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Optimal Recovery of Resources from Wastewater Treatment: Aspects of the Developing World.

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Thesis submitted in fulfilment of the requirements for the degree of

Doctor (PhD) in Applied Biological Sciences

Titel van het doctoraat in het Nederlands: optimale herwinning van grondstoffen uit afvalwaterzuivering: aspecten uit ontwikkelingsgebieden

Cover illustration by Tim Lacoere

Please refer to this work as:

Nansubuga I (2015) Optimal recovery of resources from wastewater treatment: Aspects of the developing world. PhD thesis, Ghent University, Belgium.

ISBN: 978-90-5989-830-1

This work was supported by the Vlaamse Interuniversitaire Raad (VLIR-OUS)

Abbreviations List

AC	Ash Content
AC_{db}	Ash Content on dry basis
ACF	Alternating Charcoal Filters
ACH	Aluminium Chlorohydrate
AD	Anaerobic Digestion
AL-WTS	Aluminium Water Treatment Sludge
APHA	American Public Health Association
ASTM	American Society for Testing Materials
BMP	Biochemical Methane Potential
BSTP	Bugolobi Sewage Treatment Plant
BW	Brewery Waste
CAS	Conventional Activated Sludge Systems
CD	Cow Dung
CFU	Colony Forming Units
CO	Carbon Oxides
COD	Chemical Oxygen Demand
CSTR	Continuously Stirred Tank Reactor
DO	Dissolved Oxygen
EABL	East African Breweries Limited
FAO	Food and Agricultural Organisation
FC	Faecal Coliforms
FC_{db}	Fixed Carbon on dry basis
FSM	Faecal Sludge Management
FSTP	Faecal Sludge Treatment Plant
GNI	Gross National Income
HHV	High Heating Value
HRAS	High Rate Activated Sludge
HRT	Hydraulic Retention Time
HSSF- CW	Horizontal Subsurface Flow Constructed Wetland
HTT	Highest Treatment Temperature
IBI	International Biochar Initiative
M & M	Major and Minor

MC	Moisture Content
MC_{db}	Moisture Content on dry basis
MDGs	Millennium Development Goals
NEMA	National Environment Management Authority
NWSC	National Water and Sewerage Corporation
PAC	Polyaluminium Chloride
PA-WTS	Polyaluminium Water Treatment Sludge
SCSTR	Semi-Continuously Stirred Tank Reactor
SRB	Sulphate Reducing Bacteria
SRT	Sludge Retention Time
STP	Sewage Treatment Plant
TAN	Total Ammonium Nitrogen
TP	Total Phosphates
TS	Total Solids
TSS	Total Suspended Solids
UN	United Nations
UNICEF	United Nations Children´s Fund
USEPA	US Environmental Protection Agency
VFA	Volatile Fatty Acids
VIP	Ventilated Improved Pit latrine
VM	Volatile Matter
VM_{db}	Volatile Matter on dry basis
VOCs	Volatile Organic Compounds
VS	Volatile Solids
WAS	Waste Activated Sludge
WHO	World Health Organisation
WRP	Water Reclamation Plant
WSP	Waste Stabilisation Pond
WTP	Water Treatment Plant
WT-PAS	Water Treatment Polyaluminium Sludge
WWTPs	Wastewater Treatment Plants

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Chapter 1 : MOVING TOWARDS SUSTAINABLE SANITATION SYSTEMS IN AFRICA: A REVIEW

This Chapter has been re-drafted after:

Nansubuga, I., Banadda, N., Verstraete, W., & Rabaey, K. Moving Towards Sustainable Sanitation Systems in Africa: A review. Journal of Water, Sanitation and Hygiene for Development. *Submitted*

1. Sustainable wastewater management, an indispensable tool to resolve water scarcity.

Water is increasingly becoming a limited resource while demand by human activities increases. According to the UN (2014), agriculture accounts for 70% of the global water withdrawal followed by industry at 20% and lastly domestic needs at 10%. Already today about 80 countries, comprising 20 percent of the world population are suffering from serious water shortage (UNEP 2008). Water shortage is made worse by increased water consumption due to the rapid growth of the global population which is projected to reach 9.3 billion in 2050 (UNDESA, 2012). Other factors like climate change also contribute to worsen this development. It is projected that by 2050, more than 40% of the global population will be living in areas subjected to severe water stress, especially in North and South Africa and South and Central Asia (UN, 2014). In Africa, by 2025 most of the countries will be in a state of water stress or scarcity (Figure 1-1). In terms of access to a safe drinking water source, a report by WHO and UNICEF (2014) concluded that 748 million people in the world still had no access to an improved source of drinking water by 2012 of whom, 325 million (43%) lived in the Sub Saharan Africa (SSA).

The increased scarcity of water as a resource continues to push the world towards an integrated approach to management of water resources. This encompasses among others, issues of climate change, environmental sustainability and water resources protection. Of peculiar interest to this study is the wastewater sector, which has a direct negative bearing on environmental sustainability and water resources quality and later, on occurrence of diseases and poor health, if not properly managed. It is estimated that 80-90% of all wastewater generated in developing countries is discharged without appropriate treatment into surface water bodies (Corcoran *et al*., 2010), which causes intensive water pollution. On the other hand, proper wastewater management would not only contribute to the water resource protection (U.S. Environmental Protection Agency, 2004), it also has great potential to supplement it and decrease competition for the already scarce water (Huertas *et al*., 2008). This therefore calls for a reform in the current wastewater management strategies to convert them into economically viable and environmentally sustainable systems. As particularly the Sub-Sahara has limited too often no water infrastructure, this approach can be implemented at Greenfield sites enabling e.g. recovery without the typical issues associated with converting existing infrastructure.

Recycling of wastewater and recovery of resources from wastewater has been suggested as a viable strategy to achieve the aforementioned objectives (Verstraete *et al*., 2009; Verstraete and Vlaeminck, 2011; Mulder, 2003). Old systems should be modified and new systems built to emphasize the resource recycling and recovery concepts. Minimal waste generation should imply minimal natural resources contamination by effluent wastewater. Also, effective reuse of wastewater would ultimately provide water that can be used for example for selected purposes such as agricultural and industrial activities, leaving enough for domestic use and any other purposes.

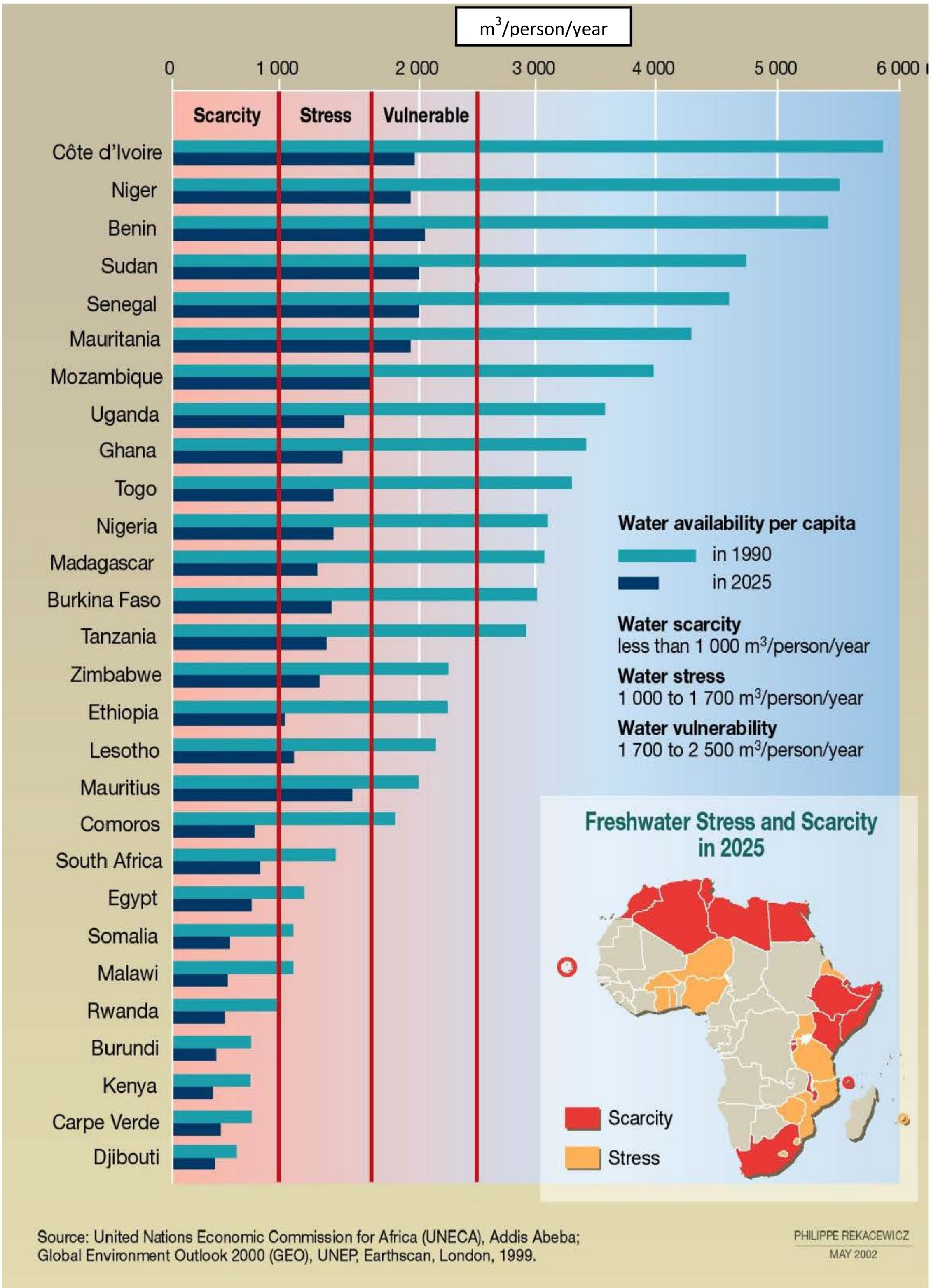


Figure 1-1: **Water Availability in Africa in 1990 and 2025. Source (UNEP, 2008).**

2. Sanitation, a prevailing predicament in Africa

While significantly increased sanitation coverage has been achieved globally (from 49% in 1990 to 64% in 2011), it is still unlikely that the millennium development goals (MDG) of

75% coverage will be achieved by 2015 (WHO and UNICEF, 2014). The contribution of Africa to the delayed target achievement of the sanitation coverage cannot be ignored. With improved sanitation coverage of 30% representing a 5 % increase from 1990 to 2011, Sub-Saharan Africa records the second lowest progress after Oceania (WHO and UNICEF, 2014). Additionally, out of the 69 countries that were not on track to achieve the sanitation MDG in 2012, 36 of them were from Sub-Saharan Africa and many like Angola and Ethiopia were among those with the lowest coverage in the world. The situation is made worse when countries such as Nigeria that had a better coverage show decreasing trends from 37 to 28% over these 22 years. Also, highlighting the deficiency of sanitation coverage is the high part of the population still practicing open defecation which was as up to 25% in the SSA in 2012 (Figure 1-2). According to WHO and UNICEF (2014), 82% of the world's 1 billion open defecating population in 2012 was housed in just 10 countries, among them are five African countries. Nigeria had the highest population of open defecators (39 million people) in Africa and it ranked 4th in the world considering countries with the highest number of open defecators in 2012. Other countries like Ethiopia, Sudan, Niger and Mozambique were ranked among the top ten in the world with a total population of 74 million people practising open defecation (WHO and UNICEF, 2014). Important to note is the difference existing between rural and urban areas in the availability of adequate sanitation with the rural areas lagging far behind. Coverage in rural areas was below 50 per cent in 2010 in most African countries. But also, in urban areas, where coverage is better the growth of slum areas poses a big challenge. In 2012 It was estimated that 863 million urban residents in the developing world live in slum conditions, compared to 650 million in 1990 and 760 million in 2000 (UN, 2014). The slums are characterised with high congestion, informal settlements, infrastructure shortages, poverty, unemployment, lack of space and inadequate urban infrastructural services including water and sanitation facilities. This creates a lot of pressure on service provision such as water, sanitation facilities, infrastructure and land which all pose a threat to social cohesion and progress. The current sanitation status poses serious health risks, which can be derived from the frequent water borne disease breakouts in many parts of Africa (WHO, 2000; Gaffga et al., 2007). Apart from that, it leads to continual deterioration of water quality in the natural water sources such as the continent's largest lake, Lake Victoria (Scheren *et al.*, 2000; Odada *et al.*, 2004; Banadda *et al.*, 2009, 2010, 2011; Komakech *et al.*, 2014). This prompts for a change in the sanitation plans and management to include strategies that will enhance sanitation coverage to all people in Africa. The silver bullet seems to be the apparent linkage of people's local needs to environmental protection programmes. Such strategies include a combination of decentralization with resource recovery, preferably enabled by local materials and reliant on simple, robust and cheap methods that are also practical and feasible especially in Africa.

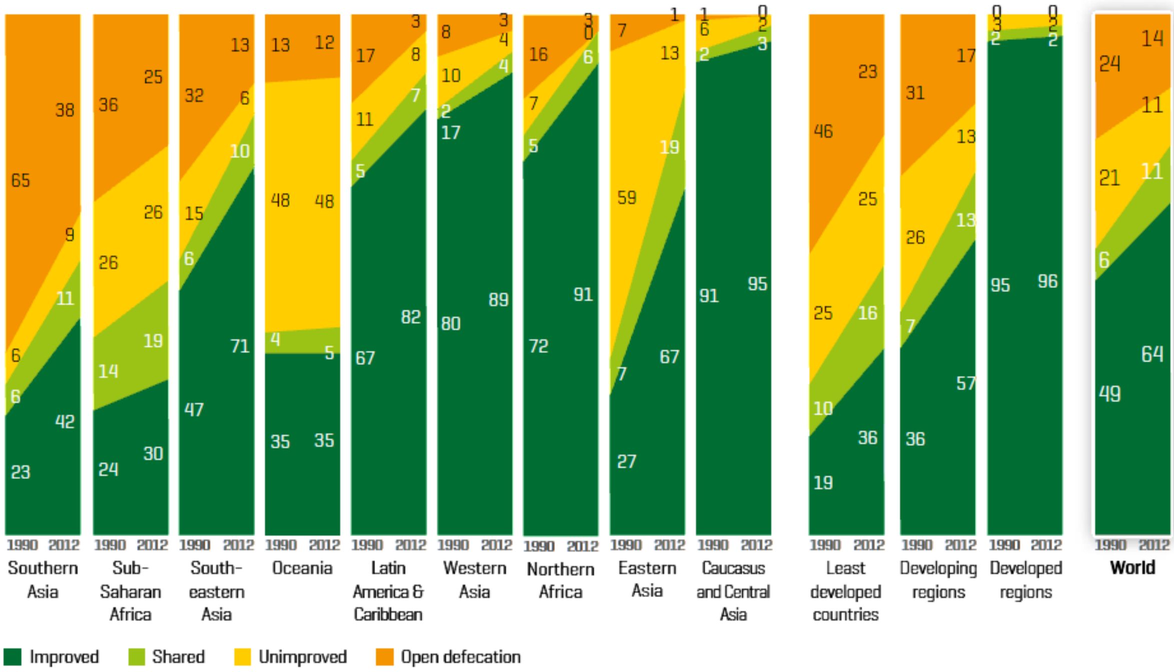


Figure 1-2: Sanitation coverage trends (%) by MDG regions, 1990–2012 (source WHO and UNICEF, 2014).

3. Benefits of providing improved Sanitation services

A total of 535 billion dollars is estimated be able to attain universal coverage of improved water and sanitation between 201 and 2015 of which USD 333 billion was estimated for sanitation (Haller, 2012). The cost of sanitation improvement can range from USD 4.88 for a simple pit latrine to a more expensive option with household sewer connection and partial treatment of wastewater at USD 10.03 per year per person served (Hutton &Haller, 2004). While these cost seem high, the costs of having no sanitation services are way beyond that. It is estimated that USD 260 billion are lost annually on a global basis due to inadequate water and sanitation (Haller 2012). With regard to health, It is now common knowledge that when water and sanitation is improved, significant health benefits are also achieved. water-borne diseases and water shed diseases are the ones most closely associated with poor water supply, poor sanitation and poor hygiene. In terms of burden of disease, these consist mainly of Infectious diarrhoea which includes cholera, salmonellosis, shigellosis, amoebiasis, and other protozoal and viral intestinal infections. In 2003, it was estimated that 54 million disability-adjusted lifeyears (DALY) or 4% of the global DALYs and 1.73 million deaths per year were attributable to unsafe water supply and sanitation, including lack of hygiene (Prüss-Üstün et al., 2004). Provision of improved sanitation services is therefore paramount. Improved sanitation can result into a number of benefits among them; health benefits due to reductions in cases and deaths associated with diarrhoeal disease and averted cases of helminths infections. This also leads to decreased costs related to healthcare services. Other economic benefits are related to savings from the health improvements. Also, time benefit would result from proximity of sanitation services, as well as reduced losses of productive time due to diseases, ultimately there is reduction in premature mortality.

The total economic benefits from providing universal sanitation would amount to USD 220 billion annually. The main contributor to the overall benefits is the value of time savings which accounts for 70% in all regions. Sub Saharan African (SSA) would contribute an important saving with USD 10 billion annually, with health care benefits also being highlighted as an important factor contributing over 37%, especially the value of saved lives. Summary results for benefit cost ratios for attaining universal access to sanitation are shown in Figure 1-3. The benefit-cost ratio (BCR) for the necessary interventions varies from 2.8 in the SSA region to 8.0 in East (E) Asia. The global economic return on sanitation spending is USD 5.5 per USD invested.

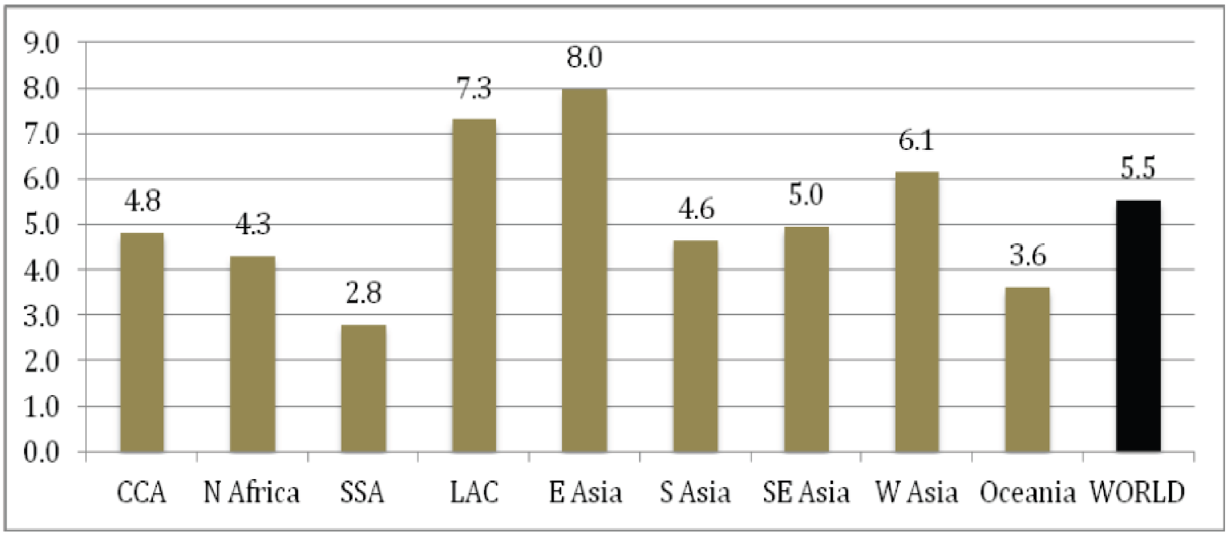


Figure 1-3: Benefit-cost ratios of interventions to attain universal access of improved sanitation (source Hutton 2012).

4. Wastewater treatment in Africa

Sanitation approaches are generally classified as centralized or decentralized (cluster system and onsite systems). The centralised system is the most expensive of them, consisting of a sewerage network with different pipe sizes required to convey sewage from a large number of households to a central wastewater treatment plant miles away from the wastewater source. There the wastewater is treated and usually disposed into a natural water body. In the decentralised systems wastewater is collected, treated and reused/disposed at or near the generation point (Massoud *et al*., 2009). The simplest form of decentralisation is the onsite system which is stationed at the wastewater generation source; this requires no sewer line network. Decentralisation can also take the form of cluster system where wastewater is collected from a small number of households in a community, in sewers usually much smaller than those in the central system, and led to a small scale treatment plant near the wastewater source (Magliaro and Lovins, 2004; USEAP, 2004). For such systems, when the treatment and disposal is far from the generation source, it becomes a centralised cluster system (USEAP, 2004).

4.1 The Central Sanitation system- often more a problem than a solution in Africa.

The central system is effective and is preferred in many developed countries but, as indicated in the preceding section, its greatest disadvantage is the capital and operational costs associated with sewer systems, making it unaffordable by the developing world (Bakir, 2001). Taking aside middle income countries like Namibia and South Africa and as an exception of Senegal, in general, the sewerage coverage in Africa is very low. Many countries such as Côte d'Ivoire, Kenya, Madagascar, Malawi, Lesotho and Uganda barely reach 10% sewerage coverage (Morella *et al.*, 2008). In many cases the central system exists in big cities and is mainly in the affluent parts of the cities, while the remaining parts are occupied with onsite systems. But, some cities with inhabitant equivalents of 1 million or more like Abuja in Nigeria, Kinshasa in DRC Congo have no public sewage coverage at all (MacDougall & McGahey, 2003) and others like Accra in Ghana have one but it is non-functional (Keraita *et al.*, 2003; Awuah & Abrokwa, 2008; Nikiema *et al.*, 2013). The sewerage coverage percentage and other dominant sanitation systems for selected cities in Africa are shown in Table 1-1. The existing central systems are predominantly waste stabilisation ponds, activated sludge systems and trickling filters (Taigbenu and Ncube, 2005; Samie *et al.*, 2009; Murray and Drechsel, 2011; Nikiema *et al.*, 2013);

Table 1-1: Sanitation coverage in some of the large cities in Africa (source IWA water WIKI)

Country	City	**Sanitation options coverage (%)**				
		Sewerage	Septic tanks	Pit latrines	other	Open defecation
Cote d'voire	Abidjan	40	20	26	-	Significant
Senegal	Dakar	30	63	5	-	Non existing
Tanzania	Dar-es Salaam	<10	20	Other	-	Significant
South Africa	Durban	54	4	4	34	Not common
Zimbabwe	Harare	1-33	47 -85		-	2-13
Uganda	Kampala	7	82%		5	Significant

(-) Data not found

Other systems include urine diversion, community ablution blocks, and other types.

The costs related to central wastewater treatment are not affordable by many households in Africa where for example in sub Saharan Africa the gross national income (GNI) is just above 1300 Euro per capita per year (World bank, 2013). The total cost (capex + opex) of sewerage network plus sewage treatment, in industrialized countries is of the order of 100 Euro per capita per year (Verstraete *et al.*, 2012). This could take up to 10% of the house hold income of many African households and it has a potential to reach 27% of the household income in some other areas (Nhapi and Gijzen, 2004). Such values are unrealistic especially when compared to those of industrialized countries like Germany, France, Belgium, Luxemburg and Netherlands where the total costs of central system is a mere 0.5% of the GNI (more than 20,000 EURO per capita per year in France, Germany and the Benelux). This usually prompts the Government to subsidize these services. For example in Uganda, the Government's new connection policy provides free connection for customers within 60 meters of the sewer mains. Where Governments don't offer support, some plants have ceased operation such as the ones in Ghana (Nikiema *et al.*, 2013; Keraita *et al.*, 2003). However, even with or without Government subsidies, for those that remain in operation, many are old and dilapidated. They have not been extended or replaced since construction and they are poorly operated and maintained with inadequate maintenance plans for the broken moving parts like pumps and motors (World bank, 2003; Taigbenu and Ncube, 2005; Nhapi *et al*. 2006; Hutton *et al*. 2007; Nikiema *et al*.,2013). Apart from that, wastewater plants in Africa have challenges of high organic loads, increasing wastewater flow rates, uncontrolled waste input, power cuts and workers who lack skills and or motivation. (Bakir, 2001; World Bank 2003; Nhapi and Gijzen, 2004; Taigbenu and Ncube, 2005; Nhapi et al. 2006; Nikiema et al., 2013). Additionally, the sewerage system requires continuous supply of electricity (Bakir, 2001; Maurer *et al*., 2006) and high volumes of would be portable water, to transport sewage (Bakir, 2001; Maurer *et al*., 2006), which cannot be sustained in many parts of Africa. As a result, many of these systems are left in a state that makes it impossible to meet their core objective. Through the central systems, large volumes of wastewater are often collected and released to the environments untreated or inadequately treated (World bank, 2003; Nhapi et al. 2006; Hutton et al. 2007), ultimately leading to the continued deterioration of the water quality in the receiving body. More so, contamination of water resources has impacted the health of many Africans as water related disease due to sewage contamination spread. Some of these plants lead to mass destructions as many people die from these diseases when there is a related disease outbreak. An example of such is the worst cholera epidemic outbreak in Africa, which occurred in Zimbabwe in 2008/2009 in which more than 1800 people died (OCHA, 2009). This coincided with a time when there was non-maintenance and breakdown of the sewerage and solid waste disposal systems.

In summary, the central system for the moment is quite a costly method and may pose more challenges than solution for Africa's poor population who may not sustain its proper maintenance. To solve these looming issues, strategies of cost minimisation such as resource recovery and recycling during wastewater treatment have to be considered. Also, there is need for inclusion of an economically viable and sustainable plan to ensure continuous maintenance and operation of these systems, during the initial concept plans of a treatment plant if a country can afford it.

4.2 The onsite system – A prevalent option offering incomplete solutions

The onsite sanitation systems are by far the most popular method in Africa, accounting for 60-100% sanitation coverage in many African cities (WHO 2000). The lack of financial resources coupled with poor urban planning cripple African Government ability to offer centralized sanitation systems. Therefore, most property developers cater for their onsite wastewater treatment systems. Although more than 70 different onsite systems exist (Ho, 2005), the onsite systems in Africa commonly occur in form of simple traditional latrines, septic tanks and Ventilated Improved Pit latrines (VIP) (Morella *et al*., 2008). While the VIP and septic tanks are recognised as improved sanitation by UN, proper management of these systems cannot be divorced from proper faecal sludge management (FSM) which caters for faecal sludge collection, treatment and final disposal/reuse (Kvarnström *et al*., 2004, Mara *et al*., 2007, 2009). In countries like Uganda, Kenya, Tanzania, Rwanda, Zambia, Zimbabwe to mention but a few, cesspool trucks are hired from the private sector players to empty full onsite pits which then transport the contents to a centralized wastewater treatment facility. The hire of cesspool trucks is an arrangement between property and truck owners, which is a big challenge due to the costs involved in hiring cesspool trucks (Katukiza *et al*., 2012). Many households cannot afford to pay for this service whose costs would consume huge amounts of their household income (Boot and Scott, 2009). According to Banadda *et al*., (2009) those that cannot afford, normally take advantage of the rainy seasons to intentionally release the contents of their sanitation systems to the environment for the rain to wash away. From the offender's point of view, this provides a perpetual opportunity to have an operational and maintenance cost free onsite wastewater treatment system. However, the communities pay a heavy price in water quality and disease control. That is why in most African cities rainy seasons are synonymous with cholera outbreaks. Apart from that, even when individuals can afford the emptying services, some cities lack a proper faecal sludge management (FSM) plan (Keraita *et al*., 2003, Katukiza *et al*., 2012) and do not have faecal sludge treatment plants (FSTP), hence, much of the onsite systems' faecal sludge ends up being poured directly into water sources (Keraita *et al*., 2003; Snyman, 2007; Strauss and Montangero, 2002).

Bringing resource recovery in the faecal management scheme may help alleviate this challenge, but some of these sanitation options like the pit latrines and VIP toilets are not truly favourable for resource recovery. On the upside, modifications to include concepts which encourage resource recovery, such as urine diversion (Kvarnström *et al*., 2006) and the Ecological sanitation (EcoSan) technology (Langergraber and Müllegger 2005) are being promoted in some areas in Africa. Other researchers have proposed resource recovery options for faecal sludge in a bid to decrease related FSM costs (Diener *et al*., 2014). Another concern with onsite systems is that they are one of the greatest contributors to ground water pollution. In pit latrines, the liquid phase of wastewater Infiltrates into the ground water and overflows during the rainy season from the excreta collection chamber, making them major causes of ground water pollution (Kulabako *et al*., 2007, Hutton *et al*., 2007). For others on site systems, pollution is as a result of structural failures for example, failing septic tanks were cited to be the third highest source of groundwater contamination in the United States (USEPA, 2005). This is mainly due to the fact that many of these systems are not properly constructed and lack the proper lining to prevent pollution. The onsite systems is likely to remain predominant for some time since it is considered to be a cheap solution for sanitation provision, but its limitations should not be ignored. Firstly, if not designed and constructed to required specifications and if a proper faecal sludge management scheme is not considered, these systems will continue to only offer partial solutions to the sanitation problems. Secondly, as urban population densities continue to rapidly increase, availability of land constrains the use of these seemingly cheaper options. Also, as more piped water is delivered to users, it is likely that per capita consumptions would increase hence increased wastewater challenges. This would ultimately require Africa's growing cities to develop affordable sewage networks in selected areas. Technological innovation aimed at decreasing costs of sewer networks systems will therefore always remain critical for sustainable sanitation management.

5. A more sustainable wastewater management scheme for Africa

The critical sanitation situation in Africa calls for radical changes in current sanitation approaches to include sustainable strategies that will enhance effective and full sanitation coverage. Experiences from other success stories that linked an environmental threat with economic opportunity e.g., the successive collection and recycling of metal scrap and plastic for cash in Uganda indicate that linking challenges and opportunities with possible monetary benefits, could be the silver bullet for a paradigm shift to achieve sustainable waste management. The new sanitation systems should have monetary benefits which ultimately make it affordable to many. Furthermore, it should be easy to use and should produce minimal waste to the environment. The continuation of the system should be ensured not only though funding plans but also through proper management that considers full participation of all stake holders to ensure it is relevant to their needs and aspirations.

5.1 Resource recovery: No longer just another option but a central strategy

Resource recovery in wastewater treatment is a key strategy that can make tremendous contribution to achieving cost effective and sustainable wastewater management. Wastewater has a number of resources which include water, nutrients and energy, whose potential profit recovery was estimated at 0.35 € per m^3 of wastewater as highlighted by Verstraete *et al*., (2009) in Table 1-2 , the prices have gone up since then.

Table 1-2: Potential products recovery from municipal "used water"in the European Union (Verstraete *et al*., 2009)

Potential recovery	**Per m^3 sewage**	**Market prices**	**Total per m^3 sewage (€)**
Water	1 m^3	€0.250/m^3	0.25
Nitrogen	0.05 kg	€0.215/kg	0.01
Methane[a]	0.14 m^3	€0.338/m^3 CH_4	0.05
Organic fertilizer[b]	0.10 kg	€0.20/kg	0.02
Phosphorus	0.01 kg	€0.70/kg	0.01
		Total	**0.35**

[a] Methane produced per m^3 of sewage was calculated on the basis of 80% organic matter recovery as biogas with 0.35 $m^3 CH_4$/kg COD removed.

