

# Compact wastewater treatment with MBBR

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## ABSTRACT

The paper describes the moving bed biofilm reactor (MBBR™) and MBBR-based processes for different wastewater treatment applications. Around 800 treatment plants in more than 50 countries are now utilizing MBBR-based processes. In the moving bed biofilm reactor the biomass is growing on plastic carriers that are freely moving in the water as a consequence of mixing by air (aerobic reactors) or mixing devices (anoxic or anaerobic reactors). A sieve is retaining the carriers that are made of a plastic material with a density close to 1 g/cm<sup>3</sup>, resulting in easy motion of the carriers up to volumetric filling fractions of 70 %. The MBBR-reactors are used in pure biofilm processes (aerobic or anoxic) as well as in integrated fixed film activated sludge (IFAS) processes and for organic matter as well as for nutrient removal (including anammox processes). In this paper focus is on the pure MBBR-based processes, particularly the combined pre- and post-denitrification MBBR process commonly used in Norway.

## 1. INTRODUCTION

Biofilm processes are increasingly being favored before activated sludge processes for biological wastewater treatment. There are several reasons for this:

- a. The treatment plant requires less space (an important cost factor)
- b. The final treatment result is less influenced by biomass separation since the biomass concentration to be separated is at least 10 times lower and there is greater flexibility in choice of biomass separation method (i.e. compact settling, flotation or filtration)
- c. The attached biomass becomes more specialized (higher concentration of relevant organisms) at a given point in the process train, because there is no biomass return

There are already 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 have all their advantages and disadvantages. The trickling filter is not volume-effective. Mechanical failures are often 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 the fluidized bed reactors show hydraulic instability. For these reasons the moving bed biofilm reactor (MBBR) was invented and developed in Norway in the late 80'ies and early 90'ties [1-2]. There are presently more than 800 treatment plants based on the MBBR in operation in more than 50 different countries all over the world out of which half of them are for municipal wastewater and the other half for various industrial wastewaters. This paper will give an overview over the process solutions that may be selected for green field plants as well as for upgrading of activated sludge plants and discuss how the use of the MBBR technology may be optimized for nutrient removal.

The MBBR was originally developed and patented as a pure biofilm reactor (without sludge recycle), but lately there has been a growing interest in using the MBBR technology for hybrid or Integrated Fixed film Activated Sludge systems (IFAS) (with sludge recycle) as well. In this paper focus will be on pure MBBR-processes. For the use of MBBR in IFAS processes the reader is referred to [3].

## 2. DESCRIPTION OF THE MOVING BED BIOFILM REACTOR (MBBR)

Contrary to most biofilm reactors, the MBBR utilizes the whole tank volume for biomass growth, as does also the activated sludge reactor. Contrary to the activated sludge reactor, however, it does not need any sludge recycle, as is also the case in other biofilm reactors. This is achieved by having the biomass grow on carriers that move freely in the water volume of the reactor, kept within the reactor volume by a sieve arrangement at the reactor outlet. Since no sludge recirculation takes place, only the surplus biomass has to be separated - a considerable advantage over the activated sludge process. The reactor may be used for both aerobic, anoxic or anaerobic processes, see Figure 1.

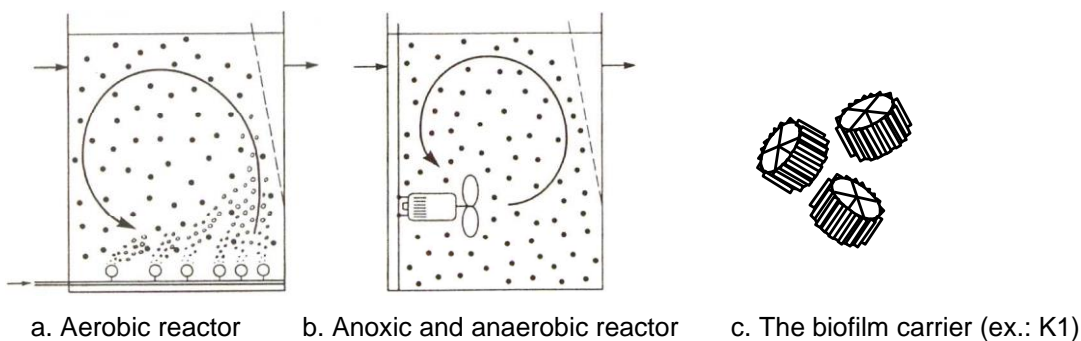


Figure 1: Principle of the MBBR and shape of the original biofilm carrier (Kaldnes K1)

The original biofilm carrier (Kaldnes K1) and most of the newer carriers (see examples in Figure 2) are made of high density polyethylene (density  $0.95 \text{ g/cm}^3$ ). Hence they are easily totally mixed in the reactor volume by the mixing set up by aeration (in aerobic reactors) and mixers in anoxic reactors (see Figure 1 and 2).

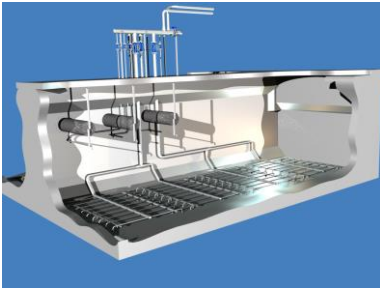


Figure 2: The moving bed biofilm reactor and various carriers supplied by AnoxKaldnes

One of the important advantages of the moving bed biofilm reactor is that the filling fraction of carrier in the reactor may be subject to preferences. At the maximum recommended filling fraction for the K1 and K3 carriers (bulk specific area  $500 \text{ m}^2/\text{m}^3$ ) in aerobic reactors of 65 %, the specific area in the reactor will be  $325 \text{ m}^2/\text{m}^3$ . For the K5 and Biofilm chip carriers, the bulk specific area is higher, but the maximum filling fraction a bit lower (60 and 55 % respectively). One may, however, use as much as needed below the maximum filling fraction. This is very convenient when using the MBBR technology for upgrading activated sludge plants.

