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AEROBIC DOMESTIC WASTE WATER TREATMENT IN A PILOT PLANT WITH COMPLETE SLUDGE RETENTION BY CROSS-FLOW FILTRATION

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Abstract—An aerobic wastewater treatment pilot plant with cross-flow filtration was operated for more than 300 days to examine whether reduced sludge production and stable treatment performance can be achieved when sludge is completely retained. The volumetric loads ranged between 0.9 and 2.0 g $COD \cdot l^{-1} \cdot day^{-1}$. Technical observations were: the oxygen transfer rate became poor at high sludge concentrations; membrane capacities declined but could be mostly sufficiently restored by cleaning. Sludge was hardly produced when the mixed liquor suspended solid (MLSS) concentration had increased to 40–50 $g \cdot l^{-1}$. Then, the sludge load was only 0.021 g $COD \cdot g\ MLSS^{-1} \cdot day^{-1}$ and only 6% of the carbon supplied was assimilated. Non-volatile compounds hardly accumulated as the fraction of inorganic compounds in sludge increased from 21.6 to 23.5% during the last 200 days, whereas the carbon, phosphor and kjeldahl nitrogen contents were stable. After 300 days the content of polluting trace elements, such as mercury, lead and cadmium, were similar to that of a conventional treatment plant supplied with this wastewater. Carbon and kjeldahl nitrogen removal was always quite satisfactory. Carbon was always removed for more than 90% and kjeldahl nitrogen that was not assimilated was completely nitrified at all times. The nitrification capacity at 30°C was constantly around 0.2 mmol $\cdot g\ MLSS^{-1} \cdot h^{-1}$, which shows that the viability of the nitrifying population did not cease. In addition, up to 40% of nitrogen supplied was lost as a result of denitrification. Hence stable treatment performance and a very low sludge production can be achieved when complete sludge retention is applied at high hydraulic loads.

Key words—activated sludge, complete sludge retention, cross-flow reactor, denitrification, domestic waste water treatment, membrane filtration, nitrification, sludge production

INTRODUCTION

Conventional aerobic treatment of domestic wastewater has several disadvantages. For instance, the production of sludge is high, nutrients are insufficiently removed and large surface areas are required as volumetric capacities are low. These drawbacks can be largely circumvented if sludge is completely, or almost completely, retained. However, the amount of sludge that can be maintained in a conventional treatment plant is limited, since the settling qualities are poor at high sludge concentrations. Therefore, enhanced sludge retention can only be achieved, if separation techniques different from secondary clarification are applied. An example of this is cross-flow filtration.

In a cross-flow reactor sludge is retained by microfiltration or ultrafiltration (see Fig. 1 for an example). This way of waste water treatment has several characteristics that differ from conventional treatment. For instance, in a cross-flow reactor waste water is sufficiently purified to be re-used for non-potable water purposes (Vial *et al.*, 1992; Langlais *et al.*, 1992; Chiemchaisri *et al.*, 1993). Moreover, volumetric capacities are typically high because the activated sludge concentration can be controlled independently of the settling qualities; hydraulic retention times as low as 2 h have satisfactorily been applied (Chaize and Huyard, 1991). So, the costs for treatment seem to be reduced as compared to conventional treatment, but the energy expenses are high due to the pressure required for filtration (Krauth and Staab, 1993). The most noticeable characteristics, however, are the low amount of sludge being produced and the excellent removal of organic substrates and kjeldahl nitrogen at high loading rates (see, e.g. Yamamoto *et al.*, 1989; Chaize and Huyard, 1991; Chiemchaisri *et al.*, 1983; Bailey *et al.*, 1994).

The low sludge production and the high removal efficiencies for organic substrates in cross-flow reactors are the consequences of the low sludge load. Bacteria primarily utilize the energy supplied with

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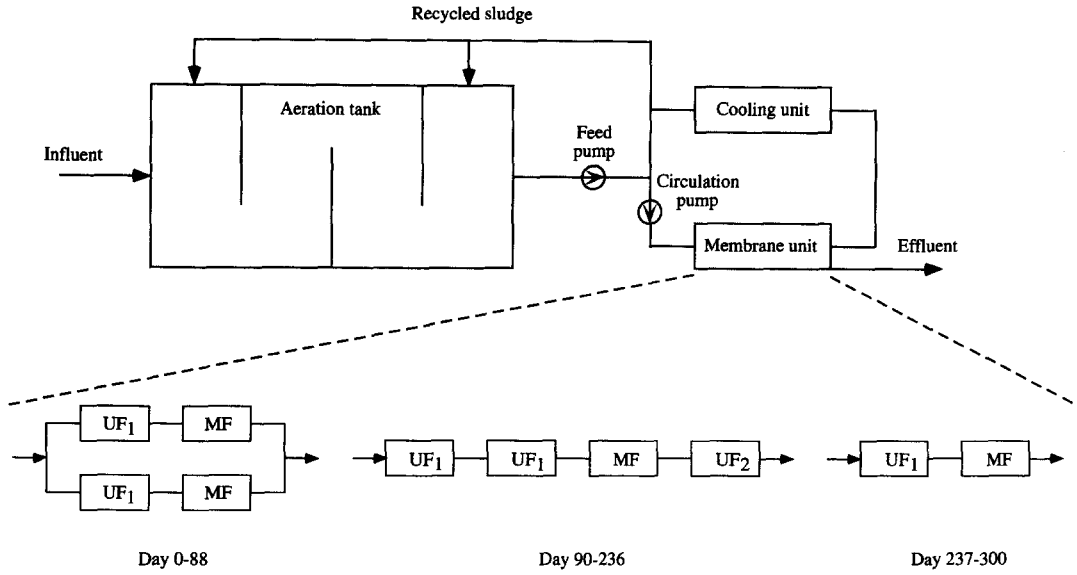

Fig. 1. Configuration of the reactor with cross-flow filtration. The characteristics of the membrane modules are listed in Table 1.

