Improving faecal sludge dewatering efficiency of unplanted drying bed

By

Richard Amankwah Kuffour (MSc.)

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CERTIFICATION

I, Richard Amankwah Kuffour hereby declare that this submission is my own work towards the PhD and that, to the best of my knowledge, it contains no material previously published by another person nor material which has been accepted for the award of any other degree of the university, except where due acknowledgement has been made in the text,

Richard Amankwah Kuffour	----------	----------
(20034866)	Signature	Date

Certified by:

Prof. Mrs. Esi Awuah	----------	----------
(Principal Supervisor)	Signature	Date
Dr F. O. K. Anyemedu	----------	----------
(Second Supervisor)	Signature	Date

Certified by:

Prof. M. Salifu	----------	----------
Head of Department	Signature	Date

DEDICATION

I dedicate this work to the Lord Almighty God.

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ABSTRACT

This thesis examines how to improve on faecal sludge dewatering efficiency of unplanted filter beds to produce biosolids and less concentrated filtrate. The experiment was conducted on bench scale drying beds constructed at the Kwame Nkrumah University of Science and Technology campus. Local sand was graded into particle size ranges of 0.1- 0.5 mm, 0.5 - 1.0 mm and 1.0 - 1.7 mm, with uniformity coefficients of 2.422, 1.727 and 2.029, respectively. They were represented as filter media, FM1, FM2 and FM3 respectively. Faecal sludge, consisting of public toilet sludge and septage collected from cesspit emptiers discharging at Dompoase treatment ponds in Kumasi, Ghana, were mixed in the ratio of 1:1 by volume, and was dewatered using the three different filter media, FM1, FM2 and FM3 to determine which one was most efficient. The Solid Loading Rate (SLR) of the faecal sludge was varied in the phase two whereby public toilet sludge and septage, mixed in the ratio of 1:1, 1:2 and 1:3 by volume representing SLR1, SLR2 and SLR3 respectively. These were dewatered on filter medium one (FM1) which was selected in phase one. Six cycles of dewatering were run for each of the phases. Percolate volume was measured every 24 hours. The total solids (TS) of the faecal sludge used for dewatering varied in every cycle so the TS was kept constant at 36.64 g/l as SLR1 and 26.93 g/l as SLR3 in all the cycles in the phase three to determine the effect of constant TS on the dewatering process. Further improvement of the dewatering was investigated by mixing sawdust with faecal sludge in phase four. Different percentages of sawdust, 50%, 100%, 150% and 0% (control) by weight of the TS of the faecal sludge, with TS of 26.93 g/l which was selected in phase two, were mixed with the faecal sludge to determine the effect of different quantities of sawdust as physical conditioner on the

dewatering of faecal sludge. The dried biosolids obtained from the different phases of dewatering were analysed for plant nutrients and heavy metals to determine their agricultural potential. The dewatering on different filter media, FM1, FM2, and FM3 showed average dewatering times of 9.8, 9.9 and 9.1 days respectively without significant differences (p=0.212). However the percolate quality showed significant differences between the different filter media in the removal of TS, TVS, SS, COD, DCOD and NH_3-N with FM1 having the highest removal efficiency for each parameter. Accumulation of organic matter in the top 10 cm of the filter bed indicated that FM1 was least likely to clog since it had the least quantity of organic matter in the sand. It also produced the largest quantity of organic matter and thus had the potential to generate the most biosolids. The faecal sludge of SLR1, SLR2 and SLR3 dewatered significantly at different average dewatering times of 7.2, 4.8 and 3.8 days respectively. Removal efficiencies at the different solid loading rates, though very high for TS, SS, TVS, COD, DCOD, NH_3-N, did not show any significant difference. Organic matter build up in the top 10 cm of the filter bed was least in SLR3, hence least likely to clog the filter bed. Again, SLR3, SLR2 and SLR1 showed the potential for annual generation of biosolids at 438, 421 and 379 (kg/m^2/year) respectively. Therefore SLR3 of faecal sludge was recommended for dewatering on the selected filter bed.

When TS of the faecal sludge for dewatering was kept constant and the number of cycles was increased to eight, FM1, FM2 and FM3, dewatering FS of SLR1 at constant TS of 36.64 g/l improved their dewatering time to 8.9, 8.8 and 8.7 days respectively while SLR3 with constant TS of 26.93 g/l, dewatering on FM1 had the shortest dewatering time of 4.4 days. SLR3 was significantly most efficient in removing TS, SS, TVS, COD and EC. Organic matter accumulation rate in the top

10cm of filter bed was least for SLR3 followed by FM1, FM2 and FM3. The percentage organic matter of the biosolid were 69, 68, 62 and 59, leading to estimation of annual organic matter production of 334, 193, 202 and 177 kg TVS/m^2/year for SLR3, FM1, FM2 and FM3 respectively.

The sawdust-faecal sludge mixture of 50%, 100%, 150% and 0% TS of faecal sludge dewatered at 5.3, 4.9, 3.9 and 5.6 days respectively with 150% being fastest to dewater. The 150% sawdust-faecal sludge treatment was most efficient with respect to removal of contaminant loads like TVS, SS, COD and NH_3-N, but was least in TS and EC removal. The 150% sawdust-faecal sludge mixture showed the least organic matter accumulation rate in the top 10cm of the filter media, followed by the 100%, 50% and the 0% treatments. The percentage TVS of the biosolids produced were 70.1, 77.3, 80.6 and 66.3 for 50%, 100%, 150% and 0% sawdust-faecal sludge mixtures, leading to annual organic matter estimation of 359, 567, 987 and 225 (kg TVS/m^2/year) respectively.

The agricultural potential of the biosolids analysed showed that, dried biosolids from the filter beds had percentage carbon (C) ranging between 28% and 43.5% while percentage nitrogen (N) in the same samples ranged between 1.82% and 3.53% leading to C/N ratio, ranging between 8.7 and 23.9. The percentage phosphorus (P) in the same samples ranged between 1.73% and 3.69% while the percentage potassium (K) values were within the range of 0.66% and 1.67%. The maximum concentration of heavy metals recorded in the dried biosolids were 0.225, 4.38, 0.024, 0.89, 0.55 and 0.208 mg/kg for Cu, Fe, Pb, Cd, Zn and Mn, respectively which were all far below the respective standards permissible in biosolids worth for use in agriculture.

Summary of criteria needed to improve dewatering efficiency of faecal sludge using unplanted filter beds

The faecal sludge generated in most cities in Ghana consists of mostly septage and public toilet sludge which contain high contaminant load. This study has shown that it is possible to dewater it using unplanted sand filter beds. Sand particle sizes ranging between (0.1 - 0.5, 0.5 - 1.0 and 1.0 - 1.7) mm with uniformity coefficients of 2.422, 1.727 and 2.029, respectively are effective, with particle size of (0.1 - 0.5) mm being most effective. Using these particle size ranges, faecal sludge of 217 – 360 kgTS/m^2 could be dewatered on the filter bed in one year which is equivalent to faecal sludge of 22 – 27 persons. Public toilet sludge (PTS) and septage ratio of 1:3 by volume is most effective in improving the dewatering time as well as generating largest volume of biosolids though other ratios like 1:1 and 1:2 are also effective. These ratios have the capacity of reducing dewatering time per cycle from 9 - 12 days to 4 – 7 days. This can dewater faecal sludge of between 379 – 532 kgTS/m^2 of filter bed. Addition of sawdust of between 50% - 150% TS of faecal sludge by weight reduced the dewatering time to less than 4 days. Addition of sawdust of 150% TS is most effective in reducing the dewatering time and contaminant load removal. About 575 kg TS/m^2 of faecal sludge can be dewatered on a filter bed of particle size range of 0.1 - 0.5 mm and a solid loading rate of 1:3 for PTS and septage in a year if sawdust of 150% TS of the faecal sludge is added. This generates almost three times the quantity of biosolids that could have been generated with very good organic bulking quality.

Key words: Faecal sludge, filter medium, dewatering time, removal efficiency.

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ABREVIATIONS

AIT	Asian Institute of Technology
C/N	Carbon Nitrogen ratio
COD	Chemical Oxygen Demand
CSIR	Centre for Scientific and Industrial Research
DALY	Disability Adjusted Life Years
DCOD	Dissolved Chemical Oxygen Demand
DS	Dry Solids
EAWAG	Swiss Federal Institute for Environmental Science & Technology
FM	Filter medium
FS	Faecal Sludge
FSTP	Faecal Sludge Treatment Plant
HLR	Hydraulic Loading Rate
IWMI	International Water Management Institute
KMA	Kumasi Metropolitan Assembly
KNUST	Kwame Nkrumah University of Science and Technology
KVIP	Kumasi Ventilated Improved Pit
LSD	Least Significant Difference
OSS	Onsite Sanitation System
PTS	Public Toilet Sludge
SANDEC	Department of Water and Sanitation in Developing Countries
SLR	Solid Loading Rate
SD-FS	Sawdust-faecal sludge
SS	Suspended Solids
TIDD	Timber Industry Development Division
TKN	Total Kjeldahl Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus
TS	Total Solids
TSS	Total Suspended Solids
TVS	Total Volatile Solids
UASB	Upflow Anaerobic Sludge Blanket

UN	United Nations
UNICEF	United Nations International Children's Education Fund
USEPA	United States Environmental Protection Agency
VIP	Ventilated Improved Pit
WC	Water Closet
WHO	World Health Organization
WRI	Water Research Institute
WSSCC	Water Supply and Sanitation Collaborative Council
WWTP	Wastewater Treatment Plant

CHAPTER ONE

1 INTRODUCTION

In the humid zone of West Africa, more people live in urban than rural areas today. In 1920, only 4% of the population lived in the urban areas (United Nation population Division 2004). The upsurge in urban population in developing countries has led to an increased generation of faecal sludge in such communities. About 44% of Ghana's total population of 22 million people now live in urban areas (up from 32% in 1994) and this number will rise as some of the urban areas have annual growth rates of more than 6% (Ghana Statistical Service (GSS), 2002). A major cause of the rapidly increasing urban population is the influx of people from rural areas in search of better livelihoods.

It is estimated that one third of the world's population, approximately 2.4 billion urban dwellers rely on on-site sanitation system (OSS) installations such as public latrines, aqua privies and septic tanks (Kootatep *et al.*, 2004). Only smaller portions of cities central business districts are linked to sewers (Strauss *et al.*, 2000). This situation is likely to last for decades to come since citywide sewered sanitation is neither affordable nor feasible for the majority of urban areas in developing countries (Koné and Strauss 2004). These OSS serve about 65-100% of urban dwellers in cities of Africa, Asia and Latin America (Strauss *et al.*, 2000). In Ghana, 85% of urban households are served by OSS (Montangero and Strauss, 2000) and that of Kumasi, the second largest city in Ghana, is 90% (Cofie, 2003). Thousands of tons of sludge from on-site sanitation installations i.e. unsewered toilets, family and public toilets and septic tanks, so called faecal sludges are disposed of untreated and indiscriminately into lanes, drainage, ditches, unto open urban spaces and into inland

waters, estuaries and the sea in different parts of Ghana and many developing countries (Strauss and Montangero, 2001). In cities like Accra, Cotonou, Kumasi, Yaoundé, Dakar, more than 500 m^3 of faecal sludge are generated and dumped in the environment untreated daily. This amount of FS contains pollution equivalent of up to about 50,000 m^3 of wastewater as a result of its high concentration, which is estimated to be 10 – 100 times that of wastewater (Koné and Strauss, 2004). Excreta may contain pathogens that cause health hazards (Feacham *et al.*, 1983; Awuah, 2006) and therefore must be properly treated in any community because, lack of adequate treatment and disposal will lead to high morbidity and mortality (Awuah, 2006). According to WHO (2005), about five million people die annually, due to lack of adequate sanitation while about 3 billion people lack access to adequate sanitation. Majority of these people live in developing countries. In Ghana for instance, enteric diseases are second only to malaria in the number of cases reported in the nation's hospitals. About 400,000 cases of enteric diseases are reported in the country annually (Ministry of Health, Ghana 1990). Diarrhoeal disease alone account for an estimated loss of 62.5 million disability adjusted life years (DALYs) or 4.3% of the total world burden of disease. 88% of this was estimated to be due to unsafe water supply, sanitation and hygiene which is mostly found in developing countries (Ezzati *et al*., 2002; Pruss *et al*., 2002; WHO, 2002).

This makes the excreta disposal situation very dramatic in developing countries in Africa, Asia and Latin America. Therefore, human excreta which are sources of pathogens must be properly disposed of to alleviate the incidence of morbidity and mortality, especially of children under the age of 5 years.

The millennium development goals seek to decrease the number of people without access to adequate sanitation by 50% by the year 2015 (Cosgrove and Rijsberman, 2000). In order to meet these goals about 460,000 people worldwide must be provided with improved sanitation daily (WHO, 2005). To achieve worldwide coverage of water supply and sanitation in 2025, a worldwide investment of $200 billion per year (assuming that conventional technologies are used) is required. These cost estimates do not include full coverage of wastewater management which is estimated at $70 – 90/cap/year, thereby expected to raise the total worldwide annual investment requirement to about $600 - 800 billion (Bos *et al.*, 2005).

There is general lack of sustainable options for treating faecal sludge (FS). In many cities the faecal sludge from OSS are collected and disposed of untreated and indiscriminately into drainage ditches, inland waters and the sea, mainly due to lack of affordable treatment systems (Heinss *et al.*,1998). Using the figure of 0.001 m^3 FS/cap/day as an average FS generation rate in urban areas (Polprasert, 1996; Metcalf and Eddy, 2003), it is calculated that 1,000 m^3 and 1600 m^3 of FS should be collected and disposed of in the cities of 1 million and 1.6 million inhabitants like Kumasi and Accra, respectively. However, it is observed that the amount collected in Kumasi is half of what is expected while that of Accra is dramatically low (Keraita, *et al.*, 2002). In Ghana for instance, it is a few urban centres that have treatment plants to treat part of faecal sludge generated. These were constructed in the 1980 and 90's and are currently facing maintenance and technical problems. Recently built facilities are equally facing maintenance, social, and political problems. Due to inappropriate and inadequate urban sanitation infrastructure, these large volumes of FS, contaminated with high pathogen load end up in water bodies which are often used as sources of

irrigation water for production of vegetables most of which are eaten raw. In Ghana, Urban-Peri-urban agriculture is mainly characterized by backyard and commercial small-scale irrigated vegetable farming. It has significant contributions to livelihoods and food security (Cofie, 2003; Amoah, 2008). For example, around Kumasi, Ghana, more than 12,000 farmers are involved in vegetable farming during the dry seasons (Cornish *et al.,* 2001) and urban farmers grow 90% of the main vegetables eaten in the city (Danso *et al.,* 2003). Most of these vegetables are eaten uncooked. These practices are responsible for many environmental and health impacts to the public. In Africa, contamination of faecal origin appears to be responsible for many pathological symptoms notably in children (Ps-Eau, 1999). Moreover, large amounts of plant nutrients in the FS which could have been utilised to improve food production is also washed into water bodies leading to eutrophication and fish kill. Critically analyzing the situation on the ground in developing and newly industrialized countries, it thus calls for provision of more but cheap and simple sanitary facilities for the treatment of faecal sludge.

New approaches in human waste management postulates that sanitation systems should, whenever feasible, be conceived and managed in a way that enables the recycling of organic matter and nutrients contained in human excreta (Winblad 1997; Esrey *et al*., 1998). A change in the sanitation management paradigm from flush-and-discharge to recycling of urine and faeces is gaining ground in Europe (Larsen and Guyer, 1996; Otterpohl *et al*. 1997 and 1999; Otterpohl 2000). As a consequence, treatment strategies and technological options for faecal sludges and solid waste will have to be developed which allow the optimum recycling of nutrients and organic matter to peri-urban agriculture, while being adapted to the local situation and needs.

Use of untreated municipal wastes to enhance productivity and soil quality could lead to significant health risks derived from the spread of excreta related diseases and chemical contamination of the food chain (Owusu-Bennoah and Visker, 1994). Nevertheless, it is considered that where sludge can improve agricultural productivity, independently of its microbial characteristics, it improves the nutritional status of a population lacking food, thus improving public health (Hoglund, 2001; IWMI, 2003)

The organic and solid content as well as ammonium and helminth egg concentrations measured in FS are normally higher by a factor of 10 or more in FS than in sewage or domeatic wastewater (Montangero and Strauss, 2000). It has been recommended that due to high variability of faecal sludge characteristics, the design of a treatment system should be based on case-to-case results, and should include a first treatment step consisting of separating solids from liquid. Methods for solids–liquid separation of FS have been reported (Pescod, 1971; Strauss *et al.*, 1997; Heinss *et al.*,1998; Strauss *et al.*, 2000). In Ghana, stabilization and settling ponds have been the method-of-choice to date. However, system based on this option prove little effective where fresh undigested and highly concentrated FS from public toilets form a major fraction of the FS delivered to the treatment plants. In most cities in Ghana, public toilet sludge (PTS) form up to 50% of the FS collected (http://www.ghanadistricts.com/districts1on1/kma/?arrow=atd&_=6&sa=5492; accessed 12/4/2011).

Many projects have financed treatment facilities by pond system to improve the situation. However, as a result of high solid content and high organic and nutrient load of faecal sludge, most treatment plants were found to be overloaded and could not

develop natural biochemical pathways to handle the overload (Koné and Strauss, 2004).

1.1 Justification of the Study

Large volumes of faecal sludge generated in urban centres of most developing countries, coupled with indiscriminate disposal of liquid and solid waste in the environment have led to rampant outbreak of sanitation related diseases in developing countries. In addition to this are the large amounts of nutrients (N and P) washed from these organic matter into water bodies causing eutrophication and consequent fish kill and the loss of the aesthetical beauty of the environment.

Harmful though the faecal sludge with its associated pathogen load could be, it can be made beneficial and very useful if well handled and treated so that incorporating the nutrients into the soil for agricultural production through recycling will not pose such hazard. Leitzinger (2000) reported that urban and peri-urban agricultural soils are usually depleted of organic matter and nutrients (N and P). According to Drangert, (1998), every person's excreta is in theory at least, nearly sufficient to grow one's own food. In a material flow study conducted in the City of Kumasi, Ghana, (Leitzinger, 2000), it was found that for urban and peri-urban agricultural soils, nutrients (N and P), organic matter, could be fully replenished by using all the human waste and recycling all the organic market waste and the wastes from breweries, timber and food processing factories and from chicken farms. However, most of these wastes would have to be treated prior to use (Leitzinger, 2000; Belevi *et al*. 2000).

Numerous options for treatment of faecal sludge are available of which solid-liquid separation using unplanted drying beds is among the potentially feasible ones. This method is among the lot prescribed for developing and newly industrialized countries. It is comparably more favourable to others with respect to logistics, land requirement and duration of cycle. In the framework of the collaborative programme entitled "Improving agricultural productivity through recycling of liquid and solid waste" a joint research collaboration between Department of Water and Sanitation in Developing Countries (SANDEC) in Switzerland, International Water Management Institute (IWMI), Kwame Nkrumah University of Science and Technology (KNUST) of Ghana and Kumasi Metropolitan Assembly (KMA) in Ghana, initiated a pilot co-composting project for treating faecal sludge together with organic solid waste prior to recycling into urban agriculture. In pursuing the goal of recycling faecal sludge into agriculture, two unplanted drying beds were constructed as an integral component, to generate dewatered biosolids for co-composting with organic solid waste.

Preliminary results showed that unplanted drying beds were effective in dewatering FS and produce biosolids after which the product of co-compost could be safely recommended for agriculture (Koné and Strauss, 2004; Maradan and Schaffner, 2003).

However, the project encountered the problems of

- long dewatering time of 7-14 days which has high cost implication for investment and operation.

- the appropriate sand quality in terms of particle size for efficient dewatering by the drying bed was not identified. Thus resulting in inadequate performance with frequent clogging and high organic and nutrient load in the percolate.

These problems therefore form the bases of this study.

1.2 Objectives of the Studies

The main focus of this research is to improve the the dewatering efficiency of faecal sludge using unplanted filter beds.

Main objective

To improve the rate of faecal sludge dewatering efficiency of unplanted drying bed to produce maximum biosolids and percolate with less concentrations.

Specific objectives

1. To determine the effective particle sizes range of sand that will improve faecal sludge dewatering efficiency of unplanted filter beds.
2. To determine the solid loading rate (SLR) of faecal sludge that will improve the dewatering efficiency of unplanted filter beds.
3. To determine the quantity of physical conditioner (sawdust) that will improve the faecal sludge dewatering efficiency of unplanted filter beds
4. To find the agricultural potential of biosolids produced from unplanted filter beds through faecal sludge dewatering

1.3 Research Questions

It is evident from literature that information on unplanted drying bed for dewatering of faecal sludge is not adequate. From the detailed review of available information on solid – liquid separation on drying beds, the following research questions must be addressed on bench scale basis to give insight into the processes associated with solid-liquid separation on drying beds.

These are:

1. What will be the effect of different ranges of sand media with respect to particle size on the dewatering of FS? In respect of this question, the research seeks to investigate the following questions:
   a) How can different ranges of particle sizes of sand be selected for dewatering?
   b) Can different ranges of particle size improve dewatering time?
   c) Do the different filter media have different contaminant load in the percolate?
   d) Can the different filter media reduce the rate of clogging?
   e) Do the different filter media accumulate different quantities of the biosolids?

2 What is the maximum solids loading rate (SLR) that can best improve FS dewatering. The following questions will be investigated to address this question.
   a) Do different solid loading rates have different dewatering time?
   b) Do the different solid loading rates produce different levels of contaminant load in the percolate?
   c) Can the different solid loading rate reduce the rate of clogging?
   d) Do the different solid loading rates accumulate different quantities of biosolids?

3 What quantity of physical conditioner (sawdust) will improve the faecal sludge dewatering performance of the unplanted drying bed? In the light of this question, the research seeks to investigate the following questions:

a) Will the addition of physical conditioners reduce the dewatering time?

b) Will the addition of different quantities of physical conditioners reduce the rate of clogging differently?

c) Will the addition of different quantities of physical conditioners remove different levels of contaminant load through dewatering process?

d) Will the addition of physical conditioners increase and improve biosolids production and removal?

4 What is the agricultural potential of the biosolid that will be generated from the dewatering processes of the unplanted filter beds? In respect of this question the following questions were investigated.

a) What is the nutrient level of biosolids produced from unplanted filter beds?

b) What are the levels of heavy metals in biosolids produced from unplanted filter beds?

Answers to the above questions will help to improve the design, construction, operation and maintenance of drying beds for effective solids liquid separation. In the present studies these research questions would mainly focus on effective solids-liquid separation using unplanted filter bed.

Hypotheses

The underlying hypotheses of the research questions outlined include:

1. The use of filter media with respect to particle size with less uniformity coefficient is expected to improve the efficiency of the FS dewatering.
2. Varying the solid loading rate is expected to identify the solid loading rate that will best enhance effective FS dewatering process.
3. The addition of different percentages of physical conditioners (sawdust) to the faecal sludge is expected to improve its dewatering and reduce clogging of the filter beds (Spychala and Blazejewski, 2003; Jing, *et al.*, 1999).
4. Biosolids generated from the unplanted filter bed is expected to have improved agricultural potential.

Expected Output

The findings of this research are expected to provide guidelines for the design of effective drying bed for dewatering of faecal sludge in Ghana and other developing countries with similar sludge characteristics.

It is hoped that this will provide a criteria for selecting appropriate filtering media of sludge dewatering, in area where the "proper sand" is not available. It will also be an option for enhancing drying bed performance for faecal sludge treatment.

This will in the broader sense contribute to the improvement of sanitation and food production through treatment of faecal sludge and application of the biosolid produced to enrich the soil, thereby contributing to ensuring the well being of people in developing countries.

CHAPTER TWO

2 LITERATURE REVIEW

2.1 Definitions and Types of Wastewater

Wastewater

Wastewater is used water, removed from residencies, institutions, and commercial and industrial establishments, together with such groundwater, surface water, and storm water as may be present (Metcalf and Eddy, 2003). According to Awuah (2009), wastewater is water containing waste components that impede the application of natural functions of water. Other definitions and concepts of wastewater are given in various reports and textbooks (Westcot, 1997; Asano and Levine, 1998; Martijn and Huibers, 2001). There are different types of wastewater depending on the source of generation and content.

Municipal wastewater

Domestic wastewater effluent made up of black water (excreta, urine and associated sludge) and grey water (kitchen and bathroom wastewater) constitute municipal wastewater. This may also include water from commercial establishments and institutions, including hospitals and may or may not contain substantial quantities of industrial effluent.

Industrial wastewater

Industrial wastewater is water polluted by industrial processes and containing high levels of heavy metals or other inorganic or organic constituents. Industrial effluent does not normally contain high levels of microbiological contaminants. According to

Veenstra and Polprasert (1998), industrial wastewater variations in flow and pollutant mass loading are affected by the production processes (batch, semi-continuous or continuous) and working hours.

Storm water

Storm water runoff is precipitation that finds its way across surfaces into receiving waters. Urban storm runoff is collected and transported in storm or combined sewers. Storm sewers carry storm water only; combined sewers also carry sanitary wastewater. A composition of storm water reflects the composition of precipitation and the surfaces with which it comes in contact (http://www.atl.ec.gc.ca/epb/issues/wstewtr.html, accessed 20/03/2009). Solid concentrations are high, Biochemical Oxygen Demand (BOD) values are relatively low, bacterial values are significant because of animal sources, and nutrient concentrations may be significant. Storm sewers are subject to illicit connections of sanitary wastewater, which can increase concentrations of all of these parameters. Storm water in many developing countries is collected in open or closed drains and freely flows into receiving water bodies. In this case it will never enter wastewaster treatment plants and therefore not affect their performance.

Sewage

Sewage is the water from toilets, bathing, washing, (pre-treated) industrial effluents and storm water (Awuah 2006). In addition to the above, storm water runoff (rain water that collects and flows from the lands surface) may be mixed with sewage prior to treatment. However, in some coastal towns, storm runoff may be collected separately and discharged untreated via storm runoff tunnels. In UK, the average

sewage is 99% water (House of Commons, 1990), and contains 1000 mg/l of impurities, of which about two thirds are organic (Lester, 1990). These impurities will either be suspended or dissolved. The main organic components are: nitrogenous compounds, protein and urea; carbohydrates including sugars, starches and cellulose; fats; oils and greases. The main inorganic compounds include: chlorine, metallic salts, and road grit from storm runoff.

Domestic sewage

Domestic sewage is the used water from home or community which includes toilets, bath, laundry, lavatory and kitchen sink waste (Salvato *et al.*, 2003). The flow rates entering the sewerage system may vary considerably over a day. This depends on the time of the day, the population size and the water use patterns (Veenstra and Polprasert, 1998). If the urine and the faeces are collected, up to 91%, 83% and 59% for N, P and K, respectively, can be recovered and recycled from the household wastewater (Vineras and Jonsson, 2002).

Black water

Black water is the wastewater flow from the toilets. Black water might contain besides urine and faeces/excreta (together sometimes called nightsoil) also some flush water. The mixture is termed as *sewage* if it ends up in a sewerage system or *septage* if it ends up in a septic tank.

Grey water (Sullage)

Grey water is the wastewater originating from the kitchen, bathroom and laundry (Veenstra and Polprasert, 1998). This is usually generated from homes, hostels, hotels

and institutions. In developed countries, it is usually channeled together with black water as sewage. In developing countries, especially Ghana, sullage is usually separated from other wastewater and it usually flows through open drains and on the ground (Awuah *et al.*, 2002).

Septage

Septage is the sludge produced in individual or community on-site wastewater-disposal systems, principally septic tanks and cesspools (Metcalf & Eddy, 2003). In water dependent on-site sanitation facilities, the human excreta is flushed out using water. The resulting wastewater is discharged into septic tanks, where the solid fraction settles out and undergoes anaerobic digestion. The contents of septic tanks comprising of settled and floating solids as well as the liquid portion is known as septage. The actual quantities and constituents of septage vary widely depending on the manner in which the facility is used. Table 2.2 shows some data on the constituents found in septage.

Nightsoil

Night-soil is mostly used to represent a mixture of human faeces and urine (Mara, 1976; Pradt, 1971; Satyanarayan *et al.* 1987; Choi *et al*., 1997). Sometimes night-soil is also used to represent a mixture of human faeces that has undergone some considerable putrefaction. Doku (2003) referred to night-soil as a mixture of human faeces and urine that is mostly fresh. Cairncross and Feachem (1993) stated that night-soil comprises of only faeces and urine plus small volumes of water if water is used for anal cleansing and pour-flushing. Mara and Cairncross (1989) referred to night-soil as mixture of human faeces and urine transported without flushing water.

Public toilet sludge

Public toilet sludge (PTS) is the term used for sludges collected from unsewered public toilets which are usually of higher consistency than septage and biochemically less stabilised (Strauss and Montangero, 2001). Doku (1998) referred to the term 'toilet sludge' as night-soil that has undergone little or partial digestion, collected from non-water dependent systems.

Faecal sludge

Faecal Sludge (FS) is sludge of variable consistency accumulating in septic tanks, aqua privies, pit latrines and unsewered public toilets. The contents comprise of varying concentrations of settleable or settled faecal solids as well as of other, non-faecal matter (e.g. kitchen grease and wastes, plastics, textiles and feminine towels). The sludges also exhibit varying degrees of biochemical stability attained through anaerobic digestion mainly, depending on the ambient temperature, retention period, and inhibition or enhancement due to the presence of other non-faecal substances.

Biosolids

The term biosolids is defined as the primarily organic solid product yielded by municipal wastewater treatment processes that can be beneficially recycled as soil amendments (EPA, 1995; NRC, 2002). Use of the term biosolids has been controversial because of the perception that it was created to improve the image of sewage sludge in a public relations campaign by the sewage industry (Rampton, 1998). Sewage sludge is defined as the solid, semi-solid, or liquid residue generated during the treatment of domestic sewage in a treatment works (NRC, 1996). The National Research Council committee considers sewage sludge as the solid, semi-

solid, or liquid residue generated during treatment of domestic sewage while biosolid is considered as sewage sludge that has been treated to meet the land application standards (NRC, 2002). In the context of this study, biosolids will be described as spadable sludge or a solid that is obtained after dewatering faecal sludge on the filter beds. Biosolids are a complex mixture that may contain organic, inorganic, and biological pollutants from the wastewaters of households, commercial establishments, and industrial facilities and compounds added or formed during various wastewater treatment processes. Such pollutants include inorganic contaminants (e.g., metals and trace elements), organic contaminants (e.g., polychlorinated biphenyls (PBCs), dioxins, pharmaceuticals and surfactants), and pathogens (e.g., bacteria, viruses, protozoa, fungi, and helminthes).

All around the world, people both in the rural and urban areas have been using human excreta for centuries to fertilise fields and fishponds and to maintain or replenish the soil organic content, that is, the humus layer. Until today, in both agriculture and aquaculture, this continues to be common in China and Southeast Asia as well as in other places in Africa (Cross 1985; Timmer and Visker, 1998; Visker, 1998; Timmer, 1999; Strauss *et al.*, 2000). Uses of biosolids have led to a strong economic linkage of urban dwellers (food consumers as well as waste producers), to the urban farmers (waste recyclers and food producers).

According to Cofie *et al.* (2006), the quantity of helminth egg of biosolids generated from drying beds is high. Hence, to inactivate the eggs and other surviving pathogens for use in agriculture, the biosolids is used for co-composting with the biodegradable fraction of municipal solid waste. Wood chips (sawdust) nonetheless will also be an

appropriate addition to the dewatered solid (biosolid). Biosolids generated in Ghana and most developing countries which is mainly faecal in origin, are expected to have little or no heavy metal concentration (Montangero and Strauss, 2002). However, increasing commercial and industrial activities coupled with indiscriminate disposal of waste have led to increasing concentrations of heavy metals in the biosolids produced (Heins *et al.*, 1998). This will be monitored in the biosolids generated in this research for possible consideration.

2.2 Characteristics of Wastewater

A typical domestic wastewater contains several substances (Metcalf and Eddy, 2003). The characteristics of wastewater depends on many variables, including the country, the type of diet, health status and water use patterns. Thus wastewater can be classified into 3 categories as weak, medium and strong as reported by Metcalf and Eddy (2003) in Table 2.1. The strength of sewage depends mostly on the amount of water used per capita per day. In countries where water is scarce, the wastewater tends to be strong. Even within the same city or country, high class communities tend to use more water than low class communities, thus high class communities generate comparatively less concentrated wastewater.

The above assertion is well manifested in one of the studies on wastewater characteristics where the characteristics of wastewater from Kwame Nkrumah University of Science and Technology (KNUST) in Kumasi was compared with that of Asafo, also a suburb of Kumasi. The wastewater of Asafo was found to be more concentrated than that of KNUST. Investigations revealed that in most homes in Asafo, inhabitants intentionally allow more people to use the toilets before flushing

the water closet, while others store water used for washing clothing and use for flushing the toilets as a way of reducing expenses on water (Kuffour, 2001). Thus the kind of wastewater found in Ghana and many other developing countries is generally the strong type (Table 2.1) or even higher. The composition of septage (Table 2.2) though having received certain level of putrefaction is still comparatively more concentrated than the strong type wastewater (Table 2.1). This is because many septic tanks have soak away which drain off most of the water content thus concentrating the resulting sludge as it stays longer in the tank.

Table 2.1: Typical composition of domestic wastewater

Parameter	Weak	Medium	Strong
BOD_5, 20^oC	110	220	400
Total organic carbon (TOC)	80	160	290
COD	250	500	1000
NH_4^+	12	25	50
NO_3^-	0	0	0
NO_2^-	0	0	0
Total Phosphorus	4	8	15
Organic phosphorus	1	3	5
Inorganic phosphorus	3	5	10
Chlorides	30	50	100
Sulphate	20	30	50
Alkalinity (as $CaCO_3$)	50	100	200
Grease	50	100	150
Total Coliforms	$10^6 – 10^7$	$10^7 – 10^8$	$10^7 – 10^9$
Volatile organics compounds	<100	100-400	>400
Total solids (TS)	350	720	1200
Dissolved solids, total (TDS)	250	500	850
Fixed	145	300	525
Volatile	105	200	325
Suspended solids (SS)	100	220	350
Fixed	20	55	75
Volatile	80	165	275
Settlable solids	5	10	20
Total Nitrogen	20	40	85
Organic Nitrogen	8	15	35

Data Source: Metcalf and Eddy (2003*). All concentrations are in mg/1 with the exception of total coliforms No./100ml and volatile organic compounds, mg/1.*

Table 2.2: Typical characteristics of septage

Constituent	Concentration, mg/l	
	Range	Typical
Total solids (TS)	5,000 – 100,000	40,000
Suspended solids(SS)	4,000 – 100,000	15,000
Volatile suspended solids (VSS)	1,200 – 14,000	7,000
BOD_5	2,000 – 30,000	6,000
COD	5,000 – 80,000	30,000
Total Kjeldahl nitrogen (TKN)	100 – 1,600	700
Ammonia-nitrogen	100 – 800	400
Total phosphorus as P	50 – 800	250
Heavy metals[a]	100 – 1,000	300

[a] Primarily iron (Fe), zinc (Zn) and aluminium (Al)

Source: Metcalf and Eddy (2003)

2.3 Historical Overview of Wastewater Treatment and Disposal

In modern societies, proper management of wastewater is a necessity, not an option. Historically, the practice of collecting and treating wastewater prior to disposal is a relatively recent undertaking.

Prior to the nineteenth century, residents of most European cities (such as London and Paris) placed "night soil" in buckets along the streets and workers emptied the waste into tanks of trucks. The waste was transported to rural areas for disposal over agricultural lands. The invention of the flush toilet in the nineteenth century drastically changed waste-disposal practices. Existing systems for transporting urban waste for disposal on agricultural lands were not adequate to handle the large volume of liquid generated by the flush toilets. Faced with this transportation problem, cities began to use natural drainage systems and storm sewers for wastewater carriage against the advice of such men as Edwin Chadwick, who in 1842 recommended "rain to the river and sewage to the soil". Construction of combined sewers was commonplace in large cities during the latter half of the nineteenth century. Since

storm drain systems naturally ended at watercourses, waterborne wastes were discharged directly to streams, lakes, and estuaries without treatment. Gross pollution often resulted, and health problems were transferred from the sewered community to downstream users of the water. (Source: Microsoft(R) Encarta(R) 97 Encyclopedia, Sewage Disposal 1993-1996).

The first modern sewerage system for wastewater carriage was built in Hamburg, Germany, in 1842 by an innovative English engineer named Lindley. Lindley's system included many of the principles that are still in use today. Most of the improvements in wastewater collection systems over the last 100 years have consisted of improved materials and the inclusion of manholes, pumping stations, and other appurtenances. (Source: Microsoft(R) Encarta(R) 97 Encyclopedia, Sewage Disposal 1993-1996).

The treatment of wastewater lagged considerably behind its collection. Treatment was considered necessary only after the self-purification capacity of the receiving waters was exceeded and nuisance conditions became intolerable. Various treatment processes were tried in the late 1800s and early 1900s, and by the 1920s, wastewater treatment has evolved into those processes in common use today. (Source: Microsoft(R) Encarta(R) 97 Encyclopedia, Sewage Disposal 1993-1996).

In most developing countries such as Ghana, Waste Management Departments are still struggling with waste collection, treatment and disposal. A large proportion of waste especially industrial waste and faecal sludge are constantly discharged directly into streams, lakes, rivers and seas without treatment. This has resulted in gross pollution

of the water bodies and actually threatening the survival of man in many cities and towns.

2.3.1 General management of wastewater and concerns

Investment in sanitation, including wastewater collection and treatment, is high on the political agenda. Millenium Development Goal no. 7 will only be achieved if proper wastewater (PTS and septage) collection and treatment systems are put in place. In many developing countries projects to improve the situation are on-going. It is very important that not only the design and construction phase of these projects are well executed, but that the operational phase also gets sufficient attention. This attention (i.e. investment in operation and maintenance and the associated cost recovery) is often lacking and as a result the project sustainability is often poor. Mariño and Boland (1999) observed:

> *'Worldwide experience in wastewater management has numerous examples of complete or partial failure. The most common management actions are, unfortunately, no action at all – by one estimate, as much as two-thirds of the wastewater generated in the world receives no treatment at all. Even more conspicuous are the many newly built wastewater treatment plants that are not properly operated and maintained, or, in some cases, that are not being operated at all. Often, inappropriate treatment levels or treatment technologies have been selected, leading to excess costs or disappointing results, or both. Often such investments are poorly targeted, providing abatement for low priority effluents, while more hazardous discharges go untreated. Many of these investments, responding to a piecemeal approach to*

river basin pollution problems, may also result in costly and ineffective solutions'

However, Parr and Horan (1994) argued that the fundamental reason for failure of wastewater management strategies is due to inappropriate technology selection.

2.3.2 The Bellagio Statement and the 3-step Approach

One of the new approaches in sanitation is the 'Bellagio Statement'(Box 1), formulated by the Environmental Sanitation Working Group of the Water Supply and Sanitation Collaborative Council (WSSCC) in 2000. Its principles are believed to be essential for achieving the objective of worldwide access to safe environmental sanitation and a healthy urban water system:

Box 1: The Bellagio Statement

1. ***Human dignity, quality of life and environmental security should be at the centre of the new approach, which should be responsive and accountable to needs and demands in the local setting.***
   - *Solutions should be tailored to the full spectrum of social, economic, health and environmental concerns*
   - *the household and community environment should be protected*
   - *the economic opportunities of waste recovery and use should be harnessed*
2. ***In line with good governance principles, decision-making should involve participation of all stakeholders, especially the consumers and providers of services.***
   - *decision-making at all levels should be based on informed choices*
   - *incentives for provision and consumption of services and facilities should be consistent with the overall goal and objective*
   - *rights of consumers and providers should be balanced by responsibilities to the wider human community and environment*
3. ***Waste should be considered as a resource, and its management should be holistic and form part of integrated water resources, nutrient flows and waste management processes.***
   - *inputs should be reduced so as to promote efficiency and water and environmental security*
   - *exports of waste should be minimised to promote efficiency and reduce the spread of pollution*
   - *wastewater should be recycled and added to the water budget*
4. ***The domain in which environmental sanitation problems are resolved should be kept to the minimum practicable size (household, community, town, district, catchment, city) and wastes diluted as little as possible.***
   - *waste should be managed as close as possible to its source*
   - *water should be minimally used to transport waste*
   - *additional technologies for waste sanitisation and reuse should be developed*

The 3 step strategic approach

The UNEP 3 Step Strategic Approach (Nhapi and Gijzen, 2005) is based on the application of the 'Cleaner Production approach' that has been so successful in the industrial sector. The authors evaluated the current urban water management system from a cleaner production point of view, and made clear the urgency to re-think our current practices in the light of sustainability (Box 2). An assessment of methods applied in industries and in solid waste management for pollution prevention and control reveals a systematic approach that is useful to wastewater management as well. A 'translation' of the cleaner production principles outlined in Box 2 to the water sector led to the development of the so called '3-Step Strategic Approach' for urban water management.

Box 2. A cleaner production approach to Urban Water Management (Nhapi and Gijzen, 2005)

The cleaner production concept, developed over the last two decades, has brought some innovative environmental thinking into the industrial sector. If we apply some of the basic principles of cleaner production to the current practices in urban water services, we may realise the need for drastic changes:

Principle 1: Use lowest amount of input material, energy or other resources per unit of product.

Practise: We supply between 130 and 350 l of drinking water per capita per day, while less than 2 litres are actually used for drinking

Principle 2: Do not use input materials of a higher quality than strictly necessary.

Practise: We use water purified to drinking water standards to flush toilets, clean floors, wash cars or to irrigate the garden.

Principle 3: Do not mix different waste flows.

Practise: Already in the household various wastewater flows are combined (urine and faecal matter, grey and black water). After disposal into the sewer this combined waste is mixed further with industrial effluents, and often times also with urban runoff. Obviously this practise makes re-use of specific components in the mixed waste flow less attractive and less feasible.

Principle 4: Evaluate other functions and uses of by-products before considering treatment and final disposal.

This approach strongly focuses on sewage management, but also considers water supply, nutrient uses and other material flows associated with the urban water cycle. The three steps include: 1) prevention, 2) treatment for reuse, and 3) planned discharge with stimulation of self-purification capacity. The steps should be implemented in chronological order, and possible interventions under each step should be fully exhausted before moving on to the next step. This strategic approach is summarized in Figures 2.1 and 2.2.

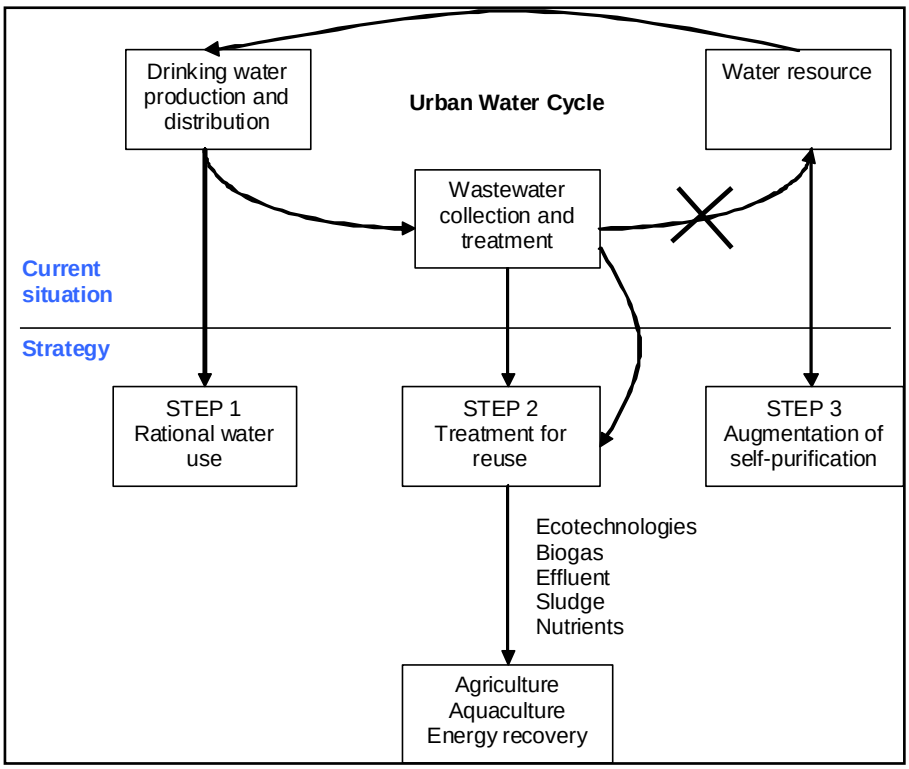


Figure 2.1: Application of the 3-Step Strategic Approach to the Urban Water Cycle

Source: Nhapi and Gijzen, (2005).

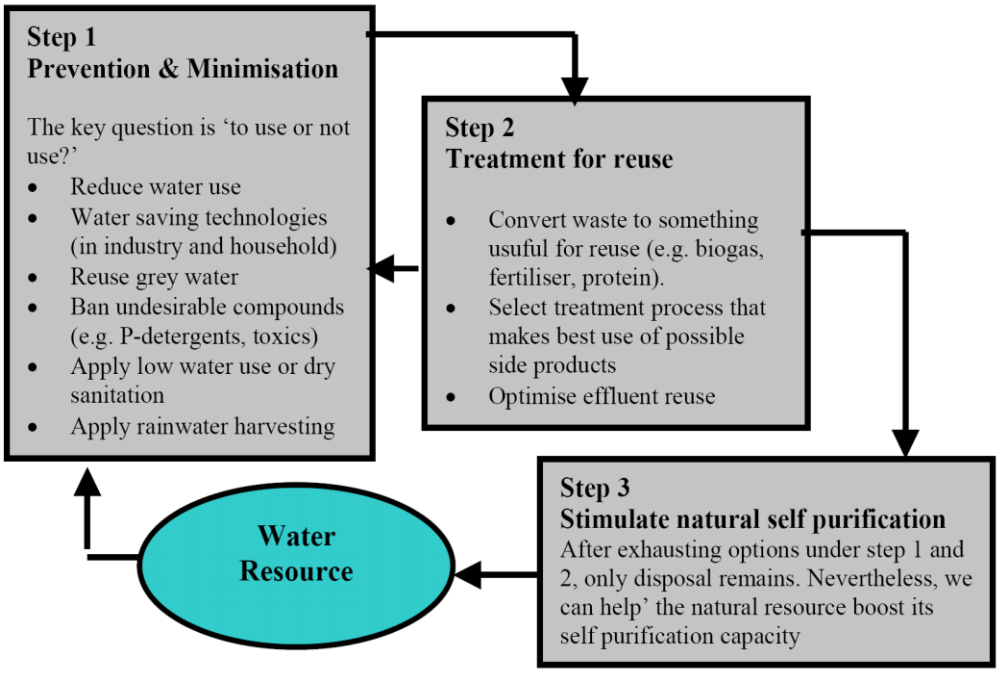


Figure 2.2: Schematic representation of 3-Step Strategic Approach to wastewater management.
Source: Nhapi and Gijzen (2005)

2.4 Criteria for Proper Wastewater Disposal

Proper disposal of sewage and other wastewater is necessary not only to protect the public's health and prevent contamination of groundwater and surface water resources, but also preserve fish and wildlife populations and to avoid creation of conditions that could detract from the attractiveness of a community, tourists establishment, resort and recreation area. The following basic criteria should be satisfied in the design and operation of excreta, sewage or other wastewater disposal system (Salvato *et al.*, 2003).

- Prevention of pollution of water supplies and contamination of shellfish intended for human consumption.
- Prevention of pollution of bathing and recreational areas.
- Prevention of nuisance, unsightliness and unpleasant odours

- Prevention of human wastes coming into contact with man, animals and food or being exposed on the ground surface accessible to children and pets
- Prevention of fly or mosquito breeding, exclusion of rodents and other animals.
- Strict adherence to standards of groundwater and surface waters; compliance with local regulations governing wastewater disposal and water pollution control.

