Reflections on anaerobic process biotechnology and its impact on water utilisation in South Africa*

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Abstract

Anaerobic processes that can be used for the prevention of salination, eutrophication, and organic pollution of water, are reviewed in this paper. Sulphate which contributes to the salination of surface water, can be converted under anaerobic conditions to H_2S, from which sulphur can be recovered. The removal of phosphates and nitrates is essential to control eutrophication. An anaerobic stage plays an important role in all biological processes by which these elements are removed. Organic material, originating from domestic and industrial effluents, and sewage sludge, are increasingly being treated in anaerobic processes because of low energy costs. Various processes and reactor types being used for this purpose are discussed.

Introduction

South Africa is semi-arid with limited water resources (Commission of Inquiry into Water Matters, 1970). The development of the country will result in water demand exceeding water supply before the year 2020 (Bekker, 1982). Hence, water has been identified as the country's most limiting natural resource which determines its human population 'carrying capacity' (Science Committee of the President's Council, 1983). The total water demand in South Africa for agriculture, housing, industrialisation and mining will increase rapidly as a result of the growing population (Science Committee of the President's Council, 1983; Marais, 1984). To protect the water quality of receiving streams, all effluents originating from domestic, industrial, agricultural and mining sources will have to be treated before disposal (e.g. Toerien, 1986).

The Commission of Inquiry into Water Matters (1970) outlined a strategy for the use of water resources, which is at present still totally applicable and can be summarised as:

- develop ways and means to use less water to do a given task;
- reuse water whenever possible; and
- develop new sources of water.

To achieve the ideal of optimal reuse of water, it is essential that applicable water treatment technologies are available in order that treated effluents can be reused directly, or be discharged back to the aquatic environment without serious and/or unacceptable water quality deterioration.

Anaerobic treatment, i.e. the use of biological processes in the absence of oxygen for the stabilisation of organic materials by conversion to methane and inorganic end-products, including carbon dioxide and ammonia, has been in use since 1881 (McCarty, 1982). Since then the application of anaerobic biological processes in waste and waste-water treatment has increased tremendously. It is, therefore, timeous to review the role of anaerobic processes in waste-water treatment, especially in South Africa, and to consider future trends. For this purpose, we will consider anaerobic treatment processes in relation to three major water quality problems facing the country: salination, eutrophication and organic pollution.

Removal of sulphates

Salination, often referred to as mineralisation, is one of the most important water quality problems in South Africa (Heynike, 1981; Water Research Commission, 1982). The average total dissolved solids (TDS) content of the water of the Vaal Barrage, one of the major water supply sources in the RSA, has increased from about 100 mg/ℓ in the early sixties to more than 400 mg/ℓ by the end of the seventies, and is expected to increase to 800 mg/ℓ by the year 2000. The cost implications of increased TDS concentrations were evaluated by Heynike (1981). He estimated an increased annual cost of some R139 million for consumers in the Pretoria-Witwatersrand-Vaal Triangle (PWV) complex, should the TDS concentration in Vaal Barrage water increase from 300 to 800 mg/ℓ. Sulphate would be an important contributor to such increased TDS concentrations.

Sulphate significantly affects the utility of many waters (Maree *et al.*, 1986). It is directly responsible for salination of receiving waters when discharged in excessive amounts but often constitutes an even greater indirect problem through salinity-associated corrosion, imparting of tastes to drinking water, scaling of pipes, boilers and heat exchangers, and giving rise to biocorrosion.

