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Aquacultural Engineering 34 (2006) 322–331
aquacultural engineering
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Design and operations of the Kaldnes moving bed biofilm reactors

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Received 26 December 2004; accepted 18 April 2005

Abstract

The moving bed biofilm reactor (MBBR) was developed in Norway in the late 1980s and early 1990s. It is covered by several patents and has been a huge success world-wide for treatment of municipal and industrial wastewaters. In addition, MBBRs have been successfully used for biological treatment of drinking water as well as for water treatment in fish farms. The MBBRs use plastic biofilm carriers of a unique design, to maximize the active biofilm surface area in the reactors. Reactors have insignificant headloss, no need for periodic backwashing and no susceptibility for clogging. This paper describes the fundamentals of the MBBR. It has a major emphasis on nitrification with the type of biofilm carrier used in fish farms, but briefly touches upon removal of organic matter and denitrification. Major factors influencing the nitrification rates in MBBRs are discussed in detail. Results from small-scale MBBR tests, as well as from commercially operated MBBRs at full scale fish farms are presented. The data are from both freshwater and marine applications.

Keywords: Biofilm carriers; Biological treatment; Freshwater systems; Marine systems; Nitrification

1. Introduction

For biological treatment of water, there are many different biofilm systems in use, such as trickling filters, rotating biological contactors (RBC), fixed media submerged biofilters, granular media biofilters, fluidised bed reactors, etc. They all have their advantages and disadvantages. The trickling filter is not volume-effective. Mechanical failures have often been experienced with the RBCs. It is difficult to get even distribution of the load on the whole carrier surface in fixed media submerged biofilters. The granular media biofilters have to be operated discontinuously because of the need for backwashing and many of the fluidised bed reactors show hydraulic instability. For these reasons, the moving bed biofilm reactor (MBBR) process (European Patent no. 0,575,314, US Patent no. 5,458,779) was developed in Norway in the late 1980s and early 1990s (Ødegaard et al., 1994, 1999).

The MBBR has been a commercial success. There are presently more than 400 large-scale wastewater

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0144-8609/$ – see front matter
doi:10.1016/j.aquaeng.2005.04.002

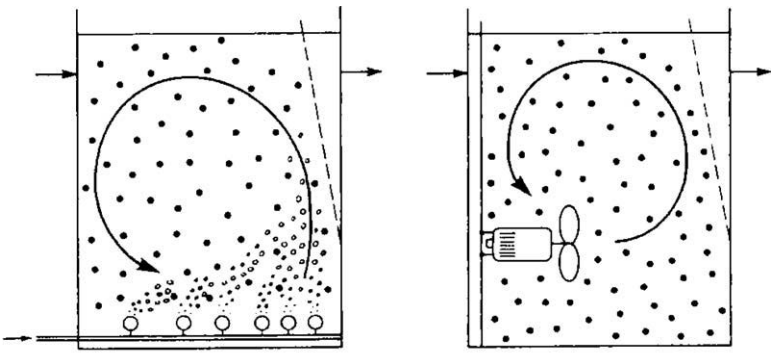

Fig. 1. Principle of the moving bed biofilm reactor (MBBR).

treatment plants based on this process in operation in 22 different countries all over the world. In addition there are several hundred small, on-site treatment units based on the MBBR—most of these in Germany. More than 50 MBBR plants are in operation at commercial fish farms, in addition to several hundred small MBBR systems for ornamental fish.

This paper describes the fundamentals of the MBBR and the design and use of MBBRs in fish farms. It has a major emphasis on nitrification, but briefly touches upon removal of organic matter and denitrification. Results from small-scale MBBR tests, as well as from commercially operated MBBRs at full scale fish farms are presented.

2. The Kaldnes moving bed biofilm reactor process

2.1. Description of reactors and biofilm carriers

The idea behind the development of the Kaldnes MBBR process was to adopt the best features of the activated sludge process as well as those of the biofilter processes, without including the worst. Contrary to most biofilm reactors, the MBBR utilises the whole tank volume for biomass growth. It also has a very low head-loss. Contrary to the activated sludge reactor, it does not need any sludge recycle. This is achieved by having the biomass grow on carriers that move freely in the water volume of the reactor and that are kept within the reactor volume by a sieve arrangement at the reactor outlet. The reactor may be used for aerobic, anoxic or anaerobic processes, as illustrated in Fig. 1.

In aerobic processes, the biofilm carrier movement is caused by the agitation set up by the air, while in anoxic and anaerobic processes a mixer (normally a horizontal shaft mounted banana mixer) keeps the carriers moving. For the aerobic reactors, a special coarse bubble aeration system has been developed. Special sieve arrangements to retain the biofilm carriers within the reactors have also been developed. This may be vertically mounted, rectangular mesh sieves, but lately the sieve is more often shaped as a cylindrical bar sieve, vertically or horizontally mounted. Proper design of aeration grids and sieves is very important for optimum performance of the MBBR process. Based on comprehensive and systematic testing, detailed guidelines for aeration and sieve design have been established and are proprietary information of the AnoxKaldnes companies.

