Sequential UASB and dual media packed-bed reactors for domestic wastewater treatment – experiment and simulation

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ABSTRACT

A wastewater treatment system composed of an upflow anaerobic sludge blanket (UASB) reactor followed by a packed-bed reactor (PBR) filled with Sorbulite® and Polonite® filter material was tested in a laboratory bench-scale experiment. The system was operated for 50 weeks and achieved very efficient total phosphorus (P) removal (99%), 7-day biochemical oxygen demand removal (99%) and pathogenic bacteria reduction (99%). However, total nitrogen was only moderately reduced in the system (40%). A model focusing on simulation of organic material, solids and size of granules was then implemented and validated for the UASB reactor. Good agreement between the simulated and measured results demonstrated the capacity of the model to predict the behaviour of solids and chemical oxygen demand, which is critical for successful P removal and recovery in the PBR.

Key words | Polonite®, simulation, Sorbulite®, wastewater, UASB

INTRODUCTION

Small villages and single houses in rural areas are generally not connected to sewerage systems and municipal wastewater treatment plants. Instead, there are many alternatives for small-scale wastewater treatment (SSWT), ranging from very poor purification in a septic tank only to advanced systems such as a sequencing batch reactor. The most common systems operating in the Scandinavian countries are the buried sand filter, soil infiltration and package plants (Jenssen et al. 2010). However, these technologies are associated with disadvantages such as clogging, high construction or maintenance costs and use of chemicals. Moreover, the soil-based systems are generally inefficient in removal of pollutants from wastewater and can be considered obsolete technology (Dey & Truax 2012).

A successful system to remove organic material from wastewater is the upflow anaerobic sludge blanket (UASB) reactor. It can handle different kinds of wastewater, including residential wastewater. Typically, this type of wastewater is classified as low strength wastewater (i.e. chemical oxygen demand (COD) 450 mg/L) (Tchobanoglous et al. 2003). Residential wastewater has been successfully treated in a UASB reactor by Lew et al. (2004), achieving a percentage removal of organic material ranging 79–89% with hydraulic retention time of 24 h and temperature of 28 °C.

In recent years, the UASB technology has been suggested for use with the septic tank (Verstraete et al. 2013), opening the way for development of new SSWT systems. Several researchers investigated the feasibility of the UASB reactor to treat municipal and residential wastewater under low temperatures (Moawad et al. 2009). It has been concluded that temperature does not significantly affect the percentage of removal of organic material; however, the mechanisms involved in its removal are not the same. At low temperature, the organic material is removed due to the process of filtration in the sludge bed, while at higher temperatures the predominant mechanism is the conversion of the organic material into biogas due to the methanogenic activity of microorganisms (Moawad et al. 2009). The use of the UASB reactor as a system for treating household wastewater could be feasible and advantageous. Currently, almost all households in areas not connected to municipal sewer systems are provided with a septic tank where most of the solids are removed. By connecting a UASB reactor to the septic tank or re-constructing the septic tank to a UASB facility the result could be a series of advantages for both the household and the environment. The most significant are the production of biogas which can be used as energy source, and the removal of organic...
material. Moreover the UASB reactor has the ability to work at low temperature and low costs of construction, maintenance and operation. The system does not require a support medium for the development of microorganisms, and has a low sludge production rate. It has also been reported that the UASB reactor can remove up to 95% of parasitic worm eggs (i.e. helminth eggs) present in domestic wastewater (Yaya-Beas et al. 2016). The main disadvantages are the low removal of pathogenic bacteria and nutrients and a long start-up time (Chong et al. 2012). The inability to remove phosphorus (P) and nitrogen (N) is a particular concern, but the UASB effluent could be treated separately in a following step consisting of a packed-bed reactor (PBR) filled with reactive filter materials.

