which could be explored further by scientific study. Moreover, the underlining fact is that plants in subsurface infiltration systems offer some level of treatment for wastewater, and this kind of technology falls under the practice of phytoremediation.

The main plant functions that were identified by the respondents are listed in Table 4.8.

Table 4.8 Responses on the roles/functions of planted native vegetation to greywater

	Specific functions of plants	*Distribution of responses, n (%)[a]*
1	Treat greywater	351 (78%)
2	Remove odour	159 (35%)
3	Remove "poison/danger"	100 (22%)
4	Remove particles	77 (17%)
5	Absorb water	22 (5%)
6	Kill germs	13 (3%)

[a]Multiple responses were allowed.

Table 4.9 Weighted average and Tukey's Hinges - plants' roles and functions

Statistics	**Percentiles**						
	5	**10**	**25**	**50**	**75**	**90**	**95**
Weighted average roles	0	0	1	1	2	4	4
Tukey's Hinges roles			1	1	2		
Weighted average beneficial functions	1	1	2	3	4	6	6
Tukey's Hinges beneficial functions			2	3	4		

The most familiar plant function was "treat greywater", followed by "remove odour" and the rest in the order presented in Table 4.8. The number of responses decreased sharply after the top two common functions, probably because respondents considered that the remaining functions were covered, or defined, as part of treating greywater. This explanation is consistent with the previous discussion of the Tukey's Hinges analysis (Table 4.9), where the majority of respondents knew only one or two plant roles, and only a minority could list more than two plant functions. Overall, we suggest that there is widespread native knowledge of greywater phytoremediation, albeit at an elementary level.

The people understand that greywater is directly used to irrigate indigenous plants, and these plants in return treat the greywater. However, the terminal state of the greywater is still haphazard disposal with no scientific proof of treatment after irrigating with plants. The haphazard final disposal of greywater is not necessarily due to reuse on plants but lack of priority for safe greywater disposal. This is because the underlying and principal intention for the use of vegetation in greywater disposal may not be for treating and/or ensuring safe disposal per se, but rather irrigating the plants for other gains (discussed in detail in the next subsection). Perhaps, it is for this reason that some respondents (a minority of 16%) claim no knowledge of what plants do to their greywater. Nevertheless, deepening local understanding to improve native phytoremediation practices seems highly worthwhile in order to reduce unsafe management of greywater.

4.2.2.3 Indigenous plants identified: numbers, types, and derived benefits

The total number of plant groups identified at the 451 houses visited during the field survey was 1,259. The plant numbers are more than twice the number of houses visited because most houses had concurrently planted two or more different plant species at the main greywater disposal sites. The descriptive statistics show the mean-standard deviation of 2.8±1.4 with the range of 5 (1–6) plants per house, with the majority (80.3%, n=362) practicing plant polyculture compared with a much smaller number (19.7%, n=89) practicing monoculture. Moreover, the number or type of plants grown was not dependent on knowledge of the treatment offered by the plants per se, but rather on the benefits derived from plants (as discussed previously). The statistics show no significant association (χ^2=6.022, p=0.304) between the number of plants grown and the known role of plants to "treat greywater".

The results presented in Table 4.10 support the view that households use vegetation in greywater disposal apparently for the benefits that plants produce, rather than wastewater treatment. Only one house reported no benefits derived from the plants, although four different plants were grown onsite for greywater disposal. More observations are made around two and more plants numbers and derived benefits, and these translate into statistically significant association between numbers of plants grown and derived plant benefits (χ^2=161.94, p<0.001).

Table 4.10 Test for association between number of plant types and plants' benefits

Number of plant types	Number of plants' benefits – Distribution, n (%)						χ^2 (p)
	None	One	Two	Three	>Three	Total	
1	0(0)	41(41.4)	18(25)	21(18.8)	9(5.4)	89(19.7)	161.94
2	0(0)	41(41.4)	25(34.7)	37(33)	16(9.6)	119(26.4)	(0.000)
3	0(0)	12(12.1)	17(23.6)	24(21.4)	41(24.6)	94(20.8)	
4	1(100)	5(5.1)	11(15.3)	18(16.1)	45(26.9)	80(17.7)	
5	0(0)	0(0)	1(1.4)	12(10.7)	53(31.7)	66(14.6)	
6	0(0)	0(0)	0(0)	0(0)	3(1.8)	3(0.7)	
Total	1(100)	99(100)	72(100)	112(100)	167(100)	451(100)	

Phi coefficient = 0.599
Cramer's V = 0.300

In all 36 different plant species were identified 1,259 times (see Figure 4.3 and Table 4.11). However, for the purpose of simplicity in this study, some plants have been regrouped based on related their local uses (e.g. vegetables – Corchorus and Amaranthus), fruit crop (e.g. mango and orange), and species (e.g. banana and plantain, basils, garden eggs and turkey berries) into 30 main plant categories (Table 4.11). All the plants identified in the survey are locally available, and households intentionally planted almost all of them. Very few were self-sowing volunteer species that were nevertheless allowed to grow with the planted species.

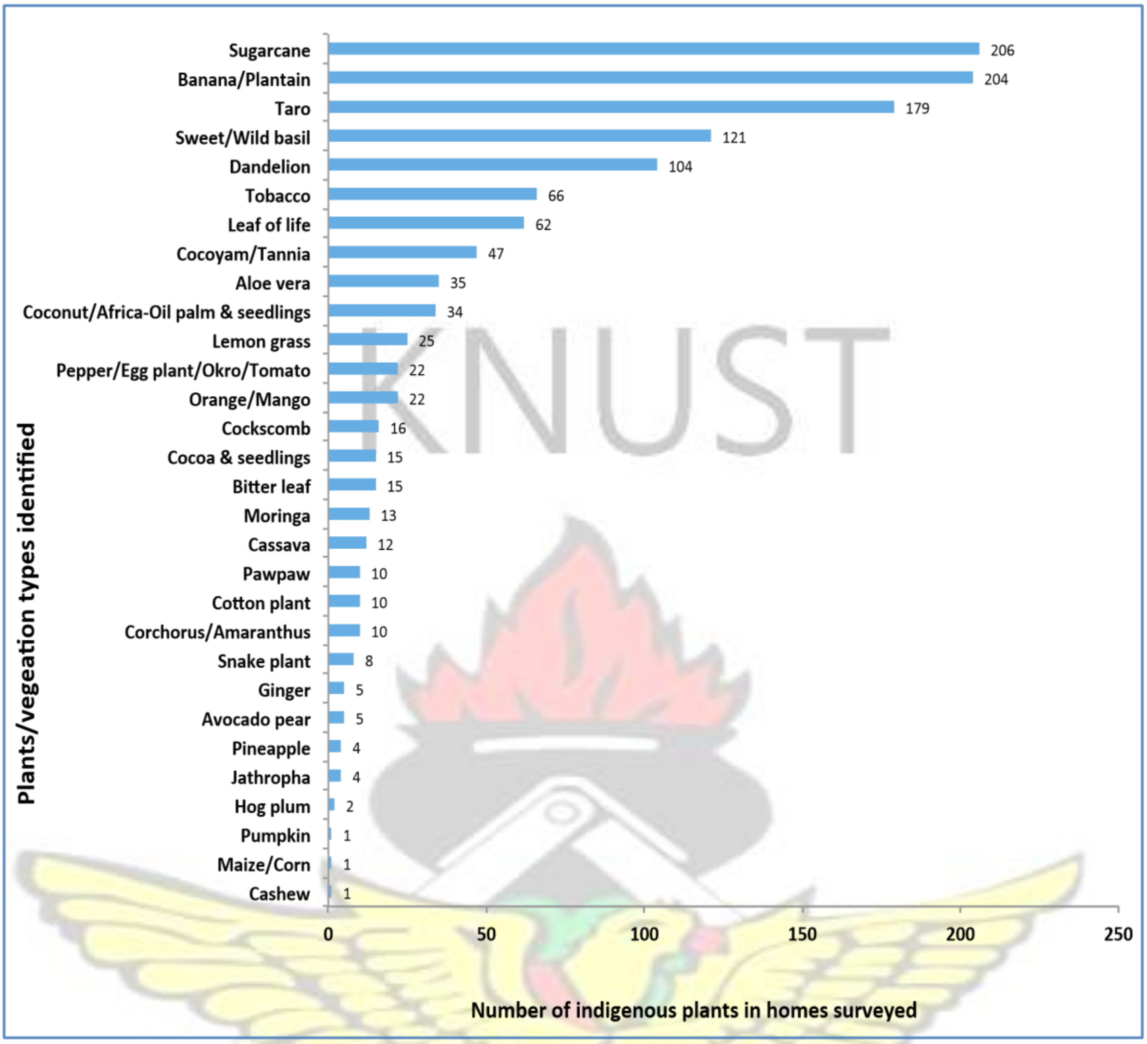

Figure 4.3 Indigenous plants used for "informal" greywater treatment

Table 4.11 Indigenous plants identified in the householder survey

	English common names	Scientific names	Local names/ how it is known
1	Aloe vera	*Aloe vera*	Aloe vera
2	Amaranthus[e]	*Amaranthus cruentus*	Alefu
3	Avocado	*Persea americana*	Paya
4	Banana/Plantain[b]	*Musa spp.*	Kwadu/Brodie
5	Bitter leaf	*Vernonia amygdalina*	Onyono/Bonyono
6	Cashew	*Anacardium occidentale*	Cashew
7	Cassava	*Manihot esculenta*	Bankye 8
	Chilli pepper[d]	*Capsicum spp.*	Mmako
9	Cockscomb	*Celosia spp.*	Akomfemtiko
10	Cocoa	*Theobroma cacao*	Kookoo
11	Coconut/Africa-Oil palm & seedlings[b]	*Cocos nucifera/Elaeis guineensis*	Kube/Abe
12	Cocoyam/Tannia	*Xanthosoma sagittifolium*	Mankeni/Menkeni
13	Cotton plant	*Gossypium hirsutum*	Asaawa dua
14	Corchorus (Jute leaves)[e]	*Corchorus olitorius*	Ayoyo
15	Dandelion	*Lactuca teraxacifolia*	Dandelion
16	Garden eggs/Wild egg plant or Turkey berries[d]	*Solanum spp.*	Nyadua/Abeduru or Kwahu nsusuaa

17	Ginger	*Zingiber officinale*	Akekaduro
18	Hog plum	*Spondias mombin*	Atoa/Atuaa
19	Jathropha	*Jatropha gossypiifolia*	Nkrandedua
20	Leaf of life	*Bryophyllum pinnatum*	Egoro
21	Lemon grass	*Cymbopogon spp.*	Fever aduro/esre
22	Maize/Corn	*Zea mays*	Aburoo
23	Mango[c]	*Mangifera indica*	Amango
24	Moringa	*Moringa oleifera*	Moringa
25	Okro[d]	*Abelmoschus esculentus*	Nnkruma
26	Orange[c]	*Citrus spp.*	Ankaa
27	Pawpaw	*Carica papaya*	Bofre
28	Pineapple	*Ananas comosus*	Abrobe
29	Pumpkin	*Cucurbita pepo*	Efre
30	Snake plant/Mother-in-law's tongue	*Sansevieria trifasciata*	Owo aduro/dua
31	Sugarcane	*Saccharum officinarum*	Ahwidie
32	Sweet basil[a]	*Ocimum basil/canum*	Akokomesa
33	Taro	*Colocasia esculenta*	Brobe/Kooko
34	Tobacco	*Nicotiana tabacum*	Bonto
35	Tomato[d]	*Solanum lycopersicum*	Nntoosi
36	Wild basil[a]	*Ocimum gratissimum*	Numnum

[a]Regrouped as sweet/wild basil.
[b]Already grouped. [c]Regrouped as orange/mango. [d]Regrouped as pepper/egg plant/okro/tomato.
[e]Regrouped as Corchorus/Amaranthus.

The top ten plants identified by the study are: sugarcane, banana/plantain, taro, sweet/wild basil, dandelion, tobacco, leaf of life, cocoyam/tannia, aloe vera, coconut/African oil palm, lemon grass, pepper/tomato/okro, and mango/orange (Figure 4.3). All the plants identified are used in Ghanaian communities for food and medicinal purposes, including the top ten plants listed. Our findings follow these plants uses, because the respondents reported that the main benefits they derive from indigenous plants were food (84%, n=379), and medicine (62%, n=281) (Figure 4.4). Thus, the results support the earlier statistical assertions that the plants are primarily grown for agronomic uses, particularly for food and medicine. This does not contradict the respondents' perception and understanding that plants help treat the greywater. However, it emphasises people's overarching interest in the basic advantages of water and nutrients supply from greywater to plants.

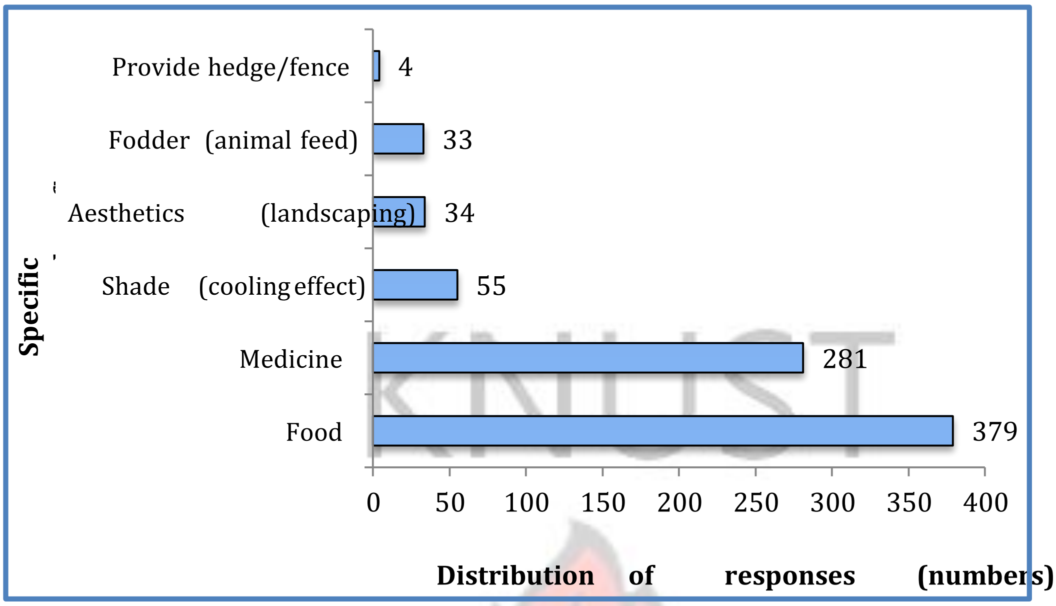

Figure 4.4 Specific benefits derived from plants used in greywater disposal

Apart from one house, all 450 households were motivated by one or more specific plants benefits, such as providing food, medicine, shade/shelter, aesthetics, fodder and hedge/fence (Figure 4.4). In a later part of the data collection, a limited number of respondents (50) were asked which of the plant parts were used for food and medicine. A small number were not using any plant parts (12%, *n*=6) whilst a sizable majority depended on the leaves (66%, *n*=29), fruits (52%, *n*=23), followed by roots and tubers (45%, *n*=20). Further interviews with the same 50 homes revealed a very low awareness of potential health risk associated with consuming plants watered with greywater (5/50, 10%). Although literature supports that greywater irrigated plants may pose some risks, including microbial and chemical loads (Benami et al., 2016; Cook, 2016), people are often not well informed. This is particularly so in Ghana where there is a dearth of knowledge on the level of risk that is associated with consumption of indigenous plants watered with greywater.

4.2.2.4 Potential for indigenous plants use in greywater treatment

The findings suggest that more can be done with the lesser known indigenous plants in the areas of green technologies. Exploring the plants with phytoremediation designs such as constructed wetlands and vegetated subsurface infiltration systems looks promising. Although this study did not examine the treatment efficacy of the existing informal treatment systems, there is strong perception from the people that their greywater disposal practices offer some agronomic benefits like food and medicine from plants. However, scientific proof about claims of treatment and also the safety of biomass consumption is required. Experts have identified that one of the gaps in vegetated filter bed technologies is finding ways to integrate value-added crops (Langergraber, 2015). Our current study suggests that we may not be too far from finding some value-added plants for such technology locally in Ghana. According to the IWA newsletter on Wetland Systems for Water Pollution Control, “a tremendous incentive for stewardship of constructed wetlands in developing countries will be created if we get a net positive cash flow off wetland crops” (Langergraber, 2015). We can say that a promising research area is emerging in Ghana, and probably in sub-Saharan Africa, where safe greywater management is a huge challenge.

Although the use of less known native plants in constructed wetlands is a grey area, we could explore monoculture systems and/or integrate these plants (sugarcane, taro, cocoyam, basil, dandelion, aloe vera, lemon grass etc.) with conventional constructed wetland plants such as reed grass (Phragmites), vetiver grass, and typha.

The planting practices identified from our survey are similar to the mixed vegetation system commonly found in natural wetlands in Ghana (Campion and Venzke, 2011; Campion and Owusu-Boateng, 2013). The potential for integrated, value-added indigenous constructed wetlands for greywater treatment is high, because they are a natural progression of existing practices amongst many of the households.

The study also revealed that very few homes (8%, n=37) have challenges with the existing practices. However, some complained of plants harbouring reptiles and other unwanted animals (36%, n=13), and that plants were badly affected by dry seasons (61%, n=22). Solutions to these challenges are straightforward and include plant husbandry and a more uniform distribution of the greywater.

It is encouraging that a substantial number of respondent households (44%, n=199) expressed interest in adopting technology changes that would improve upon their existing practices. The majority of these optimists (n=164) were willing to pay for improvements to their informal treatment systems. Eighteen people could not give an explicit amount, but indicated a willingness to “buy at any price”. However, the majority were willing to pay an extra GHS 5–500 with a mean of GHS 66 (around USD 15) suggesting this figure (or the median value of GHS 50, i.e. about USD 12) defines the acceptable upper price point of new technologies). While the lower quartile (25th percentile) was willing to pay GHS 20, the upper quartile (75th percentile) was ready with GHS 100 per a treatment system. Hence, improvements and/or new technology must be both affordable and socially acceptable. Examples of these technologies could include constructed wetlands, bioretention, bioinfiltration, biofilter, rain gardens and cells in homes, communities, towns and urban neighbourhoods, by adopting similar systems used in Singapore for treating and discharging surface runoff from drainage areas like parks, roadside, planting verges, civic squares etc. (Hunt et al., 2015).

4.3 Characteristics of greywater from the experimental site[3]

4.3.1 Profile of study households

The profile data and some basic information about greywater disposal, use of cleansing agents and others are presented in Table 4.12. All five households had their adult female heads, between the ages of 37 and 47 years, available for the interview. This was encouraging to support reliable and valid responses representative of households. The average household size of 4.8 quite reflects the city and regional census figures (GSS, 2013a), and also a similar study in the city (Dwumfour-Asare et al., 2017). The average age of young persons was 7 years old, including a 3-monthold baby. Family membership with younger children, especially under 3 years, is relevant because of potential influence on greywater characteristics through contamination from washing bodies and diapers (Katukiza et al., 2015).

All households received water supply from the main urban water utility provider, Ghana Water Company Limited (GWCL). The households were not metered but billed on a flat rate of GHS 25 (US$ 5.2) per month (Table 4.12), which was around GHS 6 (US$ 1.2) per capita/month. No definite reason was given for non-metered water supply to these bungalows, however, it is the university's internal arrangement with

[3] This Chapter has been submitted as a paper for publication with details as: **Dwumfour-Asare, B.**, Nyarko, K. B., Awuah, E. & Essandoh, H. M. K. (2018). Residential greywater flows and pollutant loads: a neighbourhood study within a university campus in Ghana (*Manuscript submitted for publication*).

its residential installations and not necessarily GWCL. GWCL bills the university with it's bulk water supply. Water consumption levels were unknown and also could not be estimated because water was accessed through in-plumbing service level (in-house connection). Households had no limitations on water usage and could consume as much water quantity as possible. Notwithstanding, a couple of families with smaller household sizes complained of unfairness inherent in the fixed rate which they perceived to be against smaller families.

All households disposed their greywater through sub drains into neighbourhood drains. Apart from disposal through point sources for kitchen, bathroom and toilet sinks, other non-point and irregular sources were identified. These non-point sources included flows generated from scrubbing and cleansing corridors, verandas, paved yards and house entrances. These greywater sources were drained into the neighbourhood drain through small open channels created in the paved floors. According to respondents, greywater from laundry are disposed of in varied ways – via bathroom discharge pipes; in few instances (ad hoc) used for flushing toilet during periods of extended water supply shortage; and also poured directly into drains outside the house. This meant that greywater from the point sources and part from laundry sources may effectively be quantified. However, households agreed to consciously discharge their laundry greywater into their bathroom and/or inspection chambers (installed along greywater sewer lines) provided near their houses.

Moreover, 3 of 5 households claimed they reused greywater for flushing toilets (Table 4.12), and this only happens on rare occasions of protracted shortages in water supply from GWCL. Thus, greywater reuse is rare because water supply is reliable and inhabitants do not see the need to practice reuse. Reuse of greywater becomes an alternative water source for flushing toilets when water supply is entirely not available.

Only one household had a member that used a special medication for the treatment of an undisclosed skin disease, otherwise no one used and/or disclosed anything further. The most used cleansing agents among households were bleach (powdered and liquid sodium hypochlorite), antiseptics, detergents and soap. The least used cleansing agents were odorizer or fragrance (for laundry), and shampoos (Table 4.12). Thus, chlorine, anionic surfactants, nutrients, cationic species, sulphates and other pollutants were expected be found in the greywater generated by inhabitants.

Table 4.12 Profile data gathered from survey with participating households

Parameters	Items measured	Values or Distribution (N=5)
Sex of respondents	Female	5 of 5
Respondent age (years)	Average (SDM)	42.2(3.96)
	Maximum	47
	Minimum	37
Household size (number)	Average (SDM)	4.8(1.3)
	Maximum	6
	Minimum	3
Youngest household member age, (years)	Average (SDM)	7.3(4.88)
	Maximum	13
	Minimum	0.25[a]
Oldest household member, age (years)	Average (SDM)	52.2(9.15)
	Maximum	67
	Minimum	43
Water source	GWCL pipe supply	5 of 5
Water consumption	Litres/day	Unknown
Amount paid for water	GHS/month	25
Toilet facility used	Water closet with septic tank	5 of 5
Greywater disposal	Discharged into drains	5 of 5
Greywater reuse (before)	Yes	3 of 5
	No	2 of 5
Specific greywater use	Flushing toilet	3 of 3
Use of any special medication	For a skin disease	1 of 5
Household cleansing & care products agents mostly used	Bleach (powder/liquid)	5 of 5
	Detergents (powdered soap)	4 of 5
	Odorizer/Fragrance	1 of 5
	Antiseptics	5 of 5
	Hair conditioners/shampoos	1 of 5

Note: SDM = standard deviation of the mean; [a]a three (3) months old baby; GWCL = Ghana Water Company Ltd; GHS = Ghana cedis (currency)

4.3.2 Greywater flow patterns and quantity generation

The greywater generation flow patterns monitored for 15hr a day for two weeks are illustrated in Figure 4.5. The general flow pattern reflects wastewater flows known in literature with characteristic peak and base flows. There were two main peak times that occurred between 7 and 9am in the mornings, and 6 and 8pm in the evenings (Tchobanoglous et al., 2004). However, there was also a third peak period in the early afternoon (12 – 2pm), which was quite conspicuous in the average flow pattern graph but appeared spike-like in daily flow patterns (Figure 4.5). The maximum peak hour flows were around 250 l/hr and 230 l/hr recorded for weeks 1 and 2 respectively. The least base flow was as low as l l/hr. Meanwhile the average hourly flows during the night (for 9hrs, 8pm – 5am) were between 14 and 48 l/hr.

Data on greywater generation is summarized in Table 4.13. The average daily generation rates were determined using both arithmetic and geometric means. The results for the separate weeks are comparable, for instance, the daily rates were around 1,270 and 1,300 l/d. The similarity was confirmed by the student t-test statistics which showed that there was no statistically significant difference ($p>0.05$) between mean generation rates for week 1 & 2 (Table 4.13). On the average, every household was generating between 250 and 260 litres per household/day (l/h/d) and that translated into 52 – 54 l/c/d. This per capita generation rate is higher than findings from a similar study in this same city, and also the quantity typically ascribed to the West African region (Table 4.14). However, our findings showed higher generation rate by a magnitude of 22 l/c/d over rates recorded for participants that accessed water supply outside their houses in a similar study (Oteng-Peprah et al., 2018b). In contrast, our results fell low by 19 – 46 l/c/d to the generation rates among counterpart participants with same water service level (house connection), previous reference, and another similar study from Accra (Mohammed et al., 2015) (Table 4.14). The disparity may be partly because of socioeconomic and cultural differences (Boyjoo et al., 2013) and/or that some of our participants possibly unconsciously missed adding their laundry greywater to the sewers provided. Moreover, greywater quantity was at the level of water scarce countries like Jordan but far lower than counterpart developing nations like Nepal and Vietnam (Table 4.14). Generally, Ghanaian greywater flows appears to confirm the assertion that per capita generation in developing countries rarely exceeds 100 l/c/d (WHO, 2006b).

