CHALMERS TEKNISKA HÖGSKOLA

CHALMERS UNIVERSITY OF TECHNOLOGY
GÖTEBORG
SWEDEN

Denitrification in a Non-nitrifying Activated Sludge System Employing Recirculation from aTertiary Nitrification Unit

ANN MATTSSON

Chalmers University of Technology
Department of Sanitary Engineering

Chalmers University of Technology
Department of Sanitary Engineering

Denitrification in a Non-nitrifying Activated Sludge System Employing Recirculation from a Tertiary Nitrification Unit

ANN MATTSSON

Dissertation no 13
ISBN 91-7197-546-2
ISSN 0346-718x
ISSN 0280-4581

Göteborg 1997

Address:
Department of Sanitary Engineering
Chalmers University of Technology
S-412 96 Göteborg, Sweden

Summary

A non-nitrifying, denitrifying activated sludge system has been demonstrated to give high denitrification rates and to be a useful component in a municipal wastewater treatment plant where partial nitrogen removal is required. One useful configuration is where clarified effluent from a non-nitrifying activated sludge system is nitrified in a tertiary nitrification unit and recirculated to the activated sludge system for denitrification.

Such a system was operated continuously in pilot scale, using influent wastewater to the Rya Wastewater Treatment Plant (in Göteborg on the West Coast of Sweden). Influent flow, recirculated flow and sludge withdrawal were controlled as in a full-scale application. Results from one year of operation indicate average effluent inorganic nitrogen concentrations in the order of 10 g N/m^3.

Recirculation of clarified effluent is a controlling factor. The variation of wastewater flow and quality influences design, operation and treatment results of such a system. Extrapolations from pilot plant data were made using simple models in order to investigate the effect of different circumstances on effluent nitrogen concentration and mass flow in a system based on recirculation of clarified and nitrified effluent to a denitrifying, non-nitrifying, activated sludge system. Effects of the capacities of the secondary settlers and the nitrification unit as well as denitrification limitations were investigated. Further, the effect of varying influent flow patterns (caused by the fraction of storm-water connected or by varying weather) on effluent ammonium concentration and nitrogen removal was illustrated. At low flows the quality and quantity of the carbon source will be crucial to nitrogen removal. At high flows the hydraulic capacities limit nitrogen removal (although effluent concentrations will still be low due to dilution). Further improvements can be made if the system is controlled and if sludge liquors are returned directly to the nitrification unit instead of to the head of the plant. Apart from the 1-2 g N/m^3 less nitrogen released, recirculation flows can be decreased. Little further improvement can be obtained by separate treatment of sludge liquor.

Denitrification rates in a denitrifying, non-nitrifying activated sludge system with a solids retention time (SRT) in the order of 3-5 days can be expected to be higher than in a nitrifying system where the SRT is in the order of 10 to 20 days or higher. This is mainly due to the higher respiration rates recorded in systems with a low SRT (and high growth rate). This was verified in the pilot plant. If the nitrate concentration was not limiting, denitrification rates were in the order of 10-15 g N/(kg VSS·h), although at high effluent nitrate concentrations. At low effluent nitrate concentrations under normal operation the average denitrification rate was 4.7 g N/(kg VSS·h). Denitrification rates were high not only where primary settled wastewater was introduced, but also in the deoxygenation zone where no new wastewater was introduced. Mass balances over the activated sludge system and estimates of nitrification in the activated sludge system due to seeding of nitrifiers from the NTF indicate nitrification, and thus denitrification, rates in the aerated tanks in the order of 0.5 - 2.5 g N/(kg VSS·h).

Preface

In this thesis a denitrifying non-nitrifying activated sludge system and its use for nitrogen removal at a municipal wastewater treatment plant is investigated. The experimental work was performed at the Rya Wastewater Treatment Plant, the regional wastewater treatment plant in Göteborg, and the theoretical work made possible by the participation of the Department of Sanitary Engineering at Chalmers University of Technology in the STAMP project. The STAMP-consortium in Göteborg consisted of several university departments and the Rya WWTP and was supported by NUTEK (Swedish National Board for Industrial and Technical Development).

The process discussed is one which has been, or will be, implemented at different wastewater treatment plants when nitrogen removal is required. However this investigation, and the experimental work involved, took place at the Rya Wastewater Treatment Plant. Naturally this influenced the investigation including pilot plant design and operation. Any reader who may be interested in the situation at the Rya WWTP is welcome to read Appendix A or refer to Balmér et al. (1997). Most pilot plant data can be found in Appendixes B and C. Some additional data used for simulations is included in Appendix D.

Acknowledgements

First I would like to thank my supervisor Peter Balmér who, apart from making sure that necessary resources for this project were available, spent hours on scientific guidance, discussion, encouragement and inspiration.

As the experimental work was part of developing a process for nitrogen removal at the Rya WWTP most of the staff of the Rya WWTP have been involved in some way or other. Many were involved in construction, operation and evaluation of pilot plants. However I would especially like to thank Christer Hansson, Lars-Erik Stefansson and Thomas Bjelke of the operations department who made sure that the pilot plant worked every day and Anette Svensson, Lena Bruto and Annika Tidlund who performed most of the sampling and analysis, thus making it possible to work out how it worked. Discussions with Peter Robinson, Doug Lumley and others have contributed significantly to working out what actually did happen.

During the course of the project several other people were involved during longer or shorter periods. The participation of Ann Rane during an important part of the pilot experiments was especially valuable and also enjoyable, thank you Ann. Doug Welch came to do his Masters Thesis and generously spent time on discussion and co-operation in the laboratory.

I would also like to thank the members of the board of GRYAAB for believing in and supporting new technology, and through them the water rate payers of the Göteborg Region.

Colleagues of the department of Sanitary Engineering at Chalmers University of Technology are thanked for friendship and for sharing their knowledge. A special thanks to professor Torsten Hedberg for engaging the department in the STAMP project.

I would like to thank the members of the STAMP project in Göteborg and also in the other Swedish STAMP-groups who, being from different fields but working with the similar projects, taught me different ways of seeing the same system. It has been inspiring to work with microbiologists, control engineers and sanitary engineers in a project where everyone knows most about something and nobody knows everything. The STAMP project has also given a valuable base for future co-operation between Swedish scientists and wastewater engineers. May this co-operation continue! And thus I thank NUTEK (Swedish National Board for Industrial and Technical Development) who, through financial support, made the STAMP project possible.

I would also like to thank my parents Pauline and Gösta Lyngå, generally for giving me two languages and specifically for correcting this one (although, of course, remaining mistakes are my own). Finally I must thank Björn and Ylva for sometimes sleeping when I planned to work and my husband Bengt for looking after them when they didn´t.

Contents

Glossary

AS	Activated sludge
BAF	Biological aerated filter
BOD$_5$	Biological oxygen demand. The amount of oxygen consumed by biological oxidation of organic matter in the sample during 5 days. Five days was chosen so as to cover the total oxygen demand of a package of water travelling down the Thames River before it reached the sea.
BOD$_7$	Biological oxygen demand. In Sweden the BOD-analysis covers 7 days in stead of five days. Doing so the analysis can be started any normal working day and finalised the same day the following week. Approximate conversion: $BOD_5 = 0.9 \cdot BOD_7$ (from constants and equations in Metcalf & Eddy, 1991).
COD	Chemical oxygen demand
DO	Dissolved oxygen
EFOR	Edb modell For Renseanlæg - a computer model for wastewater treatment based on the activated sludge model No. 1 (Henze et al., 1987)
DOC	Dissolved organic carbon
GRYAAB	Göteborgsregionens Ryaverks Aktiebolag - owners and operators of the Rya WWTP and the main tunnel system of the region. GRYAAB is owned by the municipalities of the Göteborg region.
HRT	Hydraulic retention time
MLSS	Mixed liquor suspended solids
NTF	Nitrifying trickling filter
NUTEK	Swedish National Board for Industrial and technical Development
ORP	Oxidation reduction potential
RAS	Return activated sludge
RBC	Rotating biological contactor
SBR	Sequencing batch reactor
STAMP	Control of Wastewater Treatment Systems - a research project initiated by NUTEK
SRT	Solids retention time
WW	Wastewater
WWTP	Wastewater treatment plant

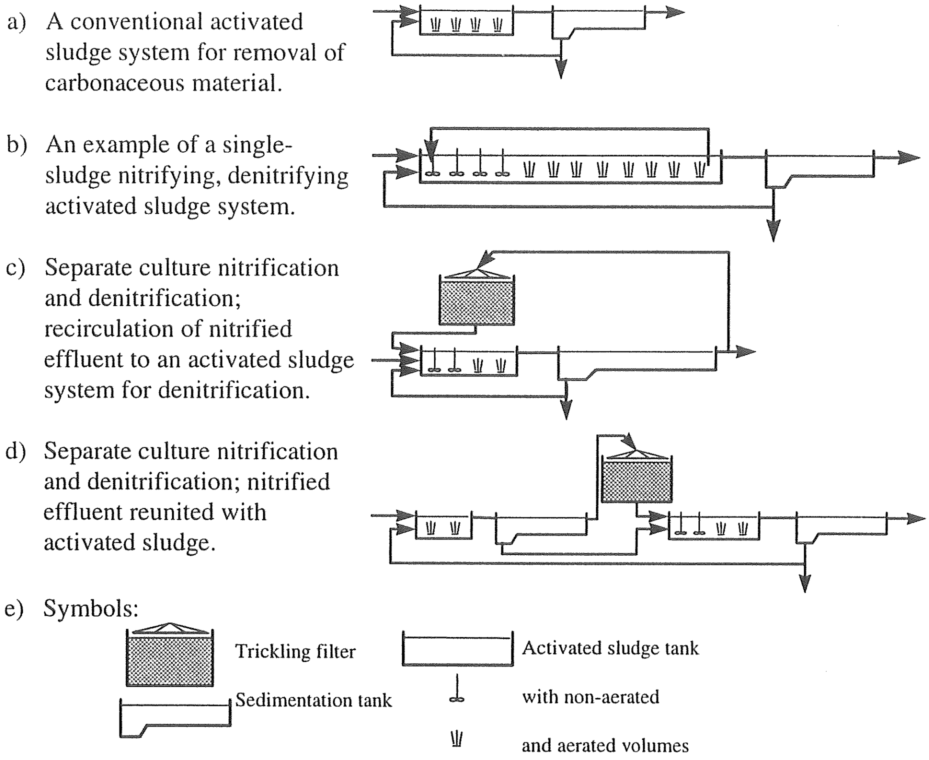
1 Introduction

A municipal wastewater treatment plant (WWTP) is generally designed and constructed to meet the standards set at the time. Land will often be reserved for future extensions in order to meet new standards or to cater for a rising population. A WWTP, built in a situation where the receiving water body is loaded with large amounts of untreated wastewater, may be constructed with screens, grit chambers and sedimentation tanks. This may improve the local environment vastly, but some years later the removal of soluble organic matter may be required. Biological treatment is then introduced, perhaps an activated sludge system or a trickling filter. Further demands on nutrient removal or removal of fine particles or bacteria as well as a changing situation with respect to sludge disposal or the price of energy and chemicals and so on cause treatment plants to be rebuilt or extended. When extending an existing municipal wastewater treatment plant several factors are to be considered, such as:

- Existing structures which may be used in the future plant. Existing structures may also complicate otherwise desirable solutions.
- The flow, temperature and composition of the wastewater to be treated. Forecasts of future wastewater composition and flow may be difficult to make.
- Site constraints. Many municipal wastewater treatment plants serving towns and cities are located close to the urban centres where land prices are high and new land for WWTP extension is scarce. Neighbours may object to location.
- Expected or stated demands on wastewater treatment results.
- Expected cost of chemicals, electricity, capital, operation etc.
- Expansion possibilities to cater for future additional loading and treatment requirements.

With many locally varying factors and varying existing wastewater treatment plants a wide variety of technical solutions for removing nitrogen may be appropriate. A common solution is single-sludge nitrification and denitrification (Figure 1.1 b). In this process nitrification (biological oxidation of ammonium to nitrate) and denitrification (biological reduction of nitrate to nitrogen gas) are performed in the same activated sludge system. The bacteria performing the different processes exist in the same bacterial culture, but operate in different environments, separated in space or time. The nitrifying environment is aerated and the denitrifying environment non-aerated. In these systems, especially when treating cold wastewater, large activated sludge volumes are needed in order to allow for the low growth rate of the nitrifying bacteria.

Another technical solution is the combination of a post-nitrifying trickling filter and a denitrifying, non-nitrifying activated sludge system. A WWTP with a non-nitrifying activated sludge system (Figure 1.1 a) can be extended for nitrogen removal if a trickling filter is built for nitrification. A wastewater treatment plant with trickling filters can, if nitrogen removal is demanded, be supplemented with a preceding non-nitrifying activated sludge system for denitrification. In order to remove nitrogen in such a system the nitrified effluent must be recirculated to the preceding activated sludge system. The recirculation flow will be an important controlling factor in nitrogen removal.



a) A conventional activated sludge system for removal of carbonaceous material.

b) An example of a single-sludge nitrifying, denitrifying activated sludge system.

c) Separate culture nitrification and denitrification; recirculation of nitrified effluent to an activated sludge system for denitrification.

d) Separate culture nitrification and denitrification; nitrified effluent reunited with activated sludge.

e) Symbols:

Figure 1.1 Principles of some processes where activated sludge systems are used.

By separating the biological processes of nitrification and denitrification into different bacterial cultures the environments may be specialised according to needs and performances of the different bacteria. In a separate, post-nitrifying, biofilm system the nitrifiers will be less threatened by competing heterotrophic growth than in a single-sludge system. In a non-nitrifying activated sludge system the organic loading can be higher than in a single-sludge system, where the slow growth of the nitrifiers must be allowed for. This is of special importance in regions where low wastewater temperature may limit nitrification. In an activated sludge system with a high growth rate the rate of oxygen respiration, and probably the denitrification rate, will be high.

However, in order to use the advantages of a denitrifying, non-nitrifying, activated sludge system, nitrate for denitrification must be produced in some other part of the system. This can be done in at least two different ways. One principle is a system where nitrified effluent from a post-nitrifying biofilm system is recirculated to a non-nitrifying, denitrifying activated sludge system, as discussed above (Figure 1.1 c). In another system the clarified effluent from a highly loaded activated sludge system is nitrified and then reunited with the activated sludge in a second, partially anoxic, activated sludge tank for denitrification (Figure 1.1 d).

The first process, post-nitrification in a trickling filter and recirculation of nitrified effluent to a non-nitrifying activated sludge system, is studied in this thesis.

One important difference between this system and a single-sludge pre-denitrifying system is that in this system all water containing nitrogen to be removed through denitrification is recirculated over the secondary settlers, whereas in a single-sludge pre-denitrification system nitrified mixed liquor is recirculated to the anoxic tanks for denitrification without sludge separation. This will influence the design of the secondary settlers. The combined effect of design of settlers, the capacity of the trickling filters and the variation of the influent ammonium concentration with flow warrants a flow dependent operational strategy. Since the effluent quality in this type of system will be dependent on operation and on influent wastewater quality variation, both of which are flow dependent, an integrated approach to design and operation is warranted.

The advantages of using a system including denitrifying, non-nitrifying activated sludge are dependent on wastewater quality and flow, demands on effluent wastewater quality and existing structures and local expansion possibilities. In this thesis aspects of design and operation of such a system are investigated.

The experimental work has been performed at the Rya WWTP, serving the Göteborg region (~770 000 equivalents of population) on the West Coast of Sweden. The WWTP is currently being extended for nitrogen removal using the process described.

2 Objectives and research strategies

The aim of this thesis is to investigate the use of a non-nitrifying activated sludge system for nitrogen removal in municipal wastewater treatment. The focus is set on a system where effluent wastewater is nitrified in a post nitrifying biofilm system and then recirculated to the non-nitrifying activated sludge system for denitrification.

This system consists of several components. However, post nitrifying biofilm systems in the form of trickling filters, rotating biological contactors, submerged filters and other systems have been studied in detail and implemented in many cases where nitrification of wastewater is demanded. Pilot plant experiments with nitrifying trickling filters have also been performed in conjunction with this study. The results of these confirm experience elsewhere and confirm that nitrifying trickling filters should work well under local conditions. As this aspect of the system is relatively well known it will not be discussed in detail in this thesis. The system demands large sedimentation capacities. However the principles governing sedimentation are not assumed to be different in this type of system from a regular activated sludge system. Therefore details of sedimentation will not be discussed in this thesis.

Two aspects of the system differ considerably from a single-sludge nitrifying denitrifying activated sludge system. Firstly, denitrification takes place in a non-nitrifying, and thus potentially highly loaded, activated sludge system. Secondly, wastewater containing nitrogen to be removed by denitrification has to be clarified before being recirculated to the nitrifying biofilm unit and back to the activated sludge system for denitrification. These aspects, and consequences of them, are considered in this thesis. The following assumptions concerning the non-nitrifying activated sludge system and the use of it are made:

- High denitrification rates can be expected in a non-nitrifying, relatively highly loaded, activated sludge system.
- In a highly loaded activated sludge system the respiration of oxygen and therefore probably of nitrate, will be high, not only where the raw wastewater enters the tank but in the entire activated sludge system.
- Partial nitrogen removal (at least 50-70 %) can be obtained in a system consisting of a non-nitrifying denitrifying activated sludge system and separate culture post-nitrifying units where nitrified effluent is recirculated to the activated sludge system for denitrification. The organic matter of the wastewater is used as the carbon source for denitrification.
- The need to recirculate effluent nitrified wastewater to the non-nitrifying activated sludge system demands extensive secondary settling area. At a wastewater treatment plant with significant flow variations due to storm-water being connected, the extra sedimentation capacity demanded for recirculation and for storm-water can be co-ordinated.
- The flow distribution to the wastewater treatment plant influences the removal of nitrogen and the nitrogen concentration of the effluent wastewater.
- Nitrogen removal may be further improved by process control.

- Due to seeding of nitrifiers from the trickling filter to the activated sludge system the "non-nitrifying" activated sludge system may not be completely non-nitrifying.

In this thesis the above assumptions are investigated in different ways. Literature is examined concerning oxygen and nitrate respiration rates in different systems and the circumstances governing these. These rates are compared with rates determined in pilot scale.

The pilot plant data are examined in order to determine denitrification rates in different parts of the activated sludge system and thus determine denitrification rates when raw wastewater is present, and the background denitrification rate where stored carbon is used. Experimental results are related to published data.

In order to demonstrate the capacity of the system to remove 50-70 % of the influent nitrogen the results of a pilot scale system operated for one year using municipal wastewater are analysed.

The results of pilot plant operation at the Rya WWTP during one year are investigated in order to determine the influence of flow and wastewater quality variation and flow dependent operation on results.

The consequences of the flow-dependent operation on nitrogen removal is related to the design of the system. Systems with for instance different sedimentation capacities or nitrification capacities will require different operational strategies. These partially flow-dependent limitations in combination with flow-dependent wastewater quality variations determine the average effluent ammonium concentration. This is illustrated using a simple steady state model on data representing the distribution of ammonium concentration and flow to the wastewater treatment plant.

The same model is used to illustrate the effect of dilution of wastewater from a collection system receiving more or less storm-water on the average effluent ammonium concentration.

The potential improvement obtained by greater control of recirculation, by the addition of a carbon source or by different handling of return sludge liquor is investigated using pilot plant data and a steady state model including estimated denitrification limitations and the recirculation of oxygen from the nitrifying trickling filter to the anoxic zone of the activated sludge tank.

Finally the results of the above investigations are evaluated to determine the usefulness of the system for nitrogen removal at a municipal wastewater treatment plant.

3 Denitrification

3.1 Introduction

This chapter gives a background to denitrification in a non-nitrifying activated sludge system treating municipal wastewater without the addition of an external carbon source. Denitrification occurs in many environments where the basic criteria for denitrification are fulfilled and denitrifiers are present. These criteria are discussed in section 3.2. The rate of denitrification will then be governed by the supply of the carbon source and nitrate which will control the abundance of denitrifiers. These conditions vary dramatically between different systems as will be illustrated in section 3.3. One system where high denitrification rates are to be expected is a highly loaded activated sludge system. In practice, however, the possibility of using a highly loaded activated sludge system for nitrogen removal at a municipal wastewater treatment plant is limited by the necessity to nitrify the wastewater prior to denitrification.

Typically about two thirds of the nitrogen content of the influent wastewater arrives at the wastewater treatment plant in the form of ammonium, having been transformed from urea during transport from the dominating source, the human being, to the wastewater treatment plant. The remaining nitrogen is mainly included in soluble or particulate organic matter. Some of the nitrogen in the organic matter is released as ammonium through biological processes in the wastewater treatment plant; the amount is mainly governed by conditions during biological wastewater and sludge treatment. On the other hand some of the organic matter and ammonium is removed from the wastewater due to biological and surface chemical processes by which it is included in the biomass or in sludge produced during mechanical or chemical treatment. In a typical municipal wastewater treatment plant, with primary and secondary treatment, but without any special provision for nitrogen removal, 15-30 % of the influent nitrogen is removed with the sludge. The main part of the remaining nitrogen leaves the plant in the form of ammonium or nitrate which, being soluble, is not removed by mechanical separation such as sedimentation, flotation or filtration. The effluent wastewater also contains a small amount of particulate matter (the amount depending on the efficiency of separation), and some soluble organic matter, both of which contain nitrogen.

In biological nitrogen removal ammonium is transformed to nitrate by autotrophic nitrifying bacteria and then transformed to nitrogen gas by heterotrophic denitrifying bacteria, Figure 3.1. The nitrifiers need oxygen and consume alkalinity. They are autotrophic bacteria with a relatively low specific growth rate. Denitrification is performed by heterotrophic bacteria using nitrate instead of oxygen for respiration. The denitrifiers need organic carbon and, if nitrate is to be used for respiration, absence of oxygen is essential. If oxygen is present it will be used in preference to nitrate.

Nitrification is performed by autotrophic bacteria. Ammonia oxidisers (often called Nitrosomonas) oxidise ammonium to nitrite:

$NH_4^+ + 3/2\ O_2 \rightarrow NO_2^- + 2\ H^+ + H_2O$

Nitrite oxidisers (often called Nitrobacter) oxidise nitrite to nitrate:

$NO_2^- + 1/2\ O_2 \rightarrow NO_3^-$

Denitrification is performed by heterotrophic bacteria:

$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$

The balanced equation will vary depending on the carbon source.

Figure 3.1 Nitrification and denitrification.

In municipal wastewater treatment a common method of nitrogen removal is to combine nitrification and denitrification in the same bacterial culture, separating the conditions needed for nitrification and for denitrification in time or space. Combining nitrification and denitrification in one activated sludge system (a single-sludge system) gives some advantages. No sludge separation is needed between the processes, and the carbon source of the wastewater can be utilised for denitrification. However, design and operation of the system will be limited by the necessity to harbour both nitrifiers and denitrifiers in the same bacterial culture. As this is the most common process for nitrogen removal in municipal wastewater treatment plants, and since many relevant observations on denitrification in activated sludge systems concern single-sludge systems, it will be discussed in section 3.3.3. Denitrification in activated sludge systems in general is discussed in section 3.3.4 and some implications concerning separate sludge denitrifying activated sludge systems are presented.

Different possibilities of including a separate sludge denitrifying non-nitrifying activated sludge system in a municipal wastewater treatment plant are discussed in section 3.4.

3.2 Influence of the environment on denitrification

3.2.1 Introduction

Some heterotrophic bacteria can use nitrate as electron acceptor in the absence of oxygen, thus reducing nitrate to nitrogen gas:

$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$

Each step in the reduction of nitrate to nitrogen gas is catalysed by a reductase. The reductases become derepressed when the bacteria are subjected to oxygen deficiency. Factors influencing repression and derepression of the different reductases may influence the course of denitrification and thus control the end product of the process.

Bacteria capable of denitrification are biochemically and taxonomically diverse (Knowles, 1982). Certain bacteria can perform the entire reduction of nitrate to nitrogen gas, while others only reduce nitrate to nitrite or perform a few steps of the total denitrification process. (Ingraham, 1981). This means that bacteria not able to perform the whole chain of denitrification may still play an important role in denitrification in a natural or constructed system with a wide variety of bacteria present. Thus the relative importance of different bacteria to denitrification in a mixed population is difficult to determine from laboratory studies using pure cultures. Other problems which arise when trying to determine the relative importance of different bacteria to denitrification can be connected with the methods used to isolate strains (which may not suit all denitrifiers) or the temperature of incubation (Ingraham, 1981).

In wastewater treatment where a wide variety of bacteria are constantly brought to the plant with the wastewater a large number of denitrifying bacteria are available to the process. Many of the bacteria in wastewater are able to denitrify and the aim of design and operation of a denitrifying process is to create an environment in which basic environmental criteria for denitrification are satisfied and a bacterial culture with a high capacity of denitrification using the carbon source present develops.

The influence of individual environmental conditions on denitrification is discussed. Most of the references are to wastewater treatment systems but when useful, references are made to denitrification in other systems.

3.2.2 Influence of environmental conditions on denitrification

The **nitrate** concentration may limit diffusion of nitrate to the denitrifying sites in soils and sediments (Knowles, 1982). However, in suspended cultures, such as activated sludge, the denitrification process for many practical purposes can be regarded as a zero-order process with respect to nitrate-nitrogen concentration (Christensen and Harremoës, 1977). According to the review of Christensen and Harremoës the saturation constant is approximately 0.1 mg NO_3-N/l.

Oxygen influences denitrification in two different ways. Firstly oxygen deficiency derepresses the synthesis of nitrogen-reducing reductases and secondly when oxygen is present oxygen and not nitrate is used as the terminal electron acceptor.

In sediments the oxygen concentration influencing denitrification may be governed by a complex interaction of oxygen diffusion, consumption of oxygen by nitrifiers and heterotrophic organisms, and oxygen production by algae (Knowles 1982, Koike and Sørensen, 1988). Investigations of the effect of oxygen consistently indicate that, in both freshwater and marine systems, an oxygen concentration of 0.2 mg/l or less is required for denitrification in the water or sediment (Seitzinger, 1988). Crucial to the bacterium is the environment to which it is actually exposed. The oxygen concentration in this environment may well vary from that of the bulk liquid. This is especially important in systems involving solid phases, such as soils, sediments and biofilms.

Reviewing literature on denitrification in wastewater treatment Christensen and Harremoës (1977) found that in suspended cultures the oxygen concentration should be below 0.5 mg O_2/l whereas in biofilm reactors bulk concentrations of 1-2 mg O_2/l seemed not to affect denitrification. Hagedorn-Olsen et al.(1994) have since shown denitrification in a biofilm reactor to be negatively affected by the bulk dissolved oxygen concentration at oxygen concentrations down to about 0.5 mg O_2/l. Lie and Welander (1994) showed denitrification rates in activated sludge to increase with decreasing oxygen concentration (measured as ORP - oxidation reduction potential) at oxygen concentrations well below 0.5 to 1.0 mg/l.

Krul (1976) showed that increased denitrification took place in activated sludge flocs and flocs of a pure culture of denitrifying bacteria at bulk liquid oxygen concentrations below 1 - 1.5 mg O_2/l whereas in a pure culture of dispersed denitrifying bacteria the oxygen concentration had to be below 0.1 mg O_2 /l for increased denitrification to occur. However even above these levels some denitrification occurred though at lower rates than those recorded at low oxygen concentrations. This indicates that denitrification can take place in the (anoxic) interior of flocs although oxygen is present in the bulk liquid. In another study (v Münch et al. 1996) denitrification at higher bulk oxygen concentrations was interpreted as an indication that aerobic denitrification can be a significant process for nitrogen removal.

Experiments performed by Simpkin and Boyle (1988) indicate that inhibition of enzyme activity rather than repression of enzyme synthesis, must be the most important effect oxygen has on denitrification in activated sludge. The authors conclude that anoxic zones in activated sludge systems only have to be designed for denitrification and not for derepression of denitrifying enzymes.

The ORP is closely related to the oxygen concentration and should thus give a good indication of the potential of denitrification. Several problems are connected with measuring the ORP. However, since on line ORP instrumentation is relatively cheap and robust and since the signal gives a sensitive response to conditions in the activated sludge tank, including nitrate and oxygen concentrations, the parameter is of interest to many operators and researchers of nitrogen removal in activated sludge systems and has been used with some success to indicate the state of the system and for process control (Lie and Welander, 1994, Lo et al., 1994, Rehmann, 1993).

As well as influencing the rate of denitrification oxygen may influence the course of denitrification if some reductases are repressed by oxygen and others are not. The most discussed consequences are denitrification to nitrite or N_2O instead of to N_2. There are some differences in sensitivity to oxygen and to time needed for derepression between different reductases. Nitrite reductase requires somewhat longer for derepression than does nitrate reductase (Knowles, 1982). This can cause a nitrite build-up during transient conditions when denitrification commences. The later reductases in the denitrification sequence are somewhat more oxygen sensitive than are earlier reductases and can take longer to undergo derepression. The increased O_2-sensitivity of later reductases can cause N_2O to be a dominant product (at least periodically) when the O_2 -concentration varies spatially, for instance in soil (Knowles, 1982). An investigation by the Swedish Environmental Agency (Naturvårdsverket, 1994) showed the release of N_2O from activated sludge systems to be very low. On average 0.15 % of the influent nitrogen was released as N_2O in the wastewater treatment plants investigated. This N_2O was mainly released during aeration, and mainly when the dissolved oxygen concentration was low. Recent modelling and experimental work (Schulthess and Gujer, 1996) also indicate the release of N_2O from denitrifying activated sludge to be low (0.021 - 0.072 % of influent nitrogen).

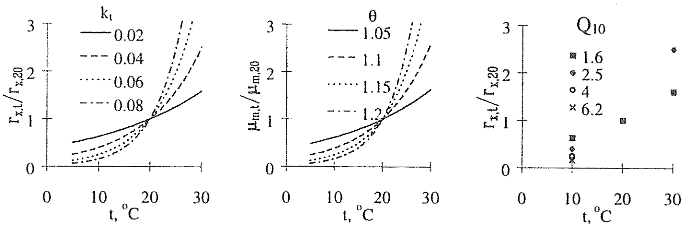
Temperature influences the rate of denitrification but may also influence the course of denitrification. In soil there appears to be a marked temperature dependence, whereas in aquatic sediments the denitrification rate varied surprisingly little with temperature (Knowles, 1982). In sediments diffusion may be rate limiting, thus masking the influence of the temperature dependency of the organisms. However, denitrification rates in sediments have a tendency to increase with temperature. In the material reviewed by Seitzinger (1988) temperature dependencies (as Q_{10}, see Figure 3.2) typically varied between 0 and 2.6 (with examples of Q_{10} from at negative values to about +4). The course of denitrification in soil is reported to be influenced by temperature. At high temperatures and very low temperatures (0 - 5 ° C) a relatively high mole fraction of N_2O and NO is reported (Knowles, 1982).

However, in the temperature regions normally relevant for wastewater treatment the influence of temperature on the rate of denitrification is more important than its influence on the course of denitrification. In wastewater treatment the influences of temperature on denitrification rate are of several kinds. Temperature influences the rate of denitrification and growth of denitrifiers in a culture (short term response) and the selection of the bacterial culture can be influenced (long term response). The short term changes of denitrification rate and growth rate follow a pattern which is well known for bacteria and can be described by the Arrhenius equation and other temperature expressions (Figure 3.2).

$$r_{x,t} = r_{x,20} \cdot 10^{k_t \cdot (t-20)}$$

$$\mu_{m,t} = \mu_{m,20} \cdot \theta^{(t-20)}$$

$$Q_{10} = \frac{r_{x,t+10}}{r_{x,t}}$$



Symbol	Explanation	Unit
t	Temperature	° C
$r_{x,t}$	Denitrification rate at the temperature t ° C	
$\mu_{m,t}$	Growth rate of denitrifiers at the temperature t ° C	
k_t, θ, Q_{10}	temperature constants	

Figure 3.2 Variation of denitrification rate with temperature (expressions used by Christensen and Harremoës (1977) and Lewandowski (1982)).

In the material reviewed by Christensen and Harremoës (1977) the temperature constant k_t typically ranged from 0.05 to 0.07 for suspended cultures and 0.02 to 0.03 for attached cultures indicating a stronger temperature dependence for suspended than for attached cultures. Lewandowski (1982) performed laboratory denitrification tests with suspended cultures using different carbon sources and found that the carbon source used for denitrification did not affect the temperature dependency of denitrification. The temperature constant (k_t) was around 0.03 whether methanol, acetone, acetic acid or no added carbon source at all was used (whereas the absolute value of the denitrification rate varied between the different carbon sources). Christensen and Harremöes (1978) on the other hand state different temperature dependencies for different carbon sources, Figure 3.10. For single sludge systems using raw wastewater and endogenous respiration the temperature constant (k_t) is set at 0.06 and 0.08 respectively indicating a greater temperature dependency for endogenous respiration than when raw wastewater supplies the carbon for denitrification. In suspended separate sludge with methanol as the carbon source a temperature constant of 0.05 was reported.

However, often what is measured is the behaviour of one particular bacterial culture, which may have been selected at 20 ° C and is then exposed to a temperature change (Dawson and Murphy, 1971, Dawson and Murphy, 1973, Lewandowski, 1982, Sutton et al. 1975). In a wastewater treatment plant, where the wastewater temperature varies gradually according to the season or rapidly during a storm, the situation may be different. Seasonal changes in a wastewater treatment plant may be difficult to relate to temperature since they may be masked or accentuated by other seasonal changes in wastewater quality or in plant operation. Christensen and Harremoës (1977 and 1978) discuss differences between long-term and short-term temperature responses of denitrification. The importance of the temperature at which the sludge is selected was demonstrated by Halmø and Eimhjellen (1981) who selected sludges for denitrification at 5 ° C and at 20 ° C, Figure 3.3. At 5 ° C denitrification rates of the sludge selected at 5 ° C were about 3 times that of the sludge selected at 20 ° C. At temperatures up to 15 ° C the rate of the low temperature sludge was higher than or equal to that of the high temperature sludge, but above 15 ° C the rate of the high temperature sludge was higher than that of the low temperature sludge. The temperature constants (k_t) of the low temperature sludge and the high temperature sludge separately were 0.05 and 0.04 respectively whereas the temperature effect in a system using the low temperature sludge at low temperatures and the high temperature sludge at high temperatures would be 0.02 corresponding to a Q_{10} value of 1.6. Also in other respects, including settling characteristics and effluent nitrate concentration, the sludges differed.

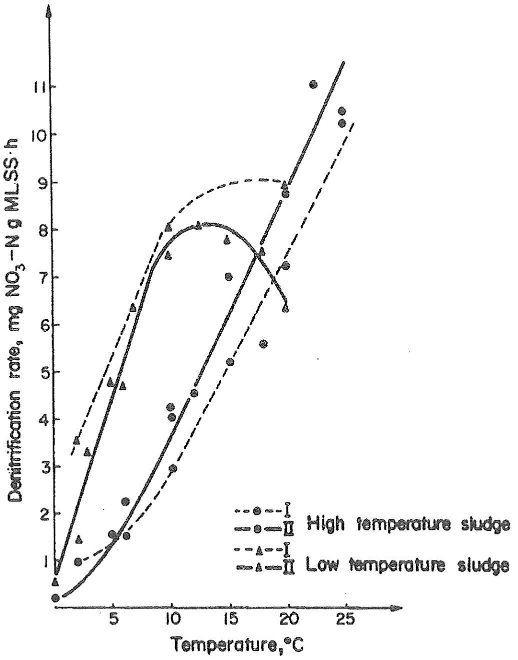


Figure 3.3 Variation of denitrification rate with temperature in activated sludge. Experimental results of Halmø and Eimhjellen, 1981.

Olezkiewsicz and Bergquist (1988), operating laboratory sequencing batch reactors for nitrification and denitrification, experienced denitrification rates dropping sharply when temperature was decreased from 5 - 7 ° C to 2 ° C . However, at 2 ° C the nitrification efficiency deteriorated drastically, and in order to obtain nitrification the food-to-micro-organism ratio was halved, giving a solids retention time of approximately 60 days (compared with 20-35 days used at the other temperatures). Consequently denitrification rates at 2 ° C would also be influenced by low organic loading (in the case of increased solids retention time) or by a deficient supply of nitrate (if the solids retention time is not increased). In the experiments of Halmø and Eimhjellen (1981) and of Oliezkiewsicz and Bergquist (1988) the temperature of the wastewater was controlled artificially, thus avoiding the effects of correlations or relations between temperature and other parameters relevant to denitrification. This will be further discussed in relation to the data of the present investigation.

As is the case for other heterotrophs, the availability of electrons in **organic carbon** compounds is one of the most important rate controlling factors for denitrifiers. The carbon source used will determine the stoichiometry of denitrification, controlling the amount of carbon source needed, the biomass production and the carbon to nitrogen ratio needed, Table 3.1. The quality and quantity of the carbon source may also control the course of denitrification.

Table 3.1 Equations of denitrification.

Carbon source	Equation	Reference
Methanol	$6NO_3^- + 5CH_3OH \rightarrow 3N_2 + 5CO_2 + 7H_2O + OH^-$	Lewandowski, 1982
Ethanol	$12NO_3^- + 5C_2H_5OH \rightarrow 6N_2 + 10CO_2 + 9\ H_2O + 12\ OH^-$ $97NO_3^- + 50C_2H_5OH \rightarrow$ $46N_2 + 5\ C_5H_7NO_2 + 75CO_2 + 84H_2O + 97\ OH^-$ (incl. assimilation)	Hamon and Fustec, 1991
Acetic acid	$8NO_3^- + 5CH_3COOH \rightarrow 4N_2 + 10CO_2 + 6\ H_2O + 8OH^-$ $34NO_3^- + 50CH_3COOH \rightarrow$ $12N_2 + 10\ C_5H_7NO_2 + 50CO_2 + 48H_2O + 34\ OH^-$ (incl. assimilation)	Hamon and Fustec, 1991
Acetone	$16NO_3^- + 5CH_3COOH \rightarrow$ $8N_2 + 15CO_2 + 7H_2O + OH^-$	Lewandowski, 1982
Methane	$5CH_4 + 8NO_3^- + 8H^+ \rightarrow 5CO_2 + 4\ N_2 + 14H_2O$	Harremoës and Christensen, 1971
Carbo-hydrate	$5CH_2O + 4NO_3^- + 4H^+ \rightarrow 5CO_2 + 2\ N_2 + 7H_2O$	Hiscock et al. 1991
Disaccharide	$5C_6H_{10}O_6 + 24NO_3^- \rightarrow$ $6CO_2 + 12\ N_2 + 18H_2O + 24HCO_3^-$	Socher and Gläser, 1992
Glucose	$C_6H_{12}O_6 + 2.8\ NO_3^- + 0.5\ NH_4^+ + 2.3\ H^+ \rightarrow$ $3.5\ CO_2 + 1.4\ N_2 + 6.4\ H_2O + 0.5\ C_5H_7NO_2$	Matějů et al., 1992
"Typical" organic matter	$C_5H_9NO + 3.36NO_3^- + 3.92H^+ \rightarrow$ $3.2CO_2 + 1.68N_2 + 3.92H_2O + 0.36C_5H_7NO_2 + 0.64\ NH_4^+$	Matějů et al., 1992
Cellulose	$5(C_6H_{10}O_5)n + 24NO_3^- \rightarrow 6nCO_2 + 12nN_2 + 13nH_2O + 24\ nHCO_3^-$	Matějů et al., 1992

In natural systems the carbon source consists of decomposing or decomposed organic matter. In sediments organic matter settles from the water column and forms a bank of organic matter available to heterotrophic organisms. The rate of denitrification varies with the supply of biologically degradable organic matter and nitrate. In natural environments, like soils and sediments, it is clear that the denitrifying activity is related to the organic carbon content. The carbon source can also affect the course of denitrification in natural systems. Knowles (1982) mentions several observed effects of the carbon source on denitrification. Different organic compounds, though giving the same denitrification rate, may give different mole fractions of N_2O of the products, suggesting that they may exert differential effects on the reductases involved. With abundant carbon and complete anaerobisis reduction proceeds significantly towards NH_4^+ rather than to gaseous products. Finally, under some conditions there is no effect of carbon addition, indicating that this factor is not rate limiting.

In wastewater treatment the carbon source may consist of wastewater organics (internal carbon source) or of some external carbon source introduced specifically in order to enhance denitrification. Raw wastewater is the most common internal carbon source, but the wastewater organics may also be treated before use in order to make them more concentrated, more readily available to denitrification or to facilitate handling.

External carbon sources may be commercial products such as methanol, ethanol, acetic acid or hydrolysed starch or industrial waste products such as whey or molasses. The main objectives when using an external carbon source are to increase the denitrification rate, to minimise sludge production in the activated sludge system and to be able to control the system. Methanol is traditionally a widely used commercial product yielding high denitrification rates and low sludge production. In order to utilise methanol efficiently, however, a specialised bacterial culture must develop. Hypomicrobium sp. are enriched in systems where methanol is used for denitrification (Nurse, 1978, Timmermans and Van Haute, 1983). In a full scale application Nyberg et al. (1992) reported this enrichment process taking one month at 10 ° C. Hallin et al. (1996) showed adaptation of activated sludge to methanol in the order of one month whereas the sludge could use acetate for denitrification without delay. This must be considered if temporary use of methanol is contemplated. Due to the toxicity of methanol some other carbon source may be preferred for denitrification in drinking water production. Ethanol, acetic acid and sucrose are mentioned for denitrification of groundwater contaminated with nitrate (Hiscock et al., 1991). The two carbon sources authorised by the French Ministry of Health for use in drinking water denitrification are ethanol and acetic acid (Hamon and Fustec, 1991).

If waste products are used as the carbon source at wastewater treatment plants they must be cheap, concentrated, uncontaminated, available and preferably not involve much transport. Waste products from local food industry, such as wine growing, and the production of wine, beer and juice may be economically advantageous (Friedrich et al., 1995). A problem with some waste products, such as slaughter-house waste is a high content of nitrogen. A Canadian survey (Monteith et al., 1980) found 27 out of 30 tested wastes to give denitrification rates equal to or greater than those obtained with methanol, using sludge from a single-sludge nitrogen removing pilot plant where methanol was used as the external carbon source. Among the useful wastes were distillery fusel oils, centrate from wine sludges, brewery wastes and wastes from vegetable processing industries.

Internal carbon sources, however, have the advantage of not adding extra matter to the treatment plant and thus not increasing the sludge production. Henze (1991) states that "in wastewater treatment the use of external carbon sources - in fact methanol - has been overrun by technologies that use carbon sources present in the wastewater - internal carbon sources". The wastewater may be used directly for denitrification in which case a process is constructed so as to make the best use of the influent wastewater carbon, for instance avoiding aeration prior to denitrification. Single-sludge activated systems for nitrogen removal utilising the influent carbon source in different ways are discussed in section 3.3.3.

One way of making use of the influent carbon source is to introduce an efficient separation of wastewater organics before the single-sludge nitrifying denitrifying activated sludge unit. This reduces the loading on the system and thus reduces the volume needed for nitrification. If the wastewater organics then can be treated so as to become readily available and afterwards can be returned to the system exactly when and where they are needed for denitrification, a compact system which does not need an external carbon source may be created (Karlsson and Smith, 1991).

The denitrification process is less sensitive to **pH** than is the nitrification process. In a pH-range from about 7-7.5 to 8 or 9 denitrification is generally stated to be relatively unaffected by pH (Christensen and Harremoës, 1977, Bryan, 1981, Knowles, 1982, Hiscock et al. 1991, EPA, 1993). As this is the range normally encountered in biological wastewater treatment, and as the supply of nitrate for denitrification is normally dependent on the more sensitive nitrification process, sensitivity of denitrification to pH is of less importance. Outside this range denitrification may be disturbed in different ways. At low pH values an inhibition of nitrogen oxide reductases may occur, especially of that which reduces N_2O. Low pH values can in this way cause decreased denitrification rate and an increase of the mole fraction of N_2O. In activated sludge the N_2O production has been shown to increase below pH 6.5 with maximum N_2O production between pH 5 and 6 (Thörn and Sörensson, 1996). However studies performed by the Swedish Environmental Protection Agency (Naturvårdsverket, 1994) as well as recent modelling and experimental work (Schulthess and Gujer, 1996, Thörn and Sörensson, 1996) indicate the release of N_2O from municipal biological wastewater treatment to be small.

Denitrifiers are considered to be much less sensitive to **inhibition** than are nitrifiers, states the EPA manual on nitrogen removal (EPA, 1993) which continues; "In general, inhibition would be expected to have a similar degree of impact on denitrification and heterotrophic respiration". This is a generally held view; since denitrifiers in municipal wastewater treatment systems depend on the sensitive nitrifiers for their supply of nitrate a slight difference in sensitivity between denitrifiers and other heterotrophic bacteria is of minor importance relative to the difference in sensitivity between nitrifiers and heterotrophic bacteria in general. However recent research within the STAMP program in Stockholm, where denitrifiers were shown to be more closely related than heterotrophic bacteria in general, suggests that denitrification in biological wastewater treatment may be more sensitive to specific inhibitors than is aerobic respiration. Members of the research group have also studied inhibition of ammonium oxidisers, nitrite oxidisers and denitrifiers to different local industrial wastewaters and found several to inhibit the denitrifiers.

An important inhibitor of denitrification is acetylene which inhibits reduction of N_2O to N_2 and is often used when denitrification is studied since it is easier to collect and analyse N_2O than to measure production of N_2. However, acetylene also has other biological effects which may influence the result; for instance inhibition of N_2-fixation and NH_4^+ oxidation. Furthermore acetylene can be utilised by certain bacteria (Seitzinger, 1988, Knowles, 1982). Some pesticides also inhibit denitrification. In the review by Knowles (1982) Vapam, Dalapon and Toluidine derivates are mentioned. Other substances mentioned by Knowles as inhibitors are azide, cyanide, 2,4-dinitrophenol, nitrapyrin (a nitrification inhibitor) and sulphur compounds.

The effect of salinity on denitrification was investigated by Clifford and Liu (1993), denitrifying spent regenerant brine in a sequencing batch reactor. However the denitrification rate was only about 10 % lower after acclimatisation, when treating brine with 0.5 M NaCl compared with a control with no added salt (the ionic strength of the brine was about two orders of magnitude higher than that of municipal wastewater).

When concentrated ammonium or nitrate solutions are treated the nitrogen compounds themselves or their corresponding bases or acids may disturb denitrification in technical systems. Abeling and Seyfried (1992) observed inhibition of denitrification at nitrous acid concentrations above 0.13 g HNO_3/m^3. At the pH value of 6.8 this corresponds to 100 g NO_2/m^3. Other effects of nitrous oxides on denitrification are reported. In soil, nitrite may cause a lag in reduction of nitrate and partially inhibit the reduction of N_2O (Knowles, 1982). Knowles also mentions a situation where nitrate in high concentrations influences the oxidation reduction potential and thus the enzymatic reactions, causing a greater mole fraction of N_2O in the products.

3.2.3 Conclusions

Denitrification in wastewater treatment is influenced by environmental factors such as pH, oxygen concentration, temperature and inhibiting substances.

Treating normal municipal wastewaters, **pH** is within an acceptable range for denitrification. Since nitrogen removal depends on nitrification the more pH-sensitive nitrifying bacteria will normally determine the sensitivity to pH.

In suspended cultures the process can be considered to be zero-order with respect to the **nitrate** concentration.

In the **temperature** range of municipal wastewater the denitrification rate is often reported to approximately double when the temperature is raised by 10 °C. Selection of bacteria and wastewater quality variation, however, may cause temperature effects in treatment plants to differ from those of denitrifying bacteria in the laboratory.

Denitrification requires **anoxia** in order for the bacteria to choose nitrate instead of oxygen as their terminal electron acceptor and in order to derepress denitrifying enzymes. The bulk oxygen concentration, below which denitrification is the dominating process, is often stated to be about 0.5 mg O_2/l. However denitrification further improves at even lower oxygen concentrations and, due to limited oxygen diffusion, denitrification in flocs or biofilms takes place at higher bulk liquid oxygen concentrations.

Inhibition of denitrifiers by pesticides, sulphur compounds, industrial wastewater etc. is documented although the inhibition of denitrification in municipal wastewater treatment is in general not considered to be a great problem. Denitrifiers are at least as sensitive to inhibiting substances as other heterotrophic organisms. In bacterial cultures containing both nitrifiers and denitrifiers the sensitivity of the nitrifiers is likely to control denitrification by limiting the supply of nitrate to denitrification. However in separate, denitrifying, cultures it is possible that inhibition of denitrifiers will be observed.

The **carbon source** for denitrification is an important parameter to control the denitrification rate of the process as well as the sludge production. The organic carbon contained in the wastewater may not be readily available to the denitrifiers. An external carbon source may be added in order to raise the denitrification rate of the system. The addition of an external carbon source will also increase the sludge production. If the organic matter of the influent wastewater is processed and the readily available matter separated from the not so readily available, more readily available matter may be added to the denitrifiers in the activated sludge system without adding to the sludge production.

3.3 Activity of denitrifiers

3.3.1 Introduction

"Denitrification occurs in essentially all rivers, lakes and coastal systems that have been studied" (Seitzinger, 1988).

Denitrification occurs in many different natural and constructed systems. However the rate of denitrification may vary by several orders of magnitude. Examples of denitrification rates observed in different systems are presented in section 3.3.2. In wastewater treatment an important denitrifying culture is the single-sludge nitrifying denitrifying activated sludge system. Some of the characteristics of this system are discussed in section 3.3.3. Finally some general observations on denitrification rates in activated sludge systems are compiled and discussed (section 3.3.4).

The methods of estimating denitrification vary considerably between different systems and different scientific disciplines. Biologists may be interested in nitrogen removal which with certainty can be attributed to denitrification whereas engineers may measure total loss of nitrogen. In some cases a mass balance approach has been used, the net loss of nitrogen or nitrate being attributed to denitrification. In other cases samples are brought to a laboratory where rate measurements are performed in vitro. In yet other experiments the rate of decrease of nitrate or nitrite through a sediment core may be used to estimate denitrification. In a modification of this technique ^{15}N labelled nitrate is added to the water overlying the sediments and the rate of $^{15}N_2$ production is measured. All methods have problems, mass balance approaches subtracting large numbers from each other, in vitro methods disturbing natural conditions. Results using different measurement techniques are quoted here.

3.3.2 Examples from different systems

In this section denitrification rates in different systems are presented and discussed. The aim is to illustrate the wide variation of rates in different natural and constructed environments. Data have been produced by a great number of scientists from several disciplines and several good reviews are available. Data from reviews as well as data from original work are discussed when useful. The denitrification rates are compiled in Table 3.2 and Figure 3.4.

Denitrification rates in **marine sediments** are in general low in deep sea sediments and higher in shallow near-shore waters where the supply of organic substrate and nitrate is higher (Koike and Sørensen, 1988). Nitrate for denitrification in sediments is to a large extent supplied by nitrification in aerobic layers of the sediments, but also by transport from the water column, and nitrate advected from groundwater through the sediments may also be important (Seitzinger, 1988). Nitrate reaches anoxic, denitrifying, parts of the sediments by diffusion. The boundary between anoxic and oxic micro-environments in estuarine sediments may move diurnally due to consumption of oxygen in the sediments and release of oxygen by daytime photosynthesis (Koike and Sørensen, 1988). The interface between nitrifying and denitrifying environments may further be increased by "bioturbation" - when worms and other large organisms disturb the sediments and thus increase the potential for nitrification and thus for denitrification (Seitzinger, 1988). The activity of denitrification is affected by transport of organic matter from land, by anthropogenic inputs elevating denitrification and by sedimentation of organic matter from the water column. The organic content may also influence denitrification indirectly; mineralisation of organic matter supplies ammonium for nitrification. Where denitrification is measured in the sediments and in the water column of lakes, denitrification is greater in the sediments (Seitzinger, 1988).

River sediments in the material collected by Seitzinger (1988) showed denitrification rates up to 0.1 g N/(m^2 ·d). However the data collected by Leonardson (1994) included major European rivers (which receive large quantities of anthropogenic organic matter) showing denitrification rates of 0.4 - 0.7 g N/(m^2 ·d).

Wetlands used for nitrogen removal range from overloaded natural systems to extensive constructed systems. An Australian wetland receiving effluent from a small wastewater treatment plant consistently removed most of the residual nitrogen, removing 0.02 g N/(m^2 ·d) (Patruno and Russel, 1994). A Swedish study, including overgrown as well as open water bodies (Fleischer and Stibe, 1991, Fleischer et al., 1991) indicated annual nitrogen retentions ranging from 0.001 to 3 g N/(m^2·d). Only a small part of the retention was estimated to be due to retention by harvested plant material and sedimentation of plant residues. The main mechanism of removal was assumed to be denitrification. Leonardson (1994) collected and evaluated Swedish and international literature on nitrogen retention in wetlands. Denitrification rates varied greatly, ranging from 0.002 to above 0.7 g N/(m^2·d), in different types of wetlands subjected to different conditions. In (for denitrification) unsuitable conditions, such as cold climate wetland forests, nitrogen retention was extremely low or even negative (nitrogen release).

Constructed wetlands may be of a surface flow or of a subsurface flow type (Crites, 1994). According to Crites a surface flow or free water surface wetland consists of basins or channels with a natural or constructed subsurface barrier to minimise seepage, and wastewater is treated as it flows through the vegetation and plant litter. The systems are typically long and narrow to minimise short-circuiting. Subsurface flow wetlands consist of channels or basins that contain gravel or sand media which will support vegetation on a bed of impermeable material. Wastewater flows horizontally through the root zone of the wetland plants below the gravel surface. Treated effluent is collected in an outlet channel or pipe. Wetland treatment systems provide a diversity of physical, chemical and biological environments and have relatively long retention times. The systems can contain aerated zones and anaerobic zones; and the hydraulic and solids residence times and presence of organic matter allow for nitrification and denitrification, thus allowing for effective nitrogen removal (Hammer and Knight, 1994). According to case histories of constructed wetlands presented by Hammer and Knight some wetlands may be designed following rational design approaches while in other cases the available area sets the limit. In the material collected by Hammer and Knight average removal rates for total nitrogen were 0.2 g N/(m^2·d) in natural wetlands receiving wastewater, 0.3 in constructed surface flow systems and 1.6 g N/(m^2·d) in constructed subsurface flow wetlands. However, the difference between surface flow systems and subsurface flow wetlands is assumed to be at least partially due to the typically higher nitrogen loading of the latter systems. In a pilot scale constructed surface flow wetland in New Zealand receiving nitrogen at a high nitrate loading the nitrogen removal rate was 5.2 to 5.5 g N/(m^2·d) of which 87 % was due to denitrification. However the removal efficiency was only about 50 %. Since the influent nitrogen was in the form of nitrate the system did not have to allow for nitrification (van Oostrom, 1995).

Among municipal wastewater treatment systems for nitrogen removal, single-sludge **activated sludge systems** are common. In these systems the activated sludge contains both nitrifiers and denitrifiers and separate environments are created for nitrifiers and denitrifiers respectively. In the nitrifying environment the oxygen concentrations is high in order to encourage nitrification, and the amount of readily available organic matter should be low in order to minimise heterotrophic activity. In the denitrifying environment lack of oxygen and abundance of readily available organic matter is necessary. In these systems anoxic and oxic conditions for denitrification and nitrification respectively are separated in space or time. Often limiting for these systems is the need to keep the wastage of activated sludge low. If the wastage of nitrifiers with the activated sludge exceeds the net growth of nitrifiers the number of nitrifiers, and thus nitrification, will decrease and eventually cease. In order to allow for the growth of nitrifiers single sludge systems are often characterised by long hydraulic and solids retention times. Reviewing the literature Christensen and Harremoës (1977) found denitrification rates in these systems to be in the order of 0.3 - 3 g N/(kg VSS·h) when no external carbon source was used. With the addition of a carbon source denitrification rates can be increased, with methanol to about 10 g N/(kg VSS·h) (Henze and Bundgaard, 1982).

A separate sludge activated sludge system using municipal wastewater as the carbon source for denitrification was suggested by Balakrishnan and Eckenfelder (1969 c), and called the contact-stabilisation-denitrification system. In this system return activated sludge was contacted with raw wastewater and then separated by sedimentation. The supernatant was nitrified in a trickling filter and returned to the activated sludge in an anoxic reactor for denitrification (see Figure 3.17). When tested in a lab-scale unit denitrification rates in the range of 1.1 to 3.3 g N/(kg SS·h) were experienced. The same principle was used by Jones et al. (1990 a, b). They proposed a system where wastewater was contacted with activated sludge in a sequencing batch reactor (SBR) (Figure 3.19). After sedimentation the supernatant was nitrified in a rotating biological contactor (RBC) or a trickling filter before being denitrified in another SBR. In the following cycle the flow would be reversed. In this way organic matter sequestered to the activated sludge during the preliminary contact period could be used for denitrification of nitrified effluent from the trickling filter. Denitrification rates between 2.6 and 4.8 g N/(kg SS·h) were recorded.

With the addition of an external carbon source, for instance methanol, higher denitrification rates are obtained. Mulbarger (1971) reported denitrification rates in the order of 8-25 g N/(kg VSS·h) in a separate sludge denitrifying activated sludge system using methanol at temperatures around 20 °C. Even higher nitrate removal rates are found in technical applications where high-nitrate wastewater is to be treated. Industrial applications with wastewater containing nitrate in excess of 1 000 g N/m^3 are fertiliser and explosives manufacture and in connection with uranium oxide fuel production (Francis and Callahan, 1975). Francis and Callahan reviewed biological denitrification of high-nitrate waste. Quoted systems with addition of external carbon exhibited denitrification rates from 10 g N/(kg SS·h) to 26 g N/(kg VSS·h). In order to determine the maximum level of nitrate concentration which might be used effectively for denitrification Francis and Mankin (1977) performed lab-scale experiments varying the nitrate concentration within a large range. When treating waste from uranium oxide production at nitrate concentrations below 1300 g N/m^3 the maximum specific removal rate of nitrate was 3.13 days^{-1} with respect to nitrate. This corresponds to about 29 g NO_3-N/(kg VSS·h). At higher nitrate concentrations the denitrification rate dropped. This may be attributable to high nitrate concentrations, but the authors point out other possible factors, such as excessive concentrations of methanol.

Using laboratory **chemostat systems** very high denitrification rates have been obtained. Blaszczyk et al. (1981) denitrified concentrated synthetic wastewater in a chemostat-type column using acetic acid. At close to complete denitrification, volumetric denitrification rates of 357 g N/(m^3·h) were obtained. This value is of the same order of magnitude as the 362 g N/(m^3·h) obtained by the same research team in a packed bed reactor. Bode et. al (1987) performed lab-scale experiments in the mesophilic (41°C) and in the thermophilic (62 ° C) range with high-nitrate feed. Using a washout reactor with a retention time between 1 and 2 hours volumetric denitrification rates were 580 and 780 g N/(m^3·h) for the mesophilic and thermophilic reactor respectively. Denitrification in the same temperature ranges was also tested in lab-scale activated sludge reactors. In these systems denitrification rates were not so high, and serious problems with sludge settlability disturbed performance in the thermophilic range.

In **biofilm systems** for denitrification, the bacterial culture is attached to a carrying medium in a reactor through which the wastewater passes. Nitrate and organic matter pass from the wastewater to the attached bacteria as the wastewater passes through the reactor. The carrying media may be fixed, as in submerged beds, or consist of mobile particles, as in fluidised beds. Discussing submerged filters la Cour Jansen et al. (1994) noted that the number of full-scale plants with denitrification in submerged filters was limited, with the result that a common base for design and operation had not yet been established. However, denitrification rates of several experimental submerged filters were collected in a diagram indicating denitrification rates of 300 - 5000 g N/(m^3·d). The authors explain the high denitrification rates by the high solids content and by the micro-organisms being highly specialised in these systems. They assume that denitrification rates would be lower in full scale domestic operation due to nitrate and carbon limitation. Denitrification rates in the same range have been experienced by others in submerged filters (Çeçen and Gönenç, 1995), in sand filters (Hultman et al., 1994) and in filters containing activated carbon (Sison et al., 1995). Biological filters are also used for denitrification of nitrate containing drinking water. A review by Matějů et al., 1992 indicate denitrification rates largely in the same range. Denitrification rates of 1 200 to 2 600 g N/(m^3·d) are reported for processes using expanded clay, polystyrene and sand in fixed beds, fluidised filters and up-flow filters using ethanol or methanol as carbon source.

If the systems above are to be compared in terms of removing nitrogen, one way is to compare nitrogen removal in terms of removal per m^2 ground surface area occupied by the "reactor" (Table 3.2 and Figure 3.4). Denitrification varies greatly between different types of systems, but also within one type of system under different loading conditions. Constructed systems can be very efficient per m^2 of ground.

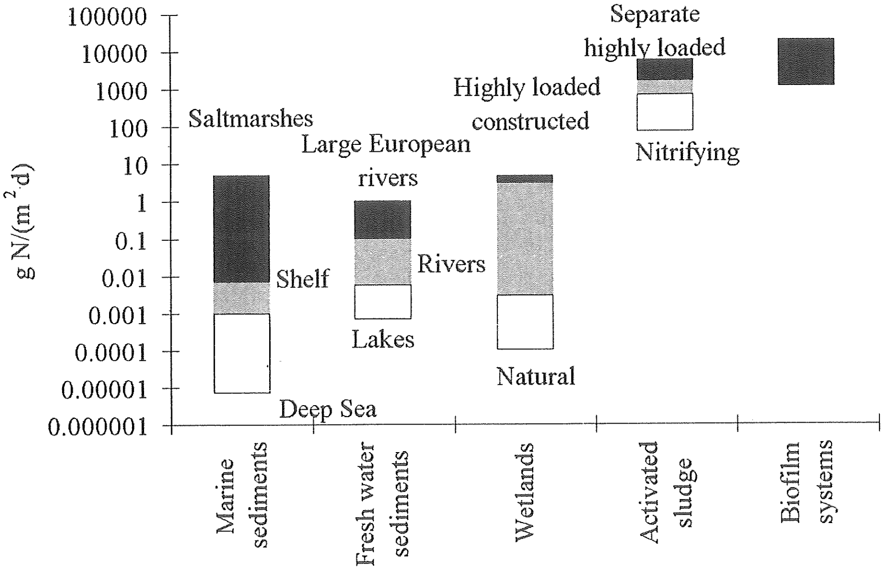


Figure 3.4 Typical denitrification efficiencies per m^2 of ground surface area in different systems (as estimated in Table 3.2).

Table 3.2 Examples of denitrification rates in different systems.