Reactive materials can be classified into two categories: industrial by-products (i.e. metallurgic slags, Sachthofer PR) and mineral-based products (i.e. Filtralite P, Polonite, Sorbulite). Among these, Polonite® and Sorbulite® have shown promising results not only in laboratory tests, but also in full-scale and commercial applications. Sorbulite is a material produced from the manufacture of autoclaved aerated concrete and Polonite is produced by controlled calcination of porous sedimentary rock from the Paleogene period. In both materials, the dominant elements are silica and calcium (Nilsson et al. 2013a). The main features of Polonite are high effectiveness in removing P (up to 100%) and bacteria (up to 100%), and the absence of clogging problems in the process. The weaknesses of Polonite are that its lifespan is significantly reduced if the incoming wastewater contains a high concentration of organic material, and that removal of N is low (21%) (Nilsson et al. 2013b). Sorbulite successfully removes P and organic material. With respect to the capacity of Sorbulite for removing total organic carbon and inorganic N, removal rates of 45% and 23%, respectively, with a particle size ranging from 2 to 4 mm, have been reported. Sorbulite also removes pathogens, but to a much lesser degree than Polonite (73%) (Nilsson et al. 2013b). The capacity of Sorbulite and Polonite to remove bacteria is promoted by their high content of calcium, which acts as a coagulant and increases the pH level of the water (Williams et al. 2009).

In order to run a serial UASB-PBR system, the UASB reactor must be optimised to achieve stable performance with high organic matter removal. Sludge escaping to the PBR will have detrimental effects on the filter medium, requiring more frequent replacement. Therefore it is important to understand the biomass kinetics in the UASB reactor. These are usually modelled based on the microbial population and availability of the substrate (Rodríguez-Gómez 2011).

However, biomass is present in the form of granules and most models do not take into account the size of the granules or the resistance that the substrate encounters on its way to the core of the granule. The latter is a key parameter for successful reactor performance (Lee et al. 2014). Based on a mass balance, a model that overcomes the aforementioned weaknesses has been developed by our research group (Rodríguez-Gómez et al. 2014). In this model, it is assumed that the substrate conversion rate parameter follows the Monod equation, although this approach has been criticised by some researchers arguing that the results do not always fit the experimental data (Parsamehr 2012). However, other researchers consider the Monod model suitable for describing bioconversion of substrate in anaerobic reactors (Monsalvo et al. 2015). The biodegradation process comprises four phases: hydrolysis, acidogenesis, acetogenesis and methanogenesis. Many researchers have reported that the limiting phase is hydrolysis and Donoso-Bravo et al. (2013) concluded that hydrolysis is better explained by the Contois model, which is a function of the substrate and biomass concentration.

Based on these facts and background, it was hypothesised that a serial UASB-PBR system can be operated with stable biological oxygen demand (BOD), P and bacteria removal. A simulation was performed using the experimental data collected in this study with the model developed by Rodríguez-Gómez et al. (2014), but using the substrate conversion parameter following the Contois model. We examined whether the model can be used, with proper adjustment, for predicting the performance of reactors developed based on UASB technology (i.e. expanded granular sludge bed, internal recirculation reactor and UASB septic tank). To the best of our knowledge, this is the first time a system composed of a UASB reactor followed by a PBR filled with dual medium (Sorbulite and Polonite) has been tested for household wastewater treatment. A main treatment goal was to meet the rigorous permissible limit on P application in Sweden for discharge of treated water to receiving water bodies, but also the limits on pH, suspended solids, N and pathogenic bacteria (Naturvårdsverket 2006).

**EXPERIMENTAL**

**UASB reactor**

The UASB reactor consisted of a polymethyl methacrylate transparent column with a height of 110 cm and an internal diameter of 10.5 cm, given a height to diameter ratio of 10.5.
This configuration was designed to enhance the granulation in the UASB reactor (Lin & Chang 2007). Sampling ports were placed throughout the height of the reactor and in the upper part a gas–solid–liquid–separator was installed.

The UASB reactor was inoculated with 5.72 L of anaerobic sludge (not granular) taken from the anaerobic digester of a large-scale wastewater treatment plant (Henriksdals Reningsverk, Stockholm). The total suspended solids (TSS) concentration was 51,263.12 mg L$^{-1}$, giving volatile suspended solids (VSS) to TSS ratio of 0.82. At the start of the experiment, the height of the sludge bed was 60% of the height of the reactor.

Since a model to predict the performance of the UASB reactor was also developed, it was necessary to carry out batch experiments to the anaerobic sludge to determine the kinetic parameters used during the simulation. This was done by following the linearisation method described by Sponza & Işık (2005) for the Contois model.