One important advantage of the moving bed biofilm reactor is that the filling fraction of biofilm carriers in the reactor may be subject to preferences. In order to be able to move the carrier suspension freely, it is recommended that filling fractions should be below 70%. One may, however, use as much as needed below this. A number of different carriers have been developed by AnoxKaldnes. Data for the three smallest carriers are listed in Table 1 and Fig. 2 shows a photo of these carriers. Most plants, and all the fish farm plants referred to in this paper, use the original Kaldnes K1 carrier. The carriers are made of polyethylene (PEHD) with a density of 0.95 g/cm^3. Since the biomass is growing primarily on the

Table 1
Data for some Kaldnes biofilm carriers

	Type of Kaldnes biofilm carrier		
	K1	K2	K3
Nominal diameter (mm)	9.1	15	25
Nominal length (mm)	7.2	15	12
Bulk density (kg/m^3)	150	95	100
Specific biofilm surface area (in bulk) (m^2/m^3)	500	350	500
Specific biofilm surface area at 60% fill (m^2/m^3)	300	210	300

protected surface on the inside of the carriers, only the effective biofilm surface area of the carriers is given in Table 1. The total surface area is significantly larger than the effective biofilm surface area.

Biofilm carriers from the first full scale MBBR plant to be in commercial operation in Norway are routinely inspected. After 15 years of uninterrupted operation no wear and tear of the carriers has been observed.

As in every biofilm process, diffusion of compounds in and out of the biofilm plays a key role. Because of the importance of diffusion, the thickness of the effective biofilm (the depth of the biofilm to which the substrates have penetrated) is important. Since this depth of full substrate penetration is normally less than 100 μm, the ideal biofilm in the moving bed process is thin and evenly distributed over the surface of the carrier. In order to obtain this, the turbulence in the reactor is of importance, both in order to transport the substrates to the biofilm and to maintain a low thickness of the biofilm by shearing forces.

2.2. Removal of organic matter

For treatment of municipal and industrial wastewater, MBBR plants have performed very well at high organic loads and high substrate concentrations (Ødegaard et al., 2004). The turbulence caused by the high air flow necessary to maintain 3 mg O_2/L in aerobic reactors at high organic loads, has been more than sufficient to maintain a fairly thin biofilm and prevent clogging of the biofilm carriers.

At fish farms, the objective has always been to nitrify, in addition to removing organic matter. Because the heterotrophic biomass that removes organic matter will out-compete the nitrifying biomass at high organic loads, MBBRs for fish farms have always been operated at very low organic loads. Even though we know that MBBRs remove a lot of organic matter in fish farm installations, measuring the organic loads, organic removal rates, and organic substrate concentrations over the MBBRs has never been a priority.

2.3. Nitrification

Nitrification in Kaldnes MBBRs has been thoroughly studied using both synthetic wastewater (Hem et al., 1994) and municipal wastewater (Rusten et al., 1995a). As for all biofilm reactors, nitrification rates are influenced by the organic load, the dissolved

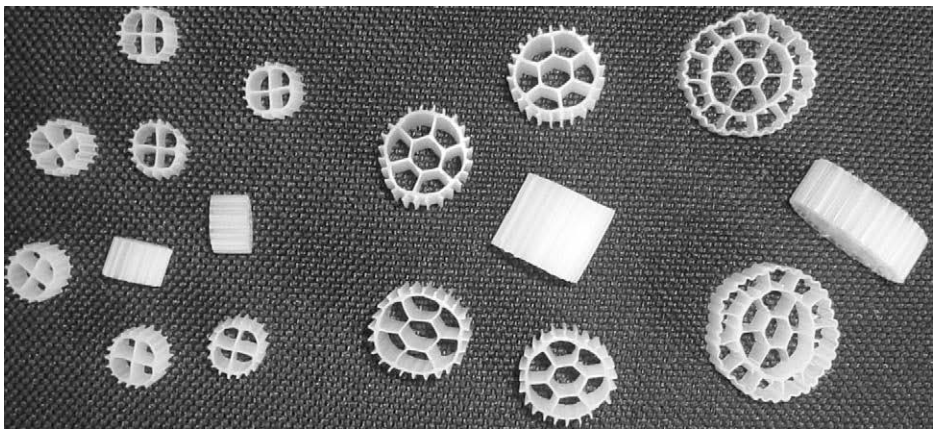

Fig. 2. Photo of (from left to right) Kaldnes type K1, K2 and K3 biofilm carriers.

