Anaerobic Pre-treatment of Sewage Under Low Temperature (15 ºC) Conditions in an Integrated UASB-Digester System

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For
the people of
Nour Shams Camp
whom I love most

ABSTRACT

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The main objective of this thesis was to amend the UASB by the means of incorporating a digester for anaerobic sewage treatment in the Middle East region.

A literature search of the various parameters that might affect the solid liquid separation process by filtration through the sludge bed of a UASB has been elaborated. The overall output of this study includes, literature review, structuring this field of science and highlighting fields where research is needed.

The investigation of sewage characteristics in Palestine revealed that the sewage is of high strength and low temperature during wintertime *ca*. 15 ºC. Accordingly, prediction based on model calculations revealed that the application of the one stage UASB reactor for sewage treatment in Palestine is only possible when designed at a HRT of more than 22 hours. Alternatively, a novel technology consisting of a Digester complementing a UASB, namely UASB-Digester system, was proposed.

The results of primary sludge stabilisation in CSTRs revealed that the major achievement of sludge stabilisation occurred at SRT ≥ 10 and 15 days at process temperatures of 35 and 25 ºC, respectively. The sludge dewatering results showed the existence of optimal SRT at 20 and 15 days at process temperatures of 25 and 35 ºC, respectively. While, the sludge settling is not affected by digestion.

Running a pilot-scale one stage UASB and a UASB-Digester system on sewage from the village of Bennekom, The Netherlands validated the technical viability of the UASB-Digester system. The UASB reactor was operated at an HRT of 6 hours and temperature of 15 °C, the wintertime sewage temperature in Palestine, and the digester was operated at 35 °C. The achieved removal efficiencies of total COD, suspended COD, colloidal COD and dissolved COD of respectively 66, 87, 44 and 30 in the UASB-Digester is significantly higher than the respectively 44, 73, 3 and 5 achieved in the one stage UASB and as high as those reported for tropical countries. The conversion in the UASB of the UASB-Digester system is substantially higher than in the one stage, viz. the percentage methanogenesis of the influent COD were 21 and 44% in the first and second systems, respectively. The sludge produced from the UASB of the UASB-Digester system is substantially lower and more stable as compared with the sludge in the one stage UASB reactor. The settleability of the sludge of the UASB, UASB of the UASB-Digester system and the digester sludges reveals high settleability with no influence of the digestion conditions. The wasted sludge from the UASB of the UASB-Digester system is better dewaterable as compared to that of the UASB reactor.

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Chapter 1

INTRODUCTION

INTRODUCTION

1.1 BACKGROUND

An adequate protection of the environment is an imperative fundamental for all living creatures. The disposal of untreated sewage is a serious environmental and public health risk, leading to water sources, soil and air pollution and consequently affects man, animal and plants. Sewage treatment by conventional means, including primary sedimentation and secondary aerobic biological treatment, is very efficient. However, this efficiency is at the price of high capital and running costs and technology requirement (El-Gohary *et al.*, 1995). The application of these expensive systems, which are popular in Europe and America, does not offer a sustainable solution for sewage treatment in less wealthy countries. Anaerobic treatment has been proven to be an admirable process and considered by many authors as the core of sustainable waste management (Zeeman and Lettinga, 1999; Hammes *et al.*, 2000; Gijzen, 2001).

The application of anaerobic technologies for sewage treatment dates back over 100 years. The first full scale application of anaerobic sewage treatment was in a reactor resembling the septic tank in the 1860's, and was called "Mouras Automatic Scavenger". Afterwards, the technological development proceeded via introducing the anaerobic filter in 1880's, the anaerobic hybrid system in about 1890 or 1891 by W. D. Scott Moncrieff and the septic tank in 1895 by Donald Cameron. The effluents of these septic tanks were often black and offensive, and contained high amounts of solids, which tend to clog the contact beds that were often used for subsequent treatment. Consequently, Harry W. Clark at Lawrence, Massachusetts, suggested in 1899 to solve this problem by digesting the sludge by itself in a separate tank. This was the first move towards separate sludge digestion. In 1905, Karl Imhoff introduced the Imhoff tank, which consisted of a settler located over a storage tank for digestion of settled sludge. By that time, anaerobic technologies were started to be shifted from treatment of wastewaters to the treatment of settled sludge. The Imhoff tank was not the complete solution as it was tall, and the digestion chamber had to be connected intimately with the sedimentation tank. Afterwards, efforts began with separate digestion of sludge which was successful in practice in 1927 when the Ruhrverband at Essen-Rellinghausen installed the first heated sludge digester. In the 1950's, the importance of the sludge retention concept for reducing the reactor size began to be recognised (McCarty, 2001).

One of the major successes in the development of anaerobic wastewater treatment was the introduction of high-rate reactors in which biomass retention and liquid retention are uncoupled (Lettinga *et al.*, 2001). The escalation of energy costs in the 1970's and the considerable costs of the construction, maintenance and operation of aerobic systems resulted in attention of anaerobic high-rate systems in terms of research and application. The success of the anaerobic high–rate systems is due to the possibility of application of a relatively high loading rate, while maintaining long SRT at relatively short HRT due to sludge immobilisation. In these systems, wastewater flows through the anaerobic sludge where purification takes place through complex bio - physical - chemical interrelated processes. Organic matter is converted into biogas (mainly methane which is a useful end product and carbon dioxide) and sludge. The main advantages and drawbacks of the anaerobic high rate systems are shown in Table 1.

Table 1. Advantages and drawbacks of anaerobic sewage treatment (Lettinga *et al*., 1993; Schellinkhout, 1993; Zeeman and Lettinga, 1999; Foresti, 2001; Gijzen, 2001; Lettinga, 2001)

Advantages	Drawbacks
– A substantial saving in operational costs as no energy is required for aeration; on the contrary energy is produced in the form of methane gas, which can be utilized for heating or electricity production. Hence, it couples the degradation of organic materials from waste to the production of energy. – The process can handle high hydraulic and organic loading rates. Thus, the applied technologies are rather compact and reduce the volume of post treatment stages. – The technologies are simple in construction and operation, consequently they are low cost technologies. – The systems can be applied everywhere and at any scale as little if any energy is required, enabling a decentralized application. This unique privilege reflects the systems flexibility, besides the fact that the decentralized mode leads to very significant savings in the investment costs of sewerage systems. – The excess sludge production is low. In addition, the sludge is well stabilized and easily dewatered due to high solids retention time (SRT). Thus, the sludge does not require extensive costly post treatment. – The valuable nutrients (N and P) are conserved which give high potential for crop irrigation and aquaculture. – Feasible for a wide range of waste and wastewaters, i.e. complex in composition, a very low and very high strength, low and high temperatures.	– Need for post treatment, depending on the requirements for effluent standards. – No experience with full-scale application at low/moderate temperatures. – Considerable amount of produced biogas i.e. CH_4 and H_2S remains in the effluent especially for low strength wastewater (sewage). – Produced CH_4 during anaerobic sewage treatment is often not utilised for energy production.

Definitely, the anaerobic high-rate systems meet the sustainability criteria set in Table 2. Different high–rate systems were developed over the last three decades including the anaerobic filter (Young and McCarty, 1969), the upflow anaerobic sludge blanket (Lettinga *et al.*, 1980), the fluidised and expanded bed reactors (Schwitzenbaum and Jewell, 1980) and the down flow stationary fixed film reactor (Murray and van den Berg, 1981) and the baffled reactors (Barber and Stuckey, 1999). Lettinga and co-workers introduced and thoroughly investigated several modified versions of the UASB system namely, the EGSB system (Man *et al.*, 1988), the UASB - septic tank reactor (Bogte *et al.*, 1993), the HUSB (Wang, 1994), the staged multi-phase anaerobic (SMPA) reactor (Lier, 1995) and the two-stage AF-AH system (Elmitwalli, 2000).

Table 2. Sustainability criteria to be met for environmental technologies (after Lettinga, 2001)

– Little if any use of mineral resources and energy.
– Enabling production of resource/ energy from wastes.
– Pairing high efficiency with long term of lives.
– Applicable at any place and at any scale.
– Plain in construction, operation and maintenance.

1.2 APPLICATION OF HIGH RATE SYSTEMS FOR SEWAGE TREATMENT

The UASB reactor is the most widely and successfully used high rate anaerobic technology for treating several types of wastewater (Foresti, 2001; Lettinga, 2001; McCarty, 2001). The success of the UASB reactor can be attributed to its capability to retain a high concentration of sludge and efficient solids, liquid and water phase separation.

1.2.1 Application of the UASB Reactor in tropical countries

The experience with the applicability of the one stage UASB reactor system for sewage treatment in tropical countries started by the pilot plant constructed in Cali-Columbia during the period 1982 - 1983. The results obtained from the operation of the Cali plant showed the feasibility of the system under the prevailing environmental and sewage characteristics (Haandel and Lettinga, 1994). After that, hundreds of UASB reactors for treatment of sewage at both full scale and pilot scale (Tables 4 and 5) have been operational (Sousa and Foresti, 1996; Kalker *et al.*, 1999; Florencio *et al.*, 2001; Foresti, 2001; Wiegant, 2001) in several tropical countries like India, Columbia, Brazil and Mexico (Alaerts *et al*., 1990; Alaerts *et al*., 1993; Agrawal *et al*., 1997; Tare *et al*., 1997; Monroy *et al*., 2000; Lettinga, 2001). The ambient temperature in these countries is rather high throughout the year (20-35 °C) (Haandel and Lettinga, 1994) and the wastewater strength is rather low.

1.2.2 Application of the UASB Reactor at moderate and low temperature

Anaerobic sewage treatment in countries with a low or moderate temperature climate is a real challenge for researchers in the field of environmental technology (Lettinga, 2001; Lettinga *et al*., 2001). However the investigations, which have been carried out, by several researchers (i.e. Lettinga and his co – workers) represented a commendable move towards the understanding of the involved complex processes and the development of a series of novel technologies. The results of several researches on bench scale and pilot scale systems operated at low temperatures have opened new perspectives (Table 5) but no full-scale application has so far been realised (Zeeman and Lettinga, 1999; Lettinga *et al.*, 2001; Zakkour *et al.*, 2001). Nevertheless, experience with the application of one stage UASB reactor system at low temperature and high influent SS concentrations as found in many Middle East countries (Lier and Lettinga, 1999) is still to be developed.

1.3 KEY ISSUES OF ANAEROBIC SEWAGE TREATMENT AT LOW/MODERATE TEMPERATURE

The design and performance of an anaerobic reactor strongly depends on the solids retention time, operational temperature, and the biodegradability and concentration of the entrapped solids, which are interrelated parameters. These factors are discussed below.

1.3.1 Solids Retention Time and Temperature

Domestic wastewater is a complex type of wastewater, characterised by a high fraction suspended solids and mostly of relatively low temperatures (Seghezzo *et al.*, 1998; Lier *et al.*, 2001). The hydrolysis of retained particles is in general considered as the rate-limiting step of the overall digestion process and is highly influenced by process temperature and solids retention time (Pavlostathis and Giraldo-Gomez, 1991; Miron *et al.*, 2000; Sanders, 2001). When using one-step UASB reactors, the SRT should be long enough to provide methanogenic conditions, which imposes a long HRT at low temperature as illustrated in Table 3 (Zeeman and Lettinga, 1999). The performance of one step UASB systems at low temperature climates (5-20 °C) is highly limited by hydrolysis of the entrapped COD (Man, 1990). Indeed, anaerobic digestion at temperatures below 15 °C is possible, but in that case a solid retention time of more than 75 days is required (Zeeman and Lettinga, 1999). Zeeman (1991) found that methanogenesis could be achieved during anaerobic digestion of cow manure at 15 °C in CSTR reactors at a retention time of 100 days, while it cannot at 50 days. Likewise, O'Rourke (1968) showed that no significant methanogenesis was provided at the digestion of sewage sludge at a SRT of 60 days at 15 °C. Even at this long SRT no complete hydrolysis is expected (O'Rourke, 1968; Zeeman, 1991). Zeeman (1991) found the hydrolysis percentage at 125 days batch digestion of cow manure to be 12, 14, 18, 27, and 45% at process temperatures of 5, 10, 15, 25 and 30 °C, respectively.

The reduction in operational temperature does not only retards the hydrolysis step but also leads to a significant decrease in the maximum growth and substrate utilisation rates (Lettinga *et al.*, 2001). Last and Lettinga (1992) found a drop in the soluble COD removal efficiency from 53% at 20 °C to 20% when temperature is reduced below 10 °C during anaerobic treatment of pre-settled domestic sewage using an EGSB reactor. The authors reported that this could be due to the lower methanogenic activity of the sludge or a limitation in the maximum possible acidification of the soluble COD fraction in the pre settled sewage.

Table 3. HRT (days) to be applied, to achieve the indicated SRT (days) assuming 50% or 75% CODss removal at the treatment of domestic wastewater with a concentration of 1 g COD/l of which 65% is suspended, at different % hydrolysis and sludge concentration in the UASB reactor of 15 g VSS/l (after Zeeman and Lettinga, 1999)

% CODss removal	% Hydrolysis of removed SS	SRT				
		25	50	75	100	150
50	25	0.28	0.56	0.84	1.12	1.68
75	25	0.42	0.84	1.26	1.68	2.52
50	50	0.19	0.38	0.57	0.76	1.14
75	50	0.28	0.56	0.84	1.12	1.68
50	75	0.09	0.18	0.27	0.36	0.54
75	75	0.14	0.28	0.42	0.56	0.84

1.3.2 Suspended solids (SS)

The SS content of the wastewater is a primary factor that may affect the performance of an anaerobic reactor (Lettinga *et al.*, 1993; Zeeman and Lettinga, 1999; Kalogo and Verstraete, 1999; Lier *et al*., 2001). The SS present in the wastewater, depending on their characteristics and concentration, can affect the anaerobic treatment in the following ways (Lettinga and Hulshoff Pol, 1991):

- Reduce the specific methanogenic activity of the sludge due to adsorption and entrapment of poorly or non-biodegradable SS,
- Formation of scum layers,
- Counteract the formation of granular sludge,
- Spontaneous and sudden washout of the sludge, in case of prolonged continuous entrapment of SS in a granular sludge bed.

Lettinga *et al.* (1983) and Grin *et al.* (1985) detected during anaerobic digestion of domestic wastewater the adsorption of finely dispersed SS to the sludge granules. This phenomenon may hinder the sludge specific activities, through wrapping the active biomass particles with a film of increasing thickness of non-biomass matter, which consequently hamper the substrate transport through the active biofilm. Man *et al*. (1986) observed dilution of the active biomass during anaerobic digestion of municipal wastewater in a granular sludge UASB reactor as a result of entrapment of the SS. Likewise, Sayed (1987) reported accumulation of SS at the treatment of slaughterhouse wastewater in a granular sludge UASB system resulting in decrease of the specific methanogenic activity of the sludge when applying loading rates above 15 kg COD/m^3.d.

Last and Lettinga (1992) investigated the feasibility of anaerobic treatment of pre settled domestic sewage for low temperature conditions using one-step UASB and EGSB systems. They reported higher maximum removal efficiencies for soluble COD in EGSB systems, viz. 53% compared to 44% for UASB systems, while for total COD the UASB systems are more efficient, viz. 65% compared to 42% for EGSB systems. The significantly better contact between sludge and wastewater in the EGSB system as compared with conventional UASB reactors is reported to be the major advantage of the EGSB as compared to the UASB. The applied high upflow velocity in the EGSB systems scour the finely dispersed SS and colloidal matter attached to the sludge which will lead to substantial increase in the sludge activities due to enhancement of substrate transport through the sludge granules. However, the results clearly show the inefficiency of EGSB systems for removing SS. Processes, which apply a high upflow velocity, such as EGSB and the Fluidised Bed (FB) reactors are unsuitable for domestic sewage treatment, unless they are combined with an adequate pre – settling/treatment (Lettinga and Hulshoff Pol, 1991).

Wang (1994) investigated the fate of removed suspended solids in a 200-litre high loaded UASB called HUSB (Hydrolysis Upflow Sludge Blanket) reactor, operated at moderate temperature conditions (9-21°C). The methanogenic activity was lost almost completely after two months of operation. The latter should be attributed to the accumulation of non - soluble substrate at the imposed high SS loading rate (2.1 kg SS/ m^3.d), high hydraulic loading rate (1 m^3/m^2.d) and low HRT (3 hrs) and consequently low SRT. Also, the sludge stabilisation deteriorated substantially due to decreased solid retention time, and therefore, the system should be provided with a sludge digester.

The application of one step UASB systems for the treatment of complex wastewater containing a high amount of SS is usually limited by accumulation of these compounds in the sludge bed, especially at low temperatures (Zeeman and Lettinga, 1999). This accumulation reduces the solid retention time and the methanogenic activity of the sludge. Lettinga and Hulshoff Pol (1991) recognised this problem when stated, "For the treatment of partially soluble complex wastewater the required removal efficiency of the SS should be given special attention". The authors concluded that to minimise the investment costs of the treatment process, it is far better to combine the SS - removal with a sludge digester. Moreover, a satisfactory stabilisation of the accumulated SS particularly when operated under lower ambient temperature is a point of concern.

1.4 PROCESS LAY OUT FOR ANAEROBIC LOW/MODERATE TEMPERATURE TREATMENT

Several researchers have proposed the application of two-step systems for anaerobic sewage treatment at low temperatures (Wang, 1994; Elmitwalli, 2000). The first reactor aims mainly at physical solids removal in order to provide good process conditions for the functioning of the second reactor. The second step is normally optimised for VFA removal (Wang, 1994). Wang (1994) compared the performance of the HUSB with the performance of a primary settling tank and concluded that the first is much more efficient in removing SS. At HRT of *ca.* 2.5 hr, SS removal efficiencies of 80% and 40% were achieved in the HUSB and the primary settling tank, respectively. Incorporation of an EGSB reactor (HRT 2 hr) after the HUSB reactor (3 hr) aiming at soluble COD removal, proved to be an attractive technology for sewage treatment at a temperature of 13 °C (Wang, 1994). Likewise, Elmitwalli (2000) investigated the performance of a two-step AF+AH system for sewage treatment at 13 °C with special emphasis on colloidal COD removal, which was shown to be limited at low temperature (Yoda *et al.*, 1985; Wang, 1994). The results of both researchers are shown in Table 6. The main disadvantage of the two-step system is that the sludge from the first high loaded step is by definition not stabilised and needs further stabilisation in a separate digester (Lettinga *et al.*, 2001; Elmitwalli *et al.*, 2002).

Sayed and Fergala (1995) investigated the feasibility of further sludge stabilisation of accumulated domestic sludge in the UASB reactor in two identical flocculent sludge UASB reactors through an intermittent mode of operation at temperature of 18 - 20 °C. The experiments were carried out using raw domestic sewage at three HRTs (4, 6, and 8 hrs). The reactors were intermittently operated for periods of two days with and without feeding. The extent of sludge stabilisation during the feedless periods was not significantly affected by the HRT, while it was strongly during the feeding period. The total COD removal efficiency was almost 60% at all applied HRTs. While, respectively 14, 11 and 2 % of the removed COD was converted to CH_4 gas at the applied HRTs during the feeding period of 8, 6 and 4 hours.

Lettinga and Hulshoff Pol (1991) proposed a novel technology consisting of an integrated high loaded UASB and sludge digester for the treatment of sewage with a high concentration of suspended solids under low temperature conditions. Wang (1994) performed preliminary investigations on the proposed UASB-Digester system, in order to improve the sludge stability and the methanogenic activity inside the HUSB reactor by recirculating the accumulated HUSB sludge over a sludge digester. The results indicated that the HUSB sludge became further stabilised after incorporating the recuperation tank. However, minor information was provided by that research.

1.5 UASB- DIGESTER SYSTEM

The previous discussion demonstrates that the application of the conventional UASB systems is highly limited under sub-optimal temperature conditions (psychrophilic conditions) and a high concentration of suspended solids. Therefore, the one stage UASB should be adapted to overcome the cold winter period in countries like those in the Middle East region. Consequently, a modified system consisting of a one stage UASB system supplemented by a sludge digester (UASB-Digester) was investigated (Fig. 1).

In the proposed system, the raw suspended solids will be partially captured in the UASB reactor and conveyed to the digester, which should be operated under optimal process conditions with respect to temperature and SRT. The methanogens enriched digested sludge is recirculated to the UASB reactor to improve the methanogenic capacity of the UASB reactor. Moreover, the finely dispersed biodegradable solids, attached to the sludge flocs in the UASB, could be degraded in the digester, enhancing the substrate transport through the sludge flocs. Optimisation of the sludge recirculation should provide complete conversion of biodegradable dissolved COD. The UASB within this concept combines three main functions: improved primary sedimentation, sludge thickener and methanogenesis. Also, the digester has three main functions: sludge stabilisation resulting in energy production and improvement of sludge dewatering characteristics, and providing methanogenic bacteria. It is worth to mention that in scaling up the proposed system, the produced biogas in the digester should be used for heating the digester.

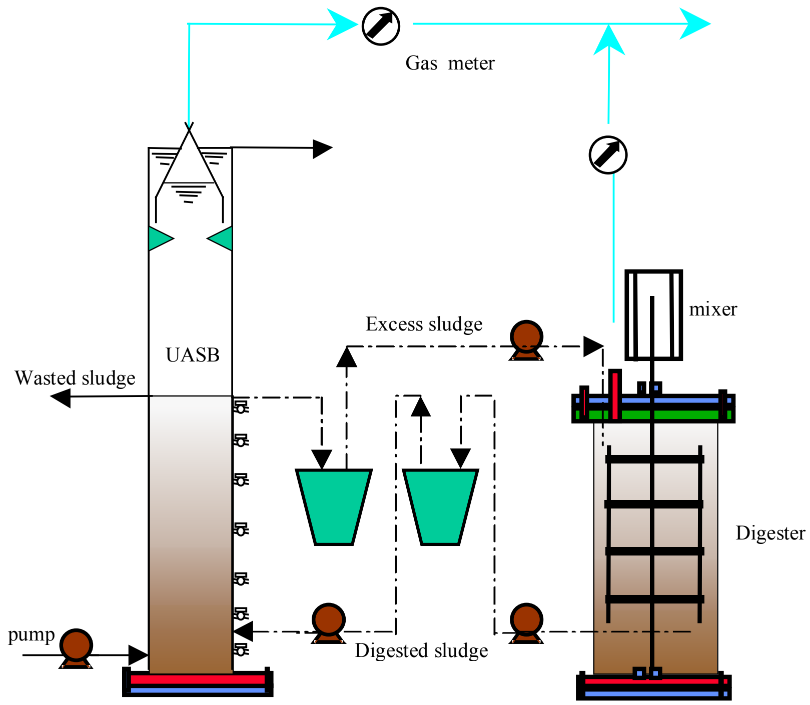

Fig 1. Schematic diagram of the UASB-digester system

Table 4. Results of anaerobic raw sewage treatment in pilot-scale reactors under tropical conditions (≥20 °C)

Reactor	Volume	Sludge	Temperature	HRT	Influent CODt	% Removal		Reference
	l		°C	(hr)	(mg/l)	CODt[$]	SS[$$]	
	106	F[+]	20-23	4	424	60	69	Vieira and Souza (1986)
	120	F[++]	20	18	550	55-75	-	Lettinga *et al.* (1980)
	118	F	20	8	500	75a	-	Grin *et al.* (1983)
	160	F	20	6	1076	64	88	Mergaert *et al.* (1992)
	106	F	21-25	4.7	265	50	73	Vieira (1988)
	106	F	35	4	300	65	61	Vieira (1988)
	8	F/G	20	10	350-500	60-75	-	Singh and Viraraghavan (1998)

[+] F: flocculent sludge
[++] F/G: mixed flocculent and granular sludge
[$]CODt: total COD
[$$]SS: suspended solids

Table 5. Results of anaerobic raw sewage treatment in full-scale reactors under tropical conditions (≥20 °C)

Reactor	Volume	Temperature	HRT	Influent CODt	% Removal		Reference
	m^3	°C	(hr)	(mg/l)	CODt[$]	SS[$$]	
UASB	64	24-26	4-6	267	65	70-85	Schellinkhout *et al.* (1985)
UASB	120	-	4.7-9	315-265	50-70	56-79	Vieira (1988)
UASB	1200	20-30	6	563	74	75	Draaijer *et al.* (1992)
UASB	6600	25	5.2	380	60-80	-	Schellinkhout & Collazos (1992)
UASB	67.5	23	7	402	74	87	Vieira *et al.* (1994)
UASB	810	30	9.7	563	67	61	Florencio *et al.* (2001)

[$]CODt: total COD
[$$]SS: suspended solids

Table 6. Results of sewage treatment in pilot-scale reactors at low temperature (≤ 20 °C)

Reactor	Volume	Sludge	Sewage	Temp	HRT	Influent CODt	% Removal		Reference
	l			°C	(hr)	(mg/l)	CODt[$]	SS(CODss)[$$]	
UASB	120	F	R[+++]	8-20	8	400	30-50	-	Grin *et al.* (1983)
UASB	120	G	R	7-8	9-14	464-700	57	72	Man *et al.* (1986)
AF	180	F	R	13-20	6	467	35-55	-	Derycke & Verstraete (1986)
UASB	110	F	R	12-18	18	465	65	(73)	Monroy *et al.* (1988)
UASB	120	G	R	12-20	7-8	190-1180	30-75	60	Man *et al.* (1988)
EGSB	120	G	S[++++]	>13	1-2	391	16-34	-	Last and Lettinga (1992)
UASB+EGSB	200+120	F, G	R	17	3,2b	697	69	79	Wang (1994)
UASB+EGSB	200+120	F, G	R	12	3,2b	507	51	67	Wang (1994)
UASB+UASB	42+4.6	F, G	R	18-20	8-4,2b	200-700	74-82	86-93	Sayed & Fergalla (1995)
AF+AH	60+65	F, F-G	R	13	2+4	528	59	(71)	Elmitwalli (2000)
AF+AH	60+65	F, F-G	R	13	3+6	528	63	(79)	Elmitwalli (2000)
AF+AH	60+65	F, F-G	R	13	4+8	461	71	(91)	Elmitwalli (2000)

[+] F: flocculent sludge

[++] F/G: mixed flocculent and granular sludge

[+++] R: raw sewage

[++++] S: settled sewage

[$] CODt: total COD

[$$] SS(CODss): suspended solids (suspended COD)

1.6 SCOPE OF THIS DISSERTATION

This Ph.D. thesis describes the results of a research concerning the assessment of anaerobic treatment of domestic sewage at a low temperature of 15 °C, the average sewage temperature in the Middle East during wintertime. The research focuses on several parameters that affect the anaerobic wastewater treatment and sludge stabilisation with special attention to optimising the novel UASB-Digester technology. Conceptual and theoretical backgrounds of the mechanisms and parameters that affect the particles separation from wastewater in mainly upflow anaerobic reactors have been addressed in **Chapter 2**. The experimental work described in **Chapter 3** deals with sewage characteristics in Palestine. In addition, different concepts based on decentralised collection and treatment has been stressed in order to sustain life in Palestine. **Chapter 4** deals mainly with optimising the digester of the UASB-Digester system in terms of sludge stabilisation. Moreover, the effects of temperature on the anaerobic biodegradability and hydrolysis rate constants have been dealt with. The effect of the degree of anaerobic digestion on sludge physical parameters, namely particle size distribution, dewaterability and settleability has been assessed in **Chapter 5**. The technical viability of the UASB-Digester system for sewage treatment at 15 °C has been tackled in **Chapter 6**. The general discussion and conclusions of the thesis are presented in **Chapter 7**.

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Chapter 2

SOLIDS REMOVAL IN UPFLOW ANAEROBIC REACTORS

SOLIDS REMOVAL IN UPFLOW ANAEROBIC REACTORS

BACKGROUND

The functioning of upflow anaerobic sludge blanket (UASB) systems depends on both physical parameters and biological processes, which determine the final removal efficiency and conversion of organic compounds. While the biological processes have been widely reported by the literature, the physical parameters and the physical-chemical mechanisms of solids removal have been scarcely reported. The mechanisms are complex and depend on various interrelated operational parameters. Better understanding of the interaction and role of these parameters is required for the development of anaerobic technologies. This desk study aims at investigating the mechanisms and parameters affecting the particle separation from wastewater, with the focus on upflow anaerobic reactors. The interaction of the various interrelated parameters and their relation to solids removal are discussed.