The carriers are kept within the reactor by an outlet sieve. This may be vertically mounted, rectangular mesh sieves, but lately the sieve is more often shaped as a cylindrical bar sieve, vertically or horizontally (now more common) mounted, see Figure 3.

Even with a medium to coarse bubble diffuser system, the oxygen transfer in an MBBR is not inferior to that of a fine-bubble diffuser system in activated sludge. This is caused by the fact that the bubbles are partly caught by the moving carriers and partly broken in contact with the carriers.



a. Typical design of an aerobic MBBR



b. Aerobic MBBR with horizontally mounted cylindrical bar sieves



c. Anoxic MBBR with horizontally mounted shaft mixers and rectangular sieves

Figure 3: Examples of MBBR reactors

Some suppliers of MBBR systems recommend medium-sized bubble aerators while others recommend fine bubble diffusers. It has been demonstrated that the oxygen transfer in a tank filled with carriers is considerably higher than without carriers and that it increases with carrier filling fraction up to approximately 60 % where values above  $15 \text{ g O}_2/\text{Nm}^3 \cdot \text{m}$  have been measured in clean water tests medium-sized bubble aerators [4].

In an activated sludge plant, the process of developing an active biomass begins with the growth of zoogeal bacterial floc which then becomes colonized by protozoa which feed on the free swimming bacteria to produce a clarified effluent. In the moving bed process, the order of colonization seems to be reversed [5]. High loading rates, around  $30 \text{ g COD}/\text{m}^2\text{d}$ , produce compact bacterial biofilms, with protozoan population either absent or limited to small free-swimming protozoa and *Vorticella* spp. Moderate loading rates, around  $10\text{-}15 \text{ g COD}/\text{m}^2\text{d}$  promote a more "fluffy" biofilm with a rich variety of ciliated protozoa. Low loading rates ( $<5 \text{ g COD}/\text{m}^2\text{d}$ ) promote a biofilm generally dominated by stalked ciliates.

It has been demonstrated [6] that the biofilm area is the key parameter in design and therefore the design rate of the process is most correctly based on effective carrier area ( $\text{g}/\text{m}^2_{\text{carrier area}} \cdot \text{d}$ ).

### 3. THE MBBR PROCESSES MOSTLY USED FOR MUNICIPAL WASTEWATERS

Figure 4 show the most commonly used MBBR processes for BOD-removal, nitrification and nutrient-removal. When P-removal is required, these processes are normally combined with chemical precipitation either as pre-precipitation (chemically enhanced primary), simultaneous precipitation (in case of IFAS) or post-precipitation.

In Figure 4 it is differentiated between pure MBBR processes and MBBR-based IFAS processes. Pure MBBR processes are often preferred in green-field plants (Figure 4a, 4c, 4e and 4h), but are also used for pre-treatment of existing activated sludge plants in order to achieve nitrification (Figure 4b, 4d, 4f, and 4g) or post-treatment (Figure 4j) in order to achieve denitrification. MBBR-based IFAS processes are mostly used for upgrading of activated sludge plant in order to achieve nitrification (Figure 4k and 4l), N-removal (Figure 4m) or N- and P-removal (Figure 4n). In this paper pure MBBR-based processes are in focus. For more information on IFAS processes the reader is referred to [3].

Pre-settling or fine screening is normally used as pre-treatment primarily to avoid clogging of the bio-reactor sieves. Pre-settling is not a prerequisite, but used to reduce load on the bio-reactor and to add flexibility to the process by allowing chemically enhanced primary treatment – if needed.

In Figure 4 the separation steps are depicted as settling tanks but other separation processes may be used in combination with the pure MBBR processes, including lamella sedimentation, microsand ballasted lamella sedimentation (Actiflo), flotation, granular media filtration and UF/MF-membrane filtration – all commonly combined with coagulation/flocculation.

