5. ATTACHED GROWTH REACTORS

5.1 Introduction

In attached growth systems the waste water is in contact with a microbial film, attached to the surface of a solid material/medium. The surface area for growth of the biofilm is increased by the use of a porous medium in the reactor. The biological reactions take place in the biofilm, while suspended bacteria are washed out of the systems.

When randomly packed reactors, are used and the waste water flows by gravity as a free surface stream, the reactor is called a *trickling filter.* The use of rotating discs, covered with biofilm, partially submerged in waste water is called a *rotating biological contactor (RBC)* process, where the biofilm development is controlled by the rate of rotation.

Other attached-culture systems are the *submerged filters with up-flow or down flow application, i.e. Up-flow Fixed Bed Reactors (UFBR)* and the *Fluidized Beds.* Thay may both have applications under certain conditions such as high-nutrient-containing waste water. Figure 5.1 shows some of the nitrifying attached growth units in use.

In the trickling filter, the medium is stationary and the waste water is passed over the biofilm in intermittent doses. In the RBC, the medium moves the biofilm alternately through water and air. Using the UFBR, the waste water is pumped up-flow through a fixed medium. The fluidized bed consists of spherical particles coated with a biofilm fluidized by up-flowing water. A segregation generally occurs; the apparent density of the particles decreases, as the thickness of the biofilm increases.

Continuous control of the biofilm is not possible with a stationary support medium. The filters therefore have to be backwashed in order to prevent clogging.

Experience has shown that many kinds of support material can be used, for example stones, gravel, sand, plastic, asbestos plates, wood, zeolites, and activated carbon particles.

In addition to the biological reactor, an attached growth system usually includes both primary and secondary clarification. Recirculation of sludge is normally not necessary in biofilm reactors, because the amount of biomass is huge compared with activated sludge systems.

The major work in an attached growth system is to establish a unit configuration, where oxygen can uniformly be supplied during the nitrifying process and

where water at the same time can pass through the support media without any limitations.

Many experiments have been carried out, assuming that the nitrifying rate per unit surface of biofilm is the same from one reactor to the other, independent of influent characteristics. Serious errors have, therefore, been made, for example with respect to scaling up. Results from pilot scale experiments with submerged filters cannot be scaled up on the assumption that the rate of substrate removal per unit m^3 is the same at full scale.

The processes for which the biofilm reactors have been used or proposed for use in waste water treatment are oxidation of organic matter, nitrification, denitrification or combinations of these.

5.2 The Biofilm

Nitrifier, denitrifier, oxidizer or a combination of these types of bacteria can attach themselves to different types of medium and grow into dense films of a viscous, gelatinous matrix called the biofilm. Waste water passes over this film in thin sheets, with dissolved organics, NH_4^+ or NO_3^- passing into the biofilm due to diffusion gradients within the film. Suspended particles and colloids cannot penetrate the surface of the biofilm, but will be decomposed on the surface of the biofilm into soluble products. Oxygen from the waste water and from air in the void spaces of the medium, provides oxygen for the aerobic reactions at the surface of the biofilm. Figure 5.3 show a diagrammatic representation of a biofilm with involved processes.

Waste products from the metabolic processes diffuse outward and are carried away by the water or air. Growth of the biofilm is restricted to the outward direction from the solid surface. As the film grows thicker (see Fig. 5.2), concentration gradients of both oxygen and nutrient develop. Eventually, when the biofilm is of an appropriate size, both anaerobic and endogenous metabolism occur in the interface of the biofilm.

In a well developed biofilm, the attachment mechanism to the solid medium is weakened, and the shearing action of the waste water flowing across the film pulls it from its attachment and washes it away. This process is called *sloughing,* and is a function of both the hydraulic and the organic loading rates. But the biofilm is quickly re-established, and, therefore, sloughing is a beneficial mechanism for development of new biofilm.

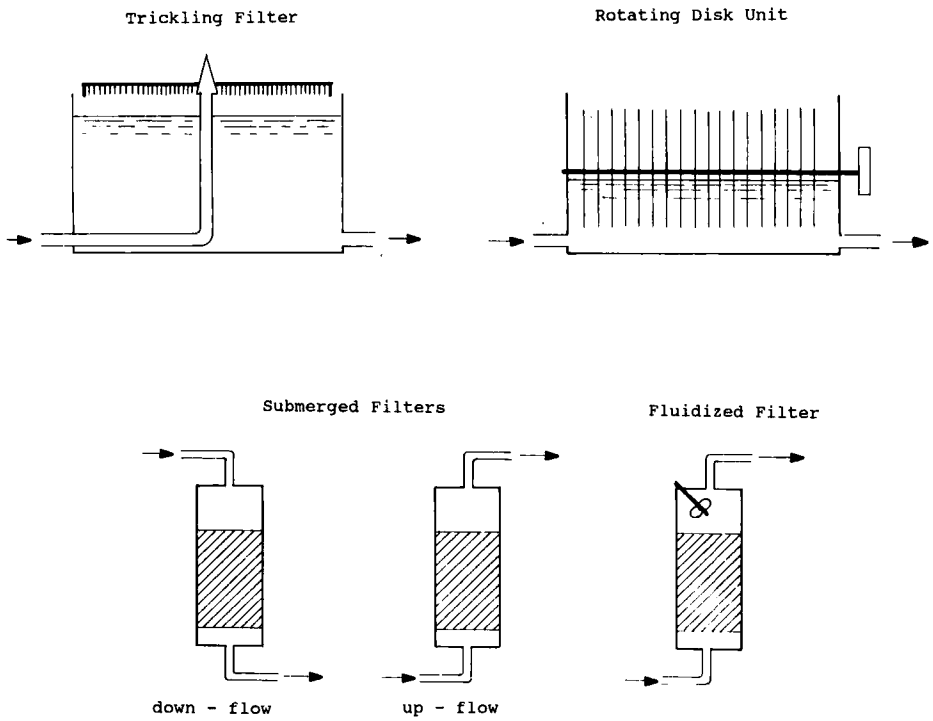


Figure 5.1 Biofilm reactors used in waste water treatment. a) Trickling filter, b) Rotating Biological Conductor (RBC), c) Submerged filters with down-flow or up-flow application, and d) Fluidized filters.

5.3 The Development of a Bacterial Biofilm

The successive steps of the development of an aerobic biofilm can be described as follows (Elmaleh and Grasmick 1985):

step 1 - The biofilm is composed of a few aerobic bacteria included in a gelatinous matrix, i.e the density is low.

step 2 - Aerobic micro-organisms grow rapidly, and the density is an increasing function of the thickness.

step 3 - As oxygen depletion begins to occur in the biofilm, an anaerobic zone appears near the solid material.

step 4 - Anaerobic and facultative bacteria grow near the solid material as aerobes decay, causing decreasing density.

step 5 - An equilibrium between the anaerobes and aerobes is reached, which means that the density is stabilized.

step 6 - Now the equilibrium is maintained, until the substrate concentration is exhausted in the deeper zone, the anaerobe bacteria will begin to decay, and parts of the biofilm will finally slough away.

step 7 - The newly developed space will be used by new aerobic bacteria which will start all over again and build new biofilm.

The rate of nutrient removal in attached-growth systems depends on the flow rate of the waste water, the organic loading rate, rates of diffusivity of nutrients into the biofilm, and temperature. The depth of penetration of both oxygen and nutrients is increased at higher loading rates. Oxygen diffusivity is usually the limiting factor. Aerobic zones of the biofilm are usually limited to a depth of 0,1 to 0,2 mm, the remaining thickness of the biofilm being anaerobic (see Fig. 5.4). Depending on hydrodynamic conditions, Atkinson and Fowler (1974), found values between 0,07 and 4 mm in thickness. When the biofilm is mechanically or hydraulically controlled, its thickness does not exceed 0,2 mm, which is the maximum depth for having a full aerobic biofilm (Grasmick 1982).

Arvin and Harremoes (1990) reported that the thickness of the biofilm is controlled by the following factors:

1. Growth of active biomass as a result of influx of the substrate.
2. Decay of active biomass.

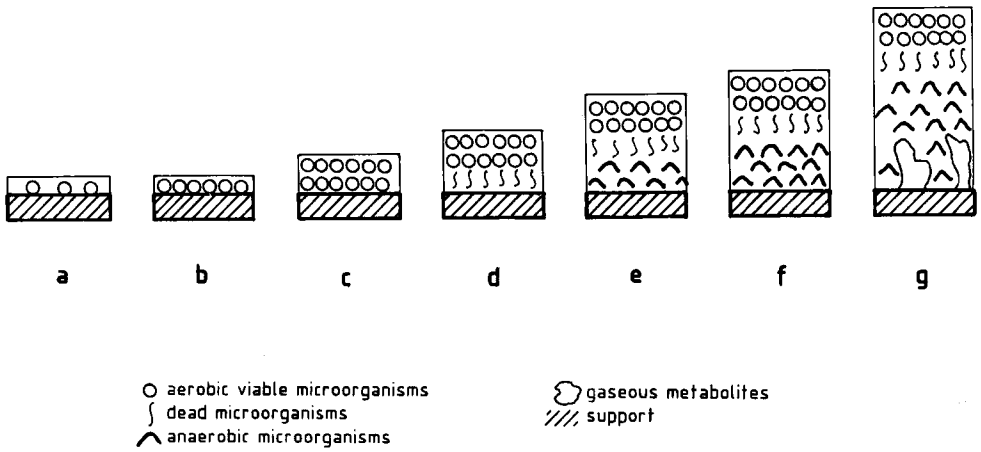


Figure 5.2 The different steps in developing a biofilm, shown as transients of a biofilm.

3. Accumulation of inert organic material from the decay of active bacteria.
4. Accumulation of polymers from the metabolism of the substrate.
5. Deposition/flocculation of suspended particles from the bulk liquid.
6. Erosion of small particles from the surface of the biofilm.
7. Sloughing of large fractions of the biofilm.

At present our ability to predict the thickness of a biofilm is relatively low.

It is also difficult to define and measure the thickness of a biofilm. Experimental measurements can be made directly, or can involve a procedure called congelation of the whole reacting medium. Accuracy in the values of a thickness less than 100 μm is 20%, but on values of about 2 mm, the discrepancy can reach 300 % (Grasmick 1982).

The number of variables affecting the growth of biomass, and subsequently the rate of substrate utilization, makes it difficult to describe the systems mathematically.

Masuda *et al.* (1987) reported that oxidizing, nitrifying and denitrifying bacteria can exist almost uniformly in the entire biofilm. Oxidation of organics, nitrification and denitrification occur in the same biofilm. Probably the denitrifying bacteria exist in the most anoxic areas, in the deeper layer of the biofilm.

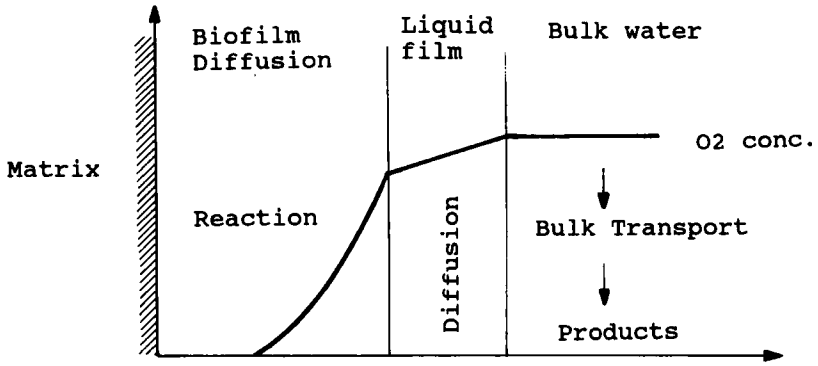


Figure 5.3 Diagrammatic presentation of a biofilm with involved processes.

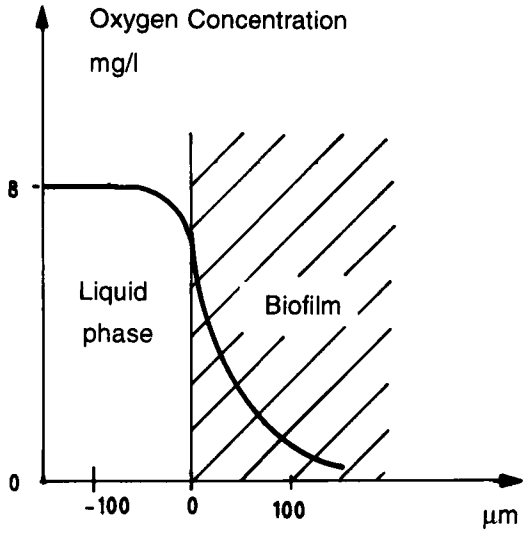


Figure 5.4 Oxygen concentration profile showing that at thickness over 100 µm the biofilm is anaerobic.

5.4 Modelling the Transport and Reactions within a Biofilm

In spite of the heterogeneity of the biofilm, it is assumed in most of the models that the substrate is transported by molecular diffusion and, therefore, that an effective diffusivity is a characteristic constant of the system. (Atkinson and Fowler 1974; Harremoes and Riemer 1975; Harremoes, 1975, 1976, 1978,; Arvin and Harremoes 1990; La Motta 1976 ; Williamson and McCarty 1976 ; Grasmick *et al.*, 1979, 1981; Rittman and McCarty 1981).

The rate of reaction in a biofilm is based on the concept of the limiting substrate. If the waste water is aerobic, the limiting substrate will consist of oxygen, organic carbon and/or ammonia.

The intrinsic reaction rate of a limiting substrate can be described, depending on the authors, as a Monod-type, first, or zero order equation.

In waste water treatment, it has been shown that the best approximation is the zero order (La Motta 1976; Riemer and Harremoes 1978; Grasmick 1982). Depending on the penetration into the biofilm, the apparent reaction rate will be zero order kinetics for full penetration, and half-order kinetics for partial penetration (Harremoes 1978). Table 5.1 presents values for the biofilm kinetics.

Arvin and Harremoes (1990) proposed that the basic feature in the biofilm model is the kinetics of the processes performed by the active bacteria in the film. This approach can be used for describing processes other than the nitrification and denitrification such as aerobic mineralization, sulphate reduction, fermentation, or methanogenesis.

Bacterial activity:

If μ_{max} and Y can be considered universal, then the bacterial activity can be described by transforming the monod equation (3.11) to:

$$\frac{dS_n}{dt} = k_x * \frac{S_n}{S_n + K_s} * X_B \qquad (5.1)$$

where:

k_x = the maximum soluble substrate (zero order) utilization rate.

The kinetic characteristic of the biofilm reactor is:

1. The diffusion resistance to the movement of the substrate into the biofilm.
2. The products developed in the biofilm.

There is a difference in the performance of a reactor depending upon whether a substrate can penetrate the biofilm fully or partly (Harremoes 1978).

The diffusion resistance:

The diffusion resistance affects both the removal rate and the order of reaction. Arvin and Harremoes (1990) explain that:

- A first order reaction in the interior of the biofilm is converted into a first order bulk reaction at a reduced rate.

- A zero order reaction in the interior of the biofilm remains a zero order bulk reaction, if the biofilm is fully penetrated, but is converted into a half-order reaction, if the film is only partly penetrated.

Assuming that the reaction rate for the nitrification and denitrification is zero order, the following kinetic equations can be used:

The biofilm where the substrate penetrates fully:

$$dS/dt = k_B * L \qquad \text{(zero order)} \qquad (5.2)$$

The biofilm where the substrate penetrates partly:

$$dS/dt = (2 * D * k_B * S)^{1/2} \qquad \text{(half order)} \qquad (5.3)$$

or:

$$dS/dt = K_{1/2order} * S^{1/2} \qquad \text{(half order)} \qquad (5.4)$$

where:

dS/dt = the reaction rate per unit surface of the biofilm.

k_B = the intrinsic reaction rate per unit volume of the biofilm.

L = the thickness of the biofilm.

D = the diffusion coefficient.

S = the substrate concentration, which can be forms of nitrogen carbon or oxygen.

The transition from zero order to half order kinetic is governed by the relative penetration of the substrate and the rate order can be determined using the following equation.

$$\beta = \left(\frac{2 \cdot D \cdot S_n}{K_B \cdot L^2}\right)^{1/2} \qquad (5.5)$$

Where ß > 1 than the biofilm is fully penetrated by the substrate.
and where ß < 1 it is partly penetrated.

The biofilm in a trickling filter treating domestic waste water will usually follow a half order kinetic.

The appearance of non-diffusible matter in the biofilm reactor.

The theories for substrate removal in the biofilm suffer from the fact that only very little is known about how the biofilm affects non-diffusible matter (Levine 1985; and Ødegaard 1987).

The two main questions concerning non-diffusible matter are:
- How can particulate matter be attached to the biofim surface?
- What is the mechanism for the extracellular degradation of the attached particulate matter?

The removal of the particulate matter depends on the following aspects (Arvin and Harremoes 1990):
1. The size and the chemical charge of the particulate.
2. The size, shape and chemical composition of the support media.
3. The surface of the biofilm.
4. The waste water flow through the biofilter.

Table 5.1 Values for the biofilm kinetic using zero order or an apparent half-order kinetic coefficient.

