Performance of upflow anaerobic sludge blanket (UASB) reactor and other anaerobic reactor configurations for wastewater treatment: a comparative review and critical updates

Abhishek Dutta, Charlotte Davies and David S. Ikumi

ABSTRACT

With the rising cost of energy and continuously increasing pressure to be more conscious of the amount of energy used in processes, aerobic wastewater treatment methods are becoming less desirable. However, anaerobic treatment is becoming more popular, especially with the development of high-rate activated sludge processes that can handle variations in operating conditions with small energy requirements. This short review aims to summarize the characteristics of anaerobic processes, with eventual emphasis on the upflow anaerobic sludge blanket (UASB) reactor and its performance under various conditions. Modeling of processes is helpful in making design decisions, and therefore the various modeling techniques applied to UASB reactors have also been included. Specific consideration has been given to anaerobic digestion model 1 (ADM1) due to its potential to be integrated with activated sludge models (ASM) to provide a single framework to describe aerobic post-treatments of the UASB effluent. Finally, an example test case involving the modeling of a pilot-scale UASB reactor has been included to indicate the usability of ADM1 in modeling anaerobic bioprocesses.

Key words | ADM1, anaerobic process, AQUASIM, upflow anaerobic sludge blanket, wastewater

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INTRODUCTION

Aerobic treatment of wastewater has been traditionally used for decades for high concentration waste. However, sludge disposal is becoming increasingly difficult due to stringent laws and aerobic processes producing a large amount of sludge (Christensen *et al.* 1984; Khan *et al.* 2011). This is with the exception of conditions where anoxic-aerobic digesters are adopted for waste activated sludge treatment. In this case, the energy savings of anaerobic digesters are sacrificed for lower technical competence requirements. This, however, encourages further nutrient removal through nitrification and phosphorus precipitation. However, for the treatment of influent primary sludge, anaerobic digestion has been accepted as a desirable alternative, the likes of which are used worldwide for the treatment of a variety of different wastewaters (Cronin & Lo 1998; Mahmoud *et al.* 2004; Ince *et al.* 2005; Miranda *et al.* 2005; Gohil & Nakhla 2006; Akbarpour Toloti & Mehrdadi 2010; Kerroum *et al.* 2010). Moreover, aerobic processes require large amounts of energy to ensure proper aeration and hence are not only expensive to install, but also costly to run. Anaerobic processes are smaller than aerobic processes and therefore require a smaller area of land in which to be installed (Christensen *et al.* 1984; Rajakumar *et al.* 2011; Chong *et al.* 2012). Further, they require a smaller power supply (Lettinga *et al.* 1980; Christensen *et al.* 1984; Akbarpour Toloti & Mehrdadi 2010; Chen *et al.* 2011; Khan *et al.* 2011), cost less to run and produce less sludge which can be preserved for a number of months, reducing the need

doi: 10.2166/aqua.2018.090

for sludge disposal (Lettinga *et al.* 1980; Sawayama *et al.* 1999; Rajakumar *et al.* 2011; Chong *et al.* 2012). In addition, anaerobic processes produce biogas which is mostly a mixture of CO_2 and methane that can be burned as a replacement of fossil fuels (Batstone *et al.* 2002a, 2002b; Chen *et al.* 2011; Thamsiriroj & Murphy 2011).

Upflow anaerobic sludge blanket (UASB) reactor is a widely used anaerobic treatment of wastewater that has numerous benefits over its alternatives, including aerobic wastewater treatments (Lettinga *et al.* 1980; Christensen *et al.* 1984; Khan *et al.* 2011) and other anaerobic treatments (Hamdi 1996; Rajeshwari *et al.* 2000; Rajakumar *et al.* 2011; Ayaz *et al.* 2012). UASB was first developed by Lettinga *et al.* (1980) and is now one of the most widely used high rate anaerobic wastewater treatment systems due to its simple design, construction (Chen *et al.* 2011; Rajakumar *et al.* 2011) and low operating costs (Christensen *et al.* 1984; Akbarpour Toloti & Mehrdadi 2010; Chen *et al.* 2011). For these reasons, and because it is able to withstand the fluctuations in pH, temperature and influent composition that are so common in industrial wastewater, full-scale UASB reactors have come into operation to treat a variety of wastewaters since its introduction. These include breweries (Ahn *et al.* 2001; Parawira *et al.* 2005), slaughterhouses (Miranda *et al.* 2005), canning (Puñal & Lema 1999), alcohol distilleries (Ince *et al.* 2005) and pharmaceutical wastewaters (Chen *et al.* 2011) as well as sewage (Schellinkhout & Collazos 1992; Schellinkhout 1993). Because of its promise as a high throughput reactor, the UASB reactor has been studied under many different conditions, including varying types of wastewater, temperature, organic loading rate (OLR) and pH. The effects and importance of mixing in the reactor have also been studied. For example, Habeeb *et al.* (2010) studied the properties attributed to the digestion process in UASB reactors. Seghezzo *et al.* (1998) compared sewage treatment in UASB and expanded granular sludge bed reactors. The impact of granule formation on wastewater treatment in UASB reactors was examined by Abbasi & Abbasi (2012), while Singh *et al.* (1999) had earlier studied the requirements of nutrients in a UASB system. The success of solids removal in UASB reactors has been described by Mahmoud *et al.* (2003) while Lin & Yang (1991) carried out a general technical review on the UASB process.

Mathematical modeling of wastewater treatment processes is a key component in the support of decision-making (Daigger 2011; Rodriguez 2011) when designing treatment plants (Henze *et al.* 2000; Morgenroth *et al.* 2002). Models can help with technology-based design decisions, optimal sizing of the wastewater treatment plants, prediction of the nature of the effluent (Daigger 2011) and the effect of the plant on the environment (Rodriguez 2011). The effect of changing different parameters on the performance of reactors can be modeled and this can be used during operation to spot and troubleshoot errors (Henze *et al.* 2000). These models have also been used extensively in teaching and research purposes as they can help researchers with experimental design to indicate where further research is needed. Anaerobic Digestion Model No. 1 (ADM1) is a general model for anaerobic digestion developed by the IWA Anaerobic Digestion Modeling Task Group (Batstone *et al.* 2002a, 2002b) and has been widely used to model a wide variety of anaerobic digestion processes (Batstone & Keller 2003; Batstone *et al.* 2006), including the UASB (Batstone *et al.* 2005; Coelho *et al.* 2006) since its first implementation in 2002. It must be noted that ADM1 is a tool that allows predictions of sufficient accuracy to be useful. Because of the varying demands in process development, operation, and optimization, a different degree of model calibration and validation is required in each case.

Within the scope of this review, it is intended to collect the experimental studies carried out on the UASB reactor, as well as to compare it with other anaerobic treatment options. Furthermore, common modeling approaches for a UASB process have been reviewed. The most common reactor type used for anaerobic digestion of wastewaters is the continuous stirred tank reactor (CSTR). Therefore, an example case to study the implementation of IWA anaerobic digestion model ADM1 has been performed on a pilot-scale anaerobic CSTR with the implications that it can be extended to advanced anaerobic reactors such as UASB.

ANAEROBIC TREATMENT OF WASTEWATER

Anaerobic continuous stirred tank reactor

A CSTR is a simple and inexpensive design for an anaerobic treatment process (Chamy *et al.* 2011). As the name suggests, it consists of a tank with a motorized stirring arm inserted usually through the top of the reactor and operated constantly to ensure continuous stirring. It can be continuously fed through the bottom of the reactor (Méndez-Acosta *et al.* 2010) or intermittently fed in a batch process (Wendland *et al.* 2007) and is often seeded with sludge before operation (Oz *et al.* 2003; Boušková *et al.* 2005; Wendland *et al.* 2007). Figure 1 shows a typical CSTR reactor for anaerobic treatment of wastewater. The anaerobic CSTR requires high hydraulic retention time (HRT) of between 15 and 20 days, compared with often less than 1 day for processes such as UASB. These reactors can also only cope with much lower OLRs (Rajeshwari *et al.* 2000). Good *et al.* (1982) found that a fixed film reactor could cope with an organic loading rate five times greater than that of a CSTR. Furthermore, the complexity of the influent to be treated affects the performance of the reactor more than in the other types of treatment described. It has been found that attached biomass reactors, such as the fluidized bed reactor, is superior in coping with shock loads. In spite of these disadvantages, Boušková *et al.* (2005) has shown that the anaerobic CSTR can cope well with step changes in temperature, and it has been used to treat a number of wastewaters including blackwater from vacuum toilets and kitchen refuse (Wendland *et al.* 2007), glucose-containing wastewater (Oz *et al.* 2003) and tequila vinasses (Méndez-Acosta *et al.* 2010), although tequila vinasses has only been treated on a pilot scale and has not yet been successfully implemented full scale. Oz *et al.* (2003) found that the anaerobic CSTR was not successful in treating pharmaceutical wastewater due to its complexity, and that an anaerobic filter reactor was much more successful.

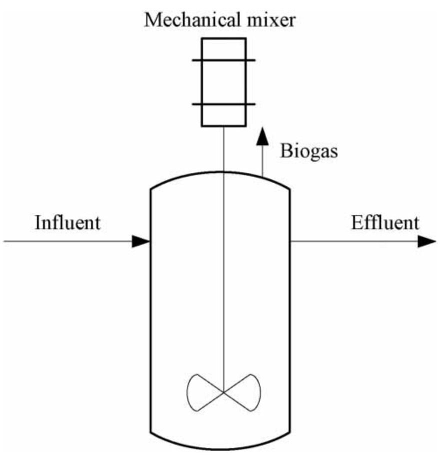

Figure 1 | Schematic of an anaerobic CSTR reactor.

Anaerobic contact process

The anaerobic contact process is an extension of the anaerobic CSTR in that it requires a continuously stirred digester followed by a settling tank (Nähle 1991; Cakir & Stenstrom 2007), as shown in Figure 2. In this settling tank, the bacteria is allowed to settle from the effluent and is able to be scraped from the bottom and recycled back to the digester (McCarty 1964; Endo & Tohya 1988; Hamdi 1996). This allows for an increase in HRT and a decrease in solid retention times (SRT) (Nähle 1991; Cakir & Stenstrom 2007). A high efficiency is needed in this recycling (McCarty 1964) which can be problematic: a degasifier is often required between the digester and the settler to overcome this problem (Cakir & Stenstrom 2007). Biogas is removed from the sludge in the degasifier to achieve better settleability of the biomass in the settling tank, but this can be expensive (Nähle 1991). The mixing in the digester can be achieved through mechanical means or by the recirculation of the biogas produced in the process (Nähle 1991; Cakir & Stenstrom 2007), but care must be taken as excessive mixing can favor the growth of acidogenic bacteria over methanogenic bacteria which can lead to lower feasible OLR (Hamdi & Garcia 1991; Hamdi 1996). The loading rates that can be applied to the contact process are higher than those that are achievable in the aerobic activated sludge process (Endo & Tohya 1988). The contact process can be quick

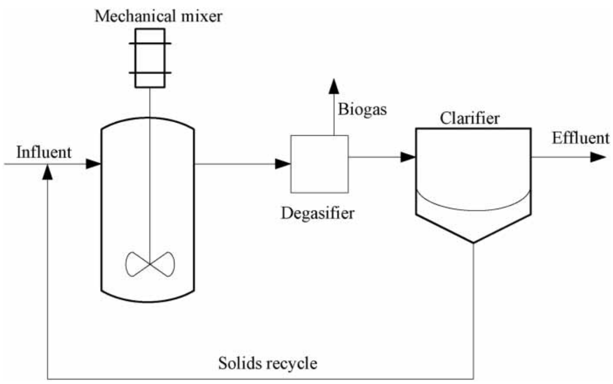

Figure 2 | Schematic of an anaerobic contactor.

to reach steady state, however this state can be more unstable compared to other processes (Hamdi & Garcia 1991). This process has been used to treat a variety of wastewaters including distilleries, breweries, meat packing, sugar, dairy and vegetable canning (McCarty 1964; Nähle 1991; Cakir & Stenstrom 2007) wastewaters.

Anaerobic filter

The anaerobic filter (AF) consists of a column which is filled with a packing material (Chian & De Walle 1977; Hickey *et al.* 1991), as can been seen in Figure 3. This packing media promotes biomass retention in the reactor, as the microorganisms can attach to the media or become entrapped. Along with those attached to the packing media, microorganisms also exist as granules or flocs in suspension, and this is where much chemical oxygen demand (COD) removal occurs (Hickey *et al.* 1991; Rajakumar *et al.* 2011). The choice of packing media can greatly affect the performance of the reactor (Manariotis & Grigoropoulos 2006a, 2006b). Characteristics of the media that are particularly important include the texture, porosity and specific surface area. It has been found that a media with high porosity is more effective than one with a lower porosity and higher SSA (Manariotis & Grigoropoulos 2006a, 2006b). The most important factor when choosing a packing media is that it is one to which bacteria can easily attach or become entrapped. Hickey *et al.* (1991) found that the AF reactor would not recover well from large pH and temperature upsets but Manariotis & Grigoropoulos (2006a, 2008) found that the AF responded well to small changes in operating conditions and restarted reasonably well after long idle periods of up to two years, although this restart time did vary with packing material and previous history of the reactor. A common issue found in literature for the AF reactor is that sludge accumulation can cause clogging (Hickey *et al.* 1991; Manariotis & Grigoropoulos 2006a) and hence periodic flushing of the reactor may be necessary. The AF reactor can be operated with up- (Hickey *et al.* 1991; Manariotis & Grigoropoulos 2006a; Rajakumar *et al.* 2011), down- or horizontal-flow (Manariotis & Grigoropoulos 2006b) and has been seen to treat low strength domestic (Manariotis & Grigoropoulos 2006b), high and low strength poultry (Rajakumar *et al.* 2011) and high acidic (Chian & DeWalle 1977) wastewaters, as well as other low and high strength (Hickey *et al.* 1991; Manariotis & Grigoropoulos 2008) wastewaters.

Anaerobic fluidized bed reactor

The anaerobic fluidized bed reactor (AFBR) consists of a column that contains a bed of fluidizing media, often sand (Heijnen *et al.* 1989; Converti *et al.* 1990; Maragno & Campos 1992; Marín *et al.* 1999). The configuration of an AFBR is shown in Figure 4. Start-up conditions are set to promote colonization of microorganisms on the bed

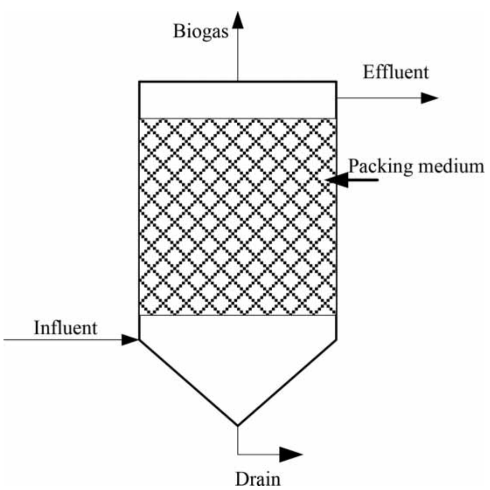

Figure 3 | Schematic of an anaerobic filter (AF).

