Civil and Environmental Research www.iiste.org
ISSN 2224-5790 (Paper) ISSN 2225-0514 (Online)
Vol.3, No.4, 2013

Effects of Different Solid Loading Rates of Faecal Sludge on the Dewatering Performance of Unplanted Filter Bed.

A. R. Kuffour[1 *], E. Awuah[1], D. Sarpong[1], F. O. K. Anyemedu[1] and D. Koné[2]

1. Department of Civil Engineering, Kwame Nkrumah University of Science and Technology (KNUST), Kumasi, Ghana.
2. Eawag: Swiss Federal Institute of Aquatic Science and Technology, Department of Water and Sanitation in Developing Countries (Sandec), Ueberlandstrasse, 133 CH-8600 Duebendorf, Switzerland.

*Email of corresponding author: rakuffour@gmail.com

Abstract

The aim of this study was to investigate which Solid Loading Rate (SLR) of faecal sludge will best improve the dewatering performance of selected sand with particle sizes range of ($\geq 0.1 \leq 0.5$) mm raised on bench scale filter beds. Public toilet sludge and septage collected from suction trucks discharging at Dompoase stabilisation ponds in Kumasi , Ghana, mixed in the ratio of 1:1, 1:2 and 1:3 by volume representing SLR1, SLR2 and SLR3 respectively, were used for the dewatering. Percolate volume was measured every 24 hour. The faecal sludge of SLR1, SLR2 and SLR3 dewatered at average dewatering times of 7, 5 and 4 days respectively. Removal efficiencies of 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 10cm of the filter bed was least in SLR3. Again SLR3 showed the highest potential for annual generation of biosolids at 438, 421 and 379 (kg/m^2 year) for SL3, SLR2 and SLR1 respectively. Therefore SLR3 of faecal sludge is recommended for dewatering on the selected filter bed.

Key words: Faecal sludge, solid loading rate, dewatering time, filter bed, percolate.

1.0 Introduction

Excreta are a rich source of organic matter and plant nutrients such as nitrogen, phosphorus and potassium. Each day, humans excrete in the order of 30g of carbon, 10-12g of nitrogen, 2g of phosphorus and 3g of potassium per capita. Most of the organic matter is contained in the faeces, while most of the nitrogen (70-80 %) and potassium are contained in urine. Phosphorus is equally distributed between urine and faeces. Ideally every person's excreta is in theory at least, nearly sufficient to grow one's own food (Drangert, 1998). Excreta is not only a fertilizer, its organic matter content, which serves as a soil conditioner and humus replenisher, an asset not shared by chemical fertilisers, is of equal or even greater importance (Winblad,1997). Faecal sludge (FS) generated in Ghana and many developing countries is mainly made up of public toilet sludge (PTS) and septage. FS is sludge of variable consistency accumulating in septic tanks, aqua privies, family latrines and unsewered public toilets. PTS is the term used for sludges collected from unsewered public toilets which are usually of higher variability than septage and biochemically less stabilized (Montangero and Strauss, 2002). Septage is the sludge produced in individual on-site wastewater-disposal systems, principally septic tanks and cesspools (Metcalf and Eddy, 1995).

Current waste management proposes that sanitation systems whenever feasible should allow for recycling of organic matter and nutrients in human excreta (Esrey *et al.,* 1998). This paradigm shift from flush and discharge to recycling of nutrients is becoming popular in Europe (Larsen and Guyer, 1996). As a consequence, treatment strategies and technological options for faecal sludges (FS) and solid waste have to be developed to allow the optimum recycling of nutrients and organic matter to improve agriculture.

Generally, there is lack of sustainable options for treating FS in many cities in developing countries. Thus faecal sludge from on-site sanitation systems (OSS) is collected and disposed-off untreated and indiscriminately into drainage ditches, inland waters and coastal waters. This is mainly due to lack of affordable treatment systems (Heinss *et al*., 1998). On the average, each person generates 1L of FS/day (Metcalf and Eddy, 1995). Therefore 1,200m^3 of FS should be collected and disposed of in a city of 1.2 million inhabitants like Kumasi. However it is observed that the amount collected at the Dompoase treatment plant in Kumasi is less than half of what is expected (Adongo, personal communication). While almost all the rest is discharged into the environment untreated. These practices are responsible for many environmental health hazards to the public. In Africa, contamination of faecal origin appears to be responsible for many enteric diseases notably in children.