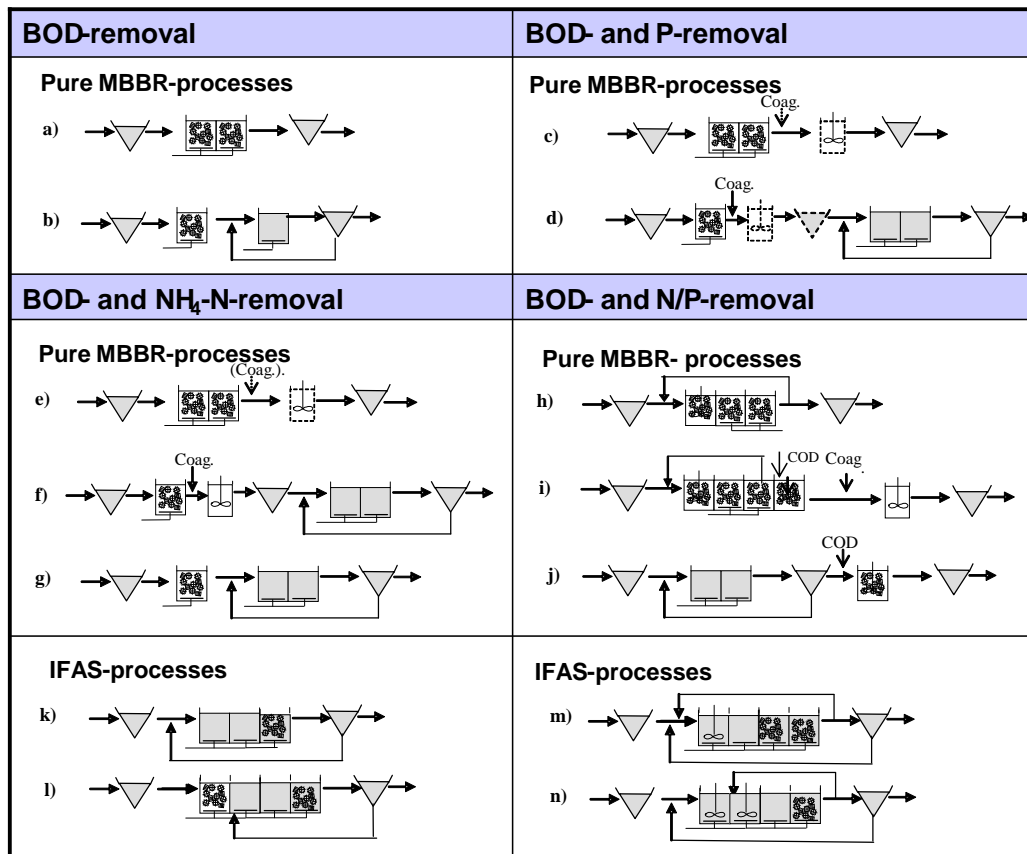


Figure 4: MBBR-based processes used for municipal wastewater treatment

## 4. BOD- AND BOD+P-REMOVAL

### 4.1 Secondary treatment

Because of the compactness, the residence time in moving bed reactors for carbonaceous matter removal will be quite low (30-90 minutes), depending on the strength of the wastewater and the effluent requirement. Biodegradable, soluble organic matter is quickly degraded. Particulate organic matter is partly caught by the irregularities of the attached biomass, hydrolysed and degraded, and partly it passes more or less unchanged through the reactor.

The most efficient use of the MBBR for secondary treatment only is, therefore, to design for a high organic loading in the first stage of the bioreactor and a lower loading on the later stages. This should be combined with a relatively low hydraulic surface loading on the settling tank - or preferably - use of coagulants (at a higher surface loading). Thus one will take advantage of the fact that the biomass will operate at a higher removal rate at a high organic loading and will remove the easily biodegradable organic matter first (in the first stage) before hydrolyzing particulate organic matter mainly in the later stages. In most European countries, phosphate removal is required and carried out by chemical precipitation. It is very common, therefore, to use the process in Figure 4c when the target is BOD/COD and P-removal. These plants will be equipped with adequately designed flocculation tanks and they may use alum, iron chloride or pre-polymerised aluminium chloride as coagulants.

It is recommended always to have a minimum of two reactor stages and since the first stage should have a higher organic loading than the second stage, a division of the total volume of 25 % in the first reactor step and 75 % in the second is recommended.

Typical design values are 8 g BOD<sub>5</sub>/m<sup>2</sup>·d at 10 °C (ca 17 g BOD<sub>5</sub>/m<sup>2</sup>·d at 20 °C) for the whole reactor (divided in at least two steps) without any post-coagulation/flocculation and 11.5 g BOD<sub>5</sub>/m<sup>2</sup>·d at 10 °C (ca 25 g BOD<sub>5</sub>/m<sup>2</sup>·d at 20 °C) for the whole reactor (divided in at least two steps) with post-coagulation/flocculation

## 4.2 The high-rate MBBR process

As a stand-alone process or for upgrading of existing (AS) plants, the high rate MBBR-process may be used. The idea behind the high rate process is that the biomass in the biofilm shall only degrade the soluble, most easily biodegradable organic matter while the particulate, more heavily biodegradable organic matter shall pass the reactor, more or less unchanged. This requires an organic loading on the MBBR that is low enough for truly soluble organic matter to be biodegraded and high enough to prevent significant hydrolysis and degradation of suspended and colloidal organic matter. In designing such a high rate process, therefore, it is especially important to know the distribution of soluble and particulate organic matter.

One must take into account that in highly loaded plants biodegradable, soluble organic matter is also produced in the process by hydrolysis. In highly loaded plants, clarification of the biomass also represents a problem since the settle-ability of the biomass decreases with increasing organic loading [6]. The key to solving this problem is either biomass flocculation by coagulant addition (metal salts or cationic polymers) and/or use of another type of biomass separation, such as flotation or filtration.

As demonstrated in Figure 5 the reaction rate will be at its maximum (and 0 order) of around 30 g BFCOD/m<sup>2</sup>·d (20 g SBOD/m<sup>2</sup>·d) at an MBBR effluent concentration of biodegradable soluble (i.e. filtered) COD (BFCOD) > 150-200 mg/l or ca. 100-150 mg SBOD/l [7]. Below this the reaction order will be close to half order, but even down to 100 mg BFCOD/l (or 70 mg SBOD/l) the removal rate will be as high as 20 g BFCOD/m<sup>2</sup>·d (ca. 15 g SBOD/m<sup>2</sup>·d) corresponding to around 60 g COD<sub>total</sub>/m<sup>2</sup>·d (ca. 40 g BOD<sub>total</sub>/m<sup>2</sup>·d). At this high rate a very small fraction of the particulate COD will be converted to soluble COD while at a load of around 5 g BFCOD/m<sup>2</sup>·d, as much as 70 % of the particulate COD may typically be converted to soluble, as demonstrated in Figure 6.

