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High-Rate Anaerobic Waste-Water Treatment Using the UASB Reactor under a Wide Range of Temperature Conditions

G. LETTINGA, L.W. HULSHOFF POL, I.W. KOSTER,
W.M. WIEGANT, W.J. DE ZEEUW, A. RINZEMA, P.C. GRIN,
R.E. ROERSMA AND S.W. HOBMA

Department of Water Pollution Control, Agricultural University, De Dreyen 12, 6703 BC Wageningen, The Netherlands

Introduction

There is a world-wide and increasing interest in anaerobic waste-water treatment (AWWT) and in anaerobic digestion for energy-production purposes. This recent development is attributable to the fact that the method combines a number of significant advantages (with few, if any, insuperable drawbacks) over conventional aerobic methods of the activated sludge type for water treatment. Benefits and drawbacks of AWWT are listed in *Table 1*.

AVAILABLE METHODS

The main reason for interest in developments in AWWT presumably has resulted from the development of simple and low-cost 'high-rate' anaerobic treatment processes able to deal with diluted as well as more concentrated effluents, and in the successful application of some of these methods, such as the anaerobic filter (AF) and the Upflow Anaerobic Sludge Blanket (UASB) process.

The loading rates of an AWWT system are primarily dictated by:

1. The amount of viable sludge which can be retained in the anaerobic reactor;
2. The contact that can be achieved between the retained sludge and the incoming waste water.

Abbreviations: AAFEB, anaerobic attached-film expanded bed; AF, anaerobic filter; AFF, anaerobic fixed film; AFFEB, anaerobic fixed-film expanded bed; AWWT, anaerobic waste-water treatment; COD, chemical oxygen demand; DSS, dry suspended solids; GSS, gas–solids separator; HRT, hydraulic retention time; RT, retention time; SS, suspended solids; STP, standard temperature and pressure; SVI, sludge volume index; UASB, upflow anaerobic sludge blanket; VFA, volatile fatty acids; VS, volatile solids; VSS, volatile suspended solids.

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Table 1. Benefits and drawbacks of anaerobic waste-water treatment over conventional aerobic methods

Benefits:	1.	Low production of (stabilized) excess sludge.
	2.	Low nutrient requirements.
	3.	No energy requirements for aeration.
	4.	Production of methane.
	5.	The process can frequently handle high space loads.
	6.	Anaerobic sludge can be preserved, unfed, for many months without any serious deterioration.
	7.	Valuable compounds like ammonia are conserved, which in specific cases might represent an important benefit (i.e. if irrigation can be applied).
Drawbacks:	1.	Anaerobic bacteria (particularly the methanogens) are very susceptible to inhibition by a large number of compounds.
	2.	The start-up of the process is slow if an adapted seed sludge is not available.
	3.	Anaerobic treatment normally demands an adequate post-treatment for the removal of remaining BOD, ammonia and malodorous compounds.
	4.	Little practical experience exists as yet, although the situation is rapidly changing in this respect.

Unlike aerobic processes, the loading rate of the process generally is not limited by the supply of any required reagent/ingredient. The more sludge that can be retained, the higher the loading rates that can be applied, provided that sufficient contact between the sludge and the waste water can be maintained.

The high retention of active sludge in the modern high-rate AWWT processes basically depends on:

1. *Bacterial sludge entrapment* in the interstices between support material present in the reactor and bacterial attachment on the external surface of this packing material, i.e. the well-known Anaerobic Filter (AF) process (e.g. Coulter, Soneda and Ettinger, 1957; Young and McCarty, 1969; Lettinga, Fohr and Janssen, 1972; Colleran *et al.*, 1982; Young, 1982, and many others);
2. *Bacterial immobilization by an attachment mechanism* to fixed surfaces, viz. the Anaerobic Stationary Fixed Film reactor (AFF) (van den Berg and Kennedy, 1981, 1983) or to mobile particulate surfaces, such as the Anaerobic Attached Film Expanded Bed (AAFEB) process which is also known as the Anaerobic Fixed Film Expanded Bed (AFFEB) (Switzenbaum and Jewell, 1978; Jewell, 1979; Schraa and Jewell, 1983) and fluidized-bed processes (Li and Sutton, 1981; Binot *et al.*, 1982; Heynen, 1983);
3. *Sludge blankets* such as the Upflow Anaerobic Sludge Blanket (UASB) process (Lettinga *et al.*, 1974, 1979a,b, 1980, 1981, 1983a,b; Lettinga, 1978; Lettinga and Vinken, 1980; Lettinga, Grin and Roersma, 1983) and the reversed flow Dorr Oliver Clarigester (Hemens, Meiring and Stander, 1962; Stander *et al.*, 1967). The latter process could be indicated as a precursor of the UASB process, although it in fact simply was applied for liquid waste-water treatment because existing clarigesters were available for that purpose.

In spite of the considerable progress made in the last two decades in the field of process technology of anaerobic treatment, only just over 60 high-rate anaerobic treatment processes have been installed in recent years (Speece, 1983). Considerably more digesters have been built for treating high-strength slurries such as manure, but for the treatment of these types of wastes in fact there is no cheap alternative such as exists for liquid low- and medium-strength wastes. Despite the fact that the feasibility of various modern high-rate processes has already been shown in large pilot plants and/or full-scale plants, it still remains difficult to convince potential users and established consultancies of the great economic value of the anaerobic treatment alternative.

Each of the high-rate processes has its own merits and limitations, depending on the extent and the ease with which the primary conditions underlying these processes can be met and on operational requirements for proper performance.

Although it is correct to designate the modern processes as high-rate processes, in fact considerable differences exist in their maximum achievable loading potentials. This is because the maximum viable sludge hold-up under high loading conditions is very different from process to process, and this is also the case with respect to the amount of contact that can be achieved between the retained sludge and the incoming waste water. Moreover, kinetic factors such as film and/or particle diffusion limitation of substrate compounds and metabolic end-products, as well as the substrate dependency of the substrate utilization rate of the organisms, have an important role.

Figure 1 represents the currently most important AWWT processes (van den Berg and Kennedy, 1983).

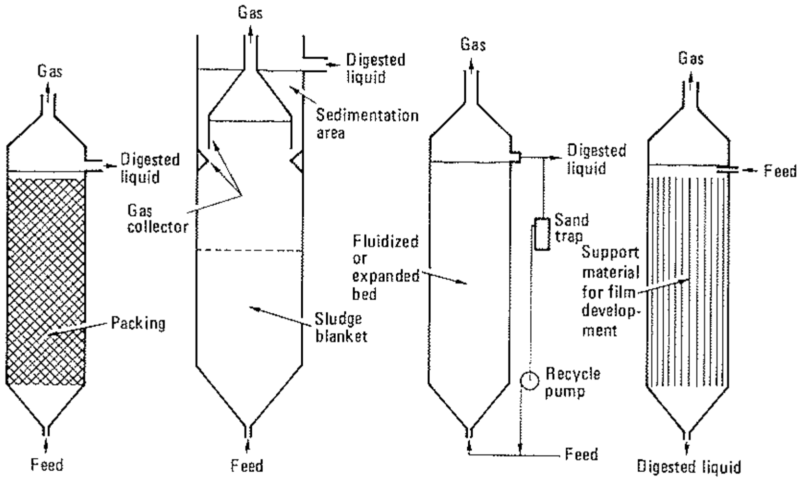

Figure 1. Diagrammatic representation of various modern high-rate waste-water treatment processes. From left to right: anaerobic filter; upflow anaerobic sludge blanket; fluidized or expanded bed; anaerobic fixed film. After Van den Berg and Kennedy (1983).

The anaerobic filter (AF) is characterized by the presence of stationary packing material in the reactor. The reactor generally is operated in an upflow mode. The type of packing material (stone, clay, plastic) seems to be less important than the shape, as an important fraction of the sludge retention is obtained by sludge entrapment in the void space between the packing material (Young and Dahab, 1982).

The anaerobic fixed film (AFF) process is also characterized by the presence of a stationary packing material in the reactor, but here the aim of the design is to avoid entrapment of suspended solids, thus making the process feasible for (pre)treatment of wastes containing a considerable amount of solids such as screened manure. In order to prevent accumulation of solids in the reactor the AFF process is mostly operated in a down-flow mode.