The search for improved and affordable treatment systems prompted the study into other alternative treatment systems that can better treat the sludge. Previous researches clearly indicate that unplanted drying beds have proved to be effective in this line of intervention but not without related problems. Among the problems associated with dewatering of faecal sludge on unplanted filter beds are long dewatering times, clogging of the filter beds, high contaminant load in the percolate, appropriate loading rates of the faecal sludge and filter materials used for the dewatering. However, these problems need to be addressed.

This study was conducted to determine the appropriate mixing ratios of public toilet sludge and septage that will improve the faecal sludge dewatering efficiency of filter beds. This could enhance the treatment of large volumes of faecal sludge generated in Ghana for a better sanitation delivery and agricultural productivity.

2.0 Materials and Methods

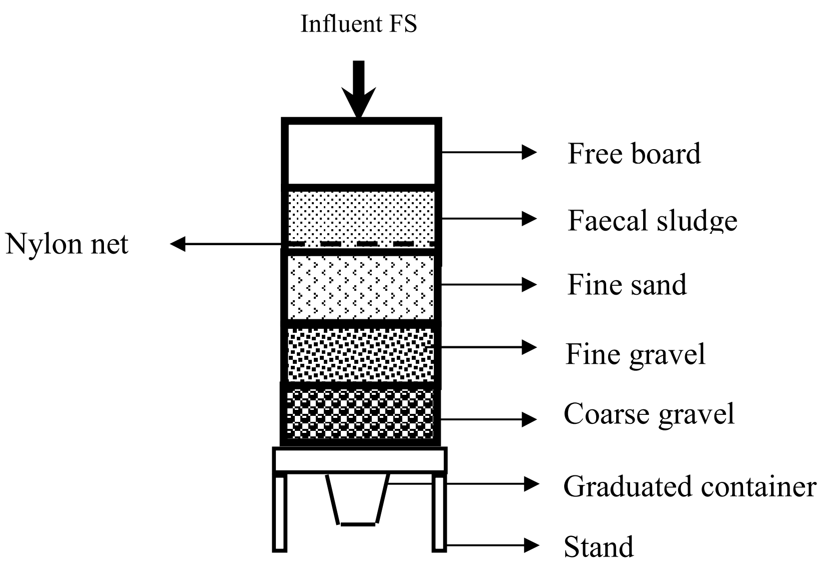
2.1 Experimental Setup

A wooden structure of size 4.5m long by 3.5m wide and 2m high was raised and roofed with iron sheets. A wooden table of 3m long, 2m wide and 0.4m high was constructed under it which served as a platform for raising miniature filter beds. Each filter bed was raised in a cylindrical plastic container having a size of 0.85m long 0.175m diameter. There were 9 of such equal filter beds consisting of 3 blocks with each replicated 3 times.

2.1.1 Filter Material and Filter Bed Preparation

Coarse and fine gravel of about 2.0-3.0 cm and 0.5-1.0 cm diameter respectively, served as base support for the filter medium (sand). The particle sizes of sand between ≥ 0.1mm - ≤ 0.5mm was obtained by sieve analyses (kuffour *et al.,* 2009). This constituted the selected filter medium based on previous research conducted. Effective sizes (Es) and uniformity coefficients (U_o) of the sand were determined. The permeability of the filter medium was determined using constant head permeability test. The base of the filter bed was filled with the course gravel to 15cm depth, followed by the fine gravel to 10cm depth, with the sand on top to 20cm depth. Nylon net was placed on the sand on which the FS was poured, to ensure easy removal of the dewatered biosolid (Figure 1).

Figure 1: Schematic diagram of the filter bed



2.1.2 Collection of Faecal Sludge (FS) and Dewatering

PTS from public toilets and septage from septic tanks were collected in separate plastic drums of about 90 liters capacity each from FS suction trucks discharging at Dompoase landfill site in Kumasi, about 20 km away from the project site. The PTS and septage were mixed in ratios of 1:1, 1:2 and 1:3 by volume, and a hydraulic loading of 20cm (equivalent to 5 liters) was applied on each filter bed after the Total Solid (TS) of each mixing had been determined. The TS of the different mixing ratios were used as the solid loading rates, ie. SLR1, SLR2 and SLR3 for 1:1, 1:2 and 1:3 respectively. By this variation, the solid content of the sludge is also varied accordingly. The volume of the percolate 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 from early January to early June 2006. Dewatering was considered complete when biosolid accumulated was spadable for possible removal from the filter bed and the percolates have ceased dripping.

