Full-scale ANANOX® system performance

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Abstract

This paper reports the results of the first experimental investigations carried out on the only existing full-scale plant that makes use of the biological treatment system known as ANANOX®. This system was first set up by the Italian research staff at ENEA (Agency for New Technologies, Energy and Environment) and consists of two biological stages in series: a two-section ABR and an anoxic section followed by an activated sludge process. The investigation aimed primarily to assess system performance under uncontrolled load conditions. In particular, system efficiency was assessed with regard to carbonaceous and nitrogen compounds in the anaerobic and anoxic phases, and the role of sulphides in the denitrification process was examined.

The results obtained show the system's ability to ensure efficiency levels that comply with stringent effluent regulations while also allowing considerable savings in running costs.

Introduction

In the wastewater treatment field, systems based on anaerobic biological processes have traditionally been adopted to stabilise both primary and secondary waste sludge (Parkin and Owen, 1986), as this application is well-suited to the main requirements of anaerobic systems. These include:

- good removal ability of the biodegradable substrates;
- efficiency levels that are not excessively high;
- high production of biogas;
- and low running costs, mainly due to the lack of a forced aeration system.

However, over the last few years, the search for "sustainable" treatment systems capable of minimising energy consumption (Jetten et al., 1997) has encouraged the use of anaerobic biological systems even for intensive wastewater treatment, where the main goal is to eliminate the biodegradable dissolved fraction in carbonaceous substrates (Lettinga et al., 1979, Pfeiffer et al., 1986). Initially these applications were used for high-strength organic wastewaters (such as those produced by the food-processing industries, e.g. sugar refineries, distilleries, cheese factories, canning factories, etc.), as the effects of the intrinsically slow anaerobic processes are less serious (Malina and Pohland, 1992). Then, following the proposal of new and more efficient plant configurations, anaerobic systems were used also for the treatment of municipal wastewater, even though this contains a low organic substrate concentration, with a significantly high percentage of suspended and colloidal solids (Lettinga et al., 1981).

These innovative plant configurations are all characterised by a high substrate removal rate per unit reactor volume (kg COD m^{-3}·d^{-1}), obtained by retaining the biomass in the reactor

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independently of the incoming wastewater (solids retention time, SRT, is higher than hydraulic residence time, HRT). High-rate anaerobic biological systems may be classified into three broad groups, depending on the mechanism used to achieve biomass detention. These are fixed film, suspended growth, and hybrid systems (Barber and Stuckey, 1999). The full-scale systems that have found a wider application are those based on the upflow anaerobic sludge blanket (UASB), which is a suspended growth system developed in the Netherlands in the early 1980s (Lettinga et al., 1980). Anaerobic biological systems arranged in series are called ABRs (anaerobic baffled reactors). The ABR uses a series of baffles to force wastewater containing organic pollutants to flow under and over (or through) the baffles as it passes from the inlet to the outlet (McCarty and Bachmann, 1992). The main advantage of the ABR stems from the ability to separate the two biological processes of acid formation and methane formation in which the anaerobic removal of carbonaceous substrate takes place (Eastman and Ferguson, 1981; Weiland and Rozzi, 1991).
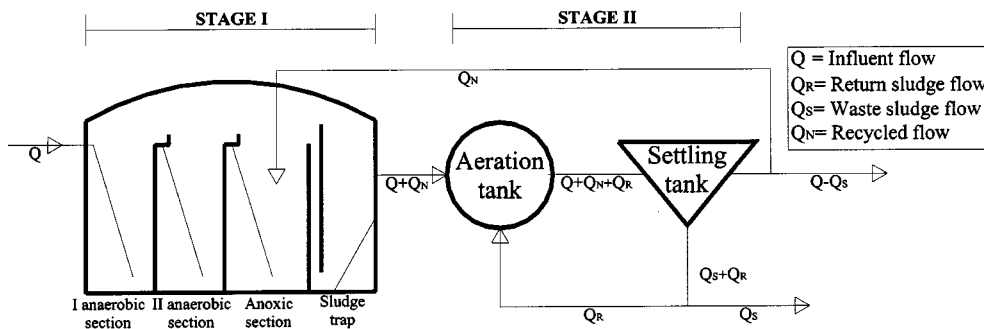
Innovative anaerobic biological systems guarantee a fairly good removal of carbonaceous matter (which may even reach high efficiency levels in the case of rapidly biodegradable substrates), but are markedly inadequate to remove nitrogen and phosphorus compounds. Consequently, use of the anaerobic system alone cannot guarantee compliance with legal standards, a goal that could be reached by using the so-called integrated systems in which anaerobic biological systems constitute only one of the stages in the treatment flow-sheet (Lettinga and Hulshoff Pol, 1991).

The integrated systems developed over the last few years differ according to the various treatment systems that they consist of and the substrates that they eliminate. With specific reference to wastewater treatment in small communities, from as far back as 1988 the research staff at Italy's ENEA Institute (Ente per le Nuove tecnologie, l'Energia e l'Ambiente) proposed the two-stage biological integrated system known as ANANOX® (ANaerobic-ANoxic-OXic - Garuti et al., 1992). The schematic diagram of the ANANOX® system is represented in Fig. 1. The first stage uses an ABR comprising two floc sludge blanket sections; one anoxic sludge blanket section; and a sludge trap. The second stage is fed

STAGE I | **STAGE II**

Q = Influent flow
Q_R = Return sludge flow
Q_S = Waste sludge flow
Q_N = Recycled flow

Figure 1
Schematic diagram of the ANANOX®
system

with the effluent from the first stage and is made up of an activated sludge aeration tank and a settling tank.

In the ABR the following processes primarily take place: ammonification of organic nitrogen; separation and hydrolysis of suspended organic solids; degradation of a part of the dissolved carbonaceous substance through the combined action of acid-forming and methane-forming micro-organisms; reduction of sulphates to sulphides through the action of sulphate reducing bacteria (SRB). The lower the COD content and the ratio between COD and sulphate concentration in the incoming wastewater, the greater the efficiency of the sulphate reduction process (Choi and Rim, 1991). The sludge trap has the only function of preventing any first-stage sludge that might have escaped from the blanket from being fed into the subsequent aerobic phase (Garuti et al., 1992).

The second stage consists of a classic activated sludge process, where oxidation of the residual carbonaceous fraction, ammonia nitrogen and sulphides occurs.

Denitrification takes place in the anoxic section of the first stage, where a portion of the effluent clarified in the second stage (and not the mixed liquor, as is often the case) is recycled. Although this solution overloads the settling tank, it makes it possible to reduce the amount of dissolved oxygen recycled to the anoxic section.

The configuration of the ANANOX® system with its anaerobic, anoxic and aerobic sections prevents biomass transfer; and the system can thus be classified as a "separated biomass" system.