Failure to observe these basic principles can result in the development of health hazards and the degradation of living conditions, recreational areas and natural resources that are essential for the wellbeing of the general public. Protection of land and water resources should be the national policy and every effort should be made to prevent their pollution by improper treatment and disposal of sewage and other types of liquid waste (Salvato *et al.*, 2003).

2.5.1 Status of Urban wastewater Disposal and Treatment in Ghana

2.5.1 Sanitation and liquid waste generation

About 63% of Ghana's population has sanitation coverage, which is more than the West Africa average of 48% and similar to the average of eastern Africa (62%) and southern Africa (63%) (WHO/UNICEF/WSSCC, 2000). While most countries in West Africa (like Senegal) show a very high disparity in provision of sanitation services between rural and urban areas, Ghana has a good balance with 62% coverage in urban areas and 64% in rural areas. According to Agodzo *et al*. (2003) the total amount of grey and black wastewater produced annually in urban Ghana has been estimated as 280 million m^3. This wastewater is derived mainly from domestic sources

as Ghana's industrial development is concentrated along the coastline where wastewater, treated or untreated, is disposed of into the ocean. In Ghana, collection and disposal of domestic wastewater is done using:

- Underground tanks such as septic tanks and aqua-privies, either at industrial facilities or at the community level and then transported by desludging tankers to treatment works or dumping sites
- Sewerage systems
- Public toilets
- Pit and improved latrines.

Less than 5% of the households in Accra and Kumasi are connected to piped sewerage systems, while 21% use floodwater drains (gutters) as open sewerage that ends up in nearby water bodies. Some of the urban dwellers discharge their faecal waste into septic tanks while kitchen and other wastes from the home are usually directed into the nearest open drain. As the majority of the urban drains are open, they often serve as defecating areas for households that do not have adequate sanitation facilities. According to the national population and housing census carried out in 2000, one third of all households in Ghana use public toilets, reflecting the absence of toilet facilities in many dwelling places. Pit latrines continue to be used in 22% of all households but an improved version, the Kumasi Ventilated Improved Pit (KVIP), is being promoted and its use is expected to rise from 7%. Bucket latrines (4%) are being phased out because they are not hygienic. It is quite striking that more than 25% of all households in Ghana have no toilet facilities, with numbers increasing to about 70% in the three northern regions. Water closets (WCs), considered to be modern toilet facilities, are used by only 9% of the households, most of them located in Accra and Kumasi

(WHO/UNICEF/WSSCC., 2000). Thus, the majority of the population in urban Ghana does not have appropriate means to manage wastewater and the costs of putting in place the required infrastructure to effectively collect and dispose of all urban wastewater are excessive.

2.6 Wastewater Treatment

More than half of all wastewater treatment plants in Ghana are in and around Accra (EPA, 2001). Two administrative regions (Brong Ahafo and Upper West) have no treatment plant, despite having several important cities and towns. But even where treatment plants are available, less than 25% (primarily in the Greater Accra, Ashanti and Eastern regions, and mostly small-capacity and/or privately owned plants) are functional (EPA, 2001). A few years ago, a large modern biological treatment plant (UASB) started operation at Accra's Korle Lagoon, in Jamestown, Accra; but, it handled only about 8% of Accra's inner-city wastewater from domestic and industrial sources (personal communication with management). The system had a capacity three times greater than what it was receiving, but was constrained by the small urban sewerage network. The plant is currently broken down as a result of lack of proper maintenance and operation and logistics (personal observation). Only about 10% of the Accra's wastewater is collected for some kind of treatment (Agozo *et al*., 2003).

There are only a few low-capacity treatment facilities (usually stabilisation ponds) functioning in most cities. To cite just a few examples: Over the last few years, Dompoase stabilisation ponds treatment plant which is Kumasi's main faecal sludge treatment plant (FSTP) was receiving less than an average of 300 m^3/ day. A research conducted on the evaluation of the performance of the treatment plant indicated that

an average of about 240 m^3 of faecal sludge was being sent to the plant daily in 2006 (Ackon, 2006). However, the treatment ponds have been filled beyond capacity for years, often without desludging for many months (personal observation). The effluent from the ponds which flows to River Oda downstream has been found to be far higher beyond effluent guidelines by EPA-Ghana (Ackon, 2006). The situation is similar in Accra with two sites loaded beyond capacity. The ocean is the third semi-official site, receiving about 40% of the excreta produced in the city. In Tamale, the first plant is still under construction while faecal sludge continues to be dumped in natural depressions and on farmlands. In most cities in Ghana, faecal sludge continues to be discharged into depressions, excavations and on farmlands as a result of lack of properly designed treatment facilities. It is apparent that city sanitation services cannot keep pace with the high urbanisation rates (Keraita *et al.*, 2003b).

2.7 Characteristics of Faecal Sludge

Faecal Sludge (FS) is sludge of variable consistency accumulating in septic tanks, aqua privies, family latrines and unsewered public toilets. These contents comprise varying concentrations of settleable or settled faecal solids as well as of other, non-faecal matter (e.g. kitchen grease and wastes, plastics, textiles and feminine towels). The sludges exhibit varying degrees of biochemical stability attained through anaerobic digestion mainly, depending on the ambient temperature, retention period, and inhibition or enhancement due to the presence of other non-faecal substances.

Organic strength, ammonium nitrogen (NH_4-N) concentrations, solids contents and helminth egg concentrations of faecal sludges also greatly differ from those of municipal wastewater collected in centralized sewerage systems and are normally higher by a factor of 10 or more. Type “A” sludges are rather fresh and exhibit high

concentrations of organics, ammonium and solids which is described to be 10 -100 times as high as sewage (Koné and Strauss, 2004; Montangero and Strauss, 2000). They originate from non-flush or pour flush family and public toilets. Type "B" are relatively weak strength and they have usually been stored for lengthy periods of time (from one year to several years) and hence have undergone biochemical stabilization to a considerable extent, e.g. Septage (Table 2.3). Table 2.4 shows typical FS characteristics, based on FS studies in Accra/Ghana, Manila/Philippines and Bangkok/Thailand. The characteristics of typical municipal wastewater as may be encountered in tropical countries are also included for comparison's sake (Table 2.3).

Table 2.3: Characteristics, classification and comparisons of on-site faecal sludge to sewage in tropical countries

Item	Type "A" (high –strength)	Type "B" (low-strength)	Sewage – for comparison's sake
Example	Public toilet or bucket latrine sludge	Septage	Tropical Sewage
Characteristics	Highly concentrated, mostly fresh FS, stored for days or weeks only	FS of low concentration; usually stored for several years; more stabilized than type "A"	
COD mg/l	20,000 – 50,000	< 15,000	500 - 2,500
COD/BOD_5	5:1 to 10:1		
NH_4-N mg/l	2,000 – 5,000	<1,000	30 – 70
TS mg/l	3.5%	< 3%	>1%
SS mg/l	30,000	≈7, 000	200 – 700
Helminth eggs, no./l	20,000 – 60,000	≈4, 000	300 – 2,000

(Source: Strauss *et al.*,(1997); Mara, 1978)

Comparison of different raw sludge data

Table 2.4 shows results of different faecal sludge analyses carried out during the collaborative field research between SANDEC and its partners. Also shown are the averages of reported data from the United States (USEPA, 1984). Faecal sludge

quality can strongly differ from place to place as many factors influence the real sludge quality as well as the analysed quality.

Table 2.4: Faecal sludge quality in different cities

Parameter	Accra Septage	Accra, Public toilet Sludge	Bangkok Septage	Manila Septage	US EPA Septage
COD	7,800	49,000	14,000	37,000	43,000
BOD	600 -1500	7,600		3,800	5,000
TS	11,900	52,500	16,000	72,000	38,800
TVS (%)	60	69	69	76	65
pH	7.6	7.9	7.7	7.3	6.9
COD/BOD	6-12	6.4	NA	9.7	9
COD/TS	0.7	0.9	0.9	0.5	1.1
Helm.eggs no/l	4,000	25,000	NA	5,700	NA

All units except pH and the ratios (COD/BOD, COD/TS) are in mg/l unless otherwise stated. NA = not available
Source: Heins *et al*. (1998)

As the septage is very inhomogeneous, consisting of a liquid phase, settled and partly settled solids, scum and dissolved solids, representative sampling from septic tanks is very difficult (Table 2.5). Raw sludge data for faecal sludges are more inhomogeneous than for wastewater and strongly vary from place to place. Only average values received from a statistically sufficient number of analyses may serve as a design basis for treatment plants. One example which illustrates the variability of septage samples is as follows: The average chemical oxygen demand (COD) concentration of 75 septage samples in Accra amounted to 6,400 mg/l with the very high standard deviation of 6,200 mg/l. Again, the COD of FS used for dewatering on different unplanted filter beds reported by Kuffour *et al.* (2009), in their dewatering experiments, had high variability (COD of 50,320 mg/l had standard deviation of 28,780 mg/l) due to high variability of septage mixed with the public toilet sludge.

The results from the monitoring and evaluation of the Achimota FSTP by Water Research Institute and Department of Water and Sanitation in Developing Countries (WRI/SANDEC) indicate that the current system could be effective in the treatment of the septage because the solids easily separate from the liquid in the settling/thickening tanks ensuring that only the liquid fraction flows into the pond system for treatment.

Table 2.5: Characteristics of Faecal Sludges from On-site Sanitation Systems in Accra, Ghana

Parameter	Public Toilet Sludge	Septage
BOD (mg/l)	8,800 (3,800 - 15,000)	630 (360 - 1,300)
COD (mg/l)	47,600 (10,400 - 97,000)	8,500 (820 - 52,000)
TS (%)	-	1.4 (0.3 - 11.4)
TVS (% of TS)	62	63
TSS (%)	6.4 (2 - 19)	0.7 (0.07 - 3.4)
VSS (% of TSS)	58	70
Helminth eggs (no./l)	29,000 (3,600 - 62,000)	4,300 (200 - 13,000)

Source: WRRI/SANDEC, 1994

The FSTP can also, to some extent, treat mixtures of nightsoil/toilet sludge and septage (the mixtures containing higher proportions of septage) if operational and design guidelines are adhered to. With regards to the nightsoil/toilet sludges the results from the monitoring and evaluation of the Achimota FSTP indicate that they are hardly conducive to solids-liquid separation, in the first step of the FSTP design. This results in the faecal sludge flowing into the pond system and causing the system to fail from overloading due to high organic strength and high concentrations of ammonia. Heinss *et al.* (1998) attribute the lack of solids-liquid separation to the fact that the nightsoil/toilet sludges are mostly fresh, i.e. undigested and highly concentrated compared with the septage. The solids-liquid separation can be improved by digesting the nightsoil/toilet sludges.

2.8 Faecal Sludge Treatment

2.8.1 Overview of FS treatment options

Proper FS treatment, either in combination with wastewater or separately, is being practiced in a few countries only (e.g. China, Thailand, Indonesia, Argentina, Ghana (Kumasi), Benin, Botswana, South Africa). In the U.S.A., most of the septage (the contents of septic tanks) is co-treated in wastewater treatment plants. In some states, notably in the Northeast, pond systems are used to separately treat septage. They typically consist of an anaerobic sedimentation pond followed by an infiltration pond. Treatment of faecal sludge is the surest way of reducing the risk associated with its reuse. Potential treatment options that may prove appropriate in developing countries are the low cost types, for example, waste stabilization ponds, drying beds (planted – constructed wetlands) and composting (Figure 2.3). There is no single best option, which may prove equally suitable under the widely varying conditions which prevail in urban areas of developing countries. Institutions/organizations like the World Health Organisation (WHO), Department of Water and Sanitation in Developing Countries (SANDEC) at the Swiss Federal Institute for Environmental Science & Technology (EAWAG)-Switzerland, Water Research Institute (WRI) of the Centre for Scientific and Industrial Research (CSIR)-Ghana, the International Water Management Institute (IWMI) and Kwame Nkrumah University of Science and Technology (KNUST) have been involved in developing guidelines for low-cost and moderately sophisticated faecal sludge treatment options. These comprise processes and technologies that prove sustainable under the varying economic, institutional and technological conditions prevailing in developing and newly industrializing countries, with paradigm that “something” (e.g. 75% instead of 95-99% helminth egg removal)

is better than “nothing” (the lack of any treatment at all or the often totally inadequate operation of existing treatment systems) (Von Sperling, 2001).

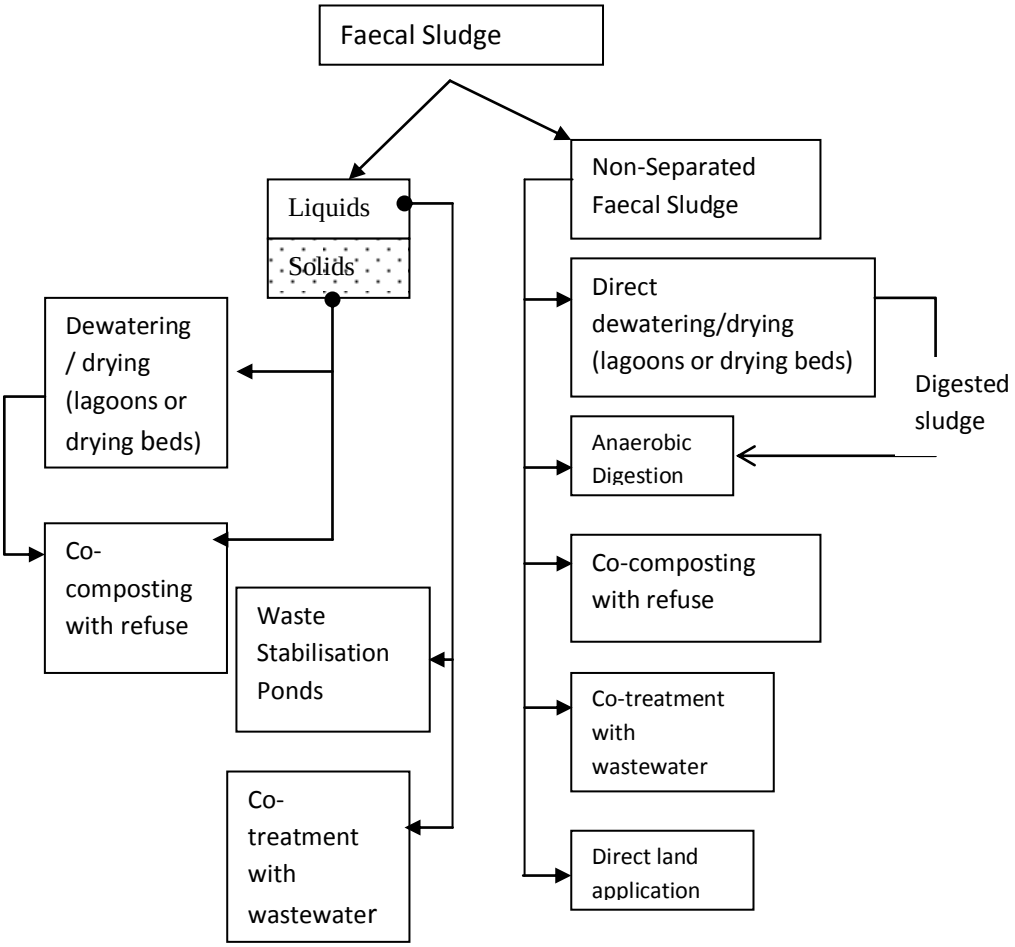


Figure 2.3 Low cost theoretical options for treating faecal sludges
Source: Strauss (1998)

Based on the general characteristics of FS, the design of treatment systems can be summarized as follows:

- A first treatment step consisting of the separation of the solids from the liquid part (e.g. drying beds or sedimentation ponds/tanks) appears meaningful as most of the organic matter is contained in the solids part. Besides, it allows concentrating the helminth eggs in the separated solids fraction.

- The fresh undigested sludge should be stabilized (Stabilization is the term used to denote the process of BOD reduction. The stabilization process can be carried out under aerobic or anaerobic conditions. eg. through primary, anaerobic treatment in a pond or a reactor). Sludges that have already attained a high level of stabilization could be directly dewatered (e.g. on planted/unplanted drying beds, sedimentation/thickening ponds) and further mineralized (through thermophilic composting).
- If the main objective is to reduce environmental pollution (e.g. of the surface waters), the treatment system should attain high removal efficiencies of organic matter and nutrients. If it is to reuse the FS, then, the treatment option should aim at creating valuable products for agriculture (i.e. stabilize and hygienise the biosolids while limiting nutrient losses) to allow safe handling by beneficiaries.

FS Treatment and regulations

In the majority of less-industrialized countries, effluent discharge legislation and standards have been enacted. The standards usually apply to both wastewater and faecal sludge treatment. They are often too strict to be attained under the unfavourable economic and institutional conditions prevailing in many countries or regions, especially in developing countries. Quite commonly, effluent standards are neither controlled nor enforced. Examples for faecal sludge treatment standards are known from China and Ghana. In the Province of Santa Fé, Argentina, e.g., current WWTP effluent standards also apply to FS treatment. For sludges used in agriculture, a helminth egg standard has been specified (Ingallinella, 1998).

Standards setting – appeal for a sensible approach

According to Vesilind (2000), "the responsibility of the regulator is to incorporate the best available science into regulatory decision making. But problems arise when only limited scientific information is available. The complexity of the environmental effect of sludge on human health leads to scientific uncertainty and makes sludge disposal difficult". The same author indicates that the standards elaborated recently by USEPA are based on the "principle of expediency" formulated by Phelps in 1948. The principle is "an ethical model that calls for a regulator to optimise the benefits of health protection while minimising costs within the constraints of technical feasibility" (Vesilind, 2000).

If this paradigm – basing environmental regulations on available technology and on (local) economic and institutional resources – has been adopted in industrialised countries, it should even more be applied to economically less advanced countries. There, the development of monitoring and enforcement systems is still lagging far behind and is more difficult to organise and implement than in industrialized countries. Therefore, replicating the strict standards or limits established in industrialized countries without taking into account the regional characteristics or necessary data pertaining to the local conditions is entirely inappropriate. A typical example is found in Ghana, where discharge levels of WWTP effluent for BOD, suspended solids, COD are cited as 50, 50 and 250 mg/l respectively whereas the receiving streams have values far higher than these. River Oda, which receives the effluent from Dompoase treatment plant, had BOD, suspended solids and COD being 200.0, 4.0 and 1648.0 mg/l respectively, at the upper course of the discharge point, as at the time of sampling in 2006. In many instances, the numerical values of certain

parameters are established without defining locally appropriate management and treatment options for wastewater and biosolids. Such options would have to take into account disposal vs. use scenarios; types of soils on which treated human wastes are spread; influence on the crops; health aspects; financial and economic factors, and institutional settings. Clearly, distinct standards and a distinct selection of treatment parameters should be stipulated depending on whether treated wastes would be used in agriculture or discharged into the environment. For reuse, hygiene-related variables (helminth eggs in biosolids and faecal coliforms in wastewater) and nitrogen are the relevant criteria whereas for discharge, variables such as COD or BOD and NH_4 are of prime importance (Mara, 1997). A sensible strategy for public health protection in biosolids use has been adopted by the EU. The general principle is to define and set up a series of barriers or critical control points, which reduce or prevent the transmission of infections. Sludge treatment options, which were found to inactivate excreted pathogens to desirable levels, are the prime element in this (Matthews, 2000). "Barrier points" such as the sludge treatment works, can be easily controlled with respect to design and operations, thereby securing the compliance of the treated biosolids with stipulated quality standards. In contrast to this, the controlling of numerical quality criteria for wastewater or biosolids requires regular monitoring. In economically less developed countries, such monitoring is often difficult and very costly to perform. Results may not be reliable and replicable as adequate routine, quality control and cross-referencing are lacking. In industrialised countries, pollution laws have been made more stringent in a stepwise manner over many decades. Concurrently, wastewater and sludge treatment technology has been upgraded stepwise to cope with an increasing number of constituents and to reduce pollution loads discharged into the environment (Johnstone and Horan, 1996).

2.9 Solids-liquid Separation

Faecal sludges typically exhibit very high total solids (TS) and suspended solids (SS) contents, compared to wastewater. The separation of the solids and the reduction in volume of the fresh FS might be desirable e.g. when treating FS in ponds, be it separately or in conjunction with wastewater. Any of these can be used as an option to produce biosolids conducive to agricultural use, and when intending the joint composting of FS solids and solid organic wastes.

Process disturbance by improper design and operation for solids separation has been repeatedly observed (Hasler, 1995; Mara *et al*., 1992). The settleability of FS can, as a first approach, be determined by settling tests in graduated cylinders at laboratory scale. Thereby, approximate information can be gained regarding (1) the rate of settling, (2) the density of the separated solids and (3) the quality of the liquid supernatant produced during the separation process.

Settling conditions in cylinders or columns are usually more quiescent and thus more favourable than in full-scale units. Therefore, a scale-up or security factor must be applied when using settling test results to size full-scale settling-thickening units. The settleability of faecal sludges varies considerably depending on the type of sludge and specific location (USEPA, 1984; Heinss *et al.*, 1998). Results from FS settling tests carried out at the Water Research Institute (WRI) in Accra have shown that Accra's septage, which has an average TS contents of 12,000 mg/l (thereof, 60 % volatile solids, TVS), exhibits good solids-liquid separability (Larmie, 1994; Heinss *et al*., 1998). Separation under quiescent conditions is complete within 60 minutes. This holds also for FS mixtures containing up to 25 % by volume of fresh, undigested

sludge from unsewered public toilets. The settling tests conducted at WRI with 4:1 mixtures of septage and public toilet sludge (SS = 4,500-18,400 mg/l), showed that theoretical SS removals of 80 % can be achieved. This resulted in SS concentrations in the supernatant of 1,200-3,500 mg/l. Investigations conducted at the full-scale settling tanks in the Achimota FSTP revealed that clear-liquid SS concentrations of ≤4,000 mg/l were achieved. The scale-up or safety factor would thus amount to 2 to 3. For septage, the cylinder tests simulated a 67-94 % removal of SS, resulting in supernatant SS of 150-700 mg/l (Larmie, 1994).

Settling tests were also conducted at AIT in Bangkok using septage of the City of Bangkok exhibiting an average SS concentration of 12,000 mg/l. Cylinder settling tests showed complete separation in 30-60 minutes and that SS concentrations of 400 mg/l in the supernatant were achieved (Koottatep, 2001; Kost and Marty, 2000). The rate of accumulation of settleable solids, hence, the required solids storage volume, is the decisive design criteria for preliminary settling/thickening units or for solids storage compartments in primary ponds. The specific volume occupied by separated solids may be assumed as 10 – 15 % of FS, depending on FS composition and on the period allowed for solids consolidation and thickening (Heinss *et al.*, 1998). Thickened solids densities in the settling/thickening tanks of the Achimota FSTP in Accra ranged from 14% TS in the settled solids layer to 16% TS in the scum layer by the end of the 4-8 weeks loading cycles (Larmie, 1994). The fairly thick scum layer was due to the share of undigested, high-strength sludges from unsewered public toilets and their associated intensive gas production causing buoyancy. In the septage settling ponds of the Alcorta (Argentina) pond scheme, TS in the settled solids amounted to about 18% after 6 months of septage loading (Ingallinella *et al.*, 2000).

Septage collected in Alcorta exhibited a SS content of approximmately 8,000 mg/l (which might be associated with an estimated TS content of 12,000-15,000 mg/l). The specific volume of accumulated solids was only 0.02 m^3/m^3 of fresh septage, hence, 5-7 times less than that found in the settling/thickening tanks of the Achimota FSTP in Accra. This was due to the higher hydraulic (and solids) loading rates (HLR) applied to the settling tanks in Accra (~ 0.7 m/d and 10 kg SS/ m^2day) as compared to the settling ponds in Alcorta (~ 0.1 m/d and 0.8 kg SS/m^2day).

Pond Treatment

The use of anaerobic ponds

Given the high organic strength frequently encountered in faecal sludges, anaerobic ponds - with or without prior solids removal in separate settling units - are a feasible option as primary units in pond treatment schemes in warm climate. Use of facultative ponds for Faecal Sludges may often not be possible due to the high ammonia levels in the sludges accumulating in unsewered public toilets with zero or low-flush installations or in latrines with so-called watertight pits. Excessive ammonia (NH_3) contents will impair or suppress algal growth (Montangero and Strauss, 2002). Primary treatment in anaerobic ponds especially in warm climatic conditions, might be the method-of-choice in developing countries. Effluent use in agriculture is not possible due to its high salinity.

Treatment Mechanism in Anaerobic Ponds

In anaerobic ponds, growth of bacteria is encouraged to breakdown particulate organic matter in absence of oxygen (anaerobically). Sludge is deposited at the bottom while scum is formed at the surface of the pond.

Anaerobic sludge stabilization involves various microbial bacteria which show great complexity of interactions. The overall process is the conversion of organic waste into methane and carbon dioxide. Bushwell stoichiometric chemical equation is given by (Bushwell and Muller, 1952):

$$\boldsymbol{C_n H_a O_b N_c} + (n - \frac{a}{4} - \frac{b}{2} + \frac{3a}{4})\boldsymbol{H_2}O = (\frac{n}{2} - \frac{b}{4} + \frac{3c}{8})\mathbf{CH_4} + \left(\frac{n}{2} - \frac{a}{8} + \frac{b}{4} + \frac{3c}{8}\right)\mathrm{C}\mathbf{O_2}2 + cN\boldsymbol{H_3}$$

Where n, a, b and c are usually whole numbers.

The process of anaerobic digestion of organic matter involves a number of transformations of the macromolecules present by several micro-organisms. Six distinct conversion processes have been identified in the degradation of particulate organic material to methane by Gujer and Zehnder (1983) and Pavlosthatis and Girado-Gomez (1991).

These six processes are: (1) Hydrolysis of particulate organic material; (2) Fermentation of amino acids and sugars; (3) Anaerobic oxidation of long chain fatty acids and alcohols; (4) Anaerobic oxidation of intermediary products such as volatile acids (with the exception of acetate); (5) Conversion of acetate to methane; (6) Conversion of hydrogen to methane.

Gujer and Zehnder (1983), Sterritt and Lester (1988) and van Haandel and Lettinga (1994) place these six distinct reactions in the conversion processes into four phases which are described below. These four processes are; hydrolysis, acidogenesis, acetogenesis and methanogenesis (Veenstra & Polprasert, 1997). The fermentation process and the activity of anaerobic oxidation throughout the pond remove about 70% of the BOD_5 of the wastewater (Ramadan & Ponce, 2003).

Hydrolysis

Hydrolysis is the first step by which microorganisms utilise complex organic molecules. In general bacteria are not able to take up particulate organic material since it first has to be broken down into soluble polymers or monomers. Thus hydrolysis or liquefaction is the first step required for microbial utilisation of the complex biopolymers. In the hydrolysis process the particulate organic matter is converted into dissolved compounds of a lower molecular weight. Proteins are degraded into amino acids, carbohydrates are transformed into soluble sugars (mono- and disaccharides) and lipids are converted to long chain fatty acids and glycerine. The monomers such as amino acids, glucose, fatty acids and glycerol are directly available to the next group of bacteria (acidogenic bacteria). It has been documented by van Haandel and Lettinga (1994) and Sterritt and Lester (1988), that exo-enzymes, excreted by fermentative bacteria, are required for this process.

Acidogenesis

This is the acid forming phase since a number of organic acids are formed. The conversion of particulate organic material into soluble compounds as a result of hydrolysis, allow the fermentative bacteria to take up the dissolved compounds. In this acid-forming phase, various short-chain organic acids including butyric acid, propionic acid, acetic acid, lactic acid are formed, together with alcohols (eg. ethanol, methanol and glycerol) and ketones (eg., acetone), acetate, and mineral compounds such as carbon dioxide, hydrogen, ammonia and hydrogen sulphide gas which are used by the methanogens . This phase is carried out by a diverse group of bacteria, most of which are obligate anaerobes. However, Mudrack and Kunst (1986) and van Haandel and Lettinga (1994), state that some facultative bacteria are also involved and

metabolise organic matter via the oxidative pathway. In this way, the little dissolved oxygen that might otherwise become toxic to the obligate anaerobic organisms is utilised.

Acetogenesis

Among the products formed in the acidogenesis stage, only acetic acid (acetate), hydrogen and carbon dioxide can be converted by the methane bacteria directly into methane. Hence in this acetogenic phase, the other products of acidogenesis are transformed into acetate, hydrogen and carbon dioxide, the final products for methane production. A larger percentage (approximately 70%) of the COD originally present is converted into methane via the acetate route. The production of acetate is generally accompanied by the formation of both hydrogen and carbon dioxide, with hydrogen being more than the carbon dioxide. Collectively, the related organisms responsible for the production of acetate and hydrogen in this phase, are known as the obligatory hydrogen-producing acetogenic (OHPA) bacteria.

Methanogenesis

Methanogenic microorganisms grow slowly in wastewater and their generation time range from 2 days at 35°C to as high as 50 days at 10°C. Hence their metabolism is usually considered rate-limiting. About two thirds of methane is derived from acetate conversion by methanogenesis. The other third is the result of carbon dioxide reduction by hydrogen. (Metcalfe and Eddy, 2003; Peavy *et al.*, 1998). In this last and final phase, methane is produced from acetate or from the reduction of carbon dioxide by hydrogen using acetotrophic and hydrogenotrophic bacteria respectively.

Acetotrophic methanogenesis

$CH_3 COOH \rightarrow CH_4 + CO_2$

Hydrogenotrophic methanogenesis:

$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O$

Methanogenesis is the final stage of the anaerobic digestion process and is of vital importance for the whole process. In particular, Henze and Harremoes (1983) state that the bacteria responsible for hydrogenotrophic methanogenesis grow faster than the bacteria which utilise acetate, and so the acetotrophic methanogenesis is the rate limiting step with respect to the transformation of particulate organic material to methane.

Settling/Thickening tanks

This is an example of solid–liquid separation using pond treatment. In settling/thickening tanks, the solids settle and accumulate at the bottom of the tank. The clarified liquid (supernatant) flows out of the tank and can further be treated (Figure 2.4). The accumulated sludge is removed periodically through draw-off pipes. Another possibility of sludge removal is manually or by front-loaders after removal of the liquid column and a period of drying.

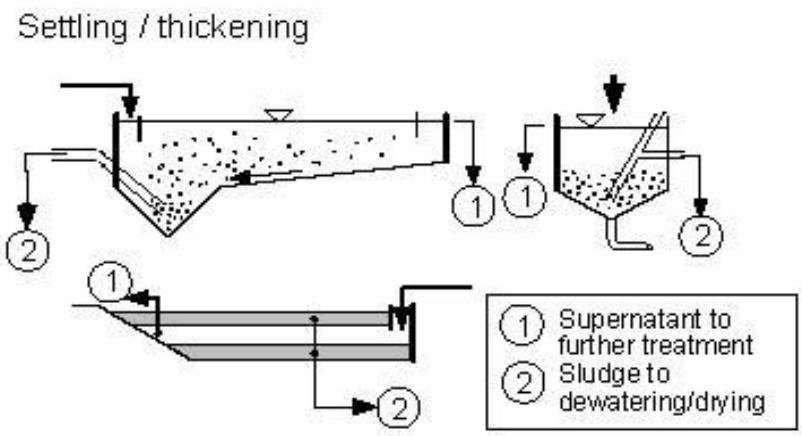


Figure 2.4: Settling/thickening tanks

Removed sludge generally requires further treatment. Settling tanks can be used for partly stabilized FS such as sludge from septic tanks. Settling tanks are not appropriate for very fresh sludge from public toilets, but may still be suitable if the fresh sludge is diluted with more stabilized sludge. The advantages of the settling tank option are; the process is simple and reliable and land requirement is low. The main disadvantage is that it is unsuitable for fresh public toilet sludge.

Drying beds

Drying beds consist of a gravel-sand filter, equipped with a drainage system. Raw or pre-settled FS is loaded on the bed and the water is drained mainly by percolation (50-80%) through the filter while part of the water is removed by evaporation (Metcalf and Eddy, 2003; Cofie, 2003; Heins *et al.*, 1998; Kim and Smith, 1997). The dewatered sludge is suitable for disposal. Further treatment for pathogen removal is necessary if the dried sludge is to be reused in agriculture. Percolate quality improves through filtration but may still require a polishing treatment. Drying beds can be used as first treatment stage and as second stage for dewatering of settled sludge removed from facilities such as settling/thickening tanks or sedimentation ponds. Public toilet sludge does not easily lend itself to dewatering on drying beds (poor dewatering characteristics). The advantages of this option are: Low moisture content of dried solids and relatively good percolate quality (compared to the effluent of settling/thickening tanks and sedimentation ponds). Technology is well known and reliable. Drying beds are simple to operate and maintain and are inexpensive to build (Kim and Smith, 1997). The disadvantages are; solids are not yet hygienically safe (unlike constructed wetlands), and have high operation and maintenance requirement, (desludging prior to new loading).

Co-composting

Composting is an aerobic bacterial decomposition process to stabilise organic wastes and produce humus (compost). Compost contains nutrients and organic carbon which are excellent soil conditioners. Composting takes place naturally on a forest floor where organic materials (leaf litter, animal wastes) are converted to more stable organic materials (humus) and the nutrients are released and made available for plant uptake. The process is slow on a forest floor, but can be accelerated under optimum conditions. The optimum conditions for composting are moisture content of about 50 %, a carbon to nitrogen ratio of about 25 to 30, and temperature of 55 ^{o}C. Because wastewater sludge is rich in nutrients, its carbon to nitrogen ratio is low (5 to 10). It is also high in moisture. Addition of dry sawdust, which is very high in carbon to nitrogen ratio (500) can adjust both the moisture and carbon to nitrogen ratio. Other waste materials that can be used for this purpose are mulched garden wastes, forest wastes and shredded newspaper. Composting can be carried out in a specially built composter, such as an inclined rotating cylinder, fed on one end with the raw materials, and the aerated product collected at the other end. As the materials are slowly tumbled over a period of about one week, they are mixed and aerated. Because bacterial decomposition produces heat, temperatures in the insulated composter can easily reach 55°C. The immature compost is then windrowed for at least 12 weeks to allow the composting process to complete, with occasional turning of the windrow.

Composting can be more simply carried out in windrows. Regular turning of the windrows assists with mixing of the materials and more importantly supply the oxygen to the bacteria. Temperatures can reach 55 ^{o}C, because compost has a good heat insulating property. Turning of the compost also ensures that all parts of the

windrow reach the required 55°C essential for pathogen destruction. Turning is required every two to three days in the first two weeks when temperature is 55°C or above. After this period frequent turning of the compost windrow is not required as less heat is generated and less oxygen is required while the compost undergoes maturation.

Pre-treated FS (e.g. solids removed from settling/thickening tanks, sedimentation ponds or drying beds) is composted together with organic solid waste (Figure 2.5). Temperatures in the heaps should reach 55-60°C and inactivate the pathogens. The produced compost constitutes a very good soil conditioner.

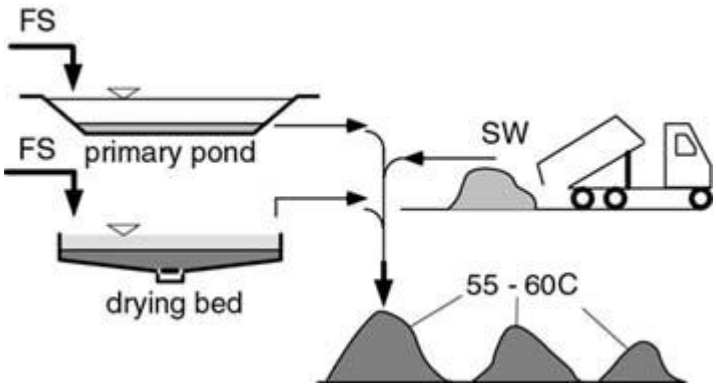


Figure 2.5: Co-composting of dried FS with organic solid waste.

Composting is a very interesting option when agricultural reuse of fecal sludge and solid waste is desired. Solid waste needs to be available in sufficient quantity and quality. A small fraction of the pit latrine sludges generated in the town of Niono, Mali (pop. = 28,000) is co-composted with sorted refuse by a micro-entrepreneur. Faecal sludges are collected manually or by tractor-drawn vacuum tanks. The compost is sold to rice and vegetable farmers (Montangero and Strauss, 1999). In Cotonou,

Benin, CREPA, initiated a co-composting scheme in October 2002, as part of action research progamme aiming at improving FS management where biosolids generated in a FS pond treatment system would be co-composted with municipal refuse (CREPA Benin, 2002). The advantage of this option is that it allows producing a good and pathogen free soil conditioner in relatively short time which is a resource recovery.

2.10 Problems encountered when co-treating FS and wastewater in waste stabilisation ponds

Where waste stabilisation ponds exist to treat municipal wastewater, and where these are used to co-treat FS, a number of problems may arise. In many cases, the problems are linked to the fact that the wastewater ponds were not originally designed and equipped to treat additional FS load. Common problems are:

- Excessive organic (BOD) loading rates may lead to overloading of the anaerobic and facultative ponds. This overloading causes odour problems and prevents the development of aerobic conditions in the facultative pond.
- Ponds may fill up with solids at undesirably fast rates due to the high solids content of FS.
- Fresh, undigested excreta and FS contain high ammonium concentrations. These may impair or even prevent the development of algae in facultative ponds. Preventative measures, such as the addition of a solids separation step ahead of the first pond, and the consideration of a maximum admissible FS load can avoid the aforementioned problems. Like in pond schemes exclusively treating FS, the (NH_4 + NH_3)-N concentration in the influent to a pond supposed to work in the facultative mode, may not exceed 400 mg/l.

Ammonia levels in faecal sludges

Average concentrations of ammonia (NH_4 + NH_3-N) in the faecal sludges collected in Accra, Ghana, range from 330 mg/l in septage to 3,300 mg/l in high-strength, rather fresh faecal sludges from unsewered, low or zero-flush public toilets (Heinss and Larmie, 1998). Hasler (1995) found average (NH_4 + NH_3)–N concentrations of 1,300 mg/l in FS from so-called watertight pits in Cotonou, Bénin. Total Kjedhal Nitrogen (TKN) levels in sludges collected from watertight pits in Ouagadougou ranged from 1,000 to 5,000 mg/l (Rehacek, 1996). NH_4 and NH_3 are in a temperature and pH dependant relationship. At 30 °C and pH 7.8, NH_3 amounts to approximately 5 % of (NH_4 + NH_3)–N. At pH 8.2, the share of NH_3 is 10 %. NH_3 is the potentially toxic component in anaerobic processes (inhibition of the methanogenic bacteria) and in facultative ponds (inhibition of algal growth). The faecal sludge treatment plant (FSTP) at Achimota in Accra comprises settling-thickening tanks followed by a series of 4 stabilisation ponds, all operating anaerobically. In the primary pond, average (NH_4 + NH_3)–N concentrations amounted to 1,000 mg NH_4-N/l during the monitoring campaigns conducted from 1994-1997.

Ammonia toxicity

Ammonia Toxicity to Methane-Forming Bacteria

Siegrist (1997) found a 50 % growth inhibition of methane-forming bacteria in digesters treating sludge at wastewater treatment plant at NH_3-N/l concentrations of 25-30 mg/l. Whether these results equally apply to anaerobic ponds is not yet clarified.

Ammonia toxicity to algae

Tolerance limits for *Chlorella vulgaris* and *Scendesmus obliquus* are 6 and 31 mg NH_3-N/l, respectively (Kriens, 1994). These algae commonly form an important share of the algal biomass in facultative ponds. Some algal species are reportedly able to adapt to and withstand concentrations of up to 50 mg NH_3-N/l under specific conditions (Mara and Pearson, 1986). In the Achimota FSTP in Accra, excessive ammonia NH_3-N concentrations of 50-70 mg/l in ponds 2 through 4 were the likely cause for the suppression of algae and, hence, of the development of facultative pond conditions with an upper, aerobic layer. The (NH_4+NH_3)-N concentration in the influent to a pond supposed to work in the facultative mode, should not exceed 400 mg/l (Heinss and Strauss, 1999). Possible methods to counteract ammonia toxicity to algae include intermittent, forced surface aeration to oxidize, lime dosing and recirculation, or a mixture thereof. The aim is to lower the ammonia concentrations and, hence, to eliminate NH_3 toxicity effects.

2.11 Sludge Dewatering Process

Sludge dewatering is a unit operation to reduce the moisture content of sludge. It is only justified to apply sludge dewatering if the cost for disposal is very high or if transportation of sludge is required over long distances. In Netherlands, cost of disposal of sludge, ranges between US$50 – 75/ton wet sludge, (Veenstra, 1999). However sludge dewatering generally aims to: Facilitate further handling of sludge, reduce transportation costs, increase the energy content of sludge for incineration, make the sludge more suitable for co-composting, reduce groundwater pollution at landfill sites or lagoon sites and allow cost effective storage of dry solids.

Dewatering processes

A wide variety of dewatering processes (Haandel and Lettinga, 1994) are available which include

i) Natural dewatering methods: these depend on evaporation and percolation which include sludge lagoons and sludge drying beds.

ii) Mechanical dewatering systems which involve filtration, capillary action, vacuum suction and centrifugal forces. Devices commonly used include, vacuum filters, centrifuges and pressure filters.

iii) Drying systems that accelerate evaporation rates in drying equipment by injecting heat energy.

Sludge dewatering methods can achieve dry solids (DS) levels between 15 and 30% while drying technologies start with input feed of 20-30% and raise the dry solids content to 80 – 89%. During dewatering, forces are exerted on the water fraction of sludge in order to separate the solids and release major water fractions (i.e. free, colloidal and capillary water). However, particle water will always remain associated with the sludge and can only be separated from the solid by intensified drying, (Veenstra, 1999).

Natural dewatering

Natural sludge dewatering depends on environmental and climatic conditions for its operation. It does not require any external source of power supply. It has been found to achieve high dry solid content of sludge and pathogen inactivation. Typical applications of this dewatering are drying lagoons and drying beds.

Drying lagoons

Sludge is allowed to dewater and dry for several months up to several years before final disposal. The dry solid (DS) levels of the final sludge may reach up to 15 – 25%. Sludge is actually stabilised before it is added to the lagoon to minimise odour release and improve the dewaterability. It has been found out that non-stabilised sludge and lime conditioned sludge are not good to be disposed of, in lagoons.

Design loading rate is given as;

2 – 4 per capita load/m^2/year, or 1.5 – 3 m^3 sludge/m^2/year, or 10 – 20 kgDS/m^2/year.

The negative effect of precipitation on sludge dry solids is strongly affected by the runoff. It is envisaged that at DS content above 10%, surface water decanting can be realised easily to increase the runoff; at DS level between 10-20% trenching at the top may improve runoff, while at DS greater than 20%, rainwater directly infiltrates into the sludge because of the occurrence of surface cracking.

The drying capacity (D) of a lagoon (Figure 2.6) is given by

D = E + R + F – P (mm/year)

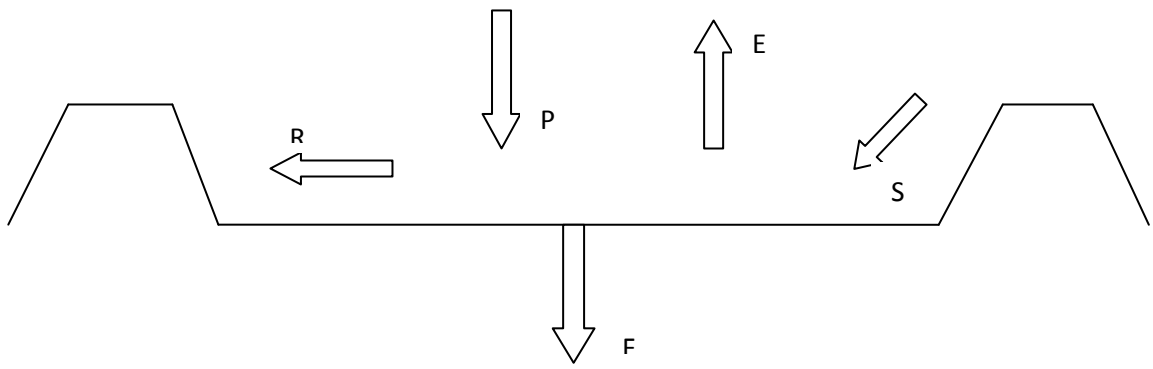


Figure 2.6: Water balance of a sludge-drying lagoon

where

F = the drainage, E = the evaporation, R = the surface runoff and P = the annual precipitation rate.

From the water balance, the filling depth of a lagoon can be calculated as

$h = (S - D) \times T$

where; h = filling depth of a lagoon (in mm); S = sludge loading rate (in mm/year); D = drying capacity of the lagoon (mm/year); T = filling period (in years). Evaporation rates of sludge may be assumed to equal those of open waters. At sludge concentration above 15% DS evaporation may be reduced. Infiltration of water depends on the soil (media) characteristics, but is generally low due to clogging at the bottom. Infiltration reduces as loading continues. Sandy soils generally have high infiltration while impermeable clay layers have low or negligible infiltration. To avoid groundwater pollution, an impermeable layer made of clay or plastic sheet at the bottom of the lagoon may be provided. Drains may also be constructed at the bottom of the lagoon to abstract the drainage water. This drainage water may be heavily contaminated, with pathogens, heavy metals and micro-pollutants which pose threat to local groundwater resources. As evaporation is the major process for removing water from sludge, climatic conditions such as precipitation, evaporation, humidity and temperature strongly affect the effectiveness of sludge lagoons.

Sludge drying beds design

They are most commonly used for small and medium sized communities in tropical environments where the evaporation rates are high. Drying beds simply consist of shallow sand beds with gravel support and drains at the bottom. The gravel bed is kept about 30 cm thick while the sand bed is about 20 cm thick. The particle sizes of the sand is about 0.1 – 2.0 mm while that of the gravel is between 5.0 – 15.0 mm. Obarska-Pempkowiak *et al.*, (2003) used sand particle size of 0.8mm but separated the gravel layer to course gravel (8-16) mm with 30 cm depth and medium gravel (2-4)

mm with 20 cm depth. Though there are slight differences in the arrangements, the principles are the same. Usually sludge is spread to a thickness of about 15 – 30cm over the sand bed. Where drying beds are very big they are often subdivided into smaller units to facilitate manual sludge removal. One unit may be 5 – 8 m wide and 15 – 50 m long to improve sludge removal. Drying times of between 1- 2 weeks have been reported in warmer countries especially in West Africa while in places like Northern USA, as minimum as 5 drying cycles a year have been reported for open beds and about 7 – 10 cycles a year for covered beds (Veenstra, 1999). Climatic variation and characteristics of sludge generally do affect the specific area requirements. This is clearly shown in (Table 2.6) below.

The performance of sand beds is often more meaningful and comparable if stated in solids loading expressed as kg DS/m^2/year rather than just m^2/capita. The average drying bed load reported from USA is about 100kgDS/m^2/year. However, between 100 – 420 kgDS/m^2/year has been reported by Pescod, (1971) in Thailand and Heins *et al.* (1998) in Ghana.

Table 2.6: Area requirements for open sand beds for sludge drying

Type of sludge	**Region**	**Area(m^2/person)**
Secondary sludge	Netherlands	0.16 - 0.33
Secondary sludge	South Africa	0.03 – 0.04
Secondary sludge	India	0.05 – 0.10
Digested sludge	India	0.05 – 0.10
Digested sludge	USA	0.11 – 0.28

Source: Arceivala, 1998

Sludge drying on open sand beds may be difficult during the wet season or winter periods. It is advisable to cover the beds or provide roofing while the sides are kept open for natural wind movements to enhance evaporation. Providing a sludge holding lagoon for temporal storage, provision of a combination of a bed cover and holding

pond as well as providing for stand-by mechanical dewatering systems are alternative solutions suggested by Veenstra (1999). However, all these solutions add extra costs to the operation and maintenance.

