Co-composting of Faecal Sludge and Solid Waste

Preliminary Recommendations on Design and Operation of Co-composting Plants based on the Kumasi Pilot Investigation

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IWMI&SANDEC

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Pilot Co-composting Plant Investigations in Kumasi
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Foreword

The present report was produced as part of the project „Co-composting of Faecal Sludge and Organic Solid Waste in Kumasi, Ghana“ that was conducted within the programme „Sustainable Solid Waste Management and Sanitation“ managed by the pS-Eau/PDM and financed by the French Ministry of Foreign Affairs.

The pilot project is co-ordinated by the International Water Management Institute (IWMI) in collaboration with the University of Science and Technology in Kumasi, the Waste Management Department (Kumasi Metropolitan Assembly) and SANDEC. Results of the investigation will help the WMD (Waste Management Department) develop its biosolids management strategy and enable the project team to develop guidelines for planners and engineers on the option of co-composting.

The project aimed at studying different aspects of co-composting faecal sludge and soild waste in the context of Kumasi:

- Technical and operational aspects
- Farmers's perception, willingness and ability to pay
- Marketing, market development and distribution pathways
- Economic aspects
- Institutional framework
- Environmental impact
- Socio-cultural aspects

This report summarises only the technical and operational aspects of the co-composting process (design and mode of operation) that were investigated at a pilot plant in Kumasi between February and June 2002. Other aspects, like farmers‘ perception with regard to compost or willingness to pay for compost are described in the project final report. The aim of this report is to have recommendations at hand for planners and engineers in developing countries who are in the process of evaluating different treatment options for faecal sludge and organic solid waste or planning a co-composting plant. Only few results were obtained so far. They are not statistically significant. Recommendations contained in this report must be therefore considered as preliminary recommendations. The monitoring of the pilot plant will continue until the end of the year and through 2003 if financial support can be found. This report will be updated with the results of the on-going monitoring.

This report consists of two parts. A literature review on human waste reuse, health aspects in particular, faecal sludge treatment, composting and co-composting (part A). Preliminary recommendations with regard to the design and mode of operation of a co-composting plant based on the results obtained so far at the Kumasi pilot co-composting plant (part B).

Abbreviations and Glossary

Abbreviations

BOD	Biochemical Oxygen Demand
COD	Chemical Oxygen Demand
FC	Faecal Coliforms
FS	Faecal Sludge
NH_4-N	Ammonium Nitrogen
NH_3-N	Ammonia Nitrogen
SS	Suspended Solids
TKN	Total Kjeldahl Nitrogen
TOC	Total Organic Carbon
TS	Total Solids
TVS	Total Volatile Solids
WSP	Waste Stabilisation Ponds
WWTP	Wastewater Treatment Plant

Glossary

Faecal sludge	Sludges of variable consistency collected from so-called on-site sanitation systems; viz. latrines, non-sewered public toilets, septic tanks, and aqua privies
Septage	Contents of septic tanks (usually comprising settled and floating solids as well as the liquid portion)
Public toilet sludge	Sludges collected from unsewered public toilets (usually of higher consistency than septage and biochemically less stabilised)
Percolate	The liquid seeping through a sludge drying bed and collected in the underdrain

Part A

Literature and State-of-Knowledge Review on Co-Composting

1. Reuse of excreta and municipal organic waste

1.1 General practices of excreta and solid waste use

All around the world, people both in rural and urban areas have been using **human excreta** for centuries to fertilise fields and fishponds and to maintain or replenish the soil organic fraction, i.e. the humus layer. Until today, in both agriculture and aquaculture this continues to be common in China and Southeast Asia as well as in various places in Africa (Cross 1985; Timmer and Visker 1998; Visker 1998; Timmer 1999; Strauss et al. 2000). Use practices 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). Chinese peri-urban vegetable farmers have reported that customers prefer excreta-fertilised rather than chemically fertilised vegetables. Thus vegetables grown on excreta-conditioned soils yield higher sales prices.

Like excreta, the use of **organic solid waste** has a long history mainly in rural areas. Traditional reuse practices of organic solid waste are shown to be especially strong in countries where population densities are high. With the growth of urban areas, the importance of managing municipal solid wastes to avoid environmental degradation and public health risks has gained significance. Although informal recycling activities of waste materials is wide spread in developing countries the treatment and use of the biodegradable organic fraction is still fairly limited. Increasingly, national and municipal authorities are now looking at ways to manage their organic solid waste. In India national legislation was adopted with the "Municipal Solid Waste (Management & Handling) Rules 2000" (Ministry of Environment and Forests 2000) whereby one section of the rules requires Urban Local Bodies to promote and implement waste segregation at source and treat organic waste.

1.2 The resource potential of human excreta and municipal solid waste

1.2.1 Excreta

Excreta are a rich source of organic matter and of inorganic plant nutrients such as nitrogen, phosphorus and potassium. Each day, humans excrete in the order of 30 g of carbon (90 g of organic matter), 10-12 g of nitrogen, 2 g of phosphorus and 3 g of potassium. 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. Table 1 shows that the fertilising equivalent of excreta is, in theory at least, nearly sufficient for a person to grow its own food (Drangert 1998). In a recent material flow study conducted in the City of Kumasi, Ghana, 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 (most of the wastes would have to be treated prior to use, though) (Leitzinger 2000; Belevi et al. 2000).

Excreta are not only a fertiliser. 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

Table 1 The Fertilization Equivalent of Human Excreta (after Drangert 1998)

Nutrient	Nutrient in kg / cap year			
	In urine (500 l/year)	In faeces (50 l/year)	Total	Required for 250 kg of cereals [1]
Nitrogen (as N)	4.0	0.5	4.5	5.6
Phosphorus (as P)	0.4	0.2	0.6	0.7
Potassium (as K)	0.9	0.3	1.2	1.2
Carbon (as C) [2]	2.9	8.8	11.7	

[1] = the yearly food equivalent required for one person
[2] = indicative of the potential for soil conditioning, normally not designated a nutrient

New approaches in human waste management postulate that sanitation systems should, whenever feasible, be conceived and managed that again enable 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.

1.2.2 Municipal organic solid waste

The resource potential of mixed municipal solid waste is more variable than for excreta as it depends on the waste composition, which varies considerably from city to city and also among city districts depending on income levels and consumer habits. Low-income countries generate significantly less waste than high- income countries. Cointreau (1985) estimates average municipal solid waste generation (mixed) between 0.4 - 0.6 kg per capita per day in low-income countries, compared to 0.7 – 1.8 kg/cap and day in high-income countries. Typically in low-income countries the biodegradable fraction is significantly higher (40-85 %) than in high-income countries (20-50 %) where municipal waste consists mainly of packaging materials (paper and plastics). Assuming a daily per-capita solid waste generation of 0.5 kg with a 60 % biodegradable fraction, 300 g/cap.day wet organic waste is being generated. Based on an assumption of 50 % water content of this organic fraction, this is equivalent to 150 grams dry organic solids/cap and day. Based on contents on a dry weight basis of 30-40 % carbon (C), 1-2 % nitrogen (N) and 0.4-0.8 % phosphorus (as P), and 1 % potassium (as K), the per-capita nutrient and carbon contributions from the organic fraction of MSW is as indicated in Table 2. The table shows that municipal organic solid waste although low in nutrients is particularly rich in organic matter can be thus be valued on its soil conditioning potential.

Table 2 The fertilization equivalent of municipal solid waste (org. fraction) before waste treatment

Nutrient	Contribution in kg / cap•year
Nitrogen (as N)	0.55 – 1.1
Phosphorus (as P)	0.2 – 0.4
Potassium (as K)	0.55
Carbon (as C) [1]	16 – 22

[1] = indicative of the potential for soil conditioning, normally not designated as a nutrient

1.3 Health consideration in re-use of human waste and solid waste

In developing countries, excreta-related diseases are very common, and faecal sludges contain correspondingly high concentrations of excreted pathogens - the bacteria, viruses, protozoa, and the helminths (worms) that cause gastro-intestinal infections (GI) in man. The actual risks to public health that occur through waste use can be divided into three broad categories - those affecting consumers of the crops grown with the waste (**consumer risk**), those affecting the agricultural workers who are exposed to the waste (**workers', farmers' risk**), and those affecting populations living near to a waste reuse scheme (**nearby population risk)**

1.3.1 Health risks related to excreted pathogens

The agricultural use of excreta or excreta-derived products such as stored or dewatered faecal sludge or co-compost can only result in an actual risk to public health if all of the following occur (WHO 1989):

(a) That either an infective dose of an excreted pathogen reaches the field or pond, or the pathogen (as in the case of schistosomiasis) multiplies in the field or pond to form an infective dose;

(b) That this infective dose reaches a human host;

(c) That this host becomes infected; and

(d) That this infection causes disease or further transmission.

(a), (b) and (c) constitute the **potential risk** and (d) the **actual risk** to public health. If (d) does not occur, the risks to public health remain potential only.

Die-off or survival of excreted pathogens is an important factor influencing transmission. In principle, all pathogens die off upon excretion. Prominent exceptions are pathogens whose intermediate stages multiply in intermediate hosts as the miracidia of e.g. *Clonorchis* or *Schistosoma* which multiply in aquatic snails and are later released into the water body. Some bacteria (Salmonellae, Shigellae and Campylobacter, e.g., have the potential to multiply outside the host primarily on food and at warm temperature. The pathogens have varying resistance against die-off, and worm eggs are among the more resistant with *Ascaris* eggs surviving longest in the extra-intestinal environment. The main factors influencing die-off are temperature, dryness and UV-light. Table 3 lists survival periods at ambient temperature in faecal sludges for temperate and tropical climates. Another important factor is the **infective dose** of a pathogen. It is the dose required to create disease in a human host. For helminths, protozoa (e.g. amoeba) and viruses, the infective dose is low (< 10^2). For bacteria, it is medium (< 10^4) to high (> 10^6).

Table 3 Pathogen Survival Periods in Faecal Sludge (after Feachem et al. 1983, Strauss 1985 and Schwartzbrod J. and L. 1994)

	Average Survival Time in Wet Faecal Sludge at Ambient Temperature[1]	
Organism	**In temperate climate (10-15 °C) [days]**	**In tropical climate (20-30 °C) [days]**
• **Viruses**	**< 100**	**< 20**
• **Bacteria:**		
-Salmonellae	**< 100**	**< 30**
-Cholera	**< 30**	**< 5**
-Faecal coliforms [2]	**< 150**	**< 50**
• **Protozoa:**		
-Amoebic cysts	**< 30**	**< 15**
• **Helminths:**		
-Ascaris eggs	**2-3 years**	**10-12 months**
-Tapeworm eggs	**12 months**	**6 months**

1 Conservative upper boundaries to achieve 100 % die-off; survival periods are shorter if the faecal material is exposed to the drying sun, hence, to desiccation

2 Faecal coliforms are commensal bacteria of the human intestines and used as indicator organisms for excreted pathogens

Scott in China conducted investigations on microbial risks from human waste use in relation to the use of human excreta in agriculture as early as the 1930-ies (Scott 1952). Rudolfs et al. (1950 and 1951) conducted later major assessments of microbial contamination of soils and plants using wastewater and sewage sludge in the U.S.. Akin et al. (1978) reported about continued work in this field done in the United States. A thorough, basic compendium on the relationships between health, excreted infections and measures in environmental sanitation has been published by Feachem et al. (1983). Strauss (1985) published a review on the survival of excreted pathogens on soils and crops –a factor of great relevance for the risk or non-risk of human waste use –. WHO, UNDP, the World Bank, in collaboration with other multi and bilateral support agencies commissioned reviews of epidemiological literature related to the health effects of excreta and wastewater use in agriculture and aquaculture in the early eighties. The results are documented in Shuval et al. (1986) and in Blum and Feachem (1985). This, in combination with the systematised assessment of gastro-intestinal infections by Feachem, aimed at developing a rational basis for the formulation by WHO of updated health guidelines in wastewater reuse (see WHO 1989).

Birley and Lock (1997) and Allison et al. (1998) have highlighted health impacts and risks of solid and human waste use in urban agriculture. While touching upon the water and excreta-related diseases, they also focused on health risks to farmers and consumers from chemical contamination of soils, occupational risks from poisoning through herbicides and pesticides and from physical injury mainly when solid wastes are recycled to agriculture. The non-pathogen related risks are discussed further below.

The epidemiological evidence on the *agricultural use of excreta* can be stated as follows (Blum and Feachem 1985):

- Crop fertilisation with **untreated** excreta causes significant excess infection with intestinal nematodes in both consumers and field workers

- Excreta treatment, e.g. through **thermophilic composting, extended storage and/or drying**, significantly reduces or eliminates the risk of transmission of gastro-intestinal infections.

Pathogen die-off or inactivation during composting is dealt with in Chapter 0

Ascaris eggs, being the most persistent of all pathogens, can be used as a hygienic indicators of treated excreta. For sludge or biosolids, Xanthoulis and Strauss (1991) proposed a nematode egg standard of ≤ 3-8 eggs/gram of dry solids. This value is based on the 1989 WHO nematode guideline of ≤ 1 egg/litre of wastewater for unrestricted irrigation.

In **municipal solid waste**, the health risk by pathogens is determined by the amount of faecal matter contained in the solid waste or by pathogenic hospital and clinical waste, which may enter the municipal solid waste stream unintentionally. Non-pathogen risks can be more significant depending on the waste composition and the way the waste is managed (or not managed).

1.3.2 Non-Pathogenic Health Risks

Chemical contamination is a potential risk associated with waste use, notably in municipal solid waste. As organic solid waste is often stored and collected together with other waste fractions, contamination of the organic fraction is easily possible by chemical constituents, heavy metals in particular. When applying the contaminated compost product, these constituents can accumulate in soils. The contamination of soils by chemicals, the potential but as yet uncertain uptake by crops, and the possible chronic and long-term toxic effects in humans are discussed by Chang et al. (1995) and by Birley and Lock (1997).

Further non-pathogen risks result from impurities of non-biodegradable origin such as glass splinters or other sharp objects contained in the compost product. Such impurities can result from insufficiently sorted municipal solid waste before or after the composting process. Birley and Lock (1999) have highlighted these risks also including indirect health risks due to the attraction and proliferation of rodents and other disease carrying vectors.

2. Faecal sludge treatment

2.1 Relevant FS characteristics and quantities

Table 4 contains the **daily per capita volumes** and **loads of organic matter, solids and nutrients** in faecal sludges collected from septic tanks and pit latrines, as well as from low or zero-flush, unsewered public toilets. Values for fresh excreta are given for comparative purposes. The figures are overall averages, actual quantities may, however, vary from place to place.

Table 4 Daily per capita volumes; BOD, TS, and TKN quantities of different types of faecal sludges (Heinss et al. 1998)

Parameter	Septage [1]	Public toilet sludge [1]	Pit latrine sludge [2]	Fresh excreta
• BOD g/cap·day	1	16	8	45
• TS g/cap·day	14	100	90	110
• TKN g/cap·day	0.8	8	5	10
• Volume l/cap·day	1	2 (includes water for toilet cleansing)	0.15 - 0-20	1.5 (faeces and urine)

1 Estimates are based on a faecal sludge collection survey conducted in Accra, Ghana.

2 Figures have been estimated on an assumed decomposition process occurring in pit latrines. According to the frequently observed practice, only the top portions of pit latrines (~ 0.7 ... 1 m) are presumed to be removed by the suction tankers since the lower portions have often solidified to an extent which does not allow vacuum emptying. Hence, both per capita volumes and characteristics will range higher than in the material which has undergone more extensive decomposition.

2.1.1 FS characteristics

In contrast to sludges from WWTP and to municipal wastewater, characteristics of faecal sludge differ widely by locality (from household to household; from city district to city district; from city to city) (Montangero and Strauss 2002).

A basic distinction can usually be made between fresh, biochemically **unstable** and “thick” vs. “thin” and biochemically fairly **stable** sludges (Heinss et al. 1998). Unstable sludges contain a relative large share of recently deposited excreta. Stable sludges are those, which have been retained in on-plot pits or vaults for months or years and which have undergone a biochemical degradation to a variable degree (e.g. septage, which is sludge from septic tanks).

Based on numerous FS monitoring studies in West Africa, Rosario (Argentina), Bangkok and Manila, the authors found that FS can often be associated with one of these two distinct categories. In contrast to fairly stable sludges, fresh undigested and biochemically unstable sludges exhibit poor solids-liquid separability.

Table 5 shows typical FS characteristics and typical characteristics of municipal wastewater as may be encountered in tropical countries. Storage duration, ambient temperature, intrusion of groundwater into vaults or pits of on-site sanitation installations; installations sizing, and tank emptying technology and pattern are important factors influencing the sludge quality.

Table 5 Faecal sludges from on-site sanitation systems in tropical countries: characteristics, classification and comparison with tropical sewage (after Strauss et al. 1997* and Mara 1978)**

Item	Type "A" (high-strength) *	Type "B" (low-strength) *	Sewage ** (for comparison purposes)
Example	Public toilet or bucket latrine sludge	Septage	Tropical sewage
Characteri-sation	Highly concentrated, mostly fresh FS; stored for days or weeks only	FS of low concentration; usually stored for several years; more stabilised than Type "A"	
COD mg/l	20, - 50,000	< 15,000	500 - 2,500
COD/BOD	5 : 1 .... 10 : 1		2 : 1
NH_4-N mg/l	2, - 5,000	< 1,000	30 - 70
TS mg/l	≥ 3.5 %	< 3 %	< 1 %
SS mg/l	≥ 30,000	≅ 7,000	200 - 700
Helm. eggs no./l	20, - 60,000	≅ 4,000	300 - 2,000

2.2 Faecal sludge treatment options

2.2.1 Treatment goals

Faecal sludge should be treated to render the treatment products (biosolids and effluent liquids) apt for discharge into the environment (including landfilling), or to produce biosolids, which may be safely used in agriculture.

In the majority of developing countries, no standards or guidelines have been set for the quality of biosolids. Standards have usually been copied from industrialised countries without taking the specific conditions prevailing in the particular developing country into account. In most if not all cases, the standards were enacted having wastewater treatment and discharge in mind. Quite commonly, in such cases, standards or the performance of infrastructure works are neither controlled nor enforced. Faecal sludges and products from their treatment were not or still not taken into special consideration nowadays, thus, applying the standards set for wastewater treatment plant effluents. In most cases, these standards are too strict to be attained even for wastewater treatment schemes under the local conditions. For FSTP, the enacted effluent standards would call for the use of sophisticated and highly capital-intensive treatment, which is unrealistic. A suitable strategy would consist in selecting a phased approach, under the paradigm that "something" (e.g. 75 % instead of 95-99 % helminth egg or COD 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).

The EU has adopted a rational strategy for public health protection in biosolids use. 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[1]. Sludge treatment options, which were found to effectively inactivate excreted pathogens to desirable levels (e.g. co-composting), are typical "barrier points", where the transmission of

1 The principle follows the "HACCP" principle, which stands for Hazard Analysis and Critical control Points. It was first developed in the U.S.A. for food safety in manned space systems

pathogens might be stopped (Matthews 2000). In Table 6, a set of effluent and plant sludge quality guidelines for selected constituents is listed. The suggested values are based on the based on the principle of defining and setting up barriers against disease transmission, which can be used as critical control points for securing safe biosolids quality. Xanthoulis and Strauss (1991) proposed a guideline value for biosolids (as produced in faecal sludge or in wastewater treatment schemes) of 3-8 viable nem. eggs/ g TS. This recommendation is based on the WHO guideline of ≤1 nematode egg/litre of treated wastewater used for vegetable irrigation (WHO, 1989), and on an average manuring rate of 2-3 tons TS/ha·year. It was used to estimate the allowable yearly deposition of eggs, based on an assumed yearly rate of irrigation (500-1,000 mm).

Examples for faecal sludge treatment standards are known from China and Ghana.

Table 6 Suggested effluent and biosolids quality guidelines for the treatment of faecal sludges (Heinss et al., 1998)

	BOD [mg/l] total	BOD [mg/l] filtered	NH_4-N [mg/l]	Helminth eggs [no./L]	FC [no./100 mL]
A: Liquid effluent *1. Discharge into receiving waters:*					
• Seasonal stream or estuary	100-200	30-60	10-30	≤ 2-5	$\leq 10^4$
• Perennial river or sea	200-300	60-90	20-50	≤ 10	$\leq 10^5$
2. Reuse:					
• Restricted irrigation	n.c.		1)	≤ 1	$\leq 10^5$
• Unrestricted irrigation	n.c.		1)	≤ 1	$\leq 10^3$
B: Treated plant sludge					
• Use in agriculture	n.c.		n.c.	≤ 3-8/ g TS 2)	3)

1) ≤ Crop's nitrogen requirement (100 - 200 kg N/ha·year)
2) Based on the nematode egg load per unit surface area derived from the WHO guideline for wastewater irrigation (WHO, 1989) and on a manuring rate of 2-3 tons of dry matter /ha·year (Xanthoulis and Strauss, 1991)
3) Safe level if egg standard is met

n.c. – not critical

2.2.2 Treatment options overview

Figure 1 provides an overview of options for faecal sludge treatment, which can be implemented by using modest to low-cost technology. They may therefore be considered as particularly sustainable for use in developing countries. They comprise:

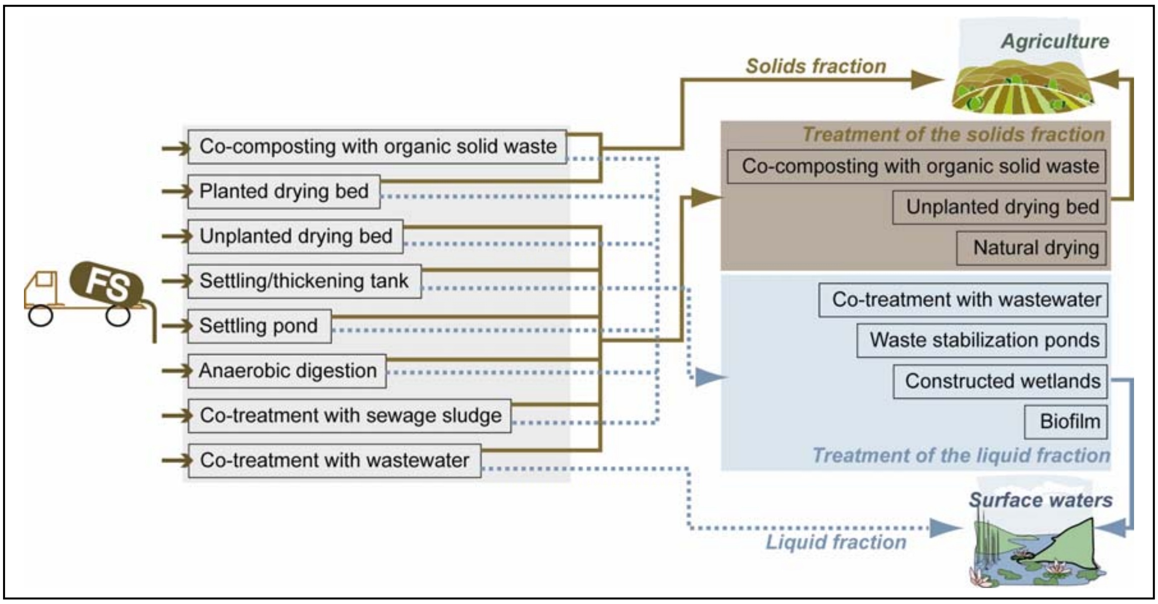

Figure 1 Overview of potential, modest-cost options for faecal sludge treatment

Some of these options were and are currently being investigated upon by EAWAG/SANDEC and its partners in Argentina, Ghana, Thailand, and The Philippines. Information can be retrieved from SANDEC's homepage[2].

The fact that faecal sludges exhibit widely varying characteristics calls for a careful selection of appropriate treatment options, notably for primary treatment. For primary treatment the separating of the solids and liquids, which make up FS, is the process-of-choice unless it is decided to co-treat FS in an existing or planned WWTP.

2.3 Solids-liquid separation and dewatering of FS

If FS is still rather fresh it has to be biochemically stabilised first for solids and liquids to become separable. Anaerobic ponds, designed to also cater for separated solids accumulation, may serve the combined purpose of stabilisation and solids-liquid separation. Solids-liquid separation of FS, which has undergone considerable biochemical stabilisation (septage), may be achieved through sedimentation and thickening in ponds or in tanks, or through filtration and drying in sludge drying beds.

Resulting from this are solids and a liquid fraction. The solids fraction, which may be designated as “biosolids" may require additional dewatering/drying to achieve spadability and to meet hygiene requirements for reuse in agriculture as a soil-conditioner and fertilizer. Table 3 lists pathogen die-off periods in faecal matter at ambient temperatures. It may be referred to for estimating additional storage periods required to render biosolids apt for use. Additional dewatering/drying might be required also for landfilling.

Additional treatment might be also necessary for the liquid fraction, to satisfy criteria for discharge into surface waters and/or to avoid long-term impacts on groundwater quality. Reference is made to the literature available on options such as waste stabilization ponds (WSP), upflow anaerobic sludge blanket clarifiers (UASB), or constructed wetlands (CW). Reuse of liquids emanating from separation processes

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2 http://www.sandec.ch/sos/references.html

can not be used for irrigation, as their salt contents exceed the salt tolerance limits of cultured plants (□3 mS/cm = 3 dS/m; FAO 1985).

2.3.1 Settling/thickening of FS

Faecal sludges typically exhibit total solids (TS) and suspended solids (SS) contents, which are very high compared with 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; as an option to produce biosolids conducive to agricultural use, and when intending the joint composting of FS solids and solid organic wastes.

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, S.A., 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.

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 that separation is complete in 30-60 minutes and that SS concentrations in the supernatant of 400 mg/l are achieved (Koottatep, 2001; Kost and Marty, 2000).

Field studies, conducted at the Achimota Faecal Sludge treatment plant in Accra/Ghana from 1993-97 reveal that the performance of the sedimentation tanks strongly depends on the plant's state of maintenance and operation. For the existing twin settling/thickening tanks, the loading and resting periods should not exceed 4 to 5 weeks each. In practice however, the tanks are emptied every 4 to 5 months. Process disturbance by improper design and operation for solids separation systems has been repeatedly observed (Hasler, 1995; Mara et al., 1992).