Pollution	*Limiting substrate*	*Conditions*	*Temp. °C*	*Intrinsic zero order rate per unit biofilm* ($kg/m^3\ S\ 10^{-3}$)	*Apparent half-order coefficient per unit biofilm volume* ($kg^{½}m^{-½}s^{-1}\ 10^{-5}$	*Apparent half-order coefficient per unit reactor volume* ($kg^{½}m^{-3/2}s^{-1}\ 10^{-3}$)	*Reference*
Milk	0_2	fixed bed	20		0,12		Grasmick *et al.* (1980)
Beef extract	0_2	fixed bed	20		0,32		Grasmick *et al.* (1980)
Methanol	methanol	rotating reactor	22		0,16 - 0,19	0,38 - 1,58	Jansen and Kristensen (1980)
Milk	TOC	rotating	20	0,27 - 0,59	0,083 - 0,18		Grasmick (1982)

From: Grasmick (1985)

The transport of non-diffusible matter into the biofilm is very slow, compared to the transport of diffusible matter.

Degradation of particulate matter outside the biofilm is conducted by extracellular enzymes, released into the waste water by the biofilm, or enzymes working on the membrane of the biofilm. This conversion of particulate matter into a soluble product, that is able to diffuse into the biofilm, may be by a special mechanism, which facilitates the penetration of the biofilm.

The liquid film diffusion.

Before any reaction takes place inside the biofilm, the substrate needs to be transferred from the bulk liquid to the solid phase. The existence of a mass transfer resistance in liquid-biofilm har been demonstrated. (La Motta 1976; Grasmick 1982)

The flux, J, of substrate into the biofilm follows Fick´s first law.

$$J = - D * dS/dx = D(S - S_S)/L_B \qquad (5.6)$$

where: S and S_S are the bulk and interfacial concentrations; L_B is the thickness of the boundary diffusion layer. L_B can be determined using a method described by Bouwer and McCarty (1985).

In practice, oxygen is always the rate limiting factor rather than the ammonia concentration, because the critical ratio between the two concentrations for performing nitrification is of the order of 0.3-0.4 mg NH_3 per mg O_2 (Gönec 1982). If the concentration of O_2 is, for example, 4 mg/l, then the concentration of NH_3 has to be smaller than 1.3 mg/l to be limiting. Table 5.2 presents values for the effective diffusivity in pure water and in biofilms.

Bacterial population dynamics in the biofilm.

If a biofilm has to oxidize carbon matter and nitrify simultaneously, the two electron donors will compete for the same electron acceptor, oxygen. Both processes will take place in the aerobic zone of the biofilm. The relative use of the limited electron

acceptor resource is determined by the population dynamic of the heterotroph and nitrifiers in the biofilm. In the aerobic zone, both types of bacteria will grow, but at different rates, determined by the available substrate and the growth rate of the different species of bacteria.

At a particular ratio of organic matter to ammonia, the nitrifiers will be outgrown by the heterotroph, and no nitrification will occur. This effect is similar to the wash-out of the nitrifiers in an activated sludge plant at too low a sludge age.
If the biofilm is not fully penetrated by oxygen (Fig. 5.4), it will be divided into an aerobic part adjacent to the bulk liquid and an underlying anaerobic part. Applying clinoptilolite as a support medium in an upflow fixed bed reactor (UFBR) (See section 5.8.1) it was possible to obtain simultaneous nitrification and denitrification (Halling-Sørensen and Hjuler 1992).

The biofilm composition is always a "mirror" of the composition of the waste water applied to the treatment system. Different zones may be developed as a function of the loading of substrate to the biofilm. According to Kinner (1983) (see Table 5.3) the most varied biofilm induced by a heavily loaded waste water can have four different layers, as follows:

1. An outer layer with heterotrophic oxidation of organic carbons, nitrification and denitrification and sulfide oxidation.
2. A microaerophilic layer with denitrification and fermentation.
3. An anaerobic layer with sulphate respiration and fermentation activity.
4. An anaerobic layer adjacent to the support material with methanogenesis and fermentation.

If the waste water becomes less heavily loaded, or possibly acquires a different composition, the biofilm will be built up of layers 1 and 2 only, or consists of layer 1 only.

At steady-state the fraction of organisms f_i, in one of the layers is given, indicating a balance between growth and decay, as:

$$r_a^* Y_i = b_i \cdot X_a * f_i \cdot I \qquad (5.7)$$

where:

r_a = the removal rate per unit surface area of substrate utilized by organism group i.

Y_i = the yield constant for organism group i.

b_i = the decay constant for the organism group i.

X_a = biomass of the whole biofilm.

f_i = the fraction of organism group i.

l = the length of the zone with organism group i.

The product $X_a * f_i * l = r_a * Y_i/b$ is derived from equation (5.7) and reflects the steady-state active biomass of organism group i per unit area of biofilm.

The pH effect in the biofilm.

Nitrification is an acidity-producing process, while denitrification is an alkalinity producing process as outlined in sections (3.4) and (4.4).
In bulk waters of low alkalinity the result can, therefore, be a significant drop in pH in the biofilm conducting nitrification. This can lead to an inhibition of the nitrification because of too low a pH.

In the denitrifying biofilm, the pH can be increased in the rear of the film to an extent where precipitation of phosphate can occur (Arvin and Christensen 1979). There is no mention in the literature of the pH in a biofilm conducting simultaneous nitrification and denitrification.

5.5 A Massbalance Equation for a Biofilm Plant

A mass-balance equation for a biofilm plant without recirculation can be outlined as follows:

$$Q_{inf} * C_{inf} - r_{x,s} * V * X_B = Q_{eff} * C_{eff} \qquad (5.8)$$

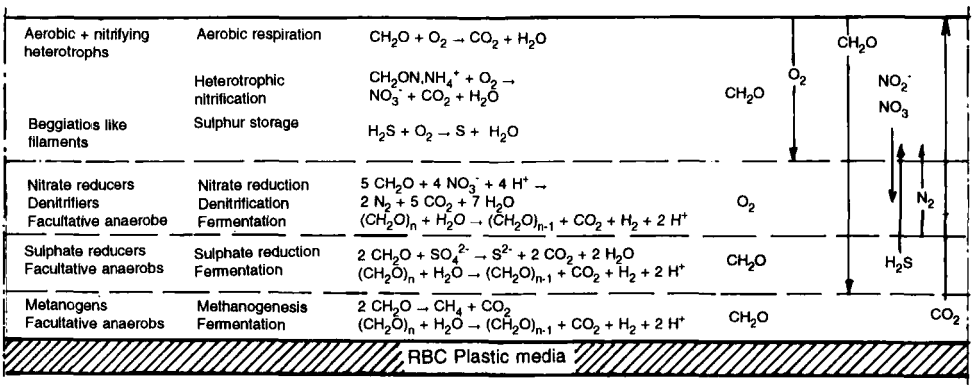
Table 5.2. Values for the effective diffusivity in pure water and in biofilms.

Pollution	*Substrate*	*Conditions*	*Temp.* (^oC)	*Effective diffusivity* 10^{-10} (m^2/s)	*Reference*
Pure water	O_2		20	15	
Milk	O_2	fixed bed	20	3 - 9	Grasmick *et al.* (1980)
Beef extract	O_2	fixed bed	20	8 - 10	Grasmick *et al.* (1980)
Wastewater	O_2	rotating cylinder	20	12 - 17	Tomlinson and Snaddon (1966)
Glucose	O_2		20	4 - 20	Matson and Charaklis (1976)
Nitrogen compounds	O_2	pure culture *Nitrobacter* *Nitrobacter + Nitrosomonas*		25	Williamson and McCarty (1976)
Methanol	methanol	rotating reactor	22	20 - 50	Jansen and Kristensen (1980)
Glucose	glucose	rotating reactor	22	1 - 5	La Motta (1976)
Milk	TOC	rotating reactor	25	6 - 50	Grasmick (1982)

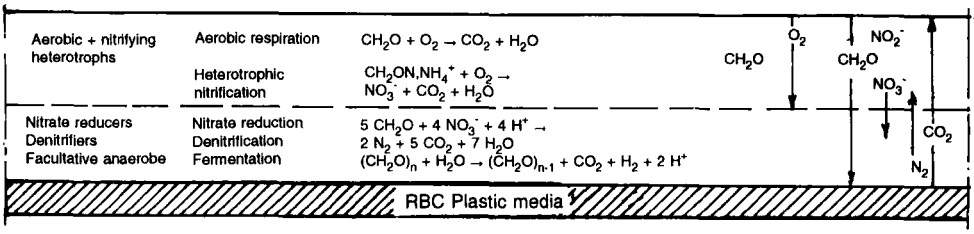
From: Grasmick (1985).

Table 5.3 The different chemical processes in the four different layers (Source Kinner 1983).

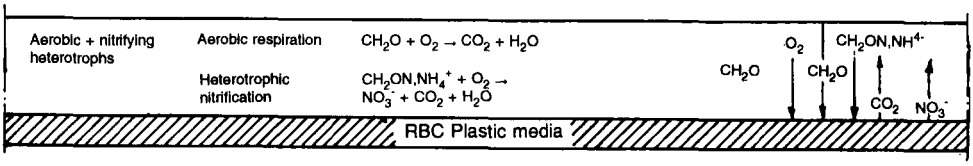
Predominant Bacteria	Metabolic process	Reactions	Limiting substrate	Reactants + products
Aerobic + nitrifying heterotrophs	Aerobic respiration	$CH_2O + O_2 \rightarrow CO_2 + H_2O$		
	Heterotrophic nitrification	$CH_2ON,NH_4^+ + O_2 \rightarrow NO_3^- + CO_2 + H_2O$	CH_2O	
Beggiatios like filaments	Sulphur storage	$H_2S + O_2 \rightarrow S + H_2O$		
Nitrate reducers Denitrifiers Facultative anaerobe	Nitrate reduction Denitrification Fermentation	$5\ CH_2O + 4\ NO_3^- + 4\ H^+ \rightarrow 2\ N_2 + 5\ CO_2 + 7\ H_2O$ $(CH_2O)_n + H_2O \rightarrow (CH_2O)_{n-1} + CO_2 + H_2 + 2\ H^+$	O_2	
Sulphate reducers Facultative anaerobs	Sulphate reduction Fermentation	$2\ CH_2O + SO_4^{2-} \rightarrow S^{2-} + 2\ CO_2 + 2\ H_2O$ $(CH_2O)_n + H_2O \rightarrow (CH_2O)_{n-1} + CO_2 + H_2 + 2\ H^+$	CH_2O	
Metanogens Facultative anaerobs	Methanogenesis Fermentation	$2\ CH_2O \rightarrow CH_4 + CO_2$ $(CH_2O)_n + H_2O \rightarrow (CH_2O)_{n-1} + CO_2 + H_2 + 2\ H^+$	CH_2O	



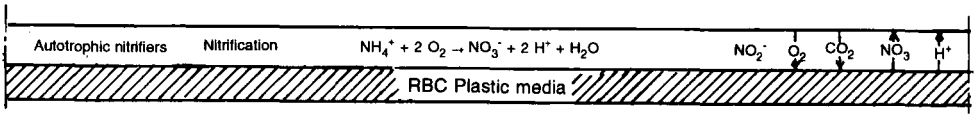
Aerobic + nitrifying heterotrophs	Aerobic respiration	$CH_2O + O_2 \rightarrow CO_2 + H_2O$		
	Heterotrophic nitrification	$CH_2ON,NH_4^+ + O_2 \rightarrow NO_3^- + CO_2 + H_2O$	CH_2O	
Nitrate reducers Denitrifiers Facultative anaerobe	Nitrate reduction Denitrification Fermentation	$5\ CH_2O + 4\ NO_3^- + 4\ H^+ \rightarrow 2\ N_2 + 5\ CO_2 + 7\ H_2O$ $(CH_2O)_n + H_2O \rightarrow (CH_2O)_{n-1} + CO_2 + H_2 + 2\ H^+$		



Aerobic + nitrifying heterotrophs	Aerobic respiration	$CH_2O + O_2 \rightarrow CO_2 + H_2O$		
	Heterotrophic nitrification	$CH_2ON,NH_4^+ + O_2 \rightarrow NO_3^- + CO_2 + H_2O$	CH_2O	



Autotrophic nitrifiers	Nitrification	$NH_4^+ + 2\ O_2 \rightarrow NO_3^- + 2\ H^+ + H_2O$		



where:

Q_{inf} = the influent flow in l/s.
C_{inf} = the influent substrate concentration in mg/l.
$r_{x,s}$ = the process reaction rate in kg substrate per kg biomass /m^3 • day.
V = the volume of the reactor in m^3.
Q_{eff} = the effluent flow in l/s.
C_{eff} = the effluent substrate concentration in mg/l.

Because it is difficult to quantify or estimate the biomass concentration X_B in a biofilm plant, it has been suggested in the literature that the term $r_{x,s} * V * X_B$ be changed to a term taking into account the volume or the area of growing bacteria.

In the literature the following removal terms have often been used.

$$r_{v,s} * V = r_{a,s} * A \qquad (5.9)$$

where

$r_{v,s}$ = the amount of substrate removal per m^3 per day, expressed as a volumetric reaction rate.
$r_{a,s}$ = the amount of substrate removal per m^2 per day (a term often used for RBC plants), expressed as a surface reaction rate.

Using one of the above terms avoids the necessity of knowing the concentration of the biomass X_B, but can simply relate the reaction rate to the present biomass X_B, under steady-state conditions, of a specific area or volume. Table 5.4 show the different units which indicate the substrate (nitrogen) removal rate.

Depending upon whether the substrate can fully penetrate the biofilm or not, the kinetic will follow zero order or ½ order kinetics. Equation (5.2) or (5.3) must then be introduced as the kinetic rate.

Applying zero order kinetics the mass-balance for the biofilm plant will be:

$$Q_{inf.} * C_{inf} - k_B * L * A = Q_{eff} * C_{eff} \qquad (5.10)$$

and for ½ order kinetics:

$$Q_{inf} \cdot C_{inf} - (2 \cdot D \cdot k_B * S)^{½} \cdot A = Q_{eff} * C_{eff} \quad (5.11)$$

where:

$$k_B = \frac{\mu_{max}}{Y_{max}} * X_B \quad (5.12)$$

The use of this equation requires also the knowledge of the biomass concentration X_B, therefore k_B is usually an experimentally found constant observed in a special set of conditions , which also makes possible the estimation of biomass.

Table 5.4 The different units which indicate the substrate (nitrogen) removal rate.

nitrification rate	term	unit
as biomass	$r_{x,s}$	kg N/kg biomass per m^3*d
as volumetric rate	$r_{v,s}$	kg N/ m^3 • d.
as surface rate	$r_{a,s}$	g N/m^2 • d.

If recirculation of the waste water is used in a biofilm plant, the following equation of biomass-balance can be used:

$$Q_{inf} \cdot X_{inf} + r_{x,x} \cdot V \cdot X_B = Q_{eff} * X_{eff} + Q_{sedimentation} \cdot X_{sedimentation} \quad (5.13)$$

where:

$r_{x,x}$ = the rate of biomass activity per unit of biomass.

The recirculation of water is used to ensure a constant water passage through the support material.

5.6 The Nitrifying Trickling Filters (NTF)

The trickling filter was introduced at the end of the last century, and is one of the first methods used for the removal of nitrogen from waste water. At first the trickling filters were only a primitive form of land treatment, where sewage was spread at intervals over sandy ground, allowing the sand to dry between each spreading.

Later, the sand was replaced by stones, but the operational procedures remained the same, with down flow application of the waste water. Trickling filters were first introduced in Great Britain. They were circular and supplied with a rotating distributor at the top of the filter, and measures were taken to secure accurate aeration.

An underdrain system is designed to carry away the treated waste water and the sloughed biomass. Several operational modes are available for trickling filters. Standard-rate filters have low hydraulic loading and do not include provision for recycling. High-rate filters maintain high hydraulic loading by recirculating portions of the effluent. Filters placed in series increase the effective depth, thus increasing the efficiency. A great number of possibilities exists for different flow regimes.

Figure 5.1 shows the basic design of a trickling filter. Modern trickling filters (sometimes called bio-towers) are packed with different types of plastic media , which allow the filters to be more efficient and also able to treat highly polluted industrial waste water. Plastic media consists of either vertical-flow or cross-flow substances.

The advantages of using plastic media are a high specific surface in addition to high void fraction and low weight, reducing the construction costs and high stability to shock-loads; this again, allows the construction and application of smaller trickling filters.

Process improvements of trickling filters, using bioflocculation components as a post-treatment following biological treatments, have produced higher quality effluents than previously. This improvement makes the trickling filter compatible with the activated sludge systems, and can produce a high quality effluent, comparable with that produced using the activated sludge process.

Trickling filters used for nitrification, are employed either to nitrify secondary effluent, or to combine organic removal with nitrification of the primary effluent. Depending upon the composition of the influent waste water, a different bacterial population will be developed.

Reduced removal efficiency of nitrogen can occur in a trickling filter for a

variety of reasons. Most important among these is the removal of biofilm by predators (worms and fly larvae), and incomplete wetting of the media. Depressed nitrification rates can, however, also result from competition between nitrifiers and heterotrophs for dissolved oxygen.

Parker and Richards (1986) determined that a soluble BOD_5 concentration above 20 mg/l was sufficient to prevent nitrification in a Nitrifiying Trickling Filter (NTF).

A typical removal rate for a conservatively loaded NTF is between 0.20 and 0.39 g N / m^2 • d. With the development of the Biofilm-Controlled-Nitrifying-Trickling-Filter (BCNTF) in the late 1980´s the reaction rate in these filters has been increased significantly.

As a comparison, the normal removal of organics with a trickling filter is in the order of 2-3 kg BOD /m^3 * d. But extreme removal rates of up to 10-20 kg BOD/m^3 • d have been reported (Audoin *et al.* 1971).

5.6.1 The Performance of Trickling Filters

The performance of trickling filters is affected by many factors, such as the hydraulic, organic and nitrogen loadings, characteristics of the influent waste water, its temperature, distribution, distribution frequency, and composition. Other factors are concentration of bacteria and macroorganisms, oxygen supply, the volume and geometric shape of the filter medium, and depth of filter.