The presence of anaerobic sections upstream of the anoxic section is a characteristic feature of ANANOX® operation, as it guarantees the availability of electron donors for the denitrification process even when there is little or no residual carbonaceous substrate in the ABR effluent. Under these conditions, the development of denitrifying heterotrophic biomass that uses short-chain fatty acids as electron donors is accompanied by the development of denitrifying autotrophic biomass that uses the sulphides produced by the SRBs in the anaerobic sections. In the presence of both these denitrifying micro-organisms, the denitrification rate is affected by physical and environmental parameters (detention time, temperature, pH, nitrate recycled, etc.) and is not influenced by possible imbalances in the C/N ratio in the incoming wastewater, a feature typical of traditional systems. With municipal wastewater the imbalance in the above ratio may even be manifested for only a few hours each day because of the oscillations in the hydraulic and organic load, which may assume important proportions for systems serving medium-sized and small communities and/or a rapidly variable population (Heduit et al., 1990). In these cases the adoption of traditional biological solutions has sometimes (Vismara, 1998) led to the inhibition of the denitrification process, a problem which can be solved only by introducing an accumulation and homogenisation (balancing) tank or by adding an external source of carbon (methanol), which has obvious economic implications (Van Haandel and Marais, 1981).

The ANANOX® system has so far been thoroughly tested on laboratory-scale pilot prototypes. In particular, during an extended investigation in 1990 and 1991 on a pilot-scale system installed at the waste treatment plant in the municipality of San Giovanni in Persiceto (Bologna Province - Italy), high values were obtained for the elimination of COD (89.6 %), Total Suspended Solids (TSS, 89.2 %) and total nitrogen (81.2 %). In addition, there was extremely small production of sludge (only 0.2 kg of TSS per kg of COD removed) and methane production equal to 0.103 m³ per kg of COD removed. Moreover, the sludge from the anoxic section was found to contain *Thiobacillus denitrificans*, which under anoxic conditions can achieve denitrification by oxidising sulphides into sulphates (Garuti et al., 1992).

Once the successful operation of the ANANOX® system had been ascertained, it was decided to assess its performance using a full-scale system operating under uncontrolled load conditions. The primary objectives of the experiments were to analyse ABR operation in relation to biomass concentration in the sludge blanket and wastewater upflow velocity, and to examine the role played by sulphides in the denitrification process.

To this end, in the municipality of San Giovanni in Persiceto, an ANANOX® system was built for the treatment of wastewater from the village of Biancolina, which has 350 population equivalents (p.e.). The system recently became operational and an initial series of experimental investigations was carried out between July and October 1998 in order to:

- systematically determine the concentrations and removal efficiencies in the various phases of the plant for COD, TSS, volatile suspended solids (VSS), and nitrogen and phosphorus compounds;
- establish the influence exerted by biomass concentration in the sludge blanket on the overall efficiency of the anaerobic phase;
- define the function of sulphides in the denitrification process.

Materials and methods

The flow sheet of the Biancolina wastewater treatment plant is shown in Fig. 2. The wastewater flows through a sanitary sewer network (total length approximately 2 km) into a pumping shaft (measuring 0.12 m × 0.12 m × 0.25 m), housing a basket screen and two submersed, open-impeller electro-pumps, with a rated power of 1.05 kW each. The pumps are controlled by a ball cock that effectively regulates the discontinuous feed of wastewater to the system.

The wastewater pumped from the pumping shaft is first subjected to micro-screening and degritting using a drum filter and a channel basin, respectively. The wastewater is uniformly distributed over the bottom of the first section of the ABR (2.80 m × 1.42 m × 2.05 m) beneath the sludge blanket via a system of six PVC pipes measuring 0.075 m in diameter. The effluent is then fed into the

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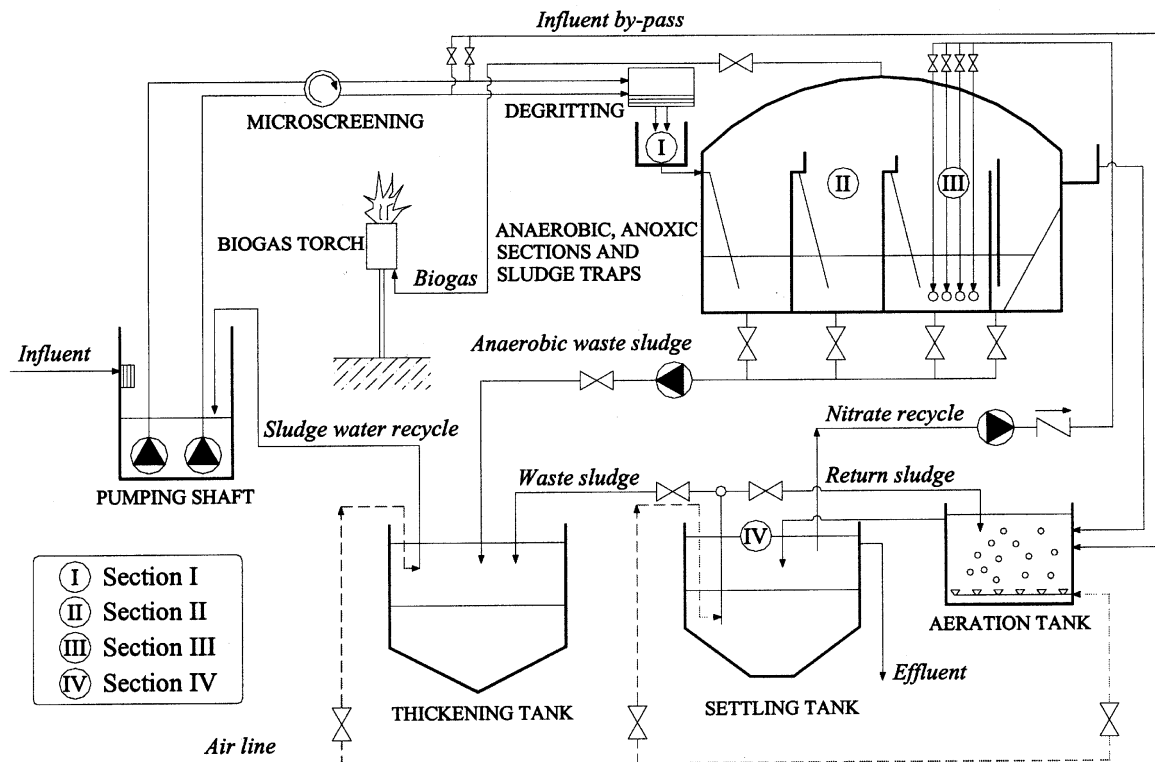
Figure 2
Flow sheet of the Biancolina wastewater treatment plant

second section of the ABR (2.80 m × 1.42 m × 1.90 m) and then into the anoxic section (2.80 m × 1.70 m × 1.75 m), in both cases using distribution systems similar to those described above. The anoxic section also receives the clarified effluent which is recycled via a self-priming centrifugal pump. The sludge trap (2.80 m × 1.37 m × 1.65 m) is fed from a weir.