PARAMETERS AFFECTING SOLIDS REMOVAL IN UPFLOW REACTORS

Several parameters are likely to have an effect on particles removal in the sludge bed of a UASB. The major parameters are related to (1) reactor operational conditions (temperature, organic loading rate, hydraulic retention time and upflow velocity), (2) influent characteristics (concentration, particle size distribution and charges) and (3) sludge bed characteristics (particle size distribution, exopolymeric substances, charges, sludge hold up). These parameters and their effects are discussed in the following paragraphs.

1. REACTOR OPERATIONAL CONDITIONS

1.1 TEMPERATURE

Temperature affects the particles removal through influencing the wastewater viscosity and conversion of organic matter. The influence of temperature on the performance of classical filters “inert filtering media based filters” is discussed. The inert based filters have fixed bed and bioconversion is negligible as compared with biological filters like UASB.

1.1.1 VISCOSITY

It is often observed that the performance of classical filters, such as deep bed filter, and sedimentation tanks is better in summer than in winter, given comparable operational conditions (Metcalf and Eddy, 1991). The reason may be that increasing wastewater temperature decreases its viscosity, and consequently decreases the hydraulic shearing force on the particles. At low temperature, the viscosity of liquids will be higher, which implies that more energy is required for mixing in for example CSTR systems. In the treatment of water and wastewater the degree of mixing is measured by the velocity gradient, G. The velocity gradient is best thought of as the amount of shear taking place, the higher the G value, the more turbulent the fluid. The velocity gradient is a function of the power input into a unit volume of water (equation 1).

$$G = \sqrt{\frac{p}{\mu \quad * \quad V}} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots(1)$$

where:
G: velocity gradient (s^{-1})
P: power input (W)
V: volume of water in the reactor (m^3)
μ: dynamic viscosity (Pa.s)

Equation (1) was developed based on the idea that more power input creates more turbulence which, leads to better mixing (Metcalf and Eddy, 1991). Based on this equation, the effect of increasing the water temperature on the factor G and therefore the fluid mixing is calculated (Table 1).

Table 1. Relation between temperature and turbulence in the reactor

Temperature	Viscosity (μ) of water in the reactor	% Increase of G from 15 ºC
ºC	Pa.s	%
15	$1.14\text{x}10^{-03}$	-
20	$1.00\text{x}10^{-03}$	7
25	$8.90\text{x}10^{-04}$	13
30	$7.98\text{x}10^{-04}$	19
40	$6.53\text{x}10^{-04}$	32

The data presented in Table 1 reveal that increasing the wastewater temperature leads to more hydraulic turbulence in a reactor. Regarding upflow reactors where no forced mixing is applied, the upflow velocity and the gas production provide mixing. Increasing the wastewater temperature will not only enhance mixing by reducing viscosity, but also more biogas will be produced and hence much more turbulence could be expected. On one hand, increasing temperature will enhance the sedimentation and better contact between sludge and solids can be expected which could lead to better entrapment and adsorption. On the other hand, increasing temperature might lead to detachment of captured solids.

1.1.2 CONVERSION OF ENTRAPPED SOLIDS

The rate of anaerobic conversion of complex organic matter is, in most cases, limited by the hydrolysis step (Pavlostathis and Giraldo-Gomez, 1991). Hydrolysis has mostly been described with first order kinetics as shown in equation 2 (Eastman and Ferguson, 1981; Pavlostathis and Giraldo-Gomez, 1991).

$$\frac{dx_{degr.}}{dt} = k_h.X_{degr} \quad \text{...}(2)$$

where:
k_h: first order hydrolysis rate constant (d^{-1})
$X_{degr.}$: biodegradable substrate (kg COD/m^3)
t: time (d)

The hydrolysis rate constant is highly dependent on temperature, since hydrolysis is a bio-chemical reaction catalysed by enzymes, which are very sensitive to temperature (Sanders, 2001). The temperature effect on the hydrolysis rate constant can be described by the Arrhenius equation (equation 3) (Veeken and Hamelers, 1999).

$$k_h = Ae^{-E/RT} \quad \text{...}(3)$$

where:
k_h: hydrolysis rate constant (d^{-1})
A: the Arrhenius constant (d^{-1})
E: activation energy (kJ.$mole^{-1}$)
R: the gas law constant (J.mole $^{-1}$. $^oK^{-1}$)
T: the absolute temperature (^oK)

It can be concluded that the operational temperature has a substantial effect on the conversion of organic matter and consequently the characteristics of the sludge bed. The results of Lawler *et al*. (1986) demonstrate the effect of anaerobic digestion on the particle size distribution of sludges. When digestion works well, particles of all sizes are reduced with a special removal of small particles (conversion into gaseous form), i.e. the specific surface area will be reduced. When digestion does not work well, large particles are broken and small particles are created which results in a larger specific surface area. Therefore, the total surface area increases in the acidogenic stage and it decreases in the methanogenic stage. The surface area of the particles affects the physical behaviour of sludge, e.g. the dewaterability by providing frictional resistance to the withdrawal of water and a surface to which water can bind. Since, digestion affects the particle size distribution of the sludge, the degree of digestion and hence the temperature and the SRT are expected to highly influence the sludge capacity for solids removal, i.e. filtration, in case of upflow reactors. Moreover, it is likely that higher conversion rate will reduce the chance that a captured particle will be detached. On the other hand, at higher conversion rate higher gas production is expected which might counteract the removal efficiency.

1.2 ORGANIC LOADING RATE (OLR)

Several authors reported that up to a certain limit, the treatment efficiency of complex wastewaters, e.g. potato maize, slaughterhouse, in high rate anaerobic reactors increases with increasing OLR. A further increase of OLR will lead to operational problems like sludge bed flotation and excessive foaming at the gas-liquid interface in the gas-liquid-solid

(GLS) separator, as well as accumulation of undigested ingredients. As a result, the treatment efficiency deteriorates (Sayed, 1987; Ruiz *et al.*, 1997; Kalyuzhnyi *et al.*, 1998). Also accumulation of biogas in the sludge bed was noticed, forming stable gas pockets that lead to incidental lifting of parts of the bed and a pulse- like eruption of the gas from this zone (Kalyuzhnyi *et al.*, 1998; Elmitwalli *et al.*, 1999).

The OLR can be varied by changing the influent concentration and by changing the flow rate (equation 4). Changing the flow rate implies changing the HRT and the upflow velocity.

$$OLR = \frac{Q * COD}{V} = \frac{COD}{HRT} \qquad \text{..(4)}$$

where:
OLR: organic loading rate (kg $COD/m^3.d$)
COD: chemical oxygen demand (kg COD/m^3)
Q: flow rate (m^3/d)
V: reactor volume (m^3)
HRT: hydraulic retention time (d)

When the solids removal efficiency in upflow reactors is related to the OLR, it becomes crucial to distinguish between these parameters. For this reason, OLR is an inadequate design parameter to assure well performance of anaerobic reactors. Increasing the loading rate by reducing HRT down to a certain value will reduce the solids removal efficiency, probably due to increasing of upflow velocity (see Table 2: case I and II; Fig. 1). Since, increasing the influent concentration can increase the OLR, it becomes more essential to maintain an adequate upflow velocity to assure good mixing (Table 2: case I and III; sec. 1.4).

Table 2. Effect of changing OLR parameters on the removal efficiency (data compiled from Kalyuzhnyi *et al.*, 1998) during anaerobic treatment of raw potato maize wastewater in a UASB reactor at 35 ºC

	COD	OLR	HRT	V up	Reduction in treatment efficiency
	g/l	g COD/l.d	d	m/d	%
Case I	8.3	4.37 → 13.8	1.9 → 0.65	0.34 → 0.98	79 → 67.1
Case II	13.1	5.24 → 7.28	2.5 → 1.8	0.25 → 0.35	80 → 80
Case III	18	3 → 13.89	6 → 1.25	0.11 → 0.51	80 → SF

The undesirable phenomena, which manifest when treating wastewater with high-suspended solids, occur due to at least one of the following situations:

- High influent concentration; this will cause gas pockets formation, while the upflow velocity is low to create adequate turbulence in the sludge bed (poor mixing).

- Low hydraulic retention time (HRT) is accompanied by high upflow velocity (Vup), which will lead to wash out of influent solids and of viable biomass.
- High solids loading rate (SLR), which imposes low SRT, will change the sludge bed composition (microbial, physical, and chemical) and cause accumulation of floatable substances (proteins and lipids).

1.3 HYDRAULIC RETENTION TIME (HRT)/ SLUDGE RETENTION TIME (SRT)

Wang (1994) reported that, during anaerobic sewage treatment in a 170 m^3 hydrolysis upflow sludge bed (HUSB) reactor, HRT in the range (2.5 –5 hours) does not seriously affect the removal rate of the suspended solids. Differently, GonÇalves *et al*. (1994) show that the removal efficiency decreased with decreasing HRT accompanied by increase of upflow velocities (see sec. 1.4). It might be argued that the HRT is an inadequate parameter for describing solids removal in upflow reactors (Fig. 1). The effect of HRT could manifest as a result of its direct relation to the liquid upflow velocity (Vup) and also to the solids contact time in the reactor and so the possibility of solids to coalesce or to be entrapped in the sludge bed. Moreover, the HRT is a major parameter, which determines the SRT (Zeeman and Lettinga, 1999). The SRT can indirectly influence the solids removal as through changing of the physical-chemical and biological characteristics of the sludge bed in addition to biogas production.

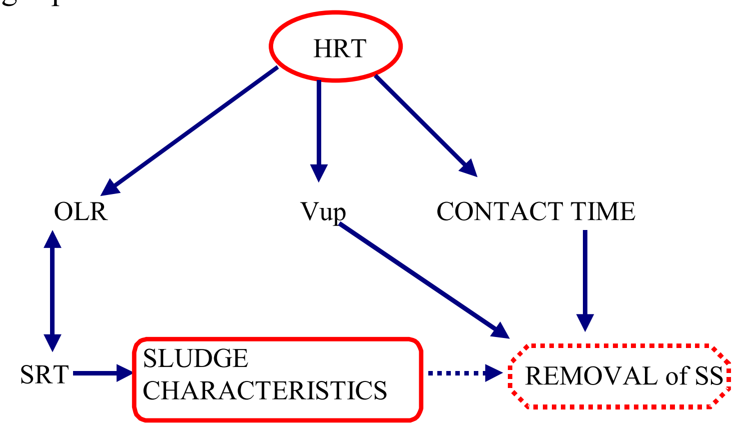

Fig. 1. The operational parameters which are expected to affect the solids removal in upflow reactor

1.4 UPFLOW VELOCITY (Vup)

The upflow velocity is one of the main factors affecting the efficiency of upflow reactors (Metcalf and Eddy, 1991; GonÇalves *et al*., 1994; Wiegant, 2001). The upflow velocity affects the sludge retention as it is based on the settling characteristics of sludge aggregates. Therefore, the upflow velocity could be a restrictive factor with respect to the required reactor volume when treating very low strength wastewater and wastewaters with high-suspended solids (Wiegant, 2001). The upflow velocity has two opposing effects. On one hand, increasing upflow velocity increases the rate of collisions between suspended particles and the sludge and thus might enhance the removal efficiency. On the other hand, increasing the upflow velocity could increase the hydraulic shearing force, which counteracts the removal mechanism through exceeding the settling velocity of more particles and detachment of the captured solids and consequently deteriorates the removal efficiency.

GonÇalves *et al*. (1994) treated sewage anaerobically at 20 ºC in an upflow anaerobic reactor (no GLS) operated at upflow velocities of 3.2, 1.7, 1.6, 0.9, 0.75 and 0.6 m/hr, corresponding to HRTs of 1.1, 2.1, 2.3, 2.8, 3.3 and 4.3 hr, respectively. They showed deterioration of removal efficiency as upflow velocity increases, varying from a value of 70% SS removal at 0.75 and 0.9 m/hr to 51% at 3.4 m/hr. The removal efficiency at an upflow velocity of 0.60 m/hr was, contradictory to these observations, only 60 % because of starting of methane production due to increase of HRT and accordingly the SRT. An increase in up flow velocity from 1.6 to 3.2 m/hr resulted in a relatively small loss in SS removal efficiency, from 55% to nearly 50%, which indicates the role of adsorption and entrapment (GonÇalves *et al.,* 1994; Zeeman *et al*., 1996). Man *et al*. (1986) found a significantly lower SS removal when the upflow velocity becomes higher than 0.50 m/hr during sewage treatment in a granular sludge-UASB reactor. Wiegant (2001), however presented data summarised from literature revealing no significant clear trend in the solids removal at increasing the upflow velocity in the range of 0.50 – 1.50 m/hr during sewage treatment in UASB reactors. These contradictory results might be explained by the occurrence of short-circuiting in the sludge bed (Wiegant, 2001).

The upflow velocity should be high enough to provide good contact between substrate and biomass, as it should be enough to disturb the gas pockets gathered in the sludge bed. The higher Vup is believed to facilitate the separation of gas bubbles from the surface of biomass (Hang and Byeong, 1990).

2. INFLUENT CHARACTERISTICS

2.1 INFLUENT CONCENTRATION

Wang (1994) noticed, during the operation of a pilot scale HUSB, that fluctuation in the influent concentration from ~180 to 700 mg COD/l at constant HRT of 2.5 hours, resulted only in a slight variation in effluent COD. Consequently, the removal efficiency increased from 20 to 60%. Similar results were reported by Chernicharo and Machado (1998), Zeeman and Lettinga (1999) and Elmitwalli *et al.* (2000). It can be concluded that there is a certain lower limit in the effluent solids concentration. Therefore, reactor performance could be clearer if described not only in terms of removal efficiency but also in terms of influent and effluent characteristics. Actually the available knowledge, about effluent characteristics, is very limited, e.g. particle size distribution, anaerobic and aerobic biodegradability and particles origin. In case the solids originate from the sludge, then the solids are stabilised, or from the influent itself then further technological development can enhance the solids removal efficiency.

The noticed increase of removal efficiency with increasing influent concentration could be due to at least one of the following reasons:

- Change in influent characteristics (increasing the percentage of settleable solids) as a consequence of for instance difference in the hydraulic regime of the wastewater stream, thus more turbulence in diluted streams and /or due to difference in the water ionic strength,
- Increase the collision opportunity of the influent solids with the sludge in the sludge bed,
- A certain amount of sludge washout that controls the amount of solids in the effluent, rather than the solids in the influent.

2.2 INFLUENT PARTICLE SIZE

There is no standard procedure to classify particles in wastewater as soluble, colloidal or suspended. Ødegaard (1999) defined the soluble fraction as the particles with a diameter d<1 nm, the colloidal 1nm<d<1 µm, and the suspended with d>1 µm. While, Wang (1994) considered the soluble, colloidal and suspended to have a diameter d <0.45 µm, 0.45 μm <d<4.4 µm, and d>4.4 µm, respectively.

The effluent quality from inert based filters is highly related to the influent characteristics (Landa *et al.*, 1997). It is well known that the treatability of wastewater depends strongly on the size distribution of the contaminants, since rates of sedimentation, adsorption, diffusion, and biochemical reactions are all influenced by particle size (Levine *et al.*, 1985; Kaminski *et al.*, 1997). The settling velocity of particulate matter is roughly proportional to the square of the particle size in accordance with Stokes' law (equation 5) (Metcalf and Eddy, 1991).

$$Vs = \frac{g(\rho_s - \rho)\phi^2}{18\mu} \qquad (5)$$

where:
Vs: settling velocity (m/s)
g: acceleration due to gravity (m/s^2)
ρs: density of particle (kg/m^3)
ρ: density of water (kg/m^3)
ϕ: diameter of particle (m)
μ: dynamic viscosity (Pa.s)

The pollutants that must be removed from wastewater are complex mixtures of particulate and soluble constituents (Levine *et al.*, 1985; Lawler, 1997). The particles in raw domestic sewage range in size from less than 0.001 to well over 100 μm and the size in settled sewage is usually less than 50 μm (Levine *et al.*, 1985). The specific size distribution of particulate organic matter in raw or settled municipal wastewater depends on several factors such as the nature of the community, climate, the length of the sewers and flow regime in there, and whether influent pumping is used (Levine *et al*., 1985). Ødegaard (1999) surveyed the contaminant distribution in Scandinavian countries (Sweden, Finland and Norway) and reveal that 70% of the organic matter of domestic sewage in these countries is suspended, 10-15 % is colloidal and 15-20 % is soluble.

Particle removal in filter media involves two distinct steps: transport and attachment (Fig. 2). The particle is firstly transported to the filter media by mechanisms such as diffusion, interception and sedimentation, before attachment takes place (Prasanthi, 1996). The transport mechanism of a particle is directly dependent on its size (Jackson, 1980; Levine *et al*., 1985; Kaminski *et al.*, 1997). The removal efficiency of particles smaller than ~1µm increases with decreasing size and is accomplished by diffusion (Jackson, 1980). While, the removal efficiency of particles >~1 µm, increases rapidly with particle size due to increase of gravitational force, in addition to interception and straining (Kaminski *et al.*, 1997). However, once captured, they will be subjected to greater shearing forces with increasing particle size. The previous discussion explains why particles in the vicinity of 1 µm are very difficult to remove in filters (Boller and Kavanaugh, 1995). Wang (1994) found that a high loaded UASB system, called the HUSB reactor, could remove the majority of the solids larger than 4.4 µm (expressed as the suspended COD), and only part of the smaller colloidal particles (expressed as COD colloidal).

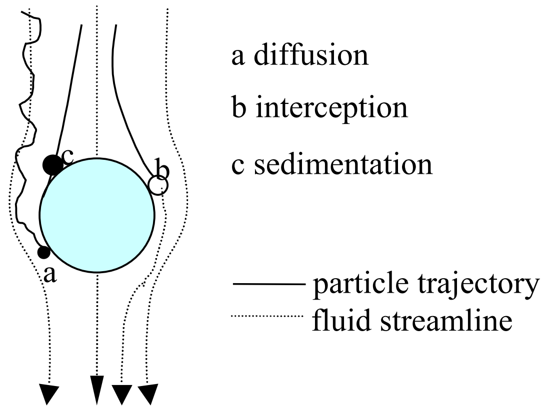

Fig. 2. Transport mechanisms (adapted from O'Melia and Tiller (1993); Stumm and Morgan, 1996)

2.3 INFLUENT PARTICLE CHARGE

Domestic sewage contains hydrophilic and hydrophobic particles. Roughly speaking, these particles consist of hydrophilic organic and hydrophobic inorganic colloids. The reason for the hydrophilic property of the organic colloids is that on their surface there are water absorbing or binding groups such as amino groups ($-NH_2$), hydroxyl groups (-OH) and carboxyl groups (-COOH) (Fig. 3). The charge of the hydrophilic particles is usually caused by dissociation of the ionisable end groups, while the charge of the hydrophobic particles is due to adsorption of anions from the water phase (Henze *et al*., 1995). Elmitwalli *et al*. (2001b) showed that particles in domestic sewage have a negative charge, which only slightly increases (less negative) as a result of digestion. The sludge solids are also negatively charged, which might partly justify the limitations of colloidal particles removal in a UASB system.

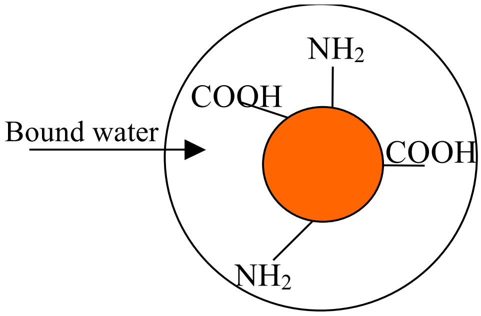

Fig. 3. A colloidal hydrophilic protein particle surrounded by water (Henze *et al.*, 1995)

3. SLUDGE BED CHARACTERISTICS

The physical-chemical characteristics of the sludge bed play a central role in its capacity to remove solids. The interaction between digestion conditions, the sludge physico-chemical characteristics and solids removal is hypothesized in Fig. 4. These relations are discussed in the following subsections.

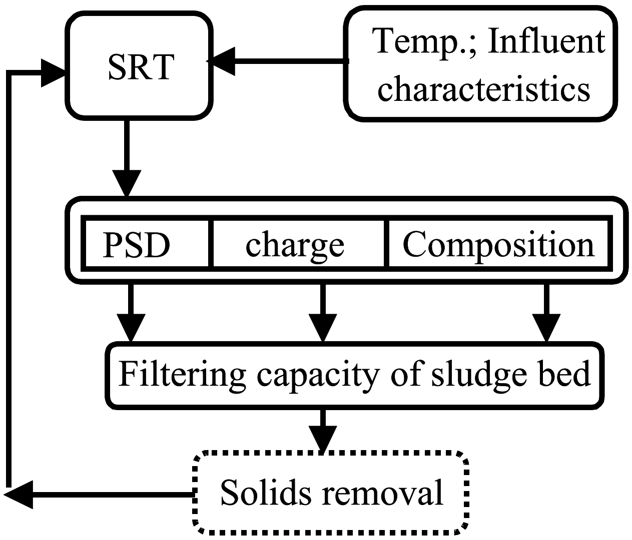

Fig. 4. Scheme of interaction hypothesis for physical removal of solids in a UASB reactor

3.1 PARTICLE SIZE DISTRIBUTION

The effluent quality from classical filters is highly related to the specific size of the filtering media (Landa *et al.*, 1997). Most studies indicate that smaller media size give more efficient removal. Meanwhile, this also could lead to more rapid head-loss development. Decreasing the media size increases the surface area, while decreasing the average pore diameter (Jackson, 1980). All these factors will tend to increase the removal efficiency, but this increase is counterbalanced by an increase in the hydraulic shear (Landa *et al.*, 1997).

The effect of particle size distribution on the performance of the upflow anaerobic sludge bed reactors is not yet clear.

3.2 EXTRACELLULAR POLYMERIC SUBSTANCES (EPS)

Extracellular polymeric substances (EPS) are high molecular weight compounds produced by microorganisms under certain conditions. Such biopolymers are believed to originate from different sources: (1) biological synthesis and excretion and (2) lysis of bacterial cells (Morgan *et al*., 1990; FrØlund, 1996), and also from the adsorption of organic matter from the incoming wastewater, e.g. cellulose and humic acids (Urbain *et al*., 1993), on floc surfaces (Morgan *et al*., 1990). The chemical composition of the EPS matrix is reported to be very heterogeneous with carbohydrate and protein as the major compounds (Morgan *et al*., 1990; FrØlund *et al*., 1996). The components of EPS extracted from activated sludge were found to be different for plants with different process design (Eriksson and Alm, 1991; Urbain *et al*., 1993; FrØlund *et al*., 1994). Chemical composition of the EPS extracted from anaerobic sludge differs from activated sludge, with protein being the most dominant fraction in anaerobic samples compared with carbohydrate in the latter (Morgan *et al*., 1990).

Morgan *et al*. (1990) investigated the differences between biopolymers extracted from activated sludge, sludge from fluidised bed and anaerobic filter and UASB digested flocculent sludge and granules. The yield of extracted polymeric material was found to differ significantly depending on the nature of the sludge sample. All the anaerobic samples, in particular the digested and the granular, yielded significantly less EPS than the activated sludge. The samples from the fluidised bed, the anaerobic filter and the UASB reactors yield intermediary amounts of EPS compared with the activated sludge and granular sludge. Activated sludge samples produced 70 - 90 mg EPS/g SS compared with 10-20 mg EPS/g SS for granular sludge which is a remarkable difference.

Jia *et al*. (1996a) examined the EPS yields in four anaerobic sludge, using acetate, propionate, butyrate and glucose, respectively, as the sole enrichment substrate. Four series of culture enrichment experiments were conducted in 135 ml glass vials, which were operated at 21 days and kept at a temperature of 35 ºC. Under steady- state conditions, the sludge content of EPS's protein (EPSp) and carbohydrate (EPSc) contents were measured. The results showed that acidogenesis of glucose produced more EPSp and EPSc than acetogenesis and methanogenesis. Harada *et al*. (1988) found that carbohydrate degrading UASB granules were larger and having higher mechanical strength than UASB granules degrading short chain fatty acids.

Harada *et al*. (1988) concluded from observation with electron microscopy that EPS excreted by acidogenic bacteria assist with cell-to-cell attachment and the enhancement of mechanical strength and structural stability. Elmitwalli *et al*. (2000) reported that the attachment of biomass and/or entrapment of solids to a reticulated polyurethane foam media, which had been used as a packing medium in an anaerobic filter, increased the colloidal particles removal efficiency. Sprouse and Rittmann (1991) showed that the growth of an anaerobic biofilm on granular activated carbon in a fluidised bed reactor enhanced solids removal. The excretion of EPS could be the main factor that promotes the solids removal in presence of a biofilm, since the EPS are believed to enhance the bio sorption of particles (Dugan, 1987; Elmitwalli, 2000). Therefore, the EPS could increase the captured solids resistance for the shearing forces.

EPS are reported to affect several physical and chemical characteristics of activated sludge, like: dewaterability (Kang *et al*., 1990), floc charge (Horan and Eccles, 1986), floc structure (Eriksson and Härdin, 1984), settleability (Forster, 1985; Goodwin and Forster, 1985; Urbain *et al*., 1993) and flocculation (Ryssov-Nielsen, 1975; Brown and Lester, 1980; Rudd *et al*., 1983; Barber and Veenstra, 1986; Eriksson and Alm, 1991; Jia *et al*, 1996a;). Also the granulation of anaerobic sludge (Jia *et al*., 1996) and the anaerobic sludge charge (Morgan *et al*., 1990) are reported to be effected by EPS. The precise function of biopolymers in relation to bioflocculation and their effect on sludge physico–chemical characteristics are not fully understood (Morgan *et al*., 1990) and sometimes the reported research results are contradictory (Urbain *et al*., 1993).

EPS are thought to influence the dewatering characteristics of sludge by forming a charged surface layer on sludge particles (Poxon and Darby, 1997). The interactions of these polymers between cells allow adjacent bacteria to aggregate by bridging cell surfaces electrostatically and physically and therefore, initiate floc formation which allows the sludge settlement (Tenny and Stumm, 1965; Morgan *et al*., 1990; Cloete and Steyn, 1988; Eriksson and Alm, 1991). Morgan *et al*. (1990) proposed that the chemical nature of the sludge surface will influence the measurable floc charge which itself affects the settling properties of the sludge.

3.3 CHARGES

The sludge surface charge is most likely a result of the EPS ionisable groups, such as amino groups (– NH_2), hydroxyl groups (– OH), carboxyl groups (– COOH) and/or through the adsorption of ions from the water phase (Sutherland, 1984; Henze *et al*., 1995; Jia *et al*., 1996b; Stumm and Morgan, 1996). The charge of these groups depends on the nature of the groups and the pH (Marshall, 1967; Jia *et al*., 1996b; Stumm and Morgan, 1996). At neutral pH, functional groups such as carboxylic groups have a negative charge, while amino groups and a like have a positive charge (Fig. 5). Elmitwalli *et al*. (2001a) showed that particles in anaerobic sludge have a negative charge.

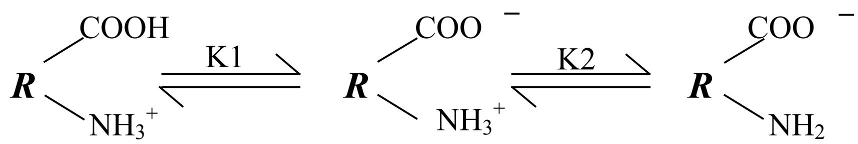

Fig. 5. Protolysis of functional amino and carboxyl groups (adapted from Stumm and Morgan, 1996)

The sludge surface charge had been reported to influence many physical-chemical characteristics of sludge like: cation exchange potential (Flemming, 1995), sludge settleability (Forster and Dallas- Newton, 1980; Eriksson and Axberg, 1981; Steiner *et al*., 1976; Magara *et al*., 1976), dewaterability (Poxon and Darby, 1997) and viscosity (Forster, 1981).