The trickling filter medium.

The requirements for the trickling filter medium are to present a large surface for the bacterial population to grow on, and provide a large enough empty space to secure aeration.

Only by applying plastic-medium is it possible to satisfy these two requirements simultaneously, because of its low weight. The geometric shape of the packing is also of importance, not only in relation to the maximum available surface for biological growth, but for its influence on the hydrodynamics of the filter; this again, influences the retention time in the filter. Table 5.5 show propoerties of the trickling filter media.

The influence of variation of nitrogen and organic load on a trickling filter.

Significant load variations of ammonium-nitrogen are normal during the course of the day. The nitrifying trickling filter must, therefore, be designed to be able to treat

peak loads, otherwise an ammonium breakthrough must be expected in the effluent. Trickling filters are specially sensitive to ammonium-nitrogen in the effluent, because of the very short hydraulic residence time coupled with the down flow system.

As the lower parts of the trickling filter obtain ammonium for only a few hours each day, it may require a long time to establish a fully developed biofilm at the bottom of the reactor. During periods of warm weather only the upper part of the trickling filter may be active, due to the higher reaction rate per unit area. Sudden temperature drops may, therefore, cause an ammonium breakthrough since the biofilm may not be developed at the bottom of the reactor. To avoid this, Boller and Gujer (1986) suggested that two trickling filters should operate in series. Their sequence should be reversed once every week to obtain a homogeneous distribution of biomass throughout the reactors.

Easily degradable organics will always be preferred by the bacteria, and the capacity of the trickling filter, treating waste water with such a composition will, therefore, be high.

Several investigators have shown that the removal per volume filter at moderate loads can be described as a linear function of the load per volume.

Thus, the performance of the trickling filter evaluated for removal of organics and nitrogen would depend on the amount of total organic load applied to the filter, rather than its concentration or the flow rate.

The oxygen transfer in a nitrifying biofilm in an NTR plant.

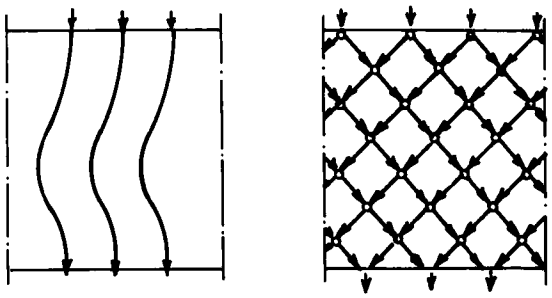
By calculating the total oxygen transfer to a nitrifying biofilm, the maximum removal of nitrogen can be determined for different types of plastic media as shown in Fig. 5.5.

Cross-flow media are predicted to produce a higher nitrification rate than vertical flow media of identical surface area, because of fluid disruption at mixing points in the cross-flow media (Parker *et al.* 1989).

Table 5.5 Properties of trickling filter media.

Medium	*Nominal size mm*	*Mass/unit volume kg/m^3*	*Specific surface area m^2/m^3*	*Void space per cent*
River rock				
Small	25-65	1250-1450	55-70	40-50
Large	100-120	800-1000	40-50	50-60
Blast-furnace slag				
Small	50-80	900-1200	55-70	40-50
Large	75-125	800-1000	45-60	50-60
Plastic				
Conventional	600 x 600 x 1200	30-100	80-100	94-97
High-specific surface	600 x 600 x 1200	30-100	100-200	94-97
Redwood	1200 x 1200 x 500	150-175	40-50	70-80

From: Metcalf and Eddy (1991)



A. Vertical Media B. Cross Flow Media

Figure 5.5 Downward flow pattern in vertical and cross-flow media. After Parker and Merrill (1984).

The hydraulic load.

The hydraulic load is a factor affecting the retention time, which is considered one of the most important factors influencing the performance of the trickling filter.

A high loading rate results in rapid growth of the biomass, and excessive growth may result in the plugging of pores and subsequent flooding of portions of the medium. Increasing the hydraulic loading rate, increases sloughing and helps to keep the bed open.

One of the limitations is the incomplete wetting of the packing at low loads and percolation at high loads. But other factors can also enhance or slow down the performance of the trickling. If diffusion in the liquid film somewhere in the filter controls the reaction rate, an increased flow rate will increase the reaction.

For plastic-packed trickling filters with a high specific surface, this effect will most likely influence the reaction rate at even normal loadings (for NTF 0.20-0.40 g N /m^2 • day). In the literature the influence of the hydraulic load on the wetted area of the filter has been suggested to be an important factor in this performance. The wetted area might vary with depth, because of an uneven distribution of biomass in the filter.

The relation of the depth and retention time for the trickling filter.

The retention time is considered to be directly proportional to depth, and therefore using the retention time automatically includes depth. Depth is normally used as a measure of total available biomass, while retention time is a measure of time of contact between organisms and substrate. The following equation is generally accepted in calculating the retention time in a trickling filter:

$$t = constant * \frac{H}{Q^n} \tag{5.14}$$

where:

Q = the flow in l/hour

H = Hydraulic retention time

n = no of recyclings

t = time

As the removal of organic pollutants from liquids takes place mainly through adsorption and absorption, the time of contact between organisms and substrate is considerably longer than the retention time of the liquid.

The removal per unit of biofilm surface sometimes increases at higher flow, which is contrary to the theory, used in most models, that only the flow influences the retention time. The same is observed when applying the SND mechanism, as shown in Section 5.8.1.

The influence of temperature on the performance of the trickling filter.

Very little information is available on the relationship between nitrification rate and temperature (see Fig. 5.6), because most studies of combined carbon oxidation and nitrification trickling filters have been carried out above 16 °C.

Data for lower temperatures can hardly be obtained because of lack of investigation, and nitrification data obtained at higher temperatures cannot be easily converted to represent performance at ten degrees lower for example, because changes in the nitrification rate will reflect changes in the relative growth rates of two

different types of organisms in the treatment plant. No information is available on the influence of temperature on the competition between nitrifiers and heterotrophs.

The interfaces of biomass, water and air also makes the trickling filters extremely sensitive to variations in temperature. Effluent quality is thus likely to show drastic seasonal changes, due primarily to changes in ambient air temperatures. The temperature of the waste water and air also determine the direction of air flow through the medium. Cool water absorbs heat from the air, and the cooled air sinks towards the bottom of the filter in a co-current fashion with the water.

Conversely, warm water heats the air, causing it to rise through the underdrain and up through the medium. Extreme cold may result in icing and destruction of the biofilm.

The effect of recirculation in a trickling filter.

Recirculation is done to ensure that a constant volume of waste water enters the plant, to dilute a strong or toxic waste, to increase the surface load, or to prolong the retention time, so that each "substrate particle" passes through the filter more than once.

In several investigations, recirculation has been proved to enhance the efficiency of the plant. The most important factor in determining the extent of recirculation is to identify which factor controls the reaction rate, because the effect of recirculation might change the control of the reaction rate from one factor to the other, for example a process controlled by liquid diffusion might become controlled by biofilm diffusion or metabolic activity.

The influence of substrate composition on performance of the trickling filter.

With a complex substrate such as domestic sewage, there will most likely exist different organic and nitrogen compounds which can be difficult to break down. Such differences in composition of the waste water are very important to take into account in the calculation of possible efficiency.

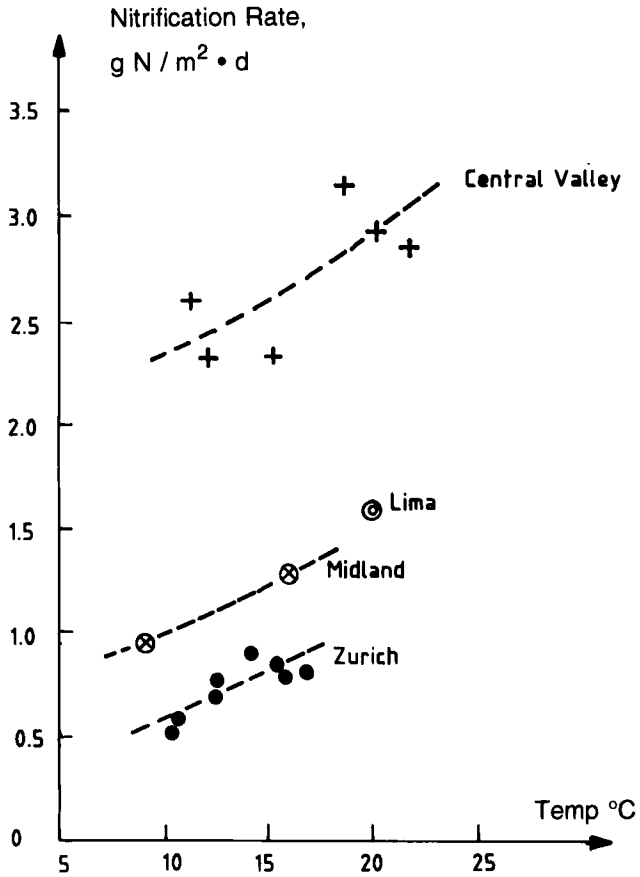


Figure 5.6 The effect of temperature on nitrification in a trickling filter. After Gujer and Boller (1986).

5.6.2 Equations for Modelling the Nitrifying Trickling Filter (NTF)

The most commonly used formula for designing a trickling filters was proposed by Erkenfelder (1961), and is as follows:

$$\frac{S_{NH_4e}}{S_{NH_4i}} = e^{-kD/Q^n} \qquad (5.15)$$

where:

$S_{NH4}e$ = effluent substrate concentration, mg/l

$S_{NH4}i$ = influent substrate concentration, mg/l

D = depth of the medium in meter, m.

Q = hydraulic loading rate $m^3/m^2 \cdot min$.

k = treatability constant relating to the waste water and the medium characteristics, min^{-1}.

n = Coefficient relating to the medium characteristics.

The values of the treatability constant k range from 0.01 to 0.1. The average value for municipal waste on plastic media is of the order of 0.06 at 20 °C (Germain 1966)

Correction for other temperatures can be made by adjusting the treatability factor k_T as follows:

$$k_T = k_{20°C}(1.035)^{T-20} \qquad (5.16)$$

The treatability factors k_T should be determined for each situation from a pilot-plant analysis of the waste water, and for the selected medium. The coefficient n for plastic media is 0.5 following Benefield and Randall (1980).

Including recirculation of the waste water into the equation, equation 5.15 can be modified to:

$$\frac{S_{NH4}e}{S_{NH4}a} = \frac{e^{-kD/Q^n}}{(1+R)-Re^{-kD/Q^n}} \qquad (5.17)$$

Table 5.6 Typical design criteria for the Trickling filter.

Item	*Low-rate filter*	*Intermediate rate filter*	*High rate filter*
Hydraulic loading $m^3/m^2 \cdot d$	1-4	4-10	10-40
Depth m	1.5-3.0	1.25-2.5	1.0-2.0
Recirculation ratio	0	0-1	1.3
Filter media	Rock, slag etc	Rock, slag etc	Rock, slag Synthetic materials
Power requirements $kW/10^3\ m^3$	2-4	2-8	1-10
Filter flies	Many	Intermediate	Few, larvae are washed away
Dosing intervals	Less than 5 min	15-60 sec	Less than 15 sec
Effluent	Usually fully nitrified	Partially nitri-nitrified	Nitrified at low loadings

FROM: Metcalf and Eddy (1991)

where:

S_{NH4},a = the content in the mixture of raw and recycled mixture applied to a medium.

R = the ratio of the recycled flow to the influent flow.

$$S_{NH_4a} = \frac{S_{NH_4o} + R * S_{NH_4e}}{1 + R} \qquad (5.18)$$

Gujer and Boller (1986) proposed the following line-fit equation for the decline in the nitrification rate with depth in a trickling filter:

$$r_{n,z,t} = r_{n,z=0,t} \cdot e^{-k \cdot z} \qquad (5.19)$$

where:

z = depth in tower in metres.

$r_{n,z,t}$ = nitrification rate at depth, g $N/m^2 \cdot d$.

$r_{n,z=0,t}$ = nitrification rate at the top of the tower, g $N/ m^2 \cdot d$.

k = empirical parameter describing the decrease of the rate with depth (k varies between 0.075 and 0.16).

t = temperature in degrees Celsius.

Gujer and Boller (1986) developed a biofilm model for predicting the surface nitrification rate as a function of the ammonia concentration in the bulk fluid, that

considered mass balance for oxygen and ammonia within the biofilm.

By combining equation 5.18 with the normal monod kinetic equation, Gujer and Boller developed the following two solutions for the design of NTF´s.

Introducing k (the empirical parameter) different from 0:

$$\frac{a * j_{n,max}(T)}{k * V_n} * (1 - e^{-kz}) = S_{ni} - S_n + N * \ln \frac{S_{ni}}{S_n} \qquad (5.20)$$

and k = 0

$$z * a * \frac{j_{n,max}(T)}{V_n} = S_{ni} - S_n + N * \ln \frac{S_{ni}}{S_n} \qquad (5.21)$$

where:

S_n = bulk liquid ammonia nitrogen concentration in mg/l.

$S_{n,i}$ = influent ammonia-nitrogen concentration in mg/l.

jn,max = maximum nitrification rate at high ammonia levels, g $N/m^2 \bullet d$.

jn(s,t) = nitrification rate at ammonia concentration g $N/m^2 \bullet d$.

N = saturation parameter mg/l.

a = specific surface area of the trickling filter media in m^2/m^3.

V_n= hydraulic load on the trickling filter in $l/m^2 \bullet s$.

Because the oxygen transfer efficiency of different plastic media differs, the following equation can be used to correct the nitrification rate for this difference:

$$j_{n,max(T)} = \frac{E * j\,O_2 max(T)}{4.3} \qquad (5.22)$$

where:

E = media effectiveness factor. The value of E depends on the media used, see Table 5.8

jO_2 max(T) = maximum surface oxygen rate for specific media design in g N/m^2 * d.

The factor 4.3 in equation (5.22) reflects the unit mass of oxygen consumption per unit mass of ammonia nitrogen oxidized.

Where recirculation is used, a repetitive solution of the above equation is necessary because recycle effects are included in both the $S_{n,i}$ and V_n terms.
The effect of the media on the nitrification rate is not considered in this modelling approach.

5.6.3 The Application of the Trickling Filter

Most trickling filters are used in single stage removal of organics. If the organic loading is lowered to about 0.16 kg BOD / m^3 • d, combined oxidation of organics and nitrification will occur, whereby a part of the influent ammonium will be nitrified (see Tables 5.7 and 5.8). But single stage nitrifying trickling filters are also becoming popular in treating secondary or tertiary influents, because the recent efforts in improving these filters have made the effluent produced of a better quality, so the NTF is comparable with the activated sludge processes in regard to nitrification efficiency and amount of the suspended solids in the effluent.

The concentration of ammonium-nitrogen must be less than 25 mg/l to obtain the best results in a conventional nitrifying trickling filter. The trickling filters are,

therefore, often used to treat municipal waste water, where BOD removal has already been accomplished. Experiments have been made with the new compact plastic media trickling filters in the treatment of industrial waste water of higher nitrogen concentration.

Combined oxidation of organics and nitrification.

Despite much interest in trickling filters, relatively little research has been made on the simultaneous organic removal and nitrification taking place in a single trickling filter unit.

The EPA (1975) showed that for rock media trickling filters, organic loading must be limited to 0.16 kg BOD / m^3 * d to attain 75% conversion of ammonium to nitrate. Nitrification decreased at a higher organic load. At an organic loading of 0.64 kg BOD / m^3 • d, nitrification of only 10 % of ammonium was obtained. This reduction in nitrification was attributed to the domination by heterotrophic bacteria of the microbial biofilm.

The difference between rock and plastic media in loading capacity, as shown in Table 5.5, was attributed to the higher specific surface area of the plastic, whereby less competition between the species of bacteria was necessary.

Wanner and Gujer (1984) showed the concentration of ammonium versus different COD concentrations for a trickling filter. They predicted that most of the organic removal occurred in the upper reaches of the trickling filter, where heterotrophic organisms dominated, and nitrifiers were absent. Nitrification occurred at the highest rates in the bottom portions of the tower where concentration of organics was the lowest, and the autotrophic population could dominate.

Nitrification only occurred in the bottom half of the reactor. The most significant nitrification occurred in the bottom 1.2 m of the filter. Most combined trickling filters do not produce nitrate before the soluble BOD concentration is less than about 20 mg/l. Figure 5.8 show the relationship between nitrification and soluble BOD_5 levels exposed to the biofilm for cross-flow media.

Nitrification in a nitrifying trickling filter (NTF).

The NTF is designed to oxidize ammonia in secondary effluents, where most of the BOD is already removed, so that the NTF can concentrate on the removal of ammonium-nitrogen. The first demonstration of the system was a pilot scheme in

Michigan (Duddles *et al.* 1974).

Typical removal rates, for conventially loaded NTF filters are as low as 0.20 to 0.39 g N/ m^2 * d as indicated by investigations in the US. Ammonia removal efficiency for rock and plastic filters at various sites in the US, applying different amounts of organic loading per unit of surface in kg BOD / 1000 m^2 • day is shown in Fig. 5.7.