Location	Addition of carbon	Denitrification rate (rates per m^2 refer to ground surface area)		Author
		g N/(m^2·d)	g N/(m^3·h)	
Deep sea sediments		$7\cdot10^{-6}$- $1\cdot10^{-3}$		Koike and Sørensen, 1988
Shelf sediments		$1\cdot10^{-3}$- $7\cdot10^{-3}$		Koike and Sørensen, 1988
Coastal bays		$7\cdot10^{-3}$-0.03		Koike and Sørensen, 1988
Estuarine sediments		0 - 0.3		Koike and Sørensen, 1988
Saltmarsh sediments		0 - 4.9		Koike and Sørensen, 1988
River sediments		0 - 0.1		Seitzinger, 1988
Lake sediments		$6.7\cdot10^{-4}$-0.058		Seitzinger, 1988
Estuarine and coastal Marine sediments		$1.7\cdot10^{-3}$-0.084		Seitzinger, 1988
ditto extremes		0-0.36		Seitzinger, 1988
Large European rivers		0.4 - 0.7		Leonardson, 1994
Natural wetlands		$1.4\cdot10^{-4}$-0.75		Leonardson, 1994
Natural wetland loaded with wastewater		0.02		Patruno and Russel, 1994
Natural or constructed wetlands		≈0.001-3		Fleischer and Stibe, 1991
Natural wetlands		0.2		Hammer and Knight, 1994
Constructed SF wetlands		0.3		Hammer and Knight, 1994
Constructed SSF wetlands		1.6		Hammer and Knight, 1994
Constructed highly loaded wetland	?	4.5-4.8		van Oostrom, 1995
Single sludge activated sludge systems		72-720*	0.75-7.5*	Christensen and Harremoës, 1977
Single sludge activated sludge systems	Yes	2 400	-25	Henze and Bundgaard, 1982
Separate sludge activated sludge systems (municipal)		380-1 150*	4-12*	Balakrishnan and Eckenfelder, 1969C
		860-1630*	9-17*	Jones et al. , 1990A
Separate sludge municipal activated sludge systems	Yes	2 000-6 000*	21-62*	Mulbarger, 1971
Act. sludge receiving NO_3^- concentration >1000g N/m^3	Yes	3 450-6 240*	36-65*	Francis and Callahan, 1975
	Yes	- 6 900*	- 72*	Francis and Mankin, 1977
Chemostat	Yes		357	Blaszczyk et al., 1981
ditto 41 °C	Yes		567	Bode et al., 1987
ditto 62 °C	Yes		775	Bode et al., 1987
Biofilm systems	Mostly	1 100- 20 200*	12-210	Fr. fig. 3, la Cour Jansen et al., 1994
ditto for drinking water treatment	Yes	4 800- 10 600*	50-110	Matějů et al., 1992

*Assuming that activated sludge tanks and biofilm systems are 4 m deep and that activated sludge tanks contain 2.5 kg VSS or 3.6 kg SS per m^3.

3.3.3 Conditions in single-sludge systems

"The results of this survey show that removal of nitrogen by the conventional treatment processes is erratic and is not correlated with carbon or solids removal...... In each instance where active nitrification was found, subsequent loss of nitrogen by denitrification was indicated." (Barth et al. 1966)

"In biological treatment of wastewaters, oxidation of carbonaceous material, nitrification, and denitrification all occur within a single process if sufficient bacterial solids retention time (SRT) is provided for development of the nitrifying organisms. Unfortunately , optimum process operating conditions for oxidation and for subsequent denitrification are thermodynamically antagonistic; that is, the presence of the more powerful oxidant oxygen (electron acceptor) suppresses the use of NO_3^- (electron acceptor) in the biological oxidation of the carbonaceous material in the wastewater." (Bishop et al. 1976)

Single-sludge systems are activated sludge systems where carbon oxidation, nitrification and denitrification are performed in the same activated sludge culture. These systems are characterised by the necessity of accommodating for nitrification and denitrification in the same culture. Nitrification was early shown to be load dependent, so that the activated sludge systems should be operated at an organic loading lower than a certain level in order to obtain nitrification (Johnson and Schroepfer, 1964). Under these conditions, however, nitrification was more or less complete. The growth rate of the culture must be low enough to allow for the slow growth of nitrifiers (Downing et al. 1964) which can be described by Michaelis type equations (Knowles et al., 1965). At the same time the denitrifiers must be supplied with a sufficient supply of organic matter for denitrification.

The growth rate of the ammonia oxidisers (nitrosomonas) is dependent on temperature, oxygen concentration, pH and ammonium concentration. The relations can be described by the equation below, Figure 3.5, depicted in Figure 3.6.

$$\mu = \mu_m \cdot \frac{S}{K_s + S} \cdot \frac{DO}{K_{O_2} + DO} \cdot e^{0.098(T-15)} \cdot (1 - 0.833(7.2 - pH))$$

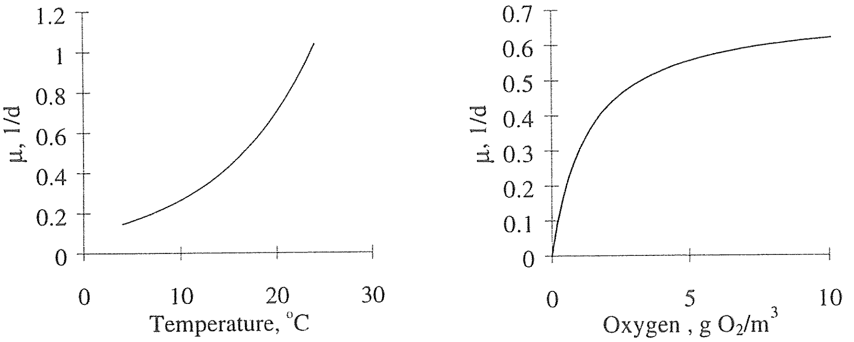
Symbol	Explanation	Unit
S	Ammonium concentration	g N/m^3
DO	Oxygen concentration	g O$_2$/m^3
T	Temperature	°C
pH	pH	
μ_m	Maximum growth rate of Nitrosomonas	1/d
μ	Growth rate of Nitrosomonas	1/d
K_s, K_{o2}	Constants	g N/m^3, g O$_2$/m^3

Figure 3.5 Growth rate of nitrosomonas (Metcalf & Eddy, 1991).

The different parameters influence design and operation of a single-sludge activated sludge system in different ways. The oxygen dependency sets technical demands on the aeration system so that a certain oxygen concentration can be held in the volume of activated sludge where nitrification is wanted. The pH dependency, in combination with the reduction of alkalinity caused by the nitrification process, can cause problems when treating water with low alkalinity relative to the ammonium content. In some cases the pH-drop and its consequences may be so great as to warrant adjustment of pH. The sensitivity to low ammonium concentration is generally of less importance, the low ammonium concentration being a consequence of nitrification. The wastewater temperature, however, is generally not a controllable parameter. In cold regions and in systems where large amounts of cold rain, groundwater and melted snow reach the wastewater treatment plants, temperature is often the key parameter controlling the growth rate of nitrifiers. Low wastewater temperatures cause low design nitrifier growth rates. In order to retain a population of nitrifiers the nitrifier growth rate must not be exceeded by the growth rate of the entire activated sludge culture, which is controlled by the relation between the aerated sludge inventory and the withdrawal of excess sludge. This can be expressed as the minimum solids retention time for nitrification (see Figure 3.7). The aerated solids retention time in an activated sludge system at 8 °C would have to be 13-17 days if nitrification is wanted whereas 3-4 days solids retention time (SRT) is reasonable for plants designed for removal of carbonaceous material.

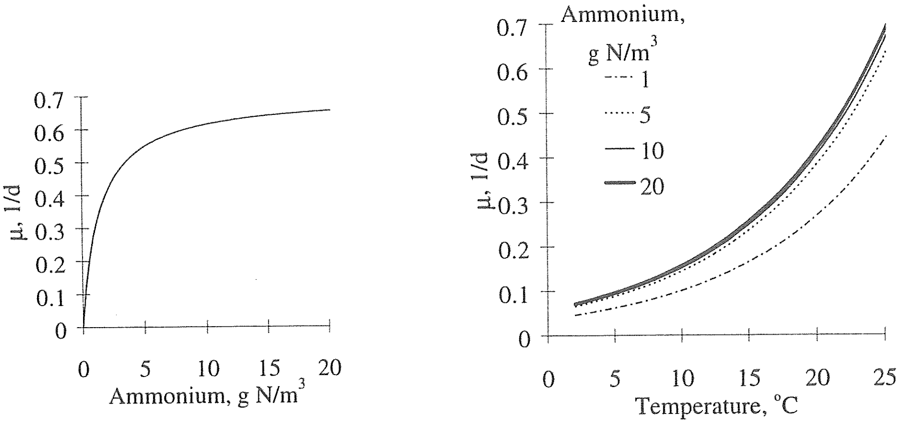
In the basic case the aerated sludge inventory is a function of the aerated volume of activated sludge and the sludge concentration. Since the sludge withdrawal must equal the net sludge production, the sludge withdrawal is controlled by the sludge yield based on the loading of the system by wastewater and process chemicals. In order to increase the SRT it is thus possible to increase the aerobic activated sludge volume, increase the concentration of the activated sludge, decrease the organic loading on the system and the chemicals added, or to decrease the sludge yield caused by these.

The activated sludge concentration can be increased to the limit set by the separation unit. The concentration may be increased selectively in a part of the system by the addition of carrying material trapped in the system (Reimann, 1990, Ødegaard et al., 1994) or through contact stabilisation where the concentrated return activated sludge is aerated separately (Jenkins and Orhon, 1972, Gujer and Jenkins, 1975, Palmgren, 1992).



a) Dependency of temperature.

b) Dependency of oxygen concentration.



c) Dependency of ammonium concentration.

d) The effect of temperature and ammonium concentration at pH 7.2 and 2 g O_2/m^3.

Figure 3.6 Growth rate of nitrifiers depending on environmental conditions. (Constants; μ_m=0.7d^{-1}, K_{O2}=1.3 g O_2/m^3 and K_s=0.6 g NH_4^+-N/m^3 at 20°C according to Metcalf & Eddy, 1991).

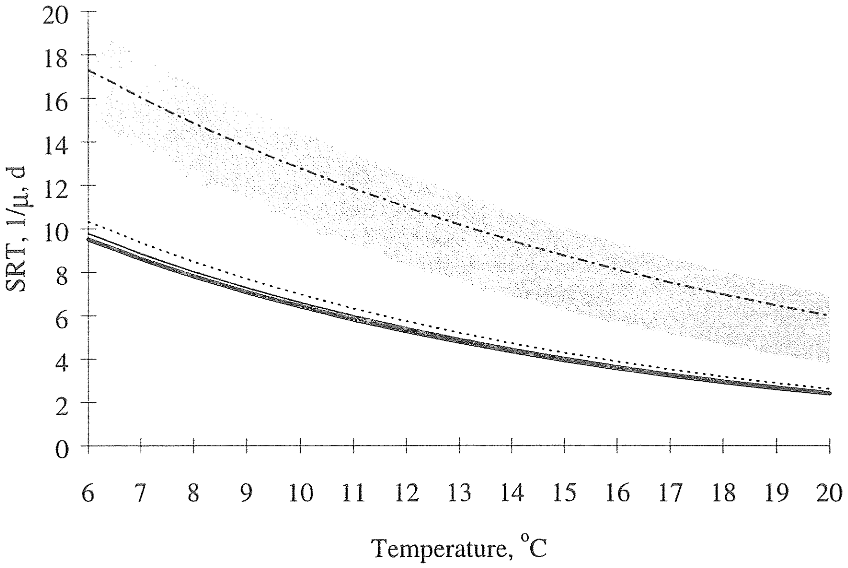


Figure 3.7 Minimum SRT for nitrification as recommended by Henze and Bundgaard (1982), shaded, and the inverse of the growth rate of nitrifiers at different ammonium concentrations. Lines and conditions as in Figure 3.6 d.

The sludge production of the system decreases if the organic loading decreases. This may be achieved by pre-treating the wastewater, through primary sedimentation or further through pre-precipitation. This may in some cases reduce the sludge production sufficiently to be able to perform nitrification in existing tanks (Nyberg et al. 1992). In this case, however, the reduction of the organic loading causes the carbon source to be insufficient for denitrification and a carbon source has to be added. The carbon source can be added when and where it is optimal for denitrification, and the type of carbon source should be chosen so as to give maximum denitrification and minimum sludge yield. Simultaneous precipitation of phosphorus will cause a greater sludge yield, the consequence being that, all else constant, the activated sludge volume will have to be greater in such a system in order to keep the same SRT. Although the methods of accommodating a high SRT may vary and the consequences on tank volumes may in some ways be manipulated, as mentioned above, the high SRT needed in cold regions generally demands large activated sludge tanks. A few typical single-sludge systems treating municipal wastewater will be described below.

In a continuous flow pre-denitrification system the first 25-50 % of the activated sludge tank is non-aerated, allowing for denitrification of nitrate contained in the activated sludge recirculated from the outlet of the activated sludge tank or returned with the return activated sludge. The readily available carbon source in the influent wastewater is used for denitrification. The remainder of the tank is aerated and allows for nitrification and aerobic carbon removal. Nitrogen removal through denitrification in this system is limited by the fraction of the nitrified mixed liquor which is returned to the first, anoxic, part of the tank. Part of the tank is often equipped for both oxic and anoxic operation so that the oxic and anoxic volumes can be varied depending on wastewater flow, quality and temperature, and on the target treatment results.

If the carbon source of the primary settled wastewater is poor, and denitrification insufficient, a carbon source can be added. The carbon source is often added to the anoxic zone in the first part of the tank. However if the wastewater is very weak, or if effluent standards are stringent, it may be advantageous to add the extra carbon source to another anoxic zone, after the aerated zone. In this case the recirculation need only be set at a level where the amount of nitrate which can be denitrified by the influent carbon source is recirculated. In this process nitrogen removal is not limited by recirculation of nitrified mixed liquor.

If a large amount of nitrate is removed later in the tank, the amount of nitrate available for removal in the preliminary anoxic zone decreases. A post denitrification system may be considered instead. In these systems nitrification takes place in the first, aerobic, part of the tank and in a later, anoxic, section of the tank denitrification takes place, usually with the aid of an added carbon source. The ultimate development in this direction is to introduce a separate denitrification unit after the activated sludge system. This, however, is no longer a single-sludge system.

In a sequencing batch reactor (SBR) for single-sludge nitrogen removal one reactor is equipped for anoxic and aerobic conditions and for sedimentation. The processes are separated in time and can be performed in any order or for any length of time suitable for treatment of the wastewater. First the reactor is filled with wastewater which is mixed with the mixed liquor collected in the bottom of the tank. Typically an anoxic period may follow, when nitrate contained in the mixed liquor from the preceding cycle is reduced using the carbon source of the influent wastewater. When the nitrate is depleted (or expected to be depleted) the aerators are turned on and the tank is run aerobically until nitrification is complete, or has reached a pre-set level. The aerators are turned off and the mixed liquor allowed to settle. After sedimentation, supernatant is pumped or siphoned out of the tank before new wastewater is added. The system may be controlled by time, allowing each process a pre-set amount of time, or by sophisticated instrumentation where the length of each phase is continuously controlled by the exact conditions in the tank. A system may be operated with any number of phases and carbon sources, and other process chemicals may be added. This type of system is often used when high-strength, intermittently produced or otherwise complicated wastewater is to be treated or for small systems. In municipal wastewater treatment, SBR-systems may be considered for main stream treatment in small municipal wastewater treatment plants or for nitrification, or nitrification and denitrification, of high strength sludge liquors. Sludge liquors are concentrated with respect to nitrogen, but may contain insufficient organic matter for denitrification, the alkalinity may be low in relation to the amount of ammonium to be nitrified and the production may be intermittent, with production only when sludge is dewatered.

Some systems combine continuous and batch operation, for example the alternating Bio-Denitro process (Bundgaard et al.,1989). In this system two connected tanks are fed in turn and aerated following a pre-set schedule. The system combines pre-denitrification and post-denitrification. When the flow is redirected, wastewater is introduced to a tank full of nitrified activated sludge, avoiding the high recirculation rate needed for a pure continuous flow pre-denitrification system. In this type of system the process conditions may relatively easily be changed by changing the sequence of flow to the different tanks and aeration of tanks. One example is the Bio-Denipho process where biological phosphorus removal is included by the addition of a third, anaerobic, tank (Isaacs et al., 1994). In a discontinuous system parameters, such as nitrification rate and denitrification rate, can be estimated using on line instruments, and the length of the cycles can be controlled in order to optimise the use of volumes, chemical dosages and energy for aeration. An advantage compared with SBR-systems is that the water level is constant which demands less pumping.

Single-sludge nitrogen removal can be performed in many different ways but some basic conditions control the systems:

- The growth rate of the activated sludge must not exceed that of the nitrifiers. This is often expressed as the minimum aerobic solids retention time for nitrification.
- The growth of the activated sludge is controlled by the amount of organic matter entering it with the wastewater and added to the system, and the sludge production caused by these and any chemicals added to the system.
- The necessary amount of anoxic activated sludge is determined by the denitrification rate of the activated sludge, which is a result of the denitrifying activity of the activated sludge and the quality and quantity of the carbon source available.
- The volume of the activated sludge reactor is determined by the amount of sludge needed in combination with the suspended solids concentration allowed for by the separation capacity of the separation unit serving the activated sludge tank.

3.3.4 Denitrification rates in activated sludge systems.

"There is no a priori reason for the proportion being constant from sludge to sludge or even for a given sludge over a period of time"* (Clayfield, 1974)

* of denitrifiers

Denitrification in activated sludge systems has been treated by many authors. During the last few decades a large part of the investigations concerns single sludge systems, which are popular due to their potential for using the carbon source contained in the influent wastewater and since they avoid an extra separation step between nitrification and denitrification.

Important rate controlling factors of denitrification are: the temperature, the supply of carbon source and the denitrifying ability of the biomass. Anoxia and presence of nitrate are required. High and low pH conditions and inhibition by different substances may disturb the pathways of denitrification or completely inhibit denitrification. These factors are governed by the environment or the wastewater. In this section the heterotrophic respiration and denitrifying capacity of the bacteria are discussed.

Weddle and Jenkins (1971) studied the activity of activated sludge at different net growth rates (the net growth rate being the inverse of the solids retention time, SRT). In their study the substrate uptake rate (using volatile suspended solids, VSS, as a base) increased by about 6 times when the net growth rate increased from 0.068 d^{-1} to 0.34 d^{-1} (corresponds to a decrease of the SRT from 15 to 3 d). At the same time the specific oxygen uptake rate (Figure 3.8 a) and the dehydrogenase activity approximately doubled. However, if the count of viable cells was used as the base for calculation the activities were independent of the net growth rate. This indicates that the viable organism content of activated sludge increases with increasing net growth rate and that, consequently, the activity of activated sludge measured as dehydrogenase activity or oxygen uptake rate will also increase with increasing net growth rate, or with decreasing SRT. However, as the substrate removal rate increased more than the oxygen uptake rate at higher organic loadings, it may also be concluded that at higher organic loadings less of the influent organic matter is degraded in the activated sludge system and more remains stored in the excess sludge. The same general pattern can be obtained using the activated sludge model No. 1 (Henze et al., 1987). Here a Danish application of the activated sludge model No. 1, "EFOR", will be used. Raising the SRT from 4 to 20 days reduces the heterotrophic respiration to about one third and respiration including nitrification to about half the respiration at 4 d SRT. The sludge yield is reduced to 80 % of the yield at 4 d SRT (Figure 3.8 d and Table 3.3).

Table 3.3 Results of simulations using EFOR (using "default" wastewater, "standard" process constants and process configuration according to "demo - recirculation").

SRT	Volume of first tank compared with total activated sludge volume	Influent wastewater flow	Excess sludge flow	Sludge concentration in aeration tank	Return and excess sludge concentration	Yield	Effluent ammonium concentration		Respiration in first tank including nitrification	Volume of first tank compared with total activated sludge volume
	V_2/V	Q_1		$C_{2,SS}$				$S_{7,N}$		$r_{x,O2}$
	-	m^3/d	m^3/d	g SS/l	g SS/l	kg SS/ kg COD	g N/m^3		g O_2/ (kg SS·h)	g O_2/ (kg SS·h)
2	1	2016	96	2.5	7.0	0.63	29*	30**	9.6*	9.6**
4	1	1008	48	2.3	6.5	0.58	28*	32**	6.5*	6.1**
8	1	504	24	2.1	5.8	0.53	2.9*	36**	6.7*	4.2**
20	1	202	10	1.7	4.9	0.46	0.9*	40**	3.8*	2.4**
30	1	134	6	1.6	4.5	0.38	0.7*	42**	2.9*	1.9**
4	0.002	1008	48	2.3	6.5	0.58	28*	32**	16.9*	16.2**
4	0.25	1008	48	2.3	6.5	0.58	28*	32**	12.2*	12.0**
4	0.50	1008	48	2.3	6.5	0.58	27*	32**	8.9*	8.5**
20	0.002	202	10	1.7	4.9	0.46	0.9*	40**	11.6*	9.4**
20	0.25	202	10	1.7	4.9	0.44	0.6*	40**	7.2*	4.8**
20	0.50	202	10	1.7	4.9	0.44	0.5*	40**	5.4*	3.4**

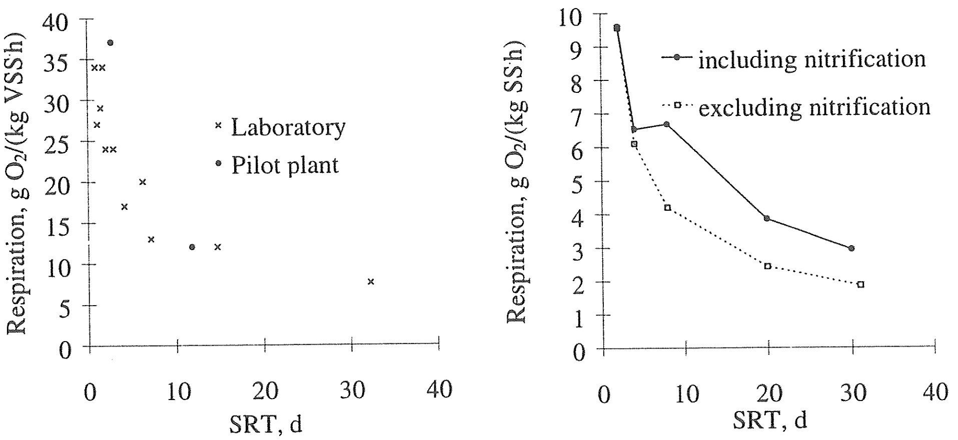
"Demo-recirculation" is a process consisting of two completely mixed aerated activated sludge tanks, with a total volume (V) of 500 m^3, in series and a final sedimentation tank.

"Default" wastewater: 530 mg COD/l, 212 mg soluble COD/l, 250 mg BOD/l, 100 mg soluble BOD/l, 41 mg soluble N/l, 50 mg Kjeldahl N/l, 30 mg Ammonium N /l, 1 mg Oxidised N/l, 16 mg P/l, 10 mg PO_4-P/l, Alkalinity 6 meq/l, Temperature 12 °C.

Symbols as used in equations in Figure 3.16

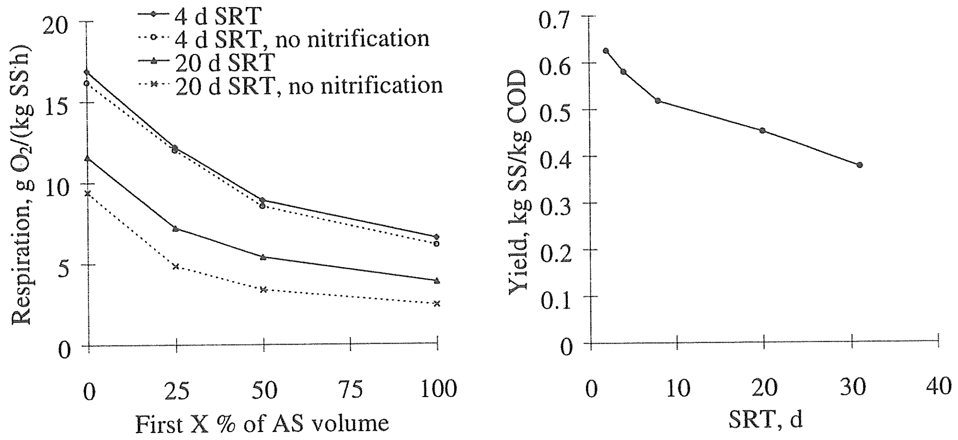
*Autotrophic growth 0.8 1/d

** Autotrophic growth 0.01 1/d



a) Experimental data of Weddle and Jenkins (1971).

b) Respiration in a completely mixed tank (simulations using EFOR).



c) Respiration in the first of two completely mixed tanks where the volume of the first tank is X % of the total activated sludge tank volume (simulations using EFOR).

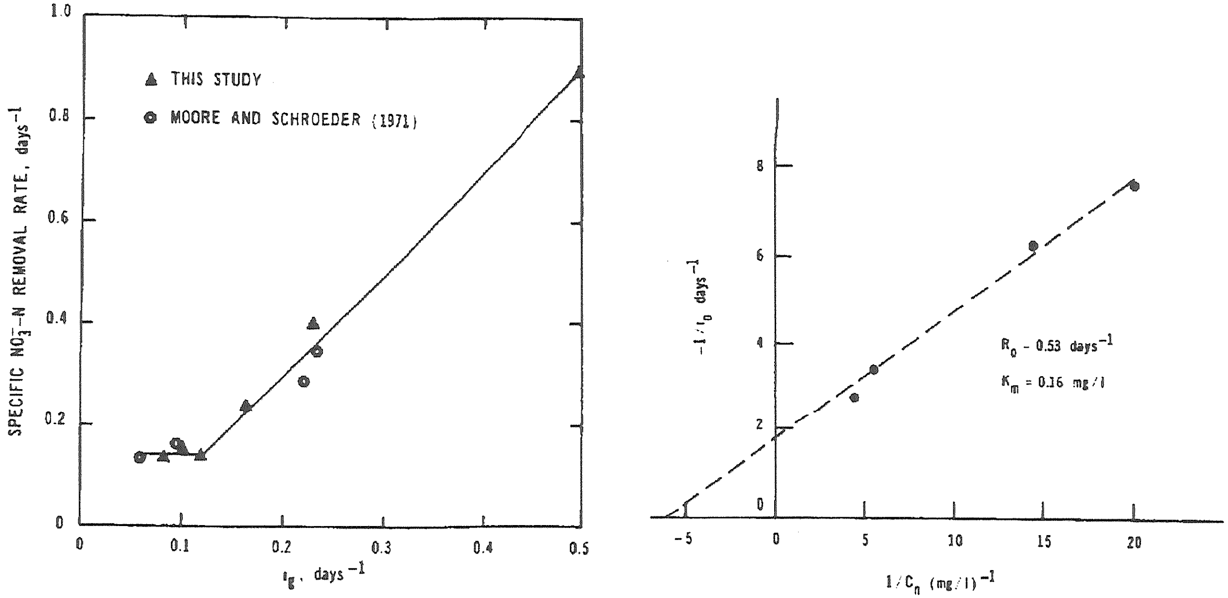
d) Yield versus SRT (simulations using EFOR).

Figure 3.8 Respiration and yield versus SRT. EFOR is a Danish implementation of the activated sludge model No 1 (Henze et al., 1987). For simulations when nitrification was avoided the standard autotrophic growth rate of 0.8 1/d was replaced by a growth rate of 0.01 1/d.

Moore and Schroeder (1970) investigated the relation between nitrate removal and growth rate in a denitrifying chemostate system and Engberg and Schroeder (1975) in continuous flow pilot and laboratory activated sludge systems. In both investigations the nitrate removal rate seemed to increase with increasing growth rate of the activated sludge. Nitrate removal rates of runs at 4-5 d SRT were more than double those of runs at 10-14 d SRT (Figure 3.9 a). The ratio between utilised methanol and removed nitrate, however, increased slightly for lower solids retention times. In the continuous flow study (Engberg and Schroeder, 1975), 2.7 -2.8 gram methanol was used per gram nitrate nitrogen removed at 4-5 d SRT, whereas at 10 -14 d SRT the ratio was 2.4-2.5, indicating a slightly poorer utilisation of the methanol at a lower SRT. However, since the data (Figure 3.9, b) can be linearised according to a Monod relationship with respect to nitrate concentration, it is indicated that these experiments were nitrate limited. The effluent nitrate concentrations ranged from 0.05 to 0.22 mg N/l, which is in the same range as or below the measured half saturation constant, 0.16 mg N/l. In order to determine denitrification capacity of the sludge at different organic loadings it would have been interesting to operate under non-nitrate limiting conditions (nitrate concentration well above 0.16 mg N/l).

The trend of higher denitrification rates at lower solids retention times (or higher organic loadings) has been reported by several authors. Literature reviewed by Jones and Sabra (1980) includes several such observations in support of the tendencies reported in a study of a single sludge system investigated by the authors. This is also in agreement with results of Argaman (1986) who, in a semi-continuous system observed denitrification rates in the order of 1 g N/(kg g SS·h) at SRT:s below 20 days, whereas when the SRT was above 20 days the average denitrification rate was 0.6 g N/(g SS·h). Assuming a ratio of VSS to SS of 0.7 these values are within the range of expected denitrification rates in single sludge systems using an internal or endogenous carbon source as presented by Henze and Bundgaard (1982), Figure 3.11.

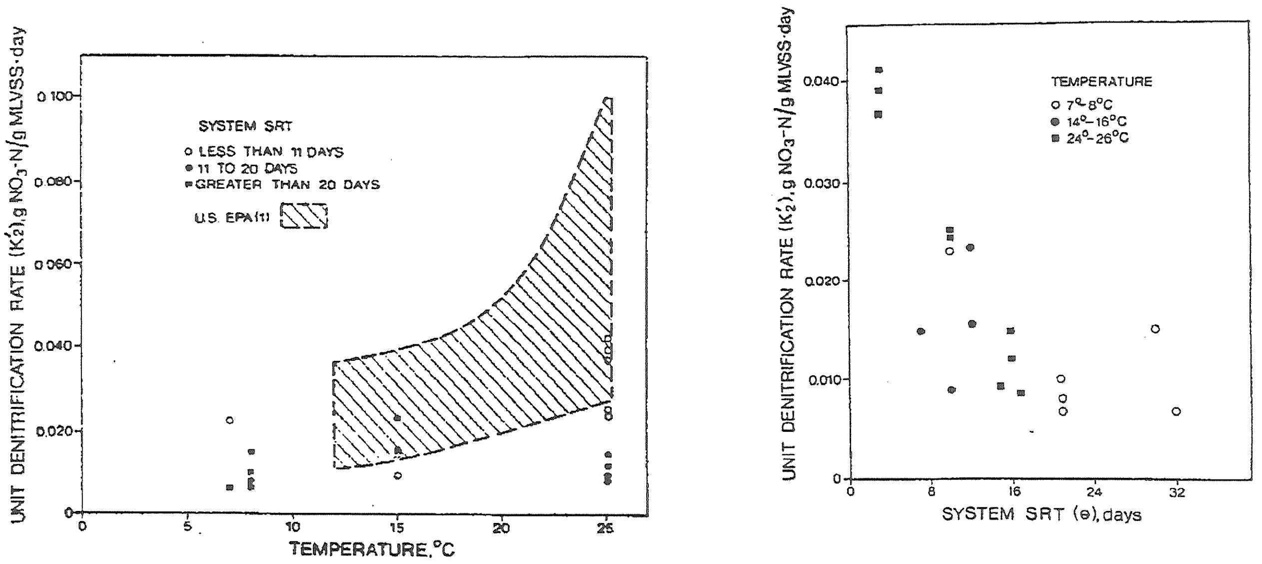
Also Sutton et al. (1978) found denitrification rates in a single sludge system to increase with decreasing system solids retention times. In fact, when plotting the data of their study against solids retention time little or no effect of temperature on denitrification rate is evident (Figure 3.10 b). Plotting the same data against temperature (Figure 3.10 a) an effect of temperature appears, but this effect can hardly be separated from the effect of SRT. In these experiments, as in others using single sludge systems, no experiments with low temperature and low SRT could be performed since these conditions would not allow for nitrification and thus would eliminate the nitrate supply for denitrification. One example is the experiments by Olezkiewsicz and Bergquist (1988) discussed in connection with temperature effects on denitrification.



a) Nitrate removal rate at different loading rates (1/θ)

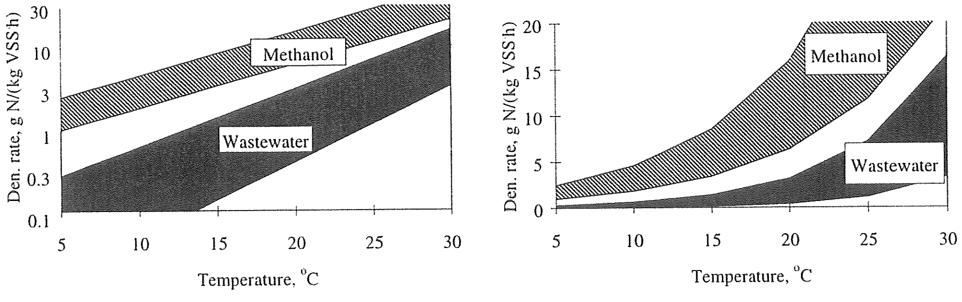
b) Linearisation of the Monod expression applied to the data (r_0 is the specific nitrate nitrogen removal rate (d^{-1}) and C_n the nitrate concentration (mg N/l) in the Monod expression, $r_0=R_0C_n/(K_m+C_n)$).

Figure 3.9 Nitrate removal and loading rates (from Engberg and Schroeder, 1975). Note: Figure a) appears to contain a typographical error. The circles should probably refer to "this study" and the triangles to “Moore and Schroeder (1971)”.



a) b)

Figure 3.10 Effect of solids retention time and temperature on endogenous denitrification rates (from Sutton et al. 1978).



a) **b)**

Figure 3.11 Denitrification rates at different temperatures in single-sludge systems using different carbon sources (redrawn from Henze and Bundgaard, 1982).

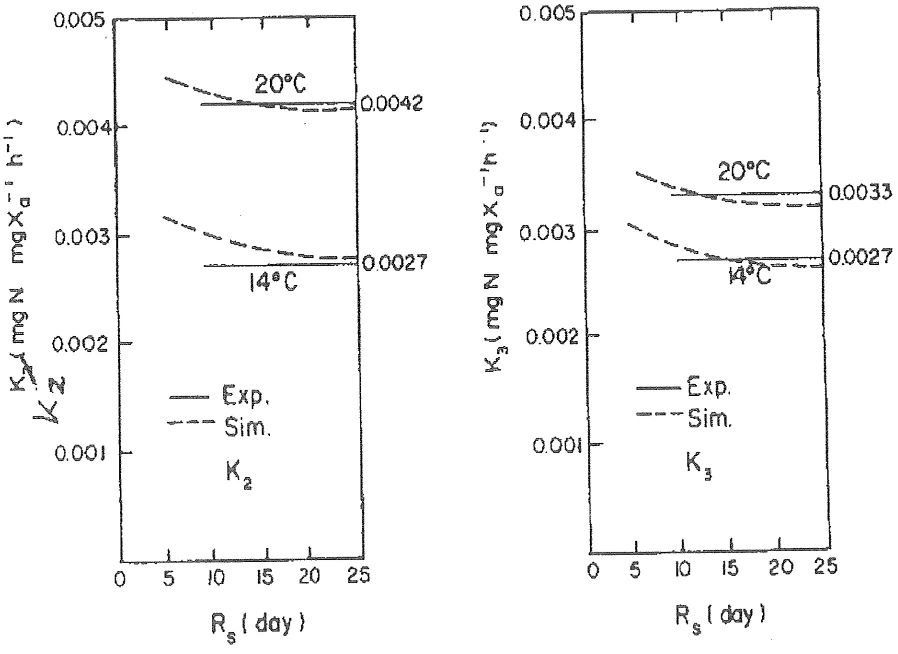


Figure 3.12 Simulated and experimental denitrification rate constants K_2 and K_3 vs. sludge age (from Van Haandel et al., 1981). K_2 is the denitrification rate observed in a plug flow system when the respiration with readily available organic matter, K_1, has ceased. K_3 is the rate of denitrification in a post-denitrification reactor without an external carbon source.

Simulations with a death regeneration model of the single sludge system (Van Haandel et al. 1981) predict higher denitrification rates at lower solids retention times. However, within the range of single sludge nitrification and denitrification, 10 - 20 days, the authors conclude that the denitrification rates remain substantially constant (Figure 3.12).

Apart from the observations and predictions of higher denitrification rates at low solids retention times some authors observe or predict that different systems or conditions produce cultures with different relationships between the denitrifying capacity and the aerobic heterotrophic activity. Argaman and Brenner (1986) studied the composition of microbial biomass in a single-sludge system and observed that the ratio of denitrifiers to the total number of heterotrophic organisms increased as the nitrate to COD removal ratio increased (Figure 3.13). In the study the removed carbon to nitrate ratio was approximately 7 g COD/g NO_3-N. Thus the authors expect that as the ratio of denitrified NO_3-N to removed COD approaches 0.15 (≈1/7) a large portion of the COD would be used by denitrifiers and thus the denitrifiers should predominate. This result, they point out, "does not change the kinetic approach used for the anoxic basin. The COD-removal in the anoxic basin is controlled by the COD concentration. The abundance of denitrifiers is a result of these kinetics, not the cause."

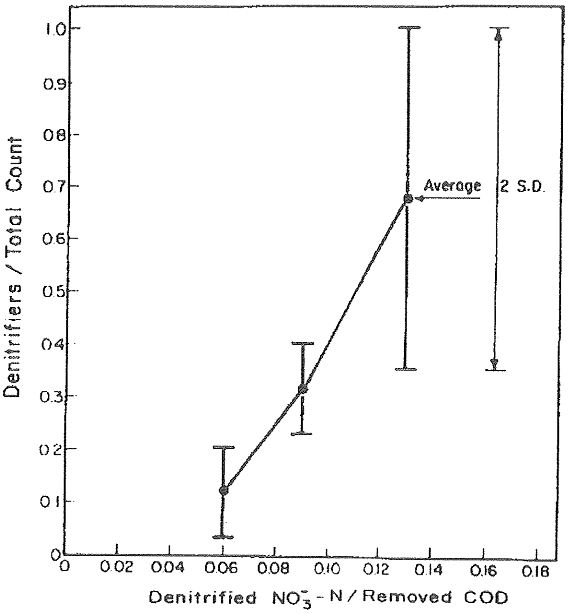


Figure 3.13 Denitrifier fraction of heterotrophs as a function of the ratio of denitrified nitrate to removed COD. (from Argaman and Brenner, 1986).

Clayfield (1974) noted that the relationship between the denitrification rate and the oxygen respiration rate varied dramatically between activated sludges from different systems. Operating conditions favouring denitrification were assumed to lead to activated sludges with a high proportion of denitrifiers. This was also recognised by

Grady et al. (1986) when developing a model for the activated sludge system. The authors comment that this could either be because the maximum growth rate is lower under anoxic conditions or because only a fraction of the heterotrophic biomass is able to function with nitrate as the terminal electron acceptor. From a modelling point of view they recommend the use of an empirical coefficient, η ($\eta < 1$), in the rate expression in order to reduce the rate under anoxic conditions. The importance of the process configuration and the influent wastewater quality to the denitrifying and oxygen reducing capacity of activated sludge has been investigated by Henze in several publications (Henze, 1986, Henze, 1987 and Henze, 1989). The denitrifying capacity and the oxygen reducing activity of activated sludges from different wastewater treatment plants were compared (Henze, 1986). In a pure oxygen plant the denitrifying capacity was only 20 % of the oxygen reducing activity (on an electron transfer basis) whereas in two denitrifying plants the ratio, η, was 0.56 and 0.58, respectively. The denitrifying capacity of the raw wastewater was also shown to vary from plant to plant and from time to time. The author states that " it is reasonable to assume that the easily degradable carbon is the source for the major part of the biomass produced in the plant. If this carbon is removed under anoxic conditions, the result will be production of denitrifying biomass."

This was further developed by Henze (1987) to a model based on a growth-decay concept (see Figure 3.14). All biomass produced in the anoxic part of the reactor is assumed to have the capacity of denitrifying. Biomass produced in the aerated reactor, whether based on influent organics or produced from substrate from decay of biomass, is assumed to have the same composition as the existing activated sludge. The biomass decay can in this way give rise to production of new biomass, which may have a different composition from the decayed biomass. If no denitrification occurs in the activated sludge system the ratio, η_1, found in the activated sludge system would equal that of the influent wastewater (η_0). A crucial element of the model of Henze is the anoxic fraction of primary production ($f_{D,anox}$) which describes how much of the influent substrate will be used directly for growth of denitrifiers under aerobic and anoxic conditions. It is stated that "The fraction is primarily a function of plant layout and operation. If the influent enters an anoxic tank and stays there for a reasonable time (say 1 h) a major fraction of the substrate will result in growth of denitrifiers". No examples of values of $f_{D,anox}$ are indicated. The size of the anoxic reactor is an important factor, governing both $f_{D,anox}$ and the fraction of secondary growth resulting in denitrifiers. However, in reality the non-aerated volumes of the activated sludge system will often not be completely efficient for denitrification since nitrate, at least periodically, may be absent in part of the tank.

The general mass balance for biomass:

Influent + Primary production + Secondary production - Decay = Surplus sludge + Effluent

For denitrifying biomass:

$$\mathit{Influent} = Q_0 \cdot X_{D0}$$

$$\mathit{Primary\ production} = Q_0 \cdot f_{D,anox}(S_{S0} + X_{S0}) \cdot Y_H$$

$$\mathit{Secondary\ production} = \left\{ \frac{\theta_{X,D}}{\theta_X} + \eta_1 \cdot \frac{\theta_{X,A}}{\theta_X} \right\} \cdot b_H \cdot Y_H \cdot (1 - f_E) \cdot V_1 (X_{D1} + X_{A1})$$

$$\mathit{Decay} = b_H \cdot V_1 \cdot X_{D1}$$

$$\mathit{Surplus\ sludge} = Q_3 \cdot X_{D3}$$

$$\mathit{Effluent} = Q_2 \cdot X_{D2}$$

For heterotrophic biomass X replaces X_D, $f_{D,ANOX}$ is excluded and secondary production takes place in the entire tank. Inserting these expressions in the mass balance of denitrifiers and introducing the "potential inlet fraction of denitrifiers" η_{P0} the following expression for the fraction of denitrifiers is produced:

$$\eta_1 = \frac{\frac{1}{\theta_X}\left\{1 + b_H \cdot \theta_X \left[1 - Y_H(1 - f_E)\right]\right\} \cdot \eta_{P0} + \frac{\theta_{X,D}}{\theta_X} \cdot b_H \cdot Y_H (1 - f_E)}{\frac{1}{\theta_X}\left\{1 + b_H \cdot \theta_X \left[1 - Y_H(1 - f_E)\right]\right\} + \frac{\theta_{X,D}}{\theta_X} \cdot b_H \cdot Y_H (1 - f_E)}$$

Where: $\eta_{P0} = \dfrac{X_{D0} + f_{D,anox}(S_{S0} + X_{S0}) \cdot Y_H}{X_{D0} + X_{A0} + (S_{S0} + X_{S0}) \cdot Y_H}$

symbol		unit	subscript	
V	Volume of reactor	m^3	0	Influent
Q	Flow	m^3/h	1	In reactor
$f_{D,anox}$	Anoxic fraction of prim. production	1	2	Effluent
S_S	Soluble substrate	kg COD/m^3	3	Surplus sludge
X_S	Suspended substrate	kg COD/m^3	A	Aerobic
Y_H	Heterotrophic yield coefficient	kgCOD/kgCOD	D	Denitrifying/anoxic
θ_X	Solids retention time	d	X	Biomass
$\theta_{X,A}$	Aerobic solids retention time	d	H	Heterotrophic
$\theta_{X,D}$	Anoxic solids retention time	d	S	Substrate
b_H	Decay coefficient	d^{-1}	E	Endogenous
f_E	Fraction of inert	1		
η_1	Fraction of denitrifiers (or denitrifying metabolic rate)	1		
X_A	Concentration of aerobic biomass	kg COD/m^3		
X_D	Concentration of anoxic biomass	kg COD/m^3		

Additional equations used:

$$Q_3 \cdot X_{D3} + Q_2 \cdot X_{D2} = \frac{V_1 \cdot X_{D1}}{\theta_X} \qquad X_{D1} = \eta_1 \cdot X_1$$

$$\theta_X = \theta_{X,D} + \theta_{X,A}$$

........and a mass balance for heterotrophic biomass

$$X = X_A + X_D$$

Figure 3.14 Estimation of the fraction of denitrifiers or the nitrate to oxygen utilisation ratio (η_1) of activated sludge (from Henze, 1987).

Without nitrate present no denitrification would take place and selection towards denitrifying biomass would occur. The problem of varying $f_{D,anox}$ is avoided by the use of η_{p0} in illustrations and discussion. Figure 3.15 shows how the fraction of denitrifiers will increase with increased anoxic fraction of the solids retention time, at constant values of η_{p0} (the potential inlet fraction of denitrifiers). In a real system the anoxic solids retention time ratio may influence the fraction of denitrifiers in different ways apart from those primarily predicted by the model. Firstly, and mainly at low anoxic retention times, the anoxic retention time may limit denitrification, $f_{D,anox}$ and thus η_{p0}. Secondly, as the anoxic retention time increases above that necessary for complete denitrification nitrate will limit the growth of denitrifiers. Thus the growth of denitrifiers using decay products should not increase even if the anoxic retention time is increased (if the recirculation is not increased so as to supply the amount of nitrate needed for denitrification). The calculated value of η_1 would, if it is not possible to determine how much of the anoxic tank is actually denitrifying, indicate the maximum fraction of denitrifiers in the system if the anoxic tank is fully utilised for denitrification.

A supplemental way of determining the maximum fraction of denitrifiers would be to compare the denitrifying activity necessary in order to denitrify the available nitrate within the anoxic tank with the aerobic respiration of the activated sludge. The amount of denitrifiers selected by the system would be expected to increase until they reach the capacity of denitrifying all the available nitrate in the anoxic tank. Above this level no further selection pressure towards a denitrifying culture exists. The minimum of the denitrifier fraction thus calculated and that yielded by the model of Henze should indicate the maximum denitrifier fraction to be expected in a system using a specific mode of operation. Above this level an increase of the anoxic solids retention time ratio does not increase denitrification and thus will not increase the fraction of denitrifiers.

The denitrification rate can be estimated using respiration, suspended solids concentration, flow and other conditions from EFOR simulations (Table 3.3) and the fraction of denitrifiers if limited by growth (Figure 3.14) or by the supply of nitrate (Figure 3.16). Thus the influence of different factors on the denitrification rate in an activated sludge system may be estimated (Table 3.4). In the growth decay model all influent soluble substrate is used by the denitrifiers ($f_{D,anox} = 1$). The first group of comparisons (1-6) estimates the influence of the solids retention time (and thus the organic loading). A decrease of the solids retention time from 20 to 4 days increases the oxygen respiration by 2.5 and the denitrification rate by about the same amount (2.2-2.9). The denitrifier fraction of the organic matter in the influent wastewater has a great influence if no denitrification takes place in the system. The denitrifying ability of the activated sludge will be a direct reflection of the denitrifying ability of the influent organic matter(11). However, if denitrification occurs in the system (7-10), a selection of denitrifiers will compensate for a lower fraction of denitrifiers in the influent wastewater, decreasing the effect of the influent wastewater.

Operation influences the activated sludge in many ways: here the recirculation ratio and the size of the anoxic tank are studied (the addition of organic matter is not considered). If the recirculation ratio is increased more nitrate enters the anoxic tank and the risk of nitrate limitation decreases. Increasing the size of the anoxic tank allows for more anoxic growth according to equations in Figure 3.14, but with a larger volume the nitrate supply may limit denitrification and thus no further selection for denitrification occurs according to equations in Figure 3.16. In the examples of influence of operation chosen here (12-16) operation influences the denitrification rates from not at all to by a factor 2.4. In case 16 the denitrifier fraction of the organic matter of the influent wastewater was higher than was necessary in the activated sludge system; no further selection towards denitrification was thus motivated.

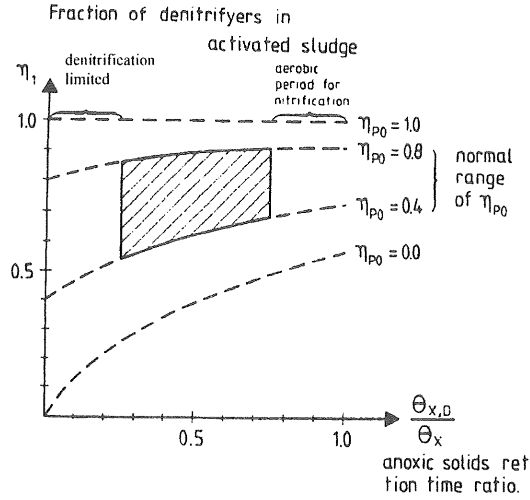
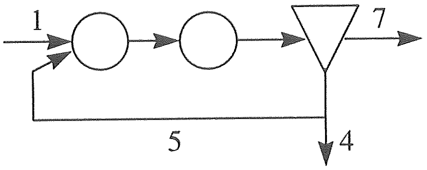
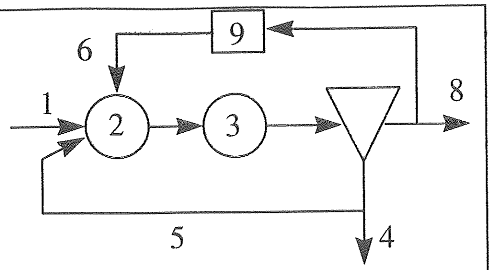


Figure 3.15 Fraction of denitrifyers expected in a single sludge nitrifying denitrifying activated sludge system (from Henze, 1987).



Non-nitrifying activated sludge system with two aerated activated sludge tanks (reactors).



Non-nitrifying activated sludge system with one anoxic (2) and one aerated (3) reactor. Nitrification in a separate unit (9).

The amount of ammonium available to nitrification, and thus to denitrification can be estimated by the effluent ammonium mass flow from a non- nitrifying activated sludge system. In a system including nitrification, part of this ammonium is nitrified and recirculated to the first, anoxic, part of the activated sludge tank. If nitrification takes place in a separate unit in the recirculation stream the maximum amount of nitrate available to denitrification will be:

$$\textit{mass flow of nitrate to anoxic zone} = S_{7,N} \cdot Q_1 \cdot \frac{Q_6}{Q_6 + Q_1}$$

The fraction of denitrifiers to which an activated sludge culture may select corresponds to the ratio between the denitrification rate if all the recirculated nitrate is reduced in the anoxic reactor and the heterotrophic respiration using oxygen had this reactor instead been aerated:

$$\eta = \frac{\dfrac{\textit{mass flow of nitrate to anoxic reactor (as oxygen equivalents)}}{\textit{mass of suspended solids in anoxic reactor}}}{\textit{specific heterotrophic oxygen respiration}} =$$

$$= \frac{\dfrac{S_{7,N} \cdot 2.86 \cdot Q_1 \cdot Q_6 / (Q_6 + Q_1)}{V_2 \cdot C_{2,SS}}}{r_{x,O2}} = \frac{S_{7,N} \cdot 2.86 \cdot Q_1 \cdot Q_6}{(Q_6 + Q_1) \cdot V_2 \cdot C_{2,SS} \cdot r_{x,O2}}$$

Using results of simulations from Table 3.3 for a situation with 4 days SRT, a recirculation 3 times the influent flow and the anoxic reactor equal to 50 % of the total activated sludge volume, the maximum fraction of denitrifiers which the system would select to would be:

$$\eta = \frac{S_{7,N} \cdot Q_1 \cdot Q_6 \cdot 2.86}{(Q_6 + Q_1) \cdot V_2 \cdot C_{ss,2} \cdot r_{x,O2}} = \frac{32 \cdot 1 \cdot 3 \cdot 1008 / 24 \cdot 2.86}{(3+1) \cdot 500 \cdot 0.5 \cdot 2.3 \cdot 8.5} = 0.59$$

Symbol		unit	Subscripts 1-9 refer to positions in flow schemes above.
V	Volume of reactor	m^3	
Q	Flow	m^3/h	
η	Fraction of denitrifiers	1	
C_{ss}	Suspended solids	kg SS/m^3	
$r_{X,O2}$	Heterotrophic oxygen respiration	g O_2/(kg SS·h)	
S_N	Soluble nitrogen	g N/m^3	

Figure 3.16 Maximum fraction of denitrifiers if limited by the supply of nitrate to the anoxic reactor.

Table 3.4 Influence of solids retention time, influent wastewater and operation of the activated sludge system on the expected denitrification rate.

Investigated factor	Chosen values of factors. The underlined one is the one varied.				Heterotrophic resp.	Denitrifier fraction			Denitrification rate	Influence of factor
Simulat. no.	Organic loading SRT	Infl. WW η	Operation/ configuration Q_6/Q_1	V_2/V	From Table 3.3 $r_{x,O2}$	Limited by growth Figure 3.14	Limited by nitrate Figure 3.16	Limiting		Ratio
	d	-	-	-	g O_2/(kg SS·h)	-	-	-	g O_2/(kg SS·h)	-
SRT										
1	4	0.2	2	0.5	8.5	0.54	**0.51**	0.51	4.3	2.9/1
	20	0.2	2	0.5	3.4	0.61	**0.44**	0.44	1.5	
2	4	0.8	3	0.5	8.5	0.89	0.59	0.8*	6.8	2.5/1
	20	0.8	3	0.5	3.4	0.90	0.50	0.8*	2.7	
3	4	0.2	2	0.25	12	**0.47**	0.72	0.47	5.6	2.2/1
	20	0.2	2	0.25	4.8	**0.52**	0.63	0.52	2.5	
4	4	0.8	2	0.25	12	0.87	**0.82**	0.82	9.8	2.6/1
	20	0.8	2	0.25	4.8	0.88	0.71	0.80*	3.8	
5	4	0.2	3	0.25	12	**0.47**	0.82	0.47	5.6	2.2/1
	20	0.2	3	0.25	4.8	**0.52**	0.71	0.52	2.5	
6	4	0.8	3	0.25	12	0.87	**0.82**	0.82	9.8	2.6/1
	20	0.8	3	0.25	4.8	0.88	0.71	0.80*	3.8	
Influent WW. η										
7	4	0.8	2	0.25	12	0.87	0.72	0.80*	9.6	1.7/1
	4	0.2	2	0.25	12	**0.47**	0.72	0.47	5.6	
8	20	0.8	3	0.25	4.8	0.88	0.71	0.8*	3.8	1.5/1
	20	0.2	3	0.25	4.8	**0.52**	0.71	0.52	2.5	
9	20	0.8	1	0.50	4.8	0.90	0.33	0.8*	3.8	2.4/1
	20	0.2	1	0.50	4.8	0.61	**0.33**	0.33	1.6	
10	4	0.8	3	0.5	8.5	0.89	0.59	0.80*	6.8	1.5/1
	4	0.2	3	0.5	8.5	**0.54**	0.59	0.54	4.6	
11 No denitrifica-		0.8						0.8*		4/1
tion in act. sludge		0.2						0.2*		
Operation										
12	4	0.2	3	0.25	12	**0.47**	0.82	0.47	5.6	1.8/1
	4	0.2	1	0.5	8.5	0.54	**0.38**	0.38	3.2	
13	4	0.8	3	0.25	12	0.87	**0.82**	0.82	9.8	1.4/1
	4	0.8	1	0.5	8.5	0.89	0.38	0.8*	6.8	
14	20	0.2	3	0.25	7.2	**0.61**	0.71	0.61	4.4	2.4/1
	20	0.2	1	0.5	5.4	0.52	**0.33**	0.33	1.8	
15	20	0.8	3	0.25	7.2	0.88	0.71	0.8*	5.7	1.3/1
	20	0.8	1	0.5	5.4	0.90	0.33	0.8*	4.3	
16	4	0.8	3	0.5	8.5	0.9	0.59	0.8*	6.8	1/1
	4	0.8	1	0.5	8.5	0.9	0.59	0.8*	6.8	

* assuming that η never is lower in the activated sludge system than in the influent wastewater (WW).

3.3.5 Conclusions

In natural and constructed anoxic systems denitrification rates vary greatly, mainly depending on the long-term supply of nitrate and organic matter. The heterotrophic and denitrifying activity in activated sludge systems has been shown to increase with increased organic loading (lower solids retention time).

The denitrification rate achievable in an activated sludge system can be assumed to be the result of at least three different groups of factors. Respiration will increase with increasing organic loading. The denitrification rate, in relation to the aerobic respiration, will depend on two main factors: the amount of denitrifiers supplied by wastewater or otherwise and the selective pressure of the system towards denitrification. The selection of denitrifiers in the activated sludge system can to a certain extent be controlled by operation and process configuration. These factors affect each other. When the supply of denitrifiers to the system is large, the selection in the system becomes less important, but when the supply of denitrifiers is poor the effect of selection will be greater.

An activated sludge system with a low solids retention time where a large fraction of the activated sludge volume is anoxic can be expected to yield a high denitrification rate. A large supply of nitrate and organic matter to the anoxic zone will increase denitrification.

3.4 Technical examples of denitrification in separate cultures

3.4.1 Introduction

Construction of a non-nitrifying activated sludge system for denitrification using wastewater organics as the only carbon source has some interesting aspects. Since the activated sludge system does not have to include nitrification the growth rate of the nitrifiers is of no concern. The solids retention time (SRT) of the activated sludge system can be in the same order as that of a conventional activated sludge system designed for removal of carbonaceous material, 3-5 days, instead of the 10-20 days which are needed when nitrification must be included.

The lower solids retention time, and thus higher organic loading, possible in a separate sludge system, should give higher denitrification rates than the single sludge system. Since the aerated SRT does not have to be high in order to allow for nitrification, a large fraction of the activated sludge tanks may be non-aerated. One consequence is that the energy input for aeration will be lower in a separate sludge denitrifying activated sludge system than in a single-sludge system, and another is that a large part of the organic matter will be used under anoxic conditions. However, the lower solids retention time gives less time for the particulate organic matter to be hydrolysed and thus become available to denitrification. A consequence of this is that the excess sludge withdrawn from the system will be less stabilised in a system with 3-4 days SRT than in a system with 10-20 days SRT. The choice of activated sludge system will affect the need for, and economy of, stabilisation of the excess sludge. Excess activated sludge from a system with a low SRT may be useful for digestion whereas sludge from a system with a very high SRT may not need stabilisation at all. Both aeration and production of methane are important parts of the overall energy budget of a WWTP (Ødegaard, 1995).

The potential advantages of the denitrifying, non-nitrifying, activated sludge system cannot be utilised without nitrate being present for denitrification. Two situations where a denitrifying, non-nitrifying, activated sludge system can be operated are (1) when the influent wastewater already contains nitrate or (2) when nitrification takes place in some other part of the system. The first situation may occur when treating industrial wastewater containing large amounts of nitrate. The second option, nitrification in some other part of the system, has been investigated by some authors.

Municipal treatment incorporating separate denitrifying activated sludge systems using wastewater organic carbon for denitrification is discussed in section 0. Since many of the advantages of separate nitrifying and denitrifying cultures can be utilised also in fixed film systems a few examples of fixed film applications will be referred to in section 0.

3.4.2 Activated sludge systems

Balakrishnan and Eckenfelder (1970) proposed a system involving denitrification in activated sludge and nitrification in a trickling filter. In this system return activated sludge was contacted with the settled raw wastewater. After sedimentation the clarified water was pumped to a nitrifying trickling filter (see Figure 3.17). The activated sludge passed on to the activated sludge tank, to which the nitrified effluent from the trickling filter was led for denitrification. The denitrifying activated sludge system was demonstrated in a laboratory activated sludge system (Balakrishnan and Eckenfelder, 1969 C) and separate stage nitrification was demonstrated in an activated sludge system (Balakrishnan and Eckenfelder, 1969 A) and in a trickling filter (Balakrishnan and Eckenfelder, 1969 B).

Discussing this process (with a separate nitrifying activated sludge unit) Barnard (1973) states that the system shows rather good overall denitrification results and that "where the form of the remaining nitrogen is of little consequence this system should work satisfactorily". However, if a nitrification unit would have to be added in order to nitrify remaining ammonium (Figure 3.18) "this would render the system so complex as to rule it out altogether". Barnard compares the system with a three sludge system with methanol used for denitrification and concludes that the investment costs would be likely to be higher than the cost saving by using no methanol. Barnard instead advocated an alternating anaerobic/aerobic single sludge system for nitrogen removal using an internal carbon source.

A discontinuous version of the process was investigated by Jones et al. (1990 A and B). The system consists of two sequencing batch reactors (SBR) and a nitrifying unit, for instance a nitrifying trickling filter (NTF) or a rotating biological contactor (RBC). The cycle begins with wastewater being fed to one of the SBR:s. After initial aeration and mixing the sludge is allowed to settle and the supernatant is pumped to the RBC. The water is nitrified in this and then returned to the other SBR where denitrification takes place using organic matter from the wastewater of the previous cycle. After sedimentation the supernatant is decanted. In the following cycle the flow is reversed (Figure 3.19).

The process was also investigated by Wanner et al. (1992) and the process of Balakrishnan and Eckenfelder was suggested under the heading "New process design for biological nutrient removal". Sekoulov et al. (1990) investigated a similar system (Figure 3.20). In this case the denitrification reactor would be housed within the secondary sedimentation tank.

The process suggested by Balakrishnan and Eckenfelder appears in an overview of two-step biological wastewater treatment plants for nitrogen removal (Bever et al., 1994 A), at Kläranlage Ahrensburg. Problems discovered during laboratory and pilot plant investigations included ammonium bypassing letting the mixed liquor bypass the NTF. Nitrification and thus nitrogen removal was not complete. This problem was minimised by thickening the bypassed sludge by flotation. Another problem was that the denitrification capacity of the sludge from the first activated sludge unit was not sufficient to denitrify more than 30 g NO_3-N/m^3. This process is also included in a split flow multiple stage treatment process suggested by Buß et al. (1994) for the WWTP Radegast (Figure 3.21). Half the influent wastewater is directed to a denitrifying trickling filter to which nitrified effluent from the following nitrifying trickling filter is recirculated. The remaining influent wastewater is contacted in a highly loaded activated sludge system. The supernatant from the sedimentation tank of the highly loaded activated sludge system is nitrified in the nitrifying trickling filter and subsequently reunited with the solids in a denitrifying activated sludge system. The system is complex but, especially if advanced control is used, it should be possible to make good use of the carbon source of the influent wastewater for denitrification.

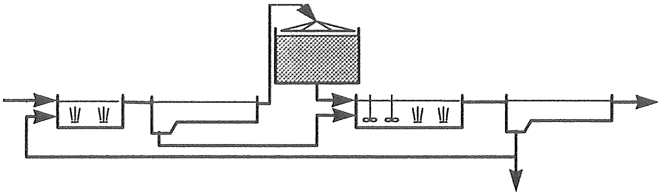


Figure 3.17 The modified activated sludge process as proposed by Balakrishnan and Eckenfelder (1970). Symbols as in Figure 1.1.