2.1.3 Design of Experiment

Filter beds comprising of 3 blocks with each block replicated 3 times containing each of the different solid loading rates (SLR1, SLR2 and SLR3) were arranged in a complete randomized block design.

2.2 Laboratory Analyses

The raw 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. 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 was analyzed for Total Solids and Total Volatile Solids (TVS) after each cycle. The parameters analyzed in the raw FS and percolate were, Chemical Oxygen Demand (COD), Dissolved Chemical Oxygen Demand (DCOD), Total Solids (TS), Total Volatile Solids (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, 1998) were used for the analyses of the parameters.

3.0 Results and Discussion

3.1 Raw Faecal Sludge (FS)

The septage used appears to have lower BOD 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. A further portion of the BOD is lost through discharge of the supernatant into soil infiltration systems or into surface drains (Heinss *et al*., 1998). Intrusion of water from underground discharge also reduces the septage BOD. This underlining factor is reflected in the reduced concentrations of the parameters as the septage ratio in the mixture increases (Table 1).

Table 1: Characteristics of raw FS (no. of cycles = 6)

	Raw Samples		
Parameter	**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.(^{o}C)	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)*

3.1.1 Solids

The raw FS from various mixtures of septage and PTS analyzed had very high values for TS, SS and TVS (Table 1). This is common with contents of public latrines (VIP, pit latrines, aqua privies etc.) since little or no water is used for flushing. TVS is very high due to high organic content of FS in Ghana and other developing countries. These values compared well with results obtained in studies conducted in Accra (Heinss *et al*., 1998).

3.1.2 Degradable Organics

The raw FS had high COD and DCOD values ranging between (43.40 – 33.47)g/l and (9.7 – 6.10)g/l respectively which decreased as the PTS ratio decreased (Table 1). Such high COD values have been recorded in FS from Bankok, Manila and Accra (Heinss *et al*., 1998). The concentrations of COD are several times (10 - 100) the strength of sewage (Strauss *et al.,* 1997). This is mainly due to the PTS which is fresh and undigested.

3.1.3 Temperature and pH

There wasn't much variation in temperature (29.8 – 29.9)^{0}C in all the mixing ratios (Table 1). They were all within the mesophillic range (25 – 40)^{0}C which is ideal for bacterial activity (Metcalf and Eddy, 1995). 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).

3.1.4 Nutrients

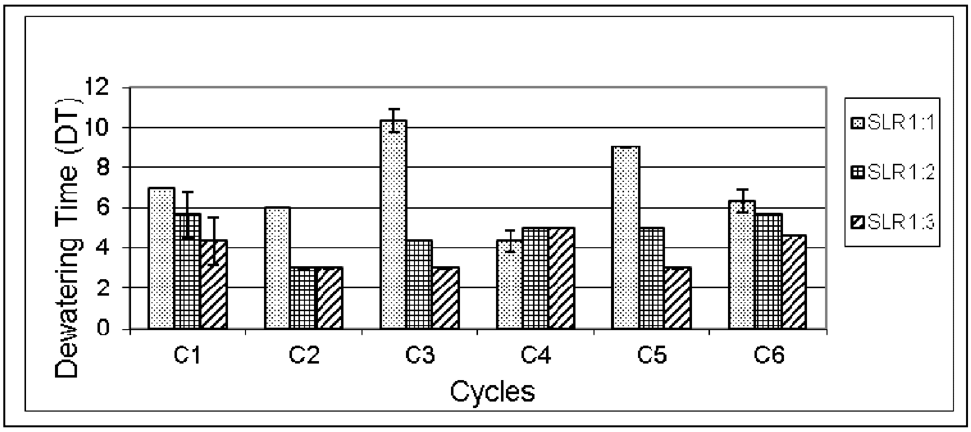
Values of NH_3-N and NO_3 (Table 1) did not differ much from the results of previous researches conducted in Accra (Larmie, 1997) and literature quoted (APHA-AWWA-WEF, 1998). The high values might result from the ammonification and mineralization of organic nitrogen (Epstein, 2003), which is a major constituent in faecal sludge. All the parameters discussed above decreased as the septage ratio in the mixture increased as shown in SL1, SL2 and SL3 respectively (Table 1). Ammonia concentrations ranging between 920 – 1310 mg/l as found in the raw sludges (Table 1) could hamper algal and bacterial growth. They also produce malodor as well as cause eye irritations at treatment plants as additional problems (McGinn, 2003)**.** TP values increased with increasing septage ratio (Table 1). High phosphorus content 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. Sludge with high contaminant loads, pose treatment challenges but has high potential of generating large quantities of biosolid that may be useful in agriculture.

