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COMBINED ANAEROBIC–AEROBIC SBR FOR THE TREATMENT OF PIGGERY WASTEWATER

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Abstract—Biological treatment of a piggery wastewater for organic carbon and nitrogen removal in a combined anaerobic–aerobic system was investigated using two laboratory-scale sequencing batch reactors. The cycle length was 24 h. In the anaerobic reactor, fed with raw wastewater and recycling from the aerobic reactor, denitrification followed by anaerobic digestion of organic carbon was observed. In the aerobic reactor, more organic carbon removal and ammonia oxidation to mainly nitrite occurred. Denitrification was also observed in the aerobic reactor during the filling period, when mixed liquor dissolved oxygen concentration was very low. Three recycle-to-influent ratios from 1 to 3 were tested. Average performances of the overall process, in the different conditions tested, were a TOC removal of 81 to 91% and 85 to 91% of TKN. 10 to 28% of the initial TKN was discharged as N-NO_x because of the low recycle-to-influent ratios used. The higher the recycle-to-influent ratio, the lower the concentrations of nitrogen oxides in the final effluent. However, the effect of this ratio was attenuated by the phenomenon of denitrification in the aerobic reactor which increased the performances of the process. © 1999 Elsevier Science Ltd. All rights reserved

Key words—anaerobic digestion, denitrification, nitrification, piggery wastewater, sequencing batch reactor

INTRODUCTION

Anaerobic digestion is widely used to remove organic matter from high strength wastewaters because of its relatively low sludge production and energy needs. Organic nitrogenous compounds present in the wastewater, such as proteins, amino-acids or urea, are mainly reduced to ammonia which is not further degraded in anaerobic conditions. The discharge of effluents containing ammonia is undesirable because it causes excessive oxygen demand in the receiving waters. Ammonia toxicity on aquatic life is also possible at high concentrations and high pH values. Complete nitrogen removal is also necessary where the receiving water is a water supply source for downstream users, since eutrophication and nitrate enrichment should be avoided.

A post-treatment may be necessary in order to remove ammonia before discharge, biological nitrification–denitrification being the most widely used process (Odegaard, 1988). In the case of a wastewater with a low COD/TKN ratio, organic carbon content of the digested effluent may be insufficient to achieve complete denitrification and the addition of an external carbon source is then required.

Earlier studies (Akunna *et al*., 1994a) have shown that nitrogen and carbon can be effectively removed from synthetic wastewaters using coupled aerobic and anaerobic filters, where methane production and denitrification will be encouraged to take place in the anaerobic filter while nitrification and final effluent polishing will take place the aerobic filter.

Attempts to carry out denitrification and methane production in a completely mixed reactor have not proved very effective (Akunna *et al*., 1992, 1993, 1994b, 1998) because nitrogen oxides have been found to inhibit (reversibly) methanogenic bacteria (Chen and Lin, 1993; Akunna *et al*., 1998; Clarens *et al*., 1998). Furthermore, dissimilatory nitrate reduction to ammonia can occur in the system depending on the type of carbon compounds present in the system. Consequently, biofilm reactors have been used in the combined denitrification/anaerobic digestion system because the fixed nature of the media enables the creation of macro- and micro-environments within the system, so that the different bacteria involved in the reactions can then

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grow and concentrate in zones within the reactor favourable to their metabolic activities (Kuroda *et al.*, 1988; Hanaki and Polprasert, 1989; Akunna *et al.*, 1994a; Lin and Chen, 1995; Chen *et al.*, 1997).

The use of biofilm reactors is good for wastewaters containing relatively small amounts of suspended solids. When the wastewater contains significant quantities of solids as in most agricultural and food processing wastewaters, the use of suspended growth reactors becomes inevitable, and presently very little has been done on nitrate reduction in suspended growth anaerobic systems. Studies with up-flow anaerobic sludge blanket (UASB) reactors have been reported (Hendriksen and Ahring, 1996), but UASB reactors have a very long start-up time for the granules to develop, thus limiting the scope of their application. This study therefore attempts to investigate the performance of an alternative suspended growth system, the sequencing batch reactor (SBR), using piggery wastewater.