The first stage of the ANANOX® system is situated under an air-tight, twin-wall PVC dome with the cavity between the walls filled with air. By regulating the air pressure it is possible to control the internal pressure of gas produced by the anaerobic and anoxic processes, in order to ensure that it can be collected safely. The gas, containing a high level of methane, is piped to the torch when the pressure inside the dome is greater than 200 mm water column.

The aerobic phase is carried out in a rectangular prismatic basin, measuring 1.40 m × 3.60 m × 3.00 m high, with 10 fine bubble diffusers fed by two blowers with a rated power of 3 kW each. In addition to the wastewater, the aerobic phase also receives the recycling sludge flow which is continuously extracted from the bottom of the settling tank by means of an air-lift system. This vertical flow tank has a rectangular base measuring 3.30 m × 3.60 m and a total height of 2.70 m. The clarified effluent overflows into the perimeter collection channels and is then discharged into a drainage channel just outside the plant area.

The waste sludge from the biological processes (both anaerobic and aerobic) is extracted periodically and transferred to a rectangular thickening tank measuring 3.05 m × 3.60 m × 3.00 m.

The arrangement of the various phases making up the system is extremely compact, covering a total surface area of 56 m².

The flow rate and temperature of the wastewater entering the plant were continuously monitored throughout the experimental period, giving constant values in terms of daily volume (12.5 m³) and temperature (15°C). The presence of ball cocks in the pumping shaft and the powerful pumps ensured that the wastewater volume

was always fed to the ANANOX® system in a number of brief pulses for a total of approximately one hour a day. During these periods, the instantaneous flow rate was equal to 10 m³·h⁻¹, corresponding to an upflow velocity in the two ABR sections of 2.5 m·h⁻¹, which is much higher than the maximum values referred to in the literature on floc sludge blankets, of approximately 0.5 m·h⁻¹ (Lettinga and Hulshoff Pol, 1991).

During the experimental period, biomass concentrations were kept constant in the anoxic and aerobic sections at 1.5 g TS·kg⁻¹ wet mass and 6 900 g TSS·m⁻³, respectively (where TS indicate total solids). Some of the operating parameters that were modified include: biomass concentration in the sludge blanket of the two ABR sections; air flow rate to the aerobic section; and daily volume of the recycling nitrates flow. In this paper, we refer to two different periods of experiments, covering the months of July/August (Period I) and September/October (Period II), the operating conditions of which are summarised in Table 1.

Biomass concentration in the ABR was reduced in order to limit sludge wash-out due to the high upflow velocity value. In the belief that it would probably improve efficiency in the anaerobic process (subsequently confirmed by experimentation), the air flow to the biological aerobic phase was reduced from 14.0 m³·h⁻¹ to 10.5 m³·h⁻¹, representing a decrease of 25%. In this connection it should be pointed out that blower operation was regulated by a timer according to the sewage flow rate and so the above air flow rate was actually maintained for only 13 h·d⁻¹.

During experimentation, the flow rate of the clarified effluent recycled to the anoxic phase was constantly monitored. Samples of the flow were taken from the system's four different characteristic sections on a weekly basis. These sections, which are illustrated in Fig. 2, are as follows: the inlet to the first ABR section (Section I); the outlet from the second ABR section (Section II); the outlet from the anoxic section (Section III); the outlet from the settling tank

TABLE 1
Operating conditions of the ANANOX® system during the two experimental periods

| Period	Upflow velocity in the two section of ABR	Biomass concentration in the sludge blanket of the ABR		Biomass concentration in the anoxic section	Biomass concentration in the aerobic section	Air flow rate to the aeration section	Daily volume of the recycling nitrates flow
		I section	II section				
Jul.-Aug. (I)	2.5 m·h⁻¹	72.1 g TS·kg⁻¹	40.0 g TS·kg⁻¹	1.5 g TS·kg⁻¹	6 900 g·m⁻³	14.0 m³·h⁻¹	10.0 m³
Sept.-Oct. (II)	2.5 m·h⁻¹	37.8 g TS·kg⁻¹	23.9 g TS·kg⁻¹	1.5 g TS·kg⁻¹	6 900 g·m⁻³	10.5 m³·h⁻¹	12.5 m³
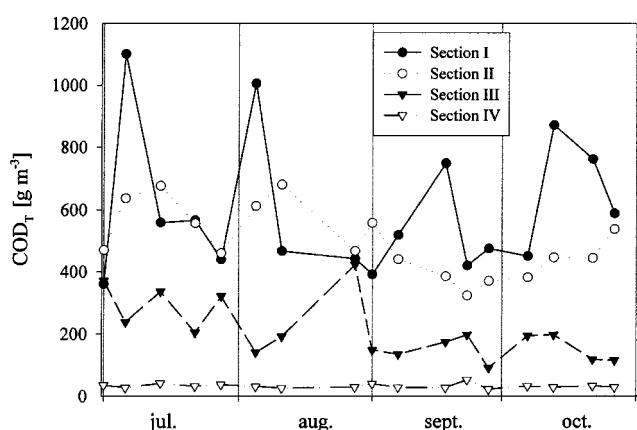
Figure 3
Measured total COD (COD$_T$) concentrations for the ANANOX® system from July to October 1998
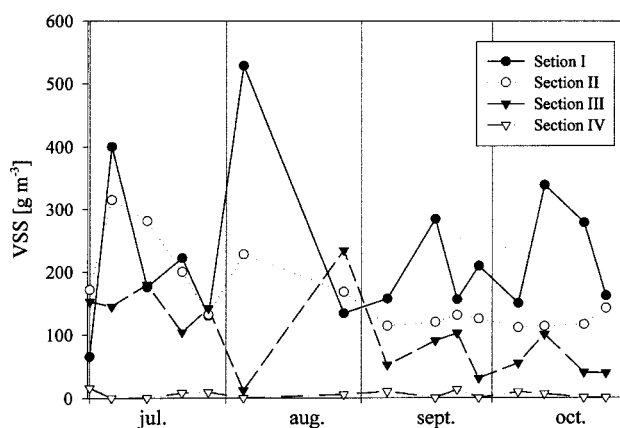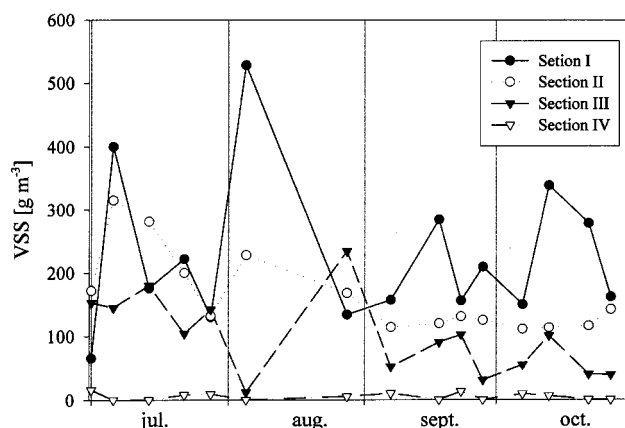