The raw wastewater generated by the daily needs of a family of four individuals living 35 km from Stockholm was deposited in a septic tank, where most of the solids were removed. The effluent from the septic tank was collected in 100-mL acid washed plastic bottles once a week and analysed immediately, except for solids and bacteria, which were analysed every second week. The COD concentration was determined according to methods CSN ISO 6060 and CSN ISO 15705. The concentrations of 7-day biochemical oxygen demand (BOD$_7$), TSS, VSS and total N were determined according to method CSN EN 1899-1/-2, CSN 757350, CSN EN 872 and CSN EN 12260, respectively. Total P was determined using the concentrations of $\text{Escherichia coli}$ and $\text{Enterococcus faecalis}$ were calculated according to methods described in SS 028167-2:1996 and EN ISO 7899-2:2000, respectively. A portable meter (Hach Lange HQ40D multi-meter) was used for measurement of pH and temperature.

### Packed-bed reactor

The PBR was a column of the same material as the UASB reactor with a height of 55 cm and an internal diameter of 10.5 cm. The medium forming the bed consisted of the reactive filter materials Sorbulite and Polonite with a particle size range of 2–4 mm and 2–6 mm, respectively. Sorbulite was added to a level of 20% of the total height of the packed bed, the remainder being filled with Polonite. Both materials were obtained from the producer and supplier, Bioptech AB, Sweden.

### Analytical procedures

Samples from the influent to the UASB reactor (wastewater from the septic tank), the effluent from the UASB reactor (influent to the PBR), and the effluent from the PBR were collected in 100-mL acid washed plastic bottles once a week and analysed immediately, except for solids and bacteria, which were analysed every second week. The COD concentration was determined according to methods CSN ISO 6060 and CSN ISO 15705. The concentrations of $\text{Escherichia coli}$ and $\text{Enterococcus faecalis}$ were calculated according to methods described in SS 028167-2:1996 and EN ISO 7899-2:2000, respectively. A portable meter (Hach Lange HQ40D multi-meter) was used for measurement of pH and temperature.

### Description of the experiment

The UASB reactor was initially started without the PBR. This provided time for acclimatisation of the microorganisms. During this period, the COD concentration at the inlet of the UASB reactor was kept around 500 mg L$^{-1}$. This was done by measuring the initial concentration of COD of the raw wastewater and when the value was lower than 500 mg L$^{-1}$, glucose was used as a carbon source to increase the COD value. Otherwise, dilution of the raw wastewater was performed.

A pulse water pump was used to discharge wastewater into the UASB reactor with an upflow velocity of 0.11 m h$^{-1}$ during the lag phase. The acclimatisation phase was assumed to have ended when the difference in percentage COD removal varied by less than 5%. In this experiment, the lag phase took 110 days. During this period, solids were observed in the effluent of the UASB reactor and the sludge bed decreased by 17%.
Following the lag phase, the PBR was connected to the UASB reactor (Figure 1). The inlet wastewater was replaced weekly with fresh wastewater collected from the septic tank and the experiment was performed at room temperature. The upflow velocity was increased up to $0.18 \text{ m h}^{-1}$, working during 8 hours per day in three intervals. The height of the Sorbulite and Polonite in the packed-bed reactor was 0.11 m and 0.44 m, respectively.

**Model development**

**Concept**

In the model, dissolved organic material (i.e. substrate) present in the wastewater enters via the base of the UASB reactor, which is inoculated with granular anaerobic biomass. Once the wastewater is in contact with the biomass, the advection process occurs from the liquid phase to the core of the granules. Transport of the substrate is disturbed by the stagnant film and by diffusion and biotransformation of substrate within the granules.

In the interior of a granule, microorganisms degrade the substrate. As a consequence of their digestion, biogas is produced and new microorganisms are also produced, raising the sludge bed level. The granules forming the sludge bed are constantly moving up and down. This occurs because the generated biogas reduces the density of the granules, causing a buoyancy effect which lasts until release of the biogas from the granule. That moment marks the beginning of the process of sedimentation. However, the distribution of the biomass along the height of the UASB reactor is described by a characteristic curve; granules are gradually re-positioned based on their density, with denser granules located at the bottom of the reactor (Loosdrecht et al. 2015).

**Reaction term**

Assuming that granules are spherical in shape (Figure 2), the expression based on a mass balance for the substrate within a granule is as follows:

$$\frac{1}{r^2} \frac{d}{dr} \left( r^2 \frac{dS_P}{dr} \right) = \frac{k_P}{D_A} S_P$$

where $r$ is the radial distance from the centre of the granule, $D_A$ is the diffusion coefficient of substrate within the granule, $k_P$ is the reaction rate constant and $S_P$ is the concentration of substrate within the granule. The term $(k_P \cdot S_P)$ refers to the substrate reaction rate within the particle, and is assumed to follow the Contois model.