[b] Organic fertilizer was calculated on the basis of 20% organic matter remaining after anaerobic digestion and the price is based on the agricultural value of organics.

The benefits of managing wastewater systems with focus on reuse and recovery are numerous. It results into decreased waste to be disposed to the environment, contributing to environmental sustainability, in particularly the preservation of the quality of water resources. Another crucial benefit would be the reduced competition for fresh water sources, when wastewater is considered as an alternative water source for different activities. It is known that up to 80 percent of the needed fresh water can be retrieved from wastewater through different optimised reuse strategies (Qin *et al*., 2006). Recovery for domestic use is not common worldwide. However, in Africa, at the Goreangab Water Reclamation Plant (WRP) in Windhoek, Namibia has recycled wastewater for domestic use for its entire design life of 30 years without any problems. It has since been replaced by a new Gorengan WRP with more multi barrier controls (Du Pisani 2006; Lahnsteiner and Lempert 2007). Globally, recovery and reuse of treated effluent has mainly been observed for agricultural use and landscape irrigation as well as the industrial sector (Brissaud, 2010). In Africa, though not widely reported, wastewater reuse is usually demand driven, it is used for irrigation in some areas that are already water stressed such as Morocco, Tunisia, Egypt, Sudan, Namibia, South Africa and Bulayo in Zimbabwe (Shuval et al., 1986; Taigbenu and Ncube, 2005; Nikiema et al,. 2013). In other areas like Kampala, Uganda, intensive crop cultivation is observed in Murchison bay, a wetland receiving wastewater effluent where farmers have simply taken advantage of the fertiliser content in wastewater. When used for irrigation, wastewater provides an added benefit of increased crop productivity (Guillaume & Xanthoulis, 1996; Asano & Levine, 1996; Vazquez-Montiel *et al.*, 1996) due to the nitrogen and phosphorous plant nutrients that are present in domestic wastewater (Verstraete *et al*., 2009; Verstraete and Vlaeminck, 2010; Mulder, 2003). Nitrogen present in domestic wastewater could theoretically cover approximately 30 percent of the current agricultural N demand (Mulder, 2003). The increased crop productivity would ultimately provide an economic benefit to the community. Lastly, as already highlighted, many wastewater plants in Africa are not meeting their objectives which among other reasons is mainly due to high costs associated with wastewater treatment. In conjunction with the aforementioned benefits, a monetary benefit from recovery of resources in wastewater appears feasible. These economic benefits indirectly contribute to achieving low system net operation cost hence increased affordability of sanitation systems. A more direct contribution to cost cutting can occur via energy recovery especially for the big central plants that demand continuous high supply of electricity. During anaerobic digestion, per kg of biodegradable organics, expressed as COD, about 0.35 L $CH4$/g COD at STP can be gained (Vandevivere and Verstraete, 2001). The recovered energy from methane can be used for powering gas engines, producing electrical and thermal energy which would go a long way to reduce operation costs. Unfortunately, not much of this energy potential is tapped in Africa. Nikiema *et al*., (2013) did not find much recovery in his assessment of plants from seven countries. Nonetheless, a few that have ventured to do so derive benefits, such as the Gabal WWTP in Egypt which is reported to cut half of its electricity costs through biogas production (Francoise, 2006).

When resource recovery is practised, one can expect decreased pollution to natural water resources, increased water availability, increased crop yield for farmers, and overall decreased costs. Hence the net outcome will be increased affordability of sanitation options, which will ultimately increase coverage and a quick progress towards the MDGs. Resource recovery in wastewater can therefore no longer be taken as just any other option, as already discussed, this is a central strategy to achieve sustainable wastewater management. A number of good examples in Africa have been cited which can be used as benchmarks for the developing world.

5.2 A vote for the decentralised cluster system for the small rural and sub urban communities in Africa

The cluster decentralised system is similar to the central system but limited to serve a smaller number of individuals and treats and disposes/reuses effluent near the point of wastewater generation. In Africa, these mainly exist to serve mainly institutions like industries, hospitals and schools and a few have been established for small communities. Decentralization in wastewater management is however increasingly gaining recognition as a major strategy towards decreasing the world's population without sanitation (Bieker *et al*., 2010; Larsen and Maurer, 2011; Lens *et al*., 2001), the benefits have been compiled by Libratalo *et al*., (2012). Among them, is the tremendous decrease in the cost when compared to the central systems. These systems require smaller size diameters and smaller collection networks than the centralised system, due to the shorter distance to the treatment location even then they are still expensive for the vast majority of people in urban areas. The collection network alone can take up between 80-90 % of the capital costs in the central system (Otis, 1996; Bakir, 2001; Maurer *et al*., 2006). Cost savings of 50% and 67% were achieved over conventional sewerage in two decentralised settled sewerage systems serving 2500 and 1500 inhabitants respectively in Columbia (Rizo-Pimbo, 1996). Twelve other similar systems in the USA registered a cost savings of 20–50% over the conventional centralised system (Otis, 1996). Also, being that these systems handle smaller volumes and are positioned not far from the wastewater source community, they can be well-suited with demands for resource recovery and reuse by the local communities served (Tchobanoglous, 2003; Raschid-Sally and Parkinson, 2004; Tchobanoglous *et al*.,2004; Ho and Anda, 2004; Ho, 2005; Hong *et al*., 2005; Weber *et al*., 2007; Brown *et al*., 2010; Lens *et al*., 2001). Additionally, small decentralised plants can quite flexibly serve a wide range of communities, they are particularly more preferable for communities with improper zoning, such as scattered low-density populated rural areas (Bakir, 2001; USEPA, 2005; Brown *et al*., 2010) but they also serve well for densely populated communities that lack space for a big plant (Nhapi, 2004; Larsen et al. 2009). In addition to that, small decentralised WWTPs can be viable with simple to moderate technology that is efficient, robust, easy to manage and maintain (Wilderer and Schreff, 2000; Parkinson and Tayler, 2003; Tchobanoglous, 2003; Tchobanoglous *et al*., 2004). Also with a decentralised system it would be possible to separate wastewater streams by pre-concentrating sewage as near as possible to the source. Pre-concentration of solids enables maximal recovery of resources as each stream can be separately treated. Examples of pre-concentration techniques include, the dynamic sand filtration (DSF), dissolved air filtration (DAF), biological sorption direct filtration, centrifugation, flocculation or a combination of any (Verstraete et al., 2009). The other option is the Adsorption Bio-Aeration method where the activated sludge acts as a flocculant (Boehnke et al., 1998).

All these beneficial attributes make the decentralised cluster system a good and practical alternative when compared to the central systems which frequently fail in Africa. It is important to note, however, that these systems also require proper management otherwise their efficient performance and expected benefits may not be realised (Liang and van Dijk, 2010).

5.3 Minimal costs and efficient technology: just part of the solution

A centralized sewage system is very effective if well operated but is also expensive and not affordable by many countries in Africa. Where it has been implemented, the subsequent operation and maintenance cost usually end up failing the functionality of the systems. The widely accepted and affordable simple onsite systems have continuously failed due to lack of an institutional arrangement to ensure proper designs and sustainable faecal sludge management, also resource recovery options are limited. The de-centralised systems are therefore forwarded as the recommended option as they have the ability to have a decreased cost in comparison to central systems and have a potential for optimizing resource recovery especially in an agricultural setting. Resource recovery has to be central and apart of the de-centralised system otherwise the population may not fully embrace it, which would consequently fail the benefits that are anticipated.

The aforementioned strategies are widely known and have worked for some regions (Asano *et al*., 1996; Maeda *et al*., 1996; Angelakis *et al*., 1999), but the question why Africa lags behind as substantially as observed today remains. It appears too simplistic to blame poverty, however, regions of similar economic status appear to be doing better than Africa. For example, in 1990, the improved sanitation coverage for Sub Saharan Africa was slightly better than Southern Asia at 24% and 23%, respectively, but 22 years later, Sub Saharan Africa was lagging behind with a 12% difference at 30% sanitation coverage (WHO and UNICEF, 2014). The considerable difference in progress highlights a structural concern beyond costs. A clear institutional management framework that ensures a continuous funding plan and knowledge dissemination (Nhapi and Gijzen, 2004; Bixio *et al*., 2006) is needed together with social acceptance. A population that has not accepted the severity of the sanitation problem will not accept sustainable more challenging solutions which stresses the need for stakeholder involvement. For example in Ghana, on comparing farm based technologies of achieving an effluent suitable for irrigation, it was observed that Interventions building on farmers' current practices and irrigation systems had the highest potential of adoption (Keraita *et al*., 2008b, 2014). The Windhoek WRP is another example of good practice in Africa, showing that complicated science and technology can be successfully implemented and managed in Namibia. Ensuring excellent water quality was but one of the reasons, but the most important part of Windhoek's direct reuse is possibly the public outreach. Widespread and continued public education campaigns including media campaigns and education of children at public schools coincided with the decision to go for water reuse. As a result the public greatly embraced and supported the project to the extent of deriving pride from it (Du Pisani, 2006; Lahnsteiner and Lempert 2007). Sustainable practices should be included in curricula for students pursuing related carriers, and the practitioners like public health specialists, environmentalists and engineers should be obliged to consider these workable solutions in their regular work. Information should be packaged in simple ways to be appreciated by local communities, political leaders and policy makers. Without public perception change, getting sanitation coverage to all people in Sub-Saharan Africa is likely to remain un-achievable. Public sensitization would also aim to address socio-cultural and religious issues about recovery of resources from a source with faecal pollution. It is of crucial importance that best practices, demonstrating the valorisation of sewage are promoted, within Africa and all over the world.

6. Objectives and Outline of this research

Clearly the current sanitation systems in Africa are not helping its failing sanitation situation, requiring a paradigm shift. Implemented methods should include key sustainable strategies like resource recovery and reuse to enhance economic gains and enhance visible local benefits. Of utmost importance is the fact that the resources thus locally recovered, should come to the benefit of the local habitants leading to a positive feedback effect. Education and demonstration of the benefits of recovery are crucial to achieve positive public response. Simple technologies that treat/separate wastewater as near as possible to the point of generation should be given priority where possible. Therefore the subsequent research chapters explore the possibility of resource recovery and reuse from wastewater treatment.

The outline of this study is summarised in a domestic wastewater management scheme (Figure 1-3). It represents a cluster decentralised system based on the M & M (major and minor) treatment concept proposed by Verstraete *et al*., (2009). The M & M sewage treatment concept advocates for zero waste generation by separating wastewater as near as possible to the source into two distinct streams; the major liquid stream consisting up to 90 % of the flow and the minor solid stream consisting of 10% of the flow. The scheme achieves a closed loop with recycling of resources derived from the two streams, by use of affordable methods which would ultimately lead to a tentatively sustainable sanitation management plan.

In **Chapter 2** the re-use of poly aluminium sludge to enhance pre-concentration of solids in wastewater treatment was investigated.

Co-digestion has been proposed for optimizing the anaerobic process and yielding higher biogas. The concept was adopted in **Chapter 3,** where the minor steam, in this case, primary sludge, was co-digested with cow dung and brewery waste.

Chapter 4 investigates the possibility of pre-concentration of the sludge by methods such as the 100-year old simple high rate oxidation sludge system (HRAS) (2-3 day solid retention time; no nitrification). After separation, the major flow is further treated with use of trickling filters whose media like charcoal are locally available to achieve an effluent that can be reused for other purposes such as crop irrigation, park irrigation, cooling of plants and other uses that require less stringent standard.

In C**hapter 5** the minor stream i.e. sludge coming from the high rate activated system is digested to recover biogas which the community can use to supplement its energy demands like cooking and lighting, or could be converted to electrical and heat energy to be used at the small wastewater plant. Furthermore the possibility of biochar formation from HRAS is explored. Biochar formation is key for sludge use in the agricultural sector and in some cases as an energy source.

In **Chapter 6,** a general discussion on all the work done, application of the concepts together with some future perspectives for further research is presented.

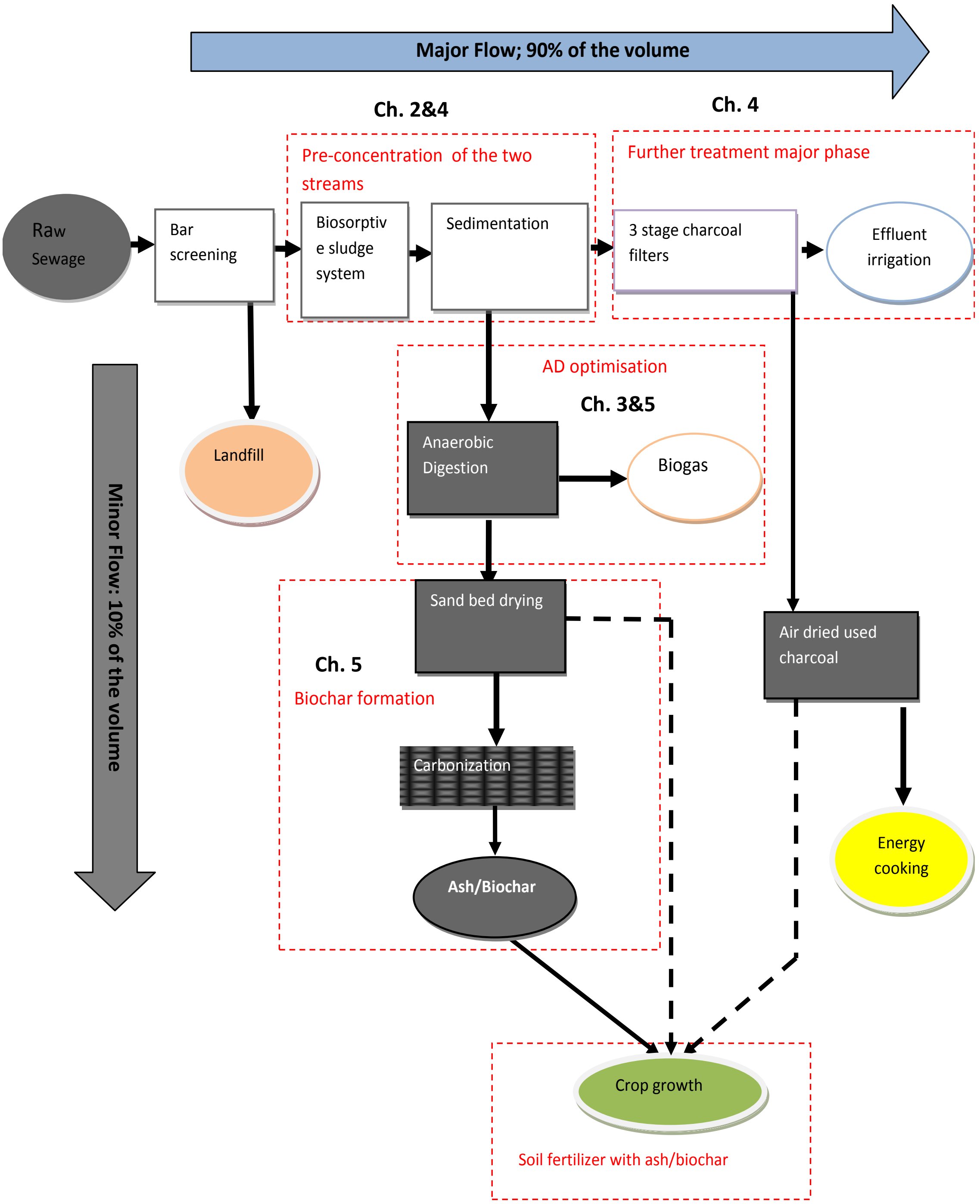


Figure 1-4: Decentralized wastewater management scheme proposed for a small agricultural community. Central in the concept, is to achieve as fast as possible separation of the used water by means of a low cost simple biosorptive sludge system (SRT 2-3 d). Chapters (Ch.) are indicated for each process where applicable.

Acknowledgements

The authors wish to acknowledge the financial support from VLIR-OUS, the Belgian scholarship body and National Water and Sewerage Corporation for further support in Uganda. Willy Verstraete and Korneel Rabaey acknowledge support from the Ghent University Multidisciplinary Research Partnership (MRP) “Biotechnology for a Sustainable Economy” (01 MRA 510W). Korneel Rabaey also acknowledges support from FWO via the FWO-MOST scheme.

Chapter 2 : EFFECT OF POLYALUMINIUM CHLORIDE DRINKING WATER TREATMENT SLUDGE ON EFFLUENT QUALITY OF DOMESTIC WASTEWATER TREATMENT

Nansubuga, I., Banadda, N., Babu, M., Verstraete, W., & Van de Wiele, T. (2013). Effect of polyaluminium chloride drinking water treatment sludge on effluent quality of domestic wastewater treatment. *African Journal of Environmental Science &Technology*. *DOI:10.5897/AJEST12.194*

Abstract

Water resources degeneration is accelerated by the discharge of untreated wastewater and its byproducts, hence, reuse of these wastes is a major contributor to sustaining fresh water for the coming decades. In this study, the reuse of polyaluminium water treatment sludge (PA-WTS) as a flocculant aid to improve the effluent quality of wastewater during primary sedimentation is evaluated and presented. PA-WTS was collected from Gabba water treatment plant (Gabba WTP) Uganda, after the coagulation-flocculation process that makes use of aluminium chlorohydrate (ACH). The average aluminium residue concentration in PA-WTS was 3.4 mg/L. During this study, batch laboratory experiments were conducted in a jar-test apparatus in which different doses of PA-WTS were added. The results obtained showed a decrease in total suspended solids (TSS), chemical oxygen demand (COD), total ammonium nitrogen (TAN), and total phosphates (TP) in the supernatant after 30 min of settlement. The optimal PA-WTS dosage of 37.5 mL/L significantly ($P<0.05$) increased the TSS, TP and COD removal efficiencies by 15, 22 and 30%, respectively. It can be concluded that the PA-WTS positively complimented the sedimentation process in the primary treatment of wastewater to achieve better effluent quality.

1. Introduction

Gabba Water Works in Kampala (Uganda) consists of three water production plants (Gabba I, Gabba II and Gabba III) and is the largest water production works in the country. It has a combined capacity to produce about 230,000 cubic meters per day. Like many other water production plants, the coagulation and flocculation process is employed for turbidity removal at Gabba water treatment plant (WTP). Recently, in a bid to improve efficiency, the Gabba WTP switched from conventional alum to aluminium chlorohydrate (ACH) which can also be referred to as poly aluminium chloride (PAC). PAC is increasingly preferred for water treatment due its lower alkalinity consumption as well as its lower dose requirement (Jiang and Graham, 1998). In other water treatment systems, PAC has a superior ability to inhibit phosphorus release in any anoxic conditions (Yonghong *et al*., 2005). The use of PAC however, still ultimately yields sludge rich in aluminium hereafter referred to as polyaluminium water treatment sludge (PA-WTS), which poses a challenge to dispose. From a chemical point of view, polyaluminum chloride (PAC) is similar to alum, except that the former contains highly charged polymeric aluminium species as well as the monomers. The solubility characteristics of PACs and alum significantly vary (Van Benschoten and Edzwald, 1990; Pernitsky and Edzwald, 2003). PACs are more soluble and have a higher pH of minimum solubility than alum which makes PAC the preferred coagulant nowadays.

When used as coagulants, both PAC and alum yield sludge containing aluminium residues, it can generally be referred to as aluminium sludge. This sludge has a gelatinous appearance, it contains aluminium with a mixture of organic and inorganic materials and hydroxide precipitates. It may also contain water treatment chemical residuals such as polyelectrolytes, powdered activated carbon, activated clay, or unreacted lime. The aluminium sludge is one of the most difficult sludges to treat because of several peculiar properties. It generally settles readily but does not dewater easily. It consists mainly of flocs with water content varying between 95 and 99%, which are the typical levels found in waterworks sludge before and after thickening (Twort *et al*., *2*000). Due to the difficulty in dewatering of the aluminium sludge, in the past the sludge was discharged into water sources, like rivers or lakes. However, nowadays the final disposal of the coagulation sludge occurs by land filling with little prospect of reuse (Hsu and Hseu, 2011).

Literature estimates the worldwide aluminium water treatment sludge to be 10,000 t/day (Dharmappa *et al*., 1997). These volumes will only keep increasing as long as aluminium compounds/complexes remain to be the major coagulant in water purification processes.

Therefore, sustainable management of such sludge continues to become an increasing concern in the water industry. The beneficial reuse of aluminium sludge is highly desirable and has continued to attract considerable research efforts. A number of researchers have already indicated that alum sludge can be a value-added raw material for beneficial reuse. Ferreira and Olhero (2002) proposed a treatment method towards recycling of aluminium rich sludge to produce high alumina refractory ceramics. Hsu and Hseu (2011) and Ulen *et al*. (2012) demonstrated that aluminium sludge can be used to reduce phosphorus availability and mobility during soil amendment. Other sets of studies for example (Yang *et al*., *2*006b; Yang, 2011) have successfully increased removal efficiency of especially phosphorus from constructed wetlands, when the dried aluminium sludge cake was used there. Also, different studies by Chao (2011) and Zhao *et al*., (2008) showed considerable phosphorus removal from stabilisation ponds and reed bed treatment systems, respectively when aluminium water treatment sludge was reused. When aluminium hydroxide sludge was discharged to a sewer in a treatment plant, phosphate removal was up to 94% (Horth *et al*., 1994). Similarly, Guan *et al*., (2005) observed that both suspended solids (SS) and COD removal efficiencies were improved by 20 and 15%, respectively, when Al-WTS was reused in primary sewage treatment.

A number of studies have already given insight into reuse of alum sludge, but many water treatment plants are now adopting PAC whose sludge characteristics differ from alum sludge. It is therefore necessary to study the possibility of re-use of sludge derived from water treatment where PAC is used. It is against this background that this study sought to explore the reuse of PA-WTS for wastewater treatment. The study aimed at studying the effect of PA-WTS on the settling ability of wastewater during wastewater treatment. PA-WTS was mixed with wastewater before settling. Low rate mixing was used to minimize energy input while at the same time enhancing flocculation. The effect of different doses of PA-WTS from Gabba water treatment plant (Kampala) on the primary treatment of wastewater was monitored.

2. Materials and methods

2.1 Sample collection

PA-WTS was collected at three instances from Gabba II water treatment plant, in February and March 2012. Gabba Water Works in Kampala is the largest water production plant complex in Uganda. It consists of three water production plants, Gabba I, Gabba II and Gabba III whose individual capacities are 70,000, 80,000 and 80,000 m^3/day, respectively.

Water treatment at Gabba WTP II is done in the order of screening, pre-chlorination, clarification, coagulation, flocculation, sedimentation, rapid gravity filtration, post chlorination and finally pH correction. The plant uses ACH ($Al_2(OH)_2Cl$) during flocculation, whose ions remain as a residue in the sludge. Domestic wastewater was collected at the inlet of Bugolobi sewage treatment plant (STP) in Kampala, Uganda. The STP is the largest sewage treatment plant in Uganda. It employs physical and biological treatment by use of screens, detritus basin, primary, settling tanks, trickling filters and clarifiers in that order.

2.2 Experimental set up

The characteristics of Gabba II PA-WTS as well as the domestic wastewater were determined at the beginning of each experimental run. Bench tests were run in which different volumes of PA-WTS were added per liter of sewage (0, 12, 25, 37.5, 50, 62.5, 75, 87.5, 100, 112.5, 125, 137.5 and 150 mL of PA-WTS per liter wastewater). These doses had a corresponding Poly-aluminum concentration of 0, 0.03, 0.07, 0.10, 0.14, 0.17, 0.20, 0.24, 0.27, 0.31, 0.34, 0.37, 0.41 mg PA/L wastewater, respectively.

2.3 Selection of the mixing time

To determine the suitable mixing time, the experiments were done at varying times of 0, 5, 10 and 20 min. The time tested was limited to 20 minutes as higher residential times would increase the cost since it would require a larger reactor in operation and a larger impeller. A mixing rate of 25 rpm was used to minimize high energy costs considering its application in the developing world. Upon mixing for the given times and rate indicated above, the mixtures were left to settle for 30 min. After the settling period, samples from the supernatant were taken and TP, COD, TAN and TSS were analysed with HACH DR 5000 Spectrometer using the standard methods (APHA, 2005). The pH was measured with a Toledo pH meter. The same parameters were determined for the wastewater prior to any treatment.

2.4 Selection of optimal dose and data analysis

The suitable mixing time selected from the procedures above was used for further experiments of determining the optimal PA-WTS dose. Bench tests for each dose were done in triplicates at this mixing time and rate, and the same parameters were measured. Removal efficiencies of the analysed parameters at different doses of PA-WTS were then compared to get the optimal sludge dose. The dose corresponding to the maximum gradient of the removal efficiency curve was selected as the optimal dose. The optimum dose and control experiments were repeated 10 times to ensure reliability of the results. An analysis of variance was done to verify the significant difference between parameters measured at the two doses**.**

3. Results and discussion

3.1 PA-WTS and untreated sewage characteristics

The characteristics of the Gabba II PA-WTS and raw wastewater from Bugolobi STP are shown in Table 2-1. The results show that the average residual aluminum in the PA-WTS was 3.4 mg/L. These are much lower doses than what has been used in other studies using alum, for example it was 313 mg Al/L alum sludge for Horth *et al*. (1994). One of the the advantages of using pre-polymerised inorganic coagulants over alum, is their lower dose requirement (Jiang and Graham, 1998). This typically yields low aluminium concentration for sludge originating from ACH coagulants in comparison to that originating from alum.

The results of the Bugolobi STP wastewater show that it is of very high strength (Metcalf and Eddy, 1991). The maximum values TSS, TP, TAN and COD of 8 samples of BSTP wastewater sampled at different times were 876, 20, 51 and 1442 mg/l, respectively. The wastewater characteristics are known to vary depending on the conditions.

Table 2-1: Average ± SD of selected parameters of the PA-WTS and raw wastewater from Bugolobi STP used in this study.

Parameter	PA-WTS	Raw wastewater
TSS **(mg/L)**	1084± 41	563 ± 179
COD **(mg/L)**	2260 ±176	1197 ± 248
TAN **(mg/L)**	11 ± 2	35 ± 13
TP **(mg/L)**	14 ± 3	15 ± 5
pH	7.2 ± 0.4	7.9 ± 0.3
Residual Aluminum **(mg/L)**	3.4 ± 0.3	ND

3.2 Selection of mixing time

Generally, the concentration of all other parameters with the exception of TP and TSS did not differ at various mixing times (Figure 2-1 (A-D)) for a mixing rate of 25 rpm. This implies that mixing time is not important for removal of TAN and COD. On the other hand, generally

TP in the supernatant at all mixing times of 0, 5, 10 and 20 minutes decreased with increased dose of PA-WTS but decreased more at 20 and 10 minutes (Figure 2-1-**C**). The concentration of TSS in the supernatant at different doses and mixing times are shown in Figure 2-1-D. Generally, TSS concentration in the supernatant at all mixing times of 0, 5, 10 and 20 minutes decreased with increased dose of PA-WTS. The final concentration of TSS at zero mixing was constantly higher than that at 5, 10 and 20 minutes for all the doses of PA-WTS. Mixing increases contact between PA-WTS flocs and suspended matter, hence more decrease of TSS is observed in the supernatant of the mixed samples. The mechanisms for removal are discussed at a later stage in this study. The mixing time of 5 minutes was selected as the suitable mixing time since it was the smallest time that could achieve more TSS decrease.

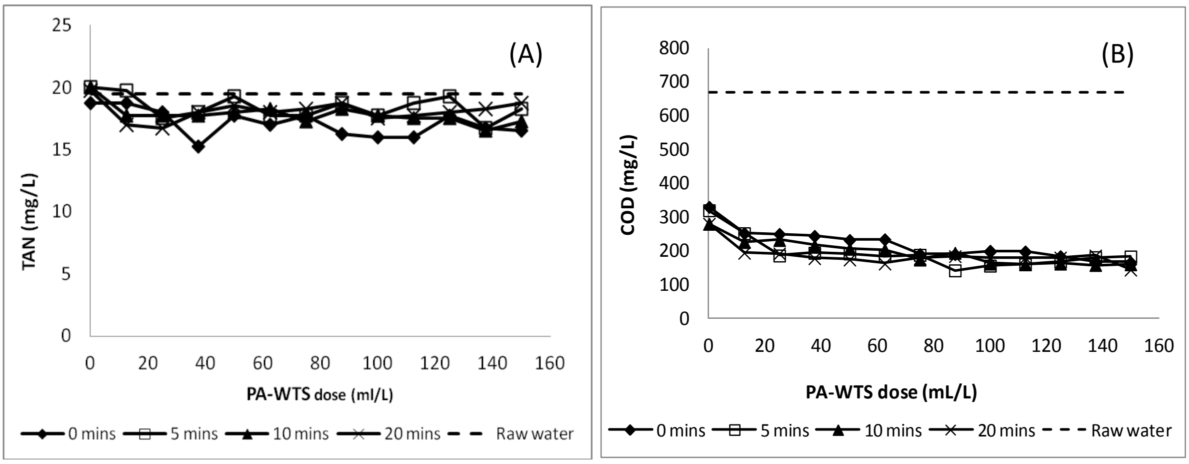


Figure 5: (A) Total Ammonium Nitrogen (TAN) and (B) Chemical Oxygen demand (COD) values for wastewater supernatant after adding different PA-WTS doses at different mixing times and settlement time of 30 minutes

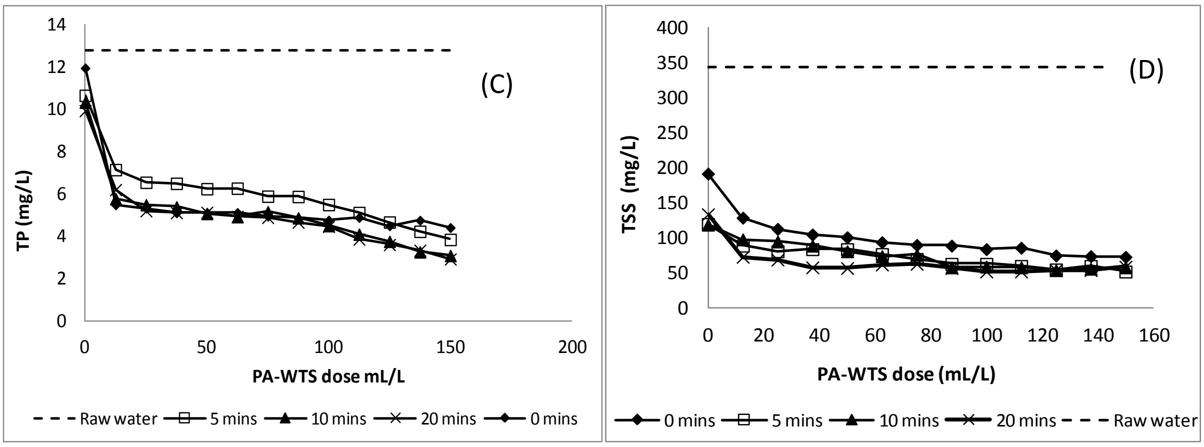


Figure 2-1: (C) Total phosphorous (TP) and (D) Total suspended solids (TSS) values for wastewater supernatant after adding different PA-WTS doses at different mixing times and settlement time of 30 minutes.