With respect to limitations on land and other associated disadvantages, this system is considered not acceptable, (Al-Muzaini and Hamoda, 1998) in places like Kuwait. However, it is recommended by Pescod (1971), Strauss *et al.* (1997) and Heinss *et al.* (1998) for low income countries like Ghana, Mali and Burkina Faso where land is available.

Determination of dewatering rate of sludge drying beds

Experience is required for obtaining optimal design at a reduced cost. The factors that affect the drying time which should be given consideration are: Extent of free drainage, climatic factors (evaporation rates from sludge and rainfall) and permissible moisture content of final outgoing sludge. The overall drying time can be computed using a simple mass balance of all incoming and outgoing moisture (Figure 2.7). The moisture to be evaporated per unit area is given by

$$q_e = (1-f_i) \times q_i + (1-f_r) \times q_r - q_d \text{ ---------- } \qquad \text{Eqn 2-1}$$

where q_e = moisture to be evaporated

q_i = moisture initially present in sludge

q_r = moisture received via rainfall

q_d = moisture remaining in final sludge

$f_{i,r}$ = fractions of qi and qr respectively that is drained from the bed, thus the $(1-f_i)$ and $(1-f_r)$ are the fractions remaining in sludge.

The model is applicable when the evaporation dominates the drying time required.

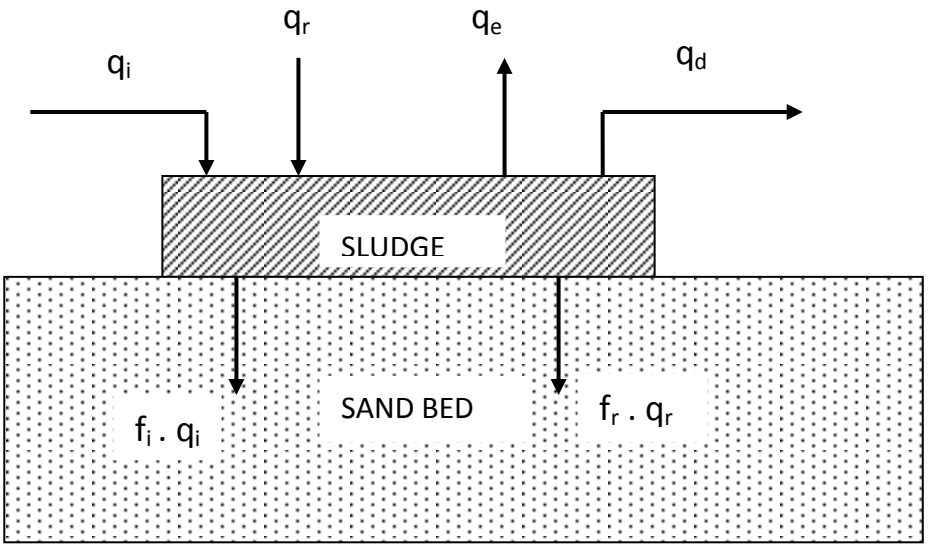


Figure 2.7: Model for estimating the overall drying time on a sludge bed.
Source: Arceivala (1998).

Generally, the above q terms can be converted from mass/area to volume/area which is m^3/m^2, simply meter (m) or milimeters (mm) (used for rainfall data). If the evaporation rate from the sludge surface is less than the rate determined from the open pan evaporation, a suitable reduction factor f_e to account for reduced evaporation is incorporated in the formula for calculation of the drying time.

$$T = \frac{q_e}{f_e \times E_w} \qquad \text{---------- Eqn 2-2}$$

Where T is the drying time in months and E_w is the evaporation rate from free water surface (mm/month). Combining Eqns. 2-1 and 2-2, the required drying time t is given as

$$t = \frac{(1 - f_i) \times q_i + (1 - f_r) \times q_r - q_d}{f_e \times E_w} \qquad \text{---------- Eqn 2-3}$$

From equation 2-3, it can be understood that the overall drying time is affected by:

- The effective evaporation rate.

- The rainfall received (which is zero for covered beds).
- The permissible moisture content of the dried sludge.

The sludge characteristics which affect the f coefficients differ for different sludges based on the characteristics. Typical values for f coefficients for sludges (between 1 – 4% DS) and at drying beds of 15 -30 cm are given in (Table 2.7) below.

Table 2.7: Typical f coefficients for stabilised and anaerobic digested sludge

Coefficient	stabilised sludge (extended aeration)	anaerobic digested sludge
f_i	0.45 – 0.65	0.8 – 0.9
f_r	0.43	0.73
f_e	0.78	0.78

Source: Arceivala (1998)

According to Arceivala (1998), coefficients f_i and f_r are affected by the nature of sludge. The higher the organic matter contents of the sludge, the lower the f coefficients. Furthermore, the presence of hygroscopic fibrous materials helps moisture retention and therefore reduces the f coefficients. There is an inverse correlation between the fi coefficient and the specific sludge resistance, Arceivala, (1998). The drainage fraction f_i can simply be determined by laboratory-scale sand column tests simulating the performance of a sludge drying bed.

2.11.1 Factors influencing dewatering characteristics

Sludge particle size

One of the primary objectives of wastewater treatment is the removal of solids. Once removed, the resultant sludge must be treated and disposed of in some innocuous manner Karr (1976). In Africa and many developing countries, the kind of sludge that should be disposed of is the faecal sludge removed from on-site sanitation systems,

(Strauss *et al.*, 2003). In many wastewater facilities the bottlenecks of the sludge handling system is the dewatering operation. It is also the least understood handling operation. In the case of faecal sludge (Cofie *et al.*, 2006), indicated that as a result of its high variability, the design for its dewatering should be based on case by case study. Many factors that influence the dewatering characteristics of sludge have been reported in the literature. Among these are cellulose content, pH and particle charge, organic content, filtrate viscosity, alkalinity, solids concentration, grease content, nitrogen content and conditioning (Caron and Mazzola, 1969; Everet, 1974; Heukelekian and Wisberg, 1956; Coackley and Dewey, 1966). Other factors include, type of sludge, compressibility coefficient, mechanical strength of particles, porosity, mixing, biological degradation and particle size, (Hatfield, 1969; Coackley and Wisberg, 1971; Gale, 1972).

Clogging

In rural areas in Poland, like in many other countries, the most popular facility for individual domestic sewage treatment and disposal is a septic tank and soil absorption system. Unfortunately, the latter is often disturbed by clogging, especially in fine and loamy sands. Clogging is a well known phenomenon in filters. The clogging layer occurs in filters, particularly with fine filtering media, during sewage (including septic tank effluent) treatment. It is unfavourable because of negative impact on hydraulic conductivity. On the other hand, a well developed biofilter enhances the treatment efficiency of the system. There are several hypotheses explaining the clogging process. Many significant conditions and factors were found by other authors. According to the literature (Blaz˙ejewski and Murat-Blaz˙ejewska, 1997), the main causes of clogging are:

- Mass accumulation within soil pores, particularly suspended solids from sewage,
- Precipitation and deposition of some materials, for example calcium carbonate (CaCO3),
- Microorganisms growing within soil pores,
- Deposition of side products of bacterial metabolism and extracellular products of bacterial cells, particularly slimes and EPS (extracellular polymeric substances).

Avnimelech and Nevo (1964) revealed that a low C:N ratio suppressed clogging. They stated that clogging was correlated with polyuronide concentration, and therefore was a function of polysaccharide production. Bouma (1979) reported that biological clogging was due to accumulation of suspended solids and biological growth in anaerobic conditions. Kristiansen (1981a) suggested that bacterial cells might be important in clogging by producing slime that would link soil particles together and reduce hydraulic conductivity. He stated that extracellular polymeric substances (EPS) play a very important role in the clogging process. Kristiansen (1981b) concluded that clogging is initiated by the deposition of suspended solids, which are linked together with soil particles by biologically produced material. Siegrist (1987) suggested that the main causes of clogging are suspended solids and BOD. These parameters were used in his empirical mathematical model of clogging. Siegrist *et al*. (1991) found out that humic substances might have significant impact on the clogging process. These substances are produced as a result of biodegradation of organic material. Some authors suggested that the clogging process is accelerated by anaerobic conditions (De Vries, 1972), and low temperature (Bihan and Lessard, 2000).

The soil clogging was investigated by many authors in various conditions. Because of specific factors, the results and conclusions were often not complete. For example: experiments with artificial sewage did not consider impact of solids. Some researches neglected the impact of live organisms. One of the aims of this research was to reduce clogging and enhance dewatering through the application of different filter particle sizes, different loading rates and different mixing ratios of sawdust.

Sand media characteristics

The media characteristics of sand based treatment systems are among the most important design criteria. The primary sand media characteristics affecting filtration performance are the effective grain size and uniformity coefficient (u_o) (Crites and Tchobanoglous, 1998). These characteristics tend to affect the retention time of liquid passing through the media and the potential for clogging.

Particle size

The larger the grain size, the faster the wastewater moves through the sand and the more the wastewater that can be filtered. However if the grain size is too large, treatment efficiency will be reduced. Boller *et al.* (1994) observed larger breakthroughs of unoxidized matter due to short retention times and instantaneous lack of oxygen when applying relatively large hydraulic loads to filter media with course grain size, especially above 1mm. The ideal sand for intermittent sand filters receiving domestic wastewater is sand with an effective size between 0.3 mm and 0.5 mm (Crites and Tchobanoglous, 1998; Ohio State University, 1999). According to Converse and Tyler (2000), clogging becomes a major concern when using sand with an effective size less than .3 mm and therefore recommended that filters using this

size of sand be lightly loaded (<56.25 litres/m^2/d). The most important feature of granular media is not the grain particles but rather the pore space in the media where suspended solids are trapped, microorganisms grow, and air and water flow bringing about treatment (Emerick, 1997).

Uniformity coefficient (u_o)

Uniformity coefficient is a numeric estimate of how sand is graded and is a dimensionless number. The uniformity coefficient is calculated by dividing D_{60} (the size of screen opening where 60% of a sample passes and 40% retained) by D_{10} (the effective particle size-that size of screen opening where 10% of a sample passes and 90% is retained). The larger the uniformity coefficient, the less uniform the sand (Eliasson, 2002). A uniformity coefficient of 4 or less is recommended for all filter media (National Small Flows Clearinghouse, 1997; Crites and Tchobanoglous, 1998; EPA, 2002). This recommendation is intended to avoid clogging at higher loading rates (Darby *et al.*, 1996).

2.12 Biosolids Produced from Drying Beds

In recent years, the fate of human and animal pathogenic microorganisms as a potential pollutant of the environment has received increased attention. Substantial quantities of these compounds and their metabolites are excreted, flushed down the drains and discarded as waste, or left over in animal feedlots. After passing to the sewer, several of these compounds are not adequately eliminated by the methods that are currently used in sewage treatment. Substantial quantities of biosolids and livestock manure end up on agricultural lands. Effective sanitation of the environment, particularly in areas of livestock production which can be source of spreading of

diseases, plays an important role in the prevention of infectious diseases. In this respect special attention should be paid to the disinfection of infected farm animal excrements (Venglovsky *et al.*, 2005*)*.

All around the world, people both in the rural and urban areas have been using human excreta for centuries to fertilise fields and fishponds and to maintain or replenish the soil organic content, that is, the humus layer. Until today, in both agriculture and aquaculture this continues to be common in China and Southeast Asia as well as in other places in Africa (Cross, 1985; Timmer and Visker, 1998; Visker, 1998; Timmer, 1999; Strauss *et al.*, 2000). Uses of biosolids have led to a strong economic linkage of urban dwellers (food consumers as well as waste producers), and the urban farmers (waste recyclers and food producers).

Biosolids generated from drying beds has high content of helminth eggs (Cofie *et al.*, 2006). Hence, to inactivate the eggs and other surviving pathogens for use in agriculture, the biosolids is used for co-composting with the biodegradable fraction of municipal solid waste. Wood Chips (Sawdust) nonetheless will also be an appropriate addition to the dewatered solid (biosolid).

Re-Use considerations

Organic matter, together with phosphorus, nitrogen and potassium as major plant nutrients are found in appreciable quantities human excreta (Table 2.7). Average nutrient contents of plant matter and cattle manure are also included for comparison. Faecal sludges, if adequately stored or treated otherwise, may be used in agriculture as soil conditioner to restore or maintain the humus layer or as fertiliser. In considering

the various options for the treatment of faecal sludge/septage, it is very important to consider its re-use. This is because: (i) Commercialisation of treated sludge can generate revenues, (ii) landfill space is not utilised for sludge disposal, (iii) faecal sludge, in contrast to sewage sludge has little chemical contamination and can therefore be considered as a valuable resource especially for soil conditioning. In the long-term the re-cycling of waste is always the preferable option. Sludge should only be disposed of, if there is neither the need for a market or soil conditioning, nor if the additional expense for providing a product suitable for agricultural use cannot be justified (Klingel, 2002).

Use of biosolids

After dewatering of faecal sludge, the biosolids produced can be applied to soil. Land application is the spreading of biosolids on or just below the soil surface. Lands to which biosolids are applied are:

(i) agricultural land,

(ii) forest land,

(iii) disturbed land and

(iv) dedicated land/disposal sites. In all four cases, the objective is to provide further biosolids treatment. Sunlight, soil microorganisms and dessication combine to destroy pathogens and many toxic organic substances.

Trace metals are trapped in the soil matrix and nutrients are taken up and converted to useful biomass (Metcalf and Eddy, 2003). In Ghana, the case of agricultural use is the most common, eg. Citrus and vegetable farmers. This is beneficial because the organic matter improves soil structure, tilth and water holding capacity, water infiltration and soil aeration (Metcalf and Eddy, 2003; Anderson, 2001). This is because of the

presence of the following: macronutrients (N, P, K) and microorganisms; resource value of FS (Table 2.8).

Table 2.8: Human excreta: Per capita quantities and their resource value

	Faeces	Urine	Excreta
Quantity and Consistency			
Gram /cap/day (wet)	250	1200	1450
Gram/cap/day (dry)	50	60	110
Including 0.35litre for anal cleansing, gram/cap/day (wet)			1800
m^3/cap/year (upon storage and digestion for ≥ 1 year in pits or vaults in hot climate)			0.04 – 0.07
Water content [%]			50 - 95
Chemical Composition	% of dry solids		
Organic matter	92	75	83
C	48	13	29
N	4 – 7	14 – 18	9 -12
P_2O_5	4	3.7	3.8
K_2O	1.6	3.7	2.7
Different organic materials	% of dry Solids		
	N	P_2O_5	K_2O
Human Excreta	9 – 12	3.8	2.7
Plant Matter	1 – 11	0.5 – 2.8	1.1 – 11
Pig manure	4 – 6	3 – 4	2.5 – 3
Cow manure	2.5	1.8	1.4

(Source: Strauss, 1985)

2.13 Sawdust Production and Faecal Sludge Treatment in Ghana

Sawdust

Sawdust is an organic waste generated usually from sawing of timber. Since sawdust is an organic material, it is obvious that man has tried to find profitable agricultural uses for this waste (Arends, 1985). Several uses of sawdust have been found including composting, as kennel bedding, in meat, fish, and vegetable markets and workshops etc. Sawdust used as bedding for animals or poultry absorbs liquid manure which contains 90% of the total nitrogen in manure, in addition to carrying the solid manure. If 2.5% of superphosphate is added, this nitrogen will not evaporate and a good

fertilizer can be obtained (Forest Products Laboratory, 1961). In temperate climate, the deep litter poultry system is an economic method of converting sawdust and shavings into an usable compost. The bacteria grab the fierce ammonium carbonate to form bodies and satisfy their needs, breaking down the celluloses and hemicelluloses and some of the lignins in the bedding material producing just as much heat as if it was burnt using the same amount of oxygen. The heat produced dries the poultry manure and becomes a useful fertilizer (Hills *et al.* 1963; Arends, 1985). Sawdust when well decomposed is a good fertilizer and a soil conditioner (Organic gardening and farming staff, 1976). Carbon – nitrogen ratio (C:N) of green sawdust is very high 500:1 or higher depending on the kind of wood. Good compost is about 30:1 C:N ratio (Bollen and Glennie, 1959). Gardeners have to add nitrogen fertilizer such as cottonseed meal, blood meal to improve the carbon-nitrogen ratio. Many users did not apply a nitrogen supplement but were still satisfied with the results (Organic gardening and farming staff, 1976). Lignin is the part of sawdust that has good fertilizing quality. The carboxyl groups of this high-molecular aromatic compound have an ability to part with hydrogen and retain absorbed ions of ammonia, calcium, magnesium, potassium, and other bases. Thus lignin prevents soil nutrients from leaching so that they are available to plants (Wilde, 1960). Wood waste therefore has a distinct advantage over most other agricultural wastes, because it continues to bind nitrogen for a longer time during decomposition due to its lignin content (Bollen and Glennie, 1959; Golueke 1978). The high lignin content of wood produces more humus than most other organic materials. Humus is necessary for the crops (Huber *et al.*, 1968), especially in the tropics where there is great demand for humus. A small-scale application of sawdust is litter for mice, guinea-pigs, rabbits and others. It is also used to cover floors of taverns, garages, machine-shops etc (Viehfutter, 1974). Red sawdust

is used for garden decoration (Kamstra and Shideler, 1978). Chemicals found in sawdust however, may affect the microorganisms that break it down as fertilizer. Lignin in the bedding material, produces just as much heat as if they actually burnt it, injuring the bacterial present (Vos, 1979).

Green sawdust will give a temporarily noxious effect because of aerobic fermentation (Fung, 1981). The acids produced in this way can be neutralized through the addition of 0.25kg of limestone per kilogram of sawdust or its equivalent of NH_3 otherwise the continued production of the acid may affect microbial activities. Lignin increases NH_3 content thereby affecting microbial activities (Arends, 1985). In natural environments, ammonia (NH_3) chemically hydrolysed or produced by bacteria can be deleterious to other organisms. Added ammonia-generating chemicals also facilitate the inactivation of pathogens in *e.g.* excreta or sewage sludge (Ghiglietti *et al*., 1997; Vinnerås *et al*., 2003).

Sawdust production in Ghana

Studies conducted by Institute of Industrial Rsearch of Center for Scientific and Industrial Research, in six regions of southern Ghana estimated sawmill residue generation in 1995, as 780,682 m^3. Over 200,329 m^3 of this constitute sawdust (Lartey *et al*. 1999; TIDD, Forest Commission, 2008). No economic use has been found for this huge volume of the sawdust yet. The only known use is its application as bedding for poultry in poultry farms in Ghana and some other African countries. This quantity accounts for less than 5% of what is being generated. The rest is thrown indiscriminately into the environment by the processing companies. Since the material occupies large area of land surface, the practice has been to burn it continuously in

order to get enough space for continual disposal. A large proportion is left unattended to for long time and gets decomposed while runoff washes large volumes into water bodies which may cause eutrophication. The timber precessing companies burn the sawdust produced from their operations, producing large volume of smoke which is a source of hazard for the people resident in the affected communities and workers as well. A visit to Kaase, Angloga, Ahwiaa, and Kwadaso, all suburbs of Kumasi where industrial wood processing takes place will witness production, accumulation, disposal and burning of large volumes of sawdust (See plates in Appendix B). Sawdust production and burning is common in almost all communities in Ghana. Furthermore there have been many incidence of fire outbreak through the burning of sawdust in certain parts of Kumasi destroying properties worth millions of cedis (personal observation). Notwithstanding the above environmental impacts associated with its production, information in the literature have shown that it could be useful in its application as a physical conditioner to the dewatering of sludge generated from wastewater treatment plants by improving the filterability and the dewaterability (Ying-Feng *et al*., 1999; Jing *et al*., 2001). It has also been found to combine well with sludge from wastewater treatment plants and faecal sludge to make very good compost which improves soil fertility (Shuval *et al*., 1981; Cofie, 2003).

The vision of soil scientists, agriculturists, sanitation experts to use faecal sludge for compost preparation is to a large extent hampered as a result of difficulties encountered in the dewatering. However, it is envisaged that the dewatering of faecal sludge can be improved by mixing with sawdust. This is based on sludge dewatering studies conducted by other researchers elsewhere (Ying-Feng *et al*., 1999; Jing *et al*

2001). What are left to be known are the mixing ratios of faecal sludge and the sawdust that will most improve dewatering time and produce quality biosolids.

Faecal sludge treatment with physical conditioners (sawdust)

Faecal sludge like other wastewater treatment plant sludges can be treated with sawdust and other physical conditioners like wheat dregs to improve its dewaterability. Addition of physical conditioners to the sludge from wastewater treatment plants (WWTP) before dewatering generally offers the advantages of improving sludge filterability, reduce chemical conditioning requirements, increase the solids content of the cake, and enhance cake detachment from filter media (Englande and Reimers, 1981; Jing *et al.*, 1999). Physical conditioners are inert materials and function as skeleton builders because; they create a permeable and rigid lattice structure within the sludge which can remain porous under high pressure filtration to allow water to be forced out under pressure. Fly ash, hydrated lime, diatomaceous earth, cement kiln dust, bagasse and pyrolysed domestic refuse are among the skeleton builders proposed in the literature to increase the dewatering rate (Zall *et al.*, 1987; Benitez *et al.*, 1994; Hwa and Jeyaseelan, 1997; Smollen and Kafaar, 1997).

They are normally much less expensive than chemical conditioners. Ying-Feng, *et al*. (2001) investigated the effect on sludge dewatering by adding the organic solids waste (wood chips or wheat dregs) to sludge after chemical preconditioning (with ferric chloride or alum). Wood chips have a low density and rigid structure, and are highly inert material as well. According to Zall *et al*. (1987) and Banitez, *et al*. (1994), wood chips enhanced sludge dewaterability probably for the same reason as when fly ash is

added; both act as a skeleton builder for conditioning sludge. These two studies concluded that poorly filterable sludges yielding highly compressible filter cakes can be conditioned with a skeleton builder to produce sludge cakes with a more rigid and incompressible structure which therefore maintains high porosity and subsequently elevates filterability. In addition to the aforementioned, wood chips are also dry, organic, and biodegradable and they are the most widely used bulking agent to provide structural support and to increase the porosity of the dewatered sludge for effective aeration in sludge composting. Moreover, conditioning wastewater sludge with wood chips, not only could improve sludge dewatering but also could produce a dewatered biosolid suitable for use in both composting and incineration. Studies conducted by Jing *et al.* (1999) showed that adding as much as 170% of wood chips (particle size 1-5 mm) decreased specific resistance (SR) by 60% which indicates an efficient improvement in dewaterability. In addition, a trace amount of mass residue (MR) as a result of nearly complete detachment of the cake from filter cloth was achieved at the same dose. Hathaway and Olexsey (1977) reported that using pulverized coal as a sludge conditioner could improve vacuum filtration efficiency and reduce sludge incineration costs. Wood chips and wheat dregs are cheaper than pulverized coal because they are waste materials, although their energy content may be lower than coal. Furthermore, wood chips are the most widely used bulking agent, and wheat dregs could be appropriate to act as amendments. Both can provide structural support in the dewatered sludge for more effective aeration during composting. Applying these two materials as sludge conditioners can also be a practical way of recycling and reusing solid waste. Various addition strategies may be suggested, based on the above findings, for different sludge management schemes. Due to the increased weight of the filter cake, high doses of skeleton builders (>100%)

should not be used if the dewatered sludge is to be directly disposed of in a landfill. If the dewatered sludge is to be incinerated, a high dose range is recommended both to enhance filtration and to eliminate the need for an auxiliary fuel in incineration. Some investigators (Vrilakis *et al*, 1999) recommended addition of wood chips dose ranging from 300% to 500% to sludge after dewatering, to achieve successful composting. Ying-Feng *et al*. (2001), suggest that applying wood chips or wheat dregs within this dose range to sludge before dewatering can greatly improve sludge filterability, while eliminating the requirement for supplemental bulking agents.

This research seeks to come out with criteria developed from the use of low cost technologies and local materials to improve on the dewatering of faecal sludge using unplanted filter beds. This will help in dewatering the large volumes of faecal sludge generated in Ghana and other developing countries in order to improve sanitation delivery. The biosolid produced from these technologies will further be hygienised for use in crop production. Information from literature and methods intended to be applied in this studies will make faecal sludge that would have been a hazard in most communities in developing countries become a useful material for improving the livelihoods of the people.

CHAPTER THREE

3 METHODOLOGY

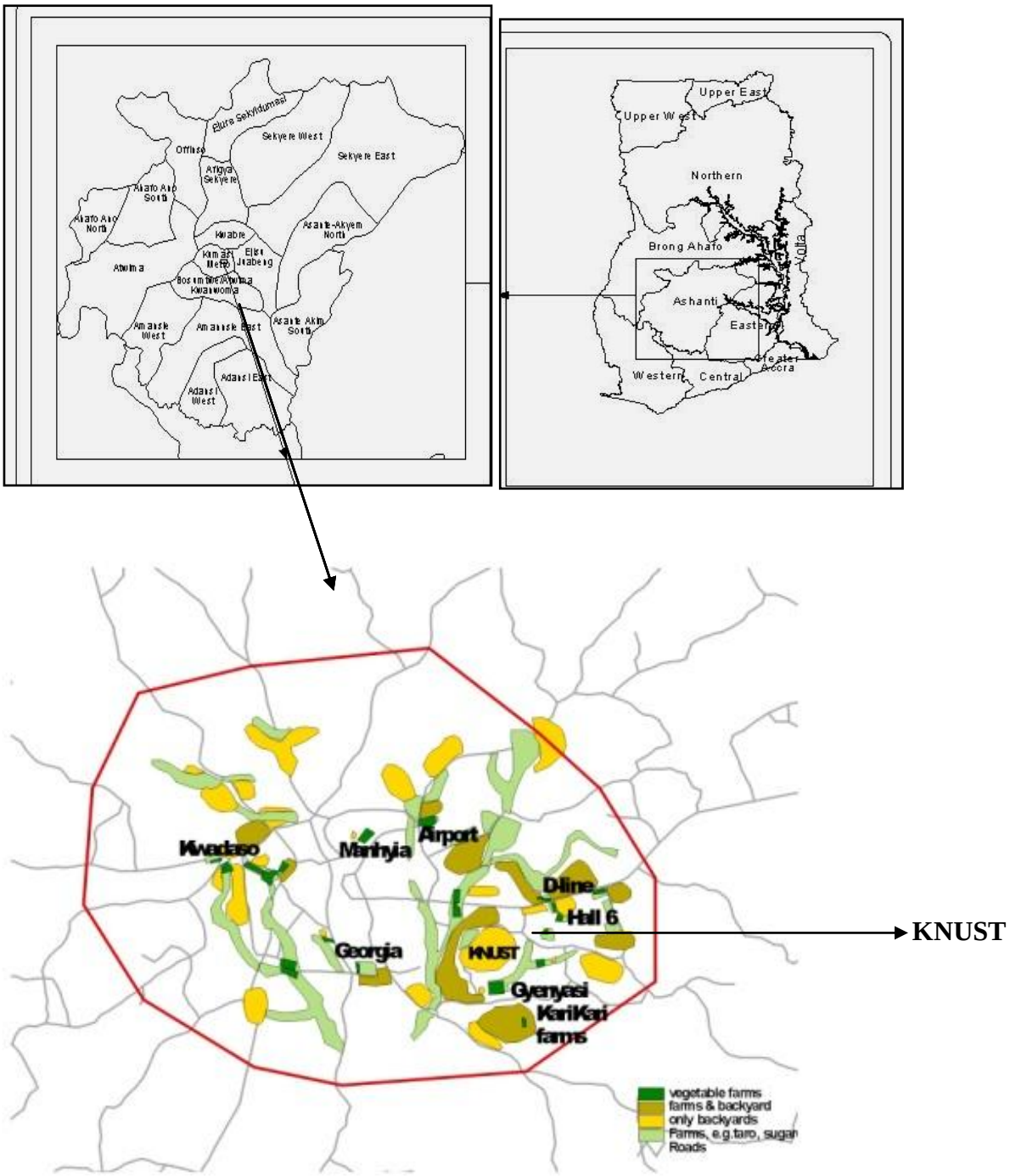
3.1 The Study Area

Ghana lies at the shores of the Gulf of Guinea in West Africa (4° 44'N and 11° 15'N; 3 ^{o} 15'W and 1° 12'E). It borders Burkina Faso to the North, Togo to the East and Côte d'Ivoire to the West. It has a population of about 19 million, with annual growth rate of 2.7%. About 44% of Ghana's total population lives in urban areas. Some urban centres have annual growth rates as high as 6%, more than twice the country's average rate (Ghana Statistical Services (GSS), 2002).

The overall national population density is 79 persons/km^2 (GSS, 2002). Agriculture is the mainstay of the Ghanaian economy, contributing 36% of the gross domestic product (GDP) and employing 60% of Ghana's labour force. The average annual per capita income of those employed in agriculture is estimated at US$390. Annual rainfall ranges from 800 mm in the coastal areas to 2,030 mm in the south western rainforests (Agodzo, 1998).

3.2 Location

The setup for the research was raised near the Water Quality Laboratory of the Civil Engineering Department of KNUST in Kumasi (Figure 3.1). All the faecal sludge used for the dewatering cycles was collected from Dompoase treatment plant also in the city of Kumasi. The city of Kumasi which currently has a metropolitan status is the capital of Ashanti region which is uniquely located at the centre of Ghana. The metropolis is located between latitudes 6^o35 N – 6^o40 N and longitude 1^o30W – 1^o35 W. The total area is approximately 254 Km^2.



Kumasi Metropolis

Figure 3.1: Maps showing location of project site (KNUST in Kumasi)

3.3 Climate

Kumasi is the capital town of the Ashanti Region and the second largest city in Ghana with a population of about one million (GSS, 2002). The city of Kumasi falls within the wet sub-equatorial climate type with the rainy season running from late February to early July and from mid-September to early November. The rains begin as late as March and it is common to experience sporadic rainfall in late November. The annual rainfall ranges between 15.1 mm in January, and 214.3 mm in June. The average temperature range is 27.8^{o}C in February to 24.3^{o}C in August (Appendix A). Kumasi is located in the forest belt of Ghana and the total average rainfalls for the wet and dry seasons in Kumasi are as follows: Main wet 680 mm, short dry 220 mm, short wet 350 mm and main dry 160 mm.

3.4 Preparation of Setup

Shelter

A wooden shelter of size 3.5 m wide by 4.5 m long and 2 m tall was raised and roofed with corrugated iron sheets. Under it was a constructed wooden platform of 2 m wide, 3 m long and 0.5 m tall which served as a bench for the drying beds. Nine filter bed plastic containers with each having a size of 0.85 m long 0.175 m diameter were moulded and used as filter bed containers. Wooden stands were also constructed to hold the filter beds in position (Plate 3-1).

Filter material collection and preparation

Coarse gravel of average diameter between 10-15 mm and fine gravel of average diameter of 5-10 mm were used as base support for the filter medium (sand). These

were collected from a rock quarry company, about 40 km from project site. The sand used was also collected from a community about 50 km from project site.

Plate 3- 2: Project setup showing filter beds being held in position by wooden stands

Filter material and drying bed preparation

Sieve analyses

The sand which was used as filter material was sieved into ranges of 3 different particle size ranges (0.1 - 0.5; 0.5 - 1.0; and 1.0 - 1.7) mm by sieve analyses (Appendix A) (National Small Flows Clearinghouse, 1997; www.scribd.com/doc/31286746/Sand-Testing-GFN, accessed 12/02/2010). These constituted the filter media (FM1, FM2 and FM3). Effective sizes (D_{10} and D_{60}) of the sand were determined in order to get the Uniformity coefficient (U_o), of the different kinds of filter media (Plate 3-2). This was obtained from the following relation.

$U_o = D_{60}/D_{10}$

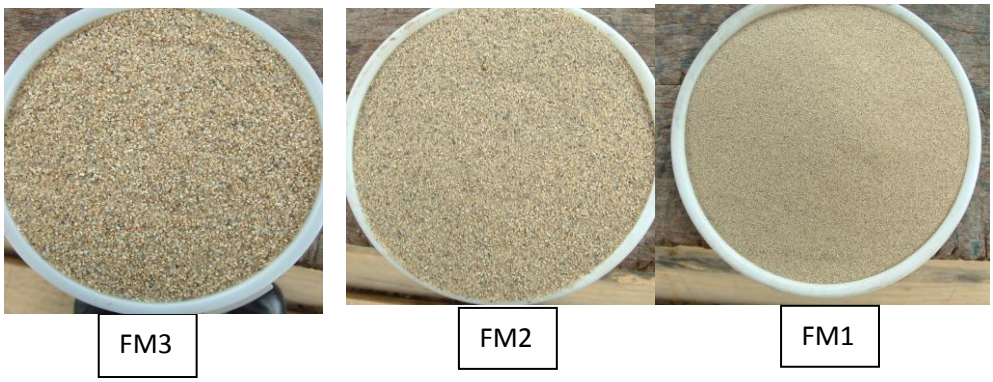


Plate 3- 3: Different filter media used for the dewatering of FS

The permeability of the three types of media obtained by the sieve analyses were determined using constant head permeameter. The base of the filer bed was filled with the coarse gravel to a depth of 15 cm, followed by the fine gravel to a depth of 10 cm. These were used to ensure a good support for the filter media, the sand, which was also filled to a depth of 20 cm. Nylon mesh (0.4 mm), was placed on the sand on which the FS was poured, to act as interphase between the faecal sludge and the sand in order to ensure easy removal of the resulting dewatered biosolid (Figure 3.2). A depth of 20cm of faecal sludge, equivalent to 5 litres was applied in each of the dewatering process.

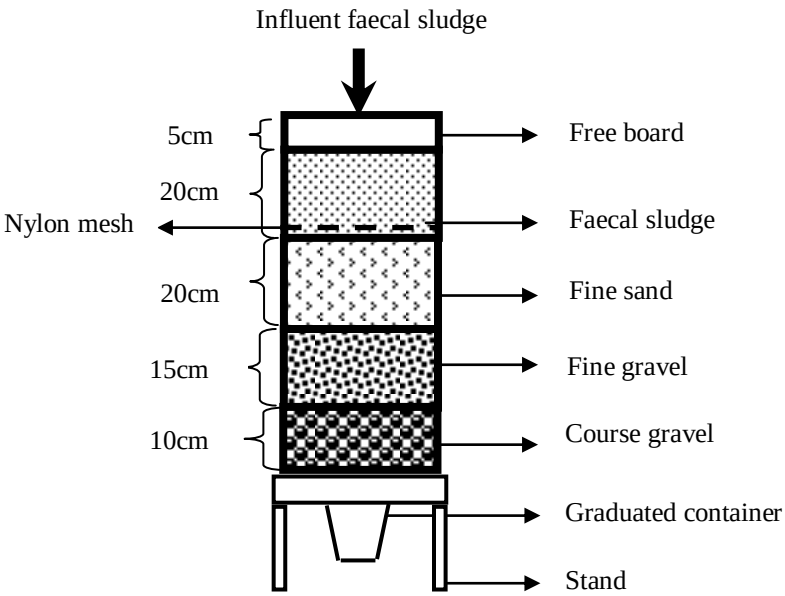


Figure 3.2: Schematic representation of bench scale filter bed

3.5 Preparation of Faecal Sludge (FS)

The faecal sludge used for the dewatering consisted of public toilet sludge (PTS) and septage. Public toilet sludge and septage were collected in separate plastic drums of about 90 litres each from Dompoase Waste Treatment Plant in Kumasi (Appendix B), about 20 km away from the project site. The drums of FS were transported to the project site for mixing (Appendix B, Plate AB6). The public toilet sludge (PTS) and the septage (Plate 3-3) were mixed in various proportions (1:1, 1:2 and 1:3) by volume, for further application on the drying beds.

Septage Public Toilet Sludge

Plate 3 - 3: Samples of septage and Public toilet sludge used for the mixing of faecal sludge

3.6 Dewatering Phases

The dewatering activities involved four main phases depending on the objective under consideration. These included, (i) dewatering of FS using filter media of different particle sizes, (ii) dewatering of FS of different solid loading rates using FM1, (iii) maintaining constant solid loading rate and dewatering on the different types of filter media and (iv) mixing different percentages of sawdust with constant solid loading rate of faecal sludge (selected from specific objective two) and dewatering on a selected filter medium (selected from specific objective one).

3.6.1 Determination of effective particle size ranges of sand that will improve FS dewatering efficiency unplanted filter beds

Design of experiment

The bench scale filter beds were set up to resemble field drying beds. There were three different types of filter media, FM1, FM2, and FM3 with each having different ranges of particle sizes of sand. Each was replicated three times to make nine (9) drying beds. They were arranged in a completely randomized block design (Figure 3.3). The FS consisting of PTS and septage, used for the dewatering were mixed in a ratio of 1:1 by volume for objective one.

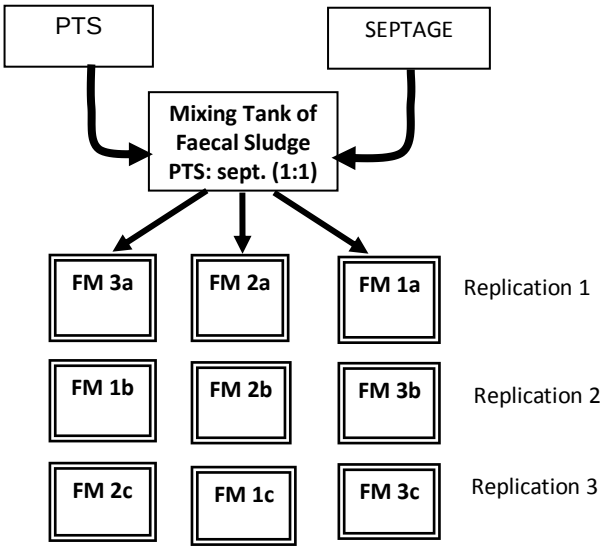


Figure 3.3: Completely randomized block design (CRBD) of filter media (FM)

The fresh septage, PTS and the mixed FS were analysed for physicochemical parameters just after collection from the treatment plant before dewatering. In-situ measurements of temperature, pH and conductivity of the filterate collected from each filter bed were made daily till the cycle was complete. The volume of the percolate and depth of sludge were measured every 24 hours while the numbers of days taken for the sludge to completely dewater were also recorded. Dewatering was considered

complete, when filterate flow from the filter bed stopped and biosolid accumulated was spadable (possible to handle) for possible removal from the filter bed. Six cycles were run for this objective covering research period from July 2005 to December 2005.

Each filterate was then kept below 4^oC separately till dewatering was complete in order to stop any microbial activity. After the dewatering, a composite sample of the percolate from each drying bed was analyzed. Biosolid as well as sand from the top 10 cm of each filter bed was analyzed for Total Solids and Total Volatile Solids (TVS) after each cycle. This is because fiteration and accumulation of particulate matter is said to be active in the top 10 cm of the filter medium, (Karr and Keinath, 1978). The parameters analyzed in the FS and percolate were, Suspended Solids (SS), Chemical Oxygen Demand (COD), Dissolved Chemical Oxygen Demand (DCOD), TS, TVS, Total Kjedhal Nitrogen (TKN), Ammonia Nitrogen (NH_3- N), Nitrate (NO_3), Total Phosphorus (TP), Electrical Conductivity (EC), Temperature and pH. Methods outlined in Standard Methods for the Examination of Water and Wastewaters (APHA-AWWA-WEF, 2005) were used for the analyses of the parameters.

3.6.2 Determination of Solid Loading Rate of faecal sludge that will improvre dewatering efficiency of unplanted filter beds

Faecal sludge collection and preparation

PTS from public toilets and septage from septic tanks were collected in separate plastic drums of about 90 litres capacity each from FS suction trucks discharging at Dompoase landfill site in Kumasi. The PTS and septage were mixed in ratios of 1:1, 1:2 and 1:3 by volume. The of production of PTS and septage in Kumasi and other urbanised areas in Ghana show increasing volumes of septage as against PTS from the

1990's to date.This has been increasing from 2:1 for PTS: septage ratio in the 1990's to 1:2 PTS:septage ratio presently. This is due to increasing construction of water closet toilet with septic tanks in households (Ackon 2006; Personal communication with Tony Mensah, Director of waste management, KMA). It was based on the above trend that the ratios of public toilet sludge and septage for this study were chosen. The TS concentration of the different mixing ratios were used as the solid loading rates, thus SLR1, SLR2 and SLR3 for 1:1, 1:2 and 1:3 respectively. The method of mixing reflects the pattern of generation of sludges in Kumasi and Ghana as a whole over the years where the volumes of generation of the PTS have been varying with the septage.

Design of experiment

The FS used for the dewatering consisted of PTS and septage mixed in the ratios of 1:1, 1:2 and 1:3 by volume, representing, SLR1, SLR2 and SLR3, respectively. Nine (9) filter beds made of the same sand, FM1, selected from objective one were arranged in 3 blocks with each block replicated 3 times. The three different solid loading rates SLR1, SLR2 and SLR3 were applied to the filter beds in a complete randomized block design arrangement (Figure 3.4).

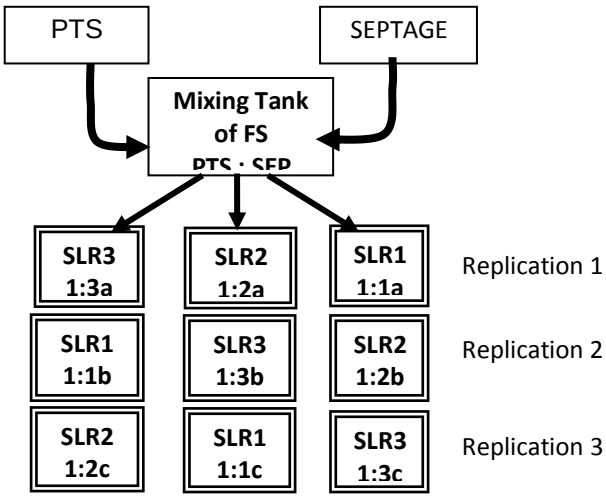


Figure 3.4: Completely randomised block design of different solid loading rates (SLR) of FS

Faecal sludge (FS) dewatering

A liquid depth of 20 cm (equivalent to 5 litres) was applied on each filter bed after the total solid (TS) of each mixing had been determined. The volume of the filterate and depth of sludge on filter beds were measured every 24 hours while the numbers of days taken for the sludge to dewater were also recorded. Six cycles were run during the research period for this objective, from early January to early June 2006. Dewatering was considered complete when filterate flow from the filter bed stopped and biosolid accumulated was spadable for possible removal from the filter bed.

Laboratory analyses

The FS was analyzed just after collection from the treatment plant before dewatering. In-situ measurements of temperature, pH and conductivity of the percolate collected from each filter bed were made daily. Each percolate was then kept below $4^{o}C$ till dewatering was complete. After the dewatering, a composite sample of the percolate from each drying bed was analyzed. Biosolid as well as sand from the top 10cm of each filter bed were analyzed for Total Solids (TS) and Total Volatile Solids (TVS) after each cycle. The parameters analyzed in the FS and filterate were, Suspended solids (SS), Chemical Oxygen Demand (COD), Dissolved Chemical Oxygen Demand (DCOD), TS, TVS, Total Kjedhal Nitrogen (TKN), Ammonia Nitrogen (NH_{3-} N), Nitrate (NO_3), Total Phosphorus (TP), Electrical Conductivity (EC), Temperature and pH. Methods outlined in Standard Methods for the Examination of Water and Wastewaters (APHA-AWWA-WEF, 2005) were used for the analyses of the parameters.

3.6.3 Effect of constant solid loading rate on FS dewatering, by different filter media

In the previous two objectives the faecal sludge sample used for the dewatering was mainly based on the hydraulic loading thus always varying the total solid content of the sample in each cycle. It was therefore decided to maintain the total solids (TS) of the sludge samples in the cycles constant in order to understand the behaviour of the dewatering of the filter beds.

Determining the constant TS of the samples

Based on the diameter of the miniature filter container designed for the dewatering and the faecal sludge liquid depth of 20 cm applied on each filter bed, the volume of faecal sludge applied at each cycle was determined to be equivalent to 5 liters indicated as standard volume (**SV).** The average TS of mixing ratio of 1:1 by volume of all the cycles run in objective one and objective two was also determined to be the **Standard TS.** In order to keep the TS of the new sample equal to the standard TS, new volume that can contain the standard TS had to be calculated. The TS of the new sample was therefore determined before dewatering started. Whenever a new sample was brought, PTS and septage were mixed in 1:1 and 1:3 ratios by volume. The 1:1 and the 1:3 ratios were selected because 1:1 ratio was used to run all the cylces for the objective one and it was also applied in objective two. It was applied to all the filter beds FM1, FM2 and FM3. The 1:3 was the best dewatering SLR in terms of dewatering time and percolate quality. It was applied to FM1 because it (FM1) was the best performing filter medium in the objective one, in terms of coupling effect of dewatering time and percolate quality. There were not enough space and time to run the other loading rates. The TS of the raw PTS, septage, and 1:1 and 1:3 ratios were

determined in an oven at 105°C for 24 hours as described in Standard Methods for examination of water and wastewater (APHA-AWWA-WEF, 2005). The TS of raw PTS and septage were also determined to check the accuracy of TS of the mixed samples. The new volume was therefore calculated from the following formula

$$NV = \left(\frac{STS}{NTS} \right) x \ SV$$

Where;

NV = New Volume (Volume of new FS to be taken for dewatering)

STS = Standard TS (average TS of the FS samples used for dewatering all the cycles)

NTS = TS of the new FS sample to be dewatered.

SV = Standard Volume used for all the previous cycles.

Calculation for 1:1 ratio

Standard TS for the 1:1 ratio was = 36.64g/l

Standard volume was = 5 litres

Therefore new volume to be taken (NV)

$$NV = \left(\frac{36.64}{NTS} \right) x \ 5$$

Calculation for 1:3 ratio

Standard TS for the 1:3 ratio was = 26.93g/l

Standard volume was = 5 litres

Therefore new volume to be taken (NV)

$$NV = \left(\frac{26.93}{NTS} \right) x \ 5$$

The NTS is the TS of the freshly mixed faecal sludge to be used for the dewatering.

In all 12 filter beds were used for dewatering at each cycle. These comprised the three different filter media, FM1, FM2 and FM3. Each had three replicates while 1:3 SLR was applied to three FM1 filter beds. Eight cycles were run.

Loading of sludge

Whenever the mixing was complete and the TS determined, a wooden stick was used to stir the sludge gently and the determined volume was measured in a graduated plastic container. A wooden board was placed slanted on the filter bed from top of container to the surface of the sand, on which the sludge sample was poured in order to reduce impact of the sludge on the surface of the sand.

Monitoring

Percolate flow into the graduated container placed under the filter bed usually began from about 2 to 30 minutes after loading depending on a number of factors. The depth of sludge on the filter bed and volume of percolate collected were monitored and measured every 24 hours. The collected percolate was kept at $4^{o}C$ continuously until dewatering was complete. Dewatering was considered complete when filterate flow from the filter bed stopped and biosolid accumulated was spadable for possible removal from the filter bed. The resulting biosolid on each filter bed was carefully removed and weighed. Each was thoroughly mixed and analysed for TS and TVS. Samples were taken from at least 10 points from the top 10cm of the sand of each filter bed. This was well mixed and analysed for TS and TVS. The percolate was well mixed and composite sample was used for the analyses of various physicochemical parameters.

