New developments in reactor and process technology for sulfate reduction

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Abstract Sulfate reduction has been regarded in the past as an unwanted process in anaerobic treatment of sulfate-rich wastewaters. Research efforts were primarily focused on H2S toxicity, competition between sulfidogenic and methanogenic microorganisms and suppression of sulfidogenesis. More recently, the potential sulfidogenesis for treating a wide range of wastestreams contaminated with oxidized sulfurous compounds and/or heavy metals was also appreciated. Heavy metals can be removed by the formation and subsequent precipitation of poorly soluble metal sulfides. Basically two approaches can be distinguished in wastewater treatment: passive treatment using low-cost technologies and active treatment in newly developed bioreactors. Both strategies are discussed.

Keywords Electron donors; heavy metals; pollution prevention; sulfate reduction; sulfur pollution

Introduction
For many years, sulfate reduction has been associated with the nuisance that it causes during the anaerobic treatment of sulfate-rich waste and wastewater. The reduction of oxidized sulfur compounds to H2S leads to a range of problems: a lower CH4 yield, production of a poor quality biogas that requires treatment, a decreased COD-removal efficiency, toxicity, bad smell and corrosion. Initially, research on sulfate reduction was focused on its negative role in anaerobic wastewater treatment (Rinzema and Lettinga, 1988). Research efforts were directed towards assessment of the toxicity of H2S (Kroiss and Plahl-Wabnegg, 1983, Koster et al., 1986), strategies to suppress sulfate reduction and exploring possibilities of steering the competition between sulfate reducing bacteria (SRB) and acetogenic and methanogenic microorganisms in the direction of methanogenesis.

Several sulfate-rich wastewaters, e.g. acid mine drainage, spoil leachates and landfill percolates often are also contaminated with heavy metals (Tichy et al., 1998). As a result of mining operations biological and physico-chemical processes give rise to the oxidation of sulfur species and the production of acidic mine drainage (AMD) waters. These wastewaters may contain a range of metals leached from the ore. The processes giving rise to pyrite oxidation, acid formation and heavy metal solubilization have been well described (Kuenen & Robertson, 1992; Johnson, 1995). The extreme low solubility of heavy metal sulfides that are formed in sulfidogenic bioreactors allows the removal of heavy metals from the water stream by precipitation. Hence, maximization of sulfide production in sulfate reducing reactor units is an attractive way to remove both sulfate and heavy metals from a wastewater (Hao et al., 1996). As these types of wastewater usually contain little or no organic matter, an alternative electron donor (e.g. synthesis gas) has to be added. Full-scale processes are currently applied for heavy metal removal from a Zn and Cd (effluent concentrations below the ppb range) contaminated groundwater at a long-standing smelter site (Scheeren et al., 1993).
Sulfate reduction in methanogenic bioreactors

Sulfide toxicity

Sulfide, produced by SRB and by fermentation of sulfur-containing amino acids, has been shown to be inhibitory for anaerobic digestion (Karhadkar et al., 1987; Hilton & Oleszkiewicz, 1988, Oude Elferink et al., 1994). Its accumulation can result in a severe inhibition of the treatment process, and can even cause a total process failure. The inhibitory effect of sulfide is presumed to be caused by unionized H$_2$S because only neutral molecules can diffuse through the cell membrane. H$_2$S may interfere with the assimilatory metabolism of sulfur, while it possibly also may affect the intracellular pH. Low pH values and low temperatures increase toxicity as they favor formation of unionized sulfide. Much of the published literature on sulfide toxicity does not take pH and adaptation into consideration, which makes general conclusions about toxicity levels difficult. Since sulfide readily reacts with most heavy metals to form insoluble metal sulfides, the toxicity of sulfide is also related to metal concentrations in the sludge.