3.3 Selection of optimal dose

To select the optimal dose, the removal efficiency of different parameters at varying PA-WTS doses was compared. The pH (data not shown) was observed to be constant with increase in the PA-WTS dose throughout the study. A pH of 8.0 was maintained in one of the sets, of experiment, while the other sets maintained a pH of 7.8. The pH has been found to affect coagulation and flocculation. Optimum pH values for re-use of alum sludge were proposed to be between 6 and 10 for simultaneous removal of TSS, turbidity, and anionic surfactants. On the other hand, the optimal pH for the removal of total COD was between 8 and 12 (Siriprpah *et al*., 2011) and optimal pH removal for phosphorus during coagulation is between 5 and 7 (Jiang & Graham, 1998). The pH between 7-8 maintained in our experiment can be said to be within an optimal range for TSS and COD removal.

All other measured parameters generally decreased with increased PA-WTS dose (Figure 2-2). The average removal efficiency of TSS in the supernatant kept increasing with increase in PAL-WTS dose. The influence of the PA-WTS dose on the COD in the wastewater is also shown (Figure 2-2). The mean COD removal efficiency in the supernatant generally increased with initial increase in PA-WTS doses. This is in agreement with other studies which showed that TSS and COD can be removed by use of alum sludge (Guan et al. 2005; Yang et al. 2011). However, our study shows a slight COD decrease after a PAW-WTS dose of 90 mL/L. The average TP removal efficiency increased slightly with the least PA-WTS dose and kept increasing slightly with further increase in the PA-WTS dose (Figure 2-2). Similar trends are shown for TAN.

As illustrated in Figure 2-2, the maximum gradient removal was observed to occur at PA-WTS doses beween 0 and 12.5 mL for TSS, 0 and 37.5 mL for TP, 0 and 37.5 mL for TAN and between 0 and 25 mL for COD. The dose of 37.5 mL PA-WTS /L was hence chosen as the optimal dose in order to cater for all doses which showed maximum gradient removal.

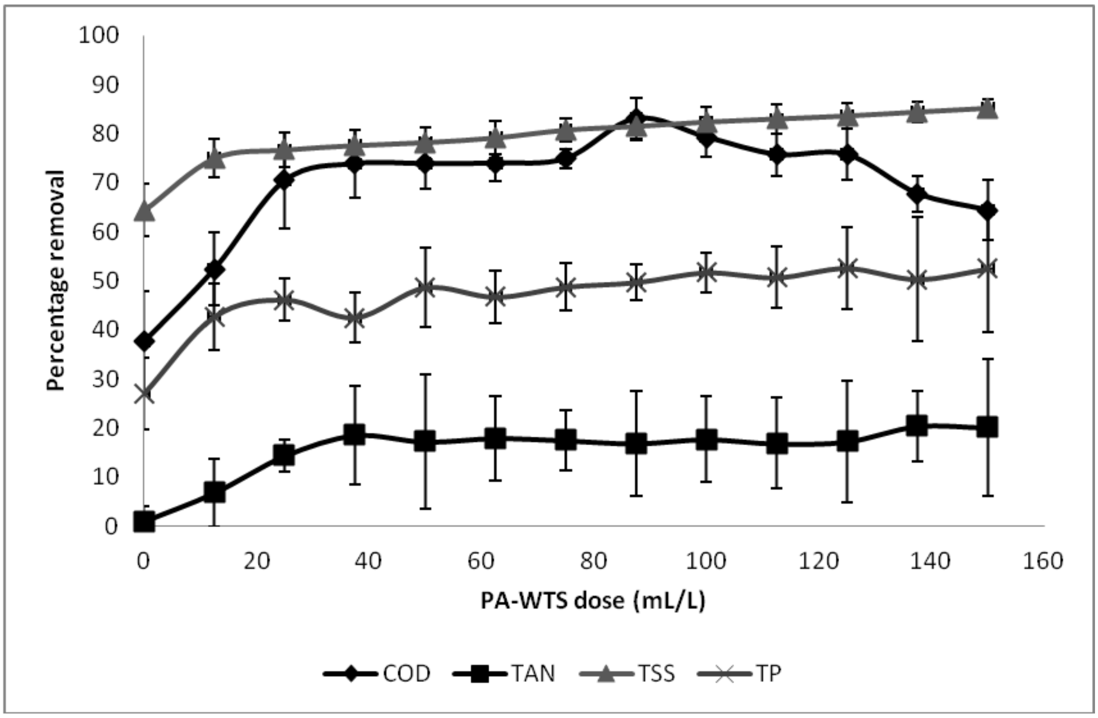


Figure 2-2: Effect of different doses of the PA-WTS on COD, TSS, TAN and TP removal efficiency from wastewater.

3.4 Comparison at Optimal dose

Experiments were repeated with the optimal PA-WTS dose (37.5 mL PA-WTS /L) in comparison to the control (0 mL PA-WTS /L). The average percentage removal efficiencies of TSS, TP, TAN and COD in the supernatant at both doses were compared and are shown in Figure 2-3. Analysis of variance test showed homogeneity for all parameters except TAN and further revealed significant difference between the measured parameters at the two doses except for TAN. It was found that the optimal PA-WTS dosage of 37.5 mL/L (0.14 mg Al /L) significantly ($P<0.05$) increased the removal efficiency of TSS from 64±6, to 78±3, TP from 26±7 to 48±8, and COD from 43± 7 to 74± 5 (Figure 2-3). TAN removal efficiency was however not significantly different for the two doses, but the trend was that it increased from 1± 3 mg/L to 19±13 mg/L **(**Figure 2-3**)**.

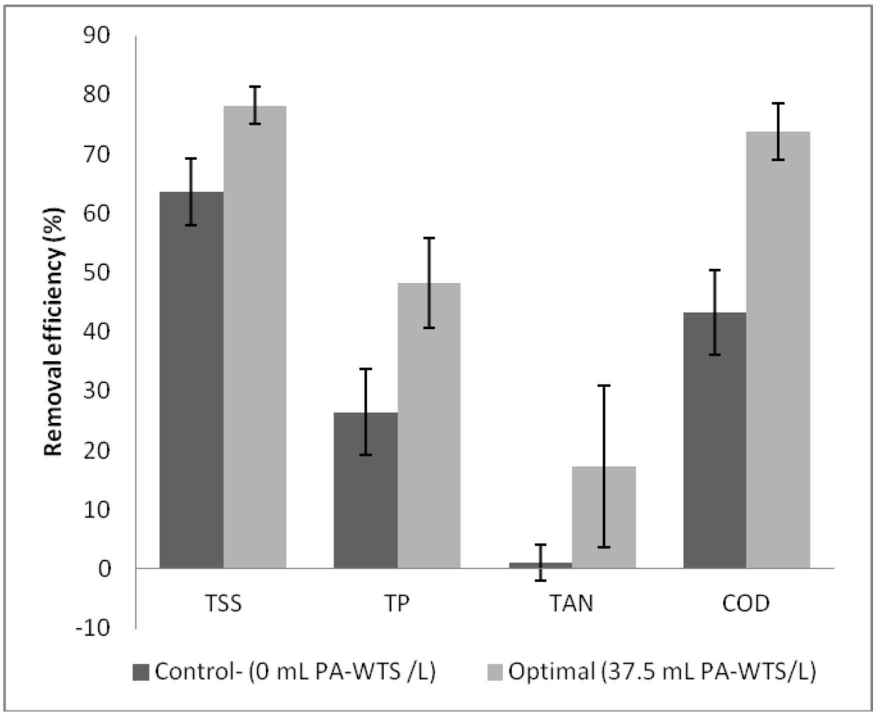


Figure 2-3: Average percentage removals ± SD of COD, TSS, TAN and TP for the control and the optimal dose at 5 minutes mixing time and settlement of 30 minutes.

The removal efficiency were calculated by comparing the decreased in selected parameters observed in the supernatant after mixing and settlement with a dose of 37.5 mL PA-WTS and comparing it to the raw water. For the Control, the supernatant values after mixing and settlement with no PA-WTS added, were compared to the raw water values. On average the removal efficiencies of TSS, TP, TAN and COD were increased by 15%, 22%, 18% and 30% respectively at the optimal dose of 37.5 mL/L (0.14 mg Al /L). These are higher removals per aluminium concentration when compared to the removal increments observed by Guan et al. (2005). The latter authors observed an increment of 20% and 15% for SS and COD respectively at a sludge dose of 18–20 mg Al/L when alum sludge was used. This may arise due to the difference in properties of the two sludges which enhance different removal mechanisms during flocculation. The four distinct mechanisms of coagulation and flocculation include double layer compression, adsorption and charge neutralization, sweep coagulation, and inter particle bridging/complexion (Amirtharajah *et al.*, 1991). Alum sludge usually yields flocs with a negative charge, which is similar to the charge in wastewater. Particulate pollutant removal efficiency in the alum sludge is therefore predominantly as a result of the sweep mechanism and not necessarily neutralisation (Guan *et al*., 2005). In contrast, the flocs formed with the high basicity non-sulfated PAC, which is typical of the sludge used in this experiment, exhibit a higher positive charge at a pH of above 7 (Pernitsky and Edzwald, 2003). This positive charge is likely to enhance neutralisation which would contribute to more particulate removal when PA-WTS is used. The neutralisation contribution may however still be small compared to the sweep mechanism since as discussed already, the residual aluminium in PA-WTS is small compared to that in the alum sludge. Another fact that could lead to higher removal when PA-WTS was used can be explained by the observations of Gregory *et al*., (2001). On comparing alum and PAC coagulants, they observed that PAC products form larger and stronger flocs than alum. It can be anticipated that larger flocs will sweep out more particulate matter than the smaller flocs. The PA-WTS used in this study can therefore be said to have sufficient floc sizes on which particulate matter attach when gently stirred and hence settle out faster than for samples without PA-WTS. Hence the supernatant TSS and COD in this study kept decreasing with increase in the sludge dose because higher doses of PA-WTS had more flocs. These could sweep out more particulate matter from the wastewater.

Evidence from literature shows that aluminium sludge can help remove phosphorus in wastewater (Horth *et al*., 1994; Yang *et al*., 2006b; Yang *et al*., 2011). The removal is accredited to adsorption and chemical precipitation enhanced by the abundant presence of aluminium ions in the sludge (Kim et al. 2003). In addition, Yang et al. (2006b) showed that the adsorption capacity can be affected by pH and the different ions present. They observed a remarkable decrease in phosphorous (P) adsorption capacity of the aluminium sludge when the pH was increased from 4.3 to 9. Compared to the mentioned studies, it is clear that the P adsorption capacity of the aluminum sludge in this study was negatively impacted by low aluminum ions in the PA-WTS combined with the pH of 7.8 and 8 that was imposed. The removal efficiency of TP was 45% (Figure 2-3) with the optimal Al dose of 0.14 mg Al/L compared to other studies which achieved more than 90% phosphorus removal. Horth et al (1994) observed phosphate removal up to 94%, at an alum sludge dose of 94 mg Al/L. Similarly, soluble phosphorus removal from a stabilisation pond went up to >90% with a dose of sludge of 131 mg/L (Yang et al. 2011).

4. Conclusions

PA-WTS was added to wastewater as a flocculant aid with an objective to determine if it will improve effluent quality during sedimentation. There was an increased removal of TSS, TP, TAN and COD in the Bugolobi STP wastewater supernatant after mixing it for 5 min at a rate of 25 rpm and allowing it to settle for 30 min. The wastewater was prior dosed with PA-WTS doses of 0, 12, 25, 37.5, 50, 62.5, 75, 87.5, 100, 112.5, 125, 137.5 and 150 mL PA-WTS/L. The study showed that an optimal dose of 37.5 mL PA-WTS /L significantly increased the removal efficiency of TSS, COD and TP from water during sedimentation. TSS, TP and COD removal efficiencies were significantly increased by 15, 22 and 30%, respectively. Based on this study, it can be concluded that incorporating PA-WTS dosing before the primary settling unit is a promising venture towards better effluent quality in wastewater treatment systems. For the existing plants, modifications done to allow mixing of PA-WTS before primary settling, would go a long way in improving effluent quality of the settling tank. While for the new plants, the design size of the settling tank can be decreased since a shorter retention time is needed with PA-WTS. Given the observed increased TSS removal efficiency of 15% and assuming the settling tank covers a third of the total cost of a simple treatment unit as described in this study. The required capital costs for the new plant can be lowered by about 5%, in addition to producing better effluent.

Acknowledgements

The authors wish to acknowledge the financial support from VLIR the Belgian scholarship body and National water and Sewerage Corporation for further support in Uganda. They also wish to acknowledge Galyaki Cyrus, Sylvia Nabateesa, Ritah Kamiti and Profilio Tebandeke for their help with field and laboratory work. Willy Verstraete acknowledges support from the multidisciplinary Research Partnership Gent Bio. Economy. The scientific responsibility is assumed by its authors.

Chapter 3 : ENHANCEMENT OF THE BIOGAS POTENTIAL OF PRIMARY SLUDGE BY CO-DIGESTION WITH COW DUNG AND BREWERY SLUDGE: THE EFFECT ON KAMPALA'S (UGANDA) WASTEWATER TREATMENT

This chapter has been redrafted after:

Nansubuga, I., Banadda, N., Babu, M., De Vriez, J., Verstraete, W., & Rabaey, K. (2015). Enhancement of biogas potential of primary sludge by co-digestion with cow manure and brewery sludge. *International Journal of Agricultural and Biological Engineering*, *8*(4), 86-94.

Abstract

Energy production from wastewater is not common in the developing world when compared to the developed world. More often, organic waste is considered a waste than a resource and is usually improperly disposed. Consequently, the quality of water resources has been compromised leading to high costs in water treatment. A study has been conducted at Bugolobi Sewage Treatment Plant (STP) where two organic wastes, cow dung and brewery sludge were co-digested with primary sludge in different proportions. The study was done in lab-scale reactors at mesophillic temperature and sludge retention time of 20 days. The aim was to evaluate the biodegradability of primary sludge generated at Bugolobi Sewage treatment plant (STP), Kampala, Uganda and try to enhance biogas production from it. When the brewery sludge was added to primary STP sludge at all proportions, the biogas production rate increased by a factor of ≥ 3. This was significantly ($p<0.001$) higher than that observed (159 to 186 mL/L.d) in the control treatment containing only STP sludge. Co-digesting STP sludge with cow dung alone did not show different results compared to the control treatment. In conclusion, Bugolobi STP sludge as such is poorly anaerobically degradable with low biogas production but co-digestion with brewery sludge, greatly enhanced the biogas production rate, while co-digestion with cow dung alone was not beneficial.

1. Introduction

The Bugolobi sewage treatment plant (BSTP) located in Kampala is the largest sewage treatment plant in Uganda. It was designed to treat 33,000 cubic meters of wastewater per day but it only receives an average flow of 12,000 cubic meters per day. The plant treats sewage using a coarse and fine screen, a detritus basin, two settling tanks in parallel, followed by trickling filters and finally by clarifiers. The sludge from the plant is left to stabilize in open semi-anaerobic digesters, before it is sent to a set of drying beds, where it is left to dry before it is sold to farmers as dry organic fertilizer. The plant, which has been in existence since the late 60s, is quite dilapidated and releases biogas that is generated at the open semi-anaerobic tanks where sludge is stabilised into the air. This contributes to greenhouse gas emissions and a lot of odour nuisance to the surrounding areas.

Fortunately, the old plant is already planned to be replaced by a new one, which will have similar treatment processes but whose sludge will undergo further treatment by anaerobic digestion. Despite the fact that a new treatment plant will be constructed, information on the performance of Kampala sewage sludge with regard to biogas production is not available. This study was carried out in order to obtain information concerning the digestibility of the sludge that is generated.

Additionally, Kampala city has a number of abattoirs whose wastes have become an environmental threat since most of it is discharged untreated in the nearby Nakivubo Channel reaching Lake Victoria. Also, a nearby brewery plant is in need of a cost friendly disposal method for its brewery waste. Co-digestion of sewage sludge with these substrates could not only enrich the operational and optimization process of the new plant, but it could also improve the environmental quality of the Northern shores of Lake Victoria

Anaerobic digestion (AD) has long been used for stabilising organic matter, such as sewage sludge and cow dung. Apart from stabilising the substrates, AD of sludge has increasingly been applied in the production of biogas (Appels *et al*., 2008). The biogas produced in the anaerobic process can be considered a valuable source of energy and electricity. Substantial effort has been geared towards optimising the AD process to increase biogas production. This has led to studies aiming at improving reactor design, optimizing AD process parameters and manipulation of substrates (Ahring 2003, Angelidaki and Sanders 2004, Lissens *et al.*, 2004, Appels *et al*., 2008). Indeed, AD has since broadened to include other waste streams, such as energy crops, fats and kitchen waste. Substrate-focused AD optimisation considers the selection of suitable substrates and their combinations (Hansen *et al*., 1998, Hamzawi *et al*., 1998, Van Lier *et al*., 2001) as well as nutrient availability (Hinken *et al*., 2008) and pre-treatment of the substrates to make them more amendable for AD (Weemaes and Verstraete, 1998; Weemaes *et al.,* 2000; Hansen *et al*. 2007; Lagerkvist *et al*. 2012; Ma *et al.,* 2011). While substrate manipulation may improve the AD process, some challenges still remain, due to the different limitations associated with the properties of the different substrate (Hansen *et al*., 1998; Callaghan *et al*., 1999). Continued studies are therefore imperative to further establish the best designs, environment and substrate mixtures to optimise biogas production in AD.

The present study was aimed at evaluating the biodegradability of primary sludge generated at Bugolobi STP. It further sought to explore the possibility of optimizing biogas recovery by means of co-digestion of the primary sludge with cow dung and brewery sludge in different proportions.

2. Materials and methods

2.1 Substrates for co-digestion

Three different feed stocks, i.e. primary STP sludge (STP sludge), cow dung (CD) and brewery waste sludge (BW) were manually mixed in different proportions and used for anaerobic digestion. STP sludge was collected from the primary settling tanks at Bugolobi STP in Kampala, Uganda. Fresh cow dung was collected from the Makerere University farm in Kampala. Water was added to the cow dung to reduce its dry matter content, thus making it easier to pour. Brewery waste sludge was collected from East African Brewery (EABL). The substrate was prepared as such that primary STP sludge was mixed with cow dung, and brewery sludge in different proportions that were labelled as follows; S_0 (100% STP sludge), S_1 (75% STP sludge and 25% cow dung), S_2 (50% STP sludge and 50% cow dung), S_3 (75% STP sludge and 25% brewery waste), S_4 (50% STP sludge and 50% brewery waste), S_5 (50% STP sludge, 25% cow dung and 25% brewery waste) and S_6 (100% brewery waste) The ratios were selected to have at least 50% STP sludge in each substrate mixture since in normal operations of the digester, priority would be given to STP sludge treatment. The substrates were stored at 4°C after mixing as feeding progressed.

2.2 Experimental set-up

The experiment to determine the biodegradability and digestibility of STP sludge, brewery sludge and cow dung mixtures was set up at laboratory scale using glass bottles with a total volume of 1 litre as anaerobic reactors. Seven anaerobic reactors, each filled with 700 mL of anaerobic inoculum sludge obtained from the East African Breweries Limited UASB wastewater treatment plant in Kampala (Uganda), were incubated at mesophilic conditions (36±1°C). The inoculum sludge was initially diluted with water in a ratio of 1:1. Each of the seven continuously stirred tank reactors (CSTR) were fed with only one of the seven substrates S_0, S_1, S_2, S_3, S_4, S_5 and S_6. The anaerobic reactors were operated for 72 days. During the start-up period, the daily organic loading rate was started at 0.71 g COD/L.d and it was gradually increased until the desired sludge retention time (SRT) of 20 days was reached. The hydraulic retention time (HRT) was also 20 days. Each reactor was performed in duplicate and the average results were reported.

2.3 Analytical techniques

2.3.1 Characteristics of the inoculum sludge and the substrate

On a weekly basis, samples were taken from the substrates and inoculum and total phosphorous (TP), chemical oxygen demand (COD) and total ammonium nitrogen (TAN) were determined using a HACH DR 5000 Spectrometer, as described in Standard Methods (APHA, 2005). The pH was measured with a Toledo pH meter. Volatile solids (VS) and total solids (TS) were also analysed according to Standard Methods (APHA, 2005).

2.3.2 Gas and pH monitoring

The biogas produced in the anaerobic reactors was captured in 2000 mL plastic transparent measuring cylinders. The cylinders were inverted in a basin with an acidic solution of water and HCl (pH < 4.3), to avoid the dissolution of CO_2. Air tight plastic tubing from each reactor was connected to an inverted cylinder. To enable direct measurement of the gas produced, the columns were graduated with volume markings and the volume of gas produced deduced from the displaced liquid volume within the columns. To enable a quick identification of potential changes in the acidic condition of the solution within the columns, this solution was treated with methyl-orange indicator. Biogas production and pH in the reactors were monitored on a daily basis for 72 days. To determine the biogas composition, the gas was collected in gas bags from each reactor, on two different days after a SRT of 20 days was reached. The samples were then taken to the College of Engineering, Design, Art, and Technology (CEDAT), Makerere University for analysis. The gas analyzer (Model GC 2000 PLUS) was then used to determine the methane and carbon dioxide percentage in the biogas. The average of the two measurements is reported.

2.3.3 Statistical methods

The results from the two experiments were considered and are reported as likely ranges.

2.3.4 Effluent sludge characteristics

Samples of the effluent from the anaerobic reactors were collected and analysed on a weekly basis for TS, VS, COD, TP and TAN.

3. Results and discussion

3.1 Feed characteristics

The composition of the raw STP sludge, cow dung, brewery sludge and the inoculum are shown in Table 3-1. Brewery sludge was slightly acidic with a pH of 4.4, while the pH in the STP sludge, cow dung and the inoculum was at neutral pH with values of 7.2, 6.8 and 7.0, respectively. In the feed mixtures S_1, S_2, S_3, S_4 and S_5 the pH was 7.1, 7.0, 6.5, 5.5 and 6.2, respectively. TAN was highest in the cow dung while COD and TP were highest in the brewery waste.

Table 3-1: Parameters of the primary STP sludge, brewery waste, cow dung and the inoculum.

Parameter	Inoculum	STP-sludge	Brewery sludge	Cow dung
COD_t (g/kgWW)	10	48	150	61
TS (g/kg WW)	14	31	62	40
VS (g/kg WW)	12	16	48	29
TAN (mg/kg WW)	48	92	67	160
TP (mg/kg WW)	238	299	655	346
pH	7.0	7.2	4.4	6.8

WW = wet weight

3.2 Operational parameters of the different reactors during stable operation at a SRT of 20 days

The operational parameters measured at a SRT of 20 days are shown in Table 3-2. The average pH at a SRT of 20 ranged between 7.0 and 7.4 for the reactors. On a few occasions the pH of digesters with substrates S_3, S_4, and S_5 decreased below 7.0, reaching a minimum pH of 6.5, 6.3 and 6.9 respectively. In such occurrences, a 0.1 N molar solution of NaOH was used to correct the pH to a range of 7.0 - 7.6. The digester with substrate S_4 required more frequent pH adjustment than the other reactors. The pH in the reactor that received 100% BW was maintained between 6.3 - 7.3, until the OLR exceeded 5.3 g COD/L.d and it subsequently reached a value of 5.5. It was not possible to maintain the pH above 7, in this reactor after that, even with the addition of the solution of 0.1 N NaOH, hence it failed at a SRT of 28 days.

The average pH at SRT of 20 for all digesters (except when 100% brewery waste was used) was in the proper range required for efficient anaerobic digestion as indicated in Table 3-2. The generally accepted range for good process efficiency is 6.5 -7.6 (Parkin & Owen, 1986). This indicates an adequate buffering capacity, as well as stable operation for the anaerobic reactors receiving substrates S_3, S_4 and S_5 that had an initial pH below 7.0. The reactor with S_6 also had an initial pH below 7.0 but failed before reaching a SRT of 20 days, due to organic overloading, as discussed later. The other three digesters (S_0, S_1 and S_2) had a constant pH ranging between 7.0 - 7.6 throughout the entire experimental period of 72 days.

The loading rate was increased slowly from 0.71 g COD/L.d and was maintained at a value of 2.0 for S_0, 2.5 for S_1, 2.7 for S_2, 3.7 for S_3, 4.9 for S_4 and 3.8 g COD /L.d for S_5 at a SRT of 20 days. At an organic loading rate of 5.3 g COD/L.d and a SRT of 28 days, the reactors that received 100% brewery waste completely failed (data not shown). Overloading during anaerobic digestion can disrupt the operational stability of the digester. Increased loading rates may cause an accumulation of fatty acids which consequently causes the pH to drop to conditions which can inhibit methanogenic activity (Appels *et al*., 2008; Chen *et al*., 2008). This implies that the loading rates at a STR of 20 days in the digesters with S_1, S_2, S_3, S_4, and S_5 did not generate residual levels of VFA that could limit the methanogenic activity.

Table 3-2: Operational parameters at a SRT of 20 days for the 6 digesters (S_0 to S_5), that reached a stable performance. S_6 is not shown as it failed before reaching a SRT of 20 days.

Parameter	STP-sludge (S_0)	75% STP : 25% Cow dung mix (S_1)	50% STP : 50% Cow dung mix (S_2)	75% STP : 25%Brewery waste-mix (S_3)	50 % STP : 50%Brewery waste mix (S_4)	50 % STP : 25% Brewery waste :25% cow dung mix (S_5)
Weight influent (g/L.d)	50	50	50	50	50	50
SRT = HRT (d)	20	20	20	20	20	20
OLR (g COD/L.d)	2	2.5	2.7	3.7	4.9	3.8
OLR (g VS/L.d)	0.8	1	1.1	1.2	1.6	1.4
Range of Biogas yield ± SD (mL gas/g COD)	159 to 186	160 to 171	129 to173	230 to 387	392 to 405	287 to 365
Average Methane yield ± SD (mL gas/g VS)	196 to 229	187 to 200	146 to 195	442 to 743	682 to 728	425 to 541
Average Biogas production rate ± SD (mL gas/L.d)	318 to 372	399 to 427	347 to 467	851 to 1430	1921 to 1937	1090 to1388
pH ± SD	7.4 to 7.5	7.3 to 7.4	7.2	7.3 to 7.6	7.0 to 7.2	7.3 to 7.5

NB. Likely averages at the SRT of 20 days for the two experiments are considered

3.3 Biogas yield

The biogas production was monitored by following the water levels in the gas columns every two days. The biogas yield (Figure 3-1) and the biogas production rates (Figure 3-2) were derived from the daily gas readings as established from each digester from one of the tests.

From these results**,** it can be noted that STP sludge alone has a low biogas yield and biogas production rate. The average biogas yield in the control digester of S_0, after a steady state SRT of 20 days was reached, ranged between 159 mL/g COD to 186 mL/g COD, indicating that biodegradability is quite low. The STP sludge had a methane yield ranging from 0.20 to 0.22 m^3/kg VS fed which is less than the range estimated by Zhao and Viraraghavan (2004) for primary and secondary sludge (0.24 - 1.01 m^3/kg VS fed) and those reported by Luostarinen *et al*., (2009), for sewage sludge (0.28 - 0.32 m^3/kg VS fed). Also, Parkin and Owen (1986) estimated the standard methane yield from primary sludge at a SRT of 20 days at a value of 643 mL/g VS fed. Primary sludge is usually composed of natural fibres, fats and other solids that settle in the primary clarifier of a wastewater treatment plant, and in contrast to waste activated sludge (WAS), it normally displays a relatively high biodegradability (Pakin and Owen, 1986, Miron et al., 1996). The results from our study indicate that the primary sewage sludge at Bugolobi STP is poorly anaerobically digestible. The reason for the poor digestibility was not determined in this study, but it is suspected to be due to factors, such as long travel times to the treatment plant. The long sewage pipe distance (average of 12 km and 100 m manhole spacing) and the high temperatures (about 24°C), favour growth of sulphate reducing bacteria (SRB). SRB can obtain energy by oxidizing organic compounds or molecular hydrogen (H_2) while reducing sulfate $(SO_4)^{-2}$ to hydrogen sulfide (H_2S).In this process the SRB could consume the organic matter which would otherwise be converted to biogas (Appels *et al.,* 2008). The long travel times can also encourage degradation before digestion given the high temperatures. The second factor that could lead to the observed low digestibility could be attributed to use of an incompatible substrate. The substrate used here originated from Brewery waste which and it may not be suitable for sewage sludge and cow dung on their own. The third factor could be due to heavy metal contamination that may originate from illegal disposal of industrial wastewater into the domestic sewer network. Further tests will be carried out to establish heavy metal content in the sewage sludge. The final factor is the C/N ratio of the substrates. This study did not determine that but optimal methane production said to occur at C/N ratio between 20 and 30 (Kayhanian & Tchobanoglous, 1992). Furture studies should consider this during substrate mixing.

The study further showed that co-digesting STP sludge with brewery waste under mesophillic conditions enhanced both biogas production and biogas yield. In general, both the biogas production rate and yields were observed to increase with an increasing ratio of brewery sludge/STP sludge. However, when the ratio was increased to 100% brewery sludge, the digester failed due to organic overloading, as discussed earlier (data not shown). The biogas yield for S_4 (50% STP sludge and 50% brewery sludge) was higher ranging between 392 to 405 mL/g COD compared to that of S_3 (75% STP sludge and 25% brewery sludge) and S_5 (50% STP sludge, 25% brewery sludge and 25% cow dung). The biogas yield of S_3 was between 230 to 387 mL/g COD while that of S_5 was between 287 and 365 mL/g COD for the two experiments. Our results show similar trends with those reported by Barbel *et al.* (2009) and Pecharaply *et al.* (2007) who observed higher biogas production with an increasing brewery: sewage sludge ratio in the substrate during co-digestion. Likewise, Callaghan *et al.* (1999) observed increased biogas production when brewery waste was co-digested with cattle slurry compared to cattle slurry alone. This is similar to our study, in the substrate with 25% cow dung,50% STP sludge and 25% brewery waste, the biogas yield than when STP sludge was digested with cow dung alone (Table 3-2). In general, organic components in brewery waste are easily biodegradable since they largely consist of sugars, soluble starch, ethanol and volatile fatty acids, which explains the observed increased biogas production when brewery was added as a co-substrate.

Co-digestion of STP with cow dung alone on the other hand did not improve biogas production. The biogas yield for S_1 and S_2 were between 160 to 171 mL/g COD and from 129 to 173 mL/g COD respectively. Methane yields showed similar trends, a methane yield of 0.19 to 0.2 m^3/kg VS fed was observed in S_1 and it was 0.15 to 0.2 m^3/kg VS for S_2. This is within the range of 0.11 - 0.24 m^3/kg VS fed, as observed by Hansen *et al.* (1998) and Sommer *et al.* (2002) when cow dung was digested. Cow dung is more difficult to digest as compared to other animal dung e.g. swine dung. Its low digestibility can be attributed to the presence of recalcitrant compounds, such as cellulose and hemicelluloses complexes with lignin (Zeeman, 1991). Since cow dung originates from the rumen where it is already partially digested (Zeeman, 1991), it is likely to lead to lower biogas yields, compared to other wastes that are directly generated without prior digestion. Li et al. (2011) have however reported values up to 0.328 m^3/kg VS fed of methane when dry cow dung was co-digested with wastewater in batch experiments. This may be due to the dung characteristics which may vary depending on the animal species or difference in the animal feed as well as due to difference in manure management practices (Hobson & wheatley, 1993). This variability consequently leads to variation of methane production during AD.