Introducing the Contois model into Equation (1) gives:

$$D_A \frac{1}{r^2} \frac{d}{dr} \left( r^2 \frac{dS_P}{dr} \right) - \frac{\mu_{\text{max}}}{Y_P} \cdot \frac{S_P \cdot X_P}{B} = 0$$

where $\mu_{\text{max}}$ is the maximum specific growth rate, $X_P$ is the concentration of microorganisms in the granule, $B$ is the amount of substrate degraded per amount of active biomass and $Y_P$ is the yield in the granule.

By assuming the concentration of substrate in the granule surface is equal to the concentration of substrate in
the bulk liquid, the reaction term ($\mathcal{R}$) is determined as (Rodríguez-Gómez et al. 2014):

$$\mathcal{R} = q \cdot 4\pi r^2 \cdot N_P$$

where $q$ represents the flux of the substrate in the granule and $N_P$ represents the number of granules per unit volume of reactor.

**Governing equations**

Once the reaction term is defined, it can be introduced into the expression describing the performance of the UASB reactor using the tanks-in-series model. This well-known model hypothetically divides the UASB reactor into several continuous stirred-tank reactors (CSTR) connected in series (Figure 3). The processes of dispersion, advection and reaction are accounted for in the simulation. Dispersion in the UASB reactor is described by the Péclet number ($Pe$), which is included in the number of CSTR ($N_{CSTR}$) containing the sludge bed (Rodríguez-Gómez et al. 2014).

$$N_{CSTR} = \frac{Pe}{2} + 1$$

The change in substrate and biomass in the UASB reactor is described by Equations (5) and (6), respectively:

$$\frac{dS_i}{dt} = \frac{Q_i}{V_i} (S_{i-1} - S_i) - \mathcal{R}_i$$

$$\frac{dX_i}{dt} = \mathcal{R}_i \cdot Y_i - K_d \cdot X_i$$

where $S$ is the concentration of the substrate (measured as COD), $Q$ is the upflow rate, $V$ is the volume of CSTR, $X$ is the biomass, $Y$ is the yield in the UASB reactor, $K_d$ represents the decay rate of the active biomass and the subscript $i$ denotes the CSTR under study.

In Equation (5), the first term on the right-hand side is the advective term and the second term is the reaction term. In Equation (6), the first term on the right-hand side represents the generation of biomass and the second term represents the decay of active biomass. The reaction term includes the radius of the granule, which is a function of the biomass and the degradation of substrate. Hence the granule size varies with time (see Rodríguez-Gómez et al. (2013) for further information).

Since the UASB reactor is divided into several CSTR of known dimensions, it is possible to determine the amount of solids contained in every CSTR; this parameter is named $\varphi$. In the model, when the amount of solids surpasses $\varphi$, the CSTR has reached its capacity to hold solids and the surplus goes to the next CSTR. Thereby, the expansion of the sludge bed in the UASB reactor can be described.

The production of biogas is influenced by the type of organic material, the properties of anaerobic microorganisms, temperature, and retention time (Zeeman et al. 2003). Its yield is strongly dependent of the substrate consumed by the anaerobic microorganisms (Zhao 2011). Therefore, a mathematical correlation to predict the biogas production can be done between Equation (5) and the yield of biogas as shown Equation (7).

$$V_{CH_4} = Y_{CH_4} \cdot \text{COD}_{UASB}$$

where $V_{CH_4}$ is the volume of biogas, $Y_{CH_4}$ represents the yield, which is the volume of methane produced by mass of COD consumed, and $\text{COD}_{UASB}$ represents the mass of COD removed in the UASB reactor. $\text{COD}_{UASB}$ can be obtained from a mass balance of substrate in the UASB reactor, from where the influent mass of substrate (COD$_i$) is assumed constant and the mass of substrate at the effluent (COD$_e$) and COD$_{UASB}$ can be derived from Equation (5).

A more complex model to predict the biogas production has been developed by Kalyuzhnyi et al. (2006), in which the gas volumetric flow was determined taking into account the
partial pressure of substrate in the gaseous phase and the mass transfer rate of substrate to the biogas. Some parameters to run the model were, however, assumed, resulting in an underestimation of the methane production.