3.6.4 Determination of quantity of physical conditioner (sawdust) that will improve the dewatering efficiency of unplanted filter beds

The physical conditioner used in this study was sawdust. It was selected because it was found to improve the porosity of sludge and its dewaterability (Ying-Feng, *et al*. 2001; Banitez *et al*., 1994). It is abundant and cheap in most parts in Ghana. It is a hazard in many communities where timber processing is common as a result of its indiscriminate disposal and burning. This is a common site in Kumasi, Ghana where these studies were conducted. The sawdust was collected in sacks from the timber processing companies where it is produced in large quantities as waste and carted by trucks for burning (Plate 3-4). The sawdust was oven dried at 105°C for 24 hours and allowed to cool. It was sieved using 5 mm metal mesh sieve, to get particle sizes of 5 mm or less. These particle sizes produce good lattice structure and porosity. It is also easier decomposed by microorganisms after dewatering (Vrilakis *et al.*, 1999).

Plate 3- 4: Sawdust collection point at the timber processing company.

Faecal Sludge collection

PTS and septage were collected in separate 90 litre drums from the Dompoase liquid waste treatment plant. The PTS and septage were thoroughly stirred using a wooden stick. A graduated plastic container was used to measure 1 volume of PTS and mixed with 3 volumes of septage. The TS of the PTS, septage and the mixed FS was

determined. This helped to determine the weight of dried sawdust that must be measured to mix with the faecal sludge (FS).

Calculation of new volume

TS of the faecal sludge was measured in g/l. This was indicated as New TS (NTS).

The standard TS was indicated as (STS).

The usual hydraulic loading of the filter beds was the Standard Volume (SV)

The new volume to be taken for the dewatering was calculated as

$$NV = \left(\frac{STS}{NTS} \right) \times SV$$

Where

NV = New Volume (Volume of new FS to be taken for dewatering)

STS = Standard TS (average TS of the FS samples used for dewatering all the previous cycles)

NTS = TS of the new FS sample to be dewatered.

SV = Hydraulic Volume used for all the previous cycles.

Determination of sawdust for dewatering

The determined TS for the 1:3 PTS: septage ratio was 26.93 g/l

The usual volume of FS applied on the filter beds was 5 litres.

The estimated quantity of sawdust determined to be added to the FS for dewatering were 50%, 100% and 150% of the solid content of the faecal sludge used.

Thus for 50% the sawdust used = (50/100) x 26.93 x 5

= 67.33 g

100%, the sawdust used = (100/100) x 26.93 x 5

= 134.65 g

150% the sawdust used = (150/100) x 26.93 x 5

= 202.0 g

Loading of filter beds

After the volume to be used had been determined, a graduated plastic container was used to measure 3 replicated quantities of the said volume. Each of the sawdust quantities determined above was mixed with one of the faecal sludge measured, and stirred with a wooden stick to ensure it was well mixed. A fourth faecal sludge measured was not mixed with any sawdust and served as control (0 % sawdust). Three (3) replicates were made for each, giving twelve (12) filter beds. They were arranged in completely randomised design (CRD), (Figure 3.5). $PC_{50\%}$, $PC_{100\%}$, $PC_{150\%}$ were the different percentages of physical conditioners (sawdust) used while the Ca, Cb, and Cc were the different replicates of control setup where no sawdust was added.

Replicate 1	Replicate 2	Replicate 3
$PC_{50\%}$	$PC_{100\%}$	$PC_{150\%}$
$PC_{100\%}$	$PC_{150\%}$	Cc
$PC_{150\%}$	Cb	$PC_{50\%}$
Ca	$PC_{50\%}$	$PC_{100\%}$

Figure 3.5: Completely Randomised Design (CRD) for the dewatering of FS mixed with different percentage of sawdust.

Six cycles of dewatering were run over a period of 7 months from June 2007 to January 2008. Percolate flow from the filter beds started between 2 and 5 minutes after filling the filter beds with the FS and the sawdust mixture.

Monitoring

The percolate flow was monitored every 24 hours by measuring the volume collected in the graduated plastic container placed under the filter bed as well as measuring the depth of sludge on the filter bed. The percolate was accumulated daily and kept under $4^{o}C$ until dewatering was complete. Dewatering was said to be complete when the filterate flow from the filter beds stopped and the biosolid was dewatered enough to be handled (spadable) to be collected from the filter bed. The resulting biosolid on each filter bed was carefully removed and weighed. Each was thoroughly mixed and analysed for TS and TVS. Samples were taken from at least 10 points from the top 10 cm of the sand of each filter bed. This was well mixed and analysed for TS and TVS. The percolate was well mixed and composite sample was used for the analyses of various physicochemical parameters. The following parameters were measured; Suspended Solids (SS), Chemical Oxygen Demand (COD), Dissolved Chemical Oxygen Demand (DCOD), TS, TVS, Total Kjedhal Nitrogen (TKN), Ammonia Nitrogen ($NH_{3-}N$), Nitrate (NO_3), Total Phosphorus (TP), Electrical Conductivity (EC), Temperature and pH. Methods outlined in Standard Methods for the Examination of Water and Wastewaters (APHA-AWWA-WEF, 2005) were used for the analyses of the parameters.

3.6.5 Nutrients and heavy metal concentrations in the different dewatered biosolids

Sample preparation

The biosolid collected from each filter bed was well mixed in a plastic bowl using a wooden stick. About 20 g was collected and oven dried at 105 ^{o}C for 24 hours. It was packaged in a clean plastic sheet and kept in refrigerator. The biosolids from all the

filter beds were treated the same way. After completing all the cycles, all the dried samples were sent to the Ecological Laboratory (ECOLAB) at University of Ghana, Legon for the analyses of nutrients (carbon, nitrogen, phosphorus and potassium), and heavy metals (copper (Cu), iron (Fe), lead (Pd), cadmium (Cd), zinc (Zn) and manganese (Mn). This is because, there was no carbon analyser to analyse carbon content and no AAS to determine the heavy metals content of the dried biosolid at KNUST. ECOLAB is also recognised internationally as well as its work.

3.7 Laboratory Analyses

3.7.1 Laboratory Analyses of contaminant loads in percolate and raw FS

pH

A digital pH measuring kit with a probe or electrode (WTW pH 323-B / Set-2) was used. The electrode was first calibrated against pH buffer 4, 7, and 10.

The percolate and raw sample under test were vigorously shaken before the insertion of the probe. There was a digital read-out when the probe was inserted in the sample indicating the pH value of the sample.

Temperature

A digital pH measuring kit with a probe (WTW pH 323-B / Set-2) was used. The kit measures both the pH and temperature of the sample. The sample under test was well shaken, and the probe inserted. There was a digital read-out of the temperature when the probe was inserted in the sample.

Conductivity (EC)

A digital conductivity measuring kit (WTW LF 323-B / Set) was used. The kit has a probe connected to it. The sample under test was well shaken, and the probe inserted. There was a digital read-out when the probe was inserted in the sample.

Mechanism

The conductivity is a measure of the ability of water sample to conduct an electric current. It is related to the concentrations of Total Dissolved Solids (TDS) or ionised substances dissolved in the water sample as well as the temperature at which measurement is made. It provides an indication of the extent of mineralisation. Conductivity is usually expressed in micro/milli siemens per centimetre (μS(mS)/cm), and for a given water sample it is related to the concentrations of TDS and major ions, and it is easy to determine on-site. The conductivity of wastewater normally exceeds 1000 mS/cm. This parameter is important to determine whether the percolate/leachate can be used as irrigation water. This is because in agricultural use, high TDS levels deteriorate the soil structure and declines crop production rates as the high TDS levels create toxicity and osmotic stress to crops grown. The conductivity guideline for discharge in Ghana is 750 μS/cm (EPA, 1997).

Turbidity

Attenuated Radiation Method (Direct reading) was used, as described in the manual of, HACH DR/2010 Portable Datalogging Spectrophotometer. (Appendix C).

Total Solids (TS)

The total solids (TS) was analysed based on the method described in the Standard Methods of water and wastewater analyses, APHA— AWWA-WE, (2005) (Appendix C).

Total volatile solids (TVS)

The total volatile solids (TVS) was analysed based on the method described in the Standard Methods of water and wastewater analyses, APHA-AWWA-WE, (2005), (Appendix C).

Suspended solids (SS)

The suspended solids (SS) was analysed based on the method described in the Standard Methods of water and wastewater analyses, APHA-AWWA-WEF (2005). (Appendix C).

Chemical Oxygen Demand (COD)

The chemical oxygen demand (COD) was analysed based on the method described in the Standard Methods of water and wastewater analyses, APHA-AWWA-WEF (2005). (Appendix C).

Ammonia-nitrogen (NH_3-N) (Distillation and Titrimetric Method)

The ammonia nitrogen (NH_3-N) was analysed using distillation titrimetric method as described in the Standard Methods of water and wastewater analyses, APHA—AWWA-WEF (2005). (Appendix C).

Nitrate-nitrogen (NO_3-N)

The nitrate nitrogen was determined using Cadmium Reduction Method, which makes use of Powder Pillows, as described in the manual of, HACH DR/2010 Portable Datalogging Spectrophotometer. (Appendix C).

Phosphorus

The Phosphorus was determined using Cadmium Reduction Method, which makes use of Phos Ver 3 (Ascorbic Acid) Method using Powder Pillows, as described in the manual of, HACH DR/2010 Portable Datalogging Spectrophotometer. See Appendix C for the detailed description of the method.

3.7.2 Nutrients and Heavy metals analyses of dewatered biosolids

The nutrients included Organic Carbon (org.C), Total Nitrogen, Phosphorus, Potassium. The heavy metals included Copper (Cu), Iron (Fe), Lead (Pb), Cadmium (Cd), Zinc (Zn) and Manganese (Mn). ECOLAB is an internationally recognised laboratory and therefore results of samples from there are internationally accepted. Moreover half of the samples sent for analyses were given shade samples which were found to be almost the same as their corresponding labelled samples.

Analyses of nutrients in dry dewatered faecal sludge

Determination of organic carbon in dried faecal sludge

A modification of the wet oxidation procedure based on the reduction of the $Cr_2O_7^{2-}$ ion by organic matter which is known as the Wakley and Black procedure was employed in this exercise. Oxidizable matter in a sludge sample is oxidized by $Cr_2O_7^{2-}$, and the reaction is facilitated by the heat generated when 2 volumes of H_2SO_4 are mixed with 1 volume of 1N $K_2\ Cr_2O_7$ solution. The excess $Cr_2O_7^{2-}$ is determined by titration with standard ferrous ammonium sulphate solution, and the quantity of substances oxidized is calculated from the amount of $Cr_2O_7^{2-}$ reduced (Okalibo *et al.*, 2002); Rayment and Higgingson, (1969).

Determination of Total Nitrogen in dried faecal sludge

Procedure Using a Block Digester

The total nitrogen in dried faecal sludge was determined using the block digester method as described by Okalibo *et al*. (2002); Rayment and Higgingson (1969), (Appendix C).

Determination of Total Phosphorus in dried faecal sludge

The total phosphorus determined by the caloric procedure for total phosphorous measurement without pH adjustment as described in, Laboratory Methods of soil and plant analysis: A Working Manual 2nd Edition. TSBF-CIAT and SACRED Africa, Nairobi, Kenya, Okalebo *et al*. (2002), (Appendix C).

Determination of potassium in dried faecal sludge

The potassium was determined using the method as described in, Laboratory Methods of soil and plant analysis: A Working Manual 2nd Edition. TSBF-CIAT and SACRED Africa, Nairobi, Kenya, Okalebo *et al*. (2002), (Appendix C).

Determination of Heavy Metals dried sludge samples

This section covered methods of digestion and determination for heavy metals in dried sludge sample. Heavy metals of concern which were determined included Copper (Cu), Iron (Fe), Lead (Pb), Cadmium (Cd), Zinc (Zn) and Manganese (Mn).

Principle

The heavy metals were measured in a digest obtained by treating samples with an acid mixture made from concentrated nitric acid, concentrated sulphuric acid, and

perchloric acid, as described in (Chapman and Pratt, 1961; Association of Official Analytical Chemist,1979), (Appendix C).

CHAPTER FOUR

4 RESULTS AND DISCUSSION

4.1 Effective Particle Size Ranges of Sand That Will Improve Faecal Sludge Dewatering Efficiency of Unplanted Filter Beds

4.1.1 Results of effective particle size ranges of sand that will improve FS dewatering efficiency

4.1.1.1 Sand (Filter Medium) Analyses

The analyses conducted on the filter medium were sieve analysis and the permeability of the sand.

Sieve Analyses of Sand

Three different ranges of particle sizes 0.1 - 0.5 mm, 0.5 - 1.0 mm, 1.0 - 1.7 mm representing FM1, FM2 and FM3 respectively were obtained from sieve analyses of local sand used for the studies which gave the respective effective sizes shown in Table 4.1.1. Based on effective sizes (D_{10}, D_{60}), the uniformity coefficient (u_o) values for the range of particle sizes were obtained. The uniformity coefficient values of FM1, FM2 and FM3 were given as 2.42, 1.73 and 2.03 respectively (Table 4.1.1). The u_o values obtained were all less than 4 as given by Metcalf and Eddy (2003), National Small Flows Clearinghouse (1997), Crites and Tchobanoglous (1998), EPA (US) (2002), and Ohio State University Bulletin (1999) as being ideal for wastewater/faecal sludge filtration.

Table 4.1.1: Effective size of sand and Uniformity Coefficient (u_o)

Particle size range (mm)	D_{10} (mm)	D_{30} (mm)	D_{60} (mm)	u_o
0.1-0.5 (FM1)	0.1064	0.1782	0.2577	2.42
0.5-1.0 (FM2)	0.4601	0.5761	0.7948	1.73
1.0-1.7 (FM3)	0.6884	0.9336	1.397	2.03

Uniformity coefficient (u_o) = D_{60}/D_{10}

Permeability Test of Sand

The initial permeability test conducted for the three different types of filter media FM1, FM2 and FM3 gave the mean permeability as 9.0 x 10^{-2}, 14.0 x 10^{-2} and 36.0 x 10^{-2} (cm/sec) respectively (Table 4.1.2). This means that the filtration was highest in FM3 followed by FM2 and then FM1. This was reflected in the average dewatering time obtained for the cycles.

Table 4.1.2: Permeability of sand

Parameters	Units	Values
Length of sample between nipple points (L)	cm	10.00
Diameter of soil sample	cm	7.60
Cross sectional area of sample (A)	cm^2	45.38

Sample Code (Particle size)		FM 1 (0.1-0.5)		FM 2 (0.5-1.0)		FM 3 (1.0-1.7)	
		Trial 1	Trial 2	Trial 1	Trial 2	Trial 1	Trial 2
Head of water in Manometer (A)	cm	56.50	56.00	35.00	36.60	24.50	25.00
Head of water in Manometer (B)	cm	78.00	79.00	77.50	80.30	48.00	47.50
Head difference (h)	cm	21.50	23.00	42.50	43.70	23.50	22.50
Volume of water	cc	920.00	1062.00	1690.00	2595.00	2290.00	2480.00
Elapsed time (t)	sec	99.90	121.66	62.01	97.84	60.72	65.88
Discharge (q)	cc /sec	9.21	8.73	27.25	26.52	37.71	37.64
Permeability (k)	cm /sec	0.09	0.08	0.14	0.13	0.35	0.37
Average Permeability	cm /sec	0.09		0.14		0.36	

Permeability k was obtained using the formula

$$k = \frac{qL}{hAt}$$

where k = permeability; q = discharge; L = Length of soil; A = area of cross section of soil

t = time elapsed.

4.1.1.2 Characteristics of faecal sludge (FS)

The faecal sludge (FS) from septage and PTS analyzed had very high values for TS, SS, COD, NH_3-N, TKN, NO_3-N (Table 4.1.3). Strauss *et al.* (1997) report that these concentrations are several (10 -100) times the strength of sewage.

Table 4.1.3: Average characteristics of faecal sludge (FS) over six cycles

Parameter	Mean values with SD	Parameter	Mean values with SD
TS (g/l)	36.64 ± 7.73	COD(g/l)	50.32 ± 28.78
TVS (g/l)	25.34 ± 9.98	DCOD(g/l)	10.98 ± 5.31
SS (g/l)	19.68 ± 6.59	NH_3-N(g/l)	2.83 ± 1.68
Conductivity (mS/cm)	22.67 ± 3.45	TKN(g/l)	3.58 ± 2.07
Temperature (^oC)	28.07 ± 1.05	NO_3(g/l)	1.38 ± 0.42
Ph	7.77 ± 0.13	TP(g/l)	1.85 ± 0.53

± *Standard deviation (SD)*

4.1.1.3 Dewatering Times of the Different Particle Sizes (FM1, FM2 and FM3)

The average dewatering times for FM1, FM2 and FM3 were 9.8, 9.9 and 9.1 days respectively. The error bars show the extent of variation in dewatering days of the different replicates, while those without error bar almost the same dewatering days. The permeabilities were 0.09, 0.14 and 0.36 cm/sec with corresponding uniformity coefficients of 2.42, 1.73 and 2.03 for FM1, FM2 and FM3 respectively. Generally, from cycle one (C1), the DT showed gradual decline until cycle five (C5) and then increased in cycle six (C6) (Figure 4.1) among all the filter media. There were no significant difference in the dewatering time of the three different filter media at 5% significance level (p = 0.212).

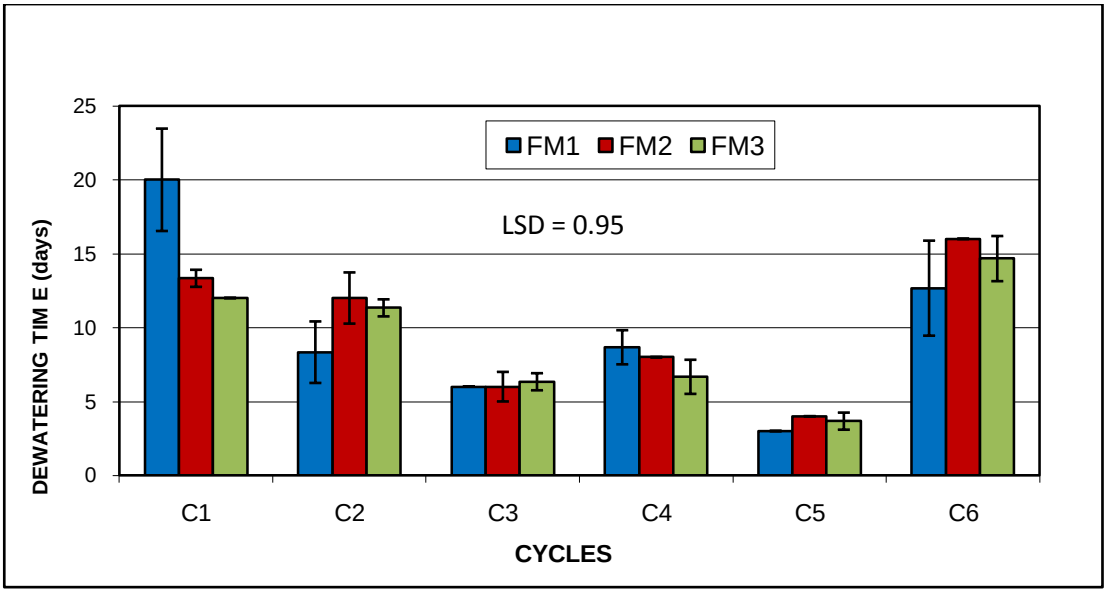


Figure 4.1: The dewatering time (days) of different particle sizes in six cycles
(Those without error bars have the sameor almost the same dewatering days for the replicates)

4.1.1.4 Physicochemical parameters removal efficiency

Total Solids (TS)

The mean percentage removals of TS from the raw sludge by FM1, FM2 and FM3 were 76.18, 74.04 and 71.51 respectively, with FM1 being the best. The corresponding average TS values in the percolate (8.62, 9.51 and 10.84) (g/l) through cycles 1-6 were high (Table 4.1.4), for discharge into the environment. This therefore, needed further treatment before being discharged because high TS in the effluent means high contaminant load whether chemical or biological which is not healthy for the receiving environment. Generally, at any cycle FM3 had the highest TS in the percolate except in C3. ANOVA of the percolate TS at 5% significance level indicated that there were significant differences between the treatments (p value = 0.0001).

Table 4.1.4: Total Solids (TS) g/l of faecal sludge (FS) and percolate with percentage removal of solids

		Percolate TS (g/l)			% TS REMOVED		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM 1**	**FM 2**	**FM 3**
C1	31.40	5.20	6.60	8.67	83.44	78.98	72.40
C2	30.14	9.07	9.07	10.20	69.92	69.92	66.16
C3	38.20	10.19	9.87	10.10	73.32	74.15	73.56
C4	43.20	6.33	8.00	10.67	85.35	81.48	75.31
C5	29.04	8.26	8.26	8.35	71.57	71.56	71.26
C6	47.87	12.69	15.25	17.06	73.48	68.14	64.37
Mean±SD	**36.64±7.73**	**8.62±2.70**	**9.51±3.02**	**10.84±3.18**	**76.18±6.52**	**74.04±5.25**	**70.51±4.32**
LSD for percolate TS = 0.57							

Total Volatile Solids (TVS)

The TVS in the percolate (Table 4.1.5) were relatively low as compared to the levels in the raw sludge in all the cycles, implying high percentage removal. The removal efficiencies of FM1, FM2 and FM3 were in the order of (78.18, 77.25 and 73.57) %, respectively (Table 4.1.5). The percentage removal of TVS by the three filter media did not follow any particular trend from cycle 1-6. However, the FM1 gave the highest removal efficiency in most of the cycles. The average TVS values of 4.41, 4.83 and 5.86 (g/l) in the percolates of FM1, FM2 and FM3, respectively (Table 4.1.5) were too high for discharge into the environment. ANOVA of the percolate TVS at 5% significance level indicated that there were significant difference between the treatments (p value = 0.0001).

Table 4.1.5: Total Volatile Solids (TVS) (g/l) of faecal sludge and percolate with percentage removal

		TVS of Percolate (g/l)			%TVS Removed		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	20.80	2.73	3.93	5.93	86.86	81.09	71.47
C2	22.96	4.13	3.20	4.00	82.00	86.06	82.58
C3	23.00	5.08	4.97	4.79	77.91	78.38	79.16
C4	38.98	2.47	3.48	5.65	93.67	91.07	85.50
C5	11.52	6.31	5.97	6.00	45.20	48.18	47.95
C6	34.76	5.75	7.40	8.77	83.46	78.72	74.76
Mean±SD	**25.34±9.98**	**4.41±1.58**	**4.83±1.62**	**5.86±1.62**	**78.18±17.00**	**77.25±15.05**	**73.57±13.54**
LSD of percolate TVS = 0.52							

Suspended Solids (SS)

There were high removal efficiencies of SS in all the filter media in all the cycles with the FM1 as the most efficient, followed by FM2 and then FM3 in the order of 95%, 93% and 89% respectively (Table 4.1.6). Although these removal efficiencies were high, the average levels of SS in the percolates (Table 4.1.6) were no near the allowable EPA Ghana, discharge levels (0.05g/l), into the environment (EPA, 2001). This implies that the SS in the effluent needs further treatment before being discharged into water bodies. ANOVA of the percolate SS at 5% significance level indicated that there were significant difference between the treatments (p = 0.0001).

Table 4.1.6: Suspended Solids (g/l) of faecal sludge (FS) and percolate with percentage removals

		Percolate SS (g/l)			%SS Removed		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	19.70	0.29	0.42	0.80	98.54	97.88	95.94
C2	19.80	0.91	0.87	1.40	95.40	95.61	92.93
C3	14.00	0.76	1.12	1.35	94.60	91.98	90.36
C4	26.00	0.87	1.70	3.17	96.67	93.46	87.82
C5	10.80	1.07	1.37	2.73	90.12	87.35	74.69
C6	27.8	1.30	1.67	2.30	95.32	94.00	91.73
Mean±SD	**19.68±6.59**	**0.86±0.34**	**1.19±0.49**	**1.96±0.92**	**95.11±2.81**	**93.38±3.58**	**88.91±7.47**
LSD of percolate SS = 0.25							

Correlation between dewatering time and suspended solids

In dewatering the faecal sludge by the different filter media, positive correlation (r = 0.89 for FM1; 0.94 for FM2 and 0.95 for FM3) was observed between dewatering time and suspended solids for filter media one, filter media two and filter media three (Figure 4.2).

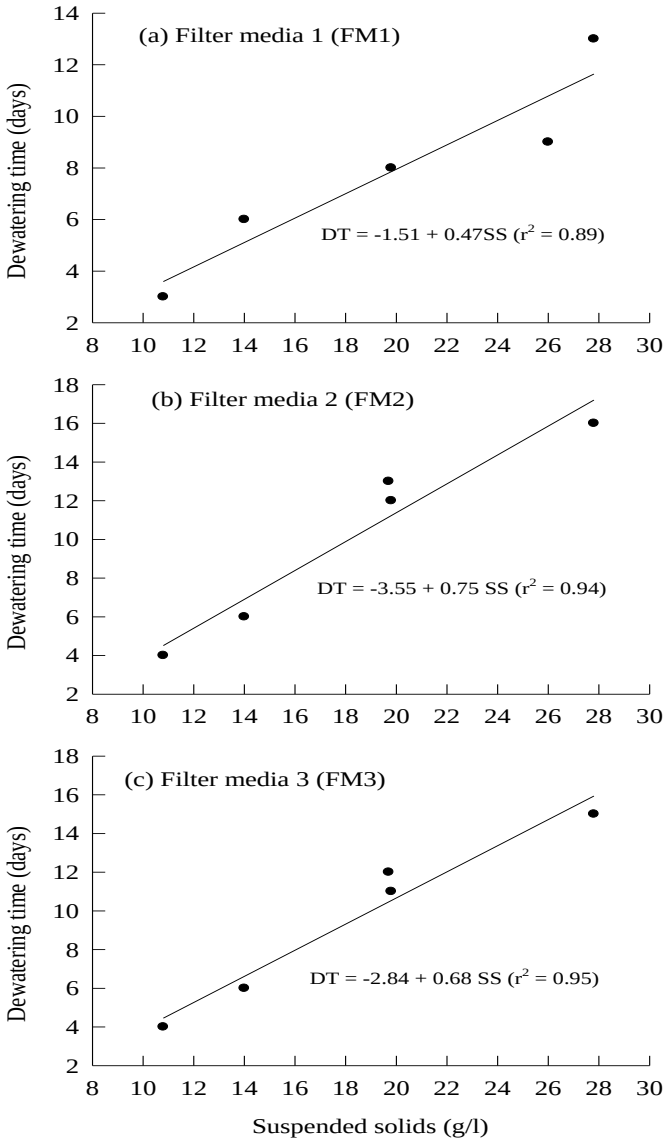


Figure 4. 2: Relationship between suspended solids and dewatering times for different filter media

Chemical Oxygen Demand (COD) and Dissolved Chemical Oxygen Demand (DCOD)

The COD of the percolate in the different filter media were relatively low, compared to the in the faecal sludge (Table 4.1.7). These were indicated in the high percentage removals (Table 4.1.7), with FM1 showing the highest removal efficiency in all the

cycles except in C2, followed by FM2 and then FM3. ANOVA of the percolate COD at 5% significance level indicated that there were significant difference between the treatments (p value = 0.0009).

The DCOD levels in the faecal sludge compared to that in the percolates from different filter media showed that the filter beds were capable of removing oxygen demanding components of FS (whether biodegradable or non-biodegradable) that passed filtration (Table 4.1.8). In almost all the cycles FM1 had the highest removal efficiency as compared to the FM2 and FM3. The average DCOD removal efficiency of FM1 was 77.74% followed by FM2 with 70.75% and FM3 with 65.80% (Table 4.1.8). ANOVA of the percolate DCOD at 5% significance level indicated that there were significant differences between the treatments (p value = 0.0002).

Table 4.1.7: Chemical Oxygen Demand (COD)(g/l) of percolate and FS with percentage removals.

		Percolate COD (g/l)			%COD Removed		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	35.36	2.21	2.76	4.17	93.76	92.19	88.21
C2	36.96	7.50	4.56	6.64	79.71	87.65	82.03
C3	107.20	6.44	7.33	8.28	93.99	93.16	92.28
C4	43.20	6.64	8.75	8.28	84.62	79.75	80.83
C5	29.04	3.21	3.91	4.33	88.93	86.53	85.10
C6	50.16	6.87	7.42	8.48	86.31	85.20	83.09
Mean±SD	**50.32±28.78**	**5.48±2.20**	**5.79±2.37**	**6.70±2.01**	**87.89±5.53**	**87.41±4.91**	**85.26±4.30**
LSD of percolate COD = 0.61							

Table 4.1.8: Dissolved Chemical Oxygen Demand (DCOD)(g/l) of FS and percolate with percentage removals

		Percolate DCOD (g/l)			%DCOD Removed		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	6.24	0.74	2.06	2.55	88.19	66.93	59.13
C2	12.32	3.39	2.99	2.96	72.46	75.73	75.95
C3	20.00	2.42	2.36	2.16	87.90	88.18	89.20
C4	5.76	1.82	2.58	3.64	68.40	55.21	36.87
C5	8.80	2.24	2.32	2.38	74.55	73.60	72.95
C6	12.77	3.20	4.49	5.02	74.97	64.86	60.72
Mean±SD	**10.98±5.31**	**2.30±0.97**	**2.80±0.88**	**3.12±1.07**	**77.74±8.31**	**70.75±11.20**	**65.80±17.94**
LSD of percolate DCOD = 0.35							

Electrical Conductivity (EC)

The conductivity values in the percolates from FM1, FM2 and FM3 were found to be 13.56, 15.85 and 16.18 mS/cm respectively, (Table 4.1.9). These values compared with the amount in the faecal sludge showed very low percentage removal efficiencies of 40.82 for FM1, 30.42 for FM2 and 28.57 for FM3, (Table 4.1.9). The removal efficiencies of the pollutants by the treatments were different (p = 0.0001).

Table 4.1.9: Electrical Conductivity (EC)(mS/cm) of faecal sludge and percolate with percentage removal.

		Percolate EC (mS/cm)			% removal of EC		
Cycles	**FS (mS/cm)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	23.2	5.74	10.65	11.70	75.26	54.08	49.57
C2	20.3	14.15	14.00	14.01	30.28	31.03	31.00
C3	22.6	16.60	16.78	16.54	26.55	25.74	26.81
C4	17.49	8.41	12.61	14.18	51.93	27.92	18.93
C5	25.2	16.95	18.53	18.77	32.74	26.46	25.52
C6	27.2	19.54	22.50	21.87	28.17	17.28	19.61
Mean±SD	**22.67±3.45**	**13.56±5.38**	**15.85±4.32**	**16.18±3.69**	**40.82±19.23**	**30.42±12.46**	**28.57±11.25**
LSD of percolate EC = 0.68							

Turbidity (NTU)

Removal efficiencies for turbidity were very high for all the filter media where FM1, FM2 and FM3 showed 97.40, 94.99 and 90.52 percentage removals respectively,

(Table 4.1.10). In all the cycles, FM1 showed the highest percentage removal (Table 4.1.10). ANOVA of the percolate turbidity at 5% significance level indicated that there were significant differences between the variables (p = 0.0001).

Table 4.1.10: Turbidity (NTU) of faecal sludge and percolate with percentage removals

		Turbidity (NTU) of Percolate			% removal of Turbidity		
Cycles	**FS (NTU)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	13700	31	291	675	99.78	97.87	95.07
C2	8760	31	291	675	99.65	96.67	92.29
C3	83500	1307	1327	1132	98.44	98.41	98.64
C4	9500	783	1494	3023	91.76	84.27	68.18
C5	5605	86	95	179	98.46	98.30	96.81
C6	17850	657	1002	1400	96.32	94.39	92.15
Mean±SD	**23153±29869**	**482±523**	**750±600**	**1181±996**	**97.40±3.03**	**94.99±5.46**	**90.52±11.24**
LSD of Turbidity of percolate = 114.7							

pH and Temperature

The average pH between the faecal sludge and the percolate did not differ much. The pH, of the faecal sludge in all the cycles was lower than that of their corresponding percolates. The temperatures of the faecal sludges in the different cycles were higher than their corresponding percolates. The variation in pH and temperature between the faecal sludge and the percolate from the different filter beds were not consistent within the cycles (Table 4.1.11). ANOVA of the percolate pH and temperature at 5% significance level indicated that there were significant differences between the filter media with respect to pH (p value = 0.0002) and temperature (p value = 0.0050).

Table 4.1.11: pH and Temperatures of faecal sludge and percolate

		pH of Percolate			FS (°C)	Temperature (°C) of percolate		
Cycles	**FS**	**FM 1**	**FM 2**	**FM 3**		**FM1**	**FM2**	**FM3**
C1	7.84	8.29	8.20	8.07	28.20	26.33	25.93	26.17
C2	7.59	7.90	8.14	8.20	27.50	25.90	26.07	25.73
C3	7.78	8.40	8.31	8.07	29.20	28.60	28.63	28.53
C4	7.93	8.50	8.26	8.04	27.20	27.10	26.97	27.63
C5	7.65	8.50	8.42	8.38	26.90	28.37	28.07	28.00
C6	7.81	8.58	8.46	8.51	29.40	28.97	28.83	28.73
Mean±SD	**7.77±0.13**	**8.36±0.25**	**8.30±0.12**	**8.21±0.20**	**28.07±1.05**	**27.54±1.28**	**27.42±1.28**	**27.47±1.24**
		LSD of percolate pH = 0.07				**LSD of percolate temperature = 0.08**		

Ammonia Nitrogen (NH_3-N) and Total Kjedhal Nitrogen (TKN)

The average ammonia nitrogen concentration in the percolates from the filter media FM1, FM2 and FM3 obtained were 0.64, 0.89 and 0.84 (g/l) respectively. These values were too high for discharge into water bodies (Table 4.1.12), even though percentage removals of 77.96, 71.14 and 71.63 for the filter beds, FM1, FM2 and FM3 respectively, were very high.

The filter media were efficient in the Total Kjeldhal Nitrogen (TKN) removal. FM1, FM2 and FM3 had percolate TKN of 0.78, 1.00 and 1.38 (g/l), with percentage removals of 78.07, 71.68 and 60.81 respectively (Table 4.1.13). FM1 showed the highest potential of ammonia nitrogen and Kjeldhal nitrogen removal followed by FM2 and then FM3 (Table 4.1.13).

Table 4.1.12 Ammonia Nitrogen (NH_3-N) (g/l) of faecal sludge and percolate with percentage removals

		Percolate NH_3-N (g/l)			% NH_3-N removed		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	3.29	0.43	1.24	1.13	86.83	62.21	65.55
C2	1.10	0.19	0.19	0.19	82.42	82.42	82.73
C3	0.81	0.25	0.25	0.29	68.72	69.55	64.20
C4	4.17	1.02	1.56	1.24	75.62	62.67	70.26
C5	2.55	0.37	0.42	0.47	85.36	83.53	81.44
C6	5.06	1.58	1.70	1.74	68.77	66.47	65.61
Mean±SD	**2.83±1.68**	**0.64±0.55**	**0.89±0.68**	**0.84±0.62**	**77.96±8.11**	**71.14±9.56**	**71.63±8.36**
		LSD of percolate =0.08					

Table 4.1.13 Total Kjedhal Nitrogen (TKN) (g/l) of faecal sludge and percolate with percentage removals

		TKN of Percolate (g/l)			% TKN removed		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	4.72	0.54	0.70	0.97	88.49	85.15	79.46
C2	1.33	0.24	0.31	0.42	82.06	76.93	68.10
C3	1.12	0.36	0.48	0.67	67.56	57.08	40.44
C4	5.16	1.23	1.57	2.17	76.23	69.59	57.98
C5	3.07	0.50	0.66	0.91	83.60	78.55	70.28
C6	6.09	1.80	2.27	3.13	70.50	62.76	48.62
Mean±SD	**3.58±2.07**	**0.78±0.61**	**1.00±0.76**	**1.38±1.05**	**78.07±8.08**	**71.68±10.52**	**60.81±14.58**
LSD of percolate = 0.09							

Nitrate Nitrogen (NO_3-N), Nitrite Nitrogen (NO_2-N) and Phosphorus (P)

The results for nitrate nitrogen analyses are presented for only cycles C1, C2 and C6 (Table 4.1.14), due to inadequate reagents during the period C3 – C5, nitrate was not analysed. Although, the nitrate levels were inconsistent in the percolate from the different filter beds in the cycles (Table 4.1.14), the average percentage removal were 56.00, 52.54, and 39.56 for FM1, FM2 and FM3, respectively (Table 4.1.14). The Nitrite Nitrogen removal efficiency was also highest in the FM1, followed by FM3 and then FM2 in the order of 90.07%, 75.30% and 71.43%, respectively (Table 4.1.15). The percentage removal of phosphorus was highest in FM2, FM1 and FM3 as 58.95%, 52.88% and 48.74%, respectively (Table 4.1.16). The removals of these nutrients did not show any consistent trend with respect to the filter beds.

Table 4.1.14 Nitrate Nitrogen (NO_3-N) g/l of faecal sludge and percolate with percentage removals

		NO_3-N (g/l) in percolate			% NO_3-N (g/l) removed		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	1.65	0.70	0.60	0.91	57.58	63.64	44.85
C2	0.9	0.30	0.47	0.56	66.67	48.15	37.78
C3	Nd	nd	nd	Nd	nd	nd	Nd
C4	Nd	nd	nd	Nd	nd	nd	Nd
C5	Nd	nd	nd	Nd	nd	nd	Nd
C6	1.6	0.90	0.87	1.02	43.75	45.83	36.04
Mean±SD	**1.38±0.42**	**0.63±0.31**	**0.64±0.20**	**0.83±0.24**	**56.00±11.54**	**52.54±9.68**	**39.56±4.66**
LSD of percolate = 0.08							

Table 4.1.15 Nitrite Nitrogen (NO_2-N) g/l of faecal sludge and percolate with percentage removals

		NO_2-N (g/l) in percolate			% NO_2-N (g/l) removed		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	0.370	0.004	0.117	0.140	98.83	68.47	62.16
C2	0.130	0.004	0.016	0.014	96.67	87.69	89.49
C3	0.300	0.029	0.036	0.088	90.44	88.00	70.67
C4	0.018	0.004	0.005	0.002	75.93	72.22	90.74
C5	0.008	0.001	0.001	0.001	90.00	86.25	90.00
C6	0.270	0.031	0.200	0.138	88.58	25.93	48.77
Mean±SD	**0.183±0.15**	**0.012±0.01**	**0.062±0.08**	**0.064±0.07**	**90.07±8.03**	**71.43±23.83**	**75.30±17.63**
LSD of percolate = 0.001							

Table 4.1.16 Phosphorus (g/l) of faecal sludge and percolate with percentage removals

		Phosphorus (P) (g/l) in percolate			% Phosphorus (P) (g/l) removed		
Cycles	**FS (g/l)**	**FM 1**	**FM 2**	**FM 3**	**FM1**	**FM2**	**FM3**
C1	2.80	0.34	0.53	0.56	87.98	81.07	79.88
C2	1.69	0.77	0.95	1.02	54.24	43.59	39.45
C3	1.50	0.83	0.72	0.80	44.36	52.20	46.47
C4	2.00	0.76	0.87	1.05	62.10	56.58	47.48
C5	1.85	1.05	1.26	1.30	43.42	31.80	29.73
C6	1.26	0.81	0.90	0.95	35.93	28.57	24.60
Mean±SD	**1.85±0.53**	**0.76±0.25**	**0.87±0.25**	**0.95±0.25**	**58.95±18.70**	**52.88±19.18**	**48.74±19.51**
LSD of percolate = 0.11							

4.1.1.5 TVS of Sand (Filter medium)

The TVS values of the sand of the different filter media (Table 4.1.17), increased gradually as the cycles advanced in all the filter media. Since TVS is a function of organic matter, its increase in the filter media is an indication of organic matter build up in the filter media. There were corresponding increases in the percentage TVS of the TS of the filter beds. From Figure 4.3, it was observed that the FM1 which had comparatively highest TVS content at the beginning of the dewatering, ended up in cycle six having the least value while the FM3 which had the least TVS value in the raw sand at the beginning of cycles had the highest TVS value after the sixth cycle. This indicates that the FM3 had the highest potential of organic matter accumulation.

Table 4.1.17: Organic matter (TVS) (g/l) of the filter beds after each cycle

Filter media	TVS raw sand	C1	C2	C3	C4	C5	C6	Mean
FM1	0.038	0.039	0.073	0.084	0.097	0.113	0.118	**0.087**
FM2	0.031	0.045	0.073	0.083	0.108	0.123	0.123	**0.093**
FM3	0.019	0.052	0.093	0.090	0.129	0.139	0.149	**0.109**
LSD of TVS in the sand of different filter media = 0.008								

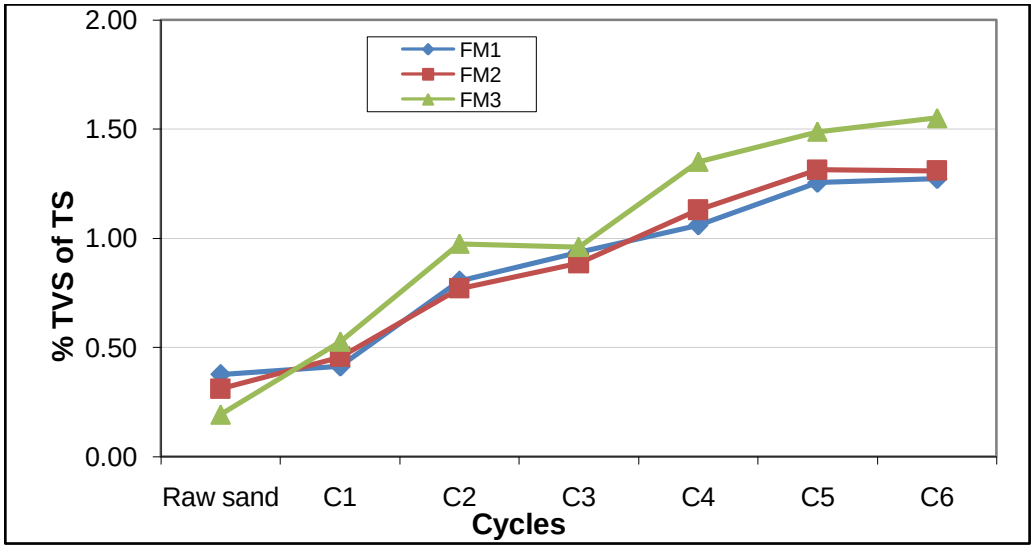


Figure 4.3: Percentage organic matter (TVS) accumulation in the filter beds after each cycle.

4.1.1.6 Biosolids accumulated by filter media

The average percentage biosolid from FS accumulated by the filter media FM1, FM2 and FM3 were 85, 81 and 80 respectively. The percentage TVS content of the biosolids accumulated on the filter media varied as, 76, 67 and 63 for FM1, FM2 and FM3 respectively (Figure 4.4). The FM1 had the highest accumulation in both the TS and the TVS, followed by FM2 and FM3.

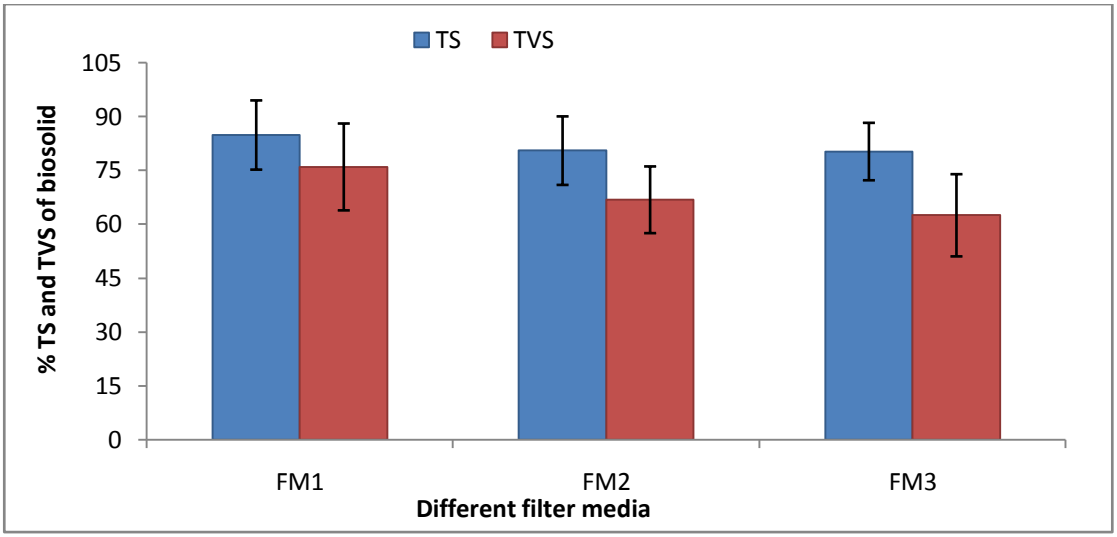


Figure 4.4: Average biosolid accumulation efficiency of the filter media

The accumulation of the fresh biosolids by the different filter media FM1, FM2 and FM3 were 410, 414 and 400 (g) respectively. Dry biosolids were 155, 148 and 147 (g) with organic matter content (TVS) being 103, 90 and 84 (g) for FM1, FM2 and FM3 respectively (Table 4.1.18). FM1 had the highest percentage TVS of biosolid accumulated by the different filter media, (Table 4.1.18). Furthermore, the percentage TVS accumulated by each filter medium from the influent FS were in the order of 76.0, 66.8 and 62.5 for FM1, FM2 and FM3 respectively (Table 4.1.18). Annual organic matter accumulation per m^2 per year for FM1, FM2 and FM3 were, 154.5, 135, and 143.5 (kgTVS/m^2 year) respectively (Table 4.1.18).

Table 4.1.18: Biosolid accumulation performance of the different filter media

Parameter	FM1	FM2	FM3
Total dry Solid of FS (g) (Influent)*	183	183	183
Total org.matter of FS (g) (Influent)*	135	135	135
Dry biosolid TS (g)	155	148	147
TVS of biosolid (g)	103	90	84
%TVS accumulated	76	67	63
TVS (org. matter) wt(g/yr).	3708	3240	3444
kg/yr.(TVS)	3.71	3.24	3.44
A (m^2)of filter bed*	0.024	0.024	0.024
kgTVS/m^2yr	154.5	135	143.5

*Same for all filter media; yr = year

4.1.2 Discussion of results on effective particle size ranges of sand that will improve FS dewatering efficiency.

4.1.2.1 Sand characteristics

Sands for municipal anaerobic digester effluent dewatering beds should ideally have effective size of between 0.30 and 0.75 mm and a uniformity coefficient less than 4, (Metcalf and Eddy, 2003; Crites and Tchobanoglous, 1998; USEPA, 2002; Ohio State University Bulletin, 1999). The uniformity coefficient (u_o) values obtained for FM1, FM2 and FM3 were all less than 4. Sand with u_o less than 4 has all the grains about the same size and therefore reduce or avoid clogging whereas sand with greatly varying sizes allow smaller particles to fill the spaces between the larger particles resulting in smaller and more convoluted pore spaces, making it easier for the filter to clog (National Small Flows Clearinghouse, 1997; Darby *et al.*, 1996). The permeabilities of FM1, FM2 and FM3 were 9.0 x 10^{-2}, 14.0 x 10^{-2} and 36.0 x 10^{-2} (cm/sec) respectively.The permeabilities of the media tended to be highest in FM3, followed by FM2 and then FM1. There are many areas in Ghana, both inland and coastal with soils (sand) having characteristics described above. The availability of the sand in many localities makes the dewatering of faecal sludge using filter beds, a feasible and promising technology to reduce large volumes of faecal sludge being disposed of indiscriminately into the environment and produce biosolids for crop production.

4.1.2.2 Characteristics of faecal sludge (FS)

The FS obtained from septage and PTS analyzed had very high values for TS, SS, COD, NH_3-N, TKN NO_3–N etc. These concentrations are several (10 -100) times higher than the strength of sewage (Strauss *et al.*, 1997). Such sludge with high

contaminant loads, pose treatment challenges but has high potential of generating large quantities of biosolid that may be useful in agriculture.