The sludge surface charge most likely depends on sludge digestion conditions, since it is directly related to the quantity and composition of the EPS content. Magara *et al.* (1976) reported that the activated sludge surface charge is strongly dependent on the EPS chemical composition and concentration. Jia *et al*. (1996b) found in anaerobic batch reactors enriched solely by propionate, butyrate and glucose that the EPS and the surface negative charge of all enriched sludge were dependent on the microorganisms growth phase. Both increase when the microorganisms are in the prolific – growth phase, having high substrate concentrations and food to microorganisms (F/M) ratio and they both decrease when the microorganisms are in the declined – growth phase. The negative surface charge increased linearly with the total EPS content, in accordance with previous findings (Morgan *et al.*, 1990). The increase of the EPS when the substrate is abundantly available had been widely reported for activated sludge as a result of increased anabolic activity (Magara *et al.*, 1976; Gulas *et al.*, 1979; Kurane *et al.*, 1986 a, b; Characklis and Marshal, 1990). Meanwhile, when the substrates are utilized or the (F/M) ratio is low the bacteria metabolise the EPS for energy and/or carbon (Jia *et al.*, 1996b). The EPS degradation under anaerobic conditions forming CO_2 and CH_4 was also reported (Ryssov-Nielsen, 1975). Using a colloid titration technique, activated sludges were found to be more negatively charged than granular sludges (Morgan *et al.*, 1990).

Forster (1981) found by the means of electrophoretic mobility measurements that activated sludge particles have a higher mobility than anaerobic digested sludge. Consequently, the authors concluded that activated sludge will probably form an expanded matrix structure, while, anaerobic sludge will be more packed with more particles per unit volume. The more highly charged particles are likely to form gel structures with poly-valent metal ions. Such a structure would have a high resistance to shear. On the contrary, Forster and Dallas-Newton (1980) found that if the negative charge of the floc surface was sufficiently large, repulsion might occur that would cause the sludge settling properties to deteriorate.

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Chapter 3

SEWAGE CHARACTERISTICS IN PALESTINE

A TOOL FOR SEWAGE MANAGEMENT AND SELECTION OF TREATMENT TECHNOLOGY

SEWAGE CHARACTERISTICS IN PALESTINE

A TOOL FOR SEWAGE MANAGEMENT AND SELECTION OF TREATMENT TECHNOLOGY

ABSTRACT

Wastewater management in Palestine has been addressed as a high priority from perspectives of both environmental protection and resource conservation. The official strategic attitude stimulates the adoption of anaerobic technologies, which are low cost and energy self-sufficient. Unfortunately, the available knowledge on sewage characteristics is limited for proper selection and sizing of treatment technologies. This work aims at increasing the knowledge on sewage characterisation as a tool for conceptual assessment of the applicability of a one stage Upflow Anaerobic Sludge Blanket reactor (UASB) as a core technology for sustainable sewage management in Palestine. Time-interval composite samples were collected from three locations in Ramallah/Al-Bireh district and analysed for several chemical and physical parameters. The available data from previous studies about sewage characteristics in Palestine were compiled. The obtained results were used as input data for model calculations to elucidate the applicability of the one stage UASB reactor in Palestine. The results revealed that the sewage of the three locations is of high strength due to high concentration of pollutants like COD, Nkj, phosphorous, sulphate, ammonia and solids. The high sewage strength is attributed to low water consumption, industrial discharges and people habits resulting in the high specific COD production (gCOD/c.d). The model calculations reveal that the process conditions in a one stage UASB reactor should be modified to overcome the sewage high solids content and low temperature during wintertime. A novel technology consisting of a UASB reactor integrated with a sludge digester is recommended. Based on the economic situation and the geographic and demographic distribution, it is recommended to consider decentralised anaerobic treatment to be the core of the Palestinian wastewater management strategy. Several treatment concepts are proposed according to sustainability criteria.

Key words—anaerobic; Palestine; sewage; technology selection; wastewater characteristics; wastewater management

INTRODUCTION

(Waste)water management in Palestine had been neglected for decades (Daibes, 2000). Domestic and industrial wastewater used to be collected mainly in cesspits or, to a much lesser extent, in sewerage networks. In many of the Palestinian villages and refugee camps, black wastewater is collected in cesspits, while grey wastewater is discharged via open channels. The majority of the collected wastewater from the sewered localities is discharged into nearby wadis without being subjected to any kind of treatment. It is estimated that about 30 % of the West Bank population is served with sewerage networks, but less than 6% is connected to treatment plants (PWA, 1997-personal communication).

As the Palestinian society is facing heavy economical burdens, the application of conventional aerobic wastewater treatment technologies is too expensive and not providing a sustainable solution for environmental protection and resource conservation. Anaerobic digestion has been widely recognized as the core of sustainable waste management (Hammes *et al.*, 2000; Zeeman and Lettinga, 1999), which has also been recognised by the Palestinian officials (PWA, 1998). The feasibility of the upflow anaerobic sludge blanket (UASB) reactor for sewage treatment has been successfully demonstrated in many tropical countries. Experience with the application of the UASB in the Middle East countries however is still limited (Zeeman and Lettinga, 1999). The main factors dictating the applicability of anaerobic technologies for domestic wastewater treatment are the sewage temperature and the characteristics and concentration of the pollutants in this sewage (Lettinga *et al.*, 1993). Also, knowledge about the wastewater characteristics is necessary for the design and operation of treatment facilities (Metcalf and Eddy, 1991) and to determine the sequence of treatment systems (Levine *et al*., 1991). The treatability of wastewater depends strongly on the size distribution of the pollutants, since most treatment processes -physical, chemical or biological- for treatment of wastewater contaminants are depend on particle size distribution (Levine *et al.,* 1985; Ødegaard, 1999). Meanwhile, the characteristics of wastewater in the West Bank have not been subjected to good analysis (Applied Research Institute – Jerusalem (ARIJ), 1997).

This research aims at increasing the knowledge on sewage characteristics and the conceptual assessment of the technical applicability of the UASB reactor as a core technology for sustainable sewage treatment in Palestine.

MATERIALS AND METHODS

Study area

Three locations were chosen for this study, all situated in the "Ramallah/Al-Bireh" district area, which is located in the central part of the West Bank and considered as one of the most important administrative centres in Palestine. Ramallah and Al-Bireh are the main urban centres for commerce and services with small and medium scale industries. According to the last census carried out by the Palestinian Central Bureau of Statistics (PCBS, 1997), the population of Ramallah and Al-Bireh are 18,017 and 27,972 inhabitants, respectively. The third location that was selected is Al-Jalazoon refugee camp, which lies about six kilometres north from the centre of Ramallah and Al-Bireh, and has a population of 6,144 inhabitants (PCBS, 1997).

According to the records of JWU (the water supplying company), the average billed water consumption per capita is 103 l/d for the total area. The consumption for the two cities Ramallah and Al-Bireh is higher amounting to 137 l/c.d, while for Al-Jalazoon, this is only 51 l/c.d.

Sewage from Ramallah and Al-Bireh is collected in sewer systems, serving about 75% of the population. The remaining is collected in septic tanks or cesspits. The treatment plant in Ramallah, which consists of aerated lagoons followed by stabilisation ponds, is over loaded and poorly maintained. The recently built treatment plant of Al-Bireh is an oxidation ditch. In Al-Jalazoon, sewage is mainly collected in open channels running along the sides of the camp's streets and discharged to down stream wades without any kind of treatment.

Sampling

Samples were collected from the sewerage outfall of the three study locations: Al- Bireh City, Ramallah City and Al- Jalazoon refugee camp. Time-interval composite samples were collected every two hours from 10 a.m. to 4 p.m. on Saturday, Tuesday and Thursday for three weeks in September 2000. Samples of four litres were collected from the mentioned locations; one-litre grab sample was gathered from each location for each cycle and combined in four-litre plastic containers. All the samples were preserved during the sampling time by storing them in special insulated boxes at 4 ^{o}C.

Analysis

TSS, VSS, NH_4^+, Nkj, total PO_4, dissolved PO_4^{3-} and SO_4^{2-} were all determined according to standard methods (APHA, 1995). Raw samples were used for measuring total COD (CODt), 4.4 μm folded paper-filtered (Schleicher and Schuell 5951/2, Germany) samples for particulate COD (CODp) and 0.45 μm membrane - filtered (Schleicher and Schuell ME 25, Germany) samples for dissolved COD (CODdis). The suspended COD (CODss) and colloidal COD (CODcol) were calculated as the difference between CODt and CODp and the difference between CODp and CODdis, respectively. Carbohydrates in raw, paper-filtered and membrane-filtered samples were determined according to Bardley *et al.* (1971). VFA were determined in membrane-filtered samples by gas chromatography. The chromatograph (Hewlett Packard 5890A, Palo Alto, USA) was equipped with a 2 m x 2 mm (inner diameter) glass column, packed with Supelco port (100-120 mesh) coated with 10% Fluorad FC 431. Operating conditions were: column, 130 ºC; injection port, 200 ºC; flame ionisation detector, 280 ºC. N_2 saturated with formic acid at 20 ºC was used as a carrier gas (30ml/min). Settleable solids were measured after settling for 30 minutes in an

Imhoff cone (Metcalf and Eddy, 1991). Biodegradability of raw samples was measured in 500 ml working volume batch reactors incubated at 30 ^{o}C for a period of 60 days. Each batch was filled with 450-ml wastewater, macro and mico nutrients and a buffer as described by Elmitwalli (2000). COD total was measured at the beginning and the end of the batch period. All measurements were determined in duplicate.

Sewage temperature was measured in situ for each grab sample by a alcohol thermometer. The ambient temperature was measured with a alcohol thermometer placed at the Al-Bireh treatment plant. Sewage temperatures during the other sampling period have been measured four times during January, February and March – winter 2000. Colour was determined by visual appearance.

Calculations

- 1 g protein, assumed as $(C_4H_{6.1}O_{1.2}N)x$ is equivalent to 0.16 g Nkj-N, 0.16g NH_4-N and 1.5 g COD (Sanders, 2001)
- 1 g carbohydrates (assumed as $C_6H_{12}O_6$) is equivalent to 1.07 g COD (Sanders, 2001)
- Biodegradability (BD) = $100*(COD_{t,\ t=0\ day} - COD_{t,\ t=60\ day})/\ COD_{t,\ t=0\ day}$

RESULTS AND DISCUSSION

Sewage characteristics

General observations. The results presented in Table 1 elucidate two main observations on sewage characteristics of the study locations. Firstly, the sewage of the three locations are classified of high strength, since, the mean values of COD, Nkj, phosphorus, sulphate, ammonia, TSS and VSS are very high according to the sewage strength classification proposed by Metcalf and Eddy (1991) and Henze (1997) and as compared with sewage characteristics in several countries in Europe, Asia and Latin America (Table 2). Secondly, the sewage characteristics of Al-Bireh City and Al-Jalazoon refugee camp are very similar and of domestic type, while, they differ from those of Ramallah City which has much higher pollutants concentrations. The Ramallah sewage characteristics are highly influenced by industrial discharges which can be clearly observed in the measured parameters (COD, carbohydrate, TSS, VSS, protein, sulphate concentrations and high wastewater temperature), and also from visual observation of the colour variation of each collected grab sample during the same sampling day due to discharges of dying factories, food and beverage industries.

Anaerobic biodegradability. The anaerobic biodegradability of the sewage of the three locations measured after 60 days are presented in Table 1. Unfortunately, the test period was far too short for achieving a stabilisation of organic matter. Elmitwalli *et al*. (2001) found that the sewage biodegradability of Bennekom village in The Netherlands was 21 % after 43 days and increased to 74% after 135 days at 30 ºC. Likewise, Kerstens (2001) found high biodegradability of 76% (after 130 days incubation at 33 ºC) of the sewage of Amman City, Jordan which is, from a socio economic point of view, very close to the Palestinian cities. The previous discussion indicates that, the biodegradability of Ramallah, Al-Bireh and Al-Jalazoon wastewater is high which is also indicated from the high VSS/TSS ratio presented in Table 3.

Size fractionation of COD. The results of COD fractions of the three locations are presented in Table 1. The results reveal that the main fraction of COD is particulate, as the suspended, colloidal and dissolved fractions of COD of all studied areas are in the ranges 49-58%, 15-22% and 25-35% of the total COD, respectively. In view of that, the percentage particulate COD (colloidal and suspended) of total COD for Ramallah, Al-Bireh and Al-Jalazoon is 65%, 75% and 71%, respectively. Wang (1994) found that the COD fraction of the particles exceeding 0.45 µm in domestic sewage represent about 70% of the total COD. Levine *et al*. (1985) found also that the COD fraction of the particles larger than 0.1 µm is in the range of 30-85% of the total amount of organic materials present in the pre-settled sewage.

Size fractionation of carbohydrate. The results of carbohydrate fractions for the three locations are presented in Table 1. The suspended, colloidal and dissolved carbohydrate fractions are in the range of 36-74%, 9-12% and 17-52% of the total carbohydrates, respectively. The suspended carbohydrate forms the major fraction for Al-Bireh and Al-Jalazoon, while the dissolved is the major fraction in Ramallah sewage. This can be explained by the existence of food industry in Ramallah like Coca-Cola and chocolates, which discharge their wastewater in the Municipal sewer system. Likewise, Elmitwalli (2000) showed that carbohydrate is mainly present in the suspended form in the domestic sewage of the village of Bennekom - The Netherlands.

Volatile fatty acids. The results of VFA and COD ratios presented in Table 3 show that around 35% of the dissolved COD are in the VFA form for Al-Bireh and Al-Jalazoon, and 27% for Ramallah. The results also show that around 25% of the total COD in Al-Bireh and Al-Jalazoon are present in hydrolysed form and a higher value of around 40% for Ramallah. The acidified fraction is around 10% for all three locations.

Nkj-N, NH_4^+-N and proteins. The Nkj-N values for Ramallah and Al-Bireh are nearly the same, but it is lower for Al-Jalazoon (Table 1). Comparing the values of Nkj-N and NH_4^+-N showed that the NH_4^+-N is the major fraction of Nkj-N. In addition, the results presented in Table 4 reveal that the highest protein content exists in Ramallah wastewater followed by

Al Bireh and the lowest concentration is found in Al-Jalazoon wastewater. This might be justified by the presence of the main district (chicken) slaughterhouses in Ramallah City.

Total phosphorus and ortho-phosphate. The measured values of both total phosphorus and ortho-phosphate are presented in Table 1. The results show that the values of total phosphorus are approximately equal to the values of ortho-phosphate for the three locations, which is in conformity with Butler *et al.* (1995).

Sulphate. The sulphate concentration in the sewage of the three studied locations (Table 1) is very high according to the classification proposed by Metcalf and Eddy (1991). This especially applies for Ramallah where the sulphate concentration range is 474-2060 mg/l. The reason for that is the presence of several industries consuming large quantities of sulphur and sulphuric acid, especially a number of dying factories. The water consumption records of the JWU reveal that the water consumption of the dying factories in Ramallah City during the year 2000 was 108,000 m^3, which represented around 10% of the total water consumption of the city. The major problem associated with the anaerobic treatment of sulphate-rich wastewater is the production of sulphide. Since sulphide can lead to several problems such as toxicity, bad smell, corrosion, deteriorated quantity and quality of the biogas and reduction of the COD removal efficiency. In practice, anaerobic treatment always proceeds successfully for wastewater's with chemical oxygen demand (COD) to sulphate ratios exceeding 10. At COD/sulphate ratios lower than 10, process failures of anaerobic reactors have been reported (Hulshoff Pol, 1998). Accordingly, the high sulphate content of Ramallah sewage might suppress its anaerobic treatment as the COD/sulphate ratio can be as low as 0.74. Nevertheless, calculation of the maximum possible H_2S production reveals that the H_2S will always be below the IC_{50} concentration (50% inhibition concentration) of 250 mg H_2S-S, which indicates the occurrence of only partial inhibition during anaerobic sewage treatment. Moreover, the high sulphate content is harmful to the sewerage system. As H_2S gas may be oxidised to H_2SO_4 that causes damages to cement containing constructions such as manholes and pipes (Metcalf and Eddy, 1991).

Table 1. Wastewater Characteristics of Ramallah City, Al - Bireh City and Al-Jalazoon refugee camp

Parameters		# Samples	Ramallah			Al-Bireh			Al-Jalazoon		
			Range	AVR	STD	Range	AVR	STD	Range	AVR	STD
COD	Total	8	1518-3812	2180	663	1411-1844	1586	125	1092-1773	1489	251
	Suspended	8	545 – 1925	1096	456	720-1209	919	157	518-990	725	153
	Colloidal	8	107- 525	323	101	171-362	274	52.4	213-440	327	71.3
	Dissolved	8	468-1482	761	297	280-464	393	62.3	258-613	438	113
Carboh.	Total	5	100-231	178.4	52.6	99-166	131	22.8	59-155	93.5	32.7
	Suspended	5	45-90	64.1	20.2	67-132	97.4	25.3	28-88	44.8	21.3
	Colloidal	5	10-35	21.6	10	9-14	11.7	1.8	4-14	9.9	3.7
	Dissolved	5	37-157	92.6	55.4	17-37	22.2	7.4	15-60	38.8	17.3
VFA as COD		2	175-199	187	12	155-162	160	3.1	100-145	123	25.4
Nkj as N		6	54-119	99.4	23.2	85-122	104	14.7	53-83	71	10
NH_4^+ as N		8	47-72	58	8.5	72-89	80.1	5	40-77	56.2	9.5
$^+$Proteins				388			224			139	
Total PO_4 as P		3	10-15	12.8	2.2	11-14	13	1.5	11-18	15	2.4
PO_4^{3-} as P		5	6-17	12.4	3.8	8-15	12.9	2.6	8-14	11.9	2.4
SO_4^{2-} as SO_4^{2-}		4	474-2060	975	742	129-151	138	9.9	143-277	213	57
TSS		5	510-1096	729	197	610-824	736	67	408-1048	630	234
VSS		5	255-892	584	209	492-676	617	66.1	364-733	480	148
Settleable solids		7	4-105	43.5	41.1	8.5-13.5	10.9	2.2	1.2-8	2.9	2.4
pH		4	7.18-8.02	7.45	0.39	7.16-7.44	7.26	0.13	7.11-7.58	7.31	0.2
T_{ww} Summer		8	26-40	30.9	3.19	24-27	25.8	0.67	20-25	23.4	1.52
Winter		4				12-13	13.13	0.63			
T_{amb}. Summer		8				21-33	27.1	3.17			
Winter		4				11-17	13.8	2.75			
Biodegradability		2	46-49	47	1.4	34-40	36	3.2	32-35	33	1.5
Colour		8	Reddish to black			Medium brown			Light brown		

All parameters have been measured in duplicate and their units are in mg/l except settleable solid in ml/l; wastewater temperatures (T_{ww}) and ambient temperature (T_{amb}) (^oC); pH no unit; Biodegradability (%); Proteins mg COD/l

$^+$Calculated

TSS and VSS. The TSS and VSS values of the three locations are high (Table 1). The results presented in Table 3 reveal high values of VSS/TSS for the three locations. Apparently, the data presented in Table 2 indicates that VSS/TSS ratio of sewage in Palestine, Jordan and Egypt are high in comparison with Turkey, Brazil and Columbia. This might be due to difference in people habits. The high CODss/VSS of Ramallah wastewater indicates high lipids content, possibly due to discharge of slaughterhouses wastewaters.

COD mass balance. The results presented in Table 4 indicate that a high fraction of biodegradable COD should be lipids (25-50 % of total COD is estimated to be lipids) as proteins, carbohydrates and lipids are the main biopolymers in sewage (Levine *et al*., 1991). This is obvious from the low values of COD subtotal and the large difference when we compare the subtotal COD in the studied areas with the sewage of Bennekom- The Netherlands. However, the sulphide COD was not accounted for.

Table 2. Sewage characteristics of different cities in different countries and continents

Parameter	[1]Palestine Al Bireh	[2]Jordan Amman	[3]Egypt Rural areas	[4]Turkey Istanbul	[5]The Netherlands Bennekom	[6] Brazil, Campina Grande Pedregal	[7]Columbia Cali
CODt	1586	1183	824.9	410	528	727	267
CODss	919	608	-	-	225	-	-
CODcol	274	174	-	-	156	-	-
CODdis	393	401	270.2	140	147	-	112
VFA-COD	160	104-177	-	-	55	-	-
NH_4^+-N	80	80	26	30	48	34	17
Nkj-N	104	109	33.8	43	70	44	24
Total P	13	-	8.9	7.2	18	11	1.3
PO_4^{3-}-P	12.9	-	3.87	4.5	14	8	-
Lipids-COD	-	443	302	-	-	-	-
Protein-COD	224	272	-	-	-	-	-
TSS	736	420	310	210	-	492	215
VSS	617	330	277	145	-	252	108
VSS/TSS	84	79	89	70	-	51	50
Temperature		16-24	-	-	8-20	24-26	24.4-25

1, this study; 2, Kerstens (2001); 3, Orhon *et al.*, 1997; 4, Tawfik (1988); 5, Elmitwalli, 2000; 6&7, Haandel and Lettinga (1994). All parameters are in mg/l except: temperatures (^oC); VSS/TSS ratio

Table 3. Percentages of hydrolysis and acidification of total COD and acidification of dissolved COD and VSS/TSS and CODss/VSS ratios for the sewage of Ramallah and Al-Bireh cities and AL-Jalazoon refugee camp-Palestine

	Parameter	Ramallah	Al Bireh	AL-Jalazoon
Acidified fraction	VFA/CODt	10	10	9
Acidified of dissolved	VFA/CODdis	27	36	35
Hydrolysed fraction	CODdis/CODt	39	28	25
	VSS/TSS	80	84	76
	CODss/VSS	1.88	1.49	1.51

Table 4. Percentages of carbohydrates, proteins and VFA out of total COD for domestic/municipal wastewater of Ramallah, Al-Bireh cities and AL-Jalazoon refugee camp-Palestine and Bennekom village-The Netherlands

Parameter	Ramallah	Al-Bireh	AL-Jalazoon	Bennekom-The Netherlands[+]
Carb-COD/CODt	8.8	8.9	6.7	12
Protein-COD/CODt	18	14	9	44
VFA/CODt	10	10	9	9
Sub total COD	36.8	32.9	24.7	65

[+] Elmitwalli (2000)

Compiled data of sewage characteristics in Palestine. Different studies have been conducted to characterise the sewage of the main cities in the West Bank as shown in Table 5. The presented data reveal that the sewage of all main cities in the West Bank is of high strength. In some cities, like Hebron and Bethlehem, the wastewater is very strong due to the effect of tannery industry discharges. There are variation of pollutant concentrations for the same location among these studies, due to the type and time of sampling, and due to different water consumption during the period of study. Our present study is the first that uses composite samples for analysis, while the previous studies were based on grab samples.

Grey wastewater. For Al-Jalazoon refugee camp, it was believed that only grey wastewater is disposed in the open channels (Amarneh, 2001). This study shows that Al-Jalazoon wastewater is of high strength and is similar to Al-Bireh wastewater COD. The characteristics of grey wastewater collected from several houses in Bilien village-Ramallah district (Mustafa, 1997) are presented with those of Al-Jalazoon refugee camp in Table 6. The substantial difference between the characteristics of both wastewaters indicates the discharge of black wastewater with the grey water in Al-Jalazoon. This is obvious from the high content of NH_4^+-N in AL-Jalazoon wastewater and confirmed by interviewed residents. The sulphate is originating from the drinking water (~ 55 mg SO_4^{2-}/l), proteins and the detergents, which contains sulphate rich compounds like sodium lauryl sulphate, ammonium sulphate, etc.

Specific waste production

The calculated specific waste production for Ramallah and Al-Bireh cities in terms of COD, N and P are presented in Table 7. In order to obtain a reference waste, the household waste production, source and composition for some countries (Germany, Denmark and Sweden) compiled from several literature sources by Henze (1997) are presented in Table 8, in addition to other data for the Dutch society (Kujawa *et al.*, 2000). The data in both Tables show that the COD specific waste production in the waterborne waste of Ramallah and Al-Bireh is substantially higher than the reference wastes (Table 8). These values are as high as the total (solid and waterborne) specific waste of the reference countries and in Ramallah it can even be two times higher. This suggests that the high COD content of sewage in Palestine and other countries in the Middle East, like Jordan (Table 2), is not only due to low water consumption, but also due to industrial discharges (Ramallah) and/or people's habits. Discarding the remaining food and used cocking oil in kitchen sinks is believed to play a central role in increasing sewage strength in Palestine. Henze (1997) shows that the application of 'clean tech cooking' can reduce the COD load of grey water from 55 gCOD/c.d to 32 gCOD/c.d.

The data in Table 8 show that kitchen waste contributes around 35% to the COD content of the household traditional waterborne wastes, and can even be much higher in case more solid waste is directed to the sewer system. The COD content of wastewater can be significantly reduced by separating part of the kitchen wastes (clean tech cocking described by Henze (1997)) and toilet wastes (Zeeman and Lettinga, 2001; Henze and Ledin, 2001; Henze, 1997). Likewise, separation of the toilet wastes (physiological wastes) from the waterborne route will lead to a significant reduction in nitrogen and phosphorous. The N specific waste production of Ramallah and Al-Bireh is similar to the reference wastes. The P specific waste production is similar to Holland, but different from those reported for Denmark, Sweden and Germany. Moreover, the presented data reveal that (77-85%) and (47-60%) of the waste waterborne N and P content originates from urine.

Table 5. Compiled sewage characteristics of the main cities in the West Bank-Palestine

City	Study	BOD	COD	Nkj	NH_4^+	Total P	PO_4^{3-}	SO_4^{2-}	TSS	pH	*Source*
Ramallah	This study, 2001		2180	99	58	12.8	12.4	975	729	7.45	
	ARIJ, 1996	525	1390	79	51		13.1	132	1290		Nashashibi, 1995
	ECPD, 1993	820	1650							6.92	Municipality
Al-Bireh	This study, 2001		1586	104	80	13	12.9	138	736	7.26	
	Nashashibi, 1995	750	1230	37	27		4.3	61			Nashashibi, 1995
	Dr. Jourdan, 1993	600	1200								Municipality
	Dr. DaHLem, 1996	575	1150			49					Municipality
Jenin	Nashashibi, 1995	1100	1440						1088		PECDAR, 1994
	BZU, 1999	1182	2020	309					1290	6.95	Municipality
Nablus	ARIJ, 1996. West	600	954			62	20.6		488	6	PECDAR, 1994
	ARIJ, 1996. East	560	1338			47	15.6		840	6.5	PECDAR, 1994
	Nashashibi, 1995	1185	2115	120	104		7.5	137	1188		Nashashibi, 1995
Tulkarm	ARIJ, 1996	250	540			17.9	6		398	6.5	PECDAR, 1994
Bethlehem	ARIJ, 1996	660	2724			141.4	45.6		688	6.5	PECDAR, 1994
	Nashashibi, 1995		2720						1080		Nashashibi, 1995
Hebron	ARIJ, 1996	520	2736			413.8	133.5		1794	6	ARIJ, 1996
	Nashashibi, 1995		3670	200	123		18.4	150			Nashashibi, 1995
	Tahboub, 2000	1008	2886	278	113	19.3	19.8	267	16346		Tahboub, 2000

Note: all units are mg/l except pH. Nkj and NH_4^+ measured as N

Table 6. Characteristics of grey wastewater of Bilien village (Mustafa, 1997) and Al-Jalazoon wastewater (this study)

Parameter	Unit	Al-Jalazoon		Bilien	
		Range	Average	range	average
COD	mg/l	1092-1773	1489	600-850	630
Nkj-N	mg/l	53-83	71	16-17	16.7
NH_4^+-N	mg/l	40-77	56.2	7-12	10
Total P	mg/l	11-18	15	15-17	16
SO_4^{2-}	mg/l	143-277	213	52-54	53
TSS	mg/l	408-1048	630	94-181	125
Settl. Solid	ml/l	1.2-8	2.9	0.3-4.5	1.7
pH		7.11-7.58	7.31	6.6-7.4	7
T_{ww}	^oC	20-25	23.4	18.5-25.4	22

Table 7. Specific production (g/c.d) and composition of waterborne household wastes in Ramallah and Al-Bireh cities, Palestine

	Ramallah	Al-Bireh
COD	166-418	155-202
N	5.9-13.0	9.3-13.4
P	0.7-1.9	0.9-1.6

Notes: calculated based on billed water consumption of 137 l/c.d of which 80% ends up in the sewer system (Nashashibi and van Duijl, 1995) and the measured parameters shown in Table 1.
Pollutants due to industrial discharges are neglected

Table 8. Production and composition of total (solid and waterborne) and traditional total waterborne household wastes, g/c.d for some countries in West Europe)

	+Denmark, Sweden and Germany						++Holland
Param.	Total		Source of waste				Total
	Solids & water-borne	Water - borne	Physiological Total (urine)	Kitchen, liquid	Wash & bath	Kitchen solids	Water- borne Total (physiological <urine>)
COD	220	130	75	45	10	90	102 (56)
N	13.3	13	11 (10)	1	1	0.3	12.9 (12.5<11>)
P	*4.5	*2.5	2 (1.5)	0.2	0.3	2	1.7 (1.4<0.8>)

+ Henze (1997)
++ Kujawa *et al.* (2000)
*The phosphorous load from the washing of clothes will be increased by some extra 1-1.5 g P/c.d, if phosphate rich detergents are used (Henze, 1997)

Treatment concepts

Generally, the geographic and demographic distribution of the Palestinian cities, villages and refugee camps were not based on proper engineering town and regional plans. The latter can be observed in all Palestinian districts, where often more than 40 villages and camps are distributed around the main cities with population ranging from less than one hundred to almost ten thousands. Unfortunately, the existence of residential neighbourhoods and industrial enterprises on fertile agricultural land is not an uncommon phenomenon in the West Bank. This miss-planned feature of the West bank makes the application of decentralised low cost anaerobic based treatment systems attractive. Because the decentralised systems enable the onsite reuse of treated effluent, which also solves the problem of effluent discharge in addition to, cost reduction of extended sewers construction and can promote urban agriculture.

