


Co-composting of Faecal Sludge and Municipal Organic Waste

A Literature and State-of-Knowledge Review

May 2003

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Foreword

This review on the combined composting of (faecal) sludges and organic solid waste was produced as part of the project entitled „Co-composting of Faecal Sludge and Organic Solid Waste in Kumasi, Ghana.“ The initial phase of the project was conducted within the programme „Sustainable Solid Waste Management and Sanitation“ which was financed by the French Ministry of Foreign Affairs and coordinated by the pS-Eau/PDM.

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 city's Waste Management Department (WMD, Kumasi Metropolitan Assembly) and SANDEC. Results of the investigation will help WMD to develop its biosolids management strategy and enable the project team to develop guidelines for planners and engineers on the option of co-composting.

Abbreviations and Glossary

Abbreviations

BOD	Biochemical Oxygen Demand
COD	Chemical Oxygen Demand
FC	Faecal Coliforms
MSW	Municipal Solid Waste
FS	Faecal Sludge
NH4-N	Ammonium Nitrogen
NH3-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

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 *Schisto-*

soma 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 **4.6**

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

Besides excreted pathogens, chemical contamination constitutes an important potential risk associated with human waste (wastewater and sludge) use. Relevant groups of chemical contaminants are (Environmental Research Foundation 2000; Holm 2001; McArdell 2002; Suter 2002):

- **Heavy metals (HM)**
- **Hormone active substances** (HAS; also termed “endocrine disrupting chemicals”, EDC). These subsume natural and synthetic estrogens, and an array of substances (and/or their degradation products) with, among them, polychlorinated biphenyl (PCB) and

chemicals used in industrial detergents, as plastic additives, in pesticides and antifoulings, body care products.

- **Antibiotics**

It may be rightfully assumed that the use and occurrence of some of these substances are still rather limited in developing countries. Others, such as heavy metals, antibiotics, and HAS (through the use of pesticides, cosmetic products and contraceptive medicine), however, may likely be found rather widely and in critical loads in urban waste streams. Antibiotics, e.g., are widely and indiscriminately used in urban societies of developing countries.

The risks associated with these substances may, in the long run, turn out to constitute a greater threat to health and be more difficult to deal with than the risks from excreted pathogens. Excreted pathogens die off once they have been shed into the environment (at varying rates, though, and with potential for transient re-growth of some bacterial pathogens under exceptional conditions). In contrast to this, heavy metals are conservative substances accumulating in the environment, particularly so in waste-amended soils. HAS and antibiotics have become of considerable concern and are now a focus of environmental research in industrialised countries, as they, too, may accumulate and persist in the environment over extended periods with potentially serious health impacts for humans and animals. They originate from human and animal excreta (naturally produced and from human and veterinary pharmaceuticals) as well as from domestic and industrial wastewater and sludges. Sludges accumulating in individual sanitation systems or produced during wastewater treatment, are the result of concentration processes taking place during storage and treatment. Hence, they constitute a sink for substances, including chemical contaminants, which are in non-dissolved form. As a tendency, therefore, keeping a close eye on the chemical quality of sludges is of particular importance, where they are used in urban agriculture regularly and over longer periods of time.

Contamination of the organic fraction of solid municipal waste by chemical constituents, notably heavy metals, must be presumed in most cases as organic solid waste is usually stored and collected together with other waste fractions. 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
Characterisation	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 %	< 0,1 ? %
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 pathogens might be stopped (Matthews 2000).

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

	BOD [mg/l]		NH_4-N	Helminth eggs	FC
	total	filtered	[mg/l]	[no./L]	[no./100 mL]
A: Liquid effluent					
1. Discharge into receiving waters:					
• Seasonal stream or estuary	100-200	30-60	10-30	≤ 2-5	≤ 10^4
• Perennial river or sea	200-300	60-90	20-50	≤ 10	≤ 10^5
2. Reuse:					
• Restricted irrigation	n.c.		1)	≤ 1	≤ 10^5
• Unrestricted irrigation	n.c.		1)	≤ 1	≤ 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

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

[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

yearly rate of irrigation (500-1,000 mm).Examples for faecal sludge treatment standards are known from China and Ghana.