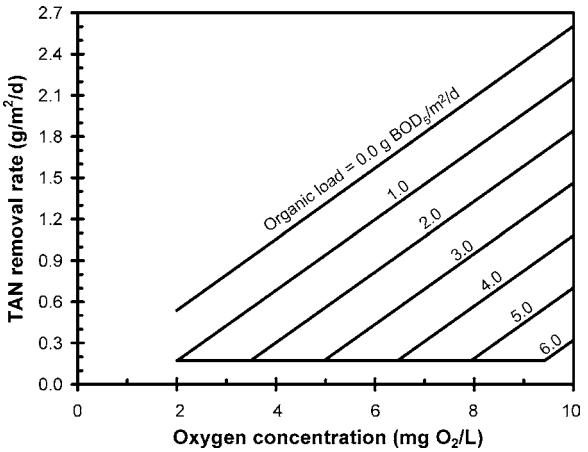

Fig. 3. Influence of organic load and reactor DO concentration on TAN removal in a Kaldnes MBBR, at 15 °C and with TAN in excess (≥2.5 mg N/L). Adapted from Hem et al. (1994).

oxygen (DO) concentration in the reactor, the total ammonium nitrogen (TAN) concentration, the temperature, the pH and alkalinity, and the previous history of the biofilm.

An example of the relationship between the TAN removal rate, the reactor DO concentration and the organic load is shown in Fig. 3, for a situation with 15 °C and excess TAN concentration (Hem et al., 1994). At an organic load of 1 g BOD_5/m^2 biofilm surface area/d, a TAN removal rate of 1 g NH_4–N/(m^2 d) was achieved at a DO concentration of about 5 mg/L. To achieve the same TAN removal rate at an organic load of 3 g $BOD_5/(m^2$ d), Fig. 3 shows that the reactor must be operated at a DO concentration of about 8 mg/L.

Due to diffusion effects in biofilms, nitrification rates are very dependent on TAN concentrations and DO concentrations. Normally oxygen will be the rate limiting substrate at high TAN concentrations, and TAN will be the rate limiting substrate at low TAN concentrations. In fish farms, the TAN concentration will normally be less than 1 mg NH_4–N/L, which for all practical purposes will make TAN the rate limiting substrate.

A model for predicting nitrification rates in MBBRs was developed a few years ago (Rusten et al., 1995a). With TAN as the rate limiting substrate, the model is as shown in the following equation:

$$r_N = k(S_N)^n \quad (1)$$

where r_N = nitrification rate as g NH_4–N/(m^2 d); k = reaction rate constant; S_N = TAN concentration in the reactor as mg NH_4–N/L; n = reaction order constant.

A reaction order constant of $n = 0.7$ was established by Hem et al. (1994), and the reaction rate constant (k) will depend on wastewater characteristics, temperature and other parameters that influence the growth of nitrifying organisms.

For transition from oxygen to TAN as the rate limiting substrate, in the absence of biodegradable organic matter, a ratio of 3.2 between DO concentration (mg/L) and TAN concentration (mg/L), as reported by Szwerinski et al. (1986), has been used for MBBR modeling. In the presence of biodegradable organic matter, heterotrophic activity in the outer layer of the biofilm will reduce the oxygen concentration available for nitrification (Harremoës, 1982). In MBBRs, the DO concentration reduction over this outer layer has been estimated to be about 0.5 mg O_2/L at very low organic loads, and increasing to about 2.5 mg O_2/L at an organic load of 1.5 g $BOD_5/(m^2$ d) (Rusten et al., 1995a).

Fig. 4 shows an example of TAN removal rate versus TAN concentration at different DO concentrations, for a MBBR at 15 °C and low organic load. The curves indicate transition from TAN limitation to DO limitation at a TAN concentration of about 0.5 mg NH_4–N/L when the reactor DO is 2 mg/L. At a reactor DO concentration of 6 mg/L, the transition from TAN

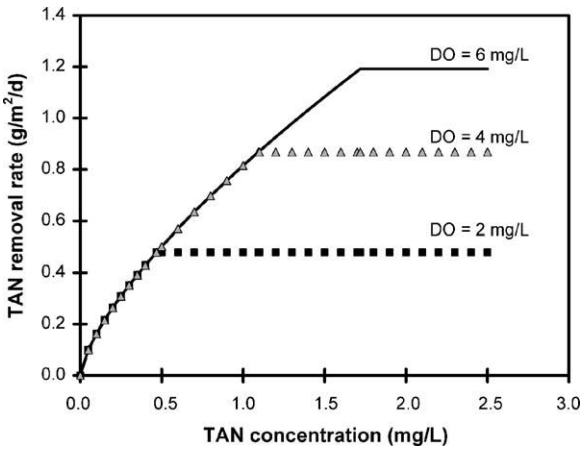

Fig. 4. Influence of TAN and DO concentrations on TAN removal in a Kaldnes MBBR at 15 °C and 0.4 g $BOD_5/(m^2$ d) organic load. Based on reaction rate equation and data from Rusten et al. (1995a).

limitation to DO limitation takes place at a TAN concentration of 1.7 mg NH_4–N/L.

Nitrifying bacteria are slow-growing organisms and it will always take a long time to reach the full nitrification potential in a biofilm reactor. Even in municipal wastewater treatment plants, with higher ammonium concentration than typically seen in fish farms, nitrification rates will still be increasing after a year of operation (Boller and Gujer, 1986). Hem et al. (1994) demonstrated that the intrinsic reaction rate in a MBBR would not only be influenced by the present concentrations and loads, but also by the history of the biofilm. When the biofilm was acclimated at a high TAN load, the reaction rate constant was more than twice as high as when the biofilm was acclimated at a low TAN load.