influent for maintenance purposes. These expenditures have to be made to remain viable, i.e. proteins and RNA must continuously be replaced, the intracellular ion concentrations has to be maintained, etc. (Stouthamer *et al.*, 1990). So, only if energy is supplied in excess, bacteria are able to grow. Therefore, sludge retention should ultimately lead to the maximal sludge concentration possible at a given load. In addition, all degradable carbon sources should be mineralized. Then, low sludge loads are combined with high loading rates. These expectations have been demonstrated for the treatment of synthetic waste water (Yamamoto *et al.*, 1989; Chiemchaisri *et al.*, 1992; Bailey *et al.*, 1994). For domestic waste water treatment, it has been shown that sludge production is greatly reduced if the sludge age is between 50 and 100 days (Chaize and Huyard, 1991). In addition, the treatment performance has been shown to be satisfactory for at least two months when sludge is completely retained (Chiemchaisri *et al.*, 1993). However, it is as yet unknown whether the treatment performance is negatively affected by the accumulation of inert material such as inorganic compounds. Also, it is still unclear whether sludge remains sufficiently viable to ensure proper treatment at loads that are fluctuating for longer periods than two months.

Kjeldahl nitrogen is also properly removed in cross-flow reactors at high loading rates (see e.g. Yamamoto *et al.*, 1989; Chaize and Huyard, 1991; Suwa *et al.*, 1992; Chiemchaisri *et al.*, 1993). This is principally a result of the nitrifying population being maintained. Autotrophic nitrifiers profit for two reasons. Since these bacteria have long generation times (see Prosser, 1989, for an overview), they are washed out in conventional treatment plants when the sludge age is kept too low. Moreover, since the sludge production is low, kjeldahl nitrogen is barely assimilated by heterotrophic bacteria, which are better competitors for kjeldahl nitrogen than nitrifiers (Hanaki *et al.*, 1990; van Niel *et al.*, 1993). Therefore, most kjeldahl nitrogen supplied is available for nitrification. Accordingly, even at hydraulic retention times as low as 2 h, kjeldahl nitrogen supplied to cross-flow reactors has been shown to be completely nitrified (Chaize and Huyard, 1991; Suwa *et al.*, 1992). Besides nitrification, nitrogen losses due to denitrification have usually been found (Yamamoto *et al.*, 1989; Suwa *et al.*, 1992; Chiemchaisri *et al.*, 1992). This denitrification activity is enhanced when the sludge concentration is increased (Suwa *et al.*, 1992). Consequently, especially in cross-flow reactors, nitrification and denitrification can be combined partly.

This study aims to investigate the sludge production and the treatment performance at complete sludge retention. For this purpose, a pilot cross-flow reactor was supplied with pre-settled domestic waste water at high loading rates for almost one year. In addition, a conventional pilot plant was operated to serve as a reference. Carbon and nitrogen flows were followed to determine the fractions of the supply spent on sludge production and mineralization. The stability of kjeldahl nitrogen removal was studied by the determination of nitrification capacities. The content of inorganic compounds was determined regularly, since severe accumulation was expected to cause membrane failure and to reduce treatment performance.

MATERIALS AND METHODS

System configuration

The configuration of the pilot plant with cross-flow filtration (membrane reactor) is shown in Fig. 1. The reactor

Table 1. Characteristics of the membrane filtration units

Configuration[1]	Time (days)	Number of pipes (per module)	Length of pipes (m)	Diameter of pipes (mm)	Surface area per module (m^2)	Velocity along membranes ($m \cdot s^{-1}$)	Recirculation flow ($m^3 \cdot h^{-1}$)	Transmembrane pressure (MPa)
2 × (UF_1–MF)	1–88	73	2.0	5.2	2.30	1–2	3–9	0.35; 0.15
UF_1–UF_1–MF–UF_2	90–236	7	2.0	14.4	0.63	±4	18–22	0.5; 0.4; 0.3; 0.2
UF_1–MF	237–290	7	3.0	14.4	0.95	±5	18–22	0.3; 0.2

[1]MF is microfiltration module, poresize is 0.1 μm; UF_1 is polysulfone ultrafiltration module, cut-off 50,000 for dextrans; UF_2 is acrylic ultrafiltration unit, cut-off 800,000 for dextrans.

consisted of four compartments and had a working volume of 613 l. The reactor was covered to enable gas analysis. Sludge was retained by cross-flow filtration. For this purpose the following tubular filter modules were used (Stork Friesland b.v., Gorredijk, The Netherlands): hydrophilic polysulfone ultrafiltration modules with a cut-off of 10,000 for polyethylene glycols and 50,000 for dextrans (UF_1); acrylic ultrafiltration modules with a cut-off of 360,000 for polyethylene glycols and 800,000 for dextrans (UF_2); hydrophilic polyvinilydene fluoride microfiltration units with a pore-size of 0.1 mm (MF). The properties of these modules are summarized in Table 1. The transmembrane pressure was obtained by recirculation of sludge; in this way, it was additionally aimed at that the reactor contents in the modules did not become overconcentrated. When the flux capacity decreased below the required flow, the membranes were cleaned with 10% Ultrasil (Stork Friesland b.v.) and/or 0.5–1.0% hypochlorite. Subsequently, the membranes were thoroughly rinsed with tap-water.

The configuration of the pilot plant operated in a conventional way (conventional reactor) was chiefly similar to that of the membrane reactor. It differed from the membrane reactor in the following aspects: sludge was separated from effluent in a secondary clarifier; sludge was only partially recycled; a selector was placed upstream of the aeration tank.

Operation conditions

The reactors were started with sludge from an oxidation ditch, in which domestic waste water was treated. Influent originated from a quarter of the city of Delft; it was completely domestic in nature. Before influent was supplied, it was screened (0.1 mm) and presettled (1 h). KOH was added to the aeration tank when the pH dropped below 6.3. The flow of compressed air was accurately controlled to enable gas analysis by mass flow controllers (type 5853E, Brooks instruments b.v., Veenendaal, The Netherlands), and was dispersed by a diffused air system. Typical flows in the membrane reactor ranged from 15 to 35 $m^3 \cdot h^{-1}$, depending on influent quality. The flow in the conventional reactor was initially 2 $m^3 \cdot h^{-1}$, but had ultimately to be increased to 6 $m^3 \cdot h^{-1}$ to keep sludge in suspension. Sludge was regularly discharged from the conventional reactor to keep the mixed liquor suspended solids (MLSS) concentration at 2.5–3.5 g $MLSS \cdot l^{-1}$. The reactor content of the membrane reactor was completely retained. The other operation conditions are listed in Table 2.

