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Review Paper

Solids removal in upflow anaerobic reactors, a review

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Abstract

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

Keywords: Anaerobic treatment; Charges; Extracellular polymeric substances (EPS); Hydraulic retention time (HRT); Organic loading rate (OLR); Physical characteristics; Particle size distribution (PSD); Solids removal; UASB; Upflow velocity

1. Background

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

2. Parameters affecting solids removal in upflow reactors

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

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2.1. Reactor operational conditions

2.1.1. Temperature

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

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

$$G = \sqrt{\frac{p}{\mu * V}} \tag{1}$$

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

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

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

Table 1
Relation between temperature and turbulence in the reactor

Temperature (°C)	Viscosity (μ) of water in the reactor (Pa s)	% Increase of G from 15 °C (%)
15	1.14×10^{-03}	–
20	1.00×10^{-03}	7
25	8.90×10^{-04}	13
30	7.98×10^{-04}	19
40	6.53×10^{-04}	32

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

$$\frac{dx_{degr.}}{dt} = k_h \cdot X_{degr.} \tag{2}$$

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

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

$$k_h = A\,e^{-E/RT} \tag{3}$$

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

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

2.1.2. Organic loading rate

Several authors reported that up to a certain limit, the treatment efficiency of complex wastewaters, e.g. potato maize, slaughterhouse, in high rate anaerobic reactors increases with increasing OLR. A further increase of OLR will lead to operational problems like sludge bed flotation and excessive foaming at the gas–liquid interface in the gas–liquid–solid (GLS) separator, as well as accumulation of undigested ingredients. As a result, the treatment efficiency deteriorates (Sayed, 1987; Ruiz et al., 1997; Kalyuzhnyi et al., 1998). Also accumulation of biogas in the sludge bed was noticed, forming stable gas pockets that lead to incidental lifting of parts of the bed and a pulse-like eruption of the gas from this zone (Kalyuzhnyi et al., 1998; Elmitwalli et al., 1999).

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

$$\mathrm{OLR} = \frac{Q * \mathrm{COD}}{V} = \frac{\mathrm{COD}}{\mathrm{HRT}} \tag{4}$$

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

When the solids removal efficiency in upflow reactors is related to the OLR, it becomes crucial to distinguish between these parameters. For this reason, OLR is an inadequate design parameter to assure good performance of anaerobic reactors. Increasing the loading rate by reducing HRT down to a certain value will reduce the solids removal efficiency, probably due to increasing of upflow velocity (see Table 2: cases I and II; Fig. 1). Since, increasing the influent concentration can increase the OLR, it becomes more essential to maintain an adequate upflow velocity to assure good mixing (Table 2: cases I and III; Section 2.1.4).

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

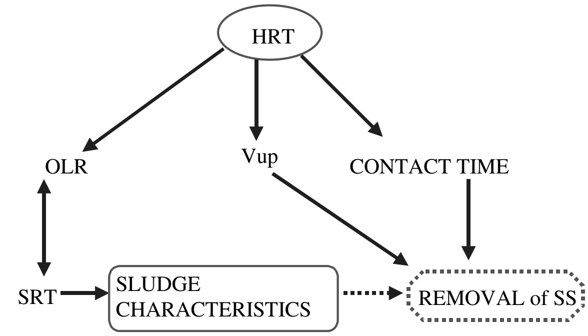

Fig. 1. The operational parameters which are expected to affect the solids removal in upflow reactor. Where: OLR, organic loading rate; SRT, sludge retention time; V_{up}, upflow velocity; HRT, hydraulic retention time; SS, suspended solids.

- High influent concentration; this will cause gas pockets formation, while the upflow velocity is low to create adequate turbulence in the sludge bed (poor mixing).
- Low HRT is accompanied by high upflow velocity (V_{up}), which will lead to wash out of influent solids and of viable biomass.
- High solids loading rate, which imposes low SRT, will change the sludge bed composition (microbial, physical, and chemical) and cause accumulation of floatable substances (proteins and lipids).