Temperature has a major effect on the nitrification rate and can be modeled by the following equation:

$$k_{T_2} = k_{T_1}\theta^{(T_2-T_1)} \quad (2)$$

where T_1, T_2 are temperatures in °C; k_{T_1} reaction rate constant at T_1; k_{T_2} reaction rate constant at T_2 and θ is the temperature coefficient.

For MBBRs, a temperature coefficient of $\theta = 1.09$ has been established (Rusten et al., 1995a).

The reaction rate constant (k) will be lower when the load of organic matter and/or particulate matter to the MBBR reactor increases, because both these factors will dilute the concentration of nitrifying bacteria in the biofilm. At low alkalinity, the reaction rate constant (k) will also decrease, due to a reduction of the pH inside the biofilm. Tests have demonstrated that a higher residual alkalinity was necessary for a thick biofilm than for a thin biofilm (Rusten et al., 1995a). This was believed to be due to a smaller pH reduction inside the nitrifying part of the biofilm, compared to the bulk water pH, for a thin biofilm than for a thick biofilm. For a thin biofilm, maximum nitrification rates were seen down to an alkalinity of 0.7 mmol/L.

2.4. Denitrification

Denitrification with MBBRs has been thoroughly tested using both internal and a variety of external carbon sources (Rusten et al., 1994, 1995b, 1996). A lot of full-scale MBBRs are used for denitrification at wastewater treatment plants. In the Netherlands, MBBRs are used for denitrification at a couple of large fish farms.

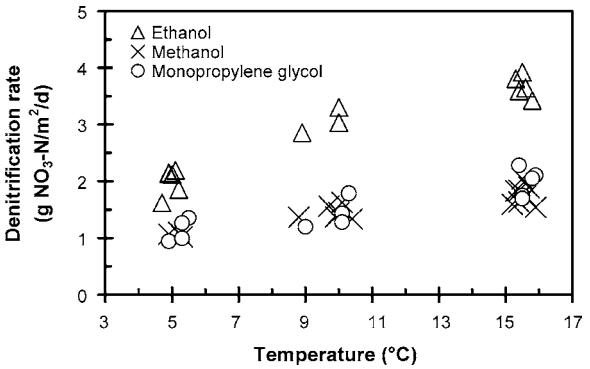

Fig. 5. Denitrification rate versus temperature, obtained with various external carbon sources (data from Rusten et al., 1996).

Systems with inadequate supply of carbon source tend to reduce some of the nitrate (NO_3) to nitrite (NO_2), instead of all the way to nitrogen (N_2) gas. Since NO_2 is very toxic to fish, it is important to minimize the NO_2 concentration by using an external and controllable carbon source. Fig. 5 shows denitrification rates in MBBRs for three different external carbon sources, when the nitrate concentration was not rate limiting (Rusten et al., 1996). Use of methanol and monopropylene glycol (MPG) gave similar denitrification rates. Use of ethanol doubled the denitrification rates. Due to the very low cost, methanol is the most commonly used external carbon source. However, starting up a denitrification reactor with methanol takes a long time (Rusten et al., 1996), because only a few bacteria can utilise methanol and enrichment of this group of bacteria takes time (Nurse, 1980). Due to the significantly shorter start-up time and higher denitrification rates, ethanol may be the best external carbon source for denitrification in fish farm applications.

3. Freshwater application of MBBRs for Atlantic salmon smolt production

The BIOFISH concept, developed by SINTEF in Norway (Eikebrokk, 1990; Eikebrokk and Piedrahita, 1997), uses a simplified recirculation technology where each fish tank at a fish farm has its own single stage biofilm reactor. From the moment the Kaldnes

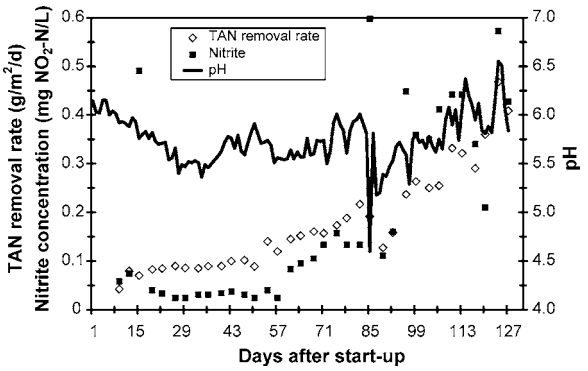

Fig. 6. TAN removal rate, nitrite concentration and pH for BIOFISH system during freshwater production of Atlantic salmon smolt. Temperature of 12.6 ± 1.2 °C and DO of 8.7 ± 1.5 mg/L. Based on data from Ulgenes (1997).

MBBR technology became commercially available, the majority of the BIOFISH systems have used MBBRs.