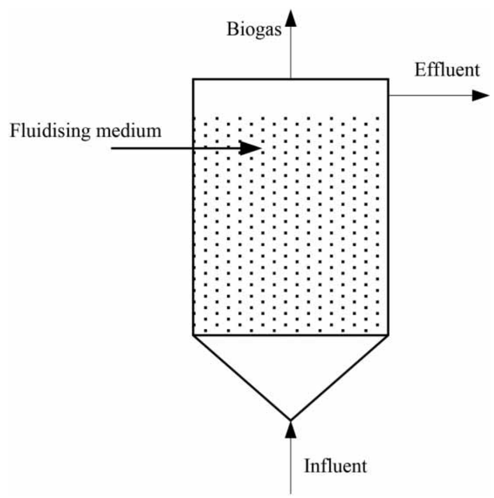

Figure 4 | Schematic of an anaerobic fluidized bed reactor (FBR).

particles (Hickey et al. 1991). The flow of wastewater is introduced at the bottom of the reactor and is high enough so as to cause fluidization of the bed (Heijnen et al. 1989; Hickey et al. 1991; Marín et al. 1999): the media particles become covered in a biofilm and hence become heavy and require significantly higher upflow velocities than in other reactors to become fluidized. The growth of the film on the media gives rise to a high biomass concentration (Tseng & Lin 1994; Mendonça et al. 2004) and high specific surface area (Carbajo et al. 2010), meaning high mass transfer efficiency (Marín et al. 1999) and reaction time (Converti et al. 1990) with no clogging of the reactor. The control of this biolayer thickness must be included in operation. A thick layer has been shown to cause washout, therefore cleaning of the media may be needed, which could incur extra expense (Heijnen et al. 1989). However, the AFBR generally finds it easy to retain the biomass in the reactor (Heijnen et al. 1989; Converti et al. 1990; Marín et al. 1999), requires smaller volume than other anaerobic wastewater treatment processes (Mendonça et al. 2004; Carbajo et al. 2010) and can resist changes to environmental conditions and toxic substances (Tseng & Lin 1994). AFBR has been used to successfully treat sewage (Mendonça et al. 2004), wine wastewater (Converti et al. 1990) and phenol-containing wastewater (Carbajo et al. 2010) as well as other low and high strength wastewaters (Tseng & Lin 1994). Wastewater from textile production has also been treated, although an additional carbon source needed to be added (Haroun & Idris 2009).

Upflow anaerobic sludge blanket reactor

The UASB reactor is generally a tubular shaped reactor, consisting of four zones (Lin & Yang 1991). The sludge bed is a layer of biomass that forms at the bottom of the reactor, and contains granules of a high settling velocity. It is where the main biochemical reactions occur (Alibhai & Forster 1986; Seghezzo et al. 1998; Schmidt & Ahring 2000). Above this forms a blanket of finely suspended flocs with lower settling velocity (Singhal et al. 1998; Kalyuzhnyi et al. 2005; Chong et al. 2012). This suspension forms due to the production of biogas in the reactor – formed from the degradation of soluble organic compounds – and is a mixture containing these gases (Lin & Yang 1991; Schmidt & Ahring 2000). Above this is the settling zone, in the upper part of the reactor. This is a quiescent zone where the granules that have been dislodged from the bed by the upflow of wastewater or gas production with a high enough settling velocity will settle back to the bed (Lettinga et al. 1980; Singhal et al. 1998). Dispersed bacteria can be washed out (Schmidt & Ahring 2000; Chong et al. 2012), but this can be minimized and even completely prevented through correct operation and conditions (Seghezzo et al. 1998). The fourth zone is the integrated three phase separator (Sawayama et al. 1999; Schmidt & Ahring 2000; Chong et al. 2012). The liquid will flow through the overflow and gas escapes through a tube (Christensen et al. 1984). This inclusive separator means that the biogas produced can be collected and used as a replacement for fossil fuels in energy production. The reactor is fed from the bottom, and the wastewater flows upwards through the reactor. This upflow and the production of biogas causes mixing and agitation, and hence provides a good contact between the wastewater and the biomass (Seghezzo et al. 1998; Chong et al. 2012). A schematic diagram of the UASB reactor is shown in Figure 5.

The COD removal occurs over the bed and blanket zones of the reactor. The main biochemical reactions occurs in the bed, and further biodegradation occurs in the blanket (Alibhai & Forster 1986; Chong et al. 2012). UASB reactors have high biomass concentrations, and hence the removal of COD is good even at highly concentrated influents, or a

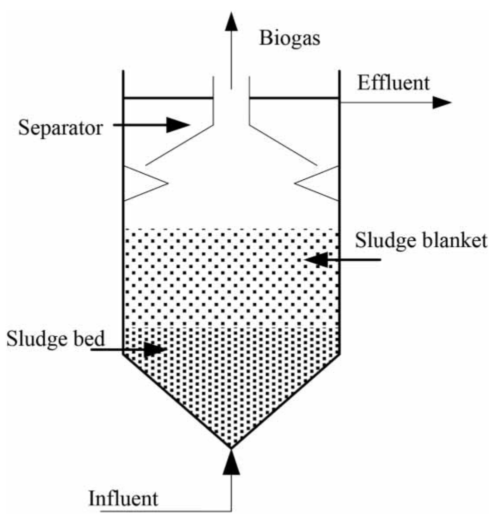

Figure 5 | Schematic of an upflow anaerobic sludge blanket (UASB) reactor.

high volume of waste (Lin & Yang 1991; Liu *et al.* 2003). Rajakumar *et al.* (2011) found that a UASB reactor could achieve a higher removal efficiency than a similar anaerobic filter reactor and Hamdi (1996) found that the UASB could cope with much higher organic loading rates than that of the anaerobic contact process and in general it was much more efficient at treating olive mill wastewaters. Short HRT, and the fact that HRT is independent of SRT in the UASB reactor, means that treatment time is reduced significantly (i.e. from days to hours). To reduce start up times, the reactor may be initially seeded: problems may occur if the seed used is not cultivated on a similar composition to that of the influent to be used (Christensen *et al.* 1984; Chong *et al.* 2012).

Successful operation of a UASB reactor depends greatly on the formation of the sludge granules that make up the sludge blanket. These granules must be well settling and significantly aggregated in order to remain in the bed in spite of the upflow velocity (Seghezzo *et al.* 1998; Schmidt & Ahring 2000; Liu *et al.* 2003). The mechanisms for the development of the granules are complicated and are not fully known, however it is suggested that the bacteria attach themselves to heavy sludge ingredients, and the growth of the bacteria concentrates around these and forms granules (Lin & Yang 1991; Liu *et al.* 2003).

Hybrid upflow anaerobic sludge blanket – anaerobic filter reactor

Guiot & van den Berg (1985) first introduced the idea of a hybrid UASB reactor by keeping the bottom two-thirds of the reactor as a UASB similar to that described, and incorporating an anaerobic filter (AF) in the top third of the reactor to treat sugar wastewater. Figure 6 shows the configuration of the hybrid reactor. A COD removal efficiency of 93% was achieved for OLRs of up to 26 kg COD m^{-3} d^{-1}. The authors also commented that the filter section achieved effective biomass retention. Hutňan *et al.* (1999) compared laboratory-scale UASB and hybrid UASB-AF reactors treating synthetic wastewaters. They found that comparable COD removal efficiency was achieved in both reactors. Biomass washout occurred in the UASB reactor at an OLR of 6 kg COD m^{-3} d^{-1} but did not occur in the hybrid reactor until an OLR of 12 kg COD m^{-3} d^{-1} was applied, which is in agreement with the observation made by Guiot & van den Berg (1985). Similar reactors were also compared by Kalyuzhnyi *et al.* (1997) for the treatment of soft-drinks wastewater. Again, both reactors performed similarly for applied OLRs up to 8 kg COD m^{-3} d^{-1} achieving COD removal efficiencies of over 80%. However, when an OLR of 12 kg COD m^{-3} d^{-1} was applied to both reactors, the efficiency of the UASB dropped to 60–70%, whereas it remained above 80% in the hybrid reactor. This was attributed to the superiority of the hybrid reactor for biomass retention. Using coal wastewater, Ramakrishnan & Surampalli (2012) observed COD removal efficiency for HRT ranging from 0.75 to 0.33 days to both reactors. The UASB reactor exhibited slightly less removal efficiency than the hybrid reactor. For example, at HRT of 0.39 days the removal efficiencies achieved were 88 and 92% in the UASB and hybrid reactors respectively. The authors attributed this to better granulation and biofilm establishment in the hybrid reactor due to the AF section. These studies highlight that UASB reactors can successfully be combined with an anaerobic filter to reduce biomass washout, with no need for an additional unit.

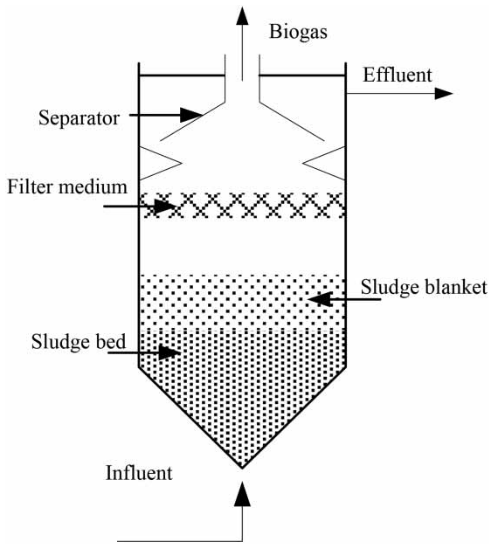

Figure 6 | Schematic of a hybrid UASB-AF reactor.

Expanded granular sludge bed reactor

Low strength influent wastewater can cause a substrate shortage inside anaerobic treatment reactors, and this can cause deterioration of the sludge in the reactor

(Yoochatchaval *et al.* 2008a, 2008b). To counter this, a modification of the UASB reactor can be considered, called an expanded granular sludge bed reactor (EGSB). It is operated in a similar way to the UASB, and the configuration of the reactor is the same as in Figure 5. However, it is operated with a higher upflow velocity, which causes the bed to expand (Seghezzo *et al.* 1998; Kato *et al.* 1999). This can increase the sludge-wastewater contact and hence reduces the problem caused by substrate shortage (Yoochatchaval *et al.* 2008a). The hydraulic mixing in the reactor is intensified and hence the reactor can be considered completely mixed (Seghezzo *et al.* 1998). The increase in upflow velocity can be achieved by effluent recirculation. Some advantages of the EGSB include the high solid retention times and the low accumulation of fats (Núñez & Martínez 1999). This type of reactor has been successfully used to treat wastewaters including: slaughterhouse (Núñez & Martínez 1999), domestic (Chunjuan *et al.* 2009), sucrose-based (Yoochatchaval *et al.* 2008a) and brewery (Xing *et al.* 2009) wastewaters, as well as other low-strength wastewaters (Yoochatchaval *et al.* 2008b). However, the increased upflow velocity required could mean higher operating costs than that of a UASB and taller reactors are required to allow for bed expansion (Kato *et al.* 1999), meaning higher initial costs as well. The COD removal in EGSB has been seen to drop at low temperatures such as 15 °C (Xing *et al.* 2009).

Anaerobic baffled reactor

The anaerobic baffled reactor (ABR) is a high rate anaerobic treatment that can be described as a series of UASBs (Bachmann *et al.* 1985; Hutňan *et al.* 1999; Dama *et al.* 2002; Manariotis & Grigoropoulos 2002; Krishna *et al.* 2007; Gopala Krishna *et al.* 2008; Liu *et al.* 2012) separated by baffles that force the wastewater to flow over and under them as can be seen in Figure 7. A mixed culture of microorganisms (Hutňan *et al.* 1999) form flocs in the upward and downward chambers (Polprasert *et al.* 1992; Manariotis & Grigoropoulos 2002) and these rise and settle with gas production, but their movement through the reactor is slow (Gopala Krishna *et al.* 2008). This means that the wastewater can come into contact with a large amount of active biomass while inside the reactor (Liu *et al.* 2012). The design of an ABR is simple

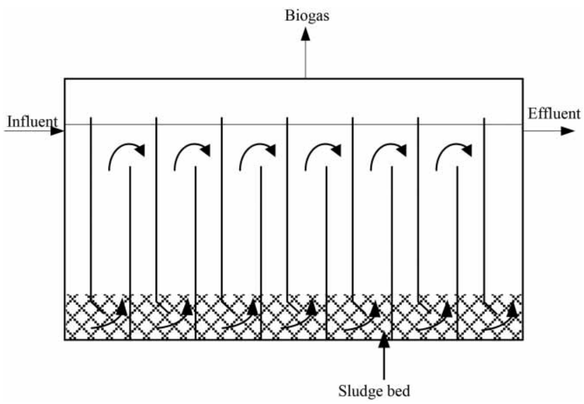

Figure 7 | Schematic of an anaerobic baffled reactor (ABR).

and inexpensive to manufacture (Polprasert *et al.* 1992; Manariotis & Grigoropoulos 2002). However, if large numbers of compartments are needed, costs can rise significantly. The number of compartments in an ABR can vary from three to 11, but 4–5 compartments is generally recommended (Manariotis & Grigoropoulos 2002). Often most of the COD removal takes place in the first two compartments. An advantage of the ABR is that the methanogenesis and acidogenesis often separate into individual compartments, allowing for specific microorganisms to dominate in each compartment (Barber & Stuckey 1999; Hutňan *et al.* 1999; Langenhoff & Stuckey 2000; Dama *et al.* 2002). ABRs are also tolerant to shock loads (Langenhoff & Stuckey 2000) and have a small amount of washout of biological solids (Bachmann *et al.* 1985; Liu *et al.* 2012) compared to similar UASB and hybrid UASB reactors (Hutňan *et al.* 1999). It has been found that ABRs should be avoided if there is a large number of particulates in the wastewater to be treated (Bachmann *et al.* 1985; Barber & Stuckey 1999). The solids can displace the biomass in the reactor and build up inside, causing reduced contact between the wastewater and biomass (Barber & Stuckey 1999). Hutňan *et al.* (1999) found the amount of $NaHCO_3$ that was needed to be added to the ABR is double compared to the UASB and hybrid reactor to maintain the pH. It has also been found that UASB reactors, when compared with ABR, perform better at lower temperatures (Ayaz *et al.* 2012). ABR reactors have been shown to successfully treat low (Barber & Stuckey 1999; Krishna *et al.* 2007), medium (Barber & Stuckey 1999) and high strength (Barber & Stuckey 1999;

Langenhoff & Stuckey 2000) wastewaters as well as slaughterhouse wastewater (Polprasert *et al.* 1992). As with other waste treatment unit operations, the selection of appropriate anaerobic systems is specific to the waste characteristics and regional constraints. Various modifications of anaerobic digestion reactors have been utilized in the past to ensure that the system could cope well with variations in temperature, and organic loading rates (e.g. CSTR), increased solid retention times (e.g. with the addition of settling tanks to CSTRs or utilization of UASB type systems), treatment of complex wastewaters (e.g. AFBR), accommodate for substrate shortage in low strength wastewaters (e.g. EGSBR and ABR).