Studies under both mesophilic and thermophilic conditions showed that granular sludge is less inhibited by H$_2$S than suspended sludges at low and neutral pH, whereas the inhibition is very similar at high pH values (Visser et al., 1995). In suspended sludges, inhibition is determined by the H$_2$S concentration both at low and high pH values (McCartney & Oleszkiewicz, 1993) and 50% inhibition of methanogens was found at H$_2$S concentrations ranging from 50 to 130 mg.l$^{-1}$. In sludge granules a 50% inhibition of methanogens was found at unionized H$_2$S concentrations of 250 and 90 mg.l$^{-1}$ at pH values of 6.4–7.2 and 7.8–8.0, respectively (Koster et al., 1986). The inhibition of methanogenic archaea (MA) is higher than the inhibition of SRB at pH values above 7.8. At a lower pH range (pH < 7.0) there is not much difference in the degree of inhibition (Koster et al., 1986). These data have however been obtained with batch assays, which are not representative for continuously fed systems. MA are more sensitive than fermentative microorganisms and acetogens to H$_2$S inhibition both in suspended (Oleszkiewicz et al., 1989) and granular (Shin et al., 1995) sludge, with the exception of syntrophic propionate degrading consortia, which are more sensitive. In a sulfate-reducing fixed bed reactor treating an acetate and sulfate mixture, process failure occurred already at H$_2$S concentrations above 50 mg.l$^{-1}$ (Stucki et al., 1993). This suggests a rather high susceptibility of acetotrophic sulfate-reducing bacteria (ASRB). In the pH range of 7.5 to 9, sulfide inhibition of ASRB is determined by the total sulfide concentration rather than the H$_2$S concentration, both in flocculent (Oleszkiewicz et al., 1989) and granular sludge (Koster et al., 1986; Visser, 1995). Besides the pH, also the COD/sulfate ratio influences the susceptibility of sludge to sulfide toxicity, because of the development of different bacterial populations (McCartney & Oleszkiewicz, 1991). In practice, anaerobic treatment always proceeds successfully for wastewaters with chemical oxygen demand (COD) to sulfate ratios exceeding 10. For such wastewaters the H$_2$S concentration in the anaerobic reactor will never exceed the presumed critical value of 150 mg.l$^{-1}$ due to the stripping effect of the biogas production (Rinzema & Lettinga, 1988). At COD/sulfate ratios lower than 10, process failures of anaerobic reactors have been reported, while in other cases the process proceeds successfully when precautions are taken to prevent sulfide toxicity (Table 1).

Competition between sulfate reducers, methanogens and acetogens

In environments where sulfate is present, SRB will compete with methanogenic consortia for common substrates like hydrogen, acetate and methanol. Compared with MA, SRB are much more versatile. Compounds like propionate and butyrate, which require syntrophic consortia in methanogenic environments, can be degraded directly by single SRB species. Kinetic properties of SRB, MA and acetogens can be used to predict the outcome of the competition for these common substrates (Lovley et al., 1982; Kristjansson et al., 1982).
For bacteria growing in suspension, Monod kinetic parameters such as the half-saturation constant ($K_s$) and the maximum specific growth rate ($\mu_{max}$) can be used. When bacterial growth is negligible, as is often the case in reactors with a dense biomass concentration, Michaelis-Menten kinetics may be used to predict which type of organism has the most appropriate enzyme systems to degrade substrates. Therefore, both the $V_{max}/K_m$ and the $\mu_{max}/K_s$ ratio give an indication of the outcome of the competition at low substrate concentrations (Robinson & Tiedje, 1984).

In most anaerobic environments, hydrogen is present as an intermediate for which SRB, MA and homoacetogens will compete. Thermodynamically, homoacetogenesis is less favorable than methanogenesis and sulfate reduction. In reactors with immobilized biomass the activity of hydrogenotrophic MA is completely suppressed within a few weeks when sulfate is added (Visser et al., 1993a). As hydrogenotrophic MA are still present in high numbers in such reactors, this effect can not simply be explained by Michaelis-Menten or Monod kinetic data. By addition of sulfate the hydrogen partial pressure becomes so low that thermodynamically hydrogenotrophic methanogenesis is no longer possible.

In marine and freshwater sediments acetate is mainly consumed by SRB when sufficient sulfate is present (Banat et al., 1981; Isa et al., 1986a, Winfrey & Zeikus, 1977; Smith & Klug, 1981). However, for anaerobic digesters it is less clear how acetate is degraded. A complete conversion of acetate by MA, even at an excess of sulfate, has been reported (Isa et al., 1986a; Qatibi et al., 1990; Visser et al., 1993a,b; Ueki et al., 1988, 1989; Yoda et al., 1987). However, in some studies a predominance of acetate-degrading SRB was found (Gupta et al., 1994; Rinzema, 1988; Visser, 1995). Work by Schönheit and others has indicated that the predominance of Desulfobacter postgatei in marine sediments could be explained by its higher affinity for acetate than Methanosarcina barkeri. The $K_m$ values were 0.2 and 3.0 mM, respectively. However, in bioreactors Methanosarcina sp. are only present in high numbers when the reactors are operated at a high acetate concentration or operated at a low pH (Grotenhuis, 1992).