It may be removed from water by desalination processes such as reverse osmosis and electrodialysis, but these are costly. Hence, increasing attention has been given to biological sulphate removal (e.g. Middleton and Lawrence, 1977; Cork, 1982; Maree and Strydom, 1985; Olthof *et al.*, 1985; Maree *et al.*, 1986). Upflow packed bed reactors were used by Maree and Strydom (1985) to establish well-developed microbial biofilms for sulphate removal from mine water with either sugar, pulp mill effluent, or sewage as energy sources. They concluded that 1,6 g sugar, 16,7 mℓ of spent sulphite liquor or 172 mℓ raw sewage sludge were necessary to remove 1 800 mg sulphate. Sulphate reduction rates ranging from 0,18 g/ℓ.d (Widdel and Pfennig, 1977) through 0,29 g/ℓ.d (Middleton and Lawrence, 1977) and 4,5 g/ℓ.d (Lavalle, 1983) up to 6,3 g/ℓ.d (Cork and Cusanovich, 1978) have been observed.

Various studies have been completed on biological processes for the treatment of sulphur species. Maree *et al.* (1986) showed that a three-stage process (anaerobic – aerobic – anaerobic) employing upflow packed bed reactors (or sludge blanket reac-

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tors) for anaerobic treatment, and an activated sludge system for aerobic treatment (Figure 1), could be used for producing reusable water from mining effluents. Sulphate was reduced from 2,5 g/ℓ to less than 0,5 g/ℓ with concomitant removal of H_2S, carbonates, complexed cyanides, phenol and heavy metals. Molasses was used as the energy source. Olthof *et al.* (1985) described the 'Biosulfix' process in which H_2S gas produced during sulphate reduction is absorbed in sodium hydroxide and converted to sodium bisulphide. The latter product can be reused in pulp manufacturing. Cork (1982) proposed that H_2S and CO_2 removal from waste gases could be achieved through the use of *Chlorobium thiosulfatophilum*, a green photosynthetic sulphur bacterium. He suggested that such a biological process has much potential and that sulphur would be a by-product.

This technology could be of economic value to South Africa. Maree (1985) pointed out that 50% of the sulphur needs of the RSA has to be imported. The price of sulphur has increased from R37,36/t in 1980 to R274/t in 1985. He calculated that sulphur can probably be produced economically from the H_2S of sulphate reduction waste-water processes. There is little doubt that the biological removal of sulphate from polluted waters in the RSA is an exciting and potentially very important anaerobic biotechnology. The increasing attention given to this process illustrates the above conclusion.

Removal of phosphates

Eutrophication is a serious water quality problem in the RSA, as elsewhere in the world (Toerien, 1977). Phosphate removal from effluents in certain 'sensitive catchments' has been adopted by the Department of Water Affairs as a means of controlling eutrophication in the RSA (Wiechers *et al.*, 1984). Modified activated sludge processes have been developed to biologically remove phosphate from waste water (Barnard, 1976; Marais *et al.*, 1983; Comeau *et al.*, 1985). The Bardenpho process (Barnard, 1976; Figure 2) and UCT process (Marais *et al.*, 1983) are being used on an increasing scale in the RSA and the rest of the world to remove phosphate from sewage effluents.

Comeau *et al.* (1985) suggested (Figure 3) that bacteria which are able to store polyphosphates play an essential role in phosphate removal in activated sludge systems. Under anaerobic conditions these bacteria (which are probably obligate aerobes) take up acetate (and other fermentation products formed under the anaerobic conditions) at the expense of polyphosphate reserves with the result that phosphate is released by the cells. The acetate is stored intracellularly as poly-β-hydroxybutyrate (PHB). Under aerobic conditions, these cells oxidise the PHB to CO_2 and H_2O, and energy is utilised to take up and store polyphosphate intracellularly.

The balance between anaerobic and aerobic zones in biological phosphate removal plants is, therefore, largely instrumental in their success in phosphate removals (Gerber and Winter, 1985). A fuller understanding of the dynamics and mechanisms of the processes occurring under anaerobic conditions is still needed.

Removal of nitrates

Nitrogen is also an important nutrient contributing to eutrophication problems (Toerien, 1977). Its removal could therefore contribute to the control of the excessive growth of algae and/or macrophytes (e.g. Ashton, 1981).