The high concentration of NH_3-N in FS is an additional problem to its treatment. This is because, algae in facultative ponds can not tolerate ammonia concentration greater than 30 mg/l (Montangero and Strauss, 2002; Awuah, 2006), though some special species are able to adapt themselves to concentrations up to 50 mg/l (Mara and Pearson, 1986). Thus ammonia concentration of 2,830 mg/l as found in this type of sludge is excessively high and will hamper algal and bacterial growth. At such concentrations, malodor and eye irritations are added challenges of the sludge treatment (McGinn *et al.,* 2003). These characteristics are major contributors to the failure of treatment of FS in pond systems in Ghana and many other developing countries and have resulted in the need to use alternatives like the unplanted drying beds for effective sludge management.

4.1.2.3 Dewatering time (DT) of the different filter media in different cycles

Generally, the DT for the entire filter media decreased from C1 to C5 until it increased in C6. The initial long dewatering times of the entire filter media especially C1, might be due to comparatively low temperatures (20.3 ^{o}C – 22.5 ^{o}C minimum and 27.4 ^{o}C - 31.6 ^{o}C maximum) and high humidity (88.5 – 91%) during the project period (Meteorological report, 2006, Ashanti Region, unpublished). Moreover, for municipal sludge dewatering, fine organic particles are created and sand-bed drainability deteriorates during the early stages of digestion leading to increase of DT (Rudolfs and Heukelekian, 1934). The fine organic particles blind the filter cake thereby resisting the flow of water. However, as digestion proceeds, the fine organic particles

are destroyed, due to agglomeration of the fine organic particles and the dead microorganisms to form bigger particles which can be removed easily and dewaterability of the sludge improves resulting in decrease of DT (Rudolfs and Heukelekian, 1934; Karr and Keinath, 1978). The state of the sludge (fresh or partially digested) also affect the dewatering time (Doku, 2003). The sharp rise in the DT in C6 might be due to factors such as average low temperature and high humidity (Veenstra, 1998). The state of the FS (level of digestion) (Doku, 2003) and possibly, the onset of clogging in the filter bed, due to accumulation of suspended solids in the filter (Siegrist, 1987). Generally, FM3 dewatered in the shortest time as compared to the others but was not significantly different, leaving the possibility that any of them could have the least dewatering time.

4.1.2.4 Dewatering efficiencies of the different filter media in removing contaminant loads

Removal of solids (TS, TVS and SS)

Total solid levels in the percolate from cycles 1-6 were high for discharge into the environment and therefore needed further treatment before discharge because high TS level in the effluent means high contaminant load whether chemical or biological which cause depletion of oxygen and eutrophication in water bodies.

Although removal efficiencies of TVS for all the filter media were high, the average TVS levels of 4.41, 4.83 and 5.86 (g/l) in the percolates of FM1, FM2 and FM3 respectively in (Table 4.1.5) were too high for discharge into the environment. High TVS is an indication of high organic matter which can cause oxygen depletion and fish kill when discharged into water bodies. Although the average TS, TVS and SS

values in the percolate were high, the filter media FM1, FM2 and FM3 were able to dewater the FS of between TS loading rate of 217 - 360 kg TS/m^2/year in 9.8, 9.9 and 9.1days respectively. This compares well with and even shows an improvement over the results of experiment conducted in Accra, Ghana, where FS of TS loading rate of 130 kg TS/m^2/yr was dewatered in 9 days (Larmie, 1994). The average percentage removals of TVS for FM1 FM2 and FM3 within the range of 73.6% - 78.2% (Table 4.1.5), compare well with the TVS removal achieved by sand bed dewatering research by Hills and Kermerle (1981). The generally high average removal efficiencies of SS between 88.9% – 95.1 % (Table 4.1.6) in all the filter media is considered very efficient for dewatering of FS. This is also comparable to studies conducted in Accra (Larmie, 1994) and in Kumasi (Esseku, 2002). Ability of FM1, to outperform FM2 and FM3 in the removal of SS, might be due to its ability to remove finer sludge particles than the other media. As particle size of sand bed increases, the dry weight recovered by dewatering reduces (Hills and Kermerle, 1981). Kumasi, where this research was conducted has the capacity of producing about 1,200 m^3 faecal sludge but currently only 300 m^3 is received at Dompoase stabilisation ponds. This volume is likely to produce 4,007,700kg total solids annually which will require between 11,133 m^2 – 18,469 m^2 of surface area to dewater effectively.

FM1 had the slowest rate of increase in dewatering time when increasing suspended solids are applied to the three filter media. This could be due to the slow rate of accumulation of organic matter in the pore spaces of the filter media of particle size of 0.1 - 0.5 mm (FM1) (Kuffour *et al.*, 2009).

Removal of salts or Reduction of Electrical Conductivity

The removal efficiencies achieved by FM1, FM2 and FM3 were comparatively low, with respect to percentage removals achieved for the solids. Conductivity is the result of ionic mobility of dissolved salts. Thus the higher the dissolved salts, the higher the electrical conductivity (Kiely,1998). Poor reduction of the electrical conductivity is an indication of poor removal of salts. High magnitude of conductivity between 13.56 – 16.8 mS/cm in the percolates will cause osmotic stress in plants when discharged into the environment. However, the poor removal of the salts means little was retained in the biosolids which indicates that salt accumulation in soils will be minmised.

Removal of Degradable Organics (COD, DCOD)

There were high percentage removals of COD with FM1 showing the highest removal efficiency in almost all the cycles. This efficiency achieved by the drying beds might be due to the ability of the filter media to remove the solids from the sludge and the availability of the dissolved organics for microbial action (Karim *et al.*, 2005; Nakhla and Farooq, 2003). The ability of FM1 to remove more solids coupled with its particles presenting the largest surface area for microbial action might have contributed to its highest performance. These efficiencies achieved by all the filter media were higher than the 50 – 69% COD removal achieved in pilot study using slow sand filtration with effective sizes of 0.31 and 0.56 mm by Farooq and Al Yousef, (1993). A good reduction in the DCOD is an indication of the availability of the dissolved substrate for bacterial action (Karim *et al.*, 2005). The removal efficiencies obtained in FM1, FM2 and FM3, well explain that some fine particles of dissolved organics were degraded by microbes. The high efficiencies of FM1 over the

FM2 and FM3 might further support the role played by the effective size of the filter media (Salvato *et al.*, 2003).

Removal of Nutrients (NH_3-N, TKN, NO_3-N, TP)

All the filter media achieved high NH_3-N removal efficiencies yet the concentrations in the percolates were very high. This might be due to the high concentrations in the influent FS and ammonification of organic-N during dewatering (Epstein, 2003). NH_3-N removal is believed to be basically due to volatilization and nitrification by nitrifying bacteria (Awuah, 2006). High levels of ammonia nitrogen could be toxic to the degrading bacteria in the wastewater. This may also affect the flora and fauna in the water bodies. Nitrification has been found to be of most sensitivity to filtration and sand size (Nakhla and Farooq, 2003). Smaller particle sizes ensure nitrification and filtration and thus improve nitrogen removal.

The filter media achieved high efficiency in the TKN removal, between 60.8% – 78.1%. FM1 showed the highest potential for TKN removal with average removal efficiency of 78.1%. This might be due to organic-N removal through high TVS removal and mineralization of organic-N. The comparatively low percentage removal of nitrate (NO_3-N) might be due to its continual production in the percolate through mineralization of organic nitrogen (Epstein, 2003). Baghat *et al*. (1999) observed that ammonia- oxidizers (*Nitrosomonas sp.*) were not only more abundant than nitrite-oxidizers (*Nitrobacter sp.*), but also better distributed throughout the filter depth. This might have contributed to the higher nitrite removal (Table 4.1.15) than nitrate removal (Table 4.1.14). Irrespective of the percentage removals of nitrate and total

phosphorus (TP) achieved, the concentrations in the effluents were still very high which were due to the high concentration of the raw sludge.

4.1.2.5 Organic matter (TVS) accumulation in the sand (Filter medium)

Since TVS is a function of organic matter (Kiely, 1998), its increase in the filter media is an indication of organic matter build up. FM3 had the highest potential of organic matter accumulation (Figure 4.3). Elliason (2002) asserted that, the most important feature in granular media is not the grain particles, but rather the pore space in the media. The treatment of wastewater occurs in the pores where suspended solids are trapped, microorganisms grow and air and water flow (Emerick, 1997). Ball (1997), determined that as the surface area of the packed spheres increases, the size of the pores per unit volume area significantly decreases with smaller diameter spheres. The above assertion indicates that FM1 had smaller pores per unit volume area to accommodate suspended solids and microorganisms than FM2 and FM3. That accounted for the rapid rate of accumulation of organic matter in FM3 more than the FM1. The organic matter accumulation is likely to fill the pore spaces between the particles of the filter bed which can subsequently result in clogging. This is because Bouma (1979) reported that biological clogging was due to accumulation of suspended solids and biological growth in anaerobic conditions. Although clogging did not take place during the running of the six cycles, there was the possibility for its occurrence since there was significant increase in the organic matter content in the filter media especially in the FM3.

4.1.2.6 Biosolids accumulated by filter media

The percentage accumulation of the biosolids by the different filter media FM1, FM2 and FM3 indicated that FM1 had the highest potential to generate biosolids (Figure 4.4). Since the same FS was applied to all the filter media, it was assumed that the TVS content of the bioslids was directly proportional to the biosolids accumulated on the respective beds. FM1 gave the highest percentage TVS of TS of biosolid accumulated by the different filter media. This was confirmed by the assertion by Ball (1997), that sand with smaller particle size (larger surface) posses decreased pores per unit volume. Increased pores per unit volume, characteristic of FM3, (Siegrist *et al*., 1991; Nevo, 1964) allowed more suspended solids and organic matter to move along with the percolate thus lesser quantity accumulated. The biosolid accumulated by FM1 proved that it had the highest potential of biosolid generation amongst the others. Annual biosolid production with respect to DT, was estimated and based on that the organic matter accumulation per m^2 per year was determined where FM1 showed the highest potential to generate biosolids of 154.5 kgTVS/m^2 year (Table 4.1.18).

4.2 The Solid Loading Rate (SLR) of Faecal Sludge That Will Improve the Dewatering Efficiency of Unplanted Filter Beds.

Loading rate is an effective tool that controls dewatering process. Dewatering process can be affected by hydraulic loading rate or solid loading rate. This research critically looked at the effect of solid loading rate on the dewatering performance of unplanted filter beds in the dewatering of faecal sludge. The solid loading rate is the total solids present in a given quantity (litre) of liquid waste. The different mixing ratios of the public toilet sludge and the septage resulted in three different faecal sludges of different solid concentrations.

4.2.1 Results of solid loading rates of faecal sludge that will improve the on dewatering efficiency of unplanted filter beds.

The three different faecal sludges were dewatered on the same filter medium (FM1) that was selected in the previous objective. The results of six dewatering cycles, physicochemical quality of the faecal sludges, percolates and biosolids have been presented in this section.

4.2.1.1 *Faecal Sludge (FS)*

The FS used consisted of PTS and septage mixed in three different ratios of 1:1, 1:2 and 1:3 by volume making SLR1:1, SLR1:2 and SLR1:3 herein referred to as SLR1, SLR2 and SLR3, respectively. The septage collected from different areas of Kumasi Metropolis appeared to have very low concentration of solids and more stabilized as compared to the PTS which was also collected from different areas in Kumasi. In all the parameters SLR1 had the highest concentration of solids with SLR3 being the least (Table 4.2.1). However, with parameters like phosphorus and nitrate-nitrogen, the SLR3 rather had the highest concentration with the SLR1 having the least (Table 4.2.1). This is because the septage which has the higher percentage in the mixing has higher concentration of these nutrients.

Table 4.2.1: Characteristics of faecal sludge (FS) of different SLR over six cycles (N = 6)

Parameter	Faecal sludge Samples		
	SLR1	**SLR2**	**SLR3**
TS (g/l)	39.41±7.19	33.21±14.66	26.93±20.25
TVS (g/l)	28.54±5.38	22.70±8.17	20.59±14.84
SS (g/l)	21.32±8.62	18.62±11.66	16.27±15.53
EC (mS/cm)	22.11±3.67	17.01±2.57	13.33±1.28
Temp.(^oC)	29.9±2.4	29.8±2.3	29.8±1.8
pH	7.9±0.2	7.8±0.2	7.8±0.2
COD(g/l)	43.40±15.48	37.67±19.05	33.47±28.94
DCOD(g/l)	9.7±7.09	7.7±4.19	6.1±3.36
NH_3-N(g/l)	1.31±0.94	1.06±0.78	0.92±0.73
TKN(g/l)	1.76±0.94	1.38±0.86	1.21±0.87
NO_3(g/l)	1.06±0.47	1.04±0.57	1.19±0.84
TP(g/l)	3.38±1.40	3.33±1.25	4.01±2.12

± = *Standard deviation (SD); N=Number of samples*

4.2.1.2 Dewatering Time

The main objective for determining the dewatering time was to find out which of the three different solid loading rates (SLR) could offer itself for most effective dewatering when performing under the same conditions. The average dewatering time for SLR1, SLR2 and SLR3 were 7, 5 and 4 respectively. Results of ANOVA indicated that there was significant difference between the different solid loading rates with respect to dewatering time at 5% significance level ($p = 0.023$) with the SLR3 outperforming the others. After continuously running six cycles, it was observed that the FS with the SLR3 dewatered fastest followed by SLR2 and then SLR1. FS of SLR1 took more number of days to dewater in almost all the cycles, except in cycle 4 (Figure 4.5). The dewatering in different cycles might have been affected by the type of sludge and prevailing conditions, since the type of FS used in each cycle differed from each other. The environmental conditions also varied for each of the cycles.

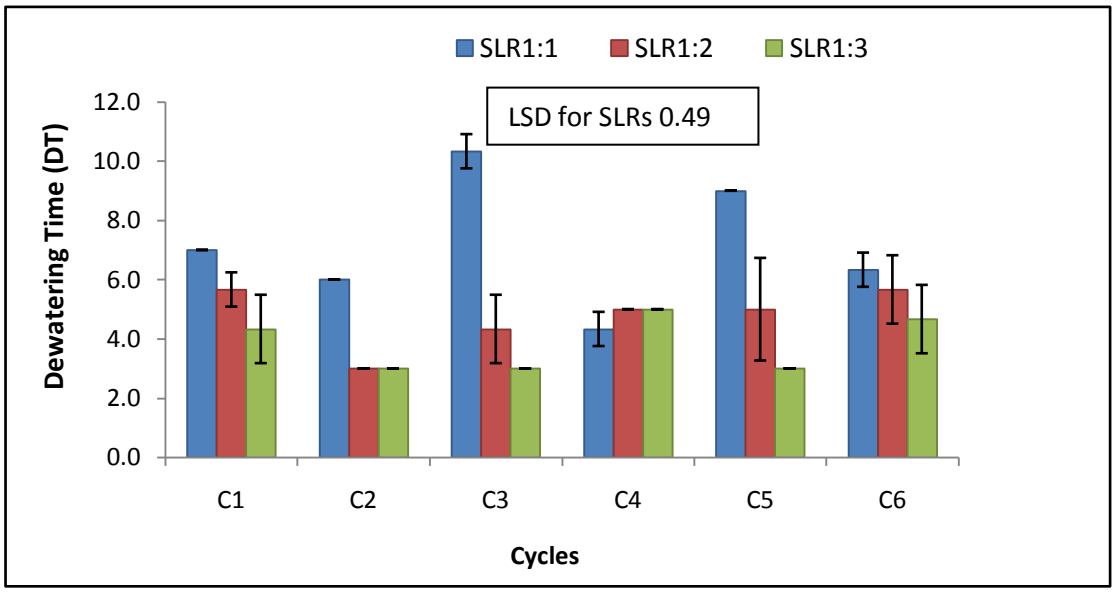


Figure 4.5: Dewatering time (days) of different SLRs in six cycles
(Those without error bars have the sameor almost the same dewatering days for the replicates)

4.2.1.3 Physicochemical Parameters of faecal sludge and percolate

Total Solids (TS)

The percentage removal of TS from SLR1 SLR2 and SLR3 as 83.42%, 80.42% and 78.10% respectively were quite high with SLR1 performing best. However, the average TS values in the percolates of SLR1, SLR2 and SLR3 (6.38, 5.83 and 4.59) g/l (Table 4.2.2) from cycles 1-6 were still high for discharge into the environment. For discharge purposes, SLR3 had the least average TS value in the percolate, but the removal efficiency for TS of this loading rate was low compared to the SLR1 and SLR2. In almost all the cycles, SLR1 showed the highest percentage removal of the TS followed by SLR2 and SLR3 (Table 4.2.2). Percentage TS removal between the three loading rates were significant ($p = 0.0001$).

Table 4.2.2: Total Solids (TS) (g/l) of faecal sludge and percolate

Cycles	Faecal sludge (g/l)			Percolate (g/l)			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	37.11	28.10	21.37	5.75	6.04	4.24	84.50	78.50	80.14
C2	28.12	20.11	16.05	6.53	5.71	4.94	76.79	71.59	69.25
C3	45.43	37.23	21.47	5.93	6.43	4.26	86.96	82.74	80.18
C4	48.77	60.56	67.99	7.93	6.43	4.72	83.75	89.39	93.05
C5	38.81	23.07	16.50	6.40	5.13	4.56	83.51	77.74	72.35
C6	38.20	30.20	18.20	5.73	5.27	4.80	84.99	82.56	73.63
Mean±SD	**39.41±7.19**	**33.21±14.66**	**26.93±20.25**	**6.38±0.83**	**5.83±0.56**	**4.59±0.29**	**83.42±3.47**	**80.42±5.99**	**78.10±8.53**

LSD of percentage removal = 2.22

Total Volatile Solids (TVS)

The average percentage removal of TVS for SLR1, SLR2 and SLR3 were 89.69, 88.52 and 86.63, respectively (Table 4.2.3). These levels of removal efficiencies were high. In almost all the cycles, SLR1 had the highest TVS removal efficiency. Although the average TVS in the percolate of SLR1 was the highest among the rest (Table 4.2.3), it offered itself for highest removal by the filter beds. The percolates TVS for the different solid loading rates showed varying concentration levels from cycle 1 – 6 with averages of 2.82 g/l, 2.64 g/l and 2.51g/l for SLR1, SLR2, and SLR3, respectively (Table 4.2.3).

Table 4.2.3: Total Volatile Solids (TVS) (g/l) of faecal sludge and percolate

Cycles	Faecal sludge			Percolate			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	25.71	21.51	17.82	1.78	1.73	3.22	93.06	91.95	86.43
C2	20.25	14.88	11.93	4.18	3.16	2.70	79.34	78.77	77.40
C3	33.44	22.73	14.19	3.98	2.40	1.86	88.11	89.44	86.91
C4	33.78	38.02	50.63	4.23	5.78	3.95	87.48	84.79	92.19
C5	31.84	16.68	14.99	0.87	0.92	1.55	97.26	94.51	89.69
C6	26.20	22.40	14.00	1.87	1.87	1.80	92.88	91.67	87.14
Mean±SD	**28.54±5.38**	**22.70±8.17**	**20.59±14.84**	**2.82±1.48**	**2.64±1.71**	**2.51±0.95**	**89.69±6.22**	**88.52±5.79**	**86.63±5.02**

LSD of percentage removal = 1.61

Suspended Solids (SS)

The average percentage removal for SLR1, SLR2 and SLR3 were 96.11, 93.84 and 93.71 respectively, (Table 4.2.4). These levels of removal efficiency achieved based on mixing ratios of PTS and septage described how effective SLR could be employed to improve FS dewatering on drying beds. The average suspended solids (SS) in the percolates of SLR1 SLR2 and SLR3 with respective values of 0.87, 0.80 and 0.78 (g/l), in (Table 4.2.4) high, with the removal efficiencies also being generally high.

Table 4.2.4: Suspended Solids (SS) g/l of faecal sludge and percolate with percentage removal

Cycles	faecal sludge			Percolate			Percentage removal		
	SLR1	**SLR2**	**SLR3**	**SLR1**	**SLR2**	**SLR3**	**SLR1**	**SLR2**	**SLR3**
C1	19.00	15.00	9.40	0.47	0.44	0.59	97.54	97.07	93.76
C2	11.40	9.90	3.40	0.27	0.32	0.22	97.66	96.80	93.63
C3	32.00	24.00	18.00	2.67	2.30	1.73	91.67	90.40	90.37
C4	31.50	40.00	46.50	0.50	0.40	0.70	98.41	99.00	98.49
C5	19.50	11.40	9.40	0.25	0.29	0.34	98.74	97.43	96.38
C6	14.50	11.40	10.90	1.07	1.07	1.13	92.64	90.64	89.60
Mean±SD	**21.32±8.62**	**18.62±11.66**	**16.27±15.53**	**0.87±0.93**	**0.80±0.79**	**0.78±0.56**	**96.11±3.11**	**95.17±3.80**	**93.71±3.41**

LSD of percentage removal = 1.16

Correlation between dewatering time and suspended solids

In dewatering different solid loading rates of faecal sludge (FS), positive correlation (r^2 = 0.81 for SLR1; r^2 = 0.57 for SLR2 and r^2 = 0.81for SLR3) was observed between dewatering time and suspended solids for all the solid loading rates where SLR1 and SLR3 showed higher correlation than SLR2, (Figure 4.6). A unit change in suspended solids caused the least increase of dewatering time in SLR3 as compared to SLR1 and SLR2 (Figure 4.6).

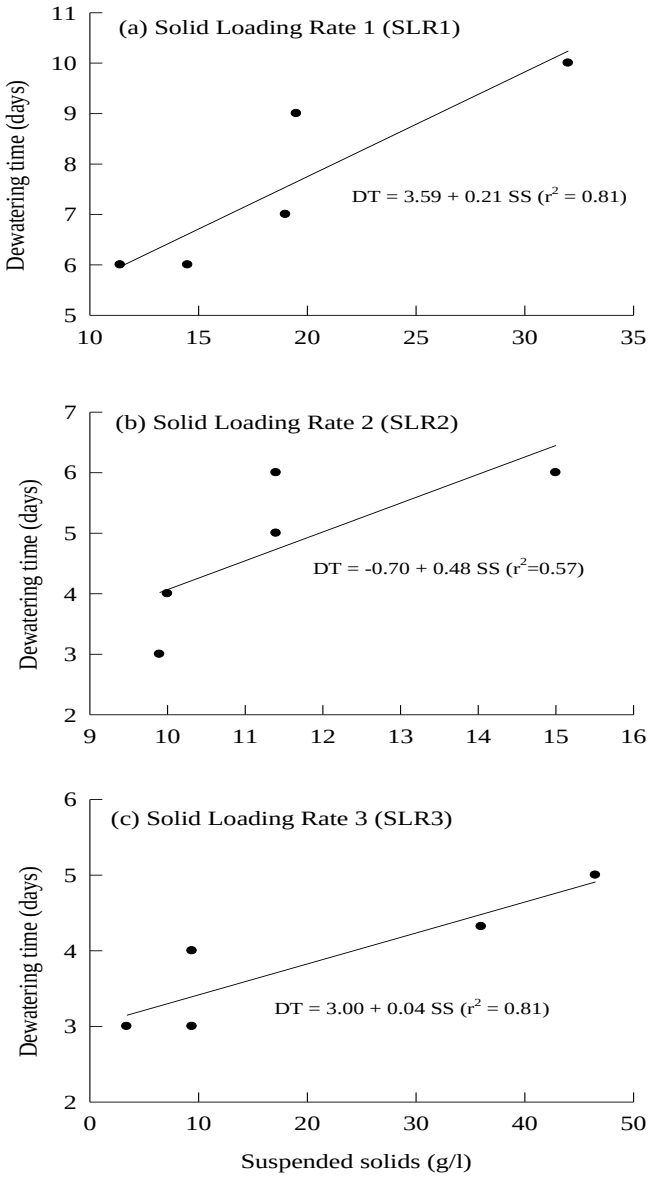


Figure 4.6: Relationship between suspended solids of solid loading rates and their dewatering times

Chemical Oxygen Demand (COD)

The chemical oxygen demand in the percolates of SLR1, SLR2 and SLR3 were 5.58, 5.04 and 4.56 (g/l), respectively, (Table 4.2.5). These were considered too high for discharge into the environment considering the Ghana EPA (2001), guideline for discharge, which recommends 250 mg/l (0.25 g/l) for COD. However,

notwithstanding the high COD levels in the percolates, the average percentage COD removal were very high for the loading rates of SLR1, SLR2 and SLR3 which were 86.44%, 84.49% and 80.10% respectively (Table 4.2.5). Although in most of the cycles, SLR1 gave the highest percentage removal, it had the least percentage removal in cycle 4. However, the COD for the faecal sludge of SLR1 was least, compared with the SLR2 and SLR3. This is because the septage used in the mixing had higher solid concentration than the PTS thus increasing the solid concentration in the faecal sludge as the septage content increased. This also indicates that the removal efficiency depended on the concentration in the raw sludge applied for dewatering.

The DCOD showed a decreasing trend in terms of percentage removal from SLR1 to SLR3, (Table 4.2.6). The SLR1 removed more DCOD than the SLR2 and SLR3. Although the DCOD removal follows the same trend as the COD, the percentages of DCOD removed were comparatively lower.

Table 4.2.5: Chemical Oxygen Demand (COD) (g/l) of faecal sludge and percolate with percentage removal

Cycles	Faecal sludge			Percolate			Percentage removal		
	SLR1	**SLR2**	**SLR3**	**SLR1**	**SLR2**	**SLR3**	**SLR1**	**SLR2**	**SLR3**
C1	69.60	42.40	33.60	8.40	8.53	7.33	87.93	79.87	78.17
C2	34.40	30.40	27.20	6.93	5.73	5.60	79.84	81.14	79.41
C3	30.00	22.00	12.00	4.27	3.73	3.20	85.78	83.03	73.33
C4	54.40	73.60	90.40	3.73	2.67	2.40	93.14	96.38	97.35
C5	32.80	24.00	16.00	4.40	4.13	4.27	86.59	82.78	73.33
C6	39.20	33.60	21.60	5.73	5.47	4.53	85.37	83.73	79.01
Mean±SD	**43.40±15.48**	**37.67±19.05**	**33.47±28.94**	**5.58±1.81**	**5.04±2.05**	**4.56±1.75**	**86.44±4.29**	**84.49±5.99**	**80.10±8.88**
	LSD of percentage removal = 1.85								

Table 4.2.6: Dissolved Chemical Oxygen Demand (DCOD) (g/l) of faecal sludge and percolate with percentage removal

Cycles	Faecal sludge			Percolate			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	24.00	16.00	12.80	4.80	4.80	4.00	80.00	70.00	68.75
C2	7.20	6.40	4.80	3.33	3.33	3.60	53.70	47.92	25.00
C3	7.20	7.20	5.60	1.33	1.60	1.60	81.48	77.78	71.43
C4	6.40	5.60	4.80	2.93	2.13	1.87	54.17	61.90	61.11
C5	5.20	4.40	3.60	3.73	2.80	2.40	28.21	36.36	33.33
C6	8.00	6.40	4.80	2.67	2.40	2.40	66.67	62.50	50.00
Mean±SD	**9.67±7.09**	**7.67±4.19**	**6.07±3.36**	**3.13±1.16**	**2.84±1.12**	**2.64±0.96**	**60.70±19.94**	**59.41±15.02**	**51.60±19.09**

LSD of percentage removal = 5.19

Electrical Conductivity (EC) and Turbidity (NTU)

The EC values in the percolates from the filtration of the SLR1, SLR2 and SLR3 were 14.99, 15.22 and 16.56 (mS/cm), respectively (Table 4.2.7). All are high compared to the EPA permissible levels guideline for discharge into the environment, (700(µS/cm, EPA Ghana, 1997). The percentage removals of 28.18, 28.35, and 25.98 from the SLR1, SLR2 and SLR3 respectively, were however low (Table 4.2.7). The salts were poorly removed in all the different solid loading rates. There was no significant difference in EC removal among the three treatments at 5% significance level (p = 0.56). This shows that the differences in their removal efficiencies were highly insignificant. The turbidity of the percolates from the SLR1, SLR2 and SLR3 were 541, 394 and 288 (NTU) respectively (Table 4.2.8). These values are very high for discharge into the environment when compared with the EPA permissible guideline of discharge which is 75 NTU, (EPA Ghana, 1997). However, the removal efficiencies from SLR1, SLR2 and SLR3 as 92.32%, 89.84% and 89.38% respectively (Table 4.2.8), were high. The turbidity of the raw sludges were so high that even these high percentage removals still left so much in the percolates. There were significant difference in the percentage removals of the treatments at 5% significance level (p value = 0.017).

Table 4.2.7: Electrical conductivity (EC) (mS/cm) of FS and percolate with percentage removal.

Cycles	Faecal sludge (mS/cm)			Percolate (mS/cm)			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	22.10	17.04	13.71	10.21	11.06	12.13	49.62	45.83	40.43
C2	25.60	18.78	15.06	18.92	18.43	18.54	27.23	19.81	17.93
C3	26.00	20.80	12.67	13.25	12.95	14.76	47.49	40.13	24.41
C4	16.53	13.80	11.78	15.42	14.67	15.43	8.21	14.54	23.80
C5	23.10	16.92	13.72	16.19	18.61	20.90	19.62	32.45	29.06
C6	19.32	14.71	12.40	15.97	15.62	16.58	16.89	17.36	20.24
Mean±SD	**22.11±3.67**	**17.01±2.57**	**13.22±1.18**	**14.99±2.96**	**15.22±2.99**	**16.56±3.06**	**28.18±16.93**	**28.35±13.01**	**25.98±8.04**

LSD of percentage removal = 3.1

Table 4.2.8: Turbidity (NTU) of FS and percolate percentage removal

Cycles	faecal sludge (NTU)			Percolate (NTU)			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	3420	1500	1210	387	402	371	88.68	73.22	69.37
C2	3800	2700	2000	700	508	398	81.59	81.20	80.11
C3	11210	5230	3130	505	309	185	95.49	94.10	94.09
C4	20710	17610	16900	627	512	391	96.97	97.09	97.69
C5	14950	10680	7880	598	337	197	96.00	96.85	97.50
C6	8920	8710	7660	428	295	187	95.20	96.61	97.56
Mean±SD	**10502±6664**	**7738±5961**	**6463±5848**	**541±121**	**394±97**	**288±108**	**92.32±6.03**	**89.84±10.16**	**89.38±11.92**

LSD of percentage removal = 2.1

Temperature (^oC) and pH

The average temperatures of the percolates for the different solid loading rates SLR1, SLR2 and SLR3 were 27.6, 27.3 and 27.1 (^oC) while the temperatures of the raw samples for the same loadings were 29.9, 29,8 and 29.8 (^oC) respectively, which were comparatively higher (Table 4.2.9). In all the cycles the raw samples produced temperatures higher than the corresponding percolates. The average pH for SLR1, SLR2 and SLR3 in the percolates were 8.4 for all while the pH for the same raw samples used for the dewatering were 7.9, 7.8 and 7.8, respectively which were lower than the pH in the percolate (Table 4.2.9). There was little variation in the pH.

Table 4.2.9: Temperature (^oC) and pH of faecal sludge and percolate

	Temperature (^oC)						pH					
	Faecal sludge			Percolate			Faecal sludge			Percolate		
Cycles	**SLR1**	**SLR2**	**SLR3**	**SLR1**	**SLR2**	**SLR3**	**SLR1**	**SLR2**	**SLR3**	**SLR1**	**SLR2**	**SLR3**
C1	29.3	29	28.7	29.0	28.2	28.6	7.6	7.6	7.7	8.4	8.4	8.3
C2	27.6	27.5	28.8	26.0	25.5	25.5	8.2	8.1	8.1	8.6	8.7	8.7
C3	29.6	29.6	29.6	27.9	27.7	26.0	8.1	7.8	7.5	8.7	8.7	8.7
C4	28.9	29.1	28.8	28.3	28.2	28.2	7.7	7.6	7.5	8.4	7.5	7.8
C5	34.6	34.2	33.5	27.4	27.3	27.4	7.9	7.9	8.0	8.0	8.5	8.6
C6	29.3	29.4	29.4	27.1	26.9	27.2	8.0	8.0	8.0	8.3	8.4	8.5
Mean±SD	**29.9±2.4**	**29.8±2.3**	**29.8±1.8**	**27.6±1.0**	**27.3±1.0**	**27.1±1.2**	**7.9±0.2**	**7.8±0.2**	**7.8±0.3**	**8.4±0.2**	**8.4±0.4**	**8.4±0.3**
	LSD of percolate = 0.4									**LSD of percolate = 0.1**		

Ammonia Nitrogen (NH_3-N) and Total Kjedhal Nitrogen (TKN)

The average Ammonia nitrogen content in the percolate of SLR1, SLR2 and SLR3 were 0.38, 0.34 and 0.30 (g/l) respectively (Table 4.2.10). These are considered too high for discharge into the environment, based on EPA permissible guideline value of 0.0015 g/l (1.5mg/l) in effluent (EPA Ghana, 1997). With regards to effluent values, ammonia nitrogen concentration in the percolate of SLR3 was the least which might be due to the corresponding raw sludge which was also the least among the raw samples. Notwithstanding the high ammonia content in the effluents, the removal efficiencies of 70.37% for SLR1, 64.40% for SLR2 and 61.81% for SLR3 (Table 4.2.10) were also high.

The TKN also showed the same trend of removal efficiency where SLR1, SLR2 and SLR3 removed 72.35%, 67.83% and 64.85 %, respectively (Table 4.2.11). The SLR1 showed the highest percentage removal. There was no significant differences between the three treatments, with respect to ammonia nitrogen removal at 5% significance level (p value = 0.088).

Table 4.2.10: Ammonia Nitrogen (NH_3-N) (g/l) of faecal sludge and percolate with percentage removal

Cycles	faecal sludge (g/l)			Percolate (g/l)			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	0.73	0.63	0.51	0.33	0.25	0.23	54.23	60.07	54.58
C2	1.05	0.90	0.86	0.15	0.15	0.14	85.73	83.67	84.17
C3	1.18	0.88	0.72	0.42	0.59	0.55	64.05	33.44	22.66
C4	3.19	2.64	2.38	0.89	0.61	0.49	72.16	77.04	79.41
C5	1.04	0.69	0.54	0.28	0.26	0.22	73.02	62.16	59.81
C6	0.71	0.62	0.51	0.19	0.18	0.15	73.02	70.00	70.24
Mean±SD	**1.31±0.94**	**1.06±0.78**	**0.92±0.73**	**0.38±0.27**	**0.34±0.21**	**0.30±0.18**	**70.37±10.52**	**64.40±17.58**	**61.81±22.23**

LSD of percentage removal = 3.23

Table 4.2.11: Total Kjeldhal Nitrogen (TKN) (g/l) of percolate and Faecal Sludge

Cycles	Faecal sludge (g/l)			Percolate (g/l)			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	1.18	0.99	0.81	0.49	0.34	0.30	59.02	65.20	63.34
C2	1.73	1.32	1.18	0.19	0.17	0.16	88.97	86.96	86.74
C3	1.73	1.25	1.00	0.54	0.69	0.66	68.94	44.84	34.08
C4	3.57	3.10	2.94	1.09	0.70	0.61	69.61	77.40	79.37
C5	1.42	0.86	0.63	0.34	0.31	0.27	76.22	63.45	57.08
C6	0.91	0.80	0.68	0.26	0.25	0.22	71.37	69.11	68.51
Mean±SD	**1.76±0.94**	**1.38±0.86**	**1.21±0.87**	**0.48±0.32**	**0.41±0.23**	**0.37±0.21**	**72.35±9.89**	**67.83±14.24**	**64.85±18.51**

LSD of percentage removal = 3.27

Nitrate Nitrogen (NO_3-N), Nitrite Nitrogen (NO_2-N) and Phosphorus (P)

The nitrate-nitrogen levels of 0.55, 0.58 and 0.53 (g/l) obtained in the percolates of SLR1 SLR2 and SL3 respectively (Table 4.2.12) were too high since maximum permissible levels in waste water effluents according to EPA guidelines in Ghana is 0.0001 g/l (0.1mg/l). The percentage removals of 62.14, 54.09 and 51.89 were observed in SLR1, SLR2 and SLR3 respectively. The SLR1 achieved the highest nitrate removal (Table 4.2.12). The percentage removal of nitrite in SLR1, SLR2 and SLR3 at 84.01, 85.42 and 71.20 (Table 4.2.13) were quite high. The SLR3 had the least concentration of nitrite in the percolate for discharge (Table 4.2.13), and its percentage removal was the least. The concentrations of phosphorus in the effluent of

all the loading rates were high (Table 4.2.14). The percentage removals of phosphorus from all the treatments were generally low.

Table 4.2.12: Nitrate Nitrogen (NO_3-N) (g/l) of faecal sludge and percolate with percentage removal

Cycles	Faecal sludge			Percolate			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	11.10	11.05	2.65	0.70	1.03	1.07	93.69	90.71	59.75
C2	0.52	0.46	0.41	0.38	0.31	0.24	25.64	31.88	40.65
C3	3.80	2.80	2.50	0.48	0.53	0.30	85.61	80.95	88.13
C4	1.30	1.00	1.15	0.49	0.68	0.56	62.44	32.17	51.01
C5	0.62	0.35	0.35	0.47	0.40	0.33	24.73	9.74	5.31
C6	4.00	2.50	2.10	0.77	0.52	0.70	80.83	79.07	66.51
Mean±SD	**3.56±4.00**	**3.03±4.06**	**1.53±1.03**	**0.55±0.15**	**0.58±0.25**	**0.53±0.32**	**62.16±30.42**	**54.09±33.55**	**51.89±27.88**

LSD of percentage removal = 5.36

Table 4.2.13: Nitrite Nitrogen (NO_2-N) (g/l) of faecal sludge and percolate with percentage removal

Cycles	Faecal sludge (g/l)			Percolate			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	0.04	0.13	0.08	0.002	0.002	0.004	93.52	98.35	95.12
C2	0.04	0.04	0.03	0.208	0.502	0.300	81.11	76.11	76.92
C3	0.04	0.03	0.02	0.005	0.002	0.002	86.02	92.24	87.80
C4	0.20	0.15	0.15	0.049	0.046	0.088	74.87	69.72	40.54
C5	0.36	0.28	0.23	0.028	0.036	0.057	88.59	87.38	75.15
C6	0.13	0.09	0.07	0.025	0.010	0.034	79.95	88.74	51.64
Mean±SD	**0.14±0.13**	**0.12±0.09**	**0.10±0.08**	**0.053±0.078**	**0.100±0.198**	**0.081±0.112**	**84.01±6.69**	**85.42±10.60**	**71.20±21.06**

LSD of percentage removal = 10.2

Table 4.2.14: Phosphorus (P) (g/l) of FS and percolate with percentage removal

Cycles	Faecal sludge (g/l)			Percolate (g/l)			Percentage removal		
	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3	SLR1	SLR2	SLR3
C1	5.65	3.70	2.94	1.86	1.95	2.31	67.08	47.39	21.54
C2	3.69	3.45	4.62	2.57	2.75	2.18	62.06	63.77	63.56
C3	2.36	1.90	1.46	0.72	0.61	0.56	69.35	67.89	61.64
C4	1.30	1.90	2.40	0.46	1.13	1.63	64.74	40.35	31.94
C5	3.95	5.00	6.60	2.92	3.54	4.26	26.16	29.17	35.48
C6	3.35	4.05	6.05	1.59	1.64	2.23	52.44	59.42	63.20
Mean±SD	**3.38±1.48**	**3.33±1.23**	**4.01±2.07**	**1.69±0.98**	**1.94±1.07**	**2.19±1.21**	**56.97±16.20**	**51.33±14.98**	**46.23±18.74**

LSD of percentage removal = 6.93

4.2.1.4 Organic matter accumulation in sand

The accumulation of organic matter in the top 10cm of the filter media of the different solid loading rates increased gradually from cycle 1 to cycle 6 (Figure 4.7). Although the increases were small, it was identified that SLR1 had the highest accumulation at the end of cycle 6.

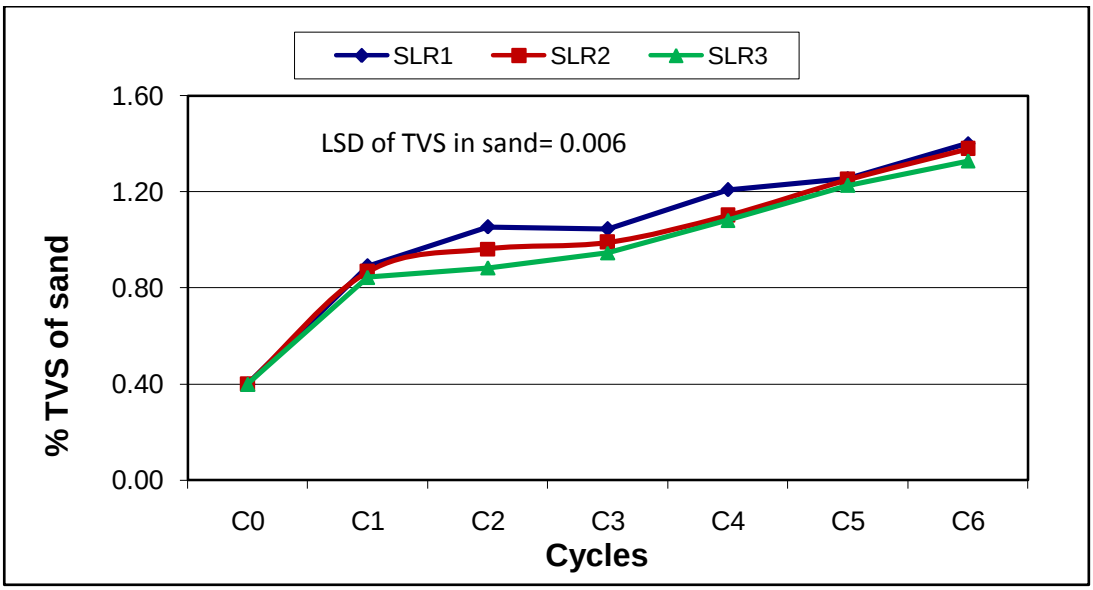


Figure 4.7 Percentage accumulation of organic matter in the filter by the different loading rates.

4.2.1.5 Biosolids accumulated

The percentage accumulated biosolids obtained from SLR1, SLR2 and SLR3 were 89, 84 and 85 respectively while the associated percentages of organic matter accumulated by the same loading regimes were 73, 67 and 67 (Figure 4.8). In both cases the SLR1 produced the highest percentage accumulation.

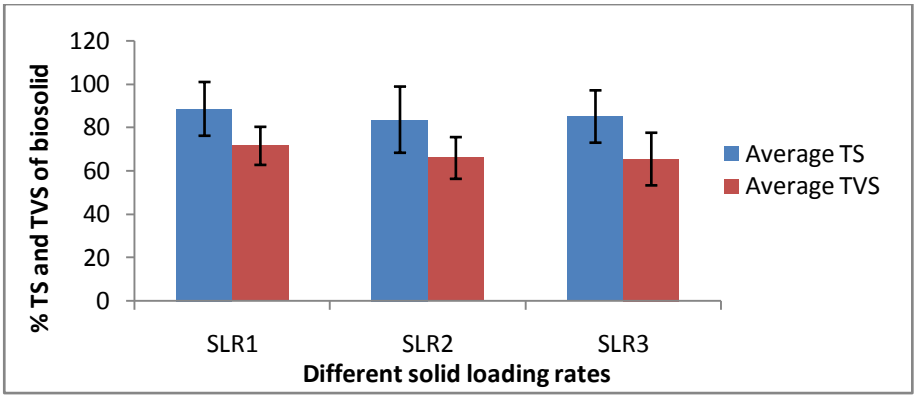


Figure 4.8: Organic matter accumulation by different solid loading rates

4.2.1.6 Annual Biosolids generation estimates

The accumulation of the biosolids from the different solid loading rates, SLR1, SLR2 and SLR3 on the filter media were 174.6, 138.9 and 114.6 (g), respectively. SLR1 had the highest potential to generate biosolids per cycle, and produced the highest percentage TVS of biosolid accumulated from the different solid loading rates, (Table 4.2.15). Furthermore, the percentage TVS accumulated by the filter media from each influent FS were in the order of 72, 66 and 65 for SLR1, SLR2 and SLR3 respectively. However, the annual organic matter generation were estimated to be 224, 228 and 256 (kgTVS/m^2/yr (kg/yr) for SLR1, SLR2 and SLR3 respectively with respect to the dewatering time and the area of filter beds (Table 4.2.15).

Table 4.2.15: Biosolid accumulation of the different SLR, (no of cycles = 6)

Parameter	SLR1	SLR2	SLR3
Ave. dry biosolid TS (g)	174.6	138.9	114.6
TVS of dry biosolid (g)	103	75	67
% TS accumulated	89	84	85
%TVS accumulated	72	66	65
Dewatering time	7	5	4
Cycles/year	52	73	91
kg/yr.(TVS) of the filter beds	5.37	5.47	6.15
Area (m^2) of filter bed	0.024	0.024	0.024
kgTVS/m^{2}yr (kg/yr)	224	228	256

yr = year

4.2.2 Discussion of results of solid loading rates of faecal sludge that will improve the on dewatering efficiency of unplanted filter beds.

4.2.2.1 Faecal sludge (FS)

The FS used consisted of septage and PTS mixed in different ratios. The septage usually has lower concentration is and more stabilized as compared to the PTS. This is based on the fact that more than 50% of the BOD load entering the septic tank is removed by anaerobic digestion during storage (Doku, 2003; Metcalf and Eddy, 2003). A further portion of the BOD is lost through discharge of the supernatant into soil infiltration systems or into surface drains, (Heins *et al*., 1998). Intrusion of water from underground discharge also reduces the septage concentration. These underlying factors are reflected in the reduced concentrations of the parameters as the septage ratio in the faecal sludge mixture increases. Because the septage is more digested and stabilised it generally dewaters easier than PTS therefore as its content increases in the mixture, the dewaterability increases. This notwithstanding, the septage may sometimes have higher solid concentration than the PTS and thus affect the solids in the mixture in the vice versa.

Solids

The FS from various mixtures of septage and PTS analyzed had very high values for TS, SS and TVS. This is common with contents of public latrines (VIP, pit latrines, aqua privies etc.) since little or no water is used for flushing (Doku, 2003). TVS is very high due to high organic content of FS in Ghana and other developing countries. These values compared well with results obtained by Heins *et al* (1998) in their studies conducted in Accra. Similar results were also obtained in Thailand and Benin (Thamarat, 2004; CREPA, 2002).

Degradable organics

The FS had high COD and DCOD which decreased as the PTS ratio decreased. Such high COD values have been recorded in FS from Bankok, Manila and Accra (Heins *et al*., 1998; Cofie, et. al., 2006). The concentrations of COD are several (10 - 100) times, the strength of sewage (Strauss *et al*., 1997). This is mainly due to the PTS which is fresh and undigested with little or no water added.

Temperature and pH

There wasn't much variation in temperature (29.8 – 29.9)^{o}C in all the mixing ratios. They were all within the mesophillic range (25 – 40)^{o}C which is ideal for bacterial activity, (Metcalf and Eddy, 2003; Miller 1996). The pH range recorded for the different mixing ratios were within optimal ranges (6.5 – 9) required for biological degradation of organic matter by microorganisms (Veenstra and Polprasert, 1997).