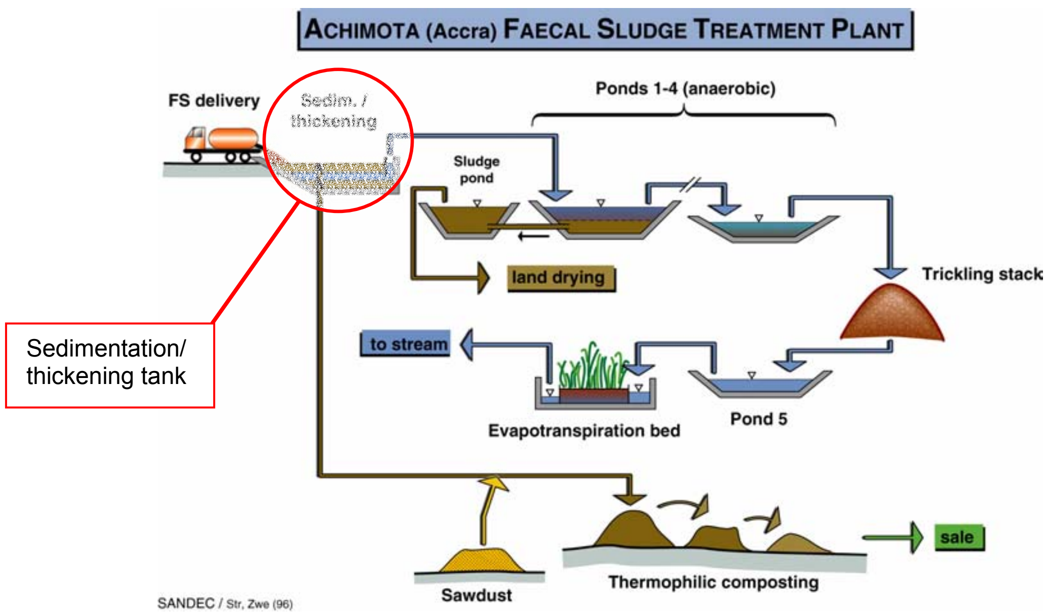

Figure 2 Scheme of the Achimota Faecal Sludge Treatment Plant

In septage settling ponds of the Alcorta (Argentina) pond scheme, TS in the settled solids amounts to about 18% after 6 months of septage loading (Ingallinella et al., 2000). Septage collected in Alcorta exhibits an SS content of approx. 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.

2.3.2 Sludge drying beds

Sludge drying beds serve to effectively separate solids from liquids and to yield a solids concentrate. Gravity **percolation** and **evaporation** are the two processes responsible for sludge dewatering and drying. In planted beds, **evapotranspiration** provides an additional effect. Unplanted and planted sludge drying beds are schematically illustrated in Figure 3.

In contrast to settling and thickening of FS, dewatering and drying of thin layers of sludge on sludge drying beds calls for comparatively long retention periods. However, organic and solids loads in the percolate of drying beds are significantly lower than in the effluent of sedimentation/thickening tanks. Hence, less extensive further treatment of percolate is required.

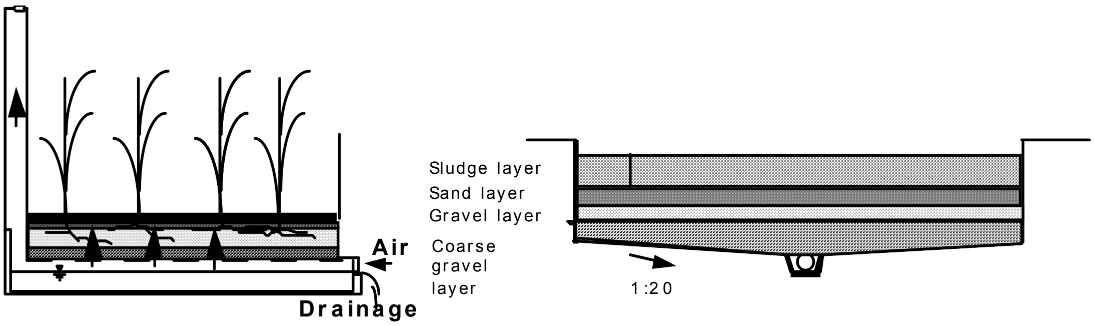

Figure 3 Planted and unplanted sludge drying beds (schematic)

From 50 - 80 % of the faecal sludge volume applied to unplanted drying beds will emerge as **drained liquid** (percolate). In planted drying beds, this ratio is likely to be much lower. Pescod (1971) conducted experiments with unplanted sludge drying beds in Bangkok, Thailand. According to the experiments, maximum allowable solids loading rates can be achieved with a sludge application depth of 20 cm. To attain a 25 % solids content, drying periods of 5 to 15 days are required depending on the different bed loading rates applied (70 - 475 kg $TS/m^2 \cdot yr$).

Results from pilot sludge drying beds (Figure 4) obtained by the Ghana Water Research Institute (WRI) in Accra indicate their suitability for septage/public toilet sludge mixtures and primary pond sludge (TS = 1.6-7 %). Experiments were conducted during the dry season with sludge application depths of $\leq$ 20 cm.

Figure 4 Pilot sludge drying beds in Accra, Ghana

Results from pilot sludge drying beds obtained by the Ghana Water Research Institute show a good applicability of sludge drying beds for septage/public toilet sludge mixtures (with p. toilet sludge shares not exceeding 30 %) and for primary pond sludge

Sludge, dewatered to ≤ 40 % TS in the Accra/Ghana experiments, still exhibited considerable **helminth egg** concentrations. This is not surprising as the drying periods amounted to 12 days at the most. To guarantee a hygienically safe product for use in agriculture, further controlled sludge drying experiments should be conducted to determine safe drying periods and required sludge dryness.

The various types of sludges revealed the following drying behaviour over a period of 8 days:

- Mixtures of public toilet sludge (unstable) and septage (stable) at a 1:4 ratio:

  Good dewaterability, drying to max. 70 % TS in eight days

- Primary pond sludge:

  Rather good dewaterability, drying to 40 % TS

- Public toilet sludge (unstable):

  Erratic results, from almost no dewaterability to 29 % TS.

2.4 Hygienic quality of biosolids

The residual concentration of helminth eggs in the biosolids is dependent on the prevalence and intensity of infection in the population from which FS or wastewater is collected and on various factors influencing parasite survival. Where biosolids use in agriculture is a practice or being aimed at, treatment or storage must be designed at reducing helminth egg counts and viability to acceptable levels. Table 3 may serve to estimate pathogen (including helminth egg) die-off in faecal sludge during storage at ambient temperature. Figure 5 allows to estimate the time required for *Ascaris* egg die-off in properly operated, thermophilic compost. Table 7 shows values for helminth egg counts and viability in untreated human wastes and in biosolids as reported in published and unpublished literature for a few selected wastewater and FS treatment schemes.

Table 7 Helminth eggs in biosolids from selected wastewater and faecal sludge treatment schemes [1]

Place and scheme	No. of helminth eggs per litre of untreated …		Helminth eggs in biosolids		Reference
	Faecal sludge	Wastewater	No. of eggs /g TS	Egg viability	
Extrabes, Campina Grande (Brazil); experimental WSP scheme	----	**1,000** (nematodes)	**1,400 – 40,000** (as distributed in sludge in a primary facult. pond; avg.= 10,000, approx.)	**2 – 8 %** (period of biosolids storage not reported but probably several years)	Stott *et al*. (1994)
Chiclayo (Peru); WSP schemes	----	**10 – 40** (mostly nematodes)	**60 – 260** (in sludge from a primary facult. pond)	**1 – 5 %** (biosolids stored for 4-5 years)	Klingel (2001)
Asian Institute of Techn. (Bangkok); pilot constructed wetland plant (planted sludge drying beds) for septage dewatering+stabilisation	**600-6,000** (septage; nematodes)		**170** (avg. nematode levels in dewatered biosolids accumulated over 3.5 years in planted sludge drying beds)	**0.2 – 3.1 %**	Koottatep and Surinkul (2000); J. Schwartzbrod (2000)

3. Municipal Organic Solid Waste Management

3.1 Relevant municipal solid waste characteristics and quantities

For composting purposes, the easily biodegradable fraction is of immediate interest. This includes food waste, vegetables and fruits, and garden wastes (sometimes referred to as yard wastes) such as grass, leaves and small woody materials. Although organic waste materials such as paper and timber may also be composted, they are more resistant to microbial degradation due to their high lignin content (Richard 1996). If these materials are included in the composting process, their particle sizes are often reduced beforehand through shredding to allow for quicker decomposition. Based on composition of solid waste of cities of low- and middle income countries as quoted in Obeng and Wright (1987) (from Algiers, Accra, Alexandria, Cairo, Sao Paolo) easily biodegradable fractions range between 44 and 87 %. (in weight). Similar average ranges (40-85 %) are also reported by Cointreau et al. (1985) for low-income countries. Data from the Kumasi Waste Management Department (2000) shows figures of 79 % biodegradable waste for the city of Kumasi.

3.2 Approaches for municipal organic solid waste treatment

Given these high amounts of biodegradable waste organic waste recycling, treatment and reuse can have considerable advantages for the city's solid waste management system. Zurbrugg and Drescher (2002) describe the potential benefits of organic waste management and as:

- ***reducing the environmental impact*** of disposal sites as the biodegradable waste fraction is largely to blame for the polluting leachate and methane generation.

- ***extending the existing landfill capacity*** as organic waste is kept out of the landfill thus providing additional volume.

- ***replenishing the soil humus layer*** with organic matter and nutrients by applying compost and thus contributing to sustainable resource management.

A further significant benefit of ***waste minimisation*** can be achieved if a decentralised approach is envisaged. In this case the organic fraction is removed from the waste stream and recycled ,as near to the source of generation as possible, thus reducing collection, transportation and disposal costs and reducing health and environmental risks resulting from inappropriate handling and management.

Current treatment and reuse practices for municipal organic solid waste– other than composting – include:

- the use of waste as source of food for urban animal livestock (Allison et al. 1998)
- direct untreated application onto soils
- production of fuel pellets as energy source
- mining of old naturally decomposed waste dumps for application on farmland (Lardinois, Van De Klundert 1993).

Although not considered a treatment option, the frequent use of municipal organic solid waste as animal feed must be mentioned here. Preferred organic waste used for urban animal livestock raising consists of fresh organic solid waste from sources such as vegetable markets, restaurants and hotels, as well as food processing industries. Health risks associated to feeding of animals with solid waste are possible disease transmission to animals and humans when feeding animals with meat waste from slaughterhouses (Lardinois, Van De Klundert 1993). Further risks to animals and humans are highlighted by Allison et al. (1998) with regard to unintentional feeding to waste with toxic content.

4. Composting and Co-composting

4.1 Process definition

Composting refers to the process by which biodegradable waste is biologically decomposed under controlled conditions by microorganisms (mainly bacteria and fungi) under aerobic and thermophilic conditions. The resulting compost is a stabilised organic product produced by the above mentioned biological decomposition process in such a manner that the product may be handled, stored and applied to land according to a set of directions for use. Important to note is that the process of "composting" differs from the process of "natural decomposition" by the human activity of "control". "Control" has the goal to enhance the efficiency of the microbiological activity, to restrict undesired environmental and health impacts (smell, rodent control, water and soil pollution) and assure the targeted product quality.

Co-composting means composting of two or more raw materials together – in this case, FS and SW. Other organic materials, which can be used or subjected to co-composting, comprise animal manure, sawdust, wood chips, bark, slaughterhouse waste, sludges or solid residues from food and beverage industries.

4.2 Why co-compost feacal sludge with municipal solid waste?

Co-composting FS and MSW is advantageous because the two materials complement each other. The human waste is relatively high in N content and moisture and the MSW is relatively high in organic carbon (OC) content and has good bulking quality. Furthermore, both these waste materials can be converted into a useful product. High temperatures attained in the composting process are effective in inactivating excreted pathogens contained in the FS and will convert both wastes into a hygienically safe soil conditioner-cum-fertilizer.

4.3 Composting systems

The technologies chosen for aerobic composting (or co-composting) will depend on the location of the facility the capital available and the amount and type of waste delivered to the site. Two main types of systems are generally distinguished which are: 1) **open** systems such as windrows and static piles and 2) **closed** "in-vessel" systems. In-vessel or "reactor" systems can be static or movable closed structures where aeration and moisture is controlled by mechanical means and often requires an external energy supply. Such systems are usually investment intensive and also more expensive to operate and maintain.

"Open” systems are the ones most frequently used in developing countries. They comprise:

- **Windrow, heap or pile composting**

  The material is piled up in heaps or elongated heaps (called windrows). The size of the heaps ensure sufficient heat generation and aeration is ensured by addition of bulky materials, passive or active ventilation or regular turning.

Systems with active aeration by blowers are usually referred to as forced aeration systems and when heaps are seldom turned they are referred to as static piles. Leachate control is provided by a sloped and sealed or impervious composting pads (the surface where the heaps are located) with a surrounding drainage system.

- **Bin composting**

  Compared to windrow systems, bin systems are contained by a constructed structure on three or all four sides of the pile. The advantage of this containment is a more efficient use of space. Raw material is filled into these wood, brick or mesh compartments and aeration systems used, are similar to those of the above described windrow systems.

- **Trench and pit composting**

  Trench and pit systems are characterised by heaps which are partly or fully contained under the soil surface. Structuring the heap with bulky material or turning is usually the choice for best aeration, although turning can be cumbersome when the heap is in a deep pit. Leachate control is difficult in trench or pit composting.

4.4 Key factors of the composting process

The **key factors** affecting the biological decomposition processes and/or the resulting compost quality are listed below. They comprise:

- Carbon to nitrogen ratio
- Moisture content
- Oxygen supply, aeration
- Particle size
- pH
- Temperature
- Turning frequency
- Microorganisms and invertebrates
- Control of pathogens
- Degree of decomposition
- Nitrogen conservation

Detailed description of the significance of the specific factors is explained more in detail in Part A *Annex 2*

The same process parameters valid for composting must be adhered to and play a role in co-composting of human waste together with solid waste. Special attention has to be paid, though, to the ratio at which human waste are co-mixed with other compostable material given their moisture as well as C and N content. Numerous mixing ratios of excreta and co-composted material are provided by Shuval et al. (1981), which are compiled in Part A *Annex 1*, together with mixing ratios collated from other publications. Dewatered or spadable sludges may be admixed at a volumetric ratio of approx. 1 (sludge) : 3 (solid organic material), whereas more liquid sludges (TS □ 5 %) may be mixed at ratios between 1:5 to 1:10.

4.5 Quality of compost

Gotaas (1956) lists ranges of the main constituents in final composts as reported in reviewed publications (Table 8). The quality varies widely and depends on the initial mixture of material to be composted.

Table 8 Ranges of constituents in finished compost (Gotaas, 1956)

Constituent	Range (% of dry weight)
• Organic matter	25 – 50
• Carbon	8 – 50
• Nitrogen (as N)	0.4 – 3.5
• Phosphorus (as P_2O_5)	0.3 – 3.5
• Potassium (as k_2O)	0.5 – 1.8

Compost which is dry (35% moisture or below) can be dusty and irritating to work with, while compost that is wet can become heavy and clumpy. The Composting Council (2000) recommends 40 % moisture for ideal product handling.

Usually, mature compost is sieved prior to sale and use. Sieves made of a wooden frame and wire mesh are suitable and can be easily made. Mesh sizes vary according to the compost users requirements. Used as plant fertiliser, a mesh size of 10-20 mm could be chosen, for use as seedling production mesh sizes may be around 3 mm. The compostable sieving residues of larger particle size are usually recycled to windrows for further composting.

4.6 Quality obtained by co-composting human waste

4.6.1 Nutrient Content

Nutrient contents of composts, which have been produced from co-composting human waste (faecal or sewage treatment plant sludge) are shown in Table 9. In theory, such compost should exhibit higher nutrients than compost, which is produced from such material as organic municipal refuse, woodchips, sawdust, i.e. material with N contents lower than in human waste. However, the data show that nutrient, notably N, contents do not range particularly high when compared with the ranges listed in Table 8, which were collated from many references and for composts produced from many different raw materials, including human waste.

Table 9 Nutrient levels in compost for which human waste was one of the raw materials

• **Nitrogen (as N)**	1.3 – 1.6 1.3 0.35 – 0.63 0.45	Shuval et al. (1981) Obeng and Wright (1987)[1] Kim, S.S. (1981)[2] Byrde (2001)[3]
• **Phosphorus (as P2O5)**	0.6 – 0.7 0.9	Shuval et al. (1981) Obeng and Wright (1987)[1] Kim, S.S. (1981)[2]
• **Potassium (K2O)**	--- 1.0	Shuval et al. (1981) Obeng and Wright (1987)[1]
• **Organic matter (% TVS)**	12 - 30	Kim, S.S. (1981)[2]
• **Carbon (C)**	46 – 50 13	Shuval et al. (1981) Byrde (2001)[3]

1	**Chosen as "typical values" by the authors in their chapter on the economic feasibility of co-composting**
2	**Raw material composed of varying ratios of FS (TS = 4 %), household waste and straw**
3	**Raw material composed of municipal solid waste and FS**

The reason for composts produced from human waste not exhibiting higher nutrient contents than other compost (as judged from the limited data available) might be due to nitrogen (ammonia) losses during pre-composting storage and treatment (e.g. by dewatering on sludge drying beds) of the human waste.

4.6.2 Control of pathogens

A good operation of aerobic composting should be able to kill all pathogenic microbes, weeds and seeds especially if the temperature can be maintained between 60 and 70 degrees for 24-hour period. The table below illustrates the thermal kill of pathogens and parasites.

Scott (1952) investigated *Ascaris* egg die-off during thermophilic composting in stacks, in which the composting material was turned every 5-10 days. The result is illustrated in Figure 5.

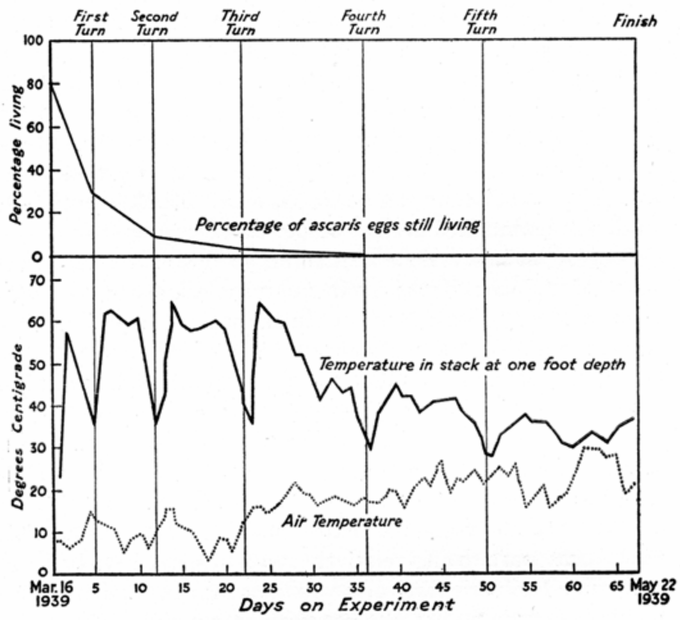

Figure 5 ***Ascaris* egg inactivation in thermophilic stack co-composting of faeces (69 % of raw material), vegetable matter (20 %), soil (10 %), and ash (1 %) (Scott 1952)**

The graph shows that complete egg die-off was achieved within seven weeks. Greater than 95 % egg die-off was achieved within little more than three weeks already, though. These periods reflect the time required for *Ascaris* eggs to "disappear" from all sections of a windrow, hence it is dependent on the composting operations. This can be achieved by windrow turning or, alternatively, by mechanically aerating a static, non-turnable, pile.

The duration for thermal inactivation of excreted pathogens at the upper temperatures attained in thermophilic composting, are much shorter, though. Table

10 lists die-off periods at temperatures constituting thermal death points for a few selected pathogens.

Table 10 Thermal Inactivation of Selected Excreted Pathogens (after Tchobanoglous et al. 1993)

Microorganism	Duration for Thermal Inactivation
Escherichia coli	Death within 1 hour at 55 °C and within 15-20 minutes at 60 °C
Salmonella sp.	Growth ends at 46 °C; death within 30 minutes at 55-60 °C and within 20 minutes at 60 °C
Entamoeba histolytica cysts	Death within a few minutes at 45 °C and within a few seconds at 55 °C
Taenia saginata	Death within few minutes at 55 °C
Ascaris lumbricoides eggs	Death in less than 1 hour at temperatures over 50 °C

A general rule of thumb for pathogen suppression is to maintain the composting process at 55°C to 65 °C for 3 consecutive days (Tchobanoglous et al., 1993).

4.7 Benefit of using compost in agriculture

The Composting Council (2000) summarises the benefits of compost as follows:

- improves soil structure, porosity and density thus creating a better plant root environment
- increases infiltration and permeability of heavy soils, thus reducing erosion and runoff
- improves water holding capacity thus reducing water loss and leaching in sandy soils
- supplies a variety of macro and micronutrients
- may control or suppress certain soil borne plant pathogens
- supplies significant quantities of organic matter
- improves cation exchange capacities of soils and growing media thus improving their ability to hold nutrients for plant use
- supplies beneficial microorganisms to soil and growing media
- improves and stabilises soil pH
- can bind and degrade specific pollutants

Addition of compost to tropical soils, which are often low in organic matter will make the soil easier to cultivate and improve its water holding capacity, preventing cracking and erosion by wind and water (Winblad and Kilama, 1978 and 1980). Obeng and Wright (1987) have summarised published information on the impact of using compost on clayey or sandy soils as shown in Table 11.

Table 11 Impact on clayey and sandy soils through the use of compost (Obeng and Wright 1987)

Impact on sandy soils	Impact on clayey soils
Water content is increased	Aeration of soil is increased
Water retention is increased	Soil permeability is increased
Aggregation of soil particles is enhanced	Potential crusting of soil surface is reduced
Erosion is reduced	Compaction is reduced

Certain microorganisms found in compost suppress detrimental organisms like root-eating nematodes and specific plant diseases. Strengthened root systems reduce the need for pesticide use (King County - Department of Natural Resources and Parks 2002).

5. Literature and case-studies on FS co-composting

5.1 Literature studies

Scott (1952) reports extensively about the combined composting of faecal matter with a variety of other organic materials as practiced in China over centuries. Experiments with material available on farms, i.e. human excreta, animal manure and crop residues focused on nutrient (notably nitrogen) conservancy and pathogen (notably helminth egg) inactivation. Scott and his co-workers found the following:

- *Ascaris* egg destruction was 95 % complete after 22 days and 100 % complete after 36 days in a stack whose contents were turned every 5-14 days and reached 60 °C after each turning.
- Nitrogen losses from raw materials and from compost exhibiting differing degrees of degradation during drying is significant. The losses found were approx. equal to the ammonia contents of the fresh material. The loss of nitrogen during co-composting amounted to about 50 % of the initial nitrogen present. The greatest loss occurred during the initial 5-10 days of composting.
- Omission of ash was assumed to have contributed to a lowering of N losses.
- Cooling the stacks with soil after the first few days of hot composting helped to considerably reduce nitrogen losses.

Shuval et al. (1981)[3] reviewed literature and collated information on historical and actual practices of co-composting "nightsoil"[3] and (sewage) sludge. Cases of excreta co-composting are reported about from India, China, Malaya, Africa (e.g. Kano, Nigeria) where fresh faecal sludge collected from bucket latrines and frequently emptied latrine vaults were co-composted. The bulking material comprised various forms of household refuse and plant residues. Most of these composting initiatives and operations are reported as having been rather successful and producing compost at a regular rate. While many of the reported schemes may not be operational anymore nowadays, since they were initiated and operated under colonial administration, considerable informal co-composting is doubtlessly being practiced in many countries around the world.

Shuval et al. (1981) and Obeng and Wright (1987)[4] reported on numerous schemes in the U.S.A. and Europe, mainly, and on windrow or open systems, in which sewage treatment plant sludge ("biosolids") are or were composted together with other organic material, notably municipal refuse. All these installations make use of lower or higher degrees of mechanization. While the biochemical and pathogen inactivation processes are the same as in non-mechanised systems, mechanised co-composting schemes are largely inappropriate for developing countries except possibly in situations where there is a high demand for the product and it can be sold at high prices.

Shuval et al. (1981) provides detailed accounts of static pile or windrow co-composting works operated with forced aeration according to the Beltsville Aerated Rapid Compost ("BARC") system developed by the U.S. Department of Agriculture research station at Beltsville, Maryland, in the 1970-ies. Several hundreds of this type of co-composting systems are in operation in the U.S.A. nowadays (Goldstein and

3 This comprises, in most reported cases, the fresh faecal material, with or without urine, collected daily from households bucket latrines or at larger intervals from latrine pits or vaults

4 Both Shuval et al. (1981) and Obeng And Wright (1987) use the term "composting" as encompassing either anaerobic, ambient-temperature degradation or "hot", aerobic and thermophilic dagrdation of organic matter. The authors of this report, however, prefer the term "composting" to exclusively designate the hot process.

Riggle, 1989). The original BARC system co-composts dewatered sewage sludge (TS = 20-25 %) and wood chips in ratios of around 1 (sludge) : 2 (wood chips). Windrows are covered with finished compost for insulation, moisture conservation and to prevent birds from feeding on fresh waste. Shuval et al. (1981) also report on a BARC-type scheme co-composting faecal sludge collected from latrine vaults in a national park with wood chips, sawdust and finished compost. The sludge (TS = 5 %) is mixed at a ratio of 1 (sludge) : 3.2 (other org. material). Finished compost contained 1.3-1.6 % nitrogen on a dry solids basis. Compost storage for one year did reportedly not lead to nitrogen losses.

Shuval et al. (1981) and Obeng and Wright (1987) also reported on economic, agronomic and marketing aspects of co-composting and its respective product. In Europe and North America, mainly digested and dewatered sewage sludge is being processed in co-composting works. Cited investigations focused on the hygienisation effect of the process, mainly, and on the fate and concentrations of heavy metals in the finished product. Shuval et al. (1981), citing Julius (1977), remarks on the importance of proper and sustained compost marketing strategies, which are to comprise the demonstration of agricultural benefits of compost on trial plots, training, extension and awareness raising.