3.2 Dewatering Time (DT) of the Different Filter Media in Different Cycles

On the average SL3, SLR2 and SLR1 achieved 4, 5 and 7 dewatering days respectively (Figure 2). The dewatering days 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 (Figure 2). These dewatering differences observed were mainly attributed to the different PTS and septage ratio. PTS from public latrines is generally known to be fresh and undigested, sometimes one week to few months old and does not lend itself to dewater easily (Montangero and Strauss, 2002). Septage from septic tanks is generally known to be fully or partially digested sometimes about one to ten years old and easy to dewater. Moreover, PTS which is fresh and undigested has high specific resistance due to high colloidal material thus resists withdrawal of water (Karr *et al.*, 1978). Therefore increase in PTS content decreases dewaterability while increase in septage content increases the dewaterability.

Moreover septage generally has lower TS strength as compared to PTS and therefore with the increase in septage content, the TS strength of the FS decreases thus favoring dewatering of sludge with the highest septage content, except few cases where the TS of the septage is higher than the PTS. The dewatering time for the different SLR did not follow any clear pattern from cycle one to cycle six (C1- C6). This is because the contaminant loads of the FS especially (TS) of each cycle differed. The performances of the different solid loading rates are clearly shown in (Figure 2). Between the period of January and May 2006, the average temperatures of the day were very high (31.5 – 33.3) ^{o}C and daily average relative humidity were comparatively low (58.5 – 61.5), (Meteorological Department of Ghana, 2006). Therefore the possible cause for the inconsistent 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.

Figure 2: Dewatering time (days) of different SLRs in six cycles



3.3 Removal Efficiencies of Filter Beds with Respect to Different Solid Loading Rates

3.3.1 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. TVS and SS removal also followed the same decreasing trend (Table 2). The levels of removal efficiencies achieved compare well with the results from pilot drying beds obtained by the Ghana Water Research Institute in Accra (Larmie, 1997). 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. 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 forms a thicker sludge cake on the surface of the filter medium. The sludge cake thus complements the filter medium in the removal of the solid particles of the sludge. However, considering the dewatering time and the various loading rates of raw sludge applied, SLR3 had the highest annual loading rate, 509 KgTS/m^2/year as compared to SLR2 and SLR3 at 502 and 425 kgTS/m^2/year respectively. The highest percentage removal of SS of 96.1% by SLR1 (Table 2) may be explained based on the fact that its high solids content due to high percentage of PTS in the mixture could lead to formation of thicker sludge cake on the surface of the filter medium. This is due to the high content of colloidal material in PTS. This thus increases the straining, compaction, and impaction abilities of the sludge cake and the filter medium.

Table 2: Percolate characteristics and percentage removal of different solid loading rates

Parameter	SLR1		SLR2		SLR3	
	Raw FS	percolate	Raw FS	percolate	Raw FS	percolate
TS	39.41±7.19	6.38±0.83	33.21±14.66	5.83±0.56	26.93±20.25	4.59±0.29
TVS	28.54±5.38	2.82±1.48	22.70±8.17	2.64±1.71	20.59±14.84	2.51±0.95
SS	21.32±8.62	0.87±0.93	18.62±11.66	1.14±1.58	16.27±15.53	0.78±0.56

± = *Standard deviation (SD)*

3.3.2 Reduction of Electrical Conductivity (EC)

The low percentage removal of electrical conductivity, 40.9%, 42.5% and 42.0 % by SLR1, SLR2 and SLR3 respectively indicate that the Solid Loading Rate does not impact much on the EC removal. This is because ionic mobility of dissolved salts increases EC. Thus increasing dissolved salts, lead to higher electrical conductivity (Kiely,1998). Therefore poor reduction of electrical conductivity is an indication of poor removal of salts. High magnitude of conductivity between 9.78 and 15.59 mS/cm in the percolates of SLR1, SLR2 and SLR3 (Table 3) will put osmotic stress on the growing plants and other organisms in the affected area when discharged into the environment. However the poor removal of the salts means little was retained in the biosolids which means a safe use of the biosolid component in agriculture.