The performance at an application for Atlantic salmon (*Salmo salar*) smolt production was closely monitored by SINTEF (Ulgenes, 1997). Fig. 6 shows some results from the start-up of this plant. The MBBR was started with virgin biofilm carriers that had not previously been exposed to fish feed or TAN, nor inoculated with nitrifying bacteria. This approach will work only if the system is started at a low load and with a low fish density in the fish tanks. Otherwise the nitrite peaks bound to appear may be detrimental to the fish. The empty bed hydraulic retention time (HRT) in the MBBR was approximately 2.5 min and the reactor had 70 % fill of the Kaldnes type K1 biofilm carriers.

The system was started at a temperature of 10 °C, but within a week the temperature was fairly stable at 12–13 °C. During the first 60 days the TAN load was low and the TAN removal rate was established at about 0.1 g NH_4–N/(m^2 d) after 2 weeks of operation. During these first 60 days nitrite concentrations where mainly below 0.05 mg NO_2–N/L, except for one very high reading of nitrite early on. However, pH values between 5.4 and 6.0 were significantly lower than the optimum pH-range for nitrifying bacteria.

As the TAN load was increased, the TAN removal rate reached 0.4–0.5 g NH_4–N/(m^2 d) after about 125 days of operation. The system was unstable, however, as shown by frequent nitrite concentrations in the 0.2–0.6 mg NO_2–N/L range. These high nitrite concentrations can be explained by the low and fluctuating pH, rapid changes in TAN load and also some rapid temperature fluctuations. At the end of the period shown in Fig. 6, the temperature dropped 6–7 °C over a 2-day-period on a couple of occasions. The highest nitrite value of 0.6 mg NO_2–N/L was observed when the pH suddenly dropped to 4.6. These results emphasize the importance of pH control, temperature control and only gradual changes in TAN load, in order to guarantee very low concentrations of nitrite in the system.

A maximum TAN removal rate of 0.18 g NH_4–N/(m^2 d) at 14 °C and a pH of 6.8, has been reported for a submerged biological filter with random biofilm media at an Atlantic salmon smolt farm (Rusten, 1989; Rusten and Harr, 1989). Compared to this the TAN removal rates shown for the MBBR in Fig. 6 were surprisingly high, taking into account the very low pH and slightly lower temperature in the MBBR system.

4. Freshwater application of MBBRs for production of brown trout and arctic char juveniles

A BIOFISH system for production of brown trout (*Salmo trutta L*) and arctic char (*Salvelinus alpinus*) juveniles was monitored over a 3-year-period (Ulgenes and Lundin, 2003). Fig. 7 shows TAN removal rates versus TAN loads for fish tanks 3 and 4. Both tanks were started directly with fish and no prior

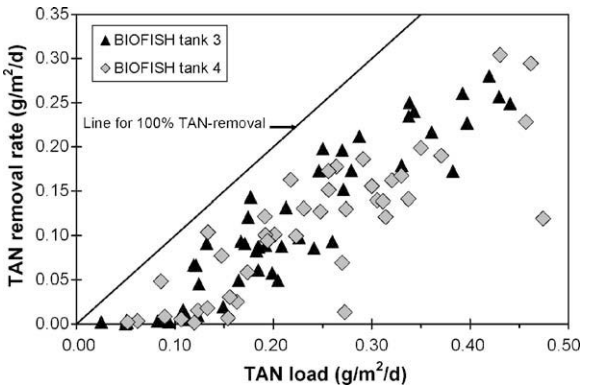

Fig. 7. TAN load and removal rate for BIOFISH system during freshwater production of brown trout and arctic char. Temperature of 8.6 ± 0.8 °C, DO of 9.7 ± 1.0 mg/L, and pH of 6.2 ± 0.2. Based on data from Ulgenes and Lundin (2003).

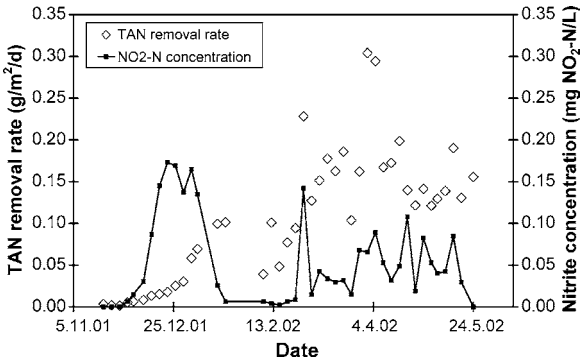

Fig. 8. TAN removal rate and nitrite concentration for BIOFISH system during freshwater production of brown trout and arctic char juveniles. Temperature of 8.5 ± 0.8 °C, DO of 9.4 ± 1.0 mg/L, and pH of 6.2 ± 0.2. Based on data from Ulgenes and Lundin (2003).

conditioning of the biofilm carriers with organic matter or TAN. Maximum TAN removal rates of 0.30 g NH_4–N/(m^2 d) were reached at a TAN load of 0.45 g NH_4–N/(m^2 d) and a temperature of 9 °C. The empty bed HRT in the MBBRs was approximately 3.5 min, and each reactor had 67% fill of the Kaldnes type K1 biofilm carriers. With an average of 6.2 the pH was significantly lower than the optimum range for nitrifying bacteria. However, the pH in the system was fairly stable due to automatic dosing of sodium carbonate (Na_2CO_3).