Nutrients

Levels of NH_3-N and NO_3-N did not differ much from the results of previous researches conducted in Accra by Larmie (1997) and literature quoted by Strauss *et al.* (1997). The high values might result from the ammonification and mineralization of organic nitrogen, (Epstein, 2003) which is one of the major constituents in faecal sludge. All the parameters discussed above decreased as the septage ratio in the mixture increased. Ammonia concentrations ranging between 920 – 1310 mg/l as found in the raw sludges could hamper algal and bacterial growth (Montangero and Strauss, 2002). They also produce malodor as well as causing eye irritations at treatment plants as additional problems (McGinn *et al*., 2003)**.**

Interestingly, total phohorus (TP) values increased with increasing septage ratio. High phosphorus in septage may be attributed to the use of detergents for cleansing toilet and water closets and also the practice of using water used for washing clothing to flush toilets in many homes during periods of water shortage (Awuah *et al.*, 2002; Personal communication with residents of Asafo, surburb of Kumasi; personal observation). Sludge with high contaminant loads, pose treatment challenges but has high potential of generating large quantities of biosolid that may be useful in agriculture.

4.2.2.2 Dewatering time (DT) of the different filter media in different cycles

The dewatering time for different loading rates were significantly different ($p < 0.023$). The SLR1 spent more number of days to dewater in almost all the cycles except cycle 4. These dewatering differences observed were mainly attributed to the different PTS and septage ratio. PTS from public latrines is generally known to be fresh (about one week to few months old), biochemically unstabilised and undigested, and does not lend itself to dewater easily, (Montangero and Strauss, 2002; Cofie *et al.*, 2006). Septage from septic tanks is generally known to be fully or partially digested (about one to ten years old), and easy to dewater (Heinss *et al.*, 1998). Moreover, PTS which is fresh and undigested has high specific resistance due to high colloidal particles thus resists withdrawal of water (Karr and Keinath, 1978). Therefore increase in PTS content decreases dewaterability while increase in septage content increases the dewaterability. Septage generally has lower TS strength as compared to PTS and therefore it's increase decreases the TS strength of the FS thus favouring dewatering of sludge with the highest septage content. SLR3 therefore had the shortest dewatering time while SLR1 had the longest dewatering time. This not withstanding, the septage

may sometimes have higher solid concentration than PTS and thus the solid content of the faecal sludge will increase as the septage increases. However, these higher sptage solids may not directly increase dewatering time. The dewatering time for the different solid loading rates varied independently in each cycle. This is because the contaminant loads of the FS especially (TS) of each cycle differed. Between the period of January and May 2006, the average temperatures of the day were very high (31.5 – 33.3) and daily average relative humidity were comparatively low (58.5 – 61.5) (Ghana Meteorological Service Report, 2006). Therefore, the possible cause for the independent dewatering times in the different cycles could be the solid content of the FS and the level of digestion attained. The filter media for the dewatering remained the same for all the three different SLR and therefore very little could be attributed to that.

4.2.2.3 Removal efficiencies of filter beds with respect to different Solid Loading Rates

Removal of solids (TS, TVS and SS)

Dewatering of SLR1, SLR2 and SLR3 showed TS removal of 83.4%, 80.4% and 78.1% in 7, 5 and 4 days respectively (Table 4.2.2). TVS and SS removal also followed the same decreasing trend. The levels of removal efficiencies achieved compare well with the results from pilot drying beds obtained by the Ghana Water Research Institute in Accra (Montangero and Strauss, 2002). Though SLR1 achieved the highest removal efficiency, it took a longer time whereas the SLR3 achieved the least removal efficiency but in a shorter time based on their respective dewatering times. This is because as dewatering time increases, particles have enough time to settle and therefore removal of solids improves. In the SLR1 the PTS content was

comparatively higher than the others and therefore formed a thicker sludge cake on the surface of the filter medium. The sludge cake thus complemented the filter medium in the removal of the sludge particles. The 96.1% removal of SS by SLR1 which was the highest might be explained based on its highest solids content resulting from high percentage of PTS in the mixture. This could lead to formation of thicker sludge cake on the surface of the filter medium (Karr and Keinath, 1978), due to the high content of colloidal particles in PTS. This thus increases the straining, compaction, and impaction abilities of the sludge cake and the filter medium (Metcalf and Eddy, 2003; Kiely, 1998). The highest positive correlation between suspended solids and dewatering time coupled with the lowest rate of increase in dewatering when suspended solids is increased showed by SLR3 is highly recommended for it to be used for faecal sludge dewatering (Figure 4.6). This might be due to the high content of septage which contains more agglomerated sludge particles, (Kuffour *et al*. 2009; Doku, 2003).

Reduction of Electrical Conductivity (EC)

The low percentage reduction of electrical conductivity, 28.18%, 28.35% and 25.98% by SLR1, SLR2 and SLR3 respectively (Table 4.2.7), indicate that the solid loading rate did not impact much on the EC reduction. This is because ionic mobility of dissolved salts increases EC. Thus increasing dissolved salts, led to higher electrical conductivity, (Kiely, 1998). Therefore poor reduction of electrical conductivity is an indication of poor reduction of salts. High magnitude of EC between 14.99 and 16.56 mS/cm in the percolates, compared to EPA permissible level of 700 μS/cm will put osmotic stress on the growing plants and other organisms in the affected area when

discharged into the environment. The poor reduction of the salts means little was retained in the biosolids which indicates a safe use of the biosolids in agriculture.

Removal of Degradable Organics (COD, DCOD)

There were high percentage removals of COD in all the different solid loading rates. The efficiencies achieved by the different solid loading rates are comparable to the removal efficiency range of 70 – 90% in pilot drying bed research conducted in Accra by WRI, Accra (Heins *et al.,* 1998) and removal efficiencies of 73.4 – 86.9% COD achieved by bench scale drying beds dewatering at Dompoasi, Kumasi, conducted by Quarshie Sharon (Quarshie 2006, MSc. Thesis) and 85 – 90% COD removal achieved during drying beds dewatering at the Buobai co-composting treatment plant (Cofie, *et al*. 2006), near Kumasi. The removal efficiency might be due to the efficiency achieved in removing solids from the raw sludge (Cofie, *et al.*, 2006) and the availability of the dissolved organics for microbial action in the filter bed (Karim *et al.,* 2005). The possibility of most solids being removed from SLR1 might have contributed to its highest removal of COD. A good reduction of DCOD of the percolate is an indication of the availability of the dissolved substrate for bacterial action during filtration (Karim *et al.,* 2005). The removal efficiencies of 60.7%, 59.4% and 51.6% by SLR1, SLR2 and SLR3 respectively (Table 4.2.6), well explains that the particles of the dissolved organics were degraded by microbes. The higher DCOD removal efficiency of SLR1 over SLR2 and SLR3 (Table 4.2.6) might further support the role played by the high content of PTS and TS concentration in the dewatering of FS. However, the average time used to achieve these removal efficiencies was comparatively high for SLR1 than the others. Thus, the longer time used for the

dewatering might have assisted in improving the removal efficiency through microbial action, sludge cake formation and removal of solids.

Removal of Nitrogen (NH_3-N, TKN)

Removal efficiencies of 70.4%, 64.4% and 61.8% resulting in percolate concentrations of 0.30, 0.34 and 0.30 g/l of NH_3-N were recorded for SLR1, SLR2 and SLR3, respectively. The reductions achieved in this study were improvement over the pilot studies of drying beds operated in Accra which achieved 50% removal of NH_4 (Larmie, 1995). It also compares well with drying beds operated in Bangkok which achieved NH_4 + NH_3 removal between 70 – 90 % during the first ten months of operation (AIT, 1998). Although the removal efficiencies were high, the percolate concentrations were still high, probably as a result of high NH_3-N in the influent FS and ammonification of organic nitrogen (Epstein, 2003). The higher overall biological activity including nitrogen and organics removal efficiencies by the filter media from the different loading rates can be attributed to higher specific surface area of filter particles and higher biomas densities (Nikhla and Farooq, 2003). Thus the more reason SLR1 with higher biomass density outperformed the others in the removal efficiency for both NH_3-N and TKN. Removal efficiencies of NH_3-N might have also been made possible through volatilization and mineralization of organic nitrogen (Awuah, 2006). The possibility of highest TKN removal from SLR1 as compared to SLR2 and SLR3 was the result of its ability to remove most TVS (organic matter) together with organic nitrogen. Furthermore a lot of organic nitrogen is mineralised to nitrates (Epstein, 2003).

Removal of nutrients, (NO_3-N, NO_2-N, TP)

The removal efficiencies of NO_3-N for SLR1, SLR2 and SLR3 which were 48.3%, 44.0% and 40.5% respectively, were considered low. This might be due to continual production of nitrate in the percolate through mineralization of organic nitrogen. Formation of NO_3 in the percolate might account for any loss through removal of organic matter thus leading to high percolate concentration in the different solid loading rates. The nitrate and nitrite removal achieved might also be attributed to removal due to both the deposition effects of suspended solids on the filter beds and fixed film growth on the sand (Jonsson *et al*., 1997). The nitrite removal efficiencies which were higher than nitrate in the different solid loading rates might be due to conversion of nitrite to nitrate through nitrification in the percolate during the dewatering process. Percentage removals of phosphorus in all the mixing ratios, were generally low but comparatively higher with respect to the results of a study where medium sand of particle sizes between 0.25 - 0.5mm achieved only about 10% reduction in PO_4^{3-} thus accounting for low removal of TP (Epstein, 2003). However the ability of SLR1 to outperform the SLR2 and SLR3 in the removal of NO_3 and TP might be due to adsorption and striation of these nutrients to sludge particles of which it recorded highest percentage removal. Dewatering of FS of the three different loading rates indicated that SLR3 was the fastest to dewater. However, physicochemical parameters removals by the different loading rates were not consistent with any of the solid loading rates.

4.2.2.4 Organic matter (TVS) accumulation in the sand (Filter medium)

An increase in TVS in the top 10cm of the filter medium is an indication of increase in organic matter content because TVS is a function of organic matter. This means that

clogging of filter beds was possible in all the mixing ratios. However, organic matter build up in the filter medium for SLR1 gradually outpaced the others as the cycles advanced to cycle six. This was due to the higher content of dissolved organics in SLR1 based on the PTS content in the mixture. Research has shown that, PTS (undigested sludge) contains high quantities of supracolloidal solids which are able to blind the sludge cake as well as the filter medium as they migrate through them increasing resistance to filtrate flow, (Karr and Keinath, 1978) and resulting in clogging. The organic matter accumulation is likely to fill the pore spaces between the particles of the filter bed which can subsequently result in clogging. Thus filter medium for SLR1 is likely to clog earlier compared to others as the cycles advance.

4.2.2.5 Accumulation of Biosolids

Considering the average TS of Faecal Sludge of the different solid loading rates, with respect to dewatering time, average annual solid loading rates of 425 kg TS/m^2/yr, 502kg TS/m^2/yr and 509kgTS/m^2/yr were possible for SLR1, SLR2 and SLR3 respectively (Table 4.2.15). This means that more sludge could be dewatered by applying SLR3. The percentage accumulation of the biosolids on the filter media from the different solid loading rates, SLR1, SLR2 and SLR3 indicates that SLR1 had the highest potential to generate biosolids per cycle. SLR1 gave the highest percentage TVS of biosolid accumulated by the filter media from the different solid loading rates. Furthermore, the percentage TVS accumulated by the filter media from each influent FS were in the order of 72, 66 and 65 for SLR1, SLR2 and SLR3 respectively. These performances showed by SLR1 proved that it had the highest potential of biosolid generation per cycle amongst the others. However, annual biosolid production with respect to dewatering time, was estimated and based on that the organic matter

accumulation per m^2 per year was determined. The potential to generate biosolids, for SLR3, SLR2 and SLR1 were 256, 228 and 224 (kgTVS/m^2 year) respectively (Table 4.2.15). This means that in dewatering where biosolid production is paramount, SLR3 is best for application.

4.3 Effect of Constant Solid Loading Rate of Faecal Sludge on Dewatering, by Filter Beds

4.3.1 Results of effect of constant solid loading rate of FS on dewatering by filter beds

This section shows the results of how the constant solid loading rate of the faecal sludge affect the different dewatering cycles. In the previous two objectives, each cycle run had different solid concentration of faecal sludge. This is because the PTS and the septage collected from the on-site toilet have varying solid concentrations (Table 4.3.1). Since each cycle had different solid concentration, it was difficult to understand the dewatering pattern of the faecal sludge, so it was decided to maintain a constant solid loading rate for dewatering in all the cycles which led to the development of this objective. The PTS and septage ratios used were, 1:1 and 1:3 by volume. The 1:1 ratio (SLR1) was dewatered on filter media FM1, FM2 and FM3 (SLR1 on FM1, SLR1 on FM2, and SLR1 on FM3) while the 1:3 was dewatered on FM1 (SLR3 on FM1) only. But for conveniency, these four treatments that were considered for dewatering were indicated simply as FM1, FM2, FM3 and SLR3 in the tables and figures and in the text. The procedure for obtaining the constant TS of the faecal sludge to be used dewatering in each cycle is well explained in pages 83 – 85.

4.3.1.1 Total Solids (TS) of different faecal sludge (FS) samples used.

The FS used for the dewatering in this phase consisted of public toilet sludge (PTS) and septage mixed in of 1:1 and 1:3 by volume (explained pages 83 – 85). These were collected over a period of 7 months from January 2007 to July 2007. The PTS being fresh, biochemically unstabilised and undigested from the public toilets had higher TS than the septage in all the cycles. The septage which was usually collected from the septic tanks was partially digested and more stabilized (Doku, 2003). It had comparatively higher water content and therefore showed comparatively low TS, due to its pre-collection storage (from 1 to several years) (Cofie *et al.*, 2006). The 1:1 and 1:3 ratios were volume combinations of the PTS and septage and therefore showed intermediate TS values between the two in all the cycles (Table 4.3.1). The constant TS used for the 1:1 ratio was 34.64 g/l which was the average TS of faecal sludge used for the six cycles in objective one (Table 4.1.4) while the constant TS for the 1:3 ratio was 26.93 g/l which was the average TS of the selected solid loading rates that were used in objective two (Table 4.2.2).

Table 4.3.1: TS (g/l) of different faecal sludge used for dewatering in each cycle (N = 8)

Cycles	PTS (g/l)	Septage (g/l)	SLR1:1(g/l)	SLR1:3 (g/l)
C1	58.33	12.04	37.14	27.56
C2	44.47	14.48	27.11	19.88
C3	89.99	2.47	49.42	33.49
C4	49.85	1.71	25.90	19.69
C5	41.25	2.43	24.00	16.30
C6	77.08	72.60	74.15	73.00
C7	18.25	11.79	13.72	10.52
C8	44.19	6.67	25.09	16.48
Mean±SD	**52.92±22.30**	**15.52±23.59**	**34.57±19.13**	**27.11±19.86**
LSD (cycles)	**13.8**	**14.8**	**11.9**	**12.4**

N = number of cycles

4.3.1.2 Dewatering Time

The FS of SLR1 which was applied to FM1, FM2 and FM3 with constant TS (36.64 g/l) in all the cycles dewatered in longer number of days in all the filter media in cycle one but the dewatering time decreased in all the filter media in the subsequent cycles to cycle 5. The DT started rising gradually in all the filter media to cycle 8. The SLR3 dewatering on FM1 also followed the same dewatering trend but at correspondingly lower DT in all the cycles (Figure 4.9). FM1 had the longest average dewatering time of 8.9 days followed by FM2 and FM3 with 8.8 and 8.7 days respectively while SLR3 had the shortest dewatering time of 4.4 days.

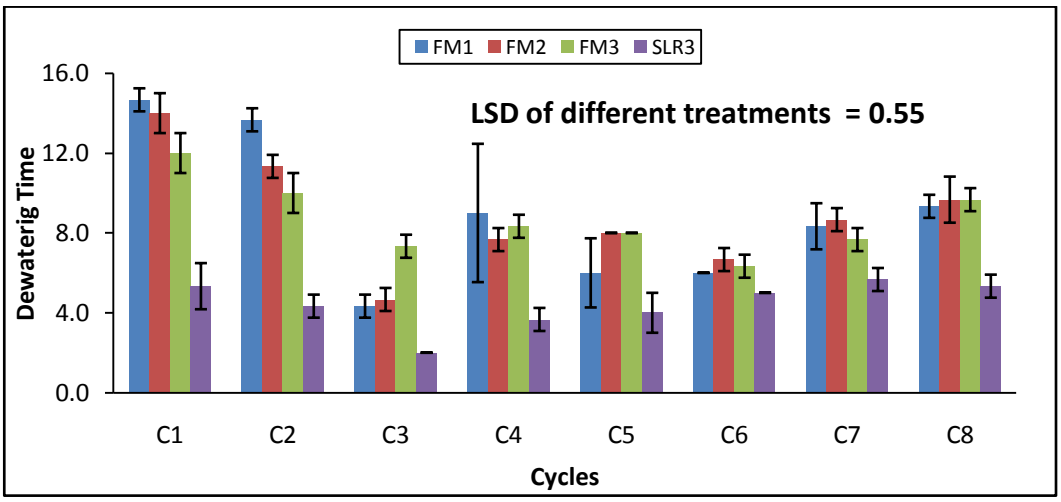


Figure 4.9: Dewatering time of different treatments of constant SLR of FS in 8 cycles
(Those without error bars have the sameor almost the same dewatering days for the replicates)

There was significant difference between SLR3 dewatered on FM1 and all the other treatments at 5% significance level (p = 0.0001) but there was no difference between FM1, FM2 and FM3 which dewatered SLR1.

4.3.1.3 Physicochemical Parameters of constant SLR of FS and percolate

Total Solids

The TS in the percolate from the different treatments (SLR1on FM1, SLR1on FM2 and SLR1on FM3) were very high with FM3 being the highest, whereas the percolate from SLR3 dewatered on FM1 had the least concentration in all the cycles (Table 4.3.2). The percentage TS removed by FM1, FM2, FM3 and SLR3 were 69.0, 65.5, 64.0 and 79.4 respectively (Table 4.3.2). Although these removals were high, the percolates concentrations remained high. The raw sludges had high corresponding concentrations (Table 4.3.2). There was significant difference in the percentage removals of TS in all the treatments at 5% significance level (p = 0.0001).

Table 4.3.2: TS of FS and percolate (g/l) with percentage removals

Faecal sludge (g/l)			Percolate (g/l)				Percentage removal of treatments			
Cycles	**SLR1**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**
C1	37.14	27.56	11.03	12.59	12.74	4.73	70.29	66.10	65.70	82.84
C2	27.11	19.88	10.01	10.50	11.03	5.19	63.09	61.27	59.30	73.91
C3	49.42	33.49	12.90	14.08	15.31	6.30	73.89	71.51	69.01	81.18
C4	25.90	19.69	12.03	12.69	12.91	5.96	53.53	50.98	50.16	69.74
C5	24.00	16.30	8.50	8.81	9.45	4.60	64.59	63.29	60.62	71.81
C6	74.15	73.00	11.16	13.94	13.76	7.17	84.95	81.20	81.44	90.18
C7	13.72	10.52	7.82	9.22	10.43	4.18	43.00	32.76	23.96	60.21
C8	25.09	16.48	12.19	13.58	13.83	6.47	51.40	45.90	44.88	60.73
Mean±SD	**34.57±19.13**	**27.11±19.86**	**10.71±1.81**	**11.93±2.12**	**12.43±1.97**	**5.57±1.05**	**69.03±13.52**	**65.50±15.34**	**64.03±17.41**	**79.44±10.56**
LSD of percolate = 0.49										

Total Volatile Solids (TVS)

The TVS concentrations were very high in the percolates of all the treatments and in all the cycles. FM1, FM2, FM3 and SLR3 showed percolate TVS concentrations of 4.87, 5.33, 6.75 and 3.21 (g/l) respectively (Table 4.3.3). Although the corresponding percentage removals of 75.64, 73.24, 67.13 and 79.15 were high, the percolate concentration remained high due to the high concentration of the raw sludges

dewatered. The percolate from the SLR3 had the highest removal efficiency of TVS while FM3 had the least. ANOVA at 5% significance level showed that, p = 0.0001 with significant differences between FM1, FM2, FM3 and SLR3 in TVS removal.

Table 4.3.3: TVS of FS and percolate (g/l) with percentage removals

Faecal sludge (g/l)			Percolate (g/l)				Percentage removal of treatments			
Cycles	**SLR1**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**
C1	29.11	18.19	4.35	4.88	7.02	3.74	85.06	83.24	75.89	79.46
C2	20.12	15.35	3.90	4.18	4.36	2.24	80.60	79.24	78.32	85.42
C3	27.74	22.92	7.35	7.83	13.36	4.62	73.49	71.76	51.86	79.83
C4	18.64	16.05	5.32	5.88	7.29	2.91	71.46	68.46	60.92	81.87
C5	15.75	9.01	3.57	4.20	4.73	1.57	77.34	73.31	69.95	82.56
C6	48.74	41.85	5.67	5.94	6.73	4.35	88.37	87.80	86.20	89.61
C7	8.01	6.60	2.96	3.12	3.64	1.75	63.10	61.07	54.61	73.44
C8	16.97	11.54	5.82	6.62	6.90	4.49	65.68	61.01	59.32	61.04
Mean±SD	**23.13±12.32**	**17.69±11.06**	**4.87±1.44**	**5.33±1.53**	**6.75±3.01**	**3.21±1.26**	**75.64±8.92**	**73.24±9.79**	**67.13±12.32**	**79.15±8.69**
LSD of percolate = 0.39										

Suspended Solids (SS)

The suspended solids in the percolate were removed to levels as low as 0.77, 1.19, 1.85 and 0.54 g/l for FM1, FM2, FM3 and SLR3 respectively, compared to the high levels in the raw sludge, (Table 4.3.4). The SS in the percolate of SLR3 was the least among the treatments, followed by FM1, FM2 and FM3. The FM1, FM2, FM3 and SLR3 had 96.5, 94.5, 91.5 and 96.5 percentage removals respectively, with FM1 and SLR3 being the highest (Table 4.3.4). ANOVA showed significant differences among the percentage removals by the treatments at 5% significance level (p = 0.0001).

Table 4.3.4: Suspended Solids of FS and percolate(g/l) with percentage removals

Faecal sludge (g/l)			Percolate (g/l)				Percentage removal of treatments			
Cycles	**SLR1**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**
C1	20.00	17.50	0.38	0.48	0.85	0.18	98.10	97.60	95.73	98.97
C2	29.00	15.00	1.55	1.99	2.92	1.22	94.64	93.13	89.93	91.87
C3	37.50	18.10	1.34	1.84	3.12	0.75	96.43	95.09	91.68	95.87
C4	15.20	9.60	0.66	1.70	3.13	0.54	95.66	88.82	79.39	94.38
C5	11.40	6.30	0.75	0.97	1.27	0.47	93.45	91.46	88.89	92.49
C6	39.00	42.00	0.67	0.97	1.51	0.69	98.27	97.52	96.14	98.37
C7	9.50	7.50	0.31	0.80	1.04	0.15	96.70	91.58	89.05	97.96
C8	12.20	8.20	0.45	0.73	0.95	0.36	96.28	93.99	92.19	95.61
Mean±SD	**21.72±11.91**	**15.53±11.65**	**0.77±0.45**	**1.19±0.57**	**1.85±1.02**	**0.54±0.35**	**96.48±1.63**	**94.54±3.06**	**91.49±5.23**	**96.49±2.66**
LSD of percolate = 0.15										

Chemical Oxygen Demand (COD)

In all the eight cycles, SLR3 had the least COD (2.92 g/l) in the percolate which is most preferred for discharge, though not up to the EPA standard of 250 mg/l (0.25 g/l) when compared to that of FM1, FM2 and FM3 which had COD values of 4.96, 6.40 and 8.27 (g/l) respectively (Table 4.3.5). FM1, FM2, FM3 and SLR3 had 86.7, 82.9, 77.9 and 87.9 percentage removals of COD respectively (Table 4.3.5), with SLR3 having the highest efficiency in COD removal. ANOVA showed significant differences in COD removal at 5% significance level (p = 0.0001) between the treatments.

Table 4.3.5: COD (g/l) of faecal sludge and percolate with percentage removals

Faecal sludge (g/l)			Percolate (g/l)				Percentage removal of treatments			
Cycles	**SLR1**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**
C1	58.15	44.02	2.96	3.34	4.70	2.09	94.92	94.26	91.92	95.26
C2	34.98	21.29	3.42	4.16	6.24	1.44	90.22	88.12	82.17	93.21
C3	50.18	33.45	10.62	11.59	13.28	6.84	78.84	76.91	73.53	79.57
C4	47.43	22.13	5.16	10.49	15.34	2.92	89.11	77.89	67.67	86.79
C5	29.48	14.34	5.15	5.90	6.77	3.54	82.52	80.00	77.04	75.33
C6	70.31	65.63	4.16	5.30	6.45	2.62	94.08	92.47	90.82	96.00
C7	18.82	18.04	3.55	5.00	6.91	2.58	81.15	73.44	63.30	85.69
C8	27.46	17.61	4.67	5.45	6.48	1.57	82.99	80.17	76.41	91.07
Mean±SD	**42.10±17.37**	**29.56±17.55**	**4.96±2.43**	**6.40±2.99**	**8.27±3.83**	**2.92±1.72**	**86.73±6.14**	**82.91±7.69**	**77.86±10.16**	**87.86±7.48**
LSD of percolate = 0.32										

Conductivity, EC (mS/cm)

In all the cycles SLR3 had the least amount of salts in the percolate. Average electrical conductivity for FM1, FM2, FM3 and SLR3 were 17.33, 18.12, 18.38 and 12.35 (mS/cm) respectively, while the percentage removals were 23.58, 20.09, 19.06 and 20.68 for FM1, FM2, FM3 and SLR3 respectively with FM1 being the highest (Table 4.3.6). All showed low removal of EC. ANOVA showed significant differences in salt removal among the treatments at 5% significance level (p = 0.0001).

Table 4.3.6: Electrical Conductivity (mS/cm) of FS and percolate with percentage removals

Faecal sludge (mS/cm)			Percolate (mS/cm)				Percentage removal of treatments			
Cycles	**SLR1**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**	**FM1**	**FM2**	**FM3**	**SLR3**
C1	23.90	15.32	15.05	15.83	17.49	10.96	37.02	33.77	26.82	28.48
C2	17.55	11.15	13.16	13.97	14.36	8.78	25.01	20.42	18.18	21.23
C3	24.30	13.82	20.73	21.53	21.64	10.23	14.69	11.40	10.96	26.00
C4	23.30	20.75	18.08	18.62	19.44	15.94	22.40	20.09	16.55	23.20
C5	20.70	12.07	15.42	16.00	16.49	10.46	25.51	22.69	20.35	13.31
C6	24.60	23.00	20.55	21.17	22.43	19.22	16.48	13.94	8.83	16.45
C7	22.00	13.05	17.46	18.34	14.94	11.10	20.62	16.65	32.11	14.97
C8	24.90	15.53	18.19	19.47	20.25	12.14	26.95	21.81	18.67	21.81
Mean±SD	**22.66**±2.50	**15.59**±4.20	**17.33**±2.66	**18.12**±2.67	**18.38**±3.02	**12.35**±3.47	**23.58**±6.94	**20.09**±6.78	**19.06**±7.65	**20.68**±5.37
LSD of percolate = 0.84										

4.3.1.4 Organic matter (TVS) accumulation in sand

The organic matter accumulation in top 10 cm of the sand of different treatments increased gradually from C1 to C8 in the filter media. The use of raw TS of sand could be deceptive due to sand effect from the filter media thus the use of TVS. The percentage organic matter accumulated in filter media of the different treatments as the cycles advanced from C1 to C8 are presented below (Figure 4.10). FM3 had the highest accumulation of organic matter in the filter medium followed by FM2 and FM1 while SLR3 showed the least organic matter accumulation (Figure 4.10).

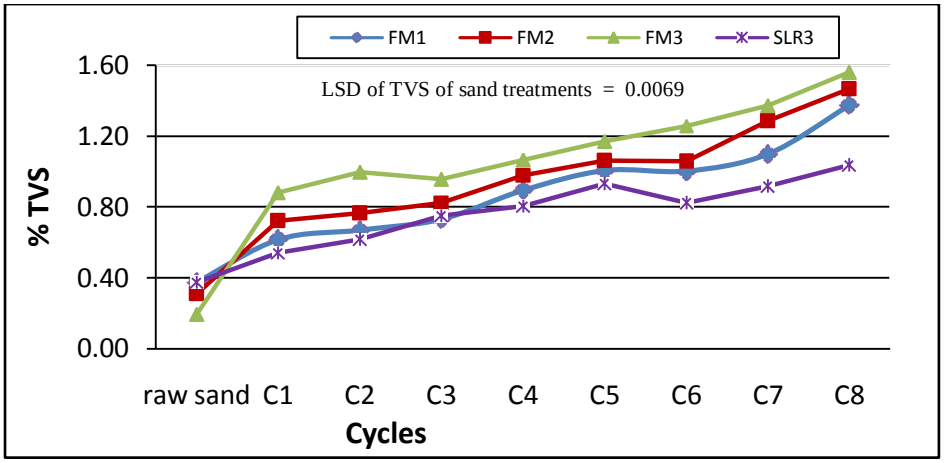


Figure 4.10: Percentage organic matter accumulation in filter media of different treatments of constant SLR of FS

4.3.1.5 Biosolids generated from dewatering

Average dry weight of biosolid obtained for FM1, FM2 FM3 and SLR3 were 166.15, 189.31, 169.97 and 140.85 (g) respectively. The dry weight of biosolid could be deceptive due to effect of sand from the filter media that usually mix with the biosolid during its collection after dewatering. Thus the corresponding organic (TVS) weights were also determined, which were 113.08, 117.16, 100.67 and 97.09 (g/l) for FM1, FM2, FM3 and SLR3 respectively (Figure 4.11). FM2 had the highest total dry weight as well as the organic weight while the SLR3 had the least values for both. However, the SLR3 had the highest percntage organic matter of the biosolids (Figure 4.11).

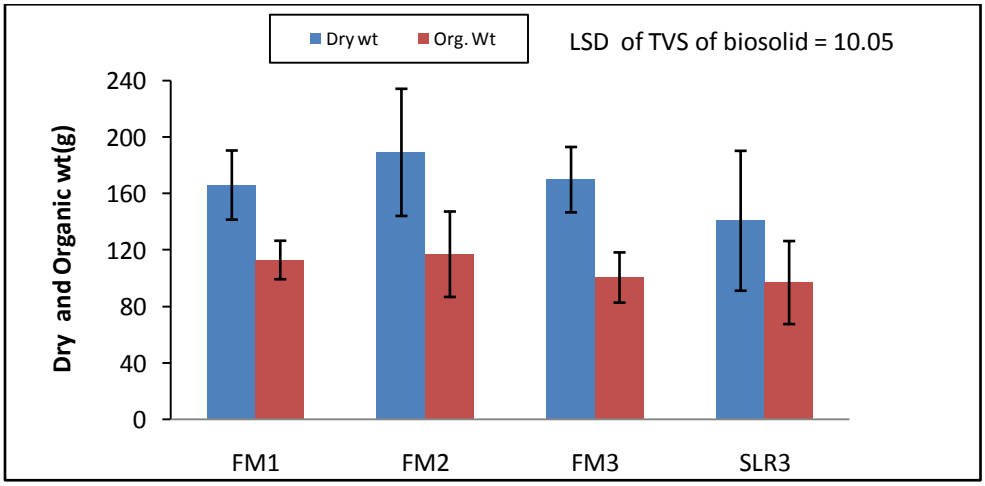


Figure 4.11: Biosolids accumulated from different treatments of constant SLR of FS

4.3.1.6 Estimate of organic matter production

The estimated organic matter production per annum based on the dewatering time of each treatment indicated that FM1, FM2, FM3 and SLR3 could generate 4.63 kg, 4.84 kg, 4.24 kg and 8.02 kg respectively, corresponding to annual per metre square production of 193, 202, 177 and 334 kg TVS/m^2/yr respectively (Table 4.3.7).

Table 4.3.7: Estimated annual organic matter production (N = 8)

Estimated parameter	FM1	FM2	FM3	SLR3
Org. Wt/cycle	113.08	117.16	100.67	97.09
DT (Days)	8.92	8.83	8.67	4.42
Est cycles/year	41	41	42	83
Org wt/year (g)	4629.05	4840.98	4239.92	8023.32
Kg Org wt/year	4.629	4.841	4.240	8.023
Area of filter bed (m^2)	0.024	0.024	0.024	0.024
Org wt (kg TVS/m^2.year)	192.877	201.708	176.663	334.305

N = Number of cycles
Org. wt = weight of organic matter: Est. = Estimated: DT= dewatering time

4.3.2 Discussion of effect of constant solid loading rate of FS on dewatering by filter beds

The characteristics of the Faecal Sludge, compared with the percolate produced from dewatering constant solid loading rate of the mixing ratios of 1:1 and 1:3 and the biosolids produced have been dicussed in this section.

4.3.2.1 Faecal Sludge (FS)

The FS used consisted of PTS and septage mixed in 1:1 and 1:3 ratios. The septage usually has very low concentration and more stabilized as compared to the PTS. The differences that result from mixing the PTS and septage are well explained in page 133. The characteristics of solids in faecal sludge used are well explained in page 134.

4.3.2.2 Dewatering time (DT) of the different filter media in different cycles

The dewatering time generally showed by the different treatments where TS of the FS were maintained constant showed an improvement in the dewatering time of FM1, FM2 and FM3 over previous studies conducted using the same particle sizes of sand (filter media) whereby the TS of the FS used was varying in the dewatering cycles (Kuffour *et al.*, 2009). The SLR3 which also dewatered in 4 days in specific objective two (SO2) where solid loading rate in the cycles were varying, increased the dewatering time to 4.4 days this time. The increase in the number of cycles from six in the previous two objectives to eight in this study, might have contributed to the increase in the dewatering time. However, the fact that FS of constant SLR1 dewatering on FM1, FM2 and FM3 could lower the dewatering times from 9.8, 9.9 and 9.1 in specific objective one (SO1) to 8.9, 8.8 and 8.7 days respectively, indicated that increasing the number of cycles was not a strong influencing factor. Therefore, other factors which might include the variations of TS in the different cycles and prevailing weather conditions (Heins *et al.*, 1998) could be counted on. Although, it is understood that the different weather conditions under which these experiments were conducted might have influenced the dewatering time, the variations in the TS of the loading rates in different cycles might have strongly contributed to the changes in the dewatering times. The fact that SLR3 of constant TS dewatering on filter bed FM1 significantly outperformed SLR1of constant TS which dewatered on FM1, FM2 and FM3 in all the cycles makes SLR3 an obvious choice in terms of commercial FS dewatering. The best constant TS range for SLR3 is 26.93 ± 20.25 g, while that of SLR1 is 36.64 ± 7.73 g.

SLR3 had significantly lowest dewatering time compared to the others. The higher volume of septage in the SLR3 might be partly responsible for this result. The pattern

of dewatering followed by all the treatments where the DT decreased from C1 to C5 support the studies conducted by Rudolfs and Heukelekian (1934) which reported that in sludge dewatering, fine sludge particles are created and drainability deteriorates during the early stages of digestion, but as digestion proceeds, the fine sludge particles are destroyed and dewaterability again improves. The increasing dewatering time from C5 to C8 might be attributed to accumulation of organic matter in the pore spaces of the sand particles and growth of microorganism resulting in onset of clogging.

4.3.2.3 Removal efficiencies of the physicochemical parameters

Removal of solids (TS, TVS and SS)

The removal efficiency of TS by SLR3 in absolute values was far higher than the FM1, FM2 and FM3 which dewatered FS of SLR1. This might be attributed to the TS differences of the two types of FS as a result of the ratio of PTS to septage. The SLR3 which had highest volume of septage with comparatively lesser amount of TS reduced the TS concentration and content of colloidal particles in the FS and therefore made it easier to be dewatered more than the SLR1 dewatering on FM1, FM2 and FM3 which had higher PTS volume. Higher septage content also contained higher digested, stabilised and agglomerated sludge particles which could be removed easier (Doku, 2003). The effect of PTS content (containing more fine sludge particles) and septage (containing lesser fine sludge particles) on the dewatering might be explained based on the research conducted earlier on particle sizes , which indicated that the original differences in sludge dewaterability were chiefly because of differences in particle size distribution (Karr and Keinath, 1978). FM1, FM2 and FM3 dewatering FS of SLR1 resulted in the FM1 outperforming the FM2 and FM3 because it had the

advantage of reduced pore volume as result of large surface area of its small sand particle size (Ball, 1997). Research on porous media has shown that, the variability in particle sizes, surface texture and charge of the porous media particles greatly influence the adhesion process (DeFlaun and Mayer, 1983; Stozky, 1986). Since FM1 had smaller particle sizes, the removal pattern confirmed the assertion by earlier researchers that, smaller particle sizes expose larger surface area compared to course particles, hence providing more adhesion sites (Stentrom and Hofner, 1982; Fontes *et al.*, 1991; Tan *et al.*, 1992).

The ability of SLR3 to outperform FM1, FM2 and FM3 with respect to TVS and SS removal would be attributed to the fact that it had higher septage volume which was more digested with majority of fine particles agglomerated for easier removal through striation, compaction and adhesion of the sludge particles (Kiely, 1998).

Removal of Degradable Organics (COD)

The SLR3 had the least COD in the percolate compared to FM1, FM2 and FM3 which is comparatively better than the others in terms of discharge into the environment. The ability of SLR3 to show the highest removal efficiency might be due to the fact that, most of its suspended solids and organics were removed due to large proportion of septage in it. The least percentage removal showed by FM3 which filtered SLR1 might be attributed to the fact that the filter allowed a lot of sludge particles to pass through the pore spaces into the percolate which contained high degradable organics. This resulted in the high COD in the percolate. Generally the average percentage removals of COD achieved by the different treatments ranging between 77.9% and 87.9% are comparable to the results obtained by Ghana Water Research Inatitute with

the use of pilot drying beds for the dewatering of faecal sludge in Accra which achieved COD removal efficiency of between 70 – 90% (Montangero and Strauss, 2002). The COD of 8.27g/l recorded in the percolate of FM1 was too high for discharge into the environment as compared to the 2.92 g/l recorded by SLR3 (Table 4.3.5). Irrespective of the high percentage COD removals achieved by the treatments, the levels in the percolate were very high, and this might be attributed to the high levels in the FS (Strauss *et al*., 1997). In terms of COD removal, SLR3 was the best, followed by FM1, FM2 and FM3 which were all significantly different from the others.

Reduction of Electrical Conductivity (EC)

Percentage removal of electrical conductivity (EC) was low in all the treatments resulting in high levels in the percolate. Ionic mobility of dissolved salts increases EC. Thus increasing dissolved salts, led to higher electrical conductivity (Kiely, 1998). Therefore poor reduction of EC is an indication of poor removal of salts. This shows that conductivity, which is a function of salinity, is not removed through the removal of solids. Increase in EC increases salinity whose high content in the FS, biosolid and the percolate renders them useless for application in agriculture (FAO/UNEP/WHO, 2006). This is because, it changes the osmotic pressure at the root zone due to high salt content. It provokes specific ion toxicity of sodium, boron or chloride. It may interfere with plant uptake of essential nutrients like potassium and nitrate. Due to antagonism with sodium, chloride and sulfates, it may destroy the soil structure by causing soil dispersion and clogging of pore spaces (FAO/UNEP/WHO, 2006). Poor removal of the salts by the dewatering process means less salts were retained in the

biosolids. This will however make the biosolid generated, suitable for soil amelioration but the percolate is not good for irrigation.

4.3.2.4 Organic matter (TVS) accumulation in the sand (Filter medium)

TVS analysed in the filter media of all the treatments indicated gradual and consistent increase from C1 to C8. An increase in TVS in the filter medium is an indication of increase in organic matter content because TVS is a function of organic matter. Meanwhile organic matter build up in the filter medium induce clogging (Largergraber *et al*., 2003; Kiely, 1998). This means that, clogging is possible in all the treatments since all of them showed increase in the organic matter build up. The highest organic matter build up in the sand of FM3 might be due to the fact that it has larger pore spaces in the filter media thus allowing rapid accumulation of sludge particles. This reasoning is supported by the fact FM2 which was second to FM3 in terms of pore spaces had the next higher TVS accumulation followed by the FM1. Ball (1997) illustrated that course particle sizes have smaller surface area but larger pore spaces which allow high and rapid accumulation of sludge particles. The SLR3 which dewatered on FM1 had higher septage volume which had most of its fines agglomerated to larger sludge particles which were removed at the surface of the filter medium (FM1) without or with little sludge accumulation within the pore spaces, thus the TVS accumulation rate in the filter medium was very low. Thus SLR3 had the least tendency to clog.

4.3.2.5 Accumulation of biosolids

Considering the constant TS of the FS applied to the different treatments with respect to dewatering time, average annual solid loading rate of FS dewatered by each

treatment was determined as 312, 315, 322, and 464 kg TS/m^2/yr for FM1, FM2, FM3 and SLR3 respectively. This means that more sludge could be dewatered by applying SLR3. Though SLR3 dewatered the least quantity of FS per cycle, its annual TS of FS dewatered was the highest due to the short dewatering time. Dry weight of FS produced by the different treatments showed that FM2 had the highest whereas SLR3 had the least. Organic matter accumulation also showed FM2 to be highest followed by FM1, FM3 and SLR3. These notwithstanding, the percentage organic matter relative to the TS showed that SLR3 had the highest percentage of organic matter accumulation. This means that the SLR3 dewatering on FM1 combined well to remove most organic matter. The annual organic matter production favoured the SLR3 which therefore makes it most suitable for application when biosolid production is paramount.

4.4 The Quantity of Physical Conditioner (Sawdust) That Will Improve the Faecal Sludge Dewatering Efficiency of Unplanted Filter Beds

4.4.1 Results of quantity of physical conditioner (sawdust) that will improve faecal sludge dewatering efficiency unplanted filter beds.

The different percentages (50%, 100% and 150%) total solids (TS) of sawdust that were mixed with the faecal sludge showed effect on the dewatering time and the physicochemical quality of the percolates produced. The results of the characteristics of the faecal sludge, FS – SD mixtures, percolates and biosolids have been presented in this section.

4.4.1.1 Characteristics of faecal sludge (FS) used for the dewatering

The FS used for the dewatering cycles of this objective consisted of PTS and septage collected from suction trucks discharging at Dompoase treatment plant in Kumasi, the capital of Ashanti region of Ghana over a period of six months from July 2007 to January 2008. The PTS was usually fresh and undigested. It had very high concentration of all the parameters while the septage from septic tanks had comparatively lower concentrations of the parameters analysed (Table 4.4.1). The SLR3 which was mixture of the PTS and the septage in the ratio of 1:3 was intermediate between the two and therefore showed concentrations corresponding the ratio (Table 4.4.1).

Table 4.4.1: Characteristics of faecal sludge (FS) used for mixing with different quantities of sawdust

	FS samples (g/l)		
Cycles	**PTS**	**Septage**	**SLR 1:3**
TS	57.672	10.090	29.446
TVS	41.08	7.16	24.194
SS	46.42	7.33	20.833
COD	57.98	11.02	31.371
TKN	7.18	1.35	3.76

4.4.1.2 Dewatering time (DT)

The dewatering time (DT) of all the treatments gradually increased from C1 to C3 but decreased in C4 and increased in C5, then finally decreased slightly in C6 (Figure 4.12). In all the cycles, the 150% sawdust treatment had the shortest dewatering time whereas the rest showed slight variations in the dewatering time in the different cycles. At the end of the sixth cycle, the average DT of 50%, 100%, 150% and 0% sawdust treatment were 5.3, 4.9, 3.9 and 5.6 days respectively. Although the DT varied among the treatments in the cycles, the average DT showed that the 150% sawdust treatment which had the highest quantity of sawdust had the shortest DT whereas the 0% sawdust (control) had the longest DT. ANOVA showed

significant differences in the dewatering time of the different treatments at 5% significance level (p = 0.0001) with 0%, 100% and 150% treatments being different.

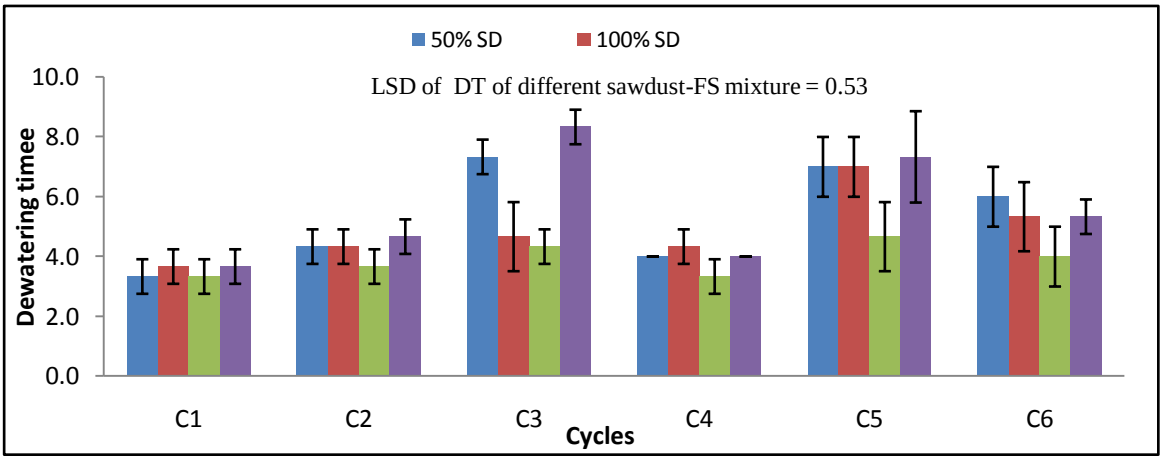


Figure 4.12: Dewatering time (DT) of different sawdust-FS mixture in six cycles
(Those without error bars have the sameor almost the same dewatering days for the replicates)

4.4.1.3 Physicochemical parameters of percolates and raw faecal sludge (FS)

Total Solids (TS)

The concentration of TS in the percolates of all the treatments, were very high. The treatment with 150% of sawdust addition had the highest TS (5.90 g/l) while the control which had no sawdust addition had the least (5.30 g/l). Considering the TS concentrations in the percolate of the different treatments and the FS, the TS removal within the cycles did not follow any particular trend (Table 4.4.2). Although the concentrations in the percolates were high, the percentage removals observed in the different treatments of sawdust-FS mixture of, 50%, 100%, 150% and 0% (control), which were 80.94, 80.30, 79.31 and 81.45 were high (Table.4.4.2). Absolute values of percentage removals indicated slight differences. ANOVA indicated significant differences among the different treatments at 5% significance level (p = 0.0016).

Table 4.4.2: Total Solids (TS) (g/l) of FS and percolate with percentage removal

	FS (g/l)	Percolate TS (g/l) of % sawdust mixture				Percentage removal			
Cycles	**SLR 3**	**50%**	**100%**	**150%**	**0%**	**50%**	**100%**	**150%**	**0%**
C1	34.09	4.36	3.88	4.23	4.75	87.21	88.62	87.59	86.06
C2	34.93	5.73	6.07	5.84	4.96	83.59	82.62	83.29	85.81
C3	26.19	4.92	5.36	6.60	4.50	81.22	79.55	74.79	82.81
C4	32.61	6.03	6.59	6.43	5.88	81.53	79.80	80.28	81.98
C5	21.25	5.07	5.37	5.12	5.04	76.15	74.74	75.91	76.27
C6	27.60	6.63	6.50	7.18	6.68	75.97	76.46	73.99	75.80
Mean±SD	**29.45±5.34**	**5.46±0.83**	**5.63±1.01**	**5.90±1.08**	**5.30±0.82**	**80.94±4.35**	**80.30±4.92**	**79.31±5.40**	**81.45±4.50**
		LSD of percolate = 0.30							

Total Volatile Solids (TVS)

The TVS in the percolates of all the treatments were high as compared to the EPA guideline. The concentrations were 2.32, 2.50, 2.26, and 2.59 (g/l) for the 50%, 100%, 150% and control respectively (Table 4.4.3). The 150% saw dust treatment had the least TVS in the percolate while the control had the highest. The concentrations in the percolate of the different treatments compared with the TVS in the faecal sludge showed that, 50%, 100%, 150% and control removed 89.9%, 89.2%, 90.2%, and 88.8% TVS respectively. The 150% sawdust treatment gave the highest TVS removal while control gave the least (Table 4.4.3).