3.3.3 Removal of Degradable Organics (COD, DCOD)

There were high percentage removals of chemical oxygen demand (COD) in all the different SLRs with SLR1, SLR2 and SLR3 showing as high as 86.4%, 84.5% and 80.1% average removal efficiencies respectively (Table 3). This efficiency achieved by the different solid loadings are comparable to the removal efficiency range of 70% – 90% in pilot drying bed research conducted in Accra by Water Research Institute (WRI), Accra. The removal efficiency might be due to the efficiency achieved in removing solids from the raw sludge and the availability of the dissolved organics for microbial action (Karim, 2005). The possibility of most solids being removed from SLR1 might have contributed to its highest performance. A good reduction in the DCOD is an indication of the availability of the dissolved substrate for bacterial action (Karim, 2005**)**. The removal efficiencies of 60.7%, 59.4% and 51.6% by SLR1, SLR2 and SLR3 respectively, well explained that the particles of the dissolved organics were degraded by microbes. The outstanding performance of SLR1 over the SLR2 and SLR3 may 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 too high for SLR1 than the others. Thus it could be reasoned that the longer time used for the dewatering might have assisted in improving the removal efficiency.

Table 3: Removal of salts and degradable organics

Parameter	SLR1		SLR2		SLR3	
	Raw FS	percolate	Raw FS	percolate	Raw FS	percolate
EC(mS/cm)	22.11±3.67	15.59±2.97	17.01±2.57	12.02±1.88	13.33±1.28	9.78±1.41
COD(g/l)	43.40±15.48	5.58±1.81	37.67±19.05	5.04±2.05	33.47±28.94	4.56±1.75
DCOD(g/l)	9.7±7.09	3.13±1.16	7.7±4.19	2.84±1.12	6.1±3.36	2.64±0.96

± = *Standard deviation (SD)*

3.3.4 Removal of Ammonia Nitrogen, Total Kjedhal Nitrogen, Nitrate-Nitrogen and Total Phosphorus (NH_3-N, TKN, NO_3-N and TP)

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 (Table 4). 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. 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 (Report by Asian Institute of Technology, 1998). Though 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 matter (Epstein, 2003). The higher nitrogen and organics removal efficiencies by the filter media from the different loading rates can be attributed to higher specific surface area of filter medium and higher biomass density of the FS (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, (Table 4). 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 (Table 4) is the result of organic nitrogen removal together with TVS and its subsequent mineralization.

The removal efficiencies of NO_3 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. The ultimate formation of NO_3 in the percolate may account for any losses through removal of TVS thus leading to high percolate concentration in the different solid loading rates. The percentage removals of phosphorus in all the mixing ratios, SLR1, SLR2 and SLR3 were 57.0%, 51.3% and 46.2% respectively (Table 4). These 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. However the ability of SLR1 to outperform the SLR2 and SLR3 with respect to removal of NO_3 and TP might be due to adsorption 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.

Table 4: Removal of nitrogen and nutrients

Parameter	SLR1		SLR2		SLR3	
	Raw FS	percolate	Raw FS	Percolate	Raw FS	percolate
NH_3-N(g/l)	1.31±0.94	0.38±0.27	1.06±0.78	0.34±0.20	0.92±0.73	0.30±0.18
TKN(g/l)	1.76±0.94	0.48±0.32	1.38±0.86	0.41±0.23	1.21±0.87	0.37±0.21
NO_3(g/l)	1.06±0.47	0.55±0.15	1.04±0.57	0.58±0.25	1.19±0.84	0.53±0.32
TP(g/l)	3.38±1.40	1.69±0.96	3.33±1.25	1.94±1.10	4.01±2.12	2.19±1.12