UPFLOW ANAEROBIC SLUDGE BLANKET REACTOR

Importance

In the past, anaerobic treatments of wastewater were not used in industry since they were not able to handle high organic loading rates and could be unstable in terms of treatment efficiency. The UASB solves these issues as it can handle high organic loading rates and adding a recycle increases the biological population development and hence start-up times are reduced (Christensen *et al.* 1984). It is also simple in design and construction, which gives rise to low initial costs, and has a low energy consumption, and therefore low running costs (Akbarpour Toloti & Mehrdadi 2010; Chen *et al.* 2011; Khan *et al.* 2011; Rajakumar *et al.* 2011; Chong *et al.* 2012). It also exhibits the classic advantages attributed to most anaerobic wastewater treatment processes such as biogas production (Sawayama *et al.* 1999; Kerroum *et al.* 2010), low sludge production (Khan *et al.* 2011; Rajakumar *et al.* 2011) and small land requirements (Christensen *et al.* 1984; Chong *et al.* 2012) compared to aerobic processes. The UASB can also withstand fluctuations in pH and hence no extra chemicals need to be added to maintain a desired pH (Akbarpour Toloti & Mehrdadi 2010; Chen *et al.* 2011). Fluctuations in influent concentration and temperature can also be handled by this reactor. The UASB has a short HRT and a long SRT compared to other processes, and hence can process large amounts of wastewater in a short time (Chong *et al.* 2012).

Experimental studies

Treatment of different sources of wastewater

There have been a wide range of publications related to the treatment of various types of wastewater by UASB reactors, at full, pilot and laboratory scale and some of these have been summarized in Table 1. Parawira *et al.* (2005) found that the full-scale UASB reactor used for their studies treated the wastewater from a brewery brewing opaque beer to a satisfactory quality so that it could be discharged into public waters. This treatment included an average COD removal of 57%; total solids reduction of 50% and settleable solids removal of 90%. They suggested brewery wastewater provided ideal conditions for the UASB influent due to its composition and strength and that the UASB reactor presented a more suitable process for the treatment of brewery wastewater than aerobic processes, due to the excessive amount of energy required for aerobic treatment of high strength wastewaters such as those from breweries. They also found that pilot-scale UASB reactors treating similar wastes could provide higher COD removals, up to 90%. Ahn *et al.* (2001) also studied a full-scale UASB reactor treating brewery wastewater, which was built for pre-treating the wastewater before discharging it to a municipal wastewater treatment plant. The desired treatment level was achieved, giving an average COD removal of 80% over the seven years it was studied. Hack (1985) studied the UASB reactor as a possible pre-treatment for an overloaded aerobic treatment plant on a brewery site. A pilot-scale UASB reactor was built to study its suitability, using the effluent wastewater from the brewery. The concentration of the influent to the reactor was, on average, 1,230 mg COD L^{-1} and it was operated at an average temperature of 22 °C. These conditions were similar to those at which the full-scale reactor would be operated and they achieved a COD reduction of 80%, meaning this anaerobic pre-treatment is a promising solution for extending the overloaded aerobic treatment plant. Cronin & Lo (1998) studied the treatment of brewery wastewater by two pilot-scale UASB reactors seeded with different amounts of activated sludge. The reactor seeded with 5.93 g volatile suspended solids (VSS) per liter of activated sludge achieved a satisfactory performance according to the requirements of the authors, reaching

Table 1 | List of experimental studies related to the treatment of various types of wastewater by UASB reactors, at full, pilot and laboratory scale

Type of wastewater	COD removal efficiency (%)	Hydraulic retention time (hours)	Organic loading rate (kg COD m^{-3} d^{-1})	Comments	Reference
Brewery	75	24	6	Effluent satisfied standards to be discharged into public waters	Parawira *et al.* (2005)
	80	6.7	2.5–7.3	Effluent satisfied standards to be discharged into municipal treatment plant	Ahn *et al.* (2001)
	80	5.6	5–15	Mentioned as a 'very promising solution' for extending the overloaded aerobic treatment plant	Hack (1985)
	91	18	–	Also achieved VSS removal of 81%	Cronin & Lo (1998)
	89.1	4	12	Laboratory-scale reactor achieved a COD removal efficiency of 89.1%	Yan & Tay (1996)
Canning	70–90	–	2–8	Fish canning	Puñal & Lema (1999)
	90–95	35	1.2–3.9	Fruit canning	Wongnoi *et al.* (2007)
	86.3	10	4–12	Fruit canning. Polishing step required to meet requirements	Diamantis *et al.* (2003)
	96	20	4.36	Fruit canning	Trnovec & Britz (1998)
	96	2.66	6.7	Tomato canning	Gohil & Nakhla (2006)
Distillery	85	–	41	Grape wine distillery	Moosbrugger *et al.* (1993)
	90	–	6–11	Alcohol distillery	Akarsubasi *et al.* (2006)
	85	–	8	Alcohol distillery	Akarsubasi *et al.* (2006)
	85	–	1–4.5	Alcohol distillery	Ince *et al.* (2005)
	>90	4–11	4–18	Seasonal alcohol distillery	Wolmarans & De Villiers (2004)
Pharmaceutical	78	24	4.79	Fed with synthetic wastewater at start up	Zheng & Hu (2002)
	77	24	4.76		Zheng & Hu (2002)
	54	10.81	33.7	Chemical synthesis pharmaceutics	Akbarpour Toloti & Mehrdadi (2010)
	86.2–91.6	33–43	6.26–10.33	Herbal pharmaceutics	Satyanarayan *et al.* (2009)
	85	16.8–40.3	12.57–21.02	Chemical synthesis pharmaceutics	Chen *et al.* (2011)
Slaughterhouse	99	19–96	1.27	Below HRT 19 hours gave poor COD removal	Borja *et al.* (1994)
	84	–	1.5	Poultry slaughterhouse	Del Nery *et al.* (2001)
	90	7.2–21.12	1.5	Pig and cattle slaughterhouse. Suspended solid removal 40%	Nacheva *et al.* (2011)
	70–92	18–27	0.86–2.43	Pig and cattle slaughterhouse. Pre-treatment needed to reduce grease concentrations	Miranda *et al.* (2005)
Sewage	85	12	2.1	TSS removal 95%	Coelho *et al.* (2006)
	44	6	1.69–2.88	Low temperature. Post-treatment by CSTR give 66% COD removal	Mahmoud *et al.* (2004)
	54	6	–	Post-treatment by CSTR gives 72% COD removal	Mahmoud (2008)

maximum COD and VSS removals of 91 and 81% respectively at an HRT of 18 hours. Yan & Tay (1996) studied the treatment of brewery wastewater by a laboratory-scale UASB reactor operating at an ambient temperature of 21.8 °C, an OLR of 12 g COD L^{-1} d^{-1} and an HRT of 4 hours, and achieved a COD removal efficiency of 89.1%.

Puñal & Lema (1999) studied the performance of a full-scale UASB reactor treating fish-canning wastewater. They found that it performed best when the influent contained a mixture of mussel- and tuna-processing wastewater compared to when it only contained tuna-processing wastewater: mussel-processing wastewater contains carbohydrates in the majority, which are easily degradable by the reactor. They found that the UASB reactor performed acceptably, yielding COD removal efficiencies of 70–90% at OLR from 2–8 kg COD m^{-3} d^{-1} but this could be achieved only through the control of the alkalinity in the reactor. Wongnoi *et al.* (2007) used pilot-scale UASB reactors to treat the wastewater from a fruit-canning factory and managed to achieve a good COD removal efficiency of between 90 and 95% when running at an HRT of 35 h and an OLR of between 1.2 and 3.9 kg COD m^{-3} d^{-1}. Diamantis *et al.* (2003) studied a pilot-UASB reactor treating wastewater from fruit canning, specifically peach canning. They found that the UASB was very effective in terms of COD removal for the examined wastewater, as it managed to achieve an average of 86.3% COD removal at an HRT of 10 h. The OLR varied between 4 and 12 kg COD m^{-3} d^{-1}. Despite this high removal, a polishing step was needed to satisfy the treatment requirements. Gohil & Nakhla (2006) studied a laboratory-scale UASB reactor for treating tomato-canning wastewater. They reached a maximum removal efficiency of 96% when operating at an HRT of 2.66 days. Moosbrugger *et al.* (1993) studied the treatment of grape wine distillery waste in a laboratory-scale UASB system, and achieved a COD removal efficiency of 85% even at loading rates as high as 41 kg COD m^{-3} d^{-1}. From this study, they concluded that grape wine distillery waste is suitable for treatment in UASB systems. Akarsubasi *et al.* (2006) compared the performance of two full-scale UASB reactors treating alcohol distillery wastewaters of similar composition. They called the two reactors IUASB (refers to the UASB reactor in Istanbul, Turkey (Akarsubasi *et al.* 2006)) and TUASB (refers to the UASB reactor in Tekirdag, Turkey (Akarsubasi *et al.* 2006)), and the latter reactor was seeded with sludge from the former reactor. IUASB performed better in terms of COD removal than TUASB by achieving 90% removal at OLRs between 6 and 11 kg COD m^{-3} d^{-1}. TUASB only reached 60–85% removal at 2.5–8.5 kg COD m^{-3} d^{-1}, the maximum removal of 85% being achieved at 8 kg COD m^{-3} d^{-1}. The authors concluded that the reason for the difference in performance was due to the smaller OLR used in TUASB and recommended that higher OLRs should be used to improve the performance of that reactor. Ince *et al.* (2005) compared the performance of these two reactors with a third reactor CUASB (refers to the UASB reactor in Canakkale, Turkey (Ince *et al.* 2005)) which was also seeded with sludge from IUASB. The CUSAB reactor achieved removal efficiencies of between 70 and 85% when operated with OLRs of between 1 and 4.5 kg COD m^{-3} d^{-1}. The efficiency was highest at the largest loading rate, but was still lower than that of the IUASB and so the authors again recommended a higher loading rate be applied to increase the efficiency. Wolmarans & De Villiers (2004) studied the seasonal treatment of alcohol distillery wastewater by a full-scale UASB reactor. The OLRs applied were between 4 and 18 kg COD m^{-3} d^{-1} and COD removal efficiencies greater than 90% were achieved. They noted that the stop–start nature of seasonal treatment posed no problems for the plant, suggesting that the UASB process should be considered as a viable option for the treatment of seasonal wastes. Zheng & Hu (2002) compared two methods of start-up and subsequent operation of laboratory-scale UASB reactors treating pharmaceutical wastewaters. The first process involved supplying the reactor with a synthetic wastewater to achieve the start-up before swapping to the pharmaceutical wastewater to be treated subsequently. Once the reactor had begun to be supplied with the pharmaceutical wastewater, it achieved a COD removal efficiency of 78% when operated with an OLR of 4.79 kg COD m^{-3} d^{-1}. The second process involved supplying the reactor with pharmaceutical wastewater from the start of operation. Start-up in this way managed to achieve a COD removal efficiency of 77.7% while operating at an OLR of 4.76 kg COD m^{-3} d^{-1}. The authors concluded that both these processes are viable for the start-up and subsequent operation of UASB to treat pharmaceutical wastewater. Akbarpour Toloti & Mehrdadi (2010) also studied a laboratory-scale UASB reactor treating chemical synthesis pharmaceutical wastewater. The UASB reactor was chosen for this particular study as the authors had noted that it had shown itself to be effective in treating high strength wastewaters, and pharmaceutical wastewaters are generally high strength. It was shown that generally an increase in OLR and a decrease in HRT increased the removal efficiency and the optimum operating conditions of an OLR of 10.81

kg COD m^{-3} d^{-1} and an HRT of 33.7 hours yielded a COD removal efficiency of 54%. Satyanarayan *et al.* (2009) set up a pilot-scale UASB reactor for treating wastewater from a herbal pharmaceutical plant. They selected the UASB for reasons in agreement with Akbarpour Toloti & Mehrdadi (2010): herbal pharmaceutical wastewaters are high strength and therefore should be well treated in a UASB system. The reactor achieved COD removal efficiencies of 86.2–91.6% when operating at OLRs between 6.26 and 10.33 kg COD m^{-3} d^{-1} and so the authors concluded that the UASB reactor is feasible for herbal pharmaceutical wastewater treatment, adding that the process is very cost effective and economical. Chen *et al.* (2011) studied the performance of a full-scale UASB reactor, also treating chemical synthesis pharmaceutical wastewater. The OLR for the full-scale reactor was between 12.57 and 33 kg COD m^{-3} d^{-1} and the average COD reduction was 52.2%, with the maximum reaching 85%.

Borja *et al.* (1994) studied a laboratory-scale UASB reactor that was treating slaughterhouse wastewater. The reactor managed to achieve a COD removal efficiency of 99% when operated at an HRT of 4 days and an OLR of 1.27 kg COD m^{-3} d^{-1}. The HRT was then reduced from 4 days and this had no significant effect on the removal efficiency until the HRT reached 0.8 days, where a significant drop of COD removal occurred. However, for HTRs above 0.8 days, the removal remained above 95%, and hence the laboratory-scale UASB was successful in treating the slaughterhouse wastewater. It is also important to study the performance of full-scale reactors to demonstrate the effectiveness of the system in industry applications. A pilot-scale UASB reactor was studied by Nacheva *et al.* (2011) for its performance in treating pig and cattle slaughterhouse wastewater from a slaughterhouse in Mexico. The reactor was operated at ambient temperature and the influent to the reactor was pre-treated for solids separation. They found that the COD removal efficiency increased with increasing OLR, reaching 90% at an OLR of 15 kg COD m^{-3} d^{-1}. In addition, the remaining total suspended solids (TSS) were reduced by more than 40% and retained in the reactor, and this did not affect the performance. Miranda *et al.* (2005) also found that a pre-treatment of the wastewater was needed to reduce the concentrations of oil and grease in the influent to the reactor, the build-up of which led to a system failure before the implementation of the pre-treatment. Once this pre-treatment had been introduced, COD removal efficiencies of between 70 and 92% were achieved when OLRs between 0.86 and 2.43 kg COD m^{-3} d^{-1} were applied. As a reference to the performance of the reactor, the authors also added that for the year after the pre-treatment was introduced, the concentration of COD in the reactor effluent was consistently below the emission standards value. However, the oil and grease concentrations remained above these standards, leading to the need of post-treatment, as also agreed by Nacheva *et al.* (2011).

Coelho *et al.* (2003) investigated the possibilities of using a UASB reactor to replace a septic tank treating sewage from a single family. The pilot-scale reactor reached a COD removal efficiency of 85% and a TSS removal efficiency of 95%, meaning that no post-treatment of the effluent was necessary. These results, when compared to the removal efficiency of the septic tank (75% COD and 75% TSS) and coupled with other benefits of the UASB, such as smaller required area and cost (in Brazil) and little maintenance compared with the ST, led the authors to conclude that the UASB can substitute the septic tank as a single-family wastewater treatment system with significant advantages. Mahmoud (2008) studied the treatment of sewage in two similar systems operating in summer. The UASB-digester system again showed significant improvement of COD removal compared to the single-stage reactor, achieving 72% COD removal efficiency as compared to 54%.