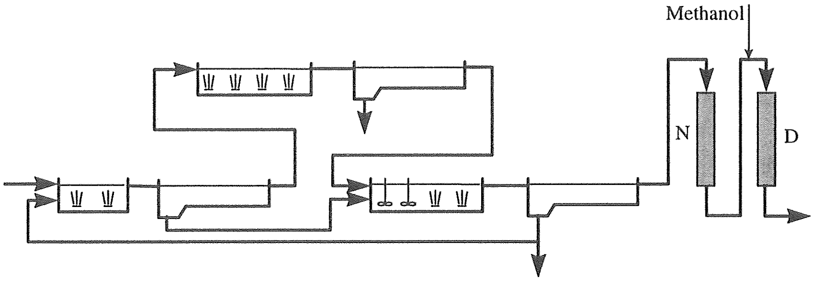
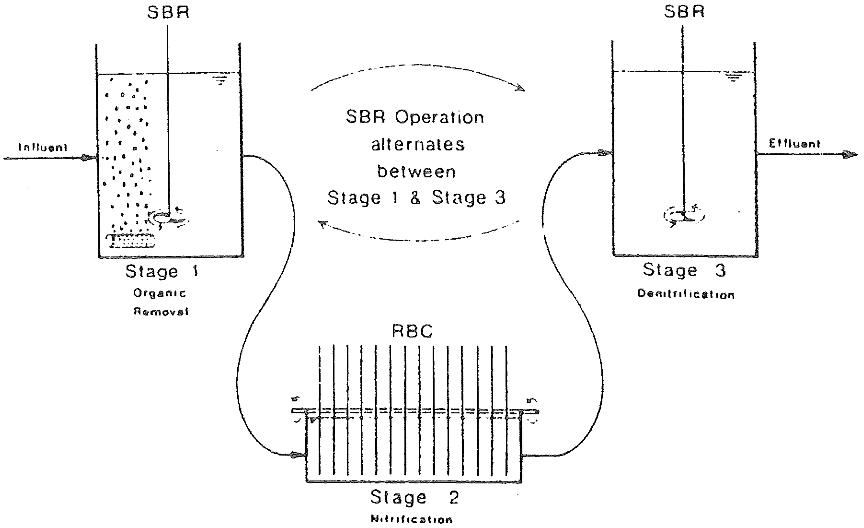


Figure 3.18 Modification of the process of Balakrishnan and Eckenfelder in order to obtain complete nitrogen removal as discussed by Barnard (1973). Symbols as in Figure 1.1 with the addition of units for nitrification (N) and denitrification (D).



Operation of the proposed three-stage system for nitrogen removal consisting of an SBR for organics removal / sequestering (Stage 1) and for denitrification (Stage 3), and an RBC for nitrification (Stage 2).

Figure 3.19 A discontinuous non-nitrifying activated sludge system (from Jones et al. 1990 A).

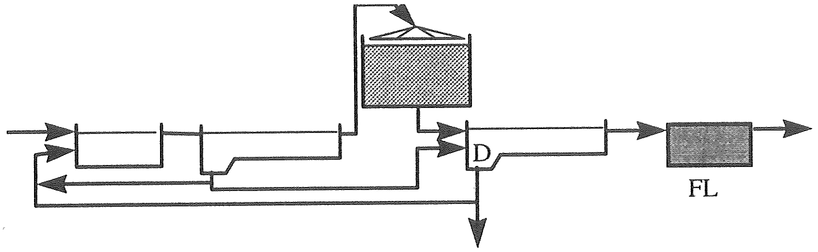


Figure 3.20 System for nitrogen removal suggested by Sekoulov et al. (1990). Symbols as in Figure 1.1 with the addition of the flotation unit (FL) and the combined sedimentation and denitrification tank (D).

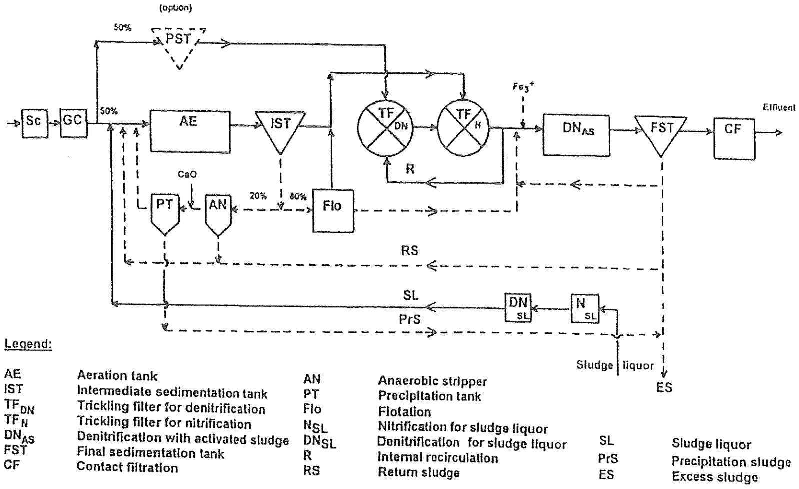


Figure 3.21 Flow chart of the WWTP Radegast (from Buß et al., 1994).

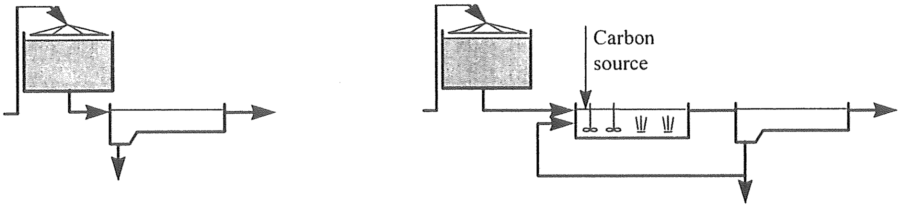
Aspegren (1995) discusses the use of a high-loaded activated sludge system for biological phosphorus and nitrogen removal at the Sjölunda WWTP (Malmö, Sweden). In one scheme recirculation of nitrified trickling filter effluent to the activated sludge system is included and in another bypassing mixed liquor past the trickling filter as suggested by Balakrishnan and Eckenfelder. The two schemes (though without biological phosphorous removal) have also been tested in pilot scale in relation to the Ingolstadt WWTP (Schreff and Wilderer, 1997).

Planning for the extension of the Kenten WWTP in Germany for nitrogen removal, Dichtl et al. (1994) presented several alternatives for integration of the trickling filters in the future treatment plant. The existing WWTP included a conventional activated sludge plant followed by partially nitrifying trickling filters. Several combinations of activated sludge systems and trickling filters were presented. In all the presented alternatives the activated sludge systems were designed for nitrification, denitrification and biological phosphorus removal. The trickling filters were either used as polishing units to remove residual ammonium or placed parallel to the activated sludge system. In these systems including nitrifying trickling filters in the process does not result in a gain in the form of smaller activated sludge tanks. In the chosen solution the trickling filters were converted into activated sludge tanks in a system involving denitrification of return sludge in converted sedimentation tanks. At another trickling filter plant in Germany, Moers-Gerdt, the trickling filters were?) included in the expansion for nitrogen removal in the form of down-stream nitrifying trickling filters aimed at nitrifying residual ammonium from the nitrifying and denitrifying activated sludge plant (Fruhen et al., 1994 B). Residual denitrification would take place in a final filter unit using an added carbon source. The authors concluded that the down-stream nitrification unit allowed a tight design of the activated sludge system.

Christensen (1991) presented various possibilities of extending a large trickling filter plant for phosphorus and nitrogen removal. Four basic schemes were presented, one of which included post nitrification in a trickling filter and recirculation of the nitrified effluent to an activated sludge system for denitrification. Phosphorus removal was to take place in a separate post precipitation step. In the process finally chosen 2/3 of the flow was led directly to an activated sludge system including biological phosphorus and nitrogen removal (Bio-Denipho, Bundgaard et al., 1989) . The remaining 1/3 of the flow was led to the existing trickling filters for removal of carbonaceous material and nitrification before being led to the anoxic zone of the activated sludge tank for denitrification. When the selection was made effluent standards set were 2 mg P/l and 5 mg N/l. In order to reduce effluent nitrogen to 5 mg N/l with a pure recirculation process the recirculation ratio would have to be high causing severe problems with oxygen recirculation. However a few years later effluent standards were changed to 8 mg N/l and 0.5 mg P/l and in order to meet this the plant was to be extended with a contact filtration unit for precipitation with $FeCl_3$ or $FeSO_4$. If these standards had been set from the start the choice of process might have been different since the slightly higher effluent nitrogen limit might have allowed for a recirculation process and the stricter effluent phosphorus limit might have been difficult to guarantee using biological phosphorus removal.

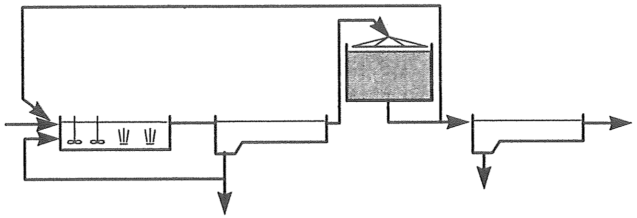
Dahlem (1986) discussed different two-step combinations of trickling filters and/or activated sludge units for removal of organic matter and in some cases for nitrification or nitrogen removal. In many cases an activated sludge system was followed by a trickling filter or another activated sludge system. Situations where multistage treatment systems may be considered are, according to Dahlem: when an existing plant is extended for nitrification or further removal of organic matter; when concentrated or difficult wastewater is to be treated; to save energy or to reduce building costs in large WWTP:s.

A system including post nitrification in trickling filters and recirculation of the nitrified effluent to a denitrifying activated sludge system has been discussed or implemented on several occasions in recent years. In most cases the process is introduced or discussed as a possibility of extending a trickling filter plant for nitrification and denitrification (Christensen, 1991, Krauth and Roth, 1991, Maisch and Schwenter, 1994, Schleypen and Nordmann, 1994) but the process may also be used when expanding a conventional activated sludge plant for nitrification and denitrification (Lyngå, 1991, Lyngå and Balmér, 1992).

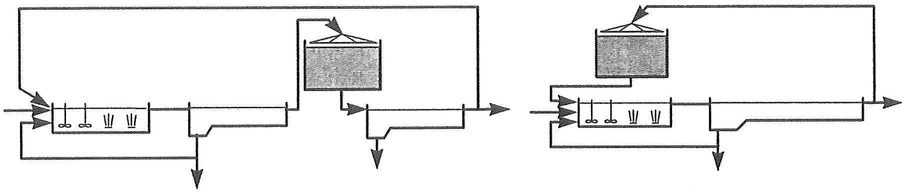


a) Existing plant, Sindelfingen

b) Post denitrification in activated sludge, Sindelfingen



c) Predenitrification in activated sludge, Sindelfingen



d) Expansion of Roth WWTP.

e) Expansion of the Rya WWTP

Figure 3.22 Alternatives for the expansion of Klärwerkes Sindelfingen for nitrogen removal (Maisch and Schwentner, 1994), the Roth WWTP (Schleypen and Nordmann, 1994) and the Rya WWTP (Lyngå, 1991, Lyngå and Balmér, 1992). Symbols as in Figure 1.1.

Maisch and Schwentner (1994) discussed possibilities of extending the Sindelfingen trickling filter plant for nitrogen removal. At present the trickling filters remove 50% of the nitrogen due to high organic loading (Figure 3.22 a). Due to a variation of influent wastewater quality nitrogen removal in the trickling filter is more complete on Tuesdays-Saturdays than on Sundays and Mondays. Plans for the extension of the trickling filter plant for nitrogen removal include different solutions depending on the price of an external carbon source. As a first step a post denitrification activated sludge unit with an external carbon source would be built (Figure 3.22 b). Later a predenitrification activated sludge system may be built. As a final solution, especially if the price of the external carbon source rises considerably, the predenitrifying activated sludge system may be extended and replace post denitrification altogether (Figure 3.22 c).

Schleypen and Nordmann (1994) describe retrofitting the Roth wastewater treatment plant according to the same principle, but in this case the final sedimentation tanks were included in the recirculation loop (Figure 3.22 d). From May to October 1993 the effluent from the plant contained 11.9 g NO_3-N/m^3 and 0.2 g NH_4-N/m^3 (median values). This process (as in Figure 3.22 c and d) is also reported as planned or existing in Kläranlage Spenge (since 1988) and in Kläranlage Brannenburg in Bavaria (since 1992) (Bever et al., 1994 B). Kläranlage Spenge is reported to remove 67 % of the influent nitrogen (Krauth and Roth, 1991).

If the form of the effluent nitrogen is of no concern, that is if the limit or target is set as removal of nitrogen or effluent concentration of nitrogen, the nitrification unit can be placed within the recirculation loop (Figure 3.22 e). Pilot investigations of this process have taken place at the Rya WWTP since 1990 (Lyngå, 1991, Lyngå and Balmér, 1992).

The systems of Figure 3.22 c, d and e include nitrification of effluent wastewater in a trickling filter. This should give stable nitrification with a lower degree of sensitivity to low wastewater temperature, toxic substances, organic and hydraulic loading than would nitrification in a single sludge activated sludge system. The recirculation of nitrified wastewater to a denitrifying non-nitrifying activated sludge system with a relatively low SRT should give relatively high denitrification rates, thus minimising the activated sludge volume. However all these systems involve recirculation of clarified effluent wastewater. All nitrogen to be removed by denitrification must be recirculated and thus will cause an additional hydraulic loading on the clarifiers. Additionally the effluent from the trickling filters will contain a considerable amount of oxygen which is recirculated to the anoxic zone of the activated sludge plant. This oxygen will be consumed by the heterotrophic organisms in preference to nitrate. If increased nitrogen removal is demanded recirculation must be increased, but when the recirculation rate is increased the amount of recirculated oxygen will increase linearly whereas the nitrogen recirculation, being a function of the recirculated fraction of effluent wastewater, will increase at a lower rate. Increasing the recirculation rate will thus also increase the ratio of recirculated oxygen to nitrate. The consequence of this is that these systems are only suitable when an intermediate level of nitrogen removal is demanded (nitrogen removal of 50 - 60 % or effluent total nitrogen standards of 10-15 g N/m^3). If stricter limits are set the systems will have to be supplemented with post denitrification.

Another system, though not with a completely separate culture, is the "hybrid process (Gamperer, 1997). A two stage activated sludge system is planned with variable sludge and water recirculation options making it process-wise quite similar to the above processes. During dry-weather conditions effluent wastewater would be recirculated to the first, highly loaded, activated sludge system and during high loading conditions the first activated sludge system would protect the second, nitrifying, activated sludge system from high organic loading.

The possibilities of combining a denitrifying non-nitrifying culture with a separate nitrification unit have in the examples discussed above focused on two main systems, one based on recirculation of nitrified effluent from a post nitrification unit (I) and the other based on quick separation of organic matter from wastewater for later use when denitrifying the nitrified supernatant (II). Some characteristics of these systems are summarised in Table 3.5. Both systems give reduced activated sludge volumes in comparison with single-sludge nitrification/denitrification systems. The nitrifying culture is less vulnerable to washout, low temperature and toxic substances. Finally both systems should give higher denitrification rates than the single-sludge system. The recirculation system is a simple, easily managed system where demands on nitrogen removal are moderate.

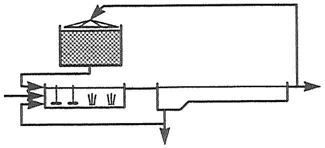
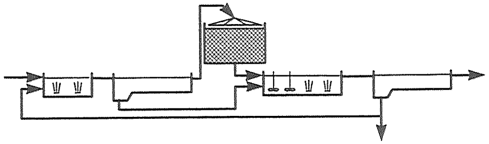
A conventional activated sludge or trickling filter plant can be converted into a nitrogen removal plant using these principles. Where further nitrogen removal is required, however, recirculation rates will become unreasonably high using system I and cause excessive hydraulic loadings on the sedimentation tanks as well as high recirculation of oxygen to the anoxic activated sludge system. In this case system I has to be supplemented with post denitrification or system II used. In system II nitrogen removal does not depend on recirculation and thus problems with oxygen from trickling filters entering anoxic tanks and with high hydraulic loading on the sedimentation tanks are smaller. However, system II needs two separation units which both have to be designed for the full flow of the biological treatment plant.

3.4.3 Other separate nitrifying and denitrifying units

Systems for separate culture denitrification and nitrification utilising internal carbon sources are realisable and exist in many configurations. The next question is which components can be used to build the systems. The separate denitrification units demonstrated in the systems in section 3.4.2 are sequencing batch reactors and continuous flow activated sludge units (and in one case trickling filters). In many ways the separation of bacterial cultures is easier in fixed film systems than in suspended cultures. Much has been published on this subject. A few examples will be given here.

Separate nitrification in the examples of section 0 has mainly been performed in nitrifying trickling filters. Post nitrification in trickling filters has been shown to be successful by several authors (Balakrishnan and Eckenfelder, 1969 B, Gujer and Boller, 1983, Parker and Richards, 1986, Boller and Gujer, 1986, Okey and Albertson, 1989, Parker et al., 1990, Andersson, 1990, Andersson et al. 1994, Parker et al. 1995). At the Rya WWTP pilot studies have also shown post nitrification in a trickling filter to be a feasible process (Mattsson and Rane, 1993, Hansson, 1994). At low organic loadings (5 g $BOD_5/(m^2 \cdot d)$), the growth rate of the heterotrophic organisms is low enough to allow for a stable nitrifying culture. In most of the above references nitrification is close to complete at ammonium loadings up to about 1.2 g $N/(m^2 \cdot d)$ under normal conditions (Parker and Richards, 1986). As packing materials range from 100 to 230 m^2/m^3 the volumetric nitrification rate ranges from 0.12 to 0.30 kg $N/(m^3 \cdot d)$. In some cases problems with grazing of biofilm by higher organisms are encountered. Parker et al. (1989) suggest the use of flooding and backwashing of the trickling filter in order to prevent the growth of predator organisms. However, predators have not been shown to cause serious problems in recent Swedish nitrifying trickling filter pilot tests (Parker et al., 1995).

Table 3.5 Systems for nitrogen removal including separate nitrification and denitrification using wastewater organics for denitrification.

Principle of system	Example of process configuration	Characteristics of configuration (+ advantages, - disadvantages)	Full-scale realised or planned applications where principle is included in configuration
I. Nitrified effluent from post nitrification unit recirculated to separate culture pre-denitrification unit.		+Smaller activated sludge volume than in single sludge systems +Nitrification independent of influent organic loading and less sensitive to wastewater temp. and toxic substances in wastewater. +High denitrification rate -N removal dependent on recirculation -Oxygen from trickling filter accounts for about 1/3 of the electron acceptor capacity of the recirculated water - High hydraulic loading on separation units.	Kläranlage Spenge (G) Kläranlage Brannenburg (G) Sindefingen (G) Roth (G) Rya WWTP(S) WWTP Radegast (G) Included in alternative schemes: Sjölunda WWTP (S) Ingolstadt (G)
II. Settled wastewater contacted with culture, separated and then reunited with nitrified supernatant for denitrification.		+ Smaller activated sludge volume than in single sludge systems. +Nitrification less sensitive to influent organic loading, wastewater temp. and toxic substances in wastewater. +High denitrification rate -Two separation steps and three biological reactors + Nitrogen removal not dependent on recirculation (-)Oxygen from trickling filter accounts for about 1/10 of the electron acceptor capacity of the nitrified water	WWTP Radegast (G) Included in alternative schemes: Sjölunda WWTP (S) Ingolstadt (G)

S=Sweden , G=Germany

Biological aerated filters (BAF) contain submerged fixed or moving carrying material upon which organisms grow. Biological aerated filters are often considered for post nitrification. Several constructions are commercially available, differing from each other in flow direction of wastewater, filter media etc. Tschui et al. (1994) compared three brands of biological aerated filters for tertiary nitrification (Biofor, Biostyr and Biocarbone) and concluded that aerated biofilters represent a reliable and efficient treatment step for tertiary nitrification. Due to high specific surface area the volumetric nitrification rates are high. Under optimised conditions nitrification rates of 0.4 to 1.5 kg N/(m^3·d) were obtained, but available results under practical operational conditions revealed lower rates. One factor may be that nitrification capacities of fixed film systems are shown to drop after a long period of ammonium deficiency. In studies of Tschui et al. (1994) and Fruhen et al. (1994 A) nitrification capacities dropped by about 30 % after 4 to 6 weeks of extremely low ammonium loading. Two-stage biofilm processes are suggested in order to first obtain removal of carbonaceous material and then nitrification (Carrand et al., 1990, Meaney and Strickland, 1994). Paffoni et al. (1990) compared two biological aerated filters (Biofor and Biocarbone) for post nitrification in Paris. They performed equally well, nitrifying 0.5 kg N/(m^3·d).

A comparison between different alternatives for post nitrification was made by Boller et al. (1994). Biofilter systems, rotating biological contactors, trickling filters and activated sludge systems were compared with respect to construction area, construction volume and energy requirement. Biofilters were deemed most advantageous with respect to construction area and volume, whereas rotating biological contactors were the best from an energy point of view. Trickling filters were calculated to need large construction volume and area and consume a large amount of energy (the high energy consumption was mainly due to the need to recirculate water in order to obtain high hydraulic loadings). However it was stated that “Conclusions from case studies may not be generalized. Local conditions, above all existing facilities, temperature and effluent requirements asking for additional treatment steps like denitrification and infiltration, can further influence the choice of the most suitable reactor for nitrification.” This is true; the data used for design of the trickling filter alternative (from Boller and Gujer, 1986) shows lower nitrification rates than does the data of Okey and Albertson (1989), Parker et al. (1990) and Andersson et al. (1994) and data from the Rya WWTP (Wik et al., 1995). Using more aggressive design, assuming a nitrification rate of 1 g N/(m^2·d) (instead of approximately 0.5), a carrying material with a specific surface area of 230 m^2/m^3 and a height of 7.2 m, the nitrifying trickling filters would need a volume of 753 m^3 (instead of 1644 m^3) and have a ground area of 105 m^2 (instead of 274 m^2). This should reduce the need for recirculation since the mean hydraulic loading without recirculation would increase from 0.7 to 1.8 m/h and the peak hydraulic loading from 1.0 to 2.6 m/h thus influencing the energy requirements. In some cases, nitrifying trickling filters may therefore be more attractive than they would appear from the comparison of Boller and Gujer.

In other contexts separate denitrification units have been demonstrated in many forms, such as suspended carrying media, biological filters and packed columns (Ødegaard et al., 1994, Meaney and Strickland, 1994, Çeçen and Gönenç, 1995, la Cour Jansen, 1994, Dee et al., 1994). In moving bed biofilm reactors (Ødegaard et al., 1994) the bacterial cultures can be kept in separate parts of the reactor by trapping the carrying material with the aid of a sieve. Here separate nitrifying and denitrifying environments, and thus separate bacterial cultures, can be created. Nitrification rates in the separate nitrifying compartment are typically 0.3 kg N/(m^3·d). Watanabe et al. (1994) demonstrated separate culture nitrification and denitrification in the same reactor by stacking two rotating biological contractors (RBC). On the top, partially submerged RBC, a nitrifying culture was maintained, whereas the bottom, submerged RBC carried heterotrophic organisms capable of denitrification.

Both the recirculation process and the process of Balakrishnan and Eckenfelder can be performed in biofilm processes. An example of the recirculation process implemented with biofilm processes is included in the process proposed for WWTP Radegast (Figure 3.21) where nitrified effluent from a nitrifying trickling filter is recirculated to a denitrifying trickling filter. Bever et al. (1994 B) list three plants (Neuhausen, Esslingen-Berkheim and Grünstadt) where nitrified effluent from a post-nitrifying activated sludge unit or trickling filter is recirculated to a denitrifying trickling filter. The denitrifying trickling filters are covered and the effluent from the trickling filter passes an airlock in order to avoid oxygenation of the trickling filters. The plants receive 30-40 g NH_4^+-N/m^3 and the effluent contains on average 0 - 2 g NH_4^+-N/m^3 and 14 - 17 g NO_3^--N/m^3. The process combination could also be implemented in biological filters or in reactors with suspended carrying media, where the processes and loadings can be adjusted separately for the nitrifying and the denitrifying unit.

One way of avoiding, or minimising, the impact of recirculation would be to use systems based on the contact stabilisation process suggested by Balakrishnan and Eckenfelder (Figure 3.17). Since these systems are not based on recirculation of nitrified wastewater a high degree of nitrogen removal is feasible. The ratio of the oxygen to nitrate mass flow to the anoxic tank will be lower than in a system based on recirculation. However, a conflict in the system is caused by the wish to maximise the removal of organic matter from the water to the trickling filter while minimising the degradation of organic matter in the contact tank. A few applications using a denitrifying activated sludge system were mentioned above (Figure 3.19, Figure 3.21). The principle of the Balakrishnan and Eckenfelder process may also lend itself well to modern biofilm technology and control systems. It may be possible to use a set of biofilm reactors, for instance biological filters or reactors filled with a suspended carrying medium, and to intermittently lead wastewater and nitrified water to them thus making use of the stored organic matter from the influent wastewater.

3.4.4 Conclusions

Two process configurations incorporating a denitrifying non-nitrifying activated sludge system in a municipal wastewater treatment plant for nitrogen removal have been suggested or implemented.

In one process configuration wastewater is contacted with activated sludge and then separated. The supernatant is nitrified in a fixed film unit before again being mixed with the activated sludge (and organic matter adsorbed on the activated sludge) in an anoxic tank for denitrification (Figure 3.17). Nitrogen removal in this system depends on the carbon source available in influent wastewater and on the amount of ammonium bypassing the trickling filter with the mixed liquor. Since the system is not based on recirculation the impact of oxygen from the nitrifying fixed film unit on denitrification should be limited. However, the system uses five different reactors: two activated sludge tanks, two sedimentation tanks and a nitrifying fixed film unit.

The other process is based on recirculation of effluent wastewater from biological treatment through a nitrifying fixed film unit to an anoxic section of the activated sludge system for denitrification (Figure 3.22 c, d and e). In this case nitrogen removal is limited by the recirculated flow and by the carbon source of the influent wastewater. This type of system should show high denitrification rates but since the system is based on recirculation the impact of oxygen from the nitrifying fixed film unit on denitrification may be serious. The system consists of three different reactors: an activated sludge tank, a sedimentation tank and a nitrifying fixed film unit. Due to recirculation the hydraulic loading on the reactors will be high. This has to be taken into account when designing the sedimentation tank(s).

Both process configurations are mainly discussed or implemented using denitrifying activated sludge systems and nitrifying trickling filters, but these could be replaced by other separate denitrifying and nitrifying unit processes. Some experience of these process configurations exists, but long-term experience of full-scale applications is sparse.

3.5 Summary

Denitrification takes place in natural and constructed systems if environmental conditions are fulfilled, bacteria capable of denitrification are present and nitrate and organic matter are supplied. The environment must be anoxic and pH, concentrations of inhibiting substances, temperature etc. not outside the ranges normally set by heterotrophic organisms.

High denitrification rates can be expected in an activated sludge system with a high organic loading.

In areas where the wastewater temperature is low, high organic loading is not easy to realise in a single sludge nitrifying and denitrifying activated sludge system, and without nitrification there is normally no nitrate supply for denitrification.

In order to use a highly loaded activated sludge system for denitrification in a municipal wastewater treatment plant, nitrification can be performed in a separate culture. Two main process configurations have been suggested, and in some cases implemented.

In the first system the wastewater is contacted with activated sludge and the organic matter and activated sludge separated from the water. The water is nitrified in a fixed film nitrification unit before being reunited with the activated sludge in a second, anoxic, activated sludge unit for denitrification. One limit to nitrogen removal is set by the amount of soluble ammonium which bypasses the nitrification unit with the activated sludge.

In the second system effluent from an activated sludge system is nitrified in a fixed film nitrification unit before being returned to the first, anoxic, part of the activated sludge tank for denitrification. One limit to nitrogen removal is set by the recirculated flow. This flow adds to the hydraulic loading of the secondary settlers. A large amount of oxygen may be recirculated to the activated sludge tank from the fixed film nitrification unit.

4 Pilot plant experiments

4.1 Introduction

The evaluation of a denitrifying, non-nitrifying, activated sludge system has involved pilot plant and laboratory experiments over a period of several years. The experiments analysed here were a part of a larger project the aim of which was to develop a system for nitrogen removal using a post-nitrifying biofilm system and a non-nitrifying activated sludge system for denitrification. The system includes nitrifying biofilm units as well as the non-nitrifying activated sludge system. The development process is schematically presented in Figure 4.1. In the experiments presented and discussed in this thesis the focus is on the denitrifying non-nitrifying activated sludge system. The experimental work was performed in the pilot plant at the Rya WWTP during four periods (A, B, C and D) between 1990 and 1994. The differences between experimental set-ups are to be found in the degree of process control, and in the extent to which realistic conditions were simulated. Each set of experiments was planned considering the results of the previous experiments. The experimental methods and aims were revised in accordance with experience gained from previous experiments and from the literature.

Apart from evaluating the assumptions made under "Objectives and research strategies" the pilot plant was operated with several other aims. These aims influenced the mode of operation and thus the experimental parameters. Since the proposed process was to our knowledge relatively untried, some points concerning design and operation of a full-scale plant were investigated:

- What is the necessary anoxic volume in order to obtain sufficient denitrification?
- Will the aerated volume be sufficient for removal of soluble organic matter?
- How serious is the problem of recirculation of oxygen from the trickling filter to the anoxic zone? Can this be remedied?
- What is the impact of varying wastewater composition and flow on nitrogen removal?
- How should the system be operated in order to make the best use of influent organic matter?

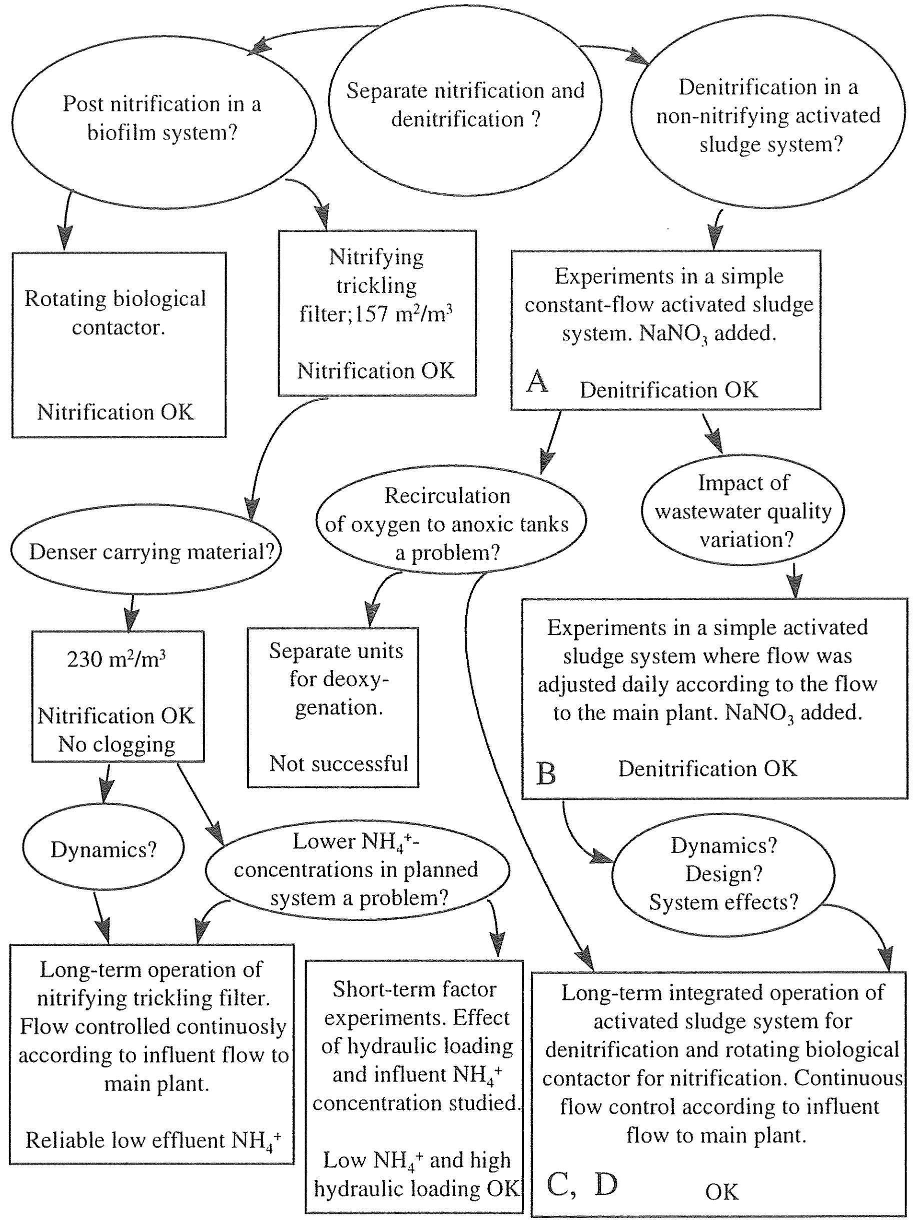


Figure 4.1 Pilot experiments leading to the construction of system for biological nitrogen removal at the Rya WWTP. Experiments A, B C and D are reported in this thesis.

4.2 Materials and methods

4.2.1 Wastewater

The pilot plant was supplied with primary settled wastewater from a pilot primary sedimentation tank to which wastewater was continuously pumped from the main plant influent channel. The sedimentation tank had a square surface and a sludge pit the shape of an inverted pyramid (surface: 12 m^2, volume: about 25 m^3). The flow was about 16 m^3/h rendering the sedimentation tank a surface loading of about 1.3 m/h and a nominal hydraulic retention time of about 1.6 h. The primary sludge was continuously pumped from the bottom of the tank and wasted. Part of the primary settled wastewater was led to the biological pilot plants. The exact amount was regulated with the aid of a weir system or pumped with a frequency-modulated pump. Excess primary settled wastewater was wasted. Scum and sludge floating on the surface were wasted regularly and the weir or pumps inspected and cleaned when necessary.

During these experiments, analysis and sampling of influent wastewater to the main plant, primary settled wastewater from the main plant and primary settled wastewater from the pilot plant were not co-ordinated. Sampling and analysis of influent wastewater to the main plant and effluent from the main plant primary settlers were governed by the criteria set by the authorities and by the need to control the plant. Effluent from the main plant primary sedimentation tanks was pumped through a long sampling pipe to the sampling station. Sampling and analysis of pilot plant primary settled wastewater, on the other hand were co-ordinated with the pilot plant operation. Thus few of the parameters analysed in these wastewater streams are quite comparable. However, with these limitations in mind, results indicate that the separation in the primary sedimentation tank of the pilot plant was somewhat inferior to that in the main plant primary sedimentation tanks (Table 4.1).

The higher COD of the wastewater treated in the pilot plant compared with that of the main plant could be a problem when applying the results of pilot plant experiments to the full scale application. The pilot plant received about 25 % more organic matter measured as COD than the main biological treatment plant. This might give unduly positive expectations on denitrification. This fact was noted quite early during the experiments and given some consideration.

The higher COD of primary settled wastewater to the pilot plant compared with that of the main plant may either be caused by poor sedimentation in the pilot plant or by particulate COD being hydrolysed. In the first case the discrepancy is no great problem to the transfer of the results to the main plant. It should not be difficult to cause poor sedimentation in the main plant primary settlers. In the second case, if the configuration of the primary sedimentation tank or the pipe transporting wastewater to it caused particulate organic matter to be hydrolysed and thus become more available to the denitrifiers this could cause results from pilot plant operation to be unduly optimistic. This may have to be compensated for by addition of an external carbon source or hydrolysed wastewater organics. Results of experiments and discussion of results will be related to the primary settled wastewater actually supplied to the pilot plant.

Table 4.1 Wastewater quality. Average of available data during pilot plant experiments (14th June 1990 to 10th February 1994).

Parameter	Unit	Influent to the Rya WWTP	Effluent from the main plant primary sedimentation tanks	Effluent from the pilot plant primary sedimentation tank
BOD_7	$g\ O_2/m^3$	155	93^*	--
COD	$g\ O_2/m^3$	443	216	271
COD (filtered)	$g\ O_2/m^3$	--	--	133
N	$g\ N/m^3$	28	--	28
NH_4	$g\ N/m^3$	18	--	20
NO_3	$g\ N/m^3$	0.1^*	--	0.2
P	$g\ P/m^3$	5.7	3.9^*	--
PO_4	$g\ P/m^3$	1.6^*	--	--

* Data cover less than half the period.

4.2.2 Experimental set-up and operation

During **experimental period A** (June 1990 - February 1991) the aerated and non-aerated activated sludge volumes were 20 and 11.2 m^3 respectively (Figure 4.4). The non-aerated tank was a completely mixed tank (surface: 4 m^2, volume: 11.2 m^3) with a central top-mounted stirrer (SCABA). Eight out of ten available 2.5 m^3 tanks in series made up the aerated activated sludge volume. Each tank was aerated by four ceramic tubular aerators (Brandol).The final sedimentation tank was a conical vertical flow tank (surface 7.4 m^3, depth 3.3 m).

The pilot plant was operated as a simple, constant flow, activated sludge plant. The primary settled wastewater passed a weir at a flow of about 2.5 l/s and was led to the non-aerated tank where it was mixed with return activated sludge (about 1.4 l/s) and final sedimentation tank supernatant (1.3 l/s) to which sodium nitrate had been added. A gravity flow pipe led the activated sludge from the non-aerated tank to the first aeration tank. The wastewater passed from one tank to the next, through an opening at the top of the tank.

The aeration was controlled automatically using a DO-probe in the fifth aerated tank regulating the total air flow to the eight compartments with a set-point of 1.5 g O_2/m^3. Distribution of air between the different aeration tanks could be set manually using valves on the distribution pipe to each compartment. The aim was to have an equal oxygen concentration in the first seven tanks while keeping a low concentration in the last tank in order to minimise recirculation of oxygen to the non-aerated tank. DO-profiles were measured manually and used to check, and if necessary adjust the distribution of air between the tanks.

A sodium nitrate solution was produced by dissolving sodium nitrate ($NaNO_3$) in effluent wastewater batch-wise in a stirred 500 litre tank. The nitrate nitrogen concentration, in the range of 30 to 70 g N/l, was calculated from a nitrate mass-balance over the tank and checked by laboratory analysis. The nitrate solution was pumped to the activated sludge system by a diaphragm dosing pump, the exact dosage calculated from the change of the fluid level in the tank with time.

Iron sulphate was stored in a stirred tank and dosed continuously with a diaphragm pump yielding about the same dosage as that of the main treatment plant.

The stirrer in the non-aerated tank had two blades and rotated at a speed of 39 min^{-1}. The effect of stirring speed on unwanted aeration of the non-aerated tank was shown to be about 0.2 g $O_2/(m^3 \cdot h)$ (see Figure 4.2). This should not affect denitrification significantly since the denitrification in the tank corresponded to a 200 times greater oxygen consumption (about 40 g $O_2/(m^3 \cdot h)$).

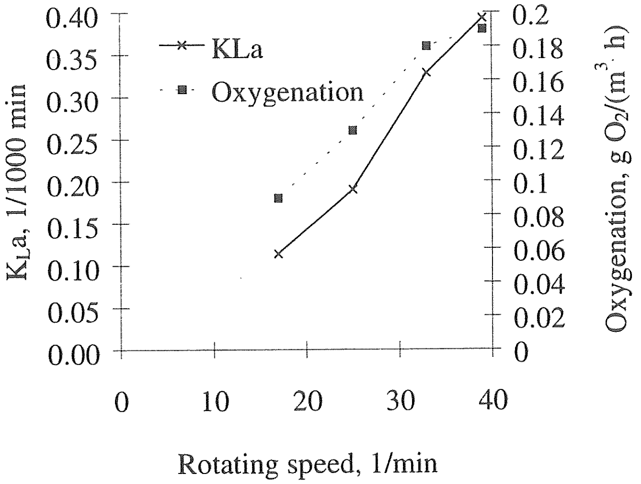


Figure 4.2 Oxygenation of the anoxic tank caused by stirring at 0 mg O_2/l (Karlberg and Eriksson, 1989). The top surface area was 4 m^2 and the volume 11.2 m^3.

Return activated sludge was pumped from the bottom of the final sedimentation tank. Waste activated sludge was extracted from the return activated sludge pipe using an automatic valve about 12 times/day. The waste activated sludge was collected in a 2.7 m^3 container and (on weekdays) measured before daily wastage to the main plant.

The influent and recycled flows, the DO-level in the aerobic compartments, the solids retention time and the hydraulic retention time were kept constant. The only operational parameter varied during this experimental period was the dosage of nitrate.

The wastewater quality and temperature, however, varied. During summer the water was warm (about 20 ° C) and concentrated. During autumn the water temperature was about 14 ° C and during winter about 12 ° C. The winter was rainy, with periods of cold and dilute wastewater.

During the main part of **experimental period B** (April - May 1992) six activated sludge tanks were used for non-aerated or aerated activated sludge (Figure 4.5). The total non-aerated volume in these experiments varied from 2.7 to 8.2 m^3 (one to three tanks). The remaining three to five tanks (7.7 to 12.9 m^3) were aerated. An overflow from the last non-aerated to the first aerated tank was used in order to avoid back-mixing of oxygenated activated sludge into non-aerated tanks. Thus the water-level in the anoxic tanks was slightly higher than in aerated tanks causing the volume of activated sludge in non-aerated tanks to be correspondingly greater than in the same tanks when aerated. During a few experiments a larger non-aerated volume was required. Then the large non-aerated tank (11.2 m^3) was used followed by only one aerated tank (this situation is not illustrated in Figure 4.5).

In order to simulate the flow conditions of a full scale plant, the influent flow was adjusted with the aid of a weir, before each 24 h test period in proportion to the influent flow to the main plant. The average influent flow to the main plant, 4 m^3/s, corresponded to 1.5 l/s in the pilot plant (with a maximum of 8 m^3/s to the main plant). The return activated sludge flow and the recirculation of final sedimentation tank supernatant were adjusted before each 24 h test period in order to make up a total flow of 3.75 l/s through the activated sludge tanks (3.75 l/s = influent + return sludge + recirculated effluent). The recirculated supernatant flow was adjusted by choosing one (out of three) hoses with different dimensions. As would be expected the method gave a rather coarse flow adjustment. Compensation was made by adjusting the flow of the return sludge pump. The flow of recirculated wastewater was measured in the tank normally used for waste activated sludge. As this measuring procedure caused a wastage of activated sludge no separate wastage of activated sludge was needed during a large part of period B. The uncertainty of the concentration of the wasted sludge caused great difficulties when attempting to estimate the solids retention time of the system.

Sodium nitrate was added to match the amount of ammonium nitrogen which would have been nitrified in the trickling filter(s) and recirculated to the non-aerated activated sludge tanks as nitrate. This amount was calculated using the effluent ammonium concentration from the main plant and the degree of recirculation in the pilot plant. In some cases the sodium nitrate dosage was increased by 50 % in order to evaluate the limit of denitrification.

When the smaller (2.7 m^3) tanks were used as non-aerated tanks they were stirred using submerged pumps.

In preparing for **experimental period C** (February - July 1993) the pilot plant was rebuilt and improved in several respects (Figure 4.6). New final sedimentation tanks were built, the ceramic aerators were replaced with medium bubble aerators rubber (not membrane), frequency modulated pumps were installed for influent and recirculated flow and the accessibility and the working environment was improved by the construction of a new system of walkways. Nitrate was supplied by a rotating biological contactor. The final sedimentation tanks were exchanged and from experimental period C onward had a circular surface of 6.2 m^2 and a water depth of about 3 m. The sludge was collected by a rotating scraper. One or two sedimentation tanks were used.

Six activated sludge tanks were used. One, two or three of them were non-aerated and the remainder were aerated.

Nitrification was performed in a 1090 m^2 rotating biological contactor (RBC), rotating at about 2 rpm. The RBC rotated in a basin containing about 7.5 m^3 of water and about 50 % of the RBC was submerged.

Primary settled wastewater was pumped to the activated sludge system in proportion to the influent wastewater flow to the main plant and the recirculated nitrified wastewater was pumped to the first non-aerated activated sludge tank at a flow calculated to keep the flow through the activated sludge system constant. Return activated sludge was pumped to the first activated sludge tank at a constant rate of about 0.7 l/s.

The first activated sludge tank was either used as a non-aerated deoxygenation tank, in which case primary settled wastewater was introduced in the second tank, or as the first anoxic tank in which case it also received primary settled wastewater. All potentially non-aerated tanks were equipped with mechanical stirrers for anoxic operation (STAMO 80-6). Potentially aerated tanks were equipped with 4 medium bubble aerators each (Cellpole ϕ 120 mm).

The mixed liquor flowed from one tank to the next through rectangular holes at the surface. At the wall separating a non-aerated tank from an aerated tank, or two non-aerated tanks from each other this hole was closed and the mixed liquor passed a V-notch weir at the top of the tank wall. This caused the water level to be higher (and thus the volume to be greater) in non-aerated than in aerated tanks. In order to verify that back-mixing from the first anoxic tank to the deoxygenation tank was of no importance a tracer study was performed. Tracer (Lithium Chloride) was added to the first anoxic tank and samples collected in the three non-aerated tanks (Figure 4.3). No significant impact on the lithium concentration in the deoxygenation tank could be seen. Thus it was concluded that back-mixing from the first anoxic tank to the deoxygenation tank was not significant.

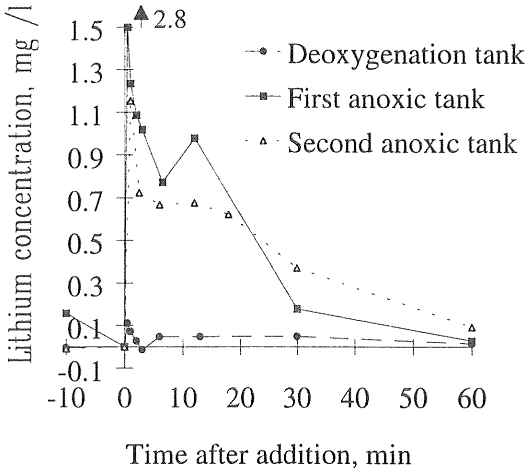
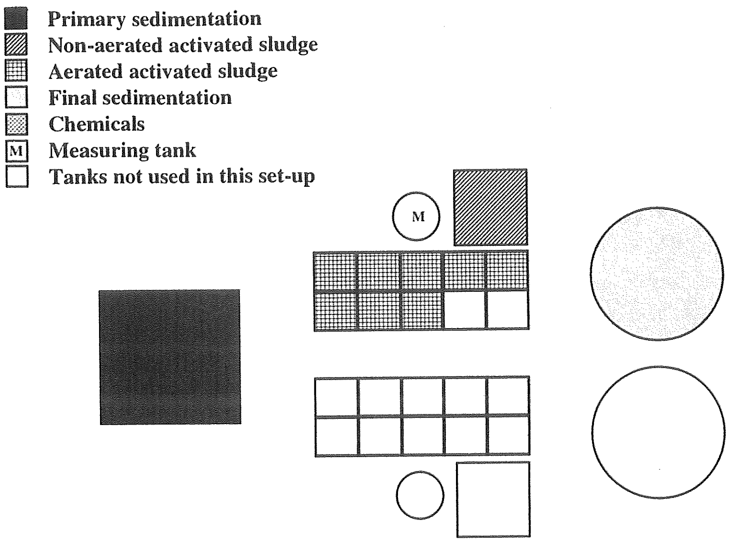


Figure 4.3 Tracer study (lithium chloride added to the first anoxic tank at 0 min).

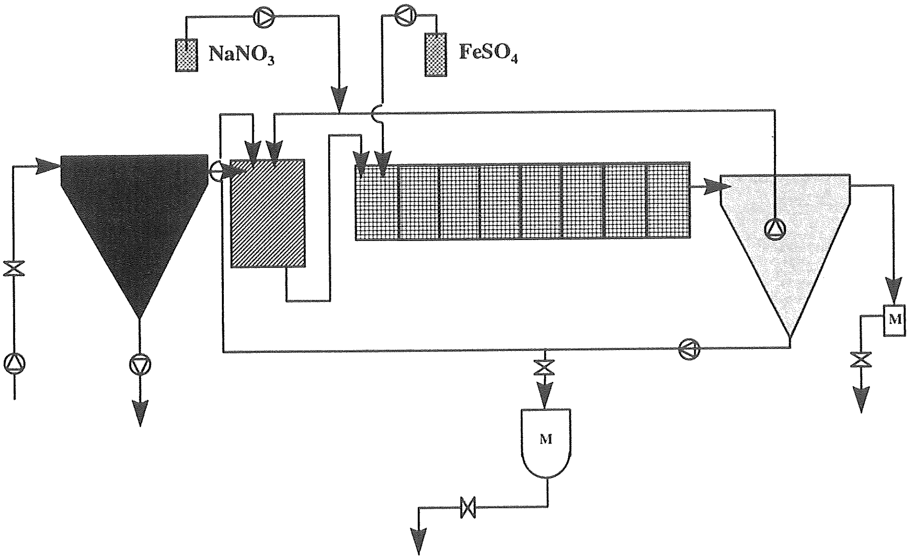
All the effluent wastewater was nitrified in a rotating biological contactor, part of it recirculated to the first compartment of the activated sludge tank and the excess wasted.

The basic experimental set-up during **experimental period D** (July 1993 - February 1994) was the same as in experimental period C. The operation was also similar, the main difference being that in experimental period D, the combinations of aerated and non-aerated tanks were limited to those in which about half the activated sludge volume was aerated (Figure 4.7). Periodically primary settled wastewater was introduced to the first non-aerated tank and periodically to the second. In the latter case the first tank was used for deoxygenation of recirculated nitrified water. Two final sedimentation tanks were used.

Operating conditions of the four experimental periods are summarised in Table 4.2.

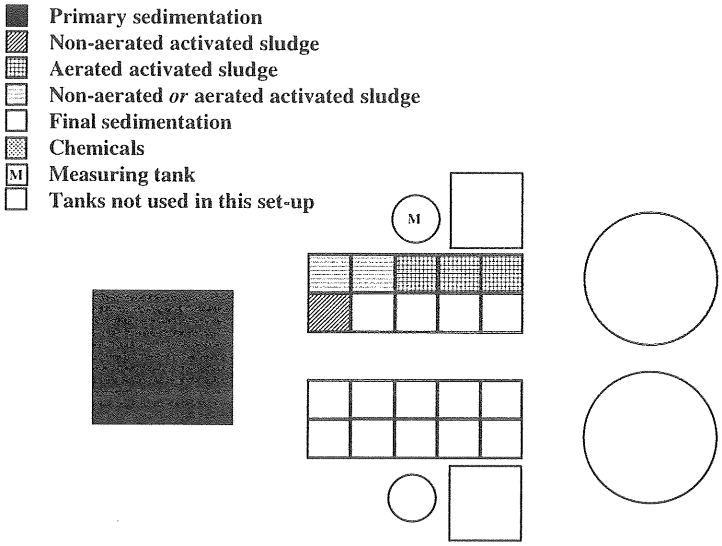


a) Pilot plant layout

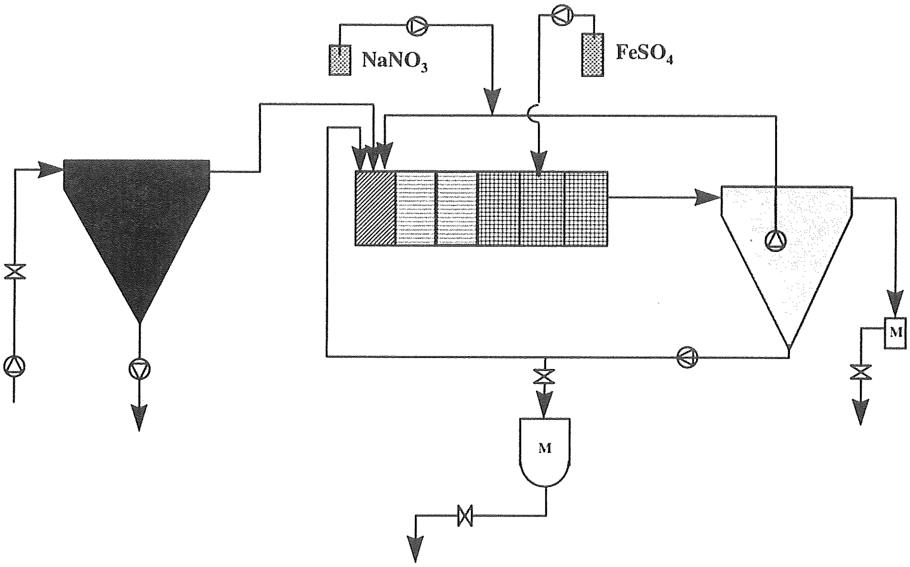


b) Process configuration

Figure 4.4 Experimental configuration, period A.

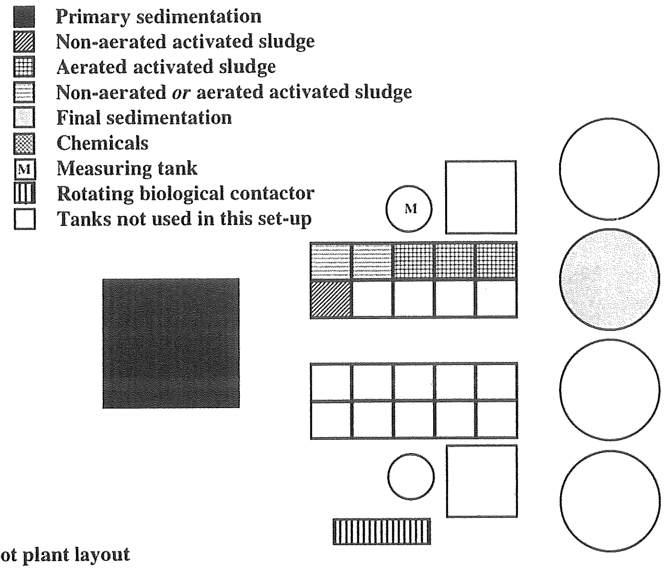


a) Pilot plant layout

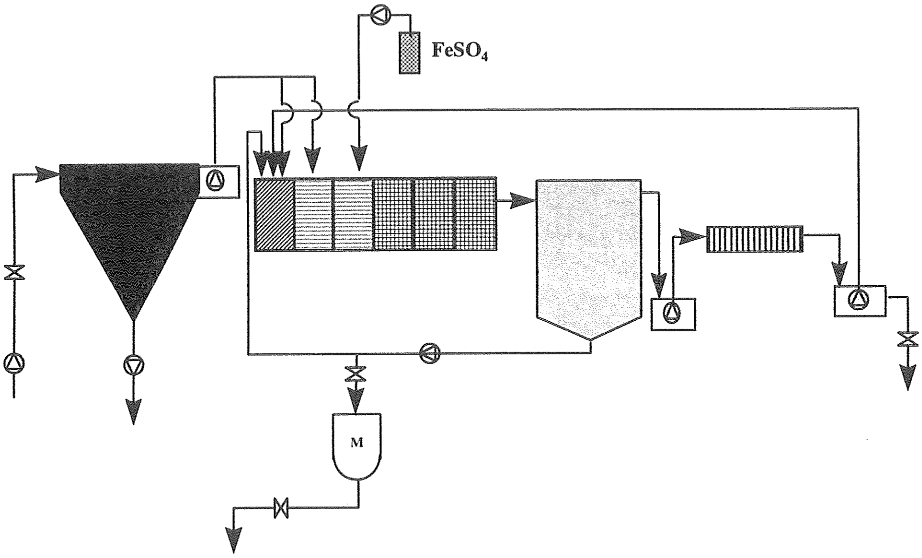


b) Process configuration

Figure 4.5 Experimental configuration, period B.

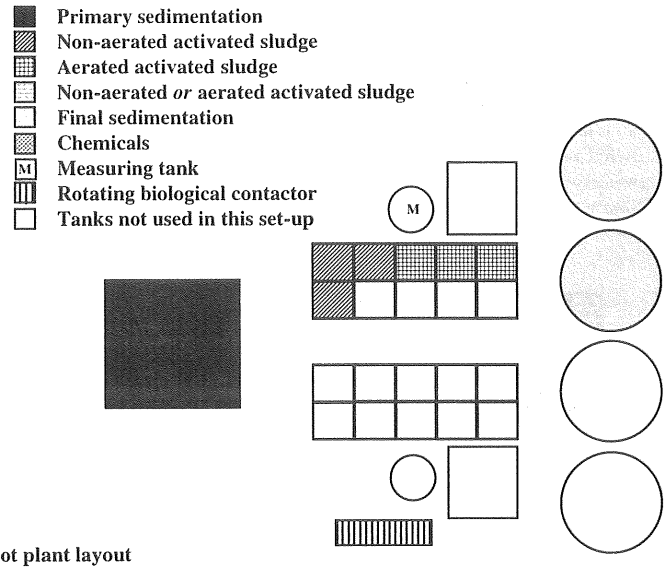


a) **Pilot plant layout**

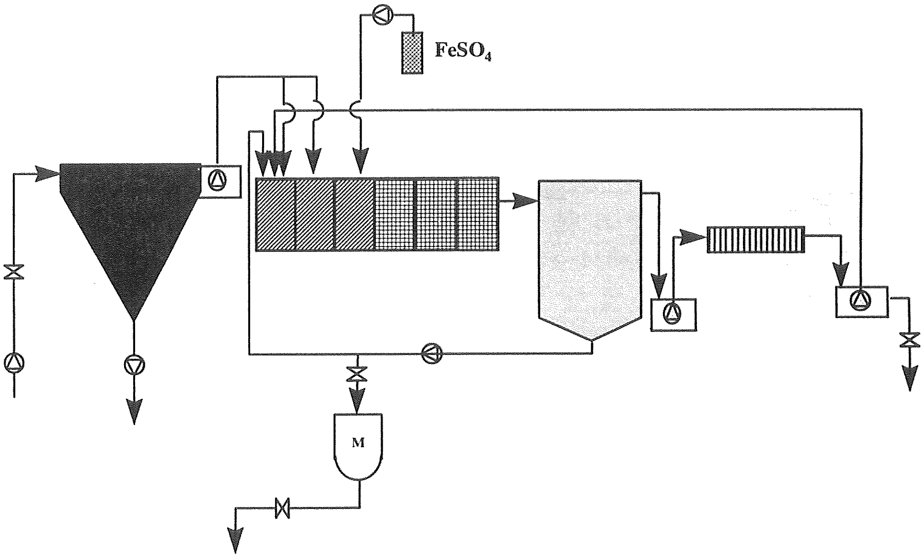


b) **Process configuration**

Figure 4.6 Experimental configuration, period C.



a) Pilot plant layout



b) Process configuration

Figure 4.7 Experimental configuration, period D.

Table 4.2 Operating conditions during the four experimental periods.

Experimental period	Volume (m^3)			Flow (see also Figure 4.15)	Nitrate for denitrification
	Deoxygenation	Anoxic	Aerobic		
A					
900614-900913 901120-901220 910107-910208	0	11.2	20	Constant	Addition of sodium nitrate
B					
920402-920514	0	2.8	12.5	Constant during 24 h but adjusted daily according to main plant influent flow	a)Addition of sodium nitrate adjusted daily to match recirculated nitrate mass flow in a full-scale plant b) as in a) but with 50 % more nitrate
	0	5.5	10.0		
	0	8.4	7.5		
	0	11.2	2.5		
C					
930211-930709	0	2.7	12.9	Continuously controlled according to main plant influent flow	Nitrification of effluent in a rotating biological contactor (RBC)
	2.7	2.7	10.3		
	0	5.5	10.3		
	2.7	5.5	7.7		
	0	8.2	7.7		
D					
930710-940210	0	8.2	7.7	Continuously controlled according to main plant influent flow	Nitrification of effluent in RBC
	2.7	5.5	7.7		

4.3 Sampling and analysis

Throughout the pilot plant experiments the system was sampled and the samples analysed (summarised in Table 4.3). Procedures are detailed below. Due to practical considerations 24-hour samples were started and collected at around 08^{00} h. A composite sample started 08^{00} h on Sunday and collected 08^{00} h on Monday represents Sunday conditions.

Table 4.3 Sampling of water and activated sludge in pilot plant.

Parameter	Unit	Sampling point (number in brackets refers to position in Figure 4.9)						
		Primary settled wastewater (1)	Deoxygenation tank (10)	Last non-aerated tank (2)	Last aerated tank (3)	Return activated sludge (5)	Effl. from final sed. tank (4)	Effluent from RBC (6)
Suspended Solids	g SS/m^3				**GS**	**GS**	**24 h**	
Ash	% of SS				**GS**			
COD	g O_2/m^3	**24 h**						
COD (filtered sample)	g O_2/m^3	**24 h**	(GS)	24 h		(GS)	24 h	(24 h)
BOD_7	g O_2/m^3						24 h	
Total Nitrogen	g N/m^3	**24 h**				**GS**	**24 h**	24h
Ammonium	g N/m^3	24 h					24 h	24 h
Nitrate + Nitrite	g N/m^3	24 h	(GS)	GSor24h			**24 h**	24 h
Nitrite	g N/m^3						24 h	24 h
Phosphorus	g P/m^3			(GS)		(GS)	24 h	
Phosphate	g P/m^3						**24h**	
Alkalinity	eq HCO_3^- /m^3	24 h					24 h	24 h
Sludge Volume	ml/l				**GS**			
Stirred Sludge Volume	ml/l				**GS**			
Initial Settling Velocity	m/h				**GS**			
Turbidity of Supernatant	NTU				(GS)			

GS = Grab sample.
24 h = 24 hour composite sample.
Bold writing indicates sampling and analysis in all experimental set-ups.
(Brackets indicate temporary sampling and/or analysis).

During experimental periods A and B sampling was mainly performed on regular working days. During experimental periods C and D the composite 24-hour samples representing Sundays were collected whenever possible since Sunday was expected to differ from other weekdays. Sampling was performed on two to four other days each week alternating between Monday, Tuesday, Wednesday and Thursday. Samples representing Friday and Saturday were normally not collected since that would mean processing samples on Saturday and Sunday.

Grab samples from the activated sludge tanks were taken by plunging a container at the end of a rod into the activated sludge tanks and then quickly pouring the contents into a sampling bottle. Return activated sludge was sampled from the pipe returning the sludge to the activated sludge tanks. Since the excess sludge was extracted from this pipe by the periodic opening of a valve, the sample of return activated sludge also represents excess sludge (except during experimental period B - when no good estimate of the composition of the excess sludge could be made).

Composite 24-hour samples of primary settled wastewater and effluent from the final sedimentation tank(s) were taken by time-controlled spoon samplers extracting a small sample every four minutes from small containers continuously fed by a high flow of the primary settled and effluent water respectively. The samples were collected in a 25 l plastic container in a refrigerator. The 25 l container and the hoses from the sampler to the refrigerator were cleaned with 3 M hydrochloric acid before each 24-hour sampling. Effluent water from the RBC (Rotating Biological Contactor) was sampled using a peristaltic sampler and collected in a plastic container. A folded filter paper was placed in the neck of the plastic container in order to remove suspended solids

During the first part of experimental period A samples from the anoxic compartment were grab samples. During the remainder of the experiments 24-hour composite samples were collected using a peristaltic carousel sampler. In order to avoid suspended solids in the samples, a pipe (diameter 100 mm) was placed in the anoxic compartment. The quiescent conditions in the pipe provided a supernatant for sampling. A folded filter paper in the neck of each sample bottle removed any remaining suspended solids. After each sampling the liquor in the pipe was stirred by a strong flow of air used to blow clean the sample hose. The sampling interval was normally 30 min. Any elevated oxygen concentration caused by the mixing of the liquor in the pipe was not detectable after 5 to 10 minutes.

Sedimentation tests and analysis of suspended solids, alkalinity and phosphate phosphorus were performed within a few hours of sampling. Samples for the remaining analysis could if necessary be frozen or otherwise conserved (after filtration if a filtered sample was required) and analysed at a later date.

For **suspended solids** a known volume was filtered through a GFA glass filter. The filter was dried in a microwave oven (6 min for wastewater and 7 min for activated sludge). After 30 min the filter was weighed and the mass of suspended solids calculated as the difference between the weight of the filter with and without the sample. For **ash content** the above-mentioned filter was incinerated at 550 °C for 1 hour and then weighed. The ash content was calculated as the difference in weight between the dried and the incinerated filter. Effects of the humidity in the laboratory on suspended solids and ash content were annulled since the filters, which were stored in the laboratory before use, were again left in the laboratory for 30 min between drying or incineration and weighing. Effects of the filters losing material to the filtrate were compensated for by comparing these with a filter through which distilled water had been filtered.