The anaerobic fixed film expanded bed (AFFEB) process and the fluidized-bed process are both characterized by the presence of mobile biomass-carrier material (e.g. sand, clay particles). Both processes require effluent recycling in order to assure a sufficient flow rate to lift the carrier material plus the attached biomass. In an expanded bed the flow rate will enlarge the sludge bed, so a good contact between waste water and biomass is ensured. In a fluidized bed the flow rate is even higher. The main difference between these systems is that in an expanded bed the sludge is present in the lower part of the reactor, whereas in a fluidized bed the sludge is distributed over almost the entire reactor volume.

The upflow anaerobic sludge blanket (UASB) process is characterized by a reactor containing no packing or any other type of biomass support material. An important feature of this design is the gas–solids separator (GSS) which provides a quiescent zone at the top of the reactor, where suspended solids (active biomass) will settle. This leads to a transport of sludge back into the actual reactor. The sludge will show good settling properties after a start-up period, followed by granulation. This will result in the sludge blanket (or sludge bed) in the lower part of the reactor.

The main factors determining the hold-up of viable biomass of the various high-rate systems are listed in *Table 2*.

Other important features of the various high-rate AWWT systems, such as the rate of start-up, the capacity of the process to remove suspended solids, the risk of clogging, the need for effluent recycle, the installation of a sophisticated feed inlet distribution system, a gas–solids separator (GSS) and the use of packing materials, are listed in *Table 3*.

With regard to the start-up procedure, each AWWT system has its own demands, especially with respect to the amount of seed sludge and the applicable loading rates. However, all the systems have in common the necessity for optimal conditions for bacterial growth as no high loading rates can be accommodated if a sufficient amount of adapted sludge has not been formed. This means that the composition of each waste water should be examined for the presence of macro-nutrients (N, P and S) and trace elements (Fe, Co and Ni seem to be particularly important). Furthermore, the possible presence of toxic or inhibitory substances in the waste water should be monitored.

Temperature and pH are also of great importance. The temperature has a

Table 2. Positive and negative factors determining the active biomass retention of the various high-rate AWWT processes under very high loading conditions*

Factors	Upflow sludge bed reactors (UASB, 'IRIS', tower reactors)	Upflow Anaerobic Filters (AF)	Downflow Anaerobic Fixed-Film systems (AFF)	Anaerobic Fixed-Film Expanded-Bed processes (AFFEB)	Fluidized-bed systems
Re-dispersion of the sludge due to: high turbulence flotation	– – – (at very high sludge loads)	–			–(?)
Disintegration of the sludge aggregates	–(?) (possibly at very high loads)				
Detachment of the bio-film		–	– ?	– ?	– – – (at incomplete acidogenesis)
Sludge bed expansion	±				
Space occupied by the packing/ carrier material		–(– – –) (depending on the packing)	–(– – –) (depending on the packing)	– – – – (70–80%)	– – (10–20%)
Surface area of the packing/ carrier material		+	++++ (100 m^2/m^3)	+++ (100 m^2/m^3)	+++++ (2000–5000 m^2/m^3)
Bed expansion required				– – (10–20%)	– – – – (approx. 50%)
Film thickness		+	+++	+++++	++

* It is assumed that all primary conditions for the optimal application of the various processes are met.

Table 3. Important features of the various high-rate anaerobic waste-water treatment systems

Features	Upflow sludge bed reactors (UASB, 'IRIS', tower reactor)	Upflow Anaerobic Filters (AF)	Downflow Anaerobic Fixed-Film system (AFF)	Anaerobic Fixed-Film Expanded-Bed (AFFEB)	Fluidized-bed systems
Rate of start-up:					
first start-up	4–16 weeks	>3–4 weeks	>3–4 weeks	>3–4 weeks	Approx. 3–4 weeks
secondary start-up	0–2 days	0–2 days	A few days?	A few days?	Uncertain
Performance with respect to the removal and the stabilization of suspended solids (SS)	Satisfactory at low and moderate loading rates	Fairly good at low SS concn. and when the filter is not clogged	Very poor	Rather poor	Very poor
Risk of channelling	Small, unless a poor feed inlet distribution system was installed	Great at high SS concn. and in clogged filters	Small	Small	Almost non-existent
Extent of effluent recycle required	Generally not required	Generally not required	Slight	Moderate	High recycle factor generally required
Sophisticated feed inlet distribution system required	For low-strength wastes and with dense sludge beds	Presumably beneficial	Not	Necessary	Essential
Gas-Solids Separator device required	Yes, essential	Could be beneficial	Not required	Could be beneficial	Beneficial
Carrier packing required	Can be beneficial in specific cases	Essential	Essential	Essential	Essential
Height-area ratio	Can be fairly high for granular sludge beds	Moderate	Moderate	Moderate (?)	Very high

major effect on the bacterial growth rate and activity; 38°C is regarded as the optimum for mesophilic treatment. With respect to the pH it may be necessary to add a buffering agent to the influent to keep the pH at a desired level of 6·5 or above.

Under good conditions conversion capacities of 8–10 kg COD/(m^3.day) should be achievable within a period of about 3 months.

It has been shown (Lettinga and Stellema, 1974) that unfed adapted methanogenic sludge can be preserved surprisingly well. Once an active adapted population has been obtained, therefore, the restart of a reactor will in general proceed within a few weeks without any major problems.

In attached-film processes the maximum sludge retention depends mainly on the surface area for sludge attachment, the film thickness, the space occupied by the carrier material and the extent to which dispersed sludge aggregates are retained. As indicated in *Table 2* there are considerable differences between the various fixed-film processes.

The main limiting factor in sludge-bed reactors is the redispersion of the sludge due to the high turbulence and the increasing tendency for flotation at high sludge loads. Depending on the type of sludge present in the reactor there may also be distinct distintegration (break-up) of the sludge aggregates at high loading rates.

In anaerobic fixed-film processes (downflow and upflow) the voidage of the packing material is a factor of prime importance with regard to sludge retention.

With respect to the amount of contact, the upflow anaerobic filter system can suffer from clogging (channelling) problems. In UASB reactors, channelling problems occur only at low loading rates and when a poor feed-inlet distribution system has been installed in the reactor. As fluidized-bed systems require a sophisticated feed-inlet distribution system, a good contact between bacterial matter is guaranteed in such systems. On the other hand the fluidized-bed systems require a high recycle factor, which may result in a distinct drop in substrate utilization rate by the active biomass because of the relatively low substrate levels prevailing throughout the reactor.

In view of what has been mentioned above, the maximum achievable loading rates with mainly soluble wastes presumably will diminish in the sequence:

Granular sludge UASB >
Fluidized bed systems >
Fixed Film Expanded Bed (AFFEB) >
Flocculant sludge UASB, upflow Anaerobic Filter, downflow Fixed Film (AFF)

So far no comparable experiments have been made at relevant scales between the various high-rate processes, although such a study is in progress at the Waste Water Technology Centre, Burlington, Canada (Hall, 1981) for a UASB process, an AFF system, an AF and a small-scale fluidized-bed reactor. In recent review papers the potential of various high-rate anaerobic treatment systems has also been evaluated (Callander and Barford, 1983; Speece, 1983; van den Berg and Kennedy, 1983).

It should be recognized that in practice the primary conditions underlying the various high-rate systems cannot always be sufficiently met. Moreover, apart from the maximum loading potentials, there are other important selection criteria with respect to the treatment system to be chosen, i.e. the stability of the process, the time required for the first and the secondary start-up, land area required, the ability to remove and to stabilize suspended solids from the raw waste water, and the capital and running costs.

ANAEROBIC MICROBIAL DEGRADATION

In anaerobic digestion four metabolic groups of bacteria can be distinguished These are: (1) hydrolytic bacteria which break down polymers such as proteins and carbohydrates into their monomers; (2) fermentative bacteria which ferment these monomers to organic acids, alcohols, carbon dioxide, hydrogen and ammonia; (3) acetogenic bacteria which convert higher VFA and alcohols into acetic acid and hydrogen, and (4) methanogenic bacteria which utilize methanol, acetate and/or hydrogen and carbon dioxide to produce methane.

In treating complex insoluble wastes, phase separation might become an attractive solution, e.g. an anaerobic treatment system consisting of a combined solids separator–liquefier–acidifier as a first step and a methanogenic reactor for converting the liquefied solids to methane as a second step. Whether or not a two-step system is more attractive than a relatively simple (although low-rate) one-step system is still an open question. In this connection one should consider the following points:

1. The hydrolysis step generally is rate limiting in the overall conversion, particularly at lower temperatures;
2. The separation of dispersed solids remaining after the liquefaction step cannot always be accomplished by simple and inexpensive means;
3. It is questionable whether or not the remaining solids will be sufficiently stabilized. If not, an additional sludge digester may be necessary.

