Technology

Chapter 9

Co-treatment of Faecal Sludge in Municipal Wastewater Treatment Plants

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Learning Objectives

- Understand the biodegradability and fractionation of organic matter and nitrogen compounds in faecal sludge.
- Understand the principles, key considerations and potential impacts of co-treatment of faecal sludge in sewer-based wastewater treatment systems.
- Determine volumes of faecal sludge that can be effectively co-treated in wastewater treatment plants.
- Understand the potential negative ramifications of co-treating faecal sludge in sewer-based wastewater treatment systems.

9.1 INTRODUCTION

The use of onsite sanitation technologies can be a sustainable solution to meet sanitation goals in a faecal sludge management (FSM) service chain, as long as the faecal sludge (FS) from these systems is collected, transported, treated, and then used for resource recovery or safely disposed of. One possibility for FS treatment is co-treatment with sewer-based wastewater treatment technologies. However, it is common knowledge that the majority of wastewater treatment plants (WWTPs) in low-income countries have failed, and improper co-treatment with FS has even been the cause of some failures. Hence, the objective of this chapter is to illustrate through modelling of WWTPs how these failures occurred, and the extreme difficulties with co-treatment that must be addressed to avoid failures. First, the chapter addresses activated sludge processes, and then anaerobic technologies including upflow anaerobic sludge blanket (UASB) reactors, digesters, and ponds. Co-treatment in ponds is also discussed in Chapter 5.

Based on the results of this chapter, co-treatment of FS with wastewater is not recommended for the vast majority of cases in low-income countries. If a co-management option is desired, a better option would potentially be co-management of FS with the sludge produced during wastewater treatment (i.e. biosolids). Many of the enduse and resource recovery options presented in Chapter 10 are appropriate for this, and could provide increased revenue from resource recovery. The tools in this chapter are relevant to evaluate existing, operational WWTPs, and for evaluating future WWTP designs.

In addition, the uncontrolled dumping of FS into sewers needs to be carefully regulated and prevented. The considerably higher solids content of FS (Chapter 2) may lead to severe operational problems such as solids deposition and clogging of sewer pipes. This is mostly because the diameter and slope of sewers are designed for the transport of municipal wastewater typically containing 250 to 600 mgTSS/L (Henze *et al.*, 2008) rather than the 12,000 to 52,500 mgTSS/L present in FS (Table 2.3). Hence, the first step in designing a co-treatment system includes determining how the FS will be transported to the treatment facility and discharged into the influent stream.

WWTPs are typically not designed for FS loadings, and process disruptions and failures are frequently observed. Common problems with co-treatment of FS in WWTPs range from the deterioration of the treated effluent quality to overloading tanks and inadequate aeration (Andreadakis, 1992; Al-sa'ed and Hithnawi, 2006; Heinss and Strauss, 1999; Strauss *et al.*, 2000; Chaggu, 2004; Harrison and Moffe, 2003; Lopez-Vazquez, 2008; Lake, 2010; Lake *et al.*, 2011; Wilson and Harrison, 2012; Still and Foxon, 2012).

Despite the potential operational problems, certain guidelines indicate that low volumes of FS could be co-treated in municipal WWTPs (ATV, 1985; USEPA, 1984, 1994). The USEPA states that that up to 3.6% of the maximum plant design capacity load can be FS (i.e. from septic tanks) (EPA, 1994). However, these recommendations are mostly based on biochemical oxygen demand (BOD_5) which does not account for the total organic and inorganic content present in FS or provide enough relevant information on the different biodegradable fractions (Henze and Comeau, 2008). Instead, chemical oxygen demand (COD) measurements are recommended to be used since total COD can be subdivided into useful organic fractions to assess the design and evaluate the performance of biological wastewater treatment processes. This chapter presents the impact of FS co-treatment in municipal WWTPs, based on expected average FS strength and COD and total nitrogen (TN) fractionations. This approach is recommended to evaluate whether co-treatment may be feasible without causing any process disruption or deterioration.

9.2 FAECAL SLUDGE BIODEGRADABILITY AND FRACTIONATION

9.2.1 Characterisation ratios

When evaluating FS characteristics to determine the potential for co-treatment, in addition to classic parameters such as COD, BOD and TSS, the ratios between these parameters also provide useful information. Ratios of parameters for public toilets and septic tanks are presented in Table 9.1.

The ranges of values in Table 9.1 are quite large and therefore only provide a rough estimation of the potential biodegradability. The ratios must also be used with caution. As compared to common values observed with wastewater, they suggest that FS is not readily biodegradable. The low VSS to TSS ratios indicate 23-50% inorganic content. The COD:BOD_5 ratio of 5.0 for public toilets indicates that, if degradable, the organics biodegrade slowly. In contrast, the COD:BOD_5 of 1.43 - 3.0 for septic tanks indicates the sludge is biodegradable, which probably is not the case, as septic tank sludge typically has a much longer storage time with significant stabilisation (e.g. years as opposed to days). This illustrates the need for a more reliable and informative method to determine the biodegradability of FS.

Table 9.1 Characterisation ratios for public toilet and septic tank faecal sludge to evaluate biodegradability for treatment purposes (calculated based on Table 2.3 and adapted from Henze *et al.*, 2008)

Ratios (g/g)	Public toilets	Septic tanks	Medium strength municipal wastewater
VSS:TSS	0.65-0.68	0.50-0.73	0.60-0.80
COD:BOD_5	5.0	1.43-3.0	2.0-2.5
COD:TKN	0.10	1.2-7.8	8-12
BOD_5:TKN	2.2	0.84-2.6	4-6
COD:TP	109	8.0-52	35-45
BOD_5:TP	17	5.6-17.3	15-20

The organic content to nitrogen ratios also indicate that organic concentrations are not sufficient for nitrogen removal by denitrification, as they are far below the lowest reported for nitrogen removal (Henze and Comeau, 2008). FS should only be considered for co-treatment in processes that include nitrogen removal if the influent wastewater has a high COD:TKN or BOD_5:TKN ratio (i.e. 12-16 and 6-8, respectively). In contrast, the COD:TP and BOD_5:TP ratios are relatively high, which suggests that there could be sufficient organic matter for biological phosphorus removal.

9.2.2 Biodegradability and fractionation

Fractionation is the breakdown of organic matter into groups based on biodegradability and physico-chemical properties. Frequently, (bio)degradability is measured by BOD_5. However, this method has limitations such as the incomplete determination of all the organics since the unbiodegradable fractions cannot be determined by this analytical technique, as underlined by Roeleveld and van Loosdrecht (2002) and Henze and Comeau (2008). Thus, the use of COD is preferred to assess the organic matter for design, control, monitoring and mathematical modelling of wastewater treatment processes. Advantages of COD over BOD_5 include: (i) a rapid analysis (e.g. hours as opposed to 5 days), (ii) more detailed and useful information including all degradable and undegradable organics, and (iii) the potential for the organics balance to be closed (on a COD basis). Of the two COD analytical determination methods, the dichromate method is preferred, as the permanganate method does not fully oxidise all organic compounds (Henze and Comeau, 2008).

The biodegradable fraction can be divided into readily and slowly biodegradable compounds. Readily biodegradable organics are assumed to be relatively small molecules that can dissolve in water and be rapidly consumed (e.g. volatile fatty acids and low molecular weight carbohydrates). Slowly biodegradable organics are considered to be more complex, and require extracellular breakdown prior to uptake and utilisation by microorganisms (Dold *et al.*, 1980). They are assumed to be colloidal and particulate compounds that can also be removed by physical-chemical means (e.g. coagulation-flocculation and settling).

The unbiodegradable fractions (often also referred to as inert) are not degraded, or degraded so slowly that they are not transformed during their transport in the sewer or residence time in WWTPs. They are also further divided into soluble and particulate organic groups. It is assumed that particulates can be removed by physical separation (e.g. settling), but the soluble unbiodegradable organics cannot be removed by biological or physical-chemical methods. Thus, when soluble unbiodegradable organics reach the sewage treatment plants, they pass through the system in the liquid phase with the same influent and effluent concentrations (Ekama, 2008). In wastewater treatment systems, the soluble unbiodegradable organics have a profound impact on effluent quality and the particulate unbiodegradable organics on sludge production and solids accumulation.