### Table 1: Measures to reduce the reactor sulfide concentration, thus allowing the integration of methanogenesis and sulfate reduction

<table>
<thead>
<tr>
<th>Measure</th>
<th>Procedures</th>
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<tr>
<td>Dilution of the influent H₂S concentration</td>
<td>Non-sulfate containing process water</td>
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<td></td>
<td>Recycle of effluent after a sulfide removal step by:</td>
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<td></td>
<td>- Sulfide stripping</td>
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<td></td>
<td>- Sulfide precipitation</td>
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<tr>
<td></td>
<td>- Biological sulfide oxidation to elemental sulfur with oxygen, nitrate or sunlight</td>
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<td></td>
<td>- Chemical oxidation to elemental sulfur</td>
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<td></td>
<td>Ferric sulfate, chelated</td>
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<td>Ferric sulfate, extractive membrane reactor</td>
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<tr>
<td>Decrease of the unionized H₂S concentration</td>
<td>Elevation of the reactor pH</td>
</tr>
<tr>
<td></td>
<td>Elevation of the reactor temperature</td>
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<td></td>
<td>Precipitation of sulfide, e.g. with iron salts</td>
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<td></td>
<td>Stripping of the reactor liquid using</td>
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<td>- High mixing degree inside the reactor</td>
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<td></td>
<td>- Recirculation of biogas after scrubbing</td>
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<td></td>
<td>- Other stripping gas (e.g. N₂ or air)</td>
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<tr>
<td>Separation of H₂S production and methanogenesis</td>
<td>Two stage anaerobic digestion</td>
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<td>Upflow staged sludge bed (USSB) reactor</td>
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<td>Selective inhibition of SRB</td>
<td>Sulfate analogues (e.g. MoO₄²⁻)</td>
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<td></td>
<td>Transition elements (e.g. Cu, Co, Zn or Ni)</td>
</tr>
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<td></td>
<td>Antibiotics</td>
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Generally, Methanosaeta sp. are the most important acetoclastic MA in anaerobic bioreactors (Grotenhuis, 1992; MacLeod et al., 1990; Morvai et al., 1992; Nishio et al., 1993). Methanosaeta sp. have a higher affinity for acetate than Methanosarcina sp.; their $K_s$ is about 0.4 mM (Jetten et al., 1992). Two abundant acetate-degrading SRB, Desulforhabdus amnigenus and Desulfofaba acetoxidans (Oude Elferink et al., 1995, 1999), have kinetic properties only slightly better than those of Methanosaeta (see Table 2).

Putting all kinetic information together it seems that the growth rate of acetate-degrading SRB is only slightly higher than that of MA. Therefore, it can be expected that the initial relative cell numbers affect the outcome of competition experiments (Omil et al., 1998). This is in particular the case for methanogenic sludge from bioreactors where a major part of the microbial biomass may consist of Methanosaeta. When methanogenic bioreactors are fed with sulfate, the few initial acetate-degrading sulfate reducers have to compete with huge numbers of aceticlastic Methanosaeta species. In Upflow Anaerobic Sludge Bed (UASB) reactors the sludge retention time can be as high as 0.5–1 year (Hulshoff Pol, 1989). Visser (1995) has simulated the competition using a maximum specific growth rate of 0.055 and 0.07 day$^{-1}$ for the methanogen and sulfate-reducing bacterium, respectively, a $K_s$ value for acetate of 0.08 and 0.4 mM acetate, respectively, and different initial ratios of bacteria. Starting with a ratio of methanogens/sulfate reducers of $10^4$, it will take already one year before the number of acetate-degrading SRB and acetate-degrading MA are equal. Nevertheless, long-term UASB reactor experiments of Visser showed that SRB are able to outcompete the MA.

**Suppression of sulfate reduction**

A complete suppression of sulfate reduction and a complete conversion of the organic substrate into methane could be considered as the most optimal option. Therefore, attempts have been made to selectively suppress sulfate reduction by using specific inhibitors, i.e. sulfate analogues (Yadav & Archer 1989), transition elements (Clancy et al., 1992) or antibiotics (Tanimoto et al., 1989). However, so far, no selective inhibitor of SRB has been found that can be used in full-scale anaerobic reactors. This implies that sulfate reduction cannot be prevented in practice.