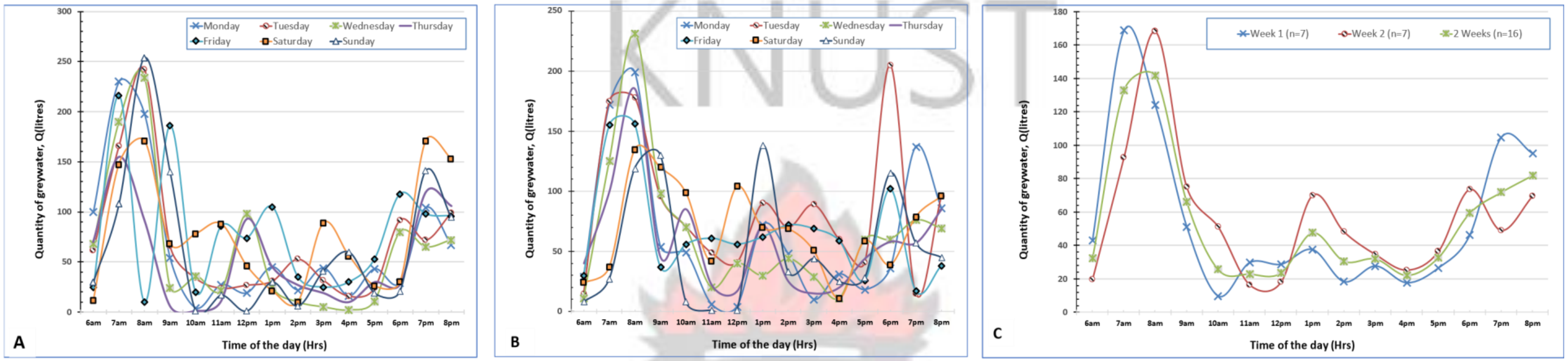

Figure 4.5 Greywater flow patterns: 15-hour flows for 1 - 2 weeks

Note: A) 1st week hourly flows; B) 2nd week hourly flows; C) 2 weeks average hourly flows

Table 4.13 Summary statistics of greywater generation rates

Greywater generation rates t-Test	Week 1 (n=7 days)				Week 2 (n=7 days)				
	x̄ a±SDMa	x̄ g±SDMg	Maximum	Minimum	x̄ a±SDMa	x̄ g±SDMg	Maximum	Minimum	p-value
Daily flows (l/d)	1,302.14±139.78	1,295.90±1.11	1,523.00	1,165.00	1,271.29±182.96	1,259.35±1.16	1,538.00	951	
Daily flow per household (l/h/d)	260.43±27.96	259.18±1.11	304.60	233.00	254.26±36.59	251.87±1.16	307.60	190.20	0.99
Daily flow per person (l/c/d)	54.26±5.82	54.00±1.11	63.46	48.54	52.97±7.62	52.47±1.16	64.08	39.63	

Note: x̄ a = arithmetic mean; x̄ g = geometric mean; SDMa = standard deviation of arithmetic mean; SDMg = standard deviation of geometric mean; na = not applicable

Table 4.14 Greywater generation rates - Ghana versus global

Study settings/site	Per capita (l/c/d)	**References**
Peri-urban, Central region of Ghana	32[a], 73[b]	(Oteng-Peprah et al., 2018b)
Urban, Kumasi of Ghana	36 – 43	(Dwumfour-Asare et al., 2017)
Urban, Accra of Ghana	98 – 100	(Mohammed et al., 2015)
West Africa	18 – 25	(Hyde and Maradza, 2013)
International	14 – 59 (Jordan), 72 (Nepal), 80-110 (Vietnam)	(Al-Mughalles et al., 2012; Oteng-Peprah et al., 2018b)
	200 (USA), 35–274 (Europe)[c], 72–225 (Asia)	(Al-Mughalles et al., 2012; Boyjoo et al., 2013)
	83 – 98 (Greece); 49 – 108 (Germany)	(Noutsopoulos et al., 2018; Sievers and Londong, 2018)

Note: a = water source is outside the house; b= water source is house connection; c = low European generation because of water savings mentality & installations.

4.3.3 Quality of greywater discharged into the environment

The quality of greywater discharged from households into the environment is presented in Table 4.15. The t-test statistics showed no significance difference existed ($p>0.05$) between the means of almost all greywater quality parameters (27 of 30) for week 1 and 2 (Table 4.15). The three parameters that differed in mean concentrations across the two weeks ($p<0.05$) were anionic surfactants, potassium, and mercury. Thus, greywater quality generally did not differ by the different sampling periods. Also, comparison of contaminant levels in the greywater with guidelines of EPA-Gh (for wastewater discharge limits in Ghana) and WHO/FAO (for non-restrictive wastewater reuse in irrigation) showed that about half of parameters, 11 of 21, exceeded the available guideline limits. The findings corroborate results in the previous sub-chapter 4.1 on greywater characteristics which demonstrated that untreated greywater in Ghana typically do not meet discharge limits.

The physical parameters that met both EPA-Gh and WHO/FAO guidelines were pH, EC, DO, and TDS but TSS failed (Table 4.15). The level of total chlorine was acceptable for discharge according to EPA-Gh but not safe for irrigation by WHO/FAO standard. The levels of main organic pollutants BOD_5 and COD were high and unsafe for the environment per EPA-Gh guideline (50 mg-BOD_5/L and 250 mg-COD/L). However, the biodegradability ratios (BOD_5:COD) range between 0.34 – 0.61 (Table 4.15), with mean values around 0.5, which matched the threshold for biodegradable organics (BOD_5:COD $\geq$ 0.50)(Kulabako et al., 2011). Untreated greywater in Ghana may have high organic contaminants dominated by COD but could still show potential for biodegradation or biological treatment as was demonstrated the previous discussions (under section 4.1).

Again, the greywater failed EPA-Gh discharge limit for nutrients like phosphorus (2 mg-P/L), ammonia (1 mg-NH_3-N/L) and ammonium (1.5 mg-NH_4^+-N/L) but not nitrate (11.5 mg-NO_3^--N/L) (Table 4.15). The nitrogen and phosphorus sources could be attributed to the use of soaps, detergents, and surfactants that are associated with household activities related to greywater generation (Li et al., 2008; Mohamed et al., 2019).

Already indicated, anionic surfactants and sulphates were expected in the greywater per the list of household products consumed (Table 4.12). The anionic surfactant levels were between 6 and 8 mg-LAS/L while the sulphates levels hovered between 13 and 15 mg/L. Sulphate levels were far below the EPA-Gh discharge limit which is quite high, 250 – 300 mg-SO_4^{2-}/L. The EPA-Gh has no discharge limits for anionic surfactants, a micropollutant commonly found in detergents (De Gisi et al., 2016) but their levels fell on the lower side of other findings (7 – 436 mg/L) in a European setting (Noutsopoulos et al., 2018). Possibly cleansing and personal care products used by our participating households did not contain much sulphate and anionic surfactants, and/or less of such products were used. It is also possible that probable laundry greywater diversions, already indicated, could affect the sulphate and anionic surfactants levels.

The EPA-Gh guideline defines discharge limits for the elemental species Cd, Pb, Hg, and Zn and same for WHO/FAO in addition to Na and Fe (Table 4.15). The Pb and Hg levels failed their discharge limits (0.1 and 0.005 mg/L respectively) while only Fe levels failed the WHO/FAO non-restrictive reuse limit (<0.1 mg-Fe/L). The reasons for high levels of Pb and Hg could be attributed to domestic commodities and building materials including plumbing materials, toys, cutlery, jewellery, coins, home maintenance products, arts and craft products, dental fillings etc (Eriksson and Donner, 2009).

No guideline limits existed for Ca and Mg but concern for their levels could be implied through the SAR values. With SAR of 0 – 3 $(meq/L)^{1/2}$, it can be implied that the levels of Ca, Mg and Na are safe, especially for the soil environment. SAR value for the greywater was 0.6, a clear indication that the levels of these elements were safe (Table 4.15).

Table 4.15 Statistical analysis of the physicochemical properties of the greywater

TDS (mg/L)	243.91	80.30	382	180.1	209.80	32.69	274.9	172	0.4	1000[a], <450[b]
TSS (mg/L)	183.08	47.27	256	108	189.06	48.09	248	96	0.64	50[a]; <50[b]
Free Chlorine (mg-CL_2/L)	2.44	0.43	3.07	1.92	2.24	0.30	2.62	1.7	0.41	na
Total Chlorine (mg-Cl_2/L)	2.49	0.44	2.96	1.95	2.4	0.28	2.78	2.04	0.68	250[a]; <1[b]
BOD_5 (mg-O/L)	346.07	86.38	454.8	216.9	320.06	66.58	421.8	223.2	0.51	50[a]
COD (mg-O/L)	737.71	196.69	1008	477	687.00	164.10	973	480	0.61	250[a]
BOD_5:COD ratio	0.48	0.09	0.61	0.34	0.47	0.08	0.55	0.36	0.93	na
Total Phosphate - PO_4^{3-} (mg/L)	9.83	5.16	18.4	4.1	7.56	1.05	8.8	6.3	0.28	na
Total Phosphorus - PO_4-P (mg/L)	3.11	1.73	6	1.3	2.47	0.35	2.9	2.1	0.36	2[a]
Nitrate (mg-N/L)	8.43	4.42	12.9	0.5	4.26	1.28	5.6	1.8	0.06	11.5[a]
Nitrite (mg-N/L)	0.03	0.02	0.05	0.01	0.04	0.02	0.06	0	0.52	na
Ammonia (mg-N/L)	9.32	4.46	14.99	4.5	6.43	2.35	10.4	4	0.12	1[a]
Ammonium (mg-N/L)	12.04	5.75	19.31	5.8	7.36	2.00	9.95	5.2	0.07	1.5[a]
Anionic surfactant (mg-LAS/L)	6.34	1.51	8.52	4.3	7.74	1.63	9.57	5.6	0.02	na
Sulphate - SO_4^{2-} (mg/L)	14.71	5.68	21	4	12.86	10.32	29	1	0.76	250-300[a]
Calcium (mg-Ca/L)	19.01	1.71	20.84	16.03	20.96	3.83	28.86	17.64	0.15	na
Magnesium (mg-Mg/L)	3.75	2.77	9.72	1.95	3.54	1.45	5.83	1.46	0.82	na
Potassium (mg-K/L)	1.47	0.28	2.01	1.15	1.14	0.14	1.33	0.94	0.03	na
Sodium (mg-Na/L)	11.72	3.73	19.41	8.87	10.46	2.36	14.47	7.31	0.43	<69[b]
Cadmium (mg-Cd/L)	0.06	0.03	0.1	0.04	0.09	0.01	0.1	0.08	0.08	<0.1[a]; 0.01[b]
Dissolved Iron (mg-Fe/L)	0.32	0.16	0.54	0.06	0.48	0.19	0.66	0.16	0.13	na
Total Iron (mg-Fe/L)	1.19	0.49	1.89	0.63	1.32	0.57	2.34	0.6	0.73	<0.1[b]
Lead (mg-Pb/L)	0.61	0.24	0.98	0.39	0.77	0.08	0.89	0.64	0.23	0.1[a]; 5.0[b]
Mercury (mg-Hg/L)	0.29	0.09	0.42	0.17	0.36	0.06	0.43	0.29	0.04	0.005[a]; 5.0[b]
Zinc (mg-Zn/L)	0.29	0.14	0.57	0.19	0.31	0.03	0.38	0.26	0.71	5.0[a]; 2.0[b]
SAR $(meq/l)^{1/2}$	0.64	0.21	1.09	0.48	0.55	0.10	0.65	0.4	0.29	0-3[b]

Parameters	Week 1 (N=7)				Week 2 (N=7)				t-test (p-value)	Guidelines
	Mean (x)	SDM	Maximum	Minimum	Mean (x)	SDM	Maximum	Minimum		EPA-Gh[a] & WHO/FAO[b]
										6-9[a]; 6.5-8.5
EC (µS/cm)	401.26	103.93	616.2	320	381.09	74.49	514.5	280	0.68	750[a]; <700[b]
Dissolved oxygen (mg-O/L)	1.09	0.71	1.84	0.1	1.20	2.44	6.69	0.1	0.93	1[a]
Turbidity (NTU)	270.14_	73.23	376	173	270.57_	42.09	346	212	0.99	75[a]
pH	6.48	0.13	6.73	6.36	6.45	0.23	6.73	6.1	0.81	[b]

Note: LAS = Linear Alkylbenzene sulfonate; SAR = sodium absorption ratio; a = Ghana's EPA discharge limits (Owusu-Ansah et al., 2015; Oteng-Peprah et al., 2018b; EPA-Gh, n.d.); b = World Health Organization/Food and Agriculture Organization's non-restrictive wastewater reuse limits (FAO, 1992; WHO, 2006a); na = not available

4.3.3.1 Specific pollutant loads and population equivalent

The daily discharge loads of key contaminants based on their concentration levels and the corresponding daily greywater generation are presented in Table 4.16. The loads varied from near zero for Cd to about 38 g/c/d for COD. Comparison of main pollutants' loads with local and international studies showed mixed trends. Comparison with similar studies shows comparable results for the discharge loads of COD and total phosphorus (TP) in Ghana, and that of COD, TSS, NH_4^+ and TP in international settings (Table 4.16). However, BOD_5 discharge load (17.6 g/c/d) was more than the value reported in a similar Ghanaian study (8 – 15 g/c/d) but less than the international figures (20 – 50 g/c/d) (Table 4.16). The disparities could be attributed to difference in socioeconomic and cultural settings including lifestyle patterns (Mohamed et al., 2013).

Based on the daily discharge load, population equivalent of the organic load was determined as about $8PE_{BOD}$. PE is the amount of oxygen-demanding substances whose oxygen consumption during biodegradation equals the average oxygen demand of the wastewater produced by one person (UN, 1997; OECD, 2007). A more recent standard measure is $1PE_{BOD}$ equals 54 g-BOD_5/d which is equivalent to
0.18 m^3/d (OECD, 2007; Henze and Comeau, 2008).

Table 4.16 Specific pollutant discharged loads in this study versus literature

Specific pollutants	Current study			Studies in literature	
	Daily load (g/d)	Household load (g/h/d)	Per capita (g/c/d)	[a]Ghana (g/c/d)	International (g/c/d)
TSS	236.61	47.32	9.86	17 – 20	10 – 30[a]
TDS	288.43	57.69	12.02		
BOD_5	422.83	84.57	17.62	8 – 15	20 – 50[a]; 28[b]
COD	906.88	181.38	37.79	24 – 48	18 – 46[a]; 49.3[b]
Ammonia	10.09	2.02	0.42		1 (TN)[b]
Ammonium	12.43	2.49	0.52	1 – 15	0.5[a]
Nitrate	8.00	1.60	0.33		
Nitrite	0.05	0.01	0		
Total Phosphate – PO_{43-}	10.94	2.19	0.46		
Total Phosphorus - PO_4-P	3.52	0.70	0.15	0.1 – 0.2	0.2 – 6[a]; 0.38[b]
Sulphate	17.01	3.40	0.71		
Anionic surfactant (LAS)	8.97	1.79	0.37		

Calcium	25.24	5.05	1.05
Magnesium	4.50	0.90	0.19
Potassium	1.67	0.33	0.07
Sodium	14.17	2.83	0.59
Cadmium	0.09	0.02	0
Iron	1.59	0.32	0.07
Lead	0.88	0.18	0.04
Mercury	0.41	0.08	0.02
Zinc	0.39	0.08	0.02

Note: g/d (gram/day); g/h/d (gram/household/day); g/c/d (gram/capita/day); TN (total nitrogen); [a](Oteng-Peprah et al., 2018b); [b](Sievers and Londong, 2018)

4.3.4 Implications of untreated greywater in the environment

The nature of greywater in this study is high strength because concentration levels for contaminants of concern were high loads of BOD_5 >300 mg/L, COD >630 mg/L, TP >2 mg/L and TN >17 mg/L as discussed under section 4.1. For TN, the sum of nitrate, nitrite, ammonia and ammonium concentrations was used as a proxy measure. While quite a number of EPA-Gh's contaminant discharge limits could not be met, the greywater also contained other contaminants of environmental concerns like anionic surfactants and some heavy metals including Cd, Pb and Hg. The state of greywater was not safe for disposal into the environment without treatment.

Meanwhile, the main outfall drains of greywater from the study neighbourhood including other places within Hall 6 residential area open into River Wiwi and its natural wetlands on campus. The river is therefore expected to receive these identified pollutant loads. The findings corroborate with other studies that KNUST community contributes to the pollution of River Wiwi (Anning et al., 2013; Amisah and Nuamah, 2014). At the same time, River Wiwi serves as irrigation water source for some urban vegetable farmers in Kumasi (Agodzo et al., 2003; Obuobie et al., 2006). Therefore, potential threat exists from polluted greywater to public health and safety is inevitable especially through urban vegetable food chain, and other means of exposure with river water uses downstream (Agodzo et al., 2003; Obuobie et al., 2006).

It is imperative to initiate responsive efforts to protect the environment against pollutants from greywater in Ghana. Key proposals may include:

- exploring low-cost onsite treatment and management options (including individual and neighbourhood cluster models) that may take advantage of the potential biodegradability inherent in Ghanaian greywater; and
- effective enforcement of local regulations on installation of onsite greywater containment, treatment and/or safe disposal especially in many places without any sewerage coverage.

These proposals are feasible because already, research and development is one of seven thematic policy areas of Ghana's Environmental Sanitation Policy (2010) as articulated by Strategic Youth Network for Development (SYND, 2013). Also, local regulations including local authority bye-laws and national building regulations underscore the obligations of occupiers (landlords and/or tenants) to safely manage their greywater (Fosu, 1996; Ghana Local Government, 1998).

4.4 Performance of experimental scale constructed wetland for greywater treatment in Ghana[4]

4.4.1 Growth and development of wetland vegetation

The growth and development of plants used in the wetlands was generally very encouraging. Plants were first planted on wetland beds in October 2017, and were well tendered and cared for including replacement of dead ones until March 1, 2018. By August 2018, the mid period of performance monitoring (June to October 2018), plants' heights, number of stems/shoots, leaves and planting density were determined (Table 4.17). All wetlands except TT1 achieved the same planting density around 35 plants per m^2 because of their equal number of total vegetation (20 plants) per bed. Generally, sugarcane (*Saccharum officinarum*) dominated in the number of leaves and height, and taro (*Colocasia esculenta*) also dominated in the number of shoots/plants especially in mixed planted beds. The average taro fell short of the maximum possible height of 1.5 m (Hall, 2017; Rana and Maiti, 2018) but some individual plant got closer by achieving heights around 1.1 – 1.3 m (Table 4.17) and the difference could be due to younger age and species difference. Though plant roots at late stages could not be observed, but in the early stages of development, taro was the most promising in root

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establishment forming long mesh-like network complexes with bed media. Also, taro was fast-growing and flourished more in colonizing the wetlands than sugarcane including mixed vegetation beds. This observation could be attributed to taro's invasive behaviour in all environments (Dana et al., 2017). In addition, taro was always the greener of the two plants in all beds.

The comparatively unimpressive growth of sugarcane could be linked to potential iron toxicity effects, potential allelochemicals released from taro especially in wetland beds with mixed vegetation, and autotoxins from sugarcane cuttings and residues (USEPA, 2003; Chou, 2010; Rout and Sahoo, 2015).

Table 4.17 shows brief descriptive characteristics of wetlands' vegetation during the time of operations and monitoring for performance.

Table 4.17 Characteristics of wetland plants around operational period

Wetlands	Plants dimension								
	Plant type	Plants	No. of leaves			Height of plants (mm)			Planting density
		Number	Avg	max	min	Avg	max	min	Number of plants per m^2
TT1 (mixed)	Sugarcane	11	5.5	11.0	2.0	37.9	100.0	16.0	33.5
	Taro	8	3.4	6.0	2.0	71.0	108.0	15.0	
TT2 (single)	Taro	20	3.2	6.0	1.0	64.3	127.0	4.0	35.3
TT3 (single)	Sugarcane	20	9.2	21.0	4.0	84.5	132.0	10.0	35.3
TT4 (mixed)	Sugarcane	8	8.8	15.0	2.0	71.8	120.0	30.0	35.3
	Taro	12	2.8	13.0	1.0	42.4	106.0	10.0	
TT5 (mixed)	Sugarcane	8	7.5	17.0	4.0	82.1	200.0	22.0	35.3
	Taro	12	2.1	5.0	1.0	32.1	99.0	5.0	
TT6 (mixed)	Sugarcane	10	6.5	14.0	2.0	50.4	100.0	14.0	35.3
	Taro	10	3.6	5.0	1.0	76.1	106.0	30.0	

Note: avg=average, max=maximum, min=minimum

4.4.2 Sedimentation pre-treatment of greywater

Table 4.18 shows contaminant levels in greywater after sedimentation and screening as the main pre-treatment operation. Contaminant levels mostly dropped between 18 and 75% for turbidity, TSS, BOD_5, COD, Ortho-P & P, NO_3-N, NO_2-N, SO_4^{-2}, AnS and Fe. Most of these reductions were statistically significant at 5% confidence level, making the pre-treatment stage relevant to the entire treatment process (OtengPeprah et al., 2018a). However, the three EC, TDS and NH_3-N saw increased levels between 9 and 61%. The effect of pre-treatment sedimentation stage was more pronounced on BOD_5 and COD in magnitude between two and three times high when compared to a similar study in India (Shegokar et al., 2015). The BOD_5 reduction was far lower than the 40% expected from sedimentation under one or more days retention time (Cooper and Findlater, 1990). The sedimentation pretreatment tended to improve slightly the BOD_5:COD biodegradability ratio from 0.60 to 0.66 on the average, higher than the reference level of 0.5 (Boyjoo et al., 2013). The average pH of pre-treated greywater was slightly lower and acidic (pH = 6.63) than the raw greywater (pH = 6.96) (Table 4.18). The slightly acidic conditions could be due to biochemical reactions (including aerobic and anaerobic metabolic processes) including nitrification and H^+ extrusion from contaminant decomposition (Bezbaruah and Zhang, 2004).

Nutrients - phosphorus (P) and nitrogen (N) showed significant reductions except the ammonia component which increased significantly (61%, $p<0.05$) (Table 4.18). Thus, more organic nitrogen may have been converted to ammonia rather than nitrification more likely due to low DO, and nitrates could be reduced via loss in gaseous nitrogen forms by denitrification (Ling et al., 2009). Also, pre-treatment sedimentation could reduce sulphate (SO_4) and anionic surfactants (63% versus 20%) but the removal efficiency was not significant for the latter (Table 4.18). The low reduction of AnS is comparable to that of phosphorus (18 – 21%), although LAS is known to be more biodegradable (Scott and Jones, 2000). The iron levels also dropped significantly likely because of conversion from soluble to insoluble and settleable forms after exposure to DO. The high reduction in DO level (78%) is not desirable because depletion of oxygen in influent greywater below 4 mg-DO/L is not conducive for biochemical processes according to National Academies of Sciences, Engineering, and Medicine (NASEM, 2015), especially in subsurface flow wetlands. However, reduced DO levels were expected due to oxidation and degradation processes which consume available oxygen in the wastewater (Mohamed et al., 2019).

When raw and pre-treated greywater were assessed with available discharge and reuse guidelines, most of the major contaminants (turbidity, TSS, BOD_5, COD, P, and NH_3-

N) failed to meet the limits (Table 4.18). For DO levels, the raw wastewater was very satisfactory (>3 mg/L) but failed after pre-treatment (<1 mg/L). The pH, EC, TDS, sulphate and iron levels in both raw and pre-treated greywater were safe for disposal and reuse for irrigation except for DO that failed after pre-treatment. Although no guideline limits exist for AnS, the levels were high (5.10±1.91 mgLAS/L) compared to most effluent wastewater (0.09 – 0.9 mg-LAS/L) (Scott and Jones, 2000), and also unsafe for most aquatic and soil organisms that may be sensitive to exposure levels around <5 mg-LAS/L (EOSCA, 2000; Abd El-Gawad, 2014). Thus, greywater from the neighbourhood still required further treatment after sedimentation pre-treatment before it could be safe for disposal and/or any reuse like irrigation.

Table 4.18 Contaminant removal after sedimentation pre-treatment

Contaminants (units)	Raw greywater (Sampled from storage tank 1), N=16				Pre-treatment (Sedimentation), N=16					Guidelines
	mean	**Std**	**maximum**	**minimum**	**mean**	**Std**	**maximum**	**minimum**	**% Rev**	**EPA-Gh[a] & WHO/FAO[b]**
pH	6.96	0.31	7.72	6.59	6.63	0.14	6.97	6.42	-	6-9[a]; 6.5-8.5[b]
EC (µS cm^{-1})	300.33	79.89	457.30	174.30	326.89	54.23	442.20	243.10	-9	750[a]; <700[b]
DO (mg L^{-1})	3.50	1.44	5.78	0.86	0.76	0.22	1.05	0.37	78*	1[a]
TDS (mg L^{-1})	149.58	39.91	227.80	86.38	163.03	26.76	220.20	122.20	-9	1000[a], <450[b]
Turbidity (NTU)	194.09	114.36	557.00	81.70	133.45	34.01	194.00	74.60	31	75[a]
TSS (mg L^{-1})	192.50	104.63	408.00	80.00	69.51	31.69	156.01	20.00	64*	50[a]; <50[b]
BOD_5 (mg-DO L^{-1})	292.81	112.55	547.00	100.00	215.25	59.96	328.00	82.00	26*	50
COD (mg-DO L^{-1})	490.31	170.22	987.00	290.00	326.63	87.24	505.00	152.00	33*	250[a]
Ortho Phosphate (mg-PO_4 L^{-1})	14.52	4.16	22.50	7.90	11.85	3.83	17.40	5.10	18	na
Phosphorus (mg-PO_4-P L^{-1})	4.74	1.34	7.31	2.60	3.73	1.44	5.65	0.88	21*	2[a]
Nitrate (mg-NO_3-N L^{-1})	9.68	6.71	23.80	0.10	5.55	2.69	10.40	1.10	43*	11.5[a]
Nitrite (mg-NO_2-N L^{-1})	0.04	0.02	0.09	0.01	0.02	0.01	0.03	0.00	56*	na
Ammonia (mg-NH_3-N L^{-1})	3.45	2.68	10.60	0.20	5.56	2.74	11.40	0.75	-61*	1[a]
Sulphate (mg-SO_4^{2-} L^{-1})	13.06	7.26	30.00	2.00	4.81	2.51	9.00	1.00	63*	250-300[a]
Anionic surfactants (mg-LAS L^{-1})	6.41	2.83	9.82	1.56	5.10	1.91	8.70	1.49	20	na
Iron (mg-Fe L^{-1})	0.37	0.28	0.97	0.01	0.09	0.08	0.28	0.00	75*	<0.1[b]

Note: N= number of samples; Std = standard deviation of the mean; %Rev = % removal or reduction; -% Rev = negative indicates increase in contaminant level; *indicates T-test is statistically significant at $p<0.05$; [a]Ghana's EPA discharge limits (Owusu-Ansah et al., 2015; Oteng-Peprah et al., 2018b; EPA-Gh, n.d.); [b]World Health Organization/Food and Agriculture Organization's non-restrictive wastewater reuse limits (FAO, 1992; WHO, 2006a); na = not available

4.4.3 Performance of treatment systems

4.4.3.1 Performance of CW under a 1-day HRT

The performance of microcosm wetlands operated on 1 day HRT is shown in Table 4.19 and Table 4.20. The results showed that effluent EC, DO and TDS levels consistently increased by 6 – 129% and these were almost always statistically significant. The effluent qualities of these mentioned parameters were somehow better in the planted beds than the unplanted although the relative performance differences were not statistically significant in most cases (Table 4.20). The acid levels mostly dropped slightly from pre-treatment (pH = 6.63 on average) to 6.65 – 6.68, and DO levels increased by 62 – 129% for most planted systems. Drop in acid levels and DO increases could be attributed to interactions (biochemical and physical processes including metabolisms) that naturally occur in the wetland beds, involving plants and microbes (Hench et al., 2003; USEPA, 2003). The increased effluent TDS and EC levels could be due to salts and dissolved organics released from the bed media and biochemical activities in the wetland beds. Also, the two parameters increased at the same levels because of their potential linear relationship (Thirumalini and Joseph, 2009; Oteng-Peprah et al., 2018a).