Analysis

Total organic carbon was determined in samples that were preserved with 0.2% sulphuric acid and stored at −20°C (Total Organic Carbon Anslyser 915-B, Beckman Instruments, Inc., Fullerton, CA); samples of sludge were homogenized by sonification before dilution and again before injection. Ammonium, nitrite and nitrate concentrations were established photometrically in samples preserved with chloroform (Autoanalyzer II, Technicon Industrial Systems, Tarrytown, NY). Kjeldahl nitrogen, phosphorus, the elements listed in Table 5, ash residue, dryweight and COD were determined according to the standard methods of American Public Health Association (1980). The reactor contents were regularly examined by microscope. Nitrification capacities of sludge samples at 30°C were determined with chemolithotrophic medium in recycling reactors as described elsewhere (Muller *et al.*, 1995). Carbon dioxide and oxygen contents in inlet and outlet gasses were determined instantaneously (URAS 10E and Magnos, respectively, Hartmann and Braun, Frankfurt, Germany). The carbon dioxide production was also determined gravimetrically. For this purpose, 40 l. h^{-1} of the outlet gas was diverted via mass flow controllers (type 5878, Brooks instruments b.v.) and passed through 0.5 M carbonate-free NaOH. Dissolved carbon dioxide was precipitated with 0.2 M $BaCl_2$. After being rinsed with distilled water, $BaCO_3$ accordingly formed was dried for 16 h at 100°C and determined gravimetrically. Oxygen transfer rates in sludge suspensions were determined from the reoxygenation curves obtained after the addition of sulphite.

RESULTS AND DISCUSSION

In a domestic waste water treatment pilot plant sludge was completely retained by cross-flow filtration for more than ten months (membrane reactor). A pilot plant that was operated in a conventional way was taken as a reference reactor (conventional reactor). The results are presented and discussed in three subsections. First, the operation conditions are given. This includes the description of the membrane units and the characterization of the influent quality. Subsequently, the development of the reactor contents are treated. Finally, the treatment performances are dealt with.

Operation conditions

The operation conditions of the membrane and conventional reactor are listed in Table 2. In both reactors, the pH was always between 6.6 and 7.1. As a consequence of heat production in the membrane system, temperatures were relatively constant around 20°C in the membrane reactor. Those in the conventional reactor decreased in time due to changes in the weather. The dissolved oxygen tension in the conventional reactor always exceeded 3 $mg \cdot l^{-1}$. However, aerobiosis in the membrane reactor was more difficult to maintain since the oxygen demand was higher. Moreover, the oxygen transfer coefficient decreased because the mixed liquor suspended solids (MLSS) concentration increased; this coefficient as a fraction of that of tap water (α factor) was 0.98 at 3 g $MLSS \cdot l^{-1}$ (as in the conventional reactor), 0.5 at 16 g $MLSS \cdot l^{-1}$, 0.3 at 26 g $MLSS \cdot l^{-1}$ and 0.2 at 39 g $MLSS \cdot l^{-1}$. Sludge ages in the membrane reactor could not be determined since sludge was completely retained. However, if the sampling volumes are taken

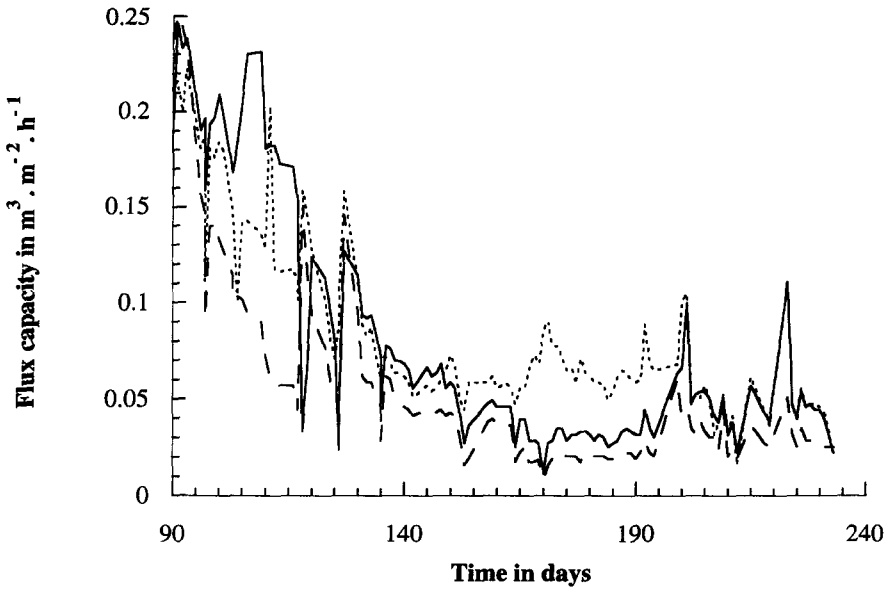

Fig. 2. Flux capacities of the membrane unit from day 90 to 236 as determined with tap water. Represented are the second ultrafiltration unit (– – –), microfiltration unit (——) and last ultrafiltration unit (- - -); the capacity of the first ultrafiltration unit was similar to the second. After cleaning the fluxes increased instantaneously.

into account, the sludge age was always higher than 3500 days. The hydraulic retention time (HRT) in the conventional reactor was 50 h, but that in the membrane reactor depended on the performance of the membrane unit (see below). During the first 220 days the average HRT ranged between 6 and 8 h, but increased to 10–15 h thereafter (see Table 2).