**Sulfate reduction in sulfidogenic bioreactors**

Biological sulfate removal is a cost-effective alternative for costly and sometimes complex physico-chemical sulfate removal methods (Maree et al., 1991). Biological sulfate removal consists of two steps with (dissimilatory) sulfate reduction to sulfide as the first one. The sulfide produced in the first stage is then biologically oxidized to elemental sulfur ($S^0$). In the sulfate-reducing stage, a complete reduction of sulfate to sulfide is desired. Channeling of reducing equivalents towards the SRB is enhanced by the ability of the SRB to effectively compete with other anaerobic bacteria for the available organic substrate and the sensitivity of the other bacteria for sulfide.

<table>
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<th>Table 2</th>
<th>Kinetic properties of acetotrophic sulfidogens and methanogens</th>
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<tr>
<td></td>
<td>$\mu_{\text{max}}$ [day$^{-1}$]</td>
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<tr>
<td>Acetotrophic methanogens</td>
<td></td>
</tr>
<tr>
<td>Methanosarcina sp.</td>
<td>0.5–0.7</td>
</tr>
<tr>
<td>Methanosaeta sp.</td>
<td>0.1–0.3</td>
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<tr>
<td>Acetotrophic sulfidogens</td>
<td></td>
</tr>
<tr>
<td>Desulforhabdus amnigenus</td>
<td>0.1–0.2</td>
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<tr>
<td>Desulfofaba acetoxidans</td>
<td>0.3–0.4</td>
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For wastewater that contains no or insufficient electron donor and carbon source for a complete sulfate reduction, addition of an appropriate electron donor is required. The selection of the electron donor depends on i) the costs of the added electron donor per unit reduced sulfate and ii) the rest-pollution of the additive in the wastestream, which should be low or easily removable. Based on the last criterion, simple organic compounds (ethanol, methanol) or synthesis gas (a mixture of $\mathrm{H}_2$, CO and $\mathrm{CO}_2$) are preferred above more complex organic substrates (e.g. molasses). Under mesophilic conditions hydrogenotrophic methanogenesis is minimal in the presence of sulfate, as SRB out-compete MA more effectively for $\mathrm{H}_2$. Using a mixture of $\mathrm{H}_2$ and $\mathrm{CO}_2$ (80%:20%), sulfate loading rates of 30 g $\mathrm{SO}_4^{2-}.\text{L}^{-1}.\text{day}^{-1}$ can be achieved at 30°C within 10 days of operation in gas-lift reactors (which provide good $\mathrm{H}_2$ transfer rates) with pumice as carrier material (to immobilize SRB) when the free $\mathrm{H}_2\mathrm{S}$ concentration is kept below 450 mg.L$^{-1}$ (van Houten et al., 1994). These experiments revealed that the hydrogenotrophic SRB were not autotrophic and needed acetate as carbon source. Acetate is formed by homoacetogens. Due to the low affinity of the homoacetogens for $\mathrm{H}_2$, it is possible that under conditions of $\mathrm{H}_2$-limitation insufficient amounts of acetate are available to the hydrogenotrophic SRB, which may result in a predominance of the hydrogenotrophic MA. $\mathrm{H}_2$ gas is too expensive to be used, but addition of synthesis gas (a mixture of $\mathrm{H}_2$, CO and $\mathrm{CO}_2$) is an economical alternative (van Houten et al., 1994), for larger applications, whereas for smaller scale installations methanol or ethanol are more preferable. With synthesis gas, it appeared that CO is not used as electron donor by SRB, and it exerts a toxic effect on SRB and thus limits the sulfate loading rate to 10 g $\mathrm{SO}_4^{2-}.\text{L}^{-1}.\text{day}^{-1}$ at CO concentrations in the gas phase between 5% and 20% (van Houten et al., 1994). With CO, layered biomass particles developed. Homoacetogenic *Acetobacterium* sp. were mainly located in the periphery, whereas sulfate reducing *Desulfovibrio* sp. were located inside the aggregate (van Houten et al., 1994). On methanogenic granular sludge fed with a sulfate-rich wastewater a predominantly sulfate reducing granular sludge will develop. Using synthesis gas, one can immobilize the sulfate reducers on the carrier material or granules and supply the gaseous substrate in a gas-lift reactor.

Weijma (2000) studied the use of methanol as electron donor and found that at pH 7.5 and under thermophilic (65°C) conditions SRB outcompete methanogenic consortia for methanol. At present this limits the potentials of methanol for full-scale applications.