Nevertheless, in specific cases, such as in the digestion of rigid agricultural residues (e.g. tomato stalks, bagasse), phase separation combined with effluent recycle from the methanogenic reactor certainly represents an attractive alternative (Rijkens, 1981).

As far as more complex soluble wastes are concerned, a separate acidogenic step might be attractive for the elimination of toxic compounds from the raw waste water, the prior removal of nitrate, sulphate or sulphite, and perhaps — in specific cases — for preventing the deterioration of granular sludge in UASB reactors, or the deterioration of the attached film in fluidized-bed systems (Heynen, 1983). However, before installing a separate acidogenic reactor, one should consider that the acid formation from soluble biodegradable compounds such as sugars proceeds very rapidly, provided that the pH is maintained above approximately 5 (Cohen *et al.*, 1980, 1982; Cohen, 1982; Zoetemeyer, 1982). On the other hand the acidification of amino acids proceeds more slowly, notwithstanding the fact that, in practice, a sufficient pre-acidification generally can be achieved in a simple holding (equalization) tank, or sometimes even in the raw waste-water supply lines.

The UASB process

To date the UASB process is by far the most widely applied high-rate AWWT system in practice. So far over 50 full-scale UASB plants have been installed, and with a few exceptions the performance of these plants is very satisfactory. *Table 4* shows an up-to-date list of the UASB plants in operation and commissioned.

Table 4. UASB plants installed and commissioned by July 1984

Type of waste water	Number of plants	Country of installation	Number of plants in country cited	Design* capacity (kg COD/ m^3/d)	Reactor volume (m^3)
Sugar beet	10	Netherlands	7	12·5–17	200–1700
		Germany	2	9,12	2300,1500
		Austria	1	8	3000
Liquid sugar	1	Netherlands	1	17	30
Potato processing	10	Netherlands	8	5–11	240–1500
		USA	1	6	2200
		Switzerland	1	8·5	600
Potato starch	3	Netherlands	2	8·5, 15	1700, 5500
		USA	1	11	1800
Maize starch	1	Netherlands	1	10–12	900
Wheat starch	3	Netherlands	1	7	500
		Ireland	1	9	2200
		Australia	1	11	4200
Alcohol	2	Netherlands	1	16	700
		Germany	1	9	2300
Yeast	1	USA	1	12	4400
Brewery	2	USA	1	14	4600
		Netherlands	1	5–10(23°C)	1400
Shellfish	1	Netherlands	1	10	2 × 50
Slaughterhouse	1	Netherlands	1	3–5	600
Dairy	1	Canada	1	6–8	450
Paper	2	Netherlands	2	8–10	1000, 740
				4(20°C)	740
Vegetable canning	1	Netherlands	1	10	375
White spirit	12	Thailand	12	15	3000

* At treatment temperature 30–35°C, unless otherwise stated.

The main conditions to be met in a UASB plant are:

1. An effective separation of the biogas, the liquid and the sludge must be accomplished;
2. The anaerobic sludge should attain an easily settleable form — preferentially it should develop as a granular sludge;
3. The raw waste water has to be introduced over the bottom of the reactor as evenly as possible.

The outline of a UASB plant is shown in diagrammatic form in *Figure 2*.

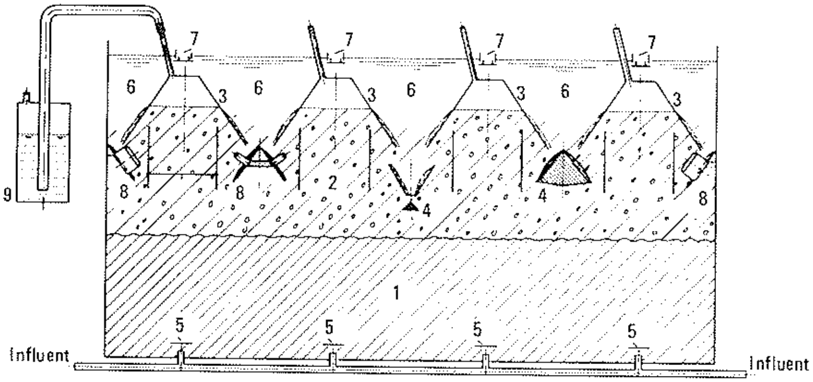

Figure 2. Diagram of a full-scale UASB plant with vertical baffles installed beneath the gas collector. 1: sludge bed; 2: bulk of the liquid with dispersed sludge forming a 'blanket'; 3: gas bowl; 4: gas seal; 5: feed-inlet distribution system; 6: settler; 7: effluent launder; 8: gas collector with exhaust pipe to (3); 9: water seal.

GAS–SOLIDS SEPARATORS (GSS)

As explained elsewhere (Lettinga and Vinken, 1980; Lettinga *et al.*, 1980, 1981, 1983a) it is essential to install a GSS device in the upper part of the reactor, irrespective of the nature of the sludge contained in the system (*see Table 5*).

The first and main objective of the GSS device is to separate the gas as effectively as possible from the mixed liquor, so that a quiescent (settling) zone is created in the uppermost part of the reactor.

It should be emphasized that the installation of a GSS device is required even if the reactor is completely filled with a highly settleable type of sludge, e.g. a granular sludge. At extremely high loads a significant fraction of the sludge will become redispersed in the liquid above the sludge bed, because of:

1. The marked turbulence caused by the brisk gas evolution in the sludge bed;
2. The increasing tendency of sludge aggregates to float, due to adherent or enclosed gas bubbles.

Sludge granules settling at a rate of 40 m/h in unloaded systems may float at very high sludge loads. In order to be able to retain the sludge aggregates in such heavily loaded systems, specific adaptations must be made to the design of the

Table 5. Main objectives of the Gas–Solids Separator (GSS)

1. The separation of the biogas from the mixed liquor and from floating sludge particles.
2. The separation of dispersed sludge particles/flocs by settling, flocculation and/or entrapment in a sludge blanket (if) present in the settler compartment.
3. Enabling the separated sludge to slide back (e.g. as large aggregates) into the digester compartment.
4. Restricting excessive expansion of the sludge blanket.

GSS device (Lettinga *et al.*, 1983b). Few, if any, flotation problems may be encountered in particulate attached film systems (AFFEB, fluid bed). Nevertheless, even here the installation of a GSS device may be very beneficial.

As explained in previous papers (Lettinga *et al.*, 1980, 1983a), the design of the GSS device can be very simple: for example, additional provisions for a circulation liquid stream or for a so-called expansion compartment, as suggested in the literature (van der Meer and de Vletter, 1982) in our opinion can be omitted.

In our opinion, an expansion of the sludge blanket into the settler compartment is in no way detrimental. In fact even the reverse may be true, as the presence of a sludge blanket in the settler may serve more or less as a lock for an extensive sludge-bed expansion into the settler compartment; it entraps separate sludge flocs/particles and it also provides a further reduction in the chemical oxygen demand (COD). Attention should be paid to the occasional strong tendency of the anaerobic sludge (and/or ingredients of the sludge) to float. For this purpose, one or more baffles should be installed in front of the effluent weir. In specific cases, e.g. in treating liquid wastes containing lipids, the installation of a skimmer should be considered for removing the scum layer which may build up continuously in such cases.

GRANULATION OF ANAEROBIC SLUDGE

One of the main features of the UASB reactor is its ability to produce a granular type of anaerobic sludge. The mechanism underlying the granulation process has been one of the main research items in our department for many years (de Zeeuw, 1982; Hulshoff Pol, Webers and Lettinga, 1983; de Zeeuw and Lettinga, 1983; Hulshoff Pol *et al.*, 1983a, 1984).

1. For cultivating a high-quality anaerobic sludge in a UASB reactor the first start-up of the process should be correct. The guidelines in *Table 6* are tentative, which means that according to our current views, this is the best way to cultivate a granular sludge. In due course these recommendations will have to be adapted in more detail to the different waste-water characteristics, e.g. waste strength, waste composition etc.
2. For a relatively easy and fast first start-up, it is advisable to use a digested sewage sludge of fairly poor specific methanogenic activity (i.e. approximately 0·05 kg COD/kg VSS per day) and an amount of seed sludge of at least 10 kg VSS/m^3. Moreover, the settleability of the sludge ingredients after the wash-out of the finely dispersed fraction of the sludge should be

Table 6. Tentative guidelines for the first start-up of a UASB plant using digested sewage sludge as seed

1. Amount of seed sludge: 10–15 kg VSS/m^3
2. Initial sludge load: 0·05–0·1 kg COD/kg VSS/day
3. No increase of the sludge load unless all VFAs are more than 80% degraded.
4. Permit the wash-out of voluminous (poorly settling) sludge
5. Retain the heavy part of the sludge

satisfactory, i.e. the SVI should be approximately 50 ml/g DSS or lower. According to de Zeeuw and Lettinga (1983) these requirements can be met by using digested sewage sludge with an original sludge concentration exceeding 75 kg DSS/m^3. So far there has been little experience with materials other than digested sewage sludge for starting up a UASB reactor under mesophilic conditions. According to results obtained by Wiegant and Lettinga (1983), cow manure might be an appropriate material for the start-up of a UASB reactor under thermophilic conditions.