± = *Standard deviation (SD)*

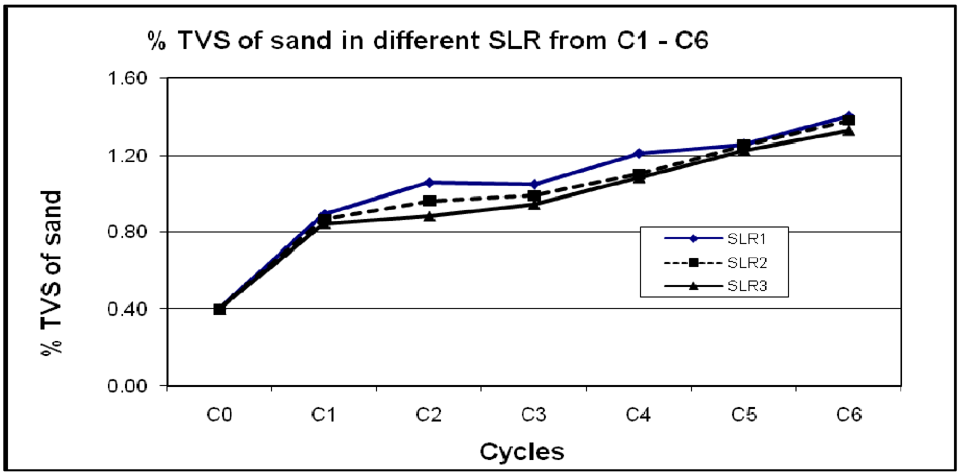
3.4 Faecal Sludge (FS) Quantities Dewatered

Considering the average TS of raw sludge of the different SLR (Table 1), 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 obtained for SLR1, SLR2 and SLR3 respectively. This means that more sludge could be dewatered by applying SLR3.

3.5 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 in the filter medium because TVS is a function of organic matter. This means that clogging of filter beds is possible in all the mixing ratios. However, the graph shows that organic matter build up in the filter medium for SLR1 gradually outpaced the others as the cycles advanced to cycle six (Figure 3). This was due to the higher content of dissolved organics in SLR1 based on the PTS content in the mixture. According to literature, 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 *et al.,* 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 dewatering cycles advance.

Figure 3: Percentage TVS in filter media of different SLR from cycles 1 – 6



3.6 Production of Biosolids

The percentage accumulation of the biosolids on the filter media from the different solid loading rates, SLR1, SLR2 and SLR3 is an indication 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, (Table 5). 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, (Table 5). These performances exhibited 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 (Table 5), 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 5). This means that in dewatering where biosolid production is paramount, SLR3 is best for application.

Table 5: Characteristics of accumulated biosolids

Parameter	SLR1	SLR2	SLR3
Total dry solid of FS (g)	197.0	166.1	134.6
Total org. matter of FS (g)	142.7	113.5	103.0
Ave. dry biosolid (bio) 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
Days in a cycle	7	5	4
Cycles/year (yr)	52	73	91
Ave dry bio/yr.(g/yr)	9103	10138	10453
TVS/filter bed/yr (g/yr).	5365	5467	6151
kg/yr.(TVS)	5.37	5.47	6.15
Area (m^2) of filter bed	0.024	0.024	0.024
kgTVS/m^2yr	224	228	256

Ave = average

kgTVS/m^2yr = kilogram total volatile solids per metre squared per year

4.0 Conclusion

All the solid loading configurations were found to be able to improve the dewatering performance of the unplanted filter bed since their average dewatering times improved over the results of previous experiments. However, the SLR3 had the shortest dewatering time but in terms of contaminant load removal, the SLR1 was most efficient. Generally there were significant differences in the dewatering times between the solid loading rates applied but they were statistically not different with respect to contaminant load removal. SLR3 had the added advantage of generating the highest volume of biosolid annually. Furthermore, the SLR3 was least likely to clog its filter medium. The solid loading rate variation was not very efficient in salts removal since the different variations showed low and inconsistent EC reduction. Dissolved salts passed through the filters. The pH and temperature were favorable 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 (US EPA, 1984). Although high removal efficiencies were achieved for all the different loading rates, the parameters in the effluent needed to be treated further either through column filters, constructed wetlands or stabilization ponds for more polishing before final discharge. Addition of physical conditioners like wood chips or rice husk before filtration is likely to increase the bulking characteristics of the sludge thereby further improving filtration.

Acknowledgements

This study was made possible by financial support from EAWAG/SANDEC. The study contributes to the SANDEC programme of low-cost options for managing FS in Developing countries to improve sanitation, health and agriculture. The authors would like to express our sincere apreciation to IWMI, for offering us space at the Kumasi office in Ghana for our office work.