Temperature

The effect of temperature on the efficiency of a UASB process has been studied, with many researchers focusing on the reactor's ability to treat wastewater at low temperatures. Some of these studies have been summarized in Table 2. Uemura *et al.* (1995), however, studied the effect of increasing the temperature of a thermophilic UASB reactor that was already operating at a temperature of 55 °C and a loading rate of 18.3 kg TOC m^{-3} d^{-1}. The wastewater contained sucrose, and as the temperature was raised to 65 °C the efficiency dropped dramatically to around 65% and started to increase slightly with the reduction of loading rate to 10 kg TOC m^{-3} d^{-1}. However, it never returned to its previous

Table 2 | List of experimental studies focusing on the effect of temperature on the efficiency of a UASB process

Temperature (°C)	COD removal efficiency (%)	Organic loading rate (kg COD m^{-3} d^{-1})	Hydraulic retention time (hours)	Comments	Reference
55	>90	18	3.9	Methane production dropped with the increase in temperature	Uemura et al. (1995)
65	65				
30	85	7.2–10.8	4	TSS removal increased with temperature, but not significantly	Farajzadehha et al. (2012)
25	81				
20	75				
32	87	0.8–1.2	6–10	TSS removal was not significantly affected by change in temperature	Singh & Viraraghavan (2003)
20	84				
15	81				
11	79				
6	40				
14–25	41	–	12	Still performed better than septic tank even at low temperature	Elmitwalli & Otterpohl (2011)
30	64		16		

efficiency of over 90%. An increase in methane production (from 100 to 130 L d^{-1}) was observed initially after the increase but then decreased drastically (to 40 L d^{-1}) after a period of time. This behavior, along with the failure to achieve the initial efficiency, was attributed to the disintegration of the sludge at the higher temperature. Farajzadehha et al. (2012) studied a laboratory-scale UASB reactor treating municipal wastewater. The wastewater was enriched with molasses and milk powder to simulate the treatment of municipal wastewater mixed with industrial wastewater (in this case, wastewater from the dairy industry) and the OLR was kept between 7 and 11 kg COD m^{-3} d^{-1}. The temperature of the reactor was decreased from 30 to 25 °C, then further to 20 °C and the COD removal efficiencies were 85, 81 and 75% respectively, indicating that the COD removal decreases with a decrease in temperature. However, 75% is still a high removal efficiency which could be satisfactory for the treatment of the wastewater. The TSS removal was observed to be slightly higher at the higher temperature (30 °C), but this was not a significant difference. The effect of reducing the temperature in a laboratory-scale UASB reactor treating municipal wastewater was also studied by Singh & Viraraghavan (2003). Two reactors were used in this study and were operated with an HRT of 6 hours. The reactors were then studied for COD removal efficiency at 6, 11, 15, 20 and 35 °C. The efficiency decreased with decreasing temperature (79, 81, 84 and 87% for 11, 15, 20 and 35 °C respectively) but these efficiencies were still considered satisfactory. However, when operating at 6 °C the removal efficiency decreased to 40%. The production of biogas accordingly decreased with the temperature. A decrease in COD removal efficiency was also observed when reducing the HRT, and it was noted that this decrease was more severe at lower temperatures. The removal efficiency of TSS was not significantly affected by the decrease in temperature, as agreed with by Farajzadehha et al. (2012). Singh & Viraraghavan (2003) attributed this to the successful formation of granules during start-up, hence causing successful entrapment of TSS. This meant that the effluent from the reactors met the criteria for discharge in terms of suspended solids, and required only a simple aerobic step to meet the criteria in terms of COD, even at temperatures as low as 11 °C. A comparison between two laboratory-scale UASB reactors treating greywater (refers to wastewater produced in households which does not include toilet wastewater, i.e. water from washing and cooking), one operating at ambient temperature (14–25 °C) and one operating at 30 °C, and a septic tank was carried out by Elmitwalli & Otterpohl (2011). They observed that the maximum COD removal efficiency was achieved by the reactor operating at 30 °C (84%), but that the removal achieved in the reactor operating at ambient temperature still greatly exceeded that achieved in the septic tank (79 and 14% respectively).

This leads to the understanding that the UASB reactor is preferable to the septic tank for treating greywater, even at low temperatures. The authors also recommended a lower HRT if operating the UASB at lower temperatures. Recent experiments on a full-scale UASB reactor (unpublished results) at Waterleau showed that although the optimum temperature for operation was at 35 °C in terms of biogas production, no significant reduction in COD removal was observed at lower temperatures of around 20 °C. These studies highlight that the UASB can be used for treatment at a variety of temperatures, though generally higher temperatures yield higher COD removal efficiencies.

Organic loading rates

The organic loading rate (OLR) applied to UASB reactors can affect the performance of the reactor. Some of the studies on the effect of varying the OLR have been summarized in Table 3. Trnovec & Britz (1998) used a laboratory-scale UASB treating canning wastewater to study the effects of changing the OLR. The OLR was increased from 2.28 to 10.95 kg COD m^{-3} d^{-1}. The increase to 3.95 kg COD m^{-3} d^{-1} was achieved by the increase of effluent COD. The increase from 3.95 kg COD m^{-3} d^{-1} to the maximum of 10.95 kg COD m^{-3} d^{-1} was achieved by the decrease of HRT. Up to the OLR of 4.36 kg COD m^{-3} d^{-1}, the removal efficiency showed a definite increase from 88 to 96%. When the OLR was increased further, the efficiency reduced but still remained high (within 88–96%). The rate of removal increased steadily with OLR. Here, the authors showed that a single UASB can cope with a wide variety of OLRs. Kalyuzhnyi *et al.* (1996) also studied a laboratory-scale UASB reactor. This was fed with synthetic wastewater at a wide variety of OLRs between 3.4 and 44.9 kg COD m^{-3} d^{-1}. At OLRs below 11 kg COD m^{-3} d^{-1}, the COD removal efficiency remained above 98%. At higher OLRs the removal efficiency was reduced, but still remained above 96%. However, at these higher OLRs, the time taken to reach the maximum efficiency was up to three times as long as at the lower OLRs. In addition, temporary disturbances such as large gas bubble formation and sludge flotation were observed immediately after a change in OLR. These results show that the UASB reactor is capable of coping with a wide range of OLRs, given the time to reach maximum removal efficiency. González *et al.* (1998) studied a laboratory-scale UASB reactor for the treatment of sugar cane molasses. They increased the OLR in a relatively small range of 2.3–7.1 kg COD m^{-3} d^{-1}. The COD removal increased from 59 to 91% while increasing the OLR from 2.3 to 5.6 kg COD m^{-3} d^{-1}. However at 7.1 kg COD m^{-3} d^{-1},

Table 3 | List of experimental studies focusing on the effect of organic loading rate (OLR) on the efficiency of a UASB process

Organic loading rate (kg COD m^{-3} d^{-1})	COD removal efficiency (%)	Comments	Reference
2.28	88	COD removal increased for ORL up to 4.36 kg COD m^{-3} d^{-1} but decreased after this. Still remained above 88%	Trnovec & Britz (1998)
4.36	96		
10.95	93		
3.4–10.2	>98	At OLR above 10.2 kg COD m^{-3} d^{-1} highest removal efficiencies took a long time to achieve	Kalyuzhnyi *et al.* (1996)
10.2–44.9	>96		
2.3	59	Biogas production also increased with increase in OLR	González *et al.* (1998)
5.6	91		
7.1	78		
10	62	OLR of 30 kg COD m^{-3} d^{-1} considered to be the upper limit	Torkian *et al.* (2003)
18	92		
27	39		
30–40	68		
0.2	96	No extra treatment was needed to satisfy treatment requirements at these OLRs	Zhou *et al.* (2009)
0.8	98		

the removal efficiency dropped to 78%. The authors also noted that the biogas production increased with OLR: a useful observation when considering the utilization of biogas produced. Torkian *et al.* (2003) used a pilot-scale UASB reactor for treating wastewater from a slaughterhouse to study the effect of OLR. The first stage of the study saw the OLR increase from 10 to 18 kg COD m^{-3} d^{-1} and a corresponding increase of COD removal efficiency from 62 to 92%. The authors, in agreement with Kalyuzhnyi *et al.* (1996), noted that after a change in OLR there incurred disturbances within the reactor that led to an initial decrease in removal efficiency. However, the system recovered quickly and hence a general increase in removal efficiency was observed. The second stage of the study saw the OLR increase to 27 kg COD m^{-3} d^{-1}. Again a decrease in removal efficiency was observed immediately after this increase, but once the system had recovered a removal efficiency of 93% was achieved and no adverse effect was caused by the large increase in OLR. Although the OLR was increased to above 30 kg COD m^{-3} d^{-1}, the removal efficiency did not increase above 68% and the methane production decreased at OLRs above 30 kg COD m^{-3} d^{-1}. It was therefore concluded that the upper limit for the OLR was 30 kg COD m^{-3} d^{-1}. However, even at OLR of 40 kg COD m^{-3} d^{-1}, no sludge washout was observed. Zhou *et al.* (2009) studied a full-scale UASB reactor treating distillery wastewater. The primary aim of their study was to observe the effects of low OLRs between 0.2 and 0.8 kg COD m^{-3} d^{-1}. High removal efficiencies (96–99%) were observed even at such low OLRs, and there was a general increase in removal efficiency with an increase in OLR. Also, according to the standards imposed by local authorities for effluent discharge, no extra treatment was needed to achieve the required effluent quality. Recent experiments on a UASB pilot reactor at Waterleau (unpublished results) saw COD removal efficiencies of around 84% at OLR up to 16 kg COD m^{-3} d^{-1}. These results indicate that the UASB reactor technology is able to cope with a variety of loading rates, and can perform with high COD removal efficiencies at low and high OLRs.

Mixing

Pol & Lettinga (1986) explained a need to achieve sufficient contact between the biomass in the reactor and influent wastewater in order to yield satisfactory COD removal, and that mixing caused by the rising of biogas bubbles produced during the process could provide this contact. They suggested that in low-rate UASB reactors, an effluent recycle or mechanical mixer could be added to the reactor configuration if insufficient mixing is achieved by biogas production. Table 4 summarizes the effects of mixing observed during some studies found in the literature. During their experiments, Lettinga *et al.* (1980) included a mechanical mixer in the form of an axle with attached blades in the reactor. The mechanical agitation was kept to a minimum, between 10 and 30 rpm, for 1 minute/10 minutes: any more would be detrimental to the sludge granules. However, after studying the results of COD removal efficiencies, the authors concluded that there seemed to be no need for any extra mixing in addition to that already achieved by biogas production. Nadais *et al.* (2006) suggested a different, but still minimal, mixing regime of 40 rpm for 10 minutes/hour, and this helped to prevent channeling in the reactor. Lin & Yang (1991) agreed that mixing achieves better contact between the sludge and wastewater and minimizes channeling in the reactor, and helps to enhance the treatment efficiency. They suggested that, in general, sufficient mixing can be achieved by the rising gases produced in the reactor, but this may not be enough at the beginning of the process, when gas production is minimal. As such, mechanical mixing may be required, the amount of which should not exceed 30 rpm for 1 minute/10 minutes, in agreement

Table 4 | List of experimental studies indicating the effects of mixing in UASB reactors

Mixing method	Comments	Reference
Mechanical mixer 10–30 rpm 1 min/ 10 min	Concluded that the extra mixing was not needed	Lettinga *et al.* (1980)
Mechanical mixer 40 rpm 10 min/1 h	Prevented channelling in reactor	Nadais *et al.* (2006)
Upflow velocity 100–700 mL h^{-1}	An increase in flowrate increased the mixing and removal efficiency	Quaff & Guha (2011)
Upflow velocity	An increase in flowrate increased mixing and removal efficiency	Peña *et al.* (2006)

with Lettinga *et al.* (1980). They also suggested that an increase in mixing, should it be required, could be achieved by a high hydraulic load, the addition of effluent recycle or the addition of gas recycle. Hickey *et al.* (1991) also suggested the use of an effluent recycle to enhance the mixing if needed, but also stated that the biogas played a greater role in the mixing than the upflow velocity. Quaff & Guha (2011) studied the performance of laboratory-scale UASB reactors treating domestic wastewater at different upflow velocities. They found that, generally, the higher flow (and hence increased mixing) gave an increase in removal efficiency: 86% COD removal was achieved at a flowrate of 700 mL h^{-1} compared to 78% at 100 mL h^{-1}. Statistical tests confirmed that the removal efficiencies were sufficiently different to suggest that the mixing had an effect on the performance. The authors also noted that a higher flowrate encouraged methane production. They concluded that at low flowrates sulfidogenic bacteria is not washed out and hence is able to colonise in the reactor and compete with the methanogenic bacteria. Peña *et al.* (2006) studied a full-scale UASB reactor treating municipal wastewater in which different upflow velocities were applied to increase the mixing in the reactor, and an increase in COD removal was observed with an increase in mixing. Statistical tests confirmed that the hydraulic loading rate, and hence mixing, affected the performance of the process.

pH

Souza (1986) found the optimum pH for anaerobic digestion to be around 7. Any pH below 6.5 or above 7.5 may be harmful to the bacteria. However, they also mention that wastewaters with low and high pH could be treated in UASB reactors, due to the buffer capacity of the process although a pH control system may be needed. There have been some studies on the effect of varying pH in UASB reactors, and some of these are summarized in Table 5. Liu *et al.* (2010) studied three UASB reactors, each operating at an HRT of 6.9 hours and an OLR of 5 kg COD m^{-3} d^{-1}. One of the reactors was kept at a pH of 7, one at 6 and one at 5, and COD removal efficiencies of 89, 49 and 10% respectively were achieved. As well as the drop in removal efficiency, a significant drop in methane production was observed, as well as high accumulation of volatile fatty acids (VFAs). Their results are in agreement with Souza (1986) that the optimum pH is around 7. A laboratory-scale UASB reactor treating winery wastewater was used by Ronquest & Britz (1999) to investigate the effect of low pH on the performance. The effect of pH is particularly important for the treatment of winery wastewaters, as they often have to undergo neutralization which is an extra cost. The HRT in the reactor was maintained at 14 hours and the pH of the influent was reduced from 6.7 to 5.0 in gradual steps. The effluent pH was measured, and was consistently

Table 5 | List of experimental studies indicating the effect of varying pH in UASB reactors

Organic loading rate (kg COD m^{-3} d^{-1})	Hydraulic retention time (hours)	pH	COD removal efficiency (%)	Comments	Reference
5	6.9	5 6 7	10 49 89	Decrease in pH resulted in a decrease in methanogenic activity	Liu *et al.* (2010)
9.09–11.05	14	6.7 5.5 5.0	98 95 88	pH of 5 considered lowest operational pH due to drop in COD removal and slow recovery time	Ronquest & Britz (1999)
6.6	6	6.75 7.20 7.25 7.40 8.30	93 98 98 98 77	UASB reactor operating at 55 °C	Visser *et al.* (1993)
16	–	7.3–7.9 8.1	97 79	At pH of 8.3 serious disintegration of granules occurred causing washout of biomass	Sandberg & Ahring (1992)

around 7 for all influent pH values, which suggests a good buffer capacity in agreement with Souza (1986). However, the COD removal started decreasing significantly at the influent pH of 5 and a slow recovery and drop in biogas production suggested that it was the lowest operational pH value. This meant that the authors could conclude that it is feasible to treat winery wastewater in a UASB reactor with little or no neutralization (depending on the source) and hence reduce processing costs. Visser *et al.* (1993) fed three thermophilic laboratory-scale UASB reactors at 55 °C with synthetic wastewater. They were all operated at HRT of 6 hours and OLR of 6.6 kg COD m^{-3} d^{-1}. The pH in each reactor was varied, so that a maximum pH of 8.3 and a minimum pH of 6.75 were used. At a pH of 8.3, the COD removal was 77%, compared to at pH of 7.2 where the maximum COD removal of 98% was achieved. Sandberg & Ahring (1992) also studied a laboratory-scale UASB reactor. This was used to treat fish-treatment wastewater, and was operated at a loading rate of 16 kg COD m^{-3} d^{-1}. The pH was varied between 7.3 and 8.3. Between pH of 7.3 and 7.9 the COD removal remained around 97%. At pH of 8.1 the removal reduced to 79% and at pH 8.3 disintegration of the sludge granules was reported resulting in washout of the biomass. Methane production also steadily decreased with an increase in pH. These studies indicate that although UASB reactors can cope with low pH relatively well, a high pH (>8) seems to cause serious problems resulting in reduced performance.