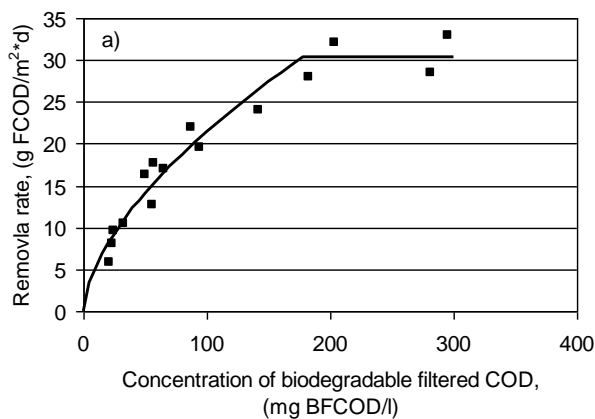


Figure 5 The removal of soluble (1 μm filtered) COD versus the concentration of soluble (i.e. filtered) biodegradable COD [7]

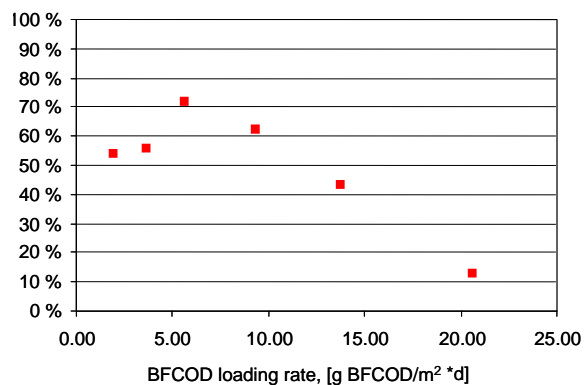


Fig. 6 Percentage of particulate COD (PCOD) converted to soluble COD and degraded versus biodegradable, soluble COD (BFCOD) loading rate [7]

If used for pre-treatment before an activated sludge plant, there are two alternatives with respect to biomass separation or not. The benefit of separating the heterotrophic biomass ahead of the proceeding activated sludge plant is that this biomass will not contribute to lowering of the sludge age in the activated sludge plant, resulting in less volume needed in the activated sludge bioreactor. It was demonstrated [8] in a fairly large scale pilot-plant (5 m<sup>3</sup>/h) based on a two compartment, high-rate MBBR followed directly with iron+cationic polymer coagulation, that the process could operate favourably at dosages as low as 5 mg polymer/g SS and 35 mg Fe/g SS – corresponding to dosages of 1 mg polymer/l and 7 mg Fe/l at a typical 200 mg SS/l in the bioreactor effluent. Cationic polymer was combined with iron for coagulation in order to operate with a very low dosage of iron and thereby minimize hydroxide precipitation and consequently sludge production. Flotation was chosen because a highly loaded bioreactor may result in poorly settle-able but easily floatable sludge and also because it is a low foot-print separation technique.

The alternative with a high-rate MBBR directly followed by an activated sludge plant (as shown in Figure 4b) is used in a plant in Wellington, New Zealand. This alternative is, of course



simpler and preferable if the activated sludge can meet the demand at the lower sludge age (as compared to alternative 4e) that will prevail. It is experienced in plants according to this process solution that the settling property of the activated sludge is significantly improved. A combination of the high rate MBBR pre-treatment and IFAS (Figure 4) may also be used favourably – as is the case in the Sharjah WWTP in UAE.

## 5. NITRIFICATION

### 5.1. Nitrification in green field MBBR plants

In a classical MBBR design (with two or more reactors in series) nitrification will be normally be limited by organic matter in the first reactor where the heterotrophs are dominating because they outcompete the nitrifiers for the oxygen. If the organic matter concentration is brought sufficiently down (for instance in the first reactor segment), the nitrifiers will establish themselves in the proceeding reactor segments. As long as the ammonia concentration is above 0.5-1.5 mg NH<sub>4</sub>-N/l (depending on oxygen concentration, see Figure 7) the nitrification rate will not be limited by ammonium, but by oxygen concentration. Only at very low ammonium concentrations the nitrification rate will be governed by ammonium. These relationships are shown in Figure 7 and 8. They are used to design MBBR-plants for nitrification.

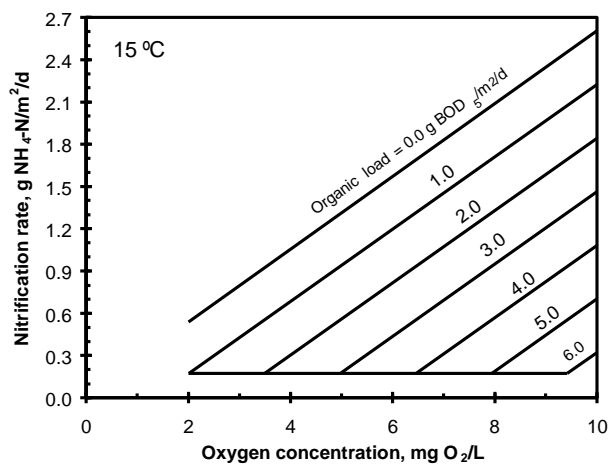


Figure 7 Influence of organic load and reactor DO on nitrification rate in MBBR's at 15°C and ammonium in excess [9]