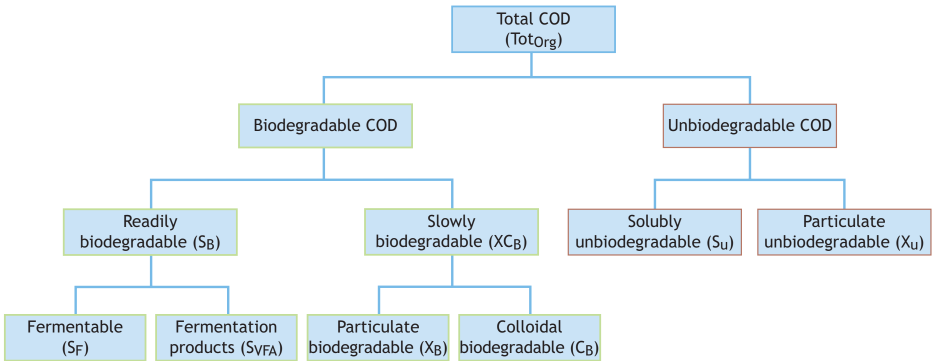

Figure 9.1 Organic matter (COD) fractionation diagram (adapted based on Melcer, 2003 and Corominas *et al.*, 2010).

It is important to underline that organic compounds contain different carbon, nitrogen and phosphorus components. It is preferable to determine and express carbon components in terms of COD (in view of the advantages of this analytical technique over others). Figure 9.1 illustrates the different COD fractions of the organic compounds as well as the common abbreviations for the different fractions (Corominas *et al.*, 2010):

X = particulate
S = soluble
C = colloidal
B = biodegradable
U = unbiodegradable
F = fermentable
VFA = products of fermentation

Thus, the total organic matter concentrations present in wastewater given as the sum of the different biodegradable and unbiodegradable COD fractions as shown in Equation 9.1:

Equation 9.1: $Tot_{Org} = S_F + S_{VFA} + X_B + C_B + X_U + S_U$ (mgCOD/L)

Recognising that organic nitrogen is the nitrogen content of the different organic compound groups, and adding the other inorganic nitrogen compounds (such as ammonia, nitrite and nitrate), the nitrogenous compounds can also be fractionated as (Figure 9.2):

Tot_N = total Kjeldahl nitrogen (TKN)
$Tot_{Ig,N}$ = total inorganic nitrogen
$Tot_{Org,N}$ = total organic nitrogen
NH_X = total free and saline ammonia
NO_X = total nitrite plus nitrate
$Tot_{Org,B,N}$ = total organic biodegradable nitrogen
$Tot_{Org,U,N}$ = total organic unbiodegradable nitrogen

Organic nitrogen can be divided into similar fractions such as COD because nitrogen is another component of the same organic groups. Thus, organic biodegradable nitrogen compounds are divided into particulate biodegradable ($XC_{B,N}$), which (bio)degrades more slowly, and soluble biodegradable ($S_{B,N}$), that is easily biodegradable.

The unbiodegradable organically bound nitrogen comprises particulate unbiodegradable and soluble unbiodegradable fractions (X_U,N and $S_{U,N}$ respectively). Because these organic groups are not degraded and remain unaffected by the biological processes, they remain intact, keeping their nitrogen (and COD and phosphorus) composition and characteristics. Therefore, in a treatment plant $X_{U,N}$ accumulates in the system and is added to the sludge mass, whereas $S_{U,N}$ leaves the plant through the effluent because it does not settle out and is not biologically removed. So, the unbiodegradable COD and organic nitrogen is simply the COD and nitrogen content of the unbiodegradable organics.

Therefore, Tot_N can be expressed as shown in Equation 9.2:

Equation 9.2: $$Tot_N = S_{NHx} + S_{NOx} + XC_{B,N} + S_{B,N} + X_{U,N} + S_{U,N} \quad (mgN/L)$$

In addition to the organic and nitrogenous compounds, wastewater also contains inorganic suspended solids (ISS) as part of the total suspended solids (Table 2.3). Bacteria are able to utilise small concentrations of ISS as trace elements or micronutrients for cell growth (e.g. magnesium, potassium and calcium compounds). However, they are not considered biodegradable. Consequently, the ISS tend to accumulate in wastewater treatment proportionally to the solids retention time (SRT) (Ekama, 2008).

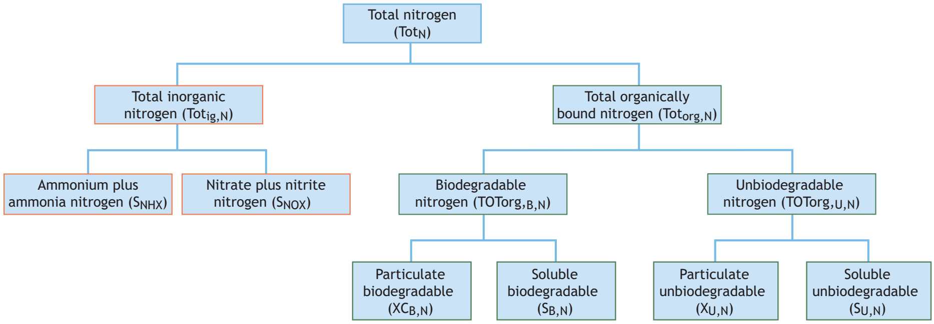

Figure 9.2 Nitrogen fractionation diagram (adapted based on Melcer, 2003 and Corominas *et al.*, 2010).

Table 9.2 Defined COD, TN and TSS concentrations for fresh and digested faecal sludge and high, medium, and low strength (Dangol, 2013; Hooijmans *et al.*, 2013)

Sludge type	Strength	COD (mg/L)	Total N (mg/L)	TSS (mg/L)
Fresh	High	250,000	5,000	100,000
	Medium	65,000	3,400	53,000
	Low	10,000	2,000	7,000
Digested	High	90,000	1,500	45,000
	Medium	45,000	400	25,000
	Low	3,000	200	1,500

9.2.3 Faecal sludge strength

FS can be classified as digested and fresh, and as high, medium and low strength, based on the COD and total nitrogen (TN) concentrations (Dangol, 2013; Hooijmans *et al.*, 2013). The values in Table 9.2 were defined by Dangol (2013) and Hooijmans *et al.* (2013) for modelling purposes based on values reported in the literature (Koné and Strauss, 2004; Heinss *et al.*, 1998; Elmitwalli *et al.*, 2006; Luostarinen *et al.*, 2007; Henze and Comeau, 2008; Halalsheh *et al.*, 2011; Ingallinella *et al.*, 2002).

Fractionations of WWTP influents have been carried out since the beginning of mathematical modeling of activated sludge systems, and examples can be readily found in the literature (Ekama *et al.*, 1986; Henze *et al.*, 1987). In contrast, literature reporting the fractionation of FS is not readily available. Examples found in literature are reported in Table 9.3. Interestingly, two different groups can be identified regardless of the strength, FS with higher fractions of biodegradable organics (up to 81% of the total COD), and FS with lower fractions of biodegradable organics (of around 43%). Consequently, the latter is more digested containing about 57% unbiodegradable organics.

Overall, the biodegradable organics in fresh FS can reach up to 82% of total COD (Table 9.3). The differences in biodegradable organics can be explained by the retention time of FS in the onsite sanitation system. Short retention times (e.g. days in public toilets) do not allow for significant stabilisation, whereas longer retention times (e.g. years in septic tanks) do. Elmitwalli *et al.* (2011), through mathematical simulations, estimated that after 90 days of accumulation in onsite systems the biodegradable fractions in fresh FS decreased from 0.81 to 0.25, whereas the unbiodegradable fractions increased from 0.19 to 0.75. This suggests the importance of matching treatment technologies to sludge types, e.g. biogas generation would be more suitable with sludge that is emptied frequently, or treated in situ. Interestingly, the COD fractionations of fresh and digested FS do not show considerable variations in spite of their strength and origin. Nevertheless, data is still limited and more studies are needed to be conclusive.

One study has reported N-fractionations of FS, as summarised in Table 9.4 (Dangol, 2013). N-fractionation of digested and fresh FS was estimated following a similar approach to Ekama (2008) for influent wastewater, and Lake (2010) for septic tank sludge. Based on the assumption that onsite systems partly function as anaerobic digesters (Montangero and Belevi, 2007), the biodegradation of organics leads to the production of fermentable organics and fermentation products (S_F and S_{VFA}, respectively) and to the release of inorganic nitrogen compounds (mostly NH_4^+ since a 6-8 pH range is usually observed) from the hydrolysis of organic nitrogen (Sötemann *et al.*, 2005). Thus, the biodegradable organic nitrogen fractions in FS can be included and therefore lumped on the free and saline ammonia (FSA) because they are eventually (and rapidly) hydrolysed. This assumption was based on the long retention times, and high solids and biomass concentrations found in onsite systems (Dangol, 2013).