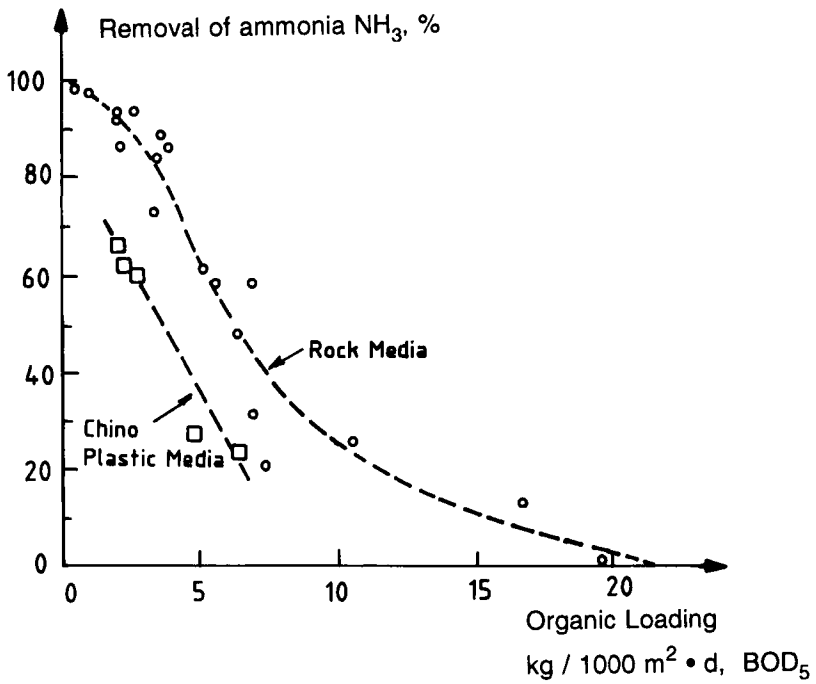


Figure 5.7 Ammonium removal efficiency for rock and plastic filters at various sites in the USA, applying different amounts of organic loading per unit of surface in kg BOD / 1000 m^2 * day. After Parker and Richards (1986).

The EPA (1975) manual showed the removal rate for the NTF to be between 0.83 and 1.50 g N / m^2 • d. A conventional design practice has been to follow the NTF with either effluent filtration or clarification.

Recognizing the costs advantages of operation and maintenance of NTF technology, studies have been undertaken to assess the factors limiting the possible

nitrification rates, and to modify the processes of the NTF.

As a result of those studies Parker *et al.* (1989) proposed the Biofilm-Controlled-Nitrifying-Trickling-Filter (BCNTF). The new design incorporated weekly flooding and backwashing of the BCNTF for predator control, and cross-flowed plastic media were applied for better oxygen transfer to the biofilm, resulting in a higher biomass content. The peak nitrification rates obtained for the BCNTF were between 2.3 and 3.2 g N /m^2 • d (0.32 and 0.44 kg N/ m^3 • d). The BCNTF process has therefore, a peak nitrification rate of about 3 times the NTF process.

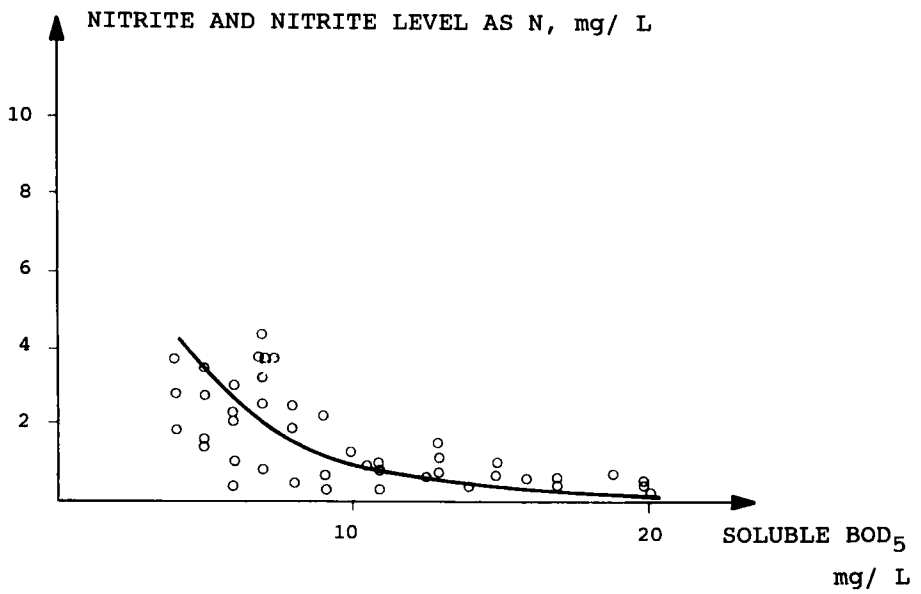


Figure 5.8 Relationship between nitrification and soluble BOD_5 level exposed to the biofilm for cross-flow media; After (Parker and Richards 1986).

Additional advantages of the BCNTF is the smaller land area needed and that it can be constructed without disruption of secondary treatment operations. These changes in design and other improvements have made the BCNTF very competitive with the nitrifying activated sludge process.

Denitrification with a trickling filter.

Trickling filters are also able to conduct denitrification, when part of the filter has low oxygen concentration, the presence of nitrate and a carbon source that can act as an electron donor in the denitrification process. Effluent recycling is predicted to be favourable to the denitrification process. Almost all NTF units can denitrify a part of the formed nitrate-nitrogen, depending on the circumstances mentioned above. Stenquist *et al.* (1974) mentioned an example where a combined trickling filter loaded with 0.36 kg BOD / m^3 • d, caused denitrification of 25 % of the ammonium-nitrogen applied to the plant, and 89 % of the ammonium-nitrogen applied was nitrified.

5.6.4 Recent Developments in the Technology of the Nitrifying Trickling Filters (NFT)

The development of the Biofilm-Controlled-Nitrifying-Trickling-Filter (BCNTF) (see Fig. 5.9) is the latest effort to enhance the nitrification rate in nitrifying trickling filter technology. The BCNTF has a peak nitrification rate of about three times that of a conventional NTF. The suspended solids (SS) from the effluent from a BCNTF are almost the same as those found in the influent. If the existing secondary effluent, therefore, is already of a high quality (i.e. the average effluent SS and BOD are less than 15 mg/l) it has been shown in the literature that applying BCNTF is less costly than using a conventional activated sludge process.
Further information about the BCNTF is presented above.

5.6.5 Nitrogen Loading Capacity and Removal Efficiency of the Different NTF-applications

Gulliecks and Cleasby (1986) proposed the curves shown in Figs 5.10A and 5.10B as design curves for application of nitrification to municipal secondary effluent, which has been settled before use of the trickling filter. The filter used for these design curves contained 6.55 m of vertical-type plastic media with a specific surface area of 88.6 m^2/m^3. Figure 5.10A is proposed for waste water with a temperature below 10 °C, and Fig. 5.10B for temperatures between 10 and 14 °C.

The curves correlate the influent ammonia-nitrogen concentration, the applied hydraulic flow in l/m^2 • s, with the expected yield in nitrification rate in kg N/m^2 * d for the trickling filter. It is important to note that the maximum range for the influent

ammonia-nitrogen and hydraulic flow on the axes of the curves. If the concentrations in a sample of waste water exceed the values on the axes of curves presented in Figs 5.10A and 5.10B, it is then necessary to use recirculation in order to achieve a mixed concentration, which is applied to the proposed curves for the use of the curves in estimations.

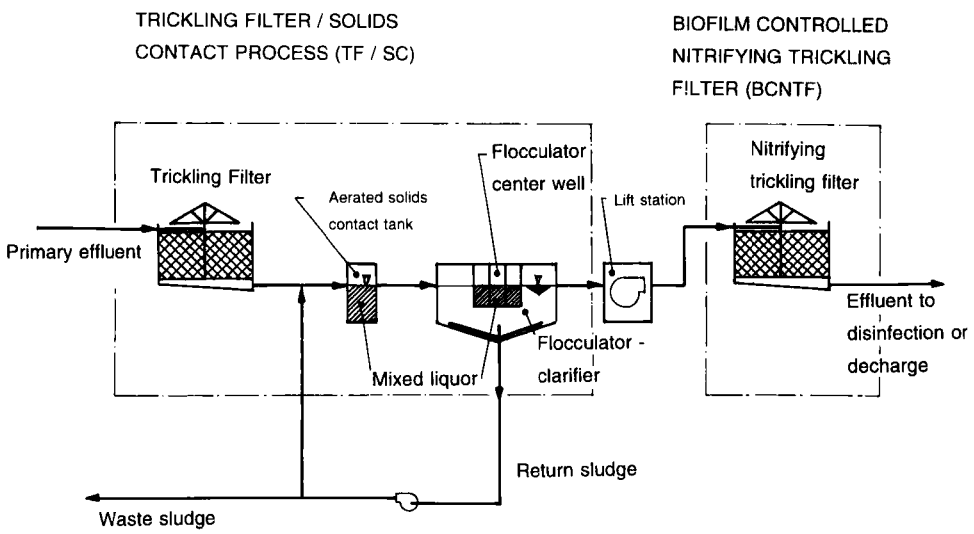


Figure 5.9 Applying Biofilm-Controlled-Nitrifying-Trickling-Filter (BCNTF) to process application of a conventional trickling filter. After Parker *et al.* (1989).

Figure 5.11 shows that there is a great variation of the peak nitrification rate at different depths in an NFT.

This decline is attributed to the patchy development of the biofilm at greater depths, caused by the absence of a continuous supply of ammonia to support biofilm development at such depths. Most peak nitrification rates are, therefore, calculated for the whole NTF, and not at certain depths in an NTF.

Table 5.7 shows the peak nitrifying rate for the main types of nitrifying trickling filters. The Table indicates that the BCNTF system developed by Parker and co-workers yields a peak nitrification rate of 2,3 to 3,2 g N / m^2 * d, which is high compared with previously developed NTF's.

Stenquist *et al.* (1974) reported that up to 25 % of denitrification (complete nitrogen loss) were found in NTF plants, depending on the design, as indicated in Section 5.6.1.

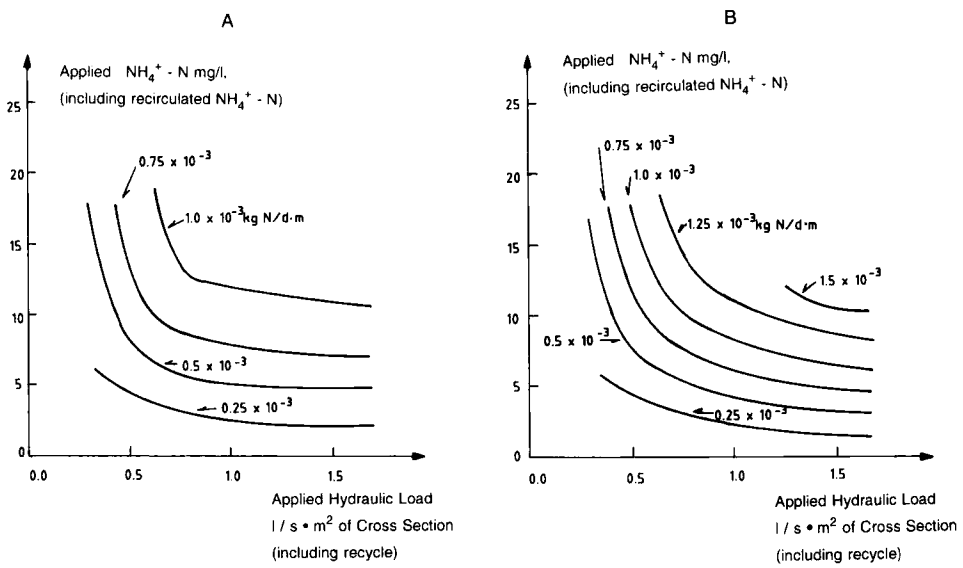


Figure 5.10A and 5.10B The predicted removal of kg N / m^2 * day of the media surface, versus the applied hydraulic load and applied ammonia-nitrogen for nitrification of a municipal secondary clarifier effluent at a waste water temperature below 10 °C (A) and between 10 ° C and 14 °C (B). After Gulliecks and Cleasby (1986).

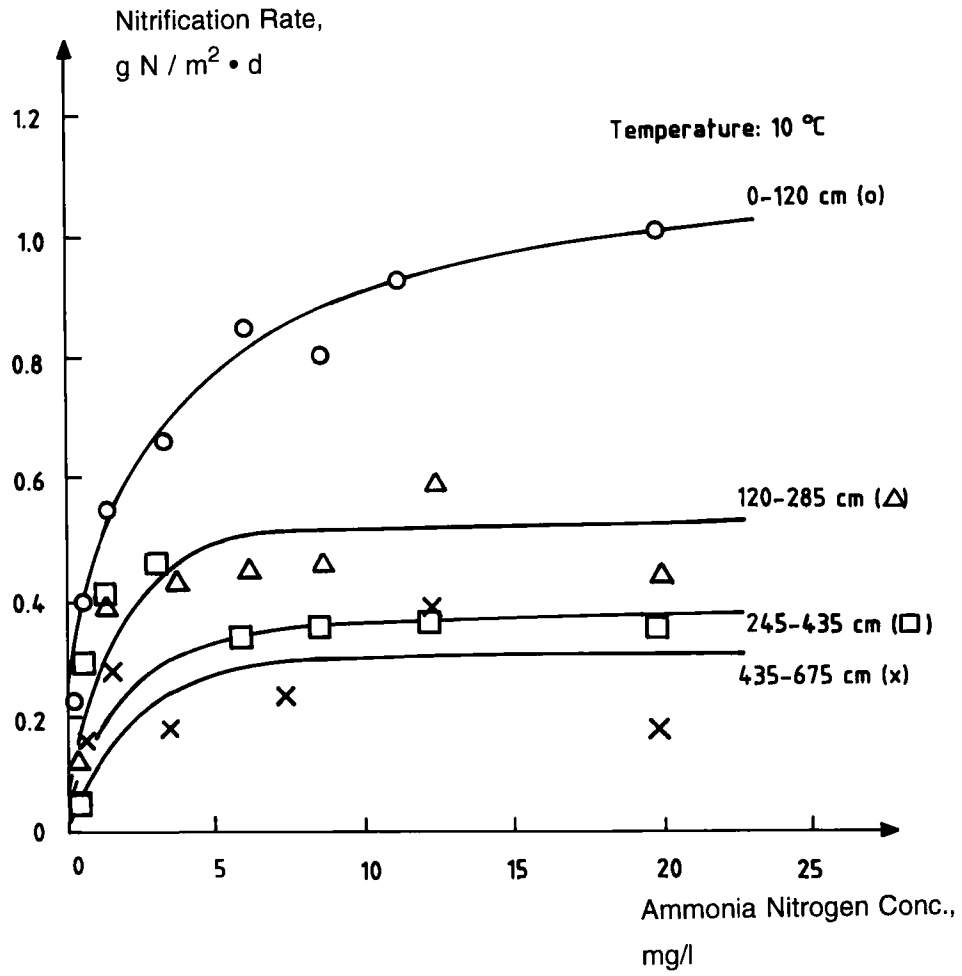


Figure 5.11 The nitrification rate as a function of ammonia concentration at four different depths in a trickling filter. After Parker *et al.* (1989).

Table 5.7 The results from different NFT´s as presented in the literature.

Organic loading	*Trickling filter media*	*Performance total possible*	*Reference*
0,16 kg BOD_5 /m^3· d	Rock	75%	EPA (1975)
0,64 kg BOD_5 /m^3· d	Rock	10%	EPA (1975)
0,36 kg BOD_5 /m^3· d	Plastic media	89% nitrification 25% denitrification	Stenquist *et al.* (1974)
2,5 kg BOD_5 /1000 m^2· d	Plastic Cross flow media	92%	Parker *et al.* (1986)
2,5 kg BOD_5 /1000 m^2· d	Plastic vertical flow media	60%	Parker (1976)
6,3 kg BOD_5 /1000 m^2· d	Plastic media Garland Texas, USA	42%	Parker *et al.* (1986)

Table 5.8 Comparison of nitrification rates for different NFT plants.

Location	*Media*	*Temp.°C*	*Nitrification*	E*	Reference
Midland, Mich	Vertical flow media	13	1,20	0,86	Duddles G.A. *et al.* (1974)
		7	0,93	0,74	"
Lima, Ohio	Vertical flow	21	1,70	1,01	Sumpayo E.F.(1973)
Bloom Township III	Vertical flow media	20	1,20	0,88	Baxter and Woodman (1973)
		17	1,10	0,82	
Zurich Switz. (3.9 m tower)	Cross flow plastic media	17-20	1,40	0,65	Richards (1988)
Zurich Switz. (6,8 m tower)	Cross flow plastic media	13	1,10	0,39	"
Central Valley	Cross flow plastic media	18	2,60	0,80	Parker *et al.* (1989)

* = Media Effectiveness factor (E).

5.6.6 Advantages and Disadvantage of the NTF

The following advantages and disadvantages can be listed for the application of a nitrifying trickling filter.

Advantages:

• Their simplicity and low operational cost make the trickling filters an attractive option for small communities in warmer climates.

• The recovery from hydraulic and substrate shock-loads is fast.

* It is possible to obtain a high content of biomass, especially when highly porous plastic media are used.

Disadvantages:

• Trickling filters achieve only with difficulty the high efficiency which is demanded by recent effluent standards in many countries.

• Most trickling filter effluent needs a polishing process, because the concentration of suspended matter at high loadings is unacceptable for meeting effluent standards.

• It is difficult to ensure an effective predator control, so the maximum nitrification rate can rarely be obtained.

5.7 Rotating Biological Contactors (RBC)

The Rotating Biological Contactor (RBC) is used for a variety of purposes: Aerobic degradation of organic material; combined organic removal and nitrification; and denitrification and nitrification of secondary and tertiary effluent (after filtration).