Anaerobic digestion has been widely applied to piggery waste treatment (Summers and Bousfield, 1980; Fischer *et al.*, 1984; Andreadakis, 1992; Dague and Pidaparti, 1992; Lo *et al.*, 1994). Anaerobic sludge treating piggery wastewater has been successfully used to carry out denitrification (Bernet *et al.*, 1996). Operating the anaerobic system in a batch mode can provide different environmental conditions, periodic anoxic and anaerobic conditions, for denitrification and methane production.

SBRs have been used in piggery wastewater treatment, either in aerobic/anoxic conditions (Jern, 1987; Fernandes *et al.*, 1991) or in anaerobic digestion (Massé *et al.*, 1996; Zhang *et al.*, 1997). This paper presents the results obtained using two SBR in a configuration favourable to nitrogen and carbon removal (Akunna *et al.*, 1994a) for the treatment of piggery wastewater under various operating conditions during a period of 2 years.

MATERIALS AND METHODS

Materials

Piggery wastewater was obtained from a pig farm located in the neighbourhood of the laboratory. The first part of the work was carried out with wastewater sieved through 1 mm diameter mesh and centrifuged at 11,000 rpm for 15 min to remove most of the suspended solids. Table 1 shows the average composition of the supernatant obtained after these preliminary treatment operations.

In the second part of the study, the raw wastewater was used without any preliminary treatment. During this period, the TSS concentration in the wastewater could be as high as 18 g l^{-1}. The aim of this study was to compare the performance with that of the first part so as to determine the effect of TSS on the performance of the system.

Experimental system

The configuration of the process is presented in Fig. 1.

Table 1. Average composition of liquid swine manure after sieving and centrifugation. TOC: total organic carbon; VFA: volatile fatty acids; C_2: acetate; C_3: propionate; C_4: butyrate; IC_4: isobutyrate; C_5: valerate; IC_5: isovalerate; TKN: total Kjeldahl nitrogen; TSS: Total suspended solids

Parameter	Average value	Standard deviation
TOC (g l^{-1})	5.86	1.33
Total VFA (% TOC)	57	
C_2 (g C l^{-1})	1.57	0.74
C_3 (g C l^{-1})	1.09	0.47
C_4+IC_4 (g C.l^{-1})	0.27	0.17
C_5+IC_5 (g C l^{-1})	0.41	0.22
TKN (g l^{-1})	3.69	0.74
NH_4-N (g l^{-1})	2.94	0.40
TSS (g l^{-1})	2.84–3.96	3.43
PH	7.68	0.14

Anaerobic reactor. The anaerobic reactor (AN) had an active liquid volume of 1.5 l. It was seeded with 0.75 l of anaerobic sludge obtained from a laboratory digester treating wine distillery wastewater. The volume was completed with 0.75 l of tap water. The temperature was kept constant at 35°C by a water jacket and mixing obtained using a magnetic stirrer maintained at a constant speed of 400 rpm.

Aerobic reactor. Two aerobic reactors were used successively in this study. The first one (N1) had an active volume of 1.5 l. It was used when wastewater flow rate was 0.1 l d^{-1}. When the organic carbon load of the system was doubled, a 4 l reactor (N2) capacity reactor (containing 3 l of mixed liquor) was used. These reactors were inoculated with sludge from another nitrifying reactor treating piggery wastewater at room temperature. Aeration was provided by compressors connected to plastic tubes placed at the bottom of the reactors. The reactors were fed with digested effluent from the anaerobic reactor. They were kept at ambient temperature (20–22°C) and mixing was carried out using a magnetic stirrer moving at 700 rpm.

Sequencing batch reactors operations

After the start-up period during which they were fed semi-continuously, the reactors were operated in a sequencing batch mode in order to increase biomass content and, consequently, total activity of the system. The cycle length was 1 day for both aerobic and anaerobic reactors with a minimum reaction time of 22 h. The filling time depended on the recycle-to-influent ratio (R) applied. The different conditions applied are detailed in Table 2. Note that $R=Q_2/Q_1$, where Q_1 and Q_2 are the raw wastewater and the recycled nitrified effluent flow rates, respectively.