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:

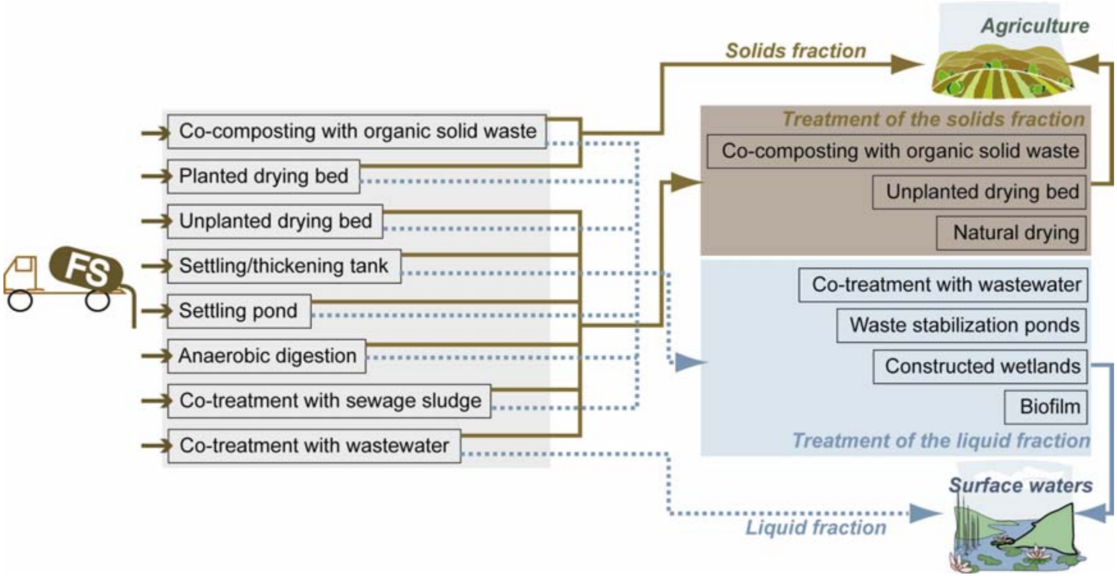

Figure 1 Overview of potential, modest-cost options for faecal sludge treatment

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. The separating of the solids and liquid fraction of FS, is the process-of-choice to condition FS for co-composting.

2.3 Pre-Treating FS for Co-Composting Solids-liquid separation and dewatering

2.3.1 Principles

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

2 http://www.sandec.ch/sos/references.html
* hydraulic ret. time = days to weeks
** hydraulich ret. time = hours

beds. Resulting from this are a solids and a liquid fraction (Figure 1). 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 in tropical and temperate climates. 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), up-flow anaerobic sludge blanket clarifiers (UASB), or constructed wetlands (CW). Liquids emanating from separation processes 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).

2.3.2 Settling/thickening of FS

Total solids (TS) and suspended solids (SS) contents in faecal sludges are by factors of 7 – 50 higher than in 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.