Table 9.3 Faecal sludge COD fractionation

Origin	Total COD (mg/L)	XCb (slowly biodegradable)		Xu (particulate unbiodegradable)		Xa (acidogenic bacteria)		Sf (fermentable organic matter)		Svfa (volatile fatty acids)		Su (soluble unbiodegradable)		Sum of bio-degradable fractions	Sum of nonbio-degradable fractions
		(mgCOD/L)	Fraction	(mgCOD/L)	Fraction	(mgCOD/L)	Fraction	(mgCOD/L)	Fraction	(mgCOD/L)	Fraction	(mgCOD/L)	Fraction		
Fresh faecal sludge															
Vacuum toilet for black water (VBW)[1]	10,000	6,940	**0.69**	1,110	**0.11**	480	0.05	240	0.02	940	0.09	290	0.03	0.81	0.19
Vacuum toilet for faeces separation (VF)[1]	65,000	42,380	**0.65**	7,215	**0.11**	3,120	0.05	2,145	0.03	8,580	0.13	1560	0.02	0.82	0.18
Dry toilet (DT)[1]	45,000	31,230	**0.69**	4,990	**0.11**	2,160	0.05	1,080	0.02	4,230	0.09	1,310	0.03	0.81	0.19
Dry toilet for faeces with urine separation (DT)[1]	20,0000	130,400	**0.65**	22,200	**0.11**	9,600	0.05	6,600	0.03	26,400	0.13	4,800	0.02	0.82	0.18
Filter bag (FB)[1]	250,000	163,000	**0.65**	27,750	**0.11**	12,000	0.05	8,250	0.03	33,000	0.13	6,000	0.02	0.82	0.18
Bio-toilet mixed with saw dust[2]			**0.80**		**0.20**		-		-		-		-	0.80	0.20
Average fractions			**0.69**		**0.13**	**-**	**0.05**	**-**	**0.03**	**-**	**0.12**	**-**	**0.03**	**0.81 ± 0.01**	**0.19 ± 0.01**
Digested faecal sludge[3]															
High strength septic sludge[4]	90,000	34,118	**0.38**	53,882	**0.60**			1,176	0.01			824	0.01	0.39	0.61
Low strength septic sludge[4]	6,000	2,235	**0.37**	3,565	**0.59**			118	0.02			82	0.01	0.39	0.61
Septic sludge[5]	2,186	568	**0.26**	1,218	**0.56**			262	0.12			138	0.06	0.38	0.62
Septic tank sludge Jordan winter (18.4 °C)[6]	2,969	1,318	**0.44**	814	**0.27**			484	0.16			353	0.12	0.61	0.39
Septic tank sludge Jordan summer (21.9 °C)[6]	6,425	615	**0.10**	2,254	**0.35**			1,949	0.30			1,607	0.25	0.40	0.60
Average fractions			**0.31**		**0.47**				**0.13**				**0.09**	**0.43 ± 0.10**	**0.57 ± 0.10**

1 Gaillard (2002); Elmitwalli *et al*. (2006); Luostarinen *et al*. (2007) 2 Lopez-Zavala *et al*. (2004)
3 Biodegradable COD fractions estimated based on the STOWA protocol (Roeleveld and van Loosdrecht, 2002) 4 Henze *et al*. (2002) 5 Lake (2010) 6 Halalsheh *et al*. (2011)

Table 9.4 Nitrogen fractionation for digested (septic tank) and fresh faecal sludge (Dangol, 2013)

Fraction	Notation	Value	
		Digested faecal sludge	Fresh faecal sludge
Free and saline ammonia (FSA)	S_{NHx}	0.20	0.46
Soluble biodegradable	$S_{B,N}$	-	-
Particulate biodegradable	$XC_{B,N}$	-	-
Organic unbiodegradable particulate nitrogen	$X_{U,N}$	0.05	0.01
Organic unbiodegradable soluble nitrogen	$S_{U,N}$	0.75	0.53
Total nitrogen	Tot_N	1.00	1.00

9.3 CO-TREATMENT IN ACTIVATED SLUDGE WASTEWATER TREATMENT SYSTEMS

9.3.1 Influence on removal efficiencies and effluent quality

When co-treating FS in activated sludge WWTPs, the COD and TN concentrations in the reactor and effluent will increase proportionally to the FS strength and influent volumes. In addition, concentrations of soluble unbiodegradable COD and TN will reduce the treated effluent quality because they cannot be removed by either physico-chemical or biological processes. Thus, influent volumes of high- and medium-strength FS will need to be limited to comply with effluent standards. As shown in Figures 9.3 and 9.4, this is confirmed through mathematical modelling of an activated sludge treatment plant with an installed capacity of 100,000 p.e. (20,000 m^3/d) treating medium strength municipal wastewater and performing biological nitrogen removal (Henze *et al.*, 2008; Dangol, 2013). As observed, the influent COD and TN concentrations increase proportionally to the volumes of FS in the influent, reaching the highest concentrations with high strength fresh FS (Figure 9.3).

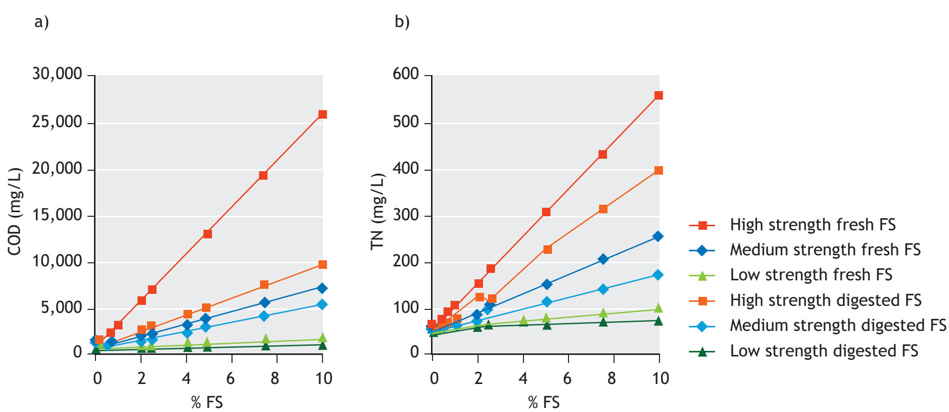

Figure 9.3 Effects of the combined discharge of municipal wastewater and faecal sludge (expressed as a percentage of the total influent discharged to the plant) on: (a) influent COD and (b) influent TN concentrations of an activated sludge wastewater treatment plant (Dangol, 2013).

It was also confirmed that the higher concentrations of soluble unbiodegradable fractions leads to higher effluent COD and TN concentrations (Figure 9.4). Thus, the soluble unbiodegradable COD and TN concentrations will set the first limit for the allowable FS volumes based on the compliance of certain effluent standards. For example, to meet the effluent requirements of <100-120 mgCOD/L and <10 mgTN/L, only 1.75-2.0% or 0.75-1.0% of the total influent could be comprised of medium- or high-strength FS, respectively. However, if plants do not have enough spare capacity (e.g. aeration, tank volumes, settling tanks and sludge handling facilities) the actual allowable volumes will probably be much lower due to the considerably higher loads discharged to the plants. For instance, 1% addition (equivalent to 200 m^3/d or 40 tankers of 5 m^3) of low strength digested FS (containing 3,000 mg COD/L as shown in Table 9.2) leads to a COD load increase of 600 kgCOD/d. This corresponds to an increase of 6,667 p.e. (assuming 1 p.e. = 90 gCOD/d), which may have a marginal effect on the 100,000 p.e. plant capacity. However, using the same approach, 1% discharge of medium- or high-strength fresh FS can equal the contribution of about 144,500 p.e. and 555,500 p.e., respectively, although this is at the upper limit of what would still allow for adequate plant operation.

Low-strength FS (e.g. from pit latrines with long residence times or infrequent emptying) does not have the same pronounced effects because of the lower concentrations of unbiodegradable COD and TN. However, assuming that there is enough spare capacity (e.g. aeration, tank volumes, settling tanks and sludge handling facilities), it will not meet the effluent requirements when it approaches 10% of the influent volume (corresponding to an increase of 66,667 p.e. and up to 222,220 p.e. for digested and fresh FS, respectively). This is similar to the recommendation of Still and Foxon (2012) of keeping the FS-to-influent wastewater ratio at no more than 1-10 to avoid a process failure at the plant.