Figure 5
Measured total suspended solids (TSS) concentrations for the ANANOX® system from July to October 1998
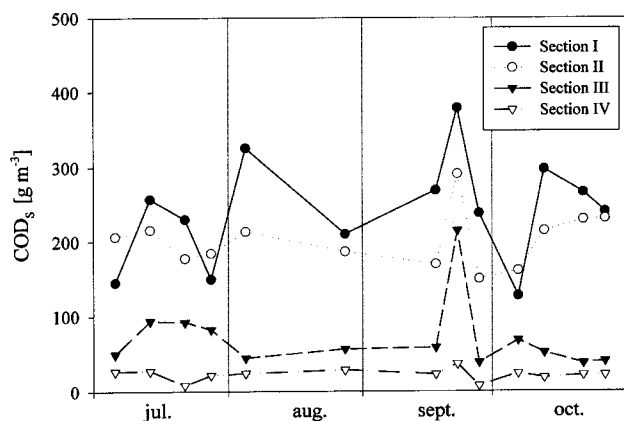
Figure 4
Measured soluble COD (COD$_S$) concentrations for the ANANOX® system from July to October 1998

Figure 6
Measured volatile suspended solids (VSS) concentrations for the ANANOX® system from July to October 1998

(Section IV). Sampling was always carried out while the pumps were operating.

The following parameters were determined for all the samples taken: total COD (COD$_T$), soluble COD (COD$_S$ – obtained by passing through an 0.2 mm filter), TSS, VSS, ammonia nitrogen (N-NH$_4^+$); the concentration of nitrate nitrogen (N-NO$_3$) was determined on the samples of Sections III and IV while the concentration of the total phosphorus (P$_T$) was measured on some of the samples taken

in Sections I and IV. All these analyses were carried out according to the procedures described in *Standard Methods* (1989).

In order to define the role played by sulphides in the denitrification process, the ABR effluent and the first stage effluent were analysed to identify the concentrations of COD$_T$, sulphates (SO$_4^{2-}$) and sulphides (H$_2$S) during a time interval (about 48 min) between two successive activations of the plant's pumping system. The concentration of N-NO$_3$ was also measured in the first stage

TABLE 2
Mean value and std dev of COD$_T$, COD$_S$, TSS and VSS in the four sampling sections

Period	Parameter	Unit	Sampling sections			
			I	II	III	IV
Jul. – Aug. (I)	COD$_T$	g·m^{-3}	597.6 (±270)	567.9 (±88)	263.5 (±102)	32.6 (±5)
	COD$_S$	g·m^{-3}	219.5 (±68)	197.3 (±16)	69.8 (±22)	22.5 (±7.5)
	TSS	g·m^{-3}	302.2 (±214)	295.1 (±87)	184.1 (±92)	12.1 (±5)
	VSS	g·m^{-3}	236.5 (±166)	213.7 (±65)	138.6 (±69)	5.4 (±6)
Sept. – Oct. (II)	COD$_T$	g·m^{-3}	603.7 (±172)	415.7 (±77)	152.3 (±40)	30.7(±9)
	COD$_S$	g·m^{-3}	259.4 (±75)	206.6 (±49)	48.9 (±12)	21.6 (±8)
	TSS	g·m^{-3}	285.4 (±108)	156.6 (±28)	72.3.6 (±27)	11.0 (±4)
	VSS	g·m^{-3}	216.8 (±73)	121.4 (±24)	58.5 (±27)	4.8 (±5)

TABLE 3
Removal efficiencies of COD$_T$, COD$_S$, TSS and VSS in the four sampling sections

Period	Parameter	h$_{II}$	h$_{III}$	h$_{IV}$	h$_{TOT}$
Jul. – Aug. (I)	COD$_T$	5.0 %	20.2 %	87.6 %	95.0 %
	COD$_S$	10.1 %	41.8 %	67.2 %	89.6 %
	TSS	2.4 %	-8.7 %	93.4 %	96.7 %
	VSS	9.6 %	-14.4 %	96.1 %	98.4 %
Sept. – Oct. (II)	COD$_T$	31.2 %	37.7 %	79.8 %	94.9 %
	COD$_S$	20.3 %	60.7 %	55.7 %	91.7 %
	TSS	45.1 %	21.4 %	84.8 %	96.1 %
	VSS	44.0 %	16.0 %	91.9 %	97.8 %

effluent, while measurements of N-NO$_3$ and COD$_T$ were also taken on the clarified effluent leaving the ANANOX® system (partly recycled), in order to make possible the application of the mass balance equations for different substrates. In particular, the determinations of H$_2$S were conducted directly on the plant system using a field instrument (Spectro2) so as to prevent it from oxidising in contact with the air. During the experimental time interval, the incoming wastewater pumping system was operating for the first 5.5 min and the nitrate recycle pump was always functioning.

Results and discussion

As already mentioned, the measurements taken refer to COD$_T$, COD$_S$, TSS, VSS, and the compounds of nitrogen and sulphur. Hereafter, the results obtained during experiments are presented and discussed separately for the various parameters, starting with the first four which are considered together.

COD$_T$, COD$_S$, TSS and VSS

The values of COD$_T$, COD$_S$, TSS and VSS measured during experiments in the four sampling sections in the ANANOX® system are presented in Figs. 3, 4, 5 and 6, respectively. The relative mean values are reported in Table 2, while Table 3 reports the removal efficiencies. In the latter table, h_{II}, h_{III}, h_{IV} and h_{TOT} are used to indicate the removal efficiencies between Sections I and II, II and

III, III and IV and I and IV, calculated using the following equations (the second equations takes into account the recycling nitrates flow):

$$\eta_{II} = \frac{S_I - S_{II}}{S_I} \cdot 100 \tag{1}$$

$$\eta_{III} = \frac{(Q \cdot S_{II} + Q_N \cdot S_{IV}) - (Q + Q_N) \cdot S_{III}}{(Q \cdot S_{II} + Q_N \cdot S_{IV})} \cdot 100 \tag{2}$$

$$\eta_{IV} = \frac{S_{III} - S_{IV}}{S_{III}} \cdot 100 \tag{3}$$

$$\eta_{TOT} = \frac{S_I - S_{IV}}{S_I} \cdot 100 \tag{4}$$

where:

$S_I, S_{II}, S_{III}, S_{IV}$ = concentrations of a generic parameter in the sampling Sections I, II, III and IV, respectively [g m^{-3}]
Q = influent daily flow rate [m^3 d^{-1}]
Q_N = daily flow rate of recycling nitrates [m^3 d^{-1}].