The model applied to the UASB reactor was solved with Matlab (Matlab R2015b Windows), following the algorithm known as ODE45 in Matlab language. The modelling results were compared with the results from the bench-scale reactor experiment.

RESULTS AND DISCUSSION

The results are presented and discussed in two sections, the first of which presents the results of the experiment (i.e. UASB reactor and PBR), focusing on the following parameters: BOD$_7$, COD, pH, total N, total P, TSS, VSS, *Escherichia coli*, *Enterococcus faecalis* and granule size. The second section presents the results of model simulations of the behaviour of the UASB reactor, focusing on the following parameters: COD, solids and granule size. Comparisons between experimental and simulated results are then made.

**Experimental results**

On average, the UASB reactor removed 88% of the influent BOD$_7$, while the BOD$_7$ removal in the PBR (i.e. Sorbulite–Polonite) was 89% of that in its influent (Table 2). The removal of organic material in the PBR was mainly attributable to the action of Sorbulite.

The BOD$_7$ removal in the whole system was 99% during this experiment (Figure 4). Similar results were obtained for the COD, reaching an observed average removal of 98% for the whole system. The mechanisms behind the organic matter removal in the filter medium, particularly in the Sorbulite, remain to be investigated. However, Chen *et al.* (2015) reported that the removal of organic material when autoclaved aerated concrete is used could be explained by the high porosity of this material. Likewise, those authors found microbial activity in the pores, which may consume not only organic material, but also nutrients.

The pH did not show significant changes in the UASB reactor. In the fixed bed reactor, however, the pH went from neutral to alkaline (Figure 5). The increase in pH was due to high Ca$^{2+}$ and OH$^-$/C0 release from both Sorbulite and Polonite.

The removal of total N was low in both the UASB and the PBR (Figure 6), reaching a removal rate of 40% for the whole system. In the UASB reactor, the low consumption of N was because anaerobic microorganisms use it for their reproduction and as an electron acceptor for their respiration (Rodríguez-Gómez 2011). Moreover, Sousa *et al.* (2008) report that N is poorly removed in anaerobic digestion, since oxygen is required for its degradation. In the

![Figure 4](https://iwaponline.com/wst/article-pdf/73/12/2959/362918/wst073122959.pdf)
PBR, the N removal could be explained by development of bacteria consuming N in the Sorbulite bed, as reported by Nilsson et al. (2013b).

On the other hand, the removal of P was very high in the system, approaching 99%. The removal of P in the PBR was 98%, whereas it was only 21% in the UASB reactor. These results are shown in Figure 7. The mechanism responsible for P removal in the UASB is biological assimilation (Ping et al. 2011). The mechanisms behind P removal in Polonite and Sorbulite are well documented (Gustafsson et al. 2008).

Phosphorus removal in Polonite has been shown to be negatively affected if the loading of organic matter is constantly high (Nilsson et al. 2013a). It is therefore recommended by the supplier of the material that the average BOD₇ value be kept below 30 mg L⁻¹. In the present experiment the value in the UASB effluent was above this limit, but Polonite was not directly affected, since particulate and dissolved organic matter (DOM) were bound in the Sorbulite layer. This was visible through the transparent reactor column, and investigation of the medium once the experiment was completed also confirmed DOM sorption (data not shown).

The TSS concentration in the incoming wastewater was 54 mg L⁻¹, of which 58% was removed in the UASB reactor and 87% in the PBR. Therefore a total removal rate of 94% was achieved in the whole system. The concentration of VSS showed similar behaviour; 44% and 89% removal in the UASB reactor and PBR, respectively. The total removal of VSS in the system was 94%. The VSS/TSS ratio in the sludge bed of the UASB reactor was stable during the experiment, with an average value of 0.74. Figures 8 and 9 show the changes in these parameters during the experiment.

The studied bacteria, Escherichia coli and Enterococcus faecalis, both underwent a reduction of 99% after treatment in the UASB-PBR system. This efficient die-off is due to the...
fact that these pathogens cannot live under high pH (Geeraerd et al. 2013), which was the prevailing condition in the PBR with the reactive materials Sorbulite and Polonite. The numbers and reductions of E. coli are shown in Figure 10 and of E. faecalis in Figure 11. Previous research by Nilsson et al. (2013a, 2013b) also demonstrated the capacity of Polonite and Sorbulite to reduce bacteria during wastewater filtration. However, their experiments were performed under high organic loading or extreme wastewater discharge, which resulted in a much lower bacterial reduction. The design flow used in our system was able to control the pathogenic bacteria concentration at below 100 CFU 100 m L\(^{-1}\), which is much lower than the EU limit for surface water of bathing water quality (Directive 2006/7/EC (EC 2006)).