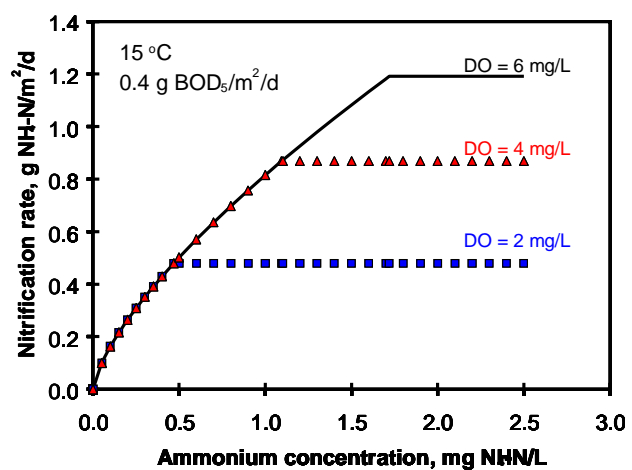


Figure 8 Influence of ammonium and DO on nitrification rate in MBBR's at 15°C and 0,4 g BOD<sub>5</sub>/m<sup>2</sup>d organic load [10]

Figure 7 demonstrates that the organic load controls nitrification and should be as low as possible. At organic loads above ca 5 g BOD<sub>5</sub>/m<sup>2</sup>d no nitrification will take place at normal O<sub>2</sub> concentrations and the first BOD-removal reactor segment may be designed for an organic load, that will lower the organic load on the second reactor segment by around 90 %, to 0,5 g BOD<sub>5</sub>/m<sup>2</sup>d.

Thereafter the proceeding nitrification reactor segments may be designed based on nitrification rate, taking the oxygen concentration into consideration according the following model [9-10]:

$$r_N = k \cdot (S_n)_n,$$

$r_N$  = nitrification rate (g NH<sub>4</sub>-N/m<sup>2</sup>· d)

$k$  = reaction rate coefficient

$n$  = reaction order constant, can be estimated at  $n = 0.7$

$S_n$  = rate-determining ammonium concentration, mg NH<sub>4</sub>-N/l  
(can be estimated at  $S_n = (DO_{bulk} - 0.5)/3.2$ )

$k$  will be dependent upon the the C/N-ratio (g BOD<sub>5</sub>/g NH<sub>4</sub>-N) on of the incoming water to the bioreactor and vary from 0,7 at C/N = 0,5 and to 0,5 at C/N = 4,5 [3]. If the preceding MBBR

reactors are designed to remove essentially all BOD then the nitrification rate coefficient for the final nitrification reactors may be set at 0,75.

Figure 7 demonstrates that the nitrification rate is linearly dependent upon the oxygen concentration, up to more than 10 mg O<sub>2</sub>/l [1-11]. Oxygen concentration may, however, limit the nitrification rate even at concentrations as low as 2-3 mg O<sub>2</sub>/l and normally one is designing for DO concentrations of 4-6 mg/l at peak load. An advantage of the linear relationship between oxygen concentration and nitrification rate is that it may very favourably be used for process control.

If extremely low values of ammonium are required, the last reactor segment must be designed for an ammonium limited rate – in which case oxygen concentration may be low since it is no longer governing the rate of nitrification (see Figure 8).

## 5.2 Upgrading of activated sludge plants for nitrification

Quite often there is a situation where an existing activated sludge plant achieving secondary treatment only, is to be upgraded in order to nitrify and reach low ammonium values as well. One obvious strategy would be to abandon the activated sludge process and convert the aeration volumes into MBBR's. Since nitrification in MBBR's normally can be achieved within a residence time of < 3 hrs, the volumes available will in most cases be more than sufficient. However, in many cases the plant-owner likes to keep the activated sludge plant and three alternative strategies for this situation exist:

1. Use a high-rate MBBR step ahead of the activated sludge reactor
  - a. With heterotrophic biomass separation after the MBBR ahead of AS (Figure 4f)
  - b. Without biomass separation after the MBBR (Figure 4g)
2. Establish an IFAS process in the AS bioreactor (Figure 4k)
3. Combine 1. and 2. (Figure 4d)

The IFAS processes (see Figure 4k and l) have become very popular for upgrading of existing activated sludge plants aiming at achieving nitrification. In this process part of (or the whole of) an activated sludge bioreactor is retrofitted into an IFAS process. By this the nitrifying biomass of the system is increased and nitrification may be achieved in a volume that otherwise could not comply with a nitrification standard. The IFAS is not an easy process to design, however, since presence of nitrifiers shifts from the biofilm on the carriers to the suspended biomass and vice versa depending on the load of organic matter and ammonium respectively. At high sludge age (low organic load) a relatively speaking larger fraction of the nitrification takes place in the suspended biomass, while at lower sludge ages a higher fraction of the nitrification takes place in the biofilm. It seems to be a much more stable process than AS-nitrification alone, because at increasing load and decreasing sludge age (resulting in decreased suspended solids nitrification, nitrification is enhanced in the biofilm. Since we are focusing on pure MBBR systems in this paper, the reader is referred to [3] when it comes to design of IFAS systems. The conclusions regarding nitrification in IFAS systems from that paper shall, however, be included here:

1. Nitrification is essentially independent of the aerobic MLSS SRT and can be achieved at SRT's in the suspended biomass as low as approximately 2 d, because it governed by the nitrification taking place in the biofilm biomass when the loading is high (low MLSS SRT). At lower organic loadings more nitrification takes place in the suspended biomass.
2. The nitrification activity of the biomass (g NH<sub>4</sub>-N<sub>removed</sub>/kg VSS·h) is far higher in the biofilm biomass (up to 8-9 g NH<sub>4</sub>-N<sub>removed</sub>/kg VSS·h, at 20°C) than in the suspended biomass (normally 2-3 g NH<sub>4</sub>-N<sub>removed</sub>/kg VSS·h, at 20 °C) at a typical influent C/N-ratio of 3-4 and DO of 4 mg O<sub>2</sub>/L.
3. As in every biofilm system, the nitrification rate is strongly influenced by reactor DO concentration. It is not advisable to design for a DO below 4-6 mg O<sub>2</sub>/L at peak load. Oxygen transfer is high (freshwater SOTR of around 12 g O<sub>2</sub>/Nm<sup>3</sup>·m provided filling the filling fraction is >50%) even with the use of medium-bubble diffusers, because of the presence of the carriers.
4. Design of nitrification reactors may be based on the design in pure biofilm systems with an additional effect from the suspended biomass – depending of the load (C/N-ratio or SRT).
5. As the temperature decreases, the fraction of the total nitrification that takes place in the biofilm biomass increases. This makes the IFAS resilient to temperature variations.

## 5. N-REMOVAL

### 5.1 N-removal in MBBR-plants

N-removal in a pure MBBR-plant may be carried out by pre-denitrification (Figure 4h), post denitrification (not shown in Figure 4) or most commonly in municipal wastewater treatment by a combined pre- and post-denitrification process, normally with chemical post-precipitation (Figure 4i)[12-13].

The combined-DN MBBR plants have normally a division between aerobic and anoxic compartments of the MBBR as indicated in Figure 9. The first compartment (comp 1) is anoxic (pre-denitrification) and receives recycled nitrate from the last aerobic compartment. The second compartment (comp 2) may be operated both anoxically (no air) and aerobically (with air). Then follow two aerobic reactors (comp 3 and 4), used for residual organic matter removal and nitrification. Compartment 5 is also used for nitrification and this compartment may be operated with or without aeration. The sixth compartment, to which carbon is added, is anoxic and for post denitrification. Finally there is optionally a small aerobic compartment for oxygenation as well as for removal of any residual biodegradable organic matter.

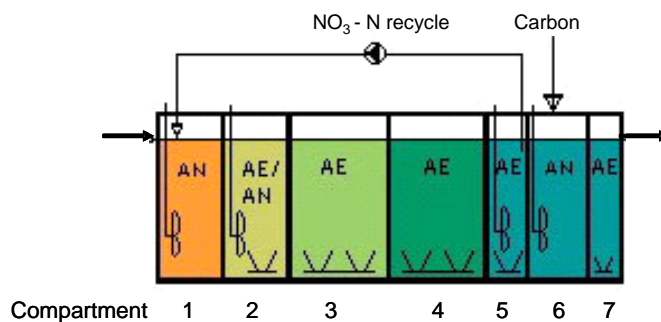


Figure 9 The typical build-up of a combined pre- and post-denitrification MBBR

This bioreactor scheme may give very high nitrogen removal efficiencies and optimal flexibility of operation. The use of external carbon source is minimized. During summer operation, for instance, less nitrification volume is needed because of higher temperatures and the raw water has normally a higher DO concentration than in the winter time. In this situation compartment 2 may be used for denitrification (stirred and not aerated). More nitrate may be returned to the pre-denitrification tank and the need for added carbon source in the post-DN tank is lower. In summer time, therefore, most of the nitrogen can be removed by pre-denitrification, reducing the consumption of external carbon source to just that what is needed to ensure that the necessary effluent nitrogen concentration is achieved.

During winter operation, more nitrification capacity is needed and compartment 2 is operated with aeration. More carbon source will have to be added to the post-denitrification tank in order to achieve the treatment goal. In a situation with extremely cold wastewater for a period, chemically enhanced primary treatment may be used to reduce the organic load to a minimum and maximizing nitrification capacity of the existing volume. The reason for the option of not aerating compartment 5, is to minimize the return of oxygen to the pre-denitrification reactor. In compartment 5 the ammonia concentration is normally low (< 3 mg/l) which means that the oxygen is no longer rate limiting for the nitrification rate, but rather the ammonia concentration. Therefore it does not matter if the DO is reduced which minimize oxygen respiration in the pre-DN compartment caused by the recirculation of oxygen.

The extent of nitrification may be controlled by the oxygen level in the nitrification reactor (compartment 4) since there is a linear relationship between nitrification rate and oxygen concentration. The extent of denitrification in the post denitrification step is governed by the availability of carbon source and nitrate measurement in the post denitrification outlet may therefore control carbon source addition and hence the final denitrification result.



## 5.2 Pre-denitrification.

A number of parameters, such as DO concentration,  $\text{NO}_x\text{-N}$  ( $\text{NO}_2\text{-N} + \text{NO}_3\text{-N}$ ) concentration, amount of biodegradable organic matter and temperature, influence the denitrification process and the optimum operation strategy. Recirculation of oxygen significantly reduces the denitrification rate. Tests have shown that at a DO concentration in the reactor of  $0.6 \text{ mg O}_2/\text{L}$ , the denitrification rate was reduced to 50 % of the potentially maximum rate [14]. Therefore, recirculation ratios should be limited to a region where DO concentrations in denitrification reactors stay low, preferably below  $0.5 \text{ mg O}_2/\text{l}$ . On-line DO measurements in denitrification reactors have successfully been used to regulate recirculation in full-scale plants. Recirculation is typically limited to 100 to 200 % of the influent flow, depending on temperature and wastewater composition.