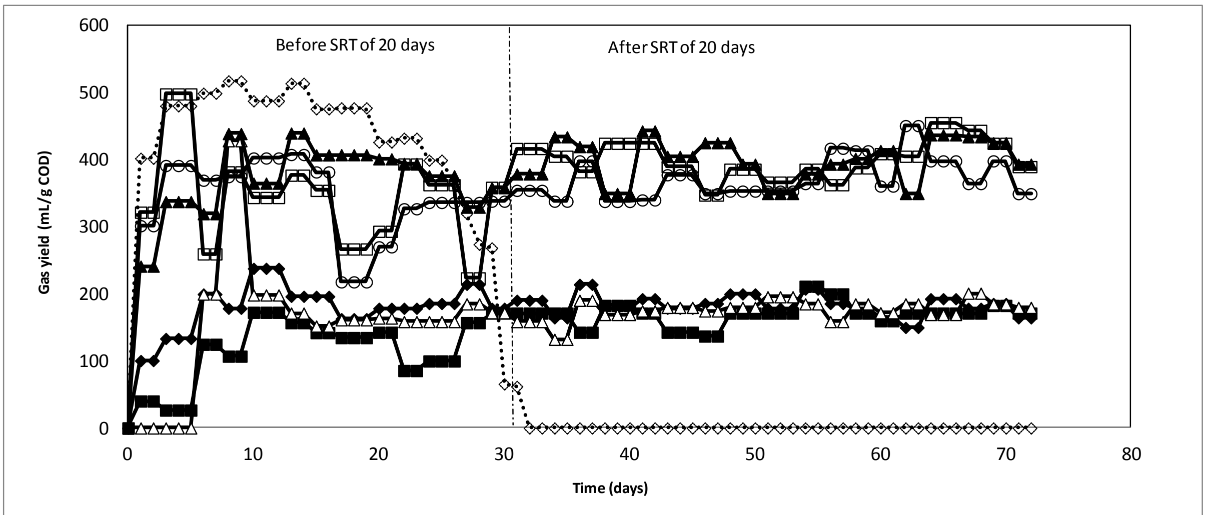


Figure 3-1: Biogas yield during the entire digestion period. (♦) 100% STP sludge, (■) 75% STP sludge and 25% Cow dung, (△) 50% STP sludge and 50% Cow dung, (□) 75% STP and 25% Brewery sludge, (▲) 50% STP sludge: 50% Brewery sludge, (○) 50

% STP sludge: 25% Cow dung: 25% Brewery sludge and (◊)100% brewery waste. (Results are for one experiment)

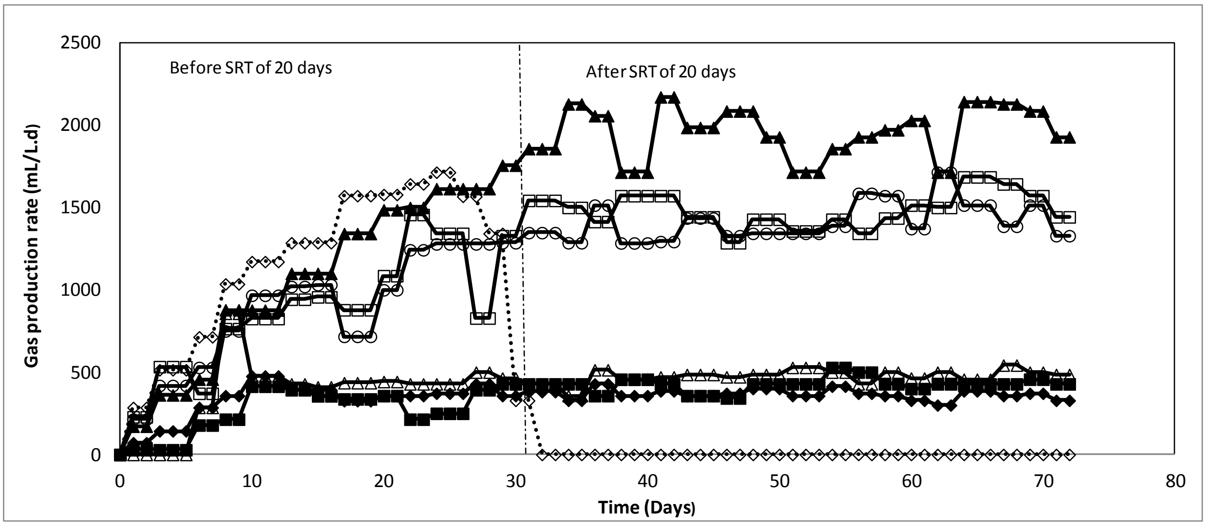


Figure 3-2: Biogas production rate during the entire digestion period. (♦) 100% STP sludge, (■) 75% STP sludge and 25% Cow dung, (∆) 50% STP sludge and 50% Cow dung, (□) 75% STP and 25% Brewery sludge, (▲) 50% STP sludge: 50% Brewery sludge, (○) 50 % STP sludge: 25% Cow dung: 25% Brewery sludge and (◊) 100% brewery waste. (Results are for one experiment)

3.4 Synergy in biodegradability

In order to determine whether synergy exists in the biodegradability of the substrates, the methane yield per g COD of each substrate was calculated from the total methane production of the mixture (Table 3-3). For example 1g of S_1 COD consists of 0.7 g STP COD and 0.3 g cow dung COD and the methane yield of the mixture was 78 mL/g COD. Since STP alone (S_0) yielded 69 mL/g COD, then from 1g of S_1, STP contributed (0.7x78) = 48 mL CH_4 while the remaining 30 mL CH_4 was contributed by cow dung. The methane yield of the cow dung in the S_1 mixture is therefore (30/0.3) = 99 mL/g COD while that of STP is 69 mL/g COD (assumed to be similar to that observed from S_0).

The maximum methane yield that can be observed from 1 g of COD is theoretically known to be 350 mL/g COD. The methane production per g COD of the individual substrates in the mixtures (Table 3-3) did not exceed 350 mL/g COD, which indicates that there was no synergy in digestibility of the substrates resulting from the co-digestion.

Table 3-3: Methane yield (mL/g COD) of the individual substrates in the different mixtures.

	STP:CD:BW ratio by volume	STP:CD:BW ratio by COD	Total CH_4 from mixture (mL/g COD)	CH_4 volume from STP in mixture (mL)	CH_4 volume from other substrate in mixture (mL)	CH_4 yield of the other substrate (s) in the mixture (mL/g COD)
S_0	100:00:00	100:00:00	69	69	0	0
S_1	75:25:00	70:30:00	78	48	30	99
S_2	50:50:00	40:60:00	73	27	46	76
S_3	75:00:25	49:00:51	202	34	168	330
S_4	50:00:50	24:00:76	231	17	214	282
S_5	31:25:25	31:20:49	175	21	154	223

- The methane yield of STP in any mixture was 69 mL/g COD.

3.5 Biogas Quality

The average methane content in the biogas in the reactors treating substrates with brewery waste was higher, i.e. 64.1 ± 3.9%, 58.3 ± 4.1% and 52.6 ± 4.6% for S_3, S_4 and S_5, respectively. The biogas produced in S_0 (100% STP sludge) showed the lowest quality with only 40.9 ± 2.5% of CH_4, followed by S_1 and S_2 were STP sludge was mixed with cow dung. The biogas from S_1 and S_2 had a methane content of 44.7 ± 3.8% and 47.5 ± 5.6%, respectively. The carbon dioxide content in the samples was in the range of 30 - 48 %. Traces of carbon monoxide and H_2S were also measured. Hydrogen sulphide is produced during hydrolysis when certain organisms break down the essential amino acid methionine (Zhu *et al.*, 1999).

The methane content observed in this study is in general quite low compared to other studies (Babel *et al.*, 2009; Davidson *et al.*, 2008; Li *et al.*, 2011). Methane percentages above 70% were reported when sewage sludge was co-digested with brewery sludge at ratios similar to our study at a SRT of 20 days during biochemical methane potential (BMP) tests (Babel *et al.*, 2009). The same study however reported methane percentages below 30% for sewage sludge alone at a SRT of 20 days, which was attributed to existence of heavy metals in the sewage sludge. Davidson *et al.* (2008), Li *et al.* (2011) and Martinez *et al.* (2012) observed methane content of 60% and more at a SRT of 21 days for sewage sludge. Li *et al.* (2011)

also reported a methane content of at least 50% for cow dung co-digested with sewage sludge.

In CSTR systems, SRTs of 20 days or more are recommended in order to to avoid washout of the methanogens, which are responsible for methane production (Appels *et al.,* 2008). While the aforementioned studies achieved higher methane contents at a SRT of 20 days, it is still possible that the same SRT of 20 days in our study was not sufficient to avoid washout of some methanogens. The difference in methane contents that were observed, compared to other studies, could also be due to the origin of the substrates and their characteristics. The presence of inhibitory elements, like heavy metals in one of the substrates cannot be ruled out, but this was not evaluated in this study.

3.6 TAN concentration in the digesters

The concentration of TAN during the experiment increased slightly in all digesters over the experimental period of 72 days. The concentrations of TAN in the control digester with S_0 increased from an initial value of 230 to 253 mg/L, for S_1 from 205 to 238 mg/L, for S_2 from 215 to 248 mg/L, for S_3 from 253 to 305 mg/L, for S_4 from 300 to 365 mg/L and for S_5 from 260 to 320 mg/L. Ammonium (NH_4^+) and free ammonia (NH_3), are produced during anaerobic digestion, mainly from proteins and amino acids. Free ammonia is the most toxic even at low levels (Appels *et al.*, 2008) but methanogenesis can be severely inhibited at concentrations exceeding 3000–4000 mg TAN/L (Chen *et al.*, 2008; Schnurer and Nordberg, 2008). The concentrations of TAN in all digesters increased during the experimental period, but none of the reactors reached inhibiting values. Therefore the TAN concentrations are not likely to have contributed to methane yield inhibition in any of the digesters.

3.7 Optimization strategies towards highest energy production

The primary sludge production rate at STP, Kampala (Uganda) is estimated at 40 m^3/day while the brewery plant, has an average daily production of 10 m^3/day. Table 3-4 presents the calculated energy potential of different options of using the substrates to which brewery waste was added, compared to the control with 100% STP sludge. Option C would give the highest energy output with a factor 11 more energy relative to the control. However, this would require 40 m^3 of each waste, which is not available from the brewery plant at the moment. This is followed by Option D and B with energy output of a factor 7 and 4 more energy compared to the control, respectively. It is important to note however that the tank volume required by option D is 1.5 times the tank volume of Option B which increases its capital cost. Operational costs may also slightly be higher in option D, considering that three different waste streams need to be handled. The increased costs may however easily be covered in a short time given the fact that the energy production in option D is almost double that of option B. Moreover, option D is a better scenario at solving problems of abattoir wastes which are increasingly polluting the fresh water sources nearby. Option D is therefore proposed as the optimal co-digestion option in this study.

Table 3-4: Electricity and heat energy potential of options where brewery sludge was added compared to one where 100% STP sludge was added.

Option	STP:BW:CW ratio	Digester Volume(m^3/day)	Biogas production rate (m^3 /day)	Electricity (KWh)	Heat energy (KWh)
A	100:0:0	800	280	560	714
B	75:25:0	1060	1272	2544	3,239
C	50:50:0	1600	3200	6400	8,160
D	50:25:25	1600	2080	4160	5,304

- The tank volume is calculated based on complete digestion of STP sludge produced at the plant at a SRT of 20 days.
- The energy is calculated based on a rule of thumb of 0.5 m^3 biogas ≈ 0.85 KWh electricity + 1.5 KWh heat energy, in a combined heat and power module.
- The options considered are B=75 % STP: 25% Brewery waste, C=50 % STP : 50% Brewery waste and D=50 % STP : 25% Brewery waste :25% cow dung mix. These are compared to the control with STP only (A=100% STP).

3.8 How do the different stakeholders benefit?

National Water and Sewerage Corporation (NWSC) is in charge of the Bugolobi sewage treatment plant and is already planning to build an anaerobic digester for the STP sludge. They would benefit from the increased energy generation. The annual electricity production estimated from option A is 173,740 kWh per year, which barely sustains the current plant electricity requirement, estimated at 230,000 kWh per year. Adapting option D will increase the electricity by a factor 7. For the new plant, whose sludge volume is estimated to be ten times the current one, option D would fully cater for its higher mechanised energy requirements. In addition, it will provide surplus electricity, which can be sold off to the National grid, hence generating extra income for NWCS with time.

For East African Breweries Limited (EABL) Uganda, the option of co-digesting STP sludge with Brewery waste provides a short term optimal solution for safe brewery sludge disposal. This would otherwise remain a concern, since it is currently quite costly for EABL to treat and get rid of this waste. The brewery plant will easily be relieved of this cost if their waste is directly fed into the anaerobic digestion process proposed. On the other hand on the long term, if EABL decided to adopt anaerobic digestion for brewery waste alone, it will be more costly as the reactor has to be designed to be operated at a higher SRT, of more than 28 days for a stable process. Adopting co-digestion of brewery waste with STP sludge provides good buffering for the process. This ensures the stability of the reactor at a lower SRT, hence providing a beneficial option.

Moreover, the proposed optimal substrates with STPS:BW:CM ratios of 50:25:25 represents a scenario which will contribute to decreased eutrophication in Lake Victoria, since it caters for the safe disposal of cow dung as well. One of Kampala's biggest abattoirs owned by Uganda meat packers is a few kilometres away from Bugolobi STP. This abattoir lacks any waste treatment and disposal facilities. The abattoir waste, a big part being is cow dung is damped on an open nearby site where it decomposes into manure, which is sometimes collected by farmers. This persistently contributes to greenhouse gas emissions and odour nuisance to the surrounding environment of which the NWSC training centre, central laboratory and the BSTP is part. Furthermore the runoff through the decomposing waste pile is discharged into the nearby Nakivubo channel that ultimately drains into Lake Victoria. This carries with it high level of phosphorous and observed in the cow dung. Utilizing the cow dung during co-digestion will therefore make a great contribution towards minimizing the nutrient load and consequently the eutrophication in the region's largest fresh water lake.

In addition to the biogas, the digestate is another rich by-product of the co-digestion process. The plant nutrient such as nitrogen, phosphorous, potassium and magnesium, as well as the trace elements essential to plant growth, are preserved in the substrate. (Kossmann et al., 1999). Possible options for utilizing these nutrients for plants include drying the sludge over drying beds and then applying it as manure when dry. This is the current practice at NWSC for the primary sludge produced. The dried manure at NWSC is very marketable and is sold to farmers at about USD 3 per tonne, a rate which could increased with a more sanitized product from the digesters. Another option could be production of biochar for fertilizer application and as a means to manage the digestate waste. This option is discussed further in chapter 5 and 6.

4. Conclusions

The results in this study have shown that the biodegradability of Bugolobi STP sludge is limited with a biogas yield of 159 to 186 mL/g COD. Co-digesting STP sludge with Brewery sludge increased the biogas production rates by a factor ≥ 3, while cow dung alone did not improve biogas production. Substrate S_4 (50% STP sludge and 50% brewery sludge) showed the highest biogas yield and production rate but S_5 (50% STP sludge, 25% brewery sludge and 25% cow dung) was selected as the optimal mixture for practical application.

Bugolobi STP sludge was co-digested with cow dung and brewery sludge in different ratios, (S_0 100% STP sludge, S_1: 75% STP sludge and 25 % cow dung, S_2: 50% STP sludge and 50 % cow dung, S_3: 75% STP sludge and 25 % brewery sludge, S_2: 50% STP sludge and 50 % brewery sludge and S_5: 50% STP sludge & 25% STP cow dung & 25 % brewery sludge). Substrate S_4 with 50% STP sludge: 50% breweries waste showed the best biodegradability with an average of 479 % increase in biogas production rate compared to the control. Due to limitation in the brewery waste supply, the STP sludge to brewery sludge ratio of S_5 was considered to be optimal for industrial application as it contributed to decreased nutrient loads for water resources. For the rural application where farmers may not have brewery waste available, other wastes like food wastes and local brew wasted could be investigated to boost biogas production.

The study has further shown the benefits that would arise if the current plant is modified to build an anaerobic digester and allow co-digestion of STP sludge with brewery waste. This presents benefits to both the brewery plant as well as NWSC. The brewery plant would be relieved of cost for discharge if their waste is directly fed into the anaerobic digestion process proposed. On the other hand, NWSC would benefit from the increased power generation that would result from co-digestion, other than using STP sludge alone.

Acknowledgements

The authors wish to acknowledge the financial support from VLIR, the Belgian scholarship body and National Water and Sewerage Corporation for further support in Uganda. We also wish to acknowledge Jo Devriez for proof reading the manuscript and Henry Mugabi (EABL), Kanyesige Christopher, Nabatesa Sylvia and Chaba Charles for the field supported. Willy Verstraete and Korneel Rabaey acknowledge support from the Ghent University

Multidisciplinary Research Partnership (MRP) "Biotechnology for a Sustainable Economy" (01 MRA 510W).

Chapter 4 : A TWO-STAGE DECENTRALISED SYSTEM COMBINING HIGH RATE ACTIVATED SLUDGE (HRAS) WITH ALTERNATING CHARCOAL FILTERS (ACF) FOR TREATING SMALL COMMUNITY SEWAGE TO REUSABLE STANDARDS FOR AGRICULTURE

This chapter has been redrafted after:

Nansubuga, I., Meerburg, F., Banadda, N., Rabaey, K., & Verstraete, W. (2015). A two-stage decentralised system combining high rate activated sludge (HRAS) with alternating charcoal filters (ACF) for treating small community sewage to reusable standards for agriculture. *African Journal of Biotechnology*, *14*(7), 593-603.

Abstract

Water scarcity increasingly drives wastewater recovery. Campaigns towards re-use of wastewater are not very common in Africa among other factors, due to a lack of efficient and cost-effective technology to treat wastewater to re-usable standards. In this study, two treatment systems, a high rate activated sludge (HRAS) system and alternating charcoal filters (ACF) are combined and used to treat wastewater to standards fit for reuse in agriculture. The charcoal can upon saturation be dried and used as fuel. Two different ACF lines were used in parallel after the HRAS: ACF1 with a residence time of 2.5 h and ACF2 with residence time of 5 h. Results showed no significant difference (α = 0.05) in the performance of the two filter lines, hence ACF1 with a higher flow rate was considered as optimal. The HRAS effectively removed up to 65% of total suspended solids (TSS) and 59% of chemical oxygen demand (COD), while ACF1 removed up to 70% TSS and 58% COD. The combined treatment system of HRAS and ACF1 effectively decreased TSS and COD on average by 89 and 83%, respectively. Total ammonium nitrogen (TAN) and total phosphates (TP) were substantially retained in the effluent with average removal percentages of 19.5 and 27.5%, respectively, encouraging reuse for plant growth.

1. Introduction

Humans depend on water for nearly all aspects of life. The diverse utilization of water coupled with population explosion across many places in the world has made it a scarce resource. Moreover, the discharge of untreated or inadequately treated wastewater leads to deterioration in the quality of fresh water sources and continues to deepen the water scarcity. Re-use of wastewater for some purposes such as agriculture is an indispensable part of integrated water management and would decrease water scarcity. This requires a change in perceptions as well as availability of simple, low cost and effective technologies. The treated wastewater should be sufficiently disinfected but not void of its nutrient content, so as to increase crop yields. In Uganda, reuse of wastewater is not widely reported; however, informal irrigation occurs in several parts of the country. For instance farmers in the Murchison Bay, which receives Kampala city's highest flow of wastewater effluent, are seen to cultivate a variety of crops. The main concern for reuse of wastewater is the health of both the farmers and the crop consumers. Unfortunately, some of the treatment methods used in developing countries may not attain sufficient disinfection, which limits reuse options (Nikiema *et al*., 2013) and may pose public health risks if improperly applied. Centralised systems common in the developing world are effective but very expensive and are not suitable for low density rural areas (Netter *et al*., 1993). These systems can cost up to € 40 per capita per year considering both capital and operational expenditure (Zessner *et al*., 2010). On the other hand, on-site systems are cheaper but have a number of limitations with regard to wastewater re-use. Also, some like pit latrines are known to increasingly pollute ground water sources (Katukiza *et al*., 2013, Nyenje *et al*., 2013). Therefore, efficacious and cost effective technology to boost wastewater reuse and recycling needs development for the developing world.

Verstraete and Vlaeminck (2011) proposed a new approach for optimal resource recovery, as opposed to the conventional wastewater management. In this approach which they label as the M & M treatment system, the wastewater is separated as near as possible to the source into two distinct streams: the major line (up to 90% of the flow) and the minor line (about 10% of the flow). The major water stream is treated to reusable standards while the minor concentrated stream can undergo additional treatment to recover energy and nutrients. Small-scale decentralised systems designed for a small number of households could provide a cost-effective method for that purpose. Such systems should focus on optimising the pre-concentration methods and further treatment of the two separate streams, to maximize resource recovery. Methods of solids pre-concentration may include the biological adsorption in a high-rate activated sludge stage (HRAS), also referred to as the A-stage of the A/B Verfahren system (Böhnke, 1977). This activated sludge process operates at high sludge loading rates (2 to 10 g bCOD $gVSS^{-1}$ d^{-1}) and low sludge retention times (hours to days), while a short hydraulic retention time of under 30 min selects for rapid incorporation of organic matter into sludge without extensive oxidation (Bohnke, 1977, Faust et al., 2014). Moreover, the 'young' A-stage sludge is easily digestible by anaerobic digestion (De Vrieze *et al*., 2013) to recover energy. The effluent from the A-stage can be further treated to achieve reusable standards by methods such as trickling filters or sand filters. For the developing world, it is important to explore locally available materials and simple technologies in order to achieve cost effective and sustainable systems. Charcoal is such a material and it is ubiquitously available in Uganda. The use of charcoal for wastewater treatment has been widely studied (Abe et al., 1993; Samkutty and Gough, 2002; Scholz and Xu, 2002; Ochieng *et al*., 2004; Sirianuntapiboon *et al*., 2007; Nkwonta *et al*., 2010; Ahamad and Jawed, 2011). Its performance compared well with other media like gravel, sand rocks and zeolite, however, attaining its continued use is still a challenge.

For this reason, this study proposes and investigates a low cost small scale wastewater treatment plant which also allows for wastewater reuse. It combines two wastewater treatment systems (Figure 4-1). The first stage is a HRAS system similar to the A-stage, to achieve pre-concentration and major organics removal, and the second stage is filtration of the liquid fraction with use of alternating charcoal filters. The wastewater is treated to meet reusable standards for agriculture. The sludge from the process could be used for biogas recovery in a subsequent study. Upon saturation the charcoal is replaced which allows for continuity of the system, the charcoal could then be dried and finally used as fuel, which originally was its primary use. This system is suitable for small communities.

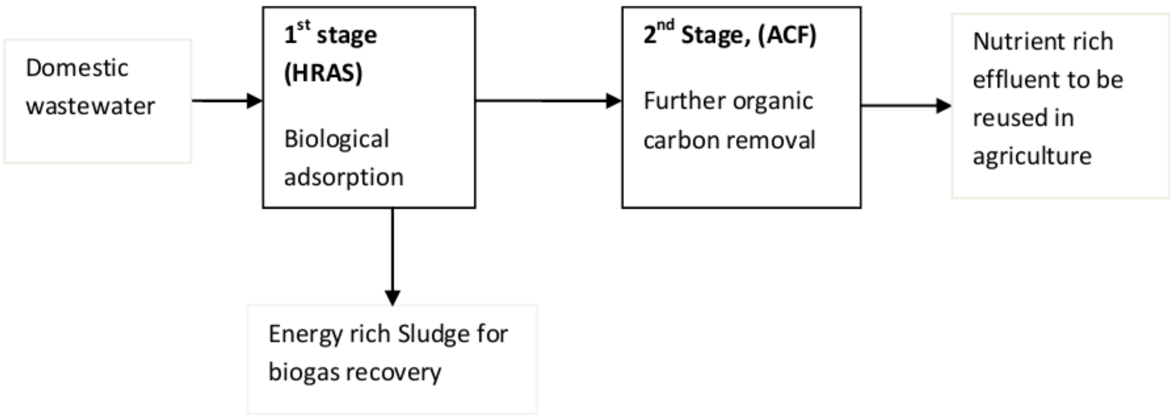


Figure 4-1: Representation of the combined processes treatment with use of high rate activated sludge (HRAS) system and the alternating charcoal filter (ACF)

2. Materials and methods

2.1 Sample collection

Raw domestic wastewater was collected from Bugolobi Sewage treatment plant (STP) in Kampala (Uganda) every two to three days for 4 months (June 2013 to October 2013). The Bugolobi STP managed by National Water and Sewerage Corporation (NWSC), is the largest sewage treatment plant in Uganda. It employs physical and biological treatment by use of screens, detritus basin, primary settling tanks, trickling filters and secondary clarifiers in that order. The plant has an average inflow of 12,000 m^3 per day mainly via the centralised sewerage pipe network. However, about 300 m^3 of the inflow is received via cesspool trucks that deliver septage from septic tanks and pit latrines around Kampala City and its outskirts. The cesspool dumping usually accounts for a sudden change in the influent wastewater quality. In this study, the wastewater was collected after the screens and grit chamber and stored at room temperature (about 24°C) in a 200 L container which continuously fed the HRAS experiment. Selected parameters of the raw wastewater characteristics and outflow of the HRAS stage were determined and are shown in Table 4-1. The maximum values of total suspended solids (TSS), total phosphates (TP), total ammonium nitrogen (TAN) and chemical oxygen demand (COD) of the Bugolobi STP wastewater sampled at different times were 794, 66, 61 and 116 mg L^{-1}, respectively.

Table 4-1: Average raw water characteristics, average operating parameters of the high rate activated sludge (HRAS). Also shows the effluent characteristics from the alternating charcoal filter 1(ACF1) with a retention time of 2.5 h and the alternating charcoal filters 2 (ACF2) with a retention time of 5 h and the removal efficiency of the HRAS combined with each of the alternating filter options.

Parameter	Raw wastewater	HRAS reactor	HRAS effluent	ACF1 effluent	ACF2 effluent	HRAS+ACF1 Average total removal (%)	HRAS+ACF2 Average total removal (%)
TSS (mg/L)	322±163	2174±932	102±49	32±22	26±19	89± 7	91±6
COD total (mg/L)	613±244		233±106	93±45	91±47	83±8	84±8
COD soluble (mg/L)	128±57		111±61	73±30	68±30	46±24	48±24
TAN (mg/L)	36±11		33±10	30±9	29±9	19±16	20±10
Ptotal (mg/L)	26±13		22±10	19±9	19±8	27±15	28±14
pH	7.2±0.2	7.4±0.2	7.5±0.2	7.6±0.1	7.6±0.1		
Temperature		21.9±0.7					
DO (mg/L)		3.7±1.6					

faecal coliform (FC) colony forming units (CFU) in the influent ranged from $3.13x10^2$ to $2.01x10^6$ CFU mL^{-1}. The wastewater characteristics are known to vary depending on the weather conditions. The variation can also be attributed to the small daily volumes (300 m^3 day^{-1}) of high strength septage received by the plant throughout the day. The reactor sludge was obtained by autonomous growth during an acclimation period of 10 days of reactor operation. The charcoal used in the study was bought from the open market, crushed into pieces ranging from 0.5 to 1.5 cm. It was then washed to remove the dust before packing it in plastic columns in the Laboratory. The porosity and dry bulk density of the packed charcoal after crushing were determined.

2.2 Experimental set-up

2.2.1. High-rate activated sludge (HRAS) experiment

A HRAS experiment was set up at laboratory scale as shown in Figure 4-2. It consisted of a continuous stirred tank reactor (CSTR) unit which was continuously aerated, a settling unit and a sludge return device. The CSTR unit had a volume of 4 L and an average hydraulic retention time (HRT) which was started at 0.5 h but was increased and maintained at 1 ± 0.3 h after 10 days. The average sludge retention time (SRT) of the CSTR was 1.5 ±0.3 days and it was loaded at an average sludge loading rate of 2.2 g bCOD/g SS per day. Two electrical aerators (Aquatic AP1, Interpet, United Kingdom) were used to supply oxygen into the CSTR which achieved an average concentration of dissolved oxygen (DO) of 3.7 ± 1.6 mg/L. A mechanical stirrer (RW16 basic, IKA Labortechnik, Germany, 60 - 2.000 rpm) was used to stir the CSTR unit. The settling unit had an effective volume of 8 L and an initial HRT of 1 h, which was increased and maintained at 2 ± 0.4 h after 10 days. The sludge from the settling unit was returned to the CSTR using a pump (Leroy Somer Varmeca, Belgium). The Recycle ratio (Qreturn/Qinfluent) of the CSTR was 1 and 2 L of sludge was removed manually every day. The wasted sludge was kept in a 5 L container at 4°C where it settled further before the clear water was poured off and the settled sludge was used in another study. Selected parameters of the influent and effluent of the HRAS experiment were measured on the samples collected three times a week.

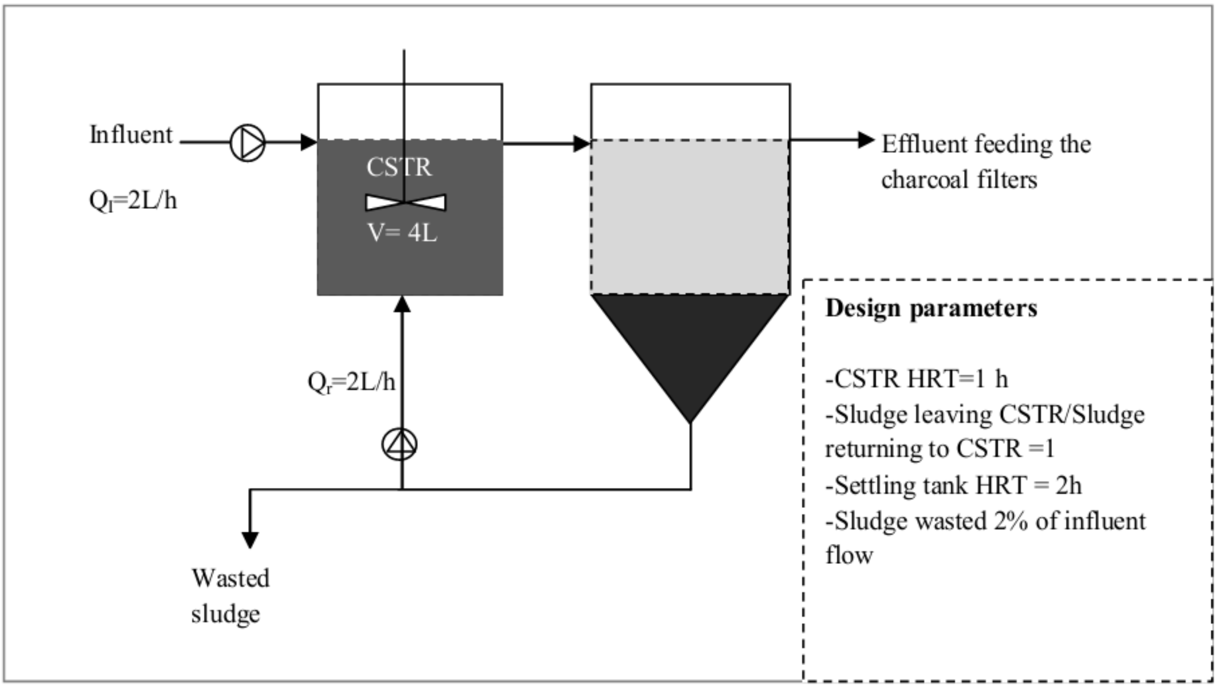


Figure 4-2: Schematic representation of the high-rate activated sludge (HRAS) set-up consisting of a completely mixed reactor (CSTR) in series with a settler.