The TSS removal efficiency ranged between 59 and 81%, and the least and highest rates were associated with TT1 and TT2 respectively (Table 4.19). Apart from TT2, none of the systems were able to meet the >80% removal efficiency usually expected of HFCW performance for TSS (Dotro et al., 2017). Most planted beds performed better than the unplanted counterparts with relative performance differences (RPDs) of 4 – 20% but not significant ($p>0.05$). Relative performance difference (RPD) is the difference in contaminant removal efficiencies between planted/treatment beds and their controls. Although RPDs are not analysed and/or presented in CW studies, it is a simple but effective way to elucidate real performance differences that may not necessarily be statistically significant (Jesus et al., 2017). In two instances, unplanted beds outperformed the planted ones (TT1 and TT5) with RPDs of -1% although not statistically significant as well (Table 4.20).

BOD_5 removal efficiencies were within 82 to 89% and also corresponded to the levels known for HFCW. Generally, all planted beds removed higher BOD_5 (≥85%) than the unplanted ones (82%) with RPDs between 3 and 6% but statistically insignificant (Table 4.20) The COD removals were somehow lower (69 – 84%) than BOD_5, and most systems including unplanted beds performed below 79%. The RPDs between unplanted and planted beds were within 3 – 15% with one significance difference ($p=0.025$) coming from TT6. Also, 1 day HRT could not achieve the expected >80% COD removal, probably due to higher fraction of not easily biodegradable organics commonly associated with greywater and shorter HRT (Dotro et al., 2017; Mohamed et al., 2019).

Almost all wetlands performed unsatisfactorily with NH_3-N removal, where two laterite-based beds (CT2 and TT1) increased effluent levels (6 – 8 mg/L) and the rest showed low removal efficiencies (<8%). The observation is known in literature (Carrasco-Acosta et al., 2019). Moreover, the planted laterite-based beds mostly performed better than their unplanted version with significant RPDs of 15 – 35% (Table 4.20). Phosphate (P) removal was low as well (29 – 34%) in gravel beds (CT1, TT4 and TT5) but quite high (69 – 86%) in the laterite-based beds. Gravels are conventional substrates noted for poor performance with respect to P removal unlike Fe rich media like laterite where Fe acts as natural capping agent by improving Pbinding capacity of substrates (Bakker et al., 2015; Yang et al., 2018). However, planted gravel betters outperformed its unplanted bed with RPD of 3 – 5% ($p>0.05$) unlike laterite-based beds with mixed performance.

Nitrate removal was also significantly high 82 – 96% for all systems, likewise nitrite removal with 100% performance except for gravel-based beds and TT6 which showed comparatively low efficiencies (50 – 63%) but not poor performance. In general, treatment systems were able to absorb nutrients (phosphorus, nitrate, ammonia and nitrite) via media, plants and microbial interactions, absorption, adsorption and uptake (Pradhan et al., 2018). While instances of ammonia increases may be due to releases from organic nitrogen (ammonification), reduction of nitrate and nitrite levels could also be attributed to nitrification and denitrification (Randerson, 2006; Ling et al., 2009). The nutrients (both N and P) removal appeared more pronounced for laterite-based beds probably because of the presence of more iron species which act as electron donor in redox reaction to be oxidized (Ge et al., 2018).

Reduction in effluent turbidity was always high and significant for all wetland beds (94 – 99%, Table 4.19). However, the laterite-gravel mixed (laterite-based) planted beds significantly outperformed their unplanted counterpart with RPDs of 2 – 4% ($p<0.05$). The improvement could be attributed to the media and presence of vegetation, likely the plants' roots forming interconnected mesh-like formation with bed media for effective filtration. Also, the microcosm wetlands significantly removed all sulphates (100%) from the effluent. Although, influent SO_4^{2-} levels were low and met Ghana's Environmental Protection Agency discharge limit, the treatment systems efficiently removed the contaminant. The bed media might have played a key role in this since controls and treatments completely removed the contaminant alike. The removal performance with AnS, a known organic xenobiotic and micropollutant of concern, was encouraging with reduction levels of 40 – 60%. All planted beds outperformed the unplanted ones (7 – 18% RPDs) with significant difference recorded for all laterite-based beds (11 – 18%, $p<0.05$). The reduction may be attributed to biodegradation (aerobic and anaerobic forms), biofiltration and adsorption (via biofilm formation, media, roots) (Konnecker et al., 2011; Ramprasad and Philip, 2016b; Yang et al., 2018).

The current results show higher AnS removal than reported in a study that used similar bed media (gravel and tezontle –laterite like media full of iron oxide) and even with their longer HRT (3 and 8 days) although different vegetation (PérezLópez et al., 2018). Meanwhile, the iron levels of effluent from all systems could not meet the guideline limit of <0.1 mg-Fe/L including the gravel beds (around 0.3 – 0.5 mg/L). High Fe releases came from laterite-based beds in excess of about 4 – 9 times more than the 1 mg/L safety level recommended for freshwater aquatic life (Vuori, 1995). While effluent Fe mainly came from wetland media especially the lateritebased beds, contributions from vegetation may not be ignored as also reported in literature (Vose, 1982; USEPA, 2003).

The oxygen transfer rate (OTR) observed in the wetland systems ranged between 5.6 and 23 g-O_2/m^2d (Table 4.19). There was no indication of transfer deficit (negative values) and almost all planted beds (5 of 6) attained the 20 g-O_2/m^2d reference level known for HSFW (Randerson, 2006). A planted bed TT1 and the two unplanted beds (CT1 & CT2) failed to meet the reference oxygen diffusion rate but the worst was CT2 (5.6 g-O_2/m^2d). Generally, the depths of the wetland beds do not appear to play a key role here since some planted gravel beds (28 cm) and laterite-based bed (30 cm) were able to attain the expected OTR. For the unplanted beds, low OTR is not too surprising since a key factor like vegetation is missing. However, it is unclear why the planted bed TT1 achieved the lowest OTR among the planted beds, may be the comparatively low planting density (Table 4.17) could be a contributing factor because the OTR was higher than its control (CT2) but not close to other planted beds. It is believed that lower diffusion rates are possible and such rates could still contribute significantly to BOD_5 removal but not ammonia by oxidation (Cooper and Findlater, 1990). Probably that could be part of the reason for increased NH_3-N levels instead of removal among the two beds - unplanted (CT2) and planted (TT1) laterite beds. Adequate oxygen transfer is necessary for ammonia stripping and conversion to nitrate although HSFWs generally achieve lower OTR (Randerson, 2006).

When the effluent quality of all wetland beds was compared with discharge and agriculture reuse guideline limits (Table 4.18), most of the parameters with available limits (10 of 13) were satisfactory – pH, EC, TDS, turbidity, TSS, BOD_5, COD, NO_3-N and SO_4. Also, all beds satisfied the limit for DO while gravel media beds satisfied the iron limit. Although removal of AnS was significant, effluent may still be toxic to some aquatic life sensitive to LAS at very low levels like ≤0.025 mgLAS/L concentrations (Abd El-Gawad, 2014).

Table 4.19 Contaminant removal efficiencies of CWs under 1 day HRT

Contaminants[a]	Wetland Treatment Cells															
	CT1		CT2		TT1		TT2		TT3		TT4		TT5		TT6	
	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev
pH (-)	6.75(0.05)	-	6.84(0.04)	-	6.65(0.04)	-	6.62(0.02)	-	6.69(0.03)	-	6.71(0.06)	-	6.72(0.10)	-	6.76(0.03)	-
EC (µS/cm)	321.44(15.97)	-13*	376.36(20.33)	-32*	377.88(27.49)	-33*	383.56(14.16)	-35*	377.94(14.01)	-33*	313.54(14.67)	-10	312.36(15.16)	-10	335.18(15.81)	-18*
DO	1.02(0.16)	-10	1.5(0.25)	-62*	1.28(0.23)	-38*	1.5(0.42)	-62*	1.55(0.21)	-67*	0.98(0.09)	-6	2.12(0.46)	-129*	1.76(0.41)	-90*
TDS	160.14(7.91)	-13*	187.32(10.35)	-32*	188.18(13.73)	-33*	191.02(7.1)	-35*	187.94(7.02)	-32*	155.78(7.48)	-10	155.64(7.6)	-10	166.82(7.85)	-18*
Turbidity (NTU)	5.43(1.87)	96*	5.89(2.01)	95*	2.73(0.87)	98*	2.85(0.87)	98*	2.44(0.76)	98*	3.74(0.78)	97*	6.79(3.29)	94*	1.4(0.25)	99*
TSS	20.8(7.69)	69*	26.4(20.71)	60*	27.2(16.59)	59*	12.8(5.22)	81*	18.41(8.3)	72*	18.4(15.13)	72*	21.6(11.87)	67*	22.41(8.29)	66*
BOD_5	36.4(9.69)	82*	35.4(16.36)	82*	24(10.84)	88*	29.6(12.56)	85*	27(12.35)	86*	26(13.29)	87*	30.2(20.49)	85*	22.8(9.04)	89*
COD	76(16.29)	73*	87.2(25.19)	69*	79.6(30.80)	72*	75.6(25.56)	74*	78.6(36.84)	72*	63.2(31.43)	78*	71(51.92)	75*	45.8(22.44)	84*
Ortho Phosphate	9.16(1.46)	29*	2.5(0.6)	81*	2.06(0.46)	84*	1.86(0.60)	86*	2.7(0.41)	79*	8.54(2.27)	34*	8.74(2.09)	32*	3.88(0.16)	70*
Phosphorus	2.98(0.49)	29*	0.82(0.20)	80*	0.7(0.14)	83*	0.62(0.16)	85*	0.9(0.16)	79*	2.78(0.74)	34*	2.84(0.68)	32*	1.28(0.04)	69*
Nitrate	0.3(0.14)	94*	0.23(0.15)	95*	0.85(0.25)	84*	0.25(0.21)	95*	0.20(0.17)	96*	0.7(0.10)	86*	0.34(0.29)	93*	0.92(0.18)	82*
Nitrite	0.01(<0.001)	50*	0	100*	0	100*	0	100*	0	100*	0.01(0.01)	63*	0.01(<0.001)	50*	0.01(<0.001)	50*
Ammonia	6.04(0.91)	1	7.77(0.64)	-28*	6.87(0.63)	-13	5.84(0.47)	4	5.96(0.60)	2	5.99(0.67)	1	5.72(0.54)	6	5.65(0.44)	7
Sulphate	0	100*	0	100*	0	100*	0	100*	0	100*	0	100*	0.33(0.18)	93*	0	100*
Anionic surf.	2.4(0.52)	47*	2.61(0.27)	42*	2.11(0.21)	53*	1.82(0.20)	60*	1.93(0.21)	57*	2.03(0.41)	55*	2.07(0.37)	54*	1.81(0.21)	60*
Iron	0.3(0.10)	-83*	6.6(0.67)	OTP*	8.02(0.81)	OTP*	9.92(0.84)	OTP*	8.34(0.44)	OTP*	0.49(0.25)	-201*	0.4(0.12)	-146*	4.10(0.33)	OTP*
OTR (g-O_2/m^2d)	18.55		5.60		13.67		20.85		20.22		20.08		21.69		22.99	

[a]All units are in mg/L unless otherwise stated; mean = average contaminant level; Std = standard deviation of the mean; %Rev = % removal or reduction; - % Rev = negative indicates increase in contaminant level; OTR = oxygen transfer rate; OTP = over 1000% increase in levels; *indicates T-test is statistically significant at $p<0.05$.

Table 4.20 Relative performance difference and T-test for CW – 1 day HRT

Parameters	Relative performance difference between planted and unplanted CW and the p-value for their T-Tests, RPD (p-value)					
	TT1	**TT2**	**TT3**	**TT4**	**TT5**	**TT6**
EC (µS/cm)	-1% (0.923)	-3% (0.534)	-1% (0.890)	3% (0.439)	3% (0.383)	14% (0.007)*
DO (mg/L)	24% (0.175)	0% (0.999)	-5% (0.760)	5% (0.602)	-118% (0.001)*	-28% (0.264)
TDS (mg/L)	-1% (0.914)	-3% (0.528)	0% (0.914)	3% (0.397)	3% (0.386)	14% (0.008)*
Turbidity (NTU)	3% (0.012)*	2% (0.015)*	3% (0.007)*	1% (0.099)	-1% (0.445)	4%(0.001)*
TSS (mg/L)	-1% (0.948)	20% (0.192)	12% (0.446)	4% (0.760)	-1% (0.902)	6% (0.699)
BOD_5 (mg/L)	6% (0.230)	3% (0.547)	4% (0.386)	5% (0.195)	3% (0.558)	6% (0.170)
COD (mg/L)	3% (0.681)	4% (0.490)	3% (0.678)	4% (0.422)	2% (0.842)	15% (0.025)*
Ortho Phosphate (mg/L)	3% (0.228)	5% (0.130)	-2% (0.552)	5% (0.622)	3% (0.723)	-11% (0.001)*
Phosphorus (mg-PO_4-P/L)	3% (0.313)	5% (0.127)	-2% (0.509)	5% (0.628)	3% (0.720)	-11% (0.001)*
Nitrate (mg-NO_3-N/L)	-12% (0.033)*	0% (0.720)	1% (0.854)	-8% (0.156)	-1% (0.185))	-13% (0.000)*
Nitrite (mg-NO_2-N/L)	0%$_a$	0%[a]	0%[a]	13% (0.141)	0% (0.347)	-50% (0.347)
Ammonia (mg-NH_3-N/L)	15% (0.055)	32% (0.001)*	30% (0.002)*	1% (0.926)	5% (0.512)	35% (0.000)*
Sulphate (mg-SO_4^{2-}/L)	0%[a]	0%$_a$	0%[a]	0%[a]	-7% (0.347)	0%[a]
Anionic surfactants (mg-LAS/L)	11% (0.012)*	17% (0.001)*	15% (0.002)*	8% (0.243)	7% (0.281)	18% (0.001)*

Note: [a] Indicates that effluent values were largely 0 mg/L as input data
*indicates T-test is statistically significant at $p<0.05$.

4.4.3.2 Performance of CW under a 2-day HRT

CWs performance after a 2-day HRT is presented in Table 4.21 and Table 4.22. The effluent levels of EC, DO and TDS followed similar patterns observed for 1 day HRT, except for CW TT5 that showed 1% reduction in EC and TDS levels instead of an increase. EC, DO and TDS largely increased in effluent levels by 2 – 133%. The significant increases were associated with DO (38 – 133%, $p<0.05$) especially in the planted beds (76 – 133%, $p<0.05$). Gravel planted beds outperformed their unplanted bed in removing effluent EC and TDS with RPDs of 6 – 10% but the differences are statistically insignificant (Table 4.22). The increased effluent TDS and EC levels could be due to salts and dissolved organics released from the bed media and biochemical activities in the wetland beds. The vegetation and porous media could contribute to the improved levels of DO through atmospheric re-oxygenation and macrophyte transmission of oxygen into the media and media-root matrix (Wang et al., 2018). The wetlands slightly reduced acidic levels of effluent to pH 6.63 – 6.83 from sedimentation pre-treatment average of pH 6.51. The wetlands removed 96 – 99% of turbidity from effluents, and the most turbid effluent was around 6 NTU in an unplanted bed, but there were marginal differences (RPDs: -1% to 2%) between planted and unplanted beds (Table 4.22). The media-root matrix may have played a key role in significantly reducing turbidity during treatment by physical mechanisms such as filtration, adsorption, and sedimentation (Kandhro, 2018; Pradhan et al., 2018). This was also observed from the wetland systems under the 1 day HRT.

Effluent TSS, BOD_5, and COD received significant removal efficiencies of 60 – 90% (Table 4.21). While wetland performance with TSS fell short of the expected >80% removal efficiencies, planted beds achieved the >80% BOD_5 removal efficiencies, and likewise 4 of 6 planted beds attained the >80% COD removal efficiencies (Dotro et al., 2017). Planted beds outperformed the unplanted with RPDs 5 – 18% for TSS ($p>0.05$), RPDs 3 – 13% for BOD_5 (but significant for gravel beds), and most RPDs 3 – 9% for COD ($p>0.05$) (see Table 4.22). Moreover, there is general improvement in COD removal efficiencies under 2 days over the 1 day HRT.

On nutrients, phosphorus removal was low with gravel beds (24 – 33%) unlike the laterite-based beds that showed significant performance of 67 – 85% (Table 4.21). The trend is similar to the findings from 1 day HRT, where only planted gravel beds performed better than the unplanted beds with RPDs between 6 and 9% although statistically insignificant (see Table 4.22). NO_3-N and NO_2-N removal efficiencies were also significant (64 – 100%) in most wetlands except for gravel beds which gave comparatively low nitrite removal efficiencies of 29% probably due to low levels microbial nitrification–denitrification and minor incorporation into plant biomass (Stottmeister et al., 2003).

However, ammonia in effluent showed general increased levels (5 – 64%) rather than removal similar to the observation under 1 day HRT. It is not clear the cause of general poor performance with ammonia removal. The highest increases were associated with the unplanted beds (56% - 64%), the worst situation probably because the beds had no vegetation. Only a monoculture sugarcane planted lateritebased (TT3) was able to remove 5% of NH_3-N, and this was consistent with the increase in HRT (Akratos and Tsihrintzis, 2007). The performance of TT3 may be attributed to its monoculture sugarcane vegetation which has high preference for NH_3-N uptake (Boschiero et al., 2018). Meanwhile, general increase in ammonia levels may be attributed lower plant and microbial metabolisms and processes like ammonification, nitrification, denitrification, and ammonia volatilization (Masi, 2004; Mthembu et al., 2013; Wu et al., 2016).

No sulphate was found in any effluent including unplanted beds after the 2 days HRT (Table 4.21), a phenomenon similar but improved results over the 1 day HRT. The 100% SO_4^{2-} removal in all wetlands could be attributed to potential anaerobic sulphate-reduction in the bed (Cooper and Findlater, 1990; Zapater-Pereyra et al., 2014), and that the 2 days HRT allowed enough time for completing the necessary redox reactions in plant and microbial metabolisms. Generally, performance with respect to AnS removal was better for the 2 days than 1 day HRT, especially for planted beds with significant performance of 66 – 74%. This confirmed that AnS are more susceptible to biodegradation in the presence of bed media, increasing retention time and vegetation

(Scott and Jones, 2000; Pérez-López et al., 2018). Moreover, all planted beds outperformed their unplanted counterpart beds with RPDs of 6 – 21%, and planted gravel beds significantly differed from unplanted (see Table 4.22).

The iron levels in effluent of all treatment systems increased above 1000% including beds with only gravel media (Table 4.21). This suggests potential releases of more iron into the saturated bed from media and probably vegetation as residence time increased. Wetland beds' ability to transfer oxygen generally decreased (0.3 – 24.12 g-O_2/m^2d) with even a worse situation recorded for an unplanted bed (-0.95 gO_2/m^2d) as HRT increased from 1 to 2 days. The two unplanted beds recorded the lowest OTRs, supporting the assertion that plants are key to contribute oxygen releases into wetland bed matrix (Stein and Hook, 2005; Randerson, 2006).

From the available discharge and irrigational reuse limits (Table 4.18), observations similar to 1 day HRT made with effluent quality. Most parameters pH, EC, DO, TDs, turbidity, TSS, BOD_5, COD, NO_3-N, SO_4^{2-} and P (for laterite-based beds) satisfied guideline limits. Effluent iron levels were unsafe for plants according to the guideline limit. Effluent AnS although <2 mg-LAS/L, yet higher than the 0.5 mg/L recommended for water bodies (Little, 1981) , because AnS could be toxic to very sensitive aquatic organisms especially those intolerant at ≤0.025 mg-LAS/L levels (Abd El-Gawad, 2014).

Table 4.21 Contaminant removal efficiencies of CWs under 2 days HRT

Contaminants[a]	Wetland Treatment Cells															
	CT1		CT2		TT1		TT2		TT3		TT4		TT5		TT6	
	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev
pH (-)	6.68(0.05)	-	6.79(0.06)	-	6.66(0.01)	-	6.63(0.04)	-	6.74(0.03)	-	6.71(0.08)	-	6.83(0.05)	-	6.71(0.02)	-
EC (µS/cm)	376.82(40.27)	-9	389.66(15.74)	-13	410.16(27.17)	-19	405.24(13.85)	-17	412.4(28.29)	-19	356(34.5)	-3	343.78(31.64)	1	390.98(8.58)	-13
DO	0.97(0.20)	-38	1.37(0.43)	-95*	1.39(0.33)	-97*	1.64(0.59)	-133*	1.54(0.27)	-118*	1.24(0.33)	-76*	1.62(0.22)	-130*	1.22(0.16)	-73*
TDS	187.7(20.04)	-9	194(7.8)	-12	204.18(13.64)	-18	201.9(6.83)	-17	205.28(13.99)	-19	177.2(17.04)	-2	171.22(15.78)	1	194.72(4.27)	-13
Turbidity (NTU)	5.65(3.24)	96*	2.14(0.19)	99*	2.01(0.75)	99*	1.76(1.2)	99*	1.75(1.23)	99*	2.13(1.31)	99*	3.26(1.99)	98*	3.14(3.05)	98*
TSS	20.81(4.38)	69*	26.4(12.52)	60*	18.41(13.45)	72*	15.2(6.58)	77*	23.21(18.64)	65*	16(5.66)	76*	16(5.66)	76*	14.41(4.56)	78*
BOD_5	52(15.57)	77*	40.2(11.69)	82*	28.4(7.23)	88*	34(5.52)	85*	28.6(4.56)	87*	23(12.41)	90*	30.4(5.37)	87*	28.2(8.41)	88*
COD	97.6(51.21)	72*	78.6(49.4)	77*	67.4(34.55)	80*	84.4(53.76)	76*	77.4(48.25)	78*	68(43.1)	80*	67.8(29.03)	80*	61.4(29.77)	82*
Ortho Phosphate	9.78(3.10)	24	1.88(0.34)	85*	2.34(0.29)	82*	2.34(1.16)	82*	2.58(0.38)	80*	8.6(3)	33	8.96(2.66)	30	4.16(0.68)	68*
Phosphorus	3.18(0.98)	24	0.62(0.08)	85*	0.76(0.11)	82*	0.78(0.36)	81*	0.84(0.13)	80*	2.8(0.99)	33	2.92(0.89)	30	1.36(0.23)	67*
Nitrate	0.54(0.34)	91*	0.5(0.5)	92*	0.90(0.5)	85*	1.2(1.04)	81*	0.7(0.36)	89*	0.31(0.28)	95*	0.34(0.28)	94*	0.58(0.51)	91*
Nitrite	0.01(<0.001)	29	0	100*	0	100*	0	100*	0	100*	0.01(<0.001)	29*	0.01(<0.001)	29	0.01(0.01)	64*
Ammonia	7.2(1.32)	-56*	7.54(1.22)	-64*	6.22(1.64)	-35	4.83(1.45)	-5	4.39(1.15)	5	6.49(0.97)	-41	5.36(1.28)	-16	6.41(1.7)	-39
Sulphate	0	100*	0.25(0.20)	93*	0	100*	0	100*	0	100*	0	100*	0	100*	0	100*
Anionic surf.	2.59(0.51)	53*	2.21(0.31)	60*	1.9(0.22)	66*	1.82(0.59)	67*	1.47(0.24)	73*	1.81(0.30)	67*	1.44(0.39)	74*	1.67(0.44)	70*
Iron	0.52(0.07)	OTP*	5.99(0.42)	OTP*	8.23(0.69)	OTP*	9.83(1.32)	OTP*	8.18(0.48)	OTP*	0.57(0.27)	OTP*	0.39(0.13)	OTP*	4.39(2.59)	OTP*
OTR (g-O_2/m^2d)	0.30		-0.95		10.34		20.20		24.12		8.91		16.60		8.93	

[a]All units are in mg/L unless otherwise stated; mean = average contaminant level; Std = standard deviation of the mean; %Rev = % removal or reduction; - % Rev = negative indicates increase in contaminant level; OTR = oxygen transfer rate; OTP = over 1000% increase in levels; *indicates T-test is statistically significant at p<0.05.

Table 4.22 Relative performance difference and T-test for CW – 2 days HRT

Parameters	Relative performance difference between planted beds and controls and the p-value for their T-Tests, RPD(p-value)					
	TT1	**TT2**	**TT3**	**TT4**	**TT5**	**TT6**
EC (µS/cm)	-6% (0.182)	-5% (0.135)	-7% (0.155)	6% (0.406)	10% (0.187)	0% (0.873)
DO (mg/L)	-2% (0.955)	-38% (0.429)	-24% (0.481)	-38% (0.157)	-92% (0.001)*	22% (0.470)
TDS (mg/L)	-6% (0.185)	-5% (0.127)	-7% (0.154)	6% (0.398)	10% (0.187)	0% (0.861)
Turbidity (NTU)	0% (0.705)	0% (0.505)	0% (0.500)	2% (0.054)	2% (0.198)	-1% (0.546)
TSS (mg/L)	12% (0.359)	17% (0.115)	5% (0.758)	7% (0.172)	7% (0.172)	18% (0.079)
BOD_5 (mg/L)	5% (0.091)	3% (0.315)	5% (0.073)	13% (0.012)*	9% (0.019)*	5% (0.099)
COD (mg/L)	3% (0.689)	-2% (0.863)	0% (0.970)	9% (0.352)	9% (0.290)	5% (0.524)
Ortho Phosphate (mg/L)	-4% (0.050)	-4% (0.421)	-5% (0.016)*	9% (0.559)	6% (0.665)	-18% (0.000)*
Phosphorus (mg-PO_4-P/L)	-3% (0.058)	-4% (0.365)	-5% (0.014)*	9% (0.558)	6% (0.671)	-18% (0.000)*
Nitrate (mg-NO_3-N/L)	-6% (0.700)	-11% (0.312)	-3% (0.949)	4% (0.178)	3% (0.336)	-1% (0.587)
Nitrite (mg-NO_2-N/L)	0%a	0%a	0%a	0% (0.242)	0% (0.141)	-36% (0.347)
Ammonia (mg-NH_3-N/L)	29% (0.187)	59% (0.013)	68% (0.003)*	15% (0.361)	40% (0.056)	25% (0.262)
Sulphate (mg-SO_4^{2-}/L)	7% (0.347)	7% (0.347)	7% (0.347)	0%a	0%a	7% (0.347)
Anionic surfactants (mg-LAS/L)	6% (0.112)	7% (0.235)	13% (0.003)*	14% (0.019)*	21% (0.004)*	10% (0.054)

Note: [a] Indicates that effluent values were largely 0 mg/L as input data

*indicates T-test is statistically significant at $p<0.05$.