5.2 Case-studies

The authors are aware of but a very few, more recently initiated schemes – pilot or full-scale – in which faecal sludge was or is being co-composted with municipal refuse or other organic bulking material. There are, doubtlessly, numerous co-composting activities and schemes in operation in developing countries, both formalised and informally operated ones, yet respective information has not been publicised. The following are schemes or practices, which are known to the authors either through retrievable literature, through personal communications or from own field visits:

- **Septage co-composting** –

  A pilot project in Massachusetts, U.S.A., initiated in 1977 (Lombardo, 1977)

- **Latrine sludge co-composting** –

  A pilot project in Port-au-Prince, Haiti, initiated in 1981 (Dalmat et al., 1982)

- **Bucket latrine sludge co-composting** –

  A full-scale demonstration project in Rini/Grahamstown, South Africa, initiated in 1990 (La Trobe and Ross, 1991 and 1992; personal observations).

- **Co-composting of latrine sludge with organic refuse** in Niono, Mali –

  Small-scale co-composting to produce compost for rice and vegetable farming (Montangero and Strauss 1999; personal observations)

- **Co-composting of biosolids from an FS pond treatment scheme** –

  A pilot-scale scheme comprising planting trials with finished compost to be initiated in Cotonou, Benin, in 2002 (CREPA-Benin, 2002).

Details on the individual case studies are described in detail in Part A *Annex 3*

6. Conclusions and open questions

6.1 Conclusions

Based on the "state of art" following conclusions regarding co-composting of FS and organic solid wastes can be made:

- Faecal sludges can be co-composted with any biodegradable, organic material if the rules of the art in process control are adhered to.
- Mixing ratios reported about in the literature vary widely, depend on the type of organic bulking material co-composted together with faecal matter, the consistency of the FS itself, the degree of dewatering prior to composting, and the co-composting practice and care.
- Reported mixing ratios of dewatered FS (TS = 20-30 %) and other, more bulky organic material tend to range from 1:2 to 1:4. For fresh, non-dewatered FS, ratios used and reported about tend range from 1: 5 – 1:10.
- Factors contributing to minimising nitrogen losses during thermophilic composting comprise:
  - Keeping the maximum temperatures below 65 °C
  - Keeping the periods of maximum temperatures as short as possible
  - Limiting the frequency of turning
  - Keeping the water content of the composting material as high as possible (50-70 %)
- Only scanty information exists on existing experiences, especially on organisational, institutional, and financial aspects of co-composting practices and schemes operated in developing countries

6.2 Open questions / researchable issues

Using the co-composting process as a treatment option for a city's faecal sludge and organic solid waste, raises the issues not only of the technological approach used, but also of the necessary organisational set-up for operation and management of the composting site as well as the delivery of feedstock (raw material) and distribution of the compost product. In cities of developing countries composting is considered not more wide-spread (Hoornweg and Thomas 1999) due to:

- inadequate knowledge or attention to the biological processes and a thereby inadequate quality and resulting nuisance potential, such as odours and rodent attraction.
- lack of markets for the product and lack of marketing skills or plans for compost.
- neglect of the economics of composting which relies on externalities, such as reduced soil erosion, reduced water pollution and avoided disposal costs.
- limited support by municipal authorities who tend to prioritise waste collection services rather than promote and support recycling activities.

Regarding the technical aspects of composting FS and MSW the following topics of research can be distinguished where more information is necessary:

6.2.1 Pre-treatment of FS by sludge drying beds

- FS handling and FS pre-treatment requirements
- Sludge drying bed performance in dry and wet weather conditions
- Maximum share of public toilet sludge (vs. septage) to allow for adequate rates of dewatering
- Appropriate options for treating the percolate of sludge drying beds

6.2.2 *Solid Waste*

- Appropriate methods of segregation at source or sorting procedures, to allow delivery or utilisation of pure organic solid waste for the co-composting process and limit risks of compost contamination by impurities and chemical constituents

6.2.3 *Co-composting*

- Maximum ratio of dewatered or thickened FS in the FS/MSW mixture which allows for proper thermophilic composting
- Process specifications required to ensure production of a hygienically safe compost
- Advantages and disadvantages of static pile vs. turnable windrow composting
- Occurrence of heavy metals in FS-derived biosolids vs. in co-compost
- Feasible operational patterns or measures to minimise nitrogen losses during co-composting

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Faecal Sludge	Other material	Remarks	References
• NS: 1 (vol.)	Village refuse 6 (vol.)	"Indore"	Shuval et al. 1981
• SI: 1 (wt.) (w=78 %)	2 wood chips (wt.) (mix) 0.7 wood chips (wt.) (base) 1.4 screened comp. (wt.) (cover)	"BARC" (sludge = 78 % w) (chips = 35 % w)	Shuval et al. 1981
• NS: 1 (vol.)	2 rice straw (vol?) 0.5 powdered bone (")	China (1940)	Shuval et al. 1981
• NS: 1 (wt.)	0.4-0.5 veg.matter + ash + soil (wt)	China (pre-war) (expt. I)	Scott, 1952
• NS: 1 (wt.)	4 veg.matter + ash + soil (wt)	China (expt. II) (NS = 90 – 95 % w?)	Scott, 1952
• NS: 1 (vol.) (w = 95 %)	1.6 wood chips (wt.) 1.5 sawdust (wt.) 1 compost (wt.)	"BARC" (NS: 95 % w)	Shuval et al. 1981
• Sludge (dewatered): 1 (wt.)	1.13 finished compost (wt.)	"BARC"	Shuval et al. 1981
• Sept. tank sludge (septage?): 1 (8 m^3) (w = 90 %)	3 (24 m^3) refuse (w = 61 %)	Windrow experiment, AIT	Pescod, Jan. 1970

NS – Nightsoil
LS – Latrine Sludge
S – Septage
SI – Sewage Sludge
w = water content

Faecal Sludge	Other material	Remarks	References
• NS: 1 (wt.)	0.6 city refuse (wt.) (wet)	Accra	GOPA, 1983
• NS: 1 (vol.)	2 city refuse (vol.) (dry)	(refuse: w = 30 %) (n/s: w = 74 %) ^{w}mixture = 57 %	
• NS: 1 (wt.)	+0.6 city refuse (wt.) (wet) → 0.5 compost (wt.)	Accra	GOPA, 1983
• S + N: 1 (wt.)	5 city refuse (wt.) (wet) (..1.7 compost)	Accra	1991 (Eiling, Neff; GOPA)
• NS: 1 (vol.) (w = 95 – 96 %)	10 city refuse (vol.)	Rini township, Cape Province South Africa	
• Faecal sludge 1 (vol.) (~ 70 % septage + 30 % BV latrine sludge)	3.5 city refuse (vol.)	Hanoi, Vietnam	Personal comm. (1994)

NS – Nightsoil
LS – Latrine Sludge
S – Septage
SI – Sewage Sludge
w = water content

SANDEC/MS
1994 and 2000

Part A Annex 2

The science of composting

C:N Ratio and other nutrients

The primary nutrients required for microorganism growth are carbon, nitrogen, phosphorus and potassium. Although bacteria also need trace amounts of sulphur, sodium, calcium, magnesium, and iron, these elements are usually present in adequate quantities and do not limit bacterial activity (Hoornweg and Thomas 2000). Carbon and nitrogen are both the most important and the most commonly limiting elements for microbial growth (occasionally phosphorous can also be limiting). The ideal ratio of C to N is between 20-30 :1. When there is too little nitrogen, the microbial population will not grow to its optimum size, and composting will slow down as nitrogen becomes a limiting factor to the growth of microorganisms. Microorganisms are forced to go through additional cycles of carbon consumption, cell synthesis, decay, etc, in order to burn off the excess carbon as CO_2 (Kiely, G., 1998; GTZ 2000). In contrast, too much nitrogen allows rapid microbial growth and accelerates decomposition, but this can create serious odour problems as oxygen is quickly depleted and anaerobic conditions occur. In addition, some of this excess nitrogen will also be given off as ammonia gas that generates odours while allowing valuable nitrogen to escape (Richard et al. 1996). The bioavailability of carbon also needs to be taken into account when considering the C/N ratio. This is commonly an issue with carbon materials, which are often derived from wood and other lignified plant materials, as increased lignin content reduces biodegradability. Thus a C/N ratio of 30 where carbon has high lignin content would be too low for ideal composting as the carbon is not easily available for microbial activity.

Mixing various feedstocks of different C/N ratios allows a control of the total C/N ratio. Some raw materials are high in carbon others high in nitrogen. In practice, the ideal combination of different feedstock types can be determined by experimentation and experience. Generally one can classify "green" high nitrogen materials and "brown" high carbon materials which in a simple recipe mixture can be mixed together in equal volumes. Examples for "green" materials are fresh grass clippings, manure, garden plants, or kitchen scraps; "brown" materials are dried leaves and plants, branches, and woody materials.

Moisture

Maintaining adequate moisture content in the composting pile is important, as hu-midity is required by microorganisms for optimal degradation. Moisture also dissipates heat and serves as a medium to transport critical nutrients. Moisture content between 40 to 60 percent by weight throughout the pile is ideal. Higher moisture levels slow the decomposition process and promote anaerobic degradation because air spaces in the pile are filled with water and can not be supplied with oxygen. Moisture levels less than 40 percent cause the microorganisms to slow their activities and become dormant or die. Moisture can be easily added during turning by sprinkling water or a mixture of urine and water in a mixing ratio of 1:4 as urine enhances the growth of the microorganisms. For best control of moisture, composting in piles covered by a roofed structure is ideal. If in an open area, at times with excessive rains, the waste pile can be made as steep as possible and be covered with a tarpaulin, plastic sheeting or gunny-bags to reduce water infiltration. In times of excessive heat and drought, the same coverings can serve to reduce evaporation. The optimal moisture level is achieved when the composting material feels damp to the touch; that is, when a few drops of liquid are released while squeezing a handful of material strongly. You can also test for moisture level content by putting a bundle of straw in the heap. If after five minutes, it feels clammy, then the moisture level is good; if still dry after five minutes, the moisture level is too low. Water droplets on the straw indicate that the heap is too wet for successful composting.

Moisture content and coarseness of material are closely interrelated in terms of displacement of air in the pores by water, promotion of aggregation and lowering of the structural strength of the material.

Particle size

The surface area of the organic material exposed to microorganisms is another factor in determining the rate of composting. Waste material shredded, chipped, or otherwise reduced in size can be degraded more rapidly. This is significant especially with slow degradable woody materials. However, care must be taken to avoid compacting the materials by too small material sizes, as this reduces the porosity of the pile and possible air circulation. The optimum particle size ranges between 25 and 75mm (1 and 3-inches). GTZ (2000) recommends chopping all materials to be composted to the length of about 5-10cm. Obeng and Wright (1987) reported that typical particle sizes should be approximately 1cm for forced aeration composting and 5cm for passive aeration and windrow composting.

The physical state and the size of particles affect the moisture content and the composting process. The coarser the material the higher the moisture content should be. A consistent particle size ensures a homogenous composting process and facilitates the further treatment of the compost.

Aeration

The air contained in the interstitial spaces of the composting mass at the beginning of the microbial oxidative activity varies in composition. The carbon dioxide content gradually increases and the oxygen level decreases. When the oxygen level falls below 10%, anaerobic microorganisms begin to exceed the aerobic ones. Fermentation and anaerobic processes take over. This implies that the aerobic microorganisms must have constant supply of fresh air to maintain their metabolic activities unaltered. The oxygen needed for composting is not only needed for aerobic metabolism and respiration by the microorganisms but also for oxidising various organic molecules present in the mass. Oxygen consumption during composting is directly proportional to microbial activity; therefore there is a direct relationship between oxygen consumption, temperature and aeration.

The greater the aeration rates the more rapid the rate of degradation. Aeration provides the necessary aerobic conditions for rapid odourless free decomposition and for destruction of pathogenic organisms by heat. The most common way for aerating the compost heap cheaply in the developing country is by turning (Winblad and Kilama, 1980). Active aeration refers to methods which actively blow air through the compost pile. Passive aeration takes advantage of the natural diffusion of air through the pile enhanced by ventilation structures such as perforated pipes in the pile, openings in the walls of composting bins and of course the particle size and structure of the raw materials in the heap. If air supply in the pile is limited, anaerobic conditions occur; thus producing methane gas and malodorous compounds such as hydrogen sulfide gas and ammonia.

The consumption of oxygen is greatest during the early stages and gradually decreases as the com posting process continues to maturity.

Temperature

In windrows which have been prepared according to the “rules of the art”, i.e. with adequate porosity, humidity, and C:N ratio, and exhibiting a minimal size to provide sufficient “body” for insulation (1x1x1 meters), thermophilic temperatures develop independently of ambient temperatures. Heat is generated in aerobic decomposition as a result of the microbial activity in the pile as the aerobic degradation of organic material is an exothermic process. As the temperature of the pile increases, different groups of organisms become active. With adequate levels of oxygen, moisture, carbon, and nitrogen, compost piles can heat up to temperatures in excess of 65 degrees Celsius. Higher temperatures begin to limit microbial activity. Temperatures above 70 °C are lethal to most soil microorganisms. If windrows don't turn hot, this is a sign of process failure and that windrows were not set up according to the rules of the art.

The thermophilic composting process goes through several temperature variations The class of bacteria involved in the degradation process are psychrophilic (5-20 °C), mesophilic (20-50 °C) and thermophilic (50-70 °C) (Kiely, 1998; Winblad and Kilama, 1980). This diversity is necessary for the stepwise decomposition of the organic substances to stable compost (humic substances and nutrients). Although composting will occur also at lower temperatures, maintaining high temperatures is necessary for rapid composting as it controls the thermo-sensitive human pathogens as well as destroys weed seeds, insect larvae, and potential plant pathogens that may be present in the waste material.

After piling the organic material, the temperature rises to 60 – 70 °C within 1-3 days. After several days of active degradation, the process slows down and the temperature remains around 50 – 55 °C. After approximately 30 days the compost process will slow down further and the temperature will drop below 50 °C.. The composting process now enters into the maturing phase with low microbiological activity at temperatures around 40 °C. As the compost becomes mature the temperature approaches the ambient temperature conditions.

Turning frequency

Usually the greater the turning frequency, the better the chances for uniform and better degradation. For quality control, it is important that all the waste has been through the thermophilic phase. This can be best controlled by regular turning. However, frequent turning may also lead to increased ammonia losses, particularly so during the first few days of thermophilic activities, when temperatures and pH is highest.

pH

Organic matter with wide range of pH (between 3 and 11), can be composted. However, good pH values for composting are between 5.5 and 8; and between 4 and 7 for the end product (Winblad and Kilama, 1980). Whereas bacteria prefer a nearly neutral pH, fungi develop better in a fairly acid environment. In the first moments of the composting process, the pH may drop to around 5 as organic acids are formed, however then microbial ammonification will causes the pH to rise into the range of 8-8.5. Only during maturation, when the ammonium compounds are nitrified to nitrate will the pH sink once more below 8. Thus, a high pH is generally the sign of immature compost.

Microorganisms and invertebrates

A properly constructed compost pile represents a interactive biological and ecological system. It involves a diversity of species that emerge in response to changes in the nutritional and environmental conditions of the pile. Chemical decomposition of organic compounds results predominantly from microorganisms. such as bacteria, actinomycetes, fungi, and some protozoans. At the first stage of composting when temperature rises through the mesophilic stage into the thermophilic range, bacterial population which can multiply rapidly while utilising simple and readily available substrates dominate. As temperature rises thermophilic bacterial populations take over. If excess heat is removed by ventilation or turning these populations will be maintained and overall rates of bacterial activity will remain high. Fungi nor actinomycetes can withstand temperatures as high as the thermophilic bacteria. When thermophilic bacteria have used up the most easily available substrates, bacterial microbial activity can no longer liberate heat fast enough to maintain high temperatures. As temperatures drop, actinomycete population increase and more complex substrates can be attacked by extracellular enzymes (Palmisano and Barlaz 1996). As temperatures drop further the remaining substrates which are even more resistant to decomposition, are degraded by fungal populations. The role of activities and appetites of various invertebrates such as mites, millipedes, beetles, earwigs, earthworms, slugs, and snails for physical and chemical decomposition is not be underestimated.

Gotaas (1956) discusses the issue of **inoculation** to enhance microbial degradation. Modern developments in science and practice of composting have, apparently, been accompanied ever since by the promotion of and continuous debate about the need and usefulness of inocula comprising specific, laboratory-cultured strains of bacteria, enzymes, “catalysts”, “hormones”, etc.. A product designated “EM” (“effective microorganisms”), has been aggressively marketed in Asia in recent years. It is used by households and applied to pits and vaults of on-site sanitation installations, as well as solid waste composting heaps and dumps, reportedly to help enhancing biochemical degradation, preventing odours and formation of large aggregates which may block appurtenances. EM is sold and used also to reportedly enhance or speed up composting processes and to prevent odour formation. The authors of this report are not aware of any independent, rigorous study, which have been done to investigate the effects and usefulness of EM in the composting process.

Early composting studies dealing with this issue and reviewed by Gotaas, appear to strongly indicate that inocula are not necessary. Gotaas argues – and in fact most composting specialists share this view – that indigenous bacterial and other microbial populations are not a limiting factor in composting. They can produce rapidly the enzymes, vitamins and other growth factors required in sufficient quantities and at adequate rates.

Nitrogen conservation

Gotaas (1956), provides a comprehensive and in-depth description of the composting process and composting operations. In particular, he also discusses problems in relation to nitrogen losses and means of conserving it. Like other nutrients (phosphorus, potassium, micronutrients), nitrogen may also be lost through leaching, yet, in contrast to those nutrients, by far the greatest portion is lost through volatilization in the form of ammonia (NH_3) and other nitrogenous gases. These losses have impact on the fertilising value of the compost product, thus influencing crop yield, farm economics and, hence, farmers’ livelihood. Ammonia losses are affected by almost all process parameters such as C/N ratio, pH, moisture, aeration, temperature, the chemical form of nitrogen in the feedstock, adsorptive capacity of the composting mixture, and windrow turning frequency.

The pH and temperature has a great effect on the ammonia (NH3) – ammonium (NH4+) equilibrium.

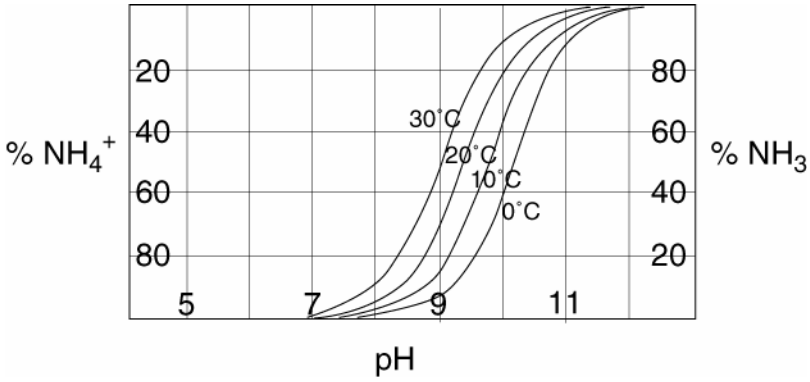


Figure 6 Ammonia – ammonium equilibrium at different temperatures and pH levels (Schreiner 1997).

The Figure 6 shows that higher pH and higher temperature moves the equilibrium in favour of ammonia. Thus higher levels of pH during the composting process or high pH in the initial feedstock might enhance ammonia volatilisation for instance if the raw material may contain appreciable portions of ash (ash exhibits a pH of 10-11).

Excessive dryness will enhance NH_3 volatilisation whereas sufficient moisture contents, like those for optimum composting, from 50-70 %, allow to keep the highly soluble ammonia in dissolved state (Gotaas, 1956).

Excessive aeration and windrow turning enhances loss of ammonia, which escapes more easily when the composting material is exposed to the atmosphere. Hence, an optimum frequency of turning must be found, which balances the need for all parts of a windrow to be subjected to hot degradation with the need to limit nitrogen loss.

A similar balance has to be strived for in temperature development. High temperatures of around 60 - max. 65 C are desirable to attain good pathogen inactivation, yet long periods of around 70 C must be avoided as ammonia formation and potential escape increases considerably at this temperature.

Degree of decomposition or compost maturity

Indicators for the degree of decomposition are: the colour and smell, the drop in pile temperature, the degree of self heating capacity, the nitrate-N / ammonium-N ratio, the amount of decomposable and resistant organic matter in the decomposed material, redox potential, and oxygen uptake.

In immature composts, when applied on soil, the microbial activity continues and there is a danger of microorganisms competing with the plants for the availability of soil nitrogen (nitrogen block). Immature compost also may contain high levels of organic acids and can damage plant growth when used for agricultural applications.

Part A Annex 3

Case studies of co-composting

Septage co-composting –Massachusetts, U.S.A

A ***septage co-composting*** pilot plant was commissioned in the state of Massachusetts in 1977 to test the feasibility of co-composting for septage collected from three neighbouring towns (Lombardi, 1977). The initiative followed prohibition by the authorities to continue the admixing of septage to the wastewater treatment plant. Septage of approximately 4 % TS was mixed with sawdust, woodchips and cow or horse manure. Mixing ratios are reported, yet conflicting figures render it difficult to know what actually used ratios were. Both forced and naturally vented, static windrows were used. Reported temperature development, however, indicates that aeration was secured and thermophilic conditions were achieved, with temperature rising to 73 °C at windrow centres within 8 days of pile formation. They levelled of to about 50 °C after 50 days. Capital cost for a full-scale septage co-composting plant serving the three towns and treating 60 m^3 of septage p. day were estimated at $ 240,000 (1977 base). The procuring of sawdust as liquid absorber was found to constitute a major O+M cost item. The authors do not avail of information whether the system is still operational, or if a full-scale system was built and has become operational as a result of the pilot works.

Latrine sludge co-composting –Port-au-Prince, Haiti

A ***pit latrine sludge co-composting*** pilot scheme was initiated at Saint Martin, a suburb of Port-au-Prince with a population density of > 2,000 persons/ha in 1981 (Dalmat et al. 1982). Both the traditional and newly constructed individual pit latrines of which several ones are attached to each other to form a toilet block are shared by several families. The latrines have traditionally been manually emptied, but tractor-drawn vacuum tanks were introduced through a donor-aided programme. A BARC-type composting system was installed, using forced-aerated windrows. Pit latrine sludge and partially composted refuse were mixed at a ratio of 5:1 to form piles of 21 m^3. No figure is given for the TS content of the pit latrine sludge, but it may be assumed to have ranged from 4-8 %. Air was drawn through the windrows at 12 minutes "on" and 8 minutes "off" cycles. Exhaust gases were pushed through a pile of finished compost to minimise odours. Windrows were covered with a layer of compost for insulation and odour control. No monitoring data were reported on the co-composting operations. Preliminary results from greenhouse planting trials indicated that the use of co-compost yielded "significantly greater plant growth and yield response" as compared to the use of refuse compost. Haitian soils reportedly have a very low organic fraction. Hence, it was anticipated that the use of co-compost would have considerable impact and be a good marketing argument. For this project, too, the authors do not have any information at hand whether the pilot project was scaled up and/or applied elsewhere in Haiti or the country, and whether such operations continue until today.

Bucket latrine sludge co-composting – Rini/Grahamstown, South Africa

An example of recent ***co-composting*** *operations using* ***bucket latrine sludge and MSW*** is the demonstration scheme at Rini near Grahamstown, South Africa (La Trobe and Ross 1992). The plant was commissioned in late 1992, following a two-year trial phase on pilot scale. The scheme became redundant, though, following the conversion of the bucket latrines into sewered toilets in 1997. In spite of this, the authors consider worthwhile to provide here a description of the plant and its operation.

Figure 7 **Sprinkling FS over refuse at the Rini/ Grahamstown (South-Africa) co-composting plant**

Figure 8 **Sieving of matured compost in a rotary sieve at the Rini/Grahamstown (South Africa) co-composting works**

The plant in which refuse and bucket latrine sludge collected from Rini (pop. =100,000) were co-composted, consisted of forced-aerated, static windrows. The faecal sludge was delivered to the station by a tractor-drawn vehicle in 20-L barrels. Approximately 20 m^3 were delivered daily. It was then screened and collected in a pump sump from where it was pumped by a macerating pump to two overhead, cone-shaped settling/thickening tanks. The tank supernatant was treated in waste stabilisation ponds, which were earlier receiving the bucket latrine sludge. The thickened FS (TS = 5 %) was gravitated over the windrow as the mixed refuse was being heaped up (Figure 8). Final windrow size amounted to around 100 m^3. The windrow was covered with finished compost for insulation and bird control. The volumetric mixing ratio was approximately 1:10 (FS:refuse). Measuring temperature at different spots of the windrow controlled the process. Temperatures of 55 °C were reached and the windrows left to react for 3 weeks. The compost was let to mature for another 3 weeks. The matured compost was sieved (Figure 9) and the rejects landfilled. The Grahamstown garden department used the compost. The finished compost was reportedly free of helminth eggs. Unfortunately, no scientific data were generated or published about this valuable co-composting experience.

Co-composting of latrine sludge with organic refuse in Niono, Mali

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 microentrepreneur. Faecal sludges are collected manually or by tractor-drawn vacuum tanks. The compost is sold to rice and vegetable farmers (Montangero and Strauss 1999). Figure 10 illustrates the processing of the FS with refuse and lime. Sieved refuse, liquid FS and lime are made up in batches of approx. 2.8 m3, let to sun-dry and then processed in the heated pelletizer (ret. period approx. 1 min.). The ratio of sieved refuse to liquid FS amounts to 1:1.3. Hence, lime (CaCO3) is added to dewater the liquid sludge.

The process allows inactivating excreted pathogens considerably, yet drying periods are too short and heating temperatures too low to achieve a reasonably safe "compost" all the time.