An example of pre-denitrification rates as a function of the biodegradable, soluble COD (BSCOD) concentrations in the reactor influent is shown in Figure 10, together with corresponding C/N-consumption ratios. The data are for reactor 1 at a pilot-plant, with  $\geq 1.1 \text{ mg NO}_x\text{-N/l}$  in the reactor [14]. With low influent BSCOD concentrations, the C/N-consumption ratios, measured as  $\text{g BSCOD/g NO}_x\text{-N}$ , were lower than with high influent BSCOD concentrations. This indicates that hydrolysis products from particulate COD were used as carbon source when available BSCOD was limited. Figure 10 also confirms that low concentrations of BSCOD in to the reactor, which may be a result of diluting the influent wastewater by high recirculation ratios of nitrified wastewater, will lead to very low denitrification rates. The optimum recirculation may therefore be less than 100%, especially at low temperatures.

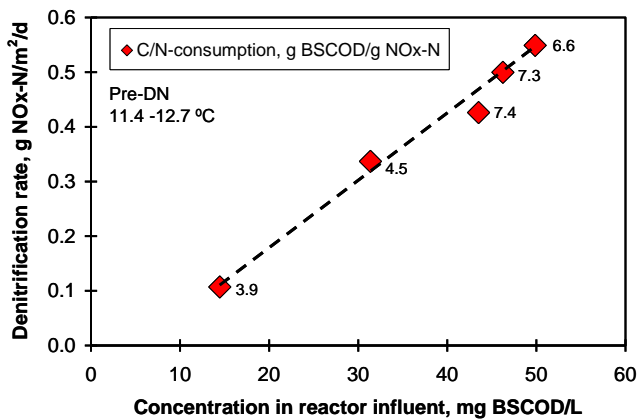


Figure 10. Denitrification rates and corresponding C/N-consumption ratios, versus BSCOD concentrations in to the pre-denitrification MBBR [14]

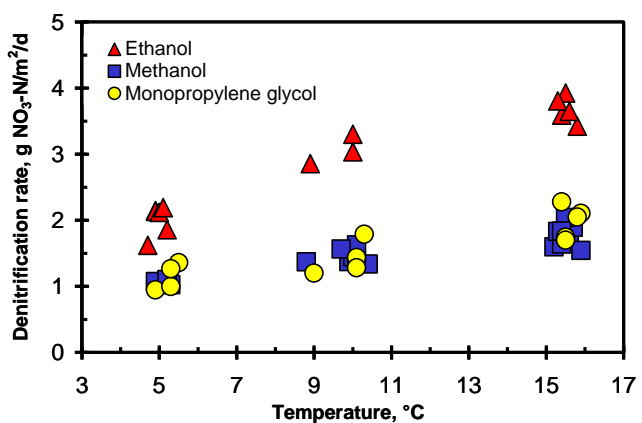


Figure 11. Denitrification rate versus temperature, obtained with various external carbon sources [15]

## 5.3 Post-denitrification.

Figure 11 shows post-denitrification rates in MBBRs for three different external carbon sources, when the nitrate concentration was not rate limiting [15]. 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, because only a few bacteria can utilize methanol and enrichment of this group of bacteria takes time. Due to the significantly shorter start-up time and higher denitrification rates, ethanol will be the best external carbon source for post-denitrification in geographic markets where ethanol is cost competitive.

All three external carbon sources showed a temperature coefficient of  $\theta = 1.07$ . C/N-ratios were also temperature dependent. This can be explained by higher specific sludge production (lower decay) at lower temperatures. The necessary C/N-ratio increased with an average of 45 % when the temperature was reduced from 15 to  $5 \text{ }^\circ\text{C}$ . For 95 % removal of nitrate, ethanol had the lowest carbon demand, with a necessary C/N-ratio of  $3.9 \text{ g COD}_{\text{added}}/\text{g NO}_3\text{-N}_{\text{equivalents}}$  at  $10 \text{ }^\circ\text{C}$ . MPG had the highest necessary C/N-ratio at  $5.6 \text{ g COD}_{\text{added}}/\text{g NO}_3\text{-N}_{\text{equivalents}}$  at  $10 \text{ }^\circ\text{C}$  [15].

Rusten and Ødegaard [12] reported very good and stable results from 4 Norwegian combined pre- and post-DN plants. For instance, the plant of the Olympic city Lillehammer, demonstrated in 2005 yearly average effluent concentrations and removals of 2.2 mg BOD<sub>5</sub>/l (99 %), 2.9 mg total N/l (92 %) and 0.12 mg total P/l (98 %). There are also several applications of post-denitrification with a pure MBBR (with carbon source addition) after a nitrifying activated sludge plant (Figure 4j) for instance the Sjölanda plant in Malmö, Sweden that reported very good and stable results [16].

#### 5.4 N-removal in IFAS processes

There are a number of process solutions for N-removal in IFAS process [3] One typical example is show in Figure 4m where nitrification is enhanced by the presence of the biofilm on the carriers while denitrification takes place in the pre-DN tank, with suspended biomass only. The anaerobic and anoxic reactors are most commonly designed without carriers, but there is an increasing interest in using carriers also in stirred reactors to reduce footprint even more. Since application of carriers means that the  $SRT_{\text{aerobic}}$  will be lower, more carbon will be available in the anaerobic and anoxic reactors which leading to higher reaction rates.

Another alternative, when the carbon source available in the raw water is not sufficient, is the use of a post-DN reactor which could be based on suspended biomass of an IFAS process alone. The design of DN-reactors in IFAS systems is equal to the one on AS-systems and based on denitrification rate, the DN-rate being depending on CN-ratio but typically being around 3 g NO<sub>x</sub>-N/kg MLSS·h when carbon is not limiting the process. For more information on IFAS, see [3].