Palestine, a country of arid or semi arid climate, suffers from water scarcity, which can be a bottleneck in future development of the country. Needless to say, speaking about water shortage is meaningless as long as the available high quality water resources are polluted. On one hand, the disposal of untreated wastewaters is a major threat of ground water pollution, the main source of potable water in Palestine. Nitrate concentrations exceeding 100 mg NO_3-N/l has been recorded for several ground water wells in The Gaza Strip (Nashashibi and van Duijl, 1995). On the other hand, concentrated human wastes (1.5 litre of faeces and urine per person per day) are diluted with large amounts of drinking water (*ca.* 100 l) in order to transport the wastes from the site of production to the site of treatment/disposal, which is a worldwide common practice in the sanitation field. Obviously, sightless adoption of classical theories of sanitation which are based on centralised, aerobic concepts will not sustain the environment in Palestine due to the following reasons:

- Huge increase in the capital and operational costs of the sewer network, pumping stations and treatment installations due to dilution.
- Pollution of other household waste streams (grey wastewater) and potable water used for flushing toilets with organics, nutrients and other salts and pathogens as the main pollutants exists in black wastewater (Zeeman and Lettinga, 2001)
- Problem of final effluent disposal, this problem manifests, due to the limitation and scatter of the agricultural land and the political sensitivity of the area. Moreover, this limits the reuse possibilities of treated wastewater and nutrients
- Limit and reduce the potential of energy recovery from the waste streams
- Production of large quantities of sludge

In many of the Palestinian villages, camps and urban areas, hardly any sanitation infrastructures have been implemented yet. Sensibly, for those places, the separation of black and grey wastewater at the household and the onsite treatment of those waste streams is a rational option which meets the sustainability concepts defined by Lettinga *et al.* (2001). Accordingly, different concepts based on decentralized collection and treatment of domestic wastewater with anaerobic as the core technology can be used in Palestine as follow (Zeeman and Lettinga, 2001):

- Collection, transport and community–on-site treatment of total sewage (grey water+ black) in a UASB or UASB-septic tank reactor: recommended for areas served with sewers network
- House–on-site treatment of total sewage or black water in a (UASB) –septic tank system, combined with transport of the effluent to a community on-site post-treatment: recommended for areas served with conventional septic tanks and demand water for irrigation outside the household
- House- on-site collection and treatment of 'night soil' and grey water: recommended for areas where little water is required for flushing toilets i.e. manually flushed toilets, which require about 2 litres of water are popular in Palestine

Application of the UASB reactor in Palestine

The final removal efficiency and conversion of organic compounds to methane gas in UASB systems depend on both physical and biological processes. For sewage, removal of suspended solids occurs by physical processes like settling, adsorption and entrapment. The subsequent hydrolysis and methanogensesis of the removed solids depends on the process temperature and the prevailing SRT. Zeeman and Lettinga (1999) developed a model for the calculation of the HRT when a certain SRT is a pre-requisite. The SRT is determined by the amount of sludge that can be retained in the reactor and the daily excess sludge production. The daily excess sludge production is determined by the biomass yield and the removal and conversion of suspended solids. At a certain temperature the SRT will determine whether methanogenesis will occur or not. So when the required SRT is known, the corresponding HRT can be calculated provided that the sludge concentration in the reactor (X), the fraction of the influent SS that is removed (R) and the fraction of the removed SS that is hydrolysed (H) are known. The HRT of a UASB reactor can be calculated with the following formulas:

$$SRT = X/Xp \tag{1}$$

X: sludge concentration in the reactor (g COD/l); 1 g VSS = 1.4 g COD
Xp: sludge production (g COD/L.d)

$$Xp = O*SS*R*(1-H) \tag{2}$$

O: organic loading rate (kg COD/m^3.d); SS = CODss / CODinf;
R: fraction of CODss removed

HRT = C/O (days) (3)
C: COD concentration in the influent (g COD/l)

HRT = (C *SS/X)*R*(1-H)*SRT (4)
SRT: sludge retention time (days)
H: fraction of removed solids that are hydrolysed

The previous model was used for the calculation of the required HRT for the application of an one stage UASB reactor for sewage treatment in Al-Bireh City. The following input data were taken into consideration:

R = 0.8, Miron (1997) found around 85% TSS removal efficiency at V_{up} = 0.56 m/hr
H = 0.15; 15% of the TS are hydrolysed at process conditions of 15 ºC and 75 days SRT [winter conditions] (Elzen and Koppes, 2000)
X = 15 g VSS/l = 21 g COD/l
SS = 0.58, (this study)
C = 1.586 g COD/l (this study)
SRT = 30 days, expected minimum SRT to achieve methanogenic conditions during winter time (Elzen and Koppes, 2000)

Accordingly, the model calculation revealed that a minimum HRT of 22 hour is required for the application of the one stage UASB reactor in Palestine to overcome the wintertime. The calculated HRT is long in comparison with normal HRT applied in the tropical countries (6-12 hours). Worth mentioning, that we have recently researched (Mahmoud *et al*., 2002) a novel technology consisting of a high loaded UASB integrated with a digester with sludge re-circulation to solve the technical applicability of the UASB system in the Middle East during the short period of winter. Promising results have been obtained which showed that the UASB-digester system could be successfully operated at a much lower overall equivalent HRT.

CONCLUSIONS AND RECOMMENDATIONS

- The sewage in Palestine is characterised of high strength and solids content.
- The application of the one stage UASB reactor in Palestine is only possible if designed at a prolonged HRT due to low solids hydrolysis during wintertime. Alternatively, the UASB-Digester system is recommended.
- The application of decentralised "one house or factory onsite and/or community onsite" is recommended for the following reasons:
  - Enabling the urban agricultural reuse of treated effluent as the majority of the agricultural land in Palestine is scattered as small agricultural lots,

- Solving the problem of effluent discharge,
- Reducing the pressure on the conventional water resources,
- Protection of downstream sewerage works,
- Reducing the sewerage work cost and consequently proper environmental protection.

- Reform in the sanitation paradigms towards the recognition of water saving appliances and separation of household waste streams is recommended.
- The factories should apply pre-treatment step before discharging their wastewater's in the municipal sewerage system. This highlights the urgent need for issuing environmental act and enforcing the implementation of environmental regulations. On the other hand, those industries that use high amounts of sulphate can recover the sulphur element, which will reduce the treatment cost" cleaner production"

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Chapter 4

ANAEROBIC STABILISATION AND CONVERSION OF BIOPOLYMERS IN PRIMARY SLUDGE

EFFECT OF TEMPERATURE AND SLUDGE RETENTION TIME

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ANAEROBIC STABILISATION AND CONVERSION OF BIOPOLYMERS IN PRIMARY SLUDGE

EFFECT OF TEMPERATURE AND SLUDGE RETENTION TIME

ABSTRACT

The effect of sludge retention time (SRT) and process temperature on the hydrolysis, acidification and methanogenesis of primary sludge was investigated in completely stirred tank reactors (CSTRs). The CSTRs were operated to maintain SRTs of 10, 15, 20 and 30 days at process temperatures of 25 and 35 ºC. Another CSTR was operated at 75 days SRT and 15 ºC, the sewage temperature in Palestine, the Middle East, during wintertime. In addition, the rates of hydrolysis and the biodegradability of primary sludge were assessed in batch reactors incubated at 15, 25 and 35 ºC. The results revealed that the major achievement of sludge stabilisation occurred between 0 and 10 days at 35 ºC and 10 and 15 days at 25 ºC. Hydrolysis was found to be the rate limiting-step of the overall digestion process, for the reactors operated at 35 and 25 ºC, except for the reactor operated at 10 days and 25 ºC. At the latter conditions, acidification of LCFA was rate limiting in comparison to hydrolysis; hydrolysis of protein and particulate carbohydrates were rate limiting of the acidification step, while methanogenesis is the rate-limiting step of the overall process. Protein hydrolysis was limited to a maximum value of 39% at 30 days and 35 ºC due to protein availability in the form of biomass. Methanogenesis occurred at 75 days and 15 ºC. The biodegradability of primary sludge was around 60%, and showed no temperature dependence. The hydrolysis of the main biopolymers and overall particulate COD of the primary sludge digested in CSTRs were well described by first order kinetics, in case hydrolysis was the rate-limiting step. Similarly, the hydrolysis of the overall particulate COD of the primary sludge digested in batch reactors were described by first order kinetics and revealed strong temperature dependence, which follows Arrhenius equation.

Keywords—acidification, anaerobic, biodegradability, carbohydrates, CSTR, hydrolysis, lipids, methanogenesis, primary sludge, protein, stabilisation

NOMENCLATURE

Water Research, 1987

C_{carbh} total carbohydrates (particulate plus dissolved carbohydrate) (g COD/l)

C_{lipid} total lipids (neutral lipids plus LCFA) (g COD/l)

$C_{protein}$ total proteins (particulate plus dissolved proteins) or (Nkj- NH_4^+-N)x1.5/0.16)g COD/l

S_{carbh} dissolved carbohydrates (g COD/l)

S_{LCFA} LCFA (long chain fatty acids) (g COD/l)
S_{COD} dissolved COD (g COD/l)
S_{VFA} volatile fatty acids (g COD/l)
X_{carbh} particulate carbohydrates (g COD/l)
X_{lipid} neutral lipids (g COD/l)
SRT sludge retention time
$CH_4(t)$ cumulative methane production at time *t* (in STP ml., i.e. ml at standard temperature of 0 °C and standard pressure of 1 atm)
CH_{4max} maximum methane yield of the primary sludge in the biodegradability batches (in STP ml.)
k_h first-order hydrolysis rate constant (d^{-1})
t time (d)
VS volatile solids (g/l)
TS total solids (g/l)
CSTR Completely Stirred Tank Reactor

INTRODUCTION

High rate anaerobic treatment has gained popularity for domestic sewage treatment, especially for developing countries, because of low costs of construction, operation and maintenance, small land requirements and low excess sludge production. Nowadays, hundreds of full-scale upflow anaerobic sludge bed (UASB) reactors are in operation for sewage treatment in several tropical countries where the temperature ranges between 20 and 35 ºC (Haandel and Lettinga, 1994; Elmitwalli, 2000; Elmitwalli *et al.*, 2002). Meanwhile, the application of the one stage UASB for treating high strength sewage at low temperature, as is the case in the Middle East countries during wintertime, is only possible when designed at a prolonged HRT (Zeeman and Lettinga, 1999). Man (1990) found that the performance of one-stage UASB systems at low temperature climates (5-20 °C) is highly limited by the hydrolysis of the entrapped solids. O'Rourke (1968) showed that no significant methanogenesis was provided at the digestion of sewage sludge in a CSTR at an SRT of 60 days at 15 °C. Several researchers suggested the application of two stage systems consisting of a high loaded first stage followed by a methanogenic stage, like the two-stage UASB system (Wang, 1994) or the two-stage anaerobic filter-anaerobic hybrid system (Elmitwalli, 2000). However, when applying high loaded reactors, the sludge produced is by definition not sufficiently stabilised and needs further treatment in a separate digester. The SRT and process temperature are the key parameters, which control the stabilisation of the organic polymers in primary sludge and consequently the sizing of the anaerobic digesters. Hydrolysis of particulate organic matter, the first step in anaerobic degradation, is usually rate-limiting in the overall degradation sequence (Eastman and Ferguson, 1981; Pavlostathis and Giraldo-Gomez, 1991). Therefore, sizing of anaerobic reactors should be based on that step. Mostly, hydrolysis is described by first order kinetics based on biodegradable organic matter and constant temperature and pH (Sanders, 2001). However, the available literature data on the effect of temperature on the hydrolysis rates of organic wastes is still scarce (Veeken and Hamelers, 1999).

In terms of chemical composition, three groups of organic polymers are the major constituents of primary sludge: carbohydrates, proteins and lipids. The particulate carbohydrates are hydrolysed to simple sugars like cellobiose and glucose (Colberg, 1988), which are further degraded to VFA. Lipids included in sewage sludge are composed mainly of neutral fats and LCFA. Neutral fats are hydrolysed to LCFA and glycerol. LCFAs are degraded via β-oxidation by H_2-producing acetogenic bacteria (Pavlostathis and Giraldo-Gomez, 1991). The occurrence of methanogenic conditions enhances lipids hydrolysis and acidification of LCFA (Palenzuela-Rollon, 1999). Proteins are hydrolysed to amino acids which are further fermented to ammonium, VFAs , carbon dioxide , hydrogen gas and reduced sulphur (Pavlostathis and Giraldo-Gomez, 1991). The acidification rate of the amino acids is faster than the rate of protein hydrolysis, the reason why amino acids occur in very low concentrations (Pavlostathis and Giraldo-Gomez, 1991; Miron *et al.*, 2000).

The aim of this research is to optimise the design of a digester complementary to a high loaded UASB treating domestic sewage in terms of sludge stabilisation. Accordingly, the effects of SRT and process temperature on the hydrolysis, acidification and methanogenesis of primary sludge were studied. The occurrence of methanogenesis at 15 ºC and SRT of 75 days was examined. Temperatures of 15 and 25 °C are the average minimum and maximum of wastewater in Palestine - the Middle East. In addition, this research aims at increasing the knowledge on the anaerobic biodegradability and hydrolysis rates of sewage particulate matter at the envisaged temperatures for operating anaerobic reactors in the Middle East.

MATERIALS AND METHODS

Experimental set-up

Continuous experiments. The envisaged system of a digester supplementing a high loaded sewage treating UASB was simplified by eliminating the UASB and alternatively primary sludge was used for feeding CSTR digesters. Two sets of eight CSTRs, with a 15-liter working volume, were operated inorder to maintain SRTs of 10, 15, 20 and 30 days at process temperatures of 25 and 35 ± 1 ºC. A third set of one, 10-liter working volume, CSTR was operated to maintain an SRT of 75 days at 15 ± 1 ºC. The temperatures were controlled by re-circulating water of pre-controlled temperature through the reactors double walls. The reactors were inoculated with primary sludge digested at 34 ºC and 20 days from the wastewater treatment plant (WWTP) of Ede, The Netherlands. The feeding stock primary sludge was collected once from the aforementioned WWTP and stored at 4 ºC. Around 20 litres of the stock was diluted weekly to 20 g TS/l and stored at 4 ºC from which the reactors were fed once daily. The reactors were intermittently mixed at 30 rpm for 10 sec/ min. Biogas was continuously collected in 10 or 5 litres gas bags and was daily measured. The pH was measured daily and the VFAs were measured at least once every two weeks and digesters temperatures were checked from time to time. After a period of at least 3SRTs of continuous operation, the monitoring parameters were stable and the reactors were considered to be at steady state conditions. Afterwards, within three weeks, four samples in duplicate were analysed for total and dissolved COD, total solids (TS) and volatile solids (VS), VFA, neutral lipids, LCFA, total and dissolved proteins, total ammonium (NH_4^+ -N), kjeldahl-nitrogen (Nkj), carbohydrates (total and dissolved). The quantity of methane produced was measured daily.

Batch experiments. The experimental set-up and procedure for determination of anaerobic biodegradability of raw primary sludge and stability of digested sludge are the same. The tests are carried out in batch reactors, serum bottles, of 500 ml with a headspace volume of 70 ml. Each bottle was filled with about 5 g COD-sludge/l, tap water and a mineral solution of macro nutrients, trace elements, and bicarbonate buffer. The composition and concentrations of the mineral solution and the experimental procedure are as described by (Elmitwalli, 2000). The collected methane gas in the headspace was regularly measured using a Mariotte displacement set-up filled with a 5% NaOH solution. The stability batches were incubated at 30 ºC, while the biodegradability was assessed at three temperatures of 15, 25 and 35 ºC. The biodegradability was measured once in triplicates, while sludge stability was measured two times in duplicate during steady state conditions of the CSTRs. Stability is defined as the maximum percentage of COD converted to CH_4 of the digested sludge.

Analytical methods

TS, VS and Nkj were determined according to standard methods (APHA, 1992). Total COD was determined according to NEN 6633 (APHA, 1992), modified for high COD concentrations as described by Miron *et al.* (2000). The soluble fractions of COD, carbohydrates and proteins were determined as described by Elmitwalli (2000). Volatile fatty acids (VFA) were measured from 0.45 µm membrane filtered samples (Schleicher and Schuell ME 25, Germany) with a gas chromatograph, described by Elmitwalli (2000). Total proteins, total lipids (neutral lipids plus LCFA), NH_4^+- N and total carbohydrates were determined according to the methods described by Elmitwalli (2000). The LCFA were eluted from the total lipids according to the method described by Sanders (2001). The COD of the LCFA was determined according to the modified COD test NEN 6633 (APHA, 1992). Prior to COD digestion, the COD tubes including the eluted samples were incubated for two hours at 105 ºC for evaporating the acetic acid. The analysed samples for total COD, total carbohydrates and total proteins were previously homogenised by ultra – Turrax

(Kika-Werk, Jank and Kunkel). Methane in the biogas was determined by gas displacement, using a 16% NaOH solution.

Calculations

Conversion factors

1 g carbohydrates (assumed as $C_6H_{12}O_6$) is equivalent to 1.07 g COD

1 g lipid is equivalent to 2.91 g COD (Sayed, 1987)

1 g protein (assumed as $(C_4H_{6.1}O_{1.2}N)x$) is equivalent to 0.16 g Nkj, 0.16 g NH_4^+-N and 1.5 g COD (O'Rourke, 1968; Sanders, 2001)

*Formulas**

$X_{\text{lipid}} = C_{\text{lipid}} - S_{\text{LCFA}}$ (1)

$X_{\text{carbh}} = C_{\text{carbh}} - S_{\text{carbh}}$ (2)

$H_{\text{protein}} = NH_4^+\text{-N x } 1.5/0.16$ (3)

$H_{\text{lipids}} = C_{0,\text{lipid}} - X_{\text{lipid}}$ (4)

$H\text{carb} = C_{0,\text{carbh}} + S_{0,\text{VFA}} - NH_4^+\text{-N}_0 \text{ x } 1.5/0.16 - X_{\text{carbh}}$ (5)

$H_{\text{total}} = S_{\text{COD}} + CH_4\text{-COD} + S_{\text{LCFA}}$ (6)

$A_{\text{lipid}} = C_{0,\text{lipid}} - C_{\text{lipid}}$ (7)

$A_{\text{carbh}} = C_{0,\text{carbh}} - C_{\text{carbh}} + S_{0,\text{VFA}} - NH_4^+\text{-N x}1.5/0.16$ (8)

$A_{\text{protein}} = NH_4^+\text{-N x } 1.5/0.16$ (9)

$A_{\text{tot}} = S_{\text{VFA}} + CH_4\text{-COD}$ (10)

$M = CH_4\text{-COD}$ (11)

Ammonification (%) = $(NH_4^+\text{-N/Nkj-N})*100$ (12)

* Calculated hydrolysis (H, g COD/l) and acidification (A, g COD/l) include hydrolysis and acidification products already present in the influent. The influent VFA is assumed to be generated from carbohydrates and proteins, but not lipids (Miron *et al.*, 2000). The index "0" refers to the influent, otherwise the effluent is meant.

Hydrolysis rate

CSTRs. The hydrolysis first order constants (k_h's) for the CSTRs are calculated based on the linearised equation (Eastman and Ferguson, 1981):

$$SRT = F_0 \text{x} \frac{SRT}{\Delta F} - \frac{1}{K_h} \quad (13)$$

Where: F_0 the biodegradable substrate, is the slope of the fitted line and ΔF is the hydrolysed substrate (H, g COD/l) that took place in the reactors as described in the following equations:

$H_{\text{protein}} = C_{0,\text{protein}} - C_{\text{protein}}$ (14)

$H_{\text{lipids}} = X_{0,\text{lipid}} - X_{\text{lipid}}$ (15)

$H_{\text{carbh}} = X_{0,\text{carbh}} - X_{\text{carbh}}$ (16)

$$H_{total} = \text{dissolved-COD } (S_{COD}) + CH_4\text{-COD} + S_{LCFA} - S_{0,COD} - S_{0,LCFA} \quad (17)$$

Batch. In case hydrolysis is rate limiting in the overall digestion process and assuming first order hydrolysis kinetics (Eastman and Ferguson, 1981), the cumulative methane production from the biodegradability batches can be described by the following equation (Veeken and Hamelers, 1999):

$$CH_4(t)=CH_{4,max}[1\text{-}exp(\text{-}k_h t)] \quad (18)$$

The $CH_{4,max}$ and k_h were estimated using non-linear least squares curve fitting of the cumulative CH_4 produced. The anaerobic biodegradability was calculated as a ratio of COD-$CH_{4,max}$ out of COD influent (added to the bottle). The lag period in gas production was excluded from the fitted data range.

Temperature. The temperature effect on the rates of biological processes can be described by the Arrhenius equation (Veeken and Hamelers, 1999):

$$k = Ae^{-E/RT} \quad (19)$$

Where: k: the reaction rate constant (d^{-1}); T: the absolute temperature in degrees Kelvin (ºK); R: the ideal gas constant (J.mole $^{-1}$.ºK^{-1}); A: the Arrhenius constant (d^{-1}); E: activation energy (kJ.mole^{-1})

RESULTS AND DISCUSSION

Continuous experiments

Total COD hydrolysis, acidification and methanogenesis

The composition and concentration of the primary sludge used in this research is presented in Table 1. The percentage particulate COD hydrolysis, acidification and methanogenes are presented in Tables 2 and 3. The results reveal that hydrolysis improved at increasing SRT ($p<0.1$) with the most increase between 0 and 10 days at 35 ºC and 10 and 15 days at 25 ºC, respectively which are in conformity with O'Rourke (1968). The results reveal that hydrolysis in the reactors operated at 35 ºC were significantly higher ($P<0.05$) than those operated at 25 ºC at similar SRT. For both sets, except that at 10 days and 25 ºC, hydrolysis was the digestion rate-limiting step as soluble COD was very low (around100 –150 mg/l) with a negligible VFA concentration. For the reactor of 10 days and 25 ºC, a high VFA concentration of around 11% of the influent COD was maintained, almost amounting to the soluble COD concentration. Therefore, methanogenesis was the rate-limiting step of the overall digestion process. The reactor of 75 days and 15 ºC could maintain methanogens after 3 SRTs.

Table 1. Composition of the influent primary sludge, Ede-The Netherlands

	prot tot g/l	Prot sol g/l	Nkj-N mg/l	NH_4-N mg/l	Carb tot g/l	Carb sol g/l	Lipids g/l	LCFA mg COD/l	COD tot mg COD/l	COD sol mg COD/l	VFA mg COD/l	TS g/l	VS g/l
Avg.	6.22	0.07	801	99	6.5	0.06	1.33	1112	29997	1923	1806.79	20.35	14.72
STD	0.51	0.02	68	3.96	0.3	0.01	0.07	155	1734	124	108.07	0.31	0.17

Main bioploymers hydrolysis and acidification

Carbohydrates. A substantial portion of carbohydrate hydrolysis occurred within the first 15 and 10 days of digestion, at process temperatures of 25 and 35 ºC, respectively (Tables 2 and 3). The results reveal that the carbohydrates in the investigated primary sludge were around 70% anaerobically biodegraded. Differently, Miron *et al.* (2000) reported for primary sludge obtained from the same source, a higher percentage of carbohydrates hydrolysis (*ca*. 90%) at 15 days and 25 ºC. This difference might be attributed to collection of the feed stock primary sludge during wintertime in the previous research and summer time in our research, when the primary sludge might be partly degraded in the sewerage system. The effluent soluble carbohydrates at all investigated conditions were constant at around 20 mg/l, which is less than 1 % of the influent carbohydrates. Therefore, hydrolysis, not acidification, was rate limiting in the carbohydrate digestion.

Proteins. The hydrolysis of proteins was followed based on ammonification. The results presented in Tables 2 and 3 reveal that the biodegradability of proteins in the studied primary sludge was low as the achieved hydrolysis at 30 days and 35 ºC was 39 %. Similarly, Miron *et al.* (2000) reported 33% hydrolysis of proteins at 15 days SRT and 25 ºC. The limitation in proteins hydrolysis in primary sludge is expected to be due to the availability of proteins in the form of biomass. This is explained by the fact that bacteria constitute around 20 - 50% of the faeces dry weight (Williams *et al.*, 1999). Considering 12 -16% of bacterial cells to consist of nitrogen (Metcalf and Eddy, 1991), 20% of the dry solids to consist of bacteria and a theoretical value of 0.16 g nitrogen per gram proteins, it can be calculated that *ca*. 60% of the proteins in the investigated sludge was present in the form of biomass.

The protein hydrolysis based on direct protein measurement (Lowry, 1951) revealed higher values than those calculated based on ammonification (Fig.1) ($p<0.1$). The calculated NH_4^+-N concentration based on the protein measurement via the Lowry method revealed that the Lowry method probably overestimated the proteins content in the fresh primary sludge and digested sludge at 10 days SRT. This is postulated to the presence of an interfering substance(s), which is degraded at longer SRT. The Lowry method is reported to be susceptible to many interfering substances like amino acids, detergents and sugars (Brown *et al.*, 1989).

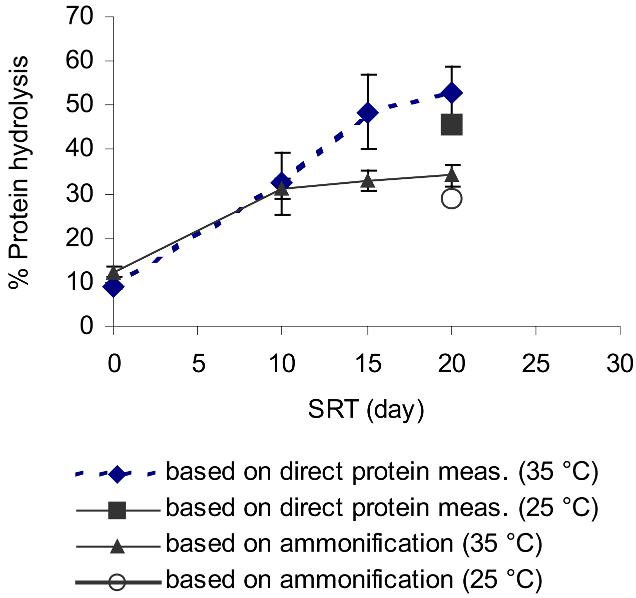

Fig. 1. Percentage of proteins hydrolysis as a function of SRT for the primary sludge digested at 25 and 35 ºC based on ammonification and direct proteins measurement using Lowry method (1951)

*Lipids***.** The results presented in Tables 2 and 3 reveal that lipid hydrolysis improved substantially with increasing SRT. The most substantial portion of hydrolysis occurred within the first 15 and 10 days of digestion at 25 and 35 ºC, respectively. The results show that around 30 % of the lipids in municipal sewage are already hydrolysed or were discharged as LCFA in the sewer system. At all investigated conditions, except 10 days at 25 ºC, LCFA were completely acidified which suggests that hydrolysis was the rate-limiting step. At 10 days and 25 ºC, acidification of LCFA was rate limiting of the acidification step. The conversion of lipids is highly affected by the occurrence of methanogenesis (Palenzuela-Rollon, 1999).