The chemical oxygen demand, **COD**, was analysed according to a dichromate-method where organic compounds are oxidised. A reagent ampoule method was used (Hach Chemical Corp.) in agreement with Swedish Standard SS 02 81 42. The oxidation causes a colour change which is quantified using a "Direct Reading Spectrophometer" (DR/2000). During experimental period A filtered samples for COD-analysis were prepared using a folded filter paper. During the remaining experiments a glass fibre filter (Munktell MGA) was used.

The 7 day biological oxygen demand, **BOD_7**, expresses the amount of oxygen needed for biological oxidation of the organic matter in a sample during the first seven days of incubation. In Sweden a 7 day incubation is used instead of the generally used 5 day incubation due to the advantage of being able to start the incubation on any regular working day and register results on the same day during the following week. The value of BOD_7 is about 10 % higher than BOD_5. The analysis was performed according to Swedish Standard SS 02 81 43. Nitrification was inhibited using allyl thiourea (ATU: $C_4H_8N_2S$).

Total nitrogen was analysed at an external laboratory according to SS 02 81 31 using a Techators Flow Injection Analyser (FIA). During most of the experimental periods **ammonium, nitrate and nitrite** were analysed using an "Auto Analyzer ^{TM} II" (TECHNICON^{TM}). For ammonium Technical Industrial Method No. 857-87 I (corresponding to Swedish Standard SS 02 81 34) was used. During experimental period A an ion specific ammonia electrode was used. The nitrate and nitrite method corresponds to Swedish Standard SS 02 81 32. As nitrite may rapidly be oxidised, analysis has to be performed within 4 h of sampling. When this was not possible using the auto analyser, nitrite was analysed using a reagent ampoule method (Accu Vac^{TM} HACH, NitriVer$^{®}$3). The colour change was quantified using a Direct Reading Spectrophotometer (DR/2000). It should be noted that when a nitrate value is given without corresponding nitrite values the "nitrate" value in reality is nitrate + nitrite. This is true during experimental period A and B and for primary settled wastewater also during experimental periods C and D.

Effluent **phosphorus** was quantified as total phosphorus (experimental periods C and D) and phosphate phosphorus (all experimental periods). During part of experimental period A biological phosphorus removal was studied in the activated sludge system. During this period phosphate in the anoxic tank was analysed as well as total phosphorus of the return activated sludge. During part of experimental period C total phosphorus of the effluent from the RBC was analysed. **Phosphate phosphorus** was analysed using reagent ampoules (Accu Vac^{TM} HACH, PhosVer Phosphate Reagent), the colour change registered using a "Direct Reading Spectrophotometer" (DR/2000). Total **phosphorus** was analysed using a simplified method. Inorganic and organic phosphorus was transformed to phosphate at 120 °C. The phosphate concentration was then quantified using a reagent pillow method (PHOSPHER3 reagent pillows, Hach 2125 99) giving a colour change (molubdene blue) quantified using a Direct Reading Spectrophotometer (DR/2000).

For **pH** an electrode "pH Meter Microprocessor" (corresponding to SS 02 81 22) was used. After registering pH, the **alkalinity** was analysed using a simplified method;

"Orion Total Alkalinity Test Kit". A comparison between this method and the Swedish Standard method (SS 02 81 39, titration of the sample to pH 5.4 with hydrochloric acid) showed a tolerable agreement between the methods for effluent wastewater (Figure 4.8).

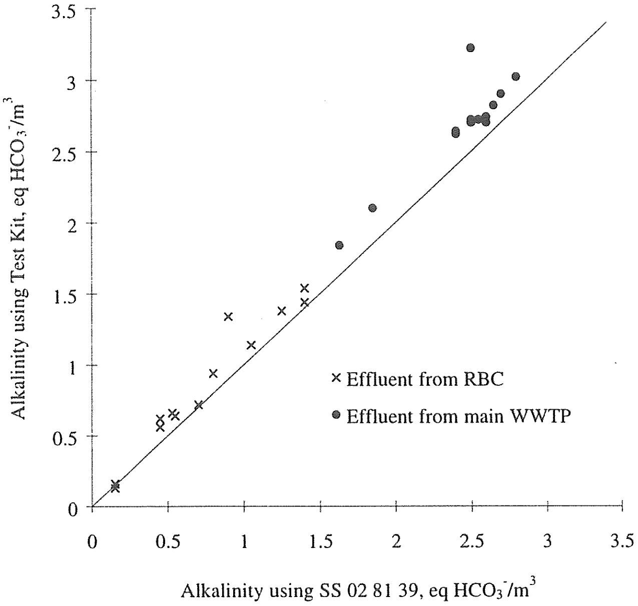


Figure 4.8 Comparison between analysis of alkalinity using "Orion Total Alkalinity Test Kit" and Swedish Standard SS 02 81 39.

The **sludge volume** was determined in a plastic settling cylinder (1 l , ϕ 62 mm). The level of the sludge blanket after 30 minutes was registered. For the **stirred sludge volume** a Triton Settling Apparatus Type 162 was used. Activated sludge (3.2 l) was poured into a Plexiglas column (ϕ 90 mm). The sludge was stirred by two vertical rods rotating at about 1 min^{-1}. During the first ten minutes the level of the sludge blanket was registered every minute and plotted. The slope of the steepest part of the curve is used to calculate the **initial settling velocity**. The sludge blanket level after 30 minutes was measured for the stirred sludge volume. During experimental period A 50 ml of the supernatant was carefully extracted with a wide-tipped pipette and the **turbidity** measured.

Some on-line instruments were used. In some cases the signals were used for control or for analysis of the process. In other cases the main purpose was to see which instruments would be useful in a future full-scale application. The **oxygen concentration** was measured in one of the aerated tanks. A WTW Oxigard was used and calibrated and/or checked one to four times every month. The signal was used for control of the airflow to the diffusers. The **oxidation reduction potential** was continuously measured in the last anoxic tank and in the last aerated tank. Effluent **pH** from the sedimentation tanks was continuously measured. These electrodes were calibrated once a week. Two **continuous ammonia sensing instruments** were tested during different periods. A system using an ion specific electrode (from Process-Styrning AB) analysed effluent wastewater from the final sedimentation tank. After filtration an ion specific electrode system (Monitor 90 from Braun Lüebbe) could be used for on-line analysis of ammonium in primary settled wastewater and effluent wastewater with quite good reliability. A **continuous nitrate analyser** from Process-Styrning AB was used without any great success as was Nitec AB:s Polystat 8810. For nitrate also, a Monitor 90 from Braun Luebbe worked quiet well on filtered water. Note that a comparison based on this material cannot be made since the Braun Luebbe instruments always analysed filtered water whereas the aim when testing the other instruments was to find a cheaper instrument which would not need a filtered sample.

4.4 Estimating some parameters of nitrogen removal

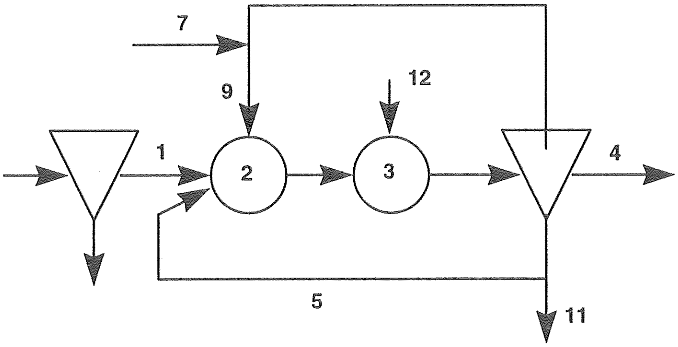
4.4.1 Introduction

Some important parameters, such as denitrification rate, carbon to nitrogen ratio and degree of denitrification, can be calculated in a number of different ways. Different methods of estimation may be used depending on which system is studied and what the result is to be used for. The choice of method may also depend on the reliability and frequency of the available data. In this section the choice of definitions used here will be presented and discussed. As far as possible the notation recommended by Grau et. al (1987) is used. Subscripts will be used when more than one substance or location is treated in the same context. Subscripts indicating location will, where necessary, be used in accordance with Figure 4.9.

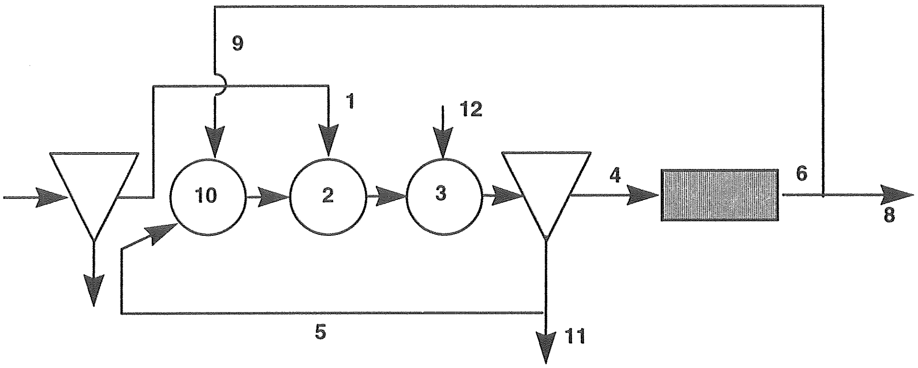
The denitrification rate is the amount of nitrate removed per unit mass of organisms per unit time. In order to estimate the denitrification rate the mass of organisms to be given credit for denitrification must be determined, the amount of nitrate depleted by denitrification calculated and possibly the influence of other oxidants quantified. In section 4.4.2 the production, recirculation and removal of nitrate and oxygen are calculated. In section 4.4.3 the mass of non-aerated activated sludge is calculated and nitrate loading and denitrification rate are defined in section 4.4.4.

Since the denitrifiers are dependent on organic matter for energy and growth the amount of organic matter measured as soluble *or* total BOD_5 *or* BOD_7 *or* COD *or* TOC available *or* consumed per unit mass of nitrate *or* total electron acceptor *or* total nitrogen available *or* total nitrogen removed is often mentioned. This is often called the carbon to nitrogen ratio. The carbon to nitrogen ratios used in this thesis are defined and discussed in section 4.4.5.

Experimental set-ups A and B



Experimental set-ups C and D



1 Primary settled wastewater
2 Anoxic zone
3 Aerated zon
4 Effluent from final sedimentation tank
5 Return activated sludge
6 Effluent from RBC
7 Sodium nitrate solution
8 Discharged wastewater in set-ups C and D
9 Recycled wastewater
10 Deoxygenation zone
11 Waste activated sludge
12 Iron sulphate solution

Figure 4.9 Denomination of flows and reactors in pilot plant. Note that in some cases during experimental periods C an D no deoxygenation tank was used (V_{10}=0 m^3). In these cases Flows 5 and 9 go directly to tank 2.

4.4.2 Nitrate and oxidant mass flow, production and removal

In order to determine the carbon to nitrogen ratio, the denitrification rate and the nitrate loading, the transport of nitrate to or removal of nitrate in the volume in question must be determined. This can be done in different ways, the chosen way depending on the purpose of the operation and the data available.

One aim of determining denitrification rates is to find out how much nitrate is removed in a system. The denitrification rate will here be used to quantify the net removal of nitrate in the activated sludge system assuming that all removal takes place in the mixed liquor in the non-aerated activated sludge tanks. An advantage of using the net removal in the activated sludge system is that it only includes analysis of water where sampling is relatively easy (primary settled wastewater, secondary effluent, RBC effluent) but not of mixed liquor where sampling is more complicated. During experimental periods A and B samples of mixed liquor were grab samples which are not easily comparable with 24-hour composite samples. However, this approach does not separate denitrification in the non-aerated tanks from any denitrification which may occur in the aerated tanks. As we will see under "results and discussion" some denitrification may take place in aerated tanks. Nitrate loadings, denitrification rates and carbon to nitrate ratios are calculated using nitrate addition and removal as defined above and in the first three equations of Figure 4.10.

When determining the supply of nitrate by nitrification in the RBC the nitrification was estimated in three different stochiometrically related ways (nitrate gain, ammonium loss and alkalinity loss, Figure 4.11). When figures based on one estimation deviated from the others by more than 20 %, they were dropped and the nitrate production calculated using the remaining two in order to reduce the influence of erroneous analysis.

Nitrate removal in the activated sludge system in experimental set-ups A and B:

$$-\Delta F_{NO3,AS} = F_{NO3,7} - Q_4 \cdot S_{NO3,4} \approx F_{NO3,7} - Q_1 \cdot S_{NO3,4}$$

and in experimental set-ups C and D:

$$-\Delta F_{NO3,AS} = \frac{Q_9}{Q_4} \Delta F_{NO3,RBC} - (Q_4 - Q_9) \cdot S_{NO3,4} \approx \frac{Q_9}{Q_1 + Q_9} \Delta F_{NO3,RBC} - Q_1 \cdot S_{NO3,4}$$

Nitrate production in the RBC:

$$\Delta F_{NO3,RBC} = Q_4 \frac{(S_{NH4,4} - S_{NH4,6}) + (S_{NO3,6} - S_{NO3,4}) + (Alk_4 - Alk_6) \cdot 7}{3}$$

Oxidant mass flow to non-aerated tanks in experimental set-ups C and D:

$$F_{OX,NA} = Q_1 \cdot S_{NO3,1} + Q_5 \cdot S_{NO3,5} + Q_9 (S_{NO3,6} + \frac{S_{O2,9}}{2.86})$$

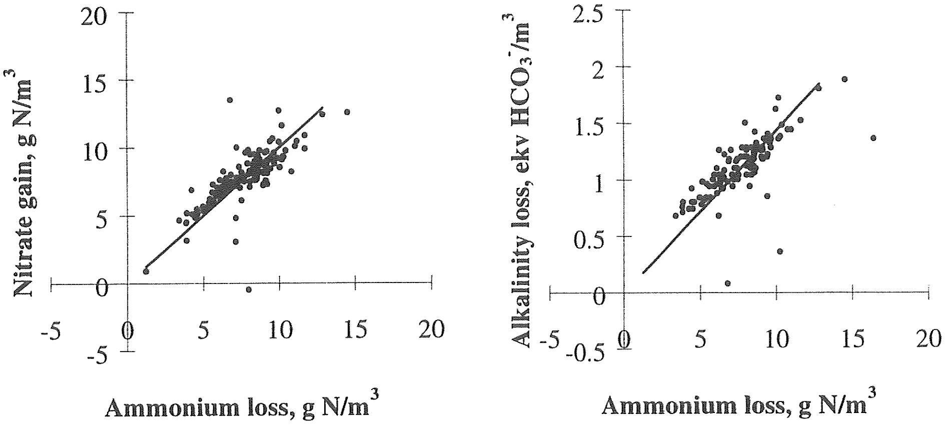
Oxidant removal in non-aerated tanks in experimental set-ups C and D:

$$-\Delta F_{OX,AS} = Q_1 \cdot S_{NO3,1} + Q_5 \cdot S_{NO3,5} + Q_9 (S_{NO3,6} + \frac{S_{O2,9}}{2.86}) - (Q_1 + Q_9 + Q_5) \cdot S_{NO3,2}$$

Symbol	Explanation	Unit
Q	Flow	m^3/h
S	Concentration	$g\ N/m^3$, $g\ O_2/m^3$
F	Mass flow	g N/h
Alk	Alkalinity	$eq\ HCO_3^-/m^3$
$\Delta F_{NO3,RBC}$	Production of nitrate in the rotating biological contactor	g N/h
$-\Delta F_{NO3,AS}$	Nitrate removal in the activated sludge system	g N/h
$-\Delta F_{OX,AS}$	Oxidant removal in the activated sludge system	g N/h

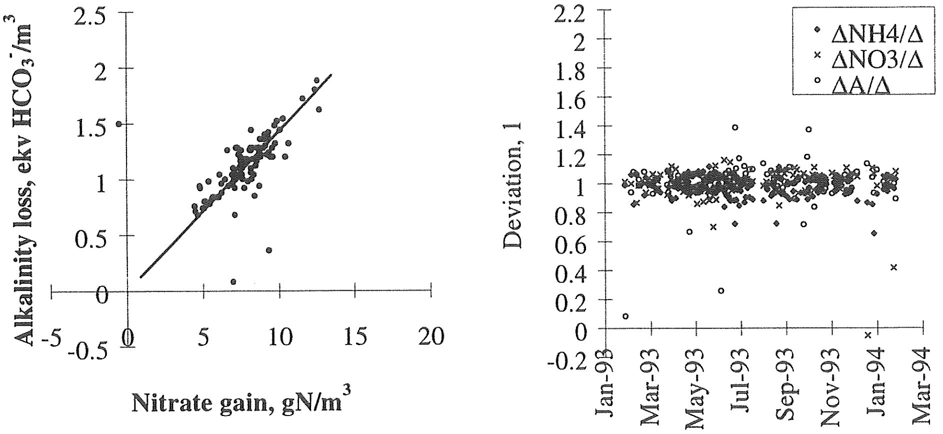
Index	
NO3	Nitrate
NH4	Ammonium
O2	Oxygen
OX	Oxidant; nitrate plus oxygen expressed as nitrate equiv.
Numbers	Position in Figure 4.9
AS	Activated sludge system
RBC	Rotating biological contactor
NA	Non-aerated activated sludge tanks

Figure 4.10 Nitrate and oxygen production and removal.



a) Nitrate gain versus ammonium loss.

b) Alkalinity loss versus ammonium loss.



c) Alkalinity loss versus nitrate gain.

d) Deviation of individual loss or gain from estimated nitrate gain (ΔA= alkalinity loss expressed as NH4-N equivalents).

Figure 4.11 Balances over the RBC during experimental periods C and D. Lines represent stoichiometric relationships.

Another aim when determining denitrification rates, or activities, can be to determine the activity of the bacteria when removing nitrate and to compare this with other bacteria observed in other systems. In this case it is useful to take into account other oxidants, such as oxygen or nitrite, which may be removed in preference to nitrate. With the above aim it is of interest to focus directly on what happens in the non-aerated tanks and not in the activated sludge tanks in general. Analysing experimental periods C and D, when nitrification took place in a RBC, it is both desirable and possible to calculate total oxidant loading in the non-aerated tanks specifically. It is desirable since the effluent of the RBC contains a significant amount of oxygen and it is possible since the mass flow of nitrate to, and the removal of nitrate in, the non-aerated tanks can be determined using only composite 24-hour samples.

When one mole of oxygen is reduced, 2 moles of electrons are accepted, whereas one mole of nitrate reduced to nitrogen gas accepts 5 moles of electrons. Taking the atomic weights of nitrogen and oxygen into account the reduction of one gram of nitrate-nitrogen corresponds to the reduction of 2.86 grams of oxygen ($\approx 16/14 \cdot 5/2$). The total oxidant mass flow to the non-aerated tanks can be expressed as the sum of the mass flow of nitrate and of oxygen expressed as nitrate equivalents. When calculating the oxidant removal all the introduced oxygen was assumed to be reduced in preference to nitrate. In this estimate nitrate in primary settled wastewater, RBC effluent and RAS and oxygen from RBC effluent is taken into account. Other oxidant mass flows are oxygen in influent wastewater and RAS (not measured) and nitrite in primary settled wastewater (not analysed), RBC effluent (analysed) and RAS (not analysed). These neglected contributions, however, are not great (Table 4.4) accounting for less than 3 % of the total oxidant mass flow. Another assumption is that all influent "nitrate" (in reality nitrate + nitrite) is actually nitrate. However assuming that half the influent "nitrate" was in fact nitrite would reduce the total oxidant mass flow by less than 0.3 %.

Table 4.4 Relative contribution of oxidants to mass flow to non-aerated tanks.

	Recirculation from RBC	Primary settled wastewater	Return activated sludge	Total
Nitrate	68.3	1.4	3.4**	73.1
Nitrite *	1.3	0.0**	0.6**	1.9
Oxygen *	24.2	0.2**	0.6**	25.0
Total	93.8	1.6	4.6	100

*Reduction of one g N as nitrite corresponds to (3/5=) 0.6 g N as nitrate and reduction of one g oxygen corresponds to ($14/16 \cdot 2/5=$) 0.35 g N as nitrate.

**Assuming primary settled wastewater to contain 0.1 g O_2/m^3 but no nitrite and return activated sludge to contain 0.5 g O_2/m^3 and nitrate and nitrite concentrations equal to those of the secondary effluent.

4.4.3 Mass of organisms

The mass of denitrifying organisms is quantified as the mass of volatile suspended solids in non-aerated activated sludge tanks. In most cases the amount of non-aerated activated sludge, $\Sigma(V_D X_b)$, was calculated assuming that the concentration of volatile suspended solids, X_b,, was the same in all the activated sludge tanks and thus equal to that of the sampling point in the last aerated tank. However when one tank was used for deoxygenation, the concentration X_b differs from that of the sampling point. The concentration in the deoxygenation tank (10) can be estimated. Assuming that the mass flow of suspended biomass with influent wastewater, waste activated sludge, effluent wastewater and recirculated wastewater is negligible compared with that of the return activated sludge the biomass concentration in the deoxygenation tank can be calculated (Figure 4.12). Estimating the biomass concentration in the deoxygenation tank from the concentration in the last aerated tank gives a concentration on average 10 % higher than when the return sludge concentration is used. These two approaches are assumed to be equally good and the error equal; and $X_{b,10}$ used for calculations will be the average of the results of the two expressions. On a few occasions however a different approach has been warranted:

- During January and February of 1994 the return sludge was pumped intermittently from the two final sedimentation tanks, causing the return sludge concentration to fluctuate considerably. During this period $X_{b,10}$ was estimated using only the concentration in the last aerated tank and the result multiplied by 0.95 in order to compensate for the systematic difference of 10 % between the equations.
- On two occasions (930503 and 930628) the registered concentration in the aerated compartments varied considerably from the concentration the day after or the day before. In these cases the estimate was based on the concentration of the return activated sludge and the result multiplied by 1.05 in order to compensate for the systematic difference.

Mass of organisms in non-aerated tanks:

$$\sum V_D X_b = V_2 X_{b,2} + V_{10} X_{b,10} = V_2 \cdot X_{b,3} + V_{10} \cdot X_{b,10}$$

Where the concentration of organisms in the deoxygenation tank (X_{10}) was estimated in two different ways:

$$X_{b,10} = X_{b,3} \frac{Q_1 + Q_5 + Q_9}{Q_5 + Q_9}$$ using the concentration in the last aerated tank and;

$$X_{b,10} = X_{b,5} \frac{Q_5}{Q_5 + Q_9}$$ using the return sludge concentration.

The average of these estimations was used.

Symbol	Explanation	Unit
Q_i	Flow through position i	m^3/h
V_i	Volume of tank i	m^3
V_D	Volume where denitrification is assumed to take place	m^3
$X_{b,i}$	Biomass in position i	kg VSS/m^3

Figure 4.12 Biomass in non-aerated tanks. Indexes refer to positions in Figure 4.9.

4.4.4 Nitrate and oxidant loadings and removal rates

Using nitrate and oxidant mass flows developed in Figure 4.10 and denitrifying biomass or mass of organisms as defined in Figure 4.12 denitrification rates, nitrate loadings and oxidant loadings and removal rates are defined in Figure 4.13.

The nitrate loading during experimental periods A and B:

$$B_x = \frac{F_{NO3,7}}{\Sigma(V_D \cdot X_b)} = \frac{F_{NO3,7}}{V_2 \cdot X_{b,3}}$$ and the denitrification rate:

$$r_x = \frac{F_{NO3,7} - F_{out}}{\Sigma(V_D \cdot X_b)} = \frac{F_{NO3,7} - Q_l \cdot S_{NO3,4}}{V_2 \cdot X_{b,3}}$$

The nitrate loading during experimental periods C and D:

$$B_x = \frac{\Delta F_{NO3,RBC} \cdot Q_9/(Q_1 + Q_9)}{\Sigma(V_D \cdot X_b)}$$ and the denitrification rate:

$$r_x = \frac{\Delta F_{NO3,RBC} \cdot Q_9/(Q_1 + Q_9) - Q_l \cdot S_{NO3,4}}{\Sigma(V_D \cdot X_b)}$$

The oxidant loading during experimental periods C and D:

$$B_x = \frac{Q_5 \cdot S_{NO3,5} + Q_1 \cdot S_{NO3,1} + Q_9 \cdot (S_{NO3,9} + \frac{S_{O2,9}}{2.86})}{\Sigma(V_D \cdot X_b))}$$ and the oxidant removal rate:

$$r_x = \frac{Q_5 \cdot S_{NO3,5} + Q_1 \cdot S_{NO3,1} + Q_9 \cdot (S_{NO3,9} + \frac{S_{O2,9}}{2.86}) - S_{NO3,2} \cdot (Q_5 + Q_1 + Q_9)}{\Sigma(V_D \cdot X_b))}$$

Symbol	Explanation	Unit
Q	Flow	m^3/h
S	Concentration	g N/m^3, g O_2/m^3
F	Mass flow	g N/h
$\Delta F_{NO3,RBC}$	Production of nitrate in the rotating biological contactor	g N/h
$-\Delta F_{NO3,AS}$	Nitrate removal in the activated sludge system	g N/h
$-\Delta F_{OX,AS}$	Oxidant removal in the activated sludge system	g N/h
Index		
NO3	Nitrate	
NH4	Ammonium	
O2	Oxygen	
Numbers	Position in Figure 4.9	
RBC	Rotating biological contactor	

Figure 4.13 Nitrate and oxidant loadings and removal rates.

4.4.5 Carbon to nitrogen ratio

The carbon to nitrogen ratio has been defined in a variety of ways, depending on the information wanted and the available data.

Carbon may actually refer to atomic carbon (Socher and Gläser, 1992). More often carbon is allowed to represent some other quantification of organic matter, like methanol, COD or BOD_5 (Akunna et al., 1993, Abufayed and Schroeder, 1986 A and B, Henze and Bundgaard, 1982 and Narkis et al., 1979). The use of oxygen equivalents to quantify the amount of organic matter needed or used for denitrification is, where wastewater treatment is concerned, practical. Denitrification is performed by heterotrophic bacteria that would alternatively use oxygen. Both denitrification and heterotrophic carbon oxidation can be quantified using oxygen equivalents. Additionally organic matter is often measured as COD or BOD in wastewater treatment plants which means that a large amount of relevant data may be available. Of the two analyses BOD and COD, BOD may appear to be the most relevant to biological degradation, but since the procedure of the analysis takes five or seven days, it is expensive and impractical as a day to day control analysis. Another disadvantage with BOD is that, being an undefined biological method, it is affected by toxic substances. On the other hand, COD gives a quick response at a low price, though the analysis involves mercury, a disadvantage from an environmental point of view. Though COD does not give an absolute quantification as to how much of the oxygen demand is actually available to the organisms it makes a mass balance expressed in O_2-equivalents possible. Currently a large amount of modelling of activated sludge systems is based on COD and extensive work is being done on defining how much of the COD is available to heterotrophic organisms. The relation between COD and BOD may vary due to wastewater composition or biological degradation at or before the treatment plant.

Here applied carbon will be quantified as COD of the filtered primary settled wastewater. While it is recognised that part of the particulate COD will eventually be available to denitrification and that part of the soluble COD is inert, the COD of the filtered sample is considered to give a closer approximation of the amount of organic matter available for denitrification than does the total COD.

The C/N ratio may also refer to *applied carbon to applied nitrogen*, *applied carbon to consumed nitrogen* or *consumed carbon to consumed nitrogen*. *Applied carbon to applied nitrogen* is a useful measure when different wastewaters are compared in relation to different processes or when the limiting C/N ratio for complete denitrification is stated (used by Abufayed and Schroeder, 1986 A , Lyngå and Balmér, 1992 and Narkis et al., 1979). *Applied carbon to consumed nitrogen* indicates how much carbon the influent wastewater must contain in order to denitrify a certain amount of nitrogen and can for instance be used when indicating how much extra denitrification is achieved with a certain addition of an external carbon source (used by Nyberg et al., 1992). In order to be able to calculate *consumed carbon to consumed nitrogen* the process must be controlled so that it is possible to determine how much organic matter was actually used for denitrification. This may be practical in a system where the only available, readily degradable, substrate is the one added (used by Akunna et al., 1993). In a municipal activated sludge system however hydrolysis, adsorption, aerobic heterotrophic activity and other processes complicate matters, making it difficult to determine how much of the substrate decrease is due to denitrification.

Nitrogen can represent nitrate nitrogen or total nitrogen. Referring to denitrification nitrate nitrogen is the most obvious choice, since this is what undergoes denitrification. If a significant amount of oxygen is introduced to the anoxic zones (as in this case through recirculation of trickling filter effluent) the oxidant addition and oxidant removal expressed as nitrate equivalents is more relevant when describing the system. However when comparing different wastewaters prior to treatment the quantity available is total nitrogen, since nitrification has not yet taken place.

In this investigation the information wanted concerns the way in which the influent organic matter limits the capacity of denitrification (Figure 4.14). To this end the influent mass flow of organic matter measured as COD of a filtered sample was chosen. Only the organic matter introduced with primary settled wastewater is included. The recirculated effluent, having passed through activated sludge treatment and in some cases an RBC, would contain a small amount of readily available organic matter. The return activated sludge, on the other hand, probably contains a considerable amount of hydrolysed or hydrolysable organic matter. The amount is probably greater than the dissolved COD but considerably less than the total COD of the return activated sludge. However all organic matter available in the system was originally introduced with the primary settled wastewater. When the applied carbon to applied oxidant ratio was calculated the nitrate plus oxygen (expressed as nitrate equivalents) mass flow to the non-aerated tanks was used. For applied carbon to removed oxidant the oxygen recirculation to the non-aerated compartments (expressed as nitrate equivalents) was included. The inverse of these ratios will also be used.

The approach above can be used for experimental periods C and D since the internal flows were sampled as 24-h composite samples. However during experimental periods A and B the nitrate concentration of the recirculated water was analysed using one daily grab sample which cannot be assumed to represent the average concentration. When a comparison of the carbon to nitrate ratio between experimental set-ups is made the net nitrate addition to the activated sludge system is used. Recirculated non-denitrified nitrate is not included.

Removed carbon to removed nitrate will not be quantified due to the difficulties of determining how much of the removed carbon was actually used for denitrification.

Applied carbon to applied oxidant to the non-aerated tanks:

$$C/N = \frac{S_{COD,1} \cdot Q_1}{Q_1 \cdot S_{NO3,1} + Q_9 \cdot (S_{NO3,9} + \frac{S_{O2,9}}{2.86}) + Q_5 \cdot S_{NO3,5}}$$

And applied carbon to removed oxidant in the non-aerated tanks:

$$C/N = \frac{S_{COD,1} \cdot Q_1}{Q_1 \cdot S_{NO3,1} + Q_9 \cdot (S_{NO3,9} + \frac{S_{O2,9}}{2.86}) + Q_5 \cdot S_{NO3,5} - (S_{NO3,2}) \cdot (Q_5 + Q_1 + Q_9)}$$

Applied carbon to net applied nitrate to the activated sludge system:

$C/N = \frac{S_{COD,1} \cdot Q_1}{F_{NO3,7}}$ during experimental periods A and B and

$C/N = \frac{S_{COD,1} \cdot Q_1}{\Delta F_{NO3,RBC} \cdot Q_9 / (Q_9 + Q_1)}$ during experimental periods C and D.

Symbol	Explanation	Unit
C/N	Carbon to oxidant ratio	g O_2/g N
Q	Flow	m^3/h
S	Concentration	g N/m^3, g O_2/m^3
F	Mass flow	g N/h
$\Delta F_{NO3,RBC}$	Production of nitrate in the rotating biological contactor	g N/h
$-\Delta F_{NO3,AS}$	Nitrate removal in the activated sludge system	g N/h
$-\Delta F_{OX,AS}$	Oxidant removal in the activated sludge system	g N/h
Index		
NO3	Nitrate	
NH4	Ammonium	
O2	Oxygen	
COD	Chemical oxygen demand	
Numbers	Position in Figure 4.9	
RBC	Rotating biological contactor	

Figure 4.14 Carbon to oxidant ratios (Other C/N ratios will be investigated. These are the main ones used.).

4.5 Comments on experimental set-up and operation

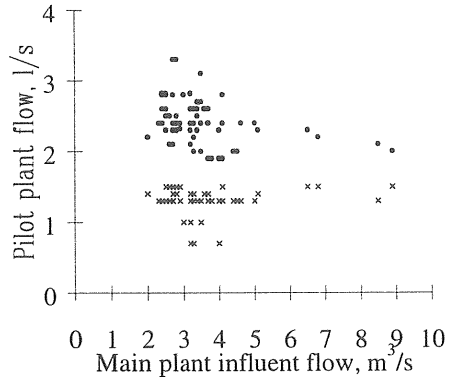
As described above, the experimental set-up and operation has been changed along the way. The changes were made when the objectives of the experiments were revised, the technical and economic possibilities improved and knowledge of the system advanced. Ideally the experiments should fulfil the following requirements:

- The data to cover a **wide range** of conditions
- The experimental conditions to be **representative** of prevailing conditions
- Process conditions to be **independent variables**
- **Methods** of sampling, measuring and analysis to be **comparable** throughout the experimental period

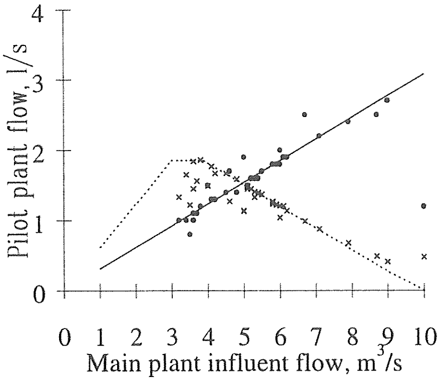
In this section these points will be discussed (this is also discussed by Mattsson, 1997). The focus will be on the influence of experimental, operational and analytical problems on some important parameters. In most of the experimental configurations the aim was that the pilot plant should represent conditions in a future non-nitrifying activated sludge system. A consequence of this is that conditions are representative of expected conditions in a full-scale plant but do not cover a wide range of conditions. Due to correlated variations in wastewater flow and quality and operation, conditions in these experiments, as in reality, may not be independent variables.

The solids retention time (**SRT**) was in general three to four days, in the same order as that of the main plant (the Rya WWTP). The suspended solids concentration was kept at the same level as that of the main plant, regulated by the extraction of surplus sludge. During experimental period B, however, a large part of the surplus sludge was extracted when measuring recirculated flow. The recirculated water was pumped from the final sedimentation tank and the suspended solids concentration varied, complicating the calculation of the surplus sludge extraction and thus of the SRT.

The method of flow regulation differed between experimental periods, varying from a constant flow set-up (A), via a set-up with constant flow adjusted every 24-hours to match influent flow to the main plant (B), to a fully automatic regulation based on main plant influent wastewater pumping (C, D). Figure 4.15 illustrates the target flow pattern and the actual influent and recirculated flows of the pilot plant as measured. During the last part of experimental period D the conditions of the future plant were hydraulically well represented. When the influent flow to the main plant was 4 m^3/s the nominal hydraulic retention time including recirculated water (**HRT**) was 1.1-1.2 h during period B and C. During period A the HRT was higher (about 1.8 h) due to lower recirculation. The net hydraulic retention time (not counting recirculated flows) as it varied with the influent flow is shown in Figure 4.16. During experimental periods B, C and D the average hydraulic conditions were representative of those of full-scale operation, but the **dynamics** were only correct during periods C and D, when automatic flow regulation was used.



Experimental period A



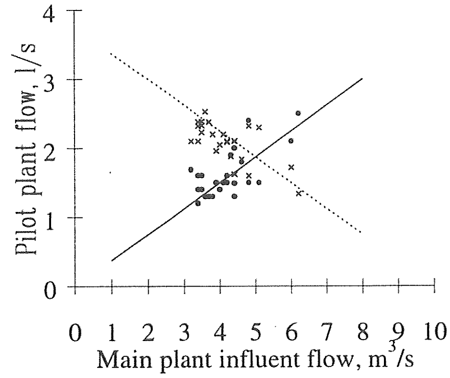
Experimental period D from January 1994.

· Influent flow

× Recirculated flow

— Target influent flow

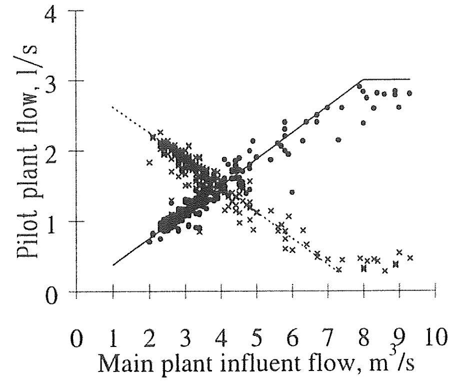
····· Target recirculated flow



Experimental period B

Note:

- "recirculated flow" during experimental period B includes the return activated sludge flow.
- In January 1994 the target flows were revised in order to match revised plans for the extension of the main WWTP.



Experimental period C and part of D.

Figure 4.15 Flow to pilot plant and recirculated flow versus influent flow to main plant.

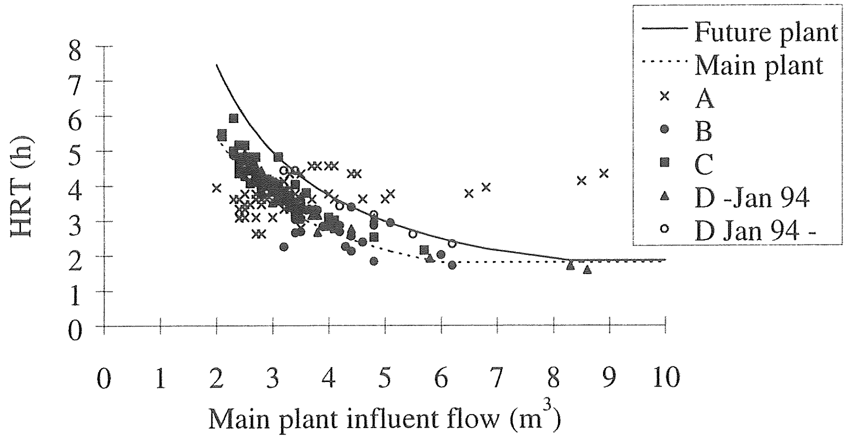
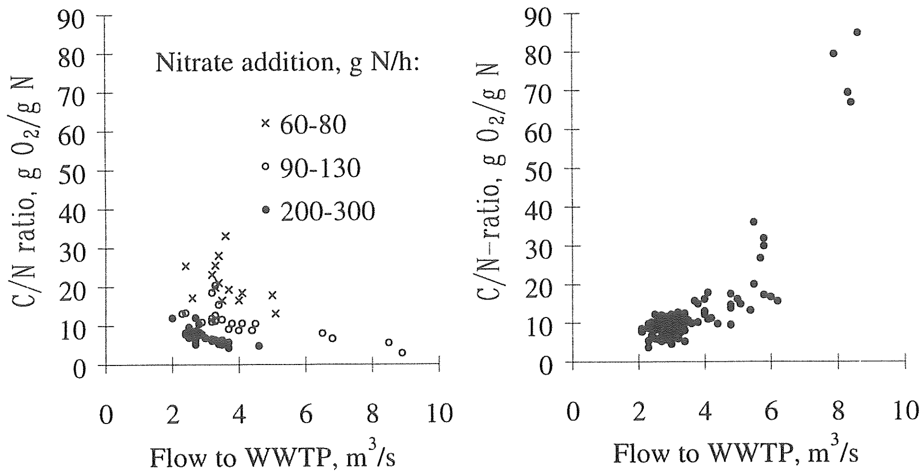


Figure 4.16 Hydraulic retention time in the activated sludge tanks of the pilot plant versus influent flow to the future and present main plant.

During experimental periods C and D the recirculated flow followed the expected recirculation in a full-scale plant and nitrification took place in a rotating biological contactor. At high flows a decreasing recirculation caused the nitrate loading to decrease whereas the COD-mass flow to the pilot plant did not decrease to the same extent. This caused the ratio of soluble COD to nitrate to increase at high flows (Figure 4.17 b). A completely different pattern was produced by the constant flow set-up used during period A. The ratio of pilot plant flow to main plant flow was higher during dry weather conditions than during rainy weather. This, together with the fact that wastewater is generally more concentrated during dry weather than during wet weather caused the **food to micro-organism** ratio to vary differently than it would in a functioning WWTP. As a consequence of the constant addition of nitrate the nitrate loadings were higher than those of the future plant at high flows and lower at low flows. Consequently, at low flows, experimental set-up A received more organic matter and less nitrate (than in a "real" WWTP) and vice versa at high flows, causing the ratio of soluble carbon to nitrate to decrease at high flows (Figure 4.17 a).

If nitrate is added to the system instead of being supplied through nitrification, as was the case during experimental periods A and B, **ammonium** and **alkalinity** are not reduced as they would be through nitrification. Effluent alkalinity during experimental period B averaged 3.7 eq. HCO_3^-/m^3, but during experimental periods C and D was only 2.1 eq.. HCO_3^-/m^3. Reduced alkalinity caused slightly lower pH when nitrate was supplied through nitrification than when dosed. The median effluent pH during experimental periods C and D was 6.9 and during experimental period B 7.1. In both cases more than 90 % of the registered effluent pH values were above 6.7.



a) Experimental period A b) Experimental periods C and D

Figure 4.17 Ratio of COD to nitrate addition or production in RBC versus flow to main plant.

Another, and probably more serious, consequence of addition of nitrate instead of nitrification, is that the recirculated effluent water was not oxygenated the way it would have been using a trickling filter. The **oxygen concentration** in the recirculated wastewater was of the order of 0 - 1 g O_2/m^3 during periods A and B and 6-9 g O_2/m^3 during periods C and D. The oxygen concentration of the water in the pilot trickling filter was in the order of 7-8 g O_2/m^3 (Mattsson and Rane 1993) comparable to conditions during periods C and D.

The fraction of the activated sludge volume under anoxic conditions varied. During experimental periods B and C one objective was to determine the optimal size of the non-aerated zones. In order to determine the influence of the size of the **non-aerated activated sludge volumes** the non-aerated tanks were sometimes very small..

The discussion above is summarised in Table 4.5 and can be concluded as follows:

- Data from experimental periods C and D are representative of full-scale operation if allowance is made for the different performance of the primary sedimentation tank. The range it covers is limited to a “normal operational range”. Data from experimental period B give a good idea of full-scale operation.

- Data from experimental period A are not representative of full-scale operation. However, results at extreme nitrate loadings may indicate the limits of the system.

Table 4.5 Were the conditions of pilot plant operation representative of those of a full-scale application?

Experimental period→ Parameter	A 900614-910208	B 920204-920514	C 930211-930709	D 930709-940210
Solids retention time	yes	?	yes	yes
Hydraulic retention time	no	yes	yes	yes
Food to micro-organism ratio	no	yes	yes	yes
Carbon to nitrate ratio	no	usually	yes	yes
Nitrate loading	no	usually	yes	yes
Ammonium concentration	no	no	yes	yes
Nitrate concentration	no	usually	yes	yes
Alkalinity	no	no	yes	yes
Recirculation of oxygen	no	no	yes	yes
Dynamics	no	no	yes	yes
Non-aerated volume	no	varied	varied	yes

5 Results and discussion

5.1 Introduction

This chapter is divided into three sections. In the first section results of pilot plant operation are presented and discussed. Nitrogen removal during one year of pilot plant operation is analysed in order to estimate the function of the system treating authentic wastewater. For this, results from experimental periods C and D are mainly used, since during these experiments the pilot plant was operated under conditions close to those expected in the future full-scale plant. The results of the pilot plant operation should give a good estimate of the capacities of the future system and give answers to some fundamental questions concerning the design of a full-scale system.

In the second section, the pilot plant data are analysed in order to find out where, how and why nitrogen was removed. For this purpose mass balances and profiles through the system are used.

In the third section, systematic effects of configuration on effluent wastewater quality are studied and modelled. The quality and flow of the influent wastewater, as well as the operation of the system, vary with the flow to the wastewater treatment plant. Some observations on these interactions are illustrated with data from the pilot plant. Finally aspects of design and operation of the system are analysed using simplified modelling.

5.2 Experimental results

5.2.1 Introduction

In this section answers to practical questions on design and operation are sought. What will the effluent ammonium and nitrate concentrations be? What size anoxic tanks are needed? Is the problem of recirculated oxygen critical? What denitrification rates can be expected? Mainly the results of experimental periods C and D will be presented, since during these experimental periods the hydraulic retention time, the recirculation rate and other important factors were close to those of the planned application. In some cases the data from previous experimental periods A and B are also included. "Effluent concentration" refers to the concentration after the final sedimentation tank.

5.2.2 Effluent ammonium concentration

The effluent ammonium concentration (Figure 5.1) was mainly governed by the degree of recirculation of nitrified effluent from the RBC (Rotating Biological Contactor) to the activated sludge system. However during February to April of 1993 the RBC did not function perfectly, probably due to overloading and insufficient aeration, causing an elevation of the effluent ammonium concentration from the final sedimentation tanks in the order of 1.0 to 2.5 g N/m$^{3.}$ The average ammonium concentration during periods C and D was 8.8 g N/m^3. If the performance of the RBC had been as good as that of the pilot nitrifying trickling filter operated at the Rya WWTP, with an effluent ammonium concentration of 0.5 mg N/m^3, the average effluent ammonium concentration from the pilot plant would have been 0.25 g N/m^3 lower, that is about 8.55 g N/m^3.

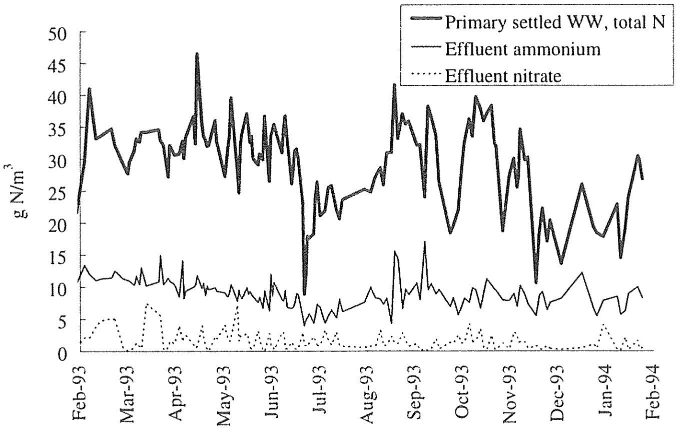


Figure 5.1 Total nitrogen concentration of primary settled wastewater and effluent ammonium and nitrate concentration from the final sedimentation tanks of the pilot plant (experimental periods C and D).

5.2.3 Effluent nitrate concentration

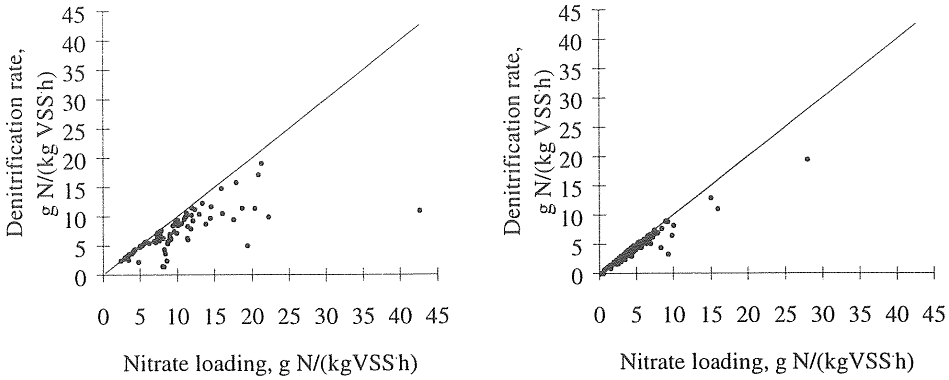
During experimental periods C and D the effluent nitrate concentration varied between 0.05 and 7.5 g N/m^3 averaging 1.5 g N/m^3. In 50 % of the samples the effluent nitrate concentration was below 1.15 g N/m^3, but 25 % of the values were above 2.3 g N/m^3. During experimental periods A and B the span was larger. The elevated effluent nitrate concentrations mainly occurred in two types of situations:

- High nitrate loading per unit of non-aerated biomass. This occurred during experimental period C, in experimental set-ups where only one or two out of six compartments were non-aerated (corresponding to 17 % or 34 % of the total activated sludge volume) and during experimental periods A and B when a large amount of nitrate was added to the system.
- Limiting carbon source. Elevated effluent nitrate concentrations on Sundays and to a lesser extent on Mondays is assumed to be caused by weak wastewater.

The influence of the nitrate loading is illustrated in Figure 5.2. A high degree of denitrification was obtained at loadings up to about 8 g N/(kg VSS·h). Above this level the degree of denitrification decreased and the highest denitrification rates measured were 10-16 g N/(kg VSS·h) at nitrate loadings of 15-25 g N/(kg VSS·h). During experimental periods C and D, when the future process was simulated, the average denitrification rate was 4.7 g N/(kg VSS·h). During experimental periods A and B, when nitrate was supplied by addition of sodium nitrate, the average denitrification rate was higher, 6.9 g N/(kg VSS·h), but the average degree of denitrification was low. However for loadings typical of experimental periods C and D, 2.5 - 8 g N/(kg VSS·h), about 90% of the added or produced nitrate was removed in both sets of data.

The effluent nitrate concentration increased with the nitrate loading (Figure 5.3). At loadings above 8 g N/(kg VSS·h) the effluent nitrate concentration frequently exceeded 5 g N/m^3. High nitrate loadings almost only occurred when nitrate was dosed to the system (during experimental periods A and B) and during experimental period C, when the non-aerated volume was less than 50 % of the total volume of the activated sludge tanks. The experimental procedures, however, influence interpretation of the data. During experimental period C the nitrate loading per non-aerated mass unit of suspended solids was manipulated by varying the number of aerated activated sludge tanks. In this case the variation of the nitrate loading per mass unit of volatile suspended solids was independent of the carbon to nitrate ratio. During experimental periods A and B, however, high nitrate loadings were systematically accompanied by low carbon to nitrate ratios, since the addition of nitrate was not matched by an addition of organic matter.

A practical conclusion to be drawn from Figure 5.3 is that if low effluent nitrate concentrations are essential, only process alternatives with a non-aerated volume corresponding to three non-aerated tanks in the pilot plant are realistic. If only one or two tanks are non-aerated the nitrate loading often exceeds 5 g N/(kg VSS·h) and elevated effluent nitrate concentrations are recorded. The average effluent nitrate concentration using two non-aerated tanks was 2.5 g N/m^3 (Table 5.1). When three tanks were non-aerated the average effluent nitrate concentrations were 0.8 and 1.6 g N/m^3 when primary settled wastewater was introduced in the second and first tank, respectively. Apparent differences in effluent nitrate concentrations between processes using two or three non-aerated tanks are statistically significant (t-test, 95 %) whereas the apparent differences between processes using one tank for deoxygenation and those where primary settled wastewater was introduced in the first tank are not significant.



a) Experimental periods A and B.

b) Experimental periods C and D.

Figure 5.2 Denitrification rate versus nitrate loading. (Nitrate loading and denitrification rate are calculated using net nitrate dosage to the system, net nitrate production in the RBC and net nitrate mass flow from the system. Solid line =complete denitrification.)

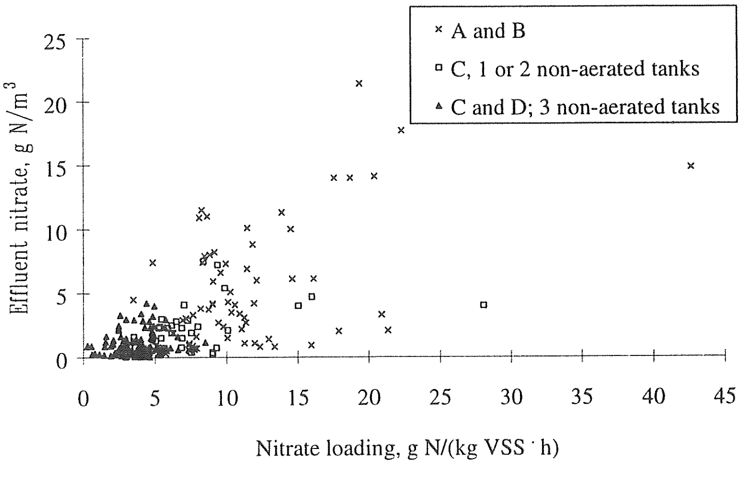


Figure 5.3 Effluent nitrate concentration as a function of the nitrate loading to the non-aerated activated sludge tanks.

Table 5.1 Nitrate loading, denitrification rate and effluent nitrate concentration, for the different ways of using the activated sludge reactor volume tested during experimental period C.

Volume (m^3)*			Volume non-aerated	Non-aerated AS	No. of days	Eff. NO_3^-	Nitrate loading on non-aerated activated sludge **		Nitrate removal	
Deox.	Anox.	Aerated	% of total	kg VSS		g N $/m^3$	g N/h	g N/ kg VSS·h	g N/h	g N/ kg VSS·h
0	2.7	12.9	17	2.9	2	3.6	64	22	44	15
2.7	2.7	10.3	34	9.5	16	2.5	66	6.9	53	5.6
0	5.5	10.3	34	7.8	12	2.5	61	7.8	52	6.7
2.7	5.5	7.7	52	13.5	16	0.8	59	4.4	55	4.1
0	8.2	7.7	52	12.8	17	1.6	61	4.8	56	4.4

*The volume of one non-aerated tank was 2.7 m^3 and one aerated tank 2.6 m^3.
** Total loading, includes recirculated non-denitrified nitrate.

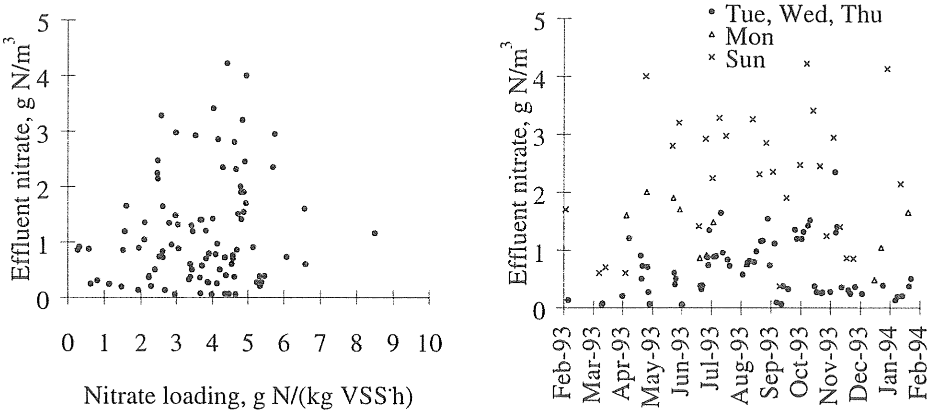
The following conclusions concerning denitrification rates and effluent nitrate concentrations can be drawn:

- Nitrate removal rates above 10 g N/(kg VSS·h) can be obtained in this system if the average effluent nitrate concentration is of no concern.
- Significant denitrification takes place in the deoxygenation zone.
- In a municipal wastewater treatment plant for nitrogen removal only systems with non-aerated zones corresponding to at least half the activated sludge volume. The total hydraulic retention time in non-aerated tanks was 1.7 h.

The influence of the carbon source on denitrification is indicated by the data from experimental periods C and D on those occasions when about 50 % of the activated sludge volume was non-aerated, thus avoiding occasions when elevated effluent nitrate concentrations were caused by high nitrate loading. Here the correlation between nitrate loading and effluent nitrate concentration is poor (Figure 5.4 a). However effluent nitrate concentrations were systematically higher on Sundays, and to a certain extent on Mondays, than on the remaining weekdays (Figure 5.4 b). This indicates a connection to wastewater quality variation. The effluent nitrate concentration on Sundays averaged 2.3 g N/m^3, on Mondays 1.3 g N/m^3 and on Tuesdays, Wednesdays and Thursdays, 0.6 g N/m^3. If Fridays and Saturdays are assumed comparable to Tuesdays, Wednesdays and Thursdays the total average effluent nitrate concentration would be 0.9 g N/m^3.

A popular criterion when quantifying the influence of the carbon source on denitrification is the carbon to nitrate supply or removal ratio. If a certain amount of carbon source can be used to denitrify a certain amount of nitrate a limiting carbon to nitrate loading ratio should exist. At carbon to nitrate ratios above that level nitrate would limit the process and the effluent nitrate concentration would be low. At lower ratios the carbon source limits denitrification and nitrate is present in the effluent. Inverting the carbon to nitrate ratio, the limiting nitrate removal per unit of carbon source can be obtained.

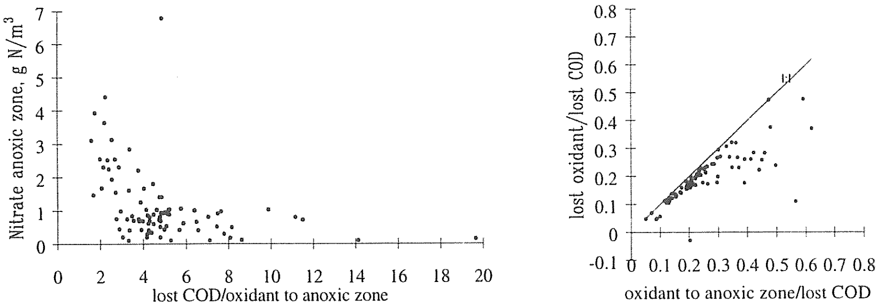
One problem is, as discussed earlier under "Pilot plant experiments", to quantify the carbon source and the nitrate loading. Here the nitrate loading will be replaced by the oxidant mass flow (which also includes an estimate of the oxygen mass flow from the RBC). It is reasonable that the soluble COD represents the most readily available organic matter and thus COD of the filtered primary settled wastewater can be used to quantify the carbon source. Up to nitrate to carbon loading ratios of 0.15 g N/g COD (carbon to nitrate ratios of 6.7) denitrification was close to complete, Figure 5.5 c and d. However, the spread of data is considerable, and, quite a few data points with low carbon to nitrate ratios show low nitrate concentrations. Assuming that some particulate matter may eventually be useful to denitrification the carbon source can be quantified as total COD in primary settled wastewater (Figure 5.5 e and f). Alternatively only soluble COD is available and some part of the soluble COD is inert. In this case a balance of influent and effluent COD of filtered samples may be appropriate (Figure 5.5 a and b).



a) Effluent nitrate concentration as a function of the nitrate loading.

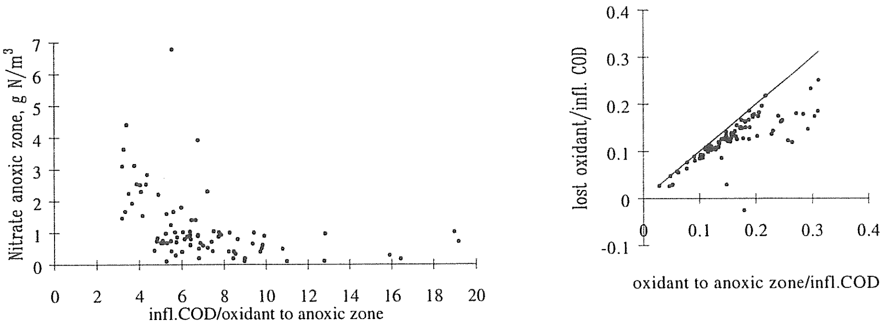
b) Effluent nitrate concentration on different days of the week.

Figure 5.4 Effluent nitrate concentration during experimental periods C and D. Data from days when three tanks, about 50% of the activated sludge volume, were non-aerated.



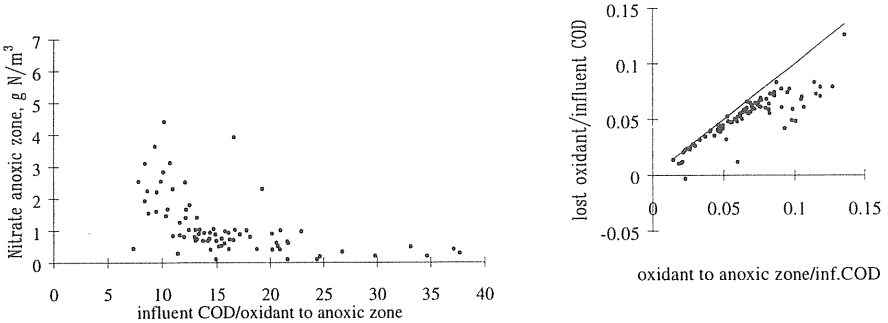
a) Lost COD = COD of filtered primary settled wastewater - COD of filtered effluent wastewater

b) Lost COD = COD of filtered primary settled wastewater - COD of filtered effluent wastewater



c) Influent COD = COD of filtered primary settled wastewater

d) Influent COD = COD of filtered primary settled wastewater



e) Influent COD = COD of primary settled wastewater

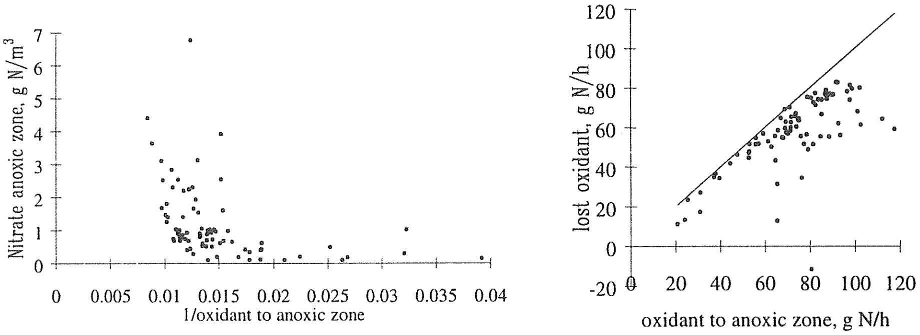
f) Influent COD = COD of primary settled wastewater

Figure 5.5 Nitrate concentration in the anoxic zone (a, c, e) and oxidant removal (b, d, f) related to carbon to oxidant (nitrate + oxygen) ratios where "carbon" is quantified in different ways.

It is not easy to determine from Figure 5.5 which carbon to oxidant ratio is the most relevant and excluding the carbon source from the diagrams gives much the same appearance (Figure 5.6). The quantification of carbon appears not to give much additional information. The main factor controlling denitrification in these experiments would appear to be the mass transport of oxidant to the anoxic zone. This does not mean that the carbon source is unimportant to denitrification, but that the influence of the carbon source on denitrification can be disguised in several ways:

- Insufficient variation of influent mass flow of organic matter.
- Variations of the influent mass flow may be further reduced by storage in the activated sludge.
- Flow dependent operation obscures the pattern. At high flows low recirculation causes less nitrate and oxygen to be recirculated and at low flows more is recirculated.
- The effect of limiting denitrification on nitrate concentrations is exaggerated by the recirculation of the non-denitrified nitrate.