Analytical methods

The redox potential (E0) in the anaerobic reactor was monitored using an Ingold pH transmitter (2400) and a combination redox electrode (Ag/AgCl reference system, KCl 3 M, $Eh_{ref}^{35°C}=199.8$ mV). Dissolved oxygen in the aerobic reactor was measured using an Ingold transmitter (4300) and a polarographic electrode. pH was monitored in both reactors with an Ingold pH-meter (2301).

Samples were centrifuged at 6000 g for 10 min before analysis to remove suspended solids. The supernatants were diluted as required prior to analysis.

Total Kjeldahl nitrogen and ammonium were determined using the titrimetric method after distillation with a Büchi apparatus (Rodier, 1975). Nitrate and nitrite were analyzed by an ion chromatography system using conductivity detection (Dionex-100). Separation and elution of the anions were carried out on IonPac AS12A analytical column utilizing a carbonate/bicarbonate eluant and

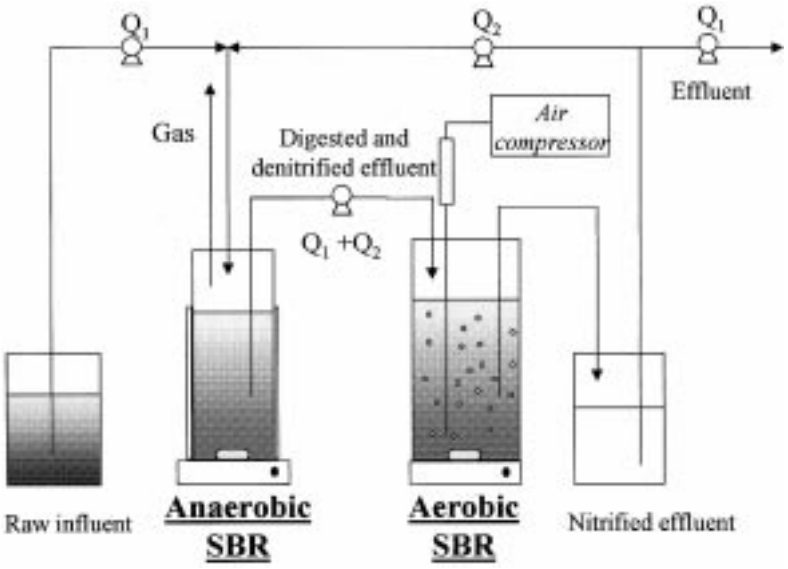

Fig. 1. Configuration of the process used in this study.

AutoSupression technology. Integration was done using a PC fitted with Peaknet Software.

Total organic carbon (TOC) was determined by UV oxidation with a Dohrman DC 80 apparatus. It involved oxidizing organic carbon compounds with potassium persulfate at low temperature. The carbon dioxide formed was detected by infrared adsorption. The waste having initially been acidified and stripped of inorganic carbon by bubbling with oxygen for 5 min. Chemical oxygen demand (COD) was measured by potassium dichromate/ferrous ammonium sulfate method. Volatile fatty acids (VFA) analysis was done using a gas chromatograph fitted with a flame ionization detector (Chrompac CP 9000) and coupled with an integrator (Shimadzu CR 3A).

Gas analysis by gas chromatography was carried out with a Shimadzu GC-8A apparatus with argon carrier using a katharometer detector. The chromatograph was coupled to a Shimadzu CR 3A integrator.

Total suspended solids (TSS) and volatile suspended solids (VSS) were determined using Standard Methods (APHA, 1992).

RESULTS AND DISCUSSION

Anaerobic reactor

Start-up. The reactor was firstly adapted to anaerobic digestion of piggery wastewater, as a fed-batch reactor, during 82 days, by a mixture of wine distillery effluent (from 100 to 50%) and piggery wastewater (from 0 to 50%). The distillery wastewater was eventually totally replaced by nitrified piggery wastewater to initiate denitrification. Figure 2 shows the performance of the anaerobic reactor during this period.