Batch experiments

Sludge biodegradability

The assessed degradation curves in the biodegradability assay results show two distinguished phases (the curves are not shown). The first is a lag phase in the methanogenic activity, followed by a first order biodegradation type of curve. The lag phases lasted for around 14, 30 and 50 days for the batches incubated at 35, 25 and 15 ºC, respectively. The lag phases are presumed as a consequence of higher hydrolysis rate than methanogenic rate at the beginning of the experiment (Veeken and Hamelers, 1999). The effective gas production lasted till the 60th and 75th day for the batches incubated at 35 and 25 ºC, respectively. For the batches incubated at 15 ºC, even after 135 days a complete degradation was not reached. The extent of primary sludge biodegradability did not show temperature dependence, but the rate significantly increased with temperature (Table 5).

The biodegradability curves excluding the lag phase could be described by first order kinetics from which the CH_{4max} and k_h could be found. These k_h values followed an Arrhenius type of behaviour (R^2 0.99) (Fig. 2) with an activation energy of 50 $kJmol^{-1}$. Veeken and Hamelers (1999) reported a similar mathematical behaviour for 6 types of solid bio-waste.

Table 2. Percentage hydrolysis (H), acidification (A) and methanogenesis (M) of the influent COD, lipids, proteins and carbohydrates and pH for primary sludge digested at 25 °C. SRT=0 stands for the influent. Standard deviations are presented between brackets

SRT (day)	0	10	15	20	30
H COD	10.00 (0.32)	23.85 (1.31)	40.70 (0.54)	41.40 (0.37)	42.10 (0.40)
A COD	6.02 (0.36)	22.42 (1.20)	39.03 (0.51)	40.97 (0.37)	41.62 (0.40)
M COD	0.00 (0.00)	10.91 (1.14)	38.97 (0.52)	40.97 (0.37)	41.62 (0.40)
H lipids	28.7 (5.30)	46.00 (7.22)	74.80 (1.60)	76.60 (0.41)	83.02 (0.36)
A lipids	0.00 (0.00)	45.23 (4.75)	74.78 (1.59)	76.57 (0.41)	83.02 (0.36)
H proteins	12.36 (0.49)	25.90 (0.29)	26.71 (0.77)	29.10 (0.54)	33.90 (4.27)
A proteins	12.36 (0.49)	25.90 (0.29)	26.71 (0.77)	29.10 (0.54)	33.90 (4.27)
H carbohydrates	12.00 (1.78)	43.40 (2.39)	57.60 (3.88)	62.40 (3.75)	64.80 (1.80)
A carbohydrates	11.28 (1.68)	42.93 (2.38)	57.36 (3.88)	62.15 (3.76)	64.01 (1.41)
pH	7.80	5.79(0.14)	6.75(0.07)	6.76(0.06)	6.76(0.06)

Table 3. Percentage hydrolysis (H), acidification (A) and methanogenesis (M) of the influent COD, lipids, proteins and carbohydrates and pH for primary sludge digested at 35 °C. SRT=0 stands for the influent. Standard deviations are presented between brackets

SRT (day)	0	10	15	20	30
H COD	10.00 (0.32)	41.10(1.42)	45.4(1.20)	47.20(1.02)	49.2(0.29)
A COD	6.02 (0.36)	40.54(1.42)	45.08(1.20)	46.84(1.01)	48.77(0.30)
M COD	0.00 (0.00)	40.48(1.42)	45.07(1.20)	46.84(1.01)	48.77(0.30)
H lipids	28.7 (5.30)	75.30(3.61)	81.20(2.97)	86.60(1.33)	87.8(1.41)
A lipids	0.00 (0.00)	73.38(3.61)	81.22(2.97)	86.58(1.33)	87.80(1.41)
H proteins	12.36 (0.49)	31.03(1.64)	33.00(1.30)	34.10(1.54)	38.70(2.60)
A proteins	12.36 (0.49)	31.03(1.64)	33.04(1.30)	34.11(1.54)	38.74(2.60)
H carbohydrates	12.00 (1.78)	56.50(5.11)	61.70(3.63)	64.60(4.25)	71.50(2.31)
A carbohydrates	11.28 (1.68)	56.26(5.12)	61.58(3.62)	64.33(4.24)	71.24(2.31)
pH	7.80	6.83(0.10)	6.92(0.09)	6.90(0.06)	6.84(0.10)

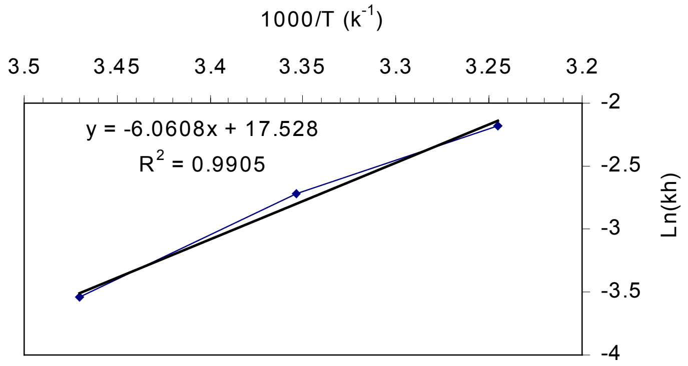

Fig. 2. Arrhenius plots for the hydrolysis rates of primary sludge

Sludge stability

The stability results and COD mass balance are presented in Table 4. The results reveal that complete stabilisation of primary sludge requires very long digestion time. After a long retention time (*ca.* 95 days) for well-digested sludge (20 days at 35 ºC), some methane was still released at an incubation temperature of 30 ºC. This justifies the gap in the COD mass balance as some batches were stopped after rather short incubation periods due to technical problems. The results reveal that the sludges in all reactors operated at 35 ºC were well stabilised as further improvement of stability with increasing SRT from 10 to 30 days was low. The improvement of stability with increasing SRT holds true for those digested at 25 ºC, with substantial improvement, as SRT increases from 10 to 15 days with negligible improvement afterwards. The results reveal that all sludges from the reactors operated at 35 ºC were better stabilised than those at 25 ºC and similar SRT. However, this difference is marginal at SRT ≥ 15 days.

Table 4. VS/TS ratio and COD mass balance calculated based on the biodegradability, stability assays and the released CH_4 from the CSTR reactors

Process temp.			25 ºC				35 ºC			
Parameter	Unit	Raw	10	15	20	30	10	15	20	30
VS/TS	-	0.72	0.69	0.63	0.63	0.62	0.64	0.63	0.62	0.60
CSRT CH_4	g COD/l	0	3.35	11.69	12.21	12.60	12.14	13.52	14.05	14.22
Stab. CH_4	g COD/l	17.10	11.53	3.82	3.70	2.76	3.76	3.09	2.72	1.96
Theor. BD.[+]	g COD/l	17.10	14.88	15.51	15.91	15.36	15.90	16.61	16.77	16.18
Stability test period	day		44	37	87	39	74	66	95	44

[+] Theor. BD = $CODCH_4$ +CODstab.; BD = biodegradability; stab.=stability

Hydrolysis mathematical description

The hydrolysis rate constants (k_h*'s*) of COD, proteins, carbohydrates and lipids based on first order kinetics for CSTRs are depicted in Table 5. On one hand, the results show that the hydrolysis step of separate polymers can be well described by first order kinetics. On the other hand, the hydrolysis of total COD at a process temperature of 25 ºC could not be described by this model, most probably due to poor hydrolysis of lipids at 10 days SRT. The pH of the reactor operated at 10 days and 25 ºC was lower than all other reactors due to the presence of VFA (Table 2). Therefore, the conditions for applying first order kinetics are not achieved. The hydrolysis rate constants derived from batch digestion showed lower values than those of the CSTRs.

Table 5. Hydrolysis rate constants (k_h) and biodegradability (f_h) of primary sludge digested in CSTRs (SRT 10 - 30 days) and batches at different temperatures, derived from the first order kinetic equation. Standard deviations are presented between brackets

	Polymer	15 ºC		25 ºC			35 ºC		
		k_h (d^{-1})	f_h	k_h (d^{-1})	f_h	R^2	k_h (d^{-1})	f_h	R^2
CSTR	COD			-	-	-	0.23	0.50	0.99
	Carbohydrates			0.10	0.50	0.95	0.16	0.51	0.99
	Proteins			0.067	0.25	0.93	0.12	0.26	0.98
	Lipids			0.11	0.89	0.66	0.20	0.98	0.99
Batches	COD	0.0289 (0.0006)	0.59 (0.002)	0.07 (0.002)	0.63 (0.005)		0.113 (0.002)	0.57 (0.006)	

Final discussion

The needed digester, complementary to high loaded systems treating sewage at low temperature, can be satisfactorily designed at SRTs of 15 or 10 days at 25 and 35 ºC, respectively. The results of the reactors operated at 25 ºC can be applied to a one stage UASB operated under tropical countries and the first stage of a two stage UASB system operated in Middle East countries during summer time, simulating the sludge bed by a CSTR (Meer, 1979). The occurrence of methanogenesis at 15 ºC at an SRT in the range of 60 - 75 days (O'Rourke, 1968; this study) reveals that the application of a one stage UASB system for sewage treatment in Middle East countries can only be achieved at a long HRT of *ca.* 20 hours (Zeeman and Lettinga, 1999). Therefore, the application of a two-stage system plus a sludge digester or a high loaded UASB system with incorporation of a sludge digester operated at the previously recommended conditions, might be profitable over the one stage system. The high hydrolysis rate - temperature dependence reveals that for the application of anaerobic treatment in the Middle East, special attention should be given to overcome the winter period (Zeeman and Lettinga, 1999). However, the non-temperature dependence of sewage sludge biodegradability reveals the potential of the application of anaerobic technologies for sewage treatment at tropical as well as subtropical climates like the Middle East region.

The results of this research confirm that sizing of anaerobic reactors for treating complex substrates like sewage should be based on the hydrolysis step, which is limiting the digestion rate (Pavlostathis and Giraldo-Gomez, 1991). The hydrolysis rate constants at full methanogenic conditions can be well described by first order kinetics. However, the hydrolysis rate constants should be measured each time for that specific waste and not adopted from literature data. Several researchers showed that particulates hydrolysis is a surface related phenomenon, which is affected by the specific surface area of the particle (the particle size), the chemical composition of the particles (*ca.* proteins, carbohydrates, and lipids) and the availability of each organic polymer to the hydrolytic enzymes (Pavlostathis and Giraldo-Gomez, 1991; Sanders, 2001). Since, these parameters are altered during digestion, the hydrolysis rate constant is preferred to be determined from SRTs close to the required design SRT. The dependence of the hydrolysis rate constants on the reactor type, viz. continuous vs. batch, reveals the limitation of the first order model, as hydrolysis is influenced by other factors, like PSD, which in turn is influenced by mixing (Sanders, 2001).

CONCLUSIONS

- The SRT and temperature have a significant influence on the hydrolysis of proteins, carbohydrates and lipids:
  - The most substantial portion of the digestion of proteins, carbohydrates and lipids occurs within the first 15 and 10 days at process temperatures of 25 and 35 ºC, respectively.

- Hydrolysis is rate limiting for all particulate substrates in the digestion of primary sludge in CSTRs operated at 25 and 35 ºC and SRTs ≥15 and ≥10 days, respectively.
- During digestion of primary sludge in CSTRs at 25 ºC and SRT = 10 days:
  - Methanogenesis is the rate-limiting step of the overall digestion process,
  - Lipids hydrolysis is limited and LCFA are partly acidified; lipids hydrolysis and acidification are affected by the degree of methanogenesis,
  - Acidification of LCFA is rate limiting of the lipids acidification step,
  - Carbohydrates hydrolysis is rate limiting of the acidification step.

- During digestion of primary sludge at 15 ºC in a CSTR, the methanogenesis could be provided at 75 days SRT.
- The amount of proteins hydrolysis in primary sludge is limited, as proteins are mainly present as part of the biomass.
- The Lowry method might overestimate the proteins content in primary sludge.
- The hydrolysis of the main biopolymers present in primary sludge can be well described by first order kinetics, in case hydrolysis is rate limiting.
- The hydrolysis rate constant of all kinds of solid substrates is significantly affected by temperature. This dependency can be described by the Arrhenius equation.
- The biodegradability of primary sewage sludge shows no temperature dependence in the range of 15 – 35 ºC.

- The hydrolysis rate constants derived from batch reactors are not recommended to be used for sizing continuous reactors.

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Chapter 5

THE INTERACTION BETWEEN DIGESTION CONDITIONS AND SLUDGE PHYSICAL CHARACTERISTICS AND BEHAVIOUR FOR ANAEROBIC DIGESTION OF PRIMARY SLUDGE

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THE INTERACTION BETWEEN DIGESTION CONDITIONS AND SLUDGE PHYSICAL CHARACTERISTICS AND BEHAVIOUR FOR ANAEROBIC DIGESTION OF PRIMARY SLUDGE

ABSTRACT

The interaction between digestion conditions and the sludge physical characteristics and behaviour was investigated for anaerobically digested primary sludge in two sets of CSTR reactors. The CSTRs were operated at sludge retention times of 10, 15, 20 and 30 days and temperatures of 25 and 35 ºC. The change of the particle size as a result of digestion was examined using wet sieve analysis (0.100, 0.125, 0.200, 0.500 and 1.000 mm). The results reveal a substantial reduction in all particle sizes with improving digestion conditions. Digestion leads to the transfer of bigger particles into smaller ones, which has a remarkable effect on the sludge physical behaviour. The majority of the raw and digested particles are smaller than 0.100 mm. The dewatering results showed the existence of an optimal SRT for dewaterability at 20 and 15 days for the reactors operated at 25 and 35 ºC, respectively. The dewaterability of sludge digested at less favourable conditions, viz. 10 days at 25 ºC deteriorates due to increase of small particles generated from destruction of larger particles. The digested sludge settling results showed a slight worsening but insignificant trend with increasing the SRT ($P<0.05$).

Keywords— Anaerobic digestion, Dewaterability, Particle size distribution, Primary sludge, PSD, Settling characteristics, SRT, SVI

INTRODUCTION

The relation between the digestion conditions (temperature and SRT) and sludge characteristics and behaviour has been scarcely reported in literature. The operational conditions are expected to largely influence the physical-chemical characteristics and consequently the physical behaviour of the sludge through influencing the organic matter conversion to methane. Anaerobic digestion changes the sludge particle size distribution (PSD), which is influenced by the operating conditions. PSD is the most important variable that affects the sludge dewaterability, e.g. dewaterability worsens with decreasing particle size (Lawler *et al.*, 1986). The objective of this research is to investigate the interaction between the CSTR digestion conditions, sludge physical characteristics (PSD) and sludge physical behaviour (dewaterability and Settleability).

MATERIALS AND METHODS

The CSTR experimental set up was described in Chapter 4.

Analytical methods

TS and VS of sludge were determined according to standard methods (APHA, 1992). Sludge volume index (SVI) was measured according to the *Dutch Standard Normalized Methods* (1969). Dewaterability of the sludge was detected by the capillary suction time (CST, in seconds) and expressed as filterability constant, which is calculated based on the CST (Vesilind, 1988). The dimensionless constant, ϕ, of the CST apparatus is 0.794.

All measurements have been carried out, four times after steady state was reached (more than 3 SRT).

Determination of PSD by sieving

1. A 250 ml sludge sample was collected and diluted 20 times with centrifuged (15 minutes at 1000 rpm) and filtered (4.4 μm paper filter) effluent of a two-step anaerobic system treating domestic sewage (Elmitwalli, 2000),
2. The diluted slurry was poured through each sieve (1.000, 0.500, 0.200, 0.125, 0.100 mm) starting with the largest size,
3. The sieve was soaked in the filtrate for 40 times, in a systematic downward-upward slow motion,
4. The retained solids were conveyed from the sieve to porcelain crucibles. The conveyance process was carried out using a gentle jet of tap water to collect all scattered particles over the sieve surface to one edge of the sieve. Then a jet of demi-water was used to push the collected particles into the porcelain crucibles with the help of a small spoon,
5. The collected solids were examined for total solids and volatile solids.

Settleability of sludge

The sludge settleability was measured by the SVI and measuring the sludge settling in the CSTR. The latter measurement was carried out according to the following procedure:

1. Mixing inside the CSTR was stopped,
2. The height of the developed 'clear' supernatant was followed with time by measuring the distance between the top of the water phase and the top of the sludge phase.

RESULTS AND DISCUSSION

Particle size distribution (PSD)

The results presented in Table 1 reveal a substantial reduction in all particles (on mass basis) within the set size intervals (0.1 – 1.0 mm) due to digestion. The mass of the particles of the digested sludge was always lower than that of the particles of identical size of the raw sludge. Nevertheless, the results of the PSD of the sludge digested at 10 days and 25 °C, clearly show the transfer of larger particles into smaller particles as is clear from particles accumulation in the size range smaller than 0.10 mm. The conversion of larger into smaller particles is also clear as the mass of the particles in the range 0.10 mm ≤ diam. <0.125 mm, of the sludge digested at sub-optimal conditions (SRT: 10 days, 25 °C) is less than those digested at more optimal temperature conditions (35 °C). Moreover, the results clarify that the main achievement in large particles (diam. ≥ 0.1) reduction could be attained by digestion at even sub optimal conditions of (SRT: 10 days, 25 °C), while the main reduction of small particles (diam.<0.1) could be achieved at improving the digestion conditions to (SRT: 10 days, 35 °C). The particles with a diameter in the range of 0.200 - 0.500 mm were dominant in all examined digested sludge samples. The same trend was shown for the inoculum primary digested sludge brought from the WWTP of Ede -The Netherlands. The VS/TS ratio illustrated in Table 2 indicates an increase of the inorganic content with a decreasing particle size, independent of the type of sludge. Moreover, the table clearly shows that the VS/TS ratio of the larger particles (diam. ≥ 0.2 mm) are either not or just slightly affected negatively by improving sludge digestion conditions. However, the VS/TS ratio of the smaller particles (diam. < 0.2 mm) is always significantly higher than that of the raw sludge. The latter indicates that the small particles from the influent are for a substantial part hydrolysed to dissolved products and replaced by new small particles produced by hydrolysis of the larger particles. This finding demonstrates the shift in PSD of big particles to particles of <0.100 mm where the main biodegradation took place. The results presented in Table 3 show that the majority of the particles in the raw as well as the digested sewage sludge are smaller than 0.100 mm.

Dewaterability

The results presented in Fig. 1 reveal a large influence of the digestion conditions on the sludge dewatering characteristics. The dewaterability results of the digested sludge in both sets of reactors, operated at 25 and 35 ºC, reveal the existence of an optimal SRT for dewaterability. The optima at 25 and 35 ºC are shown at an SRT of 20 and 15 days, respectively ($P<0.05$). The dewaterability of the digested sludge in the digester operated at 10 days SRT and 25 ºC deteriorated in comparison with the raw influent ($P<0.05$), which is in agreement with results presented by Miron *et al.* (2000). This digester was under stress as it is clear from the digestion results (Chapter 4). Whereas, the reactors operated at 15 and 30 days and 25 ºC operational temperature were not different from the raw sludge at the 0.05 level. At an SRT of 30 days at both process temperatures, 25 and 35 ºC, the dewaterability of digested sludge did neither differ from each other nor from the raw sludge at a 0.05 level.

Table1. Particle size distribution (PSD) and percentage particles reduction (R%) of raw and anaerobic digested primary sludge in CSTRs operated at different conditions (SRT, Temp.). The PSD is expressed as sludge mass (VS, mg) accumulated over sieves of different mesh sizes. Standard deviations are presented between brackets

Size (mm)	Sludge type					
	Raw	10,25	10, 35	15, 35	20,35	30,35
1.000≤diam.	426(17.0)	169(14.8)	141(15.3)	144(6.9)	140(23.2)	94(7.1)
0.500≤diam.<1.000	418(17.4)	166(23.5)	137(4.6)	129(5.1)	106(7.8)	93(8.1)
0.200≤diam.<0.500	404(5.2)	204(5.2)	203(5.0)	183 (4.7)	164(3.4)	156(2.5)
0.125≤diam.<0.200	143(8.3)	106(7.2)	104(4.7)	103 (1.6)	89(2.5)	88(1.8)
0.100≤diam.<0.125	98(4.1)	53(4.8)	70(2.7)	64(5.2)	61(3.0)	65(1.4)
Sum diam.≥0.100	1489(35)	698(1.2)	655(7)	623(10)	560(9)	496(0.35)
R% diam.≥0.100	0	53	56	58	62	67
diam.<0.100	2191(36)	2503(0.5)	1737(7.2)	1628(13.43)	1562(23.76)	1451(1.06)
R% diam.<0.100	0	-14	21	26	29	34

For all sludges, sample size of 250 ml was used

Table 2. VS/TS ratio of raw and digested primary sludge accumulated over sieves of different mesh sizes. The sludge was digested in CSTRs operated at different conditions (SRT, Temp.). Standard deviations are presented between brackets

Size (mm)	Sludge type					
	Raw	10,25	10, 35	15, 35	20,35	30,35
1.000≤diam.	0.92(0.003)	0.91(0.007)	0.90(0.037)	0.89(0.009)	0.91(0.025)	0.89(0.012)
0.500≤diam.<1.000	0.92(0.003)	0.89(0.003)	0.87(0.030)	0.86(0.004)	0.86(0.006)	0.85(0.007)
0.200≤diam.<0.500	0.82(0.004)	0.85(0.003)	0.78(0.010)	0.79(0.004)	0.79(0.006)	0.79(0.012)
0.125≤diam.<0.200	0.61(0.009)	0.83(0.006)	0.68(0.022)	0.72(0.007)	0.71(0.017)	0.68(0.028)
0.100≤diam<0.125	0.50(0.009)	0.77(0.015)	0.60(0.013)	0.65(0.013)	0.66(0.009)	0.60(0.024)

For all sludges, sample size of 250 ml was used

Table 3. Percentage pass of particles (measured as VS) through different mesh sizes for raw and digested sludge in CSTRs operated at different digestion conditions (SRT, Temp.). Standard deviations are presented between brackets

Size (mm)	Sludge type					
	Raw	10,25	10, 35	15, 35	20,35	30,35
diam.<1.000	88(0.46)	95(0.46)	94(0.45)	94(0.31)	93(1.10)	95(0.37)
diam.<0.500	77(0.94)	90(0.27)	88(0.30)	88(0.53)	88(0.70)	90(0.05)
diam.<0.200	66(1.08)	83(0.11)	80(0.29)	80(0.42)	81(0.49)	82(0.18)
diam.<0.125	62(0.85)	80(0.10)	76(0.23)	75(0.61)	76(0.36)	78(0.09)
diam.<0.100	60(0.96)	78(0.04)	73(0.29)	72(0.44)	73(0.56)	74(0.02)

For all sludges, sample size of 250 ml was used

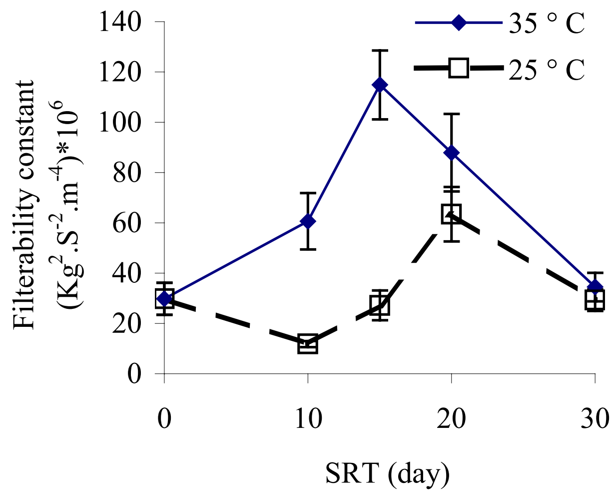

Fig. 1. Dependency of primary sludge dewaterability expressed as filterability constant according to (Visilind, 1988), on anaerobic digestion conditions (SRT: 0, 10, 15, 20 and 30 days; Temp. 25 and 35 °C) in CSTRs. SRT=0 stands for the raw influent

Settling characteristics

The results presented in Fig. 2 show that the SVI was insignificantly affected by the digestion conditions at the 0.05 level. The results however show a slightly worsening trend of sludge settling characteristics with digestion. The results of sludge settling obtained by follow up of sludge settling inside the CSTR (Fig. 3) confirm that digestion does not affect the sludge settling characteristics.

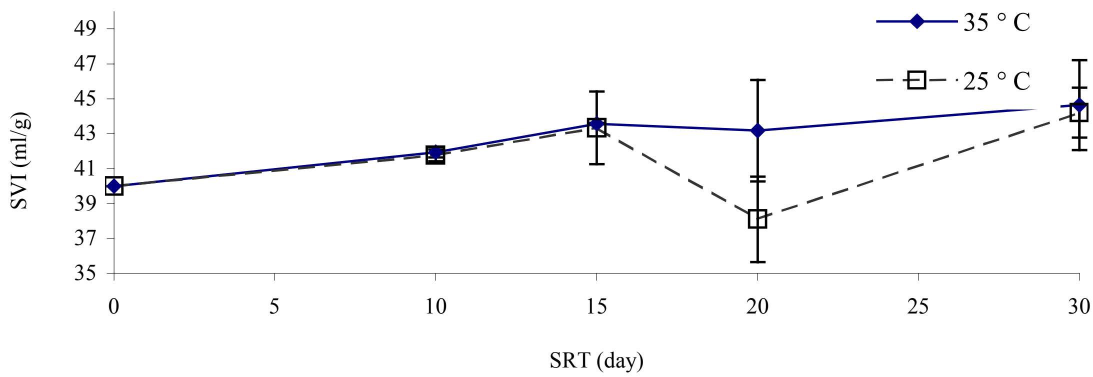

Fig. 2. Dependency of primary sludge settleability expressed as SVI, on anaerobic digestion conditions (SRT: 0, 10, 15, 20 and 30 days; Temp. 25 and 35 °C) in CSTRs. SRT=0 stands for the raw influent

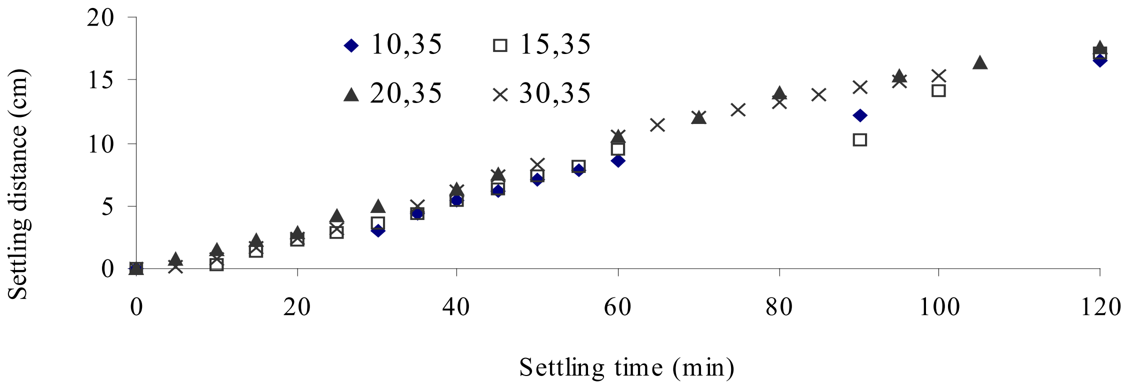

Fig. 3. Dependency of digested anaerobically primary sludge settleability measured as the developed clear distance inside the CSTR reactors after switching off the mixer, the CSTRs were operated at different conditions (SRT: 0, 10, 15, 20 and 30 days; Temp. 35 °C). The working height of the CSTR is 35 cm. SRT=0 stands for the raw influent

Discussion

The results presented reveal that sludge digestion conditions highly influence the sludge physical characteristics and behaviour. The worsening of the dewaterability of the sludge digested at 10 days SRT and 25 ºC is probably due to the coupled effect of bigger particles transfer to the pool of small particles and poor digestion conditions (Chapter 4). This result confirms the finding of Lawler *et al.* (1986) who showed that when digestion does not work well, large particles are broken but small particles are created with a consequent gain in specific surface area, resulting in decreased dewaterability. Meanwhile, the improvement of dewaterability with improving digestion conditions up to optima can be explained by gasification of the small particles and consequently the reduction of specific surface area. That is because, the results indicate that the main improvement of organic matter conversion upon improving the digestion conditions from (10 days, 25 °C) to (10 days, 35 °C) takes place in the particles smaller than 0.100 mm, which confirms the results of Lawler *et al.* (1986). Karr and Keinath (1978) found that particles in the range of 0.001-0.100 mm had the most significant effect on dewaterability.