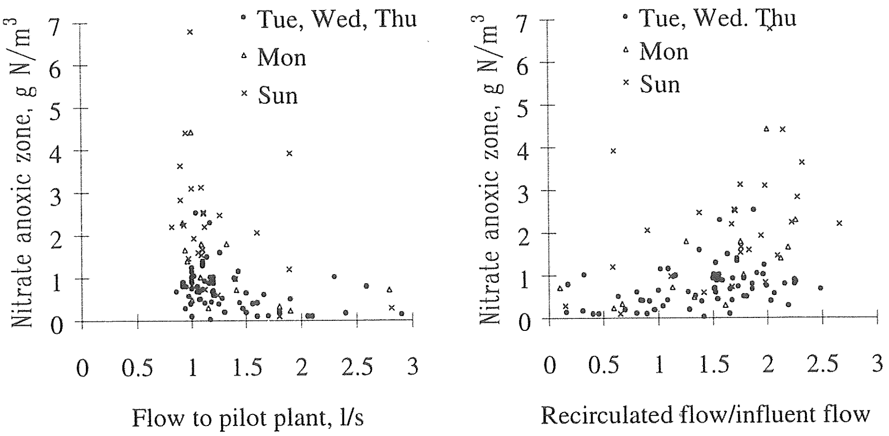
The first point, insufficient variation, should be an advantage from an operational point of view and if the supply is further equalised through storage so much the better. With an even supply of carbon the system should be easier to control. The interplay between flow dependent wastewater quality and flow dependent operation can be illustrated by the apparent dependency of the effluent nitrate concentration on the wastewater flow and on the recirculation ratio (Figure 5.7). Since all these parameters (recirculation ratio, flow, oxidant supply to the anoxic zone, carbon to nitrate ratio) are related it is not surprising that they are all more or less correlated and the effect of an individual parameter difficult to distinguish. However in the experiments poor denitrification often occurred when twice or more than the influent flow was recirculated.



a)

b)

Figure 5.6 Effect of oxidant supply on nitrate concentration and on oxidant removal.



a) Nitrate concentration versus the flow to the pilot plant.

b) Nitrate concentration versus the recirculation ratio.

Figure 5.7 Nitrate concentration in the anoxic zone.

The fourth point, that the effect of poor denitrification on the nitrate concentration and on the carbon to nitrate ratio is exaggerated by recirculation can be illustrated using mass flows of nitrate, oxygen and COD (Table 5.2). Nitrate production in the RBC was similar on different days of the week (column A), but nitrate removal in the non-aerated tanks (column F) was lower on Sundays than on Tuesdays, Wednesdays and Thursdays. The difference between produced nitrate in the RBC and reduced nitrate in the non-aerated tanks caused an extra amount of nitrate to be recirculated in the system (column G). This amount of recirculated nitrate increased until steady state was reached, when the extra amount of nitrate produced equalled the effluent mass flow of nitrate. Under these conditions the extra nitrate production (column G) gave an extra nitrate mass flow of 20 g N/h on Sundays, 13 g N/h on Mondays and 7 g N/h on Tuesdays, Wednesdays and Thursdays (column H). These values agree well with the measured mass flow of effluent nitrate recirculated via the RBC and by return activated sludge to the non-aerated tanks (column B). The internal recirculation of nitrate accounts for about 30 % of the nitrate mass flow to the non-aerated tanks on Sundays, 20 % on Mondays and under 10 % on other days. It also causes an exaggeration of the differences in carbon to nitrate ratios between days with poor denitrification (Sunday) and days with good denitrification (Tuesday , Wednesday, Thursday, compare columns M and P).

Table 5.2 Mass flows of nitrate, oxygen and COD. Experimental periods C and D, when three activated sludge tanks were not aerated.

	A NO_3^- production in RBC g N/h	B Recirculation of eff. NO_3^- g N/h	C NO_3^- to non-aerated tanks g N/h	D O_2 to non-aerated tanks g N/h	E=C+D NO_3^- + O_2 to non-aerated tanks g N/h	F NO_3^- rem. in non-aerated tanks g N/h	G=A-F Produced NO_3^- in RBC not reduced g N/h	H Calculated rec. of NO_3^- to non-aerated tanks g N/h
Sun.	46	19	63	20	83	34	12	20
Mon.	45	11	50	17	67	37	8	13
Tue. Wed. Thu.	48	5	52	17	69	44	4	7

	I COD of filt. prim. settled WW g O_2/h	J COD of filtered sec. Effluent g O_2/h	K=I-J Lost filtered COD in biological treatment g O_2/h	L COD of primary settled WW g O_2/h	M=I/E Carbon to oxidant ratio g O_2/g N	N=K/E Carbon to oxidant ratio g O_2/g N	O=L/E Carbon to oxidant ratio g O_2/g N	P=I/(A+D) Carbon to oxidant ratio g O_2/g N
Sun.	390	160	230	1080	4.7	2.8	13	5.9
Mon.	480	175	305	1180	7.2	4.6	18	7.7
Tue. Wed. Thu.	535	190	345	1250	7.8	5.0	18	8.2

The following conclusions concerning the influence of the carbon source on denitrification can be drawn:

- The average effluent nitrate concentration was higher on Sundays and Mondays (2.3 and 1.3 g N/m^3 respectively) than on Tuesdays, Wednesdays and Thursdays (0.6 g N/m^3). The weighted average effluent concentration is 0.9 - 1.0 g N/m^3.
- At a ratio of the nitrate (+ oxygen) loading to the loading of COD with primary settled wastewater (filtered sample) up to about 0.15 g N/g O_2 (corresponding to a carbon to oxidant ratio of 6.7 g O_2/g N), denitrification was close to complete giving low effluent nitrate concentrations.
- Above this ratio, elevated effluent nitrate concentrations were frequently observed.
- Ratios of nitrate (+ oxygen) to COD loading above 0.2 g N/g O_2 were almost only observed on Sundays and Mondays.
- Elevated nitrate concentrations often occurred at recirculation ratios at or above 200% of influent flow.
- With this process, flow, operation and wastewater quality are related in a way which makes it difficult to positively separate the effect of the carbon source on denitrification from effects of flow and operation.
- Poor denitrification causes more nitrate to circulate in the system, exaggerating the effect of the nitrate to COD ratio.

5.2.4 Effluent total nitrogen concentration

The effluent total nitrogen content consists of ammonium, nitrate and organic nitrogen. Factors controlling these components are presented in Table 5.3. Addition of the different components gives an effluent total nitrogen content of 11-11.4 g N/m^3. Since, in the experiments, primary settled wastewater contained 27.6 g N/m^3 this corresponds to a removal of nearly 60 % of the nitrogen in primary settled wastewater. The results of a full-scale, optimised plant should be better. The recirculation flow can be optimised and a carbon source can be added. In chapter 5.4 a few of these possibilities will be investigated.

Table 5.3 Effluent nitrogen concentration from the system and discussion of factors influencing removal.

Component	Effluent concentration g N/m^3	Factors affecting effluent concentration	Means of reducing effluent concentration	Causes of increased effluent concentration
Ammonium	8.6*	Mass flow of nitrogen /ammonium from primary treatment. Release/uptake of ammonium in activated sludge system. Degree of recirculation. Capacity of nitrification unit.	Less influent nitrogen. Higher degree of recirculation. Higher capacity in nitrification unit. Improved control of system.	More influent nitrogen. Lower recirculation capacity than expected. Lower nitrification capacity than expected. Lower flow due to less dilution. Higher flow due to more wastewater.
Nitrate	0.9-1.0*	Carbon source. Recirculation of nitrate. Recirculation of oxygen. Mass of non-aerated activated sludge. Denitrifying activity of activated sludge.	Add carbon source. Minimise oxygen recirculation. Larger non-aerated activated sludge volumes. Improved control of system.	Deterioration of wastewater carbon source.
Organic nitrogen	~1.5-1.8**	Removal of particles. Production of soluble organic matter. Removal of soluble organic matter.	Final effluent filtration. Improved flocculation and sedimentation.	Deterioration of flocculation and settling properties.
Total nitrogen	~11-11.4			

* Average from pilot plant operation during experimental periods C and D when 50 % of the activated sludge volume was non-aerated, assuming good performance of the nitrifying unit. Data are weighted in order to compensate for different numbers of samplings on different weekdays.

** Effluent from main plant 1993 (sedimentation in the pilot plant was probably not representative of a full-scale plant).

5.2.5 Other parameters

Although the focus of interest when operating the pilot plant was on nitrogen removal other parameters were also registered. These are summarised in Table 5.4 and will briefly be commented on below.

The sedimentation characteristics of the activated sludge were tolerable, the average stirred sludge volume index 70 ml/g SS. However the effluent suspended solids concentration was not monitored the way it would be in a full-scale plant. This, together with the differences in design of the sedimentation tank may explain the slightly higher than expected effluent suspended solids concentration, 24 g SS/m^3, and total phosphorus concentration, 0.5 g P/m^3.

The dosage of iron sulphate was 21 g Fe/m^3 resulting in an effluent phosphate concentration of 0.04 g P/m^3. Probably the dosage could have been decreased and the effluent concentration lowered if more emphasis had been placed on optimising these parameters.

Table 5.4 Results during experimental periods C and D (930211-940210). Average of all registered values- no compensation for the fact that some days of the week were sampled more often than others.

Parameter	Position	Unit	Average	Standard deviation.	Number of values
Flows	Influent	l/s	1.4 l/s	0.4	356
	Return activated sludge	l/s	0.73 l/s	0.03	360
	Recirculated effluent	l/s	1.6 l/s	0.4	350
	Excess sludge	m^3/d	1.3 m^3/d	0.3	169
Dosage of iron	-	gFe/m^3	21		40
Temperature	Primary settled WW	°C	14.1	3.0	224
	In activated sludge tank	°C	14.9	2.9	224
Suspended solids	Activated sludge tank	kg SS/m^3	2.5	0.5	196
	Deoxygenation tank	kg SS/m^3	3.9	1.2	124
	Return activated sludge	kg SS/m^3	11.4	2.3	184
	Effluent	g SS/m^3	24	29	141
Ash content	Activated sludge	% of SS	43	6	187
COD	Primary settled WW	g O_2/m^3	285	94	143
COD filtered sample	Primary settled WW	g O_2/m^3	118	35	142
	Return activated sludge	g O_2/m^3	72	34	94
	Deoxygenation zone	g O_2/m^3	46	12	74
	End anoxic zone	g O_2/m^3	54	31	141
	Effluent	g O_2/m^3	37	10	141
BOD	Effluent	g O_2/m^3	8.6	3.9	71
Phosphate	Effluent	g P/m^3	0.04	0.04	139
Phosphorus	Effluent	g P/m^3	0.49	0.42	142
Ammonium:	Primary settled WW	g N/m^3	21.5	6.2	141
	End anoxic zone	g N/m^3	7.5	1.7	50
	Effluent	g N/m^3	8.9	2.3	142
	RBC	g N/m^3	1.1	1.0	139
Nitrate:	Primary settled WW	g N/m^3	0.2	0.3	141
	Deoxygenation zone	g N/m^3	2.7	2.1	68
	End anoxic zone	g N/m^3	1.4	1.2	141
	Effluent	g N/m^3	1.6	1.5	142
	RBC	g N/m^3	9.4	2.3	135
Nitrite:	Effluent	g N/m^3	0.4	0.3	142
	RBC	g N/m^3	0.4	0.2	136
Total nitrogen:	Primary settled WW	g N/m^3	29.0	6.8	142
	Effluent	g N/m^3	12.3	2.7	143
	RBC	g N/m^3	11.8	2.9	142
	Return activated sludge	g N/m^3	342	105	139
Alkalinity:	Primary settled WW	eq/m^3	3.7	0.6	140
	Effluent	eq/m^3	2.1	0.3	140
	RBC	eq/m^3	1.0	0.3	127
Oxygen:	Aeration tank	g O_2/m^3	3.0	1.0	162
	RBC	g O_2/m^3	8.6	0.6	135
pH effluent			6.9	0.3	303
Ox. red. potential:	Last anoxic tank	mV	-109	115	303
	Last aerated tank	mV	-10	143	298
Last aerated tank:	SV	ml/l	307	140	48
	SSV	ml/l	178	41	48
	ISH	m/h	3.1	1.0	48

5.3 Nitrogen removal

5.3.1 Introduction

In this section the mechanisms of nitrogen removal in the non-nitrifying activated sludge system will be examined. First, factors controlling denitrification in the non-aerated tanks are examined. The effect of wastewater organics on the denitrification rate will be investigated in section 5.3.2.

In section 5.3.3 a mass balance of the system will be made in order to establish the role played by different parts of the system in nitrogen removal. The possibility of nitrification in the aerated tanks is evaluated as is the extent to which seeding of nitrifiers from the nitrifying unit may influence nitrification in the activated sludge system.

5.3.2 Denitrification rates in non-aerated activated sludge

During 1993 and 1994, profiles of ammonium and nitrate were measured through the system on a number of occasions. Samples were collected from each activated sludge tank, from return activated sludge, from influent and effluent flow, and before and after the RBC. The samples were filtered immediately and analysed for ammonium, nitrate and in some cases COD. With knowledge of flows and suspended and volatile suspended solids concentrations, denitrification rates in the individual tanks can be estimated. However, some conditions of the measurements should be observed:

- Profiles give a momentary picture of the process, not taking dynamics into account.
- Most of the profiles were measured on Tuesdays, Wednesdays or Thursdays. In nearly all cases most of the recirculated nitrate was removed in the activated sludge system. The total nitrogen removal was therefore limited by the recirculation of nitrate, not by the carbon source.
- The effluent nitrate concentration was in most cases slightly lower than that of the corresponding 24-hour sample (average 0.23 g NO_3-N/m^3 compared with 0.55 g NO_3-N/m^3). This may be caused by methodological differences of sampling. It might also be due to the fact that most profiles were sampled late morning or early afternoon (10^{00} - 14^{00}). A systematic diurnal variation of nitrate or organic loading may cause a systematic difference between 24-hour samples and grab samples.

Average mass flows of nitrate and ammonium for configurations with about 50 % of the activated sludge volume non-aerated are presented in Figure 5.8 and Figure 5.9. About 90% of the recirculated nitrate was removed in the non-aerated tanks. The nitrate removal was greatest in the first anoxic tank, where recirculated nitrate and raw wastewater were first mixed, and lower in each of the following tanks, where the nitrate concentration may be limiting or the most readily degradable organic matter from the wastewater may have been consumed. A small amount of nitrate was removed in the third non-aerated tank and in connection with the final sedimentation tank.

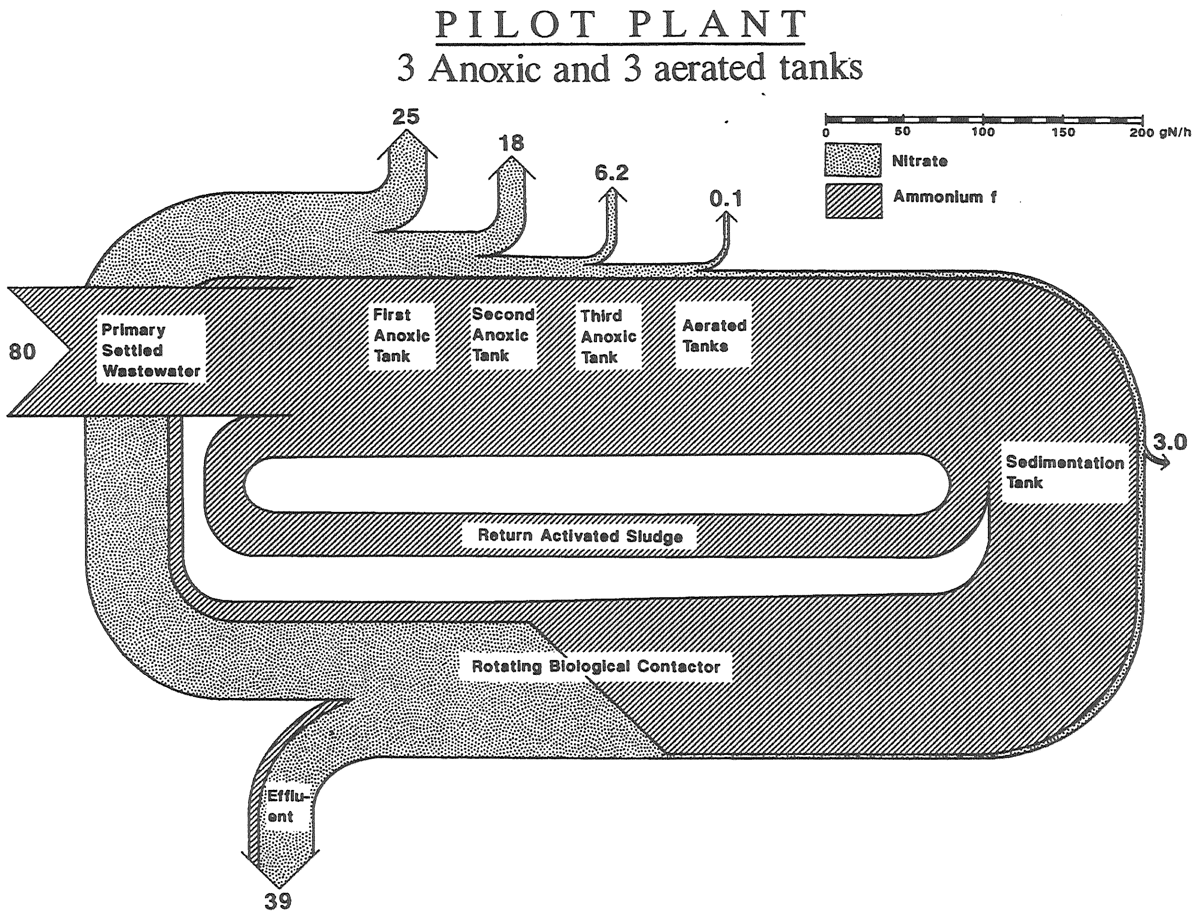


Figure 5.8 Mass flow of inorganic nitrogen in the pilot plant when three anoxic tanks were used.

Average of 8 profiles measured when wastewater, return activated sludge and recirculated water were mixed in the first anoxic tank.

PILOT PLANT

1Deoxygenation, 2 anoxic and 3 aerated tanks

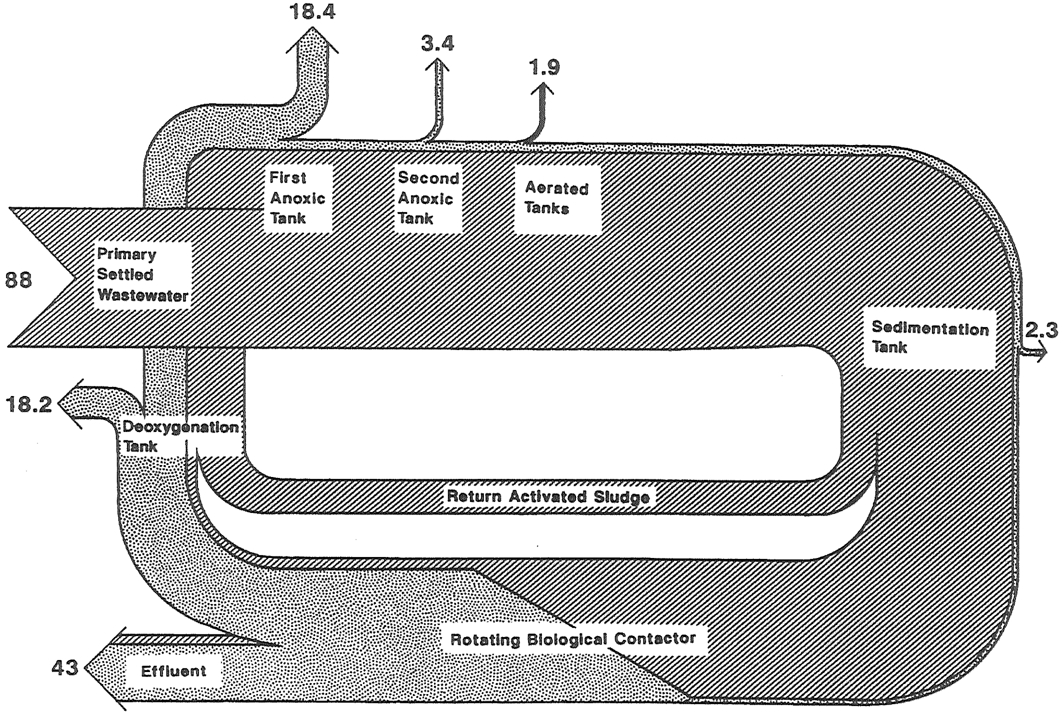


Figure 5.9 Mass flow of inorganic nitrogen in the pilot plant when one deoxygenation tank and two anoxic tanks were used.

Average of 17 profiles measured when the first non-aerated tank was used for deoxygenation. Symbols as in Figure 5.8

When one tank was used as a deoxygenation tank 40 % of the recirculated nitrate was already removed in this tank.

Denitrification rates in each tank can be estimated from the profile data. The discussion will focus on two different rates:

- The rate in the first anoxic tank, where primary settled wastewater was added, represents a high denitrification rate with a readily available carbon source.
- The rate in the deoxygenation tank where no wastewater was added represents the background activity of the activated sludge.

In the following, "the first anoxic tank" denotes the tank where primary settled wastewater is introduced. In some cases this tank is preceded by a deoxygenation tank.

In the deoxygenation tank the average nitrate removal rate was 3.2 g N/(kg VSS·h) and an almost equal amount of oxygen (equal on an electron transfer basis) was removed giving a removal rate of nitrate and oxygen of 6.3 g N/(kg VSS·h). The nitrate removal rate in the first anoxic tank was around 5.5 g N/(kg VSS·h) with no major difference whether this was preceded by a deoxygenation tank or not(Table 5.5). If oxygen is reduced in preference to nitrate and a certain amount of oxygen is recirculated either to the first anoxic tank or to a deoxygenation tank, denitrification in the anoxic tank would be assumed to be better if it was preceded by a deoxygenation tank. The deoxygenation tank, however, had another impact on the first anoxic tank. Not only was oxygen reduced, but also nitrate, reducing the nitrate loading on the first anoxic tank from 11.7 g N/(kg VSS·h) to 8.1 g N/(kg VSS·h). Nitrate and oxygen loading is a factor which appears to influence the removal of nitrate and oxygen (see Figure 5.10). The correlation between loading and nitrate removal is not only a reflection of the fact that low loadings may cause low nitrate concentrations and thus low reduction rates. The coefficient of correlation between removal rate of nitrate and nitrate concentration was close to 0 (-0.04) in the deoxygenation tank and 0.4 in the first anoxic tank indicating that this factor was less important than the total nitrate and oxygen loading (correlation coefficients between nitrate and oxygen loading, and nitrate and oxygen removal were 0.9 and 0.8 in the first anoxic tank and deoxygenation tank respectively).

Table 5.5 Denitrification rates of the deoxygenation tank and the first anoxic tank.

	Data included	Number of profiles included	Nitrate loading g N/(kg VSS·h)	Denitrification rate g N/(kg VSS·h)	Nitrate + oxygen loading g N/(kg VSS·h)	Nitrate + oxygen removal g N/(kg VSS·h)
Deoxygenation tank	All data	16	9.4	3.2	11.9	6.3
	NO_3-conc. > 1 mg N/l	14	8.9	3.0	12.5	6.1
First anoxic tank	All data	8* 16**	11.7* 8.1**	5.4* 5.6**	15.7* 8.1**	9.4* 5.6**
	NO_3-conc. > 1 mg N/l	8* 5**	11.7* 10.5**	5.4* 5.0**	15.7* 10.5**	9.4* 5.0**

*The first anoxic tank was not preceded by a deoxygenation tank

**The first anoxic tank was preceded by a deoxygenation tank

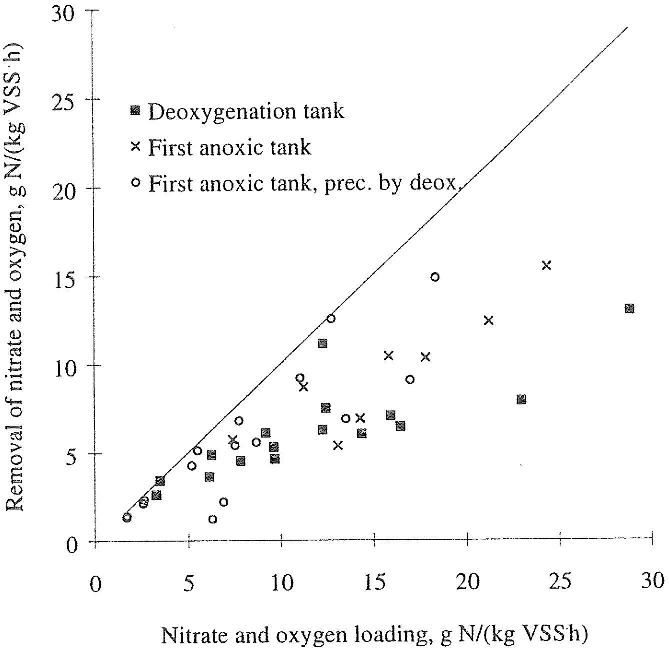
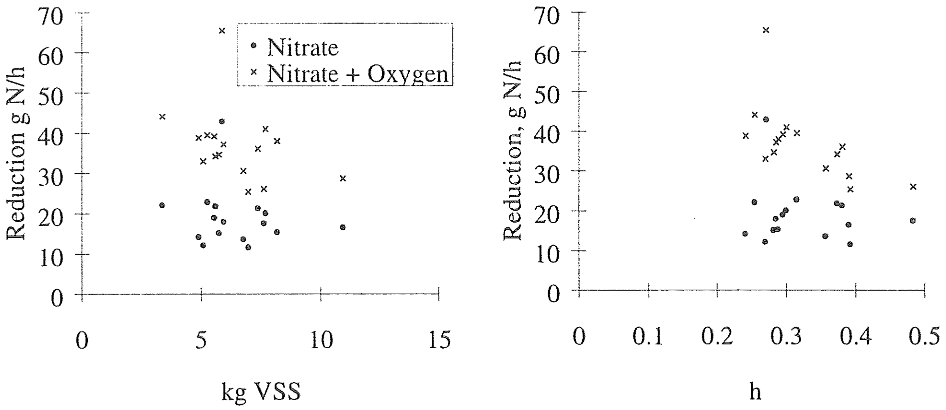


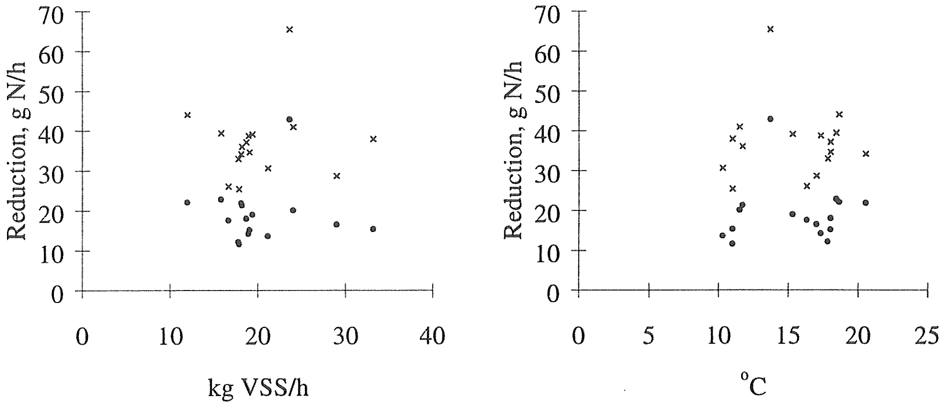
Figure 5.10 Removal rate and loading of nitrate and oxygen in the deoxygenation tank and in the first anoxic tank.

The nitrate and oxygen removal in the deoxygenation tank exhibited some unexpected behaviour. A biological process performed by a mass of bacteria may be expected to be influenced by the amount of bacteria present, the supply of substrate, the temperature, the retention time and other factors. Apparently, these factors individually were of little importance (see Figure 5.11). The amount of bacteria, characterised as the total amount of volatile suspended solids (VSS), showed little influence on nitrate removal. The nitrate removal rate was apparently also independent of the retention time in the deoxygenation tank, whereas the nitrate plus oxygen removal decreased with increased retention time. This was caused by systematic variations of the system. At high recirculation flows the retention time decreased (since the volume remained constant) but the amount of recirculated oxygen increased, since this was proportional to the recirculated flow. Since in all cases a substantial nitrate removal occurred, all the recirculated oxygen can also be assumed to have been removed. Since the nitrate removal, and to a certain extent the oxygen removal, was apparently independent of the amount of bacteria and of the retention time, it would be reasonable to assume that the removal was limited by the supply of readily available carbon.



a) Mass of volatile suspended solids in deoxygenation tank.

b) Retention time in deoxygenation tank.



c) Mass flow of return activated sludge to deoxygenation tank.

d) Water temperature.

Figure 5.11 Nitrate removal, and removal of nitrate + oxygen, as a function of various biologically relevant parameters.

The recirculated nitrified effluent contained very little organic matter. The effluent filtered COD was 41 g O_2/m^3, further decreased to 35 g O_2/m^3 in the RBC. The other supply of organic matter for reduction of nitrate and oxygen was the return activated sludge. However, if the available organic matter was entrapped in or produced by the activated sludge, it is reasonable to expect the supply of carbon for denitrification to be proportional to the mass flow of volatile suspended solids recirculated to the deoxygenation tank. However, this was not supported by data (Figure 5.11 c). The removal of nitrate and oxygen did not increase with the transport of volatile suspended solids to the deoxygenation tank. The average nitrogen removal was 19 g N/h and the oxygen removal was 18 g N/h expressed as nitrate equivalents.

There are several possible explanations for nitrate removal being apparently independent of biologically relevant factors. One possibility is that the above mentioned factors are relevant, but are masked or counteracted by other systematic variations of the system. In the system several parameters are related:

- the influent flow and the recirculated flow
- the influent flow and the influent wastewater quality
- the recirculated flow and the supply of nitrate to the deoxygenation tank
- the recirculated flow and the VSS-concentration of the deoxygenation tank
- the recirculated flow and the amount of recirculated oxygen
- the influent and recirculated flow and the carbon and nitrate supply to the activated sludge system

One scenario of related factors is that at high influent flows to the plant the recirculated flow is lower, causing the return sludge to be less diluted by recirculated water and thus causing the volatile suspended solids concentration in the deoxygenation tank to be higher and thus potentially giving better nitrate and oxygen removal. At the same time the quality of the influent carbon source may be inferior at high flows, due to processes in the collection system, thus decreasing the amount of available carbon for reduction of nitrate and oxygen.

Another possibility is that some part of the nitrate and/or oxygen removal is not caused by direct biological respiration, but through some biochemical or chemical process. One such mechanism could be reduction and oxidation of iron or iron complexes. Since phosphorus is removed by the addition of iron sulphate in the activated sludge system the activated sludge contains a large amount of iron. Approximately 4 kg of iron reaches the deoxygenation tank per hour. In order to reduce 50 g O_2 about 350 g Fe^{2+} would be oxidised to Fe^{3+}. This corresponds to about 9 % of the recirculated 4 kg/h of iron. A recent investigation (Nielsen, 1996 B) indicate that reduction of Fe^{3+} and oxidation of Fe^{2+} may be important reactions in biological wastewater treatment. He observed that "reoxidation of Fe(II) to Fe(III) was very rapid" and further that "Oxidation of Fe(II) by nitrate and nitrite seemed to take place in activated sludge and should be investigated in more detail".

An attempt to calculate the temperature dependency of the maximum nitrate and oxygen removal rate in the first anoxic tank and in the deoxygenation tank was undertaken. A linear regression was made using the linearised form of the temperature dependency equation (Figure 5.12). The spread of data was considerable (Figure 5.13 a). However, as it is the temperature dependency of the maximum denitrification rate which is sought, data where some limitation can be expected may be excluded. Data obtained under limiting conditions were excluded using the following criteria:

- Low nitrate concentration. Data obtained when the nitrate concentration in the tank was 1.0 mg N/l or lower were excluded.
- High influent flow. Data obtained at high main plant influent flow were excluded on the assumption that a wastewater quality variation, possibly in combination with the above mentioned systematic effects, may cause denitrification limitation. Data were

excluded if the flow to the main plant was 4 m^3/s or more (the average flow was just below 4 m^3/s).

The latter restriction can be motivated if high flow is correlated to poor carbon source or excessive oxygen transport to the plant. As can be seen in Figure 5.13 b the spread of data is still considerable rendering poor regressions. The regression coefficients (r^2) are 0.3 and 0.2 for the nitrate and oxygen reduction rates of the deoxygenation tank and the first anoxic tank respectively. The temperature constant calculated thus (k_t) is 0.02 in both tanks. The nitrate and oxygen reduction rates at 20 °C are 7.9 g N/(kg VSS·h) in the deoxygenation tank and 10.4 g N/(kg VSS·h) in the first anoxic tank. The nitrate and oxygen removal rates of those values which were excluded are on average lower than the calculated regression lines. For the deoxygenation tank five values were excluded due to high influent flow. The nitrate and oxygen removal rates of these are on average 2 g N/(kgVSS·h) or 32 % lower than the regression line. In the first anoxic tank the nitrate concentration was below 1 g N/m^3 on 11 occasions with nitrate and oxygen removal rates on average 3 g N/(kg VSS·h) or 35 % lower than the regression line. An even greater decrease was displayed by the 6 values with high flow where the nitrate and oxygen removal rate was 4.6 g N/(kg VSS·h) or 60 % lower than the regression line.

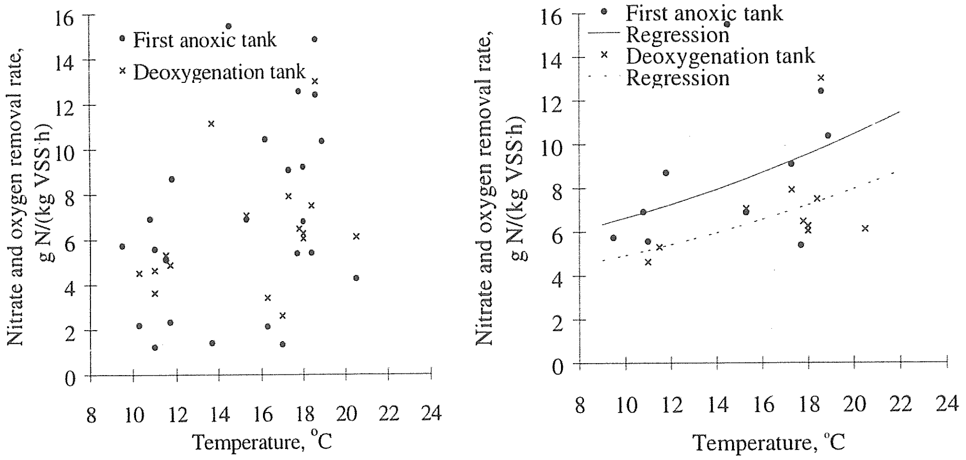
$$r_{x,t} = r_{x,20} \cdot 10^{k_t \cdot (t-20)}$$

linearised:

$$\log r_{x,t} = \log r_{x,20} \cdot k_t \cdot (t-20)$$

Symbol	Explanation	Unit
t	Temperature	° C
$r_{x,t}$	Denitrification rate at the temperature t ° C	
k_t	Temperature constant	

Figure 5.12 Temperature dependency of denitrification.



a) All data.

b)Normal or low flow, nitrate concentration above 1 mg N/l.

Figure 5.13 Nitrate and oxygen removal rates and temperature in the first anoxic tank and in the deoxygenation tank.

Conclusions:

- The average nitrate and oxygen removal rate in the deoxygenation tank was 6.3 g N/(kg VSS·h). The nitrate removal rate was 3.2 g N/(kg VSS·h). No clear dependencies of nitrate and oxygen removal on factors normally expected to influence biological respiration in an activated sludge tank were observed (such as temperature, volatile suspended solids and retention time).
- The average nitrate removal rate in the first anoxic tank was 5.5 g N/(kg VSS·h) and in those cases when oxygen was recirculated to the first anoxic tank the removal rate of nitrate and oxygen was 9.4 g N/(kg VSS·h).
- A dependency of the nitrate and oxygen removal rate on temperature may be observed. A temperature constant (k_t) of 0.02 can be estimated in the first anoxic tank and in the deoxygenation tank. The regression coefficients, however, are poor. Several factors may limit the removal of nitrate and oxygen, masking or counteracting the effect of temperature.

5.3.3 Nitrification and denitrification in aerated activated sludge

The nitrate and ammonium profiles evaluated in the previous section can be used when estimating denitrification rates in the non-aerated tanks, where no nitrification should occur. In the aerated tanks, however, matters may be complicated by nitrification. In order to quantify the removal of nitrogen by different mechanisms in the system, mass balances of nitrogen are set up for each month during experimental period D (Table 5.6). During this period three out of six activated sludge tanks were non-aerated and high effluent nitrate concentrations due to insufficient non-aerated volume were thus avoided. Nitrogen concentrations and flows used for the mass balances were monthly weighted averages (weighted so as to take into account the varying numbers of samplings on different days of the week).

The influent nitrogen mass flow, 131 g N/h, was reduced in the system by 76 g N/h to 55 g N/h. Of the 10.6 g N/m^3 in the effluent, 1.2 g N/m^3 was organic nitrogen. The effluent inorganic nitrogen (9.4 g N/m^3) consisted of

- nitrate not denitrified in the activated sludge system; 1.0 g N/m^3
- nitrate from the RBC not recirculated to the activated sludge system; 7.2 g N/m^3
- ammonium not nitrified in the RBC and not recirculated; 1.2 g N/m^3

In a system where only water to be recirculated is introduced to the nitrification unit, the main part of the effluent inorganic nitrogen would have been in the form of ammonium. Some of the transformations leading to nitrogen removal can be determined using a mass balance approach. The nitrate which leaves the RBC, is recirculated to the activated sludge system and is not observed in the last anoxic tank, can be assumed to have been removed by denitrification. This accounted for 33 g N/h or 44 % of the removed nitrogen. The small amount of nitrate in the influent wastewater was also assumed to be removed in the anoxic tank. Denitrification of nitrate in influent wastewater and return activated sludge amounted to 4 g N/h or about 5 % of the total nitrogen removal. With the excess sludge 18 g N/h was removed, accounting for 24 % of the nitrogen removal from the system. A varying amount of nitrogen was lost in the RBC. The differences in concentration between influent and effluent total nitrogen of the RBC varied between 0.2 to 1.3 g N/m^3 (on average 0.6 g N/m^3).

The remaining, as yet unexplained, loss of nitrogen, 14 g N/h, took place in the activated sludge system. This accounted for 18 % of the total nitrogen removal in the system. This loss can have different explanations;

- Systematic error in sampling or analysis
- Nitrification and denitrification in activated sludge system
- Other biological or chemical processes

The loss of 14 g N/h in the high flow passing through the activated sludge tank corresponds to a removal of about 1 g N/m^3. Assuming that all the error is due to more nitrate than registered having been supplied to the activated sludge system, then the nitrate concentration of the water from the RBC would have to have systematically contained 2.5 g N/m^3 more nitrate than measured. The total nitrogen concentration of the excess sludge would have to have been nearly double the measured concentration in order to account for the unexplained nitrogen loss. This does not seem reasonable, although a combination of small systematic errors cannot be excluded. Adsorption of ammonium to activated sludge in anoxic zones (Nielsen, 1996 A) and release of ammonium through hydrolysis in the activated sludge system (Henze and Mladenovski, 1991) may be important mechanisms, but they do not cause net removal of nitrogen from the activated sludge system.

Nitrification and denitrification in the aerated tanks of the activated sludge system will now be considered, although the activated sludge system had an aerated solids retention time between 1 and 1.5 days which would normally be too low to allow a significant population of nitrifiers to develop. However in this case a seeding effect of nitrifiers from the RBC could be expected. Daigger et al. (1993) developed equations for estimating the effect of seeding in a similar system, based on the growth rate of nitrifiers recommended by EPA and the effect of the addition of nitrifiers from a constantly sloughing trickling filter. Their contribution was subsequently discussed by Parker et al. (1994) who pointed out that the growth rate of nitrifiers suggested by EPA was intended for design of activated sludge systems using safety factors in order to provide protection from site to site variations, and was not intended as a precise test of whether nitrification would occur in any specific situation. They also questioned that nitrifiers would be continuously sloughed in stoichiometric quantities. Daigger et al. responded (1995) that although the authors were unaware of direct measurements of nitrifiers in trickling filter effluents, logically "seeding of nitrifiers from a trickling filter accomplishing combined carbon oxidation and nitrification must occur to some extent". However they assume that since models based on the assumption work, the approach is useful.

Table 5.6 Mass balance of nitrogen in the system during experimental period D.

				Influent[1]	Effluent[1]					Removed[1]						Calculated [1,2]		
	Influent flow, m³/h	Recirculated flow, m³/h	Temperature, °C	Total	Total	Nitrified in the RBC but not recirculated	Not nitrified in the RBC	Not denitrified in activated sludge	Organic nitrogen	Total	Nitrate from RBC, de-nitrified in anoxic tanks	Nitrate in influent and RAS, lost in anoxic tanks	With excess sludge	Removed in RBC	Unexplained removal in AS	Net nitrate removal in aerated tanks	Nitrification	Denitrification
Jul.	4.7	6.1	18	**111.5**	**38.9**	**27.0**	**3.5**	**4.9**	**3.0**	**72.5**	**27.7**	**4.1**	**10.9**	**10.9**	**24.9**	**0.5**	**24.4**	**24.9**
1993				23.8	8.2	5.8	0.7	1.0	0.6	15.5	5.9	0.9	2.3	2.3	5.3			
Aug.	5.7	5.0	18	**140.3**	**58.5**	**39.9**	**5.6**	**6.2**	**6.1**	**81.8**	**28.5**	**3.5**	**16.2**	**8.8**	**24.9**	**-1.7**	**26.6**	**24.9**
1993				24.8	10.2	7.1	1.0	1.1	1.1	14.5	5.0	0.6	2.9	1.6	4.4			
Sep.	4.0	6.5	18	**138.3**	**51.7**	**35.9**	**5.5**	**4.3**	**5.8**	**86.6**	**45.4**	**3.4**	**16.1**	**3.6**	**18.2**	**5.8**	**12.4**	**18.2**
1993				34.3	12.8	8.9	1.4	1.1	1.4	21.5	11.3	0.8	4.0	0.9	4.5			
Oct.	4.8	5.7	16	**141.6**	**56.8**	**36.3**	**4.0**	**7.7**	**8.1**	**84.8**	**40.1**	**4.6**	**21.9**	**4.9**	**13.3**	**-8.2**	**21.5**	**13.3**
1993				29.4	11.6	7.5	0.8	1.6	1.7	17.6	8.3	1.0	4.5	1.0	2.8			
Nov.	4.1	6.5	14	**125.3**	**47.9**	**34.2**	**3.1**	**4.2**	**5.7**	**77.4**	**42.5**	**3.3**	**20.9**	**1.9**	**8.9**	**6.6**	**2.3**	**8.9**
1993				30.7	11.6	8.4	0.8	1.0	1.4	19.0	10.4	0.8	5.1	0.5	2.2			
Dec.	7.1	3.8	11	**132.4**	**71.3**	**47.0**	**13.7**	**2.8**	**4.7**	**61.1**	**26.6**	**5.8**	**21.8**	**4.4**	**2.6**	**-3.5**	**6.1**	**2.6**
1993				18.6	9.6	6.6	1.9	0.4	0.7	8.6	3.7	0.8	3.1	0.6	0.4			
Jan.-Feb.	6.0	4.6	9	**126.7**	**62.1**	**37.2**	**8.8**	**5.0**	**9.6**	**64.7**	**23.1**	**4.5**	**18.1**	**14.1**	**4.9**	**-0.5**	**5.4**	**4.9**
1994				21.2	10.1	6.2	1.5	0.8	1.6	10.8	3.9	0.8	3.0	2.4	0.8			
Ave.	5.2	5.5	15	**130.9**	**55.3**	**36.8**	**6.3**	**5.0**	**6.1**	**75.6**	**33.4**	**4.2**	**18.0**	**6.9**	**13.9**	**-0.1**	**14.1**	**13.9**
				26.1	10.6	7.2	1.2	1.0	1.2	15.3	6.9	0.8	3.6	1.3	2.9			
% of infl. N				100	42	28	5	4	5	58	26	3	14	5	11		11	11

[1] **Bold figures refer to mass flows or removals; g N/h.** Normal font figures refer to concentrations; g N/m^3.

[2] Calculated assuming that the unexplained removal of nitrogen in the activated sludge tanks was caused by nitrification and denitrification in the aerated compartments.

Though the uncertainties concerning the constant sloughing of biofilm and use of growth rates as pointed out by Parker et al. are recognised, the approach may give an indication of the influence of seeding in this system, and the principles will be applied here based on the following assumptions:

- In steady state over long periods the amount of nitrifiers leaving the RBC is equal to the amount produced in the RBC. The fraction of these nitrifiers to be returned to the activated sludge system is equal to the fraction of wastewater returned.
- The nitrifier yield per g of ammonium nitrified in the trickling filter equals that in the activated sludge system (Y_n, g COD/g N).
- The activity of nitrifiers introduced from a fixed film system to an activated sludge system is no different from the activity of nitrifiers grown in the activated sludge system.
- The growth rate of nitrifiers in the activated sludge system is the inverse of the minimum SRT for nitrification expressed as a temperature related curve (Figure 3.7).
- The ammonium concentration does not limit nitrification in the activated sludge system. (The effluent ammonium concentration is around 7 g N/m^3.)

Using the assumptions above, the effect of seeding of nitrifiers on nitrification in the activated sludge system can be estimated by combining a mass balance of nitrifiers over the activated sludge system with an expression of the yield of the nitrifiers grown in the activated sludge system (Figure 5.14).

The amount of ammonium oxidised by nitrification in the activated sludge system can be calculated with knowledge of the amount of ammonium nitrified in the flow recirculated from the RBC. However the assumptions mentioned above limit the range where the approach is valid. The nitrification in the activated sludge system must not be so large as to lower the ammonium concentration to levels where ammonium becomes rate-limiting in the activated sludge system. Also it can be seen that at aerated solids retention times approaching the minimum SRT for nitrification, the expected nitrification in the activated sludge system will approach infinity. In the following the aerated SRT:s are well below the minimum SRT for nitrification.

A comparison of the unexplained nitrogen loss in the activated sludge system and the nitrification in activated sludge estimated thus shows that, although the spread is wide, the nitrification needed to explain the nitrogen loss in question is in the same order of magnitude as the nitrification estimated according to Figure 5.14 (see Table 5.7). The average estimated nitrification was about 300 g N/d and the nitrification which would explain the unknown nitrogen sink was about 340 g N/d.

The mass balance of nitrifying biomass over the activated sludge system can be expressed as follows:

0 = from the RBC + growth in the activated sludge system - wastage with the excess activated sludge

$$0=\Delta F_{RBC}\cdot Y_n + X\cdot f_{OX}\cdot\frac{1}{SRT_{\min}} - X\cdot\frac{1}{SRT}\Rightarrow$$

$$0 = \Delta F_{RBC}\cdot Y_n + X\cdot\left(\frac{f_{OX}}{SRT_{\min}} - \frac{1}{SRT}\right) \qquad (1)$$

The relationship between the amount of ammonium used for nitrification and the production of nitrifiers in the activated sludge system can be expressed as follows:

$$\Delta F_{AS}\cdot Y_n = \frac{X\cdot f_{OX}}{SRT_{\min}} \quad\Rightarrow\quad X = \Delta F_{AS}\cdot Y_n\cdot\frac{SRT_{\min}}{f_{OX}} \qquad (2)$$

Inserting (2) in (1) yields:

$$0= \Delta F_{RBC}\cdot Y_n + \Delta F_{AS}\cdot Y_n\cdot\frac{SRT_{\min}}{f_{OX}}\cdot\left(\frac{f_{OX}}{SRT_{\min}} - \frac{1}{SRT}\right)\Rightarrow$$

$$0= \Delta F_{RBC} + \Delta F_{AS}\cdot\frac{SRT_{\min}}{f_{OX}}\cdot\left(\frac{f_{OX}}{SRT_{\min}} - \frac{1}{SRT}\right)\Rightarrow$$

$$\Delta F_{AS} = \frac{\Delta F_{RBC}}{\dfrac{SRT_{\min}}{SRT\cdot f_{OX}} - 1} \qquad (3)$$

Expression (3) can be used to estimate nitrification in the activated sludge system at solids retention times below SRT$_{\min}$.

Symbol		
F	Mass flow of ammonium	g N/d
ΔF_{RBC}	Ammonium nitrified in the RBC and then returned to the AS system	g N/d
ΔF_{AS}	Ammonium nitrified in the AS system	g N/d
f_{ox}	Fraction of the activated sludge which is in aerated tanks	1
SRT	Solids retention time	d
SRT_{min}	Minimum SRT for nitrification	d
X	Amount of nitrifiers in the AS tanks	g COD
Y_n	Yield of nitrifiers when oxidising ammonium	g COD/g N

Figure 5.14 Estimate of the influence of seeding of nitrifying biomass to a highly loaded activated sludge system.

Table 5.7 Nitrification and denitrification in aerated activated sludge tanks based on system nitrogen loss and on expected nitrification caused by seeding from the RBC.

Month 1993-1994	Estimate of nitrification in aerated activated sludge according to equations of Figure 5.14.									Nitrogen conversions in aerated activated sludge according to mass balances of Table 5.6.			
	A Temp	B SRT	C SRT_{min}	D MLSS	E VF	$F=D\cdot E\cdot V\cdot f_{ox}/100$ Aerated sludge	G ΔF_{RBC}	H ΔF_{AS}	I=H/(F·24) Nit. rate.	J Den	K Nit	L=J/(F·24) Den. rate	M=K/(F·24) Nit. rate
	°C	d	d	kgSS/m^3	%	kg VSS	gN/d	gN/d	gN/(kg VSS·h)	gN/d	gN/d	gN/(kg VSS·h)	gN/(kg VSS·h)
Jul.	18	3	4	2.2	56	9.7	840	473	2.0	598	586	2.6	2.5
Aug.	18	2.4	4	2.8	51	11.1	840	340	1.3	597	638	2.2	2.4
Sept.	18	2.5	4	2.4	51	9.5	1390	596	2.6	437	298	1.9	1.3
Oct.	16	2.2	5	2.3	59	10.5	1030	276	1.1	319	516	1.3	2.0
Nov.	14.5	2.4	6	2.4	59	11.0	1310	311	1.2	213	55	0.8	0.2
Dec.	11	1.9	9.5	2.6	60	12.1	600	64	0.2	62	146	0.2	0.5
Jan.	9	2.1	11.5	2.5	60	11.6	680	65	0.2	118	130	0.4	0.5
Ave.	15	2.4	6	2.5	57	10.8	956	303	1.2	335	338	1.3	1.3

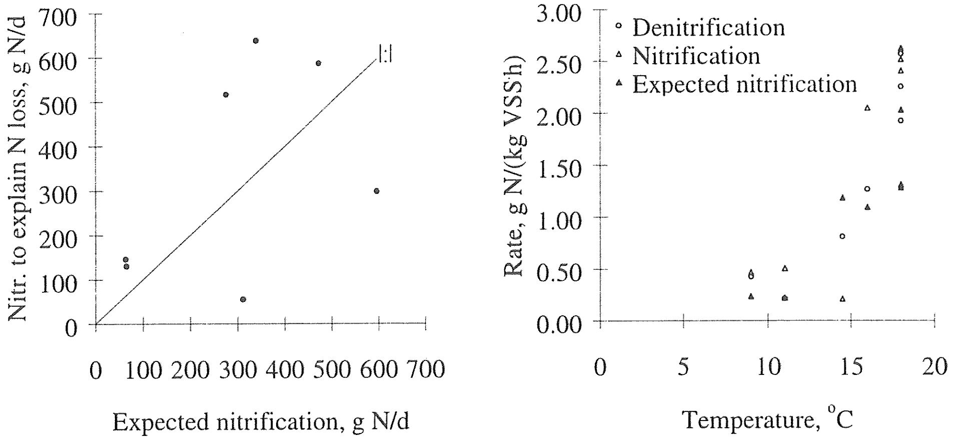
The activated sludge volume was 15.9 m^3 of which 48 % was aerated ($f_{ox} = 0.48$)

C: lower level of range indicated by Henze and Bundgaard, 1982. G: from Table 5.6. H: estimated according to Figure 5.14.

Although the prediction of the nitrification rate for individual months (Figure 5.15 a) is not impressive it is interesting to note that the nitrification rate variation with temperature is similar whether the nitrification rate is based on the nitrogen loss or on the expected nitrification due to seeding (Figure 5.15 b). The otherwise unexplained nitrogen loss in the system is temperature dependent in the same way as it would be if it were caused by nitrification. The calculated denitrification rates (0.2 to 2.6 g N/(kg VSS·h)) are of the same order of magnitude as the nitrification rates, which is to be expected if denitrification is limited by the supply of nitrate from nitrification.Some conclusions can be drawn from the mass balances of nitrogen during experimental period D (when about 52 % of the activated sludge was non-aerated).

- On average 58 % of the nitrogen in the primary settled wastewater was removed. Half the nitrogen removal can be attributed to denitrification in the non-aerated tanks, a quarter is removed with the excess sludge and the remaining quarter in the RBC and in the activated sludge system. The removal in the RBC may be due to denitrification in the bottom of the trough of the RBC.

- About 11 % of the nitrogen removal takes place in the activated sludge system but cannot be explained by denitrification of recirculated nitrate or withdrawal of sludge. This removal could be explained by nitrification and denitrification in the aerated tanks if the influence of the seeding of the activated sludge system by nitrifiers produced in the RBC is taken in account. The temperature dependency of the otherwise unexplained removal of nitrogen agrees with estimates of nitrification due to seeding of the activated sludge system.



a) Nitrification in aerated activated sludge tanks.

b) Nitrification and denitrification rates in aerated activated sludge tanks.

Figure 5.15 Comparison of nitrification needed to explain nitrogen loss in activated sludge system with nitrification expected due to seeding of activated sludge system with nitrifiers from RBC.

5.4 System effects on effluent quality

5.4.1 Introduction

The system chosen for nitrogen removal differs from the single-sludge pre-denitrification activated sludge system in several ways. One feature is the non-nitrifying activated sludge system, which can be operated at a low solids retention time, giving high denitrification rates. Another feature is nitrification in a fixed film system a low organic loading, where the low growth rate of the nitrifiers is of less importance than in a suspended mixed culture. In combining the two, however, oxygen from the trickling filter will be used by heterotrophic organisms in the activated sludge system in preference to nitrate, decreasing the denitrification capacity of the system. Since all water to the trickling filter has to pass final clarification, the hydraulic loading on the clarifiers will be increased accordingly. At wastewater flows close to the maximum capacity of the clarifiers very little, or no water at all, can be recirculated to the activated sludge plant. Most of the time, however, the flow is lower and other factors limit the removal of nitrogen.

Mainly due to factors connected with the recirculation of clarified effluent, the operational possibilities and the effects of these, including handling of sludge liquor, differ from those of a single sludge pre-denitrification system.

In section 5.4.2, system effects on the process caused by wastewater quality and quantity are analysed.

In section 5.4.3, the ways in which design parameters connected with the nitrifying trickling filter influence the average effluent ammonium concentration are analysed and discussed.

In section 5.4.5, the operational strategy, with special reference to handling of sludge liquors, is analysed and discussed.

5.4.2 Influence of wastewater quality variation with flow

The impact of flow-related parameters on wastewater treatment is made up of several different components. The composition of the wastewater varies depending on the mix of storm-water, groundwater, and domestic and industrial wastewater. For some parameters this may be seen as a simple dilution of the domestic wastewater with storm-water and infiltration/inflow. For other parameters this is not the case. If physical, biological and chemical processes in the catchment system are flow-related the variation of influent wastewater quality becomes more complex. Additionally, bottlenecks at the wastewater treatment plant may necessitate overflow or other flow-related variations in the operation of the wastewater treatment plant.

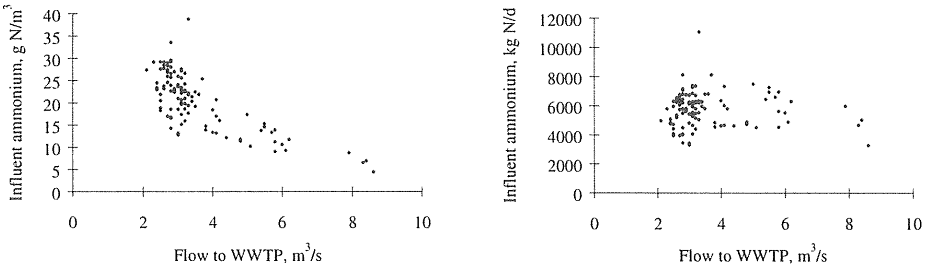
In the case of post nitrification in a trickling filter and recirculation of nitrified effluent to the activated sludge system for denitrification, the recirculated flow can be varied, allowing for extensive adjustment of the operation in relation to influent wastewater quantity and quality. However for the pilot plant operation a simple operational strategy was chosen. During experimental period C the flow through the secondary settlers was set at a fixed value. The amount of water recirculated to the activated sludge system was set as the maximum flow through the sedimentation tanks minus the influent flow.

In the following, the combined effect of varying influent wastewater quality and wastewater treatment plant operation on effluent results will be discussed with data from experimental periods C and D as a reference. During these experimental periods the operation of the pilot plant closely followed that of a full-scale plant.

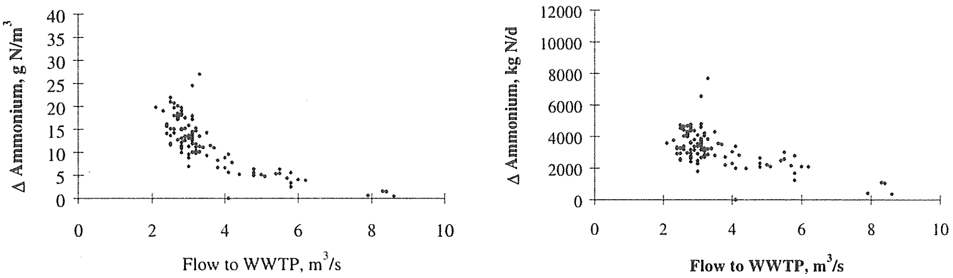
One important parameter where the influent wastewater mass flow is close to constant is ammonium (Figure 5.16). About 6 000 kg N/d of ammonium nitrogen reached the activated sludge system roughly independently of the flow. However, in this process the treatment of the wastewater depends on the flow. At high flows the capacity of the settlers sets a limit to how much water can be recirculated to the NTF, causing recirculation to decrease with increasing influent flow. The mass flow of ammonium returned to the trickling filter and nitrified will thus decrease with increased flow to the WWTP , the result being that the effluent concentration is close to constant despite influent variations. The effluent ammonium mass flow, however, will increase with increased flow.

This could influence the receiving water in different ways depending on whether the total nitrogen loading or the toxicity of ammonium is the important factor. If the important factor is the toxic effect of ammonium in the receiving water, the strategy of removing more ammonium at low flows would seem advantageous. Low flows at the WWTP would often coincide with periods of dry weather when the dilution of effluent ammonium would be low.

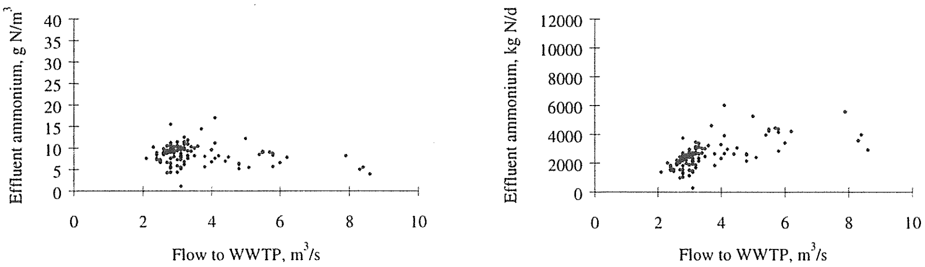
For soluble organics (analysed as COD of a filtered sample) the connection between concentration and flow is not so simple. Influent COD decreases with increased flow (Figure 5.17), but the influent mass flow appears to increase slightly with flow. Being transported to the wastewater treatment plant the wastewater organics may be subjected to a number of processes. In the sewers particles may settle and later be resuspended causing delayed transport to the treatment plant. Organic matter is consumed in aerobic sewers by biofilm processes and by suspended growth. Özer and Kazimer (1995) modelled removal in long aerobic sewer lines based on respiration in the suspension and on the biofilm on the surface of the pipes. In Göteborg, wastewater may typically be transported for about 1.5 to 2 hours in sewers (typically 1 km; ϕ 225 mm + 1 km; ϕ 400 + 3 km; ϕ 800, Bernt Persson personal communication) before reaching the tunnel system. This would, according to the model of Özer and Kazimer, reduce COD by about 60 mg O_2/l. Several factors may influence the results. The water velocity was 0.5 m/s in Özer and Kazimer´s experimental studies and modelling; in Göteborg a typical value is stated as 1 m/s. The oxygen transfer in an experimental gravity sewer was studied by Balmér and Tagizadeh-Nasser (1995) and was postulated to depend on energy dissipation and the mean hydraulic depth in the sewer. Many variables may affect the energy dissipation. Such variables may be: slope, relative depth , construction material, distance between manholes and connections, fraction of the system as combined or separate systems etc. Jensen (1995) compared several empirical models for air-to-water oxygen transfer in gravity sewers and suggested an improved equation based on water velocity, hydraulic mean depth and slope. Gall et al. (1995) modelled the effect of a reactive sewer on a wastewater treatment plant, based on oxygen consumption in the sewer, indicating a potentially large impact on oxygen consumption in the activated sludge system and on sludge production.



a) Influent ammonium concentration.
b) Influent ammonium mass flow.

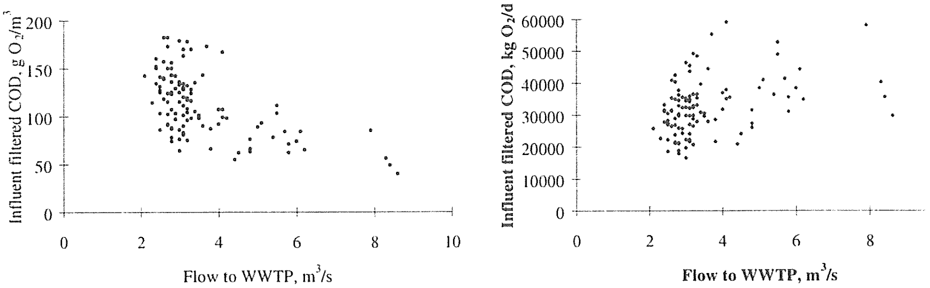


c) Removed ammonium concentration.
d) Removed ammonium mass flow.

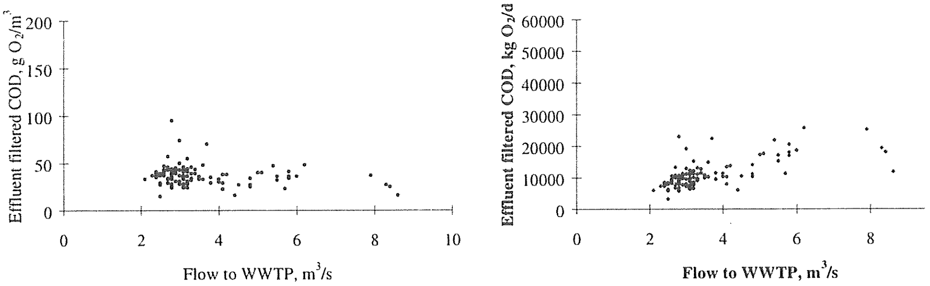


e) Effluent ammonium concentration.
f) Effluent ammonium mass flow.

Figure 5.16 Influent and effluent ammonium concentration of the pilot activated sludge system and influent and effluent ammonium mass flow. The mass flow is plotted using the flow to the main WWTP.



a) COD of influent filtered sample. b) Influent COD mass flow.



c) COD of effluent filtered sample. d) Effluent COD mass flow.