3. The granulation process of anaerobic sludge on a VFA mixture as substrate can be divided into three phases:
   Phase 1, the initial phase of start-up, with space loads up to 2 kg COD/m^3 per day. The wash-out of sludge is limited to the very fine fraction of the seed sludge. The sludge bed expands as a result of the hydraulic load and the gradually increasing gas production.
   Phase 2 (at space loads up to approximately 5 kg COD/m^3 per day), during which a distinct wash-out of sludge occurs, mainly because of excessive expansion of the sludge bed; most of the sludge washed out is very flocculant. Approximately 40 days after the start of the experiment, distinct granules can be observed already in the retained part of the sludge. As a result of the marked loss of sludge from the reactor, the sludge loading rate increases rapidly during phase 2, but the system can still handle the load applied in a fairly satisfactory manner because of the sharp increase in the specific activity of the sludge. At the end of phase 2 the sludge wash-out declines through the formation of granular sludge which will be retained within the reactor. In fact, a selection between the heavier sludge granules and the dispersed and flocculant sludge ingredients occurs in the reactor.
   Phase 3 (space loads exceeding 5 kg COD/m^3 per day) where the wash-out of flocculant sludge becomes increasingly smaller than the yield as a result of newly formed granules. The loading rate can now be greatly increased and ultimately space loads exceeding 50 kg COD/m^3 per day can well be accommodated, when the main part of the reactor has filled up with the granular sludge.
4. Depending on the nature of the seed sludge, the composition of the substrate and the conditions applied during the start-up, different types of granular sludge may develop (Hulshoff Pol *et al.*, 1983b):
   Sarcina granules, which develop when a high concentration of acetic acid is maintained in the reactor;
   'Rod' granules, which consist predominantly of rod-shaped bacteria in fragments of approximately five cells. This type of sludge develops on potato-processing waste and sugar-beet wastes in full-scale plants, but also on VFA substrates when the digested sewage sludge has been enriched with a small amount of (crushed) granular sludge of the 'rod' type;
   'Filamentous' granules, which mainly consist of long multicellular rod-shaped bacteria. These granules develop on pure VFA substrates and digested sewage sludge of a relatively high specific methanogenic activity (i.e. exceeding about 0·12 kg COD/kg VSS per day) and relatively low DSS content, i.e. less than 40 kg/m^3;

'Spiky' granules, which are very uniform in shape and size, and contain up to 60% $CaCO_3$. These granules are up to 1 mm in length and less than 0·5 mm thick. The spiky granules develop on maize-starch waste in a 900 m^3 full-scale UASB plant. The settleability of the granules varies widely, depending on their size and shape, their bacterial composition and their ash content. Settling velocities are in the range of 2–90 m/h in unloaded systems.

4. For an optimal start-up it is imperative that all essential ingredients for growth are present in sufficient amounts and in an available form. This has been demonstrated clearly in pilot-plant experiments with rendering wastes, which frequently have a phosphate deficit (de Zeeuw, 1982).
5. It is very important that the waste water is free from inhibitory concentrations of toxic compounds.
6. Recent experimental results (L.W. Hulshoff Pol, unpublished work) indicate that the rate of start-up can be enhanced significantly by increasing the operational temperature (from 30°C to 38°C).
7. Sludge granulation proceeds more easily on a mainly sucrose substrate than on a mainly VFA substrate of the same COD level (Hulshoff Pol *et al.*, 1984).
8. The rate of start-up can be enhanced significantly by supplying a suitable carrier material to the seed sludge, e.g. anthracite particles (L.W. Hulshoff Pol, unpublished work). Huysman and co-workers also observed a considerable enhancement of the rate of methanogenesis by using specific carrier materials (Huysman *et al.*, 1983).

THE IMPORTANCE OF THE FEED-INLET DISTRIBUTION SYSTEM

For achieving high loading rates it is essential to prevent channelling in the sludge bed as much as possible. The risks of severe channelling are greatest when the process is applied to the treatment of cold and/or dilute waste water, because the gas production may be too low for adequate mixing in those instances. The risk of channelling is also great when the height of the sludge bed is low, the number of feed-inlet points is small and when the sludge exerts a high settleability. In these cases, but particularly in applying the process at low ambient temperatures, the installation of a more sophisticated feed-inlet system is a prerequisite for correct performance. On the basis of the relatively sparse information available, some rough guidelines for the required number of feed-inlet points are presented in *Table 7*. It will be obvious that, with the

Table 7. Rough guidelines for the number of feed-inlet nozzles required in a UASB reactor

Type of sludge	Area (m^2) per nozzle
1. Dense flocculant sludge (exceeding 40 kg DS/m^3)	One at loads less than 1–2 kg COD/m^3/day
2. Thin flocculant sludge (less than 40 kg DS/m^3)	Five at loads exceeding approx. 3 kg COD/m^3/day
3. Thick granular sludge	One at loads of approx. 1–2 kg COD/m^3/day

installation of highly sophisticated feed-inlet distribution systems conforming to those installed in fluid-bed reactors, sufficient contact between waste water and retained sludge is guaranteed under almost all conditions and (a fairly important point) without applying fluidization.

Despite the fact that very simple feed-inlet distribution systems have been installed in the first full-scale UASB plants (i.e. only one feed-inlet nozzle every 5–10 m^2) excellent results are obtained in these plants (Pette *et al.*, 1980; Pette and Versprille, 1981), e.g. over 90% COD reduction at COD loads as high as 16 kg/m^3 per day with sugar-beet wastes at 30°C and liquid-retention times of less than 4 h. However, after a prolonged period of standstill the secondary start-up may require a longer period than would have been necessary if the reactor had been equipped with a more sophisticated feed-inlet system.

THE MAXIMUM SLUDGE HOLD-UP

The more sludge the reactor contains, the higher the loading potential of the system. Ultimately, the maximum sludge hold-up of a given reactor will be dictated mainly by the organic loading rate applied, which in turn is related to numerous factors, including the feed-inlet system employed in the reactor.

The application of the UASB system and its potential

Although the UASB process was originally developed for the treatment of mainly soluble low- and medium-strength waste waters, it would be a serious misapprehension to conclude that the process is applicable only to these categories. Satisfactory results have already been achieved with complex wastes, at optimal and suboptimal mesophilic temperatures.

The potential of the UASB concept for treating mainly soluble liquid wastes has been demonstrated in both full-scale and pilot-plant UASB reactors, as well as in numerous bench-scale UASB experiments with various types of wastes, e.g. from sugar beet (soured as well as unsoured), bean blanching, sauerkraut, alcoholic fermentations, potato processing, as well as composite VFA wastes etc.

As mentioned before, in treating mainly soluble materials there are few reasons for applying phase separation. In the absence of toxic compounds, presumably a simple one-step treatment process is the most attractive, both for economic reasons as well as from the point of view of process stability and performance. In this connection one should consider that overloading usually (both in a one-step as well as in a two-step process) results in a poor performance. The main criterion for good performance of a one-step or two-step UASB process is correct operation.

ELIMINATION OF TOXIC AND POTENTIALLY TOXIC SUBSTANCES

A valid reason for applying phase separation to soluble wastes is the prior elimination of hazardous or toxic compounds such as SO_3^{2-}, SO_4^{2-}, formaldehyde, CN^- etc. A compound of particular interest is formaldehyde, because it kills bacterial matter. Therefore, in treating wastes containing formaldehyde, it is essential that bacterial growth exceeds bacterial kill. If this condition is not met, very unpleasant surprises will occur in due course, depending on the overcapacity present in the reactor at the start of the process (Jans, 1980; de Becker, Jans and Piscaer, 1983).