9.3.2 Effects on oxygen demand

Aerobic treatment systems have limited aeration capacities. Co-treatment with FS can result in a severe increase in the oxygen demand due to the high concentrations of biodegradable COD and TN of FS. As observed in Figure 9.5, the effects of influent FS are so high that they can increase the relative oxygen demand (ΔFO_{TOT}) by 200%, with only 1% high-strength FS by volume in the influent, or 2% with medium-strength fresh FS. Prior to co-treatment with FS, the oxygen demand of the FS needs to be determined to evaluate whether the plant has enough aeration capacity to avoid process disruption.

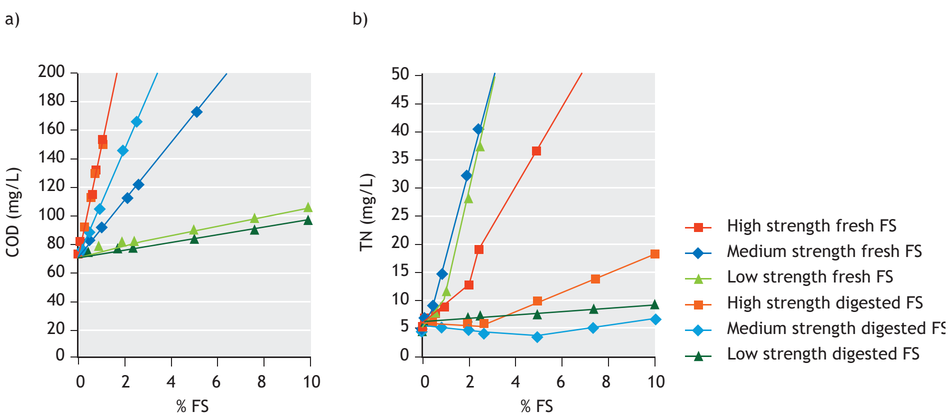

Figure 9.4 Effects of the combined discharge of municipal wastewater and faecal sludge (expressed as a percentage of the total influent discharged to the plant) on: (a) COD and (b) TN concentrations in the effluent of an activated sludge wastewater treatment plant.

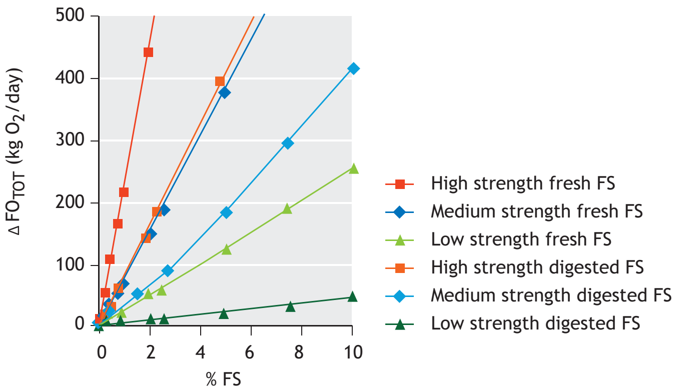

Figure 9.5 Relative increase in oxygen demand in an activated sludge wastewater treatment plant as a function of the combined discharge of municipal wastewater and different faecal sludge volumes (expressed as a percentage of the total influent discharged to the plant) (Dangol, 2013).

9.3.3 Impact on sludge generation

The accumulation of TSS is the limiting parameter for the co-treatment of FS. If the increase exceeds the maximum capacity, the plant can experience serious operational problems ranging from overloading of aeration and secondary settling tanks (with associated solid-liquid separation problems) to a considerable decrease in the oxygen transfer efficiency (which can lead to insufficient aeration and therefore to oxygen limiting conditions). As illustrated in Figure 9.6, at FS influent volumes as low as 0.5% for medium- and high-strength FS and of 2.5% for low-strength, the plant is overloaded and exceeds the maximum concentration of 5 kgTSS/m^3 recommended for the operation of aeration tanks (Metcalf and Eddy, 2003).

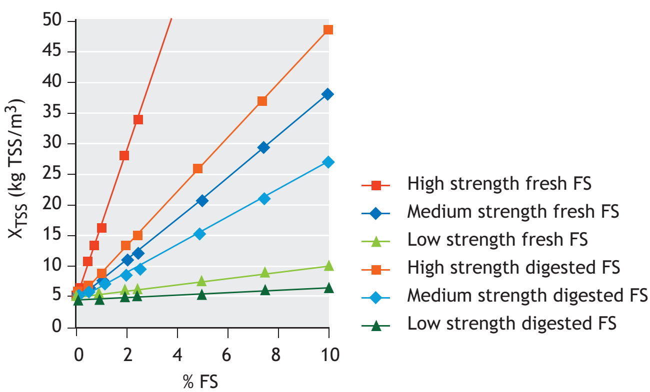

Figure 9.6 Increase in total suspended solids concentrations in the aeration tank of an activated sludge wastewater treatment plant expressed as a function of the combined discharge of municipal wastewater and of different volumes of faecal sludge (expressed as a percentage of the total influent discharged at the plant).

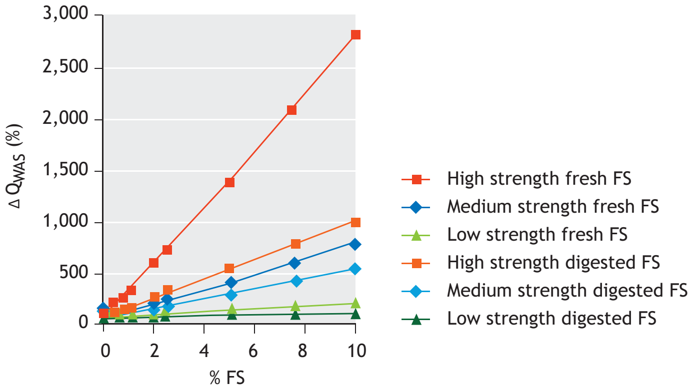

Figure 9.7 Increase in sludge production in an activated sludge wastewater treatment plant as a function of the combined discharge of municipal wastewater and of different volumes of faecal sludge (expressed as a percentage of the total influent discharged at the plant) (Dangol, 2013).

In addition, the increase in TSS and mixed liquor suspended solids (MLSS) concentrations will also result in increased volumes of waste sludge. There must be sufficient capacity in the sludge handling and disposal/enduse facilities of the plant to deal and cope with the higher sludge volumes generated, which frequently is not the case (Still and Foxon, 2012). For example, as shown in Figure 9.7, with a 100% increase in sludge production (ΔQ_{WAS}) the sludge handing facilities need to double their capacity for the co-treatment of 10% low-strength (by volume), 1% medium-strength, and 0.5% high-strength FS (Dangol, 2013).

9.3.4 Impact on aeration requirements

The increased accumulation of solids from co-treatment of FS can also lead to a reduction in the oxygen transfer efficiency. This will further increase the aeration requirements and reduce the aeration capacity of the plant. If the aeration capacity is exceeded, this will lead to oxygen limiting conditions, the creation of unaerated sections and serious operational problems. As shown in Figure 9.8, if the influent contains 2% high- or medium-strength FS by volume the demand on the aeration capacity will increase by 200%, and if it is 10% influent low-strength digested sludge this will increase by 100% (Dangol, 2013).

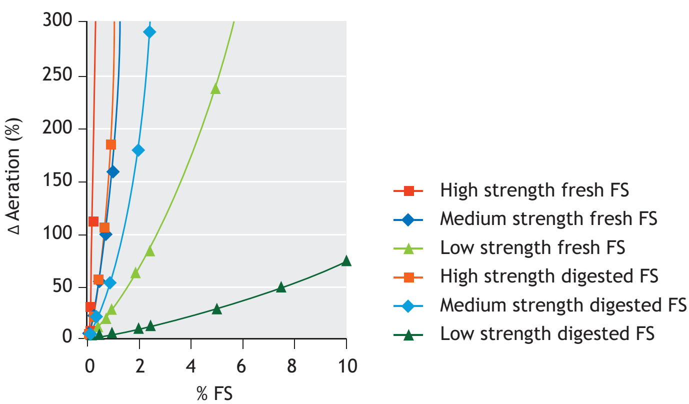

Figure 9.8 Estimation of the minimum increase in aeration requirements in an activated sludge wastewater treatment plant as a function of the combined discharge of municipal wastewater and of different volumes of faecal sludge (expressed as a percentage of the total influent discharged at the plant) (Dangol, 2013).