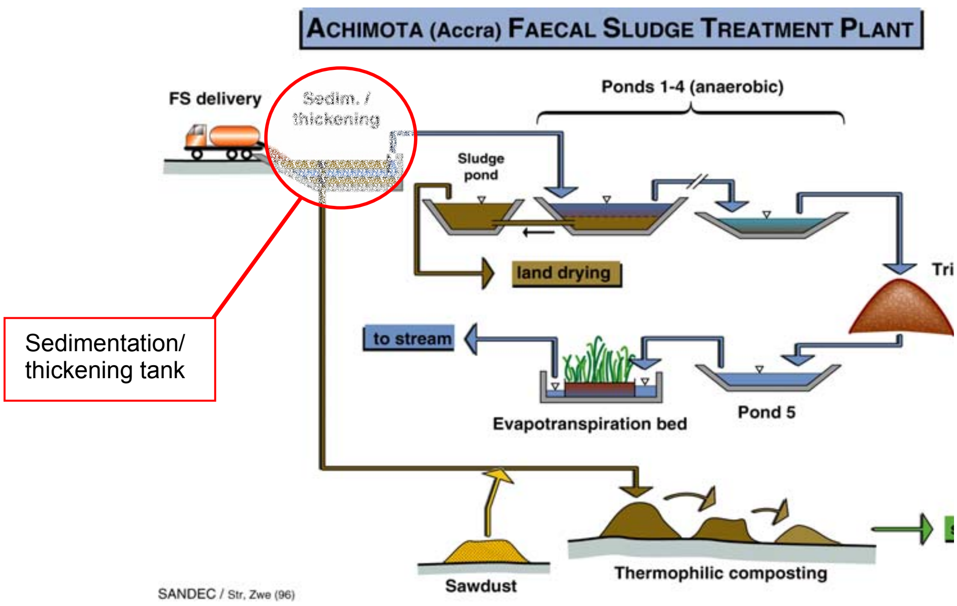

Figure 2 Scheme of the Achimota Faecal Sludge Treatment Plant

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 at most 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).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.

2.3.3 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.

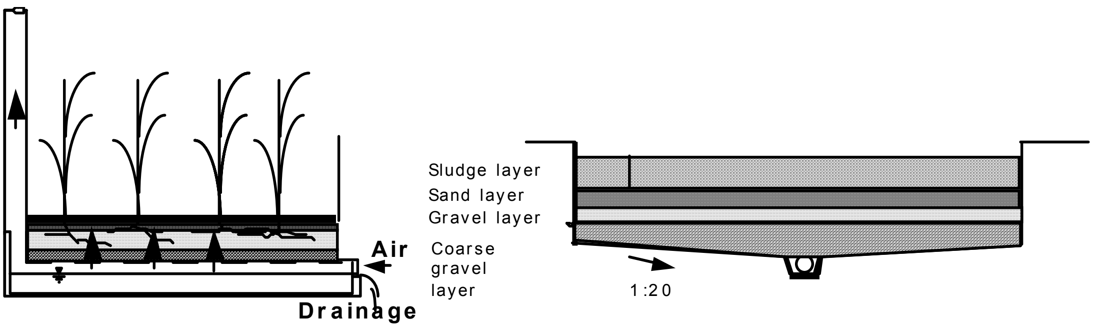

Figure 3 Planted and unplanted sludge drying beds (schematic)

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 usually in the order of 60 %. Pescod (1971) conducted experiments with unplanted sludge drying beds in Bangkok. According to these 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 were are required depending on the solids loading rates applied (70 - 475 kg $TS/m^2 \cdot yr$).

Results from FS dewatering experiments on 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 ≤ 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. Further rain-protected storage of the dewatered solids of several months is required to attain a hygienically safe product for use in agriculture

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 in tropical and temperate climates. Figure 5 allows to estimate the time required for *Ascaris* egg die-off in properly operated, thermophilic composting. 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	Waste-water	No. of eggs /g TS	Egg viability	
Extrabes, Campina Grande (Brazil); experimental WSP scheme	----	1,000 (nematodes)	1,400 – 40,000 (in the settled solids accumulated across 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)
Kumasi Ghana; faecal sludge from unsewered public toilets and from septic tanks	900 – 6,900 (7 samples) (Ascaris + Trichuris)	----	20 – 85 (3 samples) (in biosolids dewatered on sludge drying beds during 1 – 3 weeks; TS = 20 %)	45 – 82 % (8 samples) (biosolids dewatered for 1-3 weeks and fresh sludge)	Gallizzi (2003)

3 Municipal Organic Solid Waste Management

3.1 Relevant municipal solid waste characteristics and quantities

It is the easily biodegradable fraction which is of immediate interest in composting. 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 needed? (Richard 1996). If these materials are included in the composting process, their particle sizes are often reduced beforehand through shredding to allow for faster 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 Annex 2.

The same process parameters valid for composting must be adhered to and play a role in co-composting of human waste with organic 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 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 of compost produced from 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) and organic solid waste 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 9, which were collated from many references and for composts produced from many different raw materials, including human waste.

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.

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

Constituent	% of dry weight	Reference
• **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

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

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.