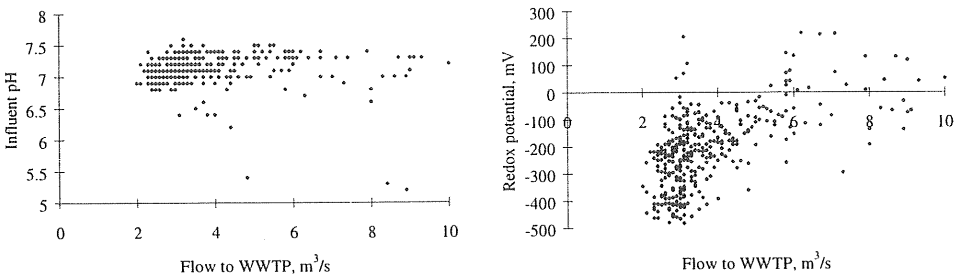
Figure 5.17 Influent and effluent COD (filtered samples) of the pilot activated sludge system and influent and effluent COD mass flow. The mass flow is plotted using the flow to the main WWTP.

An example of storage and reaction in the collection system is the performance of gully pots in dry weather and during rain. Water is stored in the gully pot from the end of one storm event and flushed out at the beginning of the next. Depending on storage time, temperature, initial concentration and composition of organic matter and suspended solids, oxygen concentration etc., different physical, chemical and biological reactions will have changed the composition by the time of the next storm (Butler et al., 1995). Morrison et al. (1995) demonstrated this cycle for a gully pot in Göteborg. This gully pot was subjected to acid storm water as the runoff from the surrounding road surface reached it. However the buffering capacity of the gully pot contents which had been built up between storm events would delay the pH drop of the effluent water while the contents of the gully pot were being exchanged. During the storm event more DOC left the gully pot than entered it.

With these mechanisms in mind it would be expected that the varying conditions at different flows would produce a wastewater where not only the content, but also the quality, of organic matter varied. Some parameters give such indications. The influent wastewater, normally containing very little nitrate, contained more nitrate at high flows than at low flows (Figure 5.20 a and b). At the main plant the oxidation reduction potential of the influent wastewater was higher at high flows than at low flows (Figure 5.18 b). This indicates that the influent wastewater is more aerobic at high flows than at low flows. Similar observations were made at the wastewater treatment plant in

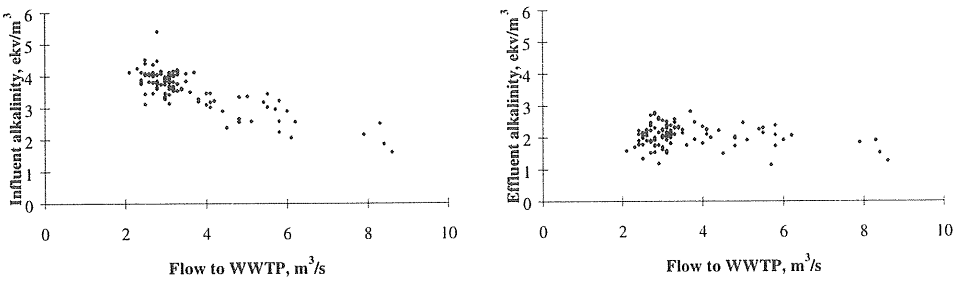
Helsingborg (in the south of Sweden) where higher oxidation reduction potentials and oxygen concentrations were observed in the influent to the anaerobic zone during high flow conditions (Christensson et al. 1995). In Malmö the ratios between volatile fatty acids and phosphorus and between COD and phosphorus decreased with decreasing COD (often during storm events) as observed by Carlsson et al. (1996). Aerobic biological processes may dominate the collection system at high flows whereas anaerobic processes may be more important at low flows.

The influent wastewater pH was quite stable between 7 and 7.5 at medium to high flows indicating that the wastewater was too strongly buffered to be influenced by the acid, but weakly buffered, storm-water. (Figure 5.18 a). The alkalinity of the influent wastewater, however, decreased with increasing flow to the main plant (Figure 5.19 a). In the process alkalinity was consumed in relation to the amount of ammonium nitrified and, in this system, more ammonium was recirculated to the nitrifying unit at low flows than at high flows. This caused the alkalinity to drop more at low flows than at high flows, and the effluent alkalinity was close to constant (Figure 5.19 b). The alkalinity produced through denitrification decreased the alkalinity drop somewhat.



a) Influent pH. b) Influent oxidation reduction potential.

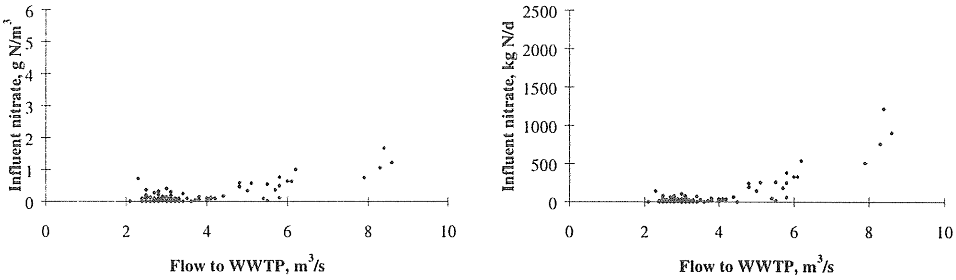
Figure 5.18 Variation of pH and oxidation reduction potential of the influent wastewater to the main WWTP (Feb. 93- Feb. 94).



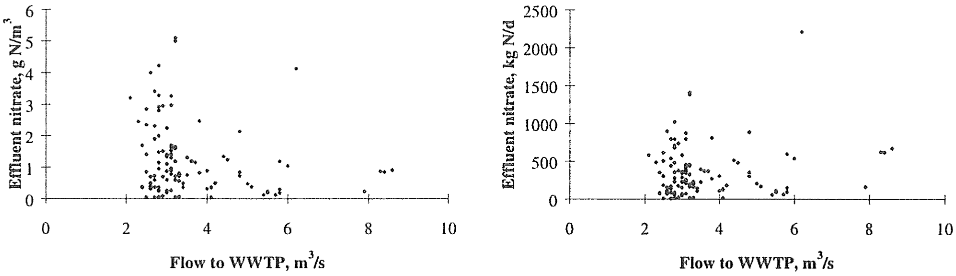
a) Influent alkalinity. b) Effluent alkalinity.

Figure 5.19 Influent and effluent alkalinity of the pilot activated sludge system during experimental periods C and D.

The effluent nitrate concentration was influenced by several factors. The increased nitrification due to higher recirculation at low flows caused the nitrate mass flow to the activated sludge system to be high at low flows. The influent COD mass flow, however, did not increase at low flows. The combined effect of increasing nitrate loading and constant or decreasing loading of organic material at low flows, was low carbon to nitrate ratios at low flows. As denitrification often was incomplete at these low flows the nitrate recirculation can be assumed to have exceeded the denitrification capacity.



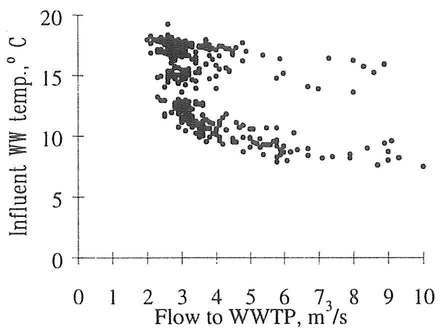
a) Influent nitrate concentration. b) Influent nitrate mass flow.



c) Effluent nitrate concentration. d) Effluent nitrate mass flow.

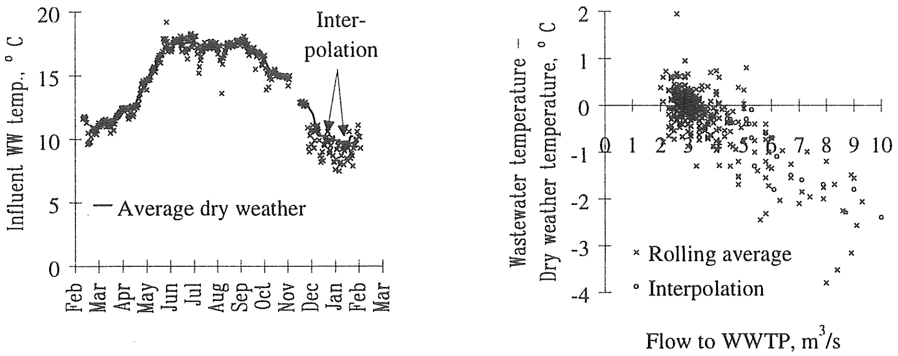
Figure 5.20 Influent and effluent nitrate of the pilot activated sludge system and influent and effluent nitrate mass flow. The mass flow is calculated using the flow to the main WWTP.

Another important parameter in biological wastewater treatment is water temperature (Figure 5.21 a). The temperature of the influent wastewater depends on the mix of water with different temperatures, e.g. domestic wastewater, industrial wastewater, groundwater and surface run-off, and on the cooling effect of the sewer system. In an attempt to relate temperature to flow the following simplifications will be made. The dry weather water temperature is assumed to vary seasonally. Figure 5.21 b shows the daily average influent wastewater temperature as it varied during the year of experimental periods C and D. In order to obtain a seasonally varying, dry weather wastewater temperature a 21 day rolling average was calculated (of those of the 21 days when the flow to the WWTP was below 3.9 m^3/s). During January 1994 the time between days when the flow was below 3.9 m^3/s was greater than 21 days. For this period an interpolation of temperatures before and after the high-flow period was used. These values are marked "interpolation" in Figure 5.21 c. At high flows (above 3.9 m^3/s) the temperature drops below the calculated dry weather temperature. At 6 m^3/s the influent wastewater temperature was 0.5 to 2.0 °C lower than the dry weather temperature and at 8 m^3/s the wastewater temperature was 1 to 3 °C lower.



Definition: Here "dry weather" was defined as occasions when the wastewater flow to the Rya WWTP was below 3.9 m^3/s. Flows above this level were significantly influenced by infiltration/inflow or storm-water. This does not mean that infiltration/inflow or storm-water did not reach the WWTP at lower flows.

a) Variation of wastewater temperature with flow



b) Temperature of influent wastewater to the main plant during experimental periods C and D.

c) Deviation of wastewater temperature from average dry weather temperature

Figure 5.21 Temperature of influent wastewater to main WWTP during experimental periods C and D.

These correlations between flow and wastewater quality and operation influence design and operation of a wastewater treatment plant for nitrogen removal.

- At low influent flows (2 - 3 m^3/s) the influent ammonium concentration was high and the remaining capacity of the final sedimentation tanks high allowing for high recirculation flow, high nitrate recirculation and thus potentially a high degree of nitrogen removal. In this situation the denitrification capacity often limited nitrogen removal. Using the full capacity of the nitrification unit and the sedimentation tanks could not improve nitrogen removal. In this situation methods for improving the denitrification capacity may be useful.

- At intermediate flows (3 - 4 m^3/s) the recirculated nitrate was almost completely removed.

- At high flows (above about 5 - 6 m^3/s) nitrogen removal was limited by the size of the final sedimentation tanks as only a limited fraction of the effluent water could be recirculated, nitrified and denitrified. At high flows, however, the influent ammonium concentration was low, so the recirculated flow would have to be extremely high in order to return enough ammonium to the trickling filter, and thus nitrate to the activated sludge system, in order to obtain substantial denitrification. Other factors may also have influenced the denitrification negatively at high flows. The temperature drop can have influenced the denitrification process negatively and the more oxidised state of the influent wastewater at high flows may have influenced the carbon source negatively. Under these conditions so little nitrogen can be removed that priority should be given to processes other than nitrogen removal, such as sedimentation and minimising overflow.

Attempting to correlate biological activities in an operating WWTP to parameters such as loading, amount of available organic matter, temperature etc. is complicated by the fact that several important parameters are correlated to each other and to flow (see Table 5.8).

Table 5.8 Summary of the correlation of some parameters to flow in a system with post nitrification in a trickling filter and recirculation of nitrified effluent to a non-nitrifying activated sludge system for denitrification.

	Influent	Effluent
Ammonium	Mass flow constant	Concentration constant
Nitrate	Concentration increases with flow	Concentration process dependent
COD, filtered sample	Mass flow constant or slightly increasing with flow	Concentration constant
pH	Stable	Stable
Alkalinity	Decreasing with increased flow	Constant
Oxidation reduction potential	-100 mV - + 200 mV at high flows -500 mV - ± 0 mV at low flows	Not measured
Temperature	1 - 3 °C lower at high flows than at low flows	

5.4.3 Influence of design on effluent ammonium concentration

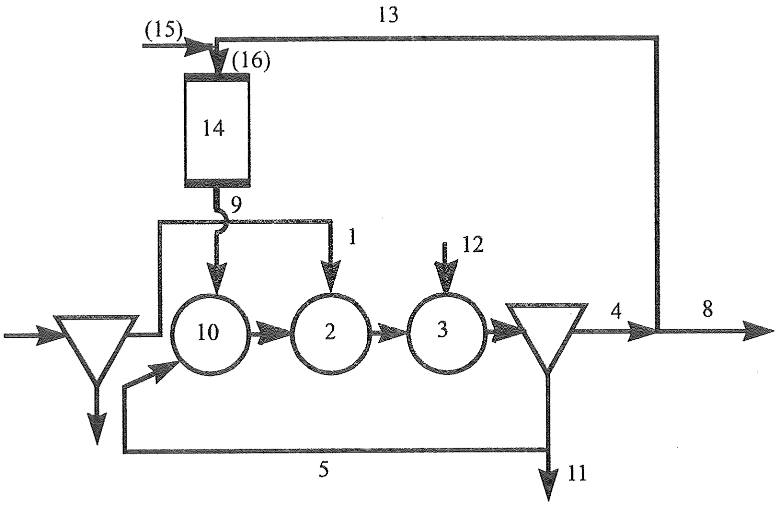
The effluent ammonium concentration from a system where nitrification takes place in a post nitrification unit and denitrification in a highly loaded activated sludge system depends on design in different ways. The nitrification of the system is mainly controlled by two factors: the supply of ammonium to the trickling filter controls the amount of ammonium available for nitrification and the nitrifying capacity of the trickling filter determines the fraction of the ammonium supplied to the trickling filter which is nitrified. Recirculation plays an important role in determining the amount of ammonium available for nitrification. At high flows recirculation is limited by the sedimentation capacity of the final clarifiers. At low flows the productive recirculation may be limited by the nitrification capacity of the trickling filters or by the denitrifying capacity of the influent wastewater. Important factors, such as influent ammonium concentration, flow and the recirculation capacity of the plant are inter-dependent. This complicates determination of the nitrogen removal capacity of the plant when treating wastewater, the quality and quantity of which varies with time. Recirculation of effluent wastewater is one of the main control handles of the process. The recirculation flow influences the process in several different ways:

- Supply of **nitrate for denitrification**. Increased recirculation supplies more nitrate to the denitrification process, within the nitrification capacity of the trickling filter.
- **Hydraulic loading** on the final **sedimentation tanks**. Increased recirculation increases the hydraulic load on the sedimentation tanks.
- **Oxygen recirculation**. Increased recirculation causes increased oxygen recirculation to the non-aerated activated sludge compartments.
- Effluent **ammonium** concentration **from trickling filters**. If the ammonium loading exceeds the nitrification capacity the effluent ammonium concentration from the trickling filter increases.
- Effluent **ammonium** concentration **from plant**. At complete nitrification in the trickling filter, the effluent ammonium concentration is a function of the fraction of the ammonium mass flow recirculated to the nitrifying trickling filter.

In a single-sludge system the design of physical structures and recirculated flow can be performed separately. In this system however, the physical structures limit the recirculation flow, which controls the treatment result. In the following, the effect of the recirculation flow and the nitrification capacity of the nitrifying trickling filter on the effluent ammonium concentration will be discussed. The average effluent ammonium concentration using different design limits will be calculated using a simple model based on the following assumptions:

- In any given situation nitrification is limited either by the supply of ammonium through recirculation or by the nitrification capacity of the trickling filters.
- No nitrification takes place in the activated sludge tanks.
- The mass flow of ammonium available to a post nitrification unit can be quantified by the effluent ammonium mass flow from a non-nitrifying activated sludge plant.
- The effluent ammonium concentration from the trickling filter when the nitrification capacity is not limiting can be neglected.

Under these conditions the effluent ammonium concentration is determined either by the recirculation rate or by the nitrification capacity of the trickling filter (Figure 5.23). The recirculation flow (Q_{13}) in this system is limited by different factors. At high flows the hydraulic loading on the final sedimentation tanks will limit recirculation. The hydraulic loading must not cause high effluent suspended solids concentrations. When designing the trickling filter, pumps and pipes supplying the trickling filter with water will be given a maximum capacity. At low influent flows it is not advisable to use the full pumping capacity since the amount of nitrate and oxygen going to the activated sludge plant would exceed the amount that could be reduced using the organics of the influent wastewater. The nitrate and oxygen loading on the activated sludge system would increase without improving the total nitrogen removal. This limit set on recirculation by the denitrification capacity, which from pilot plant data can be estimated as 200 % of the influent flow, can be exceeded if the carbon source for denitrification is improved. These constraints on the recirculated flow can all be expressed as a function of the flow to the WWTP (Figure 5.24).



1 Primary settled wastewater
2 Anoxic zone
3 Aerated zone
4 Effluent from secondary settler
5 Return activated sludge
8 Effluent
9 Recycled wastewater
10 Deoxygenation zone

11 Waste activated sludge
12 Iron sulphate solution
13 To NTF
14 NTF
(15 Extra NH_4^+ contribution to NTF)
(16 total to NTF)

Figure 5.22 Denotation of flows and processes (15 and 16 are used in section 5.4.5).

When recirculation is the limiting factor the effluent ammonium concentration is determined by the relation between the recirculated flow and the influent flow:

$$S_{8,recirculation} = S_4 = \frac{F_4}{Q_4} = \frac{F_4}{Q_1 + Q_{13}}$$

When the nitrification capacity of the NTF limits nitrification the effluent ammonium concentration can be calculated as follows:

$$S_{8,nitrification} = \frac{F_4 - \Delta F_{\max}}{Q_1}$$

Since either the nitrification capacity of the trickling filter or the recirculation will limit the total nitrification the effluent ammonium concentration will be equal to the higher of the two concentrations calculated above:

$$S_8 = \max(S_{8,nitrification}, S_{8,recirculation}) = \max(\frac{F_4}{Q_1 + Q_{13}}, \frac{F_4 - \Delta F_{\max}}{Q_1})$$

Symbol	Explanation	Unit
Q_1	Influent flow	m^3/s
Q_4	Flow from final clarifiers	m^3/s
Q_{13}	Flow to trickling filter	m^3/s
$S_8=S_{13}=S_4$	Effluent ammonium concentration	$g\ N/m^3$
F_4	Mass flow of NH_4^+ from a non-nitrifying activated sludge system	$g\ N/s$
ΔF_{max}	Nitrification capacity of trickling filter	$g\ N/s$
indexes	refer to position in Figure 5.22	

Figure 5.23 Estimate of effluent ammonium concentration.

The hydraulic loading on the final sedimentation tanks limits recirculation at high influent flows:

$$Q_{13} \le Q_{4,\max} - Q_1$$

The pumping capacity to the NTF is limited:

$$Q_{13} \le Q_{13,\max}$$

The limitation on recirculation by the denitrification capacity can be expressed as a multiple of influent flow:

$$Q_{13} \le a \cdot Q_1$$

Combining these limitations the recirculation rate on each occasion will be:

$$Q_{13} = \min(Q_{4,\max} - Q_1, Q_{13,\max}, a \cdot Q_1)$$

Symbol	Explanation
Q_1	Influent flow
$Q_{4,max}$	Maximum flow from final clarifiers
Q_{13}	Flow to trickling filter
$Q_{13,max}$	Maximum flow to trickling filter
a	Recirculation ratio limited by the capacity of denitrification
Indexes	refer to positions in Figure 5.22

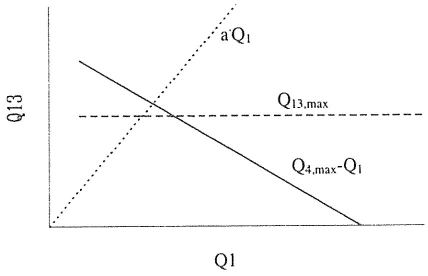


Figure 5.24 Limits to recirculation.

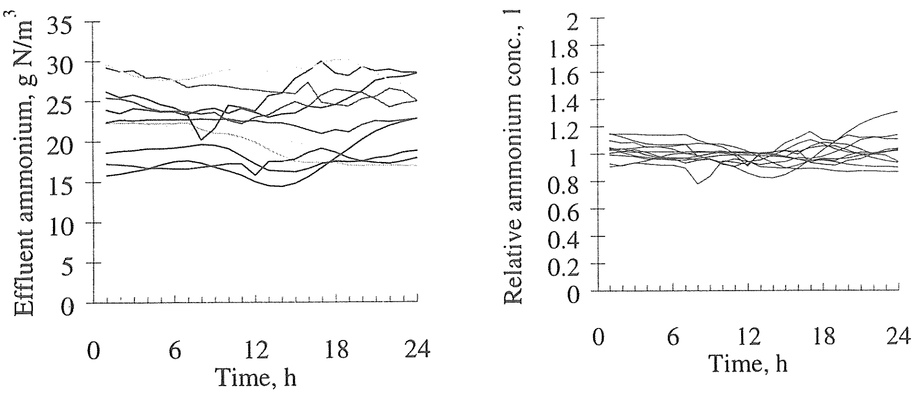
Input data needed for the model above are a set of corresponding values of influent flow to (Q_I) and mass flow of ammonium from a non-nitrifying activated sludge system (F_4). Here data from the Rya WWTP will be used. The recirculated flow is a function of influent flow, where the function is determined by the limits of the system (Figure 5.24). The aim is to investigate how the average effluent ammonium concentration responds to design constraints. Input data should be representative of the different situations to which the treatment plant is subjected. However each calculation is made on a steady state basis. Thus input data are sought where many data points representing different conditions are available and where the sampling time of each data point is short enough to represent close to constant conditions at the plant. A compromise has to be made between several aspects. First, data must be available and must be representative of the system studied. Secondly, the time span of sampling or measuring must be long enough to equalise stochastic fluctuations of input data and short enough to avoid equalisation of effects which may systematically influence the output data.

The catchment area of the Rya WWTP is large, causing the transport time from the wastewater producers to the WWTP to vary considerably. Diurnal variations of wastewater quality close to the wastewater producers are to a large degree equalised at the treatment plant. The diurnal variation of the effluent ammonium concentration from the present, non-nitrifying, activated sludge plant is small (Figure 5.25). The flow pumped to the wastewater treatment plant may vary during a 24-hour period due to the large volumes in the tunnel system available for equalisation of flow. However the pumped flow during 24 hours will, in most cases, correspond to the wastewater production (including storm-water etc.). The time-scale of treatment of wastewater in the future plant will be in the order of 12 to 24 hours including recirculation. The time-scales used for sampling and processing data at the WWTP basically range from 1 minute to 1 week. Here 24 hours suits the time-scale of variation of ammonium concentration, flow and operation of the plant. Data chosen for these simulations are:

- The ammonium concentration of 153 grab samples of effluent wastewater from the present activated sludge plant. These were analysed as being samples of influent to the pilot nitrifying trickling filter during studies performed from August 1991 to October 1993. Since the sample was taken after the activated sludge plant, it takes into account any transformations between organic nitrogen and ammonium which take place in the activated sludge system.

- The average flow of wastewater pumped to the wastewater treatment plant during the 24-hour period (calendar day) of the above-mentioned grab sample.

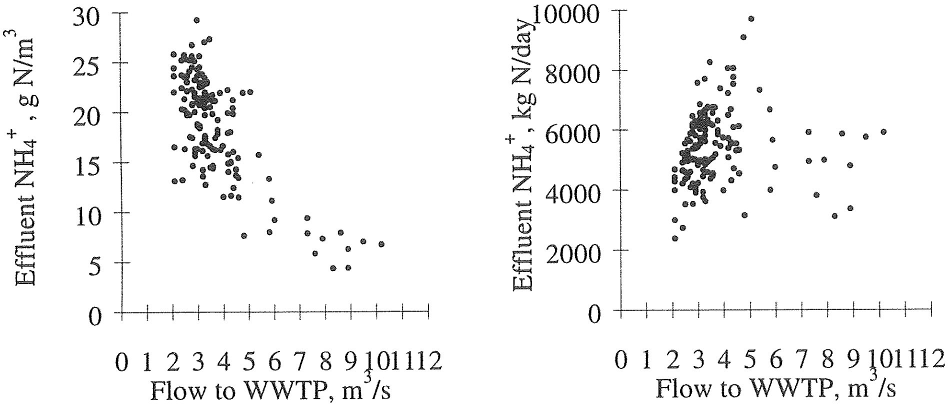
The data are illustrated in Figure 5.26 (see also appendix D). About 74% of the ammonium mass flow available for nitrification reaches the plant at flows between 2 and 4 m^3/s, covering 75 % of the recorded occasions (Table 5.9). One may conclude that a process for nitrogen removal should perform well at flows below 4 m^3/s, when a large amount of nitrogen is available for treatment. Although 20 % of the wastewater reaches the plant at these flows above 5 m^3/s, improvements at these flows only influence 10 % of the ammonium mass flow.



a) Concentration.

b) Concentration relative to 24-hour average.

Figure 5.25 Diurnal variation of effluent ammonium concentration from the Rya WWTP during normal flow conditions measured by a continuous ammonium analyser.



a) Ammonium concentration

b) Ammonium mass flow.

Figure 5.26 Effluent ammonium from the Rya WWTP.

Table 5.9 Percentage of wastewater pumped to the Rya WWTP at or below flow, and effluent ammonium mass flow at or below flow.

Flow to WWTP (m^3/s)	≤ 2	≤ 3	≤ 4	≤ 5	≤ 6	≤ 7	≤ 8	≤ 9	≤ 10	≤ 11
Occasions	0	34	75	**90**	93	93	96	99	99	100
Water	0	24	62	79	85	85	**90**	97	98	100
Ammonium nitrogen	0	31	74	**90**	94	94	97	99	99	100

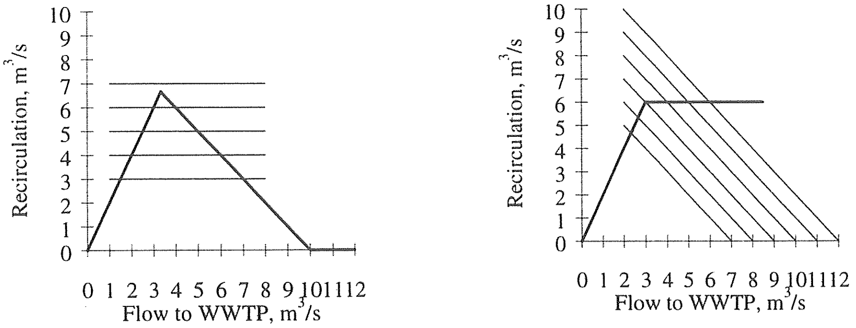
If the effluent ammonium concentration is calculated for each set of input data above, using the equations of Figure 5.23 and varying the limitations on recirculation (Figure 5.24), the influence of the design limits on the average effluent ammonium concentration can be estimated. If the target effluent standard is set at 10 g N/m^3 , and 1 g N/m^3 in the final effluent is nitrate and 1 g N/m^3 is organic nitrogen, then at the most 8 g N/m^3 as ammonium can be accepted. Four limitations will be analysed:

- The **capacity of the final sedimentation tanks**. At present the capacity of the sedimentation tanks is approximately 6 m^3/s. In the future plant new sedimentation tanks will be stacked on top of the present ones, increasing the capacity to approximately 10 m^3/s.
- The **pumping capacity** of wastewater from the final clarifiers to the nitrifying trickling filters.
- The **nitrification capacity of the trickling filters**. If the nitrifying trickling filters are 7.2 m high on the available ground area, the volume will be approximately 15 500 m^3. Using a cross-flow material with a specific surface of 230 m^2/m^3 and assuming a specific nitrification rate of 1 - 1.2 g N/(m$^2 \cdot$d), a total nitrification capacity of 3 600 - 4 300 kg N/d can be expected.
- **Limitations on denitrification due to insufficient carbon source.** Pilot plant operation indicates that at most 60 to 70 % removal of soluble nitrogen can be expected. If this amount is removed through recirculation of nitrified effluent to the activated sludge system it corresponds to a maximum recirculation about twice the flow to the WWTP.

In the following, one or two of these limitations will be analysed at a time. For each combination of limitations the average effluent ammonium concentration will be estimated as the average of the 153 effluent concentrations calculated using the model for each set of data.

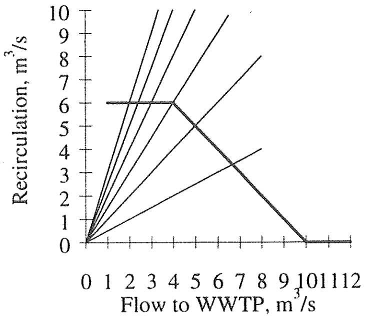
First the influence of the nitrification rate and the pumping capacity to the trickling filters is analysed. The maximum capacity of the final sedimentation tanks is set to 10 m^3/s and the denitrification capacity will not allow for a greater recirculation than twice the influent flow. The average effluent ammonium concentration is calculated for nitrification capacities ranging from 0 to 5000 kg N/d and pumping capacities to the trickling filters from 0 to 7 m^3/s (Figure 5.27 a). At a nitrification capacity of 4000 kg N/d and a pumping capacity of 6 m^3/s an average effluent ammonium concentration of 7-8 g N/m^3 can be expected (Figure 5.28). At lower nitrification capacities the effluent ammonium concentration increases rapidly and decreasing the pumping capacity to 4 m^3/s will increase effluent ammonium concentrations to about 8.5 g N/m^3. Increasing the pumping capacity to 7 m^3/s, on the other hand, will not improve the results significantly if the nitrification capacity is not at the same time increased to 5000 kg N/d. It may be concluded that a pumping capacity to the NTF of 6 m^3/s and a nitrification capacity of 4000 kg N/d gives an effluent ammonium concentration of 7-8 g N/m^3 and that increasing the pumping capacity will not significantly improve results. A small improvement can be obtained if the nitrification capacity is increased or if both capacities are increased.

Next the effect of the capacity of the final sedimentation tanks and the nitrification capacity are analysed keeping the pumping capacity to the nitrifying trickling filter constant at 6 m^3/s (Figure 5.27 b). At a nitrification capacity of 4000 kg N/d, the process is not very sensitive to a decrease of sedimentation capacity from 10 to 9 m^3/s, and improved sedimentation to 12 m^3/s does not decrease the effluent ammonium concentration (Figure 5.29). In order to benefit from a sedimentation capacity increased to 11-12 m^3/s the nitrification capacity must be at least 4 500 to 5 000 kg N/d. These improvements influence the system at high flows and as discussed earlier high flows only account for a small part of the ammonium mass flow (Table 5.9). Decreasing the capacity of the sedimentation tanks to the present 6 m^3/s, would however increase the effluent ammonium concentrations considerably, in this case to about 10.5 g N/m^3.



a) Variation of the capacity of pumping to the trickling filter.

b) Variation of the capacity of the final sedimentation tanks.



When not varied the capacity of the final sedimentation tanks is set at 10 m^3/s, the pumping capacity to the trickling filters at 6 m^3/s and denitrification limits recirculation to twice the influent flow.

c) Variation of the capacity of denitrification.

Figure 5.27 Recirculation strategies based on varied pumping capacity to the trickling filter, sedimentation capacity and capacity of denitrification .

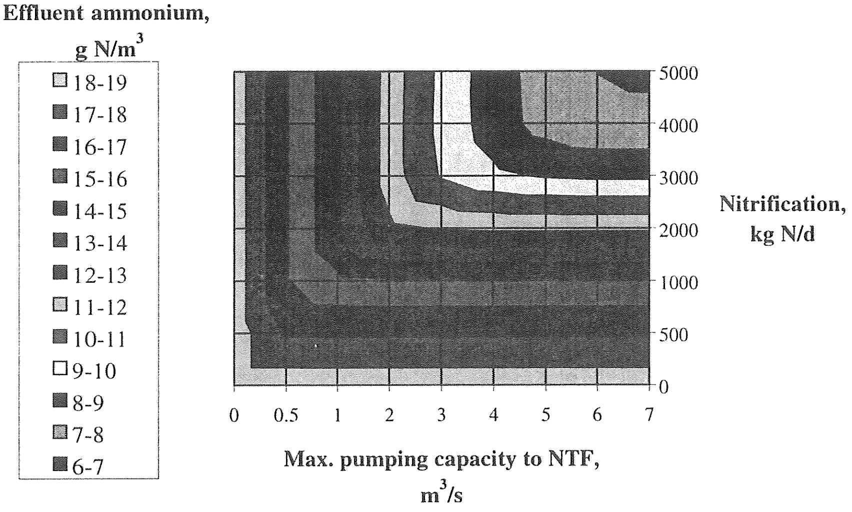


Figure 5.28 Calculated average effluent ammonium concentration. (Capacity of clarifiers = 10 m³/s, recirculation at low flows = twice the influent flow.)

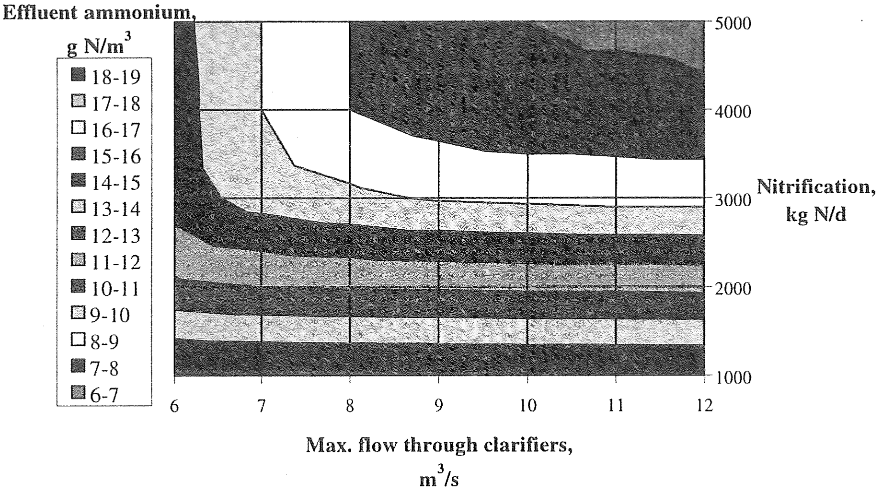


Figure 5.29 Calculated average effluent ammonium concentration. (Pumping capacity to trickling filter =6 m³/s, recirculation at low flows = twice the influent flow).

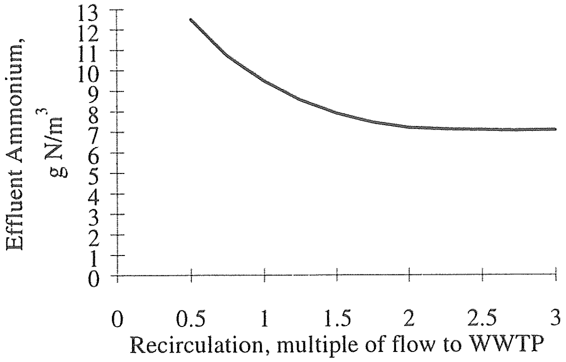


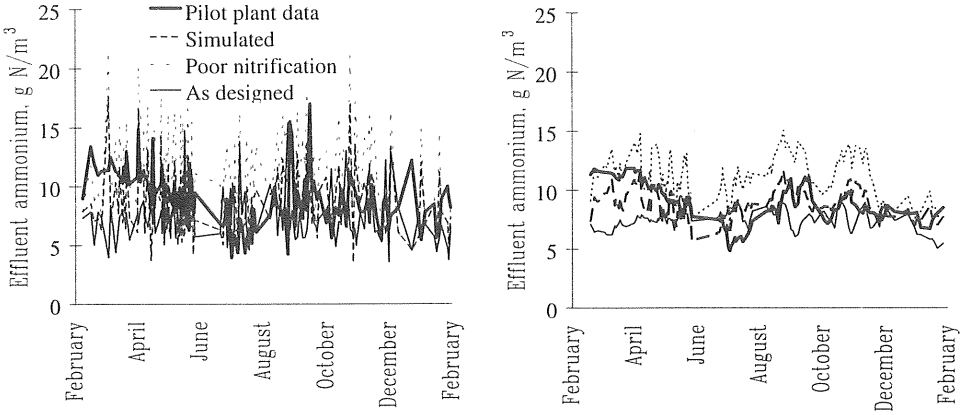
Figure 5.30 Calculated average effluent ammonium concentration.

At low flows, the useful recirculation is limited by denitrification. So far we have assumed that about double the influent flow can be recirculated and that the nitrate in this flow is denitrified in the activated sludge system. Now the effect of different maximum recirculation rates in terms of multiples of the influent flow will be studied. This may have to involve the addition of an external carbon source in order to remove all the recirculated nitrate, the details of which will not be discussed here. The capacity of the final sedimentation tanks is fixed at 10 m^3/s, the pumping capacity at 6 m^3/s and the nitrification capacity at 4 000 kg N/d; and the impact of the denitrification capacity in terms of multiples of influent flow which may be recirculated is varied (Figure 5.27 c). Increasing the recirculation ratio from twice the influent flow does not improve the results significantly (Figure 5.30). If the recirculation ratio on the other hand has to be decreased the effluent ammonium concentration will increase. If the maximum recirculation is set at 100 % of influent flow or less, the average effluent ammonium concentration will exceed 9 g N/m^3. If this is the case the system is not likely to meet a total nitrogen effluent standard of 10 g N/m^3 (considering the contributions of organic nitrogen and nitrate nitrogen). It is not surprising that these changes have a large impact since they affect the system at low flows, where a large share of the ammonium mass flow is to be found.

Sufficiently low effluent ammonium concentration (7.2 g N/m^3) is obtained with the following combination of capacities:

- Nitrification capacity: 4000 kg N/d
- Capacity of sedimentation tanks: 10 m^3/s
- Capacity of pumps to NTF: 6 m^3/s
- Recirculation limit due to denitrification: twice the influent flow

Ideally the results of simulation are compared with experimental data for verification. Unfortunately there are some practical difficulties. Firstly the data needed, corresponding flows and effluent ammonium concentration from a non-nitrifying activated sludge system, do not exist for the period of pilot plant operation. In the pilot plant, where nitrification took place in the RBC, the effluent ammonium concentration was already influenced by nitrification and the effluent from the main plant was not analysed for ammonium on a daily basis (only on weekly samples). Secondly, effluent ammonium concentrations from pilot plant operation, where the aim was low and stable effluent ammonium concentrations, do not vary greatly. Consequently any correlation between pilot plant data and simulated results is likely to be drowned in noise. However in the following the effluent ammonium concentration under the pilot plant conditions will be estimated and compared with analysed concentrations. The input ammonium mass flow distribution of previous estimates will be combined with relevant flow data. Conditions of estimates are detailed in Table 5.10 and results illustrated graphically in Figure 5.31.



a) Data

b) Smoothed data; 5 point rolling average.

Figure 5.31 Analysed and estimated effluent ammonium concentration from pilot plant.

Correlation between data and estimated values for individual data points is not strong. This is not surprising since input data are combined from two different sets of data. However it is interesting to note a general agreement between pilot plant data and simulated values of Figure 5.31 b and also that the average effluent ammonium concentrations of the period agree tolerably (8.9 and 8.7 g N/m^3, Table 5.10). During two periods data and estimated values disagree:

- During the first few months (especially February and March) nitrification in the RBC was poor. Effluent ammonium concentrations were above the expected, they agree better with estimates based on a poor nitrification case.
- During July, which is the industrial holiday month in Sweden, effluent ammonium concentrations are overestimated. This is likely to be caused by an overestimate of the available ammonium mass flow. The distribution was used without considering such seasonal changes.

Excluding these two periods the estimated average effluent ammonium concentration agrees well with the pilot plant data. Using the design arrived at earlier ("As designed" Table 5.10) an average effluent ammonium concentration of 7.6 g N/m^3 can be predicted.

Some conclusions can be drawn:

- The combined effects of operation, structural design and wastewater quality variation are important when predicting the treatment results of a system where nitrogen removal depends on recirculation of clarified secondary effluent.
- A simple steady state model using a distribution of input data can be used to predict the average effluent ammonium concentration from such a system.
- The model can be used to demonstrate effects of limited sedimentation capacity, nitrification capacity (of NTF), pumping capacity to NTF and denitrifying capacity of influent wastewater on effluent ammonium concentration. An example design was demonstrated, where a combination of the above limitations was shown to meet the effluent ammonium concentration limit set.
- The model can satisfactorily explain the average effluent ammonium concentration from the pilot plant during experimental periods C and D.

Table 5.10 Average measured and estimated effluent ammonium concentrations from pilot plant.

	Unit	Pilot plant data	Estimated; normal nitrification	Estimated; poor nitrification	As designed
CONDITIONS of estimate of effl. NH_4-N concentration acc. to Figure 5.23 and Figure 5.24.					
Influent flow; Q_1	m^3/s	-	Flow of pilot plant scaled to main plant scale		Main plant flow
Recirculated flow; Q_{13}	m^3/s	-	Recirculation flow of pilot plant scaled to main plant scale		As designed: $Q_{13} \leq 6$ $Q_{13} \leq 10-Q_1$
Input effl. NH_4-N mass flow (F_4)	kg N/d	-	Mass flow distribution of Figure 5.26.		
Cap. of nitrification ΔF_{max}	kg N/d	-	3500	2500	4000
RESULTS: Average effluent ammonium concentration					
The entire period	g N/m^3	8.9	8.7	11.1	7.6
February-March	g N/m^3	11.3	8.8	11.0	6.8
May-February	g N/m^3	8.4	8.5	11.0	7.7
July	g N/m^3	6.2	7.5	9.8	8.0
May-June, August-February	g N/m^3	8.7	8.7	11.2	7.6

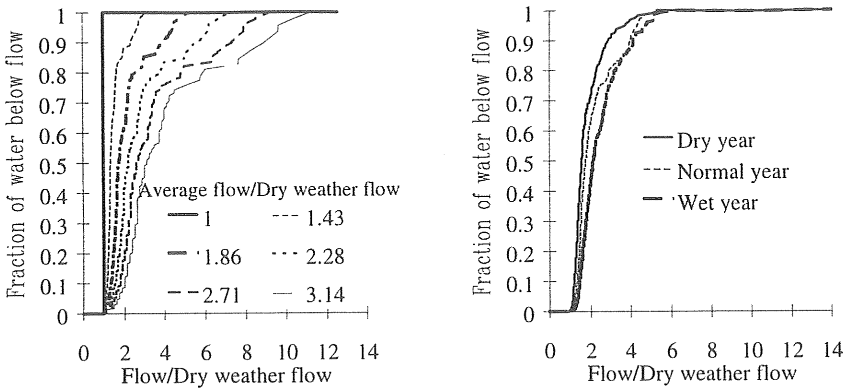
Symbols refer to constants and variable used in equations in Figure 5.23 and Figure 5.24

5.4.4 Influence of average flow on effluent ammonium concentration

In the previous section a wastewater treatment plant based on recirculation of effluent from a nitrifying trickling filter to a non-nitrifying activated sludge system was designed to give a certain average effluent ammonium concentration. The estimates were based on a given distribution of flow and effluent ammonium mass flow. This generally happens once, or a few times, in the history of a WWTP. During the remaining years the plant design is given but flow and quality of the wastewater may vary. The changes may be caused by varying weather or by changes in the collection system.

The characteristics of different collection systems cause the contribution of rain and snow to vary depending on the contributing surface area and other characteristics of the leaking system. One way of characterising a system with different degrees of dilution is to use the flow distribution used above and assume that it is made up by domestic wastewater, industrial wastewater, infiltration/inflow and storm water as suggested by Metcalf and Eddy (1991). The flow of storm water and infiltration/inflow is assumed to vary whereas the domestic and industrial wastewater flow is assumed to be constant. The domestic and industrial wastewater is assumed to contain all the ammonium. The ammonium-free additional flow can, for each data point, be multiplied by a factor giving a new distribution with a different average flow (Figure 5.32 a). This illustrates what could happen if a town, in the same place and with the same weather, had a more or less leaky collection system. In reality flow variations from year to year are more likely to be caused by varying weather. Figure 5.32 b shows flow distributions to the Rya WWTP from three different years. During the dry year (1996) 70 % of the influent wastewater reached the plant at flows less than twice the dry weather flow whereas during the wet year (1988) this was true for only about 40 % of the influent wastewater. The flow distribution of the normal year (1993) is quite similar to that of the data from 1990-1992 used in the previous section.

In the following an attempt is made to quantify the combined effect of varying flow distributions and plant capacities. The ammonium mass flow distribution is assumed to be the same as that used in section 5.4.3 and the average effluent ammonium concentration is calculated using the same equations. The average effluent ammonium concentration is calculated from WWTP:s with different capacities when treating wastewater with the distributions constructed above. The WWTP:s chosen (Table 5.11 and Figure 5.33 a and b) are the wastewater treatment plant described in the previous section(I), the same plant assuming that hydraulic limits and nitrification capacity can be increased somewhat(II), the same plant with unlimited denitrification capacity for instance due to the addition of a carbon source (III), a plant to receiving large quantities of storm water (IV) and plant I with good nitrification and unlimited denitrification (V).



a) More or less leaky collection systems b) Varying weather

Figure 5.32 Flow distributions to a WWTP receiving a varying amount of water. (Dry weather flow is here the domestic and industrial wastewater flow. No storm water or infiltration/inflow included).

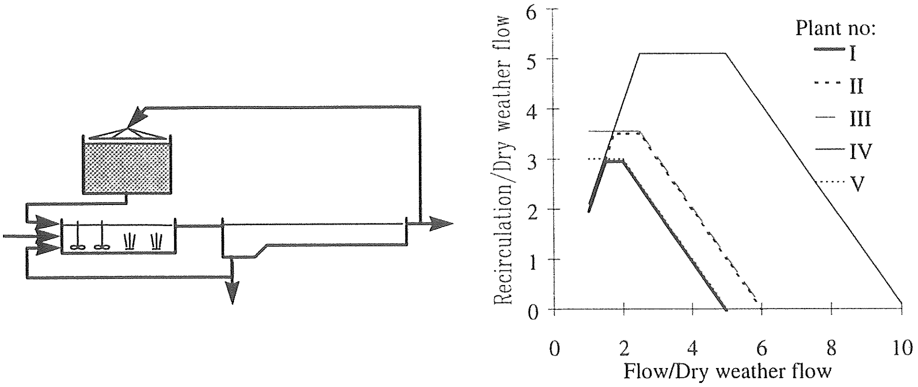
Table 5.11 WWTP capacities.

		Hydraulic capacities expressed as multiples of domestic and industrial WW flow*		Denitrification capacity	Nitrification capacity expressed as % of effluent ammonium
Plant No	Description	Sedimentation	Trickling filter	multiples of flow	from activated sludge system
I	As described in section 5.4.3	5	3	2	73*
II	Plant I stretching hydraulic and nitrification capacities	6	3.5	2	92*
III	Plant II with unlimited denitrification	6	3.5	10	92*
IV	Plant designed for high flows	10	5	2	73*
V	Plant I with good nitrification and unlimited denitrification	5	3	10	92*

* For comparison with the previous section the domestic and industrial WW flow is 2 m^3/s and nitrification capacities of 73 and 92% of effluent ammonium from the activated sludge system correspond to 4000 and 5000 kg N/d.

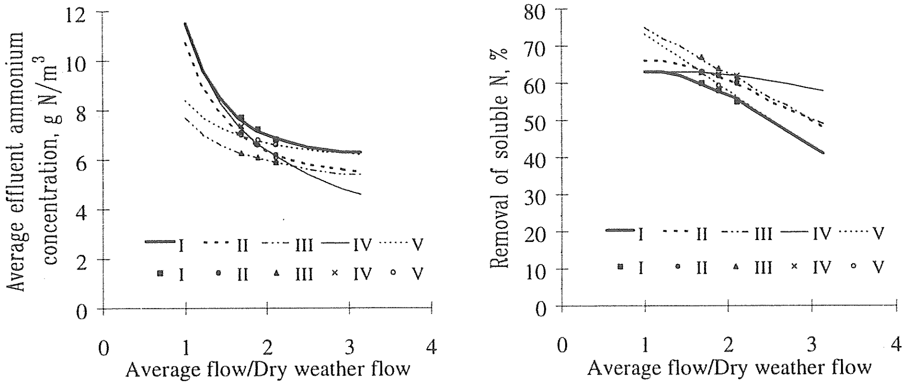
Now how would the plant the design of which was arrived at in section 5.4.3 (plant I of Table 5.11) react to changes of the average flow? Within the range of variation between wet years and dry years the average effluent ammonium concentration (Figure 5.33 c) would be stable around 7 g N/m^3 while removing 55-60 % (Figure 5.33 d) of the soluble nitrogen. However in a system with significantly more contributing storm water and infiltration/inflow the hydraulic capacities would limit nitrogen removal and a lower fraction of the soluble nitrogen would be removed (although the effluent ammonium concentration would still be low). In this case slightly better removal can be obtained by stretching the hydraulic capacities to their limits (II), but in order to obtain a significant improvement the plant has to be designed for significantly higher hydraulic capacities (IV). An often desired development in an existing collection system is that less storm-water should reach the WWTP. In this system (I) a consequence of such a development is that the removal of nitrogen increases to above 60 %; however since the wastewater is less diluted, the average effluent ammonium concentration will increase. At the low flows prevailing in such a system the capacity of denitrification will limit recirculation. In a system with improved denitrification (V) for instance through the addition of a carbon source, more water can be recirculated at low flows, removing around 60 % of the soluble nitrogen while keeping the average effluent ammonium concentration around 7 g N/m^3.

It can be concluded that a system based on recirculation of clarified effluent must be designed considering the effects of the flow pattern created by the collection system and the weather conditions. The plant designed in the previous section is robust enough to cope with the varying flows caused by the weather varying from year to year. However significant changes in the collection system, such as disconnection of a large amount of storm-water or increased storm-water flows, could require design or operation to have to be changed. At decreased flows the addition of a carbon source may solve the problem. At increased flows effluent ammonium concentrations will decrease, but the removal of ammonium will also decrease if the hydraulic capacities of the system are not increased.



a) WWTP configuration.

b) Limits to recirculation.



c) Influence of flow on average effluent ammonium concentration.

d) Influence of flow on average removal of soluble N.

Figure 5.33 Effect of flow on nitrogen removal in a system based on recirculation from an NTF to a non-nitrifying, denitrifying, activated sludge system. In figures c) and d) the lines refer to varying flow due to a varying fraction of the storm-water and infiltration/inflow reaching the collection system and the points to variations registered at the Rya WWTP during the three model years of 1988, 1993 and 1996.

5.4.5 Influence of operational strategy on results

With knowledge of the wastewater quality and process conditions, operation can be controlled to optimise treatment. Where nitrogen removal is concerned it is appealing to control operation to utilise the available carbon source for denitrification. One such possibility is to control operation in order to nitrify exactly the amount of nitrogen which can currently be removed by denitrification. Another possibility is to control the supply of organic mater and/or nitrogen in space or time in order to maximise nitrogen removal. In a system including recirculation of nitrified secondary effluent to a non-aerated activated sludge tank for denitrification some possibilities of optimisation exist within the limits set by the system. In this section some different options of optimising the system will be investigated using pilot plant data and Rya WWTP data as examples. An attempt is made to quantify the potential benefit of optimising the process within the limits set by design. Note that here the hydraulic capacity of the secondary settlers, the nitrification capacity of the trickling filter and the pumping capacity to them will be considered as fixed by design.

The return liquor from the sludge treatment process contains a large amount of ammonium which may be controlled internally. Table 5.12 shows average flows and concentrations of return liquors from the sludge treatment process and influent water to the Rya WWTP. The return liquor flows are low, the supernatant from the thickeners accounts for about 1 % of the pumped flow to the treatment plant and the sludge liquor from the band filter presses for about 0.5 % of the flow. However the ammonium concentrations are high. The sludge liquor from the band filter presses accounts for about 12 % of the total ammonium mass flow to the wastewater treatment plant and the supernatant from the thickeners for about 2.5 %. As these liquors make up a substantial share of the total soluble nitrogen mass flow to the biological treatment, and as they can to a certain extent be controlled at the WWTP they are obvious targets when additional nitrogen removal is needed. At least three different strategies are available:

- Separate treatment of return liquors (nitrification or nitrogen removal).
- Return liquors returned to the main plant when capacity for nitrogen removal is good.
- Return liquors returned to the main plant at a location where it may best be treated.

The first option, treatment of return streams, can be performed in a number of different ways including biological nitrogen removal, ammonia stripping and precipitation of magnesium ammonium phosphate (MAP). The remaining options concern treating the nitrogen from sludge treatment processes in the main plant, but controlling the return stream in space or time so as to make the best use of the capacity of the treatment plant.

If the capacity of denitrification can be estimated in advance it should be possible to co-ordinate return streams in time. The pilot plant experiments show the capacity of denitrification to be poor on Sundays and to some extent on Mondays. If no sludge is dewatered when the denitrification capacity of the plant is poor, but instead stored and dewatered when the capacity for denitrification at the WWTP is better, the capacity of denitrification may be used more efficiently. At the Rya WWTP the digestors are followed by a storage tank with the capacity to store 2 to 3 days' production of digested sludge before dewatering. The storage tank is intended for storage of sludge so as to minimise the consequences of operational problems; however here we will simulate using them to buffer the ammonium loading on the biological treatment system. Implementing this in reality is dependent on exact knowledge in advance of diurnal variations and transport times. Experience from the Littleton-Englewood WWTP along these lines showed success after the performance of lithium studies in order to determine detention times (Farmer and Caudill, 1995).

Table 5.12 Average flows and concentrations at the Rya WWTP (1994)

		Influent (incl. rec. sludge liquor)	Influent (excl. rec. sludge liquor)	Supernatant from thickeners	Filtrate from sludge dewatering
Flow	m^3/d	370 000	364 300	3 700	2000
	% of influent	100	98.5	1.0	0.5
COD	g O_2/m^3	310		5 000	1 200
	kg O_2/d	120 000	95 000	18 000	2 400
	% of influent	100	82	16	2.1
BOD_7	g O_2/m^3	120		730	370
	kg O_2/d	46 000	43 000	2 700	750
	% of influent	100	93	5.9	1.6
Nitrogen	g N/m^3	25		220	550
	kg N/d	9 400	7 500	810	1100
	% of influent	100	80	8.6	12
Ammonium	g N/m^3	17		45	370
	kg N/d	6 400	5 500	160	750
	% of influent	100	86	2.5	12
Phosphorus	g P/m^3	4.5		35	33
	kg P/d	1 700	1 500	130	66
	% of influent	100	88	7.5	3.9

Another alternative is to return the nitrogen-rich water to the plant at the location where it is best treated. Assume that the influent wastewater contains a certain amount of organic matter, with the capacity of reducing a certain amount of nitrate (or a corresponding amount of oxygen). The organic matter should as far as possible be used to reduce nitrate and not oxygen. In this process, where nitrification takes place in a trickling filter and the nitrified water is returned to the activated sludge system for denitrification, the nitrate concentration of the recirculated water will be lower at higher recirculation rates. The oxygen concentration of the recirculated flow will, however, be constant. Thus higher recirculation will increase the nitrate mass flow to the anoxic zone, however the oxygen mass flow will increase even more, and thus decrease the amount of organic matter available for denitrification. If the filtrate from the band filter presses is returned to the inlet of the trickling filter instead of to the head of the plant, more ammonium can be nitrified in the trickling filter since the supply of ammonium to the trickling filter less often will be limited by the capacity of the final clarifiers. Thus, more nitrate can be recirculated to the activated sludge tanks without increasing the amount of recirculated oxygen and water significantly.

The effect of the mode of operation can be estimated using a simple model (Figure 5.34) based on the following assumptions:

- A certain amount of nitrogen in the wastewater is (after the activated sludge system) available for nitrification. This "available ammonium" includes the net release of ammonium in the activated sludge system. This is expressed as $S_{NH4,I}$ g N/m^3 (gram nitrogen in primary settled wastewater which will be in the form of ammonium after leaving the activated sludge system).
- The nitrification capacity of the trickling filter is set at a certain maximum value; B g N/s.
- The denitrification capacity (the capacity to reduce nitrate and oxygen) is controlled by influent wastewater organics and can be expressed as A g N per m^3 influent wastewater.
- Recirculated water from the trickling filter contains 8 g O_2/m^3, which causes a decrease of denitrification capacity by (8/2.86 =) 2.8 g N per m^3 of recirculated nitrified water.
- The effluent ammonium concentration from the trickling filter is assumed to be 0.3 g N/m^3 more than the concentration indicated by the nitrification capacity.
- Even if denitrification is not limited by the denitrification capacity (as defined above) there will be a small amount of nitrate in the effluent. At full utilisation of the denitrification capacity the effluent nitrate concentration will be 0.5 g N/m^3. Below this level the effluent nitrate concentration will be proportional to the degree of utilisation of the capacity of denitrification.
- The small additions to the effluent nitrate and ammonium concentrations from the denitrification unit and the nitrification unit respectively are not included in the internal mass flows.
- The flow of the return liquor from the band filter presses is disregarded.

The mass flow of ammonium to the trickling filter is the sum of the recirculated ammonium and the mass flow of ammonium going directly to the trickling filter:

$$F_{NH4,16} = Q_1 \cdot S_{NH4,1} \frac{Q_{13}}{Q_1 + Q_{13}} + F_{NH4,15}$$

The amount of ammonium nitrified is either limited by the capacity of nitrification or by the supply of ammonium:

$$\Delta F_{NH4,NTF} = \min(F_{NH4,16}, B)$$

However the effluent from the trickling filter will always be assumed to contain 0.3 mg N/l ammonium extra. The effluent ammonium concentration is approximately:

$$S_{NH4,8} = \frac{Q_1 \cdot S_{NH4,1} + F_{NH4,15} - \Delta F_{NH4,NTF}}{Q_1} + 0.3 \frac{Q_{13}}{Q_1 + Q_{13}}$$

The nitrate mass flow to the activated sludge tank equals the ammonium loss in the trickling filter:

$$F_{NO3,6} = \Delta F_{NH4,NTF}$$

The removal of nitrate in the activated sludge system is either limited by the capacity of denitrification or by the supply of nitrate from the trickling filter:

$$\Delta F_{NO3} = \min(A \cdot Q_1, \Delta F_{NH4,NTF} + {}^{S_{O2,9}}\!/_{2.86} \cdot Q_{13}) - {}^{S_{O2,9}}\!/_{2.86} \cdot Q_{13}$$

The effluent nitrate concentration will be governed by denitrification. However when the supply of nitrate and oxygen is lower than the capacity of denitrification the effluent nitrate concentration is proportional to the degree of utilisation of the capacity of denitrification (at full utilisation; 0.5 mg N/l):

$$S_{NO3,8} = \max(\frac{\Delta F_{NH4,NTF} - \Delta F_{NO3,AS}}{Q_1}, \frac{\Delta F_{NO3,AS} + {}^{S_{O2,9}}\!/_{2.86} \cdot Q_9}{A \cdot Q_1} \cdot 0.5)$$

Symbol	Explanation	Unit
Q	Flow	m^3/s
S	Concentration	g/m^3
F	Mass flow	g/s
$S_{NH4,1}$	N in influent wastewater in the form of NH_4^+ in effluent from activated sludge system	$g\ N/m^3$
A	Denitrification capacity of influent wastewater	$g\ N/m^3$
B	Nitrification capacity of trickling filter	g N/s
$S_{O2,9}$	Oxygen in trickling filter effluent (8 $g\ O_2/m^3$)	$g\ O_2/m^3$
$F_{NH4,15}$	Extra ammonium going directly to NTF (sludge liquor from band filter presses)	g N/s
$\Delta F_{NH4,NTF}$	Loss of ammonium in trickling filter (nitrification)	g N/s
$\Delta F_{NO3,AS}$	Loss of nitrate in activated sludge (denitrification)	g N/s
Index		
1-16	Location indicated in (Figure 5.22)	
15	Ammonium-rich sludge liquor pumped directly to trickling filter	
16	Total to trickling filter (13+15)	
NH4	Ammonium nitrogen	
NO3	Nitrate nitrogen	
O2	Oxygen	
NTF	Nitrifying Trickling Filter	
AS	Activated Sludge	

Figure 5.34 Estimate of the effect of recirculation on effluent nitrate and ammonium nitrogen concentration.

The input to the model consisted of data from five different weeks of pilot plant operation. For these weeks data were available for 2 to 5 days and for the remaining days input data were chosen based on flow and on knowledge of the relationships between influent concentrations and flow. These weeks represent different conditions. The frequency of each type of week in reality will not be evaluated here, the main merit of the different weeks being that they represent different situations. The effluent ammonium and nitrate concentration of each day of these weeks was calculated using the model described above for different operational strategies. Each week will be described briefly below (for data and model constants refer to Table 5.14).

- During **week I,** in May 1993, the flow was low, influent ammonium concentrations were high. Since the effluent contained nitrate on Sunday and Monday, we can assume that the capacity of denitrification was limiting on these days. The ratio between influent soluble COD and nitrogen varied between 3.3 and 5.5 g O_2/g N, with the lowest values on Sunday and Monday.
- During **week II**, in August to September 1993, the flow was higher, ammonium concentrations lower, but judging by the fact that the effluent nitrate concentrations were high despite low recirculation, the capacity of denitrification was limiting. The ratio between influent soluble COD and nitrogen was 5.0 and 3.8 g O_2/g N on Tuesday and Thursday, respectively.
- **Week III**, in November 1993 was, like week I, a dry week.
- During **week IV,** in December 1993, flow was high, recirculation rates low, and the carbon to nitrogen ratio as soluble COD 3.8 - 4 g O_2/g N. The effluent nitrate concentration on Sunday, when the flow was highest, was significant.
- The last week studied, **week V**, in January to February 1994, was a wet week. The effluent nitrate concentration was high on Sunday, and low on Tuesday indicating low denitrification capacity on Sunday and sufficient on Tuesday. Carbon to nitrogen ratios were on Sunday 3.3 g O_2/g N and on Tuesday 4.6 g O_2/g N as soluble COD.

For each day of the chosen weeks, when data were available, the model constants "denitrification capacity" and "available ammonium" were chosen to fit effluent ammonium and nitrate concentrations. For the remaining days of the same week constants were chosen in relation to those of the days where data were available (Table 5.14). Using these model constants and varying the operational strategies according to Table 5.13 the effluent ammonium and nitrate concentration could be calculated for the different combinations of operational strategy and day.

When the recirculation flow was chosen as the flow giving the lowest effluent soluble nitrogen concentration, each case was calculated for recirculation flows from 1 to 6.5 m^3/s (at steps of 0.5 m^3/s), and the recirculation giving the lowest effluent concentration of ammonium plus nitrate nitrogen was chosen (within the limits set by sedimentation and pumping to the NTF). Applying this in practice is dependent on reliable knowledge of the influent wastewater quality and process capacities. This is probably only possible with advanced instrumentation and process control, the details of which will not be discussed here. When pilot plant conditions are mentioned the flows will in the following always be expressed as if they had concerned the main plant (4 m^3/s in the main plant corresponds to about 1.5 l/s in the pilot plant).

Table 5.13 Operational strategies.