Potential detrimental effects caused by insufficient aeration supply include:

- Low dissolved oxygen (DO) concentrations in the aeration tank (< 1.5 mgO_2/L), or even oxygen depletion (0 mgO_2/L), resulting in incomplete oxidation of organics, a deterioration of effluent quality, high COD concentrations in the effluent, and leading to incomplete (at DO < 1 mg/L) or even cessation (at 0 mgO_2/L) of nitrification and, under extreme oxygen deficiency (for several hours), to the inactivation of bacteria.
- Creation of anaerobic pockets within the aerobic tanks resulting in the reduction of the net SRT of the aerobic system (SRT^{net}_{aer}). Such a reduction will be inversely proportional to the size of the anaerobic section(s). In particular, the SRT^{net}_{aer} drops below the minimum SRT of nitrifying organisms, this will result in the washout of nitrifying bacteria and cessation of nitrification.
- Proliferation of filamentous bacteria if the DO concentrations are below 1.5-2.0 mgO_2/L, to the detriment of desired heterotrophic and nitrifying bacteria (Martins *et al.*, 2004). Filamentous bacteria also lead to bulking sludge that does not settle well, and affects the biomass retention capacity in the secondary settling tanks. This results not only in a major increase in effluent TSS and VSS concentrations, and therefore reduced effluent quality, but also in several sludge loss from the system via the effluent. It could ultimately affect the whole operation of the treatment plant if the actual SRT drops below the minimum required values for biomass growth (Ekama, 2010).
- Partial nitrification of the high N load in FS could also result in accumulation of high nitrite concentrations (>100 $mgNO_2$-N/L) due to the oxygen limiting conditions, which would be toxic to desired heterotrophic and nitrifying bacteria. High nitrite concentrations can also have significantly negative impacts on the receiving water body where the plant effluent is discharged.

9.3.5 Impact on secondary settling tanks

Increased TSS resulting from co-treatment of FS can also overload secondary settling tanks (clarifiers). This results in problems with solids-liquid separation, solids being washed out in the effluent, and reduced biomass within the system making it difficult to maintain a stable SRT. As illustrated in Figure 9.9, the minimum surface area (A^{MIN}_{SST}) for settling tanks increases considerably with the addition of FS. 1-2% FS by volume of high- and medium-strength FS, either fresh or digested, can result in an increase in required area of more than 300% (Dangol, 2013). For low strength FS at 5 to 10%, the required area is 200% larger. Prior to co-treatment with FS, it is very important to evaluate the A^{MIN}_{SST} to determine if an adequate area is available, without assuming deterioration in sludge settleability (Ekama and Marais, 1986, 2004; Ekama *et al.*, 1997).

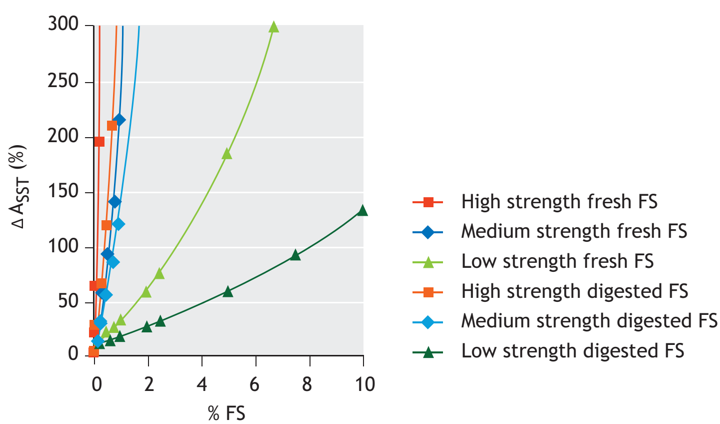

Figure 9.9 Estimation of the minimum area of the secondary settling tank required for an activated sludge wastewater treatment plant as a function of the combined discharge of municipal wastewater and of different volumes of faecal sludge (expressed as a percentage of the total influent discharged at the plant).

Table 9.5 Maximum faecal sludge volumes that can be co-treated under steady- and dynamic conditions in an activated sludge plant performing biological nitrogen removal without causing any process disruption or effluent deterioration (Dangol, 2013)

Faecal sludge type and strength	Under steady state conditions (%)	Under dynamic conditions (%)	Approximate ratio between maximum allowable faecal sludge volumes under steady-state to dynamic conditions
Digested FS			
Low-strength FS	3.75	0.64	6.0
Medium-strength FS	0.375	0.375	1.0
High-strength FS	0.25	0.25	1.0
Fresh FS			
Low-strength FS	0.375	0.125	3.0
Medium-strength FS	0.25	0.025	10.0
High-strength FS	0.125	0.025	5.0

9.3.6 Effects of the dynamic discharge of faecal sludge

Another complication when co-treating FS is the very dynamic nature of the influent FS. FS flow rates will tend to be much more dynamic than wastewater because they are not just dependent on diurnal patterns, they are also dependent on factors such as the working schedule of service providers, the customer demand for collection services, and the season. The result is heavy peak loads during the busiest times that can overload the treatment plant. Based on modelling, Dangol (2013) concluded that, under dynamic conditions, the maximum volumes that can be co-treated in an activated sludge plant without causing any process disruption or (effluent) deterioration sometimes need to be up to 10 times lower than those allowable under steady-state conditions (Table 9.5). Dangol (2013) conducted further modelling to evaluate the discharge of FS during off-peak hours (e.g. following a similar dynamic discharge pattern during the night) and the potential contribution of primary sedimentation tanks. The modelling showed no improvement in plant performance under dynamic conditions. This illustrates the importance of equalisation tanks to ensure a more even loading, and the need to distribute influent FS evenly through the entire day to approach steady-state conditions.

9.4 PRACTICAL CONSIDERATIONS FOR CO-TREATMENT OF FAECAL SLUDGE IN ACTIVATED SLUDGE SYSTEMS

Overall, the co-treatment of FS in activated sludge WWTPs can lead to severe operational problems at FS influent volumes as low as 0.025% of the total influent wastewater flow rate (which is equivalent to only one tanker of 5 m^3 per day). Thus, if the co-treatment of FS is to be employed, a very careful evaluation of the WWTP capacity needs to be made to determine which unit operation is the bottleneck of the plant (out of aeration, secondary settling tanks or sludge treatment) and how the plant is likely to fail. This will require a careful assessment and the implementation of defined measures to avoid any process disruption and deterioration of the plant. Considerations that need to be taken into account include:

- Required effluent standards. To estimate the minimum effluent COD and TN concentrations to verify the compliance with the required effluent standards.

- Maximum TSS concentrations in aeration tanks. To calculate the maximum expected TSS to evaluate if the aeration tanks will be overloaded.
- Maximum sludge production. To evaluate if the sludge handling and disposal facilities have the capacity to deal with the increase in sludge waste generation.
- Maximum installed aeration capacity. To estimate the aeration requirements based on the increase in oxygen demand and decrease in oxygen transfer efficiency. For existing plants, the DO concentration needs to be carefully monitored to maintain a concentration of at least 2 mgO_2/L.
- Secondary settling tanks. To determine the minimum surface area required for the operation of the settling tanks for the observed sludge settleability (in terms of the sludge volume index -SVI- or any other similar parameter).
- Existence and performance of equalisation tanks. To allow an even discharge of FS to the sewage plant for the longest period possible (e.g. over 24 h).

For new WWTPs that expect to receive certain volumes of FS or that are *a priori* designed to co-treat FS, the previous aspects can be used and applied to adapt the design depending upon the discharge volumes, type and strength of the FS. However, the design will probably lead to larger tank volumes, larger settling tanks, and higher installed capacity for aeration and sludge handling, treatment and disposal. For instance, compared to municipal wastewater treatment alone, for 1% FS co-treatment (regardless of the strength), the tank volumes will need to be 300% larger, the aeration capacity at least 200% higher, the secondary settling tanks 5 times larger and the sludge facilities 4 times larger. These aspects will undoubtedly considerably increase the capital and operational costs of the plant, along with the operational capacity. These considerations should be weighed carefully alongside other less expensive and more robust options presented in this book.