The here proved phenomena of the existence of optimal conditions for sludge dewaterability and the afterwards decline has also been slightly pointed out in literature. Lawler *et al.* (1986) found during anaerobic digestion of mixed primary sludge and secondary sludge (weight of activated sludge is % 20 of total solids) in CSTRs operated at 35 °C, a worsening trend of dewaterability with increasing SRT from 7 to 21 days , but this trend was not statistically significant at a 0.05 level. Similarly, Nellenschulte and Kayser (1997) showed during anaerobic digestion of primary sludge in CSTRs operated at 35 ºC, that after 15 days, the degradation of organic substances has advanced to such an extent that an optimal dewatering result was achieved. Further extension of the SRT did not result in any significant change of the dewatering result.

The decline of the dewaterability after the optimum might be due to increase of very small particles, which are normally referred to as fines due to biomass decay, and accumulation of inorganic material after long SRT (Nellenschulte and Kayser, 1997). Lawler *et al*. (1986) reported during anaerobic digestion of activated sludge in a CSTR operated at 10 days and 35 °C, some surface area reduction occurring in the particles larger than 2 μm, but the surface area increased in the smaller size range. Several authors reported that the dewatering result of sewage sludges is mainly determined by the amount of fines (Karr and keinath, 1978; Lawler *et al*., 1986; Nellenschulte and Kayser, 1997), viz. the dewaterability can be improved when the fines in digested sludge are reduced and the specific surface of all particles decreases. Moreover, the possible increase of the sludge flocs negative electrical charges with digestion might also be a reason for worsening of dewaterability results after the optima. The sludge charges are reported to influence the dewatering characteristics of sludge (Poxon and Darby, 1997) due to alteration of the electrostatic forces between the water molecules and the sludge particles.

Nellenschulte and Kayser (1997) found that increasing the fines (1.9- 3.9 ηm) content in the sludge by means of mechanical disintegration, resulted in a significant increase of the total negative charge of sludge particles, expressed by zeta potential viz. after agitating the sludge for 15 minutes, the zeta-potential changed from –4.5 to –13.9 mV. Moreover, they showed that, the longer the digestion period, the higher becomes the fine particles content in the digested sludge. After a digestion period of 15 days at 35 °C, the fines content of primary sludge increased from 2,031 particles to 2,080 particles and to 2,511 particles per gram solids after a digestion period of 25 days, which might suggest that solids become more negatively charged with digestion. Different to this speculation, Elmitwalli *et al*. (2001) reported during anaerobic batch sewage digestion at 35 °C for a period of 135 days, a slight decrease in the negative zeta-potential, viz. from –22.6 to –19.5 mV.

The slight worsening trend of the digested sludge settling characteristics with digestion might be due to increase in sludge particles electrical charges. Forster and Dallas-Newton (1980) found that increasing the sludge floc electro negativity causes the sludge settling characteristics to deteriorate, due to particle repulsion. The influence of digestion conditions on the sludge electrical charges and the influence on the physical behaviour need to be investigated.

CONCLUSIONS

- Anaerobic digestion conditions have a substantial influence on the sludge physical characteristics and behaviour.
- Particle size plays an important role in sludge physical behaviour, i.e. dewaterability.

- The majority of particles in both raw and digested sludge is smaller than 0.100 mm.
- Anaerobic digestion at full methanogenic conditions, improves dewaterability up to optima.
- The sludge settling characteristics is slightly affected due to digestion. In practice, this effect is expected to be negligible.
- More research is recommended into the interaction between sludge physical-chemical characteristics and the digestion conditions with special emphasis on electrical charges and physical behaviour.

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Chapter 6

ANAEROBIC SEWAGE TREATMENT IN A ONE-STAGE UASB AND A COMBINED UASB-DIGESTER SYSTEM

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ANAEROBIC SEWAGE TREATMENT IN A ONE-STAGE UASB AND A COMBINED UASB-DIGESTER SYSTEM

ABSTRACT

The potential of a novel technology consisting of a UASB complemented with a digester (UASB-Digester) for mutual sewage treatment and sludge stabilisation under low temperature conditions was investigated. The performance of the UASB-Digester system was compared with a one stage UASB. The UASB reactor was operated at a HRT of 6 hours and controlled temperature of 15 °C, the average sewage temperature in the Middle East countries during wintertime, while the digester was operated at 35 °C. The UASB-Digester provided substantially better removal efficiencies and conversion than the one stage UASB reactor ($p<0.05$). The achieved removal efficiencies in the UASB-Digester and the one stage UASB for CODt, CODss, CODcol and CODdis are "66, 87, 44 and 30" and "44, 73, 3 and 5"% for both systems, respectively. The wasted sludge from the UASB-Digester is much more stabilised. The performance of the UASB digester is as good as that achieved in tropical countries. Therefore, the anaerobic sewage treatment at low temperature in a UASB-Digester system is recommended.

Key words—Anaerobic treatment; hydrolysis; low temperature; methanogenesis; Middle East; one stage UASB; sewage; sludge re-circulation; suspended solids; UASB-Digester

INTRODUCTION

Anaerobic digestion has been broadly recognised as the core of sustainable waste management (Zeeman and Lettinga, 1999; Hammes *et al.*, 2000). The UASB reactor is the most widely and successfully used high rate anaerobic system for the treatment of several types of industrial (high to medium strength) wastewater and low strength domestic wastewater (Singh *et al.*, 1996; Lettinga, 2001). It has been successfully used for sewage treatment in several tropical countries like India, Colombia and Brazil at both full scale and pilot scale (Haandel and Lettinga, 1994). The ambient temperature in these countries is rather high and constant along the year ranging between 20 and 30 °C, and the sewage strength is low.

In the Middle East region, the ambient temperature fluctuates between winter and summer (average sewage temperature 15 and 25 °C during winter and summertime, respectively). Moreover, sewage is characterised by high concentration with a high fraction of suspended solids (Chapter 4). For that region and others of moderate temperature climates, experience with anaerobic treatment so far is still developing. The performance of the one-stage UASB systems at low temperature climates (5-20 °C) is highly limited by the hydrolysis of entrapped solids, which accumulate in the sludge bed when high loading rates are applied (Man, 1990; Sanz and Fdz-Polanco, 1990; Lettinga *et al.*, 1993; Zeeman and Lettinga, 1999). Consequently, the amount of excess sludge will increase leading to a lower solids retention time (SRT) which might limit methanogens growth (Haandel and Lettinga, 1994) resulting in a poor soluble COD removal and deterioration of sludge stability (Wang, 1994).

The anaerobic sewage treatment is certainly not limited to regions of hot climates. However, at low temperatures, more development of the reactor technology is needed (Lettinga and Hulshoff Pol, 1991; Wang, 1994; Sayed and Fergala, 1995; Lettinga, 1996; Kalogo and Verstraete, 1999; Elmitwalli, 2000). Two stage systems consisting of a sequential high loaded first stage followed by a methanogenic stage (granular sludge) have been proposed for stable process performance during sewage treatment at low temperature (Wang, 1994; Elmitwalli, 2000). However, when applying high loaded reactors, the produced sludge is by definition not stabilised and needs further stabilisation in a separate digester. Lettinga and Hulshoff Pol (1991) proposed a novel technology consisting of an integrated high loaded UASB and a digester for the treatment of sewage with a high concentration of suspended solids under low temperature conditions. The solids, which are entrapped in the sludge bed, are conveyed to a digester operated at optimal conditions where they can be stabilised and the digested sludge (methanogenic sludge) is recirculated to the UASB reactor to improve its methanogenic capacity.

In the present work, the anaerobic sewage treatment using a UASB-Digester system has been investigated at a wastewater temperature of 15 °C. The UASB was firstly operated without incorporating the digester aiming at identifying the best location in the sludge bed for taking and re-circulating sludge and comparison of the performance of the UASB with the UASB-Digester system. The UASB-Digester was operated aiming at performance validation and demonstration in addition to formulation of criteria for good performance and further improvement.

MATERIALS AND METHODS

Experimental set-up

The experimental work has been carried out over two successive periods. Firstly, a pilot - scale of a one stage flocculent sludge UASB reactor (volume, height, diameter: 140 l, 325 cm, 23.5 cm) was operated at 15 °C without incorporating the digester. Afterwards, the UASB was modified to the UASB-Digester system by incorporating a CSTR digester (working volume 106 l) operated at 35 °C. A schematic diagram of the experimental set-up is illustrated in Fig. 1. The UASB reactor and the digester were constructed from Plexiglas and PVC tubes, respectively. The temperatures of the reactors content were controlled by re-circulating water in tubes wrapping the reactors through external thermostats. 5-cm thick rock wool sheets were wrapped around the UASB reactor for thermal isolation. Taps were installed over the whole UASB height at ± 25 cm for sludge discharge, re-circulation and analysis. The digester content was continuously mixed at 8 rpm.

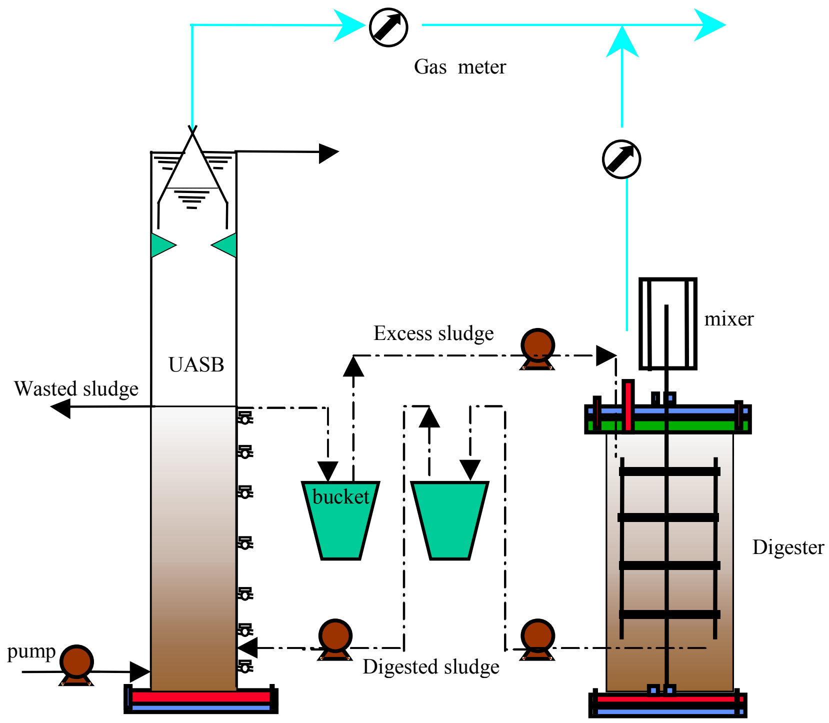

Fig. 1. Schematic diagram of the UASB-Digester pilot plant

Pilot plants operation and start up

The UASB reactor was initially inoculated with an anaerobic sludge discharged from a 6-m^3 research pilot - scale USAB reactor, located in our research hall. This reactor is operated at ambient temperature, treating domestic sewage originating from the village of Bennekom - The Netherlands. The digester was inoculated with digested primary sludge from the wastewater treatment plant of Ede-The Netherlands. During the operation of the one stage UASB system, the excess sludge was fed to the digester to accelerate the digester start up, without sludge re-circulation.

The sludge bed was kept below tap 9, *ca.* 190 cm from the UASB bottom, by once daily opening this tap for discharging the sludge accumulated above. The discharged sludge was collected in 10 litre buckets, from which the sludge was immediately fed to the digester by a peristaltic pump. At the same time, the digester effluent was pumped out to another bucket, while a third pump was re-circulating it to tap 1 of the UASB, *ca.* 15-cm from the bottom. Sludge was wasted from the UASB at tap 9 after a settling period of half an hour after finishing the recirculation process. The one stage UASB and the UASB-Digester were operated for 81 and 83 days, respectively of which the first 35 and 33 days were (arbitrarily) considered as start-up periods. The reactors behaviour during the steady state period met the criteria for steady state set by Noyola *et al.* (1988) and Polprasert *et al.* (1992). Noyola *et al.* (1988) considered 'steady state' of an anaerobic reactor treating domestic wastewater to be achieved after an operation period of 10 times the new HRT with a minimum of 2 weeks. Polprasert *et al.* (1992) considered 'steady state' to be achieved once the effluent COD concentrations at the anaerobic treatment of slaughterhouse wastewater varied within ±10%. Elmitwalli (2000) considered these criteria satisfactory for achieving steady state conditions.

Sewage

The used sewage of the village of Bennekom-The Netherlands, comes from a combined sewer system and is continuously pumped to our experimental hall. The concentration of this sewage was slightly (about 9%) increased by being continuously pumped to a 350 l settler at a flow rate of 55 l/hr of which 12 l/h was overflowed to the sewer system and 43 l/h was pumped during 8:00 am - 21:00 p.m. to a buffer tank, from which the UASB was continuously fed. The content of the buffer tank was continuously mixed at 60 rpm. Composite samples for 48, 48 and 72 hours of the buffer tank and the UASB effluent were collected in containers stored in a refrigerator at 4 °C.

On-line measurement of methanogenic activity of the UASB-reactor

Around 1000 litre of wastewater was stored in the influent holding tank for which sodium acetate was added to achieve a final concentration of around 350 mg VFA-COD/l. After around two HRT, the biogas production was monitored from time to time and analysed for gas composition. Once the stored wastewater was almost finished, the holding tank was refilled as mentioned above. Upon finishing the experiment, the solids content of the sludge bed was assessed and the biogas composition was analysed. The test was performed over six days at the end of the steady state period.

Analytical methods

COD analysis was carried out colorimetrically using the micro-method as described by Jirka and Carter (1975). Raw samples were used for COD total (CODt); filtered samples through 4.4 μm folded paper-filters (Schleicher & Schuell 5951/2, Germany) for COD filtrate (CODpf) and through 0.45 μm membrane filters (Schleicher and Schuell ME 25, Germany) for dissolved COD (CODdis). The COD suspended (CODss) and COD colloidal (CODcol) were calculated by the difference between "CODt and CODpf" and "CODpf and

CODdis", respectively. VFA were determined for membrane filtered samples by a HP model 5890 A gas chromatograph equipped with supelcoport (11-20 mesh) coated with 10% Fluorrad FC 431. The temperature of the injector, the column and the FID were 200, 130 and 280 ºC, respectively. N_2 at 20 ºC saturated with formic acid was used as a carrier gas. The biogas composition (CH_4, CO_2, N_2, O_2) was determined using a Fisons 8000 gas chromatograph equipped with a Teflon column (1.5m x 2mm, Chromosorb 108, 60-80 mesh) connected in parallel to a steel column 1.2 m x 2mm, Mol sieve 5A, 60-80 mesh). The injected sample volume was 100 µm, the carrier gas was helium and the oven and TCD temperatures were 40 and 100 ºC, respectively. The Kjeldahl nitrogen (Nkj), sludge volume index (SVI), capillary suction time (CST), total solids (TS), volatile solids (VS), suspended solids (SS) and volatile suspended solids (VSS) were measured according to the *Dutch Standard Normalised Methods* (1969). Total ammonium (NH_4^+- N) was determined with an automatic analyser (Skalar 1520).

Total lipids (neutral lipids plus LCFA) were determined according to the Soxlett extraction method (APHA, 1992). Carbohydrates (total, suspended, colloidal and dissolved) were determined according to Bardley *et al.* (1971) with glucose as standard. The analysed sludge samples for total COD, total carbohydrates and Nkj were previously homogenised by ultra – Turrax (Heidolph, DIAX 900). Sludge stability was measured in serum bottles (500 ml) incubated at 30 ºC, as described by Elmitwalli (2000). Methane in the headspace was determined by the gas displacement method using 5% NaOH solution. Dewaterability of the sludge was detected by the capillary suction time (CST, in seconds) and expressed as filterability constant (X), which is calculated based on the CST (Vesilind, 1988).

Calculations

The equations which have been used for describing the conversion processes and solids retention times (SRT) are presented below.

<u>*One stage UASB*</u>

$$Hcarb\ (\%) = 100\ \frac{(Ccarb_{(inf)} - Ccarb_{(eff)})\ x\ Q_{(inf)} - Ccarb_{(w)}\ x\ Q_{(w)}}{Ccarb_{(inf)}\ x\ Q_{(inf)}} \qquad (1)$$

$$Am\ (\%) = 100\ \frac{NH4(eff)\ x\ Q(eff) + Q(w)\ x\ NH4(w) - Q(inf)\ x\ NH4(inf)}{Nkj(inf)\ x\ Q(inf)} \qquad (2)$$

$$H(\%) = 100(\frac{CH_4\text{-}COD + CODdis_{(eff)} - CODdis_{(inf)}}{CODt_{(inf)} - CODdis_{(inf)}}) \qquad (3)$$

$$A(\%) = 100(\frac{CH4\ \text{-}COD + VFA\ \text{-}COD\ _{(eff)} - VFA\ \text{-}COD\ _{(inf)}}{CODt\ _{(inf)} - COD\ \text{-}VFA\ _{(inf)}}) \qquad (4)$$

$$M(\%) = 100(\frac{CH4\ \text{-}COD}{CODt\ _{(inf)}}) \qquad (5)$$

$$SRT = \frac{M_{SB}}{Q_{(w)} \; x \; X_{(w)}} \tag{6}$$

UASB-Digester

$$Hcarb_{(U-dig)} (\%) = 100 \; \frac{(Ccarb_{(inf)} - Ccarb_{(eff)}) \; x \; Q_{(inf)} - Q_{(w)} * Ccarb_{(w)}}{Qinf \;\; x \; Ccarb_{(inf)}} \tag{7}$$

$$Hcarb_{(dig)} (\%) = 100 \; \frac{Q_{(exc)} \; x \; Ccarb_{(exc)} - Q_{(dig)} \; x \; Ccarb_{(dig)}}{Q_{(inf)} (Ccarb_{(inf)} - Ccarb_{(eff)})} \tag{8}$$

$$Hcarb_{(UASB)} (\%) = Hcarb_{(U-dig)} - Hcarb_{(dig)} \tag{9}$$

$$Hcarb_{(digester)} (\%) = 100 \, \frac{Ccarb_{(exc)} - Ccarb_{(dig)}}{Ccarb_{(exc)}} \tag{10}$$

$$Hnitrog. (digester) (\%) = 100 \, \frac{(Nkj(exc) - NH4(exc)) - (Nkj(dig) - NH4(dig))}{(Nkj(exc) - NH4(exc))} \tag{11}$$

$$Hlipids_{(digester)} (\%) = 100 \, \frac{Clipids_{(exc)} - Clipids_{(dig)}}{Clipids_{(exc)}} \tag{12}$$

$$Am (\%) = 100 \, \frac{Q(inf) \; x \; NH4(eff) - Q(inf) \; x \; NH4(inf) - Q(dig) \; x \; NH4(dig) + Q(exc) \; x \; NH4(exc) + Q(w) \; x \; NH4(w)}{Q(inf) \; x \; Tkj(inf)} \tag{13}$$

$$SRT_{(UASB)} = \frac{M_{SB}}{Q_{(exc)} \; x \; X_{(exc)}} \tag{14}$$

$$SRT_{(dig)} = HRT = \frac{V_{(dig)}}{Q_{(dig)}} \tag{15}$$

Where: *H*, percentage hydrolysis; *A*, percentage acidification; *M*, percentage methanogenesis; *Am*, percentage ammmonification (ammonia production in the UASB reactors due to particulate-N hydrolysis and acidification as a percentage of the influent Nkj-N); Ccarb, total carbohydrates; w, wasted sludge from the UASB; *inf*, influent; *eff*, effluent; *exc*, excess sludge (sludge conveyed from the UASB to the digester); dig, digested sludge (digester effluent); M_{SB}, total sludge mass (g VS/l) in the sludge bed (average of 5 measurements); *Q*, flow rate (l/d); equations 2, 3 and 4 are also valid for the UASB and digester of the UASB-Digester system; CH_4 as COD = CH_4 as $COD_{(liquid\ form)}$+ CH_4 as $COD_{(gas\ form)}$

RESULTS AND DISCUSSION

Removal of various COD fractions

The main characteristics of the sewage used in this research are presented in Table 1. The reduction of the influent COD during the performance of the UASB-Digester run in comparison with the one stage UASB is due to dilution with rainwater.

Table 1. Characteristics of the influent sewage (Bennekom - The Netherlands)

[+]parameter	UASB		UASB-Digester	
	[++]n =16	n = 6	n = 20	n = 6
CODt	721 (171)		460 (122)	
CODss	398 (167)		251 (100)	
CODcol	151 (46)		124 (24)	
CODdis	172 (42)		86 (37)	
COD-VFA	76 (38)		34 (25)	
Total carbohydrates		68 (22)		99 (29)
Suspended carbohydrates		41 (3)		76 (27)
Colloidal carbohydrates		15 (9)		11 (2)
Dissolved carbohydrates		12 (4)		12 (2)
Nkj-N		72 (12)		50 (13)
NH_4^+-N		57 (14)		43 (8)
pH	7.4 (0.1)		7.3 (0.2)	

[+] All parameters are in mg/l except pH; [++] number of measurement except pH (daily)

The mean values of effluent CODt and fractions and the removal efficiencies of the UASB and the UASB-digester systems are depicted in Table 2. The results clearly reveal that the UASB-Digester system achieved substantially better physical removal efficiencies of all COD fractions as compared to the one stage UASB system ($p<0.05$) and a more stable performance as can be seen by the low standard deviations.

Although, the removal of VFA-COD in the UASB of the UASB-digester was very high, the removal of CODdis was limited (30%) (Table 1). The achieved effluent soluble COD of almost 50 mg COD/l is obviously the lowest achievable soluble COD value in anaerobic sewage treatment of the village of Bennekom (Elmitwalli, 2000; Last *et al.*, 1992). Apparently, the soluble fraction of the effluent COD (partly) consisted of soluble microbial products (SMP), which are resistant to anaerobic degradation (Aquino and Stuckey, 2001). Nevertheless, the rainwater apparently did not only reduce the VFA concentration due to dilution but also due to reduction of VFA production in the sewer system. The non-VFA/VFA ratio in the run of the one stage UASB (dry weather flow) was 1.26 and increased to 1.53 during the performance of the UASB-Digester run (rainy period). That might be attributed to reduction of sewage retention time in the sewer system and/or reduction of organic matter contact with the biofilm, which probably covers the lower part of the sewer.

The substantial improvement of CODcol removal achieved in the combined system over the one-step UASB reactor might be due to the creation of better digestion conditions in the combined system. Elmitwalli (2000) found that the colloidal particles in the sewage are highly biodegradable and attributed the reported poor colloidal removal during anaerobic sewage treatment at low temperature to poor physical removal. Similarly, the improvement of CODss removal might result from the better digestion conditions, which can also justify the enhancement of CODcol removal, since colloids may be generated from the suspended solids (Elmitwalli, 2000). Accordingly, our results suggest the existence of a capturing - de-capturing mechanism for solids removal in the sludge bed. Consequently, the solids physical removal might not only be affected by the characteristics of the influent particles and the sludge but also by the digestion conditions.

Hydrolysis, acidification and methanogenesis

The percentage hydrolysis, acidification and methanogenesis are depicted in Table 3. In the one stage UASB reactor, not hydrolysis, but methanogenesis was limiting the overall conversion of organic matter to methane. The effluent contained a high amount of soluble COD amounting to 162(47) mg COD/l of which 80(40) mg COD/l was in the form of VFA. Differently, in the UASB of the UASB- Digester system, hydrolysis was the limiting step of the overall digestion processes. Moreover, the results clearly reveal that the conversion in the UASB of the UASB-Digester system is substantially higher than in the one stage UASB. On one hand, this can be attributed to the existence of better methanogenic conditions in the UASB of the UASB-Digester system, which might have enhanced the hydrolysis step through, e.g. improving the contact between the substrate and the hydrolytic enzymes due to biogas production (Sanders, 2001). On the other hand, the reduction of the OLR from 2.88 (0.69) g COD/l.d in the one stage UASB run to 1.84 (0.49) g COD/l.d in the UASB-Digester run, due to the lower influent COD, is also a factor that might have enhanced the conversion in the UASB-digester run. The COD mass balance presented in Fig. 2 show that the main COD conversion takes place in the UASB reactor.

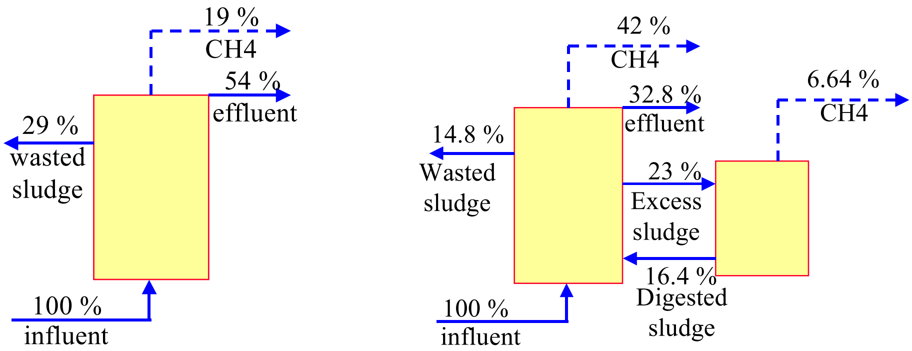

Fig. 2. COD mass balance of a one stage UASB (right) and a UASB-digester system (left) over a period of respectively 47 and 45 days. Both UASB reactors were operated at HRT=6 hours; T = 15 °C and the digester at SRT = 21 days and T = 35 °C.

Methane gas recovery

The results of the gas composition analysis revealed high methane content in the biogas released from the UASB and the UASB of the UASB-Digester system amounting to 73.38(1.21) % and 82.89(0.66) %, respectively. These values are similar to those reported in literature (Vieira and Souza, 1986; Elmitwalli, 2000). The recovered methane in the gas form from the one stage UASB, UASB of the UASB-digester and the whole UASB-digester systems amounted to 0.09, 0.15 and 0.17 Nm^3/kg CODremoved. Around 50 and 55 % of the total produced methane gas (Henry's law) in the UASB and the UASB of the UASB-Digester system respectively were lost in the effluent. The here presented gas production values of the UASB-Digester system are similar to those reported by Lettinga *et al.* (1993) (0.19 Nm^3/kgCOD removed) and by Uemura and Harada (2000) (0.16 Nm^3/kgCOD removed).

Characteristics of the retained sludge in the UASB reactors and excess, digested and wasted sludges

The characteristics of the excess, digested and wasted sludges are presented in Table 4 and Fig. 3. The results show a substantially higher VS/TS ratio of the excess sludge of the UASB reactor in comparison with that from the UASB of the UASB-Digester system, which indicates better stability of the latter. This was confirmed by the results of the stability test. The sludge digested in the digester has substantially higher stability than the excess sludge from the UASB of the UASB-Digester system due to dilution of the sludge bed with the influent raw solids. The relatively low SVI of the UASB, UASB of the UASB-Digester system and the digester sludges reveals high settleability with no influence of the digestion conditions.