Strategy No. - and brief description	Recirculation flow	Sludge liquor from band filter presses	Carbon source added
1 - Flow control	Determined by Figure 5.24*	Returned to head of plant	no
2 - Best recirculation	Within limits determined by Figure 5.24* giving the lowest effl. soluble nitrogen content	Returned to head of plant	no
3 - Move rec. of sludge liquor in time	Within limits determined by Figure 5.24* giving the lowest effl. soluble nitrogen content	Returned to head of plant. On Sun. it is stored and returned on Tue. through Fri. instead	no
4. - Rec. sludge liquor to NTF	Within limits determined by Figure 5.24* giving the lowest effl. soluble nitrogen content	Returned to NTF	no
5 - Move rec. of sludge liquor in time and space	Within limits determined by Figure 5.24* giving the lowest effl. soluble nitrogen content	Returned to NTF. On Sun. it is stored and returned to NTF on Tue. through Fri. instead	no
6 - Add carbon source	Within limits determined by Figure 5.24* giving the lowest effl. soluble nitrogen content	Returned to head of plant	yes
7 - Rec. sludge liquor to NTF and add carbon source	Within limits determined by Figure 5.24* giving the lowest effl. soluble nitrogen content	Returned to NTF	yes
8 - Remove nitrogen from recirculated sludge liquor	Within limits determined by Figure 5.24* giving the lowest effl. soluble nitrogen content	All ammonium removed from sludge liquor in a separate unit	no
9 - remove nitrogen from sludge liquor and add carb. source.	Within limits determined by Figure 5.24* giving the lowest effl. soluble nitrogen content	All ammonium removed from sludge liquor in a separate unit	yes

* $Q_{4,max} = 10\ m^3/s$, $Q_{13,max}=6\ m^3/s$, a=2

In order to illustrate the mechanisms of the system the results will first be discussed in detail for two examples, the Sunday and the Tuesday of week III (the 7:th and 9:th November). For strategy 1, results are shown for each day. For the remaining strategies a comparison of weekly averages will be made.

On Sunday 7th November 1993 the influent flow was 2.4 m^3/s, and in the pilot plant a flow corresponding to 5.6 m^3/s was recirculated. The effluent ammonium concentration was 10.2 g N/m^3 and the effluent nitrate concentration 2.4 g N/m^3. If the available ammonium concentration is set at 31 g N/m^3 and the denitrification capacity at 25 g N/m^3 , the effluent ammonium and nitrate concentrations will be 10.0 and 2.8 g N/m^3 respectively, which is close to the analysed values. Figure 5.35 a shows the increase of the nitrate mass flow to the anoxic zone with increased recirculation. If 5 m^3/s is recirculated the full nitrification capacity of the trickling filter is utilised. Increased recirculation beyond this point does not increase the nitrate mass flow to the anoxic zone. Denitrification was however limited by the capacity of denitrification even below this level. The maximum denitrification capacity corresponded to 60 g N/s. This level would be reached by the total oxidant mass flow at about 4.5 m^3/s. At recirculation rates exceeding this level an increasing amount of the denitrification capacity is used by oxygen causing the effluent nitrate concentration to increase (Figure 5.35 b). In the pilot plant the recirculation was higher, causing elevated effluent nitrate concentrations. Decreasing recirculation to 4.8 m^3/s (twice the influent flow, strategy 1) would have lowered the effluent inorganic nitrogen concentration by 1 g N/m^3 (to 11.8 g N/m^3). A slight additional improvement could have been obtained by choosing the best recirculation (4.5 m^3/s, strategy 2) yielding an effluent inorganic nitrogen concentration of 11.5 g /l.

If the ammonium content of the water from the band filter presses does not reach the main plant (as is the case on Sundays using strategies 3, 5, 8 and 9, the nitrate mass flow to the anoxic zone decreases at the same recirculation. Without exhausting the capacity of denitrification, the recirculation could have been increased to 5.5 m^3/s and the effluent inorganic nitrogen concentration brought down to 9.0 g N/m^3 (not illustrated).

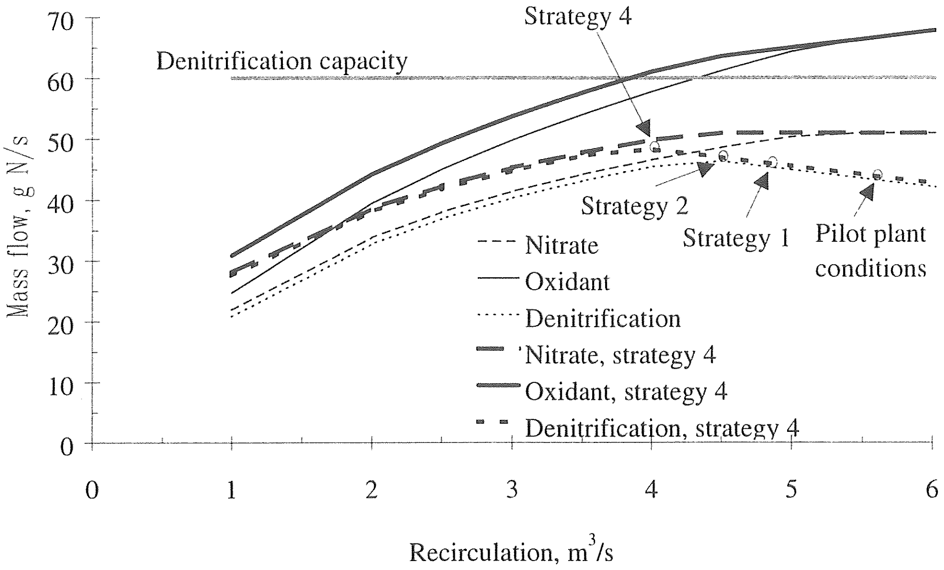
If the ammonium mass flow from the band filter presses, instead of being removed completely, had been returned to the trickling filter, the amount of nitrate returned to the anoxic zone could have been increased without increasing the amount of oxygen returned. The recirculation of water could have been minimised and the available carbon utilised better (Figure 5.35, strategy 4). At 4 m^3/s the effluent soluble nitrogen concentration would be 11.0 g N/m^3. If, however, the recirculation is increased above the optimal point more oxygen will be recirculated, consuming the carbon source available for denitrification. At 4.5 m^3/s the full nitrification capacity is used if the sludge liquor from the band filter presses is added to the trickling filter. Above this point, returning sludge liquor directly to the trickling filter instead of to the head of the plant gives no advantage in terms of nitrogen removal.

If a carbon source is added in order to give unlimited denitrification, strategy 6, the recirculation can be increased until the capacity of nitrification sets the limit without the effluent nitrate concentration increasing. The effluent ammonium concentration at 5.5 m^3/s is 9.9 g N/m^3 and, with a small amount of effluent nitrate, the effluent inorganic nitrogen concentration may be 10.3 g N/m^3.

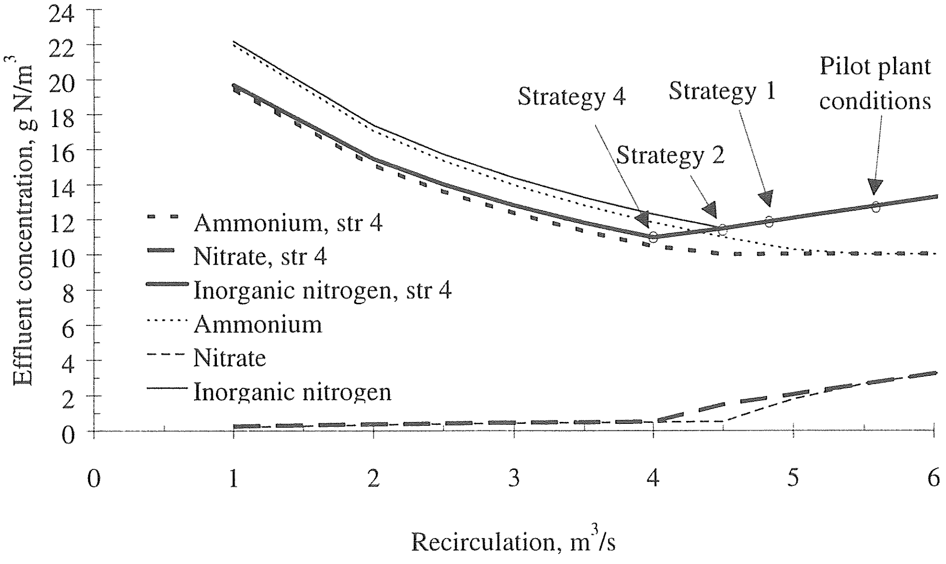
On the following Tuesday, the 9th November 1993, the situation was different (Figure 5.36). The effluent nitrate concentration was low indicating that the denitrification capacity was not limiting. Setting the available soluble nitrogen content at 25 g N/m^3 and the denitrification capacity at 35 g N/m^3 gave 9.8 g N/m^3 effluent ammonium and 0.3 g N/m^3 nitrate (measured values 9.6 and 0.2 g N/m^3 respectively). In this case, since neither nitrification nor denitrification is limiting, an increase of recirculation to the maximum pumping capacity to the trickling filter, 6 m^3/s, would give the lowest effluent soluble nitrogen concentration (strategies 1 and 2). If the sludge liquor from the band filter presses is returned to the trickling filter instead of to the head of the plant more ammonium can be nitrified at the same recirculation (strategy 4). A decrease of effluent inorganic nitrogen to 8.5 g N/m^3 can be obtained at a recirculation flow of 5.5 m^3/s. If, in addition to the sludge water due to the plant on Tuesday, the sludge water from

Sunday is stored and returned at the plant on Tuesday, the effluent concentration will increase since the nitrification capacity of the trickling filter would limit further nitrogen removal.

In the same way, the model constants denitrification capacity and available ammonium were chosen to fit the available data of all the days of the model weeks or, in absence of data, set at reasonable values in relation to available data from other days during the same week (Table 5.14). The main purpose of the operation was to construct reasonable models of weeks with different characteristics upon which to test the operational strategies. For the days when data were available a comparison could be made with modelled data (using actual flows and recirculation rates, see Table 5.14). Model results for the first operational strategy, recirculated flow controlled by influent flow, are also shown in Table 5.14. The results of all operational strategies are shown as weekly averages of effluent inorganic nitrogen in Table 5.15. Figure 5.37 shows the differences between the results of each operational strategy and strategy 2.

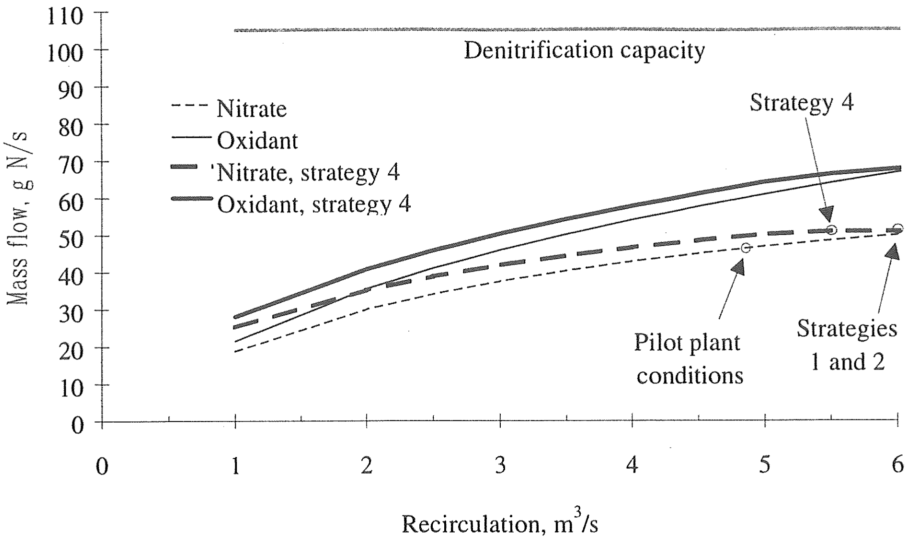


a) Mass flow to and denitrification in activated sludge system.

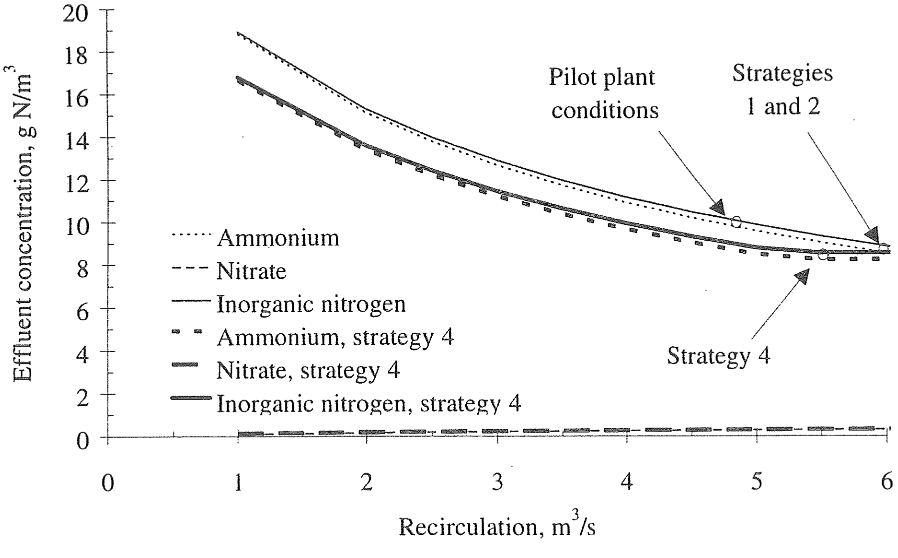


b) Effluent concentration from activated sludge system.

Figure 5.35 Results of simulations using data from Sunday 7th November 1993.

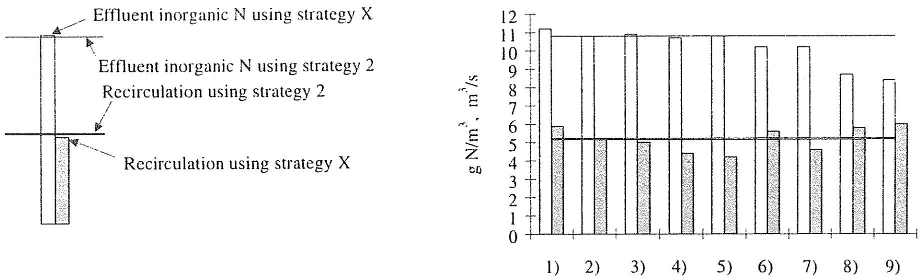


a) Mass flow to and denitrification in activated sludge system



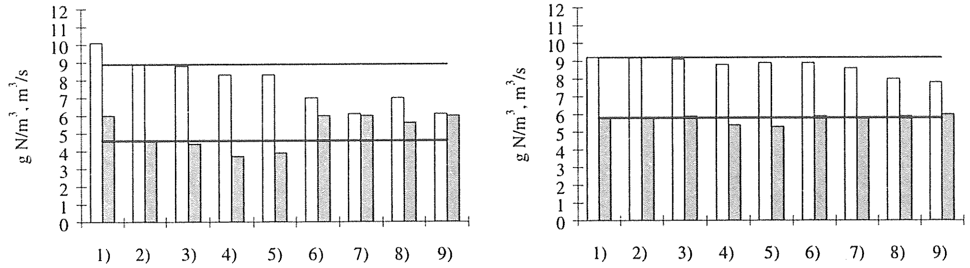
b) Effluent concentration from activated sludge system.

Figure 5.36 Results of simulations using data from Tuesday 9th November 1993.



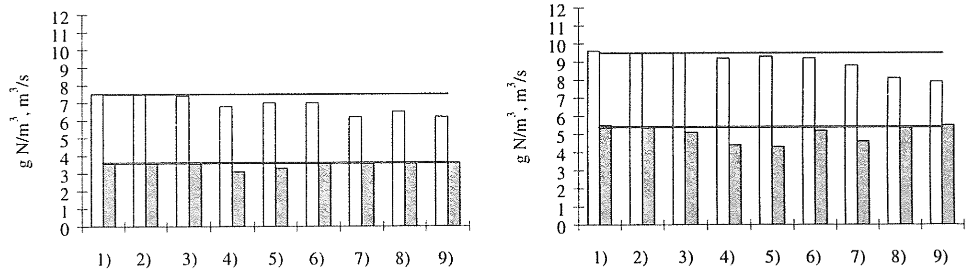
Legend

Week I



Week II

Week III



Week IV

Week V

Figure 5.37 Average effluent inorganic nitrogen concentration and recirculation rate using different operational strategies.

Table 5.14 Modelling and model constants (capacity of nitrification 4 400 kg N/d).

Date/ week	Day	Flow$^{(1,3)}$ (Q_I)	$NH_4^{(4)}$ ($SNH_{4,I}$)	Dencap. (A)	Pilot plant results $^{(1)}$			Modelled pilot plant results $^{(1,2)}$			Results of strategy 1		
					Q_{rec}	NH_4	NO_3	Q_{rec}	NH_4	NO_3	Q_{rec}	NH_4	NO_3
		m^3/s	gN/m^3	gN/m^3	m^3/s	gN/m^3	gN/m^3	m^3/s	gN/m^3	gN/m^3	m^3/s	gN/m^3	gN/m^3
93-05-09	Sun.	2.7(2.7)	28(28)	21	5.7	9.8	4.0	5.7	9.3	3.8	5.4	9.5	3.3
	Mon.	3.1(3.1)	26(26)	19	5.3	10.6	2.0	5.3	9.8	2.2	6	9.8	2.8
I	Tue.	3(2.6)	27(24)	35	5.8	8.7	0.7	5.8	8.6	0.4	6	10.2	0.32
	Wed.	3(2.6)	27(23)	35	5.8	10.2	0.3	5.8	8.6	0.4	6	10.2	0.32
	Thu.	3(3.3)	27(24)	35	5.1	9.6	0.06	5.1	11.8	0.3	6	10.2	0.32
	Fri.	3(3.2)	27	35							6	10.2	0.32
	Sat.	3(2.8)	27	35							6	10.2	0.32
93-08-29	Sun.	3.2(3.3)	19(20.7)	12	4.7	8.1	3.3	4.7	8.0	3.2	6	6.8	5.6
	Mon.	3.2(3.4)	19	13.5							6	6.8	4.1
II	Tue.	3.2(3.1)	19(20)	15	4.8	7.3	0.8	4.8	7.6	0.9	6	6.8	2.6
	Wed.	3.2(3.2)	19(22.2)	15							6	6.8	2.6
	Thu.	3.2(3.2)	19	15	4.8	8.2	1.0	4.8	7.6	0.9	6	6.8	2.6
	Fri.	3.2(3.4)	19	15							6	6.8	2.6
	Sat.	3.2(2.9)	19	15							6	6.8	2.6
93-11-07	Sun.	2.4(2.4)	31(29)	25	5.6	10.2	2.4	5.6	10.0	2.8	4.8	10.5	1.3
	Mon.	3(3.1)	24	24							6	8.2	0.4
III	Tue.	3(3.1)	25(23)	35	4.8	9.6	0.2	4.8	10.0	0.3	6	8.5	0.3
	Wed.	3(2.9)	25	35	4.9	9.4	0.3	4.9	9.5	0.3	6	8.5	0.3
	Thu.	3(2.2)	25	35							6	8.5	0.3
	Fri.	3(3.9)	25	35							6	8.5	0.3
	Sat.	3(3.3)	25	35							6	8.5	0.3

Continued on next page

Table 5.14 Continued from previous page.

Date/ week	Day	Flow$^{(1,3)}$ (Q_1)	$NH_4^{(4)}$ ($SNH_{4,1}$)	Dencap. (A)	Pilot plant results $^{(1)}$			Modelled pilot plant results $^{(1,2)}$			Results of strategy 1		
					Q_{rec}	NH_4	NO_3	Q_{rec}	NH_4	NO_3	Q_{rec}	NH_4	NO_3
		m^3/s	gN/m^3	gN/m^3	m^3/s	gN/m^3	gN/m^3	m^3/s	gN/m^3	gN/m^3	m^3/s	gN/m^3	gN/m^3
93-12-05	Sun.	7.9(7.9)	6.5(6.9)	2	1.2	5.5	0.9	1.2	5.7	0.3	2.1	5.2	0.9
	Mon.	7(6.0)	10	3.5							3	7.1	0.7
IV	Tue.	6(6.7)	12(11)	9	2.1	8.7	0.3	2.1	8.4	0.2	4	7.3	0.4
	Wed.	6(7.7)	12	9							4	7.3	0.4
	Thu.	6(5.9)	12(15)	9	2.4	9.2	0.2	2.4	8.6	0.3	4	7.3	0.4
	Fri.	6(4.8)	12	9							4	7.3	0.4
	Sat.	6(5.9)	12	9							4	7.3	0.4
94-01-31	Sun.	4.8(4.8)	12.5(12)	7.5	5.4	6.2	2.1	5.4	6.0	2.3	5.2	6.2	2.0
	Mon.	5.5(5.8)	16	13							4.5	8.9	0.4
V	Tue.	5.5(5.5)	16(15)	20	4.4	8.9	0.2	4.4	9.0	0.2	4.5	8.9	0.2
	Wed.	3.8(3.7)	23	29							6	9.8	0.3
	Thu.	3.8(4.1)	23	29							6	9.8	0.3
	Fri.	3.8(3.8)	23	29							6	9.8	0.3
	Sat.	3.8(3.6)	23	29							6	9.8	0.3

1 Flow to pilot plant and recirculated flow scaled to match flow to main plant.

2 Results modelled using actual flow and actual recirculation. Available NH_4^+ and denitrification capacity as used for modelling.

3 Flow used for modelling (actual flow to pilot plant).

4 Available ammonium used in model (ammonium concentration in primary settled wastewater to pilot plant).

Symbols refer to variables and constants used in Figure 5.34.

Table 5.15 Modelled effluent inorganic nitrogen (ammonium + nitrate) concentration for different operational strategies (Capacity of nitrification 4 400 kg N/d).

	Dry, spring, I 93-05-09		Summer, II 93-08-29		Dry, winter, III 93-11-07		Very wet, winter, IV 93-12-05		Wet , winter, V 94-01-31	
	N_{ef} (Q_{rec})	Lim.	N_{ef} (Q_{rec})	Lim.	N_{ef} (Q_{rec})	Lim.	N_{ef} (Q_{rec})	Lim.	N_{ef} (Q_{rec})	Lim.
1. Flow control	11.2 (5.9)	p(d)	10.1 (6.0)	p	9.2 (5.8)	p(d)	7.5 (3.6)	s	9.6 (5.5)	p(s)
2. Best recirculation	10.8 (5.2)	n(d)	8.9 (4.6)	d	9.2 (5.8)	p(d)	7.5 (3.6)	s(d)	9.5 (5.4)	p(sd)
3. Move rec. in time	10.9 (5.0)	n(d)	8.8 (4.4)	d	9.1 (5.9)	p(d)	7.4 (3.6)	s(d)	9.5 (5.1)	n(sd)
4. Rec. to TF	10.7 (4.4)	n(d)	8.3 (3.7)	d	8.8 (5.4)	n(dp)	6.8 (3.1)	s(d)	9.2 (4.4)	n(sd)
5. Move in time and space	10.8 (4.2)	n(d)	8.3 (3.9)	d	8.9 (5.3)	n(dp)	7.1 (3.3)	s(d)	9.3 (4.3)	n(sd)
6. Carbon source	10.2 (5.6)	n(p)	7.0 (6.0)	p	8.9 (5.9)	p	7.0 (3.6)	s	9.2 (5.2)	n(s)
7. Rec.to TF+add. carb.	10.2 (4.6)	n	6.1(6.0)	p	8.6 (5.8)	p(n)	6.2 (3.6)	s	8.8 (4.6)	n(s)
8. Nit. rem. rec. water	8.7 (5.8)	p(d)	7.0 (5.6)	p(d)	8.0 (5.9)	p(d)	6.5 (3.6)	s	8.1 (5.4)	p(sd)
9. Nit rem. rec. w. + add. c.	8.4(6.0)	p	6.1(6.0)	p	7.8(6.0)	p	6.2(3.6)	s	7.9(5.5)	

N_{ef} = effluent nitrate + ammonium concentration, g N/m^3

Q_{rec} = recirculation flow, m^3/s

Lim. = recirculation limited by nitrification, denitrification, sedimentation or pumping to nitrifying trickling filter. Most common limitation indicated first (limitation one or more days in brackets).

Table 5.16 Difference between results of each control strategy and strategy 2 (Capacity of nitrification 4400 kg N/d).

	Dry, spring, I 93-05-09		Summer, II 93-08-29		Dry, winter, III 93-11-07		Very wet, winter, IV 93-12-05		Wet , winter, V 94-01-31	
	N_{ef}, g N/m^3	Q_{rec}, m^3/s	N_{ef}, g N/m^3	Q_{rec}, m^3/s	N_{ef}, g N/m^3	Q_{rec}, m^3/s	N_{ef}, g N/m^3	Q_{rec}, m^3/s	N_{ef}, g N/m^3	Q_{rec}, m^3/s
1. Flow control	+ 0.3	+0.7	+1.2	+1.4	+0.0	+0.0	∓0.0	+0.0	+0.1	+0.2
2. Best recirculation	0	0	0	0	0	0	0	0	0	0
3. Move rec. in time	+ 0.1	-0.2	-0.1	-0.3	-0.1	+0.1	-0.0	∓0.0	+0.1	-0.2
4. Rec. to TF	- 0.2	-1.0	-0.6	-0.9	-0.4	-0.4	-0.7	-0.5	-0.3	-0.9
5. Move in time and sp.	-0.0	-1.0	-0.6	-0.7	-0.3	-0.5	-0.4	-0.3	-0.2	-1.0
6. Carbon source	-0.6	+0.4	-1.9	+1.4	-0.3	+0.1	-0.4	+0.0	-0.3	-0.1
7. Rec.to TF+add. carb.	-0.6	-0.6	-2.8	+0.9	-0.6	0.0	-1.3	+0.0	-0.6	-0.7
8. Nit rem rec. water	-2.1	+0.6	-1.9	+1.4	-1.2	+0.1	-1.0	+0.0	-1.4	+0.0
9. Nit rem. rec. wat. + add. c.	-2.4	+0.6	-2.8	+1.4	-1.4	+0.2	-1.3	+0.0	-1.6	+0.1

Table 5.17 Results of some of the operational strategies assuming the nitrification capacity to be 5 500 kg N/d.

	Dry, spring, I 93-05-09		Summer, II 93-08-29		Dry, winter, III 93-11-07		Very wet, winter, IV 93-12-05		Wet , winter, V 94-01-31	
	N_{ef} (Q_{rec})	Lim.	N_{ef} (Q_{rec})	Lim.	N_{ef} (Q_{rec})	Lim.	N_{ef} (Q_{rec})	Lim.	N_{ef} (Q_{rec})	Lim.
2. Best recirculation	10.1(5.6)	p(d)	8.9(4.6)	d	9.2(5.8)	p(d)	7.4(3.6)	s	9.1(5.3)	p(sd)
3. Move rec. in time	9.9(5.6)	p(d)	8.7(4.4)	d	9.0(5.9)	p(d)	7.4(3.6)	s	9.0(5.4)	p(sd)
4. Filter press water to TF	9.3(5.4)	p(d)	8.3(3.7)	d	8.3(5.7)	p(d)	6.7(3.4)	s(d)	8.3(4.4)	p(sd)
6. Add carb source	9.4(6.0)	p	7.0(6.0)	p	8.8(6.0)	p	7.0(3.6)	s	8.8(5.5)	p(s)
7. Rec. to TF+add . carb	8.4(6.0)	p	6.1(6.0)	p	7.8(6.0)	p	6.2(3.6)	s	8.0(5.5)	p(s)
9. Nit rem rec wat + add c	8.4(6.0)	p	6.1(6.0)	p	7.8(6.0)	p	6.2(3.6)	s	7.9(5.5)	p(s)

N_{ef} = effluent nitrate + ammonium concentration, g N/m^3

Q_{rec} = recirculation flow, m^3/s

Lim. = recirculation limited by nitrification, denitrification, sedimentation or pumping to nitrifying trickling filter. Most common limitation indicated first (limitation one or more days in brackets).

During two of the weeks, I and II, results were significantly better when the best recirculation rate was chosen (strategy 2) than when the standard recirculation rate was chosen (strategy 1). During week I standard recirculation rates would have caused too much ammonium to be returned to the trickling filter in relation to the nitrification capacity and during week II the denitrification capacity was low and the standard recirculation caused too much nitrate and oxygen to be returned to the activated sludge system. During week II effluent nitrogen concentrations could be decreased by 1.2 g N/m^3 while decreasing the recirculated flow by, on average, 1.4 m^3/s. During the remaining weeks, however, sedimentation or pumping to the trickling filter limited the process.

The third control strategy, saving the sludge liquor from Sunday and spreading the mass flow of it out over Tuesday to Friday, did not prove successful. In some cases nitrification (weeks I and V) or denitrification (week II) limited the process during Tuesday through Friday and thus the extra contribution from Sunday could not be removed.

Using strategy 4, pumping the sludge liquor to the trickling filter, the results were consistently better. Compared with strategy 2 effluent nitrogen concentrations were reduced by 0.2 to 0.7 g N/m^3 at 0.4 to 1.0 m^3/s lower recirculation flows. The benefit of pumping the sludge liquor to the NTF was in some cases limited by the nitrification capacity. Strategy 5, a combination of strategies 3 and 4, was not successful since the effect of the limitations of denitrification or nitrification caused by the individual strategies was increased when combining them.

If a carbon source can be added, all recirculated nitrate can be removed (strategy 6). For all the weeks the carbon source was limiting at some point and thus effluent nitrogen could be decreased by 0.4 to 1.9 g N/m^3. During week II, when carbon was limiting every day, effluent inorganic nitrogen could be decreased by 1.9 g N/m^3. However, in order to make use of the unlimited denitrification capacity recirculation flows have to be increased.

The greatest improvement which can be obtained, if no separate treatment of water from filter presses is to be introduced, is to return the water from the filter presses to the NTF and make sure denitrification is not limited (strategy 7). Effluent inorganic nitrogen concentrations can be reduced by 0.6 - 2.8 g N/m^3 to 6.1 - 10.2 g N/m^3. The results of strategy 7 are in several cases close to what can be expected if all nitrogen is removed from the filter press water (strategy 8). A combination of strategies 6 and 8 (removal of nitrogen from filter press water and addition of carbon source) indicates a limit to what is obtainable within the structural limits of the designed treatment plant. Effluent soluble nitrogen concentrations of 6.1 to 8.4 g N/m^3 could be obtained.

In this analysis it was assumed that the nitrification capacity was 4 400 kg N/d. Assuming a higher nitrification capacity changes the effect of some of the strategies. Table 5.17 shows results of strategies 2, 3, 4, 6, 7 and 9 assuming a nitrification capacity of 5 500 kg N/d. At this capacity the effect of strategy 3 is small but that of strategy 4 quite substantial. Strategy 4 (where sludge liquor is pumped strait to the trickling filter) would lower effluent nitrogen concentrations by 0.6 to 0.9 g N/m^3 at 0.1 to 0.9 m^3/s lower recirculation flow (compared with strategy 2). It can also be observed that at this nitrification capacity in the trickling filter there is practically no difference between the results of strategies 7 and 9. This is obvious since if there are no constraints on nitrification and denitrification, it is of no importance to nitrogen removal if the nitrogen removal takes place in the main plant or if part of it takes place in a separate return stream treatment unit.

As mentioned earlier, the weeks chosen for simulation were mainly chosen on the merits of being different and that data were available for model calibration. Therefore no definite conclusions can be drawn as to how different operational strategies influence average annual effluent concentrations. They can however be used to illustrate some points specific to this process and some with general relevance.

- In this process improvements affecting one of the constraints have the potential of improving effluent results by about 0.4 g N/m^3. One such improvement is the recirculation of ammonium-rich sludge water to the trickling filter instead of to the head of the plant. This decreases the constraint imposed upon the system by the limited recirculation. Another example is the addition of a carbon source in order to remove limitations on denitrification and a third is improvements which may increase the nitrification capacity of the trickling filter.
- If two of the constraints are affected simultaneously, improvements are of the order of 0.7 g N/m^3. This is for instance the case when the nitrification capacity of the trickling filter is increased and a carbon source is added.
- If denitrification is not limiting, the ammonium-rich sludge liquor is returned to the trickling filter and if the nitrification capacity is ample the effluent soluble nitrogen content can be decreased by 1-2 g N/m^3. No further nitrogen removal can be obtained by treating the sludge liquor separately.
- A process improvement in one part of the system is often limited by a constraint in another part of the system.
- In a complex system, with knowledge of the variation of wastewater quality and quantity, modelling can be a useful tool for analysis of the effect of operational strategies.
- Although the gain in terms of improved nitrogen removal may not be great the economy of the system may in some cases be improved by choosing the optinal operational strategy. For instance when returning the sludge liquor to the trickling filter instead of to the head of the plant the recirculation flow could be decreased by 0.4 to 1.0 m^3/s if the system can be controlled.
- Treatment of concentrated return liquors may be achieved without constructing additional units if the capacities of the treatment plant are used optimally.

6 Concluding discussion

A non-nitrifying, denitrifying activated sludge system has been demonstrated to give high denitrification rates and to be a useful component in a municipal wastewater treatment plant where partial nitrogen removal is required. One useful configuration is where clarified effluent from a non-nitrifying activated sludge system is nitrified in a tertiary nitrification unit and recirculated to the activated sludge system for denitrification (Figure 6.1 a). Recirculation of the clarified effluent is a limiting factor. The local variation of wastewater flow and quality influences design, operation and treatment results of such a system.

In the following the assumptions made under "thesis objectives and research strategies" (*italics*) are discussed.

- *" High denitrification rates can be expected in a non-nitrifying, relatively highly loaded, activated sludge system. In a highly loaded activated sludge system the respiration of oxygen and therefore probably of nitrate, will be high, not only where the raw wastewater enters the tank but in the entire activated sludge system."*.

Denitrification rates of the order of 10-15 g N/(kg VSS·h) were measured in a pilot denitrifying non-nitrifying activated sludge system when nitrate was not limiting. However when operating the system for nitrogen removal, where nitrate was supplied by recirculation of clarified nitrified effluent, the system was operated at lower average nitrate loadings. In this range the average denitrification rate was 4.7 g N/(kg VSS·h). When calculated in different parts of the system removal rates were high:

- In the first anoxic tank, where primary settled wastewater was added, the average measured denitrification rate was 5.5 g N/(kg VSS·h) and the nitrate and oxygen removal rate corresponded to 9.4 g N/(kg VSS·h).
- In the "deoxygenation tank", where return activated sludge and nitrified RBC effluent were mixed, the average measured denitrification rate was 3.2 g N/(kg VSS·h) and the nitrate and oxygen removal rate was about 6 g N//(kg VSS·h).
- If unexplained nitrogen removal in the activated sludge system were due to nitrification and denitrification in aerated tanks (DO about 3 g O_2/m^3) denitrification rates in aerated tanks would be on average 1.3 g N/(kg VSS·h).

These rates are higher than the 0.3 - 3.0 g N/(kg VSS·h) reported from single-sludge nitrifying denitrifying activated sludge systems using municipal wastewater as their only carbon source (Christensen and Harremoës, 1977).

In a nitrifying activated sludge system the low growth rate of the nitrifiers sets an upper limit to the bulk activated sludge growth rate and thus a lower limit to the solids retention time. Depending on design temperature, safety margins and wastewater quality a single-sludge nitrifying-denitrifying activated sludge system is often designed for 15-20 days SRT or more, whereas a non-nitrifying activated sludge system may be operated with an SRT in the order of 3 to 5 days. This is the main explanation of the higher denitrification rates recorded in the non-nitrifying activated sludge system than in single-sludge systems. Aerobic respiration rates increase at higher organic loadings and (specifically in activated sludge systems) at lower solids retention times. The anoxic respiration rates (when nitrate is used instead of oxygen) also increase with increased loading and decreasing solids retention times. This is demonstrated or indicated in a number of different ways:

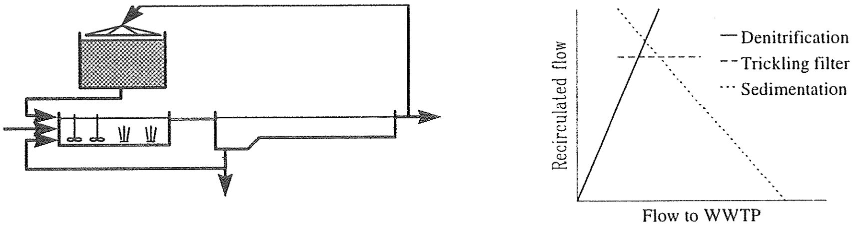
- Denitrification takes place in a wide range of natural as well as constructed systems if environmental conditions allow (anoxia, presence of nitrate and carbon source etc.). However the rate of denitrification varies by several orders of magnitude between low loaded systems and highly loaded systems.
- Observed denitrification rates in non-nitrifying activated sludge systems without an external carbon source of the order of 1 to 5 g N/(kg SS·h) are reported (Balakrishnan and Eckenfelder, 1969 C, Jones et al. 1990 A and B).
- In activated sludge systems several authors have observed higher aerobic or anoxic respiration rates at higher loadings and lower SRT:s (Moore and Schroeder, 1970, Weddle and Jenkins, 1971, Engberg and Schroeder, 1975, Sutton et al., 1978, Argaman, 1986). Aerobic or anoxic respiration rates are often 1.5 to 2.5 times higher at solids retention times of 3 to 5 days than at 10 to 20 days. However in some cases evaluation of respiration rates is complicated by the fact that the systems were operated with nitrification.
- Using the activated sludge model No. 1 (Henze et al., 1987) in a Danish application (EFOR) the heterotrophic respiration can be estimated at different solids retention times with or without nitrifiers in the system. If nitrification was allowed the respiration doubled when the SRT was reduced from 20 to 4 days. If nitrification was suppressed heterotrophic respiration was trebled at the same reduction of SRT.
- The denitrification rate of the activated sludge system can be assumed to be the result of at least three different groups of factors. Respiration increases with increased organic loading. The denitrification rate, in relation to the aerobic respiration, will depend on two main factors: the amount of denitrifiers supplied and the selective pressure of the system towards denitrification. Selection of denitrifiers in the activated sludge system can to a certain extent be controlled by operation and process configuration.

"Partial nitrogen removal (at least 50-70 %) can be obtained in a system consisting of a non-nitrifying denitrifying activated sludge system and separate culture post-nitrifying units where nitrified effluent is recirculated to the activated sludge system for denitrification. The organic matter of the wastewater is used as the carbon source for denitrification. The need to recirculate effluent nitrified wastewater to the non-nitrifying activated sludge system demands extensive secondary settling area. At a wastewater treatment plant with significant flow variations due to storm-water being connected the extra sedimentation capacity demanded for recirculation and for storm-water can be co-ordinated. The flow distribution to the wastewater treatment plant influences the removal of nitrogen and the nitrogen concentration of the effluent wastewater."

One year of pilot plant operation of the system using wastewater from the Rya WWTP influent channel demonstrated an average effluent nitrogen concentration of 11.4 g N/m^3 using an RBC for nitrification of effluent wastewater from the activated sludge system to which part of the nitrified effluent was recirculated. Compared with primary settled wastewater 58 % of the total nitrogen was removed. The organic matter of the wastewater was used as the carbon source for denitrification.

The recirculation rate to the activated sludge system is a key control handle of the process. It controls the supply of nitrate to the activated sludge system and is itself limited by the construction of the system and by wastewater quality (Figure 6.1 b). A design example, using a simple steady state model and Rya WWTP data, was used to illustrate the effect of flow on such a system:

- At high flows the capacity of the final sedimentation tanks limits nitrogen removal by limiting the recirculation flow.
- At low to normal flows the capacity of the nitrification unit (depending on design and operation) or denitrification (depending on wastewater quality) are more likely to limit nitrogen removal.
- At very low flows denitrification is likely to limit nitrogen removal if no carbon source is added.



a) Process configuration.

b) Limitations to recirculation to the activated sludge system.

Figure 6.1 A system for nitrogen removal where secondary effluent is nitrified in a trickling filter and recirculated to a non-aerated part of the activated sludge system for denitrification.

The effects of these limitations on the average effluent concentration varies depending on the distribution of wastewater flow and quality. At the Rya WWTP, the average flow is about twice the domestic and industrial wastewater flow (not including storm water or infiltration/inflow). Here a system where the capacity of the secondary settlers is five times the domestic and industrial wastewater flow and the tertiary nitrification unit can receive three times the domestic and industrial wastewater flow, can reduce the effluent ammonium concentration to 7-8 g N/m^3. In a system receiving more storm water this design would give equally low (or lower) effluent ammonium concentrations, although at a lower removal ratio. In order to remove more nitrogen the hydraulic capacities would have to be substantially greater. In a system receiving less storm water the denitrification capacity is likely to limit nitrogen removal, frequently demanding addition of a carbon source in order to obtain low effluent nitrogen concentrations.

"Nitrogen removal may be further improved by process control."

Various modes of operation and process control were simulated using a simple model in order to determine potential process improvements within the limits set by physical design. Simulations indicate that in certain situations (dry weather) results may be improved by knowledge of the denitrification capacity and thus of the optimal recirculation rate. The effects of three different limitations were investigated:

- Addition of a carbon source: remove limitation due to limited denitrification capacity.
- Ammonium rich sludge liquor pumped to nitrification unit instead of to head of plant: reduce limitation due to circulation.
- Improved nitrification capacity in trickling filter: reduce impact of limited nitrification.

If these limits were manipulated individually the effluent inorganic nitrogen concentration could be decreased by about 0.4 g N/m^3. If all three measures were taken simultaneously the improvement could be of the order of 1 to 2 g N/m^3. No further removal could be obtained by separate treatment of sludge liqour.

"Due to seeding of nitrifiers from the trickling filter to the activated sludge system the "non-nitrifying" activated sludge system may not be completely non-nitrifying."

Mass balances through the pilot plant showed 58 % of the nitrogen supplied by primary settled wastewater to have been removed. Denitrification in non-aerated tanks accounts for 29 %, removal with waste activated sludge for 14 %, unexplained removal in the RBC for 5 % and unexplained removal in the activated sludge system for 11%. The magnitude of the unexplained removal in the activated sludge system is in agreement with the nitrification which may be expected in a system with seeding of nitrifiers from a tertiary nitrifying unit to an activated sludge system with a low aerated solids retention time (1-1.5 days). The removal of nitrogen this way would be in the order of 0.5 to 2.5 g N/m^3. This however is in conflict with the previous findings, where a model based on the assumption that there was no nitrification in the activated sludge system gave a tolerably good estimate of the average effluent ammonium concentration.

7 Recommendations for future research

The work described here together with published data confirms that relatively high denitrification rates can be obtained in a non-nitrifying activated sludge system. However it is obvious that in a system where secondary settled wastewater is recirculated to the activated sludge system, influent wastewater flow and quality influences the effluent wastewater quality and should be accounted for when designing and operating such a system. This has been illustrated using simple steady state models on a distribution of influent data. In order to further clarify these effects detailed mechanistic models could be used, while still performing the calculation for a range of representative data.

The described system is extremely flow dependent. However the effect of flow, and flow dependent wastewater quality aspects, should be considered carefully also when planning and operating other types of wastewater treatment systems. If the mechanisms influencing wastewater treatment are to be understood fully it is important to understand the role of the separate effects of flow, operation, wastewater quality and temperature and the combined effects of these factors on wastewater treatment.

The system has been shown to be a potentially useful system when partial nitrogen removal is required and where the wastewater flow and quality variation is within a certain range. It would be of interest to expand the conclusions in order to cover other flow patterns and effluent standards. A comparison could be made with other systems where the "niche" of each system in terms of economy, effluent standards, wastewater quality etc. was outlined.

The principle of the system, nitrification and denitrification in separate cultures in order to obtain one environment with a low organic loading suitable for nitrification and one with a high organic loading suitable for denitrification, could lend itself well to continuous or discontinuous biofilm systems. In such systems the main problems of the system, separation of organic matter and recirculation of oxygen may be minimised.

8 References

Abeling, U. and Seyfried, C.F. (1992) Anaerobic-aerobic treatment of high-strength ammonium wastewater - nitrogen removal via nitrite. *Wat. Sci. Tech.*, Vol. 26, No. 5-6, pp. 1007-1015.

Abufayed, A. A. and Schroeder, E. D. (1986 A) Performance of SBR/denitrification with a primary sludge carbon source. *Journal WPCF*, Vol. 58, No. 5, pp. 387-397.

Abufayed, A. A. and Schroeder, E. D. (1986 B) Kinetics and stoichiometry of SBR/denitrification with a primary sludge carbon source. *Journal WPCF*, Vol. 58, No. 5, pp. 398-405.

Akunna, J.C., Bizeau, C. and Moletta, R. (1993) Nitrate and nitrite reductions with anaerobic sludge using various carbon sources: glucose, glycerol, acetic acid, lactic acid and methanol. *Water Research*, Vol. 27, No. 8, pp. 1303-1312.

Andersson, B. (1990) Tentative nitrogen removal with fixed bed processes in Malmö sewage treatment plant. *Wat. Sci. Tech.*, Vol. 22, No. 1/2, pp. 239-250.

Andersson, B., Aspegren, H., Parker, D. S. and Lutz, M. P. (1994) High rate nitrifying trickling filters. *Wat. Sci. Tech.*, Vol. 29, No. 10-11, pp. 47-52.

Argaman, Y. (1986) Nitrogen removal in a semi-continuous process. *Water Research*, Vol. 20, No. 2, pp. 173-183.

Argaman, Y. and Brenner, A. (1986) Single-sludge nitrogen removal: modeling and experimental results. *Journal WPCF*, Vol. 58, No. 8, pp. 853-860.

Aspegren, H. (1995) Evaluation of a High Loaded Activated Sludge Process for Biological Phosphorus Removal, PhD thesis at the *Department of Water and Environmental Engineering, Lund Institute of Technology*, Sweden.

Balakrishnan, S. and Eckenfelder, W. W. (1969 A) Nitrogen relationships in biological treatment processes-I. Nitrification in the activated sludge process. *Water Research*, Vol.3, pp. 73-81.

Balakrishnan, S. and Eckenfelder, W. W. (1969 B) Nitrogen relationships in biological treatment processes-II. Nitrification in trickling filters. *Water Research*, Vol.3, pp. 167-174.

Balakrishnan, S. and Eckenfelder, W. W. (1969 C) Nitrogen relationships in biological treatment processes-III. Denitrification in the modified activated sludge process. *Water Research*, Vol. 3, pp 177-188.

Balakrishnan, S. and Eckenfelder, W. W. (1970) Nitrogen removal by modified activated sludge process. *Journal of the Sanitary Engineering Division, Proceedings of ASCE*, April 1970, pp. 501-512.

Balmér, P., Ekfjorden, L., Lumley, D. and Mattsson, A. (1997) Upgrading for nitrogen removal under severe site restrictions. Presented at the IAWQ conference: *Upgrading of water and waste water systems*, Kalmar, Sweden, May 25-28, 1997.

Balmér, P. and Tagizadeh-Nasser, M. (1995) Oxygen transfer in gravity flow sewers. *Wat. Sci. Tech.,* Vol. 31, No. 7, pp. 127-135.

Barnard, J. L. (1973) Biological denitrification. *Wat. Pollut. Control,* 6,72 (1973), pp. 705-720.

Barth, E. F., Mulbarger, M., Salotto, B. V. and Ettinger, M. B. (1966) Removal of nitrogen by municipal wastewater treatment plants. *Journal WPCF*, Vol. 38, No. 7, pp. 1208-1219.

Bever, J., Boll, R., v. d. Emde, Gujer, W., Huber, L., Imhoff, K., Kalbskopf, K. -H., Kayser, R., Krauth, K., Lemmer, H., Mann, Th, Matche, N., Sarfert, F., Schleypen, P. Teichgräber, B., Zibinski, E. and Zeiss, V. (1994 A) Umgestaltung zweistufig biologischer Kläranlagen zur Stickstoffelimination. Arbeitsbericht des ATV-Fachausschusses 2.6 "Aerobe biologische Abwasserreinungsverfahren" ATV-Arbeitsbereichte, *Korrespondenz Abwasser*, Vol. 41, No. 1, pp. 95-100.

Bever, Bode, Dorias, Schleigel, Schwentner and Wolf. (1994 B) Denitrifikation bei Tropfkörperanlagen. Arbeitsbericht der ATV-Arbeitsgruppe 2.6.3 "Tropf- und Tauchkörper" im ATV-Fachausschuß 2.6 "Aerobe biologische Abwasserreinungsverfahren" ATV-Arbeitsbereichte, *Korrespondenz Abwasser*, Vol. 49, No. 11, pp. 2077-2081.

Bishop, H. D., Heidman, J. A. and Stamberg, J. B. (1976) Single-Stage nitrification-denitrification. *Journal WPCF*, Vol. 48, No. 3, pp. 520-532.

Blaszczyk, M., Przytocka-Jusiak, M., Kruszewska, U. and Mycielski, R. (1981) Denitrification of nitrites and nitrates in synthetic medium with different sources of organic carbon, I. Acetic acid. *Acta Microbiologica Polonica*, Vol. 30, No. 1, pp. 49-58.

Bode, H., Seyfried, C. F. and Kraft, A. (1987) High-rate denitrification of concentrated nitrate wastewater. *Wat. Sci. Tech.*, Vol. 19, Rio, pp. 163-174.

Boller, M. and Gujer, W. (1986) Nitrification in tertiary trickling filters followed by deep-bed filters. *Water Research*, Vol. 20, No. 11, pp. 1363-1373.

Boller, M., Gujer, W. and Tschui, M. (1994) Parameters affecting nitrifying biofilm reactors, *Wat. Sci. Tech.*, No. 10-11, pp. 1-11.

Bryan, B. A. (1981) Physiology and biochemistry of denitrification. Denitrification, nitrification and atmospheric nitrous oxide. *C. C. Delwiche (ed)Wiley & Sons Ltd.*, New York.

Buß, S., Einfeldt, J., Günter, H. and Werner, T. (1994) Upgrading of wastewater treatment plants to achieve advanced standards concerning nutrient removal. *Wat. Sci. Tech.*, Vol. 29, No. 12, pp. 49-58.

Bundgaard, E., Andersen, K.L. and Petersen, G.(1989) Bio-Denitro and Bio-Denipho systems - Experiences and advanced model development: The Danish systems for biological N and P removal. *Wat. Sci. Tech.*, Vol. 21, Brighton, pp. 1727-1730.

Butler, D., Xiau, Y. Karunaratne, S. H. P. G. and Thedchanamoorthy, S. (1995) The gully pot as a physical chemical and biological reactor. *Wat. Sci. Tech.*, Vol. 31, No. 7, pp. 13-24.

Carlsson, H., Aspegren, H. And Hilmer, H. (1996) Interactions between wastewater quality and phosphorus release in the anaerobic reactor of the EBPR process. *Water Research*, Vol. 30, No. 6, pp. 1517-1527.

Carrand, G., Capon, B., Rasconi, A. and Brenner, R. (1990) Elimination of carbonaceous and nitrogenous pollutants by a twin-stage fixed growth process. *Wat. Sci. Tech.*, Vol. 22, No. 1/2, pp. 261-272.

Çeçen, F. and Gönenç, I. E.(1995) Criteria for nitrification and denitrification of high-strength wastes in two upflow submerged filters. *Water Environment Research*, Vol. 67, No. 2, pp. 132-142.

Christensen, K. (1991) Extension of a large trickling filter plant for combined nitrogen and phosphorus removal (Ejby Molle wastewater treatment plant, Odense, Denmark). *Wat. Sci. Tech.*, Vol. 23, Kyoto, pp. 791-800.

Christensen, M. Henze and Harremoës, P. (1977) Biological denitrification of sewage: a literature review. *Prog. Wat. Tech.*, Vol. 8, Nos. 4/5, pp 509-555.

Christensen, M. Henze and Harremoës, P. (1978) Nitrification and denitrification in wastewater treatment. *Water Pollution Microbiology*, Vol. 2, pp. 391-414. (R. Mitchell, Ed.) John Wiley & Sons, New York.

Christensson, M., Jönsson, K., Lee, N., Johansson, P., Welander, T. and Østgaard, K. (1995) Evaluation of Biological Phosphorus Removal in full Scale at Öresundsverket in Helsingborg. (In Swedish - summary in English) *VA-Forsk*, Rapport nr 1995-04, ISSN; 1102-5638, ISBN; 91-88392-59-7.

Clayfield, G. W. (1974) Respiration and denitrification studies on laboratory and works activated sludges. *Wat. Pollut. Control* (1974), pp. 51-76.

Clifford, D. and Liu, X. (1993) Biological denitrification of spent regenerant brine using a sequencing batch reactor. *Water Research*, Vol. 27, No. 9, pp. 1477-1484.

la Cour Jansen, J., Jepsen, S-E. and Dahlgren Laursen, K. (1994) Carbon utilization in denitrifying biofilters. *Wat. Sci.Tech.*, Vol. 29, No. 10-11, pp. 101-109.

Crites, R. W. (1994) Design criteria and practice for constructed wetlands. *Wat. Sci. Tech.*, Vol. 29, No. 4, pp. 1-6.

Dahlem, H. W. (1986) Moderne mehrstufige verfahrenskombinationen bei der kommunalen und industriellen abwasserbehandlung. *Schriftenreihe WAR* 25, pp 121 - 160, Technischen Hochschule Darmstadt, ISSN-0721-5282.

Daigger, G. T., Norton, L. E., Watson, R. S., Crawford, D. and Sieger, R. B. (1993) Process and kinetic analysis of nitrification in coupled trickling filter / activated sludge processes. *Water Environment Research*, Vol. 65, No. 6, pp. 750-758.

Daigger, G. T., Norton, L. E., Watson, R. S., Crawford, D. and Sieger, R. B. (1995) Closure to discussion of: "Process and kinetic analysis of nitrification in coupled trickling filter / activated sludge processes, 65, 750 ((1993) Discussion by Parker, D. and Richards, J. T., 66, 934 (1994). *Water Environment Research*, Vol. 67, No. 3, pp. 380 - 382.

Dawson, R. N. and Murphy, K. L. (1972) The temperature dependency of biological denitrification. *Water Research,* Vol. 6, pp. 71-83.

Dawson, R. N. and Murphy, K. L. (1973) Factors affecting biological denitrification of wastewater. *Advances in water pollution research.* Proceedings of Jerusalem conference. Pergamon Press, Oxford, pp. 671-680.

Dee, A., James, N., Strickland, J., Upton, J. and Cooper, P. (1994) Pre- or post-denitrification at biological filter works? A case Study. *Wat. Sci. Tech.*, Vol. 29, No. 10-11, pp. 145-155.

Dichtl, N., Engelhardt, N., Firk, W. and Koppetsch, J. (1994) Conventional and unconventional integration of trickling filters in a process for biological nutrient removal. *Wat. Sci. Tech.*, Vol. 29, No. 12, pp. 81-88.

Downing, A. L., Painter, H. A. and Knowles, G. (1964) Nitrification in the activated sludge system. *J. Inst. Sew. Purif.,* 1964, pp. 130-153.

Engberg, D. J. and Schroeder, E. D. (1975) Kinetics and stoichiometry of bacterial denitrification as a function of cell residence time. *Water Research*, Vol. 9, pp. 1051-1054.

EPA (1993) Manual, Nitrogen Control, *United States Environmental Protection Agency*, EPA/625/R-93/010.

Farmer, G. and Caudill, C. (1995) Trickle down theories - meeting stringent effluent limits for ammonia requires innovative trickling filter and process adjustments. *Operations Forum*, April 1995, pp. 20-23.

Fleischer, S. and Stibe, L. (1991) Drainage basin management - reducing river transported nitrogen. *Verh. Internat. Verein. Limnol.*, 24, pp. 1753-1755.

Fleischer, S., Stibe, L and Leonardson.(1991) restoration of wetlands as a means of reducing nitrogen transport to coastal waters. *AMBIO*, Vol. 20, No. 6, Sept. 1991, pp. 271-272.

Francis, C. W. and Callahan, M. W. (1975) Biological denitrification and its' application in treatment of high-nitrate waste water. *J. Environ. Qual.*, Vol. 4, no. 2, pp. 153-163.

Francis, C. W. and Mankin, J. B. (1977) High nitrate denitrification in continuous flow-stirred reactors. *Water Research*, Vol. 11, pp. 289-294.

Friedrich, J., Klein-Schnell, H. -P., Roßwag, P. and Schmitt, R.(1995) Kostenbesparungen auf Abwasserreinigungsanlagen durch Einsatz externer Kohlenstoffquellen bei der Denitrifikation. *Korrespondenz Abwasser,* Vol. 42, No. 2, pp. 248-256.

Fruhen, M., Böcker, K., Eidens, S., Haaf, D., Liebeskind, M. and Schmidt, F. (1994 A) Tertiary nitrification in pilot-plant plug-flow fixed-film reactors with long-term ammonium deficiency. *Wat. Sci. Tech.*, Vol. 29, No. 10-11, pp 61-67.

Fruhen, M., Kühn, W. and Dohmann, M. (1994 b) Upgrading of a wastewater treatment plant utilizing existing trickling filters and a new filter stage. *Wat. Sci. Tech.*, Vol. 29, No. 12, pp. 59-67.

Gall, B., Takács, I. and Patry, G. (1995) The effect of organic reactions in a collection system on wastewater treatment plant performance. *Wat. Sci. Tech.*, Vol. 31, No. 7, pp. 25-31.

Gamperer, T. (1997) Enhanced 2-stage activated sludge. *World Water and Environmental Engineering,* Vol. 20, No. 3, pp. 20-21.

Grady, C. P. L., Gujer, W., Henze, M., Marais, G. v. R. and Matsuo, T. (1986) A model for single-sludge wastewater treatment. *Wat. Sci. Tec*h., Vol. 18, No. 6, Copenhagen, pp. 47-61.

Grau, P., Sutton P. M., Henze M., Elmach, S., Grady, C. P., Gujer, W. and Koller, J (1987) Notation for use in the description of wastewater treatment processes. *Water Research,* Vol. 21, No. 2, pp. 135-139.

Gujer, W. and Boller, M. (1983) Operating experience with plastic media tertiary trickling filters for nitrification. Workshop on "Design and operation of large wastewater treatment plants" *IAWPRC,* September 19-23, 1993, Vienna, Austria

Gujer, W. and Jenkins, D. (1975) A nitrification model for the contact stabilization activated sludge process. *Water Research,* Vol. 9, pp. 561-566.

Hagedorn-Olsen, C., Møller, I. H., Tøttrup, H. and Harremoës, P. (1994) Oxygen reduces denitrification in biofilm reactors. *Wat. Sci. Tech.*, Vol. 29, No. 10-11, pp. 83-91.

Hallin, S., Rothman, M. And Pell, M. (1996) Adaptation of denitrifying bacteria to acetate and methanol in activated sludge. *Water Research,* Vol. 30, pp. 1445-1450.

Halmø , G. and Eimhjellen, K. (1981) Low temperature removal of nitrate by bacterial denitrification. *Water Research.*, Vol. 15, pp. 989-998.

Hammer, D. A. and Knight, R. L. (1994) Designing constructed wetlands for nitrogen removal. *Wat. Sci. Tech.*, Vol. 29, No. 4, pp. 15-27.

Hamon, M. and Fustec, E.(1991) Laboratory and field study of an insitu groundwater denitrification reactor. *Research Journal WPCF,* Vol. 63, No. 7, pp. 942-949.

Hansson, E. (1994) Nitrifikation i biobädd - dess beroende av varierande flöde och ammoniumbelastning. *GRYAAB* Rapport 1994:2.(In Swedish)

Harremoës, P and Christensen, M. H.(1971) Denitrifikation med methan, *Vand* 1, Februar, pp. 7-11.(In Danish)

Henze, M (1986) Nitrate versus oxygen utilization rates in wastewater and activated sludge systems. *Wat. Sci. Tech.,* Vol. 18, Copenhagen, pp. 115-122.

Henze, M. (1987) Theories for estimation of the fraction of denitrifiers in combined nitrifying-denitrifying treatment plants. *Water Research,* Vol. 21, pp. 1521-1524.

Henze, M. (1989) The influence of raw wastewater biomass on activated sludge oxygen respiration rates and denitrification rates. *Wat. Sci. Tech.*, Vol. 21, Brighton, pp. 603-607.

Henze, M. (1991) Capabilities of biological nitrogen removal processes from wastewater. *Wat. Sci. Tech.*, Vol. 23, Kyoto, pp. 669-679.

Henze, M. and Bundgaard, E. (1982) Bemessung von kombinierten Nitrifikations- und Denitrifikationsanlagen. *gwf-wasser/abwasser,* Vol. 123, No. 5, pp. 240-246.

Henze, M., Grady, C. P. L. jr, Gujer, W., Marais, C.v.R. and Matuo, T. (1987) Activated sludge model No.1. Scientific and Technological Report No. 1 IAWPRC, London.

Henze, M. and Mladenovski, C.(1991) Hydrolysis of particulate substrate by activated sludge under aerobic, anoxic and anaerobic conditions. *Water Research*, Vol. 25, No. 1, pp. 61-64.

Hiscock, K. M., Lloyd, J. W. and Lerner, D. N. (1991) Review of natural and artificial denitrification of groundwater. *Water Research*, Vol. 25, No. 9, pp. 1099-1111.

Hultman, B., Jönsson, K. and Plaza E. (1994) Combined nitrogen and phosphorus removal in a full-scale continous up-flow sand filter. *Wat. Sci. Tech*, Vol. 29, No. 10-11, pp. 127-134.

Ingraham, J, L. (1981) Microbiology and genetics of denitrifiers. Denitrification, nitrification and atmospheric nitrous oxide. C. C. Delwiche (ed)Wiley & Sons Ltd. New York.

Isaacs, S. H., Henze, M., Søberg, H. and Kümmel, M. (1994) External carbon source as a means to control an activated sludge nutrient removal process. *Water Research*, Vol. 28, No. 3, pp. 511-520.

Jenkins, D. and Orhon, D. (1972) The mechanism and design of the contact stabilization activated sludge process. *Advances in Water Pollution Research*, Proceedings of the Sixth International Conference held in Jerusalem, June 18-23 1972, Edited by S. H. Jenkins, Pergamon press, pp. 353-365.

Jensen, N. A. (1995) Empirical modelling of air-to water oxygen transfer in gravity sewers. *Water Environment Research*, Vol. 67, No. 6, pp. 979-991.

Johnson, W. K. and Scroepfer, G. J. (1964) Nitrogen removal by nitrification and denitrification. *J. Water Pollut. Control Fed.*, Vol. 36, pp. 1015-1036.

Jones, P. H. and Sabra, H. M. (1980) Effect of systems solids retention time (SSRT or sludge age) on nitrogen removal from activated sludge systems. *Wat. Pollut. Control* (1980) pp. 106-116.

Jones, W.L., Schroeder, E. D. and Wilderer, P. A. (1990 A) Denitrification in a batch wastewater treatment system using sequestered organic substances. *Research Journal WPCF,* Vol. 62, No. 3, pp. 259-267.

Jones, W.L., Wilderer, P. A., Schroeder, E. D. (1990 B) Operation of a three-stage SBR system for nitrogen removal from wastewater. *Research Journal WPCF*, Vol. 62, No. 3, pp. 268-274.

Karlberg, H. and Eriksson, P. (1989) Syresättningshastighet i de anoxiska zonerna 890731. *(Internal report at GRYAAB in Swedish).*

Karlsson, I. and Smith, G. (1991) Pre-precipitation facilitates nitrogen removal without tank expansion. *Wat. Sci. Tech.*, Vol. 23, Kyoto, pp. 811-817.