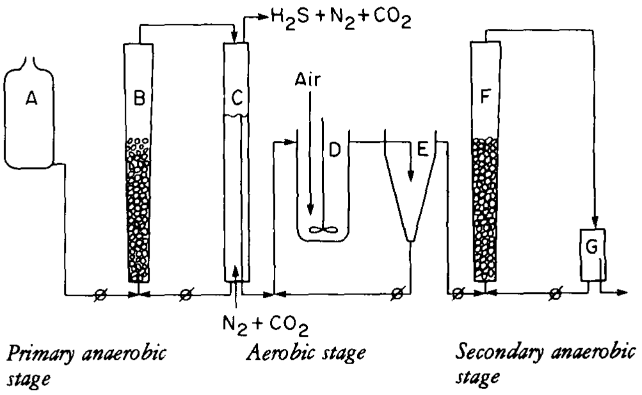


Figure 1
Schematic diagram of laboratory-scale plant used for biological sulphate reduction (Maree et al.*, 1986)*

- A. *Feed tank containing mine water and 3 mℓ/ℓ molasses*
- B. *Primary anaerobic reactor*
- C. *Stripping tower*
- D. *Aeration tank*
- E. *Settling tank*
- F. *Secondary anaerobic reactor*
- G. *Overflow chamber*

The activated sludge process was modified by Barnard (1974, 1975) to remove nitrogen in the Bardenpho process (Figure 2). Ammonia is oxidised to nitrate in the aerobic stage and is then recycled to an anoxic stage in which denitrification takes place (Barnard, 1975). In this way, effluents with low inorganic nitrogen concentrations can be discharged (Gerber and Winter, 1985). The sequence of nitrification – denitrification and the presence of fermentative metabolic end-products in the anoxic zone of activated sludge systems are prerequisites for denitrification. Most modified activated sludge systems for phosphate removal incorporate the removal of nitrate (e.g. Marais *et al.*, 1983).

Stabilisation of organic material

Most waste waters contain appreciable quantities of organic matter which, if not stabilised, will result in anaerobic conditions in receiving waters, proliferation of undesirable micro-organisms and many other detrimental effects. The stabilisation of organic matter is therefore a corner-stone of waste-water treatment and anaerobic biotechnology is an important component.

Anaerobic digestion of domestic waste waters

This process, in which a large proportion of the organic matter is converted to methane and CO_2 has been used for about 100 years to reduce the mass and putrescible nature of suspended organic material settled from municipal waste waters (McCarty, 1982). One of the first significant contributions was made by Louis Mouras who developed the 'Mouras Automatic Scavenger' (an air-tight chamber in which liquefaction of suspended organic material occurred) (McCarty, 1982).

This was followed by the development of 'Talbot tanks' in the USA in 1894 and the septic tank by Cameron in the UK in 1895 (McCarty, 1982). Methane gas from waste disposal tanks was already used for heating purposes in the 1890's in the UK and India (McCarty, 1982).