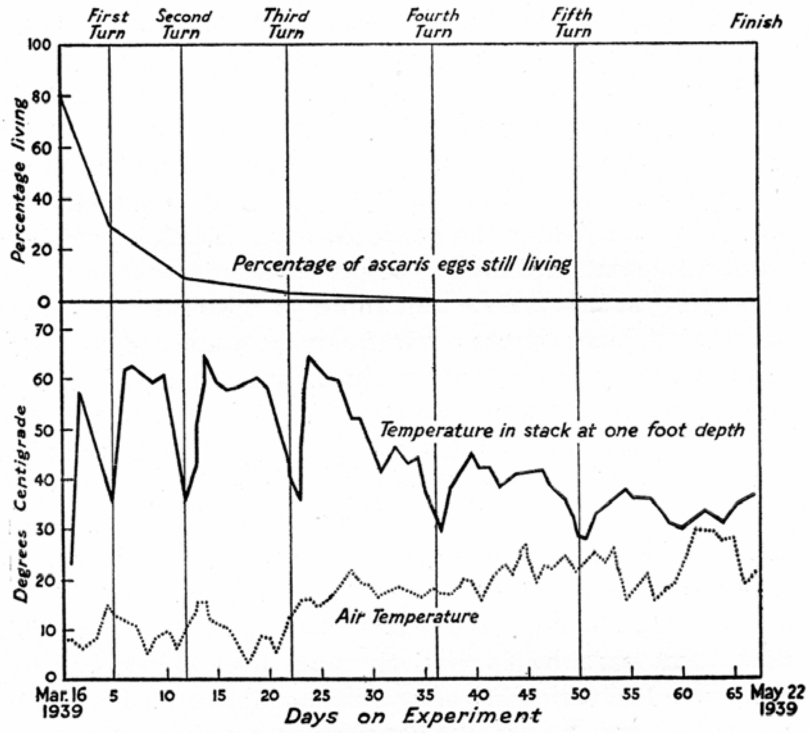

Figure 5 Ascaris egg inactivation in thermophilic stack co-composting of faeces (69 % of raw material), vegetable matter (20 %), soil (10 %), and ash (1 %) (Scott 1952)

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).

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

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.

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).

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

5 Literature and case-studies on FS co-composting

5.1 Case-studies of co-composting

5.1.1 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.

5.1.2 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.

5.1.3 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 6 Sprinkling FS over refuse at the Rini/ Grahamstown (South-Africa) co-composting plant

Figure 7 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.