The presence of sulphate or sulphite will give rise to the formation of hydrogen sulphide; both compounds are used as an electron acceptor by sulphate-reducing bacteria. Although sulphite is an intermediate product in the conversion of sulphate to sulphide, sulphate reduction will not cause accumulation of sulphite. This is because the formation of sulphite requires energy, whereas the sulphite reduction delivers energy. Consequently, sulphate is always completely converted to sulphide (Schlegel, 1981).

Generally the reduction of sulphite to sulphide is beneficial for the anaerobic purification process. It means that higher concentrations of sulphite can be accepted in the influent solution, than the concentration at which this compound becomes toxic for the methanogenic bacteria. This is because methanogens tolerate sulphide at higher concentrations than sulphite (Wijbenga *et al.*, 1983). Moreover, part of the sulphide is removed from the liquid phase by the biogas.

In a number of full-scale UASB installations a separate acidogenic reactor has been installed with the main purpose of eliminating sulphite from waste water, e.g. plants treating waste water from maize starch (Zeevalkink, 1982) and from potato starch (Wijbenga *et al.*, 1983).

The formation of sulphide, however, should not be neglected in case the influent contains sulphite or sulphate. The reduction of sulphate to sulphide is a clear disadvantage, because sulphide is much more toxic to methanogenic bacteria than sulphate. Inhibition by sulphide is mainly caused by the undissociated acid, which means that the pH has a strong influence on the allowable sulphide concentration. Recently published experimental data indicate a decrease of 25–90% of the methanogenic activity of acetate-splitting bacteria, when 150 mg/ℓ of undissociated hydrogen sulphide is present in the liquid (Mulder, 1982; Kroiss and Plahl-Wabnegg, 1983). The potential for increased tolerance by adaptation appears to be low (Parkin, Kocher and Miller, 1981). However, reactors with a high sludge-retention time — i.e. UASBs and anaerobic filters — seem to be relatively insensitive to inhibition by sulphide (Parkin and Speece, 1983). In these systems an attractive volumetric loading rate can be achieved when the methanogenic activity of the sludge is still relatively low. With yeast-production waste water for example, loading rates up to 14 kg COD/m^3 per day could be applied to a UASB, at hydrogen sulphide concentrations of approximately 90 mg/ℓ. A COD-removal efficiency of 60–80% could still be reached under these circumstances (Mulder, 1982).

The concentration of sulphate or sulphite that can be tolerated in the influent solution depends mainly on the COD of the influent and on the pH that is maintained in the reactor. On the other hand, the COD determines the amount of sulphide that is produced. Theoretically, complete sulphate reduction is possible at $COD:SO_4^{2-}$ ratios above 0·67 gram/gram (this is equivalent to 0·6 gram COD/gram SO_3^{2-}). However there are strong indications that acetate present in the influent or formed in the reactor by acetogenic bacteria, is not (completely) used as an electron donor by sulphate-reducing bacteria (Buswell, Pagans and Sollo, 1949; Middleton and Lawrence, 1977; Mulder, 1982; Hoeks *et al.*, 1983). This means that for waste water containing mainly carbohydrates, a COD:SO_4^{2-} ratio of approximately 2·0 gram/gram is required for complete sulphate reduction.

Secondly, the COD determines how much hydrogen sulphide can be removed from the liquid by the biogas. In analysing the literature, it appears that, with influents containing 2–4 gram sulphate per litre and operating at neutral pH, in almost all cases a fairly satisfactory performance can be achieved at COD:SO_4^{2-} ratios exceeding 7·5–10 gram/gram (*see Table 8*). At higher sulphate concentrations this minimum will be slightly higher. On the other hand, a lower ratio can be tolerated, when the anaerobic reactor is operated at higher pH levels.

Recent experiences with 'acid water' from the industrial production of fatty acids, indicate that even extremely unfavourable COD:SO_4^{2-} ratios — of an order of magnitude of 0·2 gram/gram — can be tolerated (Hoeks *et al.*, 1983). Under such circumstances, the COD of the influent solution limits the applicability of anaerobic digestion. Above a critical COD, depending mainly on the pH in the reactor and the desired loading rate, the sulphide concentration will become too high for the methanogens.

In all cases of (expected) inhibition by sulphide, operation in the pH range of 7·5–8·0 (usually considered to be suboptimal) is the primary control strategy. Dilution of the influent solution and stripping or precipitation of sulphides should be considered as secondary options.

SOLUBLE WASTES

In treating mainly soluble wastes, generally very high space-loading rates can be applied in UASB plants. Some results achieved with VFA substrates, an alcoholic waste and potato-processing waste are summarized in *Table 9*. In looking at these experimental results one should bear in mind that the sludge bed in most of these experiments occupied only approximately 60% of the total reactor volume. Consequently, it is apparent that considerably higher loading rates could be applied at 30°C. Another method of increasing the loading rates of a UASB reactor could be by raising the temperature from 30°C to 38°C because this should be accompanied by approximately double the specific activity of the bacterial sludge at the same overall sludge retention. This implies a doubling of the maximal loading rate, but this has yet to be proved.

In applying very high sludge loads, appropriate adjustments have to be made to the design of the GSS because, under these conditions, a considerable fraction of the sludge granules will be redispersed in the liquid medium above the sludge bed, because of the marked turbulence brought about by rising gas bubbles, as well as the increasing tendency of the granules to float.

Table 8. Literature data concerning the performance of anaerobic treatment systems with wastes containing high concentrations of sulphate

Origin of Waste	Characteristics of waste			Loading rate	Temp.	Efficiency		Remarks	Reference
	COD (g/ℓ)	BOD (g/ℓ)	SO_4^{2-} (g/ℓ)	(kg/m^3/day)	(°C)	BOD (%)	COD (%)		
Rum* distillery	100–135	20–35	3·5	3·8 (COD)	36		71–85	50 % diluted waste	Hiat, Carr and Andrews (1973)
Rum distillery	54·6		1·3	3·9 (COD)	35		77–83		Roth and Lentz (1977)
Beet molasses distillery	59·5	32	±2.4	3·2 (BOD) 5·9 (COD)	35	95·9			Basu and Leclerc (1973)
Alcohol distillery		30–40 (5–9% VS)	2–10	2.5 (VS) 6 (VS)	35 65	80–90		Inhibition at SO_4>5g/ℓ	Hideo Ono (1964)
Yeast waste	1·5–8·5 1·5–8·5		0·5–1·0 0–5	15 (COD) 10 (COD)	30–35 30		60–80 70–90		Mulder (1982)
Yeast waste	16.1	8·8	1·0	8 (COD)	30		60–61		Hansford and Richter (1975)
White water†	6·4	2·12	1·5	1·4 (BOD)	35		68		Rudolf and Arnberg (1952)

* K^+: 8·4 g/ℓ; Ca^{2+}: 1·9 g/ℓ

† Paperboard-mill waste water

Table 9. Results obtained with UASB reactors using granular or mainly granular seed sludge, and VFA solutions, alcoholic waste water and potato-processing waste as feed

Substrate characteristics			Experimental conditions						COD reduction	
Type	COD (mg/ℓ)	Soured (%)	Medium used for growth of seed-sludge inoculum	Reactor volume	Volume of sludge bed	COD sludge load (kg/kg VSS/day)	Temp. (°C)	COD load (kg/m³/day)	Total (%)	Filtered effluent (%)
VFA	1000 C_2 1000 C_3	100	VFA	30 ℓ	15–20ℓ	2·3	30	62		80–90
Alcoholic*		0	Sugarbeet waste	2·7ℓ	~ 1·5	0·6–0·7	30	22		>95
				28 ℓ	~10	0·7	30	14		85–90
Potato processing	2·5–4·2	6–16	Digested sewage sludge	6 m³	<2 m³	0·27	19	3–5	88	95
	3·3–5			6 m³	<2 m³	0·65	26	10–15	86	95
	3·5–4·5			6 m³	<2 m³	0·97	30	15–18	83	95
	3·5–7·1			6 m³	~4 m³	1·45	35	25–45	84	93

* Methanol 51%; ethanol 27%; propanol 12%; butanol 10%

Table 10. Results obtained with three types of complex wastes using flocculant-sludge UASB reactors

Waste	Influent COD* total (mg/ℓ)	Influent COD* soluble (mg/ℓ)	Temperature applied (°C)	COD load applied (kg/m³/day)	COD reduction† achieved: Filtered (%)	COD reduction† achieved: Unfiltered (%)	Volume of reactor
Domestic sewage	322–950	235–460	15–20°C	max. 2·0	30–80		6 m³
			9–12°C	max. 2·0			
Calf-fattening	9500	3800	30°C	4	93	90	25 ℓ
			25°C	2	90	85	25 ℓ
Slaughterhouse	1500–2200	600–1100	30°C	2·5–3·0	75–85	65–80	30 m³
			20°C	1·5–2·5	78–85	55–75	