With time, the biomass attached to itself and formed irregularly shaped aggregates ranging in size from fine particles to 0.4 mm in diameter. In the sludge bed, spaces occupied by water and biogas were observed as a consequence of the paths created by biogas. From time to time samples of biogas, suggested to be dominated by methane, were taken from a collector attached to the UASB reactor and burned to ensure that the gas was flammable.

The Swedish regulations on discharge of treated wastewater from households usually recommend 90% removal of BOD\(_7\) and total P and 50% removal of total N. Development of a small-scale, serial UASB septic tank and PBR system would be feasible for areas where private treatment plants have to be operated. Such decentralised systems in Sweden have requirements on effluent concentrations of 30 mg L\(^{-1}\) for BOD\(_7\), 1 mg L\(^{-1}\) for total P and 40 mg L\(^{-1}\) for total N (Naturvårdsverket 2006). The results of this experiment are promising in terms of whether the UASB-PBR system can meet the regulations for BOD\(_7\) and total P. For total N, the removal in the experiment was 10% less than the recommended level (in percentage terms). However, the concentration of total N at the outlet of the system was lower than recommended in the regulations.

**Simulation results**

Simulation results were plotted every 10 weeks, except for granule size, for which data were presented every 2.5 weeks. The parameter studied in the simulations was the concentration of organic material (measured as COD). The batch culture experiments for determination of the Contois kinetic parameters resulted in: \(\mu_{\text{max}} = 0.109\ \text{d}^{-1}\), \(B = 0.486\), \(Y = 0.13\) and \(K_d = 0.0068\ \text{d}^{-1}\).

The parameters used to run the simulation were taken from the experiment, except for the initial radius of the
granule. It was assumed to be $1.00 \times 10^{-4}$ m, since the biomass inoculated into the reactor was in the form of sludge (i.e. non-granulated). Moreover, the diffusion coefficient was assumed to be $3.00 \times 10^{-7}$ m$^2$ h$^{-1}$, which corresponds to the value used by Rodríguez-Gómez et al. (2016) in simulations of a UASB reactor. Table 3 shows the parameters used during the present simulation.

### Substrate (COD)

The concentration of COD at the outlet of the UASB reactor decreased with time; the same behaviour was observed in the experiment. During the first interval, the removal of COD was lower in the simulation (63%) than in the experiment (76%). Gradually, the removal of COD in the simulation increased until it reached 91% in the last interval; this value was slightly higher than that obtained in the experiment (89%). Figure 12 compares the results on removal of COD obtained from the simulation and the experiment.

### VSS in the sludge bed

Figure 13 compares the predicted concentration of VSS in the sludge bed according to the simulation and that obtained in the experiment. The concentration of VSS was the object of the study, because it was assumed that VSS represents the microorganisms responsible for degradation of the substrate.

The concentration of VSS in the sludge bed was underestimated by the simulation. However, a similar increase in the concentration of VSS in every interval was observed for both the simulation and the experiment. The TSS can easily be determined because of the assumption $\text{VSS} = \text{TSS} = 0.74$, which was the average ratio value at the end of the experiment. Regarding the granule, it was assumed that it was formed of active and inactive biomass (i.e. TSS). The final radius of the granule in the simulation was determined to be 0.1165 mm (Figure 14), whereas granules with radius up to 0.2 mm were measured at the termination of the experiment.

### Sludge bed expansion

One of the main characteristics of the UASB reactor is the low rate of growth of biomass. However, after a long time of operation, there will be abundant biomass in the reactor and a discharge will be needed. The simulation showed the profile of expansion of the sludge bed in the reactor. In this case, the value of $\varphi$ was 10%. Figure 15 shows the profile of the sludge bed with time. At the beginning, biomass represented a total of eight CSTR. The height of the sludge bed expanded as reproduction of microorganisms occurred, reaching 8.5
CSTR with biomass during the simulation. At the end of the experiment, however, the height of the sludge bed was 66 cm, which corresponded to 9.6 CSTR with biomass. In the sludge bed, discontinued channels and trapped bubbles of biogas were observed. Eventually small bubbles attached to a trapped bubble, increasing the size and buoyancy, and finally these bubbles made their way through the sludge bed. Consequently, agitation of the biomass occurred.