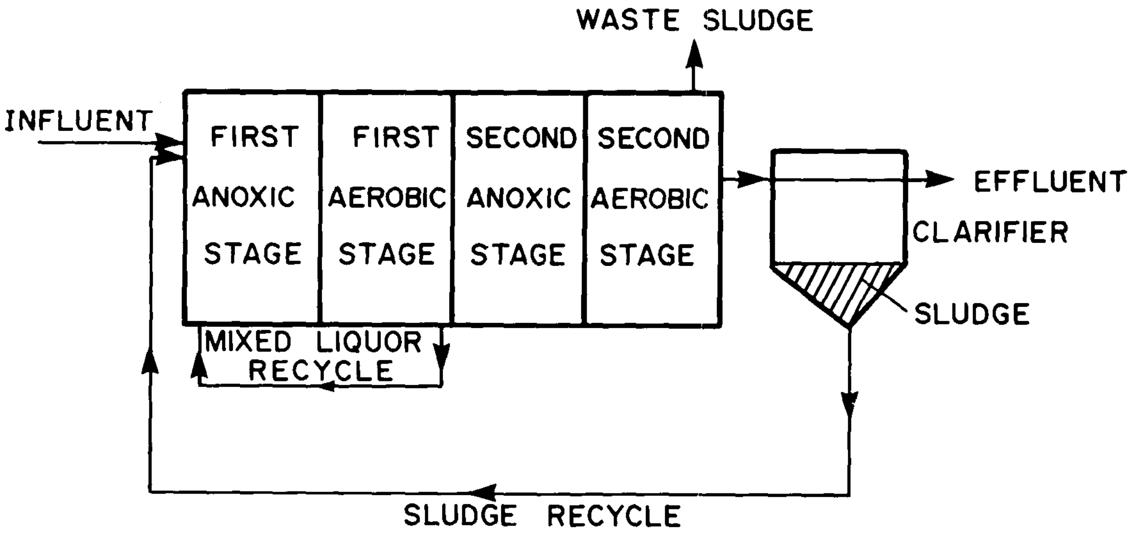


Figure 2
Flow schematic of the Bardenpho$^{(R)}$ process for biological phosphate and nitrogen removal.

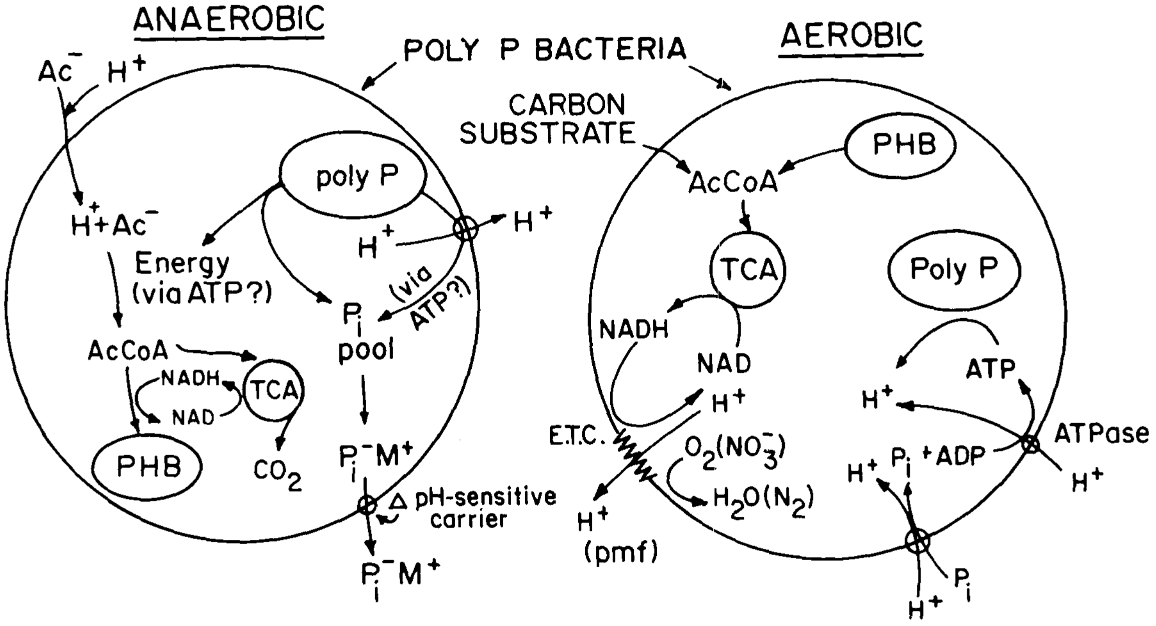


Figure 3
Postulated model for aerobic and anaerobic metabolism of polyphosphate bacteria (after Comeau et al., *1985).*

In 1904 Travis developed a two-stage process in which suspended solids were separated from the waste water and treated in a 'hydrolysing' chamber (McCarty, 1982). The Travis tank was modified in Germany by Imhoff and the Imhoff tank gained rapid favour, especially in the USA (McCarty, 1982).