Operation. The reactor was then coupled to the aerobic reactor. Complete denitrification was observed in the anaerobic reactor for all the loading rate and the recycle-to-influent ratio studied. Figure 3 shows the results obtained during a cycle when R was 2 and influent flow rate was $0.2\,l\,d^{-1}$. Filling time for raw wastewater and nitrified effluent (from aerobic reactor) were 65 and 20 min, respectively.

The amount of nitrogen oxides (mainly nitrites, which accounted for about 97% of the total nitrogen oxides) fed into the anaerobic reactor when R was 2 corresponded to a concentration of about 250 mg NO_x-N l^{-1} in the reactor. All the added nitrogen oxides were eliminated especially during the 20-min filling period. A corresponding reduction of VFA was also noticed during this period.

A significant increase in the redox potential (from -350 to -112 mV) was observed during filling with nitrified effluent. It has been shown by earlier studies that this increase was not due to the presence of oxidized nitrogenous compounds but rather to denitrifying activity (Bernet *et al.*, 1997). It took almost 3 h for the redox potential to go down to a value of about -300 mV, which was a much more conducive environment for methanogenesis. No significant changes in TOC concentration was observed before and after filling (even though TOC consumption should have taken place during the reduction of nitrogen oxides) probably due to the hydrolysis of the VSS in the reactor which might have replenished consumed soluble TOC.

Gas was produced in the anaerobic digestor throughout the entire period. It was initially planned to shorten the treatment cycle, but when continuous gas production was observed during the initial cycle, it was decided to leave it as it was. The gas produced during this cycle was composed of N_2 (70%), CH_4 (27%) and CO_2 (3%). The abundance of nitrogen in the digestor gas suggested that denitrification was the main biological process occurring in the system. The presence of methane in the biogas indicated that anaerobic digestion occurred with

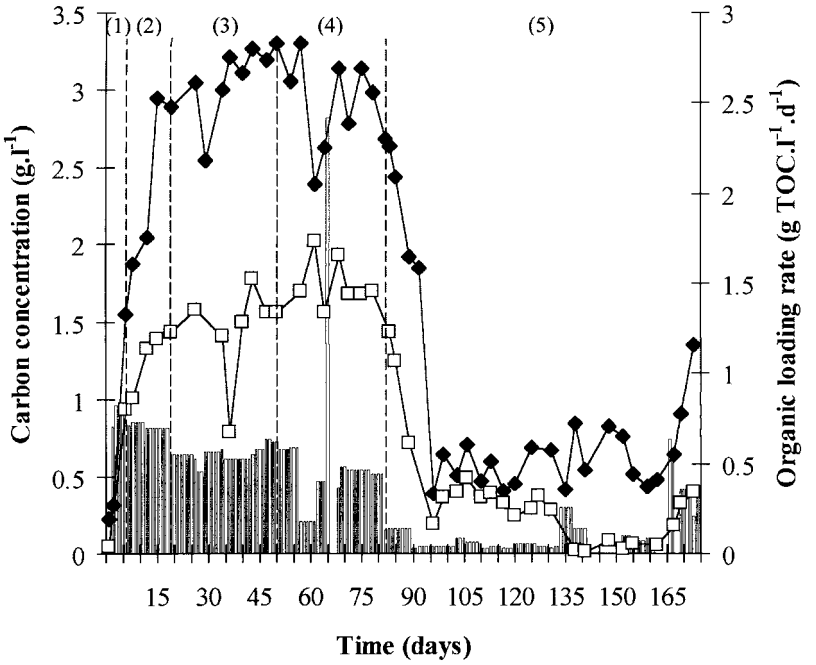

Fig. 2. TOC and VFA concentrations in the anaerobic reactor during the start-up period. □: carbon load, -◆-: TOC, -□-: VFA-C.

denitrification so that both processes contributed to TOC removal.

As shown in Fig. 4, there was a relationship between gas composition in the anaerobic SBR and the nitrogen oxides loading rate. The performance of the reactor was sensitive to the fed composition. In the presence of high nitrite and/or nitrate concentrations, organic matter was firstly used as electron donor for denitrification. During this period a high nitrogen concentration was detected in the biogas. When the nitrogen oxides were absent or in a low concentrations in the influent, organic matter was then mainly converted to biogas and an increase in methane concentration was observed in the gas produced, together with a decrease in the proportion of nitrogen gas produced.