4.4.3.3 Performance under a 3-day HRT

Performance efficiencies under the 3 days HRT are shown in Table 4.23 and Table 4.24. Effluent pH was 6.61 – 6.90 quite close to influent mean pH 6.70. For effluent EC, DO and TDS, the usual phenomenon of increased levels is observed in magnitude between 10 and 201%. As already stated, increased EC and TDS in effluent could be due to release of dissolved materials (salts and organics) from bed media, vegetation and decomposition of greywater. Apart from planted bed TT1 (RPD: -9 to -10%), the rest performed better than the unplanted with RPDs between 2 and 49% although not statistically significant (Table 4.24). DO levels in almost all planted beds, except TT4, were significantly high and this was expected for planted CW beds (Ling et al., 2009). Significant rise in DO levels was a sign of occurrence of oxygen diffusion from plant roots and wetland bed surfaces to the porous rootmedia matrix. However, planted gravel bed TT4 was not expected to record a low DO level (0.75 mg/L), and even lower than its control CT1 (0.83 mg/L) although there was improvement in influent DO. Turbidity removal efficiencies were significantly high with performance between 96 and 99% and best systems were vegetated laterite-based beds (98-99%, $p<0.05$). Planted beds were almost always marginally better or at worst equal to the unplanted in removing turbidity (RPDs: 0 – 1%, $p>0.05$). Perhaps, the high components of sand, silt and clay with large particulate surface area and comparatively low porosity, likely contributed to improved filtering property of laterite-based beds.

Almost all CW beds including controls except TT2 performed below the expected >80% TSS removal efficiency although current performance levels were all significant (Table 4.23). However, unplanted beds mostly performed better than the planted with RPDs of -2 to -9% ($p>0.05$) except for TT2 (RPD 6%, $p=0.420$) and TT6 (RPD 4%, $p=0.475$). For BOD_5 and COD, removal efficiencies were significant within the range of 83 – 90% ($p<0.05$) and that met the >80% expectation, with more improvement over the results from previous HRTs. The differences in removal efficiencies between unplanted and planted beds were mostly marginal in favour of planted beds with RPDs of 0 – 4% ($p>0.05$) and vegetation could play a role in the marginal differences.

On nutrients, gravel beds (CT1, TT4 and TT5) continued their low and statistically insignificant performance levels with respect to phosphorus (P) removal (20 – 34%,

$p > 0.05$) even after a 3-day resident time. Notwithstanding, P removal levels were higher than the expected 10 – 20% (Dotro et al., 2017), implying that the performance levels of the current wetlands are appreciable. Nitrite removal was statistically significant although characteristically low for gravel beds (55 – 66%, $p < 0.05$) including their control (TT4, TT5 and CT1) unlike the laterite beds – TT6 (77%, $p < 0.05$) and the rest 100% ($p < 0.05$). Nitrate removal continues to be significantly high for all treatment systems (83 – 96%, $p < 0.05$) and the trend is similar to earlier findings from 1 and 2 days HRTs. The removal mechanisms for these nutrients are also similar as already discussed. Performance with ammonia removal was mixed: 4 of 6 planted beds (TT1, TT4, TT5 & TT6) fixed or increased the contaminant levels (2 – 40%) while the other two, monoculture vegetation wetlands TT2 (taro) and TT3 (sugarcane), removed 3 and 33% respectively. TT3 has consistently removed NH_3-N from 2 – 33% along the 1 to 3 days HRT (Table 4.23 and Table 4.24), and this observation may partly be due to the sugarcane plant which has high preference for ammonia/ammonium (Robinson et al., 2011).

Sulphate removal as usual was impressive (100%, $p < 0.05$) for most CW beds including controls except for TT5 (50%, $p > 0.05$), TT6 (73%, $p < 0.05$) and TT4 (91%, $p < 0.05$) Table 4.23. In these three cases, the unplanted beds performed better with RPDs of -9 to -50% (Table 4.24). It appears that increasing residence time released potentially absorbed SO_4^{2-} back into effluent water for these beds TT4, TT5 and TT6. Meanwhile, performance of wetlands with surfactants removal was significantly high (61 – 75%, $p < 0.05$) similar to the levels observed in the 2 days HRT. The controls (unplanted beds) still gave comparatively close and low removal efficiencies although significant (56% & 57%, $p < 0.05$). The relatively better AnS removal performance from planted beds suggests that the contaminant is better removed after increasing residence time and with the influence of vegetation (Scott and Jones, 2000; Ramprasad and Philip, 2016b; Pérez-López et al., 2018).

Iron levels increased in all effluents after 3 days resident time but the effluent levels this time was about twice lower for gravel beds than in the 2 days HRT. Also, all laterite beds gave effluent iron levels well over 1000% similar to the previous findings.

Also, CW beds performed better in OTRs this time than during the 2-day HRT except for TT1. CT2 was able to recover from a deficit of -0.95 g-O_2/m^2d under 2 days HRT to 3.5 g-O_2/m^2d. All planted beds have shown higher OTR than their controls (unplanted ones) especially for laterite-based beds except for the gravel bed TT4 under the 3 days HRT. TT2 and TT3 which are laterite-based monoculture wetlands of taro and sugarcane respectively, have shown consistent increases in OTR as HRT increased beyond 1 day and the highest performance came from monoculture sugarcane bed (TT3). On the other hand, unplanted laterite-based bed (control CT2) showed all-time low OTR probably due absence of vegetation and low porosity than gravel beds.

In a similar trend, effluent quality of planted CW microcosm models satisfied available discharge and irrigational reuse limits for pH, EC, DO, TDs, turbidity, TSS, BOD_5, COD, NO_3-N, and SO_4^{2-} except the usual failures with effluent ammonia, phosphorus and iron levels. Also, AnS although generally <2 mg-LAS/L, effluent could still be toxic to more sensitive ecological organisms whose tolerance levels are ≤0.025 mg-LAS/L (Abd El-Gawad, 2014).

Table 4.23 Contaminant removal efficiencies of CWs under 3 days HRT

Contaminants[a]	Wetland Treatment Cells															
	CT1		CT2		TT1		TT2		TT3		TT4		TT5		TT6	
	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev	mean(Std)	% Rev
pH (-)	6.74(0.04)	-	6.76(0.09)	-	6.61(0.02)	-	6.62(0.01)	-	6.72(0.07)	-	6.65(0.08)	-	6.90(0.02)	-	6.74(0.03)	-
EC (µS/cm)	384.04(35.11)	-13	428.10(29.96)	-26*	461.40(19.96)	-35*	406.78(16.39)	-19*	421.12(20.98)	-23*	374.10(25.44)	-10	363.76(20.66)	-7	401.48(26.79)	-18
DO	0.83(0.18)	-31	1.13(0.31)	-80*	1.07(0.35)	-70*	1.21(0.25)	-92*	1.45(0.40)	-130*	0.75(0.13)	-19	1.90(0.58)	-201*	1.02(0.38)	-61
TDS	191.34(17.47)	-13	214.46(14.18)	-26*	229.84(9.94)	-35*	202.76(8.27)	-19*	209.86(10.50)	-23*	186.38(12.66)	-10	181.20(10.36)	-7	131(102.28)	23
Turbidity (NTU)	4.11(1.44)	97*	3.24(1.50)	98*	2.02(0.54)	99*	1.17(0.28)	99*	1.83(0.70)	99*	4.43(3.64)	97*	5.54(4.63)	96*	3.21(2.45)	98*
TSS	19.20(11.80)	75*	19.21(7.69)	75*	26.41(10.43)	65*	14.40(10.04)	81*	26.40(16.15)	65*	25.60(20.91)	66	20.81(13.38)	73*	16.01(5.66)	79*
BOD_5	31.80(11.37)	86*	31.60(10.85)	86*	36.80(17.25)	84*	30(11.34)	87*	32.40(13.72)	86*	25.80(15.32)	88*	23.40(7.27)	90*	28.40(12.46)	87*
COD	55.80(17.98)	84*	60(19.08)	82*	60.20(16.77)	82*	56.80(5.40)	83*	53.40(17.74)	84*	45.80(16.08)	86*	52.40(23.83)	85*	53.20(16.45)	84*
Ortho Phosphate	6.78(5.13)	31	2.43(1.91)	75*	2.76(2.24)	72*	2.38(1.92)	76*	3.11(2.03)	68*	6.74(3.16)	31	6.51(3.43)	34	4.04(1.98)	59*
Phosphorus	2.21(1.68)	20	0.80(0.58)	71	0.89(0.74)	68	0.77(0.61)	72	1.01(0.67)	64	2.20(1.02)	21	2.15(1.12)	23	1.33(0.64)	52
Nitrate	0.58(0.42)	89*	0.43(0.28)	92*	0.77(0.47)	86*	0.50(0.44)	91*	0.20(0.35)	96*	0.48(0.08)	91*	0.92(0.35)	83*	0.6(0.39)	89*
Nitrite	0.01(0.01)	66*	0	100*	0	100*	0	100*	0	100*	0.01(<0.001)	55*	0.01(<0.001)	55*	0.01(0.01)	77*
Ammonia	5.07(3.49)	0	7.48(1.56)	-47	7.10(1.21)	-40	4.94(1.65)	3	3.42(2.27)	33	5.54(2.0)	-9	5.18(1.98)	-2	5.52(1.52)	-9
Sulphate	0	100*	0	100*	0	100*	0	100*	0	100*	0.50(0.71)	91*	2.80(2.68)	50	1.50(2.12)	73*
Anionic surf.	2.37(1.04)	57*	2.38(0.40)	56*	2.13(0.47)	61*	1.58(0.43)	71*	1.54(0.79)	72*	1.89(0.71)	65*	1.34(0.53)	75*	1.50(0.52)	72*
Iron	0.61(0.37)	-500	7(0.92)	OTP*	9.72(2.48)	OTP*	9.23(2.07)	OTP*	7.78(0.87)	OTP*	0.68(0.45)	-500	0.56(0.30)	-500	4.87(0.80)	OTP*
OTR ($g\text{-}O_2/m^2d$)	21.66		3.5		5.79		22.84		34.04		18.79		21.77		18.65	

Note: [a]All units are in mg/L unless otherwise stated; mean = average contaminant level; Std = standard deviation of the mean; %Rev = % removal or reduction; - % Rev = negative indicates increase in contaminant level; OTR = oxygen transfer rate; OTP = over 1000% increase in levels; *indicates T-test is statistically significant at $p<0.05$.

Table 4.24 Relative performance difference and T-test for CW – 3 days HRT

Parameters	Relative performance difference between planted beds and controls and the p-value for their T-Tests, RPD (p-value)					
	TT1	**TT2**	**TT3**	**TT4**	**TT5**	**TT6**
EC (µS/cm)	-10% (0.072)	6% (0.200)	2% (0.681)	3% (0.622)	6% (0.298)	8% (0.177)
DO (mg/L)	10% (0.782)	-12% (0.690)	-50% (0.204)	12% (0.454)	-17% (0.004)*	19% (0.608)
TDS (mg/L)	-9% (0.082)	7% (0.150)	3% (0.576)	3% (0.621)	6% (0.297)	49% (0.108)
Turbidity (NTU)	1% (0.125)	2% (0.016)*	1% (0.093)	0% (0.860)	-1% (0.528)	0% (0.980)
TSS (mg/L)	-9% (0.250)	6% (0.420)	-9% (0.395)	-8% (0.567)	-2% (0.846)	4% (0.475)
BOD_5 (mg/L)	-2% (0.584)	1% (0.825)	0% (0.921)	3% (0.502)	4% (0.201)	1% (0.676)
COD (mg/L)	0% (0.986)	1% (0.728)	2% (0.587)	3% (0.381)	1% (0.805)	2% (0.563)
Ortho Phosphate (mg/L)	-3% (0.805)	0% (0.972)	-7% (0.599)	0% (0.989)	3% (0.926)	-16% (0.225)
Phosphorus (mg-PO_4-P/L)	-3% (0.832)	1% (0.955)	-8% (0.605)	1% (0.988)	2% (0.947)	-19% (0.208)
Nitrate (mg-NO_3-N/L)	-6% (0.675)	-1% (0.809)	4% (0.260)	2% (0.616)	-6% (0.202)	-3% (0.277)
Nitrite (mg-NO_2-N/L)	0%[a]	0%[a]	0%[a]	-11% (0.879)	-11% (0.292)	-23% (0.347)
Ammonia (mg-NH_3-N/L)	7% (0.677)	50% (0.036)*	80% (0.011)*	-9% (0.800)	-2% (0.953)	39% (0.079)
Sulphate (mg-SO_4^{2-}/L)	0%[a]	0%[a]	0%[a]	-9% (0.347)	-50% (0.048)*	-27% (0.347)
Anionic surfactants (mg-LAS/L)	5% (0.393)	15% (0.016)*	15% (0.067)	9% (0.422)	19% (0.085)	16% (0.017)*

Note: [a] Indicates that effluent values were largely 0 mg/L as input data

*indicates T-test is statistically significant at p<0.05.

4.4.3.4 Performance of CW beds based on key effluent contaminants

MANOVA comparison test on effluent TDS, TSS, BOD_5, COD, Ortho-P, P, NO_3-N, NH_3-N and AnS is presented on the use of plants (Table 4.25) and media (Table 4.26) in the CWs. However, the MANOVA test based on Pillai's statistics showed significant influence of the presence of plants and media on effluent quality. Thus, for plants the Pillai's Trace = 1.252, $F(18, 222)=20.633$, $p<0.0001$, Partial Eta Squared = 0.626, with power to detect effect = 1. Media likewise, showed Pillai's Trace = 1.666, $F(18, 222)=61.590$, $p<0.0001$, Partial Eta Squared = 0.833, with power to detect effect = 1.

The MANOVA results on the absence-presence of macrophytes in CW confirmed earlier findings on individual planted CWs and their unplanted counterparts along the three different HRT. Planted beds outperformed the unplanted ones by giving comparatively lower effluent concentrations for most contaminants except NO_3-N. For the remaining eight contaminants listed in (Table 4.25), planted beds' removal efficiencies were better than unplanted beds although the differences were mostly statistically insignificant except for BOD_5, NH_3-N and AnS ($p<0.0001$, Table 4.25). For these three, the mean (±standard deviation) of effluent concentrations (mg/L) were BOD_5: planted – 28.3±11.3 against unplanted – 37.9±13.6; NH_3-N: planted – 5.6±1.5 against unplanted – 6.9±1.9; and AnS: planted – 1.8±0.5 against unplanted 2.4±0.5.

The performance of CW beds in terms of media type showed that laterite-based beds gave comparatively lower effluent concentrations for Ortho-P/P, AnS and BOD_5 than the gravel beds (Table 4.26). However, apart from the effluent Ortho-P/P which gave significant mean difference between the two media types ($p<0.0001$), the rest (BOD_5 & AnS) was not statistically significant ($p>0.05$, 0.364 – 0.832) (Table 4.26). Also, gravel beds performed better with lower effluent concentrations for the remaining five contaminants (TDS, TSS, COD, NO_3-N and NH_3-N) but this showed no significant mean differences between the performance of the two media ($p>0.05$) except in the removal of TDS ($p<0.05$). This strongly suggests that laterite-based media are competitive to gravel, a commonly used bed media in constructed wetland technologies.

Table 4.25 MANOVA comparison test between planted and unplanted CW beds

Dependent Variable			Mean Difference (I-J)	Std. Error	Sig (p-value)	95% Confidence Interval for Difference	
Parameters	I-planted	J-unplanted				Lower Bound	Upper Bound
TDS	Planted	Unplanted	-3.553	6.403	.580	-16.234	9.127
TSS	Planted	Unplanted	-2.488	2.451	.312	-7.341	2.366
BOD_5	Planted	Unplanted	-9.622*	2.510	.000	-14.592	-4.653
COD	Planted	Unplanted	-11.311	6.730	.095	-24.638	2.016
Ortho-P	Planted	Unplanted	-.848	.715	.238	-2.264	.568
P	Planted	Unplanted	-.272	.232	.244	-.732	.188
NO_3-N	Planted	Unplanted	.127	.094	.179	-.059	.314
NH_3-N	Planted	Unplanted	-1.215*	.339	.000	-1.886	-.545
AnS	Planted	Unplanted	-.655*	.101	.000	-.856	-.454

*Mean difference is significant at p<0.05

Table 4.26 MANOVA comparison test between gravel and laterite-based CW beds

Dependent Variable			Mean Difference (I-J)	Std. Error	Sig (p-value)	95% Confidence Interval for Difference	
Parameters	I-planted	J-unplanted				Lower Bound	Upper Bound
TDS	Gravel	Laterite-based	-19.885*	5.435	.000	-30.648	-9.123
TSS	Gravel	Laterite-based	-.571	2.201	.796	-4.930	3.788
BOD_5	Gravel	Laterite-based	.507	2.380	.832	-4.206	5.219
COD	Gravel	Laterite-based	-1.573	6.089	.797	-13.631	10.485
Ortho-P	Gravel	Laterite-based	5.466*	.401	.000	4.672	6.259
P	Gravel	Laterite-based	1.775*	.130	.000	1.517	2.033
NO_3-N	Gravel	Laterite-based	-.009	.085	.919	-.177	.159
NH_3-N	Gravel	Laterite-based	-.153	.319	.632	-.784	.478
AnS	Gravel	Laterite-based	.096	.105	.364	-.112	.304

*Mean difference is significant at p<0.05

4.5 Constructed wetland for greywater treatment – linking performance to design features, and model for predicting effluent quality[5]

4.5.1 Multivariate analysis of constructed wetland performance

The multivariate analysis of variance (MANOVA) was used to handle complexities inherent in the trends associated with effects of wetland and HRT on contaminant removal efficiencies. First, the general MANOVA results from both SPSS and RStudio platforms were compared (Table 4.27 and Table 4.28) and both results yielded inference that converged. The two-way MANOVA test from SPSS showed a significant multivariate main effect from Wetlands and HRTs but no significant effect from their interactions, using the Pillai's Multivariate test statistics since the Box M test was significant (Nimon, 2012). Thus a) for Wetlands: Pillai's Trace = 1.790, $F(63, 658) = 3.590$, $p<0.001$, Partial Eta Squared = 0.256, with power to detect effect =1.00 (certainty); b) for HRT: Pillai's Trace =0.449, $F(18, 178) = 2.859$, $p<0.001$, Partial Eta Squared = 0.224, with power to detect effect = 0.999; and c) Wetland – HRT Interactions: Pillai's Trace = 1.146, $F(26, 864) = 1.00$, $p=0.486$, Partial Eta Squared = 0.127, with power of 1.0 (Table 4.27).

Similarly, MANOVA results from RStudio showed significant multivariate main effect from Wetlands and HRTs but no significant effect from their interactions, thus d) with df = 63 and MATS test statistics = 628.848, a paramBS (MATS) $p<0.001$ were recorded for Wetlands; e) for HRT, df = 18 and MATS test statistics = 96.336, a paramBS (MATS) $p<0.001$ were noted; and f) for Wetland – HRT Interactions, also a paramBS (MATS) $p=0.766$ was recorded for df=126 and MATS test statistics = 142.066 (Table 4.28).

Thus, these findings (Table 4.27 and Table 4.28) confirmed that the data primarily could be considered parametric and for quality assurance purposes, it rightly fitted MANOVA tests. Again, the multivariate tests for between group effects from all tools – parametric, semi-parametric and nonparametric (i.e. Independent-Samples Kruskal-

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Wallis ANOVA test), gave results that converged into the same conclusions (Table 4.29 and Table 4.30).

Moreover, from the two-way MANOVA results above, it can be confidently stated that designed wetlands and HRT conditions had effect on effluent water quality (also illustrated in Table 4.27). Thus, about 22% and 26% of the variance in effluent water quality parameters based on the Partial Eta Squared (Brown, 2008; Denis, 2019) were accounted for by the effect of Wetland and HRT group differences respectively (Table 4.27). Only the Roy's Largest Root statistics showed a significant main effect from Wetlands – HRT interactions. This multivariate statistics can be ignored because it is known to be associated with high Type I error rates due its preference for reporting the upper bound on F-test that yields a lower bound on the significance level (Grice and Iwasaki, 2007). This significant multivariate omnibus test for interactions effect may have come from the significant effect on TDS (p=0.022), the only individual water quality parameter that was significantly influenced by wetlandHRT interactions as identified in the univariate tests (Table 4.29). Again, the none interaction effect is an indication that there is no significant difference in wetland performance that is directly dependent on HRTs. Although this is further interrogated in the next sections, the development could be positive to the design considerations. Simply because it could imply that a 1 day HRT could achieve comparable performance efficiencies like the 2 and 3 days HRTs.

Given the significance of the overall multivariate test, univariate main effects (Between-Subject effects) were also examined (Table 4.29 and Table 4.30). Both the parametric MANOVA and the nonparametric Kruskal-Wallis ANOVA tests gave comparable results except a marginal and very narrow statistical decision for different stance on NO_3-N. While the nonparametric test strongly indicated significant statistical differences among wetlands (p=0.026, Table 6) on effluent NO_3-N, the parametric almost declared same with p=0.055. Generally, significant univariate main effects for wetlands (p<0.05) were obtained for specific effluent water quality parameters – TDS, BOD_5, Ortho-P, P, NH_3-N and AnS, and also similar effects from HRT (p<0.05) were obtained for TDS, COD and AnS (Table
4.29 and Table 4.30).

In effect, wetland cells and different HRTs had significant impact on these specific effluent water qualities. The implication is that wetland cells did not have the same performance levels with respect to the specified effluent qualities and likewise the different HRTs to three listed contaminants. Therefore, the initial assumption that the performance of all wetland cells will be the same is no more valid at least on the 6 of 9 tested contaminants. Also, the assumption that the different HRTs will equally impact effluent quality levels among the wetlands is not valid for at least 3 of 9 tested contaminants. Again, it was observed that two other effluent water qualities came close to be statistically influenced by the wetlands and HRT. Removal performance on NO_3-N was almost significantly different across wetlands (p=0.055) and indeed it was as already stated (Table 4.29 and Table 4.30), while NH_3-N somehow also came close for the different HRTs with p=0.093 (Table 4.29). It is therefore not surprising that multiple comparison tests (to be discussed shortly) revealed significant differences between pairs of wetland cells and HRTs. Probably factors other than just different wetland features and HRT may have contributed to these mixed trends and happenings, especially when several environmental and operational conditions like air/oxygen, pH, temperature, wind & light intensities, humidity etc. and their combinations were not controlled (Kadlec and Wallace, 2009; Papaevangelou et al., 2016).

Also, MANOVA Post-Hoc Comparisons tests were examined to check if the levels of individual wetlands and likewise HRTs significantly differed on the various effluent water qualities.

Table

Table 4.27 MANOVA test from SPSS

Effect		Value	F	Hypothesis df	Error df	Sig. (p-value)	Partial Eta Squared	Observed Power
Wetlands	Pillai's Trace	1.790	3.590*	63.000	658.000	.000	.256	1.000
	Wilks' Lambda	.057	5.286	63.000	501.729	.000	.336	1.000
	Hotelling's Trace	5.911	8.096	63.000	604.000	.000	.458	1.000
	Roy's Largest Root	4.485	46.846	9.000	94.000	.000	.818	1.000
HRT	Pillai's Trace	.449	2.859*	18.000	178.000	.000	.224	.998
	Wilks' Lambda	.590	2.956	18.000	176.000	.000	.232	.999
	Hotelling's Trace	.631	3.052	18.000	174.000	.000	.240	.999
	Roy's Largest Root	.503	4.974	9.000	89.000	.000	.335	.999
Wetlands:HRT interactions	Pillai's Trace	1.146	1.000	126.000	864.000	.486	.127	1.000
	Wilks' Lambda	.273	1.010	126.000	686.613	.460	.134	.999
	Hotelling's Trace	1.485	1.016	126.000	776.000	.439	.142	1.000
	Roy's Largest Root	.455	3.118	14.000	96.000	.000	.313	.994

Note: * Pillai's Trace test is significant at p<0.001

Table 4.28 MANOVA test from R

Effect	Wald-type statistics (WTS)			Modified ANOVA-type Statistics (MATS)	Resampling p-values	
	Test statistics	df	p-value	Test Statistics	paramBS (WTS)	paramBS (MATS)
Wetlands	1427.426	63	0.000	628.848*	0.000	0.000
HRT	202.072	18	0.000	96.336*	0.000	0.000
Wetlands:HRT interactions	754.35	126	0.000	142.066	0.986	0.766

Note: * MATS test is significant at p<0.001

4.29 Tests of between – subject effects (Univariate analyses)

Effect source: independent variables	Dependent variables: Effluent water quality	Type III Sum of Squares	df	Mean Square	F	Sig. (p-value)	Partial Eta Squared	Observed Power parameters
Wetlands	TDS	29349.288	7	4192.755	7.402**	.000	.351	1.000
	TSS	1186.326	7	169.475	1.180	.321	.079	.483
	BOD_5	2618.767	7	374.110	2.560*	.018	.157	.866
	COD	6816.767	7	973.824	.916	.498	.063	.376
	Ortho-P	877.934	7	125.419	27.598**	.000	.668	1.000
	P	92.644	7	13.235	27.511**	.000	.667	1.000
	NO_3-N	2.492	7	.356	2.060	.055	.131	.767
	NH_3-N	90.750	7	12.964	5.973**	.000	.303	.999

Table

Effect source: independent variables	Dependent variables: Effluent water quality parameters	Type III Sum of Squares	df	Mean Square	F	Sig. (pvalue)	Partial Eta Squared	Observed Power
	AnS	11.888	7	1.698	7.529**	.000	.354	1.000
HRT	TDS	9246.104	2	4623.052	8.162*	.001	.145	.955
	TSS	128.964	2	64.482	.449	.640	.009	.121
	BOD_5	374.617	2	187.308	1.282	.282	.026	.272
	COD	9856.817	2	4928.408	4.635*	.012	.088	.770
	Ortho-P	12.075	2	6.037	1.328	.270	.027	.281
	P	1.292	2	.646	1.343	.266	.027	.283
	NO_3-N	.533	2	.266	1.542	.219	.031	.320
	NH_3-N	10.576	2	5.288	2.436	.093	.048	.480
	AnS	1.609	2	.804	3.566*	.032	.069	.650
Wetlands * HRT Interactions	TDS	16173.809	14	1155.272	2.040*	.022	.229	.932
	TSS	986.087	14	70.435	.490	.933	.067	.281
	BOD_5	1778.983	14	127.070	.869	.594	.113	.513
	COD	4378.383	14	312.742	.294	.994	.041	.171
	Ortho-P	46.802	14	3.343	.736	.734	.097	.432
	P	4.665	14	.333	.693	.776	.092	.405

	NO_3-N	4.259	14	.304	1.761	.056	.204	.881
	NH_3-N	27.871	14	1.991	.917	.543	.118	.541
	AnS	1.834	14	.131	.581	.874	.078	.336
Error	TDS	54377.596	96	566.433				
	TSS	13787.474	96	143.620				
	BOD_5	14031.600	96	146.163				
	COD	102068.400	96	1063.213				
	Ortho-P	436.275	96	4.545				
	P	46.183	96	.481				
	NO_3-N	16.584	96	.173				
	NH_3-N	208.355	96	2.170				
	AnS	21.655	96	.226				

Note: *Test is statistically significant at p<0.05, **Test is statistically significant at p<0.001, AnS = anionic surfactants

4.30 Nonparametric I-Samples Kruskal-Wallis Tests - CWs & HRTs

Null Hypothesis	Significant & Decision			
	Wetlands		HRT	
	p-value	Decision	p-value	Decision

Table

The distribution of TDS is the same across categories	0.000**	Reject the null hypothesis	0.000**	Reject the null hypothesis
The distribution of TSS is the same across categories	0.221	Retain the null hypothesis	0.848	Retain the null hypothesis
The distribution of BOD_5 is the same across categories	0.029*	Reject the null hypothesis	0.366	Retain the null hypothesis
The distribution of COD is the same across categories	0.457	Retain the null hypothesis	0.047*	Reject the null hypothesis
The distribution of Ortho-P is the same across categories	0.000**	Reject the null hypothesis	0.576	Retain the null hypothesis
The distribution of P is the same across categories	0.000**	Reject the null hypothesis	0.572	Retain the null hypothesis
The distribution of NO_3-N is the same across categories	0.026*	Reject the null hypothesis	0.331	Retain the null hypothesis
The distribution of NH_3-N is the same across categories	0.000**	Reject the null hypothesis	0.228	Retain the null hypothesis
The distribution of AnS is the same across categories	0.000**	Reject the null hypothesis	0.026*	Reject the null hypothesis

Note: *Test is statistically significant at $p<0.05$, **Test is statistically significant at $p<0.001$, AnS = anionic surfactants

4.5.2 Influence of CW design features on performance

4.5.2.1 Influence of hydraulic retention time

The multivariate analysis in the study found no statistical significant effect of HRT and wetland interactions although treatment wetlands are functional by HRT (Conn and Fiedler, 2006; Papaevangelou et al., 2016). A real-world phenomenon was not confirmed by statistics in this case, however, HRT showed significant effect on effluent TDS, COD and AnS as already presented, and therefore a MANOVA Posthoc pairwise comparison test was carried out to isolate significant effect from the different HRTs.