The membrane unit was configured to have overcapacity. Since the volumetric loads demanded could not be maintained, the configuration was changed twice. Until day 88 the membrane unit consisted of a pair of one ultrafiltration and one microfiltration module connected in parallel (see Fig. 1 and Table 1). When the sludge concentration had increased to 16 g $MLSS \cdot l^{-1}$, the velocity along the membranes appeared to be too low and the diameter of the pipes proved to be too small for sludge circulation. For this reason, the units, which had been silted up, were exchanged for 3 ultrafiltration units and 1 microfiltration unit connected in series that had larger pipe diameters (see Table 1). Again the capacity of the membranes declined (see Fig. 2), but the net surface specific flux demanded ($0.17\ m^3 \cdot m^{-2} \cdot h^{-1}$) could be maintained until day 120. Thereafter, fluxes could be partially restored by cleaning, as shown by the sudden increases of the capacities in Fig. 2. However, the effects of regeneration declined in time, probably as a result of irreversible fouling of the pores. At day 220 fluxes could no longer be maintained. Accordingly, on day 237 the membranes were again exchanged. Since the flux capacity of the last configuration was lower, the target HRT was increased to 10 h. Additionally, the velocity along the membranes was raised (see Table 1). This HRT could be maintained until day 270. Subsequently, it had to be increased to 15 h as a result of a decline in flux capacities. For all membrane configurations, the theoretical energy consumption for filtration was about 600–750 $W \cdot m^{-3}$ treated water when the modules functioned properly.

The influent quality varied according to changes in rainfall. Figure 3a, b illustrates that the highest concentrations of organic carbon (TOC) and ammonium, which represent dry weather conditions, were in the ranges 13–17.5 mM and 4–6.5 mM, respectively. Kjeldahl nitrogen consisted of 81% ammonium and the organic carbon to nitrogen ratio was stable at 2.2 $mol \cdot mol^{-1}$. The chemical oxygen demand (COD) of influent was rarely determined but was in agreement with the COD/TOC ratio that had been stable for years at 44.4 $g \cdot mol^{-1}$ for this influent. Accordingly, the average loads varied from 0.9 to 2.0 g $COD \cdot l^{-1} \cdot day^{-1}$ in the membrane reactor and 0.2–0.3 g $COD \cdot l^{-1} \cdot day^{-1}$ in the conventional reactor (see Table 2). The sludge load in the conventional reactor was relatively stable at 0.09 g COD·g $MLSS^{-1} \cdot day^{-1}$ as the sludge content was kept at 3–4 g $MLSS \cdot l^{-1}$; the sludge load in the membrane reactor decreased when the sludge content increased (see below).

Development of reactor contents

The development of the reactor contents was characterized by following the sludge concentration, the chemical composition and the biological properties. Since the experiments lasted very long and conditions were changing, the results were classified in several periods. The first distinguishing mark was given by two failures in the membrane reactor.

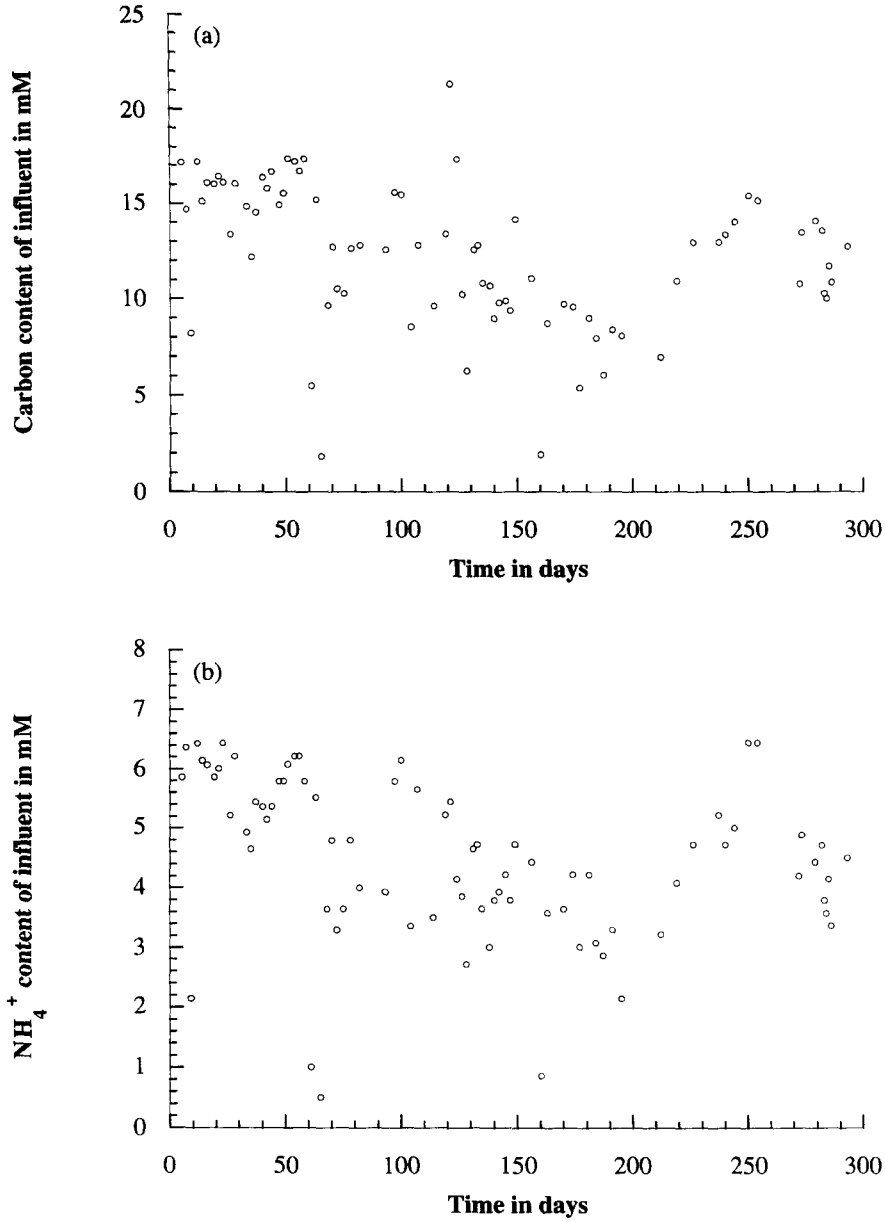

Fig. 3. Carbon (a) and ammonium (b) content of influent.