TAN removal rates close to zero in Fig. 7 were from the first days after start-up, as seen for fish tank 4 in Fig. 8. At an average temperature of about 8 °C, the nitrite peak in this system appeared from 20 to 50 days after start-up. Due to an initial fish density as low as 15 kg/m^3, however, the nitrite peak during start-up never went above 0.17 mg NO_2–N/L and no stress symptoms were seen on the fish. The nitrite peak of 0.14 mg NO_2–N/L after slightly more than 3 months of operation can be explained by a sudden doubling of both the TAN load and the TAN removal rate. In such cases, nitrifying systems are far outside the optimum steady state situation and tend to show increased nitrite concentrations.

5. Marine application of MBBRs for farming of turbot

Bench-scale tests and full-scale tests of MBBRs have been carried out at commercial land-based fish farms for turbot (*Scopthalmus maximum*) in Spain (Rusten, 2001a) and Portugal (Rusten, 2001b). The plant in Spain had two MBBRs in parallel, with a total wet volume of 600 m^3, 5.0 m water depth and 50% fill of Kaldnes type K1 biofilm carriers. The salinity at this plant was 24‰. The plant in Portugal had two biofilters. Biofilter 1 had four reactors in series, and biofilter 2 had three reactors in series. Each reactor had a wet volume of 33 m^3, 3.0 m water depth and a fill of Kaldnes type K1 carriers that ranged from 55 to 72 %. The salinity at the plant in Portugal varied from 21 to 24‰.

5.1. Bench-scale tests

Nitrification tests with biofilm carriers, taken from the two biofilter reactors in Spain and from reactor 1 of biofilter 1 in Portugal, were carried out in March 2001. Both plants were started with virgin biofilm carriers at low loads. The biofilter in Spain had been in operation for 4 months, and the biofilters in Portugal had been in operation for about 2 years. The bench-scale reactors were spiked with ammonium, in order to see what maximum TAN removal rates that could be obtained with the existing nitrifying biomass on the biofilm carriers. The results are summarized in Table 2. The biofilm carriers from the Spanish fish farm were accustomed to a concentration at or below 0.1 mg NH_4–N/L, while the biofilm carriers from the Portuguese fish farm were taken from a full-scale reactor with concentrations normally from 0.4 to 0.6 mg NH_4–N/L. In addition to the relatively short time from start-up of the plant in Spain, the differences in ammonium concentrations that the nitrifying bacteria had been exposed to led to a maximum, potential TAN removal rate that was more than 10 times higher for biofilm carriers from the Portuguese fish farm than for biofilm carriers from the Spanish

Table 2
TAN removal rates with ammonium in excess in bench-scale MBBRs with biofilm carriers from fish farms for turbot. Rates are temperature compensated to 15 °C

Location	Salinity (‰)	pH	DO concentration (mg O_2/L)	TAN removal rate at 15 °C (g TAN/(m^2 d))
Spain	24	7.5	7.5	0.07
Portugal	21	7.6	6.9	0.95

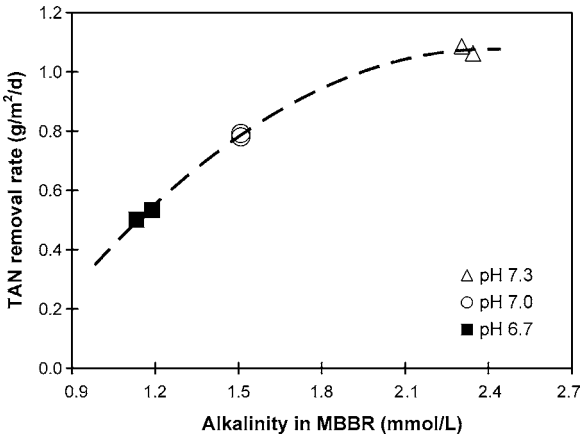

Fig. 9. Influence of alkalinity and pH on the TAN removal rate, using nitrifying biofilm carriers from a fish farm for turbot. Salinity of 23‰, temperature of 15.6–16.2 °C, DO of 6.3–6.8 mg/L, and TAN concentrations of 3.0–4.5 mg/L.

fish farm. The actual TAN removal rates in the full-scale reactors at the time the tested carriers were sampled from the reactors were 0.03 g NH_4–N/(m^2 d) for the plant in Spain and 0.25 g NH_4–N/(m^2 d) for the plant in Portugal.