2.2.2. The Alternating Charcoal Filter (ACF)

The effluent from the HRAS was fed into the ACF for further treatment as shown in Figure 4-3. It was fed into two separate ACF lines, each with three charcoal filter columns placed in series. The filter columns were 25 ± 3 cm long and had a volume of 1 L of charcoal. The charcoal particles in the filters ranged between 0.5 to 1.5 cm. The packed filters had porosity of 48% and dry bulk density of 0.3 g cm^{-3}. The residence time in the filter lines differed with filter line 1 (ACF1) having a residence time of 2.5 h, while filter line 2 (ACF2) had a residence time of 5 h. After every 30 days, the top filter column 1 (F_1) was emptied and refilled with fresh charcoal and moved to the last position in the series while filter column 2 (F_2) and filter column 3 (F_3) went a position up in the series to become F_1 and F_2, respectively. This means that all filters were replaced every 90 days and this continued for the rest of the experimental period. Wastewater samples were taken from the effluent of the last filter columns three times a week; and chemical oxygen demand (COD), TSS, total ammonium nitrogen (TAN), Total phosphorus (TP), and CFU were measured.

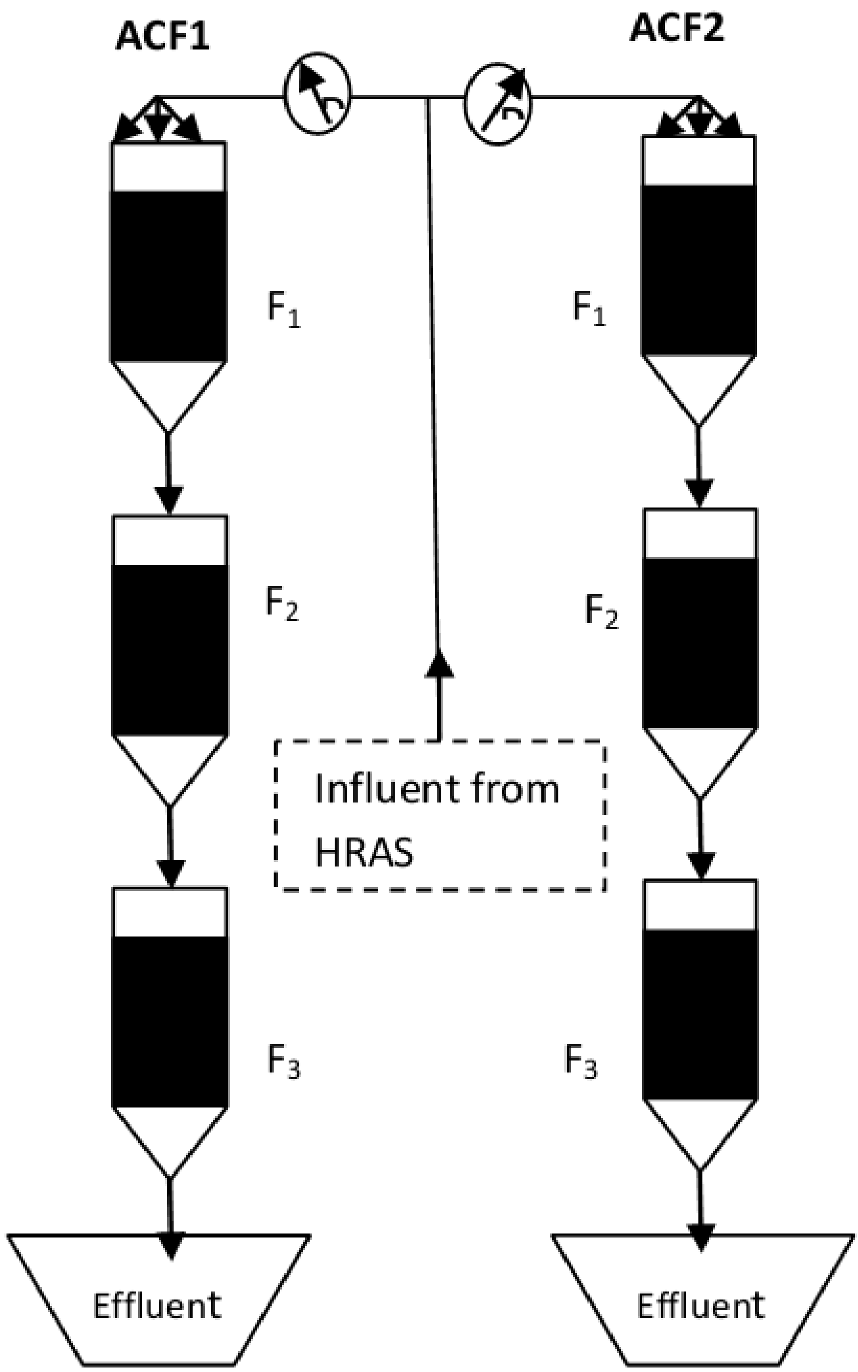


Figure 4-3: Schematic representation of the setup of the alternating charcoal filter 1 (ACF1) with a retention time of 2.5 h and the alternating charcoal filters 2 (ACF2) with a retention time of 5 h.

2.3 Analytical methods

The influent and effluent samples of the HRAS and the ACF were measured for organic matter, total nitrogen and phosphorous. Total phosphorus (TP), chemical oxygen demand (COD) and total ammonium nitrogen (TAN) were analyzed using HACH DR 5000 Spectrometer as described in the standard methods (APHA, 2005). The pH was measured with a pH meter (Teledo, USA) while volatile Solids (VS) and total solids (TS) were analysed according to standard methods (APHA, 2005). Faecal coliform Colony forming units (CFU) were determined using the Colilert-18 protocol (Idexx Laboratories, 2012) and dissolved oxygen (DO) was determined with use of a DO meter (HACH, UK). The Kruskal-Wallis non-parametric test was used to verify if there was a significant difference between the measured influent and effluent parameters of the HRAS and the ACF.

3. Results

3.1 Performance of the HRAS reactor

In the HRAS reactor, the wastewater had an average pH of 7.4 ± 0.2, dissolved oxygen of 3.7 ± 1.6 mg L^{-1} and temperature of 21.9 ± 0.7°C (Table 1). Figure 4-4 shows the performance of the HRAS over the entire 140 days of the experimental run. To evaluate the performance of the HRAS, consideration is only given to the period after day 10 when the HRT in the CSTR and the sedimentation tank were maintained at 1 ± 0.3 and 2 ± 0.4 h, respectively. Regardless of the variation observed in the influent TSS concentration (131 to 794 mg L^{-1}), the effluent concentrations were less variable ranging between 30 to 250 mg L^{-1}. This corresponded to an average TSS removal of 65%. The average influent COD was 613 ± 244 mg L^{-1} of which about 21% was soluble while the average effluent concentration was 233 ± 104 mg L^{-1} of which about 48% was soluble COD. This led to an average removal efficiency of 59% for total COD and 15% for soluble COD. The HRAS slightly decreased TAN and TP with an average removal efficiency of 11 and 17%, respectively.

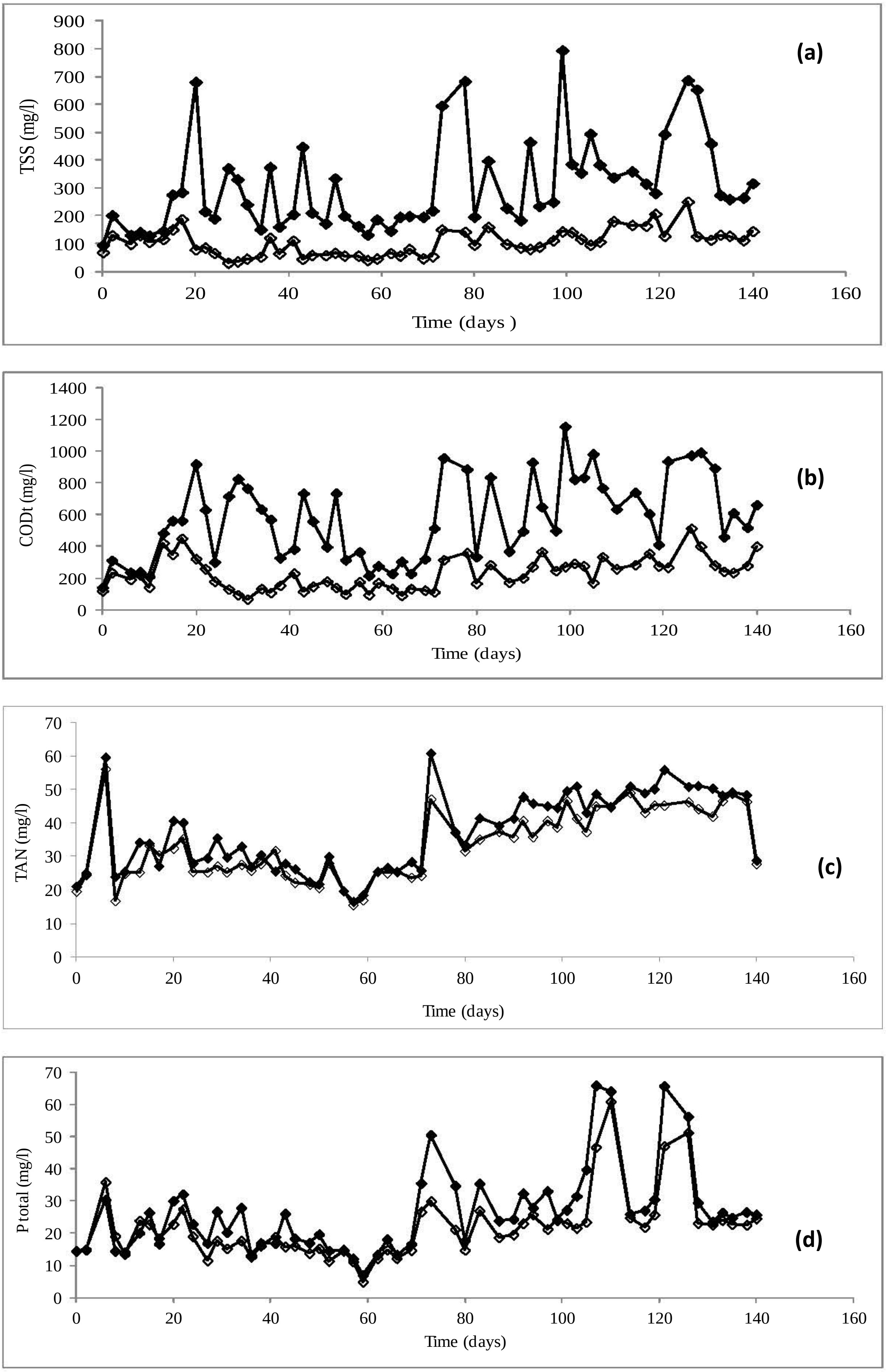


Figure 4-4:Influent (♦) and effluent (◊) concentrations of (a) the total suspended solids (TSS), (b) the total chemical oxygen demand (COD_t), (c) the total Ammonium nitrogen (TAN) and (d) the total phosphorous (P_{total}), in the High rate activated sludge system during the entire study period.

3.2 Performance of the ACF reactor

The effluent of the HRAS was fed to two ACF reactors for further treatment. ACF1 had a residence time of 2.5 h while ACF2 had a residence time of 5 h. Figure 4-5 shows the performance of the two filter lines over the entire 140 days of the experiment. For consistency, the period after day 10 was considered for evaluation of the performance of the filters. The average TSS concentration of the effluent from ACF1 and ACF2 were 32 ± 22 and 26 ± 19 mg L^{-1}, respectively. This corresponds to an average removal efficiency of 70% for ACF1 and 76% for ACF2. The concentration of total COD of the effluent from ACF1 was on average 93 ± 45 mg L^{-1} of which 78% was soluble COD, for ACF2 the average total COD was 91 ± 47 mg L^{-1} of which 74% was soluble. This corresponds to a total COD removal efficiency of 58 and 60%, observed for ACF1 and ACF2, respectively, while for soluble COD, it was 27 and 30%, respectively. Like in the HRAS reactor, the removal of TAN and TP was low in both filter lines. The average removal of TAN was 11 and 13% in ACF1 and ACF2, respectively, while the average TP removal was 12% in ACF1 and 13% in ACF2. Statistical analysis showed that there was no significant difference (α =0.05) in the performance between ACF1 and ACF2 in removal of all the above considered parameters.

3.3 Overall performance of the combined treatment system.

In general, the combination of the HRAS and ACF registered high COD and TSS removal efficiencies (Table 4-1). The overall average TSS removal was 89% ± 7 and 91% ± 6 when the HRAS was combined with ACF1 and ACF2, respectively. The same combinations attained average total COD removals of 83% ± 8 and 84% ± 8 and average soluble COD removal of 46% ± 24 and 48% ± 24, respectively. The overall removal of TP and TAN was generally lower compared to TSS and COD: the combination of HRAS with ACF1 obtained an average TAN removal of 19% ± 16 while with ACF2 it was 20% ± 10. TP removal was 27% ± 15 and 28% ± 14 for the HRAS combination with ACF1 and ACF2, respectively. There was no significant difference (α=0.05) in the performance of the two filters. CFU counts were monitored from day 34 up to the end of the experiment. The HRAS influent CFU counts varied widely from 3.13×10^2 to 2.01×10^6 CFU mL^{-1}. During the experimental study period, the HRAS system achieved on average 1 log decrease of CFU and a further 2 log decrease was achieved by the ACF treatment system.

4. Discussion

4.1 High rate activated sludge (HRAS) system

Bohnke et al. (1997) proposed that the HRT of HRAS should be 30 min or less. However, at that HRT which was used in the first 10 days of the experiment, the performance of our HRAS unit was insufficient, with COD and TSS removals going below 40 and 45%, respectively, hence the HRT was increased to 1 h. The HRAS reactor thereafter effectively removed TSS and total COD by an average of 65 and 59%, respectively. The results in this study are similar to those observed in other studies (Bohnke *et al*., 1997; Zamalloa *et al*., 2013, Faust et al., 2014). Apart from biological uptake and degradation, removal in the HRAS systems is partially due to physico-chemical processes which include adsorption and bio-flocculation (Bohnke *et al*., 1997; 1998). The contribution of physic-chemical processes on the overall removal is a result of the short SRT and high sludge loading rate of HRAS processes, which alter the kinetics of substrate removal (Larrea *et al*., 2002, Makinia *et al*., 2006). The adsorption of particulate substrates may act as a buffer against fluctuations in organic loads (Bunch and Griffin, 1987), which ensures that the effluent sent to the second stage had a more stable composition for optimal filter performance (Bohnke *et al*., 1997). TP and TAN were removed to a lower extent in comparison to TSS and COD. TAN and TP removal is generally known to be low in HRAS and other high rate activated sludge processes. To ensure sufficient removal of these compounds, additional treatment is typically incorporated after such systems. Zamalloa *et al*., (2013) applied a flocculant in the HRAS to decrease phosphates while Bohnke *et al*., (1997) ensured TAN and TP removal in a second activated sludge stage at low sludge loading rates. For this study however, since the final effluent from the treatment system is proposed for reuse in agriculture, there would be no need for removal of TP and TAN. The sludge generated in the HRAS is known to be highly degradable (2010; De Vrieze *et al*., 2013).

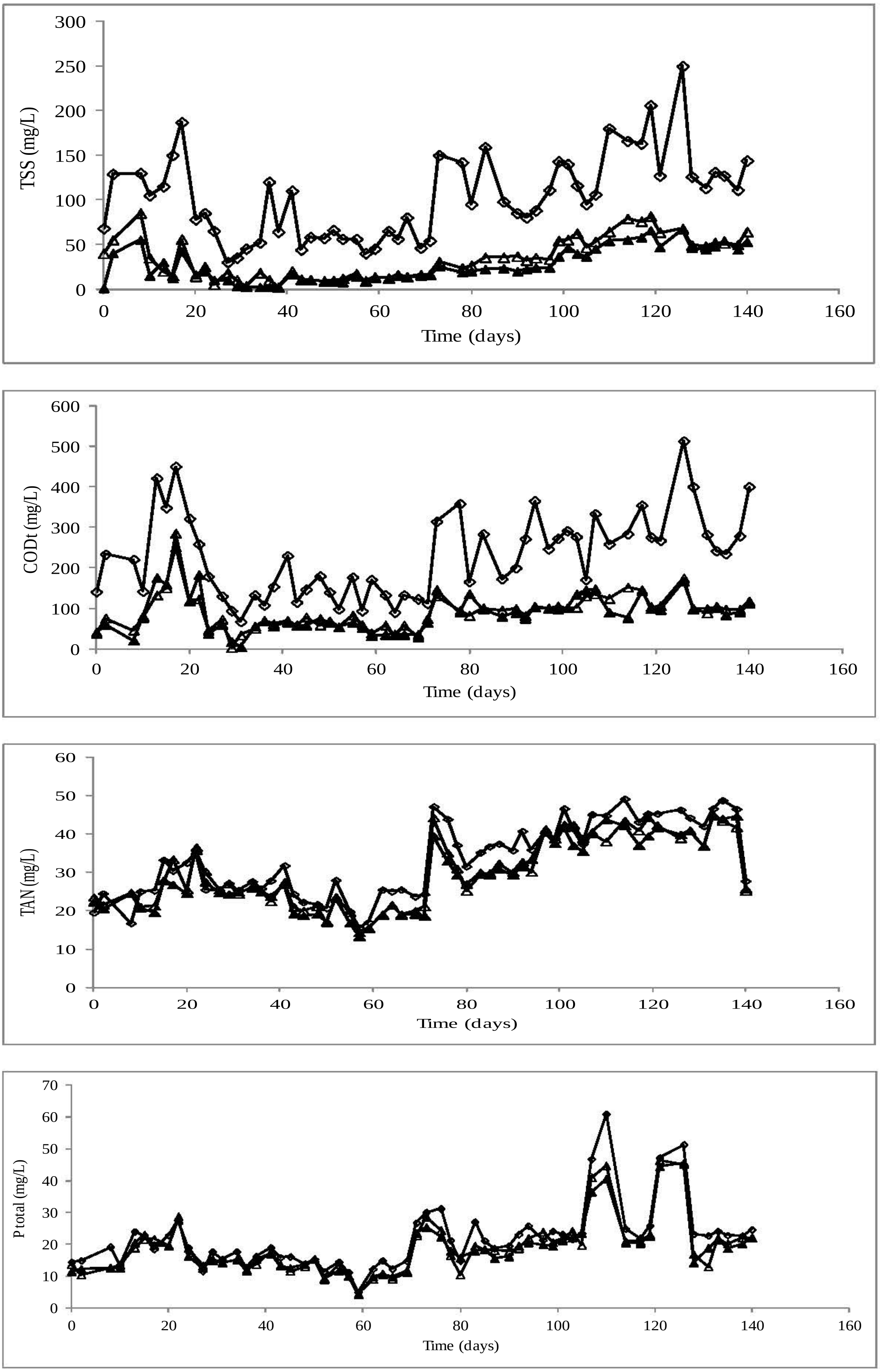


Figure 4-5: Concentrations of (a) the total suspended solids (TSS), (b) the total chemical oxygen demand (COD_t), (c) the total Ammonium nitrogen (TAN) and (d) the total phosphorous (P_{total}) in the Influent (◊), ACF1 Effluent (∆) and ACF2 Effluent (▲) during the entire study period

4.2 Alternating Charcoal Filters (ACF) system

The charcoal filters benefited from the HRAS stage which had an effective treatment and produced a more uniform effluent (TSS and COD did not vary as much as they did in the influent). The two filters had similar performance in which they effectively removed TSS and total COD by an average of 73 and 59%, respectively. Similar to the HRAS, a limited removal was observed for TAN and TP, so the final effluent still contained sufficient nutrients for plant growth. Removal mechanisms of pollutants by the charcoal filter are similar to those in other filters. These include physical filtration, sedimentation, adsorption and biological degradation due to biofilm development. When compared to other filter materials like gravel and rocks however, charcoal has a number of essential properties such as a high number of many micro pores on the surface, high porosity and a high specific surface area of 200 to 300 m^2/g (Darmstadt et al., 2000). The higher specific surface area and porosity in charcoal enhances sedimentation and other filtration processes in charcoal filters (Ochieng and Otieno, 2006) and the micro-pores provide good conditions for micro-organisms to attach. Also, like granulated carbon, charcoal is a good adsorbent and has been widely used as such in wastewater and water treatment (Abe *et al*., 1993; Khalfaoui *et al*., 1995, Kamal and Mohammad, 2012). Due to its adsorbent properties, charcoal can accumulate sufficient organic matter and nutrients for biomass to grow. It is believed that in the first few days before biofilm growth, adsorption is responsible for most of the COD removal. After some time, biofilm grows onto the charcoal and is able to contribute to the organics removal. All these processes contribute to the high efficiency of TSS and COD removal observed throughout the filter's operation. In addition, the small-sized charcoal particles used in this study are cheap, light and easily available at charcoal making stores as waste, and hence offers a cost-effective filter medium for application in the developing world. Actually, the cost for regular replacement of the charcoal are quite reasonable, they are only of the order of 9% of the total cost $capita^{-1}$ $year^{-1}$. Unlike other media however, charcoal is not easy to clean in case of clogging, which would potentially limit its application for prolonged operation times. Therefore, it is proposed in this study that the charcoal filters be used in series and be moved up the chain as the first filter is replaced every month. A charcoal filter is replaced every 30 days which also allows biofilm growth before it is removed. As demonstrated in this study, such an alternating use of charcoal filters ensures consistently high removal efficiency for both TSS and COD. Interestingly, the spent charcoal can be sun dried and subsequently used for fuel. Thus, the charcoal can be used in a coherent sustainable way. Protective wear should be used while handling the charcoal.

4.3 Overall Performance

Overall, the combination of the HRAS with each of the filters showed an effective system for the removal of TSS and COD. It produced an effluent whose average values of TSS and COD met the National effluent standard as required by the National Environment Management Authority (NEMA). NEMA is the regulatory body of effluent discharge in Uganda and its standards require both the TSS and COD of the effluent to be below 100 mg L^{-1}. The combination of the HRAS and the ACF also showed that it could on average achieve a 3 log decrease of CFU mL^{-1} from the influent. The removal efficiency of CFU is at least 60% in an activated sludge process or biofilm process (Farrell *et al*., 1990). The treatment system in this study performed as well as expected achieving 99.9% (3 log decrease) of CFU for the combined systems of the HRAS and the ACF. In porous media systems, pathogen removal is partially achieved by straining and sorption, which are largely determined by the filter pore sizes, hydraulic loading and clogging (Stevik *et al*., 2004). Straining would be predominant with small pore sizes (when bacteria sizes are bigger than the pore sizes), low hydraulic loading and where clogging has occurred, otherwise adsorption would take over. With the charcoal particle sizes up to 1.5 cm it is clear that adsorption was the most important mechanism of pathogen removal at the beginning of the experiment. However, with time, clogging brought about straining as the other pathogen removal mechanism. Also, the continued running of experiment allowed accumulation of macro-organisms which contribute to pathogen removal through predation. With the influent ranging from 3.13×10^2 to 2.01×10^6 FC mL^{-1}, it was possible to achieve the NEMA effluent standard of 10^2 CFU mL^{-1} for more than half of the samples (53%). Given that on average, a 2 log decrease of CFU can be achieved by the ACF system alone which consists of three filter columns, it would be possible to increase percentage of compliance by increasing the number of filter columns in the ACF system. Further studies could aim at optimising the system with regard to additional filters required to achieve 100% compliance of the CFU effluent to NEMA standards. Furthermore, with the effluent proposed to be reused in agriculture, it should also meet the standards for reuse. The World Health Organisation (WHO) guidelines require at least a 6 log decrease of pathogens from the wastewater source considering a level of contamination of 10^6 CFU mL^{-1} in the untreated wastewater (WHO, 2006). On the other hand, designing a plant to achieve a log decrease of 6 or more, only to eliminate pathogen contamination would be too expensive. It would include additional processes like chemical coagulation, flocculation and disinfection, which would generally preclude its application in many developing countries. It is therefore important that wastewater reuse strategies for pathogen removal are not just based on wastewater treatment alone. Instead, a multiple control approach should be adopted to effectively eliminate or inactivate the various microorganisms spread through different routes. WHO (2006) proposes different control measures such as cooking and washing of foods before consumption, that can be combined to achieve a total log decrease sufficient to eliminate risk of pathogen infection. Non edible crops could also be considered.

4.4 Preliminary estimation of costs

The preliminary cost estimates of the HRAS/ACF treatment system serving a small farming community of 10 houses, each with 5 inhabitants is shown in Table 4-2.

Table 4-2: Capital and operational cost estimation of HRAS/ACF system. Assuming a small agricultural community of 10 houses, with 5 inhabitants producing 100 L of wastewater $IE^{-1}day^{-1}$.

Capital Costs	**€**
HRAS CSTR[a]	60
HRAS Settler[b]	110
Charcoal filter[c]	114
Filter material[cd]	5
HRAS/ACF Instrumentation[e]	100
Total Capital cost	389
	7.8 € Capita-^{1}
Operational costs	€/m^3/d
ACF material[df]	0.012
Electricity costs[g]	0.003
Labour costs[h]	0.093
Total operational cost	0.1
	3.6 € $Capita^{-1}$ $year^{-1}$
Annualised overall cost for the treatment system[i]	4.9 € $Capita^{-1}$ $year^{-1}$

[a]Wastewater flow rate plus recycle of 0.4 m^3h^{-1}, requires a durable plastic water tank of 0.5 m^3, volume price according to a local plastic water tank manufacturer is 60 €.

[b]For a HRT of 2 h, the settling tank volume required is at least 0.8 m^3. Use a durable plastic water tank of 1 m^3 volume, local manufacture's price is 110 €.

[c]For a flow rate of 0.2 m^3h^{-1} (no recycle), total charcoal volume required is 0.5 m^3 (0.2 m^3 per filter). Use 3 plastic tanks of 0.25 m^3 a local price of € 38 each.

[d]A bag of charcoal (0.33 m^3) costs between € 10 - 20 depending on the season. However, a bag of the small pieces (< 2 cm) arising from the charcoal making process is wasted or sold at 3 €.

[e]HRAS/ACF instrumentation (pump, aerator and pipe work) is estimated at 100 €.

[f]Material in only one filter is replaced monthly.

[g]Based on a consumption of 18 Wh/d/m^3 wastewater treated. Installed power of 6 W/m^3 reactor is assumed (10 m hydraulic head, for a flow rate of 5 m^3/d and a pump efficiency of 60%) and 3 h pumping at an electricity cost of 0.09 €/kWh.

[h]Cheap unskilled labour is required to monitor pump operation time and change material.

[i]A life span of 10 years was considered and a real interest rate of 10%.

The costs are based on the lab-scale reactor operational conditions and use of locally available but durable material in Uganda. These estimations indicate that the system can treat wastewater at an overall (capital and operational) annualised cost of 5 € capita^{-1} year^{-1}. This estimate excludes the sludge line treatment. If it is included, it could be possible to recover an additional value from electricity generated estimated at 1 € capita^{-1} year^{-1} for sludge with at least 3 to5 kg DW/m^3 (Verstraete and Vlaeminck, 2011) through anaerobic digestion. The overall (capital and operational) cost of the HRAS/ACF system is less than a third the overall cost of a small scale (10,000 to 50,000 IE) conventional activated sludge system (CAS), which is estimated at about 18 to 24 € capita^{-1} year^{-1} (Zessner *et al*., 2010), excluding sludge treatment. It was also less than half the cost of the waste stabilisation pond (WSP) and the horizontal subsurface flow constructed wetlands (HSSF-CW) which can cost about 13 and 14 € capita^{-1} year^{-1}, respectively, in East Africa (Mburu *et al*., 2013). Apart from the already mentioned added value that could arise from anaerobic digestion of the sludge, the proposed system offers the community other benefits which include fuel that can be derived from the sun dried used charcoal. Furthermore, a nutrient rich effluent would go a long way to boost crop productivity for farmers.

5. Conclusions

The combination of the HRAS and the ACF can effectively remove TSS and COD from domestic wastewater to meet the NEMA discharge standards. The treatment system achieved the NEMA effluent standard for CFU for more than half of the samples. However, it would be possible to attain higher CFU removal if more filter columns are added in the ACF system. Further research is proposed to optimize the system in order to achieve 100% compliance to the CFU standard. TAN and TP were largely retained in the effluent, allowing nutrient reuse by crops. The proposed treatment system has an estimated cost which is less than half the cost of other systems such as, the small-scale CAS, WSP and HSSF-CW. It further offers a nutrient-rich effluent which will advance the re-use of wastewater for agriculture through generation of higher crop yields and profits. The novel design is therefore suggested for further development as a technology for wastewater treatment and reuse to benefit small agricultural communities. In order to effectively eliminate microorganisms and reduce pathogen transmission, it is recommended that the effluent be reused in an agricultural setting with a multi-barrier approach for example it can be used for non edible crops or where food has to be washed and or cooked before consumption.

Acknowledgements

The authors wish to acknowledge the financial support from the Vlaamse Interuniversitaire Raad (VLIR), and National water and Sewerage Corporation (NWSC) for further support in Uganda. They also wish to acknowledge Adrianus Van Haandel for critically reading through the manuscript, Nabatesa Sylvia, Chabba Charles and the team at Bugolobi STP and Central Laboratory of NWSC, Kampala, for their support in the laboratory. Willy Verstraete and Korneel Rabaey acknowledge the support from the multidisciplinary Research Partnership Gent Bio-Economy. Francis Meerburg was supported by the research Foundation Flanders (FWO).

Chapter 5 : DIGESTION OF HIGH RATE ACTIVATED SLUDGE COUPLED TO BIOCHAR FORMATION FOR SOIL IMPROVEMENT IN THE TROPICS

This chapter has been redrafted after:

Nansubuga, I., Banadda, N., Ronsse, F., Verstraete, W., & Rabaey, K. (2015). Digestion of high rate activated sludge coupled to biochar formation for soil improvement in the tropics. *Water Research*, 81, 216-222.