Around day 40 and day 90 sludge was lost as a result of membrane leakage and excessive foaming, respectively (see Fig. 4). This foaming was caused by detergents, which were insufficiently removed after a cleaning routine of the membrane unit. The three periods accordingly distinguished were also used to divide the results of the conventional reactor. The results of the membrane reactor in the last period were further split into the five parts as presented below; the development of the sludge concentration and the sludge load justified these distinctions.

Generally, the sludge concentration in the membrane reactor increased more slowly as time elapsed (see Fig. 4). Around day 40 and day 90 the sludge concentration abruptly declined because of experimental failures (see above). After day 93 the results were split in five periods. During the first period (from day 93 to 112) the sludge concentration increased rapidly (see Fig. 4). At the second period (from day 113 to 162), the sludge concentration accumulated more slowly. The third period (from day 163 to 220) showed a temporary stabilization of the sludge concentration at around 40 g MLSS·l^{-1}. This resulted from the lower carbon nitrogen concentrations in influent due to rainfall (see Fig. 3). Subsequently, the sludge concentration again increased from day 221 and 270. Finally, from day 271 until day 330 the sludge concentration remained at about 50 g MLSS·l^{-1}.

Table 2. Operating characteristics of the reactor with cross-flow filtration and the conventional reactor

Time (days)	HRT (h)	Carbon load ($mmol \cdot l^{-1} \cdot day^{-1}$)	Nitrogen load ($mmol \cdot l^{-1} \cdot day^{-1}$)	T (°C)	Dissolved O_2 in first compartment ($mg \cdot l^{-1}$)	Dissolved O_2 in last compartment ($mg \cdot l^{-1}$)
Membrane reactor						
0–35	7.4	44.7	20.6	21.4	1.3	2.8
43–82	7.7	37.4	16.1	23.0	1.4	3.6
93–112	6.6	44.5	21.1	22.4	0.7	1.6
113–162	6.6	38.2	16.8	19.7	1.1	1.7
163–220	7.1	23.8	11.2	18.0	0.6	4.3
221–270	9.9	28.0	13.5	19.1	0.8	4.6
271–300	14.8	19.5	8.5	22.7	0.9	3.1
Conventional reactor						
0–35	48.3	7.4	3.4	20.7	≥ 3.0	≥ 3.0
43–82	50.5	6.4	2.7	16.3	≥ 3.0	≥ 3.0
93–270	50.0	5.3	2.5	11.7	≥ 9.0	≥ 9.0

The accumulation of sluge was relative to the decreasing sludge load (see Fig. 5). When sludge concentrations increased rapidly, the sludge load declined sharply (see Figs 4 and 5). During these periods, much of the carbon supplied was built into sludge; from day 93 to 112 this even accounted for more than 60% of the organic carbon supplied (see Table 3). When the average load was 0.021 g COD·g $MLSS^{-1} \cdot day^{-1}$, i.e. between day 163 and 220 and between day 271 and 300, sludge concentrations had almost stabilized. Then, the incorporation of carbon and nitrogen into sludge was less than 6 and 2%, respectively (see Table 3). During the intermediate period (from day 221 to 270) the sludge load was 60% higher, which caused a threefold increase in the fraction of carbon assimilated. This additionally shows that even after 80 days of stabilization the bacterial population responded well to environmental changes. The sludge load in the conventional plant was intermediate (0.09 g COD·g $MLSS^{-1} \cdot day^{-1}$). This resulted in the assimilation of 22–35% of the carbon and about 10% of the nitrogen supplied.

Stabilization of sludge concentrations in membrane reactors have been reported previously. With synthetic waste water, however, sludge loads were determined to be much higher, i.e. 0.1 g COD·g $MLSS^{-1} \cdot day^{-1}$ and more (Yamamoto *et al.*, 1989; Chiemchaisri *et al.*, 1992). These loads were even higher than in the conventional reactor. This disagreement becomes even more pronounced when the composition of the synthetic waste water is taken into account. Since this contained glucose and peptone, which are high in energy content, sludge loads should have been lower. Therefore, it is likely that bacteria with higher growth efficiencies than common sludge bacteria had been selected. More in accord with our data are the results of Chaize and Huyard (1991). In this study, a stable sludge concentration was maintained when domestic wastewater was treated at 0.08 g COD·g $MLSS^{-1} \cdot day^{-1}$ with a sludge retention time of 100 days.

The chemical composition of sludge was examined by the regular determination of ash residue and carbon content. At times, the phosphorus and

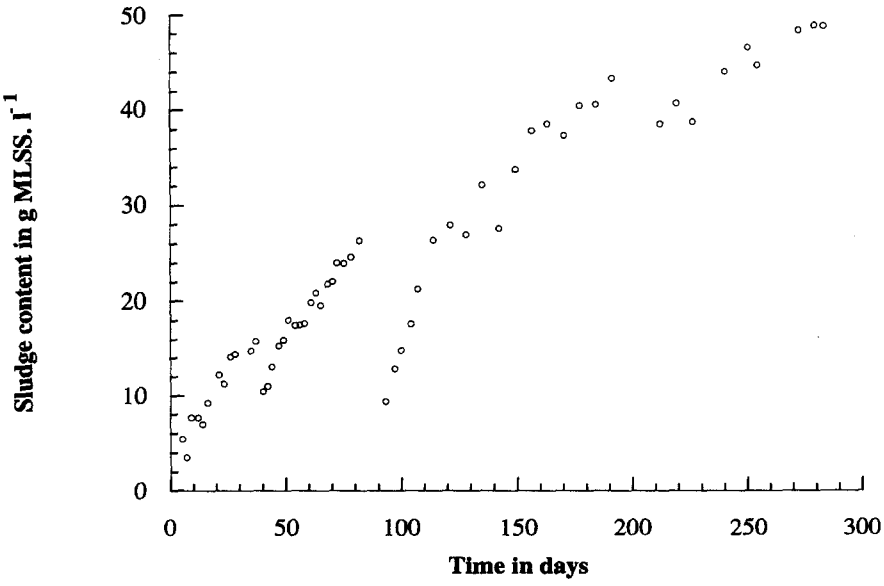

Fig. 4. Development of the sludge concentration in the membrane reactor with complete sludge retention. At day 36 and 83 sludge is lost as a result of membrane leakage and excessive foaming, respectively.