A rotating biological contactor (RBC) consists of a series of closely spaced rotating circular discs made of different kinds of materials, for example plastic, wood, and galvanized plates.

These discs are approximately 40 % immersed in a tank through which waste water flows continuously. The discs are mounted on a shaft which usually rotates through the water at a velocity of 1 rpm. A layer of biological growth, depending on the composition of the waste water, builds up on the wet surface of the discs and forms a biofilm ranging from 1-2 mm in thickness. The formation of a fully developed biofilm takes from 1 to 4 weeks. As the discs rotate through the waste water, the ammonium content is nitrified, and the organic carbon content is oxidized by various microorganisms. Excess growth on the discs is disposed of at the same time. The discarded biofilm is washed out of the unit and removed during a secondary clarification. As the biofilm is passed out of the liquid and through the air, oxygen is absorbed to keep the growth aerobic.

An RBC treatment plant will generally consist of a number of shaft trains, each operating as a completely mixed, fixed-film biological reactor. Each train is generally set up in a number of stages, separated by baffles for more efficient treatment and stability. By doing so, it is possible to achieve a high degree of nitrification. Figure 5.12 shows a flow diagram of the rotating biological contactor process.

5.7.1 The Performance of the RBC

The factors affecting nitrification in the RBC process are the same as in other nitrifying plants, namely, organic concentration, influent nitrogen concentration and composition, waste water temperature, DO concentration, pH and alkalinity, and influent flow and load variability.

Most empirical design procedures are based on the assumption that significant nitrification does not begin in an RBC system until the bulk liquid soluble BOD_5 has been reduced to 15 mg/l. In combined carbon oxidation-nitrification units this will

typically first be encountered in the third or fourth stage, depending on strength, organic loading rate and temperature of the influent.

Hydrogen ions are produced during nitrification. In poorly buffered RBC nitrifying systems, alkaline chemicals such as lime, soda ash, and sodium hydroxide may have to be added to the waste water in order to maintain a sufficient alkalinity to prevent a sudden decrease in pH and thereby a decrease of the nitrification rate.

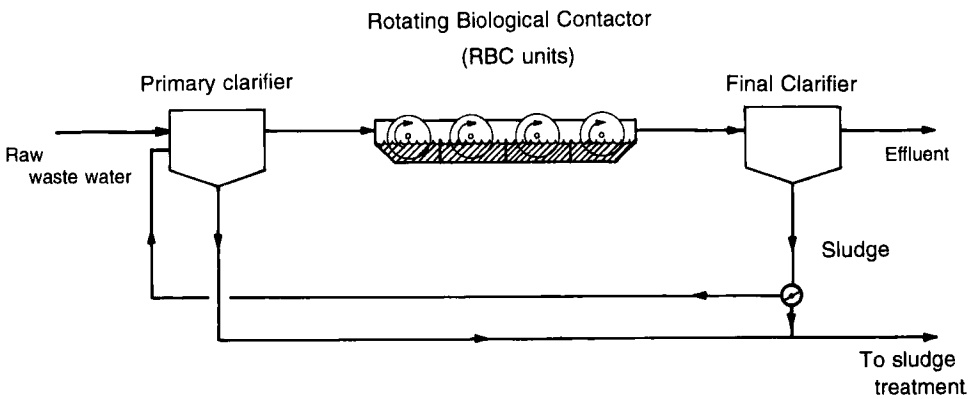


Figure 5.12 Flow diagram for the rotating biological contactor process.

The media in an RBC.

The RBC media must serve several purposes according to Antonie (1976): It must:

1. provide a surface area for the development of a large, fixed, suitable biomass,
2. provide vigorous contact of the biological growth with the waste water,
3. aerate the waste water efficiently,
4. provide a positive means of continuously removing excess biomass, and
5. agitate the mixed liquid to keep the discharged solids in suspension and thoroughly mix each stage of treatment.

Many different materials for RBC media have been used over the years, from the wooden slats of Poujelet in 1916 to the plastic discs used today. The discks are usually 2-3 m in diameter and 1.2 cm thick.

Today the discs are made from a high-density polyethylene in alternating flat

and corrugated sheets, which are bonded together. This design provides more than twice the surface area per unit volume than flat sheets.

The rotational speed of an RBC.

The rotation of discs serves a varity of purposes in the RBC process:

1. It provides a contact between the biomass and waste water.
2. Removal of excess biomass.
3. Mixing of the liquid and aeration of the waste water.

If the rotation rate is increased, the effects mentioned above are enhanced to a point above which further increase is not productive.

The optimum rotational speed for the RBC varies depending upon the composition of the waste water and the disc size of the RBC.

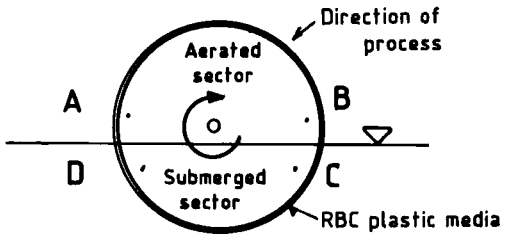
In practice, most RBC units are operated at 1.0 to 1.4 rpm.

The aeration of an RBC.

In some RBC facilities, aeration equipment has been installed, either to drive the RBC shafts or to provide supplemental aeration. RBC with aeration facilities usually results in a thinner biofilm on the discs in the first compartments because of the stripping action of the bubbles, thereby allowing more of the biofilm to remain aerobic.

It appears, however, that the dissolved oxygen in the mixed liquid has little effect on the transfer of oxygen into the biofilm (see Fig. 5.13). A study of the mass transfer of oxygen in the biofilm indicates that very little of the oxygen utilized by the microorganisms in the film comes from the bulk liquid in the RBC tank; it comes from the atmosphere, when the disc surface is exposed to air. The transfer of oxygen to the biofilm is better increased by lengthening the exposure time to air or reducing the thickness of the liquid film on the disc by more efficient emptying than by aerating the waste water. Figure 5.13 show the relative concentrations of oxygen and substarte for the loading condition and RBC rotational spped as a function of the media.

a



b

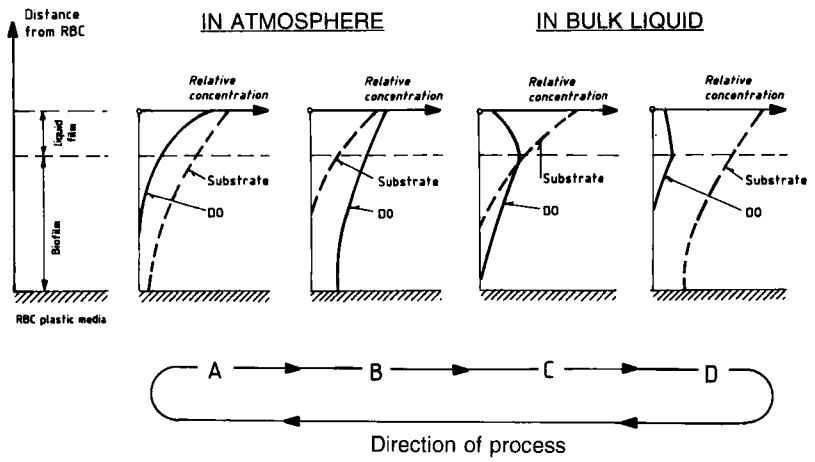


Figure 5.13 The relative concentrations of oxygen and substrate for the loading condition and RBC rotational speed as a function of location of the media.

The arrangement of multiple RBC units.

Staging of RBC media is recommended to maximize the removal of ammonium. In secondary treatment applications, three or four stages are generally provided for each stream. For small installations, four stages can be provided on a single shaft by installing three inter-stage baffles within the tank, and introducing the flow parallel to the shaft. Installations requiring two RBC units may be placed in series with a single baffle in each tank, thus providing four stages. Four or more units can be placed in series, with each unit becoming a single stage. Various schemes of staging RBC units are shown in Fig. 5.14.

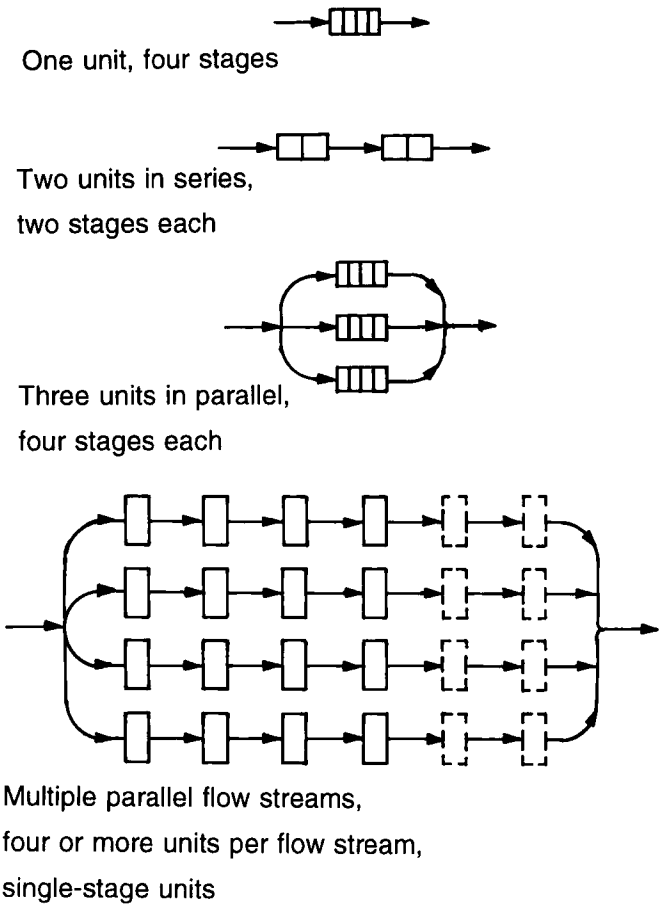


Figure 5.14 Various schemes of staging RBC units.

The biomass of the RBC.

If an RBC is supplied with secondary influent, the unit will be divided into four sections. The first section will not be able to accomplish nitrification, because of a high content of organic matter and, therefore, no nitrifying population will be able to develop.

Both nitrification and organic oxidation will be carried out in the second section. The waste water content of ammonium is high and, therefore, the nitrification is relying upon on the oxygen content in the waste water and on the size of the nitrifying biomass, developed in relation to the size of the heterotrophic biomass.

In the third section most of the organic load in the waste water is oxidized, so

that this section will function as the nitrifying section. As in trickling filter processes, nitrification will only proceed after the carbon concentration has been substantially reduced. Only the oxygen content in the waste water will limit the nitrifying rate.

In the fourth section the ammonium content is so low that it is not the oxygen content that limits the nitrification rate, but the content of ammonia itself.
It is, therefore, important to have full control over the organic content in the different parts of an RBC plant, and to design at least four or five modules in series, if nitrification is required, because as illustrated in Section (3.13) ammonia itself can inhibit the nitrification rate.

5.7.2 Equations for Modelling the RBC Reactor

Matsuo and Yamamoto (1985); Watanabe (1985) and Gujer and Boller (1990) have all modelled the process of RBC units.

Gujer and Boller (1990) proposed a model containing two levels: a microscopic level and a macroscopic level. The microscopic level considered the transport and reaction processes within the biofilm. The macroscopic level described the system as a whole. Mixing conditions within the individual compartments, influent and effluent transport processes, gas exchange processes, exchange of substrate, nutrients and biomass within the biofilm, and reactions catalyzed by biomass in suspension were considered as important factors in the performance of the RBC.

Three submodels, a kinetic model, a biofilm-model, and a reactor compartment model were proposed to take account of the above factors.

The kinetic sub-model.

The equations proposed by Gujer and Boller (1990) for the kinetic sub-model were the same as outlined in Section 5.4 for the biofilm kinetics, depending on either zero order or half order kinetics.

The biofilm sub-model.

The biofilm sub-model takes the following variables into account: the dissolved components, the particulate components, the removed biomass, the surface flocculation and the thickness of the biofilm. The different equations used in this sub-model are shown in Table 5.9 and the relevant constants are given in Table 5.10.

The reactor sub-model.

The reactor sub-model takes into consideration the rotation of the RBC and the design of the reactor compartment.

Variation in the concentration of dissolved oxygen in the depth of the biofilm, due to rotation of the RBC, depends upon the processes of diffusion and reaction. The depth of penetration of dissolved components due to molecular diffusion is given by:

$$L_i = (D_i' \bullet t)^{1/2} \qquad (5.23)$$

where:

L_i = Depth of penetration of compartment i during time t in metres.

D_i' = Effective diffusion coefficient within the biofilm, assumed to be 80% of the value in pure water.

t = time.

Table 5.9 The different equations used in the biofilm sub-model for the RBC.

Process	*Equation*	*Symbols*
Transport of dissolved components	$J_{S_i} = -D_i \cdot \frac{d S_i(z)}{dz}$ where J = 0 for z = L_B	z = depth of biofilm; z = 0 at surface and z = L_B at support material. J = flux of component **i** due to molecular diffusion within the biofilm. S_i (z) = concentration of dissolved component **i** at biofilm depth z. D_i = effective diffusion coefficient within the biofilm, assumed to be 80% of the value in pure water.
Dissolved component	$\frac{d S_i}{dt} = \frac{dJ_{S_i}}{dz} + r_{S_i}$	$r_{s,i}$ = transformation rate of the dissolved component **i** per unit volume of biofilm.

Table 5.9 (continued)

Process	*Equation*	*Symbols*
Particulate components	$\frac{dX_i}{dt} = -\frac{dJ_{x,i}}{dz} + r_{x,i}$ where $J_{x,i}(z) = X_{i(z)} \cdot \int_{L_B}^{z} \sum_i (r_{x,i}/X_{tot})\, dZ$	X_i = concentration of particulate species **i** within the biofilm. X_{tot} = sum of all particulate species concentration. $J_{z,i}$ = flux of particulate species **i** within the biofilm. $r_{x,i}$ = rate of production of particulate species **i** within the biofilm.
Surface flocculation	$J_{FLOC,i} = k_{FLOC} \cdot X_i$	$J_{FLOC,i}$ = flux of particulate material **i** flocculated form from the bulk liquid to the surface of the biofilm. k_{FLOC} = flocculation mass transfer coefficient.

Table 5.9 (continued)

Process	*Equation*	*Symbols*
	$J_{SHEAR,I} = K_{SHEAR} \cdot L_B \cdot X_{B,i}$	$J_{SHEAR,i}$ = flux of particulate material **i**, sheared from the surface of the biofilm to the reactor bulk liquid.
	K_{SHEAR} = 0,10 d^{-1} for primary effluent = 0,05 $^{-1}$ for secondary and tertiary effluent.	$K_{SHEAR,I}$ = Shear rate constant. $X_{B,i}$ = concentration of particulate material **i**, at the surface of the biofilm (z=0)
Biofilm thickness	$\frac{dL_B}{dt} = \int_o^{L_B} \sum_i (r_{X,i}/X_{tot})\, dz + \sum (J_{FLOC,i} - J_{SHEAR,i})/X_{tot}$	

Table 5.10 Different constants proposed for use in the Biofilm sub-model.

	Symbol	*Unit*
Heterotrophic organisms:		
Maximum growth rate	$\mu_{max,}H$	2,00 d^{-1}
Saturation coefficient		
for COD	K_s	10,0 g/m^3
for NH_4^+-N	K_{NH4}^+	0,1 g/m^3
for O_2	K_{HO}	0,1 g/m^3
for HCO_3^-	K_{HA}	0,10 g/m^3
for NO_3^-	K_{HNO}	0,50 g/m^3
Denitrification coefficient	DEN	0,70 g/m^3
Decay rate	b_H	0,35 d^{-1}
Yield coefficient (COD/COD)	Y_H	0,57 g/g
Fraction particulate decay product	f_I	0,08
Nitrogen content of biomass (N/COD)		0,06 g/g
Decay product		0,05 g/g
Nitrosomonas		
Maximum growth rate	$\mu_{max,}N$	0,35 d^{-1}
Saturation coefficient		
for NH_4-N	K_{NH4}	0,70 g/m^3
for HCO_3^-		0,20Mol/m^3
for O_2		0,20 g/m^3
Decay rate	b_N	0,05 d^{-1}
Yield coefficient (COD/NO_2^--N)		0,18 g/g
Nitrobacter		
Maximum growth rate	$\mu_{max,}nb$	0,60 d^{-1}

Table 5.10 (continued)

Saturation coefficient		
for NH_4^+-N	K_{NH4}^+	0,05 g/m^3
for NO_2^--N	K_{NO2}^-	0,50 g/m^3
for O_2	K_{O2}	0,10 g/m^3
Decay rate	b_{NB}	0,09 d^{-1}
Yield coefficient (COD/NO_3^--N)	Y_{NB}	0,06 g/g

Diffusion coefficients within a biofilm *correlated for temperature (10 °C) and reduction to 80% of values in pure water.*

Dissolved oxygen	$106 \cdot 10^{-6}$ m^2/d
Degradable COD	$31 \cdot 10^{-6}$ m^2/d
Ammonium	$86 \cdot 10^{-6}$ m^2/d
Nitrate	$85 \cdot 10^{-6}$ m^2/d
Nitrite	$84 \cdot 10^{-6}$ m^2/d
Bicarbonate	$53 \cdot 10^{-6}$ m^2/d

Rate constants for biofilm surface reactions

Flocculation rate	K_{FLOC}	0,10 d^{-1}
Shear rate constants	K_{SHEAR}	
Primary effluent		0,10 d^{-1}
Secondary and tertiary effluent		0,05 d^{-1}

Gujer and Boller (1990) demonstrate that it is only at a very slow rotation speeds and a low concentration of residual pollutants, that the effect of the rotation of the support material must be considered.