An examination of the figures and tables reveals that COD$_T$ in the untreated sewage is variable during experiments. Nevertheless the mean values related to the experimental Periods I and II are very close to one another (597.6 g COD$_T$ m^{-3} and 603.7 g COD$_T$ m^{-3}).
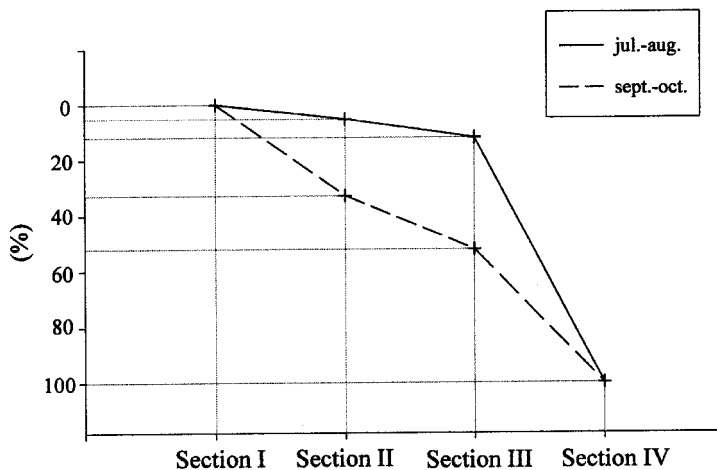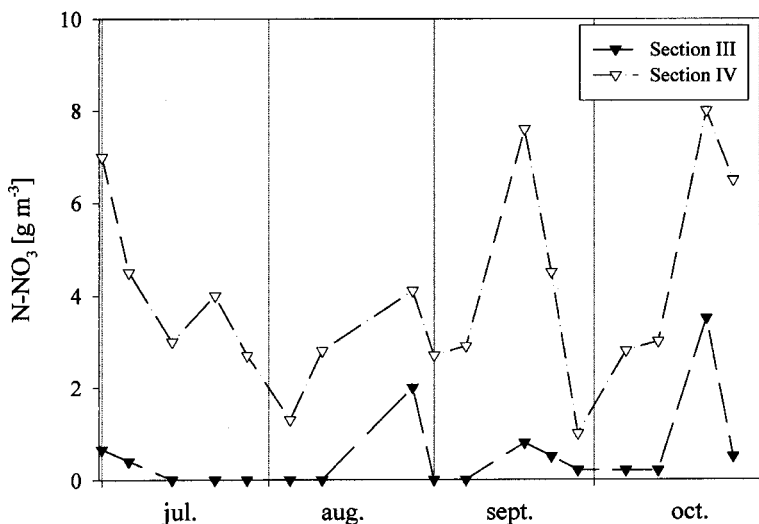
Figure 7
COD$_T$ overall removal efficiency (%) repartition between the individual phases of the ANANOX® system
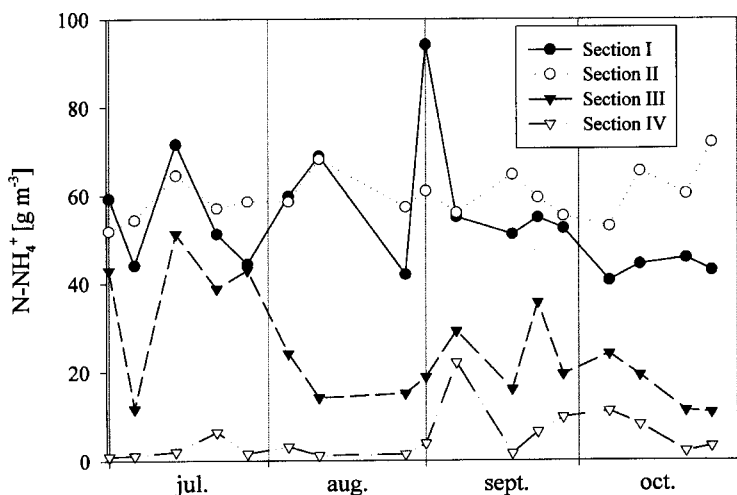
Figure 8
Measured ammonia nitrogen (N-NH$_4^+$) concentrations for the ANANOX® system from July to October 1998

Figure 9
Measured nitrate nitrogen (N-NO$_3$) concentrations for the ANANOX® system from July to October 1998

The COD$_T$ values measured in Section II present a much more uniform pattern because of the damping imposed by hydraulic factors and also the biochemical processes developing in the ABR. However, the concentrations detected during the Period II were generally lower than those in the Period I, with the mean value falling from 567.9 g COD$_T$ ·m^{-3} to 415.7 g COD$_T$ ·m^{-3}. Correspondingly, the COD$_T$ mean efficiency removal increased from 5% to 31.2%. This result must be attributed to the reduced biomass concentration in the ABR sludge blanket which made it possible to restrict solids wash-out to the ABR effluent; this was responsible for the high values of COD$_T$ measured in Section II during the Period I, sometimes even higher than those of the incoming sewage. This result is confirmed by the TSS measurements. In fact, despite the mean concentration in the incoming sewage being very close during the two experimental periods, and the different nature of the suspended solids in sections I and II, the removal of TSS in the ABR improved markedly, from Period I to Period II, increasing from 2.4% to about 45%. Moreover, the latter value must be considered to be satisfactory as it was obtained at a high upflow velocity (2.5 m·h^{-1}). This can be also confirmed by analysing the COD$_S$ measurements in Section II, which were more uniform than those of COD$_T$, with mean values ranging from 197.3 g COD$_S$·m^{-3} for Period I to 206.6 g COD$_S$·m^{-3} for Period II. These measurements are not affected by biomass wash-out, but stem from the set of biological processes to which the carbonaceous substance in the untreated sewage is subjected in the ABR, i.e. degradation of the dissolved fraction and hydrolysis of the suspended fraction.

The COD values measured in Section III were much lower than those in the ABR effluent, because of both the dilution due to the clarified effluent recycled to the anoxic section and the consumption of the carbonaceous substrate in the denitrification process. During trials, COD$_S$ removal across the anoxic phase increased from 41.8% for Period I to 60.7% for Period II. This increase is due to the increase in the daily volume of clarified effluent recycled to the anoxic section.