Case Study 9.1: Co-treatment in activated sludge wastewater treatment plants in eThekwini, South Africa

(Adapted from Still and Foxon, 2012 and Wilson and Harrison, 2012)

In spite of the apparent relatively low volumes of FS from pit latrines, two activated sludge WWTPs located in eThekwini, South Africa experienced serious operational problems caused by the high loads of organics, nitrogen compounds and suspended solids (Wilson and Harrison, 2012). A complete inactivation of the nitrification process was observed in one of the plants, which took several months to recover (Still and Foxon, 2012). A hypothesis suggests that the excessive nitrogen load discharged into the plant was the main reason (Still and Foxon, 2012). Although the causes of the problems are unclear, it cannot be discounted that the aeration capacity was exceeded as a consequence of the high loads discharged, resulting in the cessation of the nitrification process as discussed in this chapter. At the other plant under study, the high solids overloading made it practically impossible to remove the excess sludge generated as it was equal to the sludge volume produced in a month. Sludge removal was limited by the number of truckloads that could be removed, increasing associated operational costs and even the willingness of the receiving landfill to accept the material (Still and Foxon, 2012). The (digested) FS from the pits rapidly accumulated in the system and, because mixed sludge waste could not be removed at the required rate, it was retained for an extended period affecting the operation of the plant. This phenomenon resembles the excess sludge increase displayed in Figure 9.7. As Still and Foxon (2012) point out, it was clearly a case of taking one solids problem and turning it into another solids disposal problem, indicating that co-treatment in an activated sludge wastewater treatment plant can rarely be sustainable or successful.

Case Study 9.2: Co-treatment of septic tank sludge in an activated sludge wastewater treatment plant on Saint Marten, Netherlands Antilles

(Adapted from Lake, 2010 and Lopez-Vazquez, 2008)

Although the high concentrations of solids, organics and nitrogen compounds in FS attract most attention, the higher concentrations of unbiodegradable compounds and low biodegradability of organics can also hinder compliance with the effluent limits. On the island of Saint Marten, a popular tourist destination in the Caribbean, there was around 10% sewerage coverage until 2010 (Lake, 2010). Wastewater and septic tank sludge (brought by tankers to the plant) (Figure 9.11) were discharged into the existing Illidge Road WWTP, located in the Cul-de-Sac district. The plant consisted of an Imhoff tank with a volume of 154 m^3 as well as a buffer tank, secondary settling tank and sludge drying beds. The plant capacity was considerably exceeded by the wastewater flow rate (of at least 65 m^3/h) and the high FS volumes that in a typical working day accounted for an equivalent of about 175 m^3/day (Lopez-Vazquez, 2008). Since the plant was obsolete, a Modified Bardenpho (A2O) process design was proposed to achieve strict discharge standards for COD, N, P and TSS (of 125, 10, 2 and 20 mg/L, respectively). Based on local space-planning development plans, different scenarios were evaluated through mathematical modelling where the effects of the expansion of the sewer network (from 10 to 85% coverage) and population growth were taken into account (Lake, 2010). This approach helped to assess their effects on wastewater composition and WWTP performance through an estimated life span of 25 years (Lake *et al.*, 2011). Due to the loads of unbiodegradadable particulate organic matter and unbiodegradable soluble organic nitrogen from the digested FS, the study suggested that the proposed plant would only be able to comply with most of the discharge limits when the FS volumes comprise of no more than 2.8% of the influent (Lake *et al.*, 2011). However, as a consequence of the high nitrogen load and slow biodegradability of biodegradable organics (highlighted in Table 9.1), the study speculates that the nitrogen limits will probably not be met at the new plant (Lake, 2010).

Figure 9.10 Faecal sludge discharge at Illidge Road wastewater treatment plant at the buffer tank.

Case Study 9.3: Co-treatment impact on the Albireh wastewater treatment plant, Palestine

(Adapted from Al-Sa'ed and Hithnawi, 2006)

Following a similar approach as on Saint Marten, an assessment using mathematical modelling was carried out at the Albireh WWTP, located in the West Bank, Palestine. The purpose was to find an explanation for the occurrence of filamentous bulking sludge and the high effluent COD and TN concentrations that did not allow the corresponding discharge limits of 90 mgCOD/L and 18 mgTN/L to be met (Al-Sa'ed and Hithnawi, 2006). Like other plants in the region, since 2000 Albireh WWTP has been co-treating septic tank sludge from some of the 35% households not connected to the sewerage network. The modelling study indicated that, when the volumes of low-strength digested FS reached 6.6% of the total influent, the plant capacity was exceeded, requiring about 50% larger tank volumes, 50% higher oxygen requirements and generating a similar percentage of excess sludge (Al-Sa'ed and Hithnawi, 2006). The higher oxygen requirements and solids overloading might have favoured the proliferation of filamentous bacteria due to difficulties in keeping adequate aerobic conditions.

Case Study 9.4: Co-treatment of FS in Manila, Philippines

(Adapted from Robbins *et al.*, 2012)

In spite of the unsatisfactory experiences with FS co-treatment in aerobic treatment plants, activated sludge systems have recently been chosen in the Philippines as the main biological treatment process for FS treatment. Manila Water's FS operations with septic tank sludge currently utilise a FS treatment with activated sludge in the Manila South septage treatment plant (Robbins *et al.*, 2012). The plant is able to treat up to 814 m^3 per day of FS. Currently, the plant handles about 40-50% of its maximum capacity, indicating that there is still room for growth. In addition, the septage management system of the Baliwag water district has decided to build a septage treatment plant that utilises a sequencing batch reactor (a variant of the activated sludge process) as a secondary treatment process (http://watsanexp.ning.com). The full operation of the plant is scheduled by the second half of 2013. This project intends to serve as a model for water district-led septage management in the Philippines. These experiences indicate that co-treatment of FS in aerobic biological systems can be feasible and satisfactory if the design is adequate to cope with the FS influent, there is adequate operator capacity and competence, and an appropriate management scheme is implemented.

9.5 ANAEROBIC CO-TREATMENT OF FAECAL SLUDGE

The co-treatment of FS and wastewater in anaerobic processes is an alternative for sludge stabilisation, volume reduction and increased dewaterability. Possibilities include upflow anaerobic sludge blanket reactors (UASB), anaerobic digesters and anaerobic ponds. Anaerobic treatment can offset treatment costs through the production of biogas, which can be used for heating or for the generation of electricity. Pathogen reduction can also be achieved with thermophilic digestion (Metcalf and Eddy, 2003).

The characteristics of FS need to be carefully considered, as fresh or less stabilised FS will have higher concentrations of biodegradable organics but possibly also of inhibiting compounds (as discussed below). Although the biogas production and utilisation is an attractive benefit, there are currently limited applications and technologies. Therefore, further research is needed for the development of anaerobic systems for the co-treatment of high strength FS (Strauss *et al.*, 2006). FS from septic tanks (digested FS) may not be appropriate for anaerobic co-treatment, depending on the level of stabilisation it has undergone. In this case, the low concentrations of biodegradable organics in digested FS will lead to low biogas production but high solids accumulation resulting in significant operational costs with limited benefits (Still and Foxon, 2012).

9.5.1 COD overloading

As explained in Chapter 3, anaerobic digestion relies on complex interactions and dependencies among diverse bacterial groups, which makes the process susceptible to variations in influent loading rates. This is particularly important when managing FS, which by nature is highly variable. Anaerobic degradation has four stages: hydrolysis, acidogenesis, acetogenesis and methanogenesis (both acetoclastic and hydrogenotrophic). The growth rate of the fermentative bacteria that carry out acidogenesis is 10 to 20 times higher than methanogens, and their process rates five times faster (van Lier, 2008). If reactors are overloaded, the faster rate of acidogenesis will result in an accumulation of acids, as the methanogenic bacteria are not able to utilise them as fast. Depending upon the buffer capacity of the system (which depends on the nitrogen content of the organics since the hydrolysis of organic nitrogen results in an alkalinity increase), this can lead to a significant drop in pH, which inhibits the growth of methanogens, and thereby results in an even greater accumulation of acids (van Lier, 2008). This results in digester failure, and is referred to as 'souring'. In this regard, Moosbrugger *et al.* (1993) developed a simple 5-point titration method to measure both the VFA and alkalinity for anaerobic digestion control and early detection of instability to avoid 'souring'.

Anaerobic treatment processes are disrupted by overloading of COD, ammonia inhibition, pH variations, and sulfide inhibition. Therefore, these factors need to be carefully monitored, and controlled, to ensure proper operation of co-treatment of FS in anaerobic treatment systems. Each of these factors is explained below, and also how they affect appropriate FS loading rates.