**Biogas**

Assuming a constant influent concentration of COD of 475.66 mg L$^{-1}$, which corresponds to the average value of the experiment, and with the model response derived from Equation (5), the cumulative COD$_{\text{UASB}}$ was determined from a mass balance of the substrate in the UASB reactor (Table 4). The average value of $Y_{\text{CH}_4}$ obtained at the end of the experiment was 0.2. This value is in agreement with a study reported by Khan *et al.* (2015).

![Figure 14](image1) | Granule size predicted in the simulation over a 50-week period.

![Figure 15](image2) | Sludge bed expansion predicted in the simulation over a 50-week period.

**Table 4** | Mass of COD retained in the UASB reactor and volume of methane predicted in the simulation.

<table>
<thead>
<tr>
<th>Week</th>
<th>COD$_{\text{UASB}}$, kg</th>
<th>$V_{\text{CH}_4}$, m$^3$</th>
</tr>
</thead>
<tbody>
<tr>
<td>10</td>
<td>$7.88 \times 10^{-1}$</td>
<td>$1.58 \times 10^{-1}$</td>
</tr>
<tr>
<td>20</td>
<td>$8.83 \times 10^{-1}$</td>
<td>$1.77 \times 10^{-1}$</td>
</tr>
<tr>
<td>30</td>
<td>$9.70 \times 10^{-1}$</td>
<td>$1.94 \times 10^{-1}$</td>
</tr>
<tr>
<td>40</td>
<td>1.05</td>
<td>$2.11 \times 10^{-1}$</td>
</tr>
<tr>
<td>50</td>
<td>1.14</td>
<td>$2.27 \times 10^{-1}$</td>
</tr>
</tbody>
</table>

Figure 16(a) shows a comparison between simulated and mean measured values of methane production over time. For 30 weeks, the production of biogas tended to increase, but in week 40 it significantly decreased and then remained almost constant until the end of the experiment. Probably this occurred due to the change in temperature, which ranged 18–27 °C and decreased to 16–22 °C for the first 30 weeks and last 20 weeks, respectively. This means that the methane generation was strongly temperature dependent, a conclusion also drawn by Zhao (2011) who varied temperature in the anaerobic digester and studied the production of biogas. The change of temperature, the profile of methane production by assuming a yield of 0.20 and the measured value of methane production during the whole experiment are shown in Figure 16(b).
The model response for methane production tended to increase from week 10 to week 20 and so on. The simulation underestimated the experimental value for weeks 10, 20, and 30. For weeks 40 and 50, however, the values were overestimated. This occurred because the simulation did not account for the change in temperature. Additionally, the $Y_{CH_4}$ and the influent mass of COD was assumed constant while in the experiments those values varied.

Despite that the change in temperature affected the biogas production, practically it did not affect the COD removal. Similar results were reported by Lew et al. (2004) who concluded that the low production of biogas was due to inactivation of microorganisms at low temperature, and the high removal of COD occurred by filtration phenomena in the sludge bed. Another explanation could be that the biogas production may be affected by resistances that the COD suffers during its transport to the core of the granule (Korsak 2011). Therefore, during the first 50 weeks the methane-producing microorganisms were dispersed in the sludge bed or they were located in the core of small granules as was demonstrated by MacLeod et al. (1990), shortening the path of the COD to the centre of granules. For the last period, however, bigger granules were formed; consequently, the bioconversion of substrate took more time.

**CONCLUSIONS**

The system composed of a UASB reactor followed by a packed-bed reactor containing Sorbulite and Polonite is a feasible system for removing organic material, total phosphorus and total nitrogen from household wastewater. Swedish standards for wastewater discharge can be met. Moreover, the system successfully removes solids and pathogenic bacteria present in the wastewater.

The model used here to describe the performance of the UASB reactor is able to predict the concentration of organic material at the outlet of the UASB reactor, and the variation in biomass forming the sludge bed. It also accurately describes the increase in granule size within time. Prediction and control of organic matter outflow is crucial for operation of a full-scale system, since it affects the efficiency and lifetime of the filter medium in the PBR.

Biogas production is strongly dependent on COD removed by the UASB reactor and temperature.

It is concluded that a serial UASB-PBR system can be operated with stable BOD, P and bacteria removal.

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