In terms of carbonaceous substance removal, the modifications made to the system's operating conditions brought about a marked improvement in the overall performance of the first stage of the ANANOX® system, with the resulting decrease in the load fed to the aerobic phase in the second stage. The efficiency of the latter gradually decreased, but nevertheless ensured consistently low COD$_T$ concentrations in the clarified effluent leaving the ANANOX® system (always below 40 g COD$_T$ ·m^{-3}) throughout the experimental period. The overall COD$_T$ removal efficiency of the ANANOX® system was always around 95%. Limiting the organic load removed in the aerobic phase has obvious benefits in terms of plant running costs. These can be clearly seen in Fig. 7 which reports the overall efficiency percentages for COD$_T$ removal attributable to the individual phases of the ANANOX® system, for

each of the two experimental periods. As can be seen in Fig. 7, during Period I the percentages of organic load removed by the anaerobic, anoxic and aerobic phases were 5.2%, 7.3% and 87.5%, respectively. During the Period II the organic load removal increased in the first two phases (anaerobic, anoxic) to 32.8% and 19.4% respectively, while the organic load removal decreased during the aerobic phase to 47.8%.

Nitrogen and phosphorus compounds

Analysis of the nitrogen compounds made it possible to characterise the reactions and the relative performance levels of: ammonification in the anaerobic process; denitrification in the anoxic process; nitrification in the aerobic process.

Ammonia nitrogen concentrations in Section I were usually lower than those measured in Section II (Fig. 8), with mean values of 51.1 g $N \cdot m^{-3}$ and 64.3 g $N \cdot m^{-3}$, respectively. This trend was achieved despite the removal of nitrogen for the synthesis of new biomass in the two sections of the ABR. It is due to the short length of the sewer network, which allows the high organic nitrogen substrate in the sewage to be converted to ammonia nitrogen primarily in the ABR.

The decrease in ammonia nitrogen concentrations in Section III is, to a small extent, due to the synthesis processes developing in the anoxic section, but must be primarily attributed to the dilution caused by the clarified effluent recycled to the anoxic section where the content of $N-NH_4^+$ is very low. The mean value of $N-NH_4^+$ measured in Section III was 25.0 g $N \cdot m^{-3}$, less than half the value measured in Section II.

In the aerobic phase of the second stage, the nitrification process always reduced the ammonia to very low concentrations (mean value 5.0 g $N \cdot m^{-3}$) with removal efficiency of 80% (Fig. 8). This result is fully justified since the organic load, characterising the activated sludge process, was always lower than 0.03 kg $COD_T \cdot d^{-1} \cdot kg^{-1}$ TSS and the sludge age was about 45 d. Under these conditions the aerobic waste sludge also turns out to be stable, as is shown by the VSS content, equal to 60% of the TSS (for the anaerobic sludge the VS/TS ratio gave a value of 62%).

The nitrate nitrogen concentrations measured in the clarified effluent recycled from the settling tank (Section IV) and in the first stage effluent (Section III) are reported in Fig. 9. As can be seen, the values are fairly consistent and with mean values of 4.0 g $N \cdot m^{-3}$ and 0.5 g $N \cdot m^{-3}$ in Section III and IV, respectively. Taking into account the dilution to which the nitrates entering the anoxic section are subjected, these values correspond to a denitrification performance of approximately 58.1%, which is certainly a satisfactory result, considering the low recycling ratio (about 1) adopted. The denitrification rate, defined as the mass of nitrates reduced in the time unit per volume unit in the anoxic section, turned out to be not too high because of the effect of both the low nitrogen loads in the
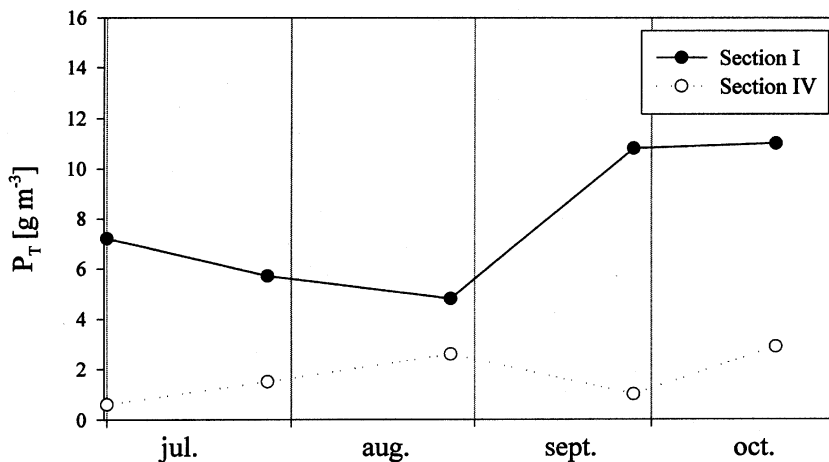

Figure 10
Measured total phosphorus (P_T) concentrations for the ANANOX® system from July to October 1998
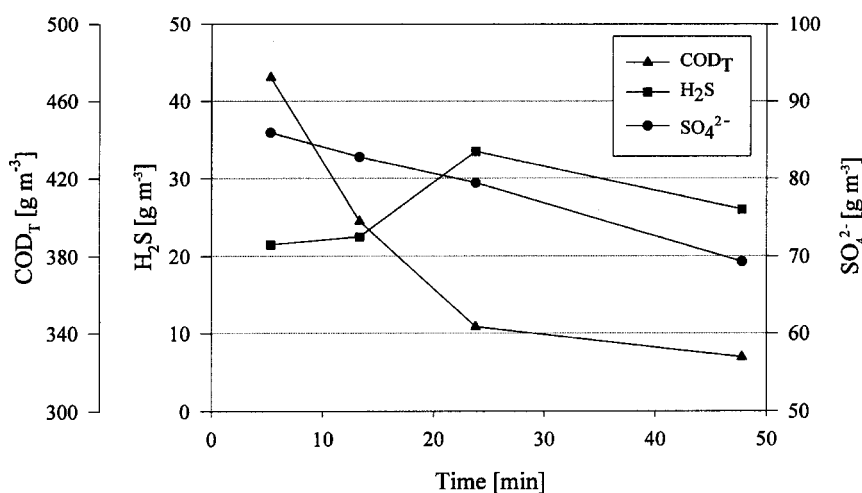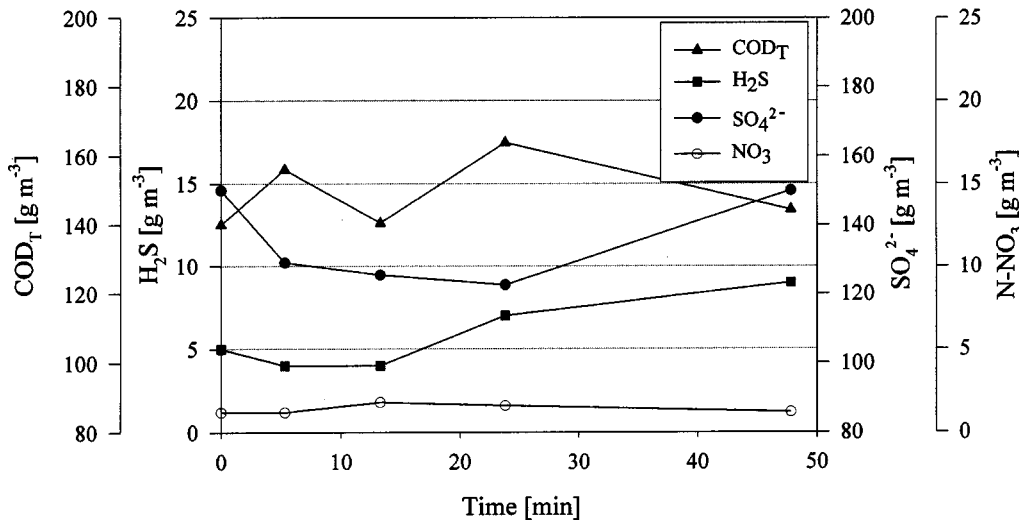