Recently the interest in the application of sulfate (or sulfite) reduction as the main step for the biological treatment of specific wastestreams from chemical, mining and galvanic industries as well as scrubbing water for flue-gas desulfurization is growing. Typical applications are sulfide generation for heavy metal removal and sulfate reduction as the first step of a two-step process for complete sulfur removal from wastestreams rich in oxidized sulfur compounds. The second step is the partial oxidation of sulfide to elemental sulfur, which can by recovered by sedimentation.

**Reactor and process technologies for sulfate reduction**

**Passive treatment of acid mine drainage**

The remediation of acidic metal-rich wastewaters using natural or constructed wetlands is a passive low-cost approach that found wide application worldwide. Often, mining operation ceased many years ago and funds are lacking for costly high-tech solutions for the treatment of the remaining acidic drainage waters. Aerobic wetlands promote oxidation of acid mine drainage (AMD), thereby causing metals (mainly iron) to oxidize and precipitate as oxides, but anaerobic wetlands are based on the reduction of sulfate and precipitation of metal sulfides. A wide range of electron donors such as manure, spent mushroom compost, peat, sawdust and woodchips can fuel this process. In some systems vegetation (e.g. cattail)
can be a continuous source of reduced carbon (Johnson, 2000). Heavy metal removal is the combined result of adsorption to the organic substratum and precipitation of metal sulfides. Therefore it is not always clear what the relative contribution of sulfate reduction is. Lee (2001) reported that in Korea the performance of an anaerobic constructed wetland, using pig manure as carbon source, could be enhanced by inoculating the wetland with SRB-rich anaerobic granular sludge from an anaerobic wastewater treatment plant at a brewery.

At the former Wheal Jane tin mine in the UK, a large-scale pilot plant passive system was installed to evaluate the most appropriate configuration for wetland remediation of this and similar drainage waters (Johnson, 2000). The plant was based on a combined treatment in an aerobic constructed wetland and a non-vegetated anaerobic compartment, a buried wetland to exclude oxygen introduction through plant roots. The aerobic wetland promoted iron and arsenic removal, and the anaerobic compartment, using mixtures of hay, sawdust and manure as both electron donor and inoculum, induced precipitation of copper, cadmium and zinc as metal sulfides. Rock filters were added as a final polishing step for Mn precipitation. The test showed that aerobic-anaerobic constructed wetlands are indeed efficient in treating AMD, especially if anoxic limestone drainage is used as pre-treatment. Zn, Cu and Cd removal was well over 99%. A serious drawback is however the large area required. Furthermore, a good understanding of the behavior of carbon sources in passive AMD treatment is still lacking (Pulles, 2000).

**Active (high-rate) treatment**

Numerous SRB reactor design studies have been reported including trench reactors (Younger et al., 1997), anaerobic filters (Chian & De Walle 1983), mixed (Maree & Hill 1989), anaerobic packed bed, fluidized bed (Umita et al., 1988), gas-lift (Du Preez & Maree, 1994; van Houten et al., 1994), sequencing batch (Herrera et al., 1991), UASB (Buisman et al., 1989; Barnes et al. (1991) and baffled (Grobicki & Stuckey, 1992) reactors. Notwithstanding the reactor type, and the particular treatment approach used, widespread application of active AMD treatment has not yet been seen.

A new low-cost process is the BioSURE Process, developed at Rhodes University in South Africa, which links AMD treatment and sewage sludge disposal (Rose et al., 2000). Sewage sludge is serving as the electron donor for the SRB and is simultaneously stabilized. Solubilization of complex carbon substrates provides the primary reaction in the “BioSURE” Process, and is effected in the Falling Sludge Bed Reactor (FSBR). In the FSBR, suspended solids settle and are then recycled to the inlet, large particles are hydrolyzed, while consumption of small organic compounds is inhibited within an increasing sulfide and alkalinity concentration gradient. After being recycled, the hydrolyzed compounds become available to sulfate reduction in a subsequent operation. Residual solids settle again and go through a further cycle of hydrolysis. The process was scaled-up to a pilot-plant at Grootvlei Mine (South Africa). Sulfide-rich process water may be blended to the influent minewater affecting the precipitation of contaminating heavy metals as metal sulfides, hydroxides and carbonates. During 18 months of operation, the process proved to be a reliable method for treating mine drainage wastewaters. Preliminary studies of sludge solubilization in the FSBR have demonstrated the role of sulfide and alkalinity, as physico-chemical effects enhancing enzymatic hydrolysis processes, and accelerating the breakdown of protein, carbohydrate and lignocellulose components in sewage sludge (Rose et al., 2000).