UASB systems can be useful in the treatment of high strength wastewaters such as brewery wastewater, carbohydrate-rich food wastes, grape wine distillery wastes, slaughterhouse wastewater and domestic wastes. However, adequate design and careful control and monitoring of the system during operation are required to ensure a successful start-up and subsequent stable operation. Optimum operation of UASB systems is noted to occur with some mixing (sufficient mixing can be achieved by the rising gases produced in the reactor) at temperatures of 35 °C (COD removal decreases with decreased temperatures) and pH of 7 (although UASB reactors can cope well with lower influent pH due to its buffering capacity, but experiences inhibitions at pH values of above 8). The UASB systems generally obtain good COD removal efficiencies, even with varying organic loading rates. However, as with other biological systems, the capacity to efficiently carry out COD removal is greatly dependent on the retention times (SRT, to ensure that sufficient biomass is retained for duration of treatment process; and HRT, to ensure that the liquid waste is in the system long enough to be treated) as dictated by the general sizing and configuration of the reactor. Moreover, it is important to note that in situations where the effluent is required to be discharged into receiving water bodies, organic loading rates require careful selection and, in most cases (depending on the regional environmental standards) a post-treatment step is required to ensure that the final effluent is compliant.

MODELING OF UASB REACTOR

Dimensional modeling

An axial dispersion model was proposed by Singhal *et al.* (1998) to describe the fluid flow through the UASB reactor. The aim of this model is to simplify the previous models describing the fluid flow, as they used multiple compartments inside the reactor. The axial dispersion model uses only two compartments that are each assumed to represent axially dispersed zones. The principles of the model rely on the Peclet number (Pe) of each compartment ($Pe \to \infty$ causes minimum axial dispersion and hence it can be assumed to be plug flow; $Pe \to 0$ causes maximum axial dispersion and hence exhibits ideal well-mixed behavior). It can also be assumed that some liquid will bypass the first zone, and this is included in the model. Peña *et al.* (2006) implemented a combined dispersion-compartmental model to describe the mixing properties inside their full-scale UASB reactor. Residence time distribution (RTD) curves were derived from tracer tests on the UASB for HRTs of 5, 6, 8, and 10 hours. At HRTs of 8 and 10 hours, the curves show that the UASB displayed characteristics of a poorly mixed reactor with dead volumes. However, at the lower HRTs of 5 and 6 hours, the curves showed better mixing in the UASB, and the curves were much closer to that of the theoretical CSTR. This shows that under-loading (longer HRTs) causes less mixing in the UASB. Kalyuzhnyi *et al.* (2005) derived a model that was intended to combine granular sludge dynamics, solid–liquid–gas interactions, biological conversions and liquid-phase equilibrium chemistry.

Their plug flow dispersion model is based on an assumption that the processes involved inside the reactor depend on the distance from the bottom of the reactor, and hence in any given cross-section of the reactor, the process characteristics are constant. The validity of the model was tested with results obtained from the pilot-scale UASB reactor treating cheese whey. It was seen that the model outputs correlated well with experimental data obtained, with the exception of effluent COD concentration at a particular time during the operation of the UASB. The model underestimated the effluent COD concentration, and therefore overestimated the methane production at the corresponding time. The authors conclude this on the simplification of the description of granule transport by the model.

Biokinetic modeling

Mathematical models for a unit operation such as anaerobic digestion aim to virtually replicate the process such that they can provide expert guidance for aiding in system design and process optimization. Most developed anaerobic digestion models incorporate a number of kinetic and stoichiometric expressions, which represent the biological interactions that are hypothetically proposed to occur during the anaerobic digestion process. The hypotheses on which these expressions are based are tested using experimental data that is tailored towards determination of the system response (i.e. simulated predictions get compared to the experimental data). Modeling approaches can either be empirical or mechanistic. Empirical models are based upon observed correlations between the performance of the plant and its main design and operating variables (i.e. established by observation when the mechanisms and/or processes operating in the system are not known or are ignored). In contrast, mechanistic models are based on some conceptualization of the biological/physical mechanisms operating in the system (Sam-Soon *et al.* 1991). In developing the mechanistic model, these determined conceptual mechanisms are used to mathematically formulate the process rates and their stoichiometric interactions with the compounds.

Dynamic modeling of anaerobic digestion has been an active area of research in the past few decades. Based on the Haldane model, a model with a single bacterial population (aceticlastic methanogens) was proposed by Graef & Andrews (1974) in which the conversion of fatty acids into biogas is considered limiting and an increase in fatty acid concentration caused a drop in pH and a rise in undissociated acetic acid concentration. However, the model was also able to predict the digester response to the entry of an external inhibitor. Hill (1982) introduced a model that was developed to describe digestion of manure and animal wastes. The model assumed that methanogenesis depends on the total fatty acids and inhibition by the total fatty acid concentration. According to this model, anaerobic digestion is inhibited with an accumulation of VFAs, causing a decrease in the rate of VFA consumption and leading to acid accumulation. Mosey (1983) introduced a four-population model with one acidogenic reaction, one acetogenic reaction, and two methanation reactions. According to their model, a sudden increase in the OLR is expected to cause an accumulation of VFAs, given that acetogens grow at a slower rate than the acidogens. Angelidaki *et al.* (1993) considered hydrolysis, acidogenesis, acetogenesis, and methanogenesis to describe the behavior of manure-fed digesters. In this model, free ammonia is assumed to inhibit methanogenesis; acetic acid is assumed to inhibit acetogenesis, and total VFA is assumed to inhibit acidogenesis. It must be noted that all biokinetic models described so far consider organic matter as a whole and do not account for the nature of the organic macromolecules in the feed composition.

The Anaerobic Digester Model 1 (ADM1) was developed by the IWA Anaerobic Digestion Task Group (Batstone *et al.* 2002a, 2002b, 2006) and is a generalized model for anaerobic wastewater treatment. The ADM1 includes multiple steps and these describe both the biochemical and physico-chemical processes. The first biochemical process is an extracellular disintegration step, where the complex organic waste disintegrates to carbohydrates, proteins, lipids and inert material. This can include many different processes. The second step is extracellular hydrolysis, where these carbohydrates form sugars, proteins form amino acids and lipids form long-chain fatty acids. This hydrolysis is the rate determining step and both these extracellular steps are assumed to be first order. The sugars and amino acids undergo acidogenesis to form organic acids, hydrogen and carbon dioxide. The organic acids, including the long-chain fatty acids, are converted to acetate, hydrogen and carbon dioxide by acetogenesis. Hydrogen and acetate then undergo methanogenesis. The methane produced, along with the CO_2, forms the biogas which can be used to provide process energy. Figure 8 summarizes these processes. The physico-chemical processes included in the anaerobic digester model are the ion association and dissociation and gas transfer. The ion association and dissociation are very rapid and can be considered as equilibrium processes and hence can be described as algebraic equations. All organic species are described in the model in terms of COD (Batstone *et al.* 2002a, 2002b; Coelho *et al.* 2006), and the nitrogenous (NH_3 and NH_4^+) and inorganic carbon (carbon dioxide and bicarbonate) compounds are described in terms of their molar concentration. In Figure 8, S denotes soluble species and X denotes particulate species. There are many advantages to using a generalized model. It can be applied to full-scale plant design and optimization, provide a common base for any further development of the model to specific applications, be used to compare outcomes to other investigations and includes the physico-chemical system. This last point is important as the physico-chemical transformations can be used to optimize the major performance variables (such as gas flow). It also allows for inhibition factors to be expressed such as pH and dissolved gas concentrations and for a pH control set point to be calculated. Based on these processes, the Task Group (Batstone *et al.* 2002a, 2002b) produced a set of equations used in the modeling of anaerobic processes. A mass balance for each component in the liquid phase is produced, including terms for the kinetic rates of each process and the rate coefficients for the specified component on each process. Mass balances for the gas phase components and rate terms for the transfer of gas components in the reactor to the headspace are also included. The physico-chemical equations are also included in the model, and can be implemented as algebraic or kinetic rate equations.

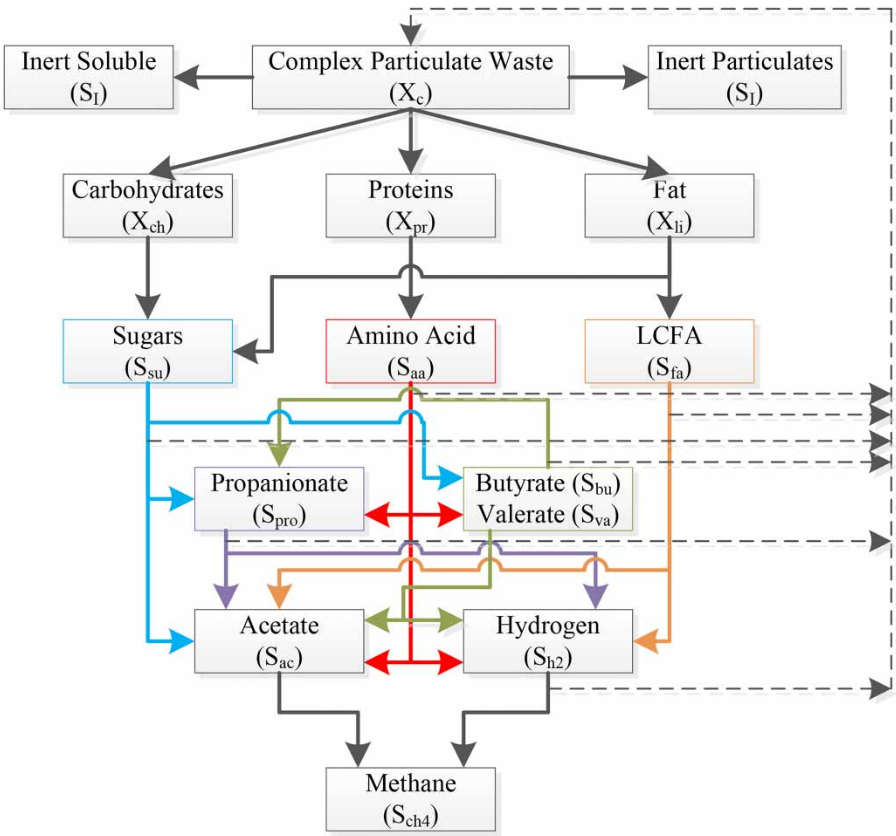

Figure 8 | Process scheme for ADM1 describing both the biochemical and physico-chemical processes adapted from Batstone *et al.* (2002a).

The ADM1 model has been used to model some of the various anaerobic wastewater treatment processes, including CSTR (Kerroum et al. 2010; Thamsiriroj & Murphy 2011) and UASB (Batstone & Keller 2003; Batstone et al. 2005; Coelho et al. 2006; Tejasen & Taruyanon 2010) reactors. Batstone et al. (2005) modeled a laboratory-scale UASB reactor using ADM1. The model was implemented in AQUASIM – a computer program designed for the simulation of aquatic systems (Reichert 1994). The response of the system to a pulse in influent acetate was studied. The model could not simulate the response of the system for a pulse of 0.8 g COD acetate at HRT of 2 days as the response was too small. However, when the pulse was increased to 1.6 g COD acetate and an HRT of 1 day, the response was larger and could be modeled. Using AQUASIM, Coelho et al. (2006) studied the accuracy of ADM1 model for a laboratory-scale UASB reactor treating dairy waste. The methane production was modeled instead of biogas for ease of implementation. The values produced by the model for methane production and effluent COD concentration correlated well with the experimental values, except for on day 80, where unexpected washout from the reactor caused experimental effluent COD concentrations to be larger than those predicted by the simulation. Batstone & Keller (2003) studied the industrial applications of ADM1, including the application to a full-scale UASB reactor treating paper mill wastewater in AQUASIM. The aim was to help towards making a decision whether dosing the reactor with HCl was economically effective: it would decrease the pH and hence decrease the precipitation of calcium carbonate to which this particular system was liable. Real measurements and model predictions of the gas flow from the reactor were plotted to validate the model which showed a good correlation (±10%). Tejasen & Taruyanon (2010) studied a laboratory-scale set up of a CSTR followed by a UASB reactor treating wastewater from an ethanol distillery using ADM1 (implemented in AQUASIM) and included an extension to allow for the reduction of sulfate. The CSTR and UASB were modeled separately, and it was found that the simulated and the experimental results for soluble COD and volatile fatty acids in the effluent correlated well. The sulfate removal was also accurately simulated by the model. However, pH measurements and gas flow from the reactor were simulated at lower values than those obtained experimentally. Despite this, the authors agreed that ADM1 was valid, and also that it is sufficient for application in the design and operation of full-scale systems. It can be concluded that ADM1 has been quite successful in the simulation of UASB processes at laboratory scale, though research into the application of the model to full-scale processes is limited.