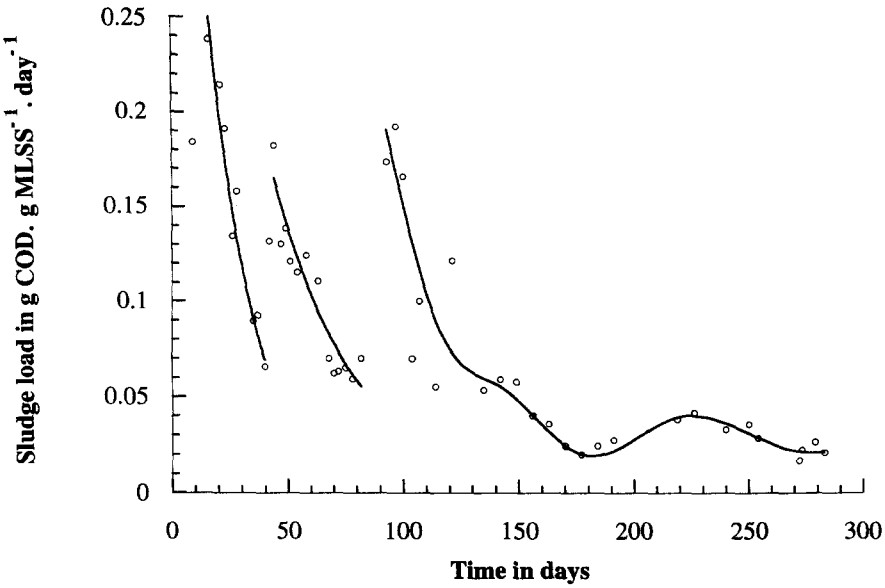

Fig. 5. Sludge loads in the membrane reactor with complete sludge retention.

kjeldahl nitrogen content were determined. In addition, the contents of several trace elements was determined at the end of the experiments. Table 4 shows that the ash residue in the membrane reactor as a fraction of the sludge concentration slightly increased. From day 93 until day 300 the fraction of non-volatile compounds increased only from 21.6 to 23.5%. This fraction was always lower than in the conventional reactor. The carbon content of sludge was somewhat higher in the membrane reactor and was stable during the whole experiment. Also, the fractions of nitrogen and phosphorus hardly changed during the experiment and were similar to those in the conventional reactor. At the end of the experiments, sludge of the membrane reactor contained a little more Cd, Cr, and Ni and less Ca, K, Mg, As, Ag and Hg than sludge from the conventional reactor, whereas the fractions of Fe, Zn, Cu and Pb were similar (see Table 5). This shows that if pre-settled and screened waste water is treated, complete sludge retention gives similar fractions of inorganic compounds as compared with sludge from conventional treatment plants. Hence, inorganic compounds will not disrupt treatment and membrane performances when sludge is completely retained.

The investigation of biological characteristics included microscopic analysis and the determination of nitrification capacities. The biomass in the membrane reactor consisted of a dense suspension of free cells, very small flocs ($<50\ \mu m$) and floc fragments. As a consequence, sludge could not be settled. Protozoa and metazoa were absent, which was probably a result of the high shear in the membrane unit. As opposed to this, the biomass in the conventional reactor was organized in firm and compact flocs with a size ranging from 100 to 1000 μm. In addition, the conventional reactor contained flagellates, ciliates, and at times rotifers, nematodes and mosquito larvae.

Nitrification capacities were determined to study the development of the nitrifying population. The nitrification capacity of the membrane reactor was determined three times: twice when the sludge

Table 3. Allocation of carbon and nitrogen supplied with influent

	Allocation of carbon				Allocation of nitrogen			
Time (days)	Effluent (%)	Sludge (%)	CO_2 (%)	Recovery (%)	Kj-N (%)	NO_3^- (%)	Sludge (%)	Recovery (%)
Membrane reactor								
0–35	9.2	26.7	ND	—	7.0	42.1	7.9	57.0
43–82	6.8	31.8	ND	—	2.8	58.2	10.0	71.0
93–112	7.1	60.9	ND	—	4.4	49.9	17.6	71.8
113–162	6.2	26.0	67.3	99.5	2.9	59.4	8.1	70.5
163–220	8.4	5.5	113.8	127.7	1.6	68.6	1.6	71.9
221–270	4.3	19.5	92.5	116.3	0.5	57.9	5.5	63.8
271–300	6.8	5.6	99.8	112.2	0.0	86.2	1.8	88.0
Conventional reactor								
0–35	10.0	22.2	ND	—	9.6	63.3	7.2	80.1
43–82	7.5	35.1	ND	—	4.1	73.0	12.2	89.3
93–270	8.3	23.1	ND	—	2.9	65.1	7.4	75.4

ND: not determined.

Table 4. Composition of sludge in the membrane and conventional reactor and maximal ammonium and nitrite consumption rates at 30°C

Time days	Ash (%)	Carbon (%)	Nitrogen (%)	Phosphorus (%)	Nitrification capacity (mmol NH_4^+ g $MLSS^{-1} \cdot h^{-1}$)	Nitrification capacity (mmol NO_2^- g $MLSS^{-1} \cdot h^{-1}$)
Membrane reactor						
0–35	20.5	44.6	6.2	2.8	ND	ND
43–82	21.0	40.1	ND	ND	0.18	0.19
93–112	21.6	42.7	7.2	ND	0.25	0.19
113–162	21.7	44.0	ND	ND	ND	ND
163–220	23.6	43.9	7.3	2.6	0.22	0.20
221–270	23.1	ND	ND	ND	ND	ND
271–300	23.5	ND	6.6	2.3	ND	ND
Conventional reactor						
0–35	26.7	38.8	6.2	2.8	ND	ND
43–82	26.6	37.8	6.7	ND	0.17	0.26
93–270	25.4	41.2	7.1	1.8	0.23	0.17