RBC's today are usually compartmentalized; each drum of rotating surface areas is calculated as an individual reactor compartment.

For the N^{th} reactor compartment the substrate balance is written as:

$$V_N * \frac{dC_{i,N}}{dt} = (Q+R) * (C_{i,N-1} - C_{i,N}) + r_{i,N} * V_N - J_{i,N} * A_N \qquad (5.24)$$

where:

V_N = Volume of reactor N (bulk water phase) in m^3.

$C_{i,N}$ = Bulk concentration of dissolved or suspended component i in reactor N in kg per m^3.

Q + R = Influent and recycle flow rate in m^3 per hour.

$r_{i,N}$ = Rate of production of component i within the bulk liquid in kg i/m^3 * day.

$J_{i,N}$ = Flux of component i into the biofilm of reactor N, in kg / m^2 * day.

A_N = Surface area of support material in reactor A in m^2.

The entire reactor system can then be modelled as a series of reactors with the option of recirculation of effluent from the last compartment to the first one. It is also possible to reverse the flow, either from the first to the last or, alternatively, from the last to the first reactor.

5.7.3 The Application of the RBC

Rotating Biological Contactors are popular in small-scale waste water treatment plants (< 100 P.E.), because of their easy maintenance, low sludge production, and low power requirements. One of the key costs in the production of an RBS is the support material. The RBC is therefore, ideal for the treatment of small volumes, which cannot easily be connected to a central treatment system for economic or geographic reasons.

The RBC can be used as a combined oxidation and nitrification system for secondary effluent, or as a tertiary nitrifying system depending on the composition of the influent waste water. Examples of anaerobic denitrification with an RBC are also presented in the literature (Hosomi *et al.* 1991).

The different functions of the RBC in the nitrogen removal processes are listed below:

A) Combined oxidation and nitrification with an RBC unit.

Treatment of industrial waste water or small scale waste water treatment plants, with an RBC treatment plant, will usually provide a unit with combined oxidation of carbonic material and nitrification of ammonium to nitrate. Because the waste water is a mixture with high carbon and ammonium contents which act as substrate for both oxidizing and nitrifying bacteria, the bacteria will compete for the space on the RBC disks.

B) Nitrification with an RBC unit.

Using the RBC as a tertiary nitrifying treatment plant has been shown to be highly efficient. The RBC units can, therefore, be expected to be used for final refinement, in an effort to reach present effluent standards for nitrogen content, because they can be integrated into existing flow schemes.

C) Denitrification with an RBC unit.

Denitrification in RBC systems may be observed in the following two situations: 1) Addition of oxygen may be reduced, either by nearly complete submersion of the rotating contactors, or by maintaining an atmosphere poor in oxygen. This situation will allow denitrification even at low levels of the organic loading, if nitrate is fed to the reactor, for example via recirculation of nitrate from the effluent waste water.

2) Denitrification may occur in the depth of a biofilm, where oxygen has fallen to insignificant levels. This requires high concentration of organics and nitrate to secure denitrifying conditions in the depth of a biofilm. Since the recirculation of nitrate results in dilution of the concentration of carbon compounds, this situation may only occur with industrial waste water where concentration of carbon compounds is high.

D) Simultaneous nitrification and denitrification (SND) with an RBC unit.

Masuda *et al.* (1991) reported the loss of nitrogen in an RBC plant treating the leachate from a sanitary landfill located at Miyazaki, Japan. The loss of nitrogen was appreciable during the summer. The RBC was covered by a hood, and during the summer the oxygen pressure in the hood was 1.8 to 1.9 atm., which was a little less than in an unhooded RBC. Matsuda and his co-workers measured the production of nitrogen gas from the biofilm, using a covered RBC, to be able to observe denitrification. In order to explain the loss of nitrogen, the authors made the following hypothesis: Nitrifiers and denitrifiers co-exist in a biofilm; the denitrifiers become active, if the transfer rate of oxygen to the biofilm decreases sufficiently to result in the formation of a micro-anaerobic environment.

Halling-Sørensen and Hjuler (1992: 1993) have observed the same occurrence in a submerged filter, using clinoptilolite as the media.(See section 5.8.1).

Masuda and his co-workers conducted a series of experiments to discover the factors which influence simultaneous nitrification and denitrification (SND).
They showed that the highest capacity of the SND in the RBC unit, in midsummer, was a total conversion of 130 g/m^3 NH_4^+ - N to 80 g/m^3 gaseous nitrogen N_2, and the remaining 50 g/m^3 was converted to NO_3^- - N. The efficiency of SND was accordingly 61,5 %.

5.7.4 Recent Development in the RBC Technology

Much effort has been made to enhance both capacity and effluent quality of the RBC treatment systems because in the near future, full nitrification will have to be adopted in many treatment plants in order to reach present effluent standards. Using the RBC as a tertiary treatment step, it can in most cases be integrated into existing flow schemes.

Boller *et al.* (1990) proposed a two-stage nitrifying RBC, including precipitation, primary clarification and a solid separation step; this was after the use of the first BOD-removing RBC´s using a cloth filter, and finally a nitrifying RBC with the possibility of reversing the flow so as to obtain a higher utilization of the surface of the biomass carrier throughout the RBC. (see Fig. 5.15). This design also provides a better possibility for fluctuations of ammonia in the waste water because of the higher nitrification potential. This type of RBC plant, with filtration before nitrification and two-sided loading of waste water, requires about 40% less surface area and volume than a conventional RBC with one-sided flow, where a nitrification capacity of 1,8-2,9 g N/ m^2 * d, is established.

To further enhance the capacity of the RBC, Wanner *et al.* (1990) proposed a packed-cage RBC. This is an RBC which is a combination of suspended and fixed-film biomass. The discs or groups of discs from a conventional RBC were replaced with a cage, packed with a random medium. The cage is equipped with tubular aeration and mixing elements.

The combination of suspended and fixed film biomass should enhance the capacity for nitrification and lower the the cost, because aeration of the activated sludge is separated from the rotation of the RBC and, therefore, an external source of air is avoided, and a large biomass is developed. This combination should make this design suitable for plants handling the treatment of 500 to 800 P.E.

To improve the effluent quality of the RBC process, Tanaka *et al.* (1991) investigated the behaviour of the fine particles throughout the processes; they found that an increase in the hydraulic retention time in the RBC reduced the amount of fine particles and increased the amount of coarse suspended solids, which are easily removed by clarification.

5.7.5 Nitrogen Loading Capacity and Removal Efficiency of the RBC-process

If nitrification is desired, loading rates should be reduced to 0,03 to 0,08 m^3 / m^2 • d. This is about one third of the capacity of an RBC when applied only to the removal of organics.

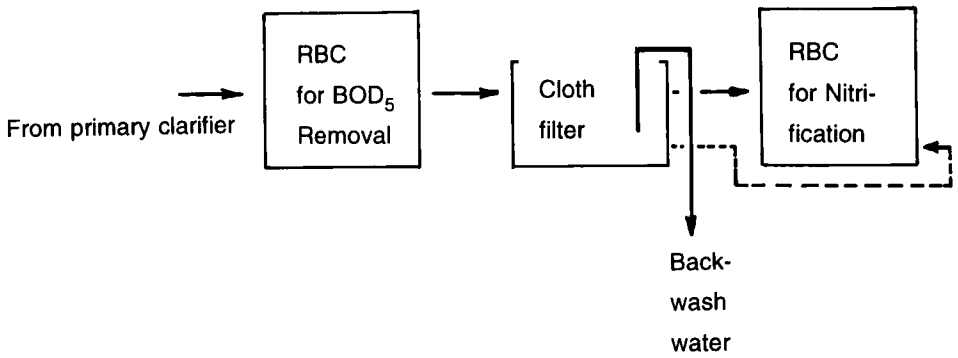


Figure 5.15 The two-stage nitrifying RBC, including precipitation, primary clarification and a solids separation step after the first BOD removing RBC´s with a cloth filter and lastly an RBC for nitrification. (From Gujer and Boller 1990).

The following mass balance equation can be used to calculate the removal of ammonium per m^2 filter per day as an average for the whole filter.
Figure 5.18 show the relationship between nitrification capacity and temperature in an RBC unit.

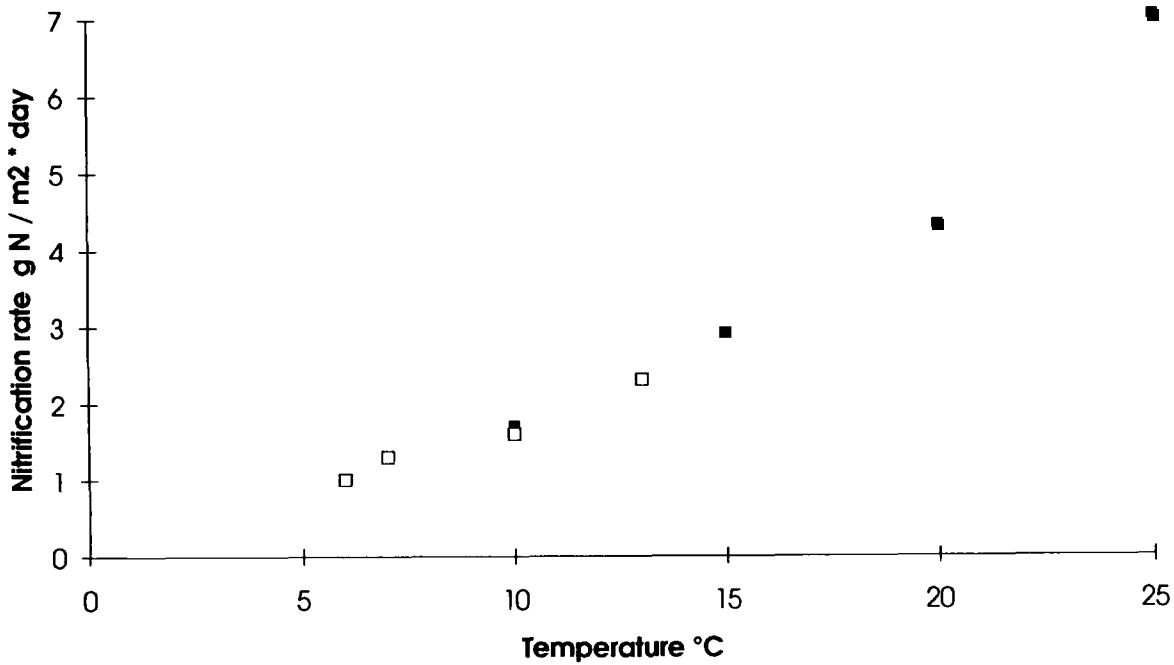
$$Q*(C_{NH_4}inflow-C_{NH_4}outflow) = \frac{dNH_4}{dt}*A' \qquad (5.25)$$

where:

Q = loading rate for waste water in m^3 /day.

A´= the total disc area in m^2.

dNH_4^+/dt = g NH_4^+ - N /(m^2 * d) removed.



Figur 5.16 The relationship between nitrification capacity and temperature in an RBC unit (partly after EPA (1984) and La Cour Jansen and Henze (1990)).

Figure 5.16 show the relationship between nitrification capacity and temperature in an RBC unit, and Table 5.11 some examples of nitrifying removal rate for the RBC using different types of waste water. Table 5.11 show the removal rate with different applications of the RBC.

Table 5.11 The removal rate for the RBC using different types of waste water.

Wastewater type	Nitrification rate g N /m^2 • d	
	maximum	minimum
Domestic	1,69	2,56
Percolate	2,42	2,66
Fertilizer industry	2,36	2,67
Leather industry	2,35	2,62
Sewage water	1,53	1,97

Source La Cour Jansen and Henze (1990)

Table 5.12 The removal rate with different applications of the RBC.

Treatment	*Treatment step*	*Capacity*	*Temp. °C*	*Process rate nitrification $gN/m^2 \cdot d$*	*Reference*
Nitrification	tertiary	800 P.E.	10°C	1,8-2,9	Boller *et al.* (1990)
Combined Nitrification/oxidation	secondary (combined RBC and activated sludge)	less than 100 P.E.	-	1,04	Wanner *et al.*(1990)
Combined nitrification/oxidation	compact RBC	-	-	0,6	Ahn and Chang (1991)
Combined nitrification/oxidation	RBC with simultaneous nitrification and denitrification	-	-	1,0	Matsuda *et al.* (1991)

5.7.6 Advantages and Disadvantages of the Nitrifying RBC

The following points summarize the major advantages and disadvantages connected with the use of the RBC in the process of nitrogen removal in treatment plants.

Advantages:

1. Only a small land area is required.

2. Ability to obtain a high content of biomass per m^3 or m^2 of disc because of the highly developed disc units and, therefore, the lower contact time with the waste water.

3. Simple operation of the equipment.

4. Ability to handle shock loads and, therefore, suitable for treatment of highly concentrated industrial waste water.

5. Ability to achieve a high degree of waste water purification, including nitrification.

6. Good performance even with a low oxygen level in bulk waste water because most oxygen is absorbed during rotation in the air phase.

7. Good performance of tertiary nitrification and, therefore, a solution to introduction of full biological removal of nitrogen in existing plants.

8. Using an RBC unit, pumping large amounts of waste water is avoided, because the water is passed slowly through the basin, where the contactor is rotating.

Disadvantages:

1. Enclosures are necessary to protect against low temperatures, rain and wind.

2. High capital cost.

3. Upsets can and do occur, because of too great wash-out of biofilm.

5. Most RBC´s are mainly designed for BOD removal, although some nitrification may occur in some plants.

4. Lack of documented operating experience.

5.8 Submerged Filters

Submerged biological filters (also known as biological aerated filters or contact aerators) are filters where the fixed material upon which the biofilm develops is continously submerged in the waste water they treat being treated.

The use of submerged filters has received renewed interest because of the development of plastic media and other sorts of filter media upon which large quantities of bacteria can grow. Also it has been shown that submerged biological filters may be very efficient at nitrification. Dillon and Thomas (1990).

Submerged filters (Fig. 5.1) can be designed both in up-flow or down-flow modes. In both cases, there is often a combination of both fixed-film and suspended growth between the filter media. Air is supplied to provide oxygen to the microorganisms, to promote mixing, and to scrub excess biofilm from the filter media to prevent irregular sloughing and plugging problems.

Because of the large biomass concentration, the contact time is often, low compared to other treatment systems to achieve the same efficiency.

Only few submerged filters are installed for nitrification. But indications are that they can be cost effective from both a capital standpoint and an operation and maintenance standpoint, that they reduce land area requirements, and that used, as teritiary treatment, have an efficiency of up to 90 percent of nitrogen removal with very low retention times.

Examples of the use on full size plants, of a biological fixed-film reactor for combined oxidation and nitrification treatment step for treatment of municipal waste water, are the systems ***Biocarbone*** and ***Biofor*** developed by respectively OTV and Degremont. (Dillon and Thomas 1990; Gilles 1990; Mange and Gros 1990; Paffoni *et al.* 1990, and Rogella and Bourbigot 1990).

The Biocarbone process use grain-sized biodagene (expanded schist) as bedvolume and the Biofor use spherical biolite as bedvolume.

The Biocarbone is a counter-current, granular media, aerobic filter with a water down-flow and an air up-flow. Its name is related to the earlier use of activeted carbon as matrix. Biofor is an abbreviation form from BIOlogical Oxygenated Reactor. Biofor is defined as an aerobic treatment using fixed biomass on a 1-5 mm granualr medium with an upflowing co-current of injected air and water (Paffoni *et al.* 1990).

These two processes primaryly differ in the fluid direction, namely a cocurrent in the Biofor process and a counter-current in the Biocarbone process.

The case study presented involves the use of simultaneous nitrification and denitrification (SND) with an upflow fixed bed, applying clinoptilolite as matrix. The combination of nitrification and denitrification in one single reactor has been described in the literature (Matsuda *et al.* 1987; 1991) The development of the process described has been conducted at the Section of Environmental Chemistry in the Royal Danish School of Pharmacy in Copenhagen, Denmark. During the summer 1992 the first pilot plant project was built as tertiary treatment step of slaughterhouse waste water.

5.8.1 Case Study; Simultaneous Nitrification and Denitrification (SND) as Tertiary Treatment Step, Using a Submerged Biofilter of Clinoptilolite

Introduction

The potential for using a simultaneous nitrification and denitrification (SND) upflow fixed bed reactor (UFBR) as a tertiary treatment step for removing nitrogen from waste water is examined in this case study. Clinoptilolite (with a grain size of 2.0-4.0 mm) was used as supporting medium for the bacterial growth. As indicated in Chapter 9 on ion-exchange, clinoptilolite is a natural zeolite which selectively sorbs NH_4^+. Furthermore the media has a porous surface, and has a high specific surface area, ideal for bacterial growth. The removal of the adsorbed ammonium from the zeolite by nitrifying bacteria allows regeneration of the zeolite surface and thus enables the same zeolite to be used repeatedly.

Thus the purpose of this case study is to explain the mechanisms and show the results of a single-stage simultaneous nitrification and denitrification (SND) reactor that biologically transforms ammonium-N to nitrogen gas, with ethanol as electron donor for denitrification.

Laboratory reactors were constructed (Fig. 5.17) of plexiglass tubes and used in three different runs using clinoptilolite as media. The loads conducted during the 3 different experimental runs, each of the duration of six months, are presented in Table 5.14.