At the turbot farms in both Spain and Portugal, the pH and alkalinity in the water will drop, due to the nitrification and very low addition of new water to the system, unless alkaline chemicals are added. It was of interest to see how the pH and alkalinity would influence the TAN removal. Therefore, a bench-scale experiment was run with biofilm carriers taken from reactor 1 of biofilter 2 at the Portuguese fish farm. The results from this test are shown in Fig. 9. Going from pH 7.3 and an alkalinity of 2.3 mmol/L to pH 6.7 and an alkalinity of 1.1–1.2 mmol/L, the nitrification rate dropped to only half of the original rate. At recirculation plants with a very high degree of recirculation, pH and alkalinity control will be a good idea.

The highest TAN removal rates in Table 2 and Fig. 9 are significantly higher than the rates reported for full-grown seawater biofilms by Bovendeur (1989).

5.2. Full-scale results

During normal operation the MBBRs at the plant in Portugal were run at empty bed HRTs from 2 to 5 min. At the highest flow rates, the cylindrical bar sieves used in the reactors had hydraulic loads that were

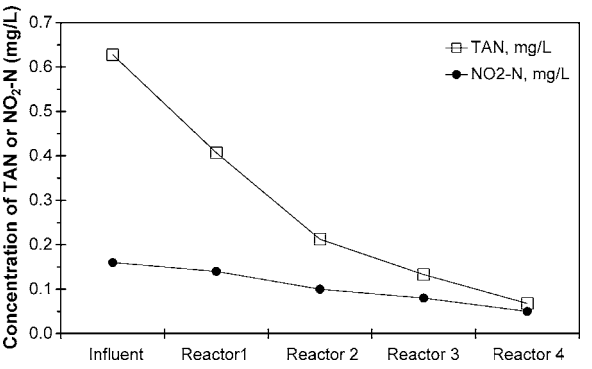

Fig. 10. Example of influent and reactor 1 through reactor 4 concentrations of TAN and NO_2–N for biofilter 1 at the water recirculation plant for turbot farming in Portugal. Salinity of 21‰, temperature of 17.4 °C and empty bed hydraulic retention time of 4.7 min per reactor.

significantly higher than recommended by Anox-Kaldnes.

An example of NH_4–N and NO_2–N concentrations in the influent and in the different reactors in biofilter 1 at the water recirculation plant in Portugal is shown in Fig. 10. Before the MBBR treated water was returned to the fish tanks the TAN concentration was reduced from 0.63 to 0.07 mg NH_4–N/L and the nitrite concentration was reduced from 0.16 to 0.05 mg NO_2–N/L.

An example of TAN removal rates versus reactor NH_4–N concentrations for the MBBRs at the plant in Portugal is shown in Fig. 11. The residual least squares best fit for the previously mentioned MBBR nitrifica-

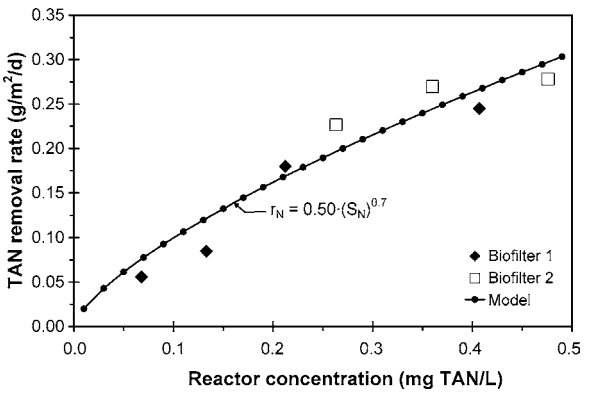

Fig. 11. Example of TAN removal rates versus reactor TAN concentrations for biofilter 1 and biofilter 2 at the water recirculation plant for turbot in Portugal. Salinity of 21‰, temperature of 17.4–17.5 °C, DO of 6.4–7.3 mg/L, and pH of 7.43–7.48.

tion model is also shown. The best fit was found for a reaction rate constant (k) of 0.50, resulting in a correlation coefficient of 0.96 between observed and predicted TAN removal rates.

Using the MBBR nitrification model on the highest TAN removal rates, shown in Table 2 and Fig. 9, gives a slightly higher reaction rate constant than found from the data in Fig. 11. Compared to reaction rate constants found for a situation with low concentrations of organic material and a mature biofilm in freshwater (Rusten et al., 1995a), the reaction rate constants found for the MBBRs at the water recirculation plant for turbot farming in Portugal indicate nitrification rates that are approximately 60 % of what can be expected in a freshwater system. However, nitrification rates in seawater are expected to be significantly lower than in freshwater (Bovendeur, 1989).

6. Conclusions

In properly designed moving bed biofilm reactors (MBBRs), the whole reactor volume is active, with no dead space or short circuiting. Different types of biofilm carriers can be used, but the dominating type is the Kaldnes type K1 carrier that has an active biofilm surface area of up to 350 m^2/m^3 reactor volume. It is very important that the design of MBBR systems, including the aeration grids and sieves, is done by engineers with proper qualifications.

MBBRs in aquaculture applications will mainly be used for nitrification. The nitrification process is influenced by the organic load, the dissolved oxygen (DO) concentration in the reactor, the total ammonium nitrogen (TAN) concentration, the temperature, the pH and alkalinity, and the previous history of the biofilm.