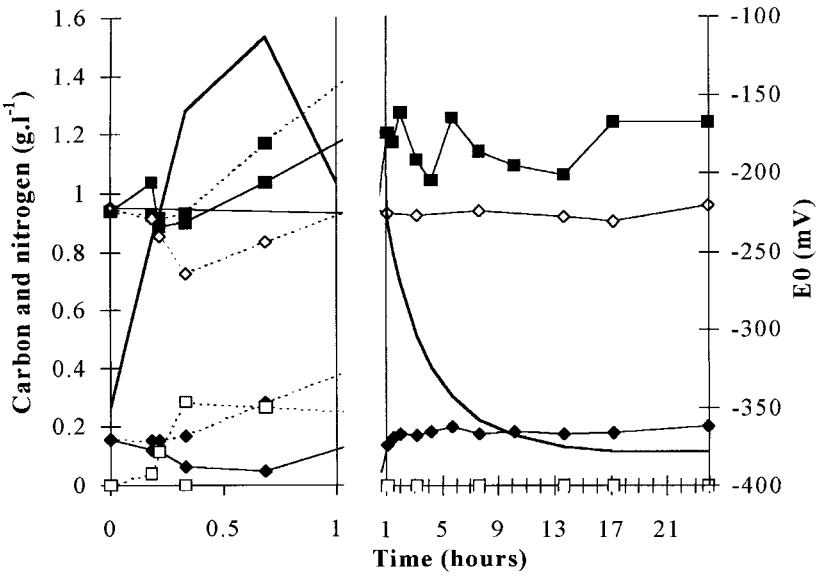

Fig. 3. Variations of main parameters during a cycle in the anaerobic SBR at $R = 2$. -■-: TOC, -◆-: VFA-C, -□-: N-NO_x, -◇-: NH_4-N, —: E0. Dash lines: concentrations introduced in the reactor; plain lines: measured concentrations in the reactor.

Table 2. Hydraulic conditions applied on the process at the different wastewater flow-rates (Q_1) and recycle-to-influent ratios (R) used

Period No.	Period (days)	Q_1(l d^{-1})	R (l l^{-1})	Anaerobic reactor volume (l)	Aerobic reactor volume (l)	Process HRT (days)
I	1–64	0.1	1	1.5	1.5	30
II	65–155	0.1	2	1.5	1.5	30
III	156–361	0.2	2	1.5	3	22.5
IV	361–429	0.2	3	1.5	3	22.5
V	429–704	0.2	3	1.5	3	22.5

It can also be noticed that denitrification was the only reduction pathway of the nitrogen oxides in the anaerobic reactor. In all experimental conditions studied dissimilatory reduction to ammonia was not observed. This could be due to the low C/N ratio of the wastewater and the high VFA fraction of the organic matter. These two parameters have been shown to favour denitrification over dissimilatory reduction to ammonia (Akunna *et al*., 1992, 1993).

Carbon and nitrogen balance. Table 3 presents the organic carbon and nitrogen balance in the reactor over a 70-day period (from 190th to 260th day), inside period III, when steady state performance was observed.

According to our results and to the literature, it is obvious that denitrification process mainly used volatile fatty acids as carbon source. From Table 3, it can be seen that 425 mg VFA-C were consumed in the anaerobic reactor. 425 mg C-VFA has a COD value of 1344 mg. Considering that stoichiometric COD/N ratios for denitrification from nitrite and nitrate are 2.86 and 1.71, respectively, 588 mg COD would be needed for denitrification. If 10% of COD consumption was used for cell synthesis, only about 50% of the VFA supplied (or 35% of TOC consumed) can then be accounted for. The rest was suspected to have been converted to methane and carbon dioxide.

The total carbon balance shows that only 194 mg C per day was removed in reactor. This unexpected low value might be probably due to significant mineralization to bicarbonate. Methanotrophic denitrification, which is usually enhanced by the presence of oxygen in the recycled effluent (Thalasso *et al*., 1997), might have contributed to the low total carbon removal obtained.