Table 5.13 Denitrification rates depending of nature of medium surface in packed-bed coloumns

Nature of medium surface	***Media trade name***	***Specific surface area (cm^2/cm^3)***	***Denitrification rates g/m^3 d (at stated temp °C)***
High porosity corrugated sheet modules or dumped media	Kock Flexirings	2,13	45 (13°C), 53 (15°C), 54 (17°C), 136 (20°C), 115 (21°C), 61 (23°C),
		3,34	336 (27°C)
	Envirotech Surface	0,89	40 (10-23°C)
	Intalox Saddelse	4,66-8,99	216-417 (20°C)
	Rashig Rings	2,59	192-304 (25°C) 100-120 (20°C)
	Filter A and B	{1,65} Polprasert and {3,11} Park (1986)	47,8 → 495 (25-35 °C)
Low-porosity media			200-400 (20 °C)

After: EPA (1975) and Metcalf and Eddy (1979)

Outline of the 3 experimental runs:

Run 1: Waste water containing Ammonium-N, and COD (Chemical oxygen demand) in the influent waste water.
Run 2: Waste water containing Ammonium-N, Nitrate-N and COD in the influent. The Nitrate-N was introduced to the waste water to see if denitrification could proceed. The clinoptilolite media do not bind nitrate-N.
Run 3: Waste water containing Ammonium-N **without applying COD** to the waste water. This should prevent denitrification from proceeding, and the influent ammonium-N should be recovered as nitrate-N.

A pilot-plant to treat, tertiary stage, industrial waste water using clinoptilolite as media, were built, at the Island of Fyn in Denmark.

Results and discussion

As indicated in Chapter 9 clinoptilolite is a natural zeolite, which selectively sorbs NH_4^+. The ionbinding capacity is 1.3 meq/g media (Jørgensen *et al.* 1976). The efficiency of fresh support matrix, is therefore high until the ion-exchange capacity is used up. The removal of ammonium from the waste water will thus decline until the introduced nitrifying biomass becomes sufficient to convert all of the influent ammonium.

The step of biomass development is critical, because if the developed ratio of nitrosomonas and nitrobacter is out of balance, breakthrough of nitrite-N (NO_2^-) will appear in the waste water and inhibit the development of nitrosemonas. If the biomass concentrations of the two bacteria species are adjusted, the nitrifying efficiency is raised. Figure 5.18 show the removal efficiency during the 10 first days of biomass development, on previously unused clinoptilolite. Because of its ionbinding capacity, clinoptilolite will bind nearly all ammonium in the first few days. After the capacity is used up, a breakthrough of ammonium will appear until the concentration of nitrifying biomass will be able to convert some of the ammonium to nitrate.

When nitrate-N is developed during nitrification and suitable conditions (anoxic and carbon source) exist, denitrifying bacteria will be developed and nitrate can be converted to nitrogen gas.

Figure 5.19 show the first six days running of a 30 mg/l ammonium influent on a clinoptilolite reactor. The first three days of treatment, a breakthrough of nitrate was observed in effluent samples, because of lack of denitrifying bacteria. For first day, 5.2 mg/l of nitrate-N was found. This amount declined the following days, due to the rapid development of denitrifying bacteria.

Efficiency

Experimental RUN 1.

Simultanous nitrification and denitrification (SND) of waste water containing ammonium and a organic source, measured as COD, in the influent waste water, was conducted during RUN 1.

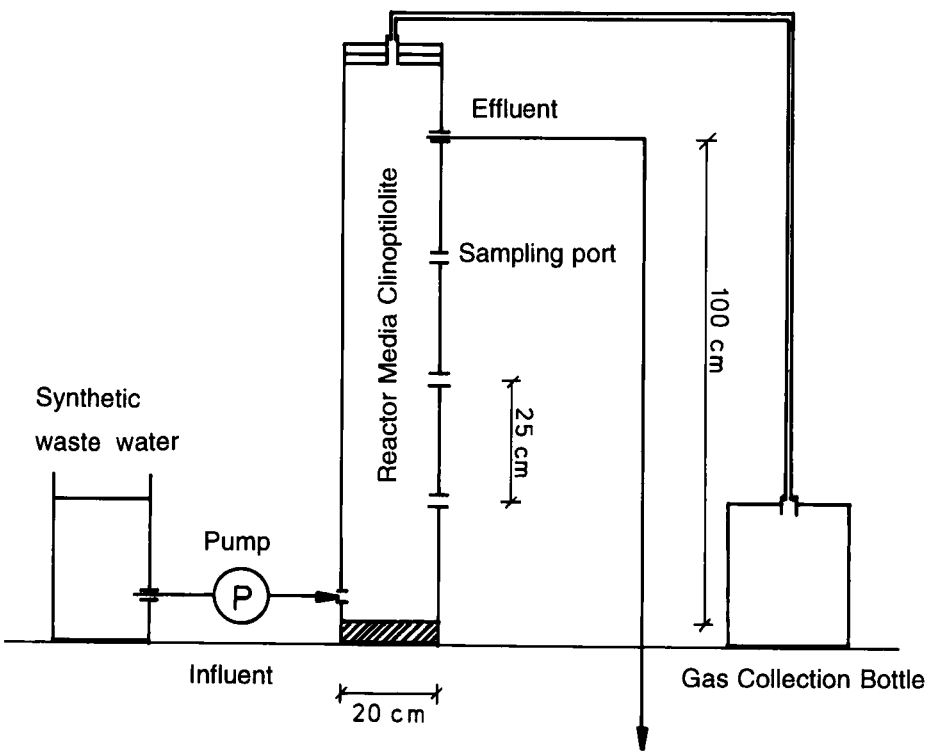


Figure 5.17 Laboratory reactor used to conduct experimental Runs 1 to 3.

Table 5.14 Loads conducted during Run 1 to 3 applying clinoptilolite as media.

	Run 1	*Run 2*	*Run 3*
NH_4^+ -N mg/l	30=> 1000	30	30=> 1000
NO_3^--N mg/l		up to 180	up to 70
COD inf. mg/L	120=>4000	800	No
pH	7,7 - 7,8	7,7 - 7,8	7,2 - 7,5
Flow L/hours	0,8 => 5,3	0,9 => 3,0	1,2 => 5,0
Oxygen conc. mg/l	2 - 3	2 - 3	2 - 3
Reactor media stones	clinoptilolite	clinoptilolite	clinoptilolite
Grain size mm	2 - 5	2 - 5	2 - 5
Void volume liter	8	8	8
Bed volume liter	30	30	30
Bed/void volume ratio	3,75	3,75	3,75
Average pore meq/g	0,44	0,44	0,44
N-ionbinding capacity	1,3	1,3	1,3
Reactor volume l	33	36	36
Reactor high	1,05	1,15	1,15
Reactor diameter m.	0,20	0,20	0,20
Intervals between samplingports. mm	250	250	250
Number of samplingports	4	4	4
SND occurred	YES	YES	NO

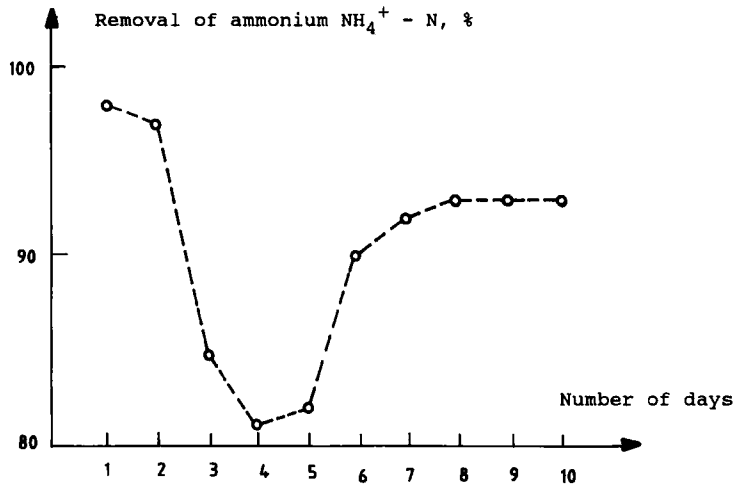


Figure 5.18 The removal efficiency during the first 10 days of biomass development on previously unused clinoptilolite.

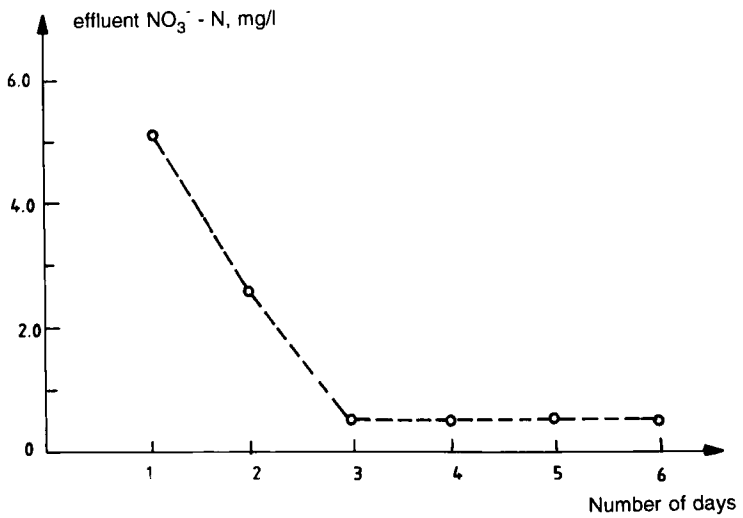


Figure 5.19 Application, during the first 6 days, of waste water containing 30 mg/l NH_4^+ - N, with a reactor of previously unused clinoptilolite. In the first three days, a breakthrough of nitrate was observed.

The removal efficiency of the simultaneous nitrification and denitrification (SND) and the effluent concentration of ammonium-N and nitrate-N were measured throughout this run. Table 5.14 summarizes the results of RUN 1 where an organic carbon source was applied to the system in stoichiometrically correct amounts. Table 5.15 shows some examples of mass balance for the simultaneous nitrification and denitrification. The amount of SND is equal to the amount of ammonium-N which in the reactor is totally converted via nitrate-N to nitrogen gas (N_2).

Figure 5.20 shows the relation between the loading of ammonium-N in kg N/m^3 voidvolume * day versus the simultaneous nitrification and denitrification (SND) reaction rate in kg N/m^3 * day. The SND reaction rate as bed volume, is calculated in kg N/m^3 * day, as:

$$SND(kgN/m3\ bed\ volume * day) = [N_{infl}] - [N_{eff}] * \frac{24}{HRT} * 8/30 \qquad (5.26)$$

where: HRT is the hydraulic Retention Time, in hours.

$N_{infl} = NH_4^+ - N_{infl} + NO_2^- - N_{infl} + NO_3^- - N_{infl}$

$N_{eff} = NH_4^+ - N_{eff} + NO_2^- - N_{eff} + NO_3^- - N_{eff}$

The factor 8/30 is the conversion factor between void volume and bed volume

The maximum amount of SND obtained during RUN 1 was 13.5 kg N/m^3 void volume * day (= 3.6 kg N/m^3 * bed volume * day) and the efficiency of SND is up to 99 % with a loading of up to 14 kg N/m^3 void volume * day under the following conditions: temperature 20 °C, DO 2-3 mg/l, pH 7.7-7.8 and stoichiometric addition of organic compound as ethanol to obtain denitrification.

Because Fig. 5.20 yields a linear relationship between loading and SND the theoretical maximum amount of SND is not found. The linear equation (5.27) can be obtained from Fig. 5.20 by linear regression. r^2 for the linear regression is 0.99.

$$SND = 0.97 * [N_{loading}] + 0.014 \qquad (5.27)$$

Data obtained from several sampling ports along the reactor presented in Fig. 5.17 yield the following equation estimating the simultaneous nitrification and denitrification, at a specific height "z" along the reactor in kg N / m^3 bed volume * day, knowing the SND at height "i".

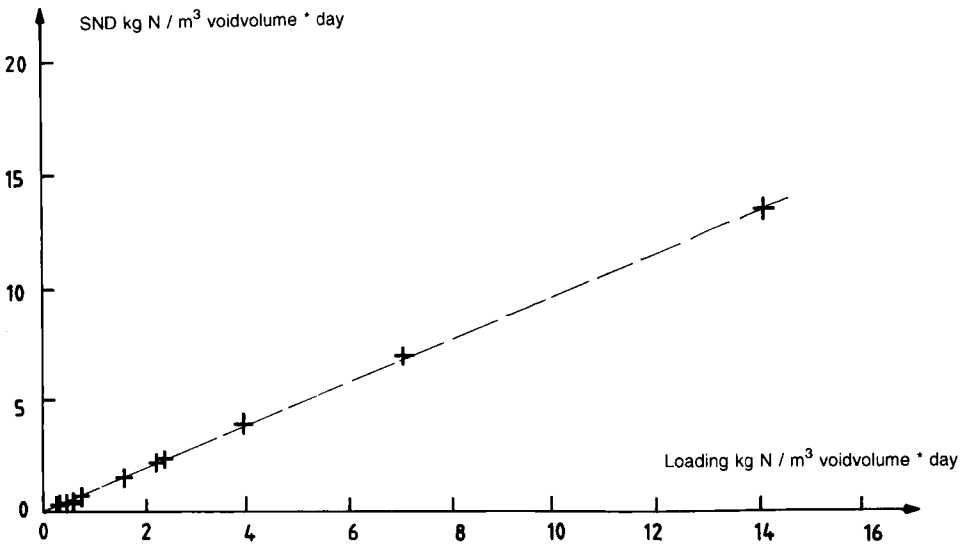


Figure 5.20 Relation between the loading of ammonium-N in kg N/m^3 bedvolume* day versus the simultaneous nitrification and denitrification (SND).

For a different substrate concentration k may be inserted in equation (5.28).

$$SND_{t=20^\circ C,h=z} = SND_{t=20^\circ C,h=i} * \theta^{k*z} * \frac{8}{30} \qquad (5.28)$$

For ammonium concentration in the influent below 100 mg/l NH^{4-} - N ; k=0.080 (s=18.1%). Between 100 and 500 mg/l NH_4^- - N; k=0.065; (s=29.2%) and between 500 and 1000 mg/l NH_4^- - N; k=0.034 (s=21.9%). s is the standard deviation.

Table 5.15 Mass balance of SND using clinoptilolite as media.

Input		***unit mg/l***		
NH_4^+ - N	30,0	100,0	500,0	1000,0
NO_3^- - N	0,2	0,4	0,7	0,4
Total N	30,2	100,4	500,7	1000,4
Output				
Removed by SND	*29,5	*96,8	*499,7	*957,4
Effluent water				
NH_4^+ - N	0,51	1,96	0,72	42,3
NO_3^- - N	0,22	1,65	0,23	0,7
Total	30,2	100,4	500,7	1000,4

* The removal by SND is found as the difference between influent and effluent waste water samples.

The relationship between the amount of SND removal in mg/l and the hydraulic retention time (HRT) is shown in Fig. 5.21, for the different feed concentrations shown in Table 5.15. From Fig. 5.21 it can be seen that the SND removal does not decrease by reduction of HRT with the ratio of HRT used during RUN 1. A greater flow through the reactor increases the daily removal capacity of the clinoptilolite medium, and the efficiency remains the same. This may be explained by considering that the surface that the bacterial population can occupy on the media is so large, that the maximum utility of the surface is not reached during the experiments. Therefore the use of a lower HRT and thus higher daily loading, yields a larger biomass, and therefore a higher capacity. Perhaps a higher flow can mechanically wash out the dead and older bacteria from the reactor and thereby also provide new surface for fast development of a fresh, new biofilm.

Table 5.16 Efficiency of SND during RUN 1, using clinoptilolite as media.