ASM-ADM interface

In some of the cases discussed previously (Diamantis et al. 2003; Gohil & Nakhla 2006; Nacheva et al. 2011), the required treatment had not been achieved by the UASB reactor alone and aerobic post-treatment steps had to be incorporated into the process to ensure satisfactory effluent quality. If the UASB process is to be modeled, it would, in these cases, also be useful to be able to model the aerobic post treatment step. The Activated Sludge Models (ASM1, ASM2, ASM2d and ASM3) developed by Henze et al. (2000) are widely used for the modeling of activated sludge treatment of wastewater. Although not a complete general model, the ASM models have grown to incorporate many of the developments in understanding the activated sludge process. The ASM models have been integrated with ADM1 to produce a single model that describes plants involving anaerobic digestion and activated sludge processes, rather than simulating each part individually. Recent approaches to overcome incompatibility problems when integrating different unit processes (in this case activated sludge linked to AD) to develop plant-wide WWTP simulation models include (i) the continuity based interfacing method (CBIM) of Vanrolleghem et al. (2005) and Tejasen & Taruyanon (2010), (ii) the 'supermodel approach' of Jones & Takács (2004, cited in Gujer et al. 1999) and Seco et al. (2004, cited in Gujer et al. 1999), (iii) the transformation based approach of Gujer et al. (1999) and (iv) the mass balances based plant-wide WWTP model approach of Ekama et al. (2006). All of these approaches have been aimed at overcoming the model interfacing difficulties caused by state variable incompatibilities. The ADM1 Scientific and Technical Report (STR) (Batstone et al. 2002b) outlines the two differences between the implementations of the two models as being the units and kinetics. It suggests that leaving the ASM unmodified and changing the ADM1 to use ASM kinetics and units could be a way around these differences and these modifications are included in the ADM1 STR. It also suggests the use of a conversion interface as an alternative way of overcoming these differences. An interface with this purpose has been developed by Yasui *et al.* (2006) for modeling anaerobic sludge digestion of sludge used in an activated sludge process. The results showed that they had successfully integrated ASM and ADM1 to model the water treatment (activated sludge) and sludge treatment (anaerobic) process. Nopens *et al.* (2009) also developed an interface for the combination of ASM1 and ADM1 for the simulation of an activated sludge plant followed by an anaerobic digester used for the treatment of sludge. They produced an interface that did not involve a general term for the input, as this had been used before and had not been successful in accurate modeling. Validation of the model produced, including the developed interface, was carried out and the model predicted realistic behavior of the plant. Kauder *et al.* (2007) combined ADM1 and ASM2d to simulate the aerobic and anaerobic phases in a sequencing batch reactor. Although the simulation results are yet to be compared with the experimental data to validate the accuracy, they have concluded that the model could save considerable time and money when it comes to laboratory work for optimization of the process. As water resource recovery modeling complexity increases, Vanrolleghem *et al.* (2014) have debated on acceptance of mass continuity interface approach as the best approach to connect unit processes in a plant-wide set up or whether adoption of a super model approach (where all components and transformations from the sub-model of each unit process are combined to form the plant-wide model) such as that of Grau *et al.* (2009) would improve the modeling process. The movement towards the supermodel approach would require description of all components according to their elemental composition, which would allow for the virtual tracking of all material elements in the models, using the principle of mass balances for the stoichiometric processes. Because many species are electroactive, Lizarralde *et al.* (2015) favored this supermodel approach that links slower reacting components which are simulated using differential equations to the faster physicochemical reactions which are calculated algebraically at each iteration. Although there appears to be no literature on integrating ADM1 simulating a UASB reactor with ASM, the above studies suggest that the interfacing or supermodel approaches are achievable.

Several anaerobic digestion models have accounted for the interaction between biological reactions of the different organism groups and the weak acid-base chemistry of the aqueous phase in which they function using separated bioprocess stoichiometry and thermodynamic equilibria based on mass and charge balance approaches towards pH prediction (e.g. the IWA ADM1 developed by Batstone *et al.* (2002b) and UCTSDM developed by Sötemann *et al.* (2005)). However, although it is possible to predict or enter the factors that cause organism inhibition (e.g. pH variations), prediction of model instability (hence early warning for failure prevention) is still a challenge in biokinetic modeling of UASB reactors and other anaerobic systems. This is mainly because most earlier anaerobic digestion models are based on one rate-limiting bioprocess (e.g. the growth of methanogens (Hill & Barth 1977) or hydrolysis of complex organics (Sötemann *et al.* 2005; Ikumi *et al.* 2015)). However, to accurately predict anaerobic model dynamics requires models to account for potential multiple rate limiting bioprocesses, which involves carrying out complete calibration for all the anaerobic digestion organism groups. This aspect is being addressed in a separate forthcoming publication by the Water Research Group, University of Cape Town, South Africa. However, recent advances in research and development of mathematical models has given motivation to and greater confidence in the application of anaerobic digestion. Improved models hold the promise of being capable of optimized operation of UASB and other anaerobic systems, including the prediction of model instability before it occurs and so prevent failure. Studies on anaerobic digestion systems have resulted in the development of explicit equations that could be integrated for the construction of mathematical models, by using fundamental kinetic relationships and application of material mass balances (McCarty & Mosey 1991; Ikumi *et al.* 2015).

EXAMPLE CASE STUDY: MODELING OF A LABORATORY-SCALE UASB

A semi pilot-scale upflow anaerobic sludge bed (UASB) reactor treating primary municipal sludge was studied at Waterleau R&D centre at Wespelaar (Belgium). The aim of the test case study was to implement ADM1 using

AQUASIM (Reichert 1994). By adding variables and processes and defining the compartments in which these processes occur, AQUASIM is able to simulate values for parameters such as COD, VFA and pH over a given period of time. AQUASIM has previously been used in literature for the implementation of ADM1. Ideal CSTR compartments were used in this implementation and the kinetic and stoichiometric parameters used were the ones recommended in the main body of the ADM1 STR (Batstone et al. 2002b). Those that were suggested to be most likely to be sensitive to the process in the STR were then subject to a sensitivity analysis, to identify the most important parameter values to be adjusted to give the best agreement between simulated and experimental results. This was done in AQUASIM using the absolute-relative sensitivity function. Those with the highest sensitivity were then subjected to parameter estimation which uses the weighted least-squares method. AQUASIM uses the secant method to convert the nonlinear problem to a linear one to allow for parameter estimation (Ding et al. 2010). The layout used in AQUASIM to model the UASB is shown in Figure 9(a). It is implemented by assuming a perfectly mixed reactor followed by a settler, to represent the settling zone. The recycle has also been included. The addition of the recycle stream is needed for a more realistic prediction of the results, as is the addition of the settling zone compartment. Based on a sensitivity analysis, the parameters chosen for estimation were $f_{pr,xc}$, $f_{si,xc}$, $f_{xi,xc}$, k_{dis}, $k_{m,ac}$ and $K_{s,ac}$ and these are summarized in Table 6. Values of influent flow, soluble COD, particulate COD, reactor volume and temperature were used to form the inputs to the model, as well as the concentrations of VFAs and ammonium in the influent. Figures 9(b)–9(e) show the experimental and simulated results of the laboratory-scale UASB reactor, plotted together for model validation. Figures 9(b)–9(e) indicate that the model can generally predict the trend of each of COD (soluble and total), VFA and biogas production respectively. However, there is generally a small over-prediction in the COD and biogas simulations and an under-prediction in the VFA simulation. This could be due to an inaccurate experimental measurement or the composition of influent. The influent fractions of carbohydrates, proteins and lipids were assumed to be similar to that found in literature for brewery wastewater (Ahn et al. 2001) and therefore more accurate results may be obtained if the influent had been examined for these fractions. The anomalous experimental result at day 61 for VFA and total COD was due to the reactor tending towards failure over a weekend where there was no measurements taken, due to a sharp reduction in temperature to 13 °C. This caused the methanogens to become less active, meaning an increase in VFA production. At day 21, there was a problem with the influent pump and there was no new influent being pumped through the reactor, except that from the recycle. It is suggested that this caused the anomalous results at days 21 and 22. It must be noted that due to problems with the measurement of biogas in the experiment the result of the experiment biogas production should be treated with caution.

CONCLUSIONS

The principles and effectiveness of various anaerobic treatment processes have been outlined, including the upflow anaerobic sludge blanket reactor. The importance of the UASB reactor has been discussed as well as its presence in industry. Various studies on the factors affecting the performance of the UASB process have led to some conclusions to be made surrounding these factors. The UASB has been successful in treating brewery, canning, distillery, pharmaceutical and slaughterhouse wastewaters, as well as sewage. It has been used successfully at temperatures in the range of 11–55 °C, but reactors operating with temperatures at the higher end of the range generally performed better. Organic loading rates of between 0.2 and 44.9 kg COD m^{-3} d^{-1} have been applied to a variety of UASB reactors, providing generally high removal efficiencies. For OLRs at the higher end of the range, the highest removal efficiencies were only achieved after a long period of time. Mechanical mixing is generally avoided in UASB reactors, as rising biogas and upflow velocities provide sufficient contact between wastewater and biomass. However, at times when this has not been enough, minimal mechanical mixing has been applied. UASB reactors have been shown to operate best at around 7 pH, but can cope with a slightly lower value such as 6.5 pH as well. pH values higher than 8 have been shown to cause problems in UASB reactors. Models specific to UASB reactors have been developed to

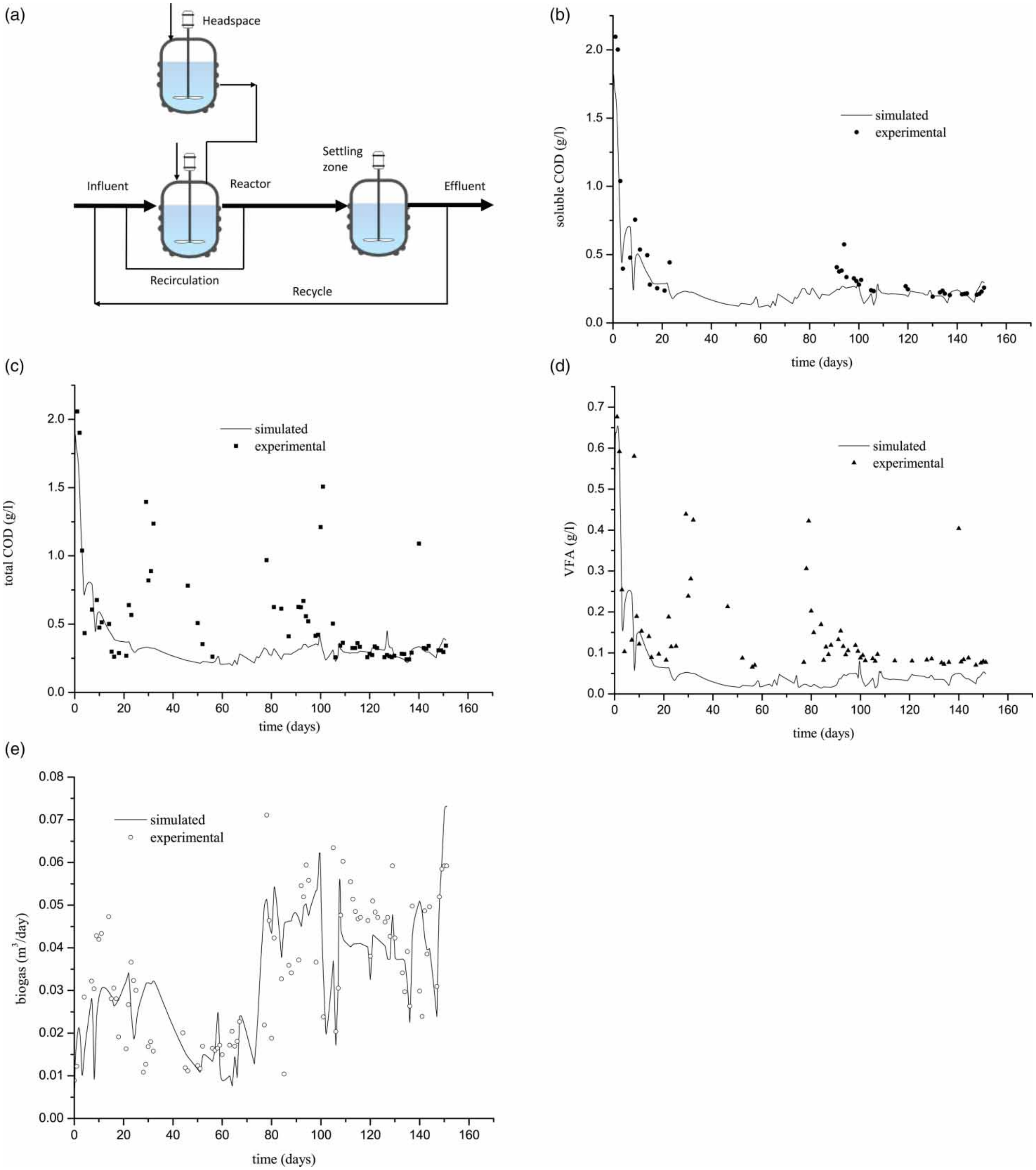

Figure 9 | (a) Layout used in the AQUASIM implementation of ADM1 adapted from Tejasen & Taruyanon (2010). (b) Comparison of experimental and simulated results for effluent soluble COD from the AQUASIM implementation. (c) Comparison of experimental and simulated results for effluent total COD from the AQUASIM implementation. (d) Comparison of experimental and simulated results for effluent VFA from the AQUASIM implementation. (e) Comparison of experimental and simulated results for biogas production from the AQUASIM implementation.

Table 6 | Values obtained from the parameter estimation using AQUASIM

Parameter	Initial value	Final value	Unit
$f_{pr,xc}$	0.2	1	–
$f_{si,xc}$	0.1	0.000505	–
$f_{xi,xc}$,	0.25	0.664	–
k_{dis}	0.4	2.03	d^{-1}
$k_{m,ac}$	8	30	d^{-1}
$K_{s,ac}$	0.15	0.284	Kg COD m^{-3}

describe flow patterns, mixing patterns and effluent characteristics successfully. ADM1 is a general model to describe anaerobic digestion and has been applied to pilot-scale UASB reactors with fairly good correlation to experimental values. The implementation of ADM1 in AQUASIM successfully provides simulated results for effluent soluble and total COD and effluent VFA, except for at times where there was a sharp reduction in temperature as the model assumed constant temperature at 32 °C.

ACKNOWLEDGEMENTS

The authors are grateful to Ron Gerards, Vice President (Technology) of Waterleau NV Wespelaar (Belgium) for providing the pilot-scale data of the UASB. CD is grateful to Gilbert Shama from the Department of Chemical Engineering, Loughborough University, UK for arranging a research stay at KU Leuven Campus Groep T through the ERASMUS exchange program.

REFERENCES

Abbasi, T. & Abbasi, S. A. 2012 Formation and impact of granules in fostering clean energy production and wastewater treatment in upflow anaerobic sludge blanket (UASB) reactors. *Renew. Sustain. Energy Rev.* **16** (3), 1696–1708.

Ahn, Y.-H., Min, K.-S. & Speece, R. E. 2001 Full Scale UASB reactor performance in the brewery industry. *Environ. Technol.* **22** (4), 463–476.

Akarsubasi, A. T., Ince, O., Oz, N. A., Kırdar, B. & Ince, B. K. 2006 Evaluation of performance, acetoclastic methanogenic activity and archaeal composition of full-scale UASB reactors treating alcohol distillery wastewaters. *Process Biochem.* **41** (1), 28–35.

Akbarpour Toloti, A. & Mehrdadi, N. 2010 Wastewater treatment from antibiotics plant. *Int. J. Environ. Res.* **5** (1), 241–246.

Alibhai, K. R. K. & Forster, C. F. 1986 Physicochemical and biological characteristics of sludges produced in anaerobic upflow sludge blanket reactors. *Enzyme Microb. Technol* **8** (10), 601–606.

Angelidaki, I., Ellegaard, L. & Ahring, B. K. 1993 A mathematical model for dynamic simulation of anaerobic digestion of complex substrates: focusing on ammonia inhibition. *Biotechnol. Bioeng.* **42** (2), 159–166.

Ayaz, S. Ç., Akça, L., Aktaş, Ö., Fındık, N. & Öztürk, İ. 2012 Pilot-scale anaerobic treatment of domestic wastewater in upflow anaerobic sludge bed and anaerobic baffled reactors at ambient temperatures. *Desalination Water Treat.* **46** (1–3), 60–67.

Bachmann, A., Beard, V. L. & McCarty, P. L. 1985 Performance characteristics of the anaerobic baffled reactor. *Water Res.* **19** (1), 99–106.

Barber, W. P. & Stuckey, D. C. 1999 The use of the anaerobic baffled reactor (ABR) for wastewater treatment: a review. *Water Res.* **33** (7), 1559–1578.

Batstone, D. J. & Keller, J. 2003 Industrial applications of the IWA anaerobic digestion model No. 1 (ADM1). *Water Sci. Technol.* **47** (12), 199–206.