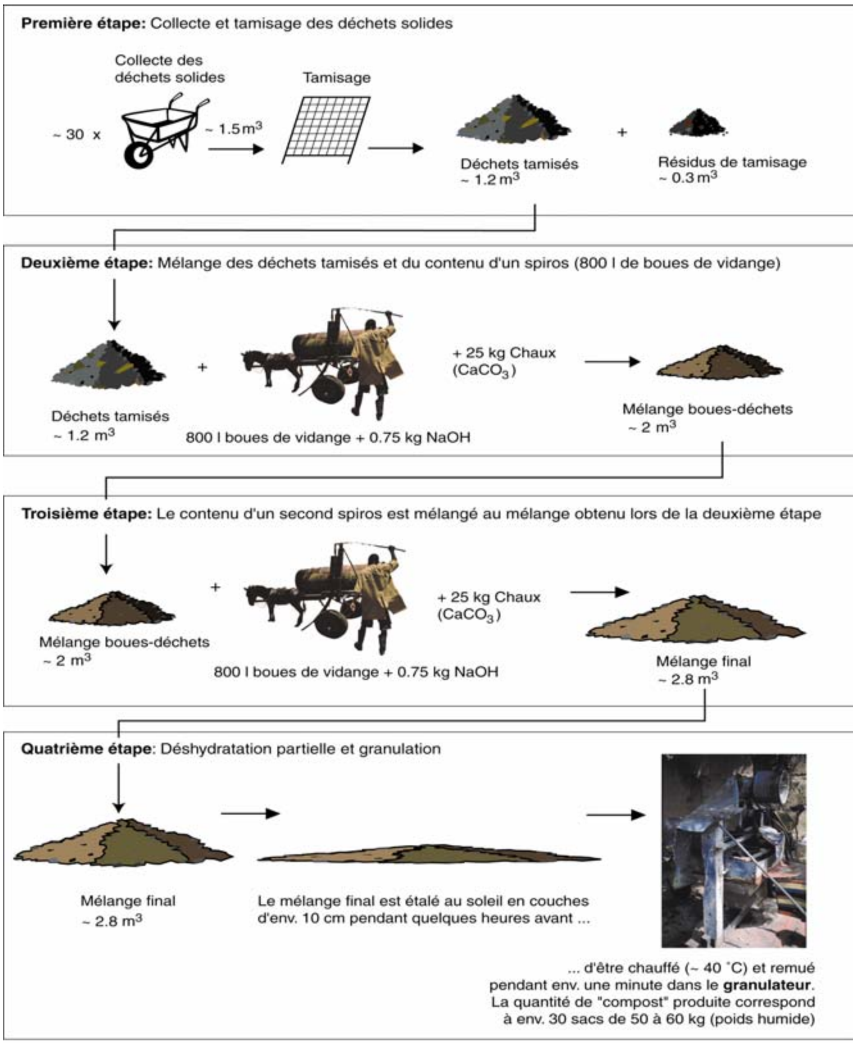


Figure 9 Co-composting of FS and sorted MSW in Niono, Mali (Montangero and Strauss 1999)

The hygienic quality of the end product may vary as a function of the concentration of parasites (helminth eggs) in the raw FS and on operational care taken during treatment.

A proposal was made to upgrade this non-thermophilic co-composting system into thermophilic processing by resorting to a scheme comprising sludge drying beds for FS dewatering followed by "hot", turnable windrow composting of the sludge cake/refuse mixture. This would enable effective and reliable inactivation of excreted pathogens. The authors do not have information, though, whether such changes of treatment technology have been effected meanwhile or not.

Co-composting of biosolids from an FS pond treatment scheme –Cotonou, Benin

A pilot co-composting scheme is currently (October 2002) being implemented in Cotonou, Benin, as part of an action research programme of CREPA aiming at improvements in FS management (CREPA Benin, 2002). **Biosolids** generated in an FS pond treatment system will be co-composted with municipal refuse. Comparative planting trials will be conducted with co-compost and other plant/soil amendments.

Part B

Pilot Co-composting Plant Investigations in Kumasi Results and Preliminary Recommendations

1. The context of the pilot project

Kumasi, located 300 km Northwest of Accra, covers 150 km^2 and counts about 1 million inhabitants. The city is an industrial centre with formal industries in timber, food processing (including beer brewing) and soap manufacturing, together with informal activities in woodworking, light engineering, vehicle repair, footwear, furniture manufacture and metal fabrication. Population growth rate (3%) and waste generation rate are high (2000 Population and Housing census, Ghana Statistical service; KMA, 2000). So far, the KMA has not been able to comprehensively manage the problem of waste in Kumasi due to high capital costs for plant and equipment, increasing operation and maintenance costs, the rapid spatial and population growth with decreasing coverage levels, and the increase in volumes of waste generated (Salifu, 1999). Disposal of these wastes have traditionally been sheer deposition on the bare soil or sometimes in water bodies creating environmental and aesthetic nightmares. The picture is even gloomier at the night soil disposal sites. Thus, there is the danger for contamination and pollution of water bodies, soil and air (through dumping, leaching and burning) and high potential for epidemic in and around the city, especially in times of inundation.

The current domestic daily waste generation in Kumasi based on 1,017,246 population (Statistical Service, 2000) is 610 tonnes. An estimated additional 250 tonnes is generated from the 2 main markets (KMA-WMD, 2000) to bring the total citywide daily generation to about 860 tonnes. Currently, the bulk of the solid waste generated in the metropolis is collected by the private sector based on a mixture of contract and franchise arrangements. Two main collection methods are employed: House-to-house and Communal Container Collection systems. The House-to-house collection service covers about 1,500 houses in selected communities of the high-cost sector The 1,500 houses serviced in comparison with the over 45,000 houses in the metropolis leaves much to be desired. The Communal Collection System entails the location of metal containers (skips) at designated sites known as transfer stations, which are shared, by a number of houses within that community. There are 124 transfer stations, which are spread over the city (Leitzinger Christopher and Gyiele Lucy, 2000). When the skips are full, they are transported and emptied at the final disposal site by skip loading trucks. Where there are no containers, households deposit their refuse temporarily on the ground. The communal containers used for the service have been found to be too high making them user-unfriendly. This results in waste being thrown about around the containers mostly by children. The current location of a temporary landfill site at Buokrom about 3.5km from the Kumasi Airport is highly undesirable and operations continue because of lack of alternative sites. Dumping of refuse result in smouldering and the sporadic outbreak of fire which created a smog cover over the surroundings of the landfill including the villages of Duase and Kenyasi.

Most residents in Kumasi (about 38%) still use public toilets. Another 26 percent use septic tanks. The unhygienic bucket latrine system caters for around 12% of the population; 8% rely on sewerage while pit latrines (10%) and the bush provides for the rest of the population (Mensah, 2002). One of the most critical waste disposal problems of the city of Kumasi is the disposal of faecal sludge (FS) from public latrines, household bucket latrines, and septic tanks. About 500m^3 of faecal sludge is produced each day in Kumasi. There has not been any proper disposal mechanism yet as FS is being dumped at the temporary disposal facility at Kaasi and then flows into the Subin River without adequate treatment.

Urban and peri-urban agriculture in Kumasi has an important socio-economic impact. It contributes to food security and increases the income of the urban poor. In a recent material flow study conducted in the City of Kumasi, Ghana, it was found that urban and peri-urban agricultural soils are greatly depleted of organic matter and nutrients (N and P). The combined production of human wastes (excreta), organic market wastes, and wastes from industries such as breweries, timber factories and chicken farms, could, if made use of in a consequent manner and upon treatment by combined composting, fully replenish the soils in the urban perimeter and considerable portions of peri-urban farms in the 40 km perimeter of the City (Leitzinger 2000; Belevi et al. 2000). In this way, significant volume reduction of waste for disposal will be achieved and the quantity of faecal sludge discharged untreated in the environment will be reduced.

The Kumasi Metropolitan Assembly (KMA) has had composting as part of its strategic planning for waste management in the city over the years but no implementation had taken place. Under the World Bank financed Urban Environmental Sanitation Project (Urban IV), two FS treatment plants (FSTP) and a sanitary landfill have been planned and are being implemented. One FSTP has been completed (to be operational very soon) at Buobai. The utilization of the organic proportion of the solid waste and the stabilized sludge from the respective facilities will not only provide nutrients for urban agricultural enhancement but will also help prolong the useful life of the landfill facility.

The Pilot Co-composting Project will provide the vital information for the planning and implementation of the large-scale project at the new landfill site. The KMA plans to utilize the compost in two ways. Some will be sold to farmers to be used as soil conditioner to improve agricultural activities and the rest used as cover materials on the landfill. The primary objective of the composting project as far as KMA is concerned is to reduce the quantity of waste to be landfilled thereby increasing the life of the landfill facility. Consequently, cost recovery will not be an objective in determining the selling price of the compost since that is likely to make it unattractive to the potential users.

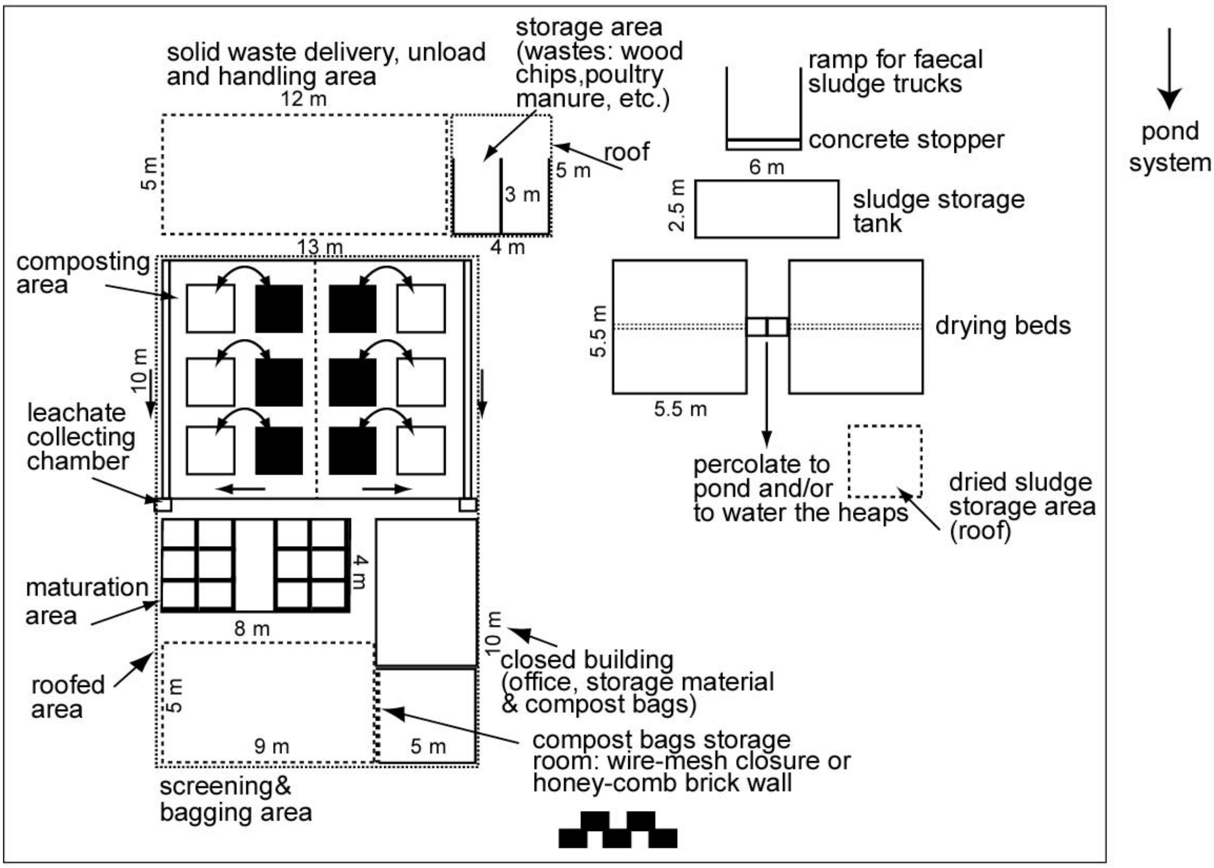
2. *The Kumasi pilot co-composting plant*

The pilot plant is operated batch-wise. The following table contains the design criteria and assumptions.

Table 1 Design criteria/assumptions

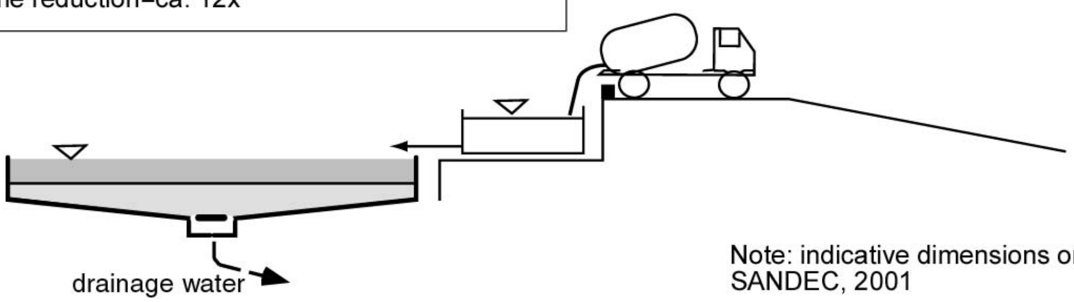
Faecal Sludge Dewatering	3 dewatering cycles per month
	3 sludge trucks per dewatering cycle (1 truck ~ 5 m3)
	15 m3 per cycle
	Ratio public toilet sludge (PTS)/septage 1:2
	1 truck containing PTS, 2 trucks containing septage
	Sludge storage tank (monitoring raw sludge mixture): 15 m3
	Sludge drying beds 15m3/30cm (fresh sludge layer): 50 m2
	Assumed sludge volume reduction through dewatering: 10%
	Assumed dewatered sludge production: 1.5 m3/cycle, 4.5 m3/month
Composting	Ratio solid waste/dewatered sludge 3:1
	Composting time: 1 month thermophilic + 1-2 months maturation
	1 composting cycle starting each month
	Sorted solid waste 3x4.5m3/month: 13.5 m3
	Unsorted solid waste delivery: ca. 27 m3/month (for 50% organic waste in household waste)
	Raw compost: 18 m3/month (4.5+13.5), 6x3m3 windrows
	Mature compost: ca. 9m3/month (volume reduction: 50%), ca. 4.5t/month

Draft Sketch Buobai Co-Composting Pilot Plant



I. sludge dewatering
sludge storage tank ca. 3 trucks x 5m3/truck = 15 m3 e.g. 2.5x6x1.2 (0.2m freeboard) tank geometry should favor solids wash out
drying beds sludge layer ca. 20-30 cm, A:60 m2 2 units 5.5mx5.5m
sludge distribution system on beds (channels?) !!!
Volume dried sludge: ca. 1-1.5 m3 volume reduction=ca. 12x

II. composting
6 heaps, 3m3/heap e.g. 2x2x1.5 "pyramid" composting period ca. 1 month
III. maturation
maturation period 1 to 2 months max. 12 heaps, volume reduction during composting about 50% ca. 12x1.5m3, e.g. in "boxes" (between walls) ca. 1mx1.5m



Note: indicative dimensions only
SANDEC, 2001

Figure 1 Sketch of the pilot plant

One person manages the treatment plant. The **plant manager**

- orders by the Waste Management Department of Kumasi (WMD) the faecal sludge and solid waste delivery at a convenient point of time,
- supervises the faecal sludge loading onto the drying beds,
- supervises desludging of the drying beds when the dewatered sludge is dry enough
- supervises the solid waste sorting
- supervises the making of the heaps as well as turning and watering of the heaps, sieving and bagging of the mature compost
- supervises the monitoring of the pilot plant
- pays the workers

Two **workers** are in charge of

- mixing the three sludge loads in the sludge storage tank
- desludging the drying beds
- sorting the solid waste
- making the compost heaps, turn and water them when necessary
- sieving and bagging the mature compost

The **WMD** is responsible to

- deliver faecal sludge and solid waste to the pilot plant and
- transport the sorted non compostable waste to the landfill

MSc students of the University of Kumasi were responsible for

- the monitoring of the drying beds and
- the monitroing of the composting

3. *Faecal sludge dewatering*

3.1. Introduction

3.1.1 The purpose of sludge dewatering prior to co-composting

The water content of fresh faecal sludge (FS) amounts to approximately 95 to 99%. Addition of fresh, undewatered faecal sludge to solid waste (SW) poses therefore practical problems unless the percentage of sludge is very low. Moreover, the water content of organic solid waste in developing countries amounts to 60-80%. As the water content of the compost windrow should amount to 40-60% to allow aerobic composting, the water content of the fresh raw sludge must be reduced. Another reason for FS dewatering is that the ratio of the per capita production of household organic SW and FS amounts to approximately 1:2 (household organic SW production: ca. 0.5 l/cap*d, FS production: ca. 1 l/cap*d, see Table 2). In case the evaluation of potential treatment options for FS and SW indicate that co-composting of FS and household organic SW would be an appropriate option for a given district, FS should be dewatered in order to obtain an adequate water content of the compost mixture. The proportion of SW should also be high enough (volume ratio SW:FS = 2:1-3:1) so as to create a porous structure in the compost windrow that will enable air to circulate through the windrow and hence aerobic conditions to develop. To obtain a SW/FS ratio of 3:1, the FS volume should be reduced through dewatering. Other organic wastes like manure or wood waste could also be mixed with dewatered FS and co-composted. The purpose of the faecal sludge dewatering step is therefore to facilitate the mixing of the sludge with the other compost feedstocks and to obtain a compost mixture with an adequate water content and structure for aerobic composting. For these reasons, a dewatering step was designed in the pilot co-composting plant.

Table 2 Daily per capita volumes of different types of FS (Heinss et al. 1998)

	Septage [1]	Public toilet sludge [1]	Pit latrine sludge [2]	Fresh excreta
Volume l/cap·day	1	2 (includes water for toilet cleansing)	0.15 – 0.20	1.5 (faeces and urine)

1 Estimates are based on a faecal sludge collection survey conducted in Accra, Ghana.
2 Estimation

3.1.2 Options for sludge dewatering

There are two main options for sludge dewatering (see part A): settling/thickening in settling tanks/ponds or dewatering on drying beds (filters). Full-scale settling/thickening tanks for faecal sludge treatment are already in operation in Accra (Ghana) (Larmie and Heinss, 98a). Pilot drying beds have already been monitored in Accra (Larmie and Heinss, 98b). In this pilot project, both sludge dewatered on drying beds and sludge thickened in settling/thickening ponds have been and will be co-composted with organic solid waste, respectively. As a full-scale FS treatment plant consisting of settling ponds as a primary treatment step already exists beside the pilot site from which thickened sludge can be removed and co-composted, pilot drying beds have been installed for sludge dewatering at the co-composting site. Main differences between both dewatering systems are the TS loading rate, the required area, the desludging operations, the attainable TS content in the dewatered/thickened sludge (Table 3) and the quality of the effluent/percolate.

Table 3 Comparison of sedimentation/thickening tanks and unplanted sludge drying beds for FS dewatering (Heinss et al. 1998)

	Attainable TS %	Assumed Loading cycle	TS loading kg TS/m^2·yr	Required area m^2/cap 1)
Sedimentation/ Thickening Tank	≤ 14	8-week cycle (4 weeks loading + 4 weeks consolidating; 6 cycles annually); two parallel settling tanks	1,200	**0.006**
Sludge Drying Bed (unplanted)	≤ 70	10-day cycle (loading-drying-removing; 36 cycles annually)	100 - 200	**0.04-0.07**

1) Assumed parameters: FS quantity = 1 litre/cap·day; TS of the untreated FS = 20 g/l

The dewaterability and thickenability of the faecal sludges are important factors determining area requirements.

3.1.3 Sludge drying beds

Unplanted drying beds consist of different layers of gravel and sand (Figure 2). Sludge dewatering and drying occurs through gravity percolation and evaporation. The percolate volume corresponds to 50-80% of the fresh sludge volume loaded onto the beds. The pilot investigations conducted in Accra/Ghana have shown that sludge can be dewatered to 40-70% TS within 8 days during the dry season. This means that biosolids dewatered on drying beds are spadable (TS≥20%) and can be therefore easily desludged. Unplanted drying beds must be desludged before each new loading as the dried sludge layer would impair the percolation of the newly loaded sludge. Furthermore, it was concluded from the Accra investigations that fresh, unstabilised sludge like the one emptied from public toilets does not lend itself to dewatering on drying beds. Public toilets are emptied every one to two weeks only (Heinss and Larmie, 98b). Hence, only septage (as it has been retained in septic tanks for several years and is therefore partly stabilised) or a mixture of septage and unstabilised public toilet sludge should be loaded onto drying beds. Loading rate of 100-200 kgTS/m2*yr and fresh?sludge layer of 20-30 cm are recommended. When loading raw sludge onto the beds, the pressure flow must be reduced so as to avoid damaging the filter layers. Drying beds retain helminth eggs efficiently. Considerable helminth eggs concentrations were found in sludge dewatered to less than 40%TS in the Accra experiments (Heinss et al. 1998). Several months of storage or further drying to less than 5% water content (Feachem et al. 1983) will ensure complete egg inactivation in biosolids and are hence necessary prior to safe reuse of biosolids as soil conditioner. Organics and solids load in the percolate of drying beds are significantly lower than in the effluent of sedimentation/thickening tanks (Heinss et al. 1998).

Table 4 Percolate of drying beds – average removal efficiencies (12 bed loadings, Accra experiments, Heinss et al. 1998)

Parameter	Removal efficiency [%]
SS	≥95%
COD	70-90%
HE	100%
NH4	40-60%

3.1.4 Objectives of the Kumasi pilot investigations

The Kumasi pilot investigation aimed at establishing recommendations for the design and operation of FS/SW co-composting systems in the Ghanaian socio-economic context. The investigation of the drying beds has the following sub-objectives:

- to know whether the design of the pilot drying bed (heights of gravel and sand layers, inlet distribution channel, sand type) is appropriate
- to know whether a mixture of public toilet sludge and septage with a ratio of 1:2 lends itself to dewatering
- to know the optimal length of the drying period to obtain biosolids which are spadable (minimum TS content of 20%), during the dry as well as the rainy season
- to know the quantity and quality of the biosolids and, based on these, the characteristics of a biosolids post-treatment system
- to know the (quantity and quality) of the percolate and, based on these, the characteristics of a percolate treatment system

3.2. Methodology

Pilot drying beds have been designed and built according to the lessons learnt from the experiment conducted in Accra between 1995 and 1997 (see Figure 2 for a sketch of the Kumasi drying beds).

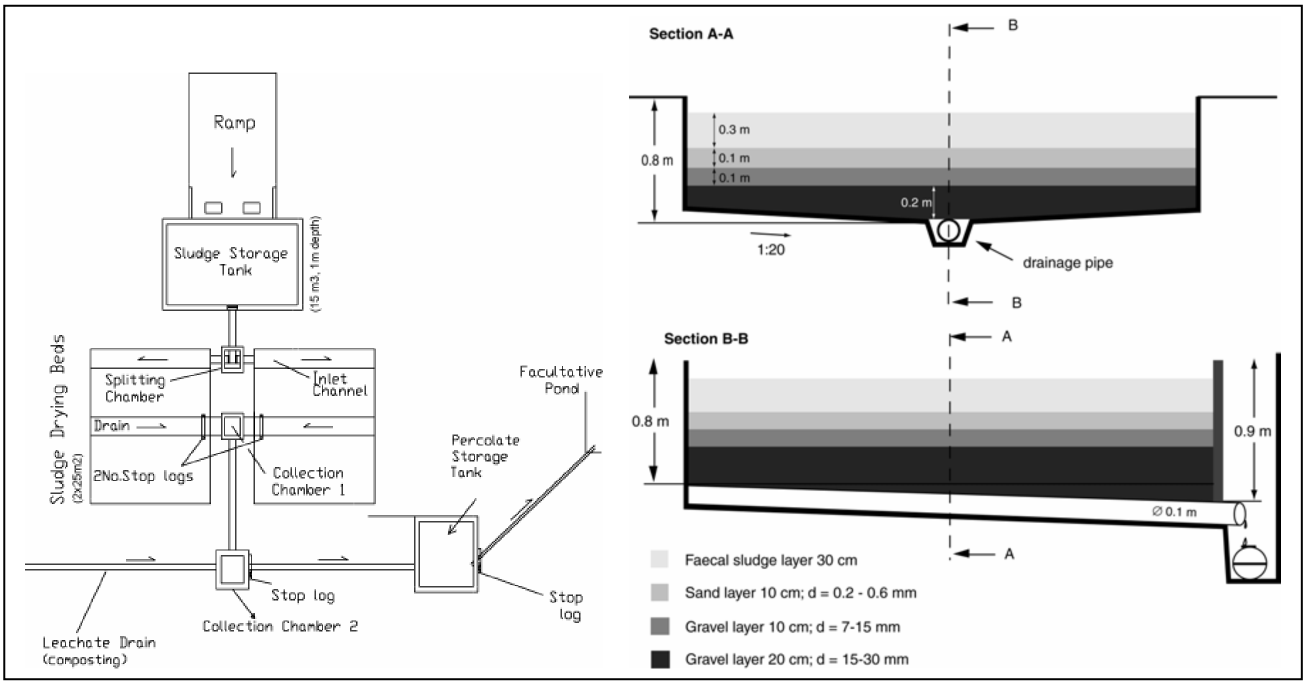


Figure 2 Sketch of the Kumasi pilot drying beds: ramp for sludge delivery, sludge storage tank, drying beds and percolate storage tank (after H. Esseku) as well as beds profile (initial design). Drying beds area: 2x25 m2

The plant is operated batch-wise. For each dewatering cycle, 3 trucks (each 5 m3 volume) discharge their FS load in the sludge storage tank. It was planned to carry out three dewatering cycles per month, corresponding to 45m3 FS/month or approximately 1,000 inhabitant equivalent. But the frequency of sludge loading had to be lowered (see chapter 3.3.1 Dewatering). Two of the three trucks were supposed to deliver septage whereas the third one public toilet sludge. This ratio could, however, not been respected (see also chapter 3.3.1 Dewatering). Dewatered sludge is desludged once it is dry enough to be shovelled out and is stored prior to co-composting with sorted organic solid waste. The percolate flows into an existing waste stabilisation pond system.