Table 4.4.3: TVS (g/l) of FS and percolate with percentage removal

	FS (g/l)	Percolate TVS (g/l) of % SD-FS mixture				Percentage removal			
Cycles	**SLR 3**	**50%**	**100%**	**150%**	**0%**	**50%**	**100%**	**150%**	**0%**
C1	30.12	2.46	2.43	2.16	2.57	91.83	91.92	92.83	91.48
C2	30.36	2.06	2.91	2.37	2.23	93.23	90.42	92.20	92.66
C3	19.76	2.34	2.39	2.38	2.48	88.16	87.89	87.97	87.44
C4	26.94	2.20	2.41	2.29	2.72	91.83	91.05	91.49	89.91
C5	15.53	1.97	2.13	1.93	2.00	87.34	86.25	87.57	87.10
C6	22.45	2.92	2.70	2.44	3.56	86.97	87.97	89.12	84.16
Mean±SD	**24.19±5.98**	**2.32±0.34**	**2.50±0.27**	**2.26±0.19**	**2.59±0.54**	**89.89±2.71**	**89.25±2.20**	**90.20±2.26**	**88.79±3.15**
		LSD of percolate = 0.30							

The percentage removals showed by all the treatments were close and had little variations. ANOVA showed no significant differences in the percentage removals by the different treatments at 5% significance level (p = 0.112).

Suspended Solids (SS)

The suspended solids (SS) in the percolates of 50%, 100%, 150% sawdust, and control treatments were 0.46, 0.45, 0.33 and 0.48 (g/l) respectively (Table 4.4.4). The 150% sawdust treatment had the least SS in the percolate of all the cycles. The control had the highest average SS in the percolates but not in all the cycles. The 150% sawdust treatment had the highest potential to remove SS. The other treatments showed equally high potential for SS removal. The 50%, 100%, 150% sawdust treatment and control showed 97.74%, 97.89%, 98.45% and 97.68% suspended solids removal efficiency respectively (Table 4.4.4). ANOVA showed significant differences among the different treatments at 5% significance level (p = 0.0001).

Table 4.4.4: Suspended Solids (g/l) of FS and percolates with percentage removal

	FS (g/l)	Percolate SS (g/l) of % SD-FS mixture				Percentage removal			
Cycles	**SLR 3**	**50%**	**100%**	**150%**	**0%**	**50%**	**100%**	**150%**	**0%**
C1	10.20	0.28	0.17	0.14	0.26	97.22	98.30	98.66	97.45
C2	13.80	0.16	0.16	0.11	0.22	98.86	98.82	99.18	98.41
C3	25.60	0.41	0.43	0.29	0.44	98.39	98.31	98.87	98.27
C4	20.50	0.45	0.49	0.24	0.36	97.80	97.63	98.84	98.25
C5	34.50	0.62	0.65	0.58	0.71	98.19	98.13	98.32	97.95
C6	20.40	0.82	0.78	0.65	0.87	96.00	96.16	96.81	95.75
Mean±SD	**20.83±8.63**	**0.46±0.24**	**0.45±0.25**	**0.33±0.23**	**0.48±0.26**	**97.74±1.02**	**97.89±0.93**	**98.45±0.85**	**97.68±1.00**
		LSD of percolate = 0.05							

Chemical Oxygen Demand (COD)

The percolate COD of the 50%, 100%, 150% and 0% (control) sawdust treatment were 1.74, 1.60, 1.53, and 1.71 (g/l) respectively (Table 4.4.5). The 150% sawdust

treatment had the least COD whereas the 50% sawdust treatment had the highest COD in the percolate. Although the COD in the percolates were high, there were high percentage removals by all the treatments. The 50%, 100%, 150%, and 0% treatments showed average COD percentage removals of 94.28%, 94.69%, 95.00% and 94.37% respectively (Table 4.4.5). The 150% sawdust treatment had the highest potential to remove COD while the 50% sawdust treatment had the least potential to remove COD during the dewatering process. There was significant difference between the removal efficiencies of the different treatments at 5% significance level (p = 0.045).

Table 4.4.5: Chemical Oxygen Demand (COD) (g/l) of FS and percolates with percentage removal

	FS (g/l)	Percolate COD (g/l) of % SD-FS mixture				Percentage removal			
Cycles	**SLR 3**	**50%**	**100%**	**150%**	**0%**	**50%**	**100%**	**150%**	**0%**
C1	33.09	2.63	1.74	2.14	2.39	92.05	94.75	93.54	92.76
C2	27.59	3.22	3.37	3.14	3.23	88.33	87.78	88.61	88.30
C3	31.82	0.89	0.95	0.63	1.02	97.21	97.01	98.00	96.81
C4	45.19	1.51	1.43	1.51	1.46	96.67	96.83	96.67	96.78
C5	25.00	0.80	0.77	0.57	0.67	96.81	96.91	97.73	97.32
C6	25.55	1.38	1.31	1.17	1.47	94.60	94.87	95.41	94.24
Mean±SD	**31.37±7.52**	**1.74±0.98**	**1.60±0.94**	**1.53±0.98**	**1.71±0.94**	**94.28±3.49**	**94.69±3.54**	**95.00±3.53**	**94.37±3.46**
		LSD of percolate = 0.16							

Nitrogen (NH_3-N and TKN)

The NH_3-N in the percolate of all the treatments was very high. The 150% sawdust showed the least concentration of NH_3-N in all the cycles except the C1 where the 100% sawdust treatment showed the least. The 0% sawdust treatment showed the highest concentration of NH_3-N in the percolate in all the cycles. The average NH_3-N in the percolate of 50%, 100%, 150% and 0% sawdust treatments were 0.61, 0.59, 0.55, and 0.71(g/l) respectively (Table 4.4.6). The 50%, 100%, 150% and 0% sawdust treatments removed 77.0%, 77.5%, 79.2% and 73.3% of NH_3-N (Table 4.4.6), which shows an increasing NH_3-N removal as the percentage sawdust increased. ANOVA

showed significant differences in NH_3-N removal by the different treatments at 5% significance level (p = 0.0001).

Table 4.4.6: Ammonia Nitrogen (NH_3-N) (g/l) in FS and percolates with percentage removal

	FS (g/l)	Percolate NH_3-N (g/l) of % SD-FS mixture				Percentage removal			
Cycles	**SLR 3**	**50%**	**100%**	**150%**	**0%**	**50%**	**100%**	**150%**	**0%**
C1	3.07	0.32	0.29	0.30	0.30	89.61	90.62	90.33	90.08
C2	2.91	0.83	0.77	0.72	0.95	71.69	73.62	75.21	67.51
C3	2.81	0.84	0.78	0.69	0.96	70.07	72.08	75.38	65.74
C4	2.84	0.36	0.34	0.30	0.40	87.38	88.19	89.57	85.93
C5	1.57	0.54	0.59	0.52	0.60	65.45	62.51	66.61	61.65
C6	3.39	0.76	0.76	0.75	1.06	77.71	77.70	77.88	68.64
Mean±SD	**2.77±0.62**	**0.61±0.23**	**0.59±0.22**	**0.55±0.21**	**0.71±0.32**	**76.99±9.77**	**77.45±10.54**	**79.16±9.19**	**73.26±11.74**
		LSD of percolate = 0.05							

The TKN in the percolates of 50%, 100%, 150% and 0% sawdust treatments were 0.73, 0.69, 0.64, and 0.85 (g/l) respectively. The 150% had the least value of TKN in the percolate while the control had the highest value (Table 4.4.7). In terms of percentage removal of TKN, the 50%, 100%, 150%, and 0% sawdust treatments recorded 79.33%, 80.04%, 81.60% and 75.91% respectively (Figure 4.4.4), indicating an increase in percentage removal with increasing percentage of sawdust addition.

Table 4.4.7: Total Kjedhal Nitrogen (TKN) of FS and percolates with percentage removal

	FS (g/l)	Percolate TKN (g/l) of % SD-FS mixture				Percentage removal			
Cycles	**SLR 3**	**50%**	**100%**	**150%**	**0%**	**50%**	**100%**	**150%**	**0%**
C1	5.01	0.42	0.37	0.38	0.42	91.69	92.51	92.51	91.68
C2	3.65	0.99	0.92	0.85	1.13	72.97	74.86	76.68	68.94
C3	3.46	1.01	0.94	0.82	1.15	70.91	72.91	76.18	66.65
C4	4.10	0.47	0.41	0.38	0.52	88.62	90.12	90.83	87.25
C5	2.33	0.62	0.66	0.59	0.69	73.55	71.55	74.87	70.59
C6	4.00	0.87	0.87	0.86	1.19	78.23	78.30	78.52	70.33
Mean±SD	**3.76±0.88**	**0.73±0.26**	**0.69±0.25**	**0.64±0.23**	**0.85±0.35**	**79.33±8.78**	**80.04±9.06**	**81.60±7.91**	**75.91±10.69**
		LSD of percolate = 0.06							

The ANOVA results indicated that there was no significant difference in the percentage removals of the different sawdust treatments at 5% significance level (p = 0.746).

Conductivity (EC (mS/cm))

The values of EC measured in the percolates of the different percentage of sawdust treatments were very high. In most of the cycles the 150% sawdust treatment recorded the highest concentration. The average EC in the percolate of 50%, 100%, 150% and 0% sawdust treatments were 9.76, 10.17, 10.39 and 9.75 (mS/cm) respectively (Table 4.4.8). The 50%, 100%, 150% and 0% sawdust treatments removed 38.24, 35.63, 34.20 and 38.30 percent EC respectively (Table 4.4.8). The control showed the highest percentage removal while the 150% sawdust treatment showed the least percentage removal, indicating a decreasing percentage removal as the percentage of sawdust added to the FS increased. ANOVA indicated significant differences among the removal efficiencies of the treatments at 5% significance level (p = 0.0001).

Table 4.4.8: EC (mS/cm) of FS and percolate with percentage removal.

	FS (mS/cm)	Percolate EC (mS/cm) of % SD-FS mixture				Percentage removal			
Cycles	**SLR 3**	**50%**	**100%**	**150%**	**0%**	**50%**	**100%**	**150%**	**0%**
C1	15.35	8.59	10.03	9.40	9.47	44.06	34.68	38.74	38.28
C2	19.49	9.77	10.16	10.59	9.88	49.85	47.87	45.68	49.32
C3	14.06	8.77	9.16	9.59	8.88	37.60	34.85	31.82	36.87
C4	14.75	9.90	10.07	10.39	9.41	32.86	31.75	29.58	36.18
C5	15.42	8.52	8.63	9.20	8.05	44.77	44.01	40.34	47.82
C6	16.30	12.99	12.94	13.20	12.82	20.29	20.59	19.04	21.33
Mean±SD	**15.90±1.91**	**9.76±1.70**	**10.17±1.49**	**10.39±1.48**	**9.75±1.63**	**38.24±10.61**	**35.63±9.62**	**34.20±9.45**	**38.30±10.07**
		LSD of percolate = 0.37							

pH and Temperature

The 50%, 100%, 150% and 0% sawdust treatments had pH, of 8.09, 8.05, 8.07 and 8.12 respectively which were all higher than the Faecal Sludge, pH of 7.68 (Table 4.4.9). Although there was no significant difference between them at 5% significance level (p = 0.775), all were higher than that of the raw sludge. Temperatures of 29.5, 29.2, 29.4 and 29.3 were recorded in the percolates of 50%, 100%, 150% and 0% sawdust treatment respectively. They were all not significantly different (p = 0.209) but lower than the temperature of the Faecal Sludge (Table 4.4.9).

Table 4.4.9: pH and Temperature (^{o}C) of FS and percolate of different SD–FS mixtures

	FS	Percolate pH of % SD-FS mixture				FS (^{o}C)	Percolate Temp. (^{o}C) of % SD-FS mixture			
Cycles	**SLR3**	**50%**	**100%**	**150%**	**0%**	**SLR3**	**50%**	**100%**	**150%**	**0%**
C1	7.97	8.06	7.93	7.88	7.96	31.1	33.1	32.8	32.7	32.7
C2	7.70	8.12	8.08	7.83	8.13	36.6	31.2	31.3	31.6	30.6
C3	7.69	8.06	7.88	7.57	8.06	30.8	27.5	26.5	27.3	27.0
C4	7.45	8.02	7.73	8.18	8.04	25.5	30.3	30.7	30.3	30.3
C5	7.63	7.59	8.02	8.25	7.79	28.0	27.3	26.0	26.7	27.3
C6	7.63	8.67	8.64	8.70	8.71	28.0	27.8	28.1	27.8	27.7
Mean±SD	**7.68**±0.17	**8.09**±0.34	**8.05**±0.31	**8.07**±0.39	**8.12**±0.31	**30.0**±3.84	**29.5**±2.36	**29.2**±2.77	**29.4**±2.49	**29.3**±2.28
		LSD of pH = 0.14					LSD of Temperature = 0.32			

4.4.1.4 Organic matter (TVS) accumulation in sand

The organic matter accumulation in filter media increased gradually from C1 to C6 in all the treatments. The 150% sawdust treatment had the least organic matter accumulation while the control (0% sawdust treatment) had the highest organic matter accumulation in the filter media. In order to avoid deception by the sand effect, the percentage organic matter of each treatment at each cycle was determined (Figure 4.13). The control had the highest percentage of organic matter accumulation while

the 150% sawdust treatment had the least organic matter accumulation in the sand in almost all the cycles.

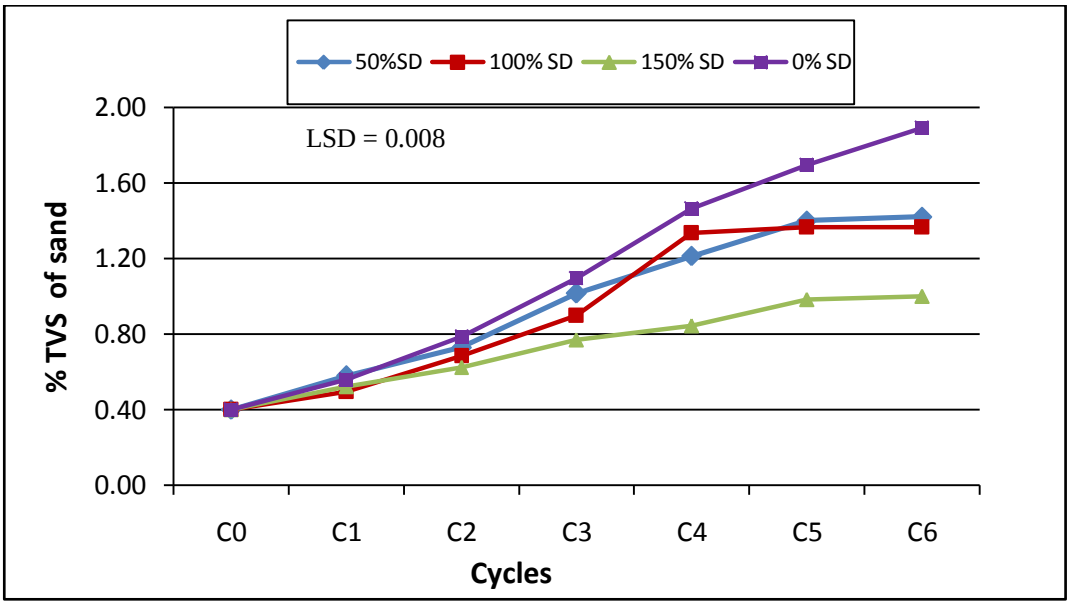


Figure 4.13: Organic matter accumulation in filter media by the different treatments

4.4.1.5 Biosolid generated from dewatering of the different treatments

Average dry weight obtained from dewatering FS mixed with 50%, 100%, 150% and 0% sawdust treatments in six cycles were 177.25, 235.12, 316.70 and 133.11 (g) respectively. Since the dry weight could be deceptive due to effect of sand from the filter media, the corresponding organic weight (TVS) of the dried solid were determined which respectively produced 125.84, 182.15, 252.35 and 82.18 (g) (Figure 4.14). Based on this the percentage organic matter (TVS) of TS of the dried biosolid for each treatment was determined . The percentage organic matter (TVS) of TS of the dried biosolid for 50%, 100%, 150% and 0% sawdust treatments were 71.10%, 77.29%, 80.57% and 66.34% respectively (Table 4.4.9).

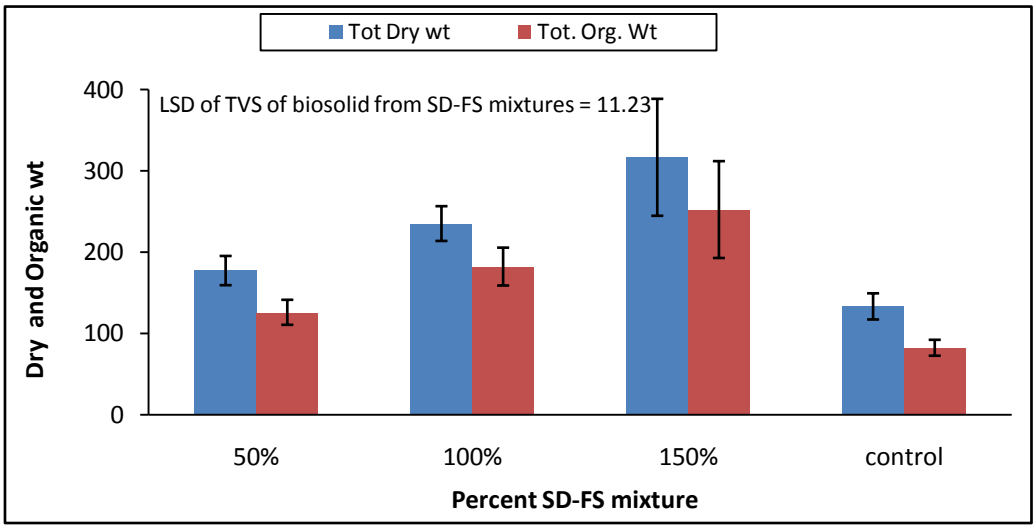


Figure 4.14: Biosolid generation from different percentage SD-FS mixtures

4.4.1.6 Annual biosolid estimation

The possible number of cycles per annum for 50%, 100%, 150% and 0% (control) SD-FS mixtures based on the DT were 68, 75, 94, and 66 respectively. This was obtained by dividing the 365 days in a year by the respective dewatering time (DT). The annual organic matter production from 50%, 100%, 150% and 0% (control) SD-FS mixtures with respect to the dewatering time were 359, 567, 987 and 225 (kg TVS/m^2year) organic matter and 505, 731, 1239 and 364 (kgTS/m^2year) dried biosolids respectively (Table 4.4.10).

Table 4.4.10: Estimated annual biosolid generation

Parameter	50%	100%	150%	Control (0%)
Tot Dry wt/cy	177.25	235.12	316.70	133.11
Tot. Org. Wt/cy	125.84	182.15	252.35	82.18
%TVS	70.10	77.29	80.57	66.34
DT	5.3	4.9	3.9	5.6
Est cycles/year	68	75	94	66
kg org wt/yr	8.61	13.60	23.69	5.40
kg dry (TS) wt/m^2/yr	505	731	1239	364
kg (TVS) wt/m^2/yr	359	567	987	225

4.4.2 Discussion of results of quantity of physical conditioner (sawdust) that will improve faecal sludge dewatering efficiency unplanted filter beds.

This section shows the discussion of the results of the dewatering of faecal sludge – sawdust mixtures.

4.4.2.1 Characteristics of raw faecal sludge (FS)

The FS used was made of a mixture of PTS (fresh and undigested) and septage (months to years old, partially or fully digested) in the ratio of 1:3 by volume, thus SLR3. All the parameters analyzed including Total solids, total volatile solids, suspended solids, chemical oxygen demand, Nitrogen etc. were characteristically high (about 10 – 100 times as wastewater) due to the freshness and low water content of PTS component (Strauss *et al*., 2004)

4.4.2.2 Dewatering time (DT) of the different SD-FS mixtures dewatered

The dewatering of FS in this study was effected through evaporation, percolation and absorption by the added quantity of sawdust. The sawdust used had the potential of absorbing and holding water. The 150% sawdust-FS mixture showed the shortest dewatering time while the control (0% sawdust) treatment had the longest DT. This might be attributed to the contribution of the water absorbing and holding capacity of the sawdust. According to Jing *et al*. (1999), sawdust (wood chips) and wheat dregs have been verified as suitable physical conditioners for both improving dewaterability and also reducing adhesion of sludge on filter walls. They are also inert materials and function as skeleton builders because they create a permeable and rigid lattice structure within the sludge which can remain porous under high pressure filtration to allow water to be forced out under pressure (Zall *et al.*, 1987; Benitez *et al.*, 1994). The decrease in DT from the control to the 150% sawdust treatment is consistent with

the research findings by Lin *et al.* (2001) which reported that increasing the dose of wood chips or wheat dregs decreased the DT and increased the sludge solid yield. Again increasing dose of sawdust decreased the sludge cake moisture. Thus the higher the dose the shorter the DT as found in this study where 0%, 50%, 100% and 150% of sawdust mixed with FS dewatered in 5.6, 5.3, 4.9, and 3.9 days respectively. The decrease in the DT might be attributed to the permeation of free water from the sludge to the wood chips which would lead to less free water in sludge to be released (Lin *et al.,* 2001). This DT is a significant improvement over previous studies where filter beds used in dewatering FS did so between 7 – 14 days (Pescod, 1971; Larmie, 1994; Heins *et al.,* 1997). There has also been improvement over previous dewatering studies of this research whereby different filter beds dewatered FS between 9 and 10 days (Kuffour *et al.,*2009) and different solid loading rates of FS dewatered in between 4 to 7 days (page 120).

4.4.2.3 Removal efficiencies of filter beds with respect to different percentages of sawdust mixture

Removal of solids (TS, TVS and SS)

The higher concentration of TS in the percolate coupled with least percentage removal of TS by the 150% sawdust treatment might be due to dissolution of solids, either salts or organics from the sawdust into the percolate thus increasing the percolate TS and further reducing the percentage removal of TS (Arends, 1985). This will depend on the type of sawdust and the quantity applied. The average TS of 5.30 g/l in the percolate of the control and 5.90g/l in the percolate of the 150% sawdust were too high for discharge into the environment. However the average percentage removal of

TS between 79.35% for 150% sawdust and 81.45% for control compares well with dewatering performance of Buobai pilot drying beds in Kumasi (Cofie, 2003).

Contrary to the TS removal, the TVS and the SS removal were highest in the 150% sawdust treatment and lowest in the 0% sawdust treatment (control) (Tables 4.4.3 and 4.4.4). The ability of the 150% sawdust treatment to show highest removal of TVS might be that the solids added to the percolate by the sawdust were not organic in nature. The percentage removal of SS between 98.45% and 97.68% were very high and compared well with the removal performances achieved by the Buobai pilot drying beds dewatering, conducted in Kumasi (Cofie, 2003). It also compares favourably with percentage removal achieved by Water Research Institute in Accra on drying bed trials where about 95% SS removal was achieved. Suspended solids removal is achieved through straining, compaction, adhesion, etc. (Metcalf and Eddy, 2003). Bouwer (1984), reported that straining happened when the diameter of the suspended particles was larger than 0.2 times the diameter of the particles constituting the porous media. The addition of sawdust provided increased surface area and additional sites for the adsorption of the sludge particles. Physical conditioning agents, such as sawdust, when added to faecal sludge, increases adsorption of sludge particles due to increased surface area provided by the particle of the sawdust. It also breaks down adhesive properties of fine particles of the sludge by distributing them evenly to enhance porosity, absorbing the moisture and providing the biosolids mass with an appropriate degree of sponginess and aeration (Tremier *et al.*, 2005; Iranzo *et al.*, 2004; Sanchez Monedero *et al.*, 2001). Therefore the filter beds loaded with the 150% sawdust-FS mixture provided most of the bulking agent (sawdust) hence created

highest porosity and absorption rate thereby reducing the quantity of percolate and increasing suspended solids removal.

Removal of degradable organics (COD)

The percentage removal of COD by all the treatments between 94 and 95% (Table 4.4.5) showed significant improvement over previous research works in Ghana. Pilot drying beds for dewatering FS at Buobai in Kumasi achieved 85 - 90% COD removal (Cofie, 2003; Cofie *et al.,* 2006). Pilot drying beds in Accra treating FS could remove between 70 and 90% COD (Heins *et al*., 1998). In other studies, settling tanks in Accra could achieve less than 60% COD removal efficiency (Heins and Larmie, 1998), whereas constructed wetland operating in Bangkok, Thaiwan treating FS, mostly septage achieved a maximum COD removal of 90% (Koottatep *et al.,* 1999a, Koottatep *et al.,* 1999b). The overarching performance of COD removal in this study might be attributed to the high aggregated sludge particles of septage that were easily removed (Doku, 2003). Further performance by the sawdust-FS mixture could be due to the provision of extra surface area by the sawdust particles which served as adsorption sites for the sludge particles. Moreover the ability of the sawdust as physical conditioner to breakdown the colloidal properties of the fine sludge particles and distributing them evenly to enhance porosity and aeration contributed to the removal of more solids and hence COD removal (Tremier *et al.,* 2005; Iranzo *et al*., 2004). That should be the reason, why 150% sawdust treatment had the highest COD removal efficiency. In spite of the outstanding performance, the COD concentrations in the percolates of all the treatments were very high (beyond EPA standard of 250 mg/l; EPA, 1997) for discharge into the environment. This was certainly due to the high COD concentration in the FS. The percolate thus needed further treatment before

discharge into the environment since high COD in effluents can cause eutrophication (Metalf and Eddy, 2003).

Removal of Nitrogen (Ammonia Nitrogen (NH_3-N) and Total Kjedhal Nitrogen (TKN)

The ability of the treatments in this study to achieve NH_3-N removal of between 73.3% and 79.2% (Table 4.4.6) and TKN removal of between 75.9% and 81.6% (Table 4.4.7) could be said to be an improved performance over the removal efficiencies achieved by drying beds in Accra which removed between 40-60% NH_3-N (Heins *etal.,* 1998). It is an improvement over dewatering by constructed wetlands in Bangkok which achieved less than 80% NH_3-N removal. It is also an improvement over earlier studies of this project where different particle sizes of filter beds removed between 60 and 75% NH_3-N (Kuffour *et al.,* 2009). The high percentage nitrogen removal might be due to the ability to remove high percentage organic matter because, according to Epstein, (2003), nitrogen component in sludge is purely organic matter, thus removal of organic matter enhances nitrogen removal. The extra removal by the sawdust-FS mixtures may be due to the ability of the lignin component in the sawdust which contains carboxyl group that have an ability to part with hydrogen and retain absorbed ions of ammonia (Wilde, 1960). Moreover, wood waste for that matter sawdust has a distinct advantage over most other agricultural wastes, because it continues to bind nitrogen for a longer time during decomposition due to its lignin content (Golueke 1978; Bollen and Glennie, 1959). This was why 150% sawdust treatment had the highest nitrogen removal. This was due to the highest percentage of sawdust added. Nitrogen removal can also be partly due to loss through volatilization as result of increased pH and temperature (Epstein, 2003). It has been reported that

ammonia lost through volatilization could reach as high as 81% in the in the first 14 days (Robinson and Polglase, 2000). However the concentrations in the percolate needed further treatment before discharge into the environment, since ammonia concentrations greater 400 mg/l is harmful in the soil (Epstein, 2003).

Reduction of Electrical Conductivity (EC)

Removal of electrical conductivity (EC) was poor in all the treatments and even more so in the sawdust treated samples since 150% sawdust showed the least percentage removal of EC. This thus supports the earlier assertion that sawdust might have contributed some salts to the percolate leading to least removal of TS in the 150% sawdust treated sample. The percolate concentrations of 9.75 and 10.39 (mS/cm) in the control and the 150% sawdust treatments respectively were too high, thus limiting the possibility of the percolate being used for irrigation. It has been reported that conductivity should be less than 3 dS/m in percolate for it to be useful for irrigation (Cofie, 2003; FAO, 1985). However, the poor reduction of conductivity in the percolate is an indication that the biosolids contain less salts and thus auger well for it to be applied for soil enrichment.

pH and Temperature

The relatively low pH in the faecal sludge of 7.68 might probably be due to the activities of acetogenic and acid forming bacteria in the sludge (Veenstra and Polprasert, 1997). Products from acetogenesis reaction may accumulate and lead to a pH decrease. Methanogenesis is the rate-limiting step in anaerobic metabolism. Optimum pH for methanogenesis amounts to 6.8 - 7.8. Based on various anaerobic digestion studies, McGarry and Pescod (1970) found that pH 6.0 probably constitutes

the absolute, lowest limit for anaerobic ponds in the tropics when treating high-strength wastes. A decrease of pH by microbial treatment of wastewater sludge has been reported by several authors including Alam *et al*. (2001); Mannan *et al*. (2005) etc. There was increase in pH from slightly neutral (7.68) in the faecal sludge to alkaline in the percolates of all the SD-FS mixtures, ranging between (8.05 – 8.12). This increase might be due to the removal of solids by the filter beds from the faecal sludge, thereby reducing organic nitrogen which further reduced the ammonium nitrogen which is the source of H^+ in the faecal sludge. The reduction of the H^+ ion in the percolate causes the increase in pH (Peavy *et al*., 2003). There was no significant difference in pH of all the percolates implying that the addition of sawdust did not have significant effect on the pH of the percolate.

The slightly lower temperatures of all the percolates ranging between (29.2 – 29.5) compared to the faecal sludge sludge of 30.0 ^{o}C might be due to reduction in microbial activities as a result of removal of microbes together with the organic matter by the filter bed. The general decrease in average temperature of the percolates in relation to the faecal sludge might be due to the fact that as BOD decreases, microbial activities also decrease resulting in fall of temperature. For every organism, there is a minimum temperature below which no growth occurs, an optimum temperature at which growth is most rapid and a maximum temperature above which growth is not possible (Madigan and Martinko, 2006). These are known as the three cardinal temperatures. There were slight variations in the temperature of the percolates but were not significantly different and therefore do not allow for drawing any valid conclusion. Nonetheless, works carried out by Snowdon *et al*. (1989) suggested that

the fluctuating trend of temperature within the percolate can be explained partly by microbial activities.

4.4.2.4 Organic matter (TVS) accumulation in the sand (Filter medium)

The TVS which is a function of organic matter increased gradually in the filter media of the different treatments from C1 to C6, though at different rates. This is an indication that clogging is possible in all the treatments. This is because according to Siegrist and Boyle (1987), the process of clogging may be caused by the accumulation of stable solid materials between or on the surface of the porous media. However, the accumulation of the organic matter was highest in the control meaning that it had the highest potential to clog its filter medium as compared to the 150% sawdust treatment which had the least accumulation of organic matter, meaning it would be the least to clog its filter medium. This was made possible by the additional surface area for adsorption provided by the sawdust treatments, thus the higher the percentage of sawdust, the lower the organic matter accumulation in the filter media. Clogging can cause hydraulic dysfunction, anoxic conditions and diminished wastewater purification (Gannon *et al*., 1991).

4.4.2.5 Accumulation of biosolids

The total dry weight and organic matter weight accumulation were high in all the treatments but were extremely higher in the sawdust treated FS, considering 150% sawdust treatment generating 1239 kg TS/m^2/yr for dry weight and 987 kgTVS/m^2/yr for organic weight as against control generating 364 kgTS/m^2/yr dry weight and 225 kgTVS/m^2/yr for organic weight annually. This variation is quite large in terms of economic analyses of biosolid production. The sawdust treatment improved the

biosolid generation. This was made clear in the percentage organic weight of the dried biosolid where 150% treatment had about 81% while the control had 66%. The higher the percentage sawdust added the higher percentage organic weight generated. This was a clear indication that the sawdust added reduced loses of biosolid into the filter media and wall effect as asserted by Lin *et al.* (2001) and Jing *et al*. (1999). Zall *et al*. (1987) and Banitez *et al.* (1994) in their studies on wood chips, concluded that poorly filterable sludges yielding highly compressible filter cakes can be conditioned with woodchips (sawdust) to produce cakes with more rigid and incompressible structure which therefore maintains high porosity and subsequently elevates filterability. It also increases the porosity of the dewatered sludge for effective aeration in sludge composting.

4.5 The Agricultural Potential of Biosolids Produced From Unplanted Filter Beds through Faecal Sludge Dewatering

4.5.1 Results of agricultural potential of biosolids produced from the dewatering FS using unplanted filter beds

The agricultural potential of biosolids generated from the filter beds through dewatering of faecal sludges, with respect to nutrients and heavy metal concentrations showed some variations after going through the different treatment processes outlined in the objectives one to four. This was evident when the dried faecal sludges and the dried biosolid from the different dewatering processes were analysed.

4.5.1.1 The quality of dried FS and dried biosolid dewatered from filter media.

The Faecal Sludge (FS) was applied for dewatering on the filter media, filter medium one (FM1), filter medium two (FM2), and filter medium three (FM3), from which the

dried biosolid was obtained. The percentage carbon (%C) in the dried biosolids from the different filter media were 36.0%, 34.0% and 33.0% for FM1, FM2, and FM3 respectively, which were lower than %C obtained in the dried FS. The percentage nitrogen (%N) in the dried biosolids obtained from FM1, FM2, and FM3 were, 3.01%, 2.45% and 2.29% respectively. The percentage phosphorus (%P) and percentage potassium (%K) in the dried biosolid from FM1, FM2 and FM3 were, 2.05%, 1.98% and 1.73%, and 1.04%, 0.93% and 0.81% respectively. All the values were lower than their corresponding values in the dried FS, (Table 4.5.1). The C:N ratio of the dried biosolids from FM1, FM2 and FM3 as 12.0, 13.9 and 14.4 were however greater than the corresponding value in the FS (Table 4.5.1). The percentage values obtained for all the nutrients decreased from FM1 down to FM3. The concentration of all the metals analysed (Cu, Fe, Pb, Cd, Zn and Mn) (mg/kg) in the dried biosolids decreased from the FM1 to FM3 (Table 4.5.1). The dried biosolids concentrations were all lower than their corresponding values in the dried FS.

Table 4.5.1: Nutrients and Heavy metals in dried biosolid from different filter media (N = 6)

	Nutrients as % TS					Heavy metals (HM) as mg/kg					
Filter media (FM)	**C:N**	**%C**	**% N**	**%P**	**%K**	**Cu**	**Fe**	**Pb**	**Cd**	**Zn**	**Mn**
FM1	12.0	36.0	3.01	2.05	1.04	0.190	4.38	0.019	0.073	0.325	0.148
FM2	13.9	34.0	2.45	1.98	0.93	0.188	3.35	0.017	0.061	0.301	0.132
FM3	14.4	33.0	2.29	1.73	0.81	0.081	2.04	0.009	0.089	0.257	0.105
FS	**10.2**	**38.10**	**3.72**	**3.06**	**2.93**	**0.116**	**4.43**	**0.022**	**0.083**	**0.496**	**0.150**
LSD	*1.91*	*2.26*	*0.65*	*0.59*	*1.00*	*0.05*	*1.12*	*0.01*	*0.01*	*0.10*	*0.02*
Limit of HM in biosolids (USA). From Salvato, 1982 (mg/kg)						95-700		200-500	10-400	1000-1800	
Limit of HM in Developing countries, Hornwerg et al, (2000). (mg/kg)						80.000		150	3.0	300	

C, N, P and K = % TS; Cu, Fe, Pb, Cd, Zn and Mn = mg/kg

4.5.1.2 The quality of dried biosolids dewatered from different solid loading rates

Public toilet sludge (PTS) and septage were mixed in the ratio of 1:1, 1:2 and 1:3 by volume representing solid loading rate one (SLR1), solid loading rate two (SLR2) and

solid loading rate three (SLR3) which were the Faecal Sludge (FS) applied for dewatering to obtain the corresponding dried biosolids (Table 4.5.2). The percentage carbon (%C) decreased from 30% in SLR1 to 28% in SLR2 but increased to 31% in SLR3, whereas in the dried FS it increased from 37.7% in SLR1 to 39.8% in SLR3 (Table 4.5.2). The percentage nitrogen (%N) in the dried biosolid decreased from 3.45% in the SLR1 to 2.23% in SLR3 and it correspondingly decreased from 3.65% to 2.98% in SLR1 to SLR3 of the dried FS. The ratio of the %C and %N resulted in an increasing C:N ratio from 8.7 in SLR1 to 13.9 in SLR3, in the dried biosolids. In the dried FS, C:N ratio increased from 10.3 for SLR1 to 10.8 for SLR2 and 13.4 for SLR3. The percentage phosphurus (%P) of 3.08% in SLR1 decreased to 2.18% in SLR3 of the dried biosolid. The percentage potassium (%K) in the dried biosolid decreased from 1.05% in SLR1 to 0.84% in SLR3. The %P and %K again decreased from 3.22 and 3.78 in SLR1 to 3.14 and 3.09 in SLR3 respectively in the dried FS (Table 4.5.2). The heavy metals analysed showed increasing values in Fe, Zn and Pb from SL1 to SLR3 in the dried biosolid and dried FS, whereas Cu, Cd and Mn decreased from SLR1 to SLR3 in the dried biosolids and the dried FS. The concentrations of the heavy metals of the dried FS samples were higher than their corresponding dried biosolid concentrations (Table 4.5.2).

Table 4.5.2: Nutrients and Heavy metals (HM) in dried dewatered biosolid from different solid loading rates (SLR) of Faecal Sludge (FS)

	Nutrients as % TS					HM as mg/kg					
Solid Loading Rate	**C:N**	**%C**	**% N**	**%P**	**%K**	**Cu**	**Fe**	**Pb**	**Cd**	**Zn**	**Mn**
SLR1 (dried biosolid)	8.7	30.0	3.45	2.18	1.05	0.225	2.127	0.014	0.059	0.282	0.026
SLR2 (dried biosolid)	9.5	28.0	2.96	2.43	0.96	0.176	2.952	0.017	0.038	0.364	0.112
SLR3 (dried biosolid)	13.9	31.0	2.23	3.08	0.84	0.142	3.986	0.018	0.039	0.451	0.208
FS (SLR1)	**10.3**	**37.7**	**3.65**	**3.22**	**3.78**	**0.145**	**4.423**	**0.022**	**0.092**	**0.532**	**0.201**
FS (SLR2)	**10.8**	**35.5**	**3.28**	**3.17**	**3.18**	**0.188**	**4.562**	**0.026**	**0.085**	**0.592**	**0.254**
FS (SLR3)	**13.4**	**39.8**	**2.98**	**3.14**	**3.09**	**0.128**	**4.506**	**0.031**	**0.076**	**0.652**	**0.230**
LSD	*1.72*	*3.82*	*0.41*	*0.37*	*1.09*	*0.03*	*0.82*	*0.01*	*0.02*	*0.11*	*0.07*

4.5.1.3 Quality of dried FS and dried biosolid dewatered from FS of constant TS

The nutrients and heavy metals analysed in dried dewatered biosolids from FS (PTS and septage) of constant solid loading rate ratios of 1:1 and 1:3 (SLR1 and SLR3) respectively (Table 4.5.3). The SLR1 was dewatered on filter media (FM1, FM2 and FM3) while the SLR3 was dewatered on FM1 only.

The percentage carbon (%C) in the dried dewatered biosolid obtained from SLR1 decreased from 33% in FM1 to 31% in FM3 whereas dried dewatered biosolid of SLR3 had a higher percentage carbon of 36.0%. The %C obtained from all the dried biosolids were lower than their corresponding values in the dried FS which were 38.4% and 42.7% for SLR1 and SLR3 respectively (Table 4.5.3). The percentage (%N) in the dried biosolid of SLR1 decreased from 3.53% in FM1 to 2.83% in FM3. The SLR3 recorded a further low %N of 2.72%. The %N in the dried FS of SLR1 was higher than that of SLR3, and they were both higher than their corresponding values in the dried dewatered biosolids. The decreasing trend of %N in the dried dewatered biosolid inversely reflected in the C:N ratio by increasing from FM1 to SLR3. The dried FS had C:N being higher in SLR1 than SLR3 while both were higher than their corresponding dried dewatered biosolids. The %P in the dried dewatered biosolids decreased from 2.35% for FM1 to 2.19 for FM3 and further decreased to 2.08 for SLR3. The %K in the dried biosolids decreased from 0.8% in FM1 to 0.66% for FM3 and further decreased to 0.64% in SLR3. The corresponding values in the dried FS were higher than the dried biosolids, however dried FS of SLR1 had higher values than the dried FS of SLR3. The concentrations of Cu and Fe were all higher in the FS of SLR1 than that of SLR3. They all had higher values than their corresponding dried dewatered biosolid samples, however the concentration decreased from FM1 to FM3.

Lead (Pb) was almost negligible but had higher values in the dried FS than their corresponding dried biosolid samples (Table 4.5.3). Dried FS of SLR1 had higher concentration of Cadmium (Cd) than the SLR3 but the concentrations of zinc (Zn) and Manganese (Mn) were both higher in SLR3 than SLR1. The concentrations of all were higher in the dried FS samples than the dried biosolids. The concentration decreased in the dried biosolids from FM1 to FM3.

Table 4.5.3: Nutrients and Heavy (HM) metals in dried dewatered biosolid from constant TS treatments (N=8)

	Nutrients as % TS					HM as mg/kg					
CONSTANT TS	**C:N**	**%C**	**% N**	**%P**	**%K**	**Cu**	**Fe**	**Pb**	**Cd**	**Zn**	**Mn**
FM1(SLR1) (db)	9.9	35.0	3.53	2.31	0.8	0.181	2.036	0.023	0.089	0.457	0.105
FM2 (SLR1) (db)	10.3	32.0	3.11	2.85	0.75	0.139	1.757	0.023	0.044	0.331	0.127
FM3 (SLR1) (db)	11.0	31.0	2.83	2.19	0.66	0.09	1.625	0.014	0.066	0.235	0.062
SLR3 (SLR3) (db)	13.2	36.0	2.72	3.08	0.84	0.122	2.986	0.024	0.039	0.55	0.208
FS (SLR1)	9.6	***34.8***	**3.61**	**3.14**	**3.15**	**0.18**	**4.837**	**0.036**	**0.090**	**0.564**	**0.182**
FS (SLR3)	12.1	**42.7**	**3.54**	**3.39**	**2.20**	**0.16**	**4.328**	**0.038**	**0.072**	**0.449**	**0.218**
LSD	*1.13*	*3.37*	*0.32*	*0.39*	*0.84*	*0.03*	*1.12*	*0.01*	*0.02*	*0.10*	*0.05*

db = dried biosolid

4.5.1.4 Quality of dried FS and dried biosolid dewatered from sawdust-faecal sludge (SD-FS) mixture.

The dried dewatered biosolids were obtained from dewatering 50%, 100%, 150% and 0% sawdust-faecal sludge (SD-FS) mixture on filter beds. The percentage carbon (%C) in the dried dewatered biosolid increased as the percentage sawdust (% SD) increased thus the control had the least %C of 34.0 while the 150% SD mixture had the highest %C of 43.5. The %C in the dried dewatered biosolids of all the treatments were lesser than the dried FS, except the dried dewatered biosolid from the 150% SD-FS mixture. The percentage nitrogen (%N) was higher in the dried FS than the dried dewatered biosolids (Table 4.5.4). The %N in the dried dewatered biosolid decreased as the percentage of sawdust increased, thus the 0% sawdust treatment had the highest

%N of 3.18 while the 150% sawdust treatment had the least %N of 1.82. This consequently affected the C:N ratio, which therefore increased as the percentage sawdust increased (Table 4.5.4). The %P in the dried biosolid of the 0% sawdust treatment (control) which was 3.04, was the highest among the treatments and it decreased as the percentage sawdust increased. The 150% sawdust treatment therefore had the least %P of 1.95. The percentage of potassium (%K) rather increased as the percentage of sawdust increased, thus the 0% sawdust treatment had the least %K of 0.94 while the 150% treatment had the highest %K of 1.67.The %P and the %K in the dried faecal sludge were higher than their corresponding dried biosolids (Table 4.5.4). Copper (Cu) and iron (Fe) in the dried biosolids decreased as the percentage sawdust increased. Lead (Pb) was extremely small in all the samples but was higher in the dried Faecal Sludge than the dewatered biosolids. Cadmium (Cd), zinc (Zn) and manganese (Mn) were all higher in the FS and decreased in the dried biosolids as the percentage sawdust increased (Table 4.5.4) thus the control had the highest concentration in the dried biosolids while the 150% SD treatments had the least concentrations.

Table 4.5.4: Nutrients and Heavy metals in dried dewatered biosolid from different sawdust-FS mixture (N = 6).

	Nutrients as % TS					Heavy metals (HM) as mg/kg					
% Sawdust Added	C:N	%C	% N	%P	%K	Cu	Fe	Pb	Cd	Zn	Mn
50% SD	15.7	37.0	2.35	3.69	1.28	0.122	2.209	0.022	0.055	0.414	0.130
100% SD	18.3	40.0	2.18	2.47	1.55	0.105	2.265	0.021	0.036	0.128	0.093
150% SD	23.9	43.5	1.82	1.95	1.67	0.069	1.530	0.015	0.045	0.076	0.052
0% SD	10.7	34.0	3.18	3.44	0.94	0.157	3.754	0.024	0.047	0.542	0.205
FS	**12.1**	**42.7**	**3.54**	**3.39**	**2.20**	**0.156**	**4.328**	**0.032**	**0.072**	**0.449**	**0.218**
LSD	*4.73*	*3.55*	*0.64*	*0.67*	*0.42*	*0.03*	*1.05*	*0.01*	*0.01*	*0.19*	*0.06*

C, N, P and K = % TS; Cu, Fe, Pb, Cd, Zn and Mn = mg/kg

4.5.2 Discussion of results of agricultural potential of biosolids produced from the dewatering FS using unplanted filter beds

Faecal sludges are usually "cleaner" than sewage treatment plant sludges, as they tend to contain less heavy metals or refractory organics. Exceptions may be found in places where septage is also collected from septic tanks serving cottage or small industrial enterprises (Montangero and Strauss, 2002).

The dried biosolids and the dried FS analysed showed organic carbon content ranging between 28.0% and 43.5%. These values of percentage carbon show significant improvement above values obtained in collaborative research conducted by IWMI, SANDEC, KMA and KNUST (Cofie, 2003), at Buobai in Kumasi, Ghana, on co-composting of dried faecal sludge (DFS) and municipal solid waste (MSW). The analyses of dried faecal sludge (DFS) in that study gave 11.4% carbon content (Cofie, 2003). Drangert *et al*. (1998), through their analyses of human excreta also obtained 11.7% carbon content. These percent carbon (%C) values compared well with municipal solid wastes of developing countries whose content have been found to contain about 40% - 85% biodegradable organics of which 30% - 40% is C. The percentage carbon of the sawdust-FS mixtures which ranged between 37.0 and 43.5% (Tables 4.51 to 4.5.4) compared well with literature compiled by Arends (1985), in which different wood materials showed carbon content of between 43.2% and 49.8%. The high carbon content of the dried FS might be attributed to the dietary content of the people in most developing countries and for that matter, Ghana, whose basic diet is mostly rough carbohydrates. It might partly depend on the type of FS generated, without addition of storm water or industrial waste as found in the developed world. The high organic carbon as found in this type of dried biosolids and FS is very

essential if the use of the biosolid for soil amelioration is paramount. Epstein (2003), asserted that organic carbon increased linearly with biosolid application, and 4 years after application, there was three times as much carbon in the high biosolid application rate soils. Again Epstein (1973; 1975), indicated that organic matter through the activity of microorganisms increases soil aggregation. Adding biosolids and sludge increased the hydraulic conductivity of soils. Organic matter through the addition of biosolids reduced bulk density and increased total porosity and moisture retention of soils (Clapp *et al.* 1986).