Indeed, the results showed significant HRT pairwise differences in four effluent qualities – TDS, COD, NH_3-N and AnS (Table 4.31), with an inclusion of NH_3-N (Table 4.29). 1 day HRT gave low effluent TDS but high AnS effluent levels which differed from both 2 & 3 days HRT at the same time; 3 days HRT with lower effluent COD levels differed from both 1 & 2 days HRT; and finally, 1 day HRT with higher effluent NH_3-N levels only differed from 3 days HRT (Table 4.31). The results imply that HRT influenced removal of greywater contaminants by moderating the wetlands although there is no statistical evidence to support CW-HRT interactions. Likely because HRT could not statistically impose its influence on all or most of the contaminants well enough to constitute deliberate efforts for statistical significance. This could be a classical situation where p-value based statistical significance does not necessarily mean practical or real-world relevance (Nakagawa and Cuthill, 2007).

Table
4.31 HRT levels pairwise comparisons

Dependent Variable	Factor levels (I)	(J)	Mean Difference (I-J)	Std. Error	Sig. (p-value)	95% Confidence Interval Lower Bound	Upper Bound
TDS	1day	2days	-17.9200*	5.32181	.001	-28.4837	-7.3563
		3days	-19.2500*	5.32181	.000	-29.8137	-8.6863
	2days	1day	17.9200*	5.32181	.001	7.3563	28.4837
		3days	-1.3300	5.32181	.803	-11.8937	9.2337
	3days	1day	19.2500*	5.32181	.000	8.6863	29.8137
		2days	1.3300	5.32181	.803	-9.2337	11.8937
COD	1day	2days	-3.2000	7.29113	.662	-17.6728	11.2728
		3days	17.4250*	7.29113	.019	2.9522	31.8978
	2days	1day	3.2000	7.29113	.662	-11.2728	17.6728
		3days	20.6250*	7.29113	.006	6.1522	35.0978
	3days	1day	-17.4250*	7.29113	.019	-31.8978	-2.9522
		2days	-20.6250*	7.29113	.006	-35.0978	-6.1522
NH_3-N	1day	2days	.1750	.32942	.596	-.4789	.8289
		3days	.6987*	.32942	.036	.0449	1.3526
	2days	1day	-.1750	.32942	.596	-.8289	.4789

Table

		3days	.5237	.32942	.115	-.1301	1.1776
	3days	1day	-.6987*	.32942	.036	-1.3526	-.0449
		2days	-.5237	.32942	.115	-1.1776	.1301
AnS	1day	2days	.2338*	.10620	.030	.0229	.4446
		3days	.2560*	.10620	.018	.0452	.4668
	2days	1day	-.2338*	.10620	.030	-.4446	-.0229
		3days	.0222	.10620	.834	-.1886	.2331
	3days	1day	-.2560*	.10620	.018	-.4668	-.0452
		2days	-.0222	.10620	.834	-.2331	.1886

Note: *Test is statistically significant at p<0.05, AnS = anionic surfactant

4.5.2.2 Influence of media on the performance of CW

On organic contaminants removal, TSS, BOD_5, COD and AnS were considered (Figure 4.6). For TSS, the gravel media showed relatively better performance than the laterite-based media in both unplanted and planted bed conditions. Gravel media gave removal efficiencies of 69 – 75% (unplanted) and 66 – 76% (planted) as against laterite-based media of 60 – 75% (unplanted) and 59 – 72% (planted) (see Figure 4.6). However, the differences between the two media types were not statistically significant ($p>0.05$, see Appendix 1). The BOD_5 removal efficiencies for gravel media were 77 – 86% (unplanted) and 87 – 90% (planted) as against those of lateritebased media with 82 – 86% (unplanted) and 84 – 88% (planted). The efficiencies from the two media types appeared comparable but unplanted laterite-based media seemingly had an edge over unplanted gravel and the observation is vice versa when the media were planted. Moreover, the comparable efficiencies were confirmed by the pairwise comparison test which showed that there was no statistically significant difference between the efficiencies of media whether unplanted ($p=0.329$) or planted ($p=0.280$), see Appendix 1.

Gravel media performance with COD removal outperformed the laterite-based beds in both unplanted (72 – 84% against 69 – 82%) and planted (78 – 86% against 72 – 82%). However, any differences in the media efficiencies were not statistically significant (for unplanted media $p=0.920$ and planted $p=0.400$). Efficiencies for AnS removal followed the BOD_5 trend with the gravel versus laterite-based media as: for unplanted (47 – 57% against 42 – 60%) and planted (55 – 67% against 53 – 66%). Thus, efficiencies appeared comparable but unplanted laterite-based media seemingly had an edge over gravel when unplanted and the observation is vice versa when the media were planted. Once again, the differences in efficiencies were not statistically significant, a reflection in mean effluent concentrations – under unplanted ($p=0.747$) and planted ($p=0.437$) conditions, see Appendix 1. In general, the two media gave comparable removal efficiencies based on the effluent organic contaminants tested although gravel sometimes had an edge over the laterite-based media.

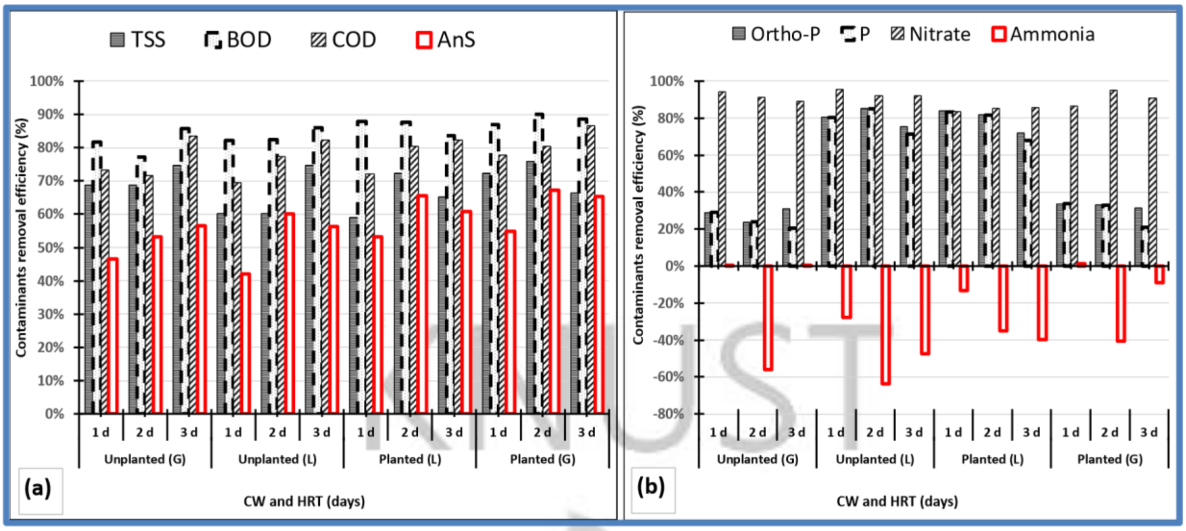

Figure 4.6 CW performance according to media type (gravel & laterite-based) Note: Performance for (a) organic contaminants and (b) nutrients

The CW efficiencies with nutrient removal in terms of phosphorus species (Ortho-P and P), NO_3-N, and NH_3-N are presented in Figure 4.6 and Appendix 1. There was clear significant difference between media efficiencies for Ortho-P/P species where gravel media poorly performed under unplanted conditions with efficiencies of 20 – 31% against 71 – 85% ($p<0.0001$) and under planted conditions with efficiencies of 21 – 34% against 68 – 84% ($p<0.0001$). The media efficiencies for NO_3-N removal were comparable. Gravel media gave 89 – 94% and 86 – 95% in unplanted and planted conditions respectively and in the same way laterite-based media gave 92 – 95% and 84 – 86%. In both unplanted and planted conditions, the effluent nitrate concentrations from the two media were not statistically significant ($p=0.432$ & $p=0.247$).

However, the removal efficiencies for NH_3-N were poor in both media types whether planted or unplanted because almost always effluent NH_3-N increased instead of reduction (Figure 4.6). Only gravel media showed instances of marginal NH_3-N removal up to 1% apart from the fact that gravel's performance was generally better than the laterite-based media. This could be the reason for significant difference in effluent concentrations between gravel and laterite-based media under unplanted conditions ($p=0.007$, Appendix 1). However, under planted conditions, there was no significant difference between the mean effluent NH_3-N ($p=0.182$). Generally, laterite-based media had superior edge over gravel for Ortho-P/P species removal while gravel had better performance for NH_3-N especially under unplanted conditions. Both media have comparable efficiencies for NO_3-N removal.

4.5.2.3 Influence of vegetation on the performance of CW

Results to show the influence of vegetation on CW performance by efficiencies and pairwise comparison tests are presented in Figure 4.7 and Appendix 2. For the organic contaminants TSS, BOD_5, COD and AnS, removal efficiencies ranged between 42 and 88%. The CW of taro – sugarcane mixed vegetation showed comparable removal efficiencies with the unplanted bed (59 – 72% against 60 – 75%) such that there was absolute no difference between the mean effluent TSS concentrations ($p=1.0$). The taro bed (77 – 81%) showed better efficiency over the unplanted (60 – 75%) and this translated into significant difference in mean effluent levels between the taro and unplanted beds ($p=0.026$). The TSS removal efficiencies of mixed (sugarcane+taro) and unplanted beds already presented as comparable appeared to be similar as well to efficiency of sugarcane bed (65 – 72%) and that was confirmed by the significant tests of $p=0.761$. Taro bed also showed a better TSS removal efficiencies than sugarcane bed and this was almost significant according to the test of difference in mean effluent concentrations ($p=0.054$). Thus, presence of these indigenous plants taro, sugarcane and their mixture influence the performance of the CWs in removing TSS. CW planted with a mixture of taro-sugarcane will show comparable performance to sugarcane planted beds but taro CW shows better performance over mixed vegetation and sugarcane beds.

The planted beds showed that taro, sugarcane and their mixture (84 – 88%) had marginal edge over the unplanted (82 – 86%) in BOD_5 removal efficiencies (Figure 4.7). However, the pairwise tests revealed no statistical significant difference ($p>0.05$, 0.150 – 0.928) in the mean effluent BOD_5 concentrations from the four CWs under consideration (Appendix 2). COD removal performance were also influenced in a similar trend like the case of BOD_5. Taro, sugarcane and their mixture beds (72 – 84%) showed marginal edge over the unplanted (69 – 82%) and such occurrences among the beds were not statistically significant ($p>0.05$, 0.602 –

0.951). Thus, vegetation by marginal edge influenced BOD_5 and COD removal performance in CW without necessarily at significant levels. All planted beds showed better AnS removal efficiencies (53 – 73%) over the unplanted bed (42 – 60%) (Figure 4.7). There was significant difference in mean effluent AnS between vegetated CWs and unplanted CW ($p<0.05$, 0 – 0.045, see Appendix 2). However, among the vegetated beds only the performance of sugarcane and mixed vegetation beds differed significantly in the effluent AnS levels ($p=0.024$) with the sugarcane being the better of the two (1.65 against 2 mg-LAS/L, see Appendix 4). However, taro and sugarcane showed comparable performance with no significant difference in effluent AnS levels ($p=0.594$). The presence of indigenous vegetation (taro and sugarcane with their mixture) influenced the performance of the CWs in removing AnS, and sugarcane better effect on performance as a monoculture over a mixture with taro which tend to have comparable performance with the mixture.

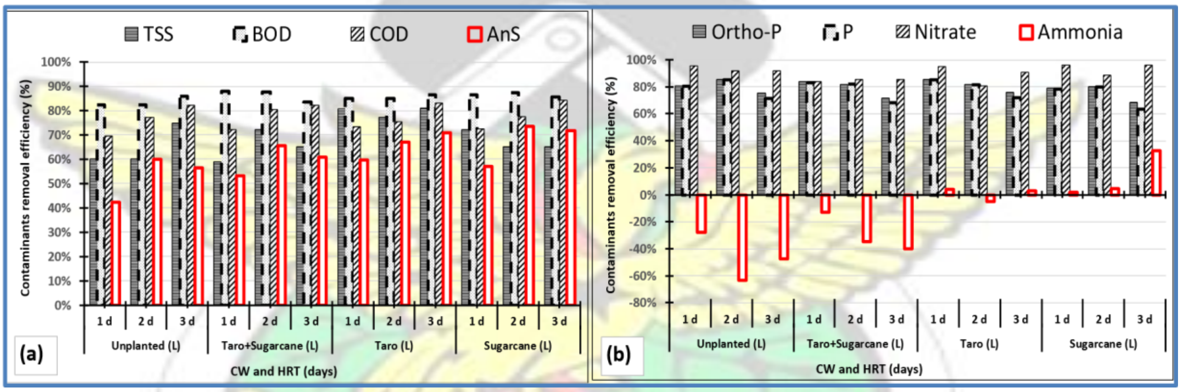

Figure 4.7 CW performance by vegetation (taro, sugarcane, and sugarcane+taro)
Note: Performance for (a) organic contaminants and (b) nutrients

The influence of vegetation on CW performance in removing Ortho-P/P, NO_3-N and NH_3-N were also tested as shown in Figure 4.7 and Appendix 2. There seemed to be no significant influence of vegetation on CW removal efficiencies for Ortho-P/P species ($p>0.05$, see Appendix 2). This is because the P removal efficiencies for vegetated (64 – 86%) and unplanted (71 – 85%) beds appeared comparable by the significance test. Similarly, vegetation may not be strong in influencing NO_3-N removal efficiencies in the CWs especially when vegetated and unplanted bed are compared. The unplanted bed showed an impressive efficiency. The results show that vegetated and unplanted beds are all comparable with no significance difference

($p>0.05$, 0.082 – 0.630). However, among the vegetation sugarcane appeared marginally better in removal efficiency (89 – 96%) than taro (81 – 95%) and significantly better than mixed vegetation (84 – 86%, $p=0.027$). In both cases of P and nitrate removal, the trend suggests that a significant role by the media (lateritebased) as was previously discussed cannot be ignored. On the contrary, vegetation influence on ammonia removal efficiencies was the most conspicuous among the three nutrient parameters. Apart from the mixed vegetation (taro+sugarcane) that showed comparable performance to the unplanted bed (-40% to -13% versus -64% to -28%), the monoculture taro (-5% to 4%) and sugarcane (2 – 33%) beds outperformed the unplanted with significance difference in effluent NH_3-N concentrations ($p<0.0001$, see Appendix 2). Also, there was significance difference in performance between the mixed vegetation and the two monoculture beds ($p<0.001$), but not between the two monoculture beds – taro and sugarcane ($p=0.258$). The indigenous vegetation (taro and sugarcane) could influence CW performance on P and NO_3-N removal but that could be undermined by bed media especially when it is laterite-based. The influence of vegetation on CW for NO_3-N and NH_3-N removal performance is pronounced especially for taro and sugarcane monoculture beds.

4.5.2.4 Influence of baffle-partitions on the performance of CW

The pairwise tests were carried out on CW cells with the same media and vegetation with only difference in the presence (baffled) or absence (unbaffled) baffle partitions. For laterite-based media, baffle partitioned bed had slightly better performance with the removal of organic contaminants TSS (66 – 79% against 59 – 72%), BOD_5 (88 – 89% against 84 – 88%), COD (82 – 84% against 72 – 82%), and AnS (60 – 72% against 53 – 66%) (see Figure 4.8 and Appendix 3). For the gravel beds, the unbaffled and baffled beds were conspicuously comparable for TSS, BOD and COD removal efficiencies which were confirmed by statistical significant tests $p>0.05$ (i.e. 0.489 – 0.903). For AnS, the baffled condition (54 – 75%) showed an edge over the unbaffled (55 – 67%) although the difference was not statistically significant ($p=0.098$). The influence of baffle partitions on the performance of laterite-based CW was only significant for the removal of AnS ($p=0.028$), The other significant tests ($p>0.05$, see Appendix 3) indicated that both unbaffled and baffled conditions gave comparable efficiencies for TSS, BOD_5 and COD removal. More likely, the results could be influenced largely also by the media as previously discussed.

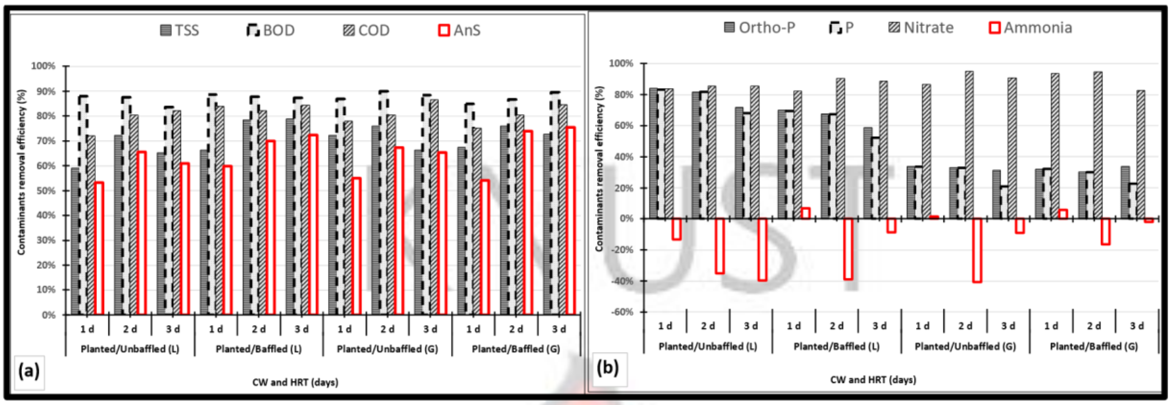

Figure 4.8 CW performance according to baffle-partitions
Note: Performance for (a) organic contaminants and (b) nutrients

Baffle partition influence on removal efficiencies of the CWs were not significant for any of the nutrient contaminants Ortho-P/P, NO_3-N and NH_3-N and the best was a marginal edge over the unbaffled condition (see Figure 4.8 and Appendix 3). For Ortho-P/P, unbaffled condition gave significantly better removal efficiencies than the baffled case in the laterite-based CWs (68 – 84% against 52 – 70%, p=0.038). However, the gravel beds gave comparable Ortho-P/P removal efficiencies between baffled and unbaffled conditions (21 – 34% against 23 – 34%, p=0.881). In the case of NO_3-N removal efficiencies, the baffled laterite-based CW gave marginal edge over the unbaffled case (82 – 91% against 84 – 86%) although that difference was not significant (p=0.359). The situation for gravel beds is that of comparable performance between baffled and unbaffled conditions (86 – 95% against 83 – 94%, p=0.325).

Although NH_3-N removal efficiencies are generally low for the design CWs, baffle partition influence on CW performance could be seen marginally in both gravel and laterite-based media beds. The baffled conditions in the two media cases marginally outperformed over the unbaffled beds (Laterite-based beds: -39% to 7% against 40% to -13%; grave beds: -16% to 6% against -41% to 1%) although the differences were not statistically significant (p>0.05, 0.109 – 0.277). Baffle partitions marginally influence the performance of CW especially in removing nutrient contaminants and such influence could be weak to significantly affect removal efficiencies.

4.6 Prediction models for effluent BOD_5, COD and AnS

The statistical summaries on fitted models, their ANOVA and coefficients tables generated for the three contaminants (as cases) are presented in Tables (Table 4.32, Table 4.33, Table 4.34, Table 4.35, Table 4.36, Table 4.37, and Table 4.38). All selected models generated were statistically significant, indicating that the models rightly predicted the effluent contaminants, and thus rejecting the null hypothesis that coefficients of the models and their R^2 are equal to zero (Denis, 2019). From the results, the last model of each case analysis appeared to be better fit, showing higher multiple regression coefficient R^2 and less error estimate than all their preceding models. Thus, the selected models in each case showed increased R^2 and decreased Standard Error of the Estimate (SEE) over preceding models.

For instance, in Table 4.32 (models for BOD_5), the R^2 increased from 0.247 to 0.368 and SEE decreased from 9.84 to 9.07 from model 1 to 2; Table 4.33 (model for COD) R^2 increased from 0.532 to 0.621 and SEE decreased from 24.53 to 22.46 (from model 1 to 3); and in Table 4.34 (model for AnS) R^2 increased from 0.236 to 0.729 and SEE decreased from 0.44 to 0.28 (from model 1 to 5). Again, the significant ANOVA tests also supported the rejection of null hypothesis that the multiple R in the population data is equal to zero. Also the collinearity statistics VIF (variable inflation factor) in all cases (Table 4.35, Table 4.36, and Table 4.37) was all far below the threshold of 10 and therefore indicated no potential collinearity problems (Field, 2013; Denis, 2019).

The selected models are therefore written down as regression equations (Table 4.38) based on the models' coefficients generated (Table 4.35, Table 4.36, and Table 4.37). All the regression coefficients were significant, i.e. rejected the null hypothesis that they were equal to zero. Only the constants for COD and AnS models showed nonsignificance ($p>0.05$), thus they were not different from zero, likely regressions lines approximately pass through the origin. However, the constants are kept in the equations because of the following: constants are not predictors and have unclear real interpretations in the absence of dummy variables; constants absorb unaccounted for biases; and constants guarantee a zero mean which is a key assumption for the residual analysis (Ogee et al., 2013).

The three prediction models (Table 4.38) account significantly for variances (37 – 73%) in the predicted parameters (BOD_5, COD & AnS) than by just chance. Also, the residual analysis showed that all fitted models gave residuals that were normally distributed and this was confirmed by the Shapiro-Wilk tests with $p>0.05$ (Table 4.32, Table 4.33 and Table 4.34).