Aerobic reactor

Start-up. The aerobic reactors were firstly fed with anaerobic effluent without recycling until full nitrification was established.

Operation. The variation of various parameters during a cycle in the aerobic reactor is presented in Fig. 5. The performance of the anaerobic reactor during the same period is shown in Fig. 3. It can be seen that DO concentration decreased quickly to a low value of about 0.6–0.8 mg O_2.l^{-1} during the filling period certainly because of the high oxygen demand caused by the introduction of anaerobic effluent. The DO concentration began to rise only after a period of up to 11 h. The increase in DO seemed to signal the end of nitrification.

Nitrite was generally the main product of nitrification. This could be due to insufficient dissolved oxygen availability and to the lower affinity for oxygen of nitrite oxidizers, compared to ammonia oxi-

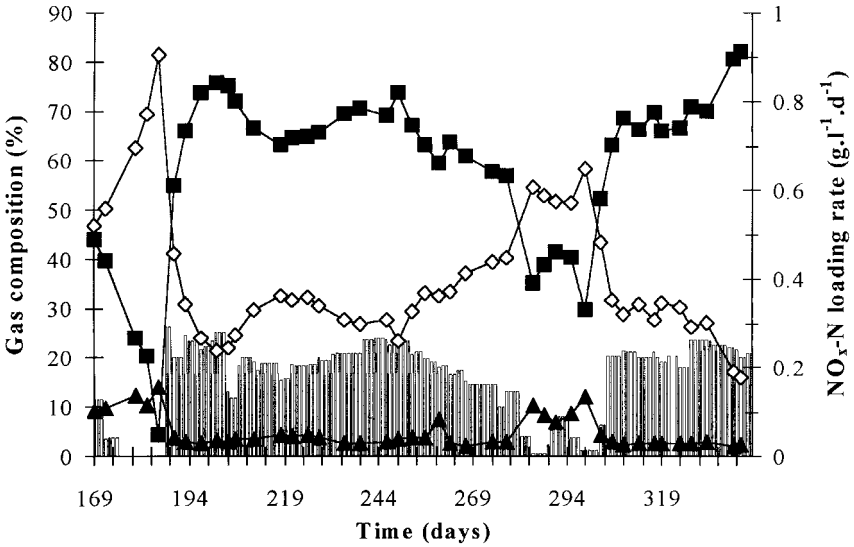

Fig. 4. Gas composition and N-oxides loading rate in the anaerobic reactor at R = 2. □: NO_x-N loading rate, -■-: N_2, -◇-: CH_4, -▲-: CO_2.

Table 3. Characterization of the process in the period III

		Q_1 ($l\ d^{-1}$)	TOC ($g\ d^{-1}$)	TC ($g\ d^{-1}$)	VFA ($g\ C\ d^{-1}$)	TKN ($g\ d^{-1}$)	NH_3-N ($g\ d^{-1}$)	NO_2-N ($g\ d^{-1}$)	NO_3-N ($g\ d^{-1}$)	Total N ($g\ d^{-1}$)
Anaerobic reactor	in (w)	0.196	1.181	1.627	0.611	0.731	0.663	0	0	0.731
	in (r)	0.420	0.284	0.423	0.005	0.193	0.075	0.324	0.012	0.529
	out	0.616	0.872	1.856	0.191	0.899	0.776	0	0	0.899
	% removal		40.5	9.5	69.0	2.7	−5.1	100	100	28.7
Aerobic reactor	in	0.616	0.872	1.856	0.191	0.899	0.776	0	0	0.899
	out	0.616	0.417	0.621	0.008	0.283	0.110	0.476	0.018	0.668
	% removal		53.6	61.1	100	73.5	81.5	–	–	25.7
Overall process	in	0.196	1.181	1.627	0.611	0.731	0.663	0	0	0.731
	out	0.196	0.133	0.198	0.002	0.090	0.035	0.151	0.006	0.247
	% removal		88.7	87.8	99.7	87.7	94.7	–	–	66.2

dizers (Hanaki *et al.*, 1990; Laanbroek and Gerards, 1993) or to free ammonia inhibition (Anthonisen *et al.*, 1976).