Flow L/h	*Retention time (HRT)*	*Loading kg N/m³ void-volume *day*	*Temp °C*	*DO mg/l*	*pH*	*NH_4^+-N Inf mg/l*	*NH_4^+-N Eff mg/l*	*Efficiency %*	*NO_3^--N Inf mg/l*	*NO_3^--N Eff mg/l*	*SND kg N/m³ void-volume *day*	*Efficiency %*	*SND Kg N/m³ support media *day*
1,2	6,66	0,108	20	2,0-3,0	7,7-7,8	30,0	1,49	95,0	0,10	1,30	0,098	90,7	0,026
2,0	4,04	0,178	20	2,0-3,0	7,7-7,8	30,0	0,22	99,3	0,10	2,12	0,165	92,7	0,044
3,6	2,22	0,324	20	2,0-3,0	7,7-7,8	30,0	0,51	98,3	0,20	0,22	0,319	98,5	0,085
0,9	8,88	0,270	20	2,0-3,0	7,7-7,8	100,0	1,44	98,6	0,40	4,63	0,255	94,4	0,068
1,5	5,33	0,450	20	2,0-3,0	7,7-7,8	100,0	1,61	99,3	0,10	3,95	0,425	70,4	0,110
2,0	4,10	0,585	20	2,0-3,0	7,7-7,8	100,0	0,70	98,6	0,70	6,25	0,412	92,5	0,182
2,5	3,25	0,738	20	2,0-3,0	7,7-7,8	100,0	1,38	98,0	0,10	1,65	0,683	96,9	0,412
5,3	1,50	1,600	20	2,0-3,0	7,7-7,8	100,0	1,96	98,0	0,40	0,49	1,550	99,9	0,598
1,5	5,33	2,250	20	2,0-3,0	7,7-7,8	500,0	0,67	99,9	0,46	0,49	2,248	99,9	0,598
2,6	3,03	3,960	20	2,0-3,0	7,7-7,8	500,0	0,71	99,9	0,42	0,30	3,956	99,9	1,052
4,7	1,70	7,059	20	2,0-3,0	7,7-7,8	500,0	0,72	99,9	0,70	0,23	7,055	99,9	1,877
0,8	10,03	2,392	20	2,0-3,0	7,7-7,8	1000,0	2,23	99,8	0,12	1,97	2,383	99,6	0,634
2,4	3,38	7,100	20	2,0-3,0	7,7-7,8	1000,0	5,51	99,4	0,26	0,66	7,059	99,4	1,878
4,7	1,70	14,118	20	2,0-3,0	7,7-7,8	1000,0	42,3	95,8	0,35	0,70	13,516	95,7	3,595

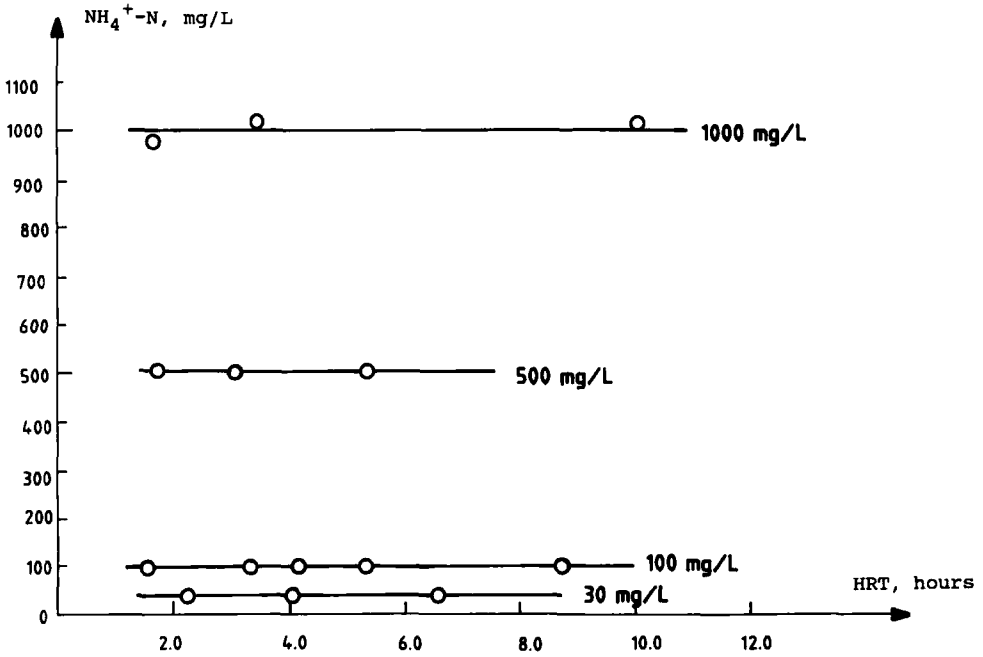


Figure 5.21 HRT vs. SND at different influent concentrations during RUN 1.

RUN 2.

The results from RUN 2 were obtained with waste water containing ammonium-N, nitrate-N and an organic compound source in form of ethanol, applied to the clinoptilolite reactor. The clinoptilolite medium does not bind nitrate as it does with ammonium.

The aim of this run was to observe if an addition of both ammonium-N and nitrate-N would be converted simultaneously to nitrogen gas.

Table 5.17 shows the efficiency of SND during RUN 2. The reactor was thus able to denitrify both the added amount of nitrate and the amount produced during the nitrification of the added ammonium.

The efficiency of Run 2 was comparable with Run 1, for the influent concentrations applied.

The conclusion is that it is possible to perform denitrification of nitrate added in excess of the nitrate produced by conversion of ammonium by nitrification.

Table 5.17 Efficiency of RUN 2.

Flow	Retention time (HRT)	Loading kg N / m^3 matrix per day	NH_4^+ - N Inf mg/l	eff mg/l	NO_3^- - N Inf mg/l	eff mg/l	SND kg N/m^3 matrix per day
1,2	6,67	0,059	30,0	0,28	30,8	7,59	0,051
1,2	6,67	0,043	30,0	0,26	14,1	0,10	0,042
3,0	2,67	0,501	30,0	0,88	177,9	105,3	0,244
1,2	6,67	0,501	30,0	1,41	99,0	30,5	0,377
0,9	8,89	0,150	30,0	1,29	70,1	0,10	0,095
1,2	6,67	0,055	30,0	0,12	27,3	0,20	0,055

Temp; 20° C, DO; 2-3 mg/l in bulk solution, pH = 7.7 - 7.8. Nitrite - N was not detected in the samples.

RUN 3

During RUN 3, ammonium-N was added to the clinoptilolite reactor *without* any continuous additon of a carbon source for denitrification. This should make it possible to recover the nitrate or nitrite produced during the nitrification process. The aim of this run was to be able to determine the efficiency of the nitrification process alone. For RUN 1 and RUN 2 the efficiencies of nitrification and denitrification are difficult to separate.

The clinoptilolite applied was fresh, to ensure that no organic compounds were left from previous experiments which would lead to uncertainty about to obtained results.

Nitrosomonas, can however (as the only bacteria in the biofilm) use CO_2 from the atmosphere to synthetize biomass (La Cour Jansen and Henze 1990).

Three times, during the 36 days of the test period, a shock-load of organic compound, in the form of ethanol, was added. On each occasion it resulted in a sudden development of the SND process.

Figure 5.22 shows the amount of ammonium-N nitrified, the amounts of produced nitrite-N and nitrate-N. Only between 25 and 30 percent of the loaded ammonium-N is nitrified. This low yield of nitrification is presumably due to the following two factors.

1) At low pH it is difficult to obtain sufficient biomass concentration to convert the amount of applied ammonium-N.

2) Because of the high nitrite-N and nitrate-N concentrations, the nitrification process can be inhibited by its own products.

At day 13 and 22 respectively 50 mg/l and 100 mg/l of COD (organic compound) were added to the waste water. At days 33, 34 and 35 1000 mg/l of COD were added.

Figure 5.22 indicates that the concentration of nitrite-N produced during the test period, was subject to great fluctuation. At the two first COD additions the concentration of nitrite-N increased and, therefore, at least some of the added COD were used to produce Nitrosomonas biomass.

The amount of produced nitrate-N increased when the nitrite concentration had reached its peak-value. This is natural because Nitrobacter (NO_2^- conversion to NO_3^-) is not developed until nitrite has been produced. Nitrite, however, both inhibits the nitrification and acts as a substrate for nitrobacter. The second step of the nitrification

At day 13 and 22 respectively 50 mg/l and 100 mg/l of COD (organic compound) were added to the waste water. At days 33, 34 and 35 1000 mg/l of COD were added.

Figure 5.22 indicates that the concentration of nitrite-N produced during the test period, was subject to great fluctuation. At the two first COD additions the concentration of nitrite-N increased and, therefore, at least some of the added COD were used to produce Nitrosemonas biomass.

The amount of produced nitrate-N increased when the nitrite concentration had reached its peak-value. This is natural because Nitrobacter (NO_2^- conversion to NO_3^-) is not developed until nitrite has been produced. Nitrite, however, both inhibits the nitrification and acts as a substrate for nitrobacter. The second step of the nitrification process (see Chapter 3) is therefore difficult to initiate.

A change in the biomass concentrations of both nitrosomonas and nitrobacter is therefore observed during the period of nitrate production. If no nitrite is produced, then nitrobacter is not developed due to a lack of the substrate that nitrobacter uses. On the other hand if the nitrite concentration is low, compared to the nitrate concentration, it was observed that both nitrosomonas and nitrobacter occurred in great amounts.

The applied shock-loads of COD seem, therefore, to have three important concequences in this investigation:

1) Maintenance of a fast formation of SND, during about 1 day.
2) Initiation of the development of *Nitrosomonas.*
3) Offering a carbon source for the synthesis of nitrobacter as soon as nitrite was available as substrate.

On days 34, 35 and 36 of the experiments, higher amounts of COD were added and a more persistent SND was introduced as during RUN 1. Both the amount of nitrite and nitrate therefore declined rapidly because there was a sufficient carbon source for the denitrification process.

Kinetics

A comparison of the nitrogen removal rate for the following submerged filters; the Biocarbone, Biofor and the SND processes, are outlined in Table 5.18 The kinetic rate of the SND process using clinoptilolite as matrix, was about three times higher than for the *Biocarbone* and *Biofor* processes, expressed as kg N / (m^3 matrix • day).

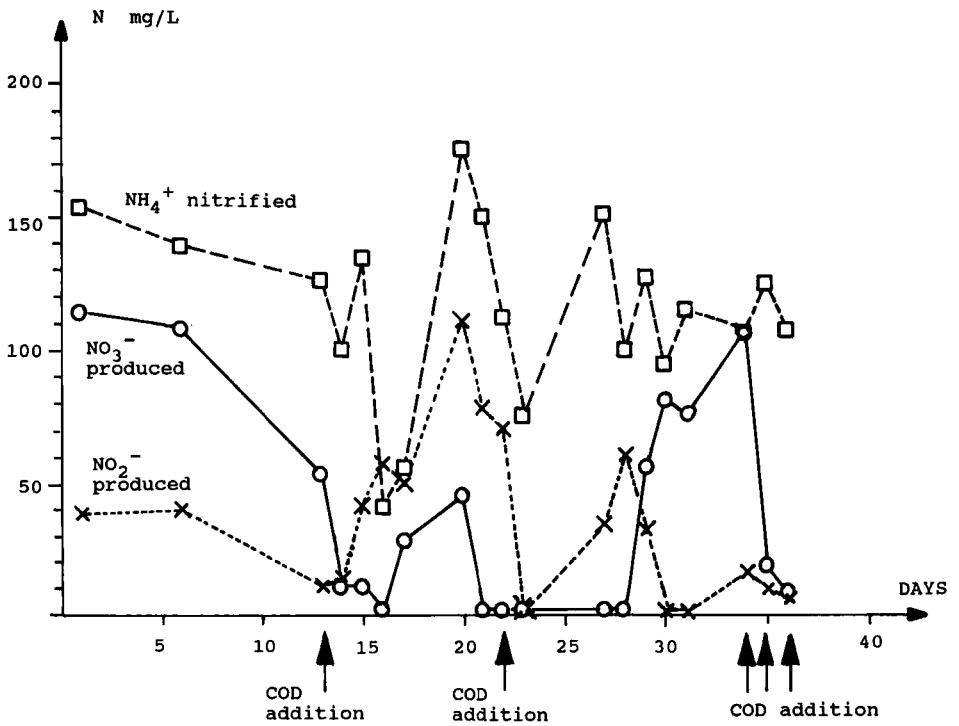


Figure 5.22 Results obtained during Run 3.

Table 5.18 The nitrification rate for the three submerged processes.

Process	***Maximum Nitrification rate kg N/m³ matrix * day***	***Reference***
Biocarbone (OTV)	0.74**	Rogella *et al.* (1990)
Biofor (Degremont)	0.75**	Paffoni *et al.* (1990)
SND*	1.7-3.4	Halling-Sørensen and Hjuler (1992; 1993)

* Only laboratory experiments

** Maximum loading 1,00 kg N / m^3 matrix per day.

Pilot-plant experiments

In Vantinge on the island of Fyen in Denmark, a pilot-plant has been built following the same concept as presented for experimental RUN 1. The only difference is that the carbon source is not ethanol, but endogeneous carbon from the waste water. Figure 5.23 shows a photo of the pilot-plant. The pilot-plant consists of 80 m^3 bedvolume of clinoptilolite distributed in six connected concrete bassins, with upflow waste water and air distribution.

The plant is used as the tertiary treatment stage for removal of nitrogen from slaughterhouse waste water. As secondary treatment stage, an activated sludge process unit for combined carbon oxidation and nitrification is used. After a secondary clarifier the waste water is pumped into the SND pilot-plant.

The total SND obtained during the first months of pilot-plant experiments were of the order of 0.45 kg N /m^3 bedvolume • day and 1.0 kg N /m^3 bedvolume • day.

Table 5.19 show the different influent and effluent concentrations found at the pilot-plant.

Table 5.19 Influent and effluent concentration of important parameters at the SND pilot-plant.

Parameter	NH_4^+-N mg/l	NO_3^--N mg/l	COD mg/l
Influent activated sludge treatment	820	30	5500
Influent SND tertiary treatment step.	450	10	1200
Effluent SND teriary treatment step.	30	3	50

Figure 5.23 Photo of the SND pilot-plant at Vantinge on the island of Fyen.

Figure 5.24 show a cross-section of a clinoptilolite stone. The porosity of the clay stone makes it possible to obtain aerobic and anaerobic conditions simultaneously. On the surface of the clinoptilolite stone oxygen diffuses into the biofilm and is used for the nitrification process. Ammonium is also diffusing towards the biofilm on the clinoptilolite stone. The ion-exchange ability of the stones binds ammonium on the surface (Jørgensen 1976; Haralambous *et al.* 1992) and nitrifying bacteria converts it to nitrate. The ion-exchange mechanisms may also play an role in the mechanisms, but is not totally clear.

The concentration of nitrate is highest at the upper layer of the biofilm which is most aerobic. Nitrate diffuses to the more anoxic areas in the lower part of the biofilm, where it is denitrified. Because of the concentration gradient a continuous diffusion to the center of the stone will take place.

Figure 5.25 is a micro-scope photo of the clinoptilolite stone covered by an SND bio-film.

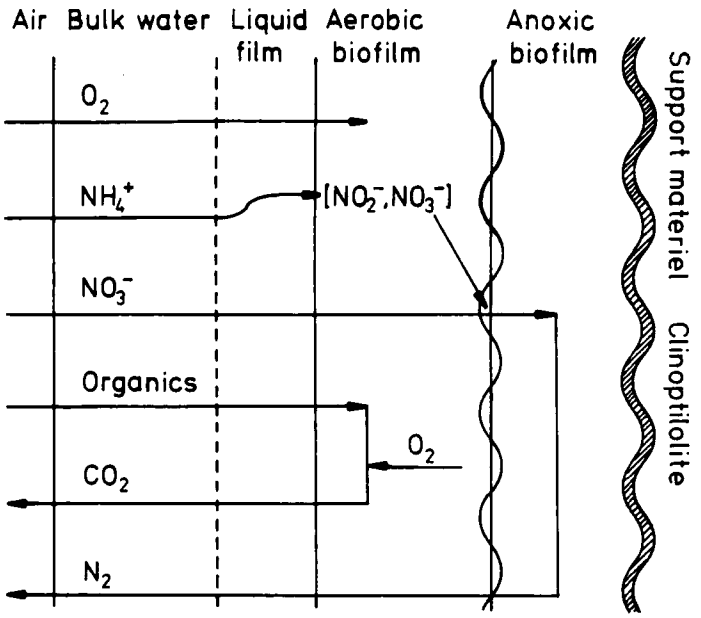


Figure 5.24 Cross-section of a clinoptilolite stone with aerobic and anoxic biofilm.

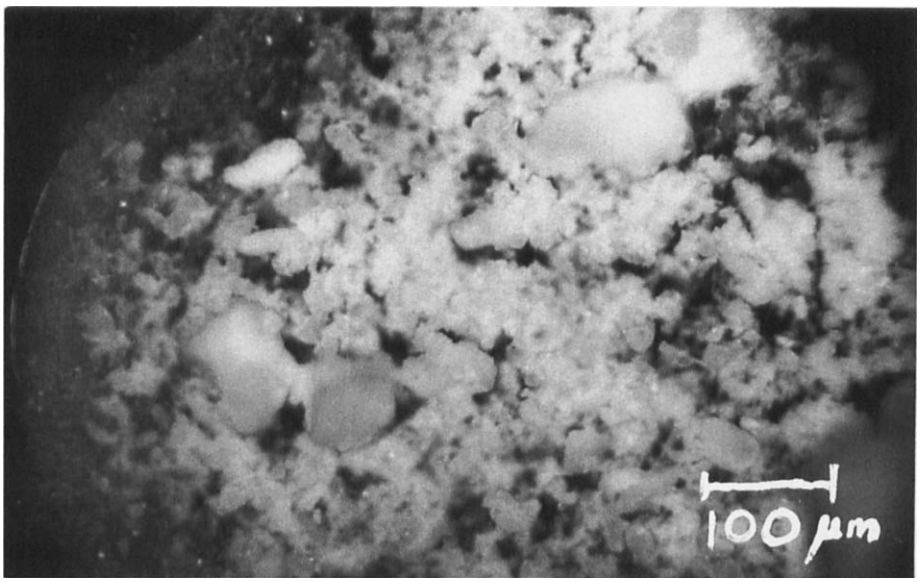


Figure 5.25 Clinoptilolite stone covered by SND bio-film, as seen under a micro scope.

Conclusions

The following conclusions can be made concerning the simultaneous nitrification and denitrification (SND) on basis of the experiments described:

1) Nitrification and denitrification occur simultaneously with different loadings.

2) The results show that a higher flow through the reactor permits greater daily loading with the same removal efficiency.

3) If no carbon source was added to the influent, nitrate and nitrite was recovered, showing that only nitrification occurs.

4) The SND is not able to treat organically bounded nitrogen.

5) The SND is relatively easy to start and fairly trouble free to maintain. In general the response of the reactor to changes is immediate and steady state conditions were apparently achieved quickly.

6) The faster nitrogen removal for this process, compared with suspended cultures, is partly due to a higher concentration of microorganisms, but it must be anticipated that some "additional effect" (i.e. ion-exchange) is needed to explain the high removal rate.

7) The pilot-plant experiments show an SND removal of 0.45 to 1.0 kg N /m^3 bed volume * day, while laboratory columns have shown up to 4 to 5 times higher efficiency.

8) This study has shown that a simultaneous nitrification and denitrification is a technologically feasible process for nitrogen control.