### 6. REMOVAL OF ORGANIC MICROPOLLUTANTS

Removal of seven active pharmaceutical substances (ibuprofen, ketoprofen, naproxen, diclofenac, clofibric acid, mefenamic acid, and gemfibrozil) was assessed by batch experiments, with suspended biofilm carriers and activated sludge from several full-scale wastewater treatment plants [17]. Biofilm carriers from full-scale nitrifying wastewater treatment plants, demonstrated considerably higher removal rates per unit biomass (i.e. suspended solids for the sludges and attached solids for the carriers) of diclofenac, ketoprofen, gemfibrozil, clofibric acid and mefenamic acid compared to the sludges. Among the target pharmaceuticals, only ibuprofen and naproxen showed similar removal rates per unit biomass for the sludges and biofilm carriers.

The results demonstrated that moving bed biofilm carriers have a pharmaceutical reduction potential superior to activated sludge. Further research is, however, needed in order to find the underlying mechanism of the enhanced removal and to fully explore the potential.

### 7. BIOMASS SEPARATION AFTER MBBR'S

Contrary to activated sludge and IFAS systems, the biomass to be separated from pure biofilm systems (like MBBR) includes only the growth since there is no recycle. This means that the SS to be separated is in the order of 150 – 250 mg SS/l in normal municipal wastewater – around 20 times less than in activated sludge and IFAS systems. This makes it possible to combine the MBBR with any of the commonly used separation technologies, settling, flotation, micro-screening, media filtration and membrane filtration.

It will not be space enough available in this paper to discuss separation of biomass from MBBR's. This is, however, done in [18]. In that paper the following conclusions were drawn:

1. The particle size distribution of the biomass leaving a MBBR is characterized by being bimodal, i.e. a large mass fraction of the particles is present in relatively large particles or aggregates of particles (30 – 300 µm) but a high number of small particles (0,1 – 1 µm) is also present and even though these do not contribute much to the volume (or mass) of particles, they may represent various challenges in separation of the biomass. The easiest way to deal with these challenges is to coagulate the fine particles ahead of separation..

2. The particle size distribution is shifted towards larger particles when the hydraulic residence time is increasing and consequently organic loading is decreasing. The higher the organic loading, therefore, the better is the effect of pre-coagulation/flocculation.
3. A wide variety of separation processes may be used. In [18] results and experiences are reported from the use of conventional sedimentation, microsand ballasted lamella settling, flotation, fine media (sand) filtration, coarse media filtration and membrane (UF) filtration. The possibility to use this variety of separation processes gives MBBR processes great flexibility. It is a considerable advantage over activated sludge processes that in most cases will have to rely on sedimentation or membrane filtration (in MBR's) for biomass separation.
4. Especially when combined with high-rate separation processes, such as flotation (DAF), microsand ballasted lamella settling (ACTIFLO<sup>®</sup>) or microscreening (Discfilter), the foot-print of the total process will be very small.

In pure MBBR-processes, P-removal is normally carried out through precipitation/flocculation directly after the MBBR and before the final separation step. The experiences from Norwegian plants with precipitation [13] show very low effluent tot P-concentrations (normally < 0,3 mg P/l). The advantage of using pure MBBR and chemical precipitation is that one is free to choose the final separation method and several plants are using compact separation processes as flotation, lamella sedimentation or microsand ballasted lamella sedimentation (Actiflo). This results in extremely low foot-print for the whole plant.

## 8. DEAMMONIFICATION (NITRITATION AND ANAMMOX PROCESSES) IN MBBR'S

Deammonification processes in MBBR's can favorably and cost-effectively be carried out in high temperature, ammonia-rich wastewaters low in biodegradable COD concentrations (typical for sludge reject waters) by combining nitritation and anammox (anaerobic ammonium oxidation). The first stage of the deammonification process, equal amounts of ammonium and nitrite nitrogen are produced in the partial nitritation route to perform in the second stage, the anammox process, which involves simultaneous biochemical removal of ammonium and nitrite by anammox bacteria under oxygen limited conditions and results in the production of N<sub>2</sub> gas. The reader is referred to the numerous papers written on this subject.

It has been demonstrated in several research projects that the MBBR is particularly useful for deammonification. The suspended carriers make the process compact and less sensitive than by use of SBR-systems. The first full scale demonstration plant was built in Hattingen in Germany [19]. The demonstration project proved the MBBR deammonification process here to be cost-effective in comparison to conventional N-removal. There are now several plants in operation in Europe and USA. The AnoxKaldnes experiences with the anammox process (trade mark ANITA Mox) can be found in [20].

Recently an IFAS-based anammox process has been implemented [20], in which the aerobic ammonium-oxidizing bacteria are enriched in the suspended activated sludge and the anaerobic ammonium-oxidizing (Anammox) bacteria are enriched in the biofilm on suspended carriers. This partition of the two groups of bacteria involved in deammonification, ensures that the anammox bacteria are not substrate-limited by nitrite. The process has recently been implemented in the side-stream process at Sjölanda WWTP (Sweden) treating reject water from sludge digestion with a removal capacity exceeding 2 kg N/m<sup>3</sup>d [20].

## 8. CONCLUSIONS

Since its introduction around 25 years ago, the Moving Bed Biofilm Reactor (MBBR) processes have gained an extraordinary position in the market of wastewater treatment and are now in most applications where biological treatment is feasible. The reason for the popularity is the simplicity and the compactness of the process and the solid scientific basis on which the process was developed. Even though the MBBR was invented and developed as a pure biofilm reactor, there is an increasing use of the reactor principle in IFAS processes – especially for upgrading and capacity increase of existing activated sludge plants.

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