UASB

To prevent overloading, the maximum COD or VSS design loading rates must not be exceeded, and reactors must have consistent and uniform feeding (Metcalf and Eddy, 2003). Figure 9.11 presents the effects of loading different FS volumes and types as a percentage of the total influent on an UASB system designed for 100,000 p.e. and operated at 25˚C. The design values that were used for the UASB were medium strength municipal wastewater as described by Henze and Comeau (2008) and used by Dangol (2013). The organic loading rate (OLR) was 3 kgCOD/m^3/d and the upflow velocity 0.83 m/h. The maximum OLR for UASB systems treating wastewater with high concentrations of particulate biodegradable organics is around 6 kgCOD/m^3/d (van Lier, 2008), which suggests that, in principle, the 100,000 p.e. plant used in this study has enough spare capacity. As illustrated in Figure 9.11, the UASB reactor can handle feedings of up to 7.5% by volume of low strength fresh FS (1,500 m^3/d equivalent to the organic load of up to 180,000 p.e.), but only 0.25% high strength fresh FS due to the high COD content (10 tankers of 5m^3 per day but with an organic load equivalent to approximately 139,000 p.e.). This means that the 100,000 p.e. UASB system, as well as other UASB plants of different capacities, could handle low strength FS but are prone to overloading with high strength FS.

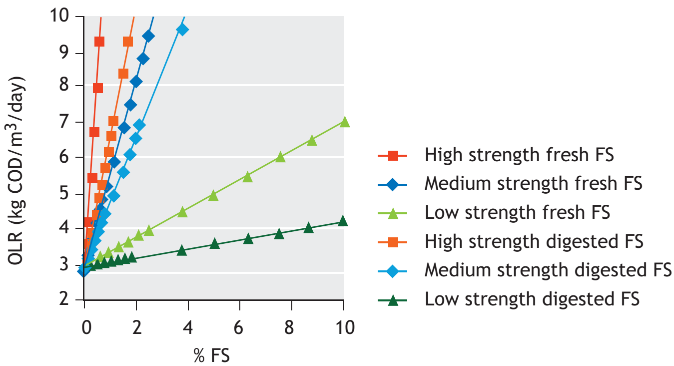

Figure 9.11 Effects of faecal sludge discharge (expressed as a percentage of the total influent discharged to the plant) on the organic loading rates of a UASB reactor designed for an average and a maximum oxygen loading rate of 3 kgCOD/m3/day and 6 kgCOD/m3/day, respectively.

Anaerobic digesters

Figure 9.12 illustrates the effect of the co-treatment of different FS types at different solids loading rates (SLR) as a percentage by volume of the total influent on an anaerobic digester. The anaerobic digester was designed to treat sludge from an activated sludge treatment plant under mesophilic conditions (35°C), with a SRT of 10 d, and a total volume of 13,750 m^3. As shown, the SRT decreases proportionally to the amount of FS being fed. Although the maximum recommended value for SLR is 4.8 kgVSS/m^3/d (Metcalf and Eddy, 2003), this needs to be monitored carefully so that FS addition does not result in a drop in the SRT below the minimum recommended, causing the reactor to fail. For example, if the anaerobic digester feeding is 1% (138 m^3/d, equivalent to 28 collection and transport trucks with a 5 m^3 volume), but it contains 10% FS, an approximate 10% reduction in the operating SRT of the digester could occur (Figure 9.12).

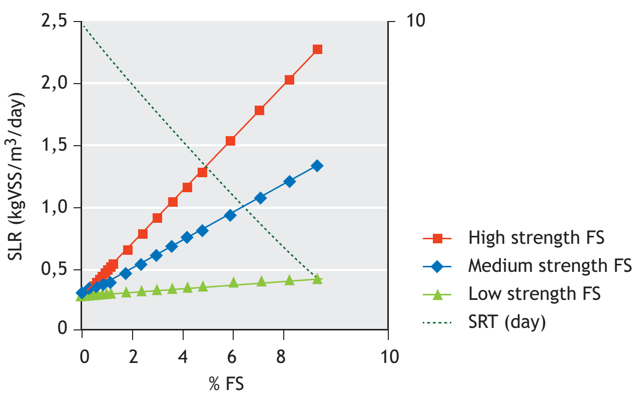

Figure 9.12 Effects of faecal sludge discharge (expressed as a percentage of the total influent discharged to the anaerobic digester) on the solid loading rates of an anaerobic digester of 13,750 m^3 designed with a sludge retention time of 10 days.

For anaerobic co-treatment in digesters, it is recommended that the feeding, including FS, is always lower than one twentieth of the digester volume (ATV, 1985). This approach would mean a maximum 5% FS loading, regardless of its strength, to prevent overloading or significant reduction in the SRT. This ratio is also based on an SRT value of 20 d commonly used for the design of anaerobic digesters (Metcalf and Eddy, 2003), which increases the reliability of the recommended approach.

Ponds

Usually, anaerobic ponds can be regarded as low loaded anaerobic systems with operational loading rates of 0.025-0.5 kgCOD/m^3/d and depths of 4 m (van Lier, 2008). For FS applications, Fernandez *et al.* (2004) suggest the pre-treatment of FS in waste stabilisation ponds (WSP) operated at maximum loading rates of 0.6 kgBOD_5/m^3/d, particularly to reduce the generation of ammonia. However, these systems may have odour problems, fast sludge accumulation (0.010 to 0.020 m^3 sludge accumulated/m^3 FS) and therefore require frequent sludge removal (Heinss *et al.*, 1998; Fernandez *et al.*, 2004). Furthermore, the loss of methane into the atmosphere, which has a 21 times higher greenhouse impact than CO_2, is an environmental impact that needs to be considered if it is not captured (van Lier, 2008). Moreover, the effluents require further polishing prior to discharge into the environment, and tend to contain high ammonia concentrations that can affect post-treatment processes in pond systems or even within the same anaerobic ponds (Strauss *et al.*, 2000). Thus, the application of anaerobic ponds for FS treatment needs to be carefully evaluated, particularly when dealing with high strength FS. This is also covered in Chapter 5.

9.5.2 Ammonia inhibition

The anaerobic co-treatment of FS can be inhibited by the high concentrations of ammonia present in FS (Still and Foxon, 2012). Among the bacteria in anaerobic reactors, methanogenic bacteria are the most sensitive to ammonia inhibition (Chen *et al.*, 2008). Inhibition of the methanogens results in lower biogas yields in spite of the availability of soluble biodegradable organics (Angelidaki *et al.*, 1993; Chaggu, 2004). Reported values for inhibition of methanogens is quite variable, and 50% reduction in activity has been observed at total ammonia concentrations between 1.7 to 14 g/L (Chen *et al.*, 2008). The broad range is due to the influence of different factors such as pH, carbon source, temperature and biomass acclimation and adaptation (Chaggu, 2004; Chaggu *et al.*, 2007; Chen *et al.*, 2008). In this regard, free ammonia (NH_3) and not ammonium (NH_4^+) has been suggested as the actual toxic agent at concentrations of 100-200 mg/L for unadapted methanogenic populations (Henze and Harremoes, 1983).

To prevent process disruptions and deterioration, Heinss and Strauss (1999) recommend limiting the volume of co-treated FS based on a total influent ammonia concentration of less than 2 g/L. However, Doku (2002) recommends limiting the maximum FS volume to reach an influent that contains less than 200 mg NH_3-N/L based on potential variations in pH (Henze and Harremoes, 1983).

Based on the total nitrogen concentrations expected in co-treatment of wastewater and fresh FS (Figure 9.3), the nitrogen concentrations would probably be higher than 200 mg/L, indicating that their volumes need to be limited to no more than 2, 5 and 8% for high-, medium- and low-strength FS, respectively.

9.5.3 pH variations

In anaerobic systems, the pH needs to be carefully monitored and kept between 7.0 and 7.5 (Chen *et al.*, 2008). The alkalinity and buffer capacity of the anaerobic systems need to be monitored to ensure that the pH remains stable (Metcalf and Eddy, 2003). pH higher than 7.5-8.0 can lead to an accumulation of free ammonia, and extreme pH levels (e.g. higher than 10.0) can fully inhibit the anaerobic biological degradation process (Chaggu, 2004; Chen *et al.*, 2008). pH values lower than 7.0 can reduce the methanogenic activity. A pH of 7.0-7.5 helps to maximise the biomass activity and reduce the potential inhibiting and toxicity effects of parameters such as ammonia and volatile fatty acids (Chen *et al.*, 2008).

Thus, to monitor, and if possible adjust, the alkalinity levels and buffer capacity of the system can help to reduce pH fluctuations and maintain an adequate pH range. However, certain practices, such as gradual feeding and the controlled addition of external compounds (including charcoal ashes to enhance pathogen removal and nutrient recovery), need to be carefully performed (Chaggu, 2004; Metcalf and Eddy, 2003). Otherwise, they may lead to pH decreases due to VFA accumulation (when overloading) or to extremely high pH levels (when overdosing alkaline or basic compounds) (Chaggu, 2004; van Lier, 2008).