Abstract

High rate activated sludge (HRAS) is well-biodegradable sludge enabling energy neutrality of wastewater treatment plants via anaerobic digestion. However, even through successful digestion a notable residue still remains. Here we investigated whether this residue can be converted to biochar, for its use as a soil improver or as a solid fuel, and assessed its characteristics and overall process efficiency. In a first phase, HRAS was anaerobicaly digested under mesophilic conditions at a sludge retention time of 20 days. HRAS digested well (57.9 ± 6.2% VSS degradation) producing on average 0.23 ± 0.04 litre CH_4 per gram VS fed. The digestate particulates were partially air-dried to mimic conditions used in developing countries, and subsequently converted to biochar by fixed-bed slow pyrolysis at a residence time of 15 minutes and at highest treatment temperatures (HTT) of 300°C, 400°C and 600°C. Subsequently, the produced chars were characterized by proximate analysis, CHN-elemental analysis, pH in solution and bomb calorimetry for higher heating value. The yield and volatile matter decreased with increasing HTT while ash content and fixed carbon increased with increasing HTT. The produced biochar showed properties optimal towards soil amendment when produced at a temperature of 600°C with values of 5.91 wt%, 23.75 wt%, 70.35 % on dry basis (db) and 0.44 for volatile matter, fixed carbon, ash content and H/C ratio, respectively. With regard to its use for energy purposes, the biochar represented a lower calorific value than the dried HRAS digestate likely due to high ash content. Based on these findings, it can be concluded that anaerobic digestion of HRAS and its subsequent biochar formation at HHT of 600°C represents an attractive route for sludge management in tropic settings like in Uganda, coupling carbon capture to energy generation, carbon sequestration and nutrient recovery.

1. Introduction

Municipal wastewater treatment plants are critical for sanitation worldwide and deliver effective nutrient and carbon removal at reasonable energy inputs of ~0.5 kWh m^{-3} treated. (Rabaey and Verstraete, 2005; Alterman et al., 2006; Bodik and Kubaska, 2013). However, considerable quantities of sludge are generated, and their treatment and disposal costs weigh heavily on the wastewater industry, besides representing a potential source of pathogens. Solutions available at the plant are; reduction of the produced quantity through better plant operation, and coupling with anaerobic digestion for energy recovery. The latter typically takes away ~40% of the sludge load. Final endpoints for sludge then include landfilling, combustion and composting for farmland utilization (Sánchez Monedero and Mondini, 2004). The latter is attractive as sewage sludge is a good fertilizer for agricultural purposes (Mendez, *et al*., 2012), due to its rich nutrient value and mineralized carbon (Hossain, *et al*., 2010). Sewage sludge can thus improve the soil structure, infiltration rate and water holding capacity (Sort and Alcañiz, 1999) or soil respiration (Hernández- Apaolaza *et al*., 2000).

While the benefits of sludge are well known (Hossain, et al., 2010), there are challenges still associated with the utilization of digested sewage sludge for agriculture. A number of studies (Jamali *et al*., 2009; Smith, 2009; Hossain *et al*., 2010; Paz-Ferreiro *et al*., 2011; Oleszczuk *et al*., 2012) have strongly criticized the direct use of sewage sludge in crop production, urging that it is of high risk. This would be due to the possible presence of toxic organic components, heavy metals and some amounts of pathogenic organisms (Wang et al., 2008) posing a threat to public health (Roy & McDonald, 2014). Moreover, sludge applied directly to the soil undergoes further decomposition releasing carbon dioxide, Nitrous oxide and methane gas which goes back into the atmosphere creating an environmental concern. Furthermore, leachate from the sludge can pollute local ground and surface water.

To mitigate the negative implications of direct application of sewage sludge onto farmlands, pyrolysis of the sewage sludge into biochar has been proposed (Chan & Xu, 2009; Lehmann *et al*., 2011; Paz-Ferreiro *et al*., 2014). The biochar concept originated from a term referred to as Tera Preta soils. These are highly sustainable fertile soils occurring on over many hectares of land in Central Amazo. These soils are richer in soil organic matter and nutrient concentrations, and have a better nutrient retention capacity than the surrounding. The soils are believed to have arisen as a result of human activity at that time which caused accumulation of plant and animal residues, ash, charcoal and various chemical elements such as P, Mg, Ca, Cu and Zn (Novotny et al., 2009). In Africa Terra Pretta soils have been observed in Benin and Liberia (Sohi et al., 2010). The formation of new *Terra Preta* sites has been already suggested to help secure food production of a fast growing population. (Glaser, 2007). With respect to the application of biochar from digested sewage sludge versus the direct land application of digested sewage sludge, there are a number of benefits. Pyrolysis significantly reduces weight and volumes (Oh et al., 2011) and the high temperatures eliminate pathogenic content and foul odour (Mendez, et al., 2012) making the product easier and safer to handle. Biochar has been used to remediate soils before, exhibiting the ability for long term amendment of physical and chemical properties of soil. It improves water infiltration (Ayodele et al., 2009), soil water retention, ion exchange capacity and nutrient retention (Laird *et al*., 2010), stabilizes pH (Van Zwieten *et al*., 2010a), and improves N use efficiency (Van Zwieten *et al*., 2010b). Biochar lowers heavy metal availability in the soil hence decreases risk of leaching of heavy metals (Méndez *et al*., 2012) and reduces plant uptake of these elements (Hossain *et al*. 2010; Moustafa *et al*., 2013). Biochar has also been widely promoted as a carbon sequestration tool as the carbon is only very slowly released (Lehmann *et al*., 2006). Other studies with regard to fuel show that biochar can suitably replace use of wood fuel and charcoal for common heating purposes (Fonts et al., 2009). Biochar production would also provide the farmer with a suitable way of managing farm waste, which if not well managed can be an environmental threat that could lead to pollution of nearby surface waters (Matteson and Jenkins, 2007). This also reduces the volume of waste and offers an easier way to handle it. Farm waste can be converted to biochar, packaged, stored and even marketed to generate more income. Biochar production in general, may contribute significantly to managing organic farm wastes in future. It is important to note though that, it will, most likely, not be able to solve poverty issues and further research is recommended to establish it's fertilizer characteristics before the biochar can be marketed as fertilizer. Being a new technology there are still a few uncertainties especially with its application on the long term. Biochar has been shown to have mixed effects on soil quality properties in the short term, as effects can be negative, e.g. reduced mineral N availability (Nelissen et al., 2013). Also the economic benefit of biochar is still not certain which may limit its economic opportunities in the developing world. However some studies suggest that biochar can potentially be produced cheaply through traditional charcoal production methods (Dickson *et al*., 2014). A number of simple and cheap technologies are highlighted by FAO (1983).

Whereas the existing studies focused on biochar from conventional activated sludge, no studies have thus far used high rate activated sludge (HRAS). HRAS is generated in the so-called AB-process which was developed by Bohnke *et al*., (1977) and which enables energy neutrality of wastewater treatment plants through the production of a considerable fraction of highly biodegradable sludge. This approach is now increasingly applied worldwide with a number of wastewater plants gradually claiming their role as energy recovery plants instead of just being nutrient and pollution removal plants (Wett *et al* ., 2007). The HRAS could thus be digested at reasonable efficiency, delivering a digestate that can be further converted to biochar. To investigate this, we obtained HRAS, subjected it to anaerobic digestion and upon air-drying produced several types of biochar. We characterized the product and assessed its value towards soil improvement based on known requirements.

2. Materials and methods

2.1 HRAS sludge source.

HRAS, as well as inoculum sludge for anaerobic digestion were collected from the municipal WWTP of Nieuwveer (Breda, the Netherlands), and stored at 4°C. This WWTP has implemented the AB-Boehnke system which consists of two stages, the A-stage which is a biosorption processes and the B-stage which consists on a nitrogen treatment step. Sludge used in this study was produced from the A-stage, herein referred to as the high rate activated sludge (HRAS). The characteristics of the inoculum sludge as well as the HRAS are described further in Table 5-1.

2.2 Anaerobic digestion of the high-rate activated sludge (HRAS)

Anaerobic digestion of the sludge from the HRAS system was done to determine its biogas formation potential. In the Laboratory, Schott bottles (1 litre) were used as anaerobic reactors. Three reactors were thus each filled with 800 mL of anaerobic inoculum sludge obtained from Breda WWTP and incubated at mesophilic conditions (36 °C) in a semi-continuously stirred reactor (SCSTR) mode. They were also, semi-continuously fed with sludge from the HRAS from Breda for 74 days. During the start-up period, the daily organic loading rate was started at 0.35 g COD/L.d and it was gradually increased up to an average value of 1.85 ± 0.63 g COD/L.d to obtain the desired sludge retention time (SRT of 20 d) after day 15. The pH, biogas production and percentage of methane in the biogas of the reactors were monitored, three times a week.

2.3 Digestate preparation and biochar production

After 20 days, effluent (digestate) collection from the anaerobic digestion experiment started. The digestate was collected every time the digester was fed (Three times a week). It was allowed to settle, the clear water was poured off, and the sludge was partially air dried for 5 to 7 days (to minimize water content without major carbon loss). It was then oven dried at 104°C and then allowed to cool in a desiccator for 20 minutes. For the pyrolysis tests, 15.3 g, 13.7 g and 13.1 g of oven dried digestate particulate sludge was packed in a vertical tube and pyrolysed at 300, 400 and 600°C respectively to form biochar. The slow pyrolysis reactions were carried out in a vertical, tubular, stainless steel reactor which was heated by an electric tube furnace (schematic in Figure 5-1). The reactor setup is similar to that used in Ronsse *et al*., (2013). The reactor tube holding the biomass sample had an inner diameter of 18.5 mm. The reactor was operated at atmospheric pressure. Each pyrolysis experiment consisted of heating the reactor at the maximum heating rate (10°C min^{-1}) until the highest treatment temperature (HTT) was reached. The reactor was then kept at the nominated HTT for a residence time of 15 minutes, before the furnace was shut off and the reactor ambiently cooled. The reactor was continuously swept with nitrogen, at a rate of 40 ml/min, to remove the gases produced during pyrolysis. The nitrogen flow was continued during cooling to purge the reactor of any remaining pyrolysis gases and to prevent any oxygen exposure to the char while still above ignition temperature.

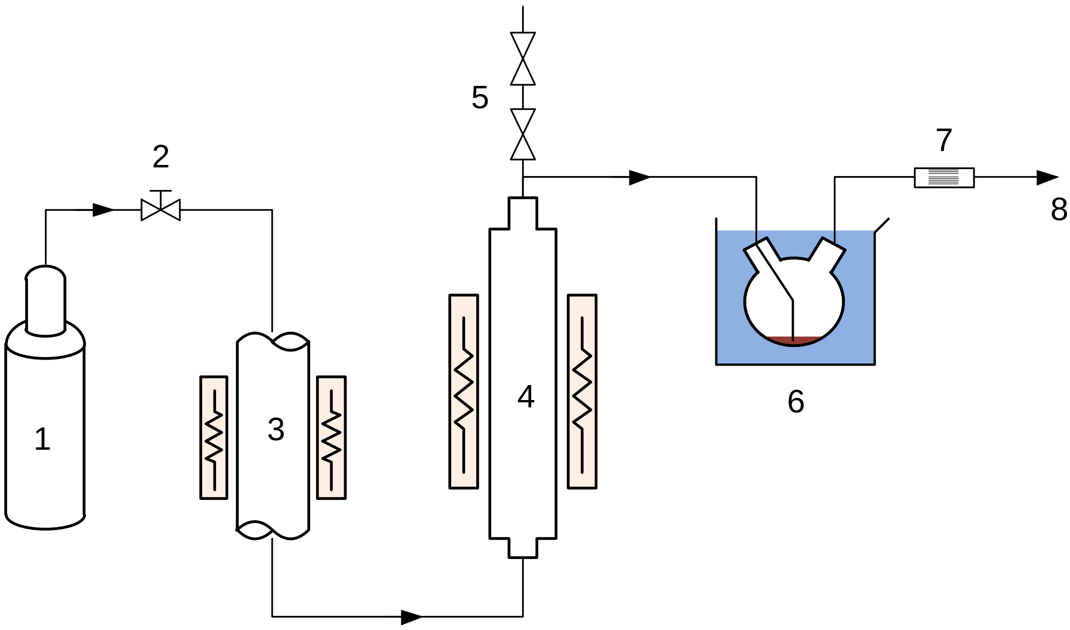


Figure 6: Slow pyrolysis set-up for the production of biochar: (1) nitrogen gas supply, (2) flow control, (3) gas preheater,(4) stainless steel pyrolysis reactor, (5) biomass lock

hopper, (6) condenser (impinger flask submerged in ice water bath) and condensate separator, (7) cotton filter, and (8) non-condensable gas vent

2.4 Analytical methods

Samples taken from the substrates (HRAS) and inoculum sludge were analysed for volatile solids (VS), total solids (TS), chemical oxygen demand (COD), and total ammonium nitrogen (TAN) as described in Standard Methods (APHA, 2005). The pH was measured with a C532 pH meter (Consort, Turnhout, Belgium). Effluent samples from the digester were also taken once a week and analysed for the same parameters. The biogas produced in the anaerobic reactors was captured in 5 L perspex gas-o-meters. Gas was transferred to the inverted cylinders through air tight plastic tubing from each reactor. Biogas production and pH in the reactors were monitored on a daily basis for 74 days. Gas samples from each reactor were taken using a syringe on a weekly basis. The biogas composition (CH_4, CO_2 and H_2) were determined with use of a compact gas chromatography (GC-2014 gas chromatograph, Shimadzu, s-Hertogenbosch, the Netherlands).

Volatile fatty acids (VFA) in the digesters were analysed once a week, they were extracted using diethyl ether and measured in a GC-2014 gas chromatograph (Shimadzu, s-Hertogenbosch, the Netherlands). The lower detection limit for VFA analysis was 2 mg L^{-1}

2.5 Biochar characterisation

The yield of the recovered biochar was expressed as weight percentages of biochar recovered to initial dried HRAS digestate used. The yield (ɳ) was calculated by equation (i):

$$\eta = \frac{W_0}{W_1} .100\% \qquad \text{(i)}$$

Where W_0 is the weight of the char recovered from the pyrolysis reactor (g) considered to be oven dried, and W_1 is the weight (g) of the oven dried HRAS digestate before pyrolysis.

Proximate analysis to determine moisture content (MC), volatile matter on dry basis (VM_{db}) and ash content on dry basis (AC_{db}) were determined according to D1762-84 (ASTM, 2007). Biochar samples of ca. 1 g in triplicate were heated in porcelain crucibles and the sample weight differences before and after heating were determined. For moisture content, samples were oven dried at 105 ° C for 2 h, while for volatile matter, samples were heated to 950 °C for 11 min (covered crucible) and for ash content 750 °C for a minimum of 2 h (uncovered crucible). Mc, VM_{db} and AC_{db} were calculated based on equations ii, iii and iv, respectively.

$$MC,\% = \left[\frac{(M_1 - M_2)}{M_1}\right] \times 100 \quad \text{(ii)}$$

Where; M_1 is the mass of the sample before oven drying, and M_2 is the mass of the sample after drying at 105°C.

$$VM_{db},\% = \left[\frac{(M_2 - M_3)}{M_2}\right] \times 100 \quad \text{(iii)}$$

Where; M_3 is the mass of sample after drying at 950°C.

$$AC_{db},\% = \left(\frac{M_4}{M_2}\right) \times 100 \quad \text{(iv)}$$

Where; M_4 is the mass of residue after drying to constant mass at 750°C.

The stable carbon fraction of the sample also termed as the fixed carbon on dry basis (FC_{db}) was determined and calculated based on equation (v).

$$FC_{db}, \% = 100\% - (VM_{db} - AC_{db}) \quad \text{(v)}$$

Elemental (CHN) analysis was performed in duplicate using a Flash 2000 Elemental Analyser. (Thermo Fisher Scientific, Waltham, MA, USA). The higher heating value (HHV) of chars and the HRAS digestate were determined in triplicate by bomb calorimetry, (Parr model 6200 Isoperibol calorimeter with a model 1108 oxygen bomb, Parr Instrument Company, Moline, IL), according to the instructions of Parr sheet no. 205M, 207M, and 442M.

3. Results and discussion

3.1 Anaerobic digestion parameters

The digesters were operated for 74 days on HRAS after a start-up with inoculum from the same plant. Table 5-1 depicts the influent, inoculum and effluent properties obtained during stable operation.

Table 5-1: Characteristics of the HRAS, inoculum sludge and the Effluent (digestate) during the study

Parameter	Inoculum	HRAS	Effluent
Total COD (g L^{-1})	26 ± 6	42 ± 9	15 ± 3
Total solids (g L^{-1})	39 ± 5	45 ± 8	20 ± 4
Volatile solids (g L^{-1})	24 ± 3	28 ± 7	11 ± 3
COD:VS ratio	1.08 ± 0.28	1.50 ± 0.65	1.36 ± 0.37
TS:VS ratio	1.63 ± 0.09	1.54 ± 0.10	1.84±0.18
TAN (g L^{-1})	2.81 ± 0.58	1.09 ± 0.25	0.65 ± 0.21

The pH over the entire 74 days remained at 7.16 ± 0.15 which is within the range of 6.5 and 7.6 required for optimal conditions for anaerobic digestion (Parkin & Owen, 1986). Residual VFA concentrations in the reactor were below 460 mg COD L^{-1} throughout the experiment, showing good conversion of hydrolysed material to methane. The trend of gas production during the entire 74 days is shown in Figure 5-2. The methanation was steadily increasing during the 20 d SRT experimental period, with an average biogas production rate of 0.68 ± 0.18 L L^{-1} d^{-1}. The average methane percentage in the biogas was 72.5 ± 4.1%, correlating to average methane production rate of 0.5 ± 0.15 L L^{-1} d^{-1}. The VS removal efficiencies in the digesters at SRT of 20 days were 57.9 ± 6.2% which is similar to those observed by De Vrieze et al., (2013), when HRAS was digested at mesophillic temperatures. HRAS digested well producing on average 0.23 ± 0.04 litre CH_4 per gram VS fed.

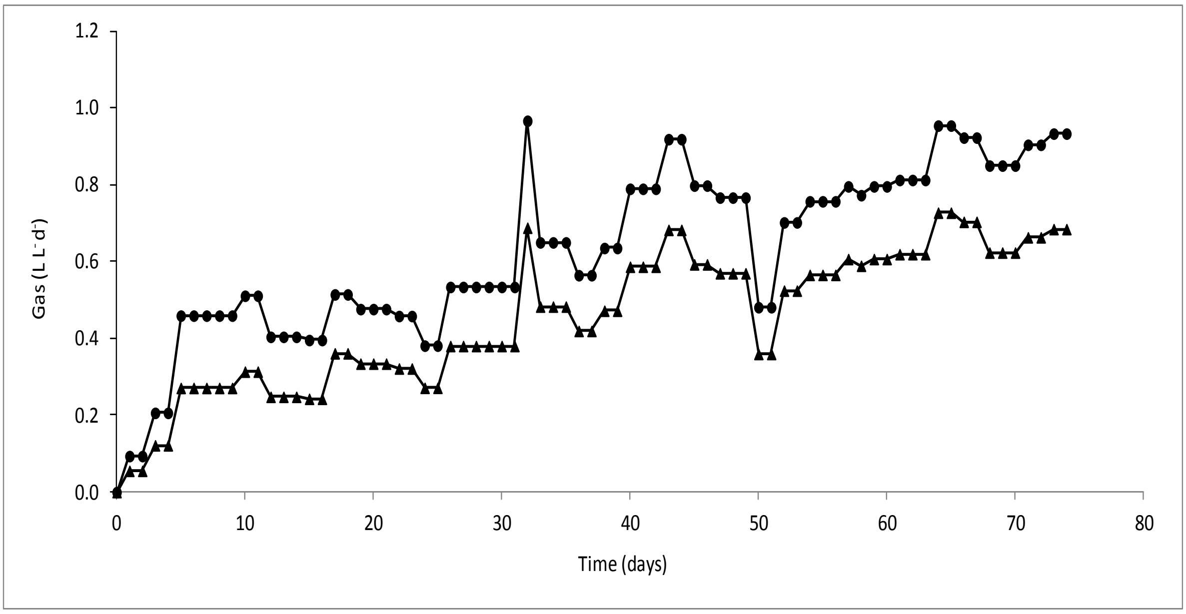


Figure 7: (a) Gas production in terms of biogas (●) and methane (▲) during the mesophillic digestion of HRAS.

3.2 Biochar yield

On average 56% of the original biomass was removed through anaerobic digestion, leaving 44% which could be converted to biochar. The general properties of the biochar produced at the different temperatures are shown in Table 5-2 and Figure 5-2. The yield of biochar is highly dependent on the pyrolysis temperature. It decreased with increased HTT as expected. Weight loss was 22.2%, 38.7% and 47.6% at HTT of 300°C, 400°C and 600°C respectively. Other studies have observed similar trends (Tsai et al., 2007; Hossain et al., 2011; Enders et al., 2012; Crombie et al., 2013; Ronsse et al., 2013). The decrease in biochar yield with increased pyrolysis temperature can be attributed to decomposition and devolatilazation of sludge constituents (Oh et al., 2012). The yield values are similar to those observed by Hossain et al. (2011) when conventional secondary sewage sludge was pyrolysed. However, compared to other biomass like wood, straw, green waste and dry algae, the HRAS digestate has higher yields. At the same HHT of 600°C, wood, straw, green waste and dry algae showed low yields of less than 26% (Ronsse et al., 2013) while that of the dried HRAS digestate here showed a yield of 53%. This may be attributed to the high ash content in the biochar.

Table 5-2: Selected properties of HRAS sludge, biochar produced at 300°C, 400°C and 600°C.

Process conditions	Proximate Analysis					Elemental Composition		
	Biochar yield (wt%)	Moisture content (wt%)	Volatile Matter on dry basis (wt%)	Ash Content on dry basis (wt%)	Fixed Carbon content on dry basis (wt%)	H/C ratio	Calorific Value HHV (MJ/Kg)	pH
HRAS Sludge	n.a	2.54±0.43	48.44±0.70	38.79±0.16	12.77±0.54	1.73±0.02	14.09±0.24	6.31±0.00
Biochar at 300 °C	77.8±5.9	0.05±0.05	33.96±0.54	49.43±0.15	16.62±0.40	1.25±0.01	14.31±0.16	6.51±0.01
Biochar at 400 °C	61.3±2.5	0.11±0.09	18.11±0.36	60.58±0.43	21.32±0.09	0.97±0.00	11.68±0.07	7.23±0.03
Biochar at 600 °C	52.4±1.5	0.49±0.15	5.91±0.02	70.35±0.15	23.75±0.15	0.44±0.00	9.26±0.11	7.73±0.02

3.3 Proximate analysis

Proximate analysis was performed to measure the key properties such as moisture content, volatile matter, fixed carbon and ash content of the biochar. The pyrolysis temperature affected the properties of HRAS biochar as shown in Table 5-2. The volatile matter was less in the biochar when compared to the dried HRAS and it decreased with increased HTT. On the other hand, both ash content (on a dry basis) and fixed carbon content (on a dry basis) where higher in the biochar than in the dried HRAS. The volatile matter decreased from 44.4% in the oven dried HRAS digestate to 5.9% in the biochar produced at 600°C. The ash content increased from 39% in the raw sludge to over 70% in the biochar produced at HTT of 600°C. This is expected as, in pyrolysis, ash remains in the solid fraction whereas the organic matter undergoes increased thermal decomposition, resulting in weight loss in the carbon-containing fraction of the feed. Other studies have shown similar trends for the different feed material used (Masek et al, 2011; Enders et al., 2012; Ronsse et al., 2013).

The observed increase in fixed carbon is supported by the fact that during slow pyrolysis, a series of devolatilization reactions occur that progressively leave behind an increasingly condensed carbonaceous matrix (Ronsse et al., 2013). Increase in fixed carbon is closely related to increased stability of char in soil (Crombie et al., 2013).

3.4 Elemental composition, higher heating value and pH of the biochar (biochar properties)

The feedstock biomass (dried HRAS digestate) had a high H:C ratio which decreased with increased temperature (Table 5-2). The H:C variation is similar to that observed by Schimmelpfennig & Glaser (2012); Sun et al. (2012) and Ronsse et al. (2013). The H:C ratio can give an indication of biochar stability

While there is still a need to develop more complete and precise methods for estimation of stable carbon, a number of methods for assessing biochar stability have already been proposed and are acceptable. After evaluating a number of proposed methods, IBI (2013) classified the existing methods as Alpha, Beta and Gama methods. Alpha methods are reliable and fast but don't provide an absolute measure of stability like the Beta methods which include incubation and accelerated oxidation tests. These are however, very tedious and lengthy. The Gamma methods verify the legitimacy of the Alpha and Beta methods through establishing strong relationships between the properties measured by them. The use of the beta and Gama methods were beyond the scope of this study. Alpha methods could be, the hydrogen to organic carbon molar ratio (H:C) (Enders et al., 2012; IBI, 2013), Oxygen to Carbon molar ratio (O:C) (Spokas, 2010) and the volatile matter (Spokas, 2010; Zimmerman, 2010; Enders et al., 2012;). Biochars with volatile matter of below 40% are considered stable (Zimmerman 2010), although at high ash content, this may be affected (Enders et al., 2012). The use of VM was discarded as a well-suited predictor of stability (IBI, 2013). Also, some studies showed that poultry waste, paper and wood biochar obtained similar H:C ratios, whereas wood biochar is known to be more stable than the others (Whitman, 2011; Enders et al., 2012). Nonetheless, the H:C ratio was suggested to give a better indication, and is fronted as an acceptable method that can be used to estimate biochar stability (IBI, 2013; 2014, Nelissen, 2013). For a given feedstock, volatile matter and H:C ratios decrease as biochar stability increases (Nelissen, 2013). Also, the labile C fraction of biochar is significantly correlated to H:C ratio and volatile matter content, indicating that it is also a good indicators for biochar stability. Biochar C content is also correlated (negatively) with the labile C fraction, indicating that when fixed C content increases, the biochar is more stable. Also higher temperature yield more stable biochar.

O:C ratios have been shown to correlate well with stability of biochars (Spokas, 2010) and they are closely related to H:C (IBI, 2014). H:C ratio was selected to predict the stability of the formed biochar in this study. According to IBI (2013; 2014), biochars with values of H/C of 0.4 and below are characterize as highly stable, they will have at least 70% organic carbon remaining in the soil, 100 years after application. With a H:C ration between 0.4 to 0.7, the biochar is considered stable, with potentially 50% organic carbon remaining in the soil after 100 years from application. The H:C of the dried sludge was 1.74 and it reduced with biochar produced at increasing HHT of 1.25 in the biochar produced at 300°C to 0.97 and 0.44 for biochar produced at 400°C and 600°C respectively. H:C ratio of ~0.4 depicts the biochar formed at 600°C to be very stable. The biochar formed at 400 and 300°C however has a H:C ratio higher than 0.7 which indicated that at these HHTs, the biomass was altered but not was not yet thermo-chemically converted (IBI 2014). Thermo-chemical conversion could however be achieved by increasing the HHT and or the residence time. Because the value of this biochar as a fuel will be low due to the high ash content, efforts should be directed to valorise it as a soil improver.

With regard to Carbon, a high retention of carbon in the biochar was observed which decreased with increasing HHT. It was 100% retention for the biochar at 300°C and 87.5 and 72.4% retention for the biochar at 400°C and 600°C respectively.

The calorific values of the biochar are expressed by their higher heating values (HHV). The calorific values are lower in the biochar when compared to the dried HRAS digestate (Figure 4), and decreased with increasing pyrolysis HTT. While uncommon, this phenomenon has been observed previously for feed materials such as algae. In those cases, the effect was attributed to high ash content (for algae, 38.2%) (Ronsse et al., 2013). It is known that energy densification from pyrolysis only occurs in the organic fraction of the feedstock. The dried HRAS digestate was generally observed to have high ash content which rose to above 70 wt% when the HTT was 600°C. This explains the low values of HHV when compared to HHV in biochar formed from other materials (Soares et al., 1997; Ronsse et al., 2013). The high ash content is caused by the solids separation technique chosen here: air-drying. We specifically opted for this option since for developing country settings, air-drying is the mode of action with sludge, rather than centrifugation, typically used in biochar studies.

The pH for the biochars was higher than the pH of the dried HRAS digestate (Table 5-2). The pH was 6.31, 6.51, 7.23 and 7.73 for the dried HRAS digestate and biochar produced at

300°C, 400°C and 600°C respectively. Biochar produced at low temperature is acidic and becomes more alkaline in nature at high temperature, in line with previous studies (Hossain, et al., 2010, Ronsse et al., 2013). The trend correlates with the increase in ash content at increased temperatures, which contributes to high pH besides a decrease in carboxyl groups and acidic groups becoming deprotonated to the conjugate bases (Ronsse et al., 2013).

4. Conclusions

Overall, our approach represents an effective way to harvest carbon. A typical A-stage removes about 80 % of incoming organics in a WWTP without major mineralization loss. About 64% of this was digested in our study without need for sludge pre-treatment and using an approach typically done on site, delivering 0.44 g organic residue and up to 11 kJ as methane gas per gram VS digested. The subsequent conversion to biochar at 600^0C HHT delivered ~0.23 g biochar. The produced biochar showed optimal properties as a soil improver when produced at a temperature of 600°C, with the H:C ratio being the lowest at 0.44 indicating a very stable biochar. With regard to fuel value, however, the biochar produced at the different temperatures had lower calorific values than the dried HRAS digestate, likely due to high ash content. Thus, a possible optimal management strategy for HRAS, would be to recover energy via anaerobic digestion and subsequently have biochar produced from the dried digestate. The biochar could be applied as a soil improver to boost agricultural productions and it may contribute significantly to managing organic farm wastes in future. However, on its own, it will, not be able to solve poverty issues in the developing world in relation to food production. Also, it recommended that further research be done, to establish the biochar characteristics of the HRAS digestate biochar before it can be marketed as a fertilizer. Additionally, there is still need for cheaper big scale production units of biochar for it to thrive the economic opportunities in the developing world. Lastly, being a new technology, there are still a few uncertainties especially with its application on the long term hence the need for further research. Where the biochar production is not favourable therefore, the digestate after anaerobic digestion, could instead be dried and directly used on land or as fuel. It is also important to explore other alternative inventive application of the biochar such as using it a filter media or as a component of black paint.

Acknowledgement

The authors gratefully acknowledge the financial support of VLIR-UOS, the Multidisciplinary Research Partnership Ghent Bio-Economy and National Water and Sewerage Corporation (NWSC), in conducting this study. The authors also kindly appreciate Robert Nachenius, Dane Dickinson and Martínez Rodríguez (Department of Biosystems Engineering, Faculty of Bioscience Engineering, Ghent University) as well as Cyrus Galyaki, (Makerere University) for their help with the biochar formation and analysis.

Chapter 6 : GENERAL DISCUSSION AND PERSPECTIVES

1. Introduction

This work aimed at exploring the innovative, resources recovery options with respect to the developing world with focus on Uganda. As such the current sanitation options were explored and new ideas proposed. The outcome of this work provides new insights that could create sustainable wastewater management strategies for the future. This work considers not only domestic wastewater but also integrates wastes from other sources to bring about better wastewater treatment performance as well as increase resource recovery benefits.