If a system for nitrification is started with virgin biofilm carriers, it is important to have a start-up period with a very low and gradual increase in TAN load. Otherwise the nitrite peak during start-up may be so high that it is toxic to the fish.

Sudden nitrite peaks may appear whenever a nitrifying system is unstable. Therefore, it is important to have good pH and temperature control, and to make sure that there are no high and sudden increases in the TAN load.

Nitrification consumes alkalinity and reduces the pH. In systems with high recirculation, it is important to have pH control by adding alkaline chemicals. Tests at a marine fish farm showed that the nitrification rate at pH 6.7 was only 50% of the nitrification rate observed at pH 7.3.

Start-up of a marine system for nitrification will take a very long time when using virgin biofilm carriers. Nitrification rates in marine systems will also be significantly lower than for comparable freshwater systems. Data from a fish farm operating at a salinity of 21–24 ‰ indicated that the nitrification rate was approximately 60% of what can be expected in a freshwater system.

On a biofilm surface area basis TAN removal rates in MBBRs have compared very favorably to rates reported in the literature.

References

Boller, M., Gujer, W., 1986. Nitrification in tertiary trickling filters followed by deep-bed filters. Water Res. 20, 1363–1373.

Bovendeur, J. (1989) Fixed-biofilm reactors applied to waste water treatment and aquacultural water recirculation systems. Ph.D. Thesis, Wageningen Agricultural University, Wageningen, The Netherlands.

Eikebrokk, B., 1990. Design and performance of the BIOFISH water recirculation system. Aquacultur. Eng. 9 (4), 285–294.

Eikebrokk, B., Piedrahita, R.H., 1997. Nitrification performance and ammonia excretion from salmonids, calculated from nitrogen mass balances in water reuse systems under field and laboratory conditions. J. Appl. Aquacult. 7 (4), 15–32.

Harremoës, P., 1982. Criteria for nitrification in fixed film reactors. Water Sci. Technol. 14, 167–187.

Hem, L.J., Rusten, B., Ødegaard, H., 1994. Nitrification in a moving bed biofilm reactor. Water Res. 28 (6), 1425–1433.

Nurse, G.R., 1980. Denitrification with methanol: microbiology and biochemistry. Water Res. 14 (5), 531–537.

Ødegaard, H., Rusten, B., Westrum, T., 1994. A new moving bed bofilm reactor—applications and results. Water Sci. Technol. 29 (10–11), 157–165.

Ødegaard, H., Rusten, B., Siljudalen, J., 1999. The development of the moving bed biofilm process—from idea to commercial product. Eur. Water Manage. 2 (3), 36–43.

Ødegaard, H., Rusten, B., Wessman, F. (2004) State of the art in Europe of the moving bed biofilm reactor (MBBR) process. Paper presented at WEFTEC'04 in New Orleans, 4 October.

Rusten, B., 1989. Start-feeding of Atlantic salmon in a water recycling plant. Aquacultur. Eng. 8 (4), 241–256.

Rusten, B., Harr, C., 1989. Design and evaluation of a water recycling plant for Atlantic salmon smolt production. In: Aquaculture—A Biotechnology in Progress, EAS, Bredene, Belgium, pp. 1143–1148.

Rusten, B., Siljudalen, J.G., Nordeidet, B., 1994. Upgrading to nitrogen removal with the KMT moving bed biofilm process. Water Sci. Technol. 29 (12), 185–195.

Rusten, B., Hem, L.J., Ødegaard, H., 1995a. Nitrification of municipal wastewater in moving-bed biofilm reactors. Water Environ. Res. 67 (1), 75–86.

Rusten, B., Hem, L.J., Ødegaard, H., 1995b. Nitrogen removal from dilute wastewater in cold climate using moving-bed biofilm reactors. Water Environ. Res. 67 (1), 65–74.

Rusten, B., Wien, A., Skjefstad, J., 1996. Spent aircraft deicing fluid as external carbon source for denitrification of municipal wastewater: From waste problem to beneficial use. In: Purdue Industrial Waste Conference Proceedings, vol. 51. Ann Arbor Press Inc., Chelsea, Michigan.

Rusten, B. (2001a) Water treatment and flow rates at the Aquacria Arousa fish farm—tests and observations from site visits in March 2001. Aquateam Report no. 01-036.

Rusten, B. (2001b) Water quality and water treatment at the Aquacria Piscicolas fish farm—tests and observations from site visit in March 2001. Aquateam Report no. 01-062.

Szwerinski, H., Arvin, E., Harremoës, P., 1986. pH-decrease in a nitrifying biofilm. Water Res. 20, 971–976.

Ulgenes, Y. (1997) Behandling og sikring av avløp fra innlandsoppdrett. SINTEF Report, Trondheim, Norway, March.

Ulgenes, Y. and Lundin, U. (2003) Valsjöbyn Fiskodling—Dokumentasjon av BIOFISH-anlegg for produksjon av settefisk. SINTEF Report STF66 A03105, Trondheim, Norway.