* COD values for domestic sewage comprise average values of 5–15 composite daily samples

† Filtered: COD reduction based on filtered effluent and raw influent; unfiltered: COD reduction based on raw effluent and influent samples

COMPLEX WASTES

In treating complex (i.e. partially insoluble) wastes, generally significantly lower loading rates can be applied, unless a major reduction in SS is not sought and the system permits high loading rates, i.e. the active biomass can be selectively retained in the reactor, and SS from the influent partly or entirely flows through the reactor. In fact, high loading rates are possible with complex wastes only when employing granular sludge-bed reactors. In flocculant sludge-bed UASB reactors the presence of poorly or non-biodegradable suspended matter in the waste water will result irrevocably in a sharp drop in the specific methanogenic activity, because the dispersed solids will be trapped in the sludge. Moreover, any significant granulation generally will not occur under these conditions. However, despite the decreased maximum loading potentials of such a flocculant sludge-bed system, it should not be concluded that a one-step anaerobic treatment system is less attractive than a two-step system, or a system in which the anaerobic reactor is combined with a primary settler. Even space-loading rates in the range of 1–4 kg COD/m^3 per day can be economically very attractive, and possibly more attractive than a (much?) higher-loaded two-step approach. This applies particularly to low-strength wastes in which the insoluble fraction is less than about 50%, but it is also true for medium- and high-strength wastes which, after hydrolysis (and acidogenesis), do not allow an easy separation of the remaining solids from the liquid.

Some relevant results obtained with raw domestic sewage (18–45% insoluble COD), slaughterhouse waste (40–50% insoluble COD) and calf-fattening waste (approximately 60% insoluble COD) are presented in *Tables 10* and *11* for flocculant and granular sludge-bed UASB reactors respectively.

RAW DOMESTIC SEWAGE

Since 1976 there have been extensive investigations in our department on the anaerobic treatment of raw domestic sewage. The results obtained so far suggest that a flocculant sludge UASB treatment process represents a very attractive alternative for developing countries in tropical areas. The results shown in *Figure 3* indicate that 60–80% COD reduction can be achieved at a treatment temperature of 20°C and liquid-retention times as low as 8 h. The results shown in *Figure 3* were obtained using a UASB reactor which was completely filled up with sludge: for this reason, no relevant data are available concerning the efficiency of the system with respect to SS reduction (Grin, Roersma and Lettinga, 1983). However, in parallel experiments it transpired that a pall-ring packed anaerobic filter (AF) was slightly — although distinctly — more effective for SS reduction than a UASB reactor. On the other hand, the long-term treatment efficiency of the AF fell behind that of the UASB reactor, because of severe channelling in the filter. Therefore at this stage we tend to prefer the UASB process to the AF as far as the anaerobic treatment of raw sewage is concerned. On the other hand, a combined UASB–AF system might perhaps represent an even more attractive solution.

Table 11. Results obtained with three types of complex wastes using granular sludge UASB reactors

Waste water			Reactor		Amount of seed sludge (g)	COD load (kg/m^3/day)	Temp (°C)	COD reduction*	
Origin	COD (g/ℓ)	Fraction dissolved (%)	Volume (litres)	Height (m)				Filtered effluent (%)	Unfiltered effluent (%)
Rendering wastes	5500	70	60	2	1500	27	30	94	64
	3000	85	60	2	1500	63	30	80	63
Slaughterhouse	1·5–2·2	50	30	1·3	1000	10	30	87	
	1·5–2·2				1000	6	20	91	
Raw sewage	0·2–0·9	5–35	120	2	3300	0·7–2·7	8–20	60–89	54–72

* Based on filtered effluent samples and unfiltered influent samples

† Averaged values measured over periods of 5–12 days

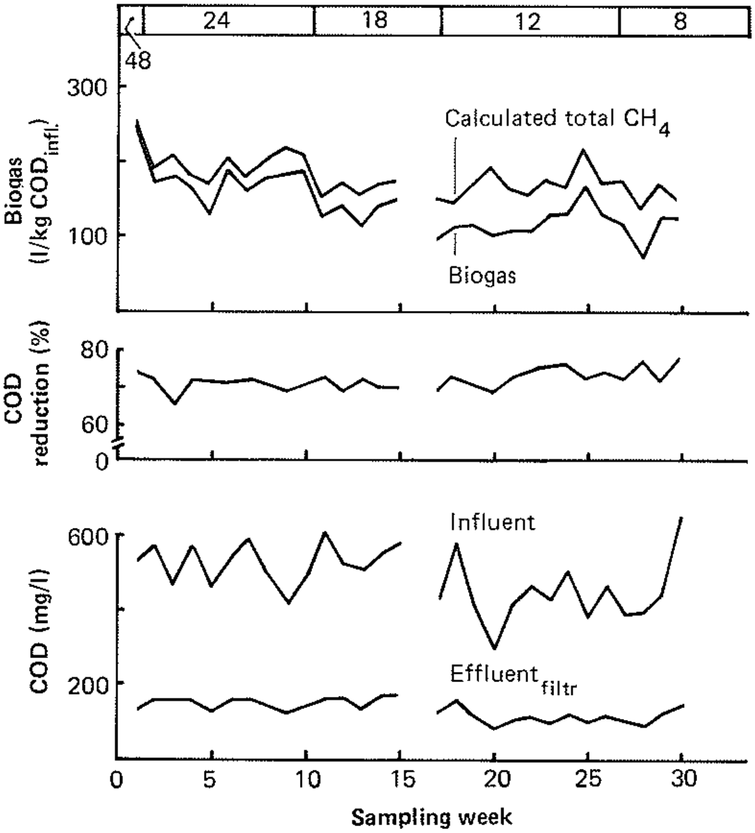

Figure 3. Results obtained with raw domestic sewage in a 6 m^3 UASB plant (May–December 1981). All experimental data were averaged over periods of 7 days. The operational temperature was 20°C. In the 'calculated' total CH_4 production the amount of dissolved CH_4 was accounted for. The numbers along the top of the Figure represent hydraulic retention time.

Results obtained in a 50 m^3 UASB with raw domestic sewage in Cali (Colombia) have shown a very similar treatment efficiency, despite the fact that the waste-water COD is only approximately 50% that of the experiments shown in *Figure 3*. On the other hand, the waste-water temperature in Cali is significantly higher (26–28°C).

In addition to the experiments shown in *Figure 3*, experiments have been performed at lower temperatures (September 1982 – April 1983). The main results obtained are summarized in *Table 12*. The maximum specific methanogenic activity of the sludge at 30°C (measured with a VFA mixture in a standardized batch test) was in the range 0·17–0·25 kg COD/kg VSS per day; no distinct drop in specific activity over the experimental period (week 1–26) could be observed. From the data in *Table 12* it is clear that the performance of the system with respect to the removal of SS is fairly poor, particularly at temperatures below 12°C: this presumably is attributable to channelling in the

Table 12. Main results obtained in a 6 m^3 UASB plant with raw domestic sewage as influent (temperature range 9·5–19°C, liquid-retention time 8 h)

	Week number		
Measurement	1–11	16–22	24–26
Temperature (°C)	19–15	11–12	9·5 –10
Effl. $COD_{filtr.}$ (mg/ℓ)	100–200	150–175	175–250
COD reduction $(\%)_{filt. effl.}$	65–80	55–70	55
COD reduction $(\%)_{raw effl.}$	40–55	30–50	30
CH_4 production (m^3/kg COD_{inf})	0·130	0·090	0·050
Excess sludge production (kg DS/kg COD_{inf})	0·195	0·172	0·271

sludge bed, resulting from the very low gas production at these low temperatures. At higher temperatures, finely dispersed matter present in the raw sewage is removed considerably more efficiently, allowing the system to be exposed to higher hydraulic and organic loads. The maximum loading potentials of flocculant sludge UASB reactors for raw sewage have not been established for temperatures exceeding 20°C.

In experiments with slaughterhouse wastes (Sayed, 1984) a very satisfactory performance of flocculant-sludge UASB systems was achieved at liquid-retention times of 8 h and at treatment temperatures of approximately 20°C. Presumably even higher hydraulic (and organic) loads can be accommodated. On the other hand, the presence of lipids in this type of waste may give rise to specific problems with respect to the build-up of scum layers at the liquid interface in the settler compartment. As the materials accumulating in the scum layer are poorly stabilized, in some cases a separate 'scum layer' digester may be required in order to get rid of these materials.