2.1.3. Hydraulic retention time/sludge retention time

Wang (1994) reported that, during anaerobic sewage treatment in a 170 m^3 hydrolysis upflow sludge bed (HUSB) reactor, HRT in the range (2.5–5 h) does not seriously affect the removal rate of the suspended solids. Differently, GonÇalves et al. (1994) show that the removal efficiency decreased with decreasing HRT accompanied by increase of upflow velocities (see Section 2.1.4). It might be argued that the HRT is an inadequate parameter for describing solids removal in upflow reactors (Fig. 1). The effect of HRT could manifest as a result of its direct relation to the liquid upflow velocity (V_{up}) and also to the solids contact time in the reactor and so the possibility of solids to coalesce or to be entrapped in the sludge bed. Moreover, the HRT is a major parameter, which determines the SRT (Zeeman and Lettinga, 1999). The SRT can indirectly

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

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

influence the solids removal as through changing of the physical–chemical and biological characteristics of the sludge bed in addition to biogas production.

2.1.4. Upflow velocity (V_{up})

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

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

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

2.2. Influent characteristics

2.2.1. Influent concentration

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

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

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

2.2.2. Influent particle size

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

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

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

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

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

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

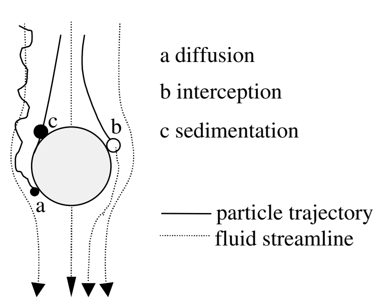

Fig. 2. Transport mechanisms (adapted from O'Melia and Tiller (1993) and Stumm and Morgan (1996)).

2.2.3. Influent particle charge

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

2.3. Sludge bed characteristics

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

2.3.1. Particle size distribution

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

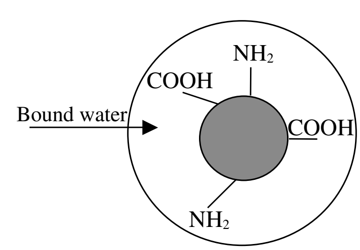

Fig. 3. A colloidal hydrophilic protein particle surrounded by water (Henze et al., 1995).

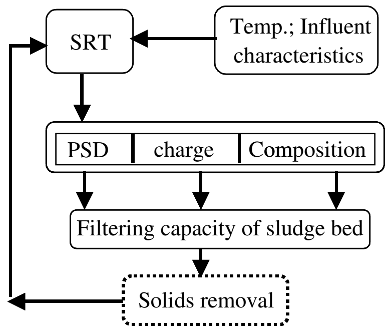

Fig. 4. Scheme of interaction hypothesis for physical removal of solids in a UASB reactor. Where: SRT, sludge retention time; PSD, particle size distribution.

effect of PSD on the performance of the upflow anaerobic sludge bed reactors is not yet clear.

2.3.2. Extracellular polymeric substances

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

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

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

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

EPS are reported to affect several physical and chemical characteristics of activated sludge, like: dewaterability (Kang et al., 1990), floc charge (Horan and Eccles, 1986), floc structure (Eriksson and Härdin, 1984), settleability (Forster, 1985; Goodwin and Forster, 1985; Urbain et al., 1993) and flocculation (Ryssov-Nielsen, 1975; Brown and Lester, 1980; Rudd et al., 1983; Barber and Veenstra, 1986; Eriksson and Alm, 1991; Jia et al., 1996a; Laspidou and Rittmann, 2002). Also the granulation of anaerobic sludge (Jia et al., 1996a,b) and the anaerobic sludge charge (Morgan et al., 1990) are reported to be effected by EPS. The precise function of biopolymers in relation to bioflocculation and their effect on sludge physico-chemical characteristics are not fully understood (Morgan et al., 1990) and sometimes the reported research results are contradictory (Urbain et al., 1993).

EPS are thought to influence the dewatering characteristics of sludge by forming a charged surface layer on sludge particles (Poxon and Darby, 1997). The interactions of these polymers between cells allow adjacent bacteria to aggregate by bridging cell surfaces electrostatically and physically and therefore, initiate floc formation which allows the sludge settlement (Tenny and

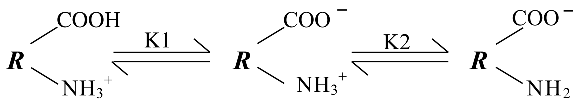

Fig. 5. Protolysis of functional amino and carboxyl groups (adapted from Stumm and Morgan (1996)).

Stumm, 1965; Morgan et al., 1990; Cloete and Steyn, 1988; Eriksson and Alm, 1991). Morgan et al. (1990) proposed that the chemical nature of the sludge surface will influence the measurable floc charge which itself affects the settling properties of the sludge.

2.3.3. Charges

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

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

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

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

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