The nitrogen balance in the aerobic reactor shows that losses of nitrogen were observed at several periods, corresponding to periods when there were breakdowns of the aeration system in the reactor. Figure 6 shows the relationship between the pH value and the nitrogen balance. The pH values of the mixed liquor were generally above 8 when significant losses (unaccounted for) of nitrogen were observed. The high pH values could be probably due to the result denitrification occurring in the reactor. High pH values might have brought about ammonia stripping. It is more than probable that at such a low DO concentration, denitrification occurred in the reactor especially at the very beginning of the cycle when biodegradable carbon from the anaerobic SBR was available. pH increase in this reactor could therefore be regarded as a good indicator of a process failure, attributable mainly to insufficient supply of oxygen.

Overall process

Table 4 presents the average performance of the process under the different experimental conditions. Good results were obtained for TOC and TKN removal. These may be compared with results obtained using anoxic/aerobic SBR (Jern, 1987; Fernandes *et al.*, 1991).

The difference between TKN and total N removal is due to nitrogen oxides lost in the final effluent. The amount of nitrogen oxides in the final effluent depended on the recycling ratio, R. High

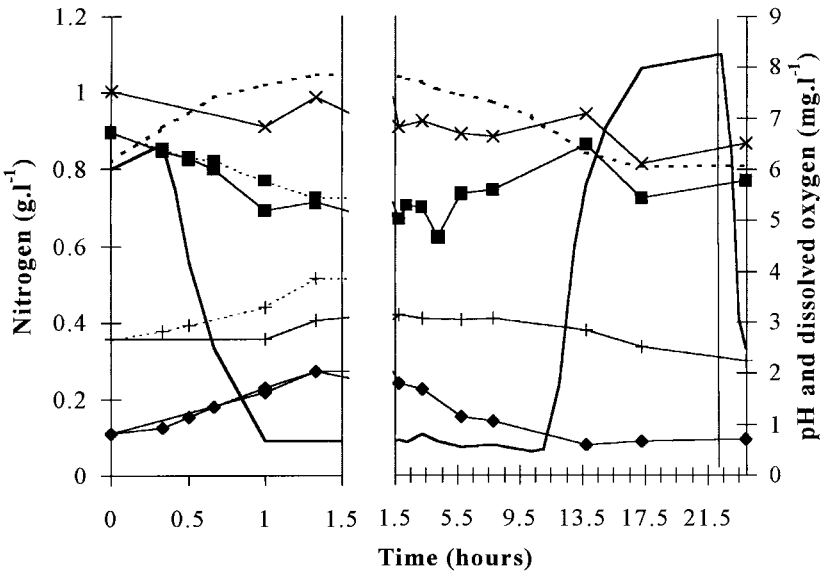

Fig. 5. Variations of main parameters during a cycle in the aerobic SBR at $R = 2$. -+-: TKN, -◆-: NH_4-N, -■-: N-NO_x, - × -: Total N, —: DO, –-: pH. Dash lines: concentrations introduced in the reactor; plain lines: measured concentrations in the reactor.