Figure 11
Trends of COD_T, H_2S and SO_4^{2-} in the second anaerobic section

incoming flow and the low recycling ratio. In any case, the concentration of $N-NO_3$ in the clarified effluent leaving the ANANOX® system was always fairly low, with values of the total nitrogen consistently less than 5 g $N \cdot m^{-3}$.

The total phosphorus concentrations were determined five times during the experimental period in Sections I and IV. The relative values are reported in Fig. 10. As can be seen, consistently low concentrations were measured in the clarified effluent, giving a mean value of 1.7 g $P \cdot m^{-3}$. The average P_T removal efficiency was around 70%, which is a satisfactory result for a plant not intended for the biological removal of phosphorus. It is hypothesised that this result can be attributed to the consumption of P in synthesis processes as well as chemical precipitation processes in the anaerobic sections. This needs to be verified by measuring the concentrations of phosphorus compounds in the effluent from the various sections and then applying a series of mass balances.

Figure 12
Trends of COD_T, H_2S, SO_4^{2-} and N-NO₃ in the anoxic section

Sulphur compounds

The trends of COD_T, H_2S, SO_4^{2-} and N-NO₃ measured during the test aiming to establish the role of sulphides in the denitrification process are reported in Figs. 11 and 12.

In particular, Fig. 11 shows that the concentration of sulphates in the second anaerobic section gradually decreased and followed a linear trend dropping from about 86 g SO_4^{2-}·m⁻³ to less than 70 g SO_4^{2-}·m⁻³ at a consumption rate, $F_{SO_4^{2-}}$, of 23.4 g SO_4^{2-} m⁻³·h⁻¹, when the plant was not being fed with wastewater. At the same time, the sulphide content increased at a mean rate, F_{H_2S}, of 7.2 g H_2S·m⁻³·h⁻¹. However, the experimental data for sulphides are less reliable because of the method adopted for the analytical measurements. Figure 11 shows also that the decrease in sulphates took place at the same time as the degradation of a portion of the organic substrate, approximately a third of the COD_T entering the system.

Analysis of Fig. 12 shows that the concentration of sulphates in the anoxic section increased after the incoming wastewater pumping system was turned off (about 5.5 min). This is due to the oxidation of the sulphides during the denitrification process. Moreover, the concentration of sulphates was much higher than those measured in the ABR effluent because of their high content in the clarified effluent recycled from the settling tank. However, contrary to what had been expected, sulphide concentrations in Section II during the test were characterised by an increasing trend, which has already been observed in the past by Gommers et al. (1988). The reason for this trend continues to elude researchers. Gommers et al. (1988) claimed that it is due to the ability of the biomass to adapt, converting its own electron donors into acceptors.

Using the values of COD_T and N-NO₃ measured in the clarified effluent recycled from the settling tank (Section IV) made it possible, on the basis of the mass balances applied between Sections II and III, to evaluate the efficiency and the reaction rates of the processes taking place in the anoxic section, i.e. denitrification, oxidation of organic carbon and sulphides. In particular, COD_T measured in Section IV was constant and equal to 20 g COD_T·m⁻³; N-NO₃ measured in Section IV had a linear trend, dropping from 6.7 g N·m⁻³ to 4.7 g N·m⁻³. The efficiency was calculated by means of the following equations (in which the time integrals are all extended to the 48 min):

$$\eta_{N-NO_3} = \frac{\int_0^{48}\{Q_N(t)\cdot[(N-NO_3)(t)]_{IV}\}\cdot dt - \int_0^{48}[Q(t)+Q_N(t)]\cdot[(N-NO_3)(t)]_{III}\cdot dt}{\int_0^{48}\{Q_N(t)\cdot[(N-NO_3)(t)]_{IV}\}\cdot dt}\cdot 100 \qquad (5)$$

$$\eta_{COD_T} = \frac{\int_0^{48}\{Q(t)\cdot[COD_T(t)]_{II}+Q_N(t)\cdot[COD_T(t)]_{IV}\}\cdot dt - \int_0^{48}[Q(t)+Q_N(t)]\cdot[COD_T(t)]_{III}\cdot dt}{\int_0^{48}\{Q(t)\cdot[COD_T(t)]_{II}+Q_N(t)\cdot[COD_T(t)]_{IV}\}\cdot dt}\cdot 100 \qquad (6)$$

$$\eta_{H_2S} = \frac{\int_0^{48}\{Q(t)\cdot[H_2S(t)]_{II}\}\cdot dt - \int_0^{48}[Q(t)+Q_N(t)]\cdot[H_2S(t)]_{III}\cdot dt}{\int_0^{48}\{Q(t)\cdot[H_2S(t)]_{II}\}\cdot dt}\cdot 100 \qquad (7)$$

where the subscripts II, III and IV are used to indicate the concentrations of the various parameters in the sampling sections identified in the same way.

The values of h_{N-NO_3}, h_{COD_T} and h_{H_2S} are equal to 65.0%, 5.5% and 18.0%, respectively. The low values of the first two efficiencies are attributable to the low strength of the incoming wastewater.

The reaction rates were calculated by comparing the mass removed of each substrate both for the duration of the test (48 min) and the section volume (8.4 m³). This gave rates of 1.17 g N·m⁻³·h⁻¹, 4.06 g COD_T·m⁻³·h⁻¹, and 0.58 g H_2S·m⁻³·h⁻¹ for denitrification, COD_T and sulphides, respectively. Consequently, the consumption of COD_T in the denitrification process, both for conversion of nitrate into nitrogen gas and sludge production, was equal to 3.47 g of COD_T per gram of reduced nitrate nitrogen. This value is lower than the values commonly encountered in traditional denitrification processes, which range from about 4 g COD_T·g⁻¹ of reduced nitrate nitrogen (Henze et al., 1997) to about 8.6 g COD_T·g⁻¹ of reduced nitrate nitrogen (Ekama and Marais, 1984).