9.5.4 Sulphide inhibition

Sulphide gas (H_2S) is generated during the anaerobic digestion of sludge that is rich in proteins, and due to (saline) groundwater intrusion or infiltration into the onsite sanitation system (Metcalf and Eddy, 2003; Lopez-Vazquez *et al.*, 2009). Sulphide is toxic to all living organisms and can easily affect the anaerobic digestion processes. Methanogenic bacteria are rather sensitive to sulphide leading to lower methane production, low quality biogas, bad smell, corrosion problems and high COD effluent concentrations (van Lier, 2008).

50% methanogenic activity has been observed at sulphide concentrations between 50-250 mgS/L, but H_2S is usually present in the gas phase (Metcalf and Eddy, 2003). Because the pK_{S1} value for H_2S to HS- dissociation is around 7.0, the pH should be maintained above 7.0 to keep the H_2S concentration low. Although relatively low sulphate concentrations can be expected due to the low FS volumes co-treated, the potential sulphide generation cannot be ignored since the anaerobic process may be prone to disruption at sulphide concentrations as low as 50 mgS/L depending upon other operational conditions (e.g. pH). However, data regarding the sulphate concentrations contained in FS is rare and therefore they need to be studied to assess their potential influence on the anaerobic processes when co-treating FS.

9.6 PRACTICAL CONSIDERATIONS FOR CO-TREATMENT OF FAECAL SLUDGE IN ANAEROBIC SYSTEMS

For any anaerobic treatment process, probably the most important operational aspect is the feeding. It needs to be supplied gradually and if possible continuously to avoid overloading and shocks (Heinss and Strauss, 1999; Metcalf and Eddy, 2003; van Lier, 2008).

For FS co-treatment in UASB reactors, the maximum OLR of design (including both wastewater and FS) must not be exceeded in order to avoid the overloading of the system. In particular, high strength FS needs to be carefully handled since the high organic content can easily overload the system. In this study, 0.25% high strength fresh FS (approximately 10 tankers of 5 m^3 per day) had an organic load equivalent to around 139,000 p.e. that led to the overloading of a 100,000 p.e. UASB plant.

Anaerobic digesters appear to be more robust for the anaerobic co-treatment of FS. Permissible loading rates for mesophilic digesters (operated at 35˚C) depend on the operational conditions but can reach up to 1.6-2.0 kgVSS/m^3/d (Heinss and Strauss, 1999; Metcalf and Eddy, 2003). Also, the feeding, including FS, needs to be limited to the maximum daily feed rate of design which depends on the applied SRT. Thermophilic anaerobic digesters (49-52ºC) are an alternative that can lead to faster hydrolysis rates (the rate limiting step in anaerobic digestion of wastewater and FS) resulting in higher biogas yields (Angelidaki *et al.*, 1993). However, they are susceptible to small temperature variations and also operating and maintenance costs are higher compared to mesophilic digesters, which make them unattractive for low-income countries (Heinss and Strauss, 1999).

Ponds appear to be cost-effective technologies for FS co-treatment when operated as low loaded systems (0.6 $kgBOD_5/m^3/d$). However, their implementation needs to be carefully evaluated because the initial investment and operational costs could be high since they have substantial land requirements and high operational costs as a consequence of the frequent desludging needed. Moreover, they can involve important environmental issues if methane is lost into the atmosphere.

Case Study 9.5: Treatment of faecal sludge in Dar es Salaam, Tanzania

(Adapted from Chaggu, 2004)

The detrimental effects of high ammonia concentrations and high pH levels need to be avoided to ensure a satisfactory performance of anaerobic digestion systems. Chaggu (2004) carried out a literature research on excreta handling in Dar es Salaam, Tanzania. He found that 50% of the filling up of pits in Dar es Salaam City was the result of a high water table and that almost 16,131 kgCOD/day from pit-latrines reached the groundwater sources. As such, he proposed to use a 3,000 L plastic tank as an experimental improved pit-latrine without urine separation for a 10-person household in Mlalakuwa settlement in Dar es Salaam City. The influent to the reactor consisted of urine and faeces in a 1.3:1 ratio. The results obtained revealed that, after 380 days of use as a daily pit-latrine, the reactor content was not yet stabilised, and 8,000 mg/L dissolved COD (but only 100 mgCOD/L as volatile fatty acids) were still present. Part of this dissolved COD was biodegradable, indicating the need for further stabilisation of the reactor content. The slow conversion of dissolved COD was assumed to be related to the inoculation of anaerobic sludge not adapted to the high ammonia concentration of 3,000 mg N/L. In the same research project, a short survey revealed that high pH values occur (up to pH 10.4) in Ecosan toilets due to addition of charcoal ashes to enhance the reduction of 'E-Coli and Ascaris eggs', but the high pH levels inhibited the anaerobic biological degradation of FS.

Case Study 9.6: Co-treatment of septage in a lab-scale UASB reactor in Ghana

(Adapted from Doku, 2002)

Although full-scale experiences are limited, Doku (2002) concluded that it is feasible to treat FS in a laboratory scale UASB reactor in Ghana provided (i) the sludge is diluted appropriately to avoid reaching high concentrations of inhibitory compounds (e.g. ammonia) and (ii) the FS is gradually and continuously fed into the reactor. Doku (2002) executed the experiments using a UASB reactor with a working volume of 50 L, operated at a mean HRT of around 12 h, and at ambient temperatures between 23.0 and 31.2 °C. The OLR was between 12.5 to 21.5 kg $COD/m^3/d$, with a relatively low upflow velocity of 0.14 m/h. The FS was diluted to a 1:6 ratio, resulting in an average total nitrogen concentration of 300 ± 50 mg/L. The average removal efficiencies were: 71% for COD, 61% for total solids, 74% for total volatile solids TVS and 73% for TSS. The calculated volume of methane in the biogas collected ranged from 4-8 L/kgCOD, not accounting for practical losses. Overall, the removal efficiencies were comparable to those obtained for a UASB reactor treating domestic sewage. However, the effluent COD concentration is too high for direct discharge and hence a form of post-treatment would be necessary. Nevertheless, full-scale studies are needed to validate the observations from this research.

9.7 CONCLUSIONS

The discharge of FS for its co-treatment in WWTPs can lead to severe operational problems when even low volumes of high-strength fresh FS are discharged (e.g. 0.25% of the total influent). This is mainly caused by the relatively higher strength of FS compared to that of municipal wastewater, which can easily lead to higher loads exceeding the plant capacity. The most common problems are the overloading of solids, COD or nitrogen compounds. They can lead to serious operational problems ranging from incomplete removal of organics to cessation of nitrification, which can take several weeks to recover. Also, the excessive solids accumulation may lead to unexpectedly highly sludge generation that can compromise the operation of the plant and increase the operational costs. Moreover, aerobic treatment systems may experience a lack of aeration capacity and severe overloading of secondary settling tanks leading to solids loss. Meanwhile, anaerobic systems are prone to inhibition by the presence of inhibitory compounds such as ammonia and pH variations. In addition, the high concentrations of soluble unbiodegradable organics and nitrogen compounds can have a serious effect on the treated effluent quality, which may hinder compliance with the required effluent standards.

If in spite of the apparent limited benefits, FS co-treatment is to be practiced in municipal WWTPs, the allowable FS volumes will probably need to be restricted to low volumes so that WWTPs do not get overloaded with total suspended solids, high COD and nitrogen loadings or high concentrations of toxic or inhibitory compounds. Moreover, FS loadings need to be added gradually and as slowly as possible to avoid overloads and shocks.

All the previous aspects need to be carefully addressed but, overall, the benefits do not seem to be attractive enough to support the co-treatment of FS with wastewater in municipal WWTPs, particularly when dealing with digested FS from septic tanks which contains low concentrations of biodegradable compounds but high concentrations of solids that will tend to overload the treatment systems. It is possible that anaerobic co-treatment of fresh FS offers certain opportunities when considering the potential recovery of resources, but further research is still needed for the development of reliable and cost-effective technologies.

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End of Chapter Study Questions

1. What are the common technical problems likely to be experienced in the co-treatment of FS in wastewater treatment plants?

2. Why is it important to determine the oxygen demand of the FS prior to co-treatment?

3. Explain the reason why the accumulation of TSS is a limiting parameter for the co-treatment of FS.