Significantly higher loading rates can be accommodated in granular-sludge UASB reactors compared with flocculant-sludge bed reactors (cf. the results in *Tables 10* and *11*). However, it should be recognized that the SS reduction in granular-sludge bed systems becomes very poor at high space loads, mainly because of the considerable turbulence resulting from the vigorous gas evolution: a primary or secondary settler therefore has to be installed in line with the anaerobic reactor. As mentioned earlier, granular sludge bed reactors require a more sophisticated feed-inlet distribution system than flocculant sludge bed reactors, particularly in applying the process for very low-strength and relatively cold waste waters. However, so far only small-scale granular sludge UASB experiments have been performed under these conditions, e.g. at a scale of 120 ℓ (reactor height 2 m) with raw domestic sewage (Lettinga *et al.*, 1983b). In addition, experiments have been started recently using an expanded granular sludge bed reactor (reactor height 2 m, reactor volume 13 ℓ) with settled domestic sewage. Although only 4 ℓ of granular sludge was supplied to the reactor at the start of the experiment, a COD removal efficiency of 45–75% could be achieved over a prolonged period (approximately 5 months) at liquid-retention times of 30–120 min. Higher treatment efficiencies were obtained at higher influent COD values, i.e. exceeding approximately 350 mg/ℓ. The treatment temperature was 18–21°C in these experiments. Part

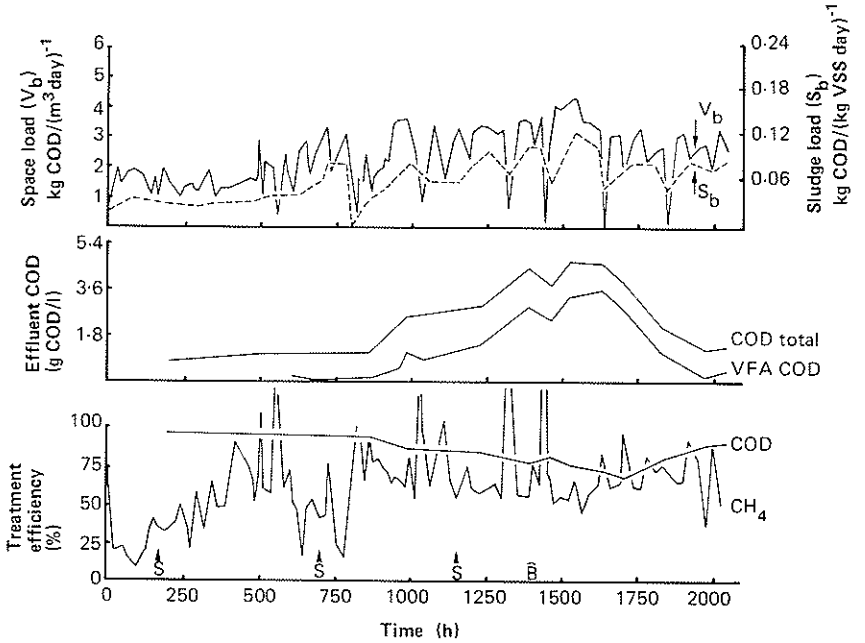

Figure 4. Anaerobic treatment of potato starch waste water in a 5·3 ℓ UASB reactor at a temperature of 14°C.

of the excess sludge accumulates above the granular sludge bed at hydraulic loads below about 4 m^3/m^3/day, but at hydraulic loads of 24 m^3/m^3/day almost all the excess sludge is washed out of the reactor.

Fairly satisfactory results were also obtained at low temperatures (14°C) with potato-starch waste water using a 5·25 ℓ UASB reactor seeded with granular sludge cultivated on sugar-beet waste water. As shown in *Figure 4*, organic space loads up to 3 kg COD/m^3/day were satisfactorily accommodated. Foaming may represent one of the main problems in treating potato-starch waste water, presumably because of the relatively slow breakdown of the proteins present in this waste. Even at higher temperatures the breakdown of the proteins constitutes the main constraint on the process.

ANAEROBIC TREATMENT UNDER THERMOPHILIC CONDITIONS

So far, the UASB process has not been applied under thermophilic conditions, but the thermophilic option without doubt will become a very attractive alternative to mesophilic treatment in the near future. This optimism is attributable not only to the major advantages of thermophilic treatment, i.e. the considerably higher conversion (and growth) rates (*Table 13*) and its higher efficiency in the reduction of pathogens, but also to the very promising results obtained in recent laboratory investigations (Wiegant and Lettinga, 1981, 1982, 1983).

Table 13. Growth rates and half saturation (substrate affinity) constants of some acetogenic and methanogenic thermophilic organisms

	Temperature			
	55°C		35°C	
Substrate	Specific growth rate (per hour)	Half saturation constant (mg/ℓ)	Specific growth rate (per hour)	Half saturation constant (mg/ℓ)
H_2/CO_2	0·532	—		
Acetate	0·046	124	0·019	300
			0·0075	42
Propionate	0·026	2·8	0·008*	7·0†
Butyrate	0·109	14‡	0·013*	7·3

* For these bacteria the highest known values (with *Desulfovibrio* spp.) are taken.
† Value determined at 33°C
‡ Detection limits generate this value

In accordance with the application of the UASB process under mesophilic and psychrophilic conditions, a prerequisite for the profitable usage of the UASB treatment concept once again is the development of sludge with a high settleability. Furthermore, it is essential that the process is properly controlled. Considerable emphasis has been placed on these two factors in our department during the past three years.

As far as the granulation of thermophilic bacterial matter is concerned, the following observations have been made:

1. Granulation of thermophilic bacterial matter proceeds easily on sucrose (unacidified!) as substrate and cow manure as seed. Using 5·35 ℓ UASB reactors (height 1 m), organic space loads up to 45 kg COD/m^3 per day are easily accommodated with 85–90% conversion efficiency (operational temperature 55°C).
2. Granulation proceeds slowly, if at all, on VFA substrates. The reason for this is not yet clear, but under mesophilic conditions, too, the granulation process did not proceed very easily on pure VFA substrates: a filamentous type of granular sludge develops under these conditions with digested sewage sludge as seed.
3. Although a thermophilic granular sludge is very difficult to cultivate on a pure VFA mixture, a sucrose-cultivated granular sludge does not deteriorate when fed with pure VFA substrates for prolonged periods. Exceptionally high space-loading rates can be satisfactorily accommodated in granular sludge thermophilic UASB reactors, i.e. up to 75 kg COD/m^3 per day.
4. A distinct break-up of granular sucrose-cultivated sludge occurs upon feeding it with a sugar-beet vinasse solution (i.e. residue after alcoholic distillation) which is 15–25% of its original strength. The undiluted sugar vinasse solution contains 119 kg/m^3 of organic pollutants (expressed as COD), 6 kg/m^3 of NH_4^+-N and 4 kg/m^3 of K^+. Despite the deterioration of the granulation (i.e. a reduction in granule size from approximately

3 mm to less than 1 mm in diameter), space-loading rates up to approximately 50 kg COD/m^3 per day can well be accommodated, mainly because the sludge retains its good settleability.

As vinasse represents a waste of particular interest for thermophilic anaerobic treatment, the feasibility of this treatment option was investigated in detail, using both intermittently fed completely mixed tank reactors (no biomass retention) and UASB reactors. In both these experiments, sucrose-cultivated granular sludge was used as seed material. The effect of the substrate concentration can be seen from the results of the experiments using completely mixed tanks (*Figure 5*). The vinasse solution apparently contains inhibitory/toxic compounds in such a concentration that direct thermophilic treatment of the undiluted waste is not feasible.