Conclusions

The ANANOX® wastewater treatment system proposed by the ENEA researchers in 1988, comprising a two-stage biological anaerobic-aerobic process, has been subjected to numerous experiments carried out primarily on laboratory-scale and pilot-scale systems. However, this paper deals with results obtained from the only full-scale plant currently operating in the municipality of San Giovanni in Persiceto (Bologna - Italy) for the treatment of sewage from the village of Biancolina. The plant is fed via a

sanitary sewer network and has sufficient capacity to meet the treatment requirements of 350 population equivalents, even though only 180 are currently linked up to the system. The plant still requires calibration in order to establish the operating conditions at which the highest efficiency can be obtained, primarily in terms of organic substance, suspended solids and nitrogen compounds so that useful insights can be gained for its successful application in other cases.

This paper reports the results of experimental investigations, which aimed essentially at defining the influence of biomass concentration on the ABR's efficiency and analysing the ways in which the denitrification process can be developed.

As far as the elimination of COD in the ABR is concerned, regulating the biomass concentration makes it possible to attain 30% better efficiency, which is thought to be highly satisfactory for municipal sewage with a low organic content and rich in suspended solids. In this regard, it should be pointed out that this performance was obtained by adopting high upflow velocities (2.5 m·h^{-1}) and fairly low organic load rates. This was $0.97 \text{ kg COD}_T \cdot \text{m}^{-3} \cdot \text{d}^{-1}$ for the first section.

As far as the denitrification process is concerned, the experimental investigations have shown the consumption of organic substrate as an electron donor to be small and that sulphur compounds play a role. In particular, the experiments have confirmed the results already obtained during tests carried out on pilot-plant systems, which showed that the sludge contained *Thiobacillus denitrificans* micro-organisms, which can achieve denitrification by oxidising the sulphides into sulphates.

Obviously the ANANOX® system must be subjected to numerous other experiments that will, in particular, help to optimise efficiency in the anaerobic phase and define the admissible values for the applied organic load and upflow velocity in the ABR. However, the first results obtained on a full-scale plant system suggest its application for the treatment of municipal wastewater as it has been seen to be capable of reaching effluent standards in line with those laid down in the most stringent regulations that are generally applied. Furthermore, the series of phases and processes in the ANANOX® system give the plant an undeniable robustness and versatility which, together with its extremely compact configuration, make its use suitable for the treatment of sewage from communities with variable capacity and when high quality standards are required for the final effluent.

Acknowledgments

References

BARBER WP and STUCKEY DC (1999) The use of the anaerobic baffled reactor (ABR) for wastewater treatment: A review. *Water Res.* **33** (7) 1559-1578.

CHOI E and RIM JM (1991) Competition and inhibition of sulfate reducers and methane producers in anaerobic treatment. *Water Sci. Tech.* **23** 1259-1264.

EASTMAN JA and FERGUSON JF (1981) Solubilization of particulate organic carbon during the acid stage of anaerobic digestion. *J. WPCF* **53** (3) 352-366.

EKAMA GA and MARAIS GvR (1984) Theory, Design and Operation of Nutrient Removal Activated Sludge Processes (Chapter 6). Water Research Commission Report No. TT 16/84.

GARUTI G, DOHANYOS M and TILCHE A (1992) Anaerobic-aerobic wastewater treatment system suitable for variable population in coastal areas: The ANANOX® process. *Water Sci. Technol.* **25** (12) 185-195.

GOMMERS JPF, BIJLEVELD W, ZUIJDERWIJK FJM and KUENEN JG (1988) Simultaneous sulfide and acetate oxidation in a denitrifyng fluidized bed reactor. *Water Res.* **22** (9) 1085-1092.

HEDUIT A, DUCHENE P and SINTES L (1990) Optimization of nitrogen removal in small activated sludge plants. *Water Sci. Tech.* **22** (3/4) 123-130.

HENZE M, HARREMOES P, LA COUR JANSEN J and ARVIN E (1997) *Wastewater Treatment, Biological and Chemical Processes.* Springer-Verlag, Berlin Heidelberg.

JETTEN MSM, HORN SJ and VAN LOOSDRECHT MCM (1997) Towards a more sustainable municipal wastewater treatment system. *Water Sci. Tech.* **35** (9) 171-180.

LETTINGA G, VAN VELSEN AFM, DE ZEEWE WJ and HOBMA SW (1979) The application of anaerobic digestion to industrial pollution treatment. In: Stafford et al. (eds.) *Anaerobic Digestion,* Applied Science Publisher, London, England. 167-186.

LETTINGA G, VAN VELSEN AFM, HOBMA SW, DE ZEEWE WJ and KLAPWIJK A (1980) Use of Upflow Sludge Blanket (USB) reactor concept for biological wastewater treatment. *Biotechnol. Bioeng.* **22** 699-734.

LETTINGA G, ROERSMA R, GRIN P, DE ZEEW WJ, HULSHOFF POL LW, HOBMA SW, VAN VERLEN AFM and ZEEMAN G (1981) *Anaerobic Treatment of Sewage and Low Strength Wastewaters, Anaerobic Digestion,* Elsevier Biomedical Press. 271-291.

LETTINGA G and HULSHOFF POL LW (1991) UASB-process design for various types of wastewaters. *Water Sci. Technol.* **24** (8) 87-107.

MALINA FJ and POHLAND FG (1992) Design of anaerobic process for the treatment of industrial and municipal wastes. *Water Quality Management Library,* Vol. **7**.

McCARTY PL and BACHMANN A (1992) United States Patent No. 5,091,315.

PARKIN JF and OWEN WF (1986) Fundamentals of anaerobic digestion of wastewater sludge. *J. Environ. Eng. Div. Am. Soc. Civil Eng.* **122** 867-920.

PFEIFFER W, TEMPER U, STEINER A, CAROZZI A and VON MUCKE J (1986) Anaerobic waste water treatment: A grown-up technology. *Proc. EWPCA Conf. On Anaerobic Treatment,* September, Amsterdam, the Netherlands. 219.

STANDARD METHODS (1989) *Standard Methods for the Examination of Water and Wastewater* (17th edn.) Edited by Clesceri LS, Greenberg AE, Trussel RR, APHA, AWWA, WPCF.

WEILAND P and ROZZI A (1991) The start-up operation and monitoring of high-rate anaerobic treatment systems: Discussers report. *Water Sci. Technol.* **24** (8) 257-277.

VAN HAANDEL AC and MARAIS GvR (1981) Nitrification and Denitrification in the Activated Sludge Process. Research Report No. W39, Dept. of Civil Eng., Univ. of Cape Town.

VISMARA R (1998) Depurazione biologica, teoria e processi. Edited by Biblioteca Scientifica Hoepli (in Italian).