The development of the first sludge-heating apparatus for separate digestion tanks improved the efficiency of stabilisation, and by 1934 about 5,6 million people in Germany alone were served by this process (Imhoff, 1938). The methane produced by the process was used to heat the digestion tank. Sufficient understanding had by then been developed to make digestion in separate heated tanks a well established process (McCarty, 1982).

Prior to 1950, most separate digesters did not employ mechanical mixing. This resulted in phase separation between solid and liquid, with a sludge layer at the bottom and a scum layer at the top of the tank (Figure 4). When mixing was introduced, it resulted in enhanced rates of digestion (McCarty, 1982). Most modern digesters employ some form of mixing (Figure 4).

Stabilisation of sewage sludges in the RSA is done mostly in separate heated anaerobic digesters. This represents one of the largest capital cost elements in the construction of local sewage treatment plants (McGlashan, 1986); therefore, effective operation of these systems or alternative treatment technologies are eagerly sought.

The process chemistry and microbiology of anaerobic digestion received much attention in the last fifty years. It is not within the scope of this paper to review this research except to refer to the importance of *'syntrophic associations'* (Bryant *et al.*, 1967; McInerney *et al.*, 1979; Boone and Bryant, 1980) in anaerobic digestion. These findings emphasise the importance of symbiotic relationships between bacteria which:

- must exist in anaerobic treatment;
- should be incorporated in practical considerations; and
- needs more research (McCarty, 1982).

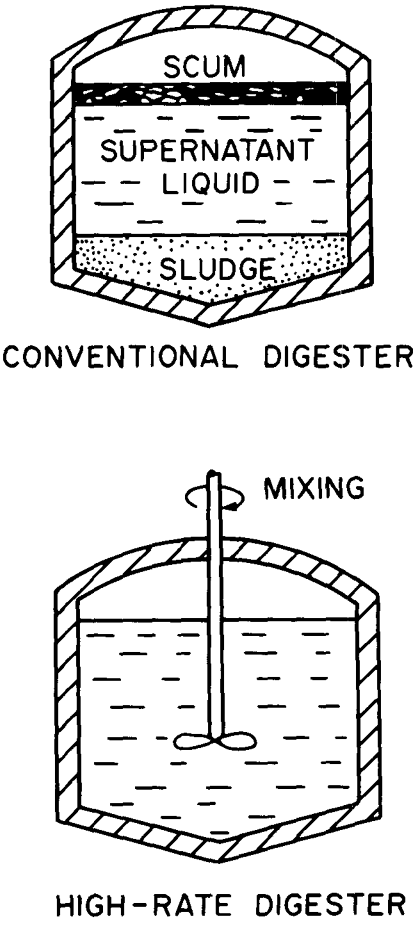


Figure 4
Conventional and high-rate separate digestion systems, commonly used for sewage sludge digestion (McCarty, 1982).

Anaerobic lagooning of domestic waste waters

An anaerobic pond system consists of an anaerobic section incorporating one or two small ponds in series followed by an aerobic section comprising one large pond followed by two or more ponds in series (Van Eck and Simpson, 1966). 'Stabilisation lagoons', which fall within the given definition, have been used for many decades to treat domestic waste water at Werribee, Melbourne (Parker *et al.*, 1950). Smells were an occasional problem and sulphides were present. Abbott (1962), using an anaerobic digestion pond system at Muizenberg, recirculated water from the secondary aerobic pond to the incoming sewage which entered a 'mixing pond'. No sulphides were detected in the Muizenberg pond system when recirculation was practised. Van Eck and Simpson (1966) indicated that anaerobic ponds have some utility, but that efficiency is impaired by low temperatures in winter.

Because of the problem of smells, the occurrence of unsightly scum layers and lower efficiencies in winter, the use of anaerobic ponds is limited. However, these systems can significantly reduce COD/BOD in waste waters with high organic loads.