Batstone, D. J., Keller, J., Angelidaki, I., Kalyuzhnyi, S. V., Pavlostathis, S. G., Rozzi, A. & Vavilin, V. A. 2002a *Anaerobic Digestion Model No.1 (ADM1)*. IWA Publishing, London.

Batstone, D. J., Keller, J., Angelidaki, I., Kalyuzhnyi, S. V., Pavlostathis, S. G., Rozzi, A. & Vavilin, V. A. 2002b The IWA anaerobic digestion model no. 1(ADM 1). *Water Sci. Technol.* **45** (10), 65–73.

Batstone, D. J., Hernandez, J. L. A. & Schmidt, J. E. 2005 Hydraulics of laboratory and full-scale upflow anaerobic sludge blanket (UASB) reactors. *Biotechnol. Bioeng.* **91** (3), 387–391.

Batstone, D. J., Keller, J. & Steyer, J. P. 2006 A review of ADM1 extensions, applications, and analysis: 2002–2005. *Water Sci. Technol.* **54** (4), 1–10.

Borja, R., Banks, C. J. & Wang, Z. 1994 Performance and kinetics of an upflow anaerobic sludge blanket (UASB) reactor treating slaughterhouse wastewater. *J. Environ. Sci. Health A Environ. Sci. Eng. Toxicol.* **29** (10), 2063–2085.

Boušková, A., Dohányos, M., Schmidt, J. E. & Angelidaki, I. 2005 Strategies for changing temperature from mesophilic to thermophilic conditions in anaerobic CSTR reactors treating sewage sludge. *Water Res.* **39** (8), 1481–1488.

Cakir, F. Y. & Stenstrom, M. K. 2007 Anaerobic treatment of low strength wastewater. In: *Proceedings of the Water Environment Federation*. WEFTEC 2007. Water Environment Federation, San Diego, USA, pp. 5178–5207.

Carbajo, J. B., Boltes, K. & Leton, P. 2010 Treatment of phenol in an anaerobic fluidized bed reactor (AFBR). Continuous and batch regime. *Biodegradation* **21** (4), 603–613.

Chamy, R., León, C., Vivanco, E., Poirrier, P. & Ramos, C. 2011 Anaerobic monodigestion of poultry manure. Determination of operational parameters for CSTR. *Water Sci. Technol.* **65** (1), 53–59.

Chen, Z., Wang, H., Chen, Z., Ren, N., Wang, A., Shi, Y. & Li, X. 2011 Performance and model of a full-scale up-flow anaerobic sludge blanket (UASB) to treat the pharmaceutical wastewater containing 6-APA and amoxicillin. *J. Hazard. Mater.* **185** (2–3), 905–913.

Chian, E. S. K. & DeWalle, F. B. 1977 Treatment of high strength acidic wastewater with a completely mixed anaerobic filter. *Water Res.* **11** (3), 295–304.

Chong, S., Sen, T. K., Kayaalp, A. & Ang, H. M. 2012 The performance enhancements of upflow anaerobic sludge blanket (UASB) reactors for domestic sludge treatment – A state-of-the-art review. *Water Res.* **46** (11), 3434–3470.

Christensen, D. R., Gerick, J. A. & Eblen, J. E. 1984 Design and operation of an upflow anaerobic sludge blanket reactor. *J. Water Pollut. Contr. Fed.* **56** (9), 1059–1062.

Chunjuan, D., Xiao, L., Suyun, C., Yanxia, W., Qingliang, Z. & Bingnan, L. 2009 Anaerobic treatment of actual domestic wastewater with EGSB reactor at ambient temperature. In: *3rd International Conference on Bioinformatics and Biomedical Engineering, ICBBE 2009*. IEEE, Beijing.

Coelho, A. L., do Nascimento, M. B., Cavalcanti, P. F. & van Haandel, A. 2003 The UASB reactor as an alternative for the septic tank for on-site sewage treatment. *Water Sci. Technol.* **48** (11–12), 221–226.

Coelho, N., Capela, I. & Droste, R. L. 2006 Application of ADM1 to a UASB treating complex wastewater in different feeding regimes. In: *Proceedings of the Water Environment Federation*. WEFTEC. Water Environment Foundation, Texas, USA, pp. 7123–7135.

Converti, D. A., Zilli, D. M., Borghi, P. M. D. & Ferraiolo, P. G. 1990 The fluidized bed reactor in the anaerobic treatment of wine wastewater. *Bioprocess Eng.* **5** (2), 49–55.

Cronin, C. & Lo, K. V. 1998 Anaerobic treatment of brewery wastewater using UASB reactors seeded with activated sludge. *Bioresour. Technol.* **64** (1), 33–38.

Daigger, G. T. 2011 A practitioner's perspective on the uses and future developments for wastewater treatment modelling. *Water Sci. Technol.* **63** (3), 516–526.

Dama, P., Bell, J., Foxon, K. M., Brouckaert, C. J., Huang, T., Buckley, C. A., Naidoo, V. & Stuckey, D. C. 2002 Pilot scale study of an anaerobic baffled reactor for the treatment of domestic wastewater. *Water Sci. Technol.* **46** (9), 263–270.

Del Nery, V., Damlanovic, M. H. Z. & Barros, F. G. 2001 The use of upflow anaerobic sludge blanket reactors in the treatment of poultry slaughterhouse wastewater. *Water Sci. Technol.* **44** (4), 83–88.

Diamantis, V. I., Parisis, A. & Aivazidis, A. 2003 Anaerobic/aerobic treatment of peach canning watewater. In: *Proceedings of the 8th International Conference on Environmental Science and Technology (CEST2003)*. IOP Publishing Ltd, Lemnos Island, Greece, pp. 245–252.

Ding, P., Hoyle, C. E. S. & Borsuk, M. E. 2010 *Parameter Optimization and Bayesian Inference in an Estuarine Eutrophication Model of Intermediate Complexity*. International Environmental Modelling and Software Society (iEMSs), Ottawa, Canada.

Ekama, G. A., Sötemann, S. W. & Wentzel, M. C. 2006 Mass balance-based plant-wide wastewater treatment plant models – Part 3: Biodegradability of activated sludge organics under anaerobic conditions. *Water SA* **32** (3), 287–296.

Elmitwalli, T. & Otterpohl, R. 2011 Grey water treatment in upflow anaerobic sludge blanket (UASB) reactor at different temperatures. *Water Sci. Technol.* **64** (3), 610–617.

Endo, G. & Tohya, Y. 1988 Ecological study on anaerobic sludge bulking caused by filamentous bacterial growth in an anaerobic contact process. *Water Sci. Technol.* **20** (11–12), 205–211.

Farajzadehha, S., Mirbagheri, S. A., Farajzadehha, S. & Shayegan, J. 2012 Lab scale study of HRT and OLR optimization in UASB reactor for pretreating fortified wastewater in various operational temperatures. *APCBEE Proc.* **1**, 90–95.

Gohil, A. & Nakhla, G. 2006 Treatment of tomato processing wastewater by an upflow anaerobic sludge blanket–anoxic–aerobic system. *Bioresour. Technol.* **97** (16), 2141–2152.

González, J. S., Rivera, A., Borja, R. & Sánchez, E. 1998 Influence of organic volumetric loading rate, nutrient balance and alkalinity: COD ratio on the anaerobic sludge granulation of an UASB reactor treating sugar cane molasses. *Int. Biodeterior. Biodegradation* **41** (2), 127–131.

Good, P., Moudry, R. & Fluri, P. 1982 Use of fixed film and CSTR reactor for anaerobic treatment of stillage of wood hydrolysate. *Biotechnol. Lett.* **4** (9), 595–600.

Gopala Krishna, G. V. T., Kumar, P. & Kumar, P. 2008 Treatment of low strength complex wastewater using an anaerobic baffled reactor (ABR). *Bioresour. Technol.* **99** (17), 8193–8200.

Graef, S. P. & Andrews, J. F. 1974 Stability and control of anaerobic digestion. *J. Water Pollution Contr. Fed.* **46** (4), 667–682.

Grau, P., Copp, J., Vanrolleghem, P. A., Takács, I. & Ayesa, E. 2009 A comparative analysis of different approaches for integrated WWTP modelling. *Water Sci. Technol.* **59** (1), 141–147.

Guiot, S. R. & van den Berg, L. 1985 Performance of an upflow anaerobic reactor combining a sludge blanket and a filter treating sugar waste. *Biotechnol. Bioeng.* **27** (6), 800–806.

Gujer, W., Henze, M., Mino, T. & van Loosdrecht, M. C. M. 1999 Activated Sludge Model No. 3. *Water Sci. Technol.* **39** (1), 183–193.

Habeeb, S. A., Latiff, A. A. B. A. & Ahmad, Z. B. 2010 A review on properties of the digestion process in the up-flow anaerobic sludge bed (UASB) reactor. *Can. J. Environ. Construct. Civil Eng.* **1** (3), 58–70.

Hack, P. J. F. M. 1985 Application of the UASB reactor for anaerobic treatment of brewery effluent. *Water Sci. Technol.* **17** (8), 1489–1490.

Hamdi, M. 1996 Anaerobic digestion of olive mill wastewaters. *Process Biochem.* **31** (2), 105–110.

Hamdi, M. & Garcia, J. L. 1991 Comparison between anaerobic filter and anaerobic contact process for fermented olive mill wastewaters. *Bioresour. Technol.* **38** (1), 23–29.

Haroun, M. & Idris, A. 2009 Treatment of textile wastewater with an anaerobic fluidized bed reactor. *Desalination* **237** (1–3), 357–366.

Heijnen, J. J., Mulder, A., Enger, W. & Hoeks, F. 1989 Review on the application of anaerobic fluidized bed reactors in wastewater treatment. *Chem. Eng. J.* **41** (3), B37–B50.

Henze, M., Gujer, W., Mino, T. & Van Loosdrecht, M. 2000 *Activated Sludge Models: ASM1, ASM2, ASM2d and ASM3*. IAWPRC scientific and technical reports no. 9, IAWPRC. IWA Publishing, London.

Hickey, R. F., Wu, W. M., Veiga, M. C. & Jones, R. 1991 Start-up, operation, monitoring and control of high-rate anaerobic treatment systems. *Water Sci. Technol.* **24** (8), 207–255.

Hill, D. T. 1982 A comprehensive dynamic model for animal waste methanogenesis. *Trans. Am. Soc. Agric. Eng.* **25**, 2129–2143.

Hill, D. T. & Barth, C. L. 1977 A dynamic model for simulation of animal waste digestion. *J. Water Pollution Contr. Fed.* **49** (10), 2129–2143.

Hutňan, M., Drtil, M., Mrafková, L., Derco, J. & Buday, J. 1999 Comparison of startup and anaerobic wastewater treatment in UASB, hybrid and baffled reactor. *Bioprocess Eng.* **21** (5), 439–445.

Ikumi, D. S., Harding, T. H., Brouckaert, C. J. & Ekama, G. A. 2015 *Plant-wide Integrated Biological, Chemical and Physical Bioprocesses Modelling of Wastewater Treatment Plants in 3 Phases (Aqueous-Gas-Solid)*. Research Report W136, Dept. of Civil Eng., University of Cape Town, Rondebosch, South Africa.

Ince, O., Kolukirik, M., Oz, N. A. & Ince, B. K. 2005 Comparative evaluation of full-scale UASB reactors treating alcohol distillery wastewaters in terms of performance and methanogenic activity. *J. Chem. Technol. Biotechnol.* **80** (2), 138–144.

Jones, R. & Takács, I. 2004 Modeling the impact of anaerobic digestion on the overall performance of biological nutrient removal wastewater treatment plants. *Proceedings of the Water Environment Federation* 244–257.

Kalyuzhnyi, S. V., Sklyar, V. I., Davlyatshina, M. A., Parshina, S. N., Simankova, M. V., Kostrikina, N. A. & Nozhevnikova, A. N. 1996 Organic removal and microbiological features of UASB-reactor under various organic loading rates. *Bioresour. Technol.* **55** (1), 47–54.

Kalyuzhnyi, S. V., Saucedo, J. V. & Martinez, J. R. 1997 The anaerobic treatment of soft drink wastewater in UASB and hybrid reactors. *Appl. Biochem. Biotechnol.* **66** (3), 291–301.

Kalyuzhnyi, S. V., Fedorovich, V. V. & Lens, P. 2005 Dispersed plug flow model for upflow anaerobic sludge bed reactors with focus on granular sludge dynamics. *J. Ind. Microbiol. Biotechnol.* **33** (3), 221–237.

Kato, M. T., Rebac, S. & Lettinga, G. 1999 Anaerobic treatment of low-strength brewery wastewater in expanded granular sludge bed reactor. *Appl. Biochem. Biotechnol.* **76** (1), 15–32.

Kauder, J., Boes, N., Pasel, C. & Herbell, J.-D. 2007 Combining models ADM1 and ASM2d in a sequencing batch reactor simulation. *Chem. Eng. Technol.* **30** (8), 1100–1112.

Kerroum, D., Mossaab, B.-L. & Hassen, M. A. 2010 Use of ADM1 model to simulate the anaerobic digestion process used for sludge waste treatment in thermophilic conditions. *Turkish J. Eng. Env. Sci* **34**, 121–129.

Khan, A. A., Gaur, R. Z., Tyagi, V. K., Khursheed, A., Lew, B., Mehrotra, I. & Kazmi, A. A. 2011 Sustainable options of post treatment of UASB effluent treating sewage: a review. *Resour. Conserv. Recycl.* **55** (12), 1232–1251.

Krishna, G. V. T. G., Kumar, P. & Kumar, P. 2007 Complex wastewater treatment using an anaerobic baffled reactor. *Environ. Progr.* **26** (4), 391–398.

Langenhoff, A. A. M. & Stuckey, D. C. 2000 Treatment of dilute wastewater using an anaerobic baffled reactor: effect of low temperature. *Water Res.* **34** (15), 3867–3875.

Lettinga, G., van Velsen, A. F. M., Hobma, S. W., de Zeeuw, W. & Klapwijk, A. 1980 Use of the upflow sludge blanket (USB) reactor concept for biological wastewater treatment, especially for anaerobic treatment. *Biotechnol. Bioeng.* **22** (4), 699–734.

Lin, K. & Yang, Z. 1991 Technical review on the UASB process. *Int. J. Environ. Stud.* **39** (3), 203–222.

Liu, Y., Xu, H.-L., Yang, S.-F. & Tay, J.-H. 2003 Mechanisms and models for anaerobic granulation in upflow anaerobic sludge blanket reactor. *Water Res.* **37** (3), 661–673.

Liu, B., Wu, W., Zhao, Y., Gu, X., Li, S., Zhang, X. & Yang, S. 2010 Effects of ethanol/ratio and pH on mesophilic sulfate reduction in UASB reactors. *Afr. J. Microbiol. Res.* **4** (21), 2215–2222.

Liu, R., Tian, Q. & Chen, J. 2012 The developments of anaerobic baffled reactor for wastewater treatment: a review. *Afr. J. Biotechnol.* **9** (11), 1535–1542.

Lizarralde, I., Fernandez-Arevalo, T., Brouckaert, C., Vanrolleghem, P. A., Ikumi, D., Ekama, D., Ayesa, E. & Grau, P. 2015 A new general methodology for incorporating physico-chemical transformations into multiphase wastewater treatment process models. *Water Res.* **74**, 239–256.