The high filterability constant of the wasted sludge of the UASB of the UASB-Digester system as compared to that of the UASB reactor demonstrates that the first sludge is better dewaterable ($P<0.05$) but the digested sludge is not ($P<0.05$). This can be explained by production of small particles during digestion and hence increase of particles specific surface area (Lawler *et al.*, 1986). The dewaterability of the wasted sludge from the UASB-Digester system is better than that reported for excess sludge from anaerobic filter (Elmitwalli, 2000) and primary sludge digested in CSTRs (Miron *et al.*, 2000)

The VS and TS charts (Fig. 3a + b) show a decline trend in concentration from bottom to top of the sludge bed with clear stratification at the border of 40% height of the bottom of the reactor. The constant VS/TS ratio over the sludge bed height indicates that the sludge is equally stabilised over the bed.

Carbohydrates, nitrogenous compounds and lipids removal / conversion

The results presented in Table 6 reveal that the carbohydrates in sewage are efficiently removed in the UASB systems. The high removal efficiency of total carbohydrates is mainly attributed to the high removal of the suspended part of carbohydrates. The results show insignificant ($p<0.05$) difference in carbohydrate conversion between the UASB and the UASB-Digester system.

The Nkj-N was partly removed in the UASB reactors due to particulate N removal, but NH_4^+- N was released as a result of protein hydrolysis (Table 6). The hydrolysis of particulate N in the studied systems could not accurately be calculated as the difference between Nkj- N and NH_4^+- N was very low and within the marginal error of the used measuring instruments. Instead the hydrolysis of particulate N was based on ammonia production (eq. 2 and 12). The results show no ammonia production in the UASB, while it was significant ($p<0.05$) in the UASB of the UASB-Digester.

In the digester, the percentage conversion of the particulate N, carbohydrates and lipids based on the digester influent were 29 (5), 25 (16) and 52 (3), respectively. Nkj removal of 6 (1.5)% occurred in the digester ($p<0.05$) which might be due to ammonia precipitation through complex formation, e.g. precipitation of ammonium as struvite ($MgNH_4PO_4.6H_2O$) (Mamals *et al.*, 1994). The low value of carbohydrate degradation in the digester is due to the relatively high carbohydrate conversion in the UASB; the same could also stand for the lipids as much more lipid hydrolysis at the applied digestion conditions was found (Chapter 4) when digesting primary sludge in a CSTR.

Table 2. Effluent COD and VFA concentrations and removal efficiencies (%) during anaerobic sewage treatment in a UASB and a UASB-digester system. Standard deviations are presented between brackets

Reactor	Effluent concentration (mg/L)					Removal efficiency (%)				
	CODt	CODss	CODcol	CODdis	VFA-COD	CODt	CODss	CODcol	CODdis	VFA-COD
UASB	390	100	128	162	80	44	73	3	5	-8
	(62)	(36)	(19)	(47)	(40)	(9)	(14)	(46)	(17)	(42)
UASB-Digester	151	32	68	50	3	66	87	44	30	95
	(34)	(24)	(17)	(10)	(3)	(6)	(5)	(15)	(36)	(8)

Table 3. Percentage of hydrolysis (H), acidification (A) and methanogenesis (M) during the treatment of domestic sewage in a one stage UASB and a UASB-digester systems. Standard deviations are presented between brackets

	⊕UASB-Digester			One stage UASB
	UASB	+Digester	Overall system	
H	44	19.99		25
	(15)	(38.2)		(7)
A	41	19.68		24
	(12)	(40.83)		(8)
M	44	19.62	47	21
	(11)	(34.23)	(15)	(5)

+ calculations are based on the digester influent and effluent

⊕ calculations are based on total methane production including dissolved fraction

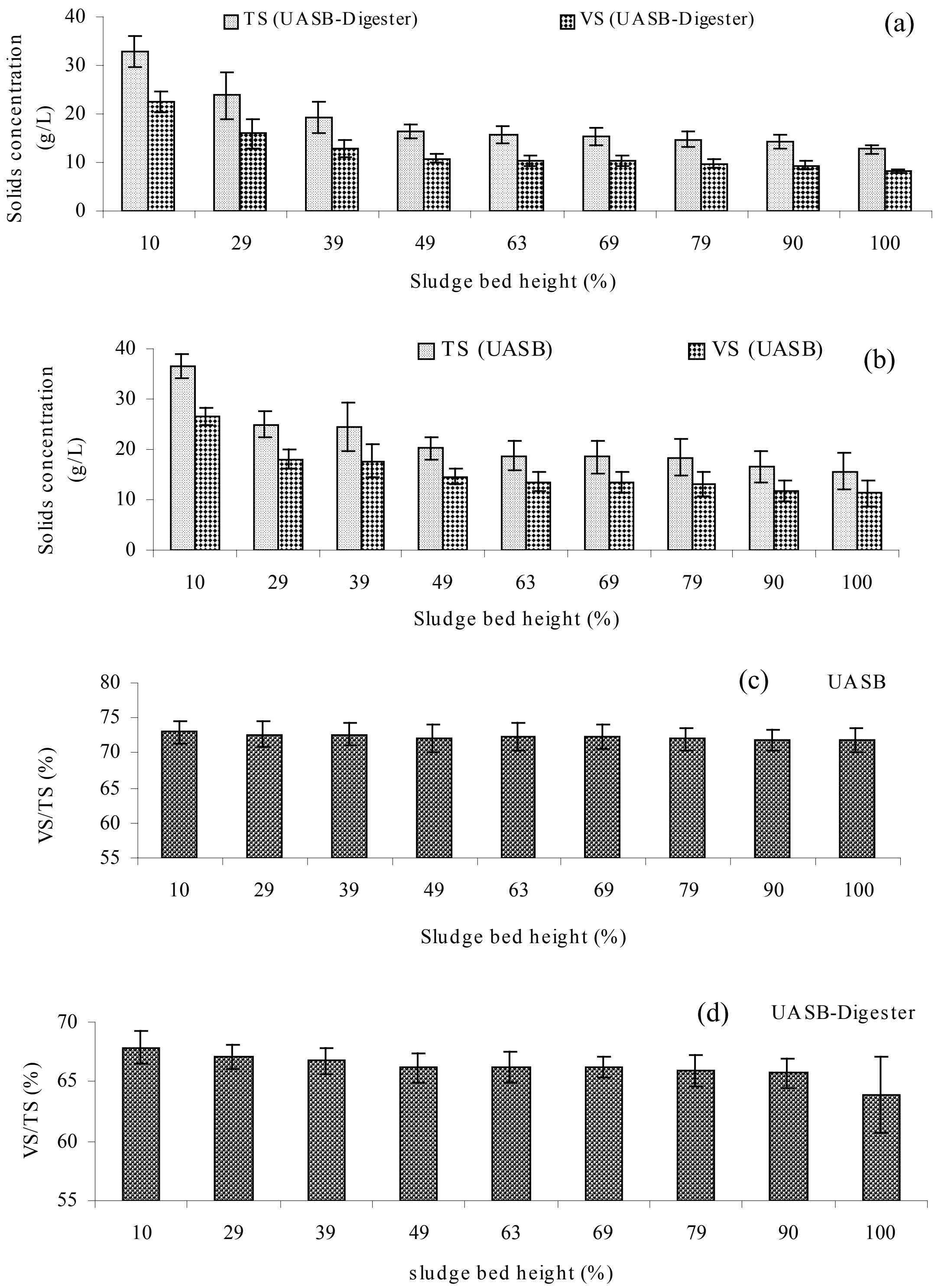

Fig. 3. Sludge profile of a UASB and a UASB of a UASB-Digester as VS and TS (a + b) and VS/TS ratio (c + d) along the sludge bed heights expressed as percentages of the total bed height. The 0-sludge height stands for the bottom of the reactors

Table 4. Characteristics of the retained and wasted sludge in the UASB system and the retained, excess and wasted sludge in the UASB-Digester system. Standard deviations are presented between brackets

Parameter	Unit	UASB	UASB-Digester	
			UASB	Digester
Retained sludge				
Concentration	g VS.l^{-1}sludge bed	16.81 (1.15)	13.63 (0.93)	5.34 (0.77)
$^{+}$SRT	day	24.27 ± 5.19	29.7 ± 9.36	21.16 ± 1.52
Meth. Activity	gCOD/gVS.d	$^{++}$0.058	$^{+++}$0.103 $^{++}$0.096	
Excess sludge				
VS/TS	%		67 (2)	
SVI	ml.g^{-1}.SS		37.35 (6.23)	
Stability	g CH_4-COD.g^{-1}COD			
COD/VS	gCOD/gVS		1.73 (0.01)	
Xx10^{6}	kg^{2}.m^{4}.s^{-2}		62.45 (17.66)	
Wasted sludge				
VS/TS	%	71 (1.13)	67.5 (2)	
SVI	ml.g^{-1}.SS	31.4 (1.7)	32.98 (1.40)	
Stability	g CH_4-COD.g^{-1}COD	45.6(0.5)	36.2(4.9)	
COD/VS	gCOD/gVS	1.87 (0.12)	1.74 (0.07)	
Xx10^{6}	kg^{2}.m^{4}.s^{-2}	29.26 (5)	59.09 (29.45)	
Digested sludge				
VS/TS	%			63.5 (0.6)
SVI	ml.g^{-1}.SS			30.09 (1.20)
Stability	g CH_4-COD.g^{-1}COD		19.9(6.6)	
COD/VS	gCOD/gVS			1.77 (0.13)
Xx10^{6}	kg^{2}.m^{4}.s^{-2}			26.14 (4.48)

$^{+}$SRT with 95% confidence interval; $^{++}$ methanogenic activity calculated on the basis of average gas production during steady state, no acetate was added. $^{+++}$ On line measured methanogenic activity as described in the materials and methods

Sludge production

The daily wasted sludge from the UASB and the UASB-Digester systems as a ratio of influent and removed CODt are presented in Table 5. The results reveal a significantly lower sludge production in the UASB-Digester as compared to the UASB system. The average sludge production in the UASB-Digester system is lower than that calculated from results given by Elmitwalli (2000) (0.58 g COD/ g CODremoved) during anaerobic sewage treatment in an anaerobic filter operated at 4 hours HRT and 13 °C, and similar to that reported by Cavalacanti *et al.* (1999) (0.15 and 0.13 g TSS/ g CODin and 0.08 and 0.07 g VSS/ g CODin) during anaerobic sewage treatment at 4 - 8 hours HRT and 25 - 28 °C.

General discussion

The results clearly reveal the high potential of the UASB-Digester system for sewage treatment at low temperature of 15 ºC as it couples wastewater treatment and sludge stabilisation. The system solves the problem of solids accumulation in the sludge bed at low temperature conditions (Man *et al.*, 1986; Lettinga *et al.*, 1993; Zeeman and Lettinga, 1999). Consequently it can be successfully applied for sewage treatment in almost all countries with low or fluctuating temperature climates like the Middle East. Ruiz *et al.* (1998) reported during anaerobic treatment of domestic sewage at 20 ºC in a bench scale UASB (2 l), an increase of the COD and SS removal efficiencies by about 5% when incorporating a digester.

The system is not only superior to the one stage UASB system for sewage treatment at low temperature climates but also to other so far proposed systems i.e. the two stage Hydrolysis Upflow Sludge Bed (HUSB) + Expanded Granular Sludge Bed (EGSB) system (Wang, 1994) and Anaerobic filter (AF) - Anaerobic hybrid (AH) (Elmitwalli, 2000) as illustrated in Table 7. The table shows that the effluent CODt of the UASB-Digester is lower than that of the HUSB-EGSB system and the AF+AH when operated at 2+4 and 3+6 hours and differs only in 15 mg COD/l from the AF-AH system when operated at 4+8 hours. Nonetheless, the excess sludge from the UASB-Digester is well stabilised, while that from both the HUSB+EGSB and AF+AH systems was poorly stabilised and consequently would need further digestion in a separate digester. Since the digester should be incorporated to the HUSB+EGSB and AF+AH systems, it is rational to compare the volume of those systems with only the volume of the UASB without including the volume of the digester. As the UASB of the UASB-Digester system, HUSB+EGSB and AF+AH were operated at 6, 5 and 12 hours, the UASB -Digester system is more compact in comparison to the AF+AH but not the HUSB+EGSB. Nonetheless, the UASB-Digester system is less complex and achieved better effluent quality. Moreover, the HUSB+EGSB system is not expected to be an attractive technology for the treatment of high strength sewage at low temperature like the case in the Middle East (CODss: ± 1000 mg COD/l) (Chapter 3). The reduction of suspended solids removal in the HUSB with decreasing temperature will certainly lead to deterioration of granular sludge in the EGSB system (Man *et al.*, 1986; Lettinga and Hulshoff Pol, 1991).

The removal efficiency of total COD in the UASB-Digester system is as high as those reported for tropical countries like Sao Paulo-Brazil by (Vieira, 1988) (70%), Bucaramanga - Columbia by (Schellinkhout *et al.*, 1988) (66 %) and Kanpur - India by (Draaijer *et al.*, 1992) (62 - 70 %).

The volume of the digester can be greatly reduced. Our previous work (Chapter 4) showed hardly any improvement in the digester performance at increasing the SRT above 10 days at 35 °C or 15 days at 25 °C. Moreover, due to solids content stratification in the sludge bed while maintaining a uniform stability, sludge with a high concentration can be conveyed from the UASB to the digester. This will lead to substantial reduction in the digester volume. Similarly, the volume of the UASB in the UASB-Digester can also be reduced as the results clearly show that it was under loaded.

CONCLUSIONS

- The here presented UASB-Digester system represents an efficient technology for anaerobic sewage (pre) treatment at low temperature conditions i.e. it provides average removal efficiencies for CODt, CODss, CODcol and CODdis of 66, 87, 44 and 30%, respectively.
- The sludge re-circulation improved both the solids physical removal and the conversion as it increased the methanogenesis from 20 % in the one stage UASB reactor to 47 % in the UASB-Digester system.
- The sludge production from the UASB-Digester system is very low, and is very well dewaterable and stabilised.

RECOMMENDATIONS

- The sludge to be re-circulated from the UASB to the digester is recommended to be taken from the first 40% height of the sludge bed.
- The sludge wastage is recommended to be from the digester not the UASB.
- The produced biogas should be reused for heating the digester content.

Table 5. Sludge production during anaerobic sewage treatment in a UASB and a UASB-digester. Standard deviations are presented between brackets

	Sludge production per influent COD			Sludge production per removed COD		
	g COD/g CODinf	g TS/g CODinf	g VS/g CODinf	g COD/g CODrem	g TS/g CODrem	g VS/g CODrem
UASB	0.31 (0.08)	0.24 (0.06)	0.17 (0.04)	0.77 (0.34)	0.58 (0.25)	0.41 (0.18)
UASB-Digester	0.143 (0.026)	0.125 (0.022)	0.082 (0.014)	0.212 (0.066)	0.185 (0.056)	0.122 (0.037)

Table 6. Carbohydrates, Nkj and NH_4^+ removal efficiencies (% of the influent), carbohydrates hydrolysis (% of the influent carbohydrates) and ammonification (ammonia production in the UASB and the UASB of the UASB-digester systems, % of the influent *Nkj*-N) during anaerobic sewage treatment in a UASB and a UASB-digester systems. Standard deviations are presented between brackets

		UASB	UASB-digester		
Removal efficiency			UASB	Digester	UASB-Digester
Carbohydrates	Total	64.27 (11)	83 (4.98)		
	Suspended	93.66 (7.34)	98.26(2.13)		
	Colloidal	20.4(45.39)	33.88(29.43)		
	Dissolved	26.6 (14.85)	34.95(16.92)		
Nkj-N		4.26(6.21)	4.82(6.19)		
NH_4^+-N		-1.8(7.63)	-17.95(5.76)		
Carbohydrates hydrolysis		23.11 (15.25)	29.57 (20.26)	16.12 (11.78)	44.47 (9.79)
Ammonification		0.21(0.62)	14.93(2.95)		

Table 7. Research results of anaerobic sewage originated from the village of Bennekom-The Netherlands at low temperature

Reactor	HRT	T	+OLR	CODt		Removal				Excess sludge	Reference
				Influent	Effluent	CODt	CODss	CODcol	CODdis	VS/TS	
	hr	°C	gCOD/l.d	(mg/l)	(mg/l)	(%)	(%)	(%)	(%)	(%)	
HUSB	3	14-21	5.6	697	432	38	65	23	-2.6	70	Wang, 1994
HUSB	3	8-13	4.1	507	319	37	49	39	16	n.m.	Wang, 1994
HUSB+EGSB	3+2	14 - 21	3.2	662	200 - 250	69	79	40	51	-	Wang, 1994
HUSB+EGSB	3+2	8 - 13	2.4	500	200 - 250	51	67	24	41	-	Wang, 1994
AF	4	13	3.1	520	234	55	82	35	38	81	Elmit., 2000
AF+AH	2+4	13	2.1	528	211	58.6	70.5	45.3	54.8	76	Elmit., 2000
AF+AH	3+6	13	1.4	528	195	63	78.8	45.8	53.6	78	Elmit., 2000
AF+AH	4+8	13	0.9	461	135	70.6	91.3	59.6	55.2	81	Elmit., 2000
UASB-Digester	6	15	1.9	460	151	66	87	44	30	++63.5	This research

n.m.: not mentioned; +for the two stage systems, OLR stands for the whole system; ++digested sludge

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Chapter 7

ANAEROBIC TREATMENT OF SEWAGE IN A UASB-DIGESTER SYSTEM

SUMMARY, DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS

ANAEROBIC TREATMENT OF SEWAGE IN A UASB-DIGESTER SYSTEM

SUMMARY, DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS

INTRODUCTION

Anaerobic digestion has been broadly recognised as the core of sustainable waste management (Zeeman and Lettinga, 1999; Hammes *et al.*, 2000). The upflow anaerobic sludge bed (UASB) reactor is the most widely and successfully used high rate anaerobic system for sewage treatment (Agrawal *et al.*, 1997; Monroy *et al.*, 2000; Lettinga, 2001). It has been successfully used for sewage treatment in several tropical countries like India, Colombia and Brazil, both at full scale and pilot scale (Haandel and Lettinga, 1994). The ambient temperature in these countries is rather high and constant along the year ranging between 20 and 30 °C, and the sewage strength is low. With respect to regions of cold climates, further development of the reactor technology is needed (Lettinga and Hulshoff Pol, 1991; Wang, 1994; Sayed and Fergala, 1995; Lettinga, 1996; Kalogo and Verstraete, 1999; Elmitwalli, 2000). The performance of the one-stage UASB systems at low temperature climates (5-20 °C) is highly limited by the slow hydrolysis of entrapped solids, which accumulate in the sludge bed when high loading rates are applied (Man, 1990; Lettinga *et al.*, 1993; Zeeman and Lettinga, 1999). Consequently, the amount of excess sludge will increase leading to a lower solids retention time (SRT) which might limit methanogenic growth (Haandel and Lettinga, 1994) resulting in a poor soluble COD removal and deterioration of sludge stability (Wang, 1994).

Two stage systems consisting of a high loaded first stage, followed by a methanogenic stage (granular sludge) have been proposed to achieve stable process performance during sewage treatment at low temperature (Wang, 1994; Elmitwalli, 2000). However, when applying high loaded reactors, the produced sludge is by definition not stabilised and needs further stabilisation in a separate digester. Lettinga and Hulshoff Pol (1991) proposed a novel technology consisting of an integrated high loaded UASB and a digester for the treatment of sewage with a high concentration of suspended solids under low temperature conditions. The problem of sewage treatment at low temperature escalates with increasing suspended solids concentrations, which is the case in several Middle East countries like Palestine and Jordan.

This thesis describes a study including experimental work on the potential of anaerobic sewage treatment in the Middle East region.

SOLIDS REMOVAL IN UPFLOW REACTORS

The desk study, which is described in **Chapter 2**, deals with the mechanisms and parameters affecting particles separation from wastewater in mainly upflow anaerobic reactors. Despite the fact that the functioning of UASB systems depends on both physical parameters and biological processes, the physical parameters have been barely reported in literature. The reason is that the underlying mechanisms are very complex and depend on various interrelated parameters. In addition, the lack of a serious attempt to gather the entire physical theme in one picture, have resulted in just a superficial understanding of this field of science. Better understanding of the interaction and role of these parameters is essential for the development of anaerobic treatment technologies. In this study, the various parameters that might affect the solid liquid separation process by filtration through the sludge bed of a UASB have been elaborated. These parameters have been classified into (1) reactor operational conditions (temperature, organic loading rate, hydraulic retention time and upflow velocity), (2) influent characteristics (influent concentration, influent particle size and influent particle charge) and (3) sludge bed characteristics (particle size distribution, exopolymeric substances, and charge). The overall output of this study includes (1) a literature review, (2) structuring of this field of science, and (3) highlighting fields where research is needed.

WASTEWATER MANAGEMENT IN PALESTINE

Results of experimental work and of a literature search dealing with sewage characteristics and management in Palestine are described in **Chapter 3**. Since sewage characteristics dictate the applicability of anaerobic treatment technologies (Lettinga *et al.*, 1993), it becomes crucial for the sake of proper selection and sizing of treatment technologies to increase the knowledge on this field, which particularly is still very limited in Palestine. The analytical work was conducted on time-interval composite samples collected from three locations in Ramallah/Al-Bireh district in the West Bank. The available data from previous studies about sewage characteristics in Palestine were compiled. The results revealed that the sewage in Palestine is of high strength according to the sewage strength classification of Metcalf and Eddy (1991) and of Henze (1997) and in comparison to the sewage characterises of several other countries. This high sewage strength can be attributed to the low water consumption, industrial discharges and habits of people, resulting in a high specific COD production (gCOD/c.d). The results were used for model calculations (Zeeman and Lettinga, 1999) to elucidate the applicability of the one stage UASB reactor in Palestine. The model calculations reveal that the application of the one stage UASB reactor is only possible when designed at a HRT of more than 22 hours to overcome the sewage high solids content and low sewage temperature during wintertime. As an alternatively, a novel technology, proposed by Lettinga and Hulshof Pol (1991), consisting of a UASB reactor integrated with a sludge digester is recommended. Based on the economic situation and the geographic and demographic distribution, it is recommended to consider decentralised anaerobic treatment to be the core of the Palestinian wastewater management strategy. Several treatment concepts are proposed according to sustainability criteria defined by Lettinga *et al.* (2001).

ANAEROBIC STABILISATION AND CONVERSION OF BIOPOLYMERS IN PRIMARY SLUDGE

Chapter 4 deals mainly with optimising the design of the digester of the UASB-Digester system in terms of sludge stabilisation. The UASB-digester system was simplified by eliminating the UASB and the primary sludge was used for feeding completely stirred tank reactors (CSTRs). The effects of sludge retention time (SRT) and temperature on the hydrolysis, acidification and methanogenesis of primary sludge were investigated. The CSTRs were operated to maintain SRTs of 10, 15, 20 and 30 days, and at process temperatures of 25 and 35 ºC. Another CSTR was operated at 75 days SRT and 15 ºC, the sewage temperature in Palestine, the Middle East, during wintertime. In addition, the rates of hydrolysis and the biodegradability of primary sludge were assessed in batch reactors incubated at 15, 25 and 35 ºC. 15 and 25 °C are the average minimum and maximum wastewater temperatures in Palestine and 35 ºC is the optimum temperature for methanogens in the mesophilic range. The results revealed that the major achievement of sludge stabilisation occurred at SRT ≥ 10 and 15 days at process temperatures of 35 and 25 ºC, respectively. Hydrolysis was found to be the rate limiting-step of the overall digestion process, for the reactors operated at 35 and 25 ºC, except for the reactor operated at 10 days and 25 ºC. At the latter conditions, acidification of LCFA was rate limiting in comparison to hydrolysis; hydrolysis of protein and particulate carbohydrates were rate limiting of the acidification step, while methanogenesis is the rate-limiting step for the overall process. Protein hydrolysis was limited to a maximum value of 39% at 30 days and 35 ºC due to protein availability in the form of biomass. Methanogenesis occurred at 75 days and 15 ºC. The biodegradability of primary sludge was around 60%, and showed no temperature dependence. The hydrolysis of the main biopolymers and overall particulate COD of the primary sludge digested in CSTRs were well described by first order kinetics, in case hydrolysis was the rate-limiting step. Similarly, the hydrolysis of the overall particulate COD of the primary sludge digested in batch reactors was described by first order kinetics and revealed a strong temperature dependence, which follows the Arrhenius equation.

THE INTERACTION BETWEEN DIGESTION CONDITIONS AND SLUDGE PHYSICAL CHARACTERISTICS AND BEHAVIOUR FOR ANAEROBIC DIGESTION OF PRIMARY SLUDGE

The results of the desk study, described in Chapter 2, indicates that the digestion conditions are likely to have a strong influence on the sludge physical–chemical characteristics and consequently the physical behaviour of the sludge. Several sludges digested in CSTRs at different conditions (Chapter 4) have been examined for particle size distribution (PSD), settleability and dewaterability as described in **Chapter 5.** The course of particle size alteration of the digested sludge was assessed with wet sieve analysis (0.100, 0.125, 0.200, 0.500 and 1.000 mm). The results reveal a substantial reduction in all particle sizes with improving digestion conditions, except particles <0.100 mm at 10 days and 25 ºC. Digestion leads bigger particles to transfer into smaller ones, which have a strong effect on the sludge physical behaviour (Lawler *et al.*, 1986). The majority of the raw and digested particles are smaller than 0.100 mm. The dewatering results of both sets of reactors showed the existence of optimal SRT for dewaterability at 20 and 15 days for the reactors operated at 25 and 35 ºC, respectively. The dewaterability of sludge digested at less favourable conditions of 10 days at 25 ºC deteriorates due to bigger particles transfer into smaller particles that accumulate in the size range diam.<0.100 mm. Moreover, the results revealed that sludge settling is not affected by digestion. In conclusion, from physical point of view, the digester of the UASB-Digester system should be operated at the conditions, which are optimal for dewaterability not settleability, as the latter is hardly affected by digestion.

ANAEROBIC SEWAGE TREATMENT IN A ONE STAGE UASB AND A UASB-DIGESTER SYSTEM

Chapter 6 describes research results demonstrating the technical viability of the UASB-Digester system for sewage treatment at low temperature. The experimental work has been carried out over two successive periods. Firstly, a pilot -scale of a one stage flocculent sludge UASB reactor (volume, height, diameter: 140 l, 325 cm, 23.5 cm) was operated at 15 °C without incorporating a digester. Afterwards, the UASB was modified to the UASB-Digester system by incorporating a CSTR digester (working volume 106 l) operated at 35 °C. The UASB reactor was operated at an HRT of 6 hours and a temperature of 15 °C, the average sewage temperature in Middle East countries during wintertime.

The UASB-Digester provided substantially better removal efficiencies than the one stage UASB reactor ($p<0.05$). The achieved removal efficiencies in the UASB-Digester and the one stage UASB for total COD, suspended COD, colloidal COD and dissolved COD are "66, 87, 44 and 30" and "44, 73, 3 and 5"% for both systems, respectively.

In the one stage UASB reactor methanogenesis was limiting the overall conversion of organic matter to methane. The effluent contained a high amount of soluble COD of 162(47) mg COD/l of which 80(40) mg COD/l was in the form of VFA. Differently, in the UASB of the UASB- Digester system, hydrolysis was the limiting step of the overall digestion processes. The conversion in the UASB of the UASB-Digester system is substantially higher than in the one stage UASB. The percentage methanogenesis of the influent COD in the UASB, the UASB of the UASB-digester and the overall UASB-Digester system were 21, 44 and 47%, respectively.

The results show that the sludge in the UASB of the UASB-Digester system is substantially more stable as compared to the sludge in the one stage UASB reactor. Also, the digester sludge has substantially higher stability than the excess sludge from the UASB of the UASB-Digester system. The settleability of the sludge of the UASB, UASB of the UASB-Digester system and the digester sludges reveals a high settleability with no influence of the digestion conditions. The high filterability constant of the wasted sludge of the UASB of the UASB-Digester system as compared with that of the UASB reactor demonstrates that the first sludge is better dewaterable ($P<0.05$) but the digested sludge is not ($P<0.05$). The daily wasted sludge results reveal a significantly lower sludge production in the UASB-Digester as compared to the UASB system, viz. 0.77 and 0.21 g COD/g COD removed.