Dual digestion of sewage sludge

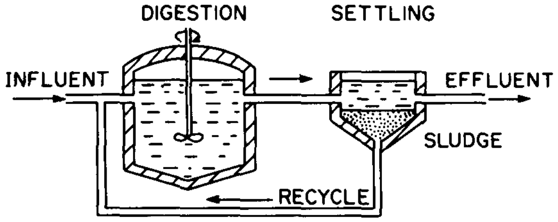
Sludge disinfection is a serious treatment problem because of the survival of pathogens and parasite eggs under anaerobic conditions in digesters and in drying beds. Dried stabilised sewage sludge can thus transmit disease and, therefore, sludge treatment processes which have the potential to eliminate pathogens during stabilisation receive research attention. 'Dual digestion' combines thermophilic aerobic digestion and anaerobic digestion in a two-step system (De Villiers, 1986). In this way the advantages of a degree of pasteurisation are coupled to lower energy requirements, process stability and reduced capital costs.

Anaerobic digestion of industrial waste waters

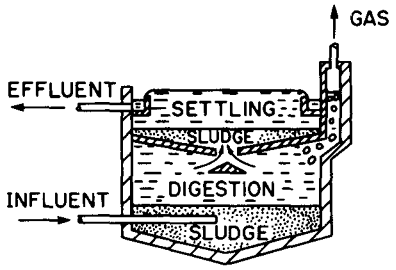
As pointed out previously, anaerobic industrial waste-water treatment uses less energy and generates less sludge than does aerobic treatment. Now that energy and sludge disposal are so costly, anaerobic treatment is gaining favour (Olthof and Oleszkiewicz, 1982). Some common misconceptions about anaerobic treatment are outlined in Table 1 and examples of industrial wastes treated successfully by anaerobic treatment in Table 2. It follows that many industrial wastes and waste streams are amenable to cost effective anaerobic treatment.

The interest in the anaerobic treatment of industrial waste waters has resulted in different reactor designs. Stander and Snyders (1950) and Stander (1966) recognised the need to maintain large bacterial populations in anaerobic digesters. The so-called 'clarigester' (Figure 5) was developed and the concept proven in full-scale treatment of winery wastes (Stander, 1966). Schroepfer *et al.* (1955) developed the 'anaerobic contact process' (Figure 5) for the treatment of packing-house wastes. Because these wastes are more dilute than municipal sludges, they used a settling tank to collect and recycle bacteria back to their reactor. Short hydraulic residence times could thus be obtained.

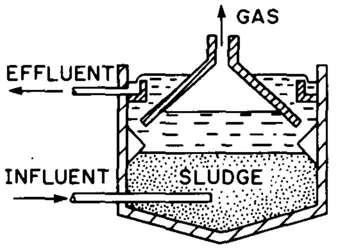
Young and McCarty (1969) developed the 'anaerobic filter' (Figure 6) which was used later to treat wheat starch waste water. It is a fixed-film reactor process which is analogous to the aerobic trickling filter process (McCarty, 1982). Switzenbaum and Jewell (1980) extended the above concept by the development of the 'anaerobic attached-film expanded bed' reactor (Figure 6) in which waste water passes in an upward direction through a bed of suspended media to which bacteria are attached. The 'upflow anaerobic sludge blanket' (Figure 5) which employs granular particles containing bacteria, was developed by Lettinga *et al.* (1979). This technology was recently imported to the RSA in order to construct a plant to treat brewery waste water at Prospecton, Natal (Hoffman, 1986).



ANAEROBIC CONTACT PROCESS



ANAEROBIC CLARIGESTER



UPFLOW ANAEROBIC SLUDGE BLANKET

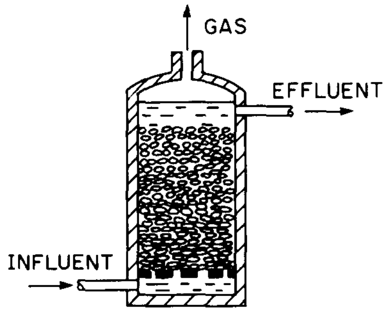
Figure 5
Suspended-growth digesters designed to maintain high bacterial populations, allowing digestion at shortened hydraulic detention times (McCarty, 1982).