Sludge dewatering has been monitored during 4 cycles between February and June 2002: cycles 1, 3, 4 and 5. Cycle 2 could not be monitored because the filter layers had been damaged during the first cycle and sludge loaded at the beginning of cycle 2 could not be retained on the beds. For each cycle, three trucks have discharged their loads (septage and public toilet sludge) in the storage tank prior to drying beds loading. This is to allow characterisation of the FS mix. Raw sludge is sampled before loading onto the beds whereas dewatered sludge and percolate are monitored during the whole dewatering period. Figure 3 illustrates the location of the sampling points. Raw sludge volume was determined prior to loading onto the beds. The raw sludge sample consists of several sub-samples taken in the sludge storage tank directly after the three sludge loads have been thoroughly stirred. Dewatered sludge samples are taken every two to three days and at the end of the drying cycle. Three points are selected on each bed. The sludge at each selected point is stirred till it becomes homogenous (if the sludge is still wet), an equal volume is taken from each point, they are mixed together and a portion taken for analysis. Percolate samples are taken daily (composite sample of one day percolate flow) and kept in the fridge. Samples taken on the first day, the last day and a composite sample (from the whole percolating period) are analysed in the laboratory. The daily flow rate was also determined.

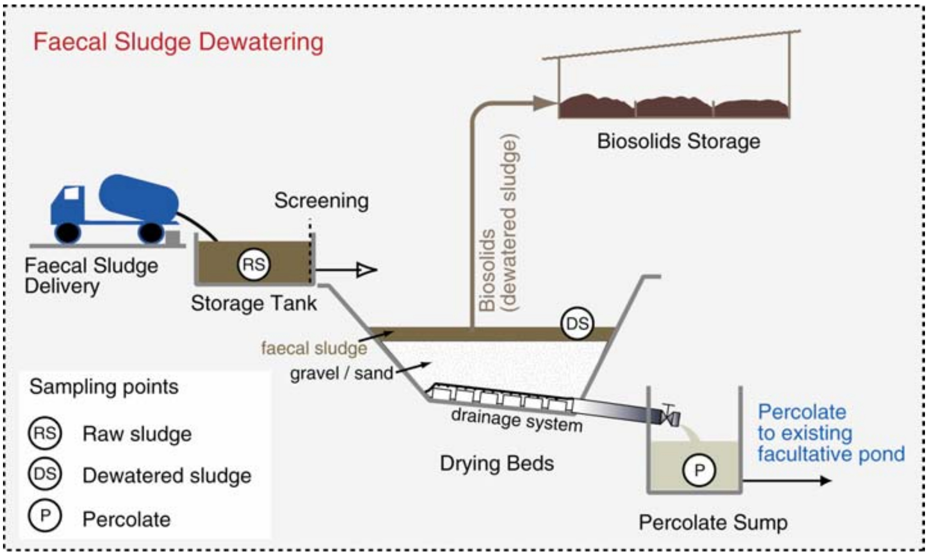


Figure 3 Faecal sludge dewatering – sampling points

COD, BOD, DBOD, SS, TS, TVS, HE, pH and conductivity were analysed in raw sludge samples (the detailed analytical procedures are given in Annex x). TS, TVS and TKN were analysed in dewatered sludge samples. Temperature, turbidity, conductivity, TS, SS, pH, DO, COD, BOD, DBOD, NH3, TKN, NO3, PO4, FC and HE were analysed in the percolate. The sampling frequency is summarised in Annex x.

3.3. Results and discussion

3.3.1 Dewatering

Drying bed design

The inlet distribution channel (figure 4) of the Kumasi pilot beds does not allow reducing the pressure flow sufficiently. The filter layers were badly damaged during the first cycle. In order to protect the filter during discharge onto the beds, splash plates were then placed under the inlet channel.

Figure 4
Inlet distribution channel (from the storage tank to the drying beds) and sludge loading

As a small sand layer is taken away of the filter when desludging the beds, the gravel and sand layer heights were modified as to the original design. The gravel layers were reduced from 30 to 25 and the upper sand layer was increased from 10 to 15 cm.

Necessary drying time – relevant factors

The following figure shows the decrease in sludge depth as well as the increase of the TS content measured in the sludge layer drying on the pilot drying beds.

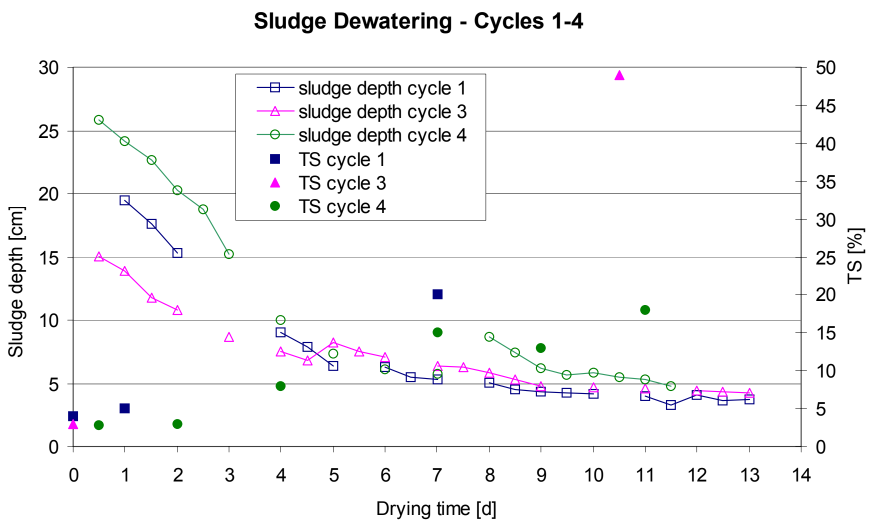


Figure 5 Decrease in sludge depth and increase in TS content measured in sludge drying on the drying beds – Results of dewatering cycles 1, 3 and 4 at the Kumasi pilot plant. Rainfall 8, 18, 111 mm for cycles 1,3 and 4, respectively.

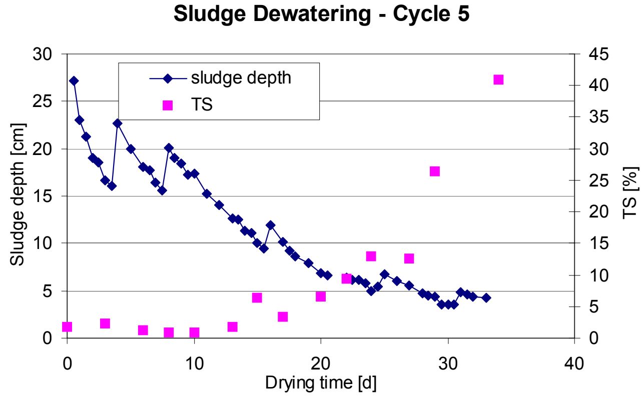


Figure 6 Decrease in sludge depth and increase in TS content measured in sludge drying on the drying beds – Results of dewatering cycle 5 at the Kumasi pilot plant. This cycle was characterised by heavy rainfall (218 mm)

Drying time observed during the Kumasi investigations, between 15 and 35 days, were much longer than the one estimated during the investigations conducted in Accra: around 8 days for a TS content in the dewatered sludge of 40%. But the Accra investigations were carried out during the dry season, only. Several factors are responsible for the slow dewatering. One important factor was the rain fall during desludging. Figure 7 illustrates the impact of rain on sludge dewatering on drying beds.

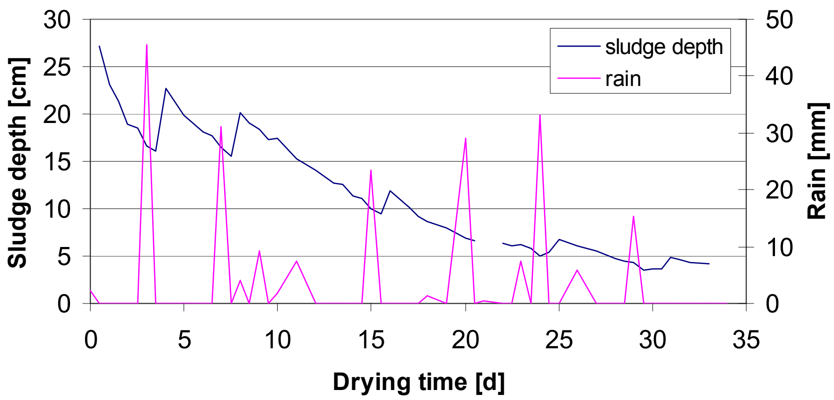


Figure 7 Sludge depth on drying beds during dewatering cycle 5. Impact of rain on sludge dewatering.

The fact that the amount of fresh, unstabilised sludge in the raw sludge mix is very high (cycles 2-5) also plays a role as unstabilised sludge does not lend itself to dewatering and does not drain rainwater efficiently. However, no correlation between the percentage of public toilet sludge in the raw sludge or the stability of the raw sludge (TVS) and the drying time can be established with the data obtained during the 4 dewatering cycles. Neither does the TS load correlate with the drying time.

Another important factor for sludge dewatering on drying beds is the characteristics of the sand. It was shown that the proportion of small sand particles (< 180 μm) in the sand used to form the upper filter layer of the Kumasi drying beds increases with time. It means that infiltration rate decreases and filter tends to clog. This could also explain why drying time increases continuously. However, the database is too scarce to determine whether the amount of rain, the change in sand characteristics or the high amount of public toilet sludge in the raw sludge loaded onto the beds is the determining factor for the drying time. It can be concluded that heavy rain, bad sand quality as well as a too high amount of public toilet sludge has lead to unreasonably long drying periods.

Cycle	Raw sludge volume	Raw sludge depth	TS load	Amount PTS	TVS	Rain	drying time
	[m3]	[cm]	[kg TS/m2*y]	[% vol. FS]	[%TS]	[mm/drying period]	[d]
1 (7.2-23.2)	12.8	26	321	38	65	8	16
2 (27.2-)	This cycle had to be stopped because of the damaged filter layers						
3 (13.3-28.3)	15	30	269	100	75	18	15
4 (8.4-20.4)	15.5	31	297	100	75	111	12
5 (23.4-28.5)	16.2	32	196	100	76	218	35

Table 5 Characteristics of raw sludge as well as quantity of rain, cycles 1 to 5. Sludge depth, TS load, percentage of public toilet sludge and hence degree of biochemical stability of the sludge as well as the quantity of rain potentially affect the drying time of sludge on drying beds. PTS: public toilet sludge

At the end of the 5^{th} cycle, a roof consisting of corrugated sheet placed on wood beams and slightly sloped so as to allow rainwater runoff was installed at the pilot plant. The roof is transported on the drying beds during rain events and during the night. It protects the biosolids efficiently but 2 persons are needed to install the roof on the drying beds. To improve the system, the roof could be installed on a rail or on wheels so that one person would be able to slide it easily on the beds when rain starts.

3.3.2 Biosolids Quantity and Quality

The average production of biosolids amounted to 1.5 m3 per cycle corresponding to 0.1 m3 biosolids per m3 raw FS (Table 6).

Helminth, notably nematode infections are highly prevalent in Kumasi. Among the pathogens causing gastro-intestinal infections, nematodes, *Ascaris* in particular, tend to be more persistent in the environment than viruses, bacteria and protozoa (Ingallinella et al. 2001). That is the reason why helminth eggs were chosen as indicators to determine hygienic quality of biosolids/compost and safety of biosolids/compost reuse in agriculture in the Kumasi co-composting project. Unfortunately, quantification of helminth eggs in biosolids could not deliver reasonable results during this first phase (cycles 1 to 5). Sample preparation for HE analysis as well as eggs count require considerable experience. No HE could be found in the percolate. The drying bed constituted an almost impermeable barrier for helminth eggs. It can be concluded that the eggs are therefore concentrated in the biosolids that need to be hygienised prior to reuse in agriculture. The subsequent co-composting should allow inactivation of the pathogens (see chapter 4). Several months of storage would also lead to hygienisation of the biosolids.

The TVS content varies between 42 and 72% of TS, the scarce database (3 data) does not allow drawing conclusions with regard to the degree of stability. In any case, the subsequent co-composting will allow stabilisation of the biosolids.

The N content of the biosolids amounts to 3% (of the TS content). The C/N ratio of the dewatered sludge was determined prior to mixing dewatered sludge with solid waste. Dewatered sludges produced during dewatering cycles 1 and 3 were mixed together prior to composting. This dewatered sludge mix was used for the first two composting cycles. The

C/N ratio was determined prior to these first two composting cycles and amounted to 29 and 27, respectively.

Table 6 Biosolids quality determined at the end of dewatering cycles 1 to 5

Cycle	Drying time	Dewatered sludge vol.	Biosolids production	Biosolids density	TS	TVS	TKN	N	C/N
	[d]	[m3]	[m3/m3]	[kg/m3]	[wt %]	[wt %]	[mg/kg]	[%TS]	[-]
1	16	1.7 1)	0.13		20				
2	This cycle had to be stopped because of the damaged filter layers								
3	15	1.7	0.14		50	50			
4	12	2	0.13		18	72	4,450	2.5	
5	35	0.9	0.06	700	41	42	13,050	3.2	
Average	**20**	**1.5**	**0.12**		**32**	**55**	**8,750**	**3**	**28**

1) Estimation based on the dewatered sludge depth on the beds prior to desludging

3.3.3 Percolate Quantity and Quality

Figure 8 shows percolate flow rate for dewatering cycles 4 and 5. Percolate flow starts between 10 minutes and 4 hours after raw sludge loading onto the beds. As those cycles were conducted during the rainy period, rain has a major impact on percolate flow. The impact is lower during cycle 5 as a tarpaulin roof has been installed on the drying beds after the second week of cycle 5. However the strong wind used to rip it off reducing its efficiency considerably. Percolate flow peaks follow rain peaks but are much less pronounced than the rain peaks.

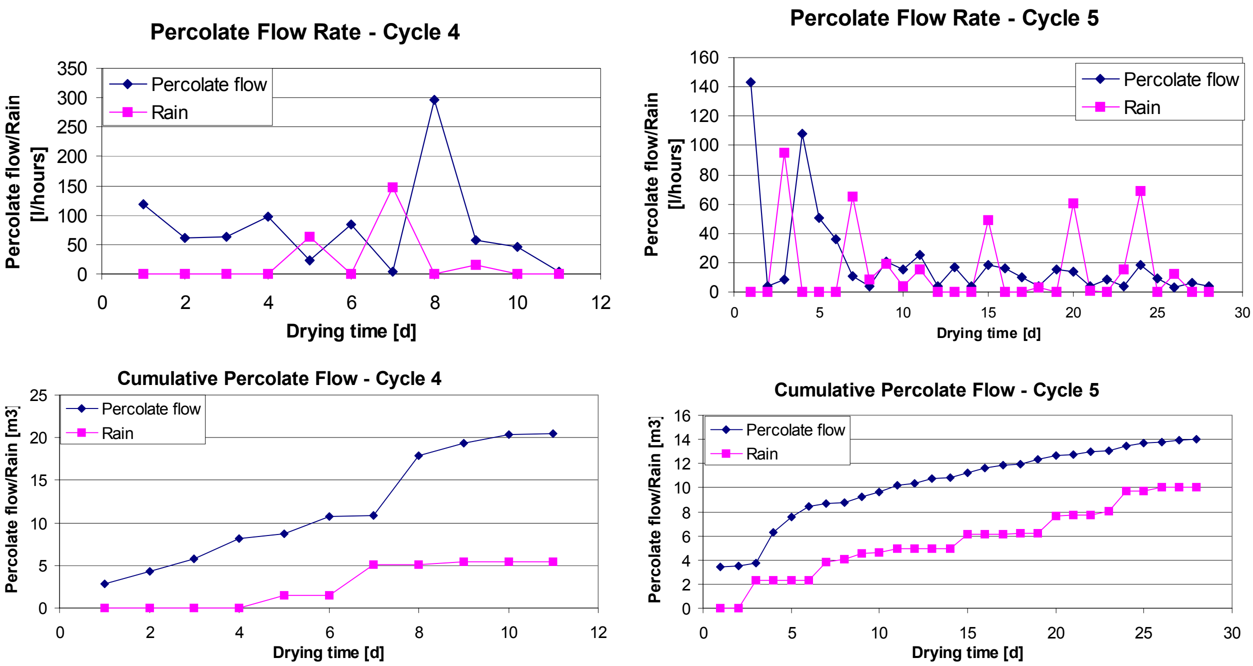


Figure 8 Percolate flow rate and cumulative percolate flow rates as well as rain as measured during dewatering cycles 4 and 5

The amount of rain in the percolate was estimated based on the beds surface area of 50 m2. This, however, does not allow the estimation of reasonable water balances over the drying beds. The main reason is that not only the amount of rain falling on the drying beds increase the percolate volume but also the rainwater flowing directly into the percolate storage tank. Another reason is that the weather station is not located at the treatment site. Local differences (local thunderstorms) could also explain why the water balances could not be established based on the available rain and percolate flow data. Heinss et al. (1998) states that from 50 to 80% of the FS volume loaded onto drying beds will emerge as drained liquid.

Concentrations measured in the percolate vary widely. Measures tend to indicate that the concentrations are higher at the beginning than at the end of the percolation (Table 7). This could be due to the fact that the percolate collected on the first day of percolation has a shorter retention time than percolate collected at a later stage of the percolating period. The percolate collected at the beginning is likely to have followed preferential pathways in the filter. It has therefore not been filtered as efficiently as percolate characterised by a longer retention period. Moreover, a shorter retention time also leads to a shorter contact time between percolate and micro-organisms present in the filter layers and that could contribute to the degradation of organic material. Another possible reason why percolate collected at the end of the percolating period is less concentrated is the dilution with rainwater.

Table 7 Raw sludge concentrations, percolate concentrations and drying beds removal efficiencies. The number of samples is written in parentheses. Composite samples of the daily percolate flow were taken each day and kept in the fridge. One composite sample was prepared with all daily composite samples taken during the percolating period. The first day, last day and the composite (whole period) samples were analysed.

	pH	DO [mg/l]	Turbidity [FAU]	Ec [mS/cm]	TS [mg/l]	SS [mg/l]	COD [mg/l]	BOD [mg/l]	DBOD[1)] [mg/l]
Raw sludge	7.9 (4)			17 (1)	27,495 (4)	12,467 (3)	35,982 (4)	10,250 (4)	9,250 (4)
First day	8.3 (4)	1.1 (4)	1926 (3)	23 (4)	5,761 (4)	363 (3)	3,659 (4)	1,100 (4)	988 (4)
Last day	7.9 (4)	1.6 (4)	257 (3)	13 (4)	4,034 (4)	140 (3)	1,775 (3)	563 (3)	390 (3)
Composite	8.4 (2)	1.6 (1)	642 (2)	7 (2)	4,155 (2)	375 (2)	2,218 (2)	560 (2)	380 (2)
Removal [%]					82 (4)	97 (3)	91 (3)	91 (3)	92 (3)

	NO3-N [mg/l]	NH3-N [mg/l]	Org. N [mg/l]	TKN [mg/l]	P [mg/l]	FC [no/100ml]	HE [no/L]
Raw sludge		1880 (1)		2195 (1)			
First day					80 (2)		
Last day	224 (1)	67 (1)	36 (1)	102 (1)	120 (1)		72 (1)
Composite	94 (1)	180 (1)	106 (1)	286 (1)	120 (1)	1.3E+05 (2)	0 (1)

1) The DBOD/BOD ratio of about 90% in the raw sludge is surprisingly high as organic matter is mainly concentrated in the solids fraction. This is likely due to an analytical error. This error also leads to an overestimation of the DBOD removal.

It is assumed that the high turbidity could have a negative impact on a subsequent pond treatment of the percolate as it reduces light penetration and hence photosynthesis and oxygen production. Because of the high conductivity, percolate cannot be used undiluted for irrigation. Conductivity should be lower than 3 dS/m (FAO, 1985). Percolate should therefore be diluted with river water prior to irrigation or be used for other purposes (e.g. brick construction) or be discharged into surface water after treatment. NH3 concentration is also variable but the average is under the limit concentration ($\leq$ 400 mg/l NH3+NH4-N, for pH<8-8.5) that can be tolerated by algae (Heinss et al., 1998). They could therefore develop in a subsequent facultative pond (in case light penetration is sufficient). The reduction of FC usually amounts to 1 log unit in such a filter (in the order of magnitude of 1×10^7 in raw sludge to 1×10^6/100 ml in the percolate). First analyses indicate a FC concentration of 1×10^5/100 ml. The FC concentration in the percolate is therefore still above the WHO guideline of $1x10^3$/100 ml for unrestricted irrigation (WHO, 1989). No HE was detected in the percolate except in one from 5 samples (72 eggs/l). The BOD/COD ratio amounts to 25-30%, that means an important fraction of the organic matter is not easily degradable.

It can be seen that the filter retains the solids efficiently (97% SS removal). The organic matter, mainly in the solids fraction, can also be reduced significantly. DBOD removal should not be so important (92%). The high removal is due to the fact that the DBOD concentration in the raw sludge is an overestimation. Approximately 85% of the TS load are retained in the biosolids whereas 15% flow out in the percolate. The removal efficiencies are similar as the ones determined during the Accra trials.

3.4 Conclusions and recommendations

According to the experience gained during the pilot experiment conducted in Kumasi, it can be recommended to increase the height of the sand layer from 10 (initial design) to 15 cm as a small layer is taken away with the dried sludge when desludging the beds. Sand should always be at the site in order to top up the sand layer that has been removed during desludging (1-2 cm). Particular care must be given to sand quality. Sand particles should have a diameter of 0.2-06 mm and not crumble. A crumbling of the sand particles would lead to a rapid clogging of the filter, making sludge dewatering ineffective. Different kind of sand should be tested prior to begin of operation in order to find an adequate type of sand. If an adequate type of sand is not available within a reasonable distance from the treatment site, another treatment option should be chosen (see Montangero and Strauss, 2002 for an overview of potential FS treatment options). Another option for sludge dewatering consists of settling/thickening tanks or ponds.

Another important factor is the reduction of pressure flow when loading sludge onto the beds. If this pressure is not sufficiently reduced, filter layers can be damaged considerably. In order to decrease the pressure flow, trucks should discharge their sludge loads in a stilling chamber. It is not necessary to construct a storage tank as it was used in this pilot project. The storage tank was built in order to make monitoring easier; it allowed the determination of the raw FS characteristics. The construction of such a tank in a full-scale, non-experimental plant would increase construction costs and operational requirements for the management of the solids settled in the tank. The stilling chamber should be followed by an inlet channel designed in such a way as so reduce the pressure flow. The inlet channel of the Kumasi drying beds is lightly sloped in the first ¾ of the length, then the slope evens up in order to slow down the sludge flow. Finally, the sludge flow should not fall directly on the filter but on a splash plate protecting the filter (see Box recommendations). In this experiment, wooden planks (15-25 cm wide) secured by heavy stones were placed below the inlet channels at one end of the drying beds (Esseku, 2002).

As also demonstrated in the Accra drying beds experiment (Heinss and Larmie, 98), undigested faecal sludge, as the one collected from public toilets, does not lend itself to dewatering. Drying beds are therefore not appropriate for the dewatering of public toilet sludge. This kind of sludge must first be stabilised through pre-treatment (e.g. anaerobic ponds or biogas digesters) or sufficiently "diluted" with partly stabilised faecal sludge such as septage prior to dewatering. Septage lends itself to dewatering. The liquid part contained in the septage rapidly infiltrates through the filter. Cracks rapidly form in the sludge layer on the top of the filter. If rain starts, rainwater can be drained through the cracks and infiltrates through the filter without increasing the drying time considerably. The combination of unstabilised sludge and rain leads to unreasonable sludge drying time on drying beds (several weeks). In this case, a roof could allow decreasing the drying time. However, a roof increases the cost of the plant and also the operational requirement – the roof cannot be fixed on the beds as evaporation is one of the mechanisms of sludge dewatering. Moreover, if the amount of unstabilised sludge is low enough (ratio public toilet sludge/septage low), the use of a roof should not be necessary as stabilised sludge can drain rainwater and is easily dewaterable. If the ratio is low enough, a dewatering time of less than 15 days should allow to obtain a spadable dried sludge layer (TS content $\geq$ 20%). More trials are necessary to determine the maximal amount of public toilet sludge that guarantee dewatering of sludge on drying beds within 10-15 days.

The depth of raw sludge loaded onto the beds should not be higher than 25-30 cm. Assuming that the sludge loaded onto the beds is characterised by a public toilet sludge/septage ratio of 1:2, the volume of sludge corresponds to 1.3 l/cap*d. For a sludge layer depth of 25 cm and a drying cycle of 15 days, the necessary area requirement is 0.08

m2/cap. This corresponds to a solids loading of 150 kg TS/m2*y. Solids loading rate on drying beds should amount to 100-200 kg TS/m2*y.

The level of decentralisation of FS treatment: the number of FS treatment plants and their respective capacity as well as their location in a given municipality should be defined in a strategic FS management plan as part of the city planning concept. Determining factors are, among others, the minimisation of distances between septic tanks, public toilets and treatment site on the one hand and between treatment site and biosolids users (farmers) on the other hand; economic aspects (economy of scale) and land use. However, such FS management plans usually don't exist. The scale of an FS treatment plant is hence given by the size of the piece of land that can be obtained for FS treatment (getting land for FS treatment is usually a difficult and very long process) and funds available for construction. Once the scale (total surface area) is defined, the number of drying bed units can be determined based on operational requirements and economic considerations.

Main operational and maintenance tasks consist in

- indicating to the truck drivers in which bed the sludge should be loaded (for a plant consisting of several beds operated in series),
- controlling that the defined septage/public toilet sludge ratio is respected,
- desludging the dewatered sludge
- storing it prior to hygienisation and reuse in agriculture
- refilling the sand layer on the top of the bed after desludging and
- cleaning the screen

Dewatered sludge (0.1 m3/m3 fresh FS) should be desludged as soon as it is spadable and be further treated (storage, further drying or co-composting with organic solid waste). Desludging the Kumasi pilot drying beds was done with shovels and is time consuming: 5 man-hours for a drying bed surface of 50 m2! For a full-scale plant, improved desludging mechanisms must be put in place, e.g. a grid/net on the filter.