This study has resulted in a better understanding of resource recovery options from wastewater with respect to energy recovery, new water recovery and nutrient recovery. The main message from this work is the need for the wastewater industry to move from the ordinary conventional centralised systems which is energy consuming, to a more decentralised system that allows for optimal recovery of resources hence bringing about a more cost effective and manageable wastewater treatment system.

This chapter integrates the obtained results with the findings from literature and identifies future challenges and critical research needs.

2. Main outcomes and positioning of this work

2.1 The decentralised system as a suitable option

Chapter 1 evaluated the existing sanitation options in Africa, highlighting the failures in onsite sanitation and the central system in most of cities. The future wastewater management plans should encourage zero waste generation through decentralisation and recovery of water, energy and nutrients. The integration of chapters 2-5 present a decentralized wastewater management scheme proposed for a small agricultural community. Central in the concept, is to achieve as fast as possible separation of the used water by means of a low cost and simple method. First, separation with use of water treatment poly-aluminium sludge (WT-PAS) was considered as demonstrated in chapter 2. When added to wastewater, WT PAS increased the settleability of particles during sedimentation which makes it a good option for the handling of sludge produced in the water treatment industry. Secondly, a biosorptive sludge system (SRT 2-3 d) also called a high rate activated sludge (HRAS) system was considered as a means for separation of the solids. This is detailed in the Chapter 4, where a decentralized concept for the treatment of domestic sewage is demonstrated. The concept consists of the HRAS system in which suspended organic matter is removed and (ii) the alternating charcoal filters (ACF)-stage, which consists of charcoal filters in series, to achieve further organic matter removal. The system showed good removal of TSS and COD while leaving some nutrients in form of nitrogen and phosphorus which makes it a good option for agricultural communities.

After separation it was demonstrated that the solids can be further treated to recover energy (chapter 3, 4 and 5) and nutrients (chapter 4 and 5), which could also be recovered from the liquid part (chapter 4). Through anaerobic digestion as demonstrated in chapter 3 and chapter 4, it was shown that primary sludge and HRAS could yield methane which could be used for energy production. On the other hand the Liquid part could be re-used for agricultural purposes. The digestate solids from the anaerobic digestion of HRAS were dried and converted to biochar (chapter 5), which showed a stable product. It was however observed that the dried sludge had more calorific value than the biochar.

The benefits attached to decentralization in wastewater management are enormous (Libratalo et al., 2012). This is why it is increasingly gaining recognition as one of the strategies towards increasing sanitation coverage. The major hindrance to implementation of the decentralised system would be the costs involved with replacing the old sewerage network. However, unlike the developed world where the central sewerage system has a wide coverage, in the developing world, the onsite system is more predominant accounting for 60-100% sanitation coverage in many African cities (WHO, 2000). This would eliminate the expenses associated with replacing the central system and therefore presents a great opportunity for the developing world to easily adopt a decentralised system, which presents a more affordable wastewater management system.

2.2 Wastewater as a resource for energy

This work has shown the energy recovery potential from the both primary sludge and HRAS. While the biodegradability of primary sludge alone was poor (chapter 3), however, the option of co-digestion has shown that other waste products could optimise the anaerobic digestion process for biogas and energy production. The feed option with 50% STP sludge and 50% brewery sludge showed the highest biogas yield and production rate but the one with 50% STP sludge, 25% brewery sludge and 25% cow dung was selected as the optimal mixture for practical application.

In chapter 5, HRAS was anaerobically digested and it showed good biogas production (0.5 ± 0.15 CH_4 L L^{-1} d^{-1} for an average OLR of 1.85 ± 0.63 g COD $L^{-1}d^{-1}$). The effluent sludge was consequently dried and converted to biochar and the calorific values determined. Interestingly, the energy potential was found to be higher in the dried sludge than in the formed biochar. Meaning, if the need was for energy, dried sludge was better to use for heating purposes and there was no need to first convert it into biochar.

2.3 Wastewater as a resource for new water and nutrients

It has been demonstrated in chapter 4 that wastewater can be treated to achieve new water that can be re-used for other purposes. The combination of the HRAS plus the ACF has offered important insight for the possibility of re-use of wastewater for agricultural purposes in an economically viable way. The system produced an effluent which was rich in nutrients and low in organic contaminants and faecal coliform. Meaning it could be re-used for irrigation of crops.

Another option for nutrient recovery was explored in chapter 5 through a proposal to utilize biochar produced from HRAS sludge. Biochar produced from the HRAS was shown to be moderately stable and therefore good for use for agricultural purposes. The biochar could be combined with the used charcoal produced in the charcoal filters (chapter 4).

3. Application of the study

At the beginning of this study we sought to contribute to strategies that could help to improve sanitation coverage in the developing world. The study aimed at exploring the use of optimised resource recovery techniques, which would also enhance monetary benefits for the users. These techniques had to be affordable and would require minimal skills for easy application in the developing world.

We have proposed optimised techniques with regard to pre-concentration of solids in wastewater, further treatment of liquid stream of the wastewater to re-usable standards and recovery of resources from sludge. The proposed techniques which are linked to existing concepts have been suitably integrated into the developing world setting to present new ideas that can enhance improved sanitation coverage for the developing world. The proposed wastewater treatment and resource recovery option has potential benefits when applied for a small organised community e.g. an agricultural community in a semi-urban setting, a prison, a hospital and learning institutions (schools). It can also benefit small and medium scale entrepreneurs and enterprises (SMEs) working in agricultural and manufacturing business. A typical setting is proposed in Figure 6-1. In this chapter the practical application and operation of the proposed techniques in the developing world are detailed.

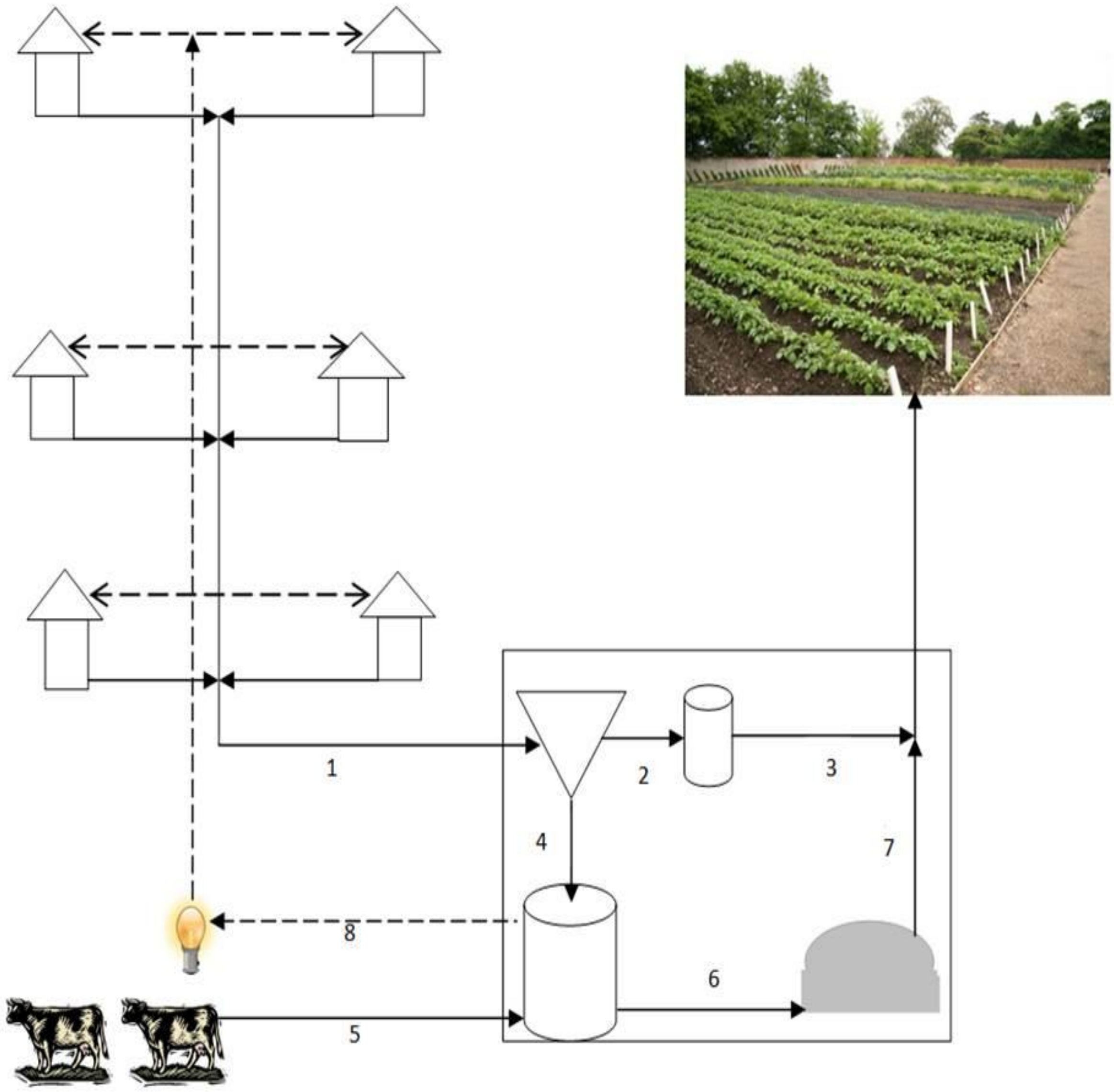


Figure 6-1: The proposed decentralised system plan: (1) The sewer line from different homes collects sewage, (2) effluent from the HRAS is led to the charcoal filters, (3) effluent from the charcoal filters goes to irrigate the agricultural land, (4) Sludge from the HRAS feeds the digester, (5) dung from cattle carried through a pipe to be co-digested with the HRAS sludge, (6) Effluent from the digester is dried and the solids sent to the kiln for biochar formation, (7) formed biochar as well as used charcoal is applied as a soil improver in the agricultural land and (8) Pipe line carries back biogas for lighting and heating in the homesteads.

3.1 Proposals for Implementation and operation

The study proposes wastewater treatment by HRAS plus the ACF where applicable, followed by anaerobic digestion of HRAS sludge (plus co-digestion with farm wastes) and the digestate be dried and converted to biochar. The other option could be co-digestion of primary sludge with farm waste and brewery waste; the digestate can be dried and converted into biochar. For practical application, two settings are considered a peri-urban setting with flushing toilets and a rural setting without flushing toilets. Different stages treatment stages are therefore considered for the two settings

3.1.1 Collection and treatment of domestic waste and other substrate

(a) Peri-urban setting

The peri-urban setting is suitable for institutions such as school, hospitals, prisons, small business entrepreneurs and a cluster of houses in the neighbourhood in a peri-urban setting. The common sanitation system would otherwise be use of septic tanks. These communities can afford to lay small size inter-connection pipes.

*Domestic waste*water: The domestic waste water can be directly connected to feed the HRAS and there after the sludge directed to the digesters with use of simple PVC pipe network. For institutions it may be easier if the HRAS+ACF treatment system as well as the digester can be positioned next to the communal toilets hence minimal pipe connection is required. For the clustered homes, (these would ordinarily otherwise use similar pipe work to connect to septic tanks) similar pipe work can be used to connect to the central treatment unit.

Manure: can be provided by neighbouring farmers who may carry it manually to the mixing chamber where the anaerobic digester is fed. Farmers can be motivated by getting a bag of biochar for every cubic meter of manure they deliver to the treatment unit.

Brewery waste: brewery waste can be collected from nearby commercialised brewery industry or from local brewers who have small scale brewing businesses. The brewery can be collected manually with use of wheel barrows and plastic containers and fed to the anaerobic digesters.

(b) Rural setting

In a rural setting, flush toilets don't exist and pipe network is not affordable. The common sanitation system would otherwise be pit latrines. For our setting however it is proposed that pit latrines are modified to feed the digesters directly or indirectly.

Sewage: For institutions such as schools it would be possible to have a pit latrine replaced with an anaerobic digester which is directly fed. Figure 6-2 shows a simple illustration of how this can be achieved. Where that is not affordable e.g. for the clustered homes with a common digester placed centrally, a special pit latrine build above ground, similar to the EcoSan toilet is proposed (Figure 6-3). With this the contents can continually be sent into a plastic containers which is manually transferred to a nearby anaerobic digester after certain period (4 to 7 days). This being a manual and time consuming activity requires commitment from the users who would be motivated by the benefit from the end products such as biogas. Protective wear is very imperative for the people involved, in order to reduce health risks associated with handling sewage.

Cow dung and brewery waste. Cow dung, brewery waste and any other digestible waste can be collected manually in Plastic containers on wheel barrows, from the farmers and brewers within the neighbourhood.

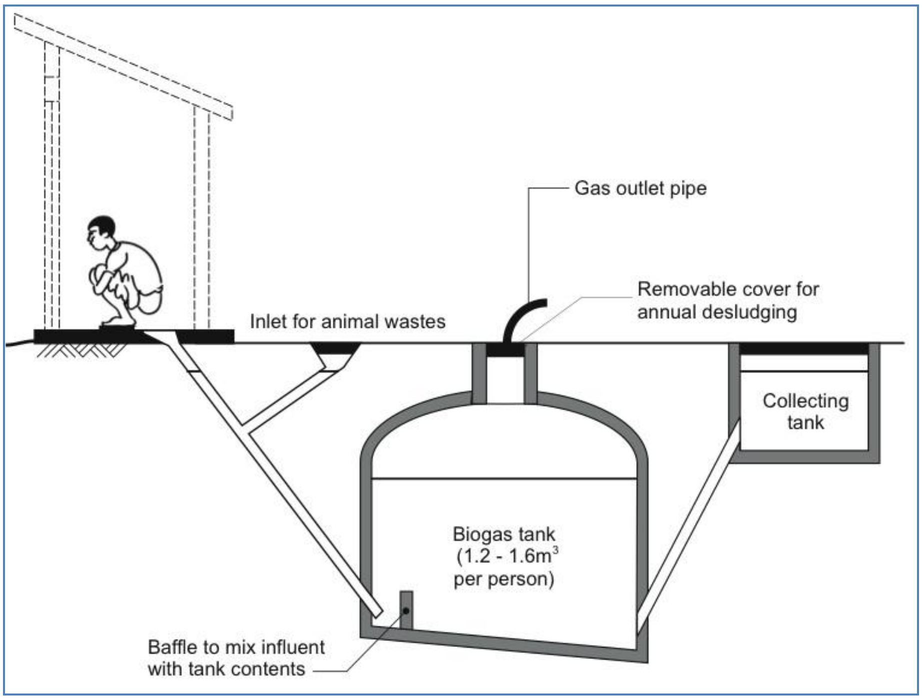


Figure 6-2: Small-scale biogas digesters receiving direct feed of sewage and having an option for other organic wastes input. Source: WELL (n.v.)

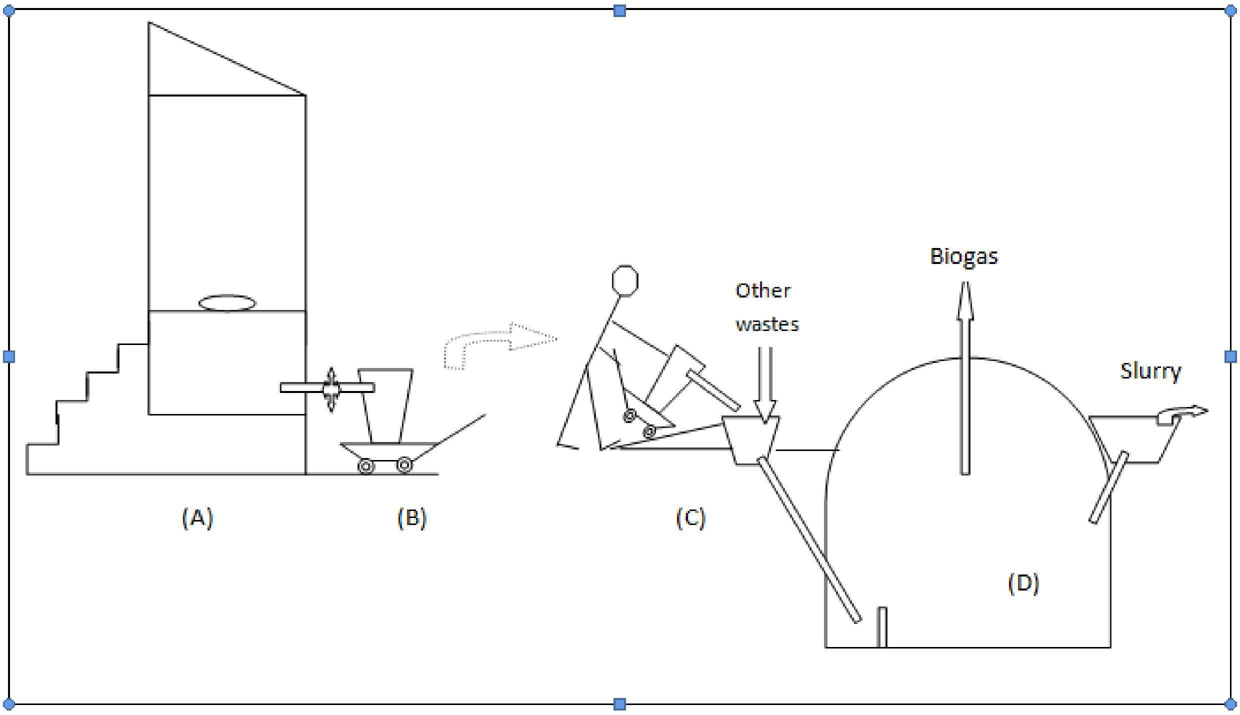


Figure 6-3: Illustration of (A) a modified pit latrine (raised above ground) with a collection chamber with an outlet pipe connected to (B) a sealed plastic container attached to a wheeler for easy transport. The pipe has two connection valves, one for the pit latrine and the other for the plastic bucket. Both can be closed off when the plastic container is de-touched. (C) Demonstrates an individual transporting and feeding their domestic waste from the plastic container into (D) a central Anaerobic digester where other homes bring their domestic wastes and farm wastes.

3.1.2 Operation of the HRAS +ACF

The mode of operation of the HRAS has been described in Chapter 4. It is important to note that this would be suitable in the peri-urban Areas for organised communities where flushing toilet systems are used for example in institutions (schools, hospitals prisons e.t.c). These would otherwise collect the waste via simple sewer network and lead it to septic tanks. The septic tank therefore is proposed to be replaced with the HRAS system. Part of the power from the biogas production can be used to run the systems mixing motor and aerator.

3.1.3 Operation of the anaerobic digestion system

Biogas digesters are of mainly types; fixed-dome plants, floating-drum plants, balloon plants, horizontal plants, earth-pit plants and ferrocement plants. The fixed dome plants and the floating dome plants are proposed here as they are the most familiar types in the developing world.

Fixed Dome: The fixed-dome plants consist of a digester with a non-movable gasholder fixed on top of the digester. When gas production starts, the slurry flows out through the outlet into collection tank. The costs of a fixed-dome biogas plant are relatively low. It is simple as no moving parts exist and have a long life of more than 20 years. These are usually constructed underground, which protects them from physical damage, saves space and helps to stabilize temperature as it's protected from the cold temperatures at night. Their construction is however labour intensive and requires skilled supervision, otherwise it may not be gas tight (Kossmann et al .,1999). The Chinese fixed-dome plant is the archetype of all fixed dome plants. Several million of them have been constructed in China (Figure 6-4) .

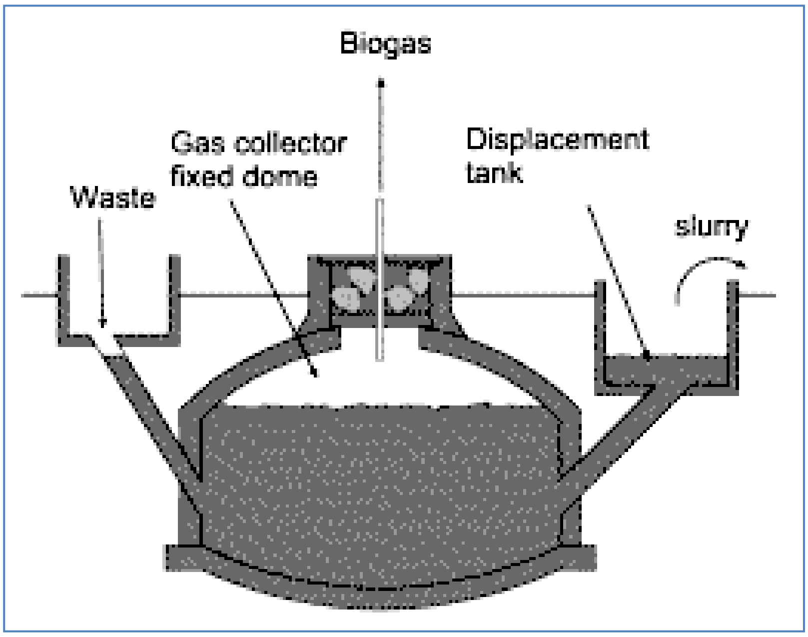
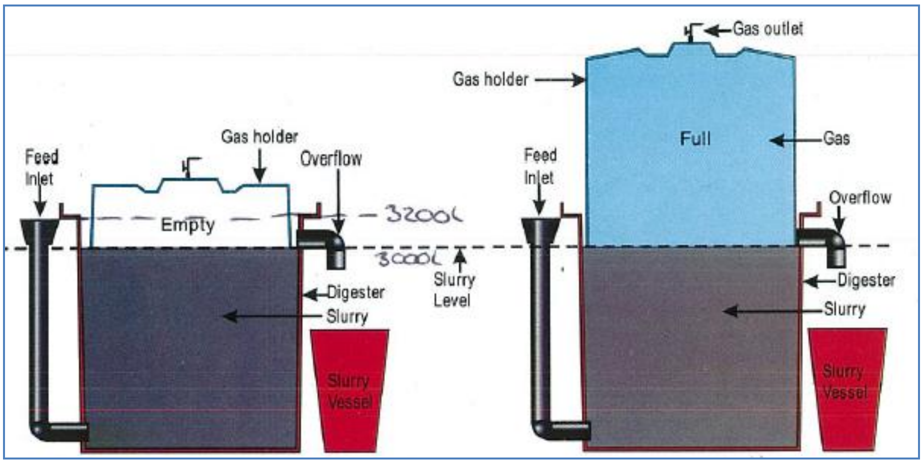


Figure 6-4: Chinese fixed dome plant. Source Kossmann et al .,1999

The floating drum: This one consists of a cylindrical or dome shaped digester and a moving, floating gas-holder, or drum. The gas-holder floats either directly in the fermenting slurry or in a separate water jacket. The drum moves up when gas is produced and sinks back when it is consumed. This type are most frequently used by small to middle-sized farms (digester size: 5-15m^3) or in institutions and larger agro-industrial estates (digester size: 20-100m^3) (Kossmann et al .,1999). Floating-drum plants are easy to install and operate. They have no issues of leaking gas, they provide gas at a constant pressure, and the stored gas-volume is immediately recognizable by the position of the drum. Figure 6-5 shows a typical plastic (polyethylene) floating drum digester on the Ugandan Local market produced by

CRESTANKS Limited Uganda (Aquasantech), it is said to have a life span of over 30 years and can be fixed on top of ground.

(a)



(b)

Figure 6-5: (a) demonstration of the feed of a poly ethylene (PE) plastic floating drum digester on the Ugandan local market (b) and during the operation the upper lid moves up as the gas is produced. The lid is heavy enough to put enough pressure on the gas as it flow out to be used for different purposes. The slurry can be collected in a plastic vessel and carried to sand drying beds. Source (Aquasantech).

Stirring the Bio-digester: While small scale digesters usually eliminate the option of stirring, it is desired that a digester is occasionally stirred during operation as optimum stirring substantially reduces the retention time and can increase gas production. A gentle daily stir on a daily basis is proposed from our study. String also helps to break up scum which could otherwise form if not stirred. This may make operation difficult as it could cause the floating drum to get stuck. The hardened scum could also form an impermeable layer which limits the gas from passing through (Kossmann et al .,1999).

Types of stirring facilities: simple manual stirring options are proposed and illustrated in Figure 6-6. The impeller stirrer and horizontal shaft, both of which originate from large scale plant practice are a suitable option especially in sewage treatment. For simple household plants, poking with a stick is the simplest and safest stirring method.

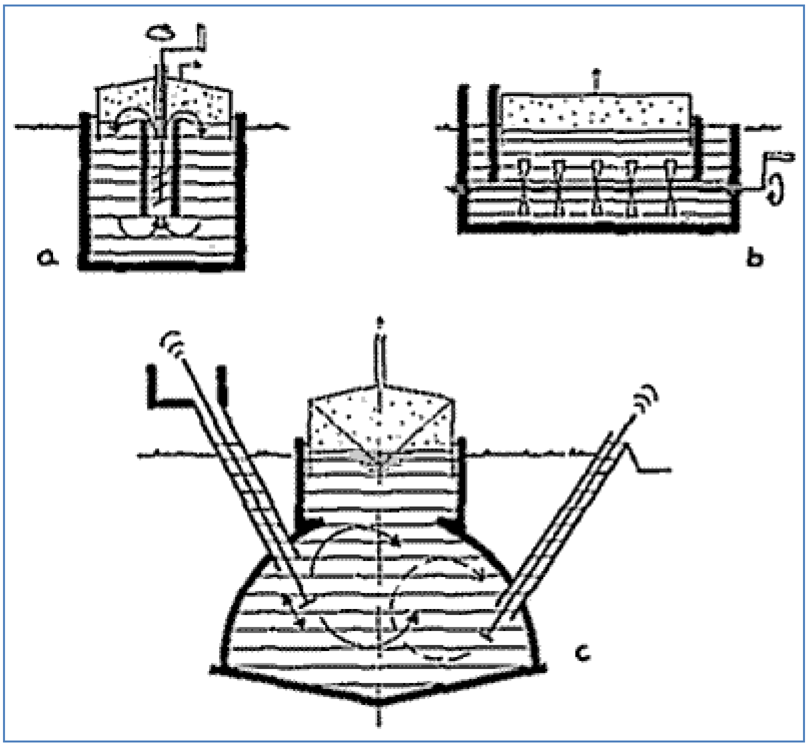


Figure 6-6: Stirring facilities in the digester, (a) The impeller stirrer, (B) the horizontal shaft and (C) poking with a stick Source: Kossmann et al .,1999

Materials: The common materials used in the construction of anaerobic digester units include; Plastics, steel, concrete and Masonry. These are continually used even in the developing world. Concrete is widely accepted acceptance especially for big scale digesters because of their unlimited useful life. Masonry on the other hand is the most frequent construction method for small scale digesters. Only well-burnt clay bricks, high quality, pre-cast concrete blocks or stone blocks are used in the construction of digesters. Cement-plastered masonry is a suitable – and inexpensive - approach for building an underground biogas digester, whereby a dome-like shape is recommended (Kossman et al., 1977). Plastic and Steel material is common for the floating drum. Steel drums are however expensive and prone to rust hence high maintenance required. Plastics or fibrous material have been introduced onto the market. These are cheaper and have a very good life span more than 20 years. Users however have to look out for the drum that could easily get stuck in the scum (Kossmann et al.,1999).

*Heating:*While known benefits are associated with achieving optimal mesophillic or thermophilic temperatures for anaerobic digestion, the high costs involved make it difficult to incorporate heating systems in the small-scale biogas plants. For the tropical countries like Uganda, the simplest way of heating is by exposing the site of the biogas plant to direct sunshine (Kossmann et al.,1999).

3.1.4 Collection and transportation of slurry:

Slurry from the digesters is proposed to be take to the drying beds before its converted into biochar. This can be collected over time in a plastic vessel and wheeled to the drying beds. This activity would require hire of a labourer to carry away the slurry at different intervals for the big institution. Alternatively a channel could be dug and overlaid with plastic sheet to lead the slurry to the drying beds (digging the channel and placing a plastic sheet can cost about 0.3 Euro per meter if the channel is less than 0.5 m deep. To reduce evaporation the channel can be covered by locally available material like wood and plastic sheet. The sludge drying bed should be positioned near the anaerobic digester for minimal cost.

3.1.5 Slurry drying and Biochar production

The sand drying beds are be built with a perforated concrete layer at the bottom which is filled with a small layer of sand to allow water in the sludge to drain out fast. They are one of the most commonly used technologies for sludge dewatering for low income countries (Tchobanoglous et al., 2003). This is because they have low capital and operational costs. However, the drying times are usually long with times of between 7 to 10 days required to achieve just about 20% dry matter in a dry tropical season (Strauss et al., 1997 and Cofie et al. 2006) and in the wet season it can go up to 50 days to achieve the same dry content. However, it is possible up to 90% DM in two weeks if the drying beds are covered (sheltered from the rain) and when the sludge is mixed frequently (Seck et al., in press). The bed could cost about 35 Euro/m^2 including construction and labour form the Ugandan market. The operational cost would ideally be cheap labour costs which can be covered by the owners or farmers. The seeping water can be led by a hand dug channel towards the gardens.

On drying, the sludge can be manually collected from the drying beds and transferred to the biochar unit which should be positioned near the beds. Biochar can be produced by use of traditional charcoal production methods. These are locally made from earth material or can be made out of old oil metallic containers. It can cost about 30 Euro to construct a 1 m^3 charcoal kiln.

3.2 Utilization of the end Products

Biochar:The sewage substrate in our study had high content of nutrients (phosphorous and nitrogen) which could still be beneficial to boost crop production when biochar is applied. During, anaerobic digestion all plant nutrients such as nitrogen, phosphorous, potassium and magnesium, as well as the trace elements essential to plant growth, are preserved in the substrate. The farmers can therefore apply the biochar directly after production, but they can also easily package and market it to create extra income in case it is produced in bulk. It is important however, that further studies are done to establish the fertilizer characteristics of the biochar produced in this study.

Biogas utilities:Biogas can be used in different ways, the common ones among them being; biogas lamp for lighting, Biogas stoves for cooking, and through an engine convertor to convert the energy to usable electricity. For use of biogas as it is, it can be transferred by use of rubber tubing to supply the different homesteads. While the engine produced electricity can be used to charge batteries, which are then used by residents to provide in house direct current.

The combined heat-energy generator: The most efficient way of using biogas is in a heat-power co-generation unit where 88% efficiency can be reached. But this is only valid for larger installations and under the condition that the exhaust heat is used. The generator can convert energy to usable power that can be fed into the normal electrical grid and used for a number of electrical needs.

Biogas Stoves or burners: The use of biogas in stoves is the best way of exploiting biogas energy from farm households in developing countries. The main prerequisite of utilizing biogas in the developing world is availability of a specially designed biogas burner. Many of these are now available and can be easily got from companies that are promoting biogas production techniques. However the relatively large differences in gas quality from different plants must be given due consideration. The stove has a commendable efficiency of about 55%, second to the heat-power combination.

Biogas /diesel engine: Biogas engines can be used if electricity is needed for other needs other than light and cooking. With the diesel electicity can be produced used for purposes such as refrigeration and battery charging. The option however also has a low enegy efficiency of 24%.