The percentage nitrogen (%N) within the range of 1.82% and 3.72% obtained in the samples from the different treatments in this studies was higher, compared to the ranges of 1.3% – 1.6%, being nitrogen levels of human wastes used as raw materials for compost (Shuval *et al.* 1981; Obeng and Wright 1987). The %N values in this study were higher than that of the FS applied in the pilot co-composting at Buobai in Kumasi which contained average N of 0.9% (Cofie, 2003). The biosolid of sawdust-faecal sludge (SD–FS) which recorded comparatively lower nitrogen values even compared better to some of the percentage nitrogen of FS mentioned earlier. Analyses of feaces and human excreta by Strauss (1985) in developing countries produced 4-7% N and 9-12% N of dry TS respectively. Although these values were comparatively higher, the analyses were made from fresh samples of feaces and excreta. The high nitrogen content of the dried biosolids obtained from the dewatering by the unplanted beds serve as good basis for its application in the soil for crop production or as good compost/co-compost material. The organic nitrogen in biosolids applied to land undergoes numerous transformations which are extremely important since they affect plant growth (Epstein, 2003). Chae and Tabatabai (1986) indicated that the rate of

nitrogen mineralization is dependent on moisture, temperature, C:N ratio and biosolids properties. Parker and Sommers (1983) found that amount of mineralisable nitrogen in biosolids was proportional to the total organic nitrogen. Since the organic matter in biosolids incorporated into the soil begin to decompose while the microorganisms utilize the carbon as a source of energy and nitrogen for cell development and growth, the C:N ratio becomes very important. Hoornweg *et al.* (2000), states that the C:N ratio of the final product (compost) should be lower than 22. Good compost is about 10:1 C:N ratio (Bollen and Glennie, 1959). The C:N obtained for these dewatering schemes which ranged between 8.7 and 23.1(Tables 4.5.1 to 4.5.4) can be described as good for composting or for direct application on the land. The highest C:N ratio recorded by 150% SD-FS mixture is high enough to make a good composting material. It compares well with C:N of 31.4 for household waste, 28.5 for Municipal waste and 21.8 for dried faecal sludge which were recorded by Cofie *et al.* (2003) in their pilot co-composting project carried out at Buobai near Kumasi.

Phosphorus (P) is an essential plant nutrient. It has been stated that, P deficiency is the second most important soil fertility problem throughout the world (Lindsey *et al.* 1989). However excessive amounts of P in the soil tend to immobilize other chemical elements such as zinc (Zn) and copper (Cu) that are also essential for plant growth (Chang, *et al.*1983). The %P obtained from the dried biosolids and dried FS which ranged between 1.73% and 3.39% compares well with percentages of 4, 3.7 and 3.8 obtained for dried feaces, urine and excreta respectively reported by Strauss (1985) in developing countries.

Epstein (2003) reported that potassium (K) in biosolids is generally low since K compounds in wastewater are soluble, they do not settle in biosolids (Epstein, 2003). This assertion is actually reflected in these analyses where the dried FS had %K ranging between 2.2 and 3.78% while the dewatered biosolids ranged low between 0.64 and 1.67% meaning that a lot of the K went into the filtrates. However all the %K obtained in the dried biosolid samples compared well with what Strauss (1985) reported, which were 1.6% for feaces and 2.7% for excreta. It also compares well with %K values of the starting compost materials of the pilot co-composting project at Buobai in Kumasi (Cofie, 2003). Potassium is essential element for plant growth. It is also important in amino acids and protein synthesis and photosynthesis (Epstein, 2003).

Copper (Cu), iron (Fe), zinc (Zn), manganese (Mn), lead (Pb) and cadmium (Cd) analysed in this research are classified as heavy metals. Copper is important in many roles including photosynthesis, respiration, enzymes, reproductive growth, and seed and fruit yield (Romheld and Marschner, 1991). Iron is essential to plants, animals and humans. It is a component of protein. It affects photosynthesis, respiration, sulphur reduction and nitrogen fixation (Romheld and Marschner, 1991). Zinc is involved in carbohydrate metabolism as well as proteins and auxins (Epstein, 2003). It is important in stabilisation and structural orientation of certain membrane proteins. Cadmium (Cd) is toxic to animals and man. It is retained in the kidney and the liver and is probably related to the mental binding protein, ‘matallothionein’ (Kagi and Valee, 1960). It has been indicated that a 25% yield reduction for various crops resulted when Cd concentration in the soil ranged from 7 to 160 μg/g dry weight (Millner *et al.*, 1976; Bingham, 1979). Lead (Pb) is non essential element to humans,

animals and plants but is toxic to humans. Generally, toxic intake is 1mg and lethal intake is 10g but the lethal intake for many animals occurs at about 10 mg (NRC, 1980). From the above discussions it is understood that low concentrations of these heavy metals in the soil may promote plant growth or their high concentration in the soils may negatively affect plants growing in the soils or animals and humans that feed on it. This therefore makes their presence in the biosolids very important especially when there is reuse consideration as fertiliser. The concentrations found in the dried biosolids and the dried FS analysed recorded low concentrations that make them insignificant to cause any concern. They were far below the samples used for the pilot co-composting at Buobai (Cofie *et al.* 2003). All were over 100 – 1000 times lower than the corresponding concentrations reported in US and Canadian biosolids (Webber and Nichols 1995; Lue- Hing *et al*, 1999). They were also about 10 – 100 times lower than their corresponding concentrations that occur in the commercial inorganic fertilisers (Bowhay, 1997).

The high level of nutrients and low concentrations of heavy metals recorded in the dried faecal sludges and the dried bioslids are good characteristics for the bioslids to be recommended for co-composting.

CHAPTER FIVE

5 CONCLUSION AND RECOMMENDATION

5.1 General Conclusions

- The general conclusions developed from this study will also serve as criteria for improving the operations and maintenance of unplanted filter beds for the dewatering of faecal sludge produced in Ghana and many developing countries.

- All the filter media with respect to particle size configuration, filter medium one, filter medium two and filter medium three (FM1, FM2 and FM3), used for the study were found to be equally good in dewatering faecal sludge. However, filter medium one (FM1) with smaller particle sizes, was most efficient of all, in terms of contaminant load removal from the faecal sludge. The three filter media had the potential to dewater between 281 and 305 kgTS/m^2/yr of fresh faecal sludge at 1:1 mixing ratio of PTS and septage, annually.

- The filter media with bigger particle sizes (FM3 and FM2) were relatively faster in accumulating organic matter within the pore spaces and thus likely to clog faster than the FM1. FM1 had the added advantage of being able to generate more biosolids than the FM2 and FM3. Average biosolid of between 135 and 154 kgTVS/m^2/yr was generated.

- The different filter media were all not efficient in salts removal since they showed low EC reduction efficiencies which were good for the biosolids generated, if application of the biosolid in agriculture is paramount.

- The pH and temperature were within favourable limits for microbial growth but the high ammonia levels in the percolate could be toxic to the microbes responsible for biological degradation. The contaminant load in the percolates from all the different filter media were all beyond permissible levels for discharge into the environment.

- All the solid loading rates were found to be able to improve the dewatering performance of the filter bed since their average dewatering times improved over the results of the different filter media. However, the SLR3 offered itself for faster dewatering and therefore had shorter dewatering time than SLR2 and SLR1 but in terms of contaminant load removal, the SLR1 was most efficient, though not in all parameters considered. SLR3 had the added advantage of generating the highest volume of biosolid annually. The different solid loading rates had the potential to dewater between 378 and 435 kgTS/m^2/year of faecal sludge annually. They were able to produce biosolid with organic matter content of between 224 and 256 kgTVS/m^2/year.

- Furthermore, the SLR3 was least likely to clog its filter medium compared to the other loading rates.

- The solid loading rate variation was not very efficient in salts removal since the different variations showed low EC reduction. Dissolved salts passed through the filters into the percolate, thus making it too high in the percolate to be applied for irrigation.

- The pH and temperature in the percolates in all the loading rates were favourable for bacterial growth but the high ammonia levels in the percolate could be toxic to bacteria responsible for biological degradation. The contaminant loads in the percolates from all the different solid loading rates were all beyond permissible levels for discharge into the environment.

- When the total solids (TS) of the faecal sludge was kept constant in all the cycles and the number of cycles increased, FM1, FM2 and FM3 dewatering SLR1 improved on their dewatering time. The SLR3 proved to be the best in terms of dewatering times and contaminant load removal under constant TS and increased number of cycles. It was the least likely to clog its filter bed and furthermore improved the percentage organic matter of the dry biosolids and generated the most biosolids. The SLR3 was able to dewater average faecal sludge of 467kgTS/m^2/year and generate biosolid with organic matter content of 334 kgTVS/m^2/year.

- Addition of sawdust to FS for dewatering had the potential to reduce the dewatering time so that more dewatering cycles could be run. Increasing the quantity of sawdust decreased the dewatering time thus 150% sawdust by weight treatment, had the highest potential to improve dewatering time. The addition of different percentages of sawdust made it possible to dewater faecal

sludge with total solids (TS) of between 367 and 525 kgTSm^2/year. The biosolids produced ranged between total solids of 505 and 1239 kgTS/m^2/year with organic matter content of between 359 and 987 kgTVSm^2/year annually.

- The sawdust-faecal sludge mixture was poor in TS and EC removal and this was found to increase with increasing percentage of sawdust. However it was found to be able to remove total volatile solids, suspended solids, COD, NH_3-N and TKN.

- The sawdust addition was found to reduce the organic matter accumulation in the filter bed with the 150% sawdust treatment being the most effective among the different treatments, thus effectively reducing the incidence of clogging. Biosolid generation on the filter bed was found to increase with increasing percentage of sawdust which increased percentage of the organic matter of the biosolids. The biosolids produced had high organic matter and should therefore be co-composted.

- The percentage carbon of all the dewatered biosolids from the filter beds were high enough to be used for composting or co-composting. It further increased with the biosolids of the faecal sludge-sawdust mixture as the percentage of the sawdust increased.
- The percentage nitrogen in all the dried dewatered biosolids obtained from the dewatering schemes was within levels to be used as compostable materials. Other nutrients like phosphorus were also within levels described as good for composting/co-composting. The concentrations of all the heavy metals

analysed in the dried dewatered biosolids were too low to be source of concern when the biosolids are to be considered as soil ameliorants. It is therefore highly recommended that the biosolids produced from faecal sludge dewatered on unplanted filter beds be used for co-composting

5.2 Limitations of the Study

- Removal of biosolids from the filter bed after the dewatering always removed some amount of sand from the filter medium while some amount of biosolid was left on the filter medium. This definitely might have affected the estimation of the dried biosolid and organic matter estimation.

- Although every effort was made to reduce wall effect, total elimination was not possible in such bench scale set up and therefore might have affected the biosolid estimated especially when extrapolated in large scale production.

- The total solids of the faecal sludge applied at different cycles with respect to the public toilet sludge and septage characteristics were always different. The fresh nature could possibly increase specific resistance (SR) of faecal sludges dewatered in different cycles. Thus resisting dewatering and removal efficiency of solids from the faecal sludge dewatered at each cycle.

- It was not possible to analyse the percolate collected from each filter bed daily so it was kept in fridge until the dewatering for the cycle was complete before

composite sample of the percolate was analysed. This could possibly affect the outcome of the percolate results.

5.3 Suggestions for Further Research

- Although high removal efficiencies were achieved for all the different dewatering schemes, the contaminant loads in the percolates were too high and needed to be treated further in systems like column filters, constructed wetlands or stabilisation ponds for more polishing before final discharge. Further investigation must be conducted on the depth of the sand as filter material to find out how it can improve the contaminant load in the percolate.

- The faecal sludge-sawdust mixtures were transferred to filter beds for dewatering immediately after mixing, which did not allow enough time for physical, chemical and biological interactions between the sludge particles, sawdust and associated microorganisms which might have improved dewatering. Further studies must be conducted to determine how time after faecal sludge-sawdust mixture will affect the dewatering performance.

- The effect of rain was eliminated from this study since it was conducted under shelter; therefore further study could be conducted to determine the effect of rain to simulate real field condition.

- The size of the bench scale filter beds and the time frame did not allow the biosolids produced from the filter beds to be processed into compost/co-compost in order to determine the final values of the carbon, nitrogen and

subsequently the carbon nitrogen ratio. Thus, it was difficult to predict the minimum and maximum percentage of sawdust that must be mixed with the faecal sludge that will produce the biosolids of required minimum and maximum carbon nitrogen ratio. Therefore further investigation must be conducted in converting the sawdust-faecal sludge biosolid into compost/co-compost.

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APPENDICES

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APPENDIX A: Tables and Figures

Climatic Data for Kumasi

Table 0-1: Climate Values of 30 Year average (1961–1990) in Kumasi

Parameter	Jan	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov	Dec.	Year
Rainfall (mm)	15.1	66.3	137.0	129.3	174.4	214.3	157.5	89.9	165.2	153.3	74.3	25.8	116.9
No of Rain days	2.0	5.0	9.0	10.0	14.0	17.0	14.0	12.0	17.0	17.0	8.0	3.0	10.7
Max. Temp. (^oC)	31.9	33.5	32.8	32.3	31.3	29.5	28.0	27.7	28.7	30.1	31.2	30.7	30.6
Min. Temp. (^oC)	20.4	22.0	22.3	22.4	22.2	21.6	21.2	21.0	21.1	21.5	21.7	20.8	21.5
Mean.Temp.(^oC)	26.2	27.8	27.5	27.3	26.8	25.6	24.6	24.3	24.9	25.8	26.4	25.8	26.1
Sunshine (h)	6.0	6.6	6.4	6.6	6.6	4.9	4.7	3.8	3.4	5.1	6.3	4.9	5.4

Table 0-2: Climate values of the monitoring period (May 2005 - May 2006)

	May	Jun	Jul	Aug.	Sep.	Oct	Nov	Dec.	Jan.	Feb.	Mar.	Apr.	May
Mean Max. Temp.	31.6	29.4	27.8	27.4	29.7	31.2	32	31.5	32.1	32.6	32.5	33.3	31.5
Mean Min. Temp.	22.5	21.6	20.9	20.3	22.2	22.1	22.8	22.6	22.6	23.1	22.5	23.3	22.7
Av. Rel. Hty (High)	88.5	91	91	91	91	89.5	90.5	90	88.5	89.5	88	88	88
Mean Rel. Hty (Low)	64.5	72	73.5	73.5	69.5	66	62.5	61	58.5	61.5	60	61.5	65

Source: Ghana Metrological Service, Kumasi (Unpublished data)

Tables of particle size analyses

DESCRIPTION: **(0.1- 0.5) = FM1**

Total Dry Weight (g) 100.0

Sieve size		Weight retained(g)	Percentage retained(%)	Percentage passing(%)
BS designation	Metric (mm)			
No. 7	2.36	0.00	0.00	
No. 14	1.18	0.00	0.00	100.00
No. 25	0.600	0.16	0.16	99.84
No. 36	0.425	4.10	4.10	95.74
No. 52	0.300	24.73	24.73	71.01
NO. 72	0.212	28.98	28.98	42.03
No. 100	0.150	22.70	22.70	19.33
No. 200	0.075	18.89	18.89	0.44

DESCRIPTION: **(0.5- 1.0) = FM2**

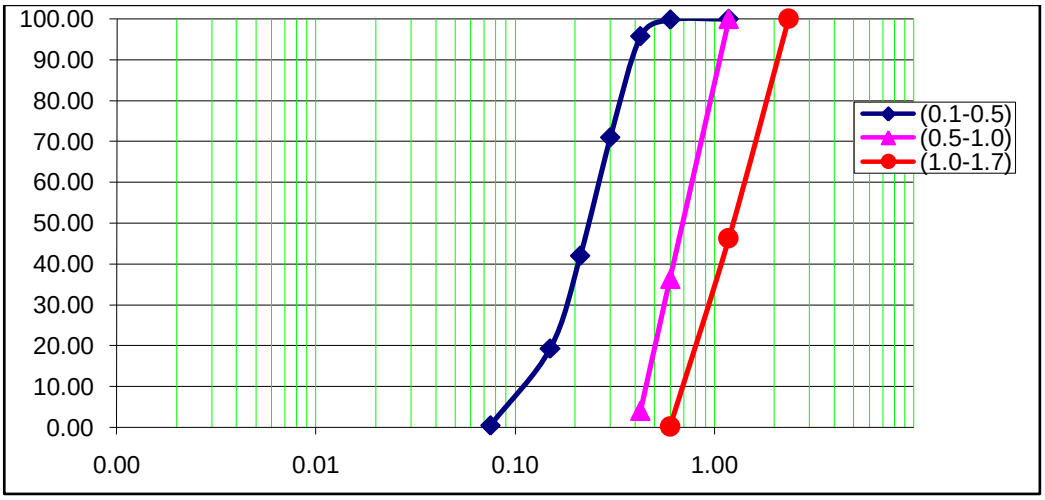
Total Dry Weight (g) 100.0

Sieve size		Weight Retained (g)	Percentage Retained (%)	Percentage Passing (%)
BS designation	Metric (mm)			
No. 7	2.36	0.00	0.00	
No. 14	1.18	0.00	0.00	100.00
No. 25	0.600	63.57	63.57	36.43
No. 36	0.425	32.31	32.31	4.12
No. 52	0.300			
NO. 72	0.212			
No. 100	0.150			
No. 200	0.075			

DESCRIPTION: **(1.0- 1.7) = FM3**

Total Dry Weight (g) 100.0

Sieve size		Weight retained (g)	Percentage retained (%)	Percentage passing(%)
BS designation	Metric (mm)			
No. 7	2.36	0.00	0.00	100.00
No. 14	1.18	53.72	53.72	46.28
No. 25	0.600	46.12	46.12	0.16
No. 36	0.425			
No. 52	0.300			
NO. 72	0.212			
No. 100	0.150			
No. 200	0.075			



Particle size analyses curves for FM1, FM2 and FM3

APPENDIX B: Plates (Photo gallery)

Plate AB 1: Sawdust production point at timber processing company.

Plate AB 2 : Sawdust heap ready for burning

Plate AB3: Sawdust being carried to burning site by labourers

Plate AB4: Sawdust burnt to ashes

Plate AB5: Suction truck discharging at Dompoase treatment plant

PlateAB6: FS in plastic drums being loaded into vehicle.

Plate AB7: Storage containers of sample and the setup

Plate AB8: Arrangements of the filter beds and setup.

Plate AB9: Percolates produced from the experiment.

APPENDIX C: Laboratory analyses

pH

A digital pH measuring kit with a probe or electrode (WTW pH 323-B / Set-2) was used. The electrode was first calibrated against pH buffer 4, 7, and 10.

The percolate and raw sample under test were vigorously shaken before the insertion of the probe. There was a digital read-out when the probe was inserted in the sample.

Temperature

A digital pH measuring kit with a probe (WTW pH 323-B / Set-2) was used. The kit measures both the pH and temperature of the sample. The sample under test was well shaken, and the probe inserted. There was a digital read-out of the temperature when the probe was inserted in the sample.

Turbidity

Attenuated Radiation Method (Direct reading)

Apparatus

HACH DR/2010 Portable Datalogging Spectrophotometer

Procedure

The spectrophotometer was switched on and the stored programme number 750 was entered by pressing the Enter button (or key). The wavelength dial was rotated until the display showed 860nm, and the display quickly showed zero sample and then FAU TURBIDITY.

The sample was well shaken and a portion of it was filtered using the filter paper. 25ml of the filtered portion of the sample (the blank) was poured into a sample cell (bottle). The blank was placed into the cell holder and the light shield closed. The Zero button was pressed and the display showed Zeroing then 0. FAU TURBIDITY. The sample was well mixed and 25ml of it was poured into another sample cell and immediately placed into the cell holder and the light shield closed. The Read button was pressed and the display showed Reading then the turbidity result in Formazin Attenuation Units (FAU).

Total Solids (TS)

Apparatus

Evaporating dish, Desiccator, Drying oven, Steam bath, Weighing Balance

Procedure

The evaporating dish was cleaned and heated in the oven at a temperature of 105°C for one (1) hour. It was then removed and cooled in the desiccator to room temperature. The empty dish was then weighed using the weighing balance and its

weight in grams recorded. The leachate sample was well shaken and a volume of 100ml measured and poured into the weighed dish. The sample was then evaporated to dryness on the steam bath, and then transferred into the oven for drying at 105°C to a constant weight for one to two hours. The dish was removed from the oven, cooled in the desiccator to room temperature, and then weighed

Calculation

$$TS\ (mg/l) = \frac{(A - B) \times 1{,}000{,}000}{V}$$

A = Weight of dish + Residue in grams.

B = Weight of empty dish in grams.

V = Volume of Sample in ml (100ml).

Total Volatile solids

Apparatus

Evaporating dish, Furnace, Steam bath, Weighing Balance

Procedure

The evaporating dish was cleaned and heated in the oven at a temperature of 105°C for at least one (1) hour. The empty dish was then weighed using the weighing balance and its weight in grams recorded. The remains of the sample on which Total solids analysis has already been done is put in the empty dish and weighed. The sample is then ashed at 550°C for 3 hours. After cooling the sample is reweighed.

Calculation

$$TVS\ (mg/l) = \frac{(A - B) \times 100}{W}$$

A = Weight of dish + Residue in grams.

B = Weight of empty dish in grams.

W = Weight of Sample grams

Suspended Solids (SS)

Principle

A well-mixed sample is filtered through a weighed standard glass fibre filter. The residue that is retained on the filter is dried to a constant weight at 105°C. The increase in weight of the filter represents the suspended solids [APHA— AWWA-WEF. (2005)].

Reagents and apparatus

Distilled water, Glass fibre filter, Membrane filter apparatus, Glass dish, Weighing Balance, Vacuum pump, Drying oven, Desiccator.

Procedure

The glass fibre filter was placed on the membrane filter apparatus. The filter flask was connected to the vacuum pump and a vacuum was applied. Three successive 20ml portion of distilled water was passed through the filter by suction to wash the filter. The filter was removed from the membrane and transferred to the glass dish. It was dried in the oven at 105°C for 1hour. The filter was then cooled in the desiccator. The glass dish with the filter was weighed using the weighing balance and the weight recorded. The filter was then placed on the membrane filter apparatus. The sample was well shaken and a measured portion taken. This portion was then diluted with distilled water to 1litre. The diluted sample was well shaken and a volume of 100ml of it was passed through the filter with the aid of the vacuum created by the vacuum

pump. The suspended solids in the sample were retained on the filter as residue. The filter was carefully removed from the membrane apparatus and transferred to the glass dish. The dish with the filter was dried in the oven at 105°C for one hour. The dish was removed from the oven, cooled in the desiccator and then weighed.

Calculation

$$SS\ (mg/l) = \left(\frac{(A - B) \times 1{,}000{,}000}{V} \right) F$$

A = Weight of dish + Filter + Residue in grams.

B = Weight of dish + Filter in grams.

V = Volume of diluted sample in ml = 100ml.

F = Dilution factor

Conductivity (EC)

A digital conductivity measuring kit (WTW LF 323-B / Set) was used. The kit has a probe connected to it. The sample under test was well shaken, and the probe inserted. The sample under test was well shaken, and the probe inserted. There was a digital read-out when the probe was inserted in the sample.

Chemical Oxygen Demand (COD)

Principle

The sample, with the organic matter to be measured, is oxidised under reflux with a known amount of dichromate in concentrated sulphuric acid with silver sulphate as a catalyst. Part of the dichromate is reduced by organic matter and the remainder is determined by titration with ferrous ammonium sulphate using ferroin as indicator.

Reagents and apparatus

Distilled water, 0.25N Potassium dichromate, Mercury (II) sulphate, Silver sulphate solution, Conc. H_2SO_4, Ferroin indicator, Ferrous ammonium sulphate, Measuring cylinder, Volumetric flask, Reflux flask, Burette, Pipettes, Heating device, Condenser.

Procedure

About 1g mercury (II) sulphate was placed in the reflux flask. The sample was well shaken and a 100ml portion taken. This portion was then diluted with distilled water to 1litre. The diluted sample was well shaken and a volume of 10ml of it was pipetted into the flask. 10ml of 0.25N Potassium dichromate followed by 20ml of concentrated sulphuric acid were added to the flask. The flask was then cooled under running water, and 1ml of silver sulphate solution was added. The process was repeated using 10ml of distilled water instead of sample as blank. The content of each flask was mixed well and the flasks fitted to the condenser with the bottom of the flasks resting on the heaters. The heaters were switched on and the mixtures boiled under open reflux for 2 hours. Flasks were allowed to cool and condenser was washed down with distilled water. Flasks were removed and 45ml of distilled water was added to each flask. Flasks were cooled under running water and 3 drops of ferroin indicator were added producing a light blue-green colour. The residual solution was then titrated with ferrous ammonium sulphate to a reddish-brown end-point.

Calculation

$$\text{COD (mg/l)} = [(A–B) \times N \times 8000/ V] \times F$$

where:

A = volume of ferrous ammonium sulphate used for the blank titration (ml)

B = volume of ferrous ammonium sulphate used for the sample titration (ml)

V = volume of diluted sample used (10ml)

N = normality of ferrous ammonium sulphate

F = dilution factor (10)

Standardisation of ferrous ammonium sulphate

The normality of ferrous ammonium sulphate is not stable; it changes with time. Thus for a COD test on any particular day, the normality was checked as illustrated below: 10ml of the standard Potassium dichromate solution was diluted with distilled water to about 100ml. 30ml of conc. sulphuric acid was added and cooled under running water. Three (3) drops of ferroin indicator was added and titrated with the ferrous ammonium sulphate to reddish-brown end-point.

Normality = (C x 0.25)/ D

where:

C = volume of Potassium dichromate (10ml)

D = volume of ferrous ammonium sulphate (ml)

0.25 = normality of the standard Potassium dichromate

Ammonia-nitrogen (NH_3-N)

(Distillation and Titrimetric Method)

Reagents and apparatus

Distilled water, Borate buffer, Indicating boric acid, 6N Sodium hydroxide solution, 0.02N Sulphuric acid, pH meter, Volumetric flask, Kjeldahl (Distillation) flask, Burette, Pipettes, Erlenmeyer flask, Heating device, Distillation apparatus

Procedure

The sample was well shaken and between 5 and 50 ml was portion taken depending on the concentration of the sample. This portion was diluted with distilled water to 300ml and poured into the Kjeldahl flask. 25ml borate buffer was added and the pH of the solution was altered to about 9.5 using 6N Sodium hydroxide solution. The process was repeated using 300ml of distilled water instead of sample as blank. 50ml indicating boric acid was pipette into the Erlenmeyer flasks. The contents of each Kjeldahl flask was mixed well, and the flasks put on the heaters and connected to the distillation apparatus which was previously steamed out with a mixture of 50ml water and 20ml borate buffer of pH 9.5. The Erlenmeyer flasks with the indicating boric acid were placed at the lower end of the condenser with the tip of the delivery tube well below the surface of boric acid. The heaters were switched on and at least 200ml of distillate containing ammonia was distilled into the boric acid. The presence of ammonia in the distillate changes the blue-black colour of the boric acid to green. The ammonia in distillate was then titrated against standard 0.02N sulphuric acid to a pale lavender end-point.

Calculation

NH3–N (mg/l) = [(A - B) x 280] / V

where:

A = volume of sulphuric acid titrated for sample (ml)

B = volume of sulphuric acid titrated for blank (ml)

V = volume of sample used (ml)

Total Kjeldahl Nitrogen (TKN)

The Total Kjeldahl Nitrogen was obtained by the sum of ammonia-nitrogen (NH_3-N) and organic-nitrogen. Thus

$$TKN\ (mg/l) = NH_3\text{-}N\ (mg/l) + \text{Organic-nitrogen}\ (mg/l)$$

The organic-nitrogen was determined as follows:

The residual solutions in the Kjeldahl flasks after the ammonia determination were allowed to cool. One Kjeldahl catalyst tablet followed by 10 ml concentrated sulphuric acid were added to each flask. The solutions were heated (or digested) under hood until white fumes of SO_3 were seen. The heating continued until the solutions became clear or pale straw. The flasks and their contents were allowed to cool. The residual solutions were diluted with distilled water to 300ml. 50ml sodium hydroxide-sodium thiosulphate solution was added gently to the solution in each distillation flask, and the pH checked to ensure that the solutions were alkaline. 50ml indicating boric acid was pipette into the erlenmeyer flasks, and at least 200ml distillate was distilled into the boric acid. The distillate was then titrated against standard 0.02N sulphuric acid to a pale lavender end-point.

Calculation

$$\text{Organic-nitrogen}\ (mg/l) = [(C - D) \times 280] / V$$

where:

C = volume of sulphuric acid titrated for sample (ml)

D = volume of sulphuric acid titrated for blank (ml)

V = volume of sample used (ml)

Nitrate-nitrogen (NO_3-N)

(Cadmium Reduction Method using Powder Pillows)

Apparatus

HACH DR/2010 Portable Datalogging Spectrophotometer

Procedure

The spectrophotometer was switched on and the stored programme number 355 was entered by pressing the Enter button (or key). The wavelength dial was rotated until the display showed 500nm, and the display quickly showed zero sample and then mg/l NO_3-N HR. The sample was well shaken and a 10ml portion of it taken and diluted with distilled water to 1 litre. 25ml of the diluted sample was poured into a 25ml sample cell. The contents of one Nitra Ver 5 nitrate reagent powder pillow was added to the prepared sample in the cell and then covered. The Shift Timer button was pressed and the cell was vigorously shaken until the timer beeps in one minute. When the timer beeps, the shaking was stopped and the Shift Timer pressed again for a five-minute reaction period to begin. A second 25ml sample cell was filled with the diluted sample, which was the blank and covered. When the five-minute period was over, the timer beeps and the display showed mg/l NO_3-N HR. The blank was placed into the cell holder and the light shield closed. The Zero button was pressed and the display showed Zeroing then 0.00 mg/l NO_3-N HR. The blank was removed and the prepared sample was then placed into the cell holder and the light shield closed. The Read button was pressed and the display showed Reading then NO_3-N result in mg/l was displayed. The procedure was repeated using demineralised (or distilled) water.

Calculation

$$NO_3\text{-N (mg/l)} = AF - B$$

where:

A = NO_3-N value obtained for the diluted sample.

B = NO_3-N value obtained for the demineralised water.

F = dilution factor of sample.

Phosphate

(Phos Ver 3 (Ascorbic Acid) Method using Powder Pillows)

Apparatus

HACH DR/2010 Portable Datalogging Spectrophotometer

Procedure

The spectrophotometer was switched on and the stored programme number 490 was entered by pressing the Enter button (or key). The wavelength dial was rotated until the display showed 890nm, and the display quickly showed zero sample and then mg/l PO_4^{3-} PV. A 10ml Cell Riser was inserted into the cell compartment. The sample was well shaken and a 10ml portion of it taken and diluted with distilled water to 1 litre. 10ml of the diluted sample was poured into a 10ml sample cell. The contents of one Phos Ver 3 phosphate powder pillow was added to the sample in the cell and immediately swirled to mix. A blue colour forms if phosphate is present. The Shift Timer button was pressed and a two-minute reaction period began. A second 10ml sample cell was filled with the diluted sample, which was the blank. When the two-minute period was over, the timer beeps and the display showed mg/l PO_4^{3-} PV. The blank was placed into the cell holder and the light shield closed. The Zero button was pressed and the display showed Zeroing then 0.00 mg/l PO_4^{3-} PV. The blank was removed and the prepared sample was then placed into the cell holder and the light shield closed. The Read button was pressed and the display showed Reading then the

orthophosphate (PO_4^{3-}) result in mg/l was displayed. Other forms of phosphorus, P, and polyphosphate (P_2O_5), can be obtained by pressing the arrow buttons in turn.

The procedure was repeated using demineralised water.

Calculation

$$PO_4\text{-P (mg/l)} = CF - D$$

where:

C = PO_4-P value obtained for the diluted sample.

D = PO_4-P value obtained for the demineralised water

F = dilution factor of sample

Nutrients and Heavy metals analyses

The nutrients included Organic Carbon (org.C), Total Nitrogen, Phosphorus, Potassium. The heavy metals included Copper (Cu), Iron (Fe), Lead (Pb), Cadmium (Cd), Zinc (Zn) and Manganese (Mn).

Total Nitrogen Determination in dried faecal sludge

Procedure Using a Block Digester (Okalibo, *et al*. (2002); Rayment and Higgingson, (1969); Black, (ed) Weighed 0.3 ≠ 0.001 g of oven dried faecal sludge samples (70 ^oC), grounded < 0.25mm, 60 mesh) into labelled, dry and clean tube. Sulphuric acid-selenium mixture was added (added 3.5 g selenium powder to 1 litre H_2SO_4 and heated to 300^oC while covering the beaker with watch glass. The original blackish colour of selenium suspension turned via green-blue to clear light yellow). 2.5 –ml digestion mixture was added (Dissolved 3.2g salicylic acid in 100ml sulphuric acid – selenium powder mixture) to each tube and the reagent batch of samples. This mixture should not be stored for more than 48hrs.

The mixture was digested at 110 ^{o}C for 1 hour. Removed, cooled and added three successive 1-ml portions of hydrogen peroxide. The temperature was raised to 330 ^{o}C and continued heating until the solution turned colourless and any remaining sand white. The content was allowed to cool. About 25ml distilled water was added and mixed well until no more sediment dissolved. Allowed to cool and made up to 50 ml with distilled water. It was allowed to settle after which a clear solution was taken from the top of the tube for analysis. Acid digestion of the dried sludge material was followed by distillation – titration.

Distillation and titration

Free ammonia is liberated from solution by steam distillation in presence of excess alkali (NaOH). The distillate is collected in a receiver (50ml conical flask) containing excess boric acid with drops of mixed indicator.

Procedure

5 ml aliquot of sample solution (digest above) was transferred to a Markham distillation apparatus. 5ml of 40% sodium hydroxide, and 100ml of distilled water was added. (Dissolve, carefully, 400 g NaOH in distilled water and then dilute to 1.0 litre= 40% NaOH. The mixture was distilled, collecting the distillate in 5ml of 2% boric acid indicator. Boric acid (2%): 2g of Boric acid in 100ml of distilled water.

The distillate was titrated with 0.014M HCl from green to reddish end point and the titre value was recorded.

Calculations

$$\%N \text{ in dried sludge sample} = \frac{\mathbf{(a-b) \times 0.014 \times 14 \times v \times 100}}{\mathbf{1000 \times w \times al}}$$

where

v = final volume of the digestion = 100ml

w = weight of the sample taken in grams

al = aliquot of the solution taken for analysis

0.014= Molarity of acid

14 = Molar weight of nitrogen

Determination of Organic carbon

Introduction

Carbon is the chief element of soil/sludge organic matter that is readily measured quantitatively. Hence, estimates of organic matter are frequently based on organic-C, which is determined mainly by two methods namely:

(1) Those based on quantitative combustion procedures wherein C is determined as CO_2, and

(2) Those based on the reduction of the $Cr_2O_7^{2-}$ ion by organic matter in which case the unreduced $Cr_2O_7^{2-}$ is measured by titration. Organic forms of sludge – C plus the carbonate forms of this element including HCO_3^- and CO_3^{2-} of any soluble salts present, constitute the total- C present in soils/dry sludges.

Values for the organic-C content of sludge may be expressed as such or may be reported as total organic matter by multiplying the figure for organic-C by the conventional 'Von Bemmlen" factor of 1.724. A modification of the wet oxidation procedure based on the reduction of the $Cr_2O_7^{2-}$ ion by organic matter which is known as the Wakley and Black procedure was employed in this exercise. Oxidizable matter in a sludge sample is oxidized by $Cr_2O_7^{2-}$, and the reaction is facilitated by the heat generated when 2 volumes of H_2SO_4 are mixed with 1 volume of 1N K_2 Cr_2O_7

solution. The excess $Cr_2O_7^{2-}$ is determined by titration with standard ferrous ammonium sulphate solution, and the quantity of substances oxidized is calculated from the amount of $Cr_2O_7^{2-}$ reduced (Okalibo, *et al.* (2002); Rayment and Higgingson, (1969)).

Reagents

1. Potassium dichromate (K_2 Cr_2O_7) IN: Dissolved 49.04g of reagent grade K_2 Cr_2O_7 (dried at 105°C) in water and diluted the solution to the volume of 1000ml.
2. Sulphuric acid (H_2SO_4), concentreated (not less than 69%). Due to presence of CI- in the sludge, 1.25g Ag_2SO_4 was added to every 100ml of the conc. H_2SO_4 employed.
3. Orthophosphoric acid (H_3PO_4): 85%.
4. Ferrous ammonium sulphate (0.2N): 78.44g Fe $(NH_4)_2(SO_4)_2$ was dissolved in 500ml of water. Add 20ml of conc. H_2SO_4 and then make up to 1 litre.
5. Barium diphnylamine sulphonate: 0.16% aqueous soln.

Procedures

1. Weigh representative sample of compost or 0.5g of soil sample screened through 0.5mm sieve into a 250 Erlenmeyer flask .
2. From a burette exactly 10ml. of dichromate solution followed by 20ml. of conc. H_2SO_4 was added. The flask was swirled and made sure that the solution was in contact with all the particles of the sludge.
3. Allowed the flask and content to stand on an asbestos sheet for 30 minutes.
4. 200ml of distilled water, 10ml of ortho phosphoric acid and finally 2ml of barium diphenylamine sulphonate indicator were added.

5. It was titrated with the ferrous ammonium sulphate solution until the colour changed to blue then to a green end-point.

Calculation of percentage of Carbon

The following formula was used: % Carbon $= \frac{\{10.0 - (X\ N\} * 0.3 * 1.33}{w}$

where X = ml of ferrous ammonium sulphate solution required for the titration.

N = normality of ferrous ammonium sulphate solution (1*10=X *blank titre, where X = 10/blank titre).

w = weight of soil sample in gram.

10 = ml of dichromate solution

1.33 = correction factor

Calculation of percent Organic matter

Multiply the percentage C by the factor 1.724 to convert to organic matter.

Caution

Whenever the end point is overshot during the titration with ferrous ammonium sulphate, more dichromate solution can be added and the titration continued; but, this volume should be accounted for in the above formula when doing the calculation.

Determination of Total Phosphorus

(Total phosphorous without pH adjustment)

Reagents

Sulphuric acid, H_2SO_4, 5N: Place one litre clean beaker on asbestos mat (or in cold water in sink). To about 500 ml distilled water, add slowly with stirring, 148 ml conc. H_2SO_4. When cool dilute to 1 litre with water.

Ammonium mollybdate/ antimony potassium tartrate solution – dissolve 12 g ammonium molybdate $(NH_4)_6Mo_7O_{24}.4H_2O$) in 250 ml of warm (50^oC) distilled water. Separately dissolve 0.291g antimony potassium tartrate ($KSb.C_4H_4O_6$) in 100ml distilled water. Add both solutions to 1000 ml of 5 N H_2SO_4 (above). Mix thoroughly and dilute with distilled water to 2 litres. Transfer to a reagent bottle. Store in a dark, cool place. The mixture (Reagent A) keeps for 2 months.

Ascorbic acid reducing agent-Dissolve 2.108-g ascorbic acid ($C_6H_8O_6$) in 400 ml of ammonium molybdate/antimony potassium tartrate solution (above) and mix well. This must be prepared as required on the day of analysis (this is adequate for about 30 samples plus the P standards). The solution keeps for about 24 hours. Larger quantities of this reducing agent may be prepared depending on the output of a specific laboratory.

Standard phosphorous stock solution, 1000 ppm P: Weigh 1.0982 g of oven- dry KH_2PO_4; dissolve and make to 250 ml mark with distilled water (1 ml = mg P).

10 ppm P working solution: Dilute 10 ml of the standard stock solution above (1000 ppm P) to 1 litre with distilled water.

Procedure:

5 ml of the supernatant clear wet-ashed digest solution was pipetted into a 50 –ml volumetric flask. About 20-ml distilled water was added to each flask. 10 ml of the ascorbic acid reducing agent was added to each flask, beginning with the standards (see below). Made to 50ml with water; it was given a stopper and shaken well. Allowed to stand for 1 hour to permit colour development. The standards and sample absorbance (blue colour) were measured at 880nm wavelength setting in a suitable colorimeter.

Standards

The standards were prepared by pipetting 0, 1, 2, 3, 4, 5, and 6 ml of the 10 ppm P working solution (above) into 50 ml volumetric flasks. 10 ml of the ascorbic acid reducing solution was added to each flask and made to the mark with distilled water. Allowed to stand for 1 hour and read the absorbance exactly like the sample solutions above. The standards contained 0, 0.2, 0.4, 0.6, 0.8, 1.0 and 1.2 ppm P respectively.

Calculations

A graph of absorbance against standard concentration (above) was plotted. The solutions concentration for each unknown and the 2 blanks were determined. The mean blank value was subtracted from the unknowns; this gave a value for corrected concentration (= c in subsequent calculations).

$$\text{P in sample (\%)} = \boxed{\frac{c \times v \times f}{w}}$$

Where c = the corrected concentration for sample solution;

w = the weight of sample taken

v = final volume of the digestion

Sample calculation

A standard graph was prepared in which absorbance was plotted on the y -axis and P concentration (ppm) was plotted on the x-axis. Based on the absorbance of the blank sample and a sample, P ppm value was assigned to each, for example assuming that the blank = 0.2 ppm and the sample = 4.05 ppm. The corrected P concentration (c) = 4.05 - 0.20 = 3.85 ppm. For a 5 ml aliquot as in the ascorbic acid method without pH adjustment and a 50 ml final dilution used in absorbency measurement:

$$\text{P in sample (\%)} = \frac{c \times 0.05}{w}$$

Determination of Potassium

(Okalibo, *et al.* (2002)

This method can also be used for the determination of Sodium, Calcium and Magnesium in dried sludge.

Principle

Total nutrient cation contents of sludge, plants and other organic materials may be measured by the complete oxidation of samples using Kjeldahl procedures followed by spectrometric analysis. Either a flame photometer or atomic absorption spectrophotometer may be used for potassium and calcium analyses but atomic

absorption is necessary for the analysis of magnesium. Standard solutions containing known mixtures of both sodium and the nutrient cations are used because of interference that may occur as a result of mutual excitation between elements.

Sludge Sample Digestion

Reagents

Analytical reagent grade ('AR') chemicals are highly recommended

1. Selenium powder, Se
2. Lithium sulphate, Li_2SO_4, H_2O
3. Hydrogen peroxide, 30%, H_2O_2 (or 100 volumes)
4. Sulphuric acid H_2SO_4, concentrated
5. Digestion mixture: Add 0.42 g selenium powder and 14 g Lithium sulphate to 350 ml 30% hydrogen peroxide and mix well. Slowly add with care 420 ml concentrated H_2SO_4 while cooling in an ice bath. Store at so C ; the mixture is stable for 4 weeks.

Procedure

1. 0.3 + 0.001 g of oven dried (70°C) was weighed, The dried sludge sample was grounded into a labelled, dry and clean digestion tube.
2. 4.4 ml digestion mixture was added to each tube and also to 2 reagent blanks for each batch of samples.
3. It was digested at 360°C for 2 hours. The solution turned colourless and any remaining sand white. Allowed the contents to cool.
4. About 25 ml distilled water was added and mixed well until no more sediment dissolves. Allow to cool.

5. It was made up to 50 ml with water and mixed well.
6. Allowed to settle so that a clear solution was taken from the top of the tube for analysis.
7. The K, was determined in the digests as described below

Determination of potassium

(Okalibo, et. al. (2002)

Reagents

1. Stock sodium solution, 1000, ppm Na. 2.541 g of dry (100 ^{o}C, 2 hour) sodium chloride (NaCl), was weighed and dissolved in distilled water and made to 1 litre. Stored in a reagent bottle.
2. Working sodium solution, 100 ppm Na. 20 ml of the above stock solution was diluted to 200 ml (this is solution 1).
3. Stock potassium solution, 1000 ppm K. 1.907 g dry (100°C, 2 hours) potassium chloride was weighed. It was dissolve and made to I litre with distilled water. It was stored in a reagent bottle.
4. Working potassium solution, 100 ppm K. 20 ml of the above stock solution was diluted to 200 ml (this is solution ii).
5. (Sulphuric acid, 1N H_2SO4). 28.0 ml of concentrated H_2SO_4 was carefully diluted to I litre with distilled water.

Potassium and sodium standards. Pipette working Na (I) and K (II) solutions to obtain a seven steps standard series as follows:

5 ml of I plus 10 ml of II into 100 ml flask, giving 5. 0 ppm Na and 10 ppm K

4 ml of I plus 8 ml of II into 100 ml flask, giving 4.0 ppm Na and 8 ppm K

3 ml of I plus 6 ml of II into 100 ml flask, giving 3.0 ppm Na and 6 ppm K

2 ml of I plus 4 ml of II into 100 ml flask, giving 2.0 ppm Na and 4 ppm K

1 ml of I plus 2 ml if II into 100 ml flask, giving 1.0 ppm Na and 2 ppm K

0.5 ml of I plus I ml of II into 100 ml flask, giving 0.5 ppm Na and 1 ppm K

0 ml of I plus 0 ml of ii into 100 ml flask, giving 0.0 ppm Na and 0 ppm K

Before making to 100 ml (with distilled water), add 3 ml of 1 N H_2 SO_4 to each flask. The standard solutions are best stored in reagent bottles.

Procedure

2 ml of the wet- digested sample solution (above) was pipetted into a 50 ml volumetric flask. It was made to mark with distilled water and mixed well. The solutions were sprayed starting with standards, the sample and blank solutions directly into the flame of the flame photometer (for Na &K) or (wavelength at 766,5nm). The amount of potassium present in the solution (c) was read off from the calibration curve prepared by plotting absorbance readings against potassium concentrations in the standard series. The operation instructions given for flame photometer was followed.

Calculation

The concentration of potassium in the plant sample expressed in percentage is calculated as follows;

$$K\ (\%)\ \text{in the sample} = \frac{(a-b) \times v \times f \times 100}{1000 \times w \times 1000}$$

$$\%K_2O = \%K \times 1.2046$$

where a = concentration of potassium in the digest; b = concentration of flask of the blank digest; w = the weight of sample; v = volume of the digest solution and f = dilution factor.

Determination of Heavy Metals in dried sludge samples

This section covered methods of digestion and determination for heavy metals in dried sludge sample. Heavy metals of concern which were determined included Copper (Cu), Iron (Fe), Lead (Pb), Cadmium (Cd), Zinc (Zn) and Manganese (Mn). Others like, mercury (Hg), nickel (Ni), arsenic (As) could not be determined due to logistics.

Principle:

The content of the heavy metals are measured in a digest obtained by treating samples with an acid mixture made from conc. nitric acid, conc. sulphuric acid, and perchloric acid, (Chapman and Pratt, 1961; Association of Official Analytical Chemist,1979). .

Reagents

1. Perchloric acid 60-62%, (2). HNO_3 Conc. AR, (3). H_2SO_4 conc.AR

Procedure

1. 1.0 g of ground dried sludge material (oven-dry, 60 ºC) was weighed into a 125 ml Erlenmeyer flask which had been previously washed with an acid and distilled water.
2. 5 ml of Ternary mixture (20ml $HClO_4$: 500ml HNO_3: 50ml H_2SO_4) was added under a fume hood.
3. The contents was mixed and heated gently at low to medium heat on a hot plate under a perchloric acid fume hood.

4. Heating continued until dense white fumes appeared (i.e. fumes of sulphuric acid).
5. It was finally, heated strongly (medium to high heat) for half a minute.
6. Allowed to cool, and then added 40-50 ml distilled water. It was boiled for half a minute on the same plate at medium heat.
7. The solution was cooled and filtered completely with a wash bottle into a 100ml Pyrex volumetric flask. It was made up to the mark with distilled water (Whatman No.42 filter paper, 9cm used).
8. The solution was stored for heavy metal determination in Atomic Absorption Spectrophotometer (AAS), Perkin-Elmer Corp. (1968).