The NTBC Research Cooperation in Canada is the developer of the Biosulphide Process. Characteristic of this biogenic sulfide system is that it is divided into two stages: a biological stage, which is isolated from the chemical stage, which is the sulfide precipitation. The bioreactor(s) become essentially a reagent generating system (the reagents being
dissolved and gaseous sulfide and alkalinity). Hydrogen is used as electron donor in the bioreactors. The separation of the chemical and biological stages in this manner has several key advantages over conventional sulfate reduction systems: i) the entire flow of water for treatment is not passing through the slowest stage of the process (the bioreactors), ii) reactions in the two stages can proceed at their optimal (and different) rates, iii) the bacterial population is not exposed to inhibitory or toxic levels of dissolved metals, iv) a greater degree of control is possible over the extent of reactions in the two stages. Several large-scale pilot projects of the Biosulphide Process have been completed, but to our knowledge no full-scale plants have been built.

The Paques Thiopaq-process for sulfate removal is a biological process in which sulfate is converted into elemental sulfur. The process consists of two biological processes which take place in separate bioreactors. First, the sulfate is converted into sulfide; subsequently, the sulfide is converted into sulfur. This sulfur can be recycled for the production of sulfuric acid. Due to the production of alkalinity during the conversion of sulfide into sulfur, influent neutralization can be achieved by recirculation of this stream eliminating the need to add large amounts of alkaline chemicals. Several full-scale sulfate removal plants are currently in operation. For instance at the synthetic fiber production plant of Akzo Nobel in Emmen, The Netherlands, the sulfate containing wastewater has been treated by the Thiopaq-process since 1995. This full-scale installation is designed to treat 40 m$^3$/h of wastewater containing 2 g/l sulfate. Effectively, 75% of the sulfate is converted to elemental sulfur. Paques has also installed a groundwater treatment system, based on combined sulfate reduction and sulfide oxidation, for the Budelco zinc refinery in the Netherlands to remove sulfate, zinc and cadmium (Scheeren et al., 1993). The system has been in operation since 1992, treating a flow of 5000 m$^3$/d. More than 99% of the Zn and Cd is removed. Both metal sulfides and elemental sulfur are returned to the smelter. The metals are recovered and the sulfur is converted to sulfuric acid. The sulfate-reducing bioreactor is an upflow sludge bed system. Presently at the same refinery, a 500 m$^3$ gas-lift reactor is in operation for the reduction of zinc sulfate to zinc sulfide, which can be returned to the smelter. The electron donor is hydrogen. This process replaces the conventional method of lime dosing, leading to large amounts of waste (jarosite). It has been demonstrated that the Thiopaq process can also be successfully used for flue-gas desulfurization, and that it can form an attractive alternative to the conventional limestone-gypsum process (Janssen et al., 2000).

Potential limitations for sulfate-reducing bioreactors are product inhibition by unionized H$_2$S, too low biomass concentrations, insufficient mixing due to the low gas production and substrate transfer limitation if hydrogen is used as electron donor. These problems can be tackled by adjusting the reactor pH (sulfide toxicity), providing carrier materials or incorporating an efficient settler in the reactor (for biomass retention), and increasing the gasflow (for enhancing the transfer of hydrogen from the gas phase to the sludge in the liquid phase).

An interesting development is the Reverse Fluidized Loop Reactor (RFLR), developed at the Regional Research Laboratory in Trivandrum, India (Haridas et al., 2000). This system is tested for biological sulfide oxidation, but it can also be used for the treatment of metal-sulfate containing wastestreams. It is based on biomass retention through biofilm formation on floating carrier material. This reactor type resembles a gas lift reactor. However in this system, water enters at the top and leaves the reactor at the bottom. While the sludge floats, settling metal sulfide precipitates can be easily discharged with the effluent.

**Conclusions**

1. At this moment no effective tools exist to prevent sulfide production during the anaerobic treatment of sulfate-rich wastewaters. Sulfate reducers can utilize the two
key intermediates in anaerobic digestion, hydrogen and acetate, more effectively than methanogens.

2. The potential benefit of using sulfidogenesis as a biological process for treating a wide range of wastestreams with oxidized sulfurous compounds as well as heavy metals is well demonstrated. Heavy metals can be separated due to the formation and subsequent precipitation of poorly soluble metal sulfides. Basically two approaches can be distinguished: passive treatment using low-cost technologies and active treatment based on new bioreactor configurations. Hydrogen is an attractive electron donor in sulfidogenic bioreactors.

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