ND: not determined.

concentration increased relatively rapidly and once when the sludge concentration had been around 40 g $MLSS \cdot l^{-1}$ for three months (see Table 4 and Fig. 4). The capacity of the conventional reactor was determined twice as a reference. Despite the differences, the capacity for ammonia and nitrite oxidation of both reactors were always around 0.2 mmol $N \cdot g$ $MLSS^{-1} \cdot h^{-1}$ at 30°C (see Table 4). This demonstrates that the content of nitrifiers in sludge in the membrane reactor is equivalent to that in conventional plants with a low loading rate. Accordingly, kjeldahl nitrogen was always sufficiently removed in the membrane reactor (see below). Moreover, this demonstrates that the increase of the nitrifying biomass concentration kept pace with the increase of the sludge content. As a consequence, the capacity as a fraction of the sludge load for nitrogen had increased. Finally, this also demonstrates that the viability of the nitrifying population was not affected by prolonged substrate limitation; the capacities could have declined as a result of the decreased sludge loads for kjeldahl nitrogen (see Table 2 and Fig. 4).

Treatment performance

The treatment performance of the reactors was followed by determining the removal of carbon and kjeldahl nitrogen. For this purpose, effluents were examined on carbon and nitrogen contents. In addition, carbon dioxide production and oxygen consumption rates were determined to study carbon mineralization in the membrane reactor. In order to interpret the results, it should be recalled that the reactors were initially filled with sludge from an oxidation ditch.

Table 5. Content of trace elements in influent and in sludge from the membrane and conventional reactor after 300 days

Element	Influent ($\mu g \cdot l^{-1}$)	Sludge membrane reactor ($g \cdot kg^{-1}$)	Sludge conventional reactor ($g \cdot kg^{-1}$)
Ag	180	0.003	0.005
As	5	0.004	0.009
Ca	ND	30.0	50.0
Cd	0.4	0.007	0.003
Cr	3	0.140	0.045
Cu	70	0.500	0.540
Fe	1000	9.3	9.5
Hg	0.1	0.001	0.002
K	ND	5.3	11.0
Mg	ND	3.3	5.9
Ni	8	0.110	0.025
Pb	17	0.110	0.140
Zn	ND	1.6	1.8

ND: not determined.

From the start of the experiment onwards, carbon was almost completely removed in both reactors. During the first 50 days, the effluent from the membrane reactor contained between 1.0 and 1.5 mM carbon (see Fig 6). Subsequently, the concentration declined to 0.5–1.0 mM and remained stable thereafter. This corresponds to about 95% of carbon being removed (see Table 3). The carbon concentration in effluent from the conventional reactor followed a similar course, but was mostly 0.2 mM higher (data not shown). Also, the fraction of carbon removed that was a little higher (see Table 3). Hence, carbon removal in a membrane reactor is at least as good as in a conventional plant with a low loading rate. High amounts of carbon being removed have also been reported for laboratory-scale treatment plants supplied with synthetic waste water (Yamamoto *et al.*, 1989; Chiemchaisri *et al.*, 1992; Suwa *et al.*, 1992; Bailey *et al.*, 1994). For domestic waste water treatment, this finding has only been confirmed by disposing some sludge (Chaize and Huyard, 1991) and with an experiment that lasted for only two months (Chiemchaisri *et al.*, 1993). In addition to these findings, this study demonstrates that carbon removal is not affected by a decay of viable heterotrophs due to prolonged sludge retention.

The rates of carbon dioxide production and oxygen consumption were regularly determined in the membrane reactor since day 121 (see Fig. 7). These rates were quite variable, since the supply of influent was discontinuous and the influent quality varied substantially (see Fig. 3a). This resulted in the fraction of carbon mineralized being calculated inaccurately, i.e. the carbon recoveries were higher than 100% (see Table 3). Therefore, the theoretical flow to carbon

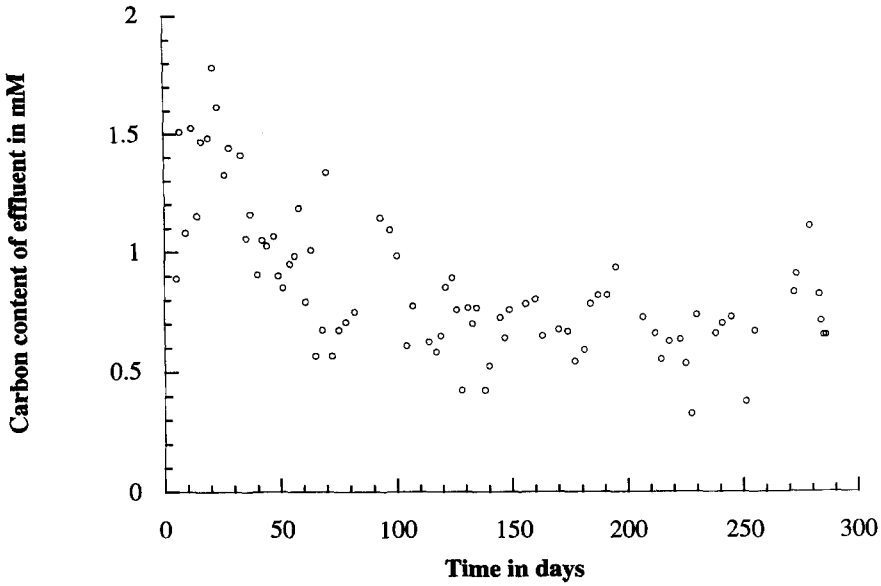

Fig. 6. Carbon content of effluent from the membrane reactor.

dioxide is a better estimate for this fraction. As shown in Table 3, this fraction should have been close to 90% when the sludge concentration had almost stabilized. Then, carbon oxidation was almost equivalent to carbon removal (see above). In the conventional reactor, carbon oxidation should have accounted for 60% of the supply (see Table 3). So, carbon mineralization is up to 50% higher at complete sludge retention as compared with conventional treatment at low loading rates.