Table 4.32 Multiple linear regression model for effluent BOD_5

Model[c]	R	R Square	Adjusted R Square	Std. Error of the Estimate	Change Statistics				
					R Square Change	F Change	df1	df2	Sig. F Change
1	.497[a]	.247	.239	9.84049	.247	28.921	1	88	.000
2	.607[b]	.368	.354	9.06792	.121	16.634	1	87	.000

a. Predictors: (Constant), NO_3-Ni; with ANOVA – $F(1, 88)=28.921$, $p<0.001$
b. Predictors: (Constant), NO_3-Ni, BODi:CODi; with ANOVA – $F(2, 87)=25.346$, $p<0.001$
c. Shapiro-Wilk (SW) normality test for model residuals, $SW(90)=0.977$, $p=0.107$
Dependent Variable: BOD_5

Table 4.33 Multiple linear regression model for effluent COD

Model[d]	R	R Square	Adjusted R Square	Std. Error of the Estimate	Change Statistics				
					R Square Change	F Change	df1	df2	Sig. F Change
1	.729[a]	.532	.524	24.52657	.532	65.828	1	58	.000
2	.761[b]	.579	.564	23.46360	.047	6.374	1	57	.014
3	.788[c]	.621	.600	22.46267	.042	6.193	1	56	.016

a. Predictors: (Constant), NO_3-Ni; with ANOVA – $F(2, 58)=65.825$, $p<0.001$
b. Predictors: (Constant), NO_3-Ni, AnSi; with ANOVA – $F(2, 57)=39.151$, $p<0.001$
c. Predictors: (Constant), NO_3-Ni, AnSi, CODi; with ANOVA – $F(3, 56)=30.543$, $p<0.001$
d. Shapiro-Wilk (SW) normality test for model residuals, $SW(60)=0.968$, $p=0.112$
Dependent Variable: COD

Table 4.34 Multiple linear regression model for effluent AnS

Model[f]	R	R Square	Adjusted R Square	Std. Error of the Estimate	Change Statistics				
					R Square Change	F Change	df1	df2	Sig. F Change
1	.486[a]	.236	.221	.44349	.236	14.864	1	48	.000
2	.610[b]	.372	.346	.40634	.136	10.178	1	47	.003
3	.764[c]	.584	.557	.33451	.211	23.352	1	46	.000
4	.834[d]	.696	.669	.28910	.112	16.585	1	45	.000
5	.854[e]	.729	.698	.27592	.033	5.403	1	44	.025

a. Predictors: (Constant), NO_3-Ni; with ANOVA – $F(1, 48)=14.846$, $p<0.001$
b. Predictors: (Constant), NO_3-Ni, Ortho-Pi; with ANOVA – $F(2, 47)=13.942$, $p<0.001$
c. Predictors: (Constant), NO_3-Ni, Ortho-Pi, CODi; with ANOVA – $F(3, 46)=21.499$, $p<0.001$
d. Predictors: (Constant), NO_3-Ni, Ortho-Pi, CODi, HRT; with ANOVA – $F(4, 45)=25.734$, $p<0.001$
e. Predictors: (Constant), NO_3-Ni, Ortho-Pi, CODi, HRT, AnSi; with ANOVA – $F(5, 44)=23.651$, $p<0.001$
f. Shapiro-Wilk (SW) normality test for model residuals, $SW(50)=0.981$, $p=0.591$
Dependent Variable: AnS (anionic surfactants)

Table 4.35 Coefficients of the regression model fitted for effluent BOD_5

Model		Unstandardized Coefficients B	Std. Error	Standardized Coefficients Beta	t	Sig.	95.0% Confidence Interval for B Lower Bound	Upper Bound	Correlations Zeroorder	Partial	Part	Collinearity Statistics Tolerance	VIF
1	(Constant)	16.761	2.380		7.043	.000	12.032	21.489					
	NO_3-Ni	2.076	.386	.497	5.378	.000	1.309	2.844	.497	.497	.497	1.000	1.000
2	(Constant)	48.438	8.071		6.002	.000	32.396	64.479					
	NO_3-Ni	2.107	.356	.505	5.920	.000	1.400	2.814	.497	.536	.505	1.000	1.000
	BODi:CODi	-47.348	11.609	-.348	-4.078	.000	-70.422	-24.273	-.337	-.401	-.348	1.000	1.000

Dependent Variable: BOD_5

Table 4.36 Coefficients of the regression model fitted for efflue COD

Model		Unstandardized Coefficients B	Std. Error	Standardized Coefficients Beta	t	Sig.	95.0% Confidence Interval for B Lower Bound	Upper Bound	Correlations Zeroorder	Partial	Part	Collinearity Statistics Tolerance	VIF
1	(Constant)	16.215	7.348		2.207	.031	1.506	30.924					
	NO_3-Ni	9.489	1.170	.729	8.113	.000	7.148	11.830	.729	.729	.729	1.000	1.000
2	(Constant)	34.414	10.069		3.418	.001	14.252	54.576					
	NO_3-Ni	11.906	1.473	.915	8.085	.000	8.957	14.854	.729	.731	.695	.577	1.732
	AnSi	-6.359	2.519	-.286	-2.525	.014	-11.403	-1.315	.309	-.317	-.217	.577	1.732
3	(Constant)	17.773	11.731		1.515	.135	-5.728	41.274					
	NO_3-Ni	9.660	1.674	.742	5.771	.000	6.307	13.013	.729	.611	.475	.409	2.442
	AnSi	-9.644	2.749	-.433	-3.508	.001	-15.151	-4.137	.309	-.424	-.289	.444	2.251
	CODi	.145	.058	.356	2.489	.016	.028	.263	.600	.316	.205	.331	3.019

Dependent Variable: COD

Table 4.37 Coefficients of the regression model fitted for effluent AnS

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.	95.0% Confidence Interval for B		Correlations			Collinearity Statistics	
		B	Std. Error	Beta			Lower Bound	Upper Bound	Zeroorder	Partial	Part	Tolerance	VIF
1	(Constant)	1.107	.144		7.677	.000	.817	1.396					
	NO_3-Ni	.087	.023	.486	3.855	.000	.042	.133	.486	.486	.486	1.000	1.000
2	(Constant)	.666	.191		3.483	.001	.281	1.050					
	NO_3-Ni	.078	.021	.435	3.728	.001	.036	.120	.486	.478	.431	.981	1.019
	Ortho-Pi	.044	.014	.372	3.190	.003	.016	.071	.432	.422	.369	.981	1.019
3	(Constant)	1.124	.184		6.119	.000	.754	1.494					
	NO_3-Ni	.109	.018	.608	5.933	.000	.072	.146	.486	.658	.564	.861	1.162
	Ortho-Pi	.094	.015	.807	6.132	.000	.063	.126	.432	.671	.583	.522	1.915
	CODi	-.004	.001	-.672	-4.832	.000	-.005	-.002	.091	-.580	-.460	.468	2.136
4	(Constant)	.073	.303		.239	.812	-.538	.683					
	NO_3-Ni	.126	.016	.701	7.664	.000	.093	.159	.486	.752	.630	.807	1.239
	Ortho-Pi	.125	.015	1.068	8.181	.000	.094	.156	.432	.773	.673	.397	2.521
	CODi	-.005	.001	-.869	-6.709	.000	-.006	-.003	.091	-.707	-.552	.403	2.484
	HRT	.387	.095	.389	4.072	.000	.196	.578	-.072	.519	.335	.741	1.350
5	(Constant)	-.435	.362		-1.200	.237	-1.165	.296					

NO_3-Ni	.162	.022	.904	7.328	.000	.118	.207	.486	.741	.575	.405	2.471
Ortho-Pi	.161	.021	1.376	7.561	.000	.118	.204	.432	.752	.593	.186	5.382
CODi	-.004	.001	-.785	-6.097	.000	-.006	-.003	.091	-.677	-.478	.371	2.696
HRT	.521	.108	.524	4.847	.000	.305	.738	-.072	.590	.380	.526	1.900
AnSi	-.109	.047	-.469	-2.324	.025	-.203	-.014	.419	-.331	-.182	.151	6.619

Dependent Variable: AnS (anionic surfactants)

For convenient use of the models as tools for design and data assessments like commonly found in literature (Rousseau et al., 2004; Dotro et al., 2017), the corresponding input and output data ranges have been defined for the regression equations (Table 4.38). This table is relevant for guidance because regression equations are only useful within the data ranges used to build them (Son et al., 2010). By the multiple regression coefficients, the strongest model is likely that for predicting AnS because of its high ability to predict 73% of variance in the effluent anionic surfactant levels, followed by COD and then BOD. The low R^2 especially with BOD model signals caution and circumspect in its application for long-term predictions (Son et al., 2010).

The AnS model is defined by the most number of predictors about five in number, and the others have three each. The model for BOD_5 rather fitted the BOD/COD ratio instead of the individual carbon species, probably because the ratio is an explicit expression of biodegradability, more meaningful and directly relates to biological degradation of BOD. The commonest predictors for all three predicted effluents are NO_3-N and COD, followed by AnS. Nitrate may be prominent in the models probably because it is a significant nitrogenous nutrient species that attempted to balance the necessary influent C/N ratios (Singh et al., 2017; Jia et al., 2018). COD is usually a large source of organic carbon which is necessary for the removal of other contaminants or substances in wetlands (Jia et al., 2018). Another key observation is the missing out of BOD_5 on effluent COD model (model 2, Table 4.38). The expectation was that influent BOD_5 or its ratio form with COD would be fitted, however, it failed and a counterpart organic contaminant AnS was rather successful. Probably, AnS better represented readily biodegradable organics because LAS in a sense is also tagged as more biodegradable (Scott and Jones, 2000).

The model for AnS (model 3, Table 4.38) needed influent nutrient phosphorus in addition to nitrate nitrogen to stimulate biological treatment in the wetlands, a probable attempt to establish a classical requisite biochemical balance between biodegradable organics and nutrients in a C/N/P ratio (Boyjoo et al., 2013). In the same model, HRT became relevant probably because removal of AnS was more sensitive to the time that predictors would spend together in a wetland bed.

The models are also associated with suppressors which are key in "removing irrelevant variance from predictors, increasing their regression weight, and also increasing overall predictability of the model" (Pandey and Elliott, 2010; Klein, 2014). These included influent BOD/COD ratio in model 1, influent COD in model 2 as a suppressor to AnS, and influent AnS model 3 as a suppressor to all predictors except COD (Table 4.38).

Table 4.38 Summary of effluent prediction models with input/output ranges

Model number	**Parameter**	**Regression model/Equations**	***Input range (mg/L)**	***Output range (mg/L)**	**R^2**
1	BOD_5	BOD_5 = 48.438 +2.107(NO_3-Ni)–47.348 (BODi/CODi)	≤ NO_3-Ni ≤10.40 1.110 0.54 ≤BODi/CODi ≤0.84	8 ≤BOD_5 ≤57	0.37
2	COD	COD =17.73+9.66(NO_3-Ni) - 9.644(AnSi) + 0.145(CODi)	1.10 ≤ NO_3-Ni ≤9.30 2.15 ≤AnSi ≤7.65 52 ≤CODi ≤458	9 ≤COD ≤145	0.62
3	AnS	AnS=0.162(NO_3-Ni)+0.161(Ortho-Pi)-0.004(CODi)+0.521(HRT)-0.109(AnSi)-0.435	1.8 ≤ NO_3-Ni ≤10.40 5.1 ≤Ortho-Pi ≤17.34 152 ≤CODi ≤505 1 ≤HRT ≤3 (days) 1.49 ≤AnSi ≤8.70	0.52 ≤AnS ≤2.90	0.73

Note: *All units are in mg/L unless otherwise stated

Chapter 5: General Discussions

5.1 Characteristics of greywater in Ghana

At least some reasonable amount of data is generated through desk and literature reviews, and laboratory analysis of greywater samples from the field. The scope of greywater source data included drains of a sewered and non-sewered urban communities, residential facilities, channels to lagoons, hotel and hostels, and others. Some of the key characteristics of Ghanaian greywater are briefly discussed.

The pH of greywater is generally within the range mostly reported, 6 to 10 – slightly acidic to alkaline but meet the Ghana's Environmental Protection Agency's (EPAGh) discharge guideline limit (El-Fadl, 2007; Mohamed et al., 2019; EPA-Gh, n.d.). For electrical conductivity (EC, 280 – 17,100 µS/cm), some greywater sources pass discharge limits with ≤1000 µS/cm but others fail it (1500 µS/cm, (EPA-Gh, n.d.)). The trend for total dissolved solids (TDS) which ranges between 170 and 2,860 mg/L, largely followed that of the EC with some greywater sources failing the EPAGh discharge limit of 1000 mg/L.

The dissolved oxygen (DO) levels as expected are not high (0.1 – 2 mg-O/L) but far below the 4 mg-O/L recommended level for water environment, and therefore Ghanaian greywater could have negative impacts on biota when released into water courses as asserted by the National Academies of Sciences, Engineering, and Medicine (NASEM, 2015).

The TSS (70 – 4,720 mg/L) and turbidity (40 – 2,880 NTU) levels varied widely across greywater sources. Very few sources comply with the EPA-Gh discharge limit for turbidity (75 NTU). For TSS, there is total failure with the discharge limit of 50 mg/L (EPA-Gh, n.d.) for all sources. High levels of greywater TSS and turbidity, perhaps could be due to the presence of solids, fabric softeners, and detergent residues (Mohamed et al., 2019).

The BOD_5 (60 – 700 mg/L) and COD (200 – 2,300 mg/L) contents exhibit high variability commonly associated with greywater (Sievers and Londong, 2018). The

BOD_5 and COD values are high and generally exceed the EPA-Gh discharge limits (50 mg-BOD_5/L and 250 mg-COD/L). The biodegradability state (BOD_5/COD ratio) is in a wide range from 0.12 to 0.62 but the average could still meet the reference level of 0.5 (Kulabako et al., 2011). The understanding is that greywater with low biodegradability ratios are generally rich in chemical contaminants like nonbiodegradable surfactants, detergents, etc, and could be generated by people with low water consumption behaviour as well (El-Fadl, 2007; Boyjoo et al., 2013; Mohamed et al., 2019). It is also noted that EPA-Gh's BOD_5 and COD discharge thresholds do not favour potential environmental biodegradability because the BOD/COD ratio is only around 0.20, far below the minimum 0.5 biodegradability reference point. Howbeit, it is nothing more can be deduced from that because the basis for EPA-Gh choosing the separate discharge limits for BOD and COD are not known.

Ghanaian greywater also contains nutrients (nitrogen and phosphorus), with appreciable levels of nitrate (0.5 – 13 mg-NO_3-N/L), nitrite (0 – 58 mg-NO_2-N/L), ammonia (2 – 15 mg-NH_3-N/L), Total Kjeldahl Nitrogen (TKN, 7 – 220 mg/L), total phosphorus/phosphate (TP, 1 – 72 mg/L). Some greywater sources fail the EPA-Gh discharge limits for nitrate (11.5 mg/L), ammoniacal nitrogen (NH_3/NH_4 1 – 1.5 mg/L) and phosphorus (2 mg/L) sometimes in high excess. High levels of ammoniacal nitrogen are mostly associated with fresh greywater, suggesting that little or no nitrification has occurred (Mohamed et al., 2019). The phosphorus and phosphate in greywater are connected to the use of household detergents, etc, while the nitrogenous components come mainly from cationic surfactants – e.g., fabric softeners and laundry disinfectants (El-Fadl, 2007; Li et al., 2008; Widiastuti et al., 2008; Mohamed et al., 2019).

Moving beyond discharge limits, knowledge of greywater organic contaminants and nutrients loads is critical for decisions on treatment options. The desirable COD:N:P ratio for biological treatment is 100:20:1 (Boyjoo et al., 2013; Mohamed et al., 2019). Those for the sampled greywater could be between 7:1:1 and 55:5:1 and these are too low for balanced biologically treatable greywater. However, most greywater by the simpler biodegradability ratio in terms of BOD_5/COD ($\geq$ 0.5) is potential positive (Kulabako et al., 2011).

The microbial contaminants which include total coliforms, faecal coliforms and *E. coli* could be very high with ranges around 1 to 10 log CFU/100ml, similar to the pathogenic loads reported in other studies (De Gisi et al., 2016; Shi et al., 2018). It is therefore not surprising that almost all the greywater sources are failing the EPA-Gh limits for *E. coli* (1 log CFU/100ml) and total coliforms (2.6 log CFU/100ml). Meanwhile, potential sources of faecal coliforms in greywater could be linked to greywater flows from bathrooms, laundries, kitchen sinks and dishwashers – e.g., washed off clothing, hands, diapers, childcare items, etc (Ottoson and Stenstrom, 2003; Gilboa and Friedler, 2008). The findings indicate that greywater flows in Ghana contain similar infectious agents to those in other studies, and the diversity and concentrations depend on the greywater sources, health status and number (diversity) of waste generators, and the geographic location and its seasonality (Mohamed et al., 2019).

Greywater also contains some cationic species – e.g., calcium and magnesium – and heavy metals (copper, cadmium, iron, lead, mercury, manganese and zinc) at varying concentrations (0 – 2 mg/L). However, for those macro elements with EPA-Gh specified discharge limits, their levels are generally low – e.g., 2.5 mg-Cu/L, <0.1 mg-Cd/L, 0.1 mg-Pb/L, and 5 mg-Zn/L. In some instances, lead and mercury could be around 0.30 mg/L, exceeding the discharge limits by about three- to eight-folds. The sources of such heavy metals are domestic commodities and building materials including plumbing materials, cutlery, jewellery, coins, home maintenance products, etc (Eriksson and Donner, 2009). Similar macro-element concentration levels are found in greywater sources world-wide (Mohamed et al., 2019). No guideline limits existed for Ca and Mg but concerns for their levels could be implied through the sodium adsorption ratio (SAR). By the World Health Organization (WHO) and Food Agriculture Organization (FAO) guidelines on non-restrictive reuse of wastewater, a SAR of 0 – 3 $(meq/L)^{1/2}$ implies that the levels of Ca, Mg and Na are safe, especially for the soil environment (FAO, 1992; WHO, 2006a). For a greywater source with data to compute its SAR, the value of 0.6 is a clear indication that the elemental levels are safe for greywater reuse.

Anionic surfactants (AnS) and sulphates (SO_4^{2-}) are also found in Ghanaian greywater as expected because of household products like detergents, soaps, and other personal care products usually consumed (De Gisi et al., 2016). The surfactants (AnS) expressed as Linear Alkylbenzene sulfonate (LAS) levels could be between 2 and 10 mg-LAS/L while the SO_4^{2-} levels could be between 1 and 29 mg/L. Sulphate levels are far below the EPA-Gh discharge limits of 250 – 300 mg-SO_4^{2-}/L. The EPA-Gh has no discharge limits for anionic surfactants, a micropollutant commonly found in detergents (De Gisi et al., 2016) but their levels are low compared to figures from European settings (7 – 436 mg/L) (Noutsopoulos et al., 2018). Possibly cleansing and personal care products used by households did not contain much sulphate and anionic surfactants, and/or less of such products are used. It is also possible that some laundry greywater sources could be diverted from the sampled flows. The levels of AnS may be low but not necessary safe because they are higher than most effluent from treatment systems (0.09 – 0.9 mg-LAS/L) (Scott and Jones, 2000), and may also be unsafe for most aquatic and soil organisms sensitive to exposure levels of <5 mg-LAS/L (EOSCA, 2000; Abd El-Gawad, 2014).

Generally, most greywater sources in Ghana could be classified as high strength (dark) greywater with BOD >300 mg/L, COD >630 mg/L and nutrients P >2 mg/L and N >17 mg/L (Boyjoo et al., 2013). Clearly, data on greywater characteristics in Ghana is seeing initial improvements, and this is positive for finding solutions to safe management (Kivaisi, 2001; Noutsopoulos et al., 2018; Mohamed et al., 2019).

On generation rates, the literature search and fieldwork about eight data points: 32, 73, 36, 43, 98, 100, 54 and 53 l/c/d. The confidence level analysis gives generation rates in the range 39 – 83 l/c/d (95% CI) for Ghana. The generation rates although comparable to other developing countries, they are still low probably due to generally low water consumption and service levels.

5.2 Use of indigenous vegetation in greywater disposal practices

The surveyed communities demonstrate that major greywater sources namely kitchen, bathroom and laundry are disposed of mainly into open spaces within compounds (46–66%), and use of septic tank and soakaway systems (4–24%). Greywater disposal into the open is mainly from the kitchen (46%, *n*=208), and laundry (66%, *n*=296) sources. The only improved or appropriate disposal practices are the use of soakage pits (0.2–23%) and septic tanks (0.2–2.4%) systems. The existing practices from these communities are basically not different from the account given in literature about general haphazard and poor approaches to greywater disposal in Ghana (GSS, 2013b; Hyde and Maradza, 2013; DwumfourAsare et al., 2017; Oteng-Peprah et al., 2018b). Clearly, the minimum requirement for greywater disposal which is by the use of at least soakaways as defined by local authorities' sanitation bylaws and the national building regulations (Fosu, 1996; Ghana Local Government, 1998), are not being followed, partly due to weak enforcement and other constriants (WHO, 2012; Antwi-Agyei et al., 2019).

The open space disposal options included vegetated open fields or niches within compounds, existing as walls/fences or at the backyard of houses where the plants are watered or irrigated directly or indirectly. Apart from the greywater being used to water plants, other uses identified (without pre-treatment) include cleaning/scrubbing floors, and watering down dust within the compound of the house. These kinds of reuses are encouraged but need some restrictions because greywater is not pretreated and associated risks from contaminants could be high (WHO, 2006a; OtengPeprah et al., 2018a; Oteng-Peprah et al., 2018b).

The 451 home-respondents use a considerable number and diverse range of plant groups numbering close to 1,260 coming from about 36 different plant species in their greywater disposal. Several of these plants e.g. sugarcane, taro, cocoyam, and lemon grass are identified as potential candidate macrophytes for application in greywater treatment technologies such as constructed wetlands (Bindu et al., 2008; Mateus et al., 2014; Madera-Parra et al., 2015; Rana and Maiti, 2018). These are local vegetation which include some food crops, are planted in the homes as mix (multicultural) vegetation systems just like commonly found in local natural wetlands (Campion and Venzke, 2011). These are also planted in laterite soils generally found in forested warm tropical regions like Ghana (especially Forest Ochrosol soil class) (Gidigasu, 1972; Asamoa, 1973).

It is also very encouraging to know that people would be willing to use and possibly pay some small amount like GHS 100 (about US$ 20) for a technology that is an improvement over their existing practices if the value-added benefits would be maintained or made better. This keen interest from respondents support the need to indigenize a feasible technology such as phytoremediation and specifically constructed wetlands that relies on vegetation to explore greywater treatment in Ghana.

5.3 Performance of experimental scale constructed wetlands for greywater treatment

The general prospects of the treatment system developed and tested look very promising. The pre-treatment sedimentation with screening was useful in removing some key contaminants like for turbidity, TSS, BOD_5, COD, Ortho-P & P, NO_3-N, NO_2-N, SO_4^{-2}, AnS and Fe to significant levels making the whole process relevant as asserted in literature (Cooper and Findlater, 1990; Oteng-Peprah et al., 2018a). For instance, the levels of some key contaminants dropped between 18 and 75%, and the least was phosphorus (18 – 21%), then anionic surfactants (AnS) 20%, BOD_5 (26%), turbidity (31%), COD (33%), nitrate (43%), sulphate (63%), and TSS (64%). However, major contaminants like turbidity, TSS, BOD_5, COD, P, and NH_3-N fail to meet the available discharge limits and therefore warrant further treatment. Although no guideline limits exist for AnS, the levels were high (≥ 5 mg-LAS/L) compared to expectations from literature 0.09 – 0.9 mg-LAS/L (Scott and Jones, 2000), and also unsafe for sensitive ecological organisms especially to those intolerant at <5 mgLAS/L (EOSCA, 2000; Abd El-Gawad, 2014).

Meanwhile the sedimentation pre-treatment marginally improved the average biodegradability BOD/COD ratio from 0.60 to 0.66 which is good for a biological treatment system like CW being applied for further treatment. It is also noted that the pH of pre-treated greywater was lower and slightly acidic (pH 6.63) than the raw (pH 6.96), likely as a result of biochemical activities by microbes and enzymes (Bijalwan and Bijalwan, 2016). A significant drawback with this pre-treatment stage is the large drop in DO levels by 78%, a clear phenomenon of oxygen depletion through consumption for aerobic degradation and nitrification (Masi, 2004; Mthembu et al., 2013; Wu et al., 2016). Also, the parameters EC, TDS and NH_3-N saw increased levels between 9 and 61%.

Effluent levels for EC, DO, TDS, and Fe always increased while NH_3–N levels showed both decreases and largely increases at some instances. Dropped acid levels and increased DO levels could be attributed to interactions (biochemical and physical processes) that naturally occur in the wetland beds involving the activities of plants and microbes (USEPA, 2003). Increases in TDS and EC levels are known in literature (Gupta and Nath, 2018), and such observations are linked to salts and dissolved organics that are released from bed media and biochemical activities. Meanwhile, the two parameters TDS and EC increased at the same levels because of their potential linear relationships (Thirumalini and Joseph, 2009; Oteng-Peprah et al., 2018a). For NH_3–N, the general increases in effluent levels were pronounced with increasing HRT especially at 2 days. This could have emerged from organic nitrogen conversion to ammonia (ammonification) rather than nitrification more likely due to relatively low DO (Ling et al., 2009). However, the monoculture sugarcane planted wetland bed shows consistent increased ammonia removal along the HRT likely due to sugarcane high preference for NH_3-N uptake (Boschiero et al., 2018).

Turbidity removal was always significantly high from the CW with performance range of 94 – 99% including control (unplanted) beds. The laterite-based CW generally performed better for all HRT and this could be linked to the high components of sand, silt and clay with potential large particulate surface area and comparatively low porosity which contribute to improved filtering property. The turbidity removal efficiencies in the current study could be among the best reported (Arden and Ma, 2018). Meanwhile, effluent iron levels from all systems at all instances did not meet the EPA-Gh's discharge limit of <0.1 mg-Fe/L including the gravel beds (with effluent 0.3 – 0.7 mg/L). High Fe releases were associated with laterite-based beds in excesses of several times more than the 1 mg/L safe level recommended for freshwater aquatic life (Vuori, 1995). While effluent Fe mainly came from CW bed media including gravels, contributions from vegetation may not be ignored either (Vose, 1982; USEPA, 2003). This is because there were instances where planted beds gave higher iron levels than their controls (unplanted beds).

SO_4^{2-} is almost always 100% removed with isolated low removal efficiencies around 50 – 93%. The low performance is mostly associated with more residence time of 3 days in planted gravel beds (50% and 91%), and a planted laterite-based bed (73%). It is expected that more HRT from the 3 days will provide potential extensive anaerobic sulphate-reduction in the beds (Cooper and Findlater, 1990; ZapaterPereyra et al., 2014) but it is the contrary. Similarly, NO_2-N removal patterns followed the trend of SO_4^{2-} quite closely with efficiencies around 29 – 100%. The low efficiencies (29 – 66%) come from the gravel bed systems, and a single lateritebased bed (50 – 77%) with baffle-partitions which in few instances behave like a gravel bed. However, low removal efficiencies might not necessarily generate significant worries because the effluent levels are generally low (0 – 0.01 mg-NO_2N/L) even though there is no discharge limit from EPA-Ghana's guideline.

The following major contaminants were also significantly removed by the CW microcosm models: TSS (59 – 81%), BOD_5 (77 – 90%), COD (69 – 86%), NO_3-N (81 – 96%), PO_4 (24 – 86%), and AnS (42 – 75%). On nitrate and phosphate, there could be the phenomena of high nitrate uptake by plants and gaseous loss via nitrification, and slow P sorption and adsorption in the bed matrix (Ling et al., 2009).

The current study shows quite competitive performance that is comparable to efficiencies reported in literature for constructed wetlands - TSS (25 – 98%), BOD_5 (63 – 99%), COD (81 – 82%), TP (24 – 74%), TN (44 – 59%), and AnS (23 – 90%) (Ling et al., 2009; Reyes-Contreras et al., 2012; Arden and Ma, 2018; Gupta and Nath, 2018; Oteng-Peprah et al., 2018a; Pérez-López et al., 2018). This means that the designed CW may be proposed for greywater treatment in Ghana to a comparable standard performance when further refined by pilot studies.