Table x summarises construction, operation and maintenance costs for the Kumasi pilot drying beds as well as full-scale drying beds. Costs are expressed in function of the TS load treated (US$/t TS*year) as well as per capita and year (US$/cap*year) for a FS mixture characterised by a septage/public toilet sludge ratio of 2 to 1.

Table 8 Construction as well as O&M costs for the Kumasi drying beds as well as a full-scale plant (drying beds) (Steiner, 2002), see also Annex x for the details.

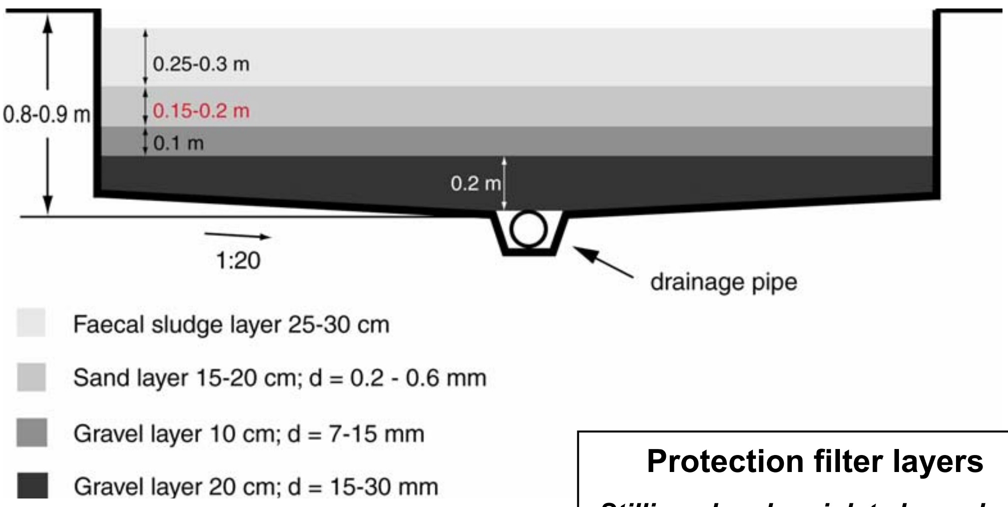
		Pilot plant (800 ie)	**Full-scale plant** (40,000 ie)
Capital cost	[US$/t TS*y]	35	16
	[US$/cap*y]	0.6	0.3
O&M cost	[US$/t TS*y]	50	31
	[US$/cap*y]	0.8	0.5
Total cost	[US$/t TS*y]	85	47
	[US$/cap*y]	1.4	0.8

Helminth eggs were identified as indicators of choice to determine whether biosolids reuse in agriculture is hygienically safe. Helminthic infections are prevalent in the region and among the pathogens causing gastro-intestinal infections, nematodes, *Ascaris* in particular, tend to be more persistent in the environment than viruses, bacteria and protozoa (Ingallinella et al. 2001). The drying beds act as a barrier for helminth eggs. No eggs were found in percolate samples. This means that HE are concentrated in the dewatered sludge layer. The number of eggs as well as their viability could not be determined in the biosolids during the Kumasi pilot project. As a consequence, dewatered sludge from the Kumasi drying beds must be hygienised prior to reuse in agriculture. This can be achieved through composting (high temperature), sun drying (water content less than 5%) or storage (eggs die-off with time). Biosolids cannot be composted without mixing them with a bulking agent (e.g. household organic waste or wood waste) as they are too dense to allow air circulation through the compost material. The bulking agent will allow forming a porous structure allowing air circulation and hence aerobic composting. C/N ratio of sludge dewatered on the drying beds has a C/N ratio of 28. An adequate C/N ratio of the compost mixture amounts to 20-35.

Drying beds retain suspended solids very efficiently, SS concentration in the percolate is reduced by 97%. The concentration of organic matter is also considerably reduced (90%) as it is mainly in the solids fraction. However, concentrations in the percolate are still high: 4,000 mg/l TS, 2,000 mg/l COD. The quality of the percolate can be compared with the one of (a rather concentrated) tropical wastewater. Percolate could be co-treated with wastewater. If co-treatment is not feasible (for example, if there is no wastewater or no wastewater treatment facilities), a percolate treatment system can be designed according to the design guidelines developed for wastewater treatment in the tropics. Laboratory batch experiments were conducted using 11 litres buckets to investigate anaerobic degradation of the percolate. Results indicate that the COD concentration can be reduced by 40% within 3 days. As expected, FC concentration could not be reduced during the anaerobic treatment (Seth, 2002). A possible percolate treatment could consist of a sand filter (reduction of turbidity, organic matter and NH3) followed by a series of pond (further reduction of COD and reduction of FC). This will reduce the organic, N and FC load in surface water. Stream water is usually used for domestic use. However, it will not allow a reduction of the conductivity. Treated percolate can therefore not be used for irrigation unless it is “diluted” with a less saline water.

Box recommendations

Drying bed profile



Raw sludge characteristics

⇒ Partly stabilised (septage or mixture septage/public toilet sludge with a low amount of public toilet sludge)

Sand characteristics

⇒ Sand particles do not crumble
⇒ Sand easily available locally

Reduction pressure flow

⇒ Stilling chamber, inlet channel and splash plates

Sizing of the beds

⇒ 15 days drying cycle
⇒ 25-30 cm sludge layer on beds
⇒ 100-200 kg TS/m2*y
⇒ 0.08 m2/cap

Drying bed removal efficiency

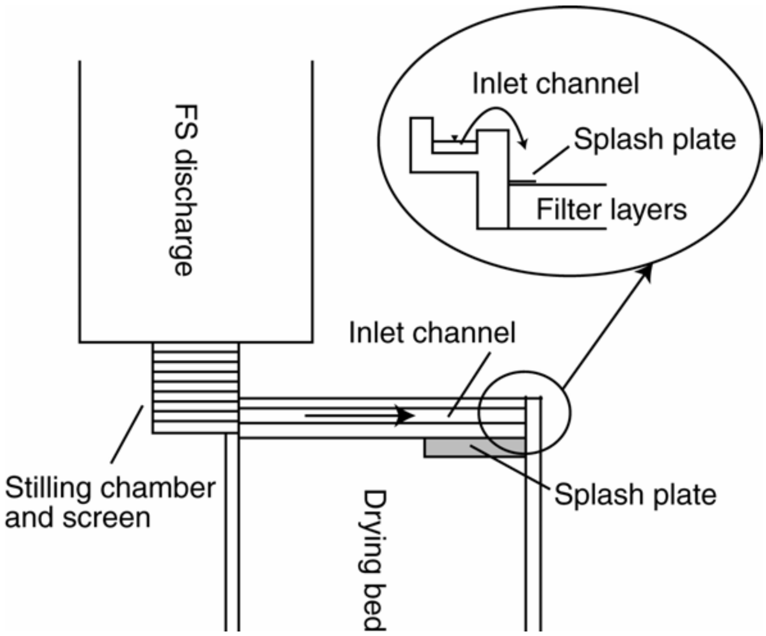
⇒ 97% SS, 90% COD, 100% HE, 40% NH4 (Accra)

Costs (full-scale plant)

⇒ Construction: 16 US$/t TS*y
⇒ O&M: 31 US$/t TS*y

Protection filter layers

Stilling chamber, inlet channel and splash plate



Biosolids

⇒ 0.1m3 biosolids/m3 fresh FS
⇒ Biosolids hygienisation necessary prior to reuse in agriculture (co-composting, sun drying or storage)

Percolate

⇒ 50-80% of raw FS volume
⇒ Quality comparable to tropical wastewater
⇒ Salinity too high for irrigation
⇒ Percolate treatment: e.g. waste stabilisation ponds or combination sand filter and wsp

4. Composting

4.1 Introduction

Household or market waste were delivered to the pilot plant, sorted, mixed with faecal sludge previously dewatered on the pilot drying beds and composted. The monitoring of the composting process aimed at:

- knowing the necessary composting/maturation time under the Kumasi conditions (climatic conditions as well as type of waste composted)
- knowing whether a dewatered sludge/solid waste mixing ratio of 1:2 or 1:3 (volume:volume) allow aerobic composting
- knowing whether helminth eggs (present in dewatered sludge) are inactivated during composting
- knowing the quality and the quantity of the mature compost
- knowing the costs of the compost production (see chapter 5 for an economic analysis)

4.2 Methodology

A pilot composting plant, consisting of a composting platform equipped with a drainage system and covered by a roof, was constructed within the framework of the co-composting pilot project. Solid waste is delivered to the pilot plant by the waste management department (WMD) of the Kumasi metropolitan assembly (KMA). Two plant workers sort the solid waste into easily biodegradable material, recyclables and reject waste using sticks and rakes. They are also responsible to make the compost heaps, turn and water them when necessary, and finally sieve and bag the mature compost. Inorganic waste (rejects) are transported to the landfill by the WMD. For access to water a well has been dug at the treatment site and is used by the workers for their personal hygiene and to water the heaps.

Two composting cycles have been monitored. Only one heap of 0.85 m3 was formed during the trial cycle whereas four heaps of 3 to 4 m3 were set up and monitored during the following first cycle. Dewatered sludge obtained during the dewatering cycles 1 and 3 (see Chapter 3) were mixed together and co-composted with solid waste in the two composting cycles. Sorted household waste was co-composted with dewatered sludge in trial cycle and both sorted household and sorted market waste were co-composted with dewatered sludge in the “first cycle”.

Temperature has been measured daily at several locations in the heaps (centre, bottom and top). Humidity was also measured daily using the squeeze test[5]. Samples were taken weekly for microbiological and physico-chemical (C, N, pH, moisture) analysis. From each heap, portions of compost are taken from the inner, outer, top, bottom and middle of the heap and mixed thoroughly before a sample is taken. A portion was then blended in order to obtain a homogenised sample prior to analysis. K, Ca, Mg, P, Pb, Mn, Cu, Zn, Fe were analysed in the compost samples taken at the end of the composting cycles. Chemical and microbiological analyses were carried out at the Soil Research Institute, Kumasi according to standard procedures (Annex x). Volume and weight of compost heaps were determined at the beginning and at the end of the composting.

[5] A handful of compost material is taken out of the heap and squeezed in the hand. Humidity is adequate if the palm of the hand gets humid but no drops of water flow out of the squeezed compost (Biocycle 2002).

4.3 Results and discussion

4.3.1 Solid waste sorting

Table 9 Solid waste sorting at the Kumasi pilot plant

Date	Type of waste	Vol. unsorted waste	Vol. sorted waste	Sorting time	Sorting time per sorted volume	Cycle/Heap	Heap volume
		[m3]	[m3 (%)]	[man-hours]	[man-hours/m3]		[m3]
21.2.02	HW	9	4.5 (50%)	70	16	Trial/(only 1 heap)	0.65
						Cycle 1/Heap 1	3*
29.3.02	MW	10	8 (80%)	60	8	Cycle 1/Heap 2	3
						Cycle 1/Heap 3	2
						Cycle 1/Heap 4	3
8.4.02	HW	8	4.5 (55%)	75	17		
26.4.02	MW	0.2	0.15 (75%)	0.5	3		
2.5.02	MW	4.5	3.5 (80%)	30	9		
10.5.02	HW			40			
15.6.02	MW	5	4.5 (90%)	14	3		

HW: household waste, MW: market waste

* A small amount was left, that's why the total (3 + 0.65) is less than 4.5 m3

Solid waste from two different sources were delivered to the treatment plant: household solid waste and market waste. Market waste is characterised by a higher amount of biodegradable waste. As a consequence, the percentage of sorted waste is higher in the case of market waste. The sorting process of market waste therefore needs less time than sorting household waste (16 and 6 man-hours/m3 sorted waste, respectively, Table 9). The sorting time of market waste varies between 3 and 9 hours per m3 of resulting biodegradable waste and this shows high variability. Rejects consist of inorganic material like (plastics), textiles, glass, pieces of metal, pottery, leather and organic material that do not decompose easily such as bones and pieces of wood. Sorting amounts to 30% (in average) of the operation and maintenance costs of the co-composting process (see chapter economic impact of co-composting). Possible measures to reduce sorting time are

- testing more efficient sorting measures (sorting table, etc.)
- source segregation (sorting at residential level) and separate collection

4.3.2 Compost heaps characteristics

Table 10 Initial characteristics of the compost heaps: quantities and ratio

Cycle		Dewatered FS		HW		MW		Compost (begin)		
		Volume (m^3)	Dry Weight (kg)	Volume (m^3)	Dry Weight (kg)	Volume (m^3)	Dry Weight (kg)	Volume Ratio	Volume (m^3)	Dry Weight (kg)
Trial		0.2	-	0.65	-	-	-	3:1 (HW:DFS)	0.85	500
Cycle 1	Heap 1	1	500	3	1600	-	-	3:1 (HW:DFS)	4	2100
	Heap 2	1	500	-	-	3	1200	3:1 (MW:DFS)	4	1700
	Heap 3	1	500	-	-	2	800	2:1 (MW:DFS)	3	1300
	Heap 4	-	-	-	-	3	1200	MW (control)	3	1200

Table 11 Initial characteristics of the compost heaps: quality

Trial	C	N	C/N
	[%dry weight]	[%dry weight]	[-]
HW	37	1.1	35
DFS	6	0.2	29
HW/DFS (3:1)	9	0.4	23

First Cycle	Humidity	C	N	C/N
	[%]	[%dry weight]	[%dry weight]	[-]
MW	69	19	0.6	33
HW	50	5	0.2	36
DFS	42	6	0.2	27
(1) 3:1 (HW:DFS)	49	4	0.1	32
(2) 3:1 (MW:DFS)	60	10	0.3	30
(3) 2:1 (MW:DFS)	53	3	0.1	28
(4) MW (control)	69	19	0.6	32

Table 11 shows that N in DFS is in similar range as in solid waste. Nitrogen dynamics, especially in the dewatering process, need to look at further to understand the processes leading to such low values.

4.3.3 Temperature changes during composting

“Trial Cycle”

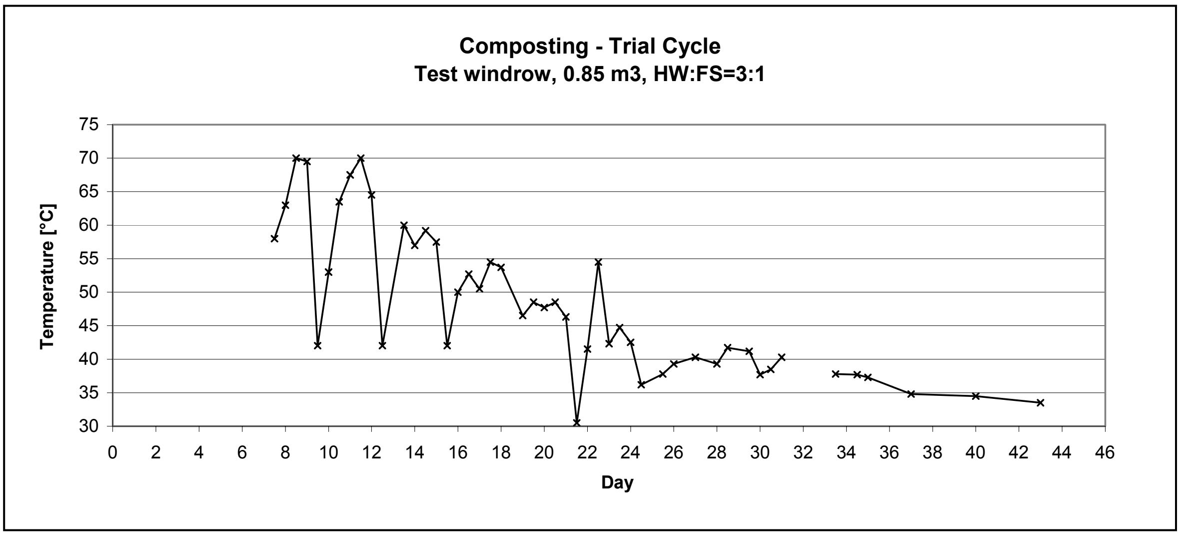


Figure 12 Temperature development during the trial cycle (temperature measured in the centre of the heap)

Figure 12 shows that the thermophilic phase (50-70°C) lasted about three weeks and was followed by a maturation phase (temperature around 40°C) of three more weeks. Although temperature data in the first week is missing, high temperatures up to 70°C can be observed also during the second week. Drops in temperature are due to the turning of the heaps. Heaps were turned at a frequency of two to three days during the thermophilic phase. The frequency was then reduced to twice weekly and finally weekly. Temperature then approached the ambient temperature conditions. To allow interaction of pathogens, the entire compost mass must be maintained at a minimum of 65°C for 2 to 3 consecutive days (Hoornweg et al., 2000). In the second and third week, the heap has been turned about 5 times. Thus, it can be concluded that all parts of the heap have been heated sufficiently and that pathogens have been inactivated. The temperature patterns are typical of a well functioning composting. Even though no data on final compost quality are available, the temperature development indicates that compost matured.

“First cycle”

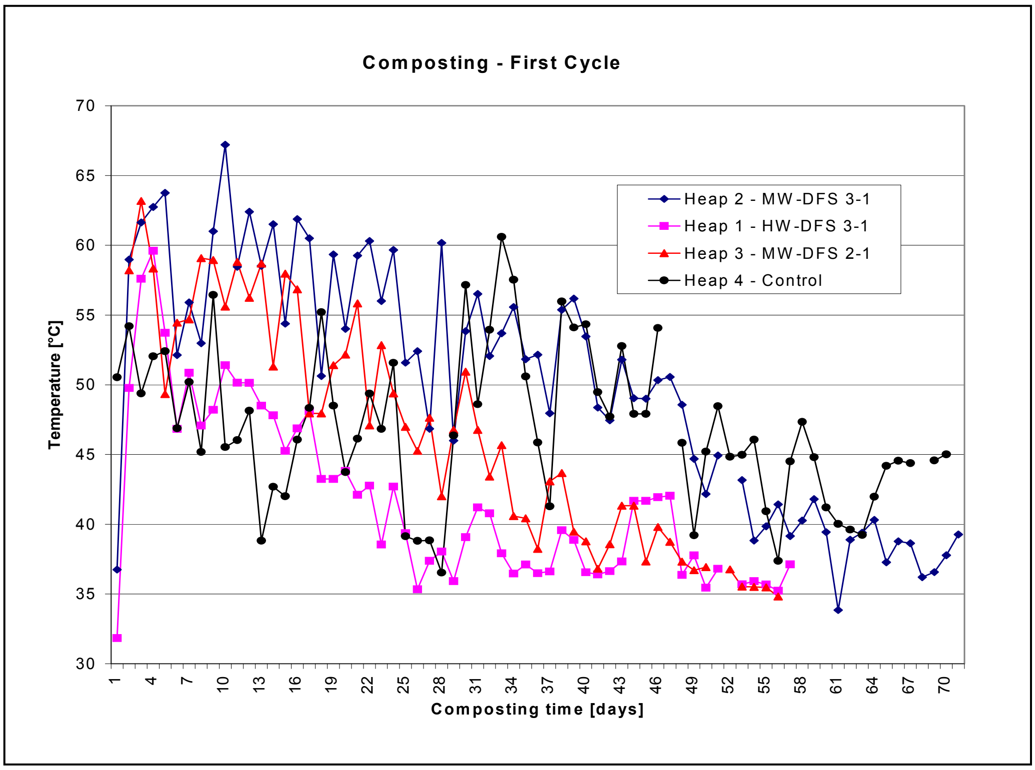


Figure 13 Temperature development during the first cycle (temperature measured in the centre of the heaps)

Temperatures vary greatly and patterns are not as clear as in the “trial”. Temperatures measured during the “first cycle” did not reach as high levels as during the “trial”. With the exception of the heap 1, temperatures have been higher than 50°C for 4 weeks at least and it can be assumed that pathogens have been inactivated. Total composting periods are longer than during the trial:

		Thermophilic phase (weeks)	Maturation phase (weeks)
Heap 1	**3:1 (HW:DFS)**	4	4
Heap 2	**3:1 (MW:DFS)**	7	> 4
Heap 3	**2:1 (MW:DFS)**	5	3
Heap 4	**MW (control)**	7	> 4

Heap 4, consisting of solid waste only, has the highest temperature after 70 days. The slower composting of this heap could be due to the fact that the content of easily available nitrogen is lower in solid waste than in sludge and/or particle sizes of solid waste are larger and thus delay degradation. The higher levels of DFS in the heap 3 do not hinder the composting process. However, more data are needed to determine whether there are significant differences between heaps characterised by a SW/DFS ratio of 3:1 and 2:1.

4.3.4 Moisture content

Table x shows the quantity of water added to the compost heaps while setting up the heaps as well as during the composting process.

Table 14 Water consumption during the composting process

		Initial humidity	Dry weight raw compost	Water added making the heaps	Water added during composting	Total	Total per t compost
		[%]	[kg]	[l]	[l]	[l]	[l/t]
Trial	**3:1 (HW:DFS)**	49*	480	n.d.	n.d.	210	438
Cycle 1	**3:1 (HW:DFS)**	49	2060	280	610	890	432
	3:1 (MW:DFS)	60	1670	350	620	970	581
	2:1 (MW:DFS)	53	1280	280	420	700	547
	MW (control)	69	1170	(moistened by rain)	525	525	449

*assumed

The average water added is 490 litres per ton raw compost (dry weight).

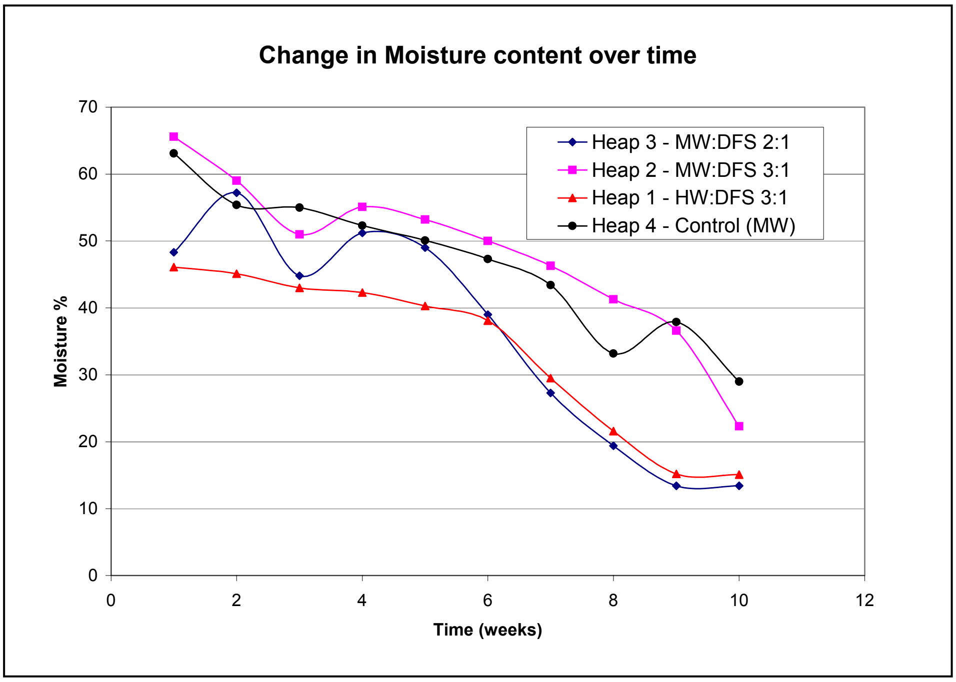


Figure 9 Changes in water content during the first cycle

Figure 9 illustrates the drop of water content in the heaps after watering of the heaps had been stopped (after the thermopihilic phase, after 6 weeks). The water content drops continuously during the maturation phase to as low as 15-30%.

4.3.5 Nitrogen

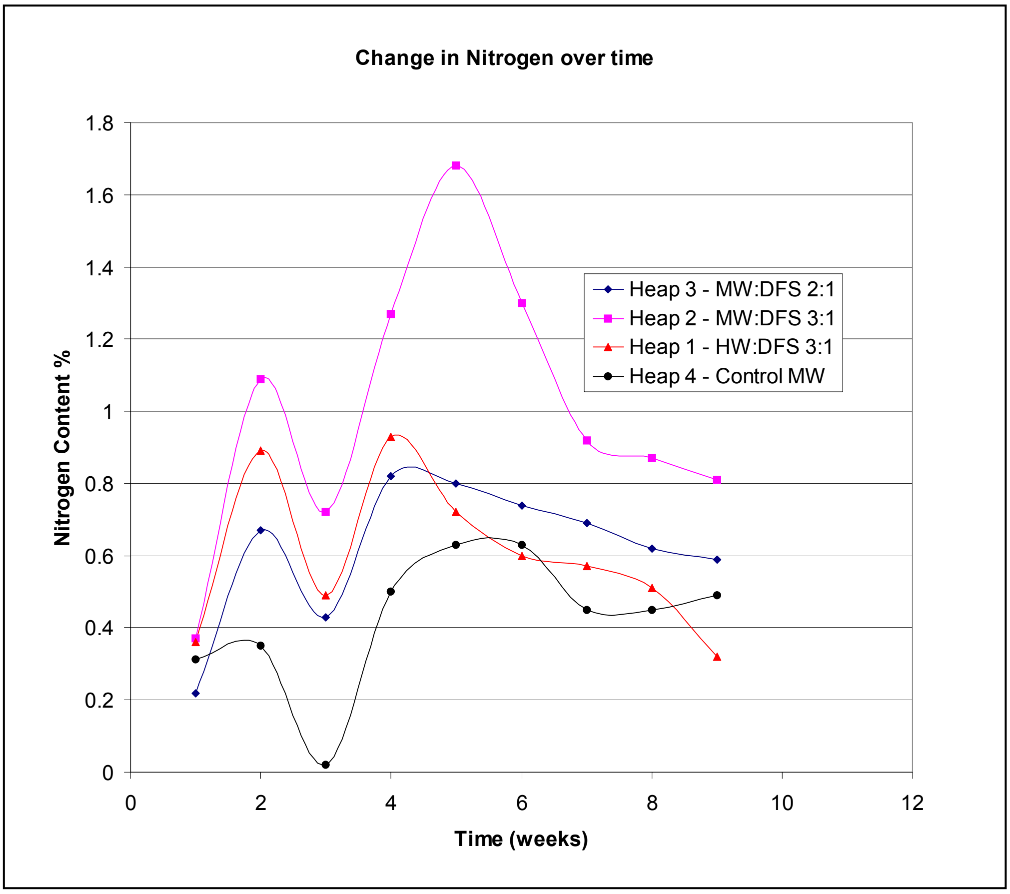


Figure 10 Nitrogen changes during the first cycle

It was expected that heaps containing dewatered sludge would have a higher N content than the control heap because of the high N content of human excreta (10%). However, figure 10 shows that the initial N contents of the 4 compost heaps are all similar. This could be caused by nitrogen losses during sludge dewatering.