Kjeldahl nitrogen was also sufficiently removed in both reactors. At all times, the nitrate concentration in the effluent from the membrane reactor was high, while the kjeldahl nitrogen content was negligible (see Fig. 8). The nitrogen compounds in effluent from the conventional reactor followed a similar trend (data not shown). So, nitrification proceeded well, which is in agreement with the stable nitrification capacities discussed above. Since nitrogen was hardly assimilated (see Table 3), kjeldahl nitrogen supplied should have completely been nitrified (Hanaki *et al.*, 1990; van Niel *et al.*, 1993). This has been demonstrated several times for treatment plants with cross-flow filtration (Yamamoto *et al.*, 1989; Chaize and Huyard, 1991; Chiemchaisri *et al.*, 1992, 1993). However, the nitrogen balances were always incomplete (see Table 3), which indicates that denitrification had occurred. Up to 40% of the nitrogen supplied should

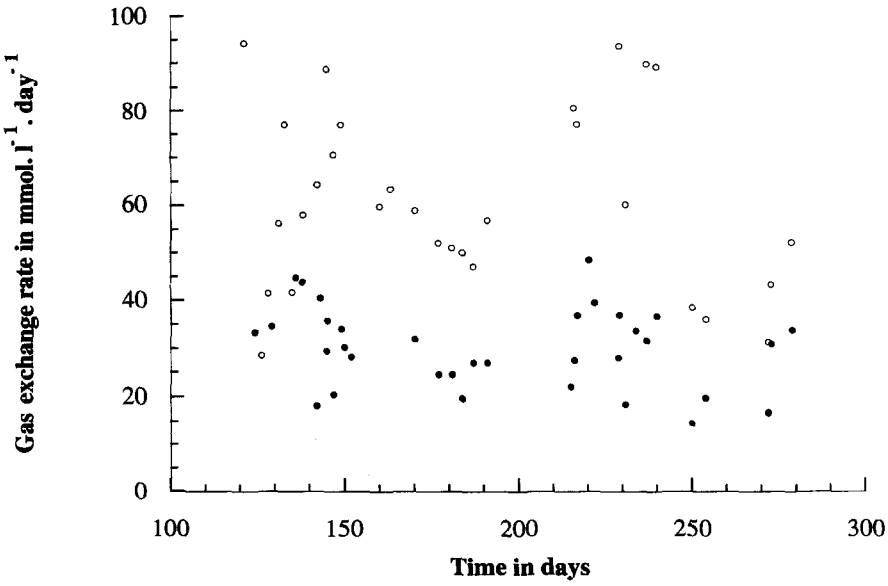

Fig. 7. Rates of carbon dioxide production (●) and oxygen consumpton (○) in the membrane reactor.

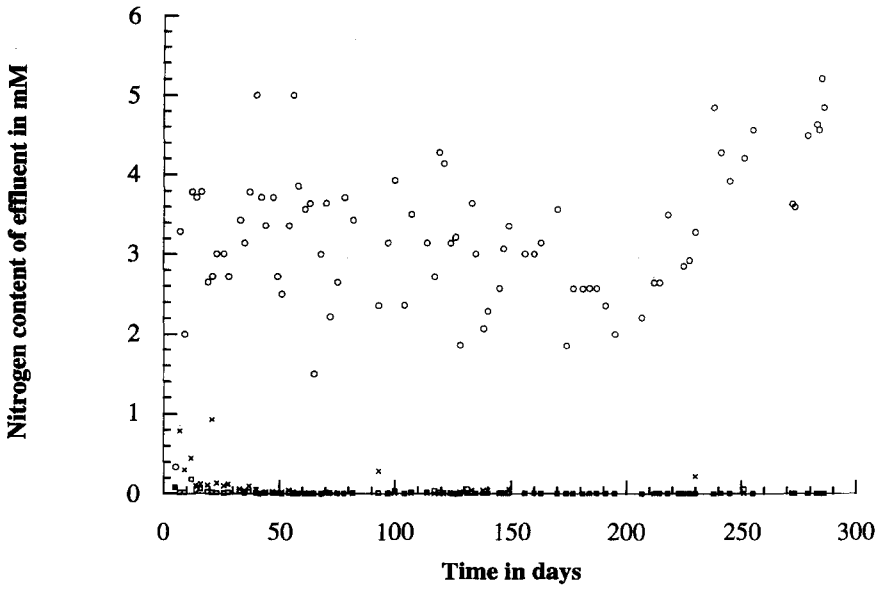

Fig. 8. Nitrate concentration (○), nitrite concentration (□) and ammonium concentration (×) in effluent from the membrane reactor.

have been denitrified in the membrane reactor (see Table 3). Denitrification could easily proceed, since oxygen demands were too high and oxygen transfer rates too low to maintain true aerobic conditions in the first compartment (see Table 2). This has also been found by Suwa *et al.* (1992), who have demonstrated that denitrification is enhanced at higher sludge concentrations, i.e. at reduced oxygen transfer rates. However, loss of nitrogen should not be considered as an advantage *per se*, as the nature of denitrification products is unknown. Both autotrophic nitrifiers and heterotrophic denitrifiers are able to produce nitric oxide and nitrous oxide (Krauth, 1993). Since these are greenhouse gasses, special attention should be paid to circumvent incomplete denitrification.

CONCLUSIONS

From a biological point of view, aerobic treatment of domestic waste water at high loading rates can be applied if sludge is completely retained. Then, the amount of sludge produced becomes very low. The treatment performance is at least as good as in a conventional plant with a low loading rate. Moreover, this performance is stable, i.e. both nitrifying and heterotrophic bacteria remain sufficiently viable to ensure proper treatment. However, cross-flow filtration is not yet an applicable alternative to conventional waste water treatment. Energy expenses are high because the transmembrane pressure has to be maintained (Krauth and Staab, 1993) and the costs for aeration increase at higher sludge concentrations. Moreover, the capacity of membranes rapidly declines as a result of fouling of the pores (Yamamoto *et al.*, 1989; Ben Aim *et al.*, 1993; Chiemchaisri *et al.*, 1993). Consequently, future research should focus on the development of retention techniques that are stable and cost effective, and, as is argued above, on nitrogen removal by enhancement of complete denitrification.

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