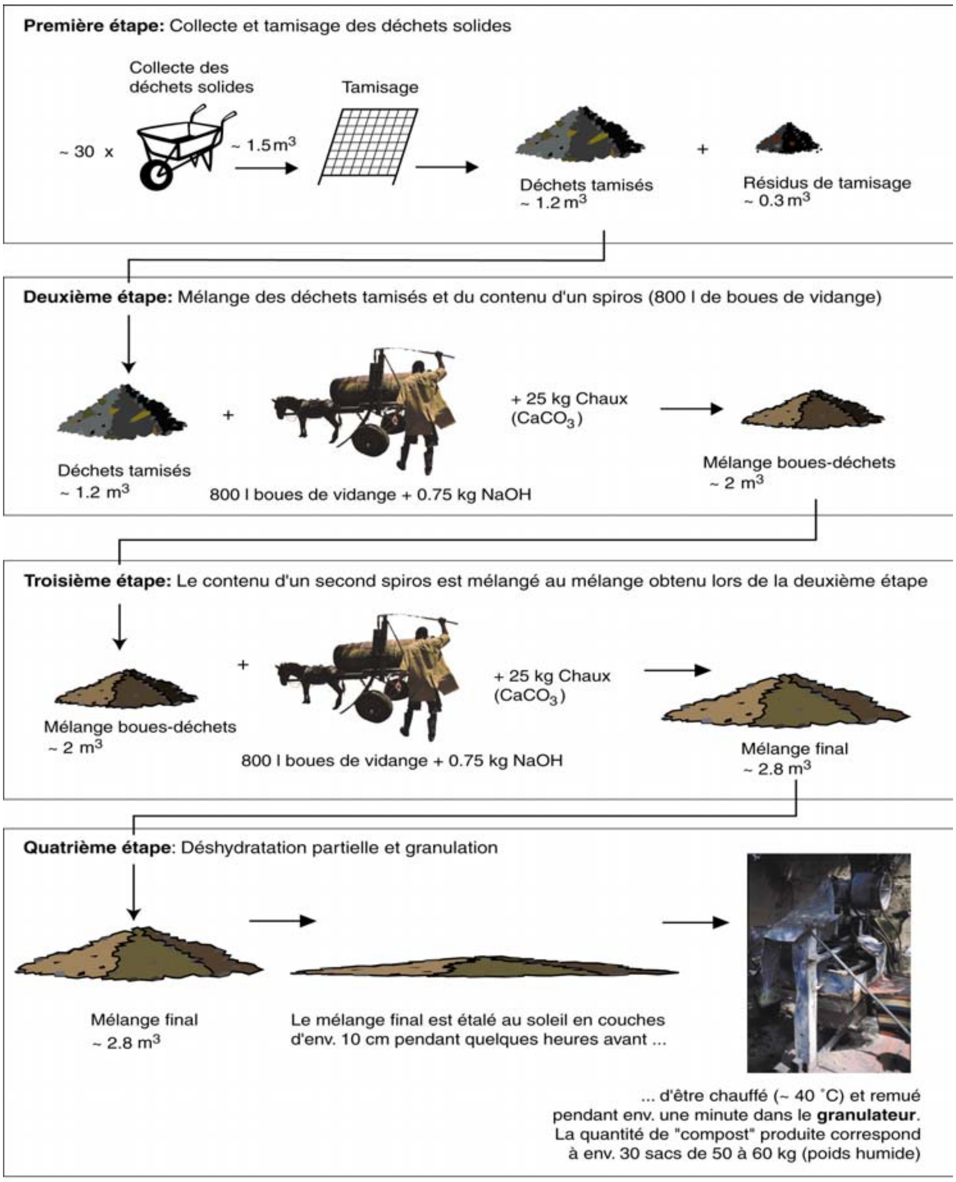

Figure 8 Co-composting of FS and sorted MSW in Niono, Mali (Montangero and Strauss 1999)

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 ($CaCO_3$) 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.

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.

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**

  Small-scale co-composting to produce compost for rice and vegetable farming in Niono, Mali (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).

5.2 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

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

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 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.

6 Conclusions and open questions

6.1 Conclusions

Based on the above "state-of-art" review the following conclusions on 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 for composting are adhered to.
- Mixing ratios reported about in the literature vary widely, depending 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 to 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 productHoornweg and Thomas (1999) list explanations why composting is not widely or successfully practiced in cities of developing countries:

- insufficient knowledge and care in carrying out composting operations leading to inadequate compost quality and resulting in nuisances potential, such as odours and rodent attraction.
- lack of markets for the product and lack of compost marketing efforts and skills.
- 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 cen-

tralised waste collection services rather than promote and support recycling activities and decentralised composting schemes.

The following issues related to FS/MSW composting warrant applied research:

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

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 to limit risks of compost contamination by impurities and chemical constituents

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

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Annex 1 Mixtures of faecal sludge and other organic material in combined composting as reported in the literature

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

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 the 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.

Ammonia (NH3) and Ammonium (NH4+) are in a pH and temperature dependant equilibrium..

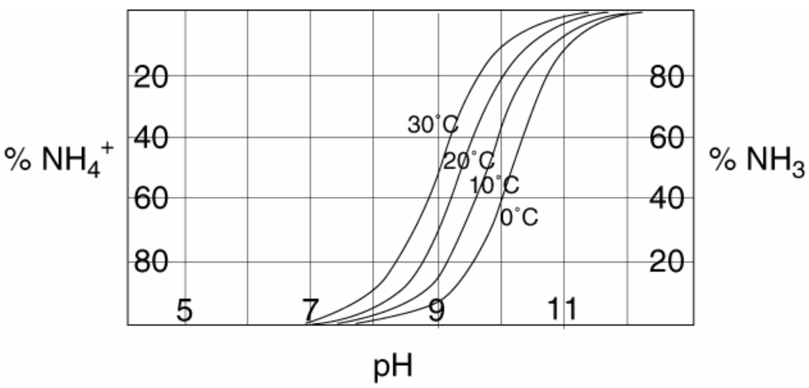

Figure 9 Ammonia – ammonium equilibrium as a function of different temperatures and pH (Schreiner 1997).

Figure shows that higher pH and higher temperature move the equilibrium in favour of ammonia. Thus, higher levels of pH developing 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 volatilization 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 high temperatures for pathogen inactivation 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 hence, nitrogen losses increase 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.

Immature composts, still exhibit, the microbial activities when applied on soil 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.