The range of oxygen transfer rates (OTRs) among wetlands are wide -0.95 to 34 gO_2/m^2d, and inconsistent, without clear trends. The OTRs sometimes fail the recommended 20 g-O_2/m^2d for subsurface constructed wetlands (Randerson, 2006) except for the monoculture wetlands of taro and sugarcane plants. Meanwhile, for the planted wetlands, none operated below the least possible transfer rate of 5 g-O_2/m^2d (Randerson, 2006) but the controls are almost always worst likely due to the absence of plants (Stein and Hook, 2005; Randerson, 2006). It is not very clear what is actually accounting for such inconsistencies, however, the results suggest that vegetation could contribute to better transfer oxygen especially when a particular macrophyte is planted alone. It is not yet known if allelopathic influence, which is "effect(s) of one plant on others through the release of chemical compounds into the environment" (Bhadoria, 2011), could be a potential factor in the case of mixed vegetated wetland bed (Asao et al., 2003; USEPA, 2003; Chou, 2010; Rout and Sahoo, 2015).

MANOVA comparison test on effluent TDS, TSS, BOD_5, COD, Ortho-P, P, NO_3-N, NH_3-N and AnS using Pillai's statistics showed significant influence of the presence of plants and media on effluent quality ($p<0.0001$). Planted beds outperformed the unplanted CWs by giving comparatively lower effluent concentrations of contaminants except NO_3-N. For the other eight effluent quality parameters, planted beds performed better than unplanted beds although the differences were mostly statistically insignificant except for BOD_5, NH_3-N and AnS ($p<0.0001$). The findings in this study contradicts the assertion that vegetation may be unnecessary for effective removal of contaminants in CWs (Mara, 2003), although the type of macrophyte may play a key role in this functionality. For instance, it is known that sugarcane has high preference for NH_3-N uptake to other nitrogenous nutrient species that could be a desirable advantage in CW (Boschiero et al., 2018).

The performance CW beds in terms of media type showed that laterite-based beds gave comparatively lower effluent concentrations for Ortho-P/P, AnS and BOD_5 than the gravel beds. Fe rich media like laterite, the Fe acts as natural capping agent by improving P-binding capacity of substrates to remove Ortho-P/P and also plays active role in the removal of organics (Bakker et al., 2015; Yang et al., 2018; Wu et al., 2019). However, apart from the effluent Ortho-P/P which gave significant mean difference between the two media types ($p<0.0001$), the rest (BOD_5 & AnS) was not statistically significant ($p>0.05$, 0.364 – 0.832). Also, gravel beds performed better with lower effluent concentrations for the remaining five contaminants (TDS, TSS, COD, NO_3-N and NH_3-N) but with no statistically significance mean differences between the two media groups ($p>0.05$) except in the removal of TDS ($p<0.0001$).

5.4 Influence of key design features on the performance of CW

A two-way MANOVA tests showed significant ($p<0.001$) influence of CW cells and HRT on effluent quality. Thus, there were differences among the wetland microcosm models and likewise among HRT levels. However, the interactions between wetland cells and HRT did not prove to be significant ($p=0.486$) enough to jointly influence effluent water quality. Moreover, using the Pillai's Multivariate test statistics because the Box M test was significant (Nimon, 2012), it is observed that about 26% variability in effluent quality could be attributed to the wetland cells while HRT could account for about 22% of same. The implication is that the different wetland cells did not have the same performance levels on the effluent qualities, and likewise the level of influence from the different HRTs (Reyes-Contreras et al., 2012; Morató et al., 2014).

Therefore, any assumption that the performance of wetland cells will be same across board is not valid, especially for at least 6 of 9 tested major contaminants (TDS, BOD_5, Ortho-P, P, NH_3-N, and AnS) in treated effluent. Also, any assumption that different HRTs will have equal impact on wetland effluent quality levels is not valid for at least 3 of 9 tested contaminants (TDS, COD, and AnS). Again, it is also observed that two other effluent water qualities came close to be statistically influenced by the wetlands (NO_3-N, $p=0.055$), and by the HRTs (NH_3-N, $p=0.093$). Probably more factors other than the considered wetland features (media, vegetation presence/types, and baffle-partitions) and HRT levels may have contributed to these mixed trends, especially when several environmental and operational conditions like air/oxygen, pH, temperature, wind & light intensities, humidity, planting density etc. and their combinations are not controlled (Kadlec and Wallace, 2009; Papaevangelou et al., 2016).

The fundamental purpose of installing wetlands and operating them under different HRT is to achieve some significant effect in removing contaminants (Conn and Fiedler, 2006; Papaevangelou et al., 2016). The MANOVA Post-hoc pairwise tests showed effluent differences among the HRTs for four effluent qualities – TDS, COD, NH_3-N and AnS. The observed phenomena are that: 1 day HRT gives low TDS but high AnS effluent levels which all differ significantly from both 2 & 3 days HRT; 3 days HRT shows lower effluent COD levels significantly different from effluent of both 1 & 2 days HRT; and finally, 1 day HRT with higher effluent NH_3-N levels only differed from 3 days HRT. The results imply that HRT played influential role in the removal of these aforementioned contaminants although there is no statistical evidence to support the existence of interaction between HRT and CW. This is likely because HRT could not statistically impose its influence on all or most of the effluent contaminant levels well enough to constitute deliberate efforts for statistical significance. This could be a classical situation where p-value based statistical significance does not necessarily mean practical or real-world relevance (Nakagawa and Cuthill, 2007).

The list below presents key understanding drawn from the findings on the comparison between the performance of the two media types (gravel and lateritebased) experimented in the current study.

1. Both media showed relatively comparable general performance other than for specific contaminants like P and Orth-P, the differences between the two media types were mostly not statistically significant ($p>0.05$).
2. Similarly, the BOD_5 removal efficiencies from the two media beds appeared comparable. While the unplanted CW of laterite-based media seemingly had an edge over unplanted gravel counterpart, the opposite is observed when the beds are planted. However, the efficiencies are not significantly different between the media types whether unplanted ($p=0.329$) or planted ($p=0.280$).
3. Gravel media beds outperform laterite-based beds in removing COD in both unplanted beds (72 – 84% versus 69 – 82%) and planted beds (78 – 86% versus 72 – 82%). Again, differences in the media efficiencies are not statistically significant ($p>0.05$).
4. Efficiencies for AnS removal followed the removal trends for BOD_5 as previously discussed. The removal efficiencies given by the two media are comparable and not significantly different ($p>0.05$). The unplanted lateritebased media seemingly have an edge over unplanted gravel and vice versa when the media were planted.
5. There was clear significant difference between efficiencies from the two media types for Ortho-P/P species where gravel media poorly performed in both unplanted and planted conditions. The reason is largely attributed to the

presence of Fe in laterite which plays significant role in P removal (Wu et al., 2019).

6. The media efficiencies for NO_3-N removal were comparable and both media types gave almost the same removal efficiencies in both planted and unplanted conditions ($p=0.432$ & $p=0.247$ respectively).
7. Generally, removal efficiencies for NH_3-N were poor in both media types. Whether planted or unplanted, effluent NH_3-N almost always increased instead of reduction.
    a. However, gravel beds showed instances of marginal NH_3-N removals up to 1% and marginally performed better than the laterite-based beds.
    b. The difference in NH_3-N removal efficiencies between the media was significant under unplanted conditions ($p=0.007$) only and not when planted ($p=0.182$). This suggests that vegetation is key to influence NH_3-N removal efficiencies especially when the plant is sugarcane.

In general, the two media types (gravel and laterite-based) gave comparable removal efficiencies based on the effluent contaminants (organics and nutrients) tested and sometimes one media type have marginal edge over the other in their removal efficiencies. The laterite-based media could compete favourably with a conventional and well-known CW media like gravel. Generally, the striking findings are that laterite-based media had superior edge over gravel for Ortho-P/P species removal while gravel had better performance for NH_3-N removal. Furthermore, both media have comparable efficiencies for NO_3-N removal.

Generally, the presence of the indigenous vegetation in the CW influenced performance in terms of contaminant removal efficiencies. At the level of individual vegetation, the influence on CW ranged between marginal and significant levels. Planted beds outperformed the unplanted ones by giving lower effluent concentrations of most contaminants except NO_3-N. For these contaminants TDS, TSS, BOD_5, COD, Ortho-P, P, NO_3-N, NH_3-N and AnS, planted beds' removal efficiencies were better than unplanted counterparts although the differences were statistically insignificant except for BOD_5, NH_3-N and AnS ($p<0.0001$). This clearly contradicts the assertion that vegetation in CW may be unnecessary for effective removal of contaminants in wastewater (Mara, 2003).

For individual vegetation, the following are considered to be impressive happenings that need attention.

1. The taro monoculture bed showed better efficiency (77 – 81%) over the unplanted (60 – 75%) and the difference in mean effluent TSS levels is significant ($p=0.026$). Again, taro bed showed a better TSS removal efficiencies than sugarcane bed and this was close to being significant ($p=0.054$).
2. The planted beds showed that taro, sugarcane and their mixture (84 – 88%) had marginal edge over the unplanted (82 – 86%) in BOD_5 removal efficiencies although not statistically significant difference ($p>0.05$, 0.150 – 0.928)
3. COD removal performance were also influenced by vegetation in a similar trend like the case of BOD_5. Taro, sugarcane and their mixed beds (72 – 84%) showed marginal edge over the unplanted (69 – 82% with $p>0.05$). Although not statistically significant but there is an indication that plants could play a key role in the difference in efficiencies.
4. All planted beds showed better AnS removal efficiencies (53 – 73%) than the unplanted beds (42 – 60%) with significant difference between the two categories (planted versus unplanted, $p<0.05$, 0 – 0.045). However, among the vegetated beds, the performance of sugarcane only and mixed plant beds differed significantly in the effluent AnS levels ($p=0.024$) with the sugarcane monoculture bed being the better of the two (1.65 against 2 mg-LAS/L).

   However, taro and sugarcane monoculture beds showed comparable performance with no significant difference in effluent AnS levels ($p=0.594$), probably monoculture conditions are better for AnS removal.
5. Vegetation is not strongly seen to influence NO_3-N and Ortho-P/P species removal efficiencies in the CWs. However, sugarcane monoculture bed appeared marginally better in removal efficiencies (89 – 96%) than taro monoculture (81 – 95%), and also significantly better than mixed vegetation beds (84 – 86%, $p=0.027$). Suggesting that sugarcane as a CW macrophyte may perform better with nitrate and phosphorus removal when planted alone.

6. Ammonia removal efficiencies influence from the presence of vegetation was very strong. The monoculture taro (-5% to 4%) and sugarcane (2 – 33%) beds outperformed the unplanted with significance difference in effluent NH_3-N levels ($p<0.0001$). Also, monoculture beds (taro only and sugarcane only) performed significantly better than mixed bed ($p<0.001$). Again, a much better performance from sugarcane monoculture bed is likely aided by the plant's high preference for NH_3-N uptake to other nitrogenous nutrient species (Boschiero et al., 2018).

Thus, indigenous plants taro, sugarcane and their mixture influence the performance of the CWs in removing contaminants. These two plants have demonstrated their ability to work as CW macrophytes especially for treatment of greywater (Masi, 2004; Mthembu et al., 2013; Wu et al., 2016). The two plants have demonstrated their ability to be applied in CW designs especially for greywater treatment. However, the vegetation applied as monoculture to the CW perform better than when mixed, probably because of the influence of potential allelopathy conditions created by the plants (Asao et al., 2003; Sampietro et al., 2007; Sampietro et al., 2018).

The statistical analysis indicated that baffle-partitions could influence performance of CW as asserted in literature (Ramprasad and Philip, 2016a; Ramprasad et al., 2017). However, the influence may be marginal and not significant on most contaminants especially organics (BOD_5 and COD) and nutrients (Ortho-P/P, OrthoP/P, NO_3-N and NH_3-N). The only contaminant that strongly demonstrated
($p<0.028$) that baffle-partitions influenced its removal efficiency is anionic surfactant (AnS). Again, the presence of baffle-partitions generally appeared weak to influence CW performance, the marginal contributions to removal efficiencies could still be relevant in the presence of other key performance factors. For instance, the influence of baffle partitions in laterite-based bed is strongly felt for removal of AnS. Thus, that baffle-partitions may not influence CW performance alone but in addition to other key factors like media, vegetation, and biofilms (Ramprasad and Philip, 2016a).

5.5 Prediction models for main organic contaminants – BOD_5, COD and AnS

Regression method of predicting wetland effluent water quality has been a design approach labelled as a "black box model" because it oversimplifies a complex reality (Rousseau et al., 2004). Such regression models are normally generated from large data sets on the performance of existing constructed wetlands (Rousseau et al., 2004; Dotro et al., 2017), and this give them some credibility as potential design tools and source of useful information.

The main organic contaminants BOD, COD and AnS are selected for fitting prediction models for their effluent levels based on a verified condition that they are not widely different (not extensively statistically significant) among treatment wetlands and along the different HRT in this study. This allows for pulling together enough data points required for robust regression analysis. The multiple regression models were fitted using Stepwise selection method because of its strength of combining both forward selection and backward elimination methods (Ghani and Ahmad, 2010; Denis, 2019).

The models generated are statistically significant ($p<0.05$), indicating that they rightly predicts the effluent contaminants, and thus rejecting the null hypothesis that coefficients of the models and their R^2 are equal to zero (Denis, 2019). Again, the associated significant ANOVA tests support the rejection of null hypothesis that the multiple R in the population data is equal to zero. The collinearity statistics VIF (variable inflation factor) in all cases are far below the threshold of 10 and therefore indicate no potential collinearity problems (Field, 2013; Denis, 2019). Only that the constants for COD and AnS models show non-significance ($p>0.05$), thus they are not different from zero, implying that their regression lines approximately pass through the origin. However, the constants are kept in the equations because of the following: 1) constants are not predictors and have unclear real interpretations in the absence of dummy variables; 2) constants absorb unaccounted for biases; and 3) constants guarantee a zero mean which is a key assumption for the residual analysis (Ogee et al., 2013).

For convenient use of the models as tools for design and data assessments like commonly found in literature (Rousseau et al., 2004; Dotro et al., 2017), the corresponding input and output data ranges are defined for guidance to make the models more useful for specified applications (Son et al., 2010). Among all fitted models, the seemingly strongest model is the one fitted for effluent AnS because of its high ability to predict 73% of variance in effluent AnS, followed by COD (62%), and then BOD_5 (37%). The relatively low R^2 especially with BOD model signals caution and circumspect in its application for long-term predictions (Son et al., 2010).

The AnS model is defined by the most number of predictors about five in number (nitrate, Ortho-P, COD, AnS and HRT), and the others have three each (with combinations of nitrate, BOD/COD, COD, and AnS). The model for BOD_5 rather fitted the BOD/COD ratio instead of the individual carbon species, probably because the ratio is an explicit expression of biodegradability, more meaningful and directly relates to the biological degradation of BOD (Boyjoo et al., 2013). The commonest predictors in all three effluent models are NO_3-N and COD, followed by AnS. Nitrate may be prominent in the models probably because it is a significant nitrogenous nutrient species that attempts to balance the necessary influent C/N ratios (Singh et al., 2017; Jia et al., 2018). Also, COD is usually a large source of organic carbon, which is necessary for the removal of other contaminants or substances in wetlands (Jia et al., 2018).

Another key observation is the missing out of BOD_5 on effluent COD model. The expectation was that influent BOD_5 or its ratio form with COD would be fitted, however, a counterpart organic contaminant AnS is rather successful. Probably, AnS better represented readily biodegradable organics because LAS in a sense is also tagged as more biodegradable and especially influenced by key wetland features like media, vegetation, and operational condition of increasing HRT (Scott and Jones, 2000).

The model for AnS needed influent nutrient phosphorus in addition to nitrate nitrogen to stimulate biological treatment in the wetlands, a probable attempt to establish a classical requisite biochemical balance between biodegradable organics and nutrients in a C/N/P ratio (Boyjoo et al., 2013). In the same model, HRT becomes relevant probably because removal of AnS is more sensitive to the time that influent predictors would spend together in a wetland bed (Scott and Jones, 2000; Pérez-López et al., 2018). The models are also associated with suppressors which are key in "removing irrelevant variance from predictors, increasing their regression weight, and also increasing overall predictability of the model" (Pandey and Elliott, 2010; Klein, 2014). These include influent levels of BOD/COD ratio, COD, and AnS.

Chapter 6: Conclusions and recommendations

6.1 Conclusions

The greywater generation rates in Ghana based on available information are within 39 – 83 l/c/d (95% CI), low and similar to values linked to other developing countries. Ghanaian greywater quality so far could be classified as polluted and loaded with diverse contaminants including BOD_5, COD, nutrients (NO_3-N, NO_2-N, NH_3-N and Ortho-P & P), SO_4^{2-}, and anionic surfactants (AnS) which generally exceed regulatory discharge limits, and therefore unsafe for the environment and public health. The untreated greywater from sewered and/or non-sewered households and communities could pose similar risks including nuisance and public health hazards. However, the greywater is biodegradable to be handled in any biological treatment systems.

There is strong perception that plants used in greywater disposal practices, call it irrigation or plant watering or disposal by vegetated subsurface infiltration, could treat greywater including the removal of odour, and removal of "poison/danger". Some key benefits derived from plant biomass are consumption for food and medicine, which are incentives for indigenous plants use in greywater disposal. Diverse range of local plants are used in the disposal of greywater and these plants identified with 36 different species. The most frequently used plants in the greywater disposal practices are sugarcane, taro, cocoyam, basil, dandelion, aloe vera, and lemon grass which are also candidate macrophytes for a phytoremediation application in constructed wetlands especially for greywater treatment.

Pre-treatment sedimentation is useful to reduce greywater contaminants by 18 – 75% to significant levels but effluent could not still meet regulatory discharge limits for major contaminants. First of all, the designed indigenized constructed wetlands have shown strong potential for effective onsite treatment of greywater to an acceptable standard. Effluent water quality generally met the available regulatory discharge limits for most contaminants tested (pH, EC, TDS, DO, turbidity, TSS, BOD_5, COD, AnS, SO_4^{2-}, NO_3-N and NO_2-N) except for NH_3-N, P and Fe.

The quality of effluent from the CWs is always influenced by wetland microcosm models and the HRT accounting for 22% and 26% of effluent quality variations respectively. The designed CW directly influences effluent levels of TDS, BOD_5, Ortho-P, P, NH_3-N and AnS, while HRT influences TDS, COD and AnS. The designed CW could achieve lower and better effluent TDS levels with 1 day HRT, however CW performance for AnS removal improves with HRT beyond 1 day. For better COD and NH_3-N removal efficiencies, CW would require 3 days HRT.

The local media used in the design of CW influence contaminant removal performance. Gravel media have an edge over laterite-based beds with respect organic contaminant removal and vice versa for nutrients especially for Ortho-P/P. Gravel media always influence CW for better removal performance with NH_3-N. The indigenous vegetation (taro, sugarcane, and mixed of both) could influence the performance of the CWs in removing organic contaminants and nutrients, and specifically TSS, anionic surfactants (AnS), NO_3-N and NH_3-N. Vegetation in monoculture conditions better influence performance of CWs than when blended. Baffle partitions could influence the performance of CW for the removal of organic contaminants especially anionic surfactants. Again, baffle partitions influence on the performance of CW in removing nutrient contaminants are rather marginal and weak to significantly affect their removal efficiencies.

Three prediction models for effluent BOD_5, COD and AnS (anionic surfactants) are appropriately fitted using multiple linear regression analysis. The prediction accuracies may not be necessarily too high especially for BOD_5 model but still valid, and may only require some caution for long-term prediction applications because of large uncertainties usually associated with regression models usage in constructed wetlands (Rousseau et al., 2004). The models have well-defined input and output ranges for potential applications as tools for simplistic designs including exploratory studies and data assessment.

6.2 Recommendations

6.2.1 Recommendations for further studies

The following recommendations are made for further research.

1. Risk assessment studies of plant biomass derived from indigenous vegetation used in greywater disposals are required to ensure public health safety of produce consumed as food and/or medicine among local inhabitants.
2. The designed systems should be piloted to ascertain performance efficiencies in settings (preferably household or neighbour cluster) without the least manipulation or control.
3. The CW designs need further improvement and better understanding of performance under long-term (e.g. 3 – 4 years) operational conditions (Dan et al., 2011; Reyes-Contreras et al., 2012).
4. Contaminant risk assessment studies are required on treated effluent and plant biomass from the designed CW before direct reuse and/or produce consumption (e.g. edible parts like corms of taro and stems of sugarcane).
5. Informative studies of comparing the performance of batch-loading and continuous-flow conditions for the designed CWs are strongly recommended.
6. Data generated in subsequent studies based on current design features should be explored on the fitted prediction models for further development and improvement.

6.2.2 Recommendations for policy and practice

The following are recommended as the necessary steps for policy and practice.

- There should be specific guidelines and standards on design and use of constructed wetlands in Ghana especially for onsite treatment of wastewater including greywater.

- Policy and decision makers are encouraged to promote proactive initiatives (technology experimentation, piloting, and scale-up forms) that introduce low-cost treatment alternatives like CWs for onsite greywater management to protect our environment from contaminant-laden greywater especially in urban Ghana.

6.3 Contribution to knowledge

The following are the listed key contributions the the study has made to the body of knowledge.

- Generation of useful data on greywater characteristics in Ghana which is indicative as a basis for further studies in this limited/grey research area.
- Identification and documentation of useful indigenous practices greywater disposal with potential for integration into a scientifically proven low-cost technology like CW.
- Extends existing knowledge that constructed wetland is applicable for treating wastewater in Ghana and requires exploration of indigenous macrophytes.
- Evidence of successful indigenization of a low-cost phytoremediation technology by integrating locally available but lesser-known macrophytes like taro & sugarcane, and also local media like gravel and lateritic soil amended with gravel.

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Appendixes

Appendix 1: MANOVA pairwise comparison tests for CW media

Dependent Variable			Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
TSS	Unplanted (G)	Unplanted (L)	-3.7367	4.37599	.395	-12.4229	4.9496
		Planted (L)	-3.7367	4.37599	.395	-12.4229	4.9496
		Planted (G)	.2647	4.37599	.952	-8.4216	8.9509
	Unplanted (L)	Unplanted (G)	3.7367	4.37599	.395	-4.9496	12.4229
		Planted (L)	0.0000	4.37599	1.000	-8.6863	8.6863
		Planted (G)	4.0013	4.37599	.363	-4.6849	12.6876
	Planted (L)	Unplanted(G)	3.7367	4.37599	.395	-4.9496	12.4229
		Unplanted(L)	0.0000	4.37599	1.000	-8.6863	8.6863
		Planted(G)	4.0013	4.37599	.363	-4.6849	12.6876
	Planted(G)	Unplanted(G)	-.2647	4.37599	.952	-8.9509	8.4216
		Unplanted(L)	-4.0013	4.37599	.363	-12.6876	4.6849
		Planted(L)	-4.0013	4.37599	.363	-12.6876	4.6849
BOD_5	Unplanted (G)	Unplanted (L)	4.3333	4.41456	.329	-4.4295	13.0962
		Planted(L)	10.3333*	4.41456	.021	1.5705	19.0962
		Planted(G)	15.1333*	4.41456	.001	6.3705	23.8962
	Unplanted (L)	Unplanted (G)	-4.3333	4.41456	.329	-13.0962	4.4295
		Planted(L)	6.0000	4.41456	.177	-2.7628	14.7628
		Planted(G)	10.8000*	4.41456	.016	2.0372	19.5628
	Planted(L)	Unplanted (G)	-10.3333*	4.41456	.021	-19.0962	-1.5705
		Unplanted (L)	-6.0000	4.41456	.177	-14.7628	2.7628
		Planted(G)	4.8000	4.41456	.280	-3.9628	13.5628
	Planted(G)	Unplanted (G)	-15.1333*	4.41456	.001	-23.8962	-6.3705
		Unplanted (L)	-10.8000*	4.41456	.016	-19.5628	-2.0372
		Planted(L)	-4.8000	4.41456	.280	-13.5628	3.9628
COD	Unplanted (G)	Unplanted (L)	1.2000	11.90637	.920	-22.4340	24.8340
		Planted(L)	7.4000	11.90637	.536	-16.2340	31.0340
		Planted(G)	17.4667	11.90637	.146	-6.1673	41.1006
	Unplanted (L)	Unplanted (G)	-1.2000	11.90637	.920	-24.8340	22.4340
		Planted(L)	6.2000	11.90637	.604	-17.4340	29.8340
		Planted(G)	16.2667	11.90637	.175	-7.3673	39.9006
	Planted(L)	Unplanted (G)	-7.4000	11.90637	.536	-31.0340	16.2340
		Unplanted (L)	-6.2000	11.90637	.604	-29.8340	17.4340
		Planted(G)	10.0667	11.90637	.400	-13.5673	33.7006
	Planted(G)	Unplanted (G)	-17.4667	11.90637	.146	-41.1006	6.1673
		Unplanted (L)	-16.2667	11.90637	.175	-39.9006	7.3673
		Planted(L)	-10.0667	11.90637	.400	-33.7006	13.5673
Ortho-P	Unplanted (G)	Unplanted (L)	6.3033*	.77842	.000	4.7582	7.8485

		Planted(L)	6.1847*	.77842	.000	4.6395	7.7298
		Planted(G)	.6133	.77842	.433	-.9318	2.1585