Mahmoud, N. 2008 High strength sewage treatment in a UASB reactor and an integrated UASB-digester system. *Bioresour. Technol.* **99** (16), 7531–7538.

Mahmoud, N., Zeeman, G., Gijzen, H. & Lettinga, G. 2003 Solids removal in upflow anaerobic reactors, a review. *Bioresour. Technol.* **90** (1), 1–9.

Mahmoud, N., Zeeman, G., Gijzen, H. & Lettinga, G. 2004 Anaerobic sewage treatment in a one-stage UASB reactor and a combined UASB-Digester system. *Water Res.* **38** (9), 2348–2358.

Manariotis, I. D. & Grigoropoulos, S. G. 2002 Low-strength wastewater treatment using an anaerobic baffled reactor. *Water Environ. Res.* **74** (2), 170–176.

Manariotis, I. D. & Grigoropoulos, S. G. 2006a Municipal-wastewater treatment using upflow-anaerobic filters. *Water Environ. Res.* **78** (3), 233–242.

Manariotis, I. D. & Grigoropoulos, S. G. 2006b Anaerobic filter treatment of municipal wastewater: biosolids behavior. *J. Environ. Eng.* **132** (1), 23–31.

Manariotis, I. D. & Grigoropoulos, S. G. 2008 Restart of anaerobic filters treating low-strength wastewater. *Bioresour. Technol.* **99** (9), 3579–3589.

Maragno, A. L. F. C. & Campos, J. R. 1992 Treatement of wastewater with a low concentration of organics using and anaerobic fluidized bed reactor. *Water Sci. Technol.* **25** (7), 179–191.

Marín, P., Alkalay, D., Guerrero, L., Chamy, R. & Schiappacasse, M. C. 1999 Design and startup of an anaerobic fluidized bed reactor. *Water Sci. Technol.* **40** (8), 63–70.

McCarty, P. L. 1964 Anaerobic waste treatment fundamentals. *Public Works* **95** (9), 107–112.

McCarty, P. L. & Mosey, F. E. 1991 Modelling of anaerobic digestion processes (a discussion of concepts). *Water Sci. Technol.* **24** (8), 17–33.

Méndez-Acosta, H. O., Snell-Castro, R., Alcaraz-González, V., González-Álvarez, V. & Pelayo-Ortiz, C. 2010 Anaerobic treatment of tequila vinasses in a CSTR-type digester. *Biodegradation* **21** (3), 357–363.

Mendonça, N. M., Niciura, C. L., Gianotti, E. P. & Campos, J. R. 2004 Full scale fluidized bed reactor for domestic wastewater treatment. performance, sludge production and biofilm. *Water Sci. Technol.* **49** (11–12), 319–325.

Miranda, L. A. S., Henriques, J. A. P. & Monteggia, L. O. 2005 A full-scale UASB reactor for treatment of pig and cattle slaughterhouse wastewater with a high oil and grease content. *Braz. J. Chem. Eng.* **22** (4), 601–610.

Moosbrugger, R. E., Wentzel, M. C., Ekama, G. A. & Marais, G. v. R. 1993 Grape wine distillery waste in UASB systems – feasibility, alkalinity requirements and pH control. *Water SA* **19** (1), 53–68.

Morgenroth, E., Arvin, E. & Vanrolleghem, P. 2002 The use of mathematical models in teaching wastewater treatment engineering. *Water Sci. Technol.* **45** (6), 229–233.

Mosey, F. E. 1983 Mathematical modeling of the anaerobic digestion process: regulatory mechanisms for the formation of short-chain volatile acids from glucose. *Water Sci. Technol.* **15** (8–9), 209–232.

Nacheva, P. M., Pantoja, M. R. & Serrano, E. A. L. 2011 Treatment of slaughterhouse wastewater in upflow anaerobic sludge blanket reactor. *Water Sci. Technol.* **63** (5), 877–884.

Nadais, H., Capela, I., Arroja, L. & Duarte, A. 2006 Treatment of dairy wastewater in UASB reactors inoculated with flocculent biomass. *Water SA* **31** (4), 603–608.

Nähle, C. 1991 The contact process for the anaerobic treatment of wastewater: technology, design and experiences. *Water Sci. Technol.* **24** (8), 179–191.

Nopens, I., Batstone, D. J., Copp, J. B., Jeppsson, U., Volcke, E., Alex, J. & Vanrolleghem, P. A. 2009 An ASM/ADM model interface for dynamic plant-wide simulation. *Water Res.* **43** (7), 1913–1923.

Núñez, L. A. & Martínez, B. 1999 Anaerobic treatment of slaughterhouse wastewater in an expanded granular sludge bed (EGSB) reactor. *Water Sci. Technol.* **40** (8), 99–106.

Oz, N. A., Ince, O., Ince, B. K., Akarsubasi, A. T. & Eyice, O. 2003 Microbial population dynamics in an anaerobic CSTR treating a chemical synthesis-based pharmaceutical wastewater. *J. Environ. Sci. Health A* **38** (10), 2029–2042.

Parawira, W., Kudita, I., Nyandoroh, M. G. & Zvauya, R. 2005 A study of industrial anaerobic treatment of opaque beer brewery wastewater in a tropical climate using a full-scale UASB reactor seeded with activated sludge. *Process Biochem.* **40** (2), 593–599.

Peña, M. R., Mara, D. D. & Avella, G. P. 2006 Dispersion and treatment performance analysis of an UASB reactor under different hydraulic loading rates. *Water Res.* **40** (3), 445–452.

Pol, L. H. & Lettinga, G. 1986 New technologies for anaerobic wastewater treatment. *Water Sci. Technol.* **18** (12), 41–53.

Polprasert, C., Kemmadamrong, P. & Tran, F. T. 1992 Anaerobic baffle reactor (ABR) process for treating a slaughterhouse wastewater. *Environ. Technol.* **13** (9), 857–865.

Puñal, A. & Lema, J. M. 1999 Anaerobic treatment of wastewater from a fish-canning factory in a full-scale upflow anaerobic sludge blanket (UASB) reactor. *Water Sci. Technol.* **40** (8), 57–62.

Quaff, A. & Guha, S. 2011 Evaluation of mixing and performance of lab-scale upflow anaerobic sludge blanket reactors treating domestic wastewater. *J. Environ. Eng.* **137** (5), 322–331.

Rajakumar, R., Meenambal, T., Rajesh Banu, J. & Yeom, I. T. 2011 Treatment of poultry slaughterhouse wastewater in upflow anaerobic filter under low upflow velocity. *Int. J. Environ. Sci. Tech* **8** (1), 149–158.

Rajeshwari, K., Balakrishnan, M., Kansal, A., Lata, K. & Kishore, V. V. 2000 State-of-the-art of anaerobic digestion technology for industrial wastewater treatment. *Renew. Sustain. Energy Rev.* **4** (2), 135–156.

Ramakrishnan, A. & Surampalli, R. Y. 2012 Comparative performance of UASB and anaerobic hybrid reactors for the treatment of complex phenolic wastewater. *Bioresour. Technol.* **123**, 352–359.

Reichert, P. 1994 AQUASIM – A tool for simulation and data analysis of aquatic systems. *Water Sci. Technol.* **30** (2), 21–30.

Rodriguez, R. 2011 *Upflow Anaerobic Sludge Blanket Reactor Modelling*. *Licentiate Thesis*, School of Chemical Science and Engineering, Royal Institute of Technology (KTH), Sweden.

Ronquest, L. C. & Britz, T. J. 1999 Influence of lower substrate pH and retention time on the efficiency of a UASB bioreactor treating winery waste water. *S. Afr. J. Enol. Viticult.* **20** (1), 35–41.

Sam-Soon, P. A. L. N. S., Wentzel, M. C., Dold, P. L., Loewenthal, R. E. & Marais, G. v. R. 1991 Mathematical modelling of upflow anaerobic sludge bed (UASB) systems treating carbohydrate waste waters. *Water SA* **17** (2), 91–106.

Sandberg, M. & Ahring, B. K. 1992 Anaerobic treatment of fish meal process waste-water in a UASB reactor at high pH. *Appl. Microbiol. Biotechnol.* **36** (6), 800–804.

Satyanarayan, S., Karambe, A. & Vanerkar, A. P. 2009 Herbal pharmaceutical wastewater treatment by a pilot scale upflow anaerobic sludge blanket (UASB) reactor. *Water Sci. Technol.* **59** (11), 2265–2272.

Sawayama, S., Yagishita, T. & Tsukahara, K. 1999 Lighted upflow anaerobic sludge blanket. *J. Biosci. Bioeng.* **87** (2), 258–260.

Schellinkhout, A. 1993 UASB technology for sewage treatment: experience with a full scale plant and its applicability in Egypt. *Water Sci. Technol.* **27** (9), 173–180.

Schellinkhout, A. & Collazos, C. J. 1992 Full-scale application of the UASB technology for sewage treatment. *Water Sci. Technol.* **25** (7), 159–166.

Schmidt, J. E. & Ahring, B. K. 2000 Granular sludge formation in upflow anaerobic sludge blanket (UASB) reactors. *Biotechnol. Bioeng.* **49** (3), 229–246.

Seghezzo, L., Zeeman, G., van Lier, J. B., Hamelers, H. V. M. & Lettinga, G. 1998 A review: the anaerobic treatment of sewage in UASB and EGSB reactors. *Bioresour. Technol.* **65** (3), 175–190.

Singh, K. S. & Viraraghavan, T. 2003 Impact of temperature on performance, microbiological, and hydrodynamic aspects of UASB reactors treating municipal wastewater. *Water Sci. Technol.* **48** (6), 211–217.

Singh, R., Kumar, S. & Ojha, C. S. 1999 Nutrient requirement for UASB process: a review. *Biochem. Eng. J.* **3** (1), 35–54.

Singhal, A., Gomes, J., Praveen, V. V. & Ramachandran, K. B. 1998 Axial dispersion model for upflow anaerobic sludge blanket reactors. *Biotechnol. Prog.* **14** (4), 645–648.

Sötemann, S. W., Ristow, N. E., Wentzel, M. C. & Ekama, G. A. 2005 A steady state model for anaerobic digestion of sewage sludge. *Water SA* **31** (4), 511–527.

Souza, M. E. 1986 Criteria for the utilization, design and operation of UASB reactors. *Water Sci. Technol.* **18** (12), 55–69.

Tejasen, S. & Taruyanon, K. 2010 Modelling of two-stage anaerobic treating wastewater from a molasses-based ethanol distillery with the IWA anaerobic digestion model No.1. *Eng. J.* **14** (1), 25–36.

Thamsiriroj, T. & Murphy, J. D. 2011 Modelling mono-digestion of grass silage in a 2-stage CSTR anaerobic digester using ADM1. *Bioresour. Technol.* **102** (2), 948–959.

Torkian, A., Eqbali, A. & Hashemian, S. 2003 The effect of organic loading rate on the performance of UASB reactor treating slaughterhouse effluent. *Resour. Conserv. Recycl.* **40** (1), 1–11.

Trnovec, W. & Britz, T. J. 1998 Influence of organic loading rate and hydraulic retention time on the efficiency of a UASB bioreactor treating a canning factory effluent. *Water SA* **24** (2), 147–152.

Tseng, S. K. & Lin, M. R. 1994 Treatment of organic wastewater by anaerobic biological fluidized bed reactor. *Water Sci. Technol.* **29** (12), 157–166.

Uemura, S., Tseng, I.-C. & Harada, H. 1995 Effect of temperature elevation from 55°C to 65°C on the performance of a thermophilic UASB reactor and characteristics of methanogenic granular sludge. *Environ. Technol.* **16** (10), 987–994.

Vanrolleghem, P. A., Rosen, C., Zaher, U., Copp, J., Benedetti, L., Ayesa, E. & Jeppsson, U. 2005 Continuity-based interfacing of models for wastewater systems described by Petersen matrices. *Wat. Sci. Tech.* **52** (1–2), 493–500.

Vanrolleghem, P. A., Flores-Alsina, X., Guo, L., Solon, K., Ikumi, D., Batstone, D. J., Brouckaert, C., Takács, I., Grau, P., Ekama, G. A., Jeppsson, U. & Gernaey, K. V. 2014 Towards BSM2-GPS-X: A plant-wide benchmark simulation model not only for carbon and nitrogen, but also for greenhouse gases (G), phosphorus (P), sulphur (S) and micropollutants (X), all within the fence of WWTPs/WRRFs. In: *Proceedings of the IWA/WEF Wastewater Treatment Modelling Seminar*, Spa, Belgium, 30 March–2 April 2014.

Visser, A., Gao, Y. & Lettinga, G. 1993 Effects of pH on methanogenesis and sulphate reduction in thermophilic (55°C) UASB reactors. *Bioresour. Technol.* **44** (2), 113–121.

Wendland, C., Deegener, S., Behrendt, J., Toshev, P. & Otterpohl, R. 2007 Anaerobic digestion of blackwater from vacuum toilets and kitchen refuse in a continuous stirred tank reactor (CSTR). *Water Sci. Technol.* **55** (7), 187–194.

Wolmarans, B. & De Villiers, G. H. 2004 Start-up of a UASB effluent treatment plant on distillery wastewater. *Water SA* **28** (1), 63–68.

Wongnoi, R., Songkasiri, W. & Phalakornkule, C. 2007 Influence of a three-phase separator configuration on the performance of an upflow anaerobic sludge bed reactor treating wastewater from a fruit-canning factory. *Water Environ. Res.* **79** (2), 199–207.

Xing, W., Zuo, J., Dai, N., Cheng, J. & Li, J. 2009 Reactor performance and microbial community of an EGSB reactor operated at 20 and 15°C. *J. Appl. Microbiol.* **107** (3), 848–857.

Yan, Y. & Tay, J. 1996 Brewery wastewater treatment in UASB reactor at ambient temperature. *J. Environ. Eng.* **122** (6), 550–553.

Yasui, H., Sugimoto, M., Komatsu, K., Goel, R., Li, Y. Y. & Noike, T. 2006 An approach for substrate mapping between ASM and ADM1 for sludge digestion. *Water Sci. Technol.* **54** (4), 83–82.

Yoochatchaval, W., Nishiyama, K., Okawara, M., Ohashi, A., Harada, H. & Syutsubo, K. 2008a Influence of effluent-recirculation condition on the process performance of expanded granular sludge reactor for treating low strength wastewater. *Water Sci. Technol.* **57** (6), 869–873.

Yoochatchaval, W., Ohashi, A., Harada, H., Yamaguchi, T. & Syutsubo, K. 2008b Characteristics of granular sludge in an EGSB reactor for treating low strength wastewater. *Int. J. Environ. Res.* **2** (4), 319–328.

Zheng, P. & Hu, B. I. 2002 Start-up strategies of UASB reactor for treatment of pharmaceutical wastewater. *J. Environ. Sci. China* **14** (2), 250–254.

Zhou, W., Wu, B., She, Q., Chi, L. & Zhang, Z. 2009 Investigation of soluble microbial products in a full-scale UASB reactor running at low organic loading rate. *Bioresour. Technol.* **100** (14), 3471–3476.

First received 12 July 2018; accepted in revised form 13 October 2018. Available online 15 November 2018