References

APHA-AWWA-WEF. Standard Methods for the Examination of Water and Wastewater, 20^{th} Edition. Washington D. C. 1998.

Asian Institute of Technology (AIT). Septage treatment in constructed wetlands and attached- growth stabilisation ponds. Interim Report, AIT/EAWAG Field Research Project. Environmental Engineering Program, February, 1998., Unpublished.

Awuah E. Pathogen removal in macrophyte and agal waste stabilisation ponds. PhD dessertation submitted the Academic Board of Wageningen University and the Academic Board of the UNESCO-IHE Institute for water education on 29th November 2006 in Delft . The Netherlands. 2006, Pp 106.

Drangert J. O. Fighting the Urine Blindness to Provide More Sanitation Options. *Water South Africa*, 1998, Vol. 24, No. 2, April.

Epstein Eliot. Land application of sewage sludge and biosolids, Lewis Publishers , CRC press company, London, 2003, pp 30-34.

Esrey, S.A., Gough, J., Rapaport, D., Sawyer, T., Simpson-Hébert, M., Vargas, J., Winblad, U. . *Ecological Sanitation*. SIDA, Swedish International Development Cooperation Agency, Stockholm ,1998**.**

Heinss, U., Larmie, S.A., Strauss, M. Solids Separation and Pond Systems for the Treatment of Septage and Public Toilet Sludges in Tropical Climate - Lessons Learnt and Recommendations for Preliminary Design. EAWAG/SANDEC Report No. 05/98. Duebendorf, Switzerland. 1998.

Karim K. Hoffmann R. Klasson T. and Al-Dahhan M. H. Anaerobic Digestion of Animal waste: Waste strength versus impact of mixing. *Bioresource Technology*, 2005, 96, 1771 – 1781.

Karr P. R., Keinath, T. M. Influence of Particle Size on Sludge Dewaterability. 1978, Pp 1911 – 1913.

Kiely G. *Environmental Engineering*. International Ed. Irwin McGraw-Hill, Malasia,1998, pp 64 – 73.

Kuffour A.R., Awuah E., Anyemedu F. O. K., Strauss M. Koné D., and Coffie O. Effect of using different particle sizes of sand as filter media for dewatering faecal sludge. *Desalination Land*. ASA-CSSA-SSSA, Madison, 2009 pp. 695–703.

Larmie, S. A. Achimota Pond Performance Study. Project report, Water Research Institute (CSIR), Accra, Ghana. 1997. Unpublished document.

Larsen, T. and Guyer, W. Separate Management of Anthropogenic Nutrient Solutions (human urine). *Water Science & Technology*, 1996**,** 34 (3-4), 87-94.

McGinn S. M., Janzen H. H. and Coates T. Atmospheric Ammonia, Volatile Fatty Acids, and Other Odorants near Beef Feedlots*, Journal of Environmental Quality*, 2003**,** 32, 1173-1182.

Metcalf, L. and Eddy, H. P. Wastewater Engineering – Treatment, Disposal and Reuse, 3rd Ed., McGraw-Hill, New York, 1995.

Meteorological Department report, Ashanti region, Ghana (2006). Unpublished data.

Montangero, A. Strauss, M.. Faecal sludge treatment. A report of Swiss Federal institute for Environmental Science and Technology, IHE Delft. 2002 pp 9 – 10.

Nikhla G. and Farooq Shaukat. Simultaneous Nitrification-denitrification in Slow Sand Filters. *Journal of Harzardous Materials*, 2003**,** B96, 291-303.

Strauss M, Larmie S. A. and Heinss U. Treatment of Sludges from On-Site Sanitation: Low-cost Options. *Water Science and Technology*, 1997**,** 35-36.

US EPA, Handbook; Septage Treatment and Disposal EPA 625/6-84-009, US Environmental Protection Agency, Centre for Environmental Research Information, Cincinnati, Ohio, 1984.

Veenstra, S. and Polprasert C. Wastewater treatment – Part 1, Lecture notes IHE, Delft, 1997, (EE01/97/1)

Winblad, U. Towards an Ecological Approach to Sanitation. Publications on Water Resources No. 5. Department for Natural Resources and the Environment, Swedish International Development Authority (SIDA), Birger Jarlsgatan 1997 61, S-10525 Stockholm, Sweden.