Dependent Variable			Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
						Lower Bound	Upper Bound
	Unplanted (L)	Unplanted (G)	-6.3033*	.77842	.000	-7.8485	-4.7582
		Planted(L)	-.1187	.77842	.879	-1.6638	1.4265
		Planted(G)	-5.6900*	.77842	.000	-7.2352	-4.1448
	Planted(L)	Unplanted (G)	-6.1847*	.77842	.000	-7.7298	-4.6395
		Unplanted (L)	.1187	.77842	.879	-1.4265	1.6638
		Planted(G)	-5.5713*	.77842	.000	-7.1165	-4.0262
	Planted(G)	Unplanted (G)	-.6133	.77842	.433	-2.1585	.9318
		Unplanted (L)	5.6900*	.77842	.000	4.1448	7.2352
		Planted(L)	5.5713*	.77842	.000	4.0262	7.1165
P	Unplanted (G)	Unplanted (L)	2.0453*	.25327	.000	1.5426	2.5481
		Planted(L)	2.0080*	.25327	.000	1.5053	2.5107
		Planted(G)	.1980	.25327	.436	-.3047	.7007
	Unplanted (L)	Unplanted (G)	-2.0453*	.25327	.000	-2.5481	-1.5426
		Planted(L)	-.0373	.25327	.883	-.5401	.4654
		Planted(G)	-1.8473*	.25327	.000	-2.3501	-1.3446
	Planted(L)	Unplanted (G)	-2.0080*	.25327	.000	-2.5107	-1.5053
		Unplanted (L)	.0373	.25327	.883	-.4654	.5401
		Planted(G)	-1.8100*	.25327	.000	-2.3127	-1.3073
	Planted(G)	Unplanted (G)	-.1980	.25327	.436	-.7007	.3047
		Unplanted (L)	1.8473*	.25327	.000	1.3446	2.3501
		Planted(L)	1.8100*	.25327	.000	1.3073	2.3127
NO_3-N	Unplanted (G)	Unplanted (L)	.1200	.15177	.431	-.1813	.4213
		Planted(L)	-.1467	.15177	.336	-.4479	.1546
		Planted(G)	.0300	.15177	.844	-.2713	.3313
	Unplanted (L)	Unplanted (G)	-.1200	.15177	.431	-.4213	.1813
		Planted(L)	-.2667	.15177	.082	-.5679	.0346
		Planted(G)	-.0900	.15177	.555	-.3913	.2113
	Planted(L)	Unplanted (G)	.1467	.15177	.336	-.1546	.4479
		Unplanted (L)	.2667	.15177	.082	-.0346	.5679
		Planted(G)	.1767	.15177	.247	-.1246	.4779
	Planted(G)	Unplanted (G)	-.0300	.15177	.844	-.3313	.2713
		Unplanted (L)	.0900	.15177	.555	-.2113	.3913
		Planted(L)	-.1767	.15177	.247	-.4779	.1246
NH_3-N	Unplanted (G)	Unplanted (L)	-1.4940*	.53794	.007	-2.5618	-.4262
		Planted(L)	-.6273	.53794	.246	-1.6951	.4405
		Planted(G)	.0960	.53794	.859	-.9718	1.1638
	Unplanted (L)	Unplanted (G)	1.4940*	.53794	.007	.4262	2.5618
		Planted(L)	.8667	.53794	.110	-.2011	1.9345

		Planted(G)	1.5900*	.53794	.004	.5222	2.6578
	Planted(L)	Unplanted (G)	.6273	.53794	.246	-.4405	1.6951
		Unplanted (L)	-.8667	.53794	.110	-1.9345	.2011
		Planted(G)	.7233	.53794	.182	-.3445	1.7911
	Planted(G)	Unplanted (G)	-.0960	.53794	.859	-1.1638	.9718
Dependent Variable			**Mean Difference (I-J)**	**Std. Error**	**Sig.**	**95% Confidence Interval**	
						Lower Bound	**Upper Bound**
		Unplanted (L)	-1.5900*	.53794	.004	-2.6578	-.5222
		Planted(L)	-.7233	.53794	.182	-1.7911	.3445
AnS	Unplanted (G)	Unplanted (L)	.0560	.17342	.747	-.2882	.4002
		Planted(L)	.4080*	.17342	.021	.0638	.7522
		Planted(G)	.5433*	.17342	.002	.1991	.8876
	Unplanted (L)	Unplanted (G)	-.0560	.17342	.747	-.4002	.2882
		Planted(L)	.3520*	.17342	.045	.0078	.6962
		Planted(G)	.4873*	.17342	.006	.1431	.8316
	Planted(L)	Unplanted (G)	-.4080*	.17342	.021	-.7522	-.0638
		Unplanted (L)	-.3520*	.17342	.045	-.6962	-.0078
		Planted(G)	.1353	.17342	.437	-.2089	.4796
	Planted(G)	Unplanted (G)	-.5433*	.17342	.002	-.8876	-.1991
		Unplanted (L)	-.4873*	.17342	.006	-.8316	-.1431
		Planted(L)	-.1353	.17342	.437	-.4796	.2089

*The mean difference is significant at p<0.05

(G) – gravel bed

(L) – laterite-based bed

AnS – anionic surfactant

Appendix 2: MANOVA pairwise comparison tests for CW vegetation

Dependent Variable	**Factors**		**Mean Difference (I-J)**	**Std. Error**	**Sig. (p)**	**95% Confidence Interval**	
	(I)	**(J)**				**Lower Bound**	**Upper Bound**
TSS	Unplanted(L)	Taro + Sugarcane(L)	0.0000	4.37599	1.000	-8.6863	8.6863
		Taro(L)	9.8680*	4.37599	.026	1.1817	18.5543
		Sugarcane(L)	1.3327	4.37599	.761	-7.3536	10.0189
	Taro+Sugarcane(L)	Unplanted(L)	0.0000	4.37599	1.000	-8.6863	8.6863
		Taro(L)	9.8680*	4.37599	.026	1.1817	18.5543
		Sugarcane(L)	1.3327	4.37599	.761	-7.3536	10.0189
	Taro(L)	Unplanted(L)	-9.8680*	4.37599	.026	-18.5543	-1.1817

		Taro + Sugarcane(L)	-9.8680*	4.37599	.026	-18.5543	-1.1817
		Sugarcane(L)	-8.5353	4.37599	.054	-17.2216	.1509
	Sugarcane(L)	Unplanted(L)	-1.3327	4.37599	.761	-10.0189	7.3536
		Taro + Sugarcane(L)	-1.3327	4.37599	.761	-10.0189	7.3536
		Taro(L)	8.5353	4.37599	.054	-.1509	17.2216
BOD_5	Unplanted(L)	Taro + Sugarcane(L)	6.0000	4.41456	.177	-2.7628	14.7628
		Taro(L)	4.5333	4.41456	.307	-4.2295	13.2962
		Sugarcane(L)	6.4000	4.41456	.150	-2.3628	15.1628
	Taro+Sugarcane(L)	Unplanted(L)	-6.0000	4.41456	.177	-14.7628	2.7628
		Taro(L)	-1.4667	4.41456	.740	-10.2295	7.2962
		Sugarcane(L)	.4000	4.41456	.928	-8.3628	9.1628
	Taro(L)	Unplanted(L)	-4.5333	4.41456	.307	-13.2962	4.2295
		Taro + Sugarcane(L)	1.4667	4.41456	.740	-7.2962	10.2295
		Sugarcane(L)	1.8667	4.41456	.673	-6.8962	10.6295
	Sugarcane(L)	Unplanted(L)	-6.4000	4.41456	.150	-15.1628	2.3628
		Taro + Sugarcane(L)	-.4000	4.41456	.928	-9.1628	8.3628
		Taro(L)	-1.8667	4.41456	.673	-10.6295	6.8962
COD	Unplanted(L)	Taro + Sugarcane(L)	6.2000	11.90637	.604	-17.4340	29.8340
		Taro(L)	3.0000	11.90637	.802	-20.6340	26.6340
		Sugarcane(L)	5.4667	11.90637	.647	-18.1673	29.1006
	Taro+Sugarcane(L)	Unplanted(L)	-6.2000	11.90637	.604	-29.8340	17.4340
		Taro(L)	-3.2000	11.90637	.789	-26.8340	20.4340
		Sugarcane(L)	-.7333	11.90637	.951	-24.3673	22.9006
	Taro(L)	Unplanted(L)	-3.0000	11.90637	.802	-26.6340	20.6340
		Taro + Sugarcane(L)	3.2000	11.90637	.789	-20.4340	26.8340
		Sugarcane(L)	2.4667	11.90637	.836	-21.1673	26.1006
	Sugarcane(L)	Unplanted(L)	-5.4667	11.90637	.647	-29.1006	18.1673
		Taro + Sugarcane(L)	.7333	11.90637	.951	-22.9006	24.3673

Dependent Variable	Factors (I)	(J)	Mean Difference (I-J)	Std. Error	Sig. (p)	95% Confidence Interval Lower Bound	Upper Bound
		Taro(L)	-2.4667	11.90637	.836	-26.1006	21.1673
Ortho-P	Unplanted(L)	Taro + Sugarcane(L)	-.1187	.77842	.879	-1.6638	1.4265
		Taro(L)	.0747	.77842	.924	-1.4705	1.6198
		Sugarcane(L)	-.5273	.77842	.500	-2.0725	1.0178
	Taro+Sugarcane(L)	Unplanted(L)	.1187	.77842	.879	-1.4265	1.6638
		Taro(L)	.1933	.77842	.804	-1.3518	1.7385
		Sugarcane(L)	-.4087	.77842	.601	-1.9538	1.1365
	Taro(L)	Unplanted(L)	-.0747	.77842	.924	-1.6198	1.4705
		Taro + Sugarcane(L)	-.1933	.77842	.804	-1.7385	1.3518
		Sugarcane(L)	-.6020	.77842	.441	-2.1472	.9432
	Sugarcane(L)	Unplanted(L)	.5273	.77842	.500	-1.0178	2.0725
		Taro + Sugarcane(L)	.4087	.77842	.601	-1.1365	1.9538
		Taro(L)	.6020	.77842	.441	-.9432	2.1472
P	Unplanted(L)	Taro + Sugarcane(L)	-.0373	.25327	.883	-.5401	.4654
		Taro(L)	.0207	.25327	.935	-.4821	.5234
		Sugarcane(L)	-.1713	.25327	.500	-.6741	.3314
	Taro+Sugarcane(L)	Unplanted(L)	.0373	.25327	.883	-.4654	.5401
		Taro(L)	.0580	.25327	.819	-.4447	.5607
		Sugarcane(L)	-.1340	.25327	.598	-.6367	.3687
	Taro(L)	Unplanted(L)	-.0207	.25327	.935	-.5234	.4821
		Taro + Sugarcane(L)	-.0580	.25327	.819	-.5607	.4447
		Sugarcane(L)	-.1920	.25327	.450	-.6947	.3107
	Sugarcane(L)	Unplanted(L)	.1713	.25327	.500	-.3314	.6741
		Taro + Sugarcane(L)	.1340	.25327	.598	-.3687	.6367
		Taro(L)	.1920	.25327	.450	-.3107	.6947
NO_3-N	Unplanted(L)	Taro + Sugarcane(L)	-.2667	.15177	.082	-.5679	.0346
		Taro(L)	-.1933	.15177	.206	-.4946	.1079

		Sugarcane(L)	.0733	.15177	.630	-.2279	.3746
	Taro+Sugarcane(L)	Unplanted(L)	.2667	.15177	.082	-.0346	.5679
		Taro(L)	.0733	.15177	.630	-.2279	.3746
		Sugarcane(L)	.3400*	.15177	.027	.0387	.6413
	Taro(L)	Unplanted(L)	.1933	.15177	.206	-.1079	.4946
		Taro + Sugarcane(L)	-.0733	.15177	.630	-.3746	.2279
		Sugarcane(L)	.2667	.15177	.082	-.0346	.5679
	Sugarcane(L)	Unplanted(L)	-.0733	.15177	.630	-.3746	.2279
		Taro + Sugarcane(L)	-.3400*	.15177	.027	-.6413	-.0387
Dependent Variable	**Factors**		**Mean Difference (I-J)**	**Std. Error**	**Sig. (p)**	**95% Confidence Interval**	
	(I)	**(J)**				**Lower Bound**	**Upper Bound**
		Taro(L)	-.2667	.15177	.082	-.5679	.0346
NH_3-N	Unplanted(L)	Taro + Sugarcane(L)	.8667	.53794	.110	-.2011	1.9345
		Taro(L)	2.3953*	.53794	.000	1.3275	3.4631
		Sugarcane(L)	3.0080*	.53794	.000	1.9402	4.0758
	Taro+Sugarcane(L)	Unplanted(L)	-.8667	.53794	.110	-1.9345	.2011
		Taro(L)	1.5287*	.53794	.005	.4609	2.5965
		Sugarcane(L)	2.1413*	.53794	.000	1.0735	3.2091
	Taro(L)	Unplanted(L)	-2.3953*	.53794	.000	-3.4631	-1.3275
		Taro + Sugarcane(L)	-1.5287*	.53794	.005	-2.5965	-.4609
		Sugarcane(L)	.6127	.53794	.258	-.4551	1.6805
	Sugarcane(L)	Unplanted(L)	-3.0080*	.53794	.000	-4.0758	-1.9402
		Taro + Sugarcane(L)	-2.1413*	.53794	.000	-3.2091	-1.0735
		Taro(L)	-.6127	.53794	.258	-1.6805	.4551
AnS	Unplanted(L)	Taro + Sugarcane(L)	.3520*	.17342	.045	.0078	.6962
		Taro(L)	.6567*	.17342	.000	.3124	1.0009
		Sugarcane(L)	.7493*	.17342	.000	.4051	1.0936
	Taro+Sugarcane(L)	Unplanted(L)	-.3520*	.17342	.045	-.6962	-.0078
		Taro(L)	.3047	.17342	.082	-.0396	.6489

		Sugarcane(L)	.3973*	.17342	.024	.0531	.7416
	Taro(L)	Unplanted(L)	-.6567*	.17342	.000	-1.0009	-.3124
		Taro + Sugarcane(L)	-.3047	.17342	.082	-.6489	.0396
		Sugarcane(L)	.0927	.17342	.594	-.2516	.4369
	Sugarcane(L)	Unplanted(L)	-.7493*	.17342	.000	-1.0936	-.4051
		Taro + Sugarcane(L)	-.3973*	.17342	.024	-.7416	-.0531
		Taro(L)	-.0927	.17342	.594	-.4369	.2516

*The mean difference is significant at $p<0.05$

(L) – laterite-based bed

AnS – anionic surfactant

Appendix 3: MANOVA pairwise comparison tests for baffle-partitions

Dependent Variable	**Factors (I)**	**(J)**	**Mean Difference (I-J)**	**Std. Error**	**Sig p-value**	**95% Confidence Interval for Difference: Lower Bound**	**Upper Bound**
TSS	Planted/ unbaffled (L)	Planted/ unbaffled (G)	4.001	4.376	.363	-4.685	12.688
		Planted/ baffled (G)	4.535	4.376	.303	-4.151	13.222
		Planted/ baffled (L)	6.398	4.376	.147	-2.288	15.084
	Planted/ unbaffled (G)	Planted/ unbaffled (L)	-4.001	4.376	.363	-12.688	4.685
		Planted/ baffled (G)	.534	4.376	.903	-8.152	9.220
		Planted/ baffled (L)	2.397	4.376	.585	-6.290	11.083
	Planted/ baffled (G)	Planted/ unbaffled (L)	-4.535	4.376	.303	-13.222	4.151
		Planted/ unbaffled (G)	-.534	4.376	.903	-9.220	8.152
		Planted/ baffled (L)	1.863	4.376	.671	-6.824	10.549
	Planted/ baffled (L)	Planted/ unbaffled (L)	-6.398	4.376	.147	-15.084	2.288
		Planted/ unbaffled (G)	-2.397	4.376	.585	-11.083	6.290
		Planted/ baffled (G)	-1.863	4.376	.671	-10.549	6.824
BOD_5	Planted/ unbaffled (L)	Planted/ unbaffled (G)	4.800	4.415	.280	-3.963	13.563
		Planted/ baffled (G)	1.733	4.415	.695	-7.029	10.496
		Planted/ baffled (L)	3.267	4.415	.461	-5.496	12.029
	Planted/ unbaffled (G)	Planted/ unbaffled (L)	-4.800	4.415	.280	-13.563	3.963
		Planted/ baffled (G)	-3.067	4.415	.489	-11.829	5.696
		Planted/ baffled (L)	-1.533	4.415	.729	-10.296	7.229
	Planted/ baffled (G)	Planted/ unbaffled (L)	-1.733	4.415	.695	-10.496	7.029
		Planted/ unbaffled (G)	3.067	4.415	.489	-5.696	11.829
		Planted/ baffled (L)	1.533	4.415	.729	-7.229	10.296

	Planted/ baffled (L)	Planted/ unbaffled (L)	-3.267	4.415	.461	-12.029	5.496
		Planted/ unbaffled (G)	1.533	4.415	.729	-7.229	10.296
		Planted/ baffled (G)	-1.533	4.415	.729	-10.296	7.229
COD	Planted/ unbaffled (L)	Planted/ unbaffled (G)	10.067	11.906	.400	-13.567	33.701
		Planted/ baffled (G)	5.333	11.906	.655	-18.301	28.967
		Planted/ baffled (L)	15.600	11.906	.193	-8.034	39.234
	Planted/ unbaffled (G)	Planted/ unbaffled (L)	-10.067	11.906	.400	-33.701	13.567
		Planted/ baffled (G)	-4.733	11.906	.692	-28.367	18.901
		Planted/ baffled (L)	5.533	11.906	.643	-18.101	29.167
	Planted/ baffled (G)	Planted/ unbaffled (L)	-5.333	11.906	.655	-28.967	18.301
		Planted/ unbaffled (G)	4.733	11.906	.692	-18.901	28.367
		Planted/ baffled (L)	10.267	11.906	.391	-13.367	33.901
	Planted/ baffled (L)	Planted/ unbaffled (L)	-15.600	11.906	.193	-39.234	8.034
		Planted/ unbaffled (G)	-5.533	11.906	.643	-29.167	18.101
		Planted/ baffled (G)	-10.267	11.906	.391	-33.901	13.367
Ortho-P	Planted/ unbaffled (L)	Planted/ unbaffled (G)	-5.571*	.778	.000	-7.116	-4.026
		Planted/ baffled (G)	-5.683*	.778	.000	-7.228	-4.138
		Planted/ baffled (L)	-1.640*	.778	.038	-3.185	-.095
	Planted/ unbaffled (G)	Planted/ unbaffled (L)	5.571*	.778	.000	4.026	7.116
		Planted/ baffled (G)	-.111	.778	.887	-1.656	1.434
		Planted/ baffled (L)	3.931*	.778	.000	2.386	5.476
	Planted/ baffled (G)	Planted/ unbaffled (L)	5.683*	.778	.000	4.138	7.228
		Planted/ unbaffled (G)	.111	.778	.887	-1.434	1.656
		Planted/ baffled (L)	4.043*	.778	.000	2.498	5.588
	Planted/ baffled (L)	Planted/ unbaffled (L)	1.640*	.778	.038	.095	3.185
		Planted/ unbaffled (G)	-3.931*	.778	.000	-5.476	-2.386
		Planted/ baffled (G)	-4.043*	.778	.000	-5.588	-2.498
P	Planted/ unbaffled (L)	Planted/ unbaffled (G)	-1.810*	.253	.000	-2.313	-1.307
		Planted/ baffled (G)	-1.854*	.253	.000	-2.357	-1.351
		Planted/ baffled (L)	-.539*	.253	.036	-1.042	-.037
	Planted/ unbaffled (G)	Planted/ unbaffled (L)	1.810*	.253	.000	1.307	2.313
		Planted/ baffled (G)	-.044	.253	.862	-.547	.459
		Planted/ baffled (L)	1.271*	.253	.000	.768	1.773
	Planted/ baffled (G)	Planted/ unbaffled (L)	1.854*	.253	.000	1.351	2.357
		Planted/ unbaffled (G)	.044	.253	.862	-.459	.547
		Planted/ baffled (L)	1.315*	.253	.000	.812	1.817
		Planted/ unbaffled (L)	.539*	.253	.036	.037	1.042
		Planted/ unbaffled (G)	-1.271*	.253	.000	-1.773	-.768

	Planted/ baffled (L)	Planted/ baffled (G)	-1.315*	.253	.000	-1.817	-.812
NO_3-N	Planted/ unbaffled (L)	Planted/ unbaffled (G)	.177	.152	.247	-.125	.478
		Planted/ baffled (G)	.027	.152	.861	-.275	.328
		Planted/ baffled (L)	-.140	.152	.359	-.441	.161
	Planted/ unbaffled (G)	Planted/ unbaffled (L)	-.177	.152	.247	-.478	.125
		Planted/ baffled (G)	-.150	.152	.325	-.451	.151
		Planted/ baffled (L)	-.317*	.152	.040	-.618	-.015
	Planted/ baffled (G)	Planted/ unbaffled (L)	-.027	.152	.861	-.328	.275
		Planted/ unbaffled (G)	.150	.152	.325	-.151	.451
		Planted/ baffled (L)	-.167	.152	.275	-.468	.135
	Planted/ baffled (L)	Planted/ unbaffled (L)	.140	.152	.359	-.161	.441
		Planted/ unbaffled (G)	.317*	.152	.040	.015	.618
		Planted/ baffled (G)	.167	.152	.275	-.135	.468
NH_3-N	Planted/ unbaffled (L)	Planted/ unbaffled (G)	.723	.538	.182	-.344	1.791
		Planted/ baffled (G)	1.312*	.538	.017	.244	2.380
		Planted/ baffled (L)	.869	.538	.109	-.198	1.937
	Planted/ unbaffled (G)	Planted/ unbaffled (L)	-.723	.538	.182	-1.791	.344
		Planted/ baffled (G)	.589	.538	.277	-.479	1.656
		Planted/ baffled (L)	.146	.538	.787	-.922	1.214
	Planted/ baffled (G)	Planted/ unbaffled (L)	-1.312*	.538	.017	-2.380	-.244
		Planted/ unbaffled (G)	-.589	.538	.277	-1.656	.479
		Planted/ baffled (L)	-.443	.538	.413	-1.510	.625
	Planted/ baffled (L)	Planted/ unbaffled (L)	-.869	.538	.109	-1.937	.198
		Planted/ unbaffled (G)	-.146	.538	.787	-1.214	.922
		Planted/ baffled (G)	.443	.538	.413	-.625	1.510
AnS	Planted/ unbaffled (L)	Planted/ unbaffled (G)	.135	.173	.437	-.209	.480
		Planted/ baffled (G)	.427*	.173	.016	.083	.772
		Planted/ baffled (L)	.387*	.173	.028	.043	.732
	Planted/ unbaffled (G)	Planted/ unbaffled (L)	-.135	.173	.437	-.480	.209
		Planted/ baffled (G)	.292	.173	.095	-.052	.636
		Planted/ baffled (L)	.252	.173	.149	-.092	.596
	Planted/ baffled (G)	Planted/ unbaffled (L)	-.427*	.173	.016	-.772	-.083
		Planted/ unbaffled (G)	-.292	.173	.095	-.636	.052
		Planted/ baffled (L)	-.040	.173	.818	-.384	.304
		Planted/ unbaffled (L)	-.387*	.173	.028	-.732	-.043
		Planted/ unbaffled (G)	-.252	.173	.149	-.596	.092

	Planted/ baffled (L)	Planted/ baffled (G)	.040	.173	.818	-.304	.384

*The mean difference is significant at p<0.05

(L) – laterite-based bed

(G) – gravel bed

AnS –anionic surfactant

Appendix 4: CW mean effluent contaminant levels from MANOVA

Dependent Variable		Mean	Std. Error	95% Confidence Interval	
				Lower Bound	Upper Bound
TDS	C1	179.727	6.145	167.529	191.925
	C2	198.593	6.145	186.395	210.791
	T1	207.400	6.145	195.202	219.598
	T2	198.560	6.145	186.362	210.758
	T3	201.027	6.145	188.829	213.225
	T4	173.120	6.145	160.922	185.318
	T5	169.353	6.145	157.155	181.551
	T6	164.180	6.145	151.982	176.378
TSS	C1	20.269	3.094	14.127	26.411
	C2	24.005	3.094	17.863	30.147
	T1	24.005	3.094	17.863	30.147
	T2	14.137	3.094	7.995	20.279
	T3	22.673	3.094	16.531	28.815
	T4	20.004	3.094	13.862	26.146
	T5	19.470	3.094	13.328	25.612
	T6	17.607	3.094	11.465	23.749
BOD_5	C1	40.067	3.122	33.870	46.263
	C2	35.733	3.122	29.537	41.930
	T1	29.733	3.122	23.537	35.930
	T2	31.200	3.122	25.004	37.396
	T3	29.333	3.122	23.137	35.530
	T4	24.933	3.122	18.737	31.130
	T5	28.000	3.122	21.804	34.196

	T6	26.467	3.122	20.270	32.663
COD	C1	76.467	8.419	59.755	93.178
	C2	75.267	8.419	58.555	91.978
	T1	69.067	8.419	52.355	85.778
	T2	72.267	8.419	55.555	88.978
	T3	69.800	8.419	53.088	86.512
	T4	59.000	8.419	42.288	75.712
	T5	63.733	8.419	47.022	80.445

Dependent Variable		**Mean**	**Std. Error**	**95% Confidence Interval**	
				Lower Bound	**Upper Bound**
	T6	53.467	8.419	36.755	70.178
Ortho-P	C1	8.573	.550	7.480	9.665
	C2	2.269	.550	1.177	3.362
	T1	2.388	.550	1.295	3.481
	T2	2.195	.550	1.102	3.287
	T3	2.797	.550	1.704	3.889
	T4	7.959	.550	6.867	9.052
	T5	8.071	.550	6.978	9.163
	T6	4.028	.550	2.935	5.121
P	C1	2.791	.179	2.435	3.146
	C2	.745	.179	.390	1.101
	T1	.783	.179	.427	1.138
	T2	.725	.179	.369	1.080
	T3	.917	.179	.561	1.272
	T4	2.593	.179	2.237	2.948
	T5	2.637	.179	2.281	2.992
	T6	1.322	.179	.967	1.677
NO_3-N	C1	.413	.107	.200	.626
	C2	.293	.107	.080	.506
	T1	.560	.107	.347	.773
	T2	.487	.107	.274	.700
	T3	.220	.107	.007	.433

	T4	.383	.107	.170	.596
	T5	.533	.107	.320	.746
	T6	.700	.107	.487	.913
NH_3-N	C1	6.103	.380	5.348	6.858
	C2	7.597	.380	6.842	8.352
	T1	6.731	.380	5.976	7.486
	T2	5.202	.380	4.447	5.957
	T3	4.589	.380	3.834	5.344
	T4	6.007	.380	5.252	6.762
	T5	5.419	.380	4.664	6.174
Dependent Variable		**Mean**	**Std. Error**	**95% Confidence Interval**	
				Lower Bound	**Upper Bound**
	T6	5.861	.380	5.106	6.616
AnS	C1	2.454	.123	2.211	2.697
	C2	2.398	.123	2.155	2.641
	T1	2.046	.123	1.803	2.289
	T2	1.741	.123	1.498	1.985
	T3	1.649	.123	1.405	1.892
	T4	1.911	.123	1.667	2.154
	T5	1.619	.123	1.375	1.862
	T6	1.659	.123	1.415	1.902

Note: AnS = Anionic surfactants
C1=unplanted gravel bed control
C2 = unplanted laterite-based bed control
T1 = mixed vegetated laterite-based bed
T2 = taro vegetated laterite-based bed
T3 = sugarcane vegetated laterite-based bed
T4 = mixed vegetated gravel bed
T5 = mixed vegetated gravel baffle-partioned bed
T6 = mixed vegetated laterite-based baffle-partitioned bed