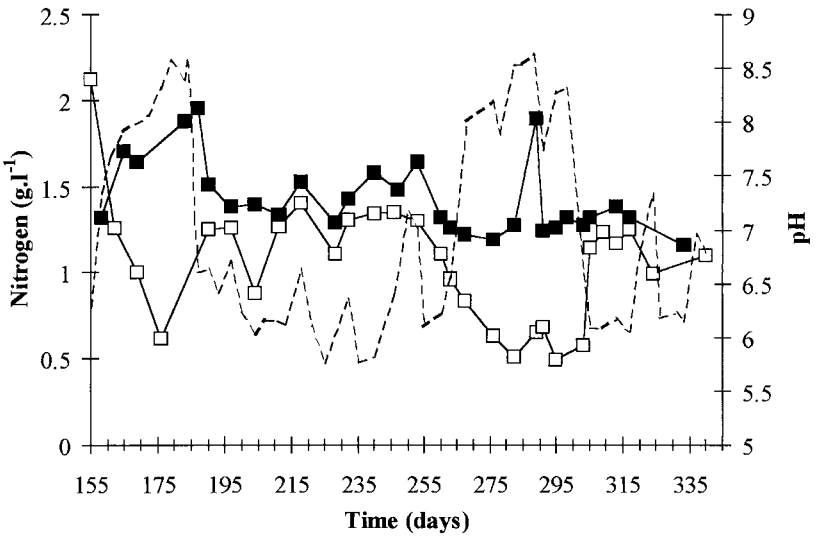

Fig. 6. Nitrogen balance in the aerobic SBR at $R = 2$. -■-: TKN_{in}, -□-: Total nitrogen, –-: pH.

values of R resulted in high total nitrogen removal and subsequently low concentrations of nitrogen oxides in the final effluent.

Assuming that the TKN removal rate in the aerobic reactor is 100% and that:

- TKN removal is only by nitrification in the aerobic reactor and
- nitrogen oxides recycled are completely denitrified in the anaerobic reactor, then,

$$N_{out} = TKN_{out} + [N - NO_x]_{out} = \frac{1}{1+R} \times TKN_w$$

where TKN_w is the TKN of raw wastewater and N_{out} is the total nitrogen in the final (or aerobic reactor) effluent.

Using the above equation, N_{out} concentrations for R values of 1, 2 and 3 correspond to theoretical total nitrogen removal of, respectively, 50, 66.7 and 75%. These values are lower than the experimental values, as shown in Table 4. Table 4 also indicated no direct correlation between TKN removal and N_{out}, suggesting that other process(es) might have contributed to nitrogen removal in the aerobic system. As already indicated in the preceding section, denitrification and/or ammonia stripping could be responsible for these extra nitrogen losses in the aerobic reactor. Assimilation of nitrogen for cell synthesis is also a possible source of nitrogen losses but it was considered negligible since only about 10% of the TOC fraction was removed in the aerobic reactor, and nitrifiers are known to have a low cell yield.

Ammonia is known to inhibit anaerobic digestion of swine wastes (Braun *et al.*, 1981; Hansen *et al.*, 1998). This inhibition has been attributed to free ammonia which increases at high pH values (de Baere *et al.*, 1984). In our system, it was believed that ammonia inhibition was reduced by the dilution of influent raw wastewater with ammonia-free effluent from the aerobic reactor.

Referring to Table 4, it will be seen that the performance of the system using centrifuged and sieved wastewaters are comparable. This evaluation was based on process parameters measured on soluble raw and treated wastewater samples. It was difficult to evaluate the performance of the system on solids

Table 4. Average performances of the process in the different conditions tested

Waste water	Q_1 ($l\,d^{-1}$)	R ($l\,l^{-1}$)	TOC removal (%)		TKN removal (%)		Total N removal (%)	
			average	standard deviation	average	standard deviation	average	standard deviation
Centrif	0.1	1	84.4	5.3	86.8	6.0	58.5	3.8
Centrif	0.1	2	90.9	1.1	90.5	2.5	75.5	7.3
Centrif	0.2	2	88.6	2.9	89.3	4.4	71.6	8.5
Centrif	0.2	3	82.0	3.6	90.3	1.8	69.2	4.8
Sieved	0.2	3	81.0	7.2	85.2	7.4	75.5	8.4

removal at such a small scale. However, it appears likely that this system results in lower sludge production when compared to an entirely aerobic process due to possible volatile solids breakdown during the anaerobic digestion phase. Further studies are necessary in this direction.

CONCLUSIONS

A system coupling two sequencing batch reactors for the biological treatment of piggery wastewaters was carried out. It can be concluded from this work that:

- It was possible to carry out denitrification in an anaerobic SBR reactor. Methane production followed denitrification.
- The efficiency of nitrogen removal was dependent on the applied recycle-to-influent ratio and was enhanced by partial denitrification and suspected ammonia stripping in the aerobic reactor.
- Overall performances of TOC and TKN removal of, respectively, 81 to 91% and 85 to 91% were achieved.

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