The removal efficiency of total COD in the UASB-Digester system is as high as those reported for tropical countries like Sao Paulo-Brazil by (Vieira, 1988) (70%), Bucaramanga - Columbia by (Schellinkhout *et al.*, 1988) (66 %) and Kanpur - India by (Draaijer *et al.*, 1992) (62 - 70 %).

FINAL DISCUSSION

The results of the research presented in this thesis, clearly show that the incorporation of a sludge digester in an anaerobic sewage treatment under conditions of low temperatures in an UASB reactor will substantially improve both the physical removal and biological conversion of the different COD fractions (Chapter 6). The observed improved physical removal and biological conversion supports our hypothesis in Chapter 2, based on literature search, viz. 'The higher the solids conversion rate, the higher will become the removal as a result of reduction of de-capturing opportunity of already captured solids in the sludge bed'.

The integrated UASB-digester reactor concept can be further optimized, especially with respect to size of the digester. The results presented in Chapters 4 and 5 show that the optimum conditions for anaerobic conversions and improving the dewaterability of the sludge in the digester are a digester temperature of 35 ºC and a sludge retention time of 15 days. In our experimental set-up, sludge from the top of the sludge bed in the UASB-reactor, which is relatively low in SS-content, was recirculated over the sludge bed. As the sludge profile in terms of VS/TS ratios do not reveal a clear change in stability, recirculation of sludge with the highest concentration, consequently from the bottom of the UASB would result in a considerable decrease in digester volume, viz. by approximately 50 %. This would mean that at an HRT in the UASB of 6 hours, consequently a needed UASB reactor volume of 0.25 m^3 reactor volume per 1 m^3 sewage to be treated per day, the required additional sludge digester volume would be 0.071 m^3 reactor volume per m^3/day sewage treated.

The research presented in this thesis has been conducted in order to assess the feasibility of anaerobic pre-treatment of domestic sewage under Middle East conditions, with special emphasis to the conditions in Palestine. The results in Chapter 3 show that the strength and consequently the concentration of the SS-fraction of the domestic sewage in Palestine is considerably higher as compared to that produced in the Netherlands. The UASB-digester system therefore should be optimized for those at particular conditions. The reactor volume (consequently the applicable HRT) of the UASB is mainly dictated by the amount of methanogenic activity that can be retained in the system. The methanogenic activity of the system should suffice to convert the dissolved COD, entering and produced in the system to methane gas. The results presented in Chapter 6 show that enough methanogenic capacity is retained to cope with an increased dissolved COD concentration, provided an sufficient amount of sludge is circulated via the digester in order to achieve here the required amount of hydrolysis, acidification and methanogenesis. Additionally, an increase of the volume of the digester might be necessary. Results of preliminary model calculations indicated, that when the sludge concentration entering the digester can be elevated from 10 to 22 gVS/l, a volume of the digester of 0.107 m^3 suffices for each m^3 of sewage to be treated per day under Palestinian conditions when operating the UASB reactor at 6 hours HRT. The sludge recirculation ratio should be increased 3.62 times of the sludge flow rate that should be applied at operating the digester at 15 days HRT, to convey the sludge from the UASB to the digester, i.e. 3.62 x 7.13 l/d. This implies a reduction in the total reactor volume (UASB+Digester) corresponding to 8.6 hours HRT as compared to 24 hours HRT proposed by Halalsheh (PhD-thesis in preparation) when applying a conventional UASB-reactor under Middle East conditions. Moreover, the effluent quality of the system will be considerably better, especially during the winter situation. Apart from the benefits of lower vulnerability and higher sustainability and self-sufficiently, this implies a very significant reduction in the costs. In addition, the water is greatly retained at the residential site enabling the residents to re-use it, e.g. for agricultural purposes. For the Palestinian people this at least comprises one mean to survive in their extraordinary difficult situation.

CONCLUSIONS

- The application of the one stage UASB reactor for sewage treatment in Palestine is only possible if designed at a long HRT, i.e. more than 22 hours.

- During anaerobic digestion of primary sludge in CSTR, the main digestion of proteins, carbohydrates and lipids occurs within the first 15 and 10 days SRT at process temperatures of 25 and 35 ºC, respectively.

- The UASB-Digester system is a technically viable technology for sewage (pre) treatment at 15 ºC.

- The enhancement of methanogenic conditions in UASB reactor, improves both solids hydrolysis and physical removal.

- Anaerobic digestion at full methanogenic conditions in CSTRs, improve dewaterability up to optima. The optima were noticed at 20 and 15 days SRT at process temperatures of 25 and 35 ºC, respectively.

RECOMMENDATIONS

- The performance of the UASB-Digester system under the environmental conditions of the Middle East countries still needs to be demonstrated along with relevant research at pilot scale. The technology needs to be optimised in terms of digestion and compactness. Afterwards, the technology can be scaled-up including biogas utilisation for heating the digester.

- The feasibility of the UASB-Digester technology should be investigated at lower temperature, e.g. 10 ºC for application in colder regions.

- Integrated research directed at stimulation of sanitation paradigms reform towards the recognition of water saving appliances and maximisation of by products reuse are needed.

- More research is needed into the interaction between sludge physical-chemical characteristics, digestion conditions and sludge physical behavior.

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Hoofdstuk 7

ANAËROBE ZUIVERING VAN AFVALWATER IN EEN UASB-VERGISTER SYSTEEM

SAMENVATTING, DISCUSSIE, CONCLUSIES EN AANBEVELINGEN

ANAËROBE ZUIVERING VAN AFVALWATER IN EEN UASB-VERGISTER SYSTEEM

SAMENVATTING, DISCUSSIE, CONCLUSIES EN AANBEVELINGEN

INLEIDING

Anaërobe vergisting wordt wereldwijd erkend als de basis voor duurzame afvalbehandeling (Zeeman en Lettinga, 1999; Hammes *et al*., 2000). Het meest verbreide, en succesvol toegepaste anaërobe systeem voor afvalwaterbehandeling is de opwaarts doorstroomde anaërobe slibbed (UASB) reactor (Agrawal *et al.*, 1997; Monroy *et al.*, 2000; Lettinga, 2001). Deze reactor is toegepast voor de behandeling van afvalwater in tropische landen als India, Colombia and Brazilië, (Haandel en Lettinga, 1994). De temperatuur in deze landen is vrij hoog en stabiel gedurende het jaar met waarden tussen 20 en 30 ^{o}C. Bovendien is het afvalwater niet erg geconcentreerd. Toepassing van UASB reactoren in regio's met een kouder klimaat vereist meer onderzoek (Lettinga en Hulshoff Pol, 1991; Wang, 1994; Sayed en Fergala, 1995; Lettinga, 1996; Kalogo en Verstraete, 19999; Elmitwalli, 2000). De prestaties van een-traps UASB systemen bij lagere temperaturen (5-20 ^{o}C) wordt sterk gelimiteerd door de hydrolyse van ingevangen vaste bestanddelen omdat deze in het slibbed accumuleren wanneer hoge belastingen worden toegepast (Man, 1990; Lettinga *et al.*, 1993; Zeeman en Lettinga, 1999). Hierdoor neemt de hoeveelheid surplusslib toe en daalt de slibleeftijd waardoor de groei van methanogenen kan worden gelimiteerd (Haandel en Lettinga, 1994). Dit resulteert vervolgens in een slechte verwijdering van opgelost CZV en een verminderde slibstabiliteit (Wang, 1994).

Voor een stabiele procesvoering bij lage temperaturen zijn twee-traps systemen voorgesteld die bestaan uit een hoogbelaste eerste trap, gevolgd door een methanogene trap (korrelslib) (Wang, 1994; Elmitwalli, 2000). Echter, het slib dat in hoogbelaste reactoren wordt geproduceerd is per definitie ongestabiliseerd en moet in een aparte vergister verder worden behandeld. Lettinga en Hulshoff Pol (1991) stelden een nieuwe technologie voor die bestaat uit een combinatie van een hoogbelaste UASB en een vergister voor de behandeling van hoge concentraties gesuspendeerde stoffen bij lage temperaturen. Het probleem van afvalwaterbehandeling bij lage temperaturen neemt namelijk toe bij hogere concentraties gesuspendeerde stoffen in het afvalwater. Dit is bijvoorbeeld het geval in verschillende landen in het Midden-Oosten, waaronder Palestina en Jordanië.

Dit proefschrift beschrijft experimenteel onderzoek naar de mogelijkheden van anaërobe zuivering van afvalwater in het Midden-Oosten.

DEELTJESVERWIJDERING IN OPWAARTS DOORSTROOMDE REACTOREN

De bureaustudie die in **hoofdstuk 2** wordt beschreven behandelt de mechanismen en parameters die de afscheiding van deeltjes in opwaarts doorstroomde reactoren beïnvloeden. Hoewel het functioneren van UASB systemen zowel door fysische als biologische processen wordt bepaald, komen de fysische parameters in de beschikbare literatuur nauwelijks aan bod. De reden hiervoor is waarschijnlijk dat de onderliggende mechanismen erg complex zijn en afhangen van meerdere samenhangende parameters. Toch is een beter begrip van deze samenhang essentieel voor de verdere ontwikkeling van anaërobe zuiveringstechnologie. In deze studie werden de verschillende parameters die van invloed op de deeltjesscheiding verder uitgewerkt. Deze parameters werden onderverdeeld in (1) operationele parameters (temperatuur, organische belasting, hydraulische verblijftijd en opstroomsnelheid), (2) eigenschappen van het te behandelen afvalwater (concentraties, deeltjesgrootte en lading van de deeltjes), (3) eigenschappen van het slibbed (deelthesgrootteverdeling, exopolymeren en lading). Deze studie heeft als resultaat (1) een literatuuroverzicht, (2) een structureel beeld van de kennis die op dit gebied beschikbaar is en (3) kennisleemtes die om nadere invulling vragen.

AFVALWATERBEHANDELING IN PALESTINA

Hoofdstuk 3 behandelt de resultaten van experimenteel en literatuuronderzoek naar de eigenschapen van het afvalwater en de afvalwaterbehandeling in Palestina. Omdat de eigenschappen van het afvalwater bepalen of anaërobe zuiveringstechnologie kan worden ingezet (Lettinga *et al.*, 1993), was het noodzakelijk meer kennis op dit gebied te vergaren zodat zuiveringsinstallaties op gedegen wijze kunnen worden geselecteerd en gedimensioneerd. Er werden mengmonsters genomen op drie locaties in het district Ramallah/Al-Bireh op de Westelijke Jordaanoever. Daar werden eerdere resultaten aan toegevoegd. Volgens classificaties, voorgesteld door Metcalf en Eddy (1991) en door Henze (1997), kan het afvalwater in Palestina in vergelijking met andere landen sterk geconcentreerd worden genoemd. De hoge concentraties kunnen worden toegeschreven aan een laag waterverbruik, industriële lozingen en een hoge CZV-productie door de inwoners (g CZV/i.e/dag). De resultaten werden gebruikt in modelberekeningen (Zeeman en Lettinga, 1999) om te beoordelen of in Palestina een-traps UASB reactor een toepasbaar systeem zou zijn. Hieruit blijkt dat dit alleen mogelijk is indien een hydraulische verblijftijd van meer dan 22 uur wordt toegepast om tijdens de wintermaanden problemen te voorkomen als gevolg van de hoge concentraties aan vaste bestanddelen in het afvalwater. Als alternatief wordt een nieuwe technologie aanbevolen, voorgesteld door Lettinga en Hulshoff Pol (1991), die bestaat uit een combinatie van een UASB en een slibvergister. Uitgaande van de economische situatie en de geografische en demografische verspreiding in Palestina, wordt tevens aanbevolen om gedecentraliseerde anaërobe technologie toe te passen. Verschillende zuiveringsconcepten worden voorgesteld volgens de duurzaamheidscriteria opgesteld door Lettinga *et al.* (2001).

ANAËROBE STABILISATIE EN OMZETTING VAN BIOPOLYMEREN IN PRIMAIR SLIB

Hoofdstuk 4 beschrijft het ontwerp en de optimalisatie van de vergister van het gecombineerde UASB-vergister systeem in relatie tot de slibstabilisatie. Het UASB-vergister systeem werd vereenvoudigd door de UASB weg te laten en een aantal volledig gemengde reactoren te voeden met primair slib. Het effect van de slibleeftijd en de temperatuur op de hydrolyse, verzuring en methanogenese van het primaire slib werden onderzocht. De reactoren werden bedreven bij slibleeftijden van 10, 15, 20 en 30 dagen en bij temperaturen van 25 en 30 ^{o}C. Tevens werd een reactor bedreven bij een slibleeftijd van 75 dagen en 15 ^{o}C, de temperatuur van het afvalwater in Palestina in de winter. De hydrolysesnelheid en biodegradeerbaarheid van het primaire slib werd bepaald in batchreactoren bij 5 en 25 ^{o}C, de gemiddelde minimum en maximum temperatuur in Palestina en bij 35 ^{o}C, de optimale temperatuur voor methanogenen in het mesofiele gebied. Uit de resultaten bleek dat de optimale slibvergisting bij 35 en 25 ^{o}C respectievelijk optreedt bij een slibleeftijd van meer dan 10 en meer dan 15 dagen. Hydrolyse was de snelheidsbeperkende stap in het vergistingsproces bij 35 en 25 ^{o}C. Alleen in de reactor bij een slibleeftijd van 10 dagen en een temperatuur van 25 ^{o}C was de verzuring van hogere vetzuren de snelheidsbeperkende stap; de hydrolyse van eiwitten en particulaire koolhydraten suikers was snelheidsbeperkend voor de verzurings terwijl de methanogenese de snelheidbeperkende stap was voor het gehele vergistingsproces. De hydrolyse van eiwitten was slechts 39% bij een slibleeftijd van 30 dagen en een temperatuur van 35 ^{o}C omdat de eiwitten vooral in de vorm van biomassa aanwezig waren. Methanogenese trad op bij een slibleeftijd van 75 dagen en een temperatuur van 15 ^{o}C. De biodegradeerbaarheid van het primaire slib was ongeveer 60% en bleek niet van de temperatuur afhankelijk te zijn. De hydrolyse van de belangrijkste biopolymeren en deeltjes CZV kon met een eerste-orde kinetiek beschreven worden, indien hydrolyse de snelheidsbeperkende stap was. Ook de hydrolyse van de deeltjes CZV in de batchexperimenten kon met een eerste-orde kinetiek beschreven worden. De snelheid was sterk afhankelijk van de temperatuur volgens de Arrhenius vergelijking.

INTERACTIE TUSSEN VERGISTINGSCONDITIES EN FYSISCHE SLIBEIGENSCHAPPEN EN GEDRAG BIJ ANAËROBE VERGISTING VAN PRIMAIR SLIB

De resultaten van de bureaustudie van hoofdstuk 2 laten zien dat de vergistingscondities een belangrijk effect hebben op de fysisch-chemische eigenschappen van het slib, en daarmee op het fysische gedrag van het slib. Verschillende slibsoorten die onder verschillende condities in volledig gemengde reactoren werden vergist (hoofdstuk 4) werden in **hoofdstuk 5** onderzocht op hun deeltjesgrootteverdeling, bezinking en ontwaterbaarheid. De deeltjesgrootteverdeling van het vergistte slib werd bepaald met zeven van 0.100, 0.125, 0.200, 0.500 en 1.000 mm. De resultaten laten zien de deeltjesgrootte aanzienlijk kleiner wordt naarmate de vergistingscondities verbeteren, behalve voor deeltjes <0.100 mm bij een slibleeftijd van 10 dagen en een temperatuur van 25 ^oC. De kleinere deeltjes die door vergisting ontstaan hebben een opvallend effect op de fysische eigenschappen van het slib (Lawler *et al.*, 1986). Het grootste deel van de ruwe en vergistte deeltjes is kleiner dan 0.100 mm. De ontwaterbaarheid van beide series reactoren liet zien dat de optimale slibleeftijd met betrekking tot ontwaterbaarheid gelijk is aan 20 dagen voor een temperatuur van 25 ^oC en 15 dagen voor een temperatuur van 35 ^oC. De ontwaterbaarheid van slib dat onder minder gunstige omstandigheden wordt vergist, dat wil zeggen bij een slibleeftijd van 10 dagen en een temperatuur van 25 ^oC, neemt af doordat accumulatie van kleinere deeltjes (< 0.100 mm) optreedt. De bezinking van het slib veranderde niet door vergisting. Geconcludeerd kan worden dat, vanuit fysisch oogpunt, een UASB-vergister systeem bedreven zou moeten worden onder condities die leiden tot een optimale ontwatering van het slib.

ANAËROBE ZUIVERING VAN AFVALWATER IN EEN EEN-TRAPS UASB EN IN EEN UASB-VERGISTER SYSTEEM

Hoofdstuk 6 beschrijft resultaten die de technische haalbaarheid van een UASB-vergister systeem aantonen bij lage temperaturen. De experimenten werden uitgevoerd in twee opeenvolgende periodes. In eerste instantie werd een UASB reactor (volume 140 l, hoogte 325 cm en diameter 23.5 cm) met vlokkig slib bedreven bij 15 ^oC zonder daar een vergister aan te koppelen. Daarna werd de UASB gecombineerd met een volledig gemengde vergister van 106 l die bij 35 ^oC werd bedreven. De hydraulische verblijftijd van de UASB was 6 uur en de temperatuur was 15 ^oC wat overeenkomt met de gemiddelde wintertemperatuur in het Midden-Oosten.

Het UASB-vergister systeem gaf een veel betere verwijdering dan de een-traps UASB reactor ($p<0.05$). De verwijdering van totaal CZV, gesuspendeerd CZV, colloïdaal CZV en opgelost CZV was respectievelijk 66, 87, 44 en 30% in het gecombineerde systeem en 44, 73, 3 en 5% in de een-traps UASB.

In de een-traps UASB was methanogenese de snelheidsbeperkende stap voor de volledige omzetting van organische stof naar methaan. Het effluent bevatte een hoge concentratie opgelost CZV van 162 mg CZV/l waarvan 80(40) mg CZV/l als vluchtige vetzuren aanwezig was. In het gecombineerde UASB-vergister systeem was hydrolyse de snelheidsbeperkende stap. Tevens bleek dat de omzetting in de UASB van dit systeem veel hoger was dan in de een-traps UASB. De percentages methanogenese in de een-traps UASB, de UASB van het gecombineerde systeem en het volledige UASB-vergister systeem waren respectievelijk 21, 44 en 47%.

De resultaten tonen aan dat het slib in de UASB van het gecombineerde systeem veel stabieler is dan het slib in de een-traps UASB. Ook is het slib uit de vergister van het gecombineerde systeem veel stabieler dan het slib uit de UASB systemen. De bezinking van het slib uit de een-traps UASB, de gecombineerde UASB en uit de vergister is goed en de vergistingscondities hebben geen effect op de bezinking. De hoge filtratieconstante van het spuislib uit de UASB van het gecombineerde systeem laat zien dat dit slib beter ontwaterbaar is dan het slib van de een-traps UASB ($p<0.05$). Echter, het slib uit de vergister is slechtere ontwaterbaar dan het slib uit de een-traps UASB ($p<0.05$). De dagelijkse hoeveelheid spuislib laat zien dat in het gecombineerde systeem veel minder slib (0.21 g CZV per g verwijderde CZV) wordt geproduceerd dan in het een-traps systeem (0.77 g CZV per g verwijderde CZV).

De verwijderingsefficiëntie voor totaal CZV in het gecombineerde UASB-vergister systeem is vergelijkbaar met waarden die in tropische landen bereikt kunnen worden, zoals 70% in Sao-Paulo in Brazilië (Viera, 1988), 66% in Bucaramanga in Colombia (Schellinkhout *et al.*, 1988) en 62-70% in Kanpur in India (Draaijer *et al.*, 1992).

SLOTDISCUSSIE

De resultaten die in dit proefschrift worden beschreven laten duidelijk zien dat bij lagere temperaturen uitbreiding van een UASB reactor met een slibvergister een verbetering geeft van de fysische verwijdering en de biologische omzetting van de verschillende CZV fractie in huishoudelijk afvalwater (hoofdstuk 6). Dit ondersteunt de hypothese, die was gebaseerd op het literatuuronderzoek van hoofdstuk 2, dat door een verbeterde biologische omzetting de kans vermindert dat deeltje die in het slibbed zijn ingevangen weer vrijkomen en met het effluent verdwijnen.

Het gecombineerde UASB-vergister systeem kan nog verder worden geoptimaliseerd, in het bijzonder de grootte van de vergister. De resultaten van hoofdstuk 4 en 5 tonen aan dat bij een temperatuur van 35 ^{o}C en een slibleeftijd van 15 dagen de condities voor anaërobe omzetting en ontwatering van het slib optimaal zijn. In de experimentele opzet werd het slib bovenin het slibbed van de UASB over de vergister gerecirculeerd. Echter, dit slib is veel minder geconcentreerd dan het slib onderin de UASB. Recirculatie van dit meer geconcentreerde slib zou een aanzienlijke reductie (~50%) van het benodigde volume van de vergister kunnen opleveren. Dit laatste betekent dat voor een hydraulische verblijftijd in de UASB van 6 uur, oftewel een reactorvolume van 0.25 m^3 per m^3 te behandelen afvalwater per dag, een extra vergistingsvolume van 0.071 m^3 per m^3 afvalwater per dag nodig is.

Het onderzoek dat in dit proefschrift wordt beschreven werd uitgevoerd om de toepassing van anaërobe zuivering van huishoudelijk afvalwater in het Midden-Oosten te bestuderen, met speciale aandacht voor de situatie in Palestina. De resultaten van hoofdstuk 3 laten zien dat het afvalwater in Palestina veel geconcentreerder is dan in Nederland, vooral wat betrfet de concentratie aan gesuspendeerde stoffen. Het gecombineerde UASB-vergister systeem dient dan ook op de condities in Palestina te worden aangepast. Het volume van de UASB wordt vooral bepaald door de methanogene activiteit die in deze reactor kan worden gehandhaafd. Deze activiteit moet voldoende zijn om het opgelost CZV, dat met het influent wordt aangevoerd, volledig in methaan om te zetten. Uit de resultaten van hoofdstuk 6 blijkt dat genoeg methanogene activiteit beschikbaar is om ook verhoogde concentraties opgelost CZV te kunnen behandelen. Echter, in dat geval moet meer slib over de vergister worden gerecirculeerd zodat de hydrolyse, verzuring en methanogenese in de vergister worden gestimuleerd. Daarnaast moet mogelijk een wat grotere vergister worden toegepast. Verkennende modelberekeningen duiden erop dat, indien de slibconcentratie in de vergister kan worden berhoogd van 10 tot 22 g VS/l, 0.107 m^3 reactorvolume nodig is per m^3 afvalwater voor een UASB die werkt bij een hydraulische verblijftijd van 6 hr. De recirculatiefactor moet dan met een factor 3.62 worden verhoogd ten opzichte van de factor die minimaal nodig is om het slibbed in de UASB op een constant niveau te houden, oftwel driemaal 7.13 l/d. Dit komt erop neer dat de hydraulische verblijftijd van het gecombineerde UASB-vergister systeem tot 8.6 uur kan worden teruggebracht in vergelijking met een verblijftijd van 24 uur voor een conventioneel systeem zoals door Halalsheh (in bewerking) is voorgesteld voor het Midden-Oosten. De zuiveringskosten zullen dan ook veel lager uitvallen. Bovendien zal de effluentkwaliteit van het voorgestelde systeem vooral in de winter veel beter zijn, de gevoeligheid van het systeem lager zijn en de duurzaamheid verbeteren. Een ander belangrijk voordeel is dat het water ter plekke aanwezig blijft zodat het beschikbaar komt voor hergebruik, bijvoorbeeld voor irrigatie. Voor het Palestijnse volk is dit één van de mogelijke manieren om onder zeer moelijke omstandigheden te overleven.

CONCLUSIES

- De toepassing van een een-traps UASB reactor voor de zuivering van afvalwater in Palestina is alleen mogelijk bij een zeer lange hydraulische verblijftijd.

- Tijdens de anaërobe vergisting van primair slib in een volledig gemengde reactor treedt het grootste deel van de vergisting van eiwitten, koolhydraten en vetten op binnen de eerste 15 dagen bij een temperatuur van 25 ^oC en binnen de eerste 10 dagen bij een temperatuur van 35 ^oC.

- Het UASB-vergister systeem is technisch haalbaar voor de (voor)behandeling van afvalwater bij 15 ^{o}C.

- Methanogene condities in een UASB reactor bevordert zowel de fysische als hydrolytische deeltjesverwijdering.

- Anaërobe vergisting onder volledige methanogene condities in volledig gemengde systemen bevordert de ontwaterbaarheid, tot een bepaald optimum is bereikt. Dit optimum werd vastgesteld bij een slibleeftijd van 20 dagen voor een temperatuur van 25 ^{o}C, en bij een slibleeftijd van 15 dagen voor een temperatuur van 35 ^{o}C.

AANBEVELINGEN

- De prestaties van een UASB-vergister systeem onder de omgevingscondities van Midden-Oosten landen moeten ter plekke nog worden aangetoond. Daarnaast is nog relevant onderzoek nodig op proefinstallatie- en praktijkschaal. De technologie moet verder worden geoptimaliseerd wat betreft de vergisting en de compactheid van het systeem. Daarna kan het systeem worden opgeschaald waarbij het biogas kan worden gebruikt voor verwarming van de vergister.

- De haalbaarheid van het UASB-vergister systeem zou ook bij nog lagere temperaturen, bijvoorbeeld 10 ^{o}C, onderzocht moeten worden.

- Er is behoefte aan geïntegreerd onderzoek, gericht op het veranderen van bestaande sanitatieconcepten, zodat waterbesparing en hergebruik worden gestimuleerd.

- Meer onderzoek is nodig naar de interactie tussen fysisch-chemische slibeigenschappen, vergistingscondities en het fysische gedrag van het slib.

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List of abbreviations

AF:	anaerobic filter
AH :	anaerobic hybrid
AVR:	average
BD:	biodegradability (%)
C:	capita
Carb:	carbohydrate
COD:	chemical oxygen demand (kg COD/m^3)
CST:	capillary suction time (s)
CSTR:	completely stirred tank reactor
diam.:	diameter (mm)
E:	activation energy (kJ.$mole^{-1}$)
EGSB:	expanded granular sludge bed
EPS:	exopolymeric substances
F/M:	food to microorganisms ratio
FB:	fluidised bed
G:	velocity gradient (s^{-1})
GLS:	gas-liquid-solid three phase separator
HRT:	hydraulic retention time (d)
HUSB:	hydrolysis upflow sludge bed
k_h:	hydrolysis rate constant (d^{-1})
LCFA:	long chain fatty acids (g COD/l)
Meth.:	methanogens or methanogenic
OLR:	organic loading rate (kg COD/m^3.d)
P:	probability
PSD:	particle size distribution
Q:	flow rate (m^3/d)
rpm:	revolutions per minute
SLR:	solids loading rate (gSS/l.d)
SMPA:	staged multi-phase anaerobic reactor
SRT:	sludge retention time
SS:	suspended solids (g/l)
STD:	standard deviation
SVI:	sludge volume index (ml/g)
T:	the absolute temperature, ºK
t:	time (d)
TS:	total solids (g/l)
TSS:	total suspended solids (g/l)
UASB:	upflow anaerobic sludge blanket
V:	volume (m^3)
VFA:	volatile fatty acids (g COD/l)
Vs:	settling velocity (m/s)
VS:	volatile solids (g/l)
VSS:	volatile suspended solids
Vup:	upflow velocity (m/h or m/d)
WWTP:	wastewater treatment plant
X:	filterability constant ($kg^2.m^4.s^{-2}$)

Greek

ρ:	density of water (kg/m^3)
μ:	dynamic viscosity (Pa.s)
ϕ:	filterability constant of the CST apparatus (dimensionless)
ρ_s:	density of particle (kg/m^3)
μm:	micrometer

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Curriculum vitae

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