Increase in N content from week 1 to week 2 as well as decrease from week 2 to week 3 is likely due to systematic analytical errors. If no nitrogen is lost during composting, the N content should theoretically increase due to loss of mass during the composting process. However, N losses, especially at the beginning of the thermophilic phase as ammonia nitrogen has not yet been mineralised and the pH is high (see Figure 12), are expected to occur. The slight decreasing trend observed starting at the week four could indicate that N losses do occur during the composting process. Determination of the total nitrogen mass in the input material as well as in the mature compost did not deliver reasonable results. More N data (total N, NH3, NO2, NO3) as well as precise weight determination of the heaps at the beginning and at the end of the composting are needed to estimate N balances over the composting process.

Concentrations at the end of the composting process are in the range indicated by Gotaas (1956): 0.4-3.5% dry weight, but rather at the lower side of the range.

4.3.6 C/N ratio

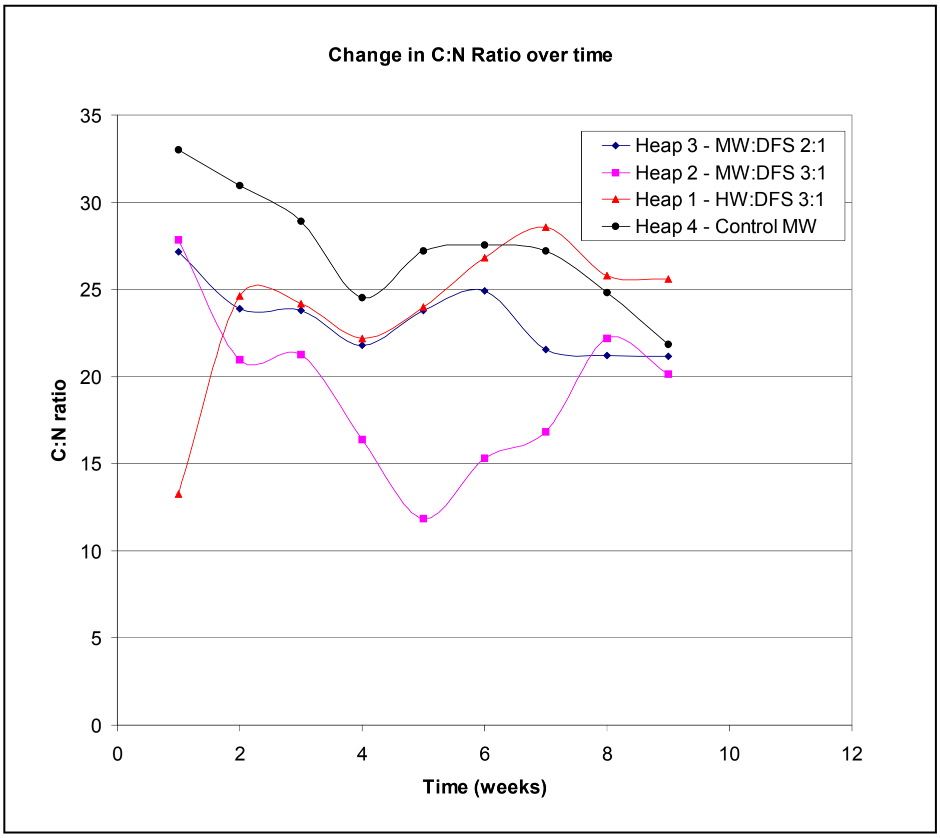


Figure 11 Change in C/N ratio during the first composting cycle

The decreasing trend reflects the carbon losses due to decomposition (production of CO2). C/N ratios are in the expected range, but the values measured at the end of the composting are rather high. Hoornweg et al. (2000) states that the C/N ratio of the final product should be lower than 22. However, temperature patterns indicate that compost maturation has been reached.

4.3.7 pH

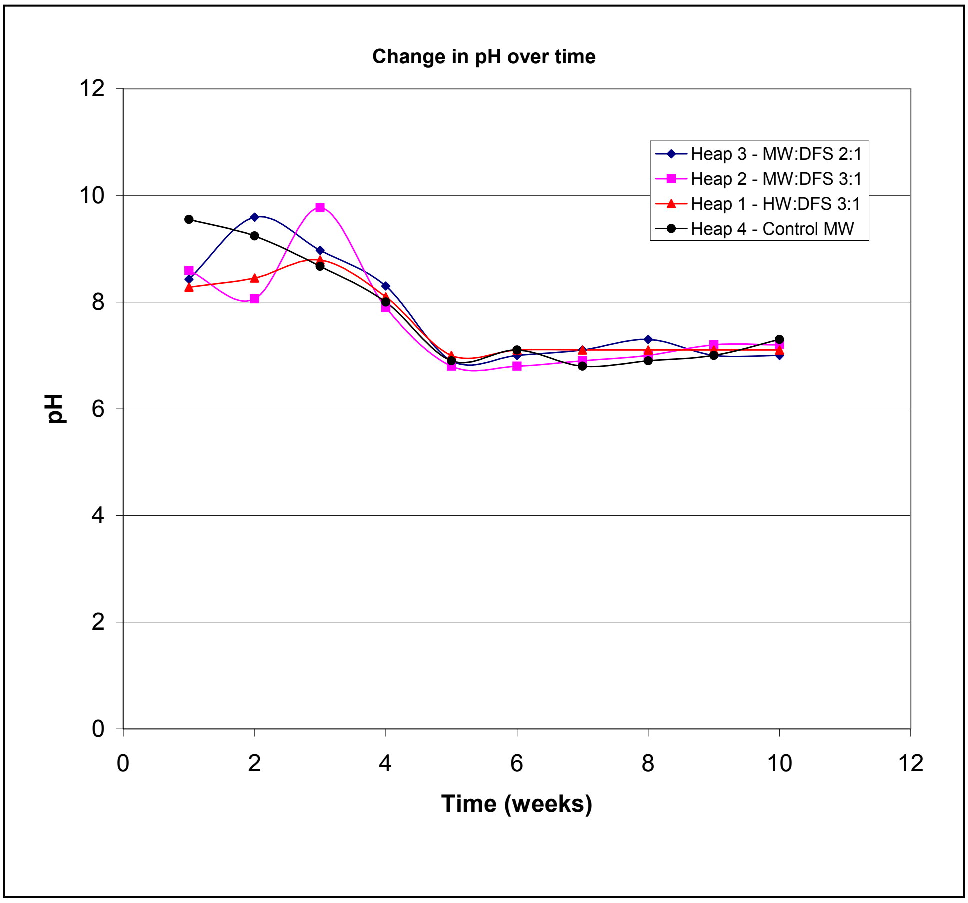


Figure 12 Change in pH during the first composting cycle

The high pH at the beginning of the composting is due to the high amount of ammonia nitrogen in the heaps. Ammonia compounds are then nitrified to nitrate causing the pH to sink. pH becomes neutral and stable from the fifth weeks of composting, corresponding to the end of the thermophilic phase. Compost characterised by a neutral pH is well tolerated by plants.

4.3.8 E.Coli

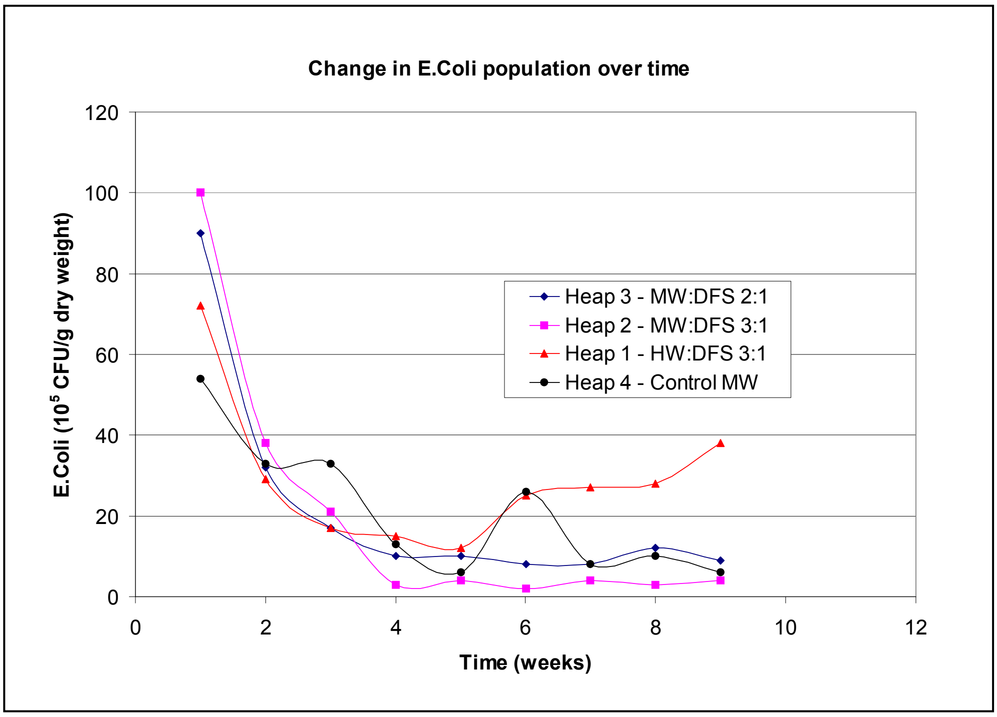


Figure 13 Change in E.Coli during the first composting cycle

The concentration decreases exponentially during the thermophilic phase then remains more or less stable. Total bacteria, fungi and clostridia were also analysed. Results indicate a similar behaviour as E.Coli. However, concentrations are surprisingly high as Escherichia coli should be inactivated within one hour at 55°C. Nematode eggs should be analysed in order to determine the hygienic quality of compost and assess the health risk of reuse. This parameter should be monitored in further composting cycles.

4.3.9 Compost quantity

Table 15 Compost quantity and reduction of compost volume and weight during composting

Cycle			Begin		End of maturation		Reduction		After sieving	Total reduction
			Volume (m^3)	Weight (kg)	Volume (m^3)	Weight (kg)	Volume (%)	Weight (%)	Weight (kg)	Weight (%)
Trial		3:1 (HW:DFS)	0.86	480	0.46	350	47	28	300	38
Cycle 1	**Heap 1**	3:1 (HW:DFS)	4	2060	2.2	1750	45	15	-	-
	Heap 2	3:1 (MW:DFS)	4	1670	1.5	1500	63	10	920	45
	Heap 3	2:1 (MW:DFS)	3	1280	1.3	830	57	35	750	41
	Heap 4	MW (control)	3	1170	1.1	670	63	43	-	-

The average volume reduction amounts to 55%. Hoornweg et al. (2000) mentions that mature compost should reduce volume of raw organic material by at least 60%. Weight reduction amounts to 26% in average. This is in the same order of magnitude as the weight reduction determined by Rytz (2001) at a composting scheme in Bangladesh. Weight reduction is due to loss of dry organic matter through decomposition (volatilisation).

The weight is not significantly reduced during the sieving of the compost. The compost contains a low amount of large pieces that do not pass through the sieve.

4.3.10 Compost quality

Table 16 Compost quality. Samples were taken at the end of the composting (after 8 and 9 weeks of composting). Results of "first cycle"

		C	N	C/N	K	Ca	Mg	P
		[% DW]	[% DW]	[-]	[% DW]	[% DW]	[% DW]	[% DW]
Heap 1	3:1 (HW:DFS)	13	0.6	21	0.2	0.9	3.2	0.4
Heap 2	3:1 (MW:DFS)	18	0.8	21	0.4	1.5	3.8	0.4
Heap 3	2:1 (MW:DFS)	11	0.4	26	0.2	4.2	1.9	0.1
Heap 4	MW (control)	11	0.5	23	0.5	2.8	2.6	0.2

		l.	Mn	Cu	Zn	Fe
		[ppm]	[ppm]	[ppm]	[ppm]	[ppm]
Heap 1	3:1 (HW:DFS)	262	58	35	140	1036
Heap 2	3:1 (MW:DFS)	99	71	39	124	504
Heap 3	2:1 (MW:DFS)	342	68	41	51	1078
Heap 4	MW (control)	1721	69	17	84	2953

Table 17 Proposed standards for MSW compost in developing countries (Hoornweg et al. 2000)

Heavy metals	Proposed standards [ppm]
Arsenic	10
Cadmium	3
Chromium	50
Copper	80
Lead	150
Mercury	1
Nickel	50
Zinc	300

Concentrations are in the expected range. As already mentioned, N content is lower than expected. The comparison of the heavy metals concentrations measured in the compost with standards proposed for developing countries reveals that only lead exceeds the proposed standards. Lead concentrations in heaps 1, 2 and 3 are in the range mentioned by Brunt et al. (1985) quoted by Waas et al. (1996) of 200-400 mg/kg dry weight whereas the concentration in heap 4 is higher (1721 mg/kg). High Pb concentrations can be due to atmospheric deposition (fuel) or by contamination by household toxic waste such as batteries.

There is no significant difference between the nutrient content of the different heaps containing only market waste (heap 4), market waste and faecal sludge (heaps 2 and 3) and household waste and faecal sludge (heap 1). Moreover, the heap containing household waste is not more contaminated by heavy metals than the others.

4.4 Conclusions and recommendations

Results obtained during the monitoring of two composting cycles indicate that the necessary composting time to obtain mature compost under the Kumasi conditions (climate, type of wastes, turning frequency) amounts to about 9 weeks. The thermophilic phase lasts 4 to 5 weeks. Heaps were turned every two days during the thermophilic phase, twice weekly and weekly during the subsequent phase. It is questionable whether such a high frequency is necessary. It increases the production cost (turning of the heaps) and also water and possibly also N losses. Further composting cycles should be monitored in order to determine the appropriate turning frequency under the Kumasi conditions.

Helminth eggs could not be monitored. However, temperature patterns indicate that all parts of the heaps have been exposed to high temperatures during a sufficient long period so as to guarantee pathogens inactivation.

Data obtained so far are scarce but they tend to indicate that there is no significant difference between the heap characterised by a SW/DFS ratio of 3:1 and the one by a ratio of 2:1. The ratio of 2:1 allows the mixing and hence the treatment of a higher realtive amount of faecal sludge and should be recommended.

It could further be observed that there is no significant difference between heaps containing market waste and heaps containing household waste. The nutrient content in both types of compost is similar. Compost produced with household waste is not more contaminated by heavy metals than market waste compost except by lead. However, more data are needed to determine statistical differences between the different kinds of compost. The only difference observed is the longer time needed (and hence higher cost) for household waste sorting than for market waste sorting.

Dry weight reduction during composting amounted to 24%. To produce 1 ton of compost (dry matter), 1.3 ton of input raw material are needed. If a mixing ratio of SW:DFS of 2:1 is chosen, 1.7 m3 sorted solid waste and 0.9 m3 dewatered faecal sludge are needed. This corresponds to approximately 2 ton of unsorted household waste and 10 m3 fresh faecal sludge. Total water volume added during composting corresponds to approximately 500 litres per ton raw compost. Compost quality measured in the pilot project is in the expected range. Plant trials will allow to demonstrate the effect of compost on soil and plant as well as spread knowledge about this type of organic fertiliser/soil conditioner among farmers in Kumasi.

5. Economic aspects of co-composting

5.1 Capital and O+M cost

The construction costs of the plant listed in Table 18 includes the ramp for vacuum trucks; a sludge storage tank (15 m^3); two parallel drying beds; a sludge storage area; a solid waste delivery, unload and handling area; a composting area (for composting, maturation, screening, bagging and compost storage); a closed building and a percolate storage tank.

Table 18 Distribution of construction costs for the co-composting pilot plant Buobai, Kumasi. (Colan Consult: Quotation from 11/2001)

Item	[US $][1)]	[%]
General items	5,500	24
Site clearance	50	0.2
Discharge bay	750	3
Sludge storage tank	1,250	6
Pipe work and splitting chamber	280	1
Sludge drying beds	2,250[2)]	10
Solid waste handling area	1,300	6
Composting area	4,150	18
Sludge storage area	70	0.3
Roofing materials	2,550	11
Percolate storage tank	1,300	6
Daywork[3)]	1,300	6
Contingencies	1,950	9
Total	**22,700**	100

1) Original prices in Cedi (7,400 Cedi = US$ 1, November 2001)
2) Original value of US$ 1,100 was modified by the author to US$ 2,250 (more concrete and reinforcement steel than quoted)
3) Contingencies for materials and contractor's equipment, overhead and profit.

Regarding cost composition, it appears that FS solids-liquid separation constitutes about 20% (without percolate storage tank, which is needed only for possibly irrigation of compost heaps) and co-composting about 35% of total expenses, while the rest is attributable other general expenditures. An important expense under general item constitutes the establishment of the site office that amounts to US$ 4,300. Concrete and reinforcement steel is causing an expensive composting area. Important outlays for the roofing material to cover the composting area is useful to protect windrows in order to limit leachate during wet season and evaporation during hot season.

Land price and percolate polishing are not included, because latter is either used to moisten the compost heaps or treated in the stabilisation ponds of the Buobai full-scale treatment plant. Land belongs to the Kumasi Metropolitan Assembly and its price was about US$ 0.5 per m^2 (Annoh 2002). Thus, with an approximate land requirement of 500 m^2, land expenditure of the pilot plant would amount to US$ 250 corresponding to 1% of investment cost. Post-treatment of the liquid effluent issue from drying beds and possibly from the composting process in waste stabilisation ponds would create additional cost in same order of magnitude as the drying beds itself (about 10% of total costs of the co-composting plant) according to Steiner (2002).

In addition to investment costs, operation and maintenance (O+M) of the faecal sludge treatment and co-composting plant cause expenditures, which occur regularly. Table 19 shows yearly O+M cost estimates based mainly on time observations by Quarshie (2002) during fieldwork on the pilot plant. Note that properly maintenance costs for general repairs will depend on quality of construction, care and dealing of the plant. Thus, difficult to estimate in which extent repairs will be needed.

Table 19 Composition of the O+M costs of co-composting in Buobai based on man-hour monitoring from Quarshie (2002). For details, please refer to appendix 6.1.

Item	[US $]	[%]
Sludge removal[1)]	100	6
Replenishment of sand[2)]	75	4
Compost turning[1)]	300	17
Waste sorting[1)]	525	29
Compost screening and bagging[1)]	100	6
Salary management[1)]	500	28
Contingencies	200	11
Annual O+M cost	**1,800**	100

[1)] field experience after Quarshie (2002) and Cofie (2002), excluding solid waste delivery and transport of remaining solid waste to landfill by the municipality
[2)] estimated by the author, working time and equipment included

Regarding to the above O+M costs, sorting of solid waste is the most time consuming and hence cost intensive of all operations on the co-compost station. According to the first experiences reported by Quarshie (2002), the sorting cost amounts to a range of US$ 2.8 – 3.8 per m^3 of sorted organic waste depending of the organic content of initial waste. Rejected material of initial waste load was 49% and 18 % for unsorted household waste and partially sorted market waste, respectively (Quarshie 2002). Hence, market waste with higher organic content is less labour intensive to sort than domestic waste.

Cofie (2002) reported current running cost of the pilot plant to 2,000,000 Cedi per month. This correspond with about US$ 2,850[6] of O+M cost per year. This amount is notably higher than estimates in Table 19 due to its research destination. Because an important amount is imputable to the monitoring program (sampling, transport, measurement of weight and volumes) and to salaries of two fulltime workers which operate the co-composting station during six days a week.

In order to compare capital and O+M cost, capital cost were annualised assuming an interest rate of 5% and a depreciation period of 15 years (formula in appendix 6.2). Annualised capital and O+M costs expressed per ton TS of FS and per capita are shown in Table 20. Assuming a yearly sludge charge of 12.5 ton, a sludge mixture of 2:1 (septage : public toilet sludge) and a daily sludge load of 14 g TS (septage) and 100 g TS (PT) per capita respectively (Heinss et al. 1998), the pilot plant treats the faecal sludge of about 800 persons.

[6] US$ 1= 8,400 Cedi, August 2002

Table 20 Annual capital and O+M costs of the Buobai co-composting plant in Kumasi. Solids-liquid separation with the help of drying beds.

	Annual cost [US$]	Cost per capita[3)] [US$]	Cost per t TS FS [US$]
Capital cost[1)]	2,187	2.7	**175**
O+M cost[2)]	1,800	2.3	**144**
Total costs	3,987	5.0	**319**

1) without land cost and percolate polishing ponds
2) ignoring potential revenue from biosolids sale
3) plant capacity ~ 800 PE (1 PE = 14 g TS/day per capity)

It is remarkable that O+M costs are almost as high as capital costs. This might be a characteristic of a low-cost option, where hardly mechanical installations or sophisticated machines are installed, but work is entirely accomplished by time intensive manual working.

5.2 Economy of scale

Investment and running cost provided in 5.1 are only valid for the pilot plant of quite small size. Hence, there is a considerable potential of economy of scale when enlarging the plant size, because its price does not rise in the same extent as its capacity. For instance, even when more waste and FS is treated, one discharge bay and one site office will still be sufficient. There is a potential of economy of scale for proper treatment items, too. For example volume of a tank gives its capacity, while its cost is determined by the concrete surface, which does not rise linearly with the volume. Economy of scale of an installation is expressed mathematically by the so-called law of economy of scale (Maystre 1985), reported in appendix 6.3.

In order to estimate the potential of the economy of scale, two co-composting plant of the same type as the pilot plant were designed for a capacity of 125 t TS and 625 t TS per year respectively. Note that no sludge storage tank was previewed for the up-scaled plants due to direct loading of drying beds. Then, the up-scaled co-composting plants were quoted on the base of the pilot plant quotation. Results showed that average specific capital cost of initial US$ 175 to US$ 80 and US$ 60 per ton TS of FS respectively. Specific O+M cost were estimated to decreases only slightly, due to the fact that man-hours were utilised to calculate them and not monthly salaries. Figure 14 illustrates results of the potential of economy of scale.

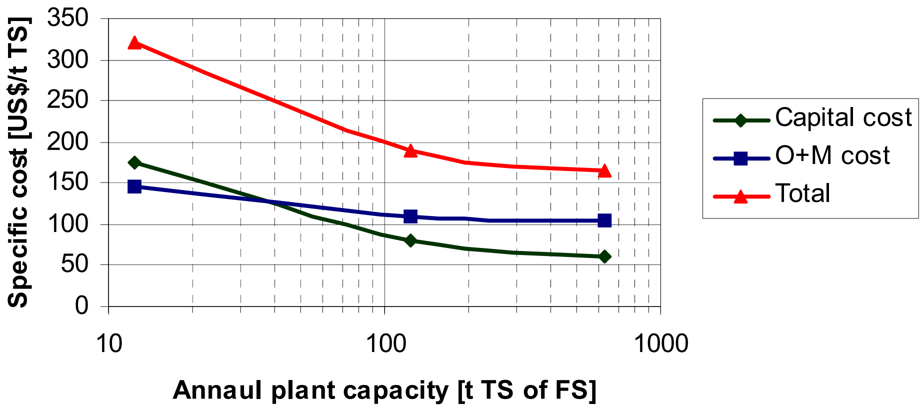


Figure 14 Economy of scale: Specific cost in function of plant size.

Considering the important potential of economy of scale, a logical consequence would be now to adopt large treatment plants. But also a few reflections listed below favour small decentralised co-composting plants. It is challenging to work out economical and functional optimum somewhere between an ultra small pilot scale and a huge full-size plant.

- With increasing plant size, the catchment area of the FS does increase, too. Hence, long distances of transport and time needed to get to the plant might be uneconomically high and encourage indiscriminate dumping of untreated FS.
- Up-scaled plant, which treats for instance 625 t TS of FS per year, would produce approximate 5,000 m^3 of compost. It will be very difficult to sell this quantity in a reasonably perimeter. And transport cost would be high probably due to long distances to buyers.
- When large quantities of solid waste have to be delivered, trucks are needed and its transport is expensive, while e.g. small entrepreneur (carts with animal or human traction) can deliver also decentralised plants.
- Investment for construction of a large plant is high and could be out of reach of disadvantaged countries. It often is easier to invest several small amounts than once a big sum.
- Planing horizon of a FSTP usually is rather small (e.g. 5-15 years) in developing countries because local conditions change quickly and capital is scarce. Therefore treatment capacity expansion is preferable to do at small increments.

5.3 Economic benefits

5.3.1 Compost sale

The cost could be reduced considerably when biosolids could be sold at reasonable prices. The compost production of the pilot plant amounts approximately to about 100 m^3/yr (50 m^3 dewatered sludge and 150 m^3 organic waste, volume reduction of 50% during composting period). With an expected sale price[7] of about US$ 5 per m^3 of compost, annual revenue

[7] According to Annoh (2002), a fertiliser bag of compost (50 kg) would cost US$ 1, one ton of compost US$5 and a tipper truck (5 m^3) full of compost, transport included, would cost US$ 40, based on a small willingness to pay poll. Jeuland (2002) estimates to sell compost in Bamako at 5,000 FCFA per m^3 corresponding to about US$ 7.5. Alter Ego (1996) gathered compost sale prices in different

would constitute US$ 40 per t TS of FS, and so it could be possible to reduce the annual treatment and production cost from 319 per t TS to around US$ 280 (reduction of 12%). This revenue is becoming important in comparison to the capital and O+M cost, when capital cost are reduced due the economy of scale (refer to the compilation 5.4).