TABLE 1
COMMON MISCONCEPTIONS ABOUT ANAEROBIC WASTE-WATER TREATMENT
(Olthof and Oleszkiewicz, 1982)

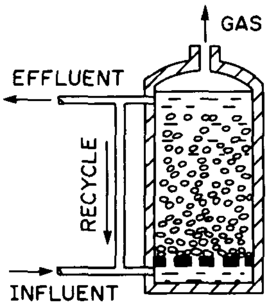
Misconceptions	Facts
Anaerobic treatment processes . . .	In fact, anaerobic treatment . . .
are applicable only to concentrated wastes, slurries and sludges;	has been applied to streams with COD as low as 1 000 mg/ℓ;
are not applicable to streams containing difficult-to-degrade organics;	may be acclimated to degrade organic compounds, even some that aerobic treatment cannot degrade;
are only applicable to streams having no suspended solids;	processes soluble wastes more quickly;
are slow, requiring 8-10 day retention times and, therefore, high reactor volume;	requires hydraulic retention times comparable to those in aerobic treatment;
are energy-inefficient, as the reactors must be heated; and	generates surplus energy when treating streams with more than 3 000 mg/ℓ COD; and
require costly chemicals for process control	requires only 10-20% of the nutrients that aerobic treatment needs, and controls alkalinity where needed by recycle

TABLE 2
EXAMPLES OF INDUSTRIAL WASTES TREATED SUCCESSFULLY BY ANAEROBIC TREATMENT
(Olthof *et al.*, 1984)

Wastes treated successfully
Methanol from cotton seeds
Yeast production
Citric acid production
Pulp and paper plant
Pharmaceutical waste
Piggery waste
Palm oil waste
Meat packing waste
Beet sugar wastes
Wine distillery
Guar gum
Barley stillage
Dairy waste
Cheese whey
Chemical industry waste



ANAEROBIC FILTER



ANAEROBIC ATTACHED-FILM
EXPANDED BED

Figure 6
Stationary-bed and expanded-bed fixed-film anaerobic reactors and treatment of relatively dilute waste waters (McCarty, 1982).

Anaerobic treatment of sludge thermal conditioning liquors

Thermal conditioning liquors (TCL) are generated at many municipal and industrial waste treatment facilities as a by-product of sludge dewatering operations (Crawford and Teletzke, 1986). As part of the thermal conditioning process, biological cells in the sludge are lysed and oxidised resulting in liquids containing high concentrations (10 000 to 20 000 mg/ℓ) of chemical oxygen demand (COD).

Hybrid anaerobic reactors, mostly with an upflow configuration having a filter above a sludge blanket reactor, have been proposed (Olthof and Oleszkiewicz, 1982; Crawford and Teletzke, 1986). For example, the HYAN reactor is in part an anaerobic filter to take advantage of its high performance and resistance to shock loadings. The sludge blanket zone is designed to produce gas and allow biomass production. Excess solids are removed from the lower zone in order to control sludge retention times and prevent sludge entering the filter bed. A small gas flow rate is used to agitate the filter zone to dislodge accumulated solids and to allow their return by gravity to their lower suspended growth zone. Recirculation liquid is withdrawn from below the filter zone and recirculated to the bottom of the sludge layers. The functions of the sludge and filter zones are, therefore, made truly interdependent (Crawford and Teletzke, 1986). The filter zone achieves high organic removals without accumulating excess solids. By continuously returning solids to the lower zone, high biomass concentrations are maintained resulting in efficient organic removal performances.