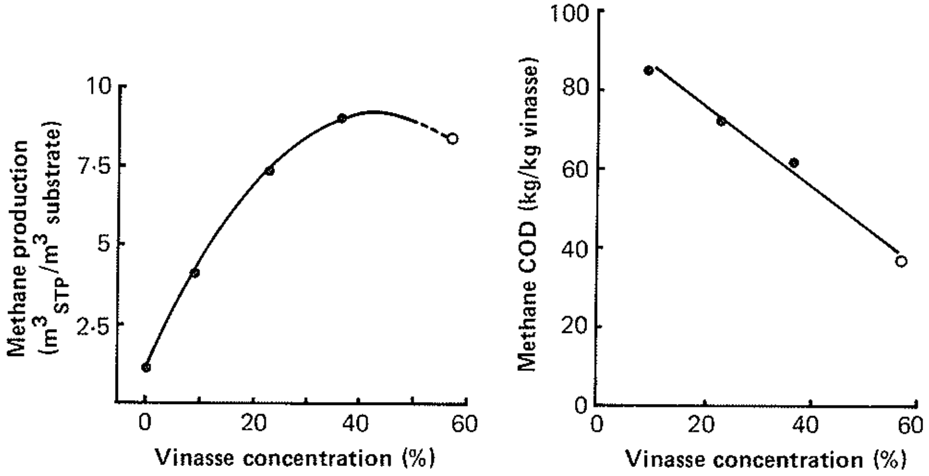

Figure 5. Effect of dilution of vinasse solution (as a percentage of the original vinasse solution) on methane production under thermophilic conditions in intermittently fed tank reactors (●, RT = 18·2 days; ○, RT = 33·3 days; temperature 55°C). Methane production is expressed as m^3 (STP)·m^{-3} vinasse in the left-hand figure, and as kg CH$_4$–COD·kg vinasse-COD^{-1} in the right-hand figure. RT = retention time.

In the UASB experiments, during a 3-month adaptation period, the substrate was shifted stepwise from sucrose to vinasse. After this adaptation period, very high loading rates could be handled by the system, as shown in *Table 14*.

In conformity with results obtained in the completely mixed tank experiments, the effluent VFA concentration is closely related to the substrate concentration applied to the system, and apparently (*see Figure 6*) in a very similar way. On the other hand, the results in *Table 14* indicate that the treatment efficiency of the system is not affected in the concentration range investigated (12·9–24·9% dilution): therefore, a very high dilution factor offers

Table 14. The performance of a 5·35 litre thermophilic (55°C) UASB reactor treating diluted vinasse solutions (seed sludge: sucrose-cultivated granular sludge). In a 3-month adaptation period the substrate was stepwise shifted from sucrose to vinasse

Loading rate	Concentration (% dilution)	Treatment eficiency*	$\frac{\text{VFA-COD}_{\text{in}}}{\text{COD}_{\text{out}}}$	Potential treatment efficiency†
(kg COD/m^3/d)	(%)	(%)	(%)	(%)
17·2	21·0	61·9	21·2	83·1
25·6	14·8	59·0	19·7	78·7
25·5	19·6	64·9	17·4	82·3
24·8	22·3	56·8	20·1	76·9
25·3	24·9	52·6	27·1	79·7
38·6	23·6	58·8	23·5	82·3
83·6	14·6	59·6	22·5	82·1
98·3‡	12·9	58·9	ND§	ND§

* 1–Filtr. effluent COD × 100%

† 1–(Filtr. effluent COD – VFA COD)/influent COD × 100%

‡ At this loading rate sludge washout occurred a few hours after the loading rate was raised to this value.

§ ND, not determined

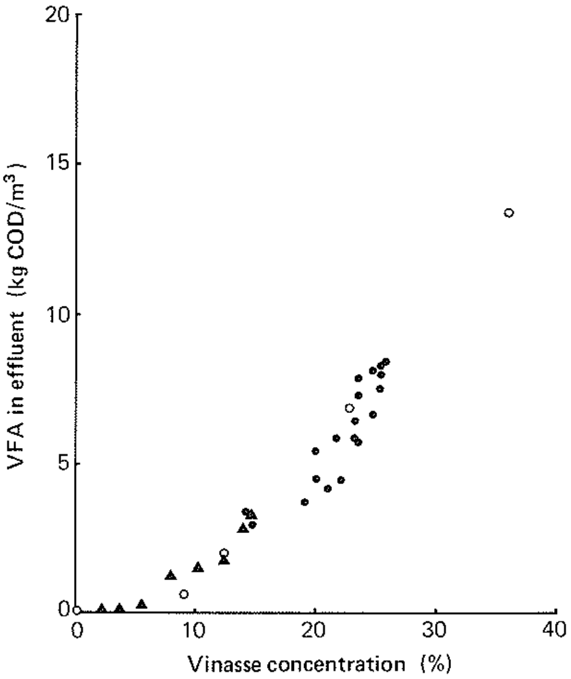

Figure 6. Effluent VFA COD concentration in relation to the influent vinasse concentration applied, as measured in completely mixed tank reactors (RT = 18·2 days (○) and in the 5·35 litre UASB experiments (*see Table 14*), during the adaptation period (▲) and after adaptation (●).

Table 15. A rough estimate of cost (in US dollars) of anaerobic treatment using the UASB reactor concept (price index 1980)

Assumptions made in the estimate:

COD load : 10 and 15 kg/m^3/day
Treatment efficiency : 90% COD-reduction
Methane yield : 0·9 kg COD-methane/kg COD-removed
Methane production ($m^3/m^3_r/y$) : 1550 at a load of 15 kg COD/m^3/day
1030 at a load of 10 kg COD/m^3/day
Interest and redemption : 15% of the capital costs
Maintenance and renewals : 2% of the capital costs
Energy requirements : 10% of the methane production
Investment costs : 1000 m^3 plant—$500 000–750 000
5000 m^3 plant—$2 000 000–3 000 000

OPERATION COSTS (in $ 1000 ×)

I. Continuous operation (365 days/year, 24 h/day)	1000 m^3 plant	5000 m^3 plant
Interest + redemption	75–112.5	300–450
Maintenance + renewals	10– 15	40– 60
Labour + supervision	15	40
Analysis + control	15	40
Total costs	115–147·5	420–590
Costs of methane gas ($/m^{3}STP)*		
1. load: 15kg COD/m^3/day	0·08 –0·105	0·06–0·085
2. load: 10 kg COD/m^3/day	0·125–0·160	0·09–0·125

II Seasonal operation (3 months/year, 24 h/day)	1000 m^3 plant	5000 m^3 plant
Interest + redemption	75–112·5	300–450
Maintenance + renewals	3–5	15– 20
Labour + supervision	5	15
Analysis + control	5	15
Total costs	88–127·5	335–500
Cost of methane gas ($/m^3 STP)		
1. load: 15 kg COD/m^3/day	0·25–0·36	0·19–0·29
2. load: 10 kg COD/m^3/day	0·38–0·55	0·29–0·43

* Net methane gas production

little, if any, benefit in this respect. Nevertheless, the results clearly show that the process is not feasible for undiluted sugar-beet vinasse solutions. However the situation presumably is more favourable for sugar-cane vinasse, because this type of vinasse contains significantly less salts (Robertiello, 1982). Consequently, the prospects for thermophilic treatment of the liquid wastes resulting from gasohol production look fairly good.

AWWT often can be regarded as a combined method of energy production and waste-water treatment. For example, approximately 40% of the energy required for the production of ethyl alcohol can be obtained from the waste-water pollutants by applying anaerobic treatment. Considering that 10 m^3 of methane can be produced from 1 m^3 of stillage, the potential methane production capacity from stillage as raw material in Brazil in 1986 at a projected fuel alcohol production of approximately 9×10^6 m^3/year may amount to $1{\cdot}2 \times 10^9$ m^3 (STP)/year. Instead of producing gasohol from the sugar cane, from the point of view of energy conservation it appears more attractive to convert the sugar cane directly to methane. From the amount of sugar cane required for the projected gasohol production in 1986, approximately $6{\cdot}6 \times 10^9$ m^3 (STP) methane per year can be produced, which is equivalent to approximately 240×10^6 GJ/year, whereas the net amount of energy produced with 9×10^6 m^3 gasohol/year is only approximately 130×10^6 GJ/year.

Costs of anaerobic waste-water treatment

It is impossible to provide exact figures for the investment and running costs of AWWT because both greatly depend on factors such as size of the plant, the need for solid foundations, labour costs, the desired degree of sophistication etc. On the basis of information available to us, the investment costs of a full-scale UASB plant — including facilities for gas utilization (i.e. a small gas store for direct use of the gas in the factory), a small control room, some heat exchangers — can be estimated at US$500 000 – $750 000 for 1000 m^3 UASB plant and at $2 000 000 – $3 000 000 for a 5000 m^3 plant. A rough estimate of the running costs and the costs of the methane gas produced is presented in *Table 15*. These data show that anaerobic treatment is indeed a very attractive method for removing organic pollutants from waste water, because this can be accomplished at a cost price for the methane of only US$0·06 – 0·16 per m^3 STP for a continuous operation, and US$0·25 – 0·43 m^3 STP for a discontinuous operation.

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