5.3.2 Averted waste transport to landfill

Due to the process of composting of solid organic waste, the quantity of organic waste has not to be transported to the landfill. Hence, there is a potential of cost saving due to less waste transport. Transport cost are composed of capital cost due to the purchase of a tipper truck or similar engine and kilometre dependent cost due to operation and maintenance cost. Following assumptions have been done in order to estimate specific capital cost of a waste transport truck:

- Truck price (second hand): US$ 20,000
- Truck life time: 10 years
- Interest rate: 5%
- Truck capacity: 8 m^3
- Yearly collected waste volume: 6,000 m^3 (3 trips/day, 250 days/year)
- Quantity of organic waste treated: 150 m^3 per year (pilot plant)

Assumed annual capacity is higher than the amount of solid organic waste treated on the pilot plant. Hence, it is assumed that truck is running at listed capacity due to its use for other transports, too, instead of standing unutilised in the garage. Thus, truck capital cost amounts to US$ 0.4 per m^3 of transported waste. This corresponds with about US$ 5 per t TS of FS because 12 m^3 organic waste are used for the co-composting process with one t TS of FS[8].

In addition to truck capital cost, kilometre dependent costs occurs during transport for truck running and maintenance (fuel, tyres, etc.) and salaries for the driver and the worker. This cost depends on the distance to the landfill and its speed through the city. Following data are assumed to estimate km dependent cost:

- Truck cost per km (fuel, O+M): US$ 0.5
- Man hour cost (driver +worker): US$ 2
- Average speed: 30 km/h
- Truck capacity: 8 m^3

Note that average speed seems to be low, but often roads are in bad conditions and congested in the city centre. And low speed is correct because waste dumping on landfill time is neglected. With these conditions, the average km dependent cost per m^3 transported waste amounts to US$ 0.07 per km. This corresponds with US$ 0.9 per t TS of FS per km (factor 12, see footnote 3). When assuming a saved transport distance of 5 km in comparison to direct landfill disposal of all waste, transport cost saving would rise to US$ 9 per t TS (10 km in total, go and back).

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cities of West Africa. Prices ranged from 7,300 to 15,000 FCFA per t compost. Corresponding about to US$ 6 to 12 per m^3 (assuming $\delta = 0.5$ t/m^3)

[8] 25 g TS/L in delivered raw FS, volume reduction of 90% during dewatering, co-composting ratio 1:3 (dewatered FS:organic waste)

The sum of truck capital and truck running cost constitute the total cost benefit due to saved transport of organic solid waste and would amount to about **US$ 14 per t TS of FS** with above listed conditions. Calculation formula is given in appendix 6.4.

If there were no composting process, dewatered faecal sludge would have to be transported to the landfill in addition to solid waste, too. But we assume that this cost saving is cancelled by the transport cost of the compost to the buyers.

5.3.3 Landfill space saving

About 150 m^3 of organic waste is co-composted with approximate 50 m^3 of dewatered FS on the pilot plant. Without composting, the sum of these two amounts would have to be disposed on the landfill. Assuming a mean density of 0.5 t/m^3 of organic waste and 1 t/m^3 of dewatered FS, a sum of 125 t would have been to discharge annually on landfill without the co-composting procedure. According to Cointreau-Levine (1997), capital and O+M cost of a simple landfill (without clay lining nor leachate collection) in low incoming areas amounts to a range of US$ 6 to US$ 10 per ton capacity (large landfill of 1,000 t/day to small landfill of 250 t/day respectively) for a 10-year landfill life. Assuming a landfill of an medium size (US$ 8 per ton), we obtain **landfill cost** of disposal of **US$ 80 per t TS of FS[9].**

Saved landfill cost are high in comparison to other benefits. But the problem is that this benefit is not really perceived because in reality, waste and dewatered FS often is not disposed on a landfill, but dumped somehow in the next environment and stored on the treatment plant site, respectively. Therefore, this saved landfill cost has to be considered as a theoretical economic (not financial) benefit if waste and dehydrated FS was disposed properly on landfill.

5.3.4 Diarrhoea reduction

The economical valuation of health benefits is a challenging issue based on several assumptions and hypotheses. In order to evaluate health benefits in monetary terms of the co-composting of FS and organic waste, it is necessary first to estimate the health impact (change of illness rates) of the FS treatment associated with waste composting. When knowing how many diseases are averted, an economic value has to be allocated to an averted disease and its associated benefits.

Several diseases like diarrhoea and parasite affections (e.g. ascaris worm) are due to the faecal-oral transmission path. Therefore, improved FSM will have a positive impact on public health to faeces related diseases. As diarrhoeal infections account for the main disease burden associated with bad sanitation conditions, diarrhoeal disease was chosen to estimate economic benefits of improved FSM. Other diseases and the positive impact of organic waste composting on public health were not considered. Direct benefit of diarrhoea infections is in the health sector due to less treatment cost. Beneficiaries are both the health facilities and the households (averted medicaments and treatment fees). But there is not only the benefit of averted health cost, but also the fact, that an averted patient remains productive during the time he would have been sick or the rest of his life if his death was averted. Figure 15 illustrates the benefits taking into account of diarrhoea reduction and the procedure to value them.

A set of health studies evaluated the impact of water supply and sanitation improvements. The review of several rigorous studies done by Esrey et al. (1991) figured out diarrhoea morbidity reduction of 36% due to improved excreta disposal. The fact that bad faecal sludge evacuation and handling compromise the success of sanitation interventions is not mentioned and not evaluated yet. Hence, the crucial question is to estimate the positive

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[9] 125t*8$/12.5t TS (annual amount of treated FS: 12.5 t TS)

impact of improved FSM on the public health. For this benefit evaluation, we estimated the health impact of FSM to 3% diarrhoea morbidity reduction. It seems to be appropriate to assume that proper FS evacuation contributes to about 10% of improved excreta disposal.

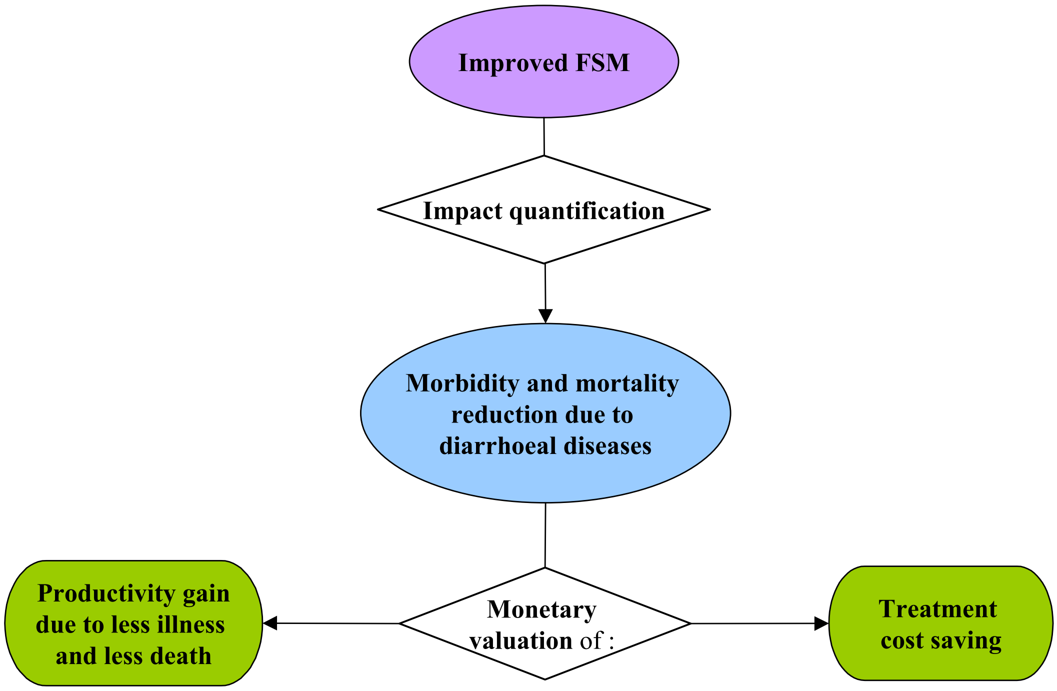

Figure 15 Schema of the process of benefit valuation of diarrhoea reduction.

Then the estimation of averted diarrhoea diseases can be done on the base of diarrhoea incidence rates by age groups and population structure provided by the WHO for the region of Ghana. With a 3% diarrhoea morbidity reduction, the averted cases per year per 1,000 persons amounts to 35 cases, most of them (64%) in the age group of children younger than five years. With the help of averted diarrhoea cases and the case-fatality-rate (number of death per diarrhoea case) provided as well by the WHO, averted death due to diarrhoea infections amounts to 2.6 persons (almost only children under five) per 100,000 inhabitants served by improved FSM (refer to appendix 6.5 for details).

The procedure to convert the health impact in monetary terms was adopted from Hutton (2002) from the Swiss Tropical Institute, who is working in collaboration with the WHO on a publication regarding economic benefits of diarrhoea reduction of water and sanitation interventions. For instance, the minimum wage of US$ 2 was taken to value a lost adult workday due to illness (one diarrhoea case causes two days of at an adult). Details of the valuation method and costs of diarrhoea treatment are given in the appendix 6.5. When assuming that the pilot plant treats about 12.5 t TS per year and serves about 800 persons (septage and public toilet sludge mixture 2:1), economic benefits of diarrhoea reductions can be converted into US$ per t TS of treated FS. Because with this mixture, about 65 persons produce one ton TS of FS. Results are given in 21.

It is interesting to note, that indirect health benefits due to higher productivity is as important than the direct health benefits of averted treatment cost. But note that these values, although given in absolute values, constitute an order of magnitude and only valid under cited conditions. Furthermore, a set of uncertainty is present:

- Estimation of the potential diarrhoea morbidity and mortality reduction due to improved FSM
- Kind of valuation method (how to allocate an economic value to gained workdays, etc.)

- Reliability of input data (e.g. treatment cost of one diarrhoea case)

Table 21 Economic benefits of diarrhoea reduction due to improved faecal sludge management.

Type of benefit	Unit	Amount
Averted health treatment cost	[US/t TS]	22
Productivity gain due to lower morbidity	[US/t TS]	12
Productivity gain due to lower mortality	[US/t TS]	17
Sum of benefits	**[US/t TS]**	**50**

Nevertheless, health benefits of diarrhoea reduction might be remarkable. When sludge mixture would change, i.e. less public toilet sludge, economic benefits would even increase, because more people could be served with the same FS treatment plant (e.g. 200 persons would produce 1 ton TS of FS if there were only septage).

5.3.5 Environmental and societal benefits

Indiscriminate disposal of untreated FS does create damaging impacts on the environment. Main problem is water, groundwater and soil pollution in sanitary and organic terms through surface flow, infiltration or washing away through drainage water. Adapted treatment of FS and its reuse in agriculture like proceed in the co-composting pilot plant, following environmental benefits are achieved:

- Reduction of the organic and pathogen charge in the environment (surface and groundwater, soil)
- Recycling of valuable nutriments and amendment of soil structure (not achieved with mineral fertiliser)

Furthermore, the collection and treatment of solid waste and FS might produce further benefits on the served population in addition to public health improvements. Improved FSM and waste collection should enable the waste disposal outside of settlements. Therefore, in the settled area, there would be two indirect benefits for the population:

- Less odour nuisance in household environment through disposed or buried FS and waste.
- Toilet holders enjoy more amenity because pit emptying became affordable and destruction of pit latrine is not needed anymore (because mechanical emptying by suction instead of manual emptying by buckets)

But, it is very difficult to estimate this kind of environmental and societal benefit. However, even if they would be tangible in some way, it would be very difficult to allocate them a monetary value. One possibility of valuation would consist in the estimation of mitigation measures. E.g. the price of a new well due to a groundwater contamination, the price of a wastewater treatment plant in order to clarify polluted surface water or the commercial price of another soil conditioner like peat. However, it is not the scope of this document to develop these challenging issues.

5.4 Cost and benefits compilation

Table 22 provides a summary of costs and benefits of the co-composting plant. Costs are adopted from 5.1 while benefits are described in 5.3. Average costs and benefits are not only expressed in US$/t TS in reference to the faecal sludge, but also in US$/t compost in reference to the sold compost of the co-composting process. Therefore an average density of final compost of 500 kg/m^3 is assumed and then four t of compost are produced per t TS of incoming faecal sludge. E.g. the pilot plant treats about 12.5 t TS per year and produces approximately 100 m^3 (~ 50 t) of compost given the co-composting ratio of 1:3 (dewatered FS : organic waste) and a volume reduction of FS on the drying beds of 90%.

Table 22 Compilation of costs and benefits of the co-composting pilot plant.

		Amount [US$/t TS of FS]	Amount [US$/t compost]	Remarks, assumptions
Costs	Capital cost	**175**	**44**	Without land cost and post treatment of effluent
	O+M cost	**144**	**36**	
	Sum of costs	**319**	**80**	
Benefits	Revenue biosolids sale	**40**	**10**	Sale price: US$ 5 per m^3
	Economy of organic waste transport	**14**	**4**	150 m^3 organic waste have not to be transported over a distance of 10 km
	Economy of landfill space	**80**	**20**	Landfill cost: US$ 8 per t waste and year
	Health benefits	**50**	**13**	Reduction of diarrhoea cases of 3%
	Sum of benefits	**184**	**47**	
Balance		**135**	**33**	Cost reduction of 62%

When taking into account the economy of scale, described in 5.2, economic benefits of co-composting may even overtake all capital and investment costs of the installation. Assuming that benefits behaviours in a linear manner to the treated amount of waste and FS, Table 23 summarises costs and benefits for three different plant sizes. But note that in order to take into account all costs, collection and haulage cost of FS and waste should be integrated to the costs, too. Because as a matter of fact, the larger the treatment plant size, the bigger specific haulage cost per t TS of FS and waste due to longer transport distances. Furthermore, capital costs do not include post treatment of the liquid-solids separation effluent. In order to get an order of magnitude, transport cost would amount to about US$ 30 per t TS of FS (Steiner 2002, for a plant size of 1,500 t TS) and post treatment of liquid to about US$ 10 per t TS (Heinss 1999, plant size of about 150 t TS). Hence, even the balance of the plant treating annually 625 t TS of FS and corresponding amount of solid waste would not be negative, i.e. costs are higher than benefits. But unquestionably, the aim of FS and solid waste treatment is not to be economically viable, but to improve public health and to protect the environment from hazards due to uncontrolled discharge of FS and solid waste.

Table 23 Summary of costs and benefits of co-composting of faecal sludge and solid waste in function of plant capacity. Costs are rounded and capacity is expressed in TS comprised in incoming and, hence, treated FS.

Yearly plant capacity	**12.5 t TS of FS and 150 m^3 organic waste**		**125 t TS of FS and 1,500 m^3 organic waste**		**625 t TS of FS and 7,500 m^3 organic waste**	
	[US$/t TS]	[US$/t compost]	[US$/t TS]	[US$/t compost]	[US$/t TS]	[US$/t compost]
Capital cost	175	45	80	20	60	15
O+M cost	145	35	110	30	105	25
Benefits	185	45	180	45	180	45
Difference of costs and benefits	135	35	10	5	-15	-5

Part B References

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(to be completed)

Part B Annex

Composting

Trial	C	N	C/N	K	Ca	Mg	P
	[%dry weight]	[%dry weight]	[-]	[ppm]	[ppm]	[ppm]	[ppm]
HW	37	1.1	35	2	10.6	10.3	0.2
DFS	6	0.2	29	1	1.3	6.8	0.5
HW/DFS (3:1)	9	0.4	23	1.4	9	6.8	0.4

Trial	Pb	Mn	Cu	Zn	Fe
	[ppm]	[ppm]	[ppm]	[ppm]	[ppm]
HW	82	82	16	39	608
DFS	147	28	7	136	359
HW/DFS (3:1)	69	23	12	91	425

First Cycle	Humidity	C	N	C/N	K	Ca	Mg	P
	[%]	[%dry weight]	[%dry weight]	[-]	[ppm]	[ppm]	[ppm]	[ppm]
MW	69	19	0.6	33	0.9	8	5.3	0.5
HW	50	5	0.2	36	0.2	8	2.8	0.4
DFS	42	6	0.2	27	0.1	0.3	9.4	1.3
(1) 3:1 (HW:DFS)	49	4	0.1	32	0.3	10.9	5.1	0.2
(2) 3:1 (MW:DFS)	60	10	0.3	30	0.6	3.9	8	0.8
(3) 2:1 (MW:DFS)	53	3	0.1	28	0.2	2.9	7	1.1
(4) MW (control)	69	19	0.6	32	1	8.1	6	0.5

First Cycle	Pb	Mn	Cu	Zn	Fe
	[ppm]	[ppm]	[ppm]	[ppm]	[ppm]
MW	1725	70	18.9	86	2943
HW	329	41	11.48	64	1044
DFS	325	53	27.46	87	487
(1) 3:1 (HW:DFS)	354	71	44.46	54	1087
(2) 3:1 (MW:DFS)	104	75	42.7	129	840
(3) 2:1 (MW:DFS)	266	63	37.92	143	1042
(4) MW (control)	1727	71	19.3	86	2962

Economic Aspects
Estimation of O+M cost in function of plant size

Operation and Maintenance cost			Pilot plant (500 m3 FS)		Full-scale (5,000 m3 FS)		Full-scale (25,000 m3 FS)	
	unit	unit price [US$]	quantity	cost [US$]	quantity	cost [US$]	quantity	cost [US$]
Sludge removal and transport to composting area (removal [3h/m3] 1 $ and transport 1 $)	[m3]	2	50	100	500	1000	2500	5000
Replenishment of sand (for times per year and bed a layer of 10 cm), sand included in prize	[m3] of replaced sand	3.0	25	75	250	750	1'250	3'750
Waste sorting (~10 h/m3) and transport to composting area	[m3] of final organic waste	3.5	150	525	1'500	5'250	7'500	26'250
Compost turning (every third days) including watering, heaping, removal to maturation heaps (5 h/m3)	[m3] of initial composted material	1.5	200	300	2'000	3'000	10'000	15'000
Compost screening and bagging (3 h/m3)	[m3] of final compost	1	100	100	1'000	1'000	5'000	5'000
Cleaning screen and storage tank, contigencies (desinfectant, masks, etc.), general repairs	sum			200		2'000		10'000
Salary plant manager	[US$]	1'000	0.5	500	1	1'000	1	1'000
Total				**1'800**		**14'000**		**66'000**

according to manhour information of the pilot plant from Sharon and Cofie (2002)
a salary of 0.25-0.4 US$ per manhour for an unskilled worker is assumed

Annualised capital cost

The equation to calculate the annual capital cost is a financial standard operation given below (e.g. Maystre 1985):

$$CC = C_{tot} \frac{(1+i)^n \cdot i}{(1+i)^n - 1}$$

with: CC = annual capital costs; C_{tot} = total costs; i = interest rate; n = depreciation period

Economy of scale

When P is the price and C the capacity of the installation, following relation can express the economy of scale:

$$P = a \cdot C^{\alpha}$$

The parameters a and α can be determined with statistics or with theoretical construction considerations. The parameter α is minor than 1. Thus, cost of the installation rises less than proportionally when capacity increases. Therefore the average specific cost ASC of a treatment plant, for instance US$ per capita or per t TS, does decrease, viz.:

$$ASC = P / C = a \cdot C^{\alpha - 1} \quad with\ \alpha < 1$$

This relation is called *law of economies of scale*. It even can be used for a whole installation like a FSTP including screen, solid-liquid separation, sludge and effluent post-treatment. In addition of an economy of scale for each unit (tank or pond, etc.), there is a possible economy of scale of other items like discharge area or screen.

However, economy of scale is not implicitly valid anymore beyond of a certain plant size. In deed, it is possible that a huge plant is more cost intensive because of the change of adopted construction techniques. For instances when high resistance materials, special reinforcement or security installation become necessary. In this case, graph of the ASC in function of plant size does comprise a so-called *zone of diseconomy*. However, regarding FS management with low cost and very simple treatment options, we probably won't be confronted to such a zone of diseconomy of scale.

Transport and landfill costs

Formula to calculate transport costs:

$$Transport\ \cos t\left[{}^{US\$}\!/\!_{m^3.km}\right]=\frac{truck\ \cos t\left[\frac{US\$}{km}\right]+\dfrac{man\ hour\ \cos t\left[{}^{US\$}\!/\!_{h}\right]}{average\ speed\left[{}^{km}\!/\!_{h}\right]}}{\text{truck capacity}\cdot\left[m^3\right]}$$

Calculation sheet of transport, truck capital and landfill costs:

	A	B	C	D	E
1	**Transport cost (truck and salaries) of waste to landfill per km**				
2	Distance (e.g FSTP to landfill and back)	[km]	1		
3	Truck km cost	[US$/km]	0.5		
4	Truck capacity	[m3]	8		
5	Average speed	[km/h]	30		
6	Salary (driver + worker)	[$/h]	2		
7	Haulage cost per m3 waste	[US$/t]	0.07	(C2)*((C3+C6/C5)/(C4))	
8	**Haulage cost per t TS**	[US$/t TS]	**0.9**	12*C7	
9					
10	**Capital cost of truck for transport**				
11	Truck capacity	[m3]	8		
12	Price of one truck	[$]	20'000		
13	Yearly transported volume	[m3]	6000		
14	Life time	[years]	10		
15	Interest rate	[%]	5		
16	Capital cost per truck	[$/year]	2590	C12*(0.01*C15/(1-(1/(1+0.01*C15)^C14)))	
17	Captial cost per m3 organic waste	[US$/m3]	0.4	C16/C13	
18	**Capital cost per t TS**	[US$/t TS]	**5**	12*C17	
19					
20	**Landfill cost**				
21	Average cost of landfill (capacity of about 250 t/day) according to Cointreau-Levine (1997)	[US$/t.yr]	8		
22	Quantity of organic waste and dewatered FS to dispose	[m3]	200		
23	Average density of disposed mixture	[t/m3]	0.625		
24	**Disposal cost per t TS**	**[US$/t TS]**	**80**	C23*C22*C21/12.5	

Health benefits calculation sheet due to diarrhoea reduction

Health and associated benefit valuation

WHO region code	**Afro D**	**Source**	**Remarks/calculation**
Costing country (for treatment)	Benin		Used by WHO
Country used for minimum wage	Ghana		Chosen because FS field research partners are here

Value of morbidity and mortality reduction

Valuation of morbidity reduction (treatment cost and productivity)

Treatment cost	[US$/case]	9.55	GPE/WHO	average cost, when 8.2% of patients go to hospital during 3.5 days
Minimum wage	[US$/day]	2	author with local partners, method according to Hutton (2002)	
Productivity loss (0-4)	[US$/case]	5		5 days off; value of productivity of carer 50% of minimum wage
Productivity loss (5-15)	[US$/case]	6		3 days off; value of one lost schoolday equivalent to minimum wage
Productivity loss (15+)	[US$/case]	4		2 days off; valued at minimum wage

Valuation of lower mortality (productivity rise)

Discounted productivity years lost (average)	[years]	20	WHO	30 years with a 3% discount rate leads to 20 years
Productivity per year	[US$/death.yr]	500	author	according to WHO annual wage for low skill worker
Value of a productive life lost	[US$/death]	10000		= years lost x productivity per year

Estimation of diarrhoea reduction (per t TS of FS)

Population structure

Country used for population structure		Ghana	
Total population	[in million]	19.3	WHO
Proportion (0-4)	[%]	15	http://www.census.gov/ipc/www/idbpyr.html
Proportion (5-15)	[%]	28	"
Proportion (15+)	[%]	57	"

Annual diarrhoea incidence rate by age

Incidence rate (0-4)	[case/capita]	5	WHO
Incidence rate (5-15)	[case/capita]	0.9	WHO
Incidence rate (age 15+)	[case/capita]	0.3	WHO

Annual reduction of diarrhoea cases rate due to improved FSM

Reduction by improved FSM	[%]	3		For comparison: Esrey et al. (1990) give a 36% diarrhoea reduction
Avoided incidence rate (0-4)	[avoided case/capita]	0.15		and WHO a 22.5 to 37.5% reduction for improved sanitation
Avoided incidence rate (5-15)	[avoided case/capita]	0.027		(excreta disposal)
Avoided incidence rate (age 15+)	[avoided case/capita]	0.009		= morbidity reduction x incidence rate

Annual diarrhoea cases avoided per t TS of FS

Population per t TS		65		assuming daily 14 g TS and 100 g TS per capita for septage
(mixture 2:1=septage:PT)				and public toilet sludge respectively
Avoided cases per t TS	(0-4)	1.4625		= avoided incidence rate x popul. proportion x population per t TS
Avoided cases per t TS	(5-15	0.4914		
Avoided cases per t TS	(15+)	0.33345		

Fatality rates per affected patient

Fatality rate (0-4)	[death/affected capita]	0.00102	WHO
Fatality rate (5-15)	[death/affected capita]	0.00021	"
Fatality rate (15+)	[death/affected capita]	0.00033	"

Annual mortality reduction per t TS of FS

Avoided fatalities (0-4)	[avoided case/ t TS]	1.5E-03		= avoided cases per t TS x fatality rate
Avoided fatalities (5-15)	[avoided case/ t TS]	1.0E-04		"
Avoided fatalities (15+)	[avoided case/ t TS]	1.1E-04		"

Monetary benefits calculation of diarrhoea reduction (per t TS of FS)

Increased productivity due to morbidity reduction

Gained productivity (0-4)	[US/t TS]	7.3125	= avoided cases x productivity loss per case
Gained productivity (5-15)	[US/t TS]	2.9484	"
Gained productivity (15+)	[US/t TS]	1.3338	"
Total productivity benefit	**[US/t TS]**	**11.6**	

Avoided health treatment cost due to reduced morbidity

Total avoided cases	[cases/t TS]	2.28735	sum of all avoided cases
Health benefit (treatment)	**[US$/t TS]**	**21.8**	= sum of avoided cases x treatment cost per case

Productivity benefit due to mortality reduction

Total avoided fatalities	[cases/t TS]	0.0017	sum of all avoided fatalities
Productivity benefit	**[US$/t TS]**	**17.0**	= sum of avoided fatalities x value of a lost productivity life

Summary of annual diarrhoea reduction benefits

Health treatment cost avoided	[US/t TS]	22
Productivity (lower morbidity)	[US/t TS]	12
Productivity (lower mortality)	[US/t TS]	17
Sum of benefits	**[US/t TS]**	**50**