Biogas Lamps: Lighting is a basic need especially for places without electricity, hence the need to promote biogas lamps for such communities. The lamps (Fig 6-7) however have only about 3% energy-efficiency meaning most of the energy is lost as they usually get very hot. Biogas lamps are controlled by adjusting the supply of gas and primary air. The aim is to make the gas mantle burn with uniform brightness. The light output (luminous flux) is measured in lumen (lm). The luminous efficiency of biogas lamps ranges from 1.2 to 2 lm/W. By comparison, the overall efficiency of a light bulb comes to 3-5 lm/W, and that of a fluorescent lamp ranges from 10 to 15 lm/W.

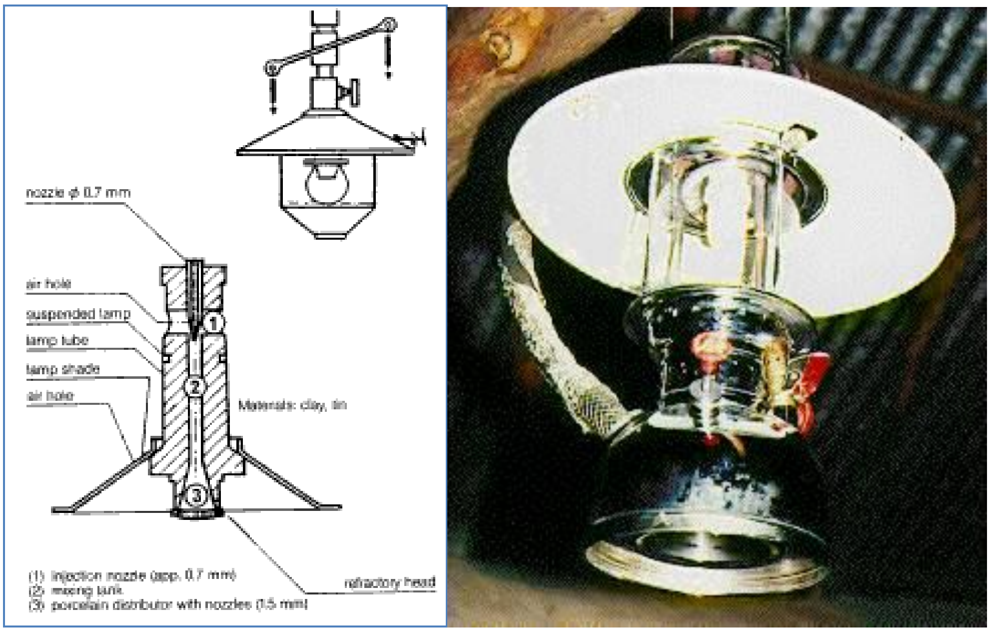


Figure 6-7: (a)Schematic structure of a biogas lamp and (b) a picture of a biogas Lamp in use.

In general, for the utilization of biogas, the following consumption rates in litres per hour (L/h) can be Assumed (Kossmann et al., 1999):

- household burners: 200-450 L/h
- industrial burners: 1000-3000 L/h
- refrigerator (100 l) depending on outside temperature: 30-75 L/h
- gas lamp, equiv. to 60 W bulb: 120-150 L/h
- biogas / diesel engine per bhp: 420 L/h
- generation of 1 kWh of electricity with biogas/diesel mixture: 700 L/h
- plastics molding press (15 g, 100 units) with biogas/diesel mixture: 140 l/h

3.3 Economic feasibility of the proposed system in the developing world

The typical system proposed in our study is decentralised consisting of a HRAS systems to enhance pre-concentration of solids, a charcoal filter to further polish the effluent from the HRAS to standards suitable for crop irrigation; an anaerobic digester to recover energy from the sludge from the HRAS and a local kiln to pyrolyse the HRAS digestate to form biochar which could be used as a soil improver. Two settings are proposed; the peri-urban setting with a pour flushing system and small bores and the rural setting with a modified pit latrine as they don't have flushing toilets. To estimate the cost of the entire proposed treatment system, the assumptions already made in chapter 4 are sustained. I.e. estimation is made for a community of 10 households, each with 5 inhabitants, where local materials like plastic tanks were considered where applicable. This puts the annualised cost of the peri-urban setting at 20.7 € Capita^{-1} year^{-1} including the sewer system and the rural setting at 13.8 including the pit latrine (Table 6.1). These costs are lower compared to other common technologies. For example a small scale conventional activated sludge system (CAS) could cost up to 24 € Capita^{-1} year^{-1} (Zessner *et al*., 2010). Then some of the preferred centralised system in Africa such as the waste stabilization pond (WSP) and the horizontal subsurface flow constructed wetland (HSSF-CW) can cost about 13 and 14 € Capita^{-1} year^{-1} respectively (Mburu *et al*., 2013). But very important to note is that this is without the sewerage network, moreover such centralised systems will require the normal sewer lines whose cost is over 17.1 € Capita^{-1} year^{-1} (based on the a capital cost of 105 € Capita^{-1} year^{-1} for Africa (WHO & UNICEF, 2000)). Ultimately these systems would cost over 41, 30 and 31€ Capita^{-1} year^{-1} for the CAS, WSP and HSSF-CW respectively.

Table 6-1: Cost Calculations for the different parts of the proposed treatment system

	Peri-urban	Cost (Euro)	Rural	Cost (Euro)
Domestic waste collection system				
Capital cost (Capita^{-1} year^{-1})	a	45	b	33
Operational cost (Capita^{-1} year^{-1})	0	0	0	0
Annualizedcost (Capita^{-1} year^{-1})		7.3		5.3
HRAS+ACF				
Capital cost (Capita^{-1} year^{-1})	Ch 4	7.8	n.a	0
Operational cost (Capita^{-1} year^{-1})	Ch 4	3.6	n.a	0
Annualizedcost (Capita^{-1} year^{-1})	Ch 4	4.9	n.a	0
Anaerobic digestion				
Capital cost (Capita^{-1} year^{-1})	c	6	c	6
Operational cost (Capita^{-1} year^{-1})	d	5	d	5
Annualizedcost (Capita^{-1} year^{-1})		6		6
Sludge drying				
Capital cost (Capita^{-1} year^{-1})	e	14	e	14
Operational cost (Capita^{-1} year^{-1})	f	0	f	0
Annualizedcost (Capita^{-1} year^{-1})		2.3		2.3
Biochar unit				
Capital cost (Capita^{-1} year^{-1})	g	1.2	g	1.2
Operational cost (Capita^{-1} year^{-1})	h	0	h	0
Annualizedcost (Capita^{-1} year^{-1})		0.2		0.2
Total Annualised cost		20.7		13.8

a- Small bore sewers cab be estimated at 44 Euro/per Capita (WHO & UNICEF, 2000)

b- Modified pit latrine can be estimated at 33 Euro per Capita similar to a pit latrine (WHO & UNICEF, 2000)

c- Total cost for a biogas plant, including all essential installations and accessories for utilizing it, but not including land, is between 50-75 US Dollar per m^3 capacity, 35-40% is this cost is the digester alone (GTZ) . From 100 L, an estimate of 30L of sludge would be produced per day. If other substrates are combined it estimated that about 60L of waste can be fed per day. A simple rule for temperature of 25°C, is to construct the digester size to be 120 fold the feed it gets. (GTZ & ISAT), hence the required size is 7 m^3 for the clustered setting proposed. Capital cost therefore is 307 Euro. (6 Euro/ Capita)

d- Running costs including unskilled labour to feed and operate the digester and repairs can be up to 250 Euro year (5 Euro/Capita) for the Ugandan market e- The drying bed could cost about 35 Euro/m^2 including construction and labour on the Ugandan market. A 20 square meter bed is sufficient for the proposed community; it will have a capital cost of 14 Euro per Capita

f- Labour may be required to carry the slurry from the digesters, one person should be hired to work on the entire system including digesters and biochar hence cost is already covered (e and Ch.4).

g- A local kiln in Uganda could cost about 30 Euro per cubic meter to construct. A kiln of 2 cubic meters is sufficient and would cost about 60 Euro (1.2 Euro per Capita)

h- Labour is required for operation of the Kiln, cost are included (e and Ch.4).

Ch. 4; Costs extracted from chapter 4

Note: To calculate the annualised cost, a life span of 10 years was considered and a real interest rate of 10%.

It is also, important to note that the total estimated cost of the system we have proposed is before considering the benefits that would arise from recovering biogas as well as biochar utilization. The benefits attached to the proposed system range from direct and indirect monetary benefits to other benefits such as waste management and pollution control. Monetary benefits can be calculated based on expenditure the Individual households could save on items like; 1). Energy by utilizing biogas instead of charcoal or electricity or other types of fuel, 2). Use of biochar as a soil improver and 3).Time saved for collecting and preparing the earlier fuel sources e.g. wood if applicable.

With regard to Waste management and pollution control, the system provides a profitable way of disposing waste which would be more acceptable to individuals. Further more the better management of farm wastes and other organic wastes, ultimately contributes to decreased nutrient loads that would otherwise end up in the fresh water sources. Ultimately, due to the mentioned benefits, the system here proposed provides an affordable safe sanitation option that could easily be gradually adopted by the poor communities in the developing world. This would lead to increased sanitation coverage in the developing regions which would ultimately contribute to economic development since as discussed in chapter one sanitation links to economic development. The contribution to economic development would include; redeemed man hours as less people fall sick, decreased expenditure on sanitation related diseases and deaths and redeemed time for accessing proper sanitation facilities.

4. Further research needs

4.1 Limitations and opportunities of biochar production

The benefits of biochar with regard to carbon storage and increased agricultural yields have been highlighted by many researchers (Lehmann 2007; Mathews 2008, Sohi 2013). Biochar can improve soil productivity by adapting the pH and increasing nutrient retention as well as improving the soil water retention ability. It can even help with the remediation of degraded soils as well carbon sequestration. However there is need for further research in order to establish the fertilizer properties of this biochar before it can be marketed on big scale as a fertilizer. Wastewater sludge is also known to accumulate heavy metals, studies are necessary to rule out heavy metal availabity to plants in case biochar is applied for agriculture.

Biochar production is still expensive although it was found to be economical for cereal crops in the sub Saharan Africa where biochar production is simulated through the traditional labour intensive charcoal pit production (Dickson et al., 2014). However, the economic benefit of the technology from an agricultural perspective is still low for short term agronomic application for the advanced technologies (McCarl et al. 2009). In general, the developed world approach for biochar production is expensive while the traditional charcoal production methods simulated for the developing world are cheaper but labour intensive. The Sub Saharan approach had low labour prices giving rise to biochar costs of about 99 to 165 USD t^{-1} Compared to 155 to 259 USD t^{-1} for the advanced pyrolysis technologies in the developed world like the North Western Europe. That may explain why the technology has not been widely adopted despite the obvious benefits and the wide attention it has already received. Moving forward, there is need for development of both simple and cheaper pyrolysis techniques that can easily be adopted even by the developing world. A traditional charcoal kiln (Figure 6-8a) is made by piling up the wood in a pit (pit kiln), and a covering with a layer of e.g. soil or bricks to keep O_2 from entering. Apart from being labour intensive it has low charcoal yields due to its poor insulation. Also the char formation is not uniform, with some being only partially pyrolysed due to non-uniform air flow. Improvements are required to achieve an easily usable system which is capable of producing high yields. An improved charcoal kiln with a chimney has been adapted in some areas (Figure 6-8b). The improvised chimney improves air flow which increases the yield; it however allows release of carbon oxides (CO) and volatile organic compounds (VOCs) to the atmosphere. This could be minimised by allowing continuous burning at the vent. The improvised chimney betters air flow and increases the yield. It is estimated that for an economically feasible use of biochar for agricultural purposes, the cost of biochar needs to come down to 12 USD t^{-1} (Galinato et al., 2011).

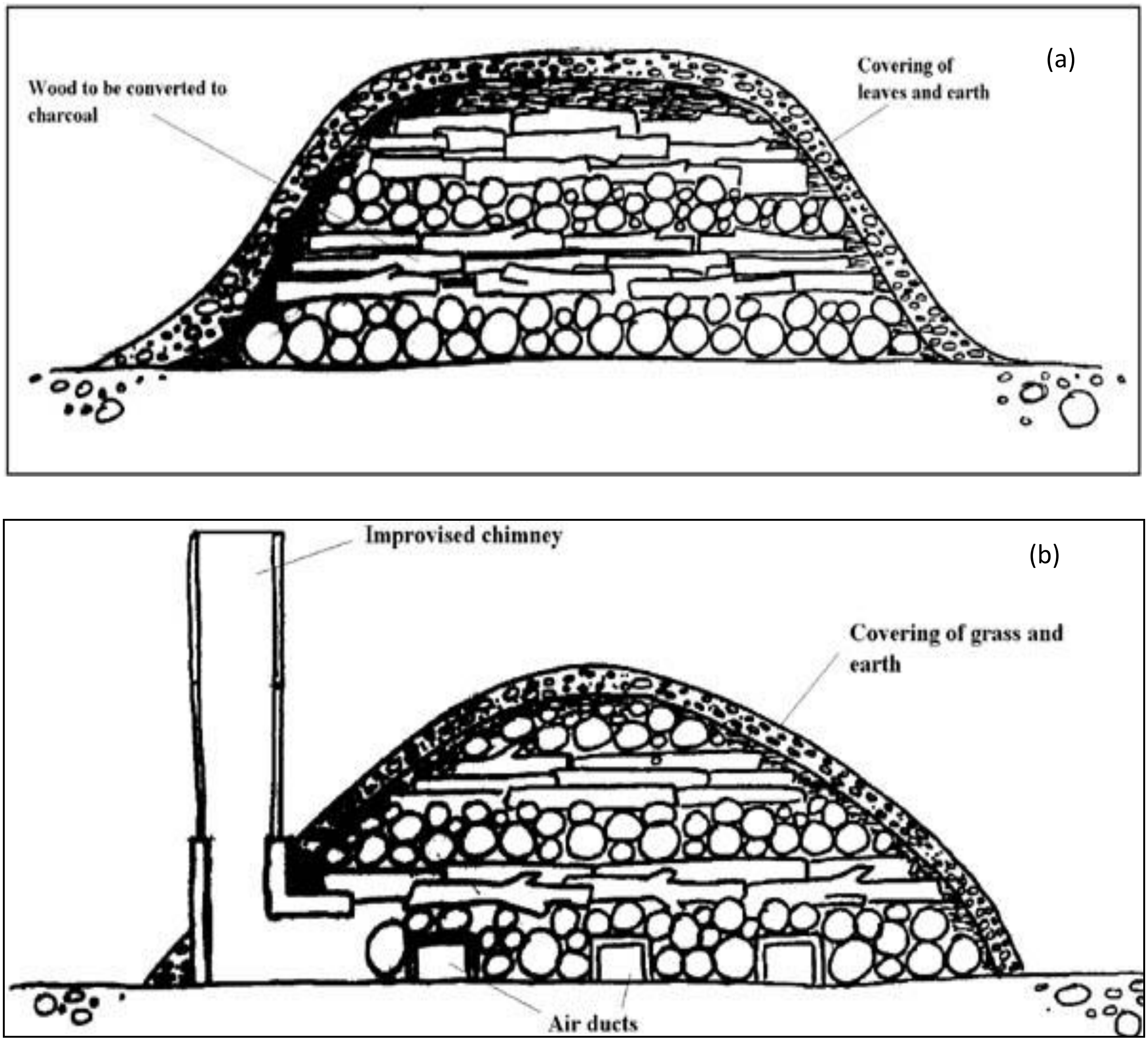


Figure6-8: (a) A traditional mound kiln used to produce charcoal, (b) An improved version of the traditional kiln Source: http://en.howtopedia.org/wiki/Biomass_(Technical_Brief).

Also, it is important to note that while pyrolysis requires some major energy input, the majority of that energy is needed to bring the biomass from room temperature to pyrolysis temperature, the actual slow pyrolysis or carbonization reaction is exothermic in nature (Mok

& Antal, 1983). In most slow pyrolysis/carbonization systems, what remains of the biomass heating value will be embedded in the non-condensable gases and vapours. For example for the pyrolysis at the optimal HHT of 600°C, 34 % of the biomas energy ended up in the gasses; These gases and vapours can be combusted and the hot flue gases recycled to provide the heat for heating the biomass whereby these processes become auto-thermal overall. In practice, one can have continuous systems or multiple batch reactor systems - in which each reactor is at a different stage of the pyrolysis cycle, allowing for proper heat integration. This would make tremendous contribution to lowering of costs, especially for the advanced technologies.

4.2 The potential of the combination of the HRAS plus the ACF system for a decentralized domestic wastewater treatment system for an agricultural community.

The HRAS +ACF system concept has been proved in this study to be potentially viable to produce effluent fit for use for agricultural purposes. There is however still need to improve the microbiological quality of the effluent which fell below required standards on certain occasions. Therefore, further studies could consider optimising the system e.g. by increasing the charcoal filter columns in series to a level where 100% faecal colifom removal can be achieved. Also, the charcoal from the filter operation if not properly handle can pose a solid waste nuisance, yet from observation, the charcoal was noted to have accumulated a number of organics onto its surface. Moreover charcoal and biochar are both carbon rich products formed in a similar way, by heating the biomass in an oxygen free or limited environment. The difference between the two is that the former is mostly appreciated as a fuel source, while the latter as a soil improver. It is therefore suggested that the used charcoal could be further crushed and applied in the soil as biochar, to increase solid production. Further research could look at the impact of use of the charcoal as a soil improver in comparison to other biochars. Further research could also look to rule out the possibility of accumulation of pharmaceutical products and heavy metal on the charcoal.

4.3 Sensitisation to change people perception

The concept of resource recovery from wastewater discussed in this study has potential to give rise to an affordable sanitation option, which would make a positive contribution towards achieving the millennium development goal of providing sanitation for all even in the developing world. The concept could however easily be rejected by people due to peoples' cultural taboos, beliefs and sometimes simply out of ignorance. Sensitization of the communities is therefore paramount to enable the Africa population to appreciate the resource recovery concept and consequently derive the expected benefits therein.

5. Conclusions

This work has explored different concept and applied them in the developing world setting for recovery of resources from wastewater treatment, while achieving cost-effective and sustainable wastewater treatment in a decentralised setting. The research has made contribution by; (i) proposing a complete decentralized domestic wastewater treatment system that optimises recovery of energy, water and nutrients from wastewater. The proposed system combines a number of technologies such as anaerobic digestion, high rate activated sludge and pyrolysis which together also ultimately lead to minimal waste generation (ii) showing that poly-aluminium drinking water treatment sludge is a valuable product that can be re-used to improve wastewater treatment (iii) showing that co-digestion of primary sludge with brewery and cow dung in the ratios of 50:25:25 respectively provided optimal anaerobic conditions and increased biogas production to volumes more than two times than when primary sludge was digested alone (iv) proving a concept in which new nutrient rich water fit for re-use for agricultural purposes is produced through treatment by a combination of the HRAS and the ACF system (iv) Showing that good quality biochar could be formed from HRAS digestate. The biochar formed is more stable compared to the dried sludge. In addition to the technology development it will require public sensitization especially in the developing world to break the barriers that may inhibit adoption of such workable solutions.

Abstract

Whilst there is significant improvement in access to sanitation globally, access to proper sanitation is still a great challenge in the developing world, especially the Sub Sahara Africa where 25% of the population still practiced open deification by 2012. The current sanitation systems have loop-holes and can barely help the situation. Wastewater is rich in a number of resources, which include water, energy and nutrients. These when rightly explored through recovery, present an opportunity to subsidise sanitation costs hence making it more affordable and consequently accessible for all. A paradigm shift from the ordinary centralised and onsite systems to a cluster decentralised systems which encourage resource recovery is pertinent to achieve more cost effective and manageable wastewater treatment system.

This work sought to explore interventions for resource recovery that are appropriate for application in the developing world. In **Chapter 1**, a review of literature was done to understand the current situation in Africa, after which, a wastewater management plan that could contribute to improvement for small agricultural communities was suggested. The plan encourages zero waste generation through decentralisation and recovery of water, energy and by-products such as nutrients and organics relevant to the local community. The subsequent chapters therefore details studies in which resources from wastewater could be recovered as new water, energy and nutrients/fertilizers or simply re-used to achieve better treatment.

The work considers not only domestic wastewater but also integrates wastes from other sources to bring about better wastewater treatment performance as well as increase resource recovery benefits. **Chapter 2** of this work explores, the re-use of polyaluminium drinking water treatment sludge (PA-WTS) as a flocculant aid to improve the effluent quality of wastewater during primary sedimentation. The results obtained showed a tremendous decrease in total suspended solids (TSS), chemical oxygen demand (COD), total ammonium nitrogen (TAN), and total phosphates (TP) in the supernatant after 30 min of settlement. The optimal PA-WTS dosage of 37.5 mL/L significantly ($P<0.05$) increased the TSS, TP and COD removal efficiencies by 15, 22 and 30%, respectively. It can be concluded PA-WTS therefore positively complimented the sedimentation process in the primary treatment of wastewater to achieve better effluent quality.

Among the many resources in wastewater and other wastes is the energy which can be recovered through biogas production. **Chapter 3** presents a study where two organic wastes, cow dung and brewery sludge were co-digested with primary sludge in different proportions.

The aim was to enhance biogas production from municipal sewage sludge. Brewery waste was found to increase the biogas production rate by a factor of ≥ 3. This was significantly ($p<0.001$) higher than that observed (336 ± 18 mL/L.d) in the control treatment containing only STP sludge. Co-digestion with brewery-waste and cow-dung improved biodegradability of municipal sewage sludge and is recommended. Apart from the increased energy recovery, digestion of other wastes with sewage sludge would also lead to cleaner cities as waste is better managed and would ultimately cut down cost of both sewage and solid waste management.

Chapter 4 explores the re-use of water for agricultural purposes with a two treatment systems; a high rate activated sludge (HRAS) system and alternating charcoal filters (ACF). Two systems were in parallel with the ACF line after the HRAS. The HRAS effectively removed up to 65% of total suspended solids (TSS) and 59% of chemical oxygen demand (COD), while ACF1 removed up to 70% TSS and 58% COD. The combined treatment system of HRAS and ACF effectively decreased TSS and COD on average by 89% and 83% respectively. Total ammonium nitrogen (TAN) and total phosphates (TP) were largely retained in the effluent with removal percentages of on average 19.5% and 27.5% respectively, encouraging reuse for plant growth. The charcoal can upon saturation be dried and used as fuel. This provides a cheap way for developing countries to counter the challenge of climate change especially in regard to water scarcity.

Another possible way of recovering nutrients and energy from wastewater treatment is by converting the sludge to biochar. In **Chapter 5** biochar formed from high rate activated sludge (HRAS) was characterised with respect to its use as a soild improver and energy. HRAS was first anaerobically digested under mesophilic conditions at a sludge retention time of 20 days. The results showed that HRAS digested well producing on average 0.5 ± 0.15 CH_4 $L^{-1}L^{-1}d$ for an average OLR of 1.85 ± 0.63 g $COD^{-1}L^{-1}d$. The produced biochar showed optimal properties as a soil improver when produced at a temperature of 400°C with values of 18.11 wt%, 21.32 wt%, 60.58 % and 0.41 for volatile matter, fixed carbon, ash content and H/C ratio, respectively. With regard to energy, the biochar had a lower caloric value than the dried HRAS digestate. Based on these findings, it can be concluded that anaerobic digestion of HRAS and its subsequent biochar formation at HHT of 400°C presents a sustainable management option for sludge in tropic settings like in Uganda.

Based on the results presented in the chapters of this work, future research needs are proposed in **chapter 6**. Among them, the need for cheaper and user friendly pyrolysis techniques that can make biochar formation sustainable in the developing world. A number of local technologies for charcoal making are already in existence and could be adopted, with optimisation aimed at increased efficiency in biochar production. With regard to the HRAS +ACF system, further studies could be dedicated to optimising the system in order to achieve complete removal of faecal coliform. Also, the used charcoal from that system could easily turn into a waste nuisance it not well managed. Yet, charcoal behaves similarly to biochar, moreover, this one also had organics adsorbed on the surface, which may increase its potential to act as a soil improver when crushed and applied to soils. Further research therefore, could consider establishing the impact of applying the crushed charcoal to soil as some form of biochar.

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Professional activities

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Oversees water and wastewater treatment and coordinates quality assurance of water supplied and wastewater discharged, within 118 towns in Uganda where NWSC operates.

2006 –2010 **Principal Engineer at the Sewerage Services department, National Water and Sewerage Corporation (NWSC), Uganda.**
I was in charge of treatment and disposal of wastewater at the five wastewater treatment plants in Kampala, Uganda under NWSC. I also co-ordinated training and research activities in water and sanitation at the department.

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Give lectures in water and environment related courses (Public health and Environmental Engineering, Engineering Hydrology and hydraulics, Sanitation Engineering, Water supply and Water and wastewater Treatment and monitoring) for student's up to under graduate level. Coordinate research in water and environment related activities at the Department of civil and Building Engineering including the supervision of students' research and projects.

Scientific contributions

A1 publications:

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Boundary River: The Case of the Akagera River." International Journal of Mathematics & Computation™ 9, no. D10 : 26-45.

Acknowledgement

When I started this journey, it seemed like a mountain too high to climb, and to get a better feel of it, I never missed the chance to attend a PhD final defense when it came my way. Seeing many finish well, kept me inspired and hopeful and finally made me agree with Barry Finlay that; "Every mountain top is within reach if you just keep climbing" I am very indebted to many people, who supported me along this journey, they are the reason I smile today as I reach this mountain top.

First and foremost I am grateful to my supervisors, Prof. Willy Verstraete, Prof. Korneel Rabaey and Prof. Noble Banadda, for the incredible support they gave me throughout the course of my PhD. My special thanks to Willy, who availed me the opportunity to join the prestigious LabMET research team, when I contacted him initially. The tips and ideas you provided as we explored the sphere of this research as well as the continued support throughout my PhD study is exceptional and will never be taken for granted. To Korneel, you took me on as your student mid way when you joined the department. Thank you for accepting to take me on, your guidance and knowledge have moulded me into a critical thinker. You believed I could achieve better than what I sometimes aimed at which gave me a lot of confidence to achieve higher. I also appreciate all the administrative arrangements you have helped me with, especially arranging resources for my final travel and stay in Belgium for my final defense. Noble, your continued guidance and discussions while back in Uganda is very much appreciated. If it were not for your support, the experiments back home may never have been completed. Thank you for believing in me and making the sandwich research less burdening than it usually is for many others. I also wish to thank Prof. Tom Vande Weille who supervised me, in the first two years of my PhD. You accepted to supervise me even when my line of study was not your first preference. I am very gratefully for the guidance you provided to shape my research into the wonder it turned out to be. And to Prof. Nico Boom and Prof. Siegfried Vlaeminck, thanks for your support at LabMET. I am also gratified to work under the supervision of Prof. Frederick Ronsse through whom I affiliated my work with the department of Biosystems Engineering. Your continued efforts to promptly react to all my queries were very humbling. Thank you for expounding my knowledge on biochar techniques.

Special thanks go to the teams that supported my research while in Belgium. I am very proud of the LabMET team who have been very inspirational in this journey of my life. The knowledge we shared during our work interactions and the smiles shared away from work cannot be exchanged for anything. Special thanks go to the all my officemates, Pieter VdA, Varvara, Keren, Jianyun, Ramiro, Carlos, Elham and Hannele. I was very fortunate to share the office with all of you, it was always a short stay at LabMET but very rewarding, because of you. Furthermore, i acknowledge other colleagues from Labmet my co-authors; Francis Meerburg, Jo-De Vrieze and all members from the AB cluster, Thanks for the academic collaborations and very useful discussions which enriched my study. I also thank Chris Callewaert for guidance in finalisation of my PhD. Further appreciation goes to Robert Nachenius, Dane Dickinson and Martínez Rodríguez from the department of Biosystems Engineering, the valuable discussions and your help with my biochar experiments greatly advanced my biochar knowledge. I also wish to acknowledge the administrative support that I received from Regine and Christine, your calmness and good spirit made you approachable at all times. Thank you for the tremendous support especially with the final arrangements before my defense given that I was mostly not in Belgium. I also appreciate the continued support from the LabMET technical team; Mike, Tim, Greet, Siska, and Renée. Special thanks to Tim for helping with my drawings and Mike for the help with equipment acquisition.

Prof. Filip Tack, thank you for chairing all my examination committee deliberations. Many thanks to Prof. Pascal BoeckxProf, Prof. Grietje Zeeman, Prof. Kevin van Geem and Dr. Steven De Meester for the time you spent to go through my manuscript and the suggestions that you gave which enriched my thesis.

Very special thanks go to the team in Uganda that supported my research. I am very appreciative to the Board of directors and entire management of NWSC for providing an environment that enhances research and innovation. Thank you for availing NWSC facilities for my research as well as further financial support while back in Uganda. Special thanks go to managing director of NWSC Dr. Eng. Silver Mugisha for supporting this research opportunity and for your guidance. You inspire many of many of us to achieve excellence.

I am also indebted to the Team from the Directorate of Business and scientific Services in NWSC- especially the Director, Dr Rose Kaggwa and Research Manager, Dr. Babu Mohammed for the support with my research while in Uganda. Special thanks go to the team from Sewerage Services Department-NWSC, especially to James Maiteki, Angelo K, Richard S, Chaba Charles and Robert Smith lubega who supported the field work and experimental set-ups in Uganda. I also thank the NWSC-Central Laboratory team especially, Christopher Kanyesigye, Robinah Muhairwe, Profilio Tebandeke, Juliet Nakanjako, Constance Kaggwa, George Kyeswa and Ritah Kamiti who supported my Laboratory analysis.

I acknowledge the financial support from VLIR-OUS, This research would never have been possible without your support. Special thanks to Micheline D'hoodge for organising my travels and stay in Belgium and ensuring that the facilitation throughout the time of my research comes on time.

Thanks also to Sylvia Nabatesa and Cyrus Galyaki, the students I supervised, who ended up making valuable contribution to this work. I wish you the very best in life.

To my dear family members, I can't ask for more, I am so blessed to have you all in my life, you really make life worth living. Dr. & Mrs. Nsubuga Mutaka thanks for always believing in me and allowing me flourish. Lonah and Michael Gaukroger, thanks for making my European stays more homely with your visits and my visits, Esther and Pheona thanks for the love and filling in for me when I was away. Davis, Abbey, Brenda and Edward thanks for the un-ending calls and keeping me encouraged. To the little ones Mykylah, Aiden, Caelan and Sian you are such a big blessings. I am proud of you all, The 'NSUS' really Rock!!. Ethan Mugabi, thanks for believing in me and for your continuous support and encouragement. And to my friends Susan Muyindie, Christine L, Linda-Medrina, and Halima A, Emmanuel and Susan O, to mention but a few. Thanks for the support; you gave me courage to continue pursuing this study, I am glad to have friends like you in my life.

Am also grateful to the loving International community church family that made my Sundays in Belgium, a big blessing. Special thanks to Leo and his family, for the very insightful discussions and in whom I found great friends, and to the Smith Family; Anne and Albert who hosted me in my first year of my PhD study.

Most importantly, I give praise to the Almighty God who guides my life and without whom I would be nothing. I praise him for enabling me finalise this journey and trust his continued guidance to even greater heights. !! Mukama Mulungi Obudde Bwona !!.