Industrial waste treatment with photosynthetic bacteria

The presence of photosynthetic sulphur bacterial blooms in anaerobic ponds receiving domestic or industrial wastes was described by Cooper (1963) and Sletten and Singer (1971). Cooper *et al.* (1975) reported significant reduction of total sulphide in settled fellmongery effluent by photosynthetic bacteria. Melcer and McFarlane (1977) found that anaerobic lagooning utilising photosynthetic bacteria was the cheapest and most effective treatment alternative (from catalytic oxidation, biological filtration and anaerobic lagooning). McFarlane and Melcer (1977) presented the theory and application of photosynthetic sulphur bacteria in anaerobic lagoons. They indicated that suitable conditions for the development of *Chromatiaceae* (a group of photosynthetic sulphur bacteria) included:

- underloaded anaerobic lagoons;
- overloaded facultative lagoons; and
- selective effluents.

Dominance of the photosynthetic bacteria can be manipulated through the use of retention time and organic loading rates. Wenke and Vogt (1981) indicated that photosynthetic sulphur bacteria of the species *Thiopedia rosea* can dominate a feedlot drainage lagoon. The bacterial chlorophyll concentration in the pond was negatively correlated with the dissolved oxygen concentrations in the lagoon, indicating the necessity of anaerobic conditions for the multiplication of *Thiopedia rosea*.

The utilisation of photosynthetic non-sulphur bacteria in foul water purification was described by Kobayashi (1975). In South Africa, the potential of this group was also investigated by Toerien (1976) and Toerien *et al.* (1982). Gaigher *et al.* (1985) indicated that photosynthetic non-sulphur bacteria could be used in the treatment of sorghum beer brewery effluent. The latter study incorporated a 2,25 m³ photosynthetic bacterial unit which discharged to an algal unit of similar capacity. Gaigher *et al.* (1985) suggested that their treatment system showed potential but its use should be further evaluated. Anaerobic treatment of high-strength wastes with photosynthetic bacteria (sulphur or non-sulphur types) appears to have potential; however, this needs further evaluation. Photosynthetic bacteria might prove to be a valuable proteinaceous by-product from waste-water treatment (Toerien *et al.*, 1982), but this topic also deserves further study.

Discussion

The application of anaerobic biotechnologies is receiving widespread attention world-wide. It is therefore ironic that South Africa is now apparently dependent upon foreign 'cutting edge' anaerobic technologies despite highly-rated research on anaerobic digestion in the 1950's and 1960's (e.g. Stander and Snyders, 1950; Stander, 1966; Hattingh *et al.*, 1967; Kotzé *et al.*, 1968; Thiel *et al.*, 1968; Toerien and Hattingh, 1969; Kotzé *et al.*, 1969). Neglect of research on anaerobic treatment in the 1970's and early 1980's has resulted in a knowledge gap which might be exacerbated by a lack of funds for water research (Toerien *et al.*, 1986). However, increased attention to anaerobic treatment processes at the National Institute for Water Research (e.g. Maree *et al.*, 1986; Ross, 1986) and the Universities of the Orange Free State (Britz, 1986), Cape Town (Marais, 1986) and Pretoria (Pretorius, 1986) could restore local expertise and experience in anaerobic biotechnology.

In this regard the utilisation of upflow anaerobic sludge blanket and anaerobic packed bed reactors for sulphate removal, stabilisation of organic wastes, and other industrial uses hold much promise. However, local experience with hybrid anaerobic reactors or anaerobic-aerobic systems is extremely limited, except for the use of sequential anaerobic and aerobic stages in phosphate-removing activated sludge systems.

Water, as a limiting factor in the RSA, dictates the need for effective waste and waste-water treatment. Economic considerations and limitations dictate the need to be effective towards energy and capital costs. Anaerobic biotechnology probably offers promise in all these areas, but has to be preceded by the necessary research into development of new and/or adaptation of existing technologies. The challenges facing South African scientists and engineers in this regard are formidable and cannot be postponed. A structured research and development plan is therefore needed.

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