Knowles, G., Downing, A. L. and Barrett, M. J. (1965) Determination of kinetic constants for nitrifying bacteria in mixed culture, with the aid of an electronic computer. *J. gen. Microb.*, Vol. 38, pp. 263-278.

Knowles, R (1982) Denitrification. *Microbiological Reviews*, Mar. 1982, pp. 43-70.

Koike, I. and Sørensen, J. (1988) Nitrate Reduction and Denitrification in Marine Sediments. *Nitrogen Cycling in Coastal Marine environments*. Blackburn, T.H. and Sørensen, J.(eds.) Wiley & Sons Ltd.

Krauth, Kh, and Roth, M. (1991) Betreibsergebnisse von anlagen zur Stickstiff- und Phosphorelimination.Kläranlagen zur Stickstoffelimination und Garantien für bel¨ftung, Herausgeber: Rolf Kayser, *Veröffentlichungen des Instituts für Siedlungswasserwirtschaft Technische Univesität Braunschweig*, Heft 50, pp. 195-229.

Krul, J. M. (1976) The relationship between dissimilatory nitrate reduction and oxygen uptake by cells of an *Alcaligenes* strain in flocs and in suspension and by activated sludge flocs. *Water Research,* Vol. 10, pp. 337-341.

Leonardson, L. (1994) Våtmarker som kvävefällor, Svenska och internationella erfarenheter/Wetlands as nitrogen sinks, Swedish and international experience. *Naturvårdsverket*, Rapport 4176 (In Swedish - 20 page summary in English).

Lewandowski, Z. (1982) Temperature dependency of biological denitrification with organic materials addition. *Water Research,* Vol. 16, pp. 19-22.

Lie, E. and Welander, T. (1994) Influence of dissolved oxygen and oxidation-reduction potential on the denitrification rate of activated sludge. *Wat. Sci. Tech.*, Vol. 30, No. 6, pp. 91-100.

Lo, C. K., Yu, C. W., Tam, N. F. Y. and Traynor, S. (1994) Enhanced nutrient removal by oxidation-reduction potential (ORP) controlled aeration in a laboratory scale extended aeration treatment system. *Water Research,* Vol. 28, No. 10, pp. 2087-2094.

Lumley, D. J. and Balmér, P. (1994) Rya tar ett kliv upp för att nå längre; Kapacitetsökning av Ryaverkets sedimenteringsbassänger (in Swedish). Presented at: Eftersedimentering och filtrering vid reningsverk, Stockholm, Sweden, August 9, 1994.(in Swedish)

Lyngå, A. (1991) Nitrifikation och denitrifikation i separata steg- ett konkurrenskraftigt alternativ för kväveavskiljning? Separate stage nitrification and denitrification-a viable alternative for nitrogen removal? *Vatten*, Vol. 47, No. 3, pp. 241-256. (in Swedish)

Lyngå, A. and Balmér, P. (1992) Denitrification in a non-nitrifying activated sludge system. *Wat. Sci. Tech.*, Vol. 26, No. 5-6, pp. 1097-1104.

Maisch, H., Schwentner, G. (1994) Umrüstung einer tropfkörperanlage zur stickstoffelimination am Beispiel des Klärwerkes Sindelfingen. *Korrespondenz Abwasser.*, Vol. 41, No. 9, pp. 1564-1578.

Matějů, V., Čižinská, S., Krejčí and Janoch, T. (1992) Biological water denitrification - a review. *Enzyme Microb. Technol.*, Vol. 14, March, pp. 170-183.

Mattsson, A. (1997) Pilot plant experiemnts used for design of acivated sludge systems-some possibilities and pitfalls. *Vatten*, Vol. 53, No. 1, pp. 57-63.

Mattsson, A. and Rane, A. (1993) Nitrifikation i biobädd - Försök på Ryaverket december 1990- augusti 1992. *GRYAAB* Rapport 1993:1.(in Swedish)

Meaney, B. J. and Strickland, J. E. T.(1994) Operating experiences with submerged filters for nitrification and denitrification. *Wat. Sci. Tech.*, Vol. 29, No. 10-11, pp. 119-125.

Metcalf & Eddy (1991) Wastewater engineering, treatment, disposal and reuse. Revised by Tchobanoglous, G and Burton, F. L. McGrae-Hill, Inc. ISBN 0-07-100824-1.

Monteith, H. D., Bridle, T. R. and Sutton, P. M. (1980) Industrial waste carbon sources for biological denitrification. *Prog. Wat. Tech.* Vol.12, Toronto, pp. 127-141.

Moore, S. F. and Schroeder, E. D.(1970) An investigation of the effects of residence time on anaeribic bacterial denitrification. *Water Research* Vol. 4, pp. 685-695.

Morrison, G. M., Revitt, D. M. and Ellis, J. B. (1995) The gully pot as a biochemical reactor. *Wat. Sci. Tec.* Vol. 31, No. 7, pp. 229-236.

Mulbarger, M. C. (1971) Nitrification and denitrification in activated sludge systems. *Journal WPCF*, October 1971, pp. 2059-2070.

v Münch, E., Lant, P. and Keller, J. (1996) Simultaneous nitrification and denitrification in bench-scale sequencing batch reactors. *Water Research*, Vol. 30, No. 2, pp. 277-284.

Narkis, N., Rebhun, M. and Sheindorf, Ch. (1979) Denitrification at various carbon to nitrogen ratios. *Water Research,* Vol. 13, pp 93-98.

Naturvårdsverket (1994). Lustgasutsläpp från kommunala reningsverk, mätningar vid tolv svenska reningsverk. Rapport 4309. (In Swedish)

Nielsen, P. H. (1996 A) Adsorption of ammonium to activated sludge (Rapid communication), *Water Research*, Vol. 30, No. 3, pp. 762-764.

Nielsen, P.H. (1996 B) The significance of microbial Fe(II) reduction in the activated sludge process. *Wat. Sci. Tech.*, Vol. 34, No. 5-6, pp. 129-136.

Nurse, G. R. (1980) Denitrification with methanol: microbiology and chemistry. *Water Research,* Vol. 14, pp. 531-537.

Nyberg, U., Aspegren, H., Andersson, B., la Cour. Jansen, J., Villadsen, I. S. (1992) Full-scale application of nitrogen removal with methanol as carbon source. *Wat. Sci. Tech.,* Vol. 26, No. 5-6, pp. 1077-1086.

Okey, R. W. and Albertson, O. E. (1989) Diffusion's role in regulating rate and masking temperature effects in fixed-film nitrification. *Journal WPCF*, Vol. 61, No. 4, pp. 500-509.

Oleszkiewicz, J. A. and Bergquist, S. A. (1988) Low temperature nitrogen removal in sequencing barch reactors. *Water Research.,* Vol. 22, No. 9, pp. 1163-1171.

van Oostrom, A. J. (1995) Nitrogen removal in constructed wetlands treating nitrified meat processing effluent. *Wat. Sci. Tech.* Vol. 32, No. 3, pp. 137-147.

Özer, A. and Kazimer, E. (1995) Substrate removal in long sewer lines. *Wat. Sci. Tech.*, Vol. 31, No. 7, pp. 213-218.

Paffoni, C., Védry, B. and Gousailles, M. (1990) Tertiary nitrification pilot plants on Parisian waste water. *Wat. Sci. Tech.*, Vol. 22, No. 1/2, pp. 347-352.

Palmgren, T. (1992) Nitrogen reduction - Volume demand. *Wat. Sci. Tech.,* Vol. 25, No. 4-5, pp. 233-240.

Parker, D., Lutz, M., Andersson, B. and Aspegren, H. (1995) Effect of operational variables on nitrification rates in trickling filters. *Water Environment Research*, Vol. 67, No. 7, pp. 1111-1118.

Parker, D., Lutz, M., Dahl, R. And Berkopf, S.(1989) Enhancing reaction rates in nitrifying trickling filters through biofilm control. *Journal WPCF,* Vol. 61, No. 5, pp. 618-631.

Parker, D. S., Lutz, M. P. and Pratt, A. M. (1990) New trickling filter applications in the U.S.A. *Wat. Sci. Tech.*, Vol. 22, No. 1/2, pp. 215-226.

Parker, D. and Richards, T. (1986) Nitrification in trickling filters. Journal WPCF, Vol. 58, pp. 896-902.

Parker, D. and Richards, J. T. (1994) Discussion of: Process and kinetic analysis of nitrification in coupled trickling filter activated sludge systems. Daigger et al., 65, 750 (1993). *Water Environment Research*, Vol. 66, No. 7, pp. 934 - 935.

Patruno, J. and Russell, J. (1994) Natural wetland polishing effluent discharging to Wooloweyah lagoon. *Wat. Sci. Tech.,* Vol. 29, No. 4, pp. 185-192.

Rehmann, W. (1993) Stickstoffelimination-Betriebserfahrungen mit der Regelung nach dem Redoxverfahren. *Korrespondenz Abwasser*-Betriebs-Info, Vol. 23, No. 3, pp. 495-497.

Reimann, H. (1990) The Linpor-process for nitrification and denitrification. *Wat. Sci. Tech.,* Vol. 22, No. 7/8, pp. 297-298.

Schleypen, P. and Nordmann, W. (1994) Stickstoff- und Phosphorelimination an einer zweistufigen belebungs-Tropfkörperanlage. *Korrespondenz Abwasser*, Vol. 41, No. 12, pp. 2241-2249.

Schreff, D. And Wilderer, P. A. (1997) Nitrogen-removal in multi-stage wastewater treatment plants by using a modified post-denitrification system. Presented at the IAWQ conference: *Upgrading of water and waste water systems*, Kalmar, Sweden, May 25-28, 1997.

Schulthess, R. v. and Gujer, W. (1996) Release of nitrous oxide (N2O) from denitrifying activated sludge: verification and application of a mathematical model. *Water Research*, Vol. 30, No. 3, pp. 521-530.

Seitzinger, S. P. (1988) Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. *Limnol. Oceanogr.*, 33 (4, part 2), 1988, pp. 702-724.

Sekoulov, I., Addiks, R. and Oles, J. (1990) Post-denitrification with controlled feeding of activated sludge as H donor. *Wat. Sci. Tech.* Vol. 22, No. 7/8, pp. 161-170.

Simpkin, T. J. and Boyle, W. C. (1988) The lack of repression by oxygen of the denitrifying enzymes in activated sludge. *Water Research,* Vol. 22, No. 2, pp. 201-206.

Sison, N. F., Hanaki, K. and Matsuo, T. (1995) High loading denitrification by biological activated carbon process (Research Note). *Water Research*, Vol. 29, No. 12, pp. 2776-2779.

Socher, M. and Gläser, K. (1992) Untersuchungen zur heterotrophen Denitrifikation. *gwf-wasser/abwasser*, Vol.133, No. 2, pp 92-98.

Sutton, P. M., Murphy, K. L. and Dawson, R. N. (1975) Low-temperature biological denitrification of wastewater. *Journal WPCF*, Vol. 47, No. 1, pp. 122-134.

Sutton, P. M., Murphy, K. L. and Jank, B. E.(1978) Kinetic studies of single sludge nitrogen removal systems. *Prog. Wat. Tech.,* Vol. 10, Nos. 1/2, pp. 241-253.

Swedish Standard - Refers to methods issued by the Swedish Standards Institution, SIS, Box 6455, 113 82 Stockholm, SWEDEN. (in Swedish)

Thörn, M. and Sörensson, F. (1996) Variation of nitrous oxide formation in the denitrification basin in a wastewater treatment plant with nitrogen removal. *Water Research*, Vol. 30, No.6, pp. 1543-1547.

Timmermans, P. and Van Haute, A. (1983) Denitrification with methanol; Fundamental study of the growth and denitrification capacity of Hyphomicrobium sp. *Water Research*, Vol. 17, No. 10, pp. 1249-1255.

Tschui, M., Boller, M., Gujer, W., Eugster, J., Mäder, C. and Stengel, C. (1994) Tertiery nitrification in aerated pilot biofilters. *Wat. Sci. Tech.,* Vol. 29, No. 10-11, pp. 53-60.

Van Haandel, A. C., Ekama, G. A. and Marais, G. v. R. (1981) The activated sludge process - 3. Single sludge denitrification. *Water Research*, Vol. 15, pp. 1135-1152.

Wanner, J., Cech, J. S. and Kos, M. (1992) New process design for biological nutrient removal. *Wat. Sci. Tech.*, Vol. 25, No. 4-5, pp. 445-448.

Watanabe, Y., Bang, D. Y., Itoh, K. and Matsui, K. (1994) Nitrogen removal from wastewaters by a bio-reactor with partially and fully submerged rotating biofilms. *Wat. Sci. Tech.* Vol. 29, No. 10-11, pp. 431-438.

Weddle, C. L. and Jenkins, D. (1971) The viability and activity of activated sludge, *Water Research*, Vol. 5, pp. 621-640.

Wik, T., Mattsson, A., Hansson, E. and Niklasson, C. (1995) Nitrification and denitrification in a tertiary trickling filter at high hydraulic loads - pilot plant operation and mathematical modelling. *Wat. Sci. Tech.*, Vol. 32, No. 8, pp. 185-192.

Ødegaard, H (1995) An evaluation of cost efficiency and sustainability of different wastewater treatment processes. *Vatten*, Vol. 51, No. 4, pp. 291-299.

Ødegaard, H., Rusten, B. and Westrum, T. (1994) A new moving bed biofilm reactor - applications and results. *Wat. Sci. Tech.*, Vol.29, No. 10-11, pp. 157-165.

Appendix A The Rya WWTP

At the Rya WWTP the question of nitrogen removal was raised around 1989 as the result of a series of events which set the local scene. A description of the WWTP and its surroundings are to be seen in the context of some events in the environment and some social and political events as summarised in Table A.1.

The Rya WWTP received wastewater from about 755 000 population equivalents in six municipalities in the Göteborg region on the west coast of Sweden. The treatment plant and main tunnel system are owned by GRYAAB (Göteborgsregionens Ryaverks Aktiebolag), a company owned by the seven municipalities of Ale, Göteborg, Härryda, Kungälv, Lerum, Mölndal and Partille. Local sewers and tunnels are owned and maintained by each municipality. The catchment area is large, causing transport times in the range of 0 to 30 hours. Collecting systems range from combined systems where storm-water and wastewater are transported in the same pipes, to separate systems where storm-water is disposed of directly. On average the Rya WWTP receives about 4 m^3/s of wastewater, consisting of 34 % domestic wastewater, 12 % industrial wastewater and the remainder storm-water, house foundation drainage or groundwater, reaching the collecting system through infiltration or direct inflow. The large proportion of rainwater and groundwater causes the total influent flow to vary considerably (see Figure A.1). The average flow of the rainy year of 1988 (1002 mm) was 4.3 m^3/s, whereas in 1996 (636 mm) the average flow was only 3.3 m^3/s.

Before extension for nitrogen removal the Rya WWTP was a conventional, non-nitrifying, activated sludge plant with simultaneous precipitation of phosphorus with iron sulphate (Figure A.4 a). Transported by gravity tunnels the wastewater reached the treatment plant 19 m below ground. After screening through 20 mm bar screens the wastewater was pumped to ground level. Primary sedimentation took place in rectangular sedimentation tanks with a total surface area of 5800 m^2 and an average surface loading of 2.4 m/h. All wastewater underwent primary treatment but, at flows exceeding 6 m^3/s, the excess was overflowed prior to secondary treatment. The annual overflow of primary settled wastewater was 3 to 8 % of the total flow.

The activated sludge system consisted of 31 parallel aeration tanks with a total volume of 39 000 m^3 and 24 final sedimentation tanks with a total surface area of 11 300 m^2, yielding an average surface loading of 1.3 m/h and a maximum surface loading (at 6 m^3/s) of 1.9 m/h. The plant could be operated in contact stabilisation mode where return activated sludge was aerated in 8 000 m^3 of the aeration volume and the remaining volume used for aerated contact with wastewater. With a total solids retention time of 3 to 4 days nitrification was seldom observed.

Table A.1 Some events influencing the progress towards nitrogen removal at the Rya WWTP.

When?	Who?	What?
1980:s	Algae, fish, oxygen	Fish found dead on beaches, poisonous algae make shellfish inedible, oxygen depletion in sediments off the coast.
1987	The second North Sea Conference	The North Sea States are urged to take effective national action against the discharges of nitrogen and phosphorus, aiming at reduction of 50 % between 1985 and 1995. (the Nordic Council, 1989)
1988	The Swedish Government and Parliament	50 % nitrogen removal prescribed at WWTP:s in sensitive areas (Laholmsbukten, Skälderviken, Öresund and Hanöbukten). (Naturvårdsverket 1996)
1988	Helsinki, HELCOM, Ministers from the countries surrounding the Baltic.	Effluent nitrogen from medium size WWTP:s (10 000 - 100 000 pe) to be reduced to 15 mg N/l and from large WWTP:s (above 100 000) to be 10 mg N/l by 1998. (To the Baltic proper, the Gulf of Bothnia, the Gulf of Finland, the Kattegatt and part of Skagerrak) (Naturvårdsverket 1996)
Summer of 1988	Seals	A large number of seals in the North Sea died and were found on the beaches of the West coast of Sweden.
1988	GRYAAB	Pilot plant experiments with nitrogen removal in a single sludge activated sludge system initiated at the Rya WWTP.
1989	Swedish Environmental Protection Agency.	Standards corresponding to at least 50 % nitrogen removal should be applied at about 70 coastal WWTP:s serving more than 10 000 pe. (Naturvårdsverket 1996)
1989-1993	GRYAAB	Pilot plant experiments with nitrogen removal in a non-nitrifying activated sludge system initiated at the Rya WWTP.
1990	The Swedish Government and Parliament	The aim of reducing nitrogen emissions to the sea by 50 % should apply to all human activities. 50 % nitrogen removal should apply to all coastal WWTP:s (greater than 10 000 pe) from the Norwegian border to the archipelago of Stockholm. For sensitive areas stricter limits should apply. (Naturvårdsverket 1996)
1990	Swedish Environmental Protection Agency	Demands of nitrogen removal (50 %) and phosphorus removal (95 %) to be applied to all coastal (or up to 30 km inland) wastewater treatment plants serving more than 10 000 population equivalents from the Norwegian boarder to Stockholm by 1995.
Early 1990:s	Local opinion	Increasing resistance to exploitation of any part of the forest adjacent to the Rya WWTP
Mid 1991- mid 1995	STU - later NUTEK- Swedish National Board for Industrial and Technical Development	Integrated research between wastewater treatment plants and universities on process control and development. The Rya WWTP takes part together with the universities in Göteborg.
1991	EES	Effluent nitrogen standards from coastal WWTP:s from the Norwegian boarder to Stockholm (classified as sensitive) to be 15 mg N/l for medium size WWTP:s (10 000 - 100 000 pe) and 10 mg N/l from large WWTP:s (above 100 000 pe). (Naturvårdsverket 1996)
1992	GRYAAB	Application filed to the Franchise Board.
1993	GRYAAB	Final design of full scale plant.
1993	Franchise Board	Preliminary permit from Franchise Board.
1993-1994	GRYAAB	Pilot plant operation continued in order to gain operational experience of the chosen process.
1997	GRYAAB	Construction work converting the Rya WWTP for nitrogen removal terminated.
1999	GRYAAB	Application of final limits on effluent nitrogen, phosphorus and BOD_7 to be applied for.
2001	Franchise Board	Final limits expected.

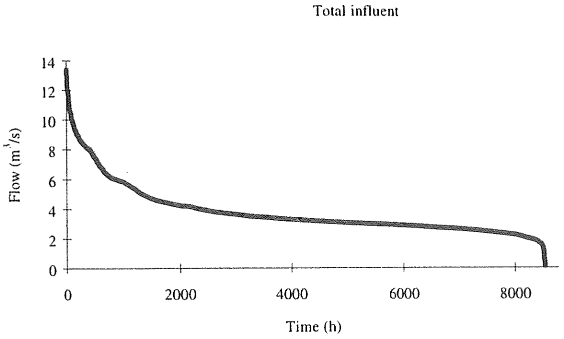


Figure A.1 Distribution of influent flow to the Rya WWTP.

Waste activated sludge, including sludge from phosphorus precipitation, was handled together with primary sludge. First it was screened in a 3 mm moving screen (Step Screen®). After thickening in four circular thickeners (surface loading 0.2 m/h, retention time 15 hours) it was digested at about 37 °C in two anaerobic digestors with a total volume of 22 600 m^3 and a retention time of 12 to 17 days. The digested sludge was either dewatered on site or pumped 6.8 km to off site dewatering units (band filter presses) where it was dewatered to a dry solids content of about 28 %. The dewatered sludge was pumped into a large cavern formerly used for storing oil, or loaded onto trucks and used in agriculture or for landscaping. The treatment results were good, especially considering the loading of the plant, meeting effluent demands well. Effluent standards were set as quarterly averages of effluent phosphorus (0.5 g P/m^3) and biological oxygen demand, BOD_7, (15 g O_2/m^3). The standards were met, normally with some margin, as illustrated in Figure A.2.

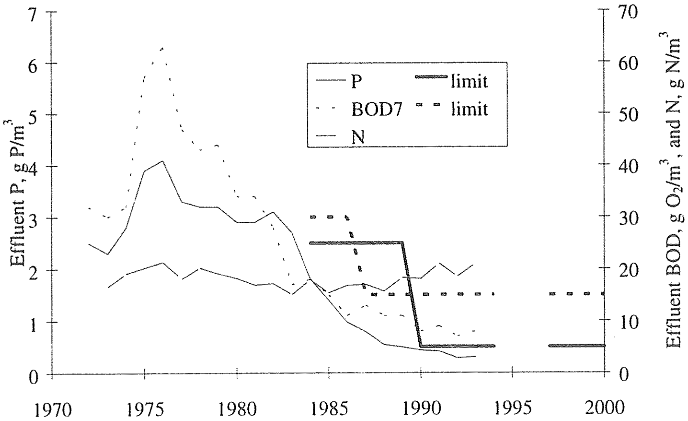


Figure A.2 Effluent wastewater quality, Rya WWTP.

The wastewater treatment plant was originally built with land area for additional treatment units, but with time virtually all on site free area has been filled in (Figure A.3). Constructed 1971 as an extremely highly loaded activated sludge plant, the plant has since been extended with improved sludge handling (1974-1980) primary sedimentation tanks and more activated sludge tanks (1982), chemical precipitation (1984-1990), polymer dosage (1989) and sludge screens and digestion tanks 1990.

The possibilities of expanding the site of the WWTP are limited. To the north the site is limited by an industrial railway line and a road connecting the industrial areas of western Hisingen with Göteborg. To the east industrial sites limit expansion. A forest (partially with the status of nature reserve), to the south and west of the treatment plant would be an unpopular direction of expansion. Originally a section of the forest (without the status of nature reserve) was set aside for expansion of the WWTP, but GRYAAB´s aim is to intrude as little as possible on the natural values of the forest.

Thus with limited ground area on site or in the immediate surroundings the focus was set on compact solutions for expansion for nitrogen removal. The relatively high overflow of primary settled wastewater warranted solutions which also included an increased hydraulic capacity for secondary treatment.

Several alternatives for nitrogen removal have been considered and evaluated for the expansion of the Rya WWTP. The most important alternatives were (see Figure A.4):

- single sludge denitrification and nitrification (pre-denitrification)
- a system including aerated biological filters
- a system where nitrification is carried out in tertiary trickling filters and denitrification in a non-nitrifying activated sludge system

These three solutions for nitrogen removal were considered for extension of the Rya WWTP. All three solutions would be able to meet the limits but economic and areal considerations caused the third alternative, post nitrification in trickling filters and pre-denitrification in a non-nitrifying activated sludge system, to be chosen for extension for nitrogen removal. With this system the entire extension could be kept within the present site boundaries (Figure A.5).

The activated sludge system would be modified to include denitrification but not nitrification. Tertiary trickling filters would nitrify the secondary effluent, which would be recirculated to the activated sludge tanks for denitrification. Separating nitrification and denitrification in different bacterial cultures, in this case nitrification in a fixed culture and denitrification in a suspended culture, means that the conditions in each culture can be modified to suit the culture in question.

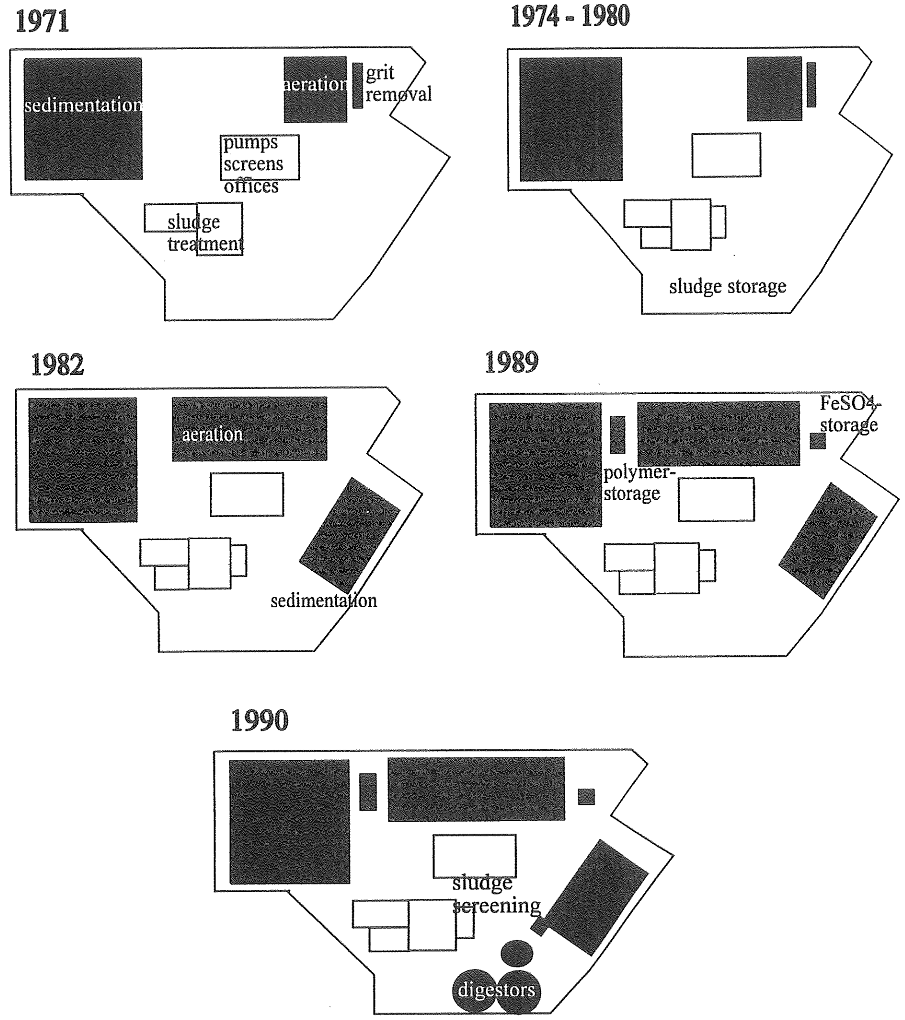


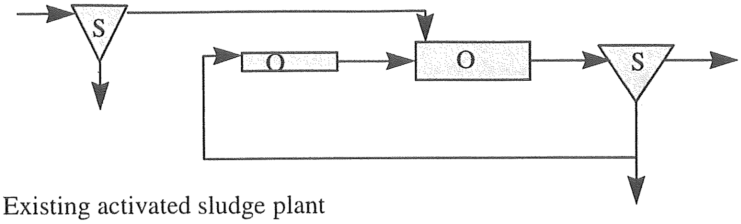
Figure A.3 Development of the Rya WWTP.

In a tertiary nitrifying trickling filter, competition from heterotrophic organisms is limited since the water to be nitrified is biologically treated and thus has a low content of organic matter (influent BOD_7 would be around 10 g O_2/m^3). The heterotrophic growth in such a trickling filter is relatively low, giving the nitrifiers a possibility to develop and maintain a stable population despite their low growth rate. Denitrification in this system takes place in a non-nitrifying activated sludge system and nitrified water is recirculated to the head of the activated sludge system and mixed with raw wastewater for denitrification. In this system the activated sludge system does not have to be designed for nitrification, thus the present activated sludge volumes yielding an SRT of 3 to 4 days should be sufficient.

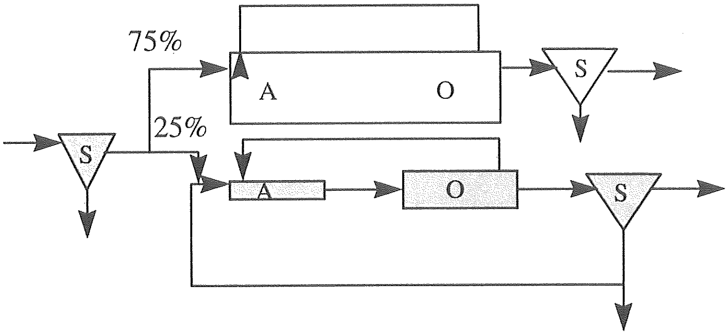
However an important difference between this system and pre-denitrification in a single sludge activated sludge system is that in this system all water to be recirculated has to pass secondary sedimentation. This causes the load on the final sedimentation tanks to increase correspondingly. With a recirculation flow equal to the influent flow the sedimentation capacity has to be doubled compared with a situation without recirculation. Another major difference is that the recirculated nitrified water in this system contains more oxygen than recirculated activated sludge (about 8 mg O_2/l compared with about 2 mg O_2/l in the activated sludge case).

In order to meet the demands on nitrogen removal a recirculation equal to or above the average flow will be needed. At the same time it is desirable to decrease overflow after primary treatment. The activated sludge tanks and the final sedimentation tanks will be built to receive a maximum influent flow of 10 m^3/s and the maximum flow to the trickling filters is set at 6 m^3/s.

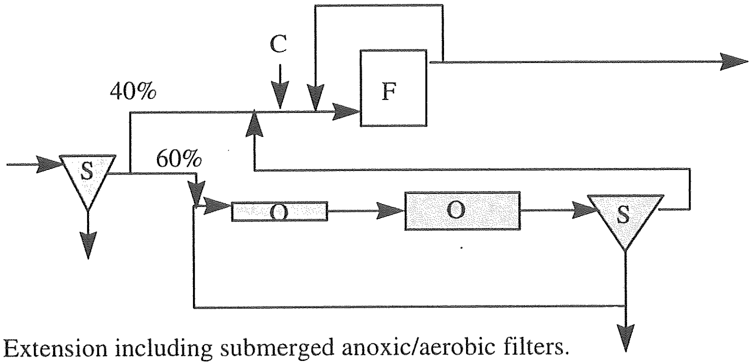
Biological treatment of flows up to 8-10 m^3/s instead of 6 m^3/s is expected to decrease overflow after primary treatment from 3-8 % to below 1 % of the total annual flow. The operational strategy will be to recirculate as much nitrate as possible to the activated sludge system within the limits set by nitrification in the trickling filters, by denitrification in the activated sludge system or by the capacity of the final sedimentation tanks. The increased need for sedimentation caused by increased recirculation and increased capacity for receiving storm-water will be met by another layer of final sedimentation tanks on top of the present ones. In this way no extra ground area will be needed for sedimentation. The newest 15 of the present 31 activated sludge tanks will be extended by about 4 m in height, giving them a total depth of 10 m. The total activated sludge volume will then be 51 600 m^3 divided into 3 lines with non-aerated zones (40 %), aerated zones (34 %) and zones which can be either aerated or non-aerated (26 %). Compared with the present 39 000 m^3 the total activated sludge volume will have increased by 30 % without using the 16 older activated sludge tanks, which instead will be used as the base for two rectangular trickling filters. The total filter base area will be about 2 300 m^2 and the height 7.2 m. The old activated sludge tanks below the filters will be used for collecting the nitrified water from the filters, the two central ones will be used as canals from which water to the filters will be pumped. The two outer ones will be used for transport of return activated sludge and trickling filter effluent to the activated sludge system. In these canals return activated sludge will be mixed with nitrified water, without aeration, for about 7 minutes before being mixed with primary settled wastewater



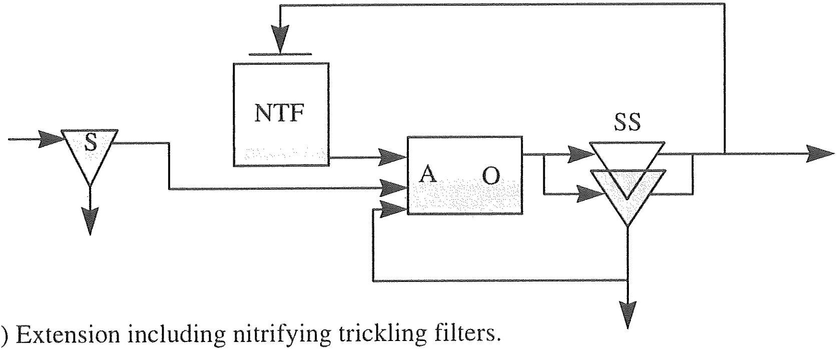
a) Existing activated sludge plant



b) Extension for pre-denitrification in activated sludge.



c) Extension including submerged anoxic/aerobic filters.



d) Extension including nitrifying trickling filters.

Figure A.4 Configurations considered for expansion for nitrogen removal. (Shaded areas indicate use of existing constructions. S = sedimentation tanks, SS = stacked sedimentation tanks, NTF = nitrifying trickling filters, F = submerged anoxic/aerobic filters, A = anoxic tanks, O = oxic tanks)

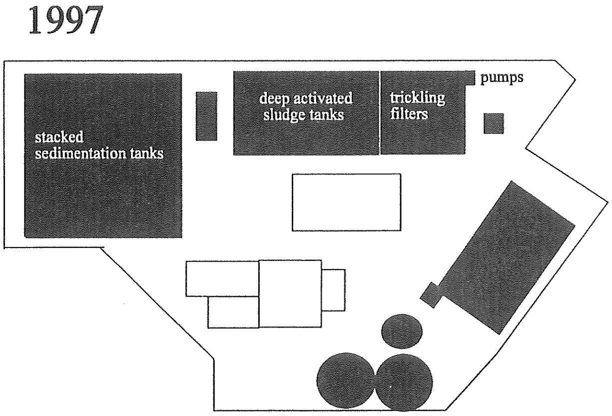


Figure A.5 Extension of the Rya WWTP for nitrogen removal (compare with Figure A.3).

References:

Naturvårdsverket (1996) Kväve från land till hav; Arbetsmaterial-Delredovisning. (In Swedish)

the Nordic Council (1990) Northern Europe´s seas, Northern Europe´s environment, *Report to the Nordic Council´*s International Conference on the Pollution of the Seas, 16-18 October, 1989. ISBN 91-38-12246-4.

Appendix B Data from pilot plant experiments, day to day results

The pilot plant was operated under different conditions during four years. During all experiments the pilot plant was operated as a denitrifying non-nitrifying activated sludge system. For more information on objectives and conditions refer to main text.

Notes to table of data:

Process: Letters refer to experimental period (A,B,C,D). Numbers (1,2,3,4,5) during experimental periods C and D are used in order to separate process configurations where the available activated sludge volume is used in different ways.

Volumes: The use of the total activated sludge tank volume is specified. Return sludge stabilisation occurred by mistake when RAS was pumped to the first non-aerated tank without the addition of recirculated water or primary settled wastewater. In the deoxygenation tank RAS and recirculated water were mixed without aeration. Anoxic tanks are non-aerated tanks treating a mixture of primary settled wastewater, RAS and recirculated water.

Flows: See figure in main text for explanation of indexes.

Waste activated sludge and addition of $FeSO_4$ and $NaNO_3$: Omitted figures indicate that no measurement was made, not that there was no withdrawal or addition.

Temperature: registered in the morning with a laboratory thermometer in the tank.

Analysis of Suspended solids, Ash content, COD, BOD, Fractions of Nitrogen and Phosphorus, alkalinity and settling of mixed liquor: refer to main text.

Estimation of suspended solids concentration in deoxygenation tank: refer to main text.

Oxygen concentration in aeration tank, oxidation reduction potential: 24-hour average of results of on-line measurement.

Oxygen concentration in effluent from RBC: Estimated in relation to temperature and performance of RBC.

Date	PROCESS	Day of the week	Volume: Total activated sludge	Return sludge stab.	Deoxygenation tank	Anoxic tank	Aerated tanks	Flow: To pilot plant (Q1)	Recirculated water (Q9)	Return activated sludge (Q5)	Waste activated sludge (Q11)	Addition of $FeSO_4$	Addition of $NaNO_3$	Temperature: Primary settled WW	Activated sludge tanks	Suspended solids: Activated sludge tanks	Deoxygenation tank (estimated)	RAS	Effluent	Ash content, activated sludge	COD: Primary settled WW	COD filtered sample: Primary settled WW	Anoxic tank	Effluent	RAS	Deoxygenation tank	RBC
Unit			m^3					l/s			m^3/d	gFe/h	g N/d	"C		kg SS/m^3				%ofSS		gO_2/m^3					
14/6 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.60	1.40	1.40	0.74	142	68	18.8	19.6			9.06	0.083		122	228	55	30			
15/6 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.26	1.40	1.40	1.61	142											71				
18/6 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.80	1.40	1.40	1.61	142	65	18.8	19.6	3.80		8.82	0.096	40	315	217	49	27			
19/6 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.82	1.40	1.40	1.61	142	65						0.120		332	255	70				
20/6 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.30	1.40	1.40	1.49	179	78	19.0	19.8	3.60		9.46	0.128	40	207	124	71	26			
4/7 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.60	1.40	1.40	1.43	195	59	18.2	19.0	3.25		8.46	0.052	47	202	209	42	27			
5/7 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.82	1.40	1.40	1.59	195	59	18.4	19.2						192	138	51				
6/7 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.42	1.40	1.40	1.68	195	59	17.4	18.2				0.052				35				
9/7 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.62	0.70	1.20		146	59	17.3	18.1				0.050		180	131					
10/7 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.30	0.70	1.20	1.84	146	68	17.8	18.6	3.49		9.68	0.044	47	181	136	35	27			
11/7 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.82	0.70	1.20	1.41	146	68	18.2	19.0				0.041		172	119	28				
12/7 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.60	0.70	1.20	1.54	146	67	18.0	18.8	3.20		8.79	0.045	47	194	142	32	26			
13/7 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.42	0.70	1.20	1.51	146	67	18.2	19.0								35				
16/7 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.10	1.30	1.40	1.57	162	63	18.2	19.0	3.20		8.89	0.057	47	220	143	45	27			
17/7 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.54	1.30	1.40	1.58	162	63	18.6	19.4				0.078		270	205	54				
18/7 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.42	1.30	1.40	1.59	142	63	19.0	19.8				0.070		245	166	59				
19/7 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.80	1.30	1.40	1.59	142	65	19.0	19.8	3.30		8.84	0.066	47	253	164	57	28			
20/7 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.60	1.30	1.40	1.53	142		19.0	19.8								63				
23/7 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.59	1.30	1.40		204	112						0.071				42				
24/7 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.60	1.30	1.40	1.59	204	112	19.2	20.0	3.40		8.06	0.075	49	264	159	55	31			
25/7 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.26	1.30	1.40	1.57	204	112	19.2	20.0						248	173	50				
26/7 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.59	204	112	19.2	20.0	3.50		8.30	0.072	49	266	170	50	35			
27/7 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.59	1.30	1.40	1.57	200	112										49				
30/7 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.82	1.30	1.40		192		19.4	20.2	3.54		7.40	0.060		247	153	47	32			
31/7 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.60	1.30	1.40	1.63	192	207	20.0	20.8	3.50			0.047	49	252	187	53				
1/8 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.82	1.30	1.40	1.59	192	211	20.2	21.0	3.53		8.75	0.064	47	261	168	54	52			
2/8 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.54	196	227	20.2	21.0	3.50			0.055	47	297	203	103				
3/8 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.64	1.30	1.40	1.53	196	235	20.2	21.0								65				
6/8 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.50	1.30	1.40		196	236	19.3	20.1	4.09		2.74	0.028	49	268	183	46	36			
7/8 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.10	1.30	1.40	1.47	196	240	20.0	20.8	4.09			0.045	49	281	163	71	36			
8/8 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.50	1.30	1.40	1.58	208	204	20.2	21.0	4.05			0.038	49	236	119	88				
9/8 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.32	1.30	1.40	1.58	196	112	19.5	20.3	4.05		9.70	0.037	49	260	146	42	33			
10/8 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.44	1.30	1.40	1.53	214	208										47				
13/8 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.30	1.30	1.40		204	204	20.0	20.8	4.06		10.02	0.038	51	255	149	52	30			
14/8 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.80	1.30	1.40	1.63	204	236	20.5	21.3	4.06			0.039	51	295	203	55				
15/8 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.30	1.30	1.40	1.54	204	224	20.8	21.6	4.17			0.083	51	292	186	63				
16/8 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.50	1.30	1.40	1.56	204	216	20.6	21.4	4.17		10.48	0.250	51	249	171	57	40			
17/8 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.47	1.30	1.40	1.48	204	225										48				
20/8 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	3.10	1.00	1.40		204	237	19.6	20.4	3.77				51	212	132	49	39			
21/8 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.60	1.00	1.40	1.52	204	245	19.4	20.2	3.77		9.94	0.068	51	251	159	46	31			
22/8 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.80	1.00	1.40	1.54	204	253	20.0	20.8	3.80				51	288	171	58	29			
23/8 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.40	1.50	1.40	1.53	204	249	20.4	21.2	3.80		9.96	0.074	51	287	224	59	36			
24/8 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	3.10	1.50	1.40	1.5	200	239	20.2	21.0	3.70				51			61				
27/8 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.50	1.50	1.40		204	226	20.0	20.8	3.60		9.90	0.042	51	310	192	52	30			
28/8 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.40	1.50	1.40	1.5	204	241	20.0	20.8	3.90				51	288	211	61	36			
29/8 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	3.30	1.50	1.40	1.6	204	237	20.2	21.0	3.90		10.26	0.017	51	300	207	53	31			
30/8 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.50	1.50	1.40		204	239	20.4	21.2						309	222	59	43			
31/8 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.90	1.50	1.40		204		20.4	21.2								62				
6/9 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	3.30	1.50	1.40	1.57	200	216	20.0	20.8	2.90		8.18	0.082	41	316	218	65	95			
7/9 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	3.30	1.50	1.40	1.57	200		19.3	20.1									40			
10/9 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.40	1.50	1.40		200		19.3	20.1	3.30		8.50	0.067	43	255	178	43	42			
11/9 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.80	1.50	1.40	1.26	179	134	20.0	20.8	3.30				43	289	223	59	40			
12/9 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.40	1.50	1.40		179	257	20.1	20.9								66	40			
13/9 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.80	1.50	1.40		179	245	20.0	20.8	3.20		8.78	0.131	43	297	231	109	39			
18/9 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.80	1.50	1.40	1.63	175		18.3	19.1	2.98		7.20	0.081	36	247	169	89	139			
20/11 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.67	0	71	13.7	14.5	3.20				34	222	147	61	26			
21/11 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.46	0	71	13.6	14.4	3.21				34	227	152	59	39			
22/11 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.54	0	71	13.8	14.6	2.65		9.16	0.113	34	246	159	53	38			
23/11 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.54	0	71	14.1	14.9	2.71				34				35			
25/11 -90	A	Sun.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40		0	71			2.28		8.06	0.094	34	198	136	48	31			
26/11 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.70	1.30	1.40		0	71	13.7	14.5	2.28				33	239	154	53	49			
27/11 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.60	1.30	1.40	1.57	0	71	14.0	14.8	2.14		8.06	0.049	32	290	194	58	42			
28/11 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40		0	71	13.8	14.6	2.40			0.098	31	282	196		49			
29/11 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.60	1.30	1.40	1.67	0	71	14.2	15.0	2.20		8.38	0.069	31	305	213	74	50			
30/11 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.80	1.30	1.40	1.11	0	69	14.3	15.1	2.11				30				48			
3/12 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.60	1.30	1.40		0	69	14.1	14.9	2.22		9.00	0.036	29	260	171	64	47			
4/12 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.54	0	260	14.1	14.9	2.27			0.034	30	280	197	56	46			
5/12 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.63	0	260	14.4	15.2	2.32		9.06	0.062	31	299	223	75	44			
6/12 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.59	0	260	14.4	15.2	2.46			0.075	32	280	190	75	55			
7/12 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.51	0	269	14.3	15.1	2.43				32				35			
9/12 -90	A	Sun.	31.2	0.0	0.0	11.2	20.0	2.60	1.30	1.40		0	269			1.85		7.36	0.092	33	187	121	72	33			
10/12 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.70	1.30	1.40		0	309	13.2	14.0	1.85		7.36	0.067	33	245	162	58	42			

Date	PROCESS	Day of the week	Volume: Total activated sludge	Return sludge stab.	Deoxygenation tank	Anoxic tank	Aerated tanks	Flow: To pilot plant (Q1)	Recirculated water (Q9)	Return activated sludge (Q5)	Waste activated sludge (Q11)	Addition of FeSO4	Addition of NaNO3	Temperature: Primary settled WW	Activated sludge tanks	Suspended solids: Activated sludge tanks	Deoxygenation tank (estimated)	RAS	Effluent	Ash content, activated sludge	COD: Primary settled WW	COD filtered sample: Primary settled WW	Anoxic tank	Effluent	RAS	Deoxygenation tank	RBC
Unit			m³					l/s			m³/d	gFe/h	g N/d	°C		kg SS/m³				%ofSS		gO₂/m³					
11/12 -90	A	Tue	31.2	0.0	0.0	11.2	20.0	2.70	1.30	1.40	1.52	0	309	13.6	14.4	2.02			0.028	33	274	172	64	46			
12/12 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.47	0	309	13.4	14.2	2.17			0.030	32	238	170	79	44			
13/12 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.72	0	261	12.7	13.5	1.95		7.34	0.030	32	261	172	58	42			
14/12 -90	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.78	0	261	13.1	13.9	2.08				32				45			
17/12 -90	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.30	1.30	1.40		0	129	13.5	14.3	3.88		8.12	0.035	15	283	185	71	53			
18/12 -90	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.40	1.30	1.40	1.53	0	94	13.8	14.6	2.09			0.040	25	354	222	74	59			
19/12 -90	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.30	1.30	1.40	1.46	0	94	13.8	14.6	2.24		8.58	0.050	25	324	210	66	55			
20/12 -90	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.30	1.30	1.40	1.48	0	109	13.7	14.5	2.49			0.053	25	309	203	64	46			
7/1 -91	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.20	1.50	1.40	1.56	125	109	9.9	10.7	1.89		9.80	0.022	39	130	91	61	35			
8/1 -91	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.30	1.50	1.40	1.54	125	109	10.4	11.2	1.98			0.021	39	146	105	59	32			
9/1 -91	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.00	1.50	1.40	1.46	125	109	10.2	11.0	1.85		6.80	0.020	39	99	43	53	36			
10/1 -91	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.10	1.30	1.40	1.61	142	104	9.5	10.3	1.88			0.017	40	110	76		24			
11/1 -91	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.00	1.30	1.40		125		9.3	10.1	1.90				42			48	17			
12/1 -91	A	Sat.	31.2	0.0	0.0	11.2	20.0		1.30	1.40		125	104										53				
13/1 -91	A	Sun.	31.2	0.0	0.0	11.2	20.0		1.30	1.40		125	110										58				
14/1 -91	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.00	1.30	1.40	1.51	125	110	10.9	11.7	1.91			0.024	38	195	134	52	37			
15/1 -91	A	Tue.	31.2	0.0	0.0	11.2	20.0	1.90	1.30	1.40	1.58	125	110	11.2	12.0	1.94			0.028	38	234	142	74	43			
16/1 -91	A	Wed.	31.2	0.0	0.0	11.2	20.0	1.90	1.30	1.40	1.62	125	110	11.5	12.3	2.21			0.028	36	214	147		41			
17/1 -91	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.00	1.30	1.40	1.56	125	110	11.6	12.4	2.11		7.40	0.036	37	229	178	64	39			
18/1 -91	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.30	1.30	1.40		125	110	11.8	12.6	2.21				42			56	53			
19/1 -91	A	Sat.	31.2	0.0	0.0	11.2	20.0		1.30	1.40		112											75				
20/1 -91	A	Sun.	31.2	0.0	0.0	11.2	20.0		1.30	1.40		112											75				
21/1 -91	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.00	1.30	1.40	1.59	112	110	10.6	11.4	2.36		8.30		42			46				
22/1 -91	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.00	1.30	1.40	1.44	112	110	11.8	12.6	2.22		8.38	0.053	44	229	161	72	42			
23/1 -91	A	Wed.	31.2	0.0	0.0	11.2	20.0	1.90	1.30	1.40	1.44	112	110	12.0	12.8	2.23			0.052	43	238	170	58	44			
24/1 -91	A	Thu.	31.2	0.0	0.0	11.2	20.0	1.90	1.30	1.40		129	110	12.5	13.3	2.33				43	246	170	64	35			
25/1 -91	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.20	1.40	1.40		129		12.4	13.2	2.50				43			79	43			
26/1 -91	A	Sat.	31.2	0.0	0.0	11.2	20.0		1.40	1.40		129											58				
27/1 -91	A	Sun.	31.2	0.0	0.0	11.2	20.0		1.40	1.40		129															
28/1 -91	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.20	1.40	1.40	1.53	129	110	12.1	12.9	2.70		10.12	0.032	42	260	159	71	36			
29/1 -91	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.20	1.40	1.40	1.57	129	110	12.2	13.0	2.96			0.047	43	312	176	74	35			
30/1 -91	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.00	1.40	1.40	1.59	125	110	12.4	13.2	3.29		11.40	0.064	41	305	171	66	42			
31/1 -91	A	Thu.	31.2	0.0	0.0	11.2	20.0	2.10	1.40	1.40	1.57	125	120	12.6	13.4	3.40			0.078	44	304	175	64	40			
1/2 -91	A	Fri.	31.2	0.0	0.0	11.2	20.0	2.00	1.40	1.40		125	155	12.6	13.4	3.25				42				47			
2/2 -91	A	Sat.	31.2	0.0	0.0	11.2	20.0		1.40	1.40		125															
3/2 -91	A	Sun.	31.2	0.0	0.0	11.2	20.0		1.40	1.40		125															
4/2 -91	A	Mon.	31.2	0.0	0.0	11.2	20.0	2.00	1.40	1.40	1.56	125	236	12.4	13.2	3.45				43			62	45			
5/2 -91	A	Tue.	31.2	0.0	0.0	11.2	20.0	2.30	1.40	1.40	1.53	125	213	12.6	13.4	3.35		10.50	0.024	43	328	215	68	46			
6/2 -91	A	Wed.	31.2	0.0	0.0	11.2	20.0	2.20	1.40	1.40	1.49	125	225	12.6	13.4	3.74			0.061	44	417	339	56	39			
4/2 -92	B	Mon.	15.5	0.0	0.0	5.5	10.0	1.40	0.65	1.40		117	42		12.3	3.57		8.50	0.512	40	304	126		52			
6/2 -92	B	Wed.	15.5	0.0	0.0	5.5	10.0	1.30	1.18	1.20		117	82		13.0	2.76		8.50	0.432	40	297	105	66	32			
10/2 -92	B	Mon.	15.5	0.0	0.0	5.5	10.0	1.30	1.13	1.40		117	45		12.3	1.66		4.86	0.224	43	222	76	47	31			
13/2 -92	B	Thu.	15.5	0.0	0.0	5.5	10.0	1.50	1.20	1.00		110	107		12.0	0.72		5.09	0.204	36	151	40	43	6			
17/2 -92	B	Mon.	15.5	0.0	0.0	5.5	10.0	1.30	1.10	1.10		110	42		12.0	1.66		4.37	0.086	41	263	99	72	48			
18/2 -92	B	Tue.	15.5	0.0	0.0	5.5	10.0	1.60	1.10	1.13		110	101		12.6	1.95		4.09	0.084	41	273	102	61	49			
24/2 -92	B	Mon.	15.5	0.0	0.0	5.5	10.0	2.50	0.58	0.76		104	63		10.8	1.56		9.84	0.126	43	219	76	67	39			
27/2 -92	B	Thu.	15.5	0.0	0.0	5.5	10.0	1.50	0.90	1.20		104	65		13.1	3.67		6.14	0.146	43	151	95	59	38			
2/3 -92	B	Mon.	15.5	0.0	0.0	5.5	10.0	1.50	0.94	1.38		95	72		12.0	3.25		5.80	0.066	43	224	60	47	30			
9/3 -92	B	Mon.	15.3	0.0	0.0	2.8	12.5	1.60	1.00	1.10		125	72		13.3	3.17		6.22	0.242	49	275	87	47	24			
12/3 -92	B	Thu.	15.3	0.0	0.0	2.8	12.5	2.10	0.72	1.00		108	45		11.6	2.42		4.76	0.154	41	179	65	51	22			
16/3 -92	B	Mon.	15.5	0.0	0.0	5.5	10.0	1.40	0.90	1.15		108	76		12.2	2.63		5.00	0.020	41	223	83	47	20			
19/3 -92	B	Thu.	15.5	0.0	0.0	5.5	10.0	1.80	0.70	1.14		108	89		13.4	3.76		4.40	0.208	43	217	96	61	32			
25/3 -92	B	Wed.	15.5	0.0	0.0	5.5	10.0	1.90	0.63	1.25		92	58		12.5	1.52		3.64	0.111	43	236	101	50	30			
26/3 -92	B	Thu.	15.5	0.0	0.0	5.5	10.0	1.60	0.67	1.42		92	105		12.8	1.61		3.54	0.025	43	269	110	56	43			
31/3 -92	B	Tue.	15.5	0.0	0.0	5.5	10.0	1.40	0.75	1.64		92	81		13.7	3.41		6.04	0.263	43	359	131	65	52			
1/4 -92	B	Wed.	15.3	0.0	0.0	2.8	12.5	1.40	0.70	1.62		108	98		13.9	2.90		4.84	0.129	43	312	200	67	52			
2/4 -92	B	Thu.	15.3	0.0	0.0	2.8	12.5	1.40	0.70	1.63		108	85		13.8	2.99		4.85	0.051	43	274	113	54	40			
6/4 -92	B	Mon.	15.3	0.0	0.0	2.8	12.5	2.00	0.43	1.20		108	44		12.2	3.64		8.89	0.207	44	251	89	48	37			
7/4 -92	B	Tue.	15.3	0.0	0.0	2.8	12.5	1.50	0.60	1.36		108	51		12.4	2.43		7.94	0.024	44	273	99	73	39			
9/4 -92	B	Thu.	15.3	0.0	0.0	2.8	12.5	1.20	0.91	1.47		108	46		13.8	3.67		5.28	0.062	39	287	117	41	35			
13/4 -92	B	Mon.	15.8	0.0	0.0	8.3	7.5	1.50	1.00	1.30		108	99		13.2	2.26		3.58	0.083	42	206	65	37	26			
14/4 -92	B	Tue.	15.8	0.0	0.0	8.3	7.5	2.40	0.75	0.85		108	54		12.6	1.52		3.14	0.304	42	270	98	57	41			
15/4 -92	B	Wed.	15.8	0.0	0.0	8.3	7.5	1.30	0.73	1.38		108	27		13.0	1.39		3.14	0.068	42	215	72	55	27			
5/5 -92	B	Tue.	13.7	0.0	0.0	11.2	2.5	1.49	0.90	1.20	1.78	108	49		13.0	2.62		4.60	0.018	43	244	105.5		52			
10/5 -92	B	Sun.	13.7	0.0	0.0	11.2	2.5	1.69	0.90	1.20	1.69	108	45		13.4	2.86		5.56	0.075	43	222	75.5		49			
11/2 -93										0.70	1.72	92	0														
14/2 -93	C5	Sun.	16.0	0.0	0.0	8.2	7.7	1.16	1.84	0.70			0			2.04		9.44	0.009	38	240	83		41			
15/2 -93	C5	Mon.	16.0	0.0	0.0	8.2	7.7	1.32	1.66	0.65	1.34		0	11.3	12.7	2.04		9.44					44				
16/2 -93	C5	Tue.	16.0	0.0	0.0	8.2	7.7	1.53	1.47	0.65	1.06		0	11.1	12.5												
17/2 -93	C5	Wed.	16.0	0.0	0.0	8.2	7.7	2.06	0.94	0.65	1.03		0	9.1	10.6	2.37		13.22	0.013	38	194	84		23			
18/2 -93	C3	Thu.	15.8	0.0	0.0	5.5	10.3	1.69	1.27	0.65	1.03		0	10.6	11.6												
19/2 -93	C3	Fri.	15.8	0.0	0.0	5.5	10.3	1.99	0.60	0.65		92	0	9.4	10.5												

Date	PROCESS	Day of the week	Volume: Total activated sludge	Return sludge stab.	Deoxygenation tank	Anoxic tank	Aerated tanks	Flow: To pilot plant (Q1)	Recirculated water (Q9)	Return activated sludge (Q5)	Waste activated sludge (Q11)	Addition of FeSO4	Addition of NaNO3	Temperature: Primary settled WW	Activated sludge tanks	Suspended solids: Activated sludge tanks	Deoxygenation tank (estimated)	RAS	Effluent	Ash content, activated sludge	COD: Primary settled WW	COD filtered sample: Primary settled WW	Anoxic tank	Effluent	RAS	Deoxygenation tank	RBC
Unit			m³					l/s			m³/d	gFe/h	g N/d	°C		kg SS/m³				%ofSS		gO₂/m³					
20/2 -93	C3	Sat.	15.8	0.0	0.0	5.5	10.3	1.80		0.65			0														
21/2 -93	C3	Sun.	15.8	0.0	0.0	5.5	10.3	1.59		0.65			0														
22/2 -93	C3	Mon.	15.8	0.0	0.0	5.5	10.3	1.34	1.60	0.70	1.01		0	9.6	10.6	3.52				37							
23/2 -93	C3	Tue.	15.8	0.0	0.0	5.5	10.3	1.29	1.71	0.70	1.03		0	10.4	11.3												
24/2 -93	C3	Wed.	15.8	0.0	0.0	5.5	10.3	1.37	1.63	0.70	1.44	100	0	11.0	11.9												
25/2 -93	C3	Thu.	15.8	0.0	0.0	5.5	10.3	1.26	1.74	0.70	1.6		0	10.7	11.9						284	112	63	39			
26/2 -93	C3	Fri.	15.8	0.0	0.0	5.5	10.3	1.31	1.69	0.70			0	10.7	11.9												
27/2 -93	C3	Sat.	15.8	0.0	0.0	5.5	10.3	1.42	1.58	0.70			0														
28/2 -93	C3	Sun.	15.8	0.0	0.0	5.5	10.3	1.19	1.81	0.70			0			2.27		11.24	0.026	63	243	84	66				
1/3 -93	C1	Mon.	15.6	0.0	0.0	2.7	12.9			0.70	1.52		0	10.6	12.0	2.27		11.24		63							
2/3 -93	C1	Tue.	15.6	0.0	0.0	2.7	12.9	1.16	1.72	0.70	1.34		0	10.9	12.1												
3/3 -93	C1	Wed.	15.6	0.0	0.0	2.7	12.9	1.32	1.90	0.70	0.78		0	11.1	12.1	2.21		10.82		61							
4/3 -93	C1	Thu.	15.6	0.0	0.0	2.7	12.9	1.18	1.82	0.70	0.87		0	11.5	12.2	1.94		9.06	0.015	63	332	127	56	44			
5/3 -93	C1	Fri.	15.6	0.0	0.0	2.7	12.9	1.39	1.61	0.70		100	0	11.3	12.3												
6/3 -93	C1	Sat.	15.6	0.0	0.0	2.7	12.9	1.16	1.84	0.70			0														
7/3 -93	C1	Sun.	15.6	0.0	0.0	2.7	12.9	1.10	1.90	0.70			0			2.96		13.30	0.016	54	283	92	73	37			
8/3 -93	C2	Mon.	15.8	0.0	2.7	2.7	10.3	1.15	1.84	0.70	1.06		0	11.3	12.5	2.96	3.98	13.30		54							
9/3 -93	C2	Tue.	15.8	0.0	2.7	2.7	10.3	1.12	2.00	0.70	1.1		0	11.5	12.3												
10/3 -93	C2	Wed.	15.8	0.0	2.7	2.7	10.3	1.10	1.60	0.70	0.68		0	11.5	12.1												
11/3 -93	C2	Thu.	15.8	0.0	2.7	2.7	10.3	1.17	1.83	0.70	0.93		0														
12/3 -93	C2	Fri.	15.8	0.0	2.7	2.7	10.3	1.12	1.88	0.70			0	11.3	12.5	2.41	3.30	11.60		40							
13/3 -93	C2	Sat.	15.8	0.0	2.7	2.7	10.3	1.17	1.82	0.70			0														
14/3 -93	C	Sun.	15.8	2.7	0.0	2.7	10.3	1.23	1.77	0.70			0						0.012		260	108	67	28			
15/3 -93	C	Mon.	15.8	2.7	0.0	2.7	10.3	1.28	1.72	0.75	0.89		0	11.2	12.3	2.51		11.72		40							
16/3 -93	C	Tue.	15.8	2.7	0.0	2.7	10.3	1.15	1.85	0.75	0.86	121	0	11.5	12.5	2.39		11.00	0.012	40	302	132	84	41			
17/3 -93	C4	Wed.	16.0	0.0	2.7	5.5	7.7	1.49	1.49	0.75	0.91		0	11.4	12.6												
18/3 -93	C4	Thu.	16.0	0.0	2.7	5.5	7.7	1.42	1.58	0.75	0.92		0														
19/3 -93	C4	Fri.	16.0	0.0	2.7	5.5	7.7	1.52	1.46	0.75		92	0	10.0	11.8												
20/3 -93	C4	Sat.	16.0	0.0	2.7	5.5	7.7	1.24	1.76	0.75			0														
21/3 -93	C4	Sun.	16.0	0.0	2.7	5.5	7.7	1.24	1.76	0.75			0			2.94	4.30	14.08	0.013	41	218	75	42	24			
22/3 -93	C4	Mon.	16.0	0.0	2.7	5.5	7.7	1.80	1.20	0.75	0.93		0	11.9	12.2	2.94	5.54	14.08		41							
23/3 -93	C4	Tue.	16.0	0.0	2.7	5.5	7.7	1.39	1.22	0.75	0.92		0	10.0	11.0												
24/3 -93	C4	Wed.	16.0	0.0	2.7	5.5	7.7	1.47	1.53	0.75	1.2		0	11.1	12.0	2.80	4.55	13.67	0.030	42	260	99	58	29			32
25/3 -93	C4	Thu.	16.0	0.0	2.7	5.5	7.7	1.25	1.75	0.75	1.19		0	11.1	12.3	2.80	4.16	13.72	0.016	42	280	124	62	32		40	
26/3 -93	C4	Fri.	16.0	0.0	2.7	5.5	7.7	1.22	1.78	0.75			0	11.5	12.7											52	
27/3 -93	C4	Sat.	16.0	0.0	2.7	5.5	7.7	1.15	1.85	0.75			0														
28/3 -93	C4	Sun.	16.0	0.0	2.7	5.5	7.7	1.06	1.94	0.75			0			3.16	4.24	14.62	0.019	41	306	97	96	45			
29/3 -93	C2	Mon.	15.8	0.0	2.7	2.7	10.3	1.18	1.82	0.75	1.23		0	11.9	13.1	3.16	4.44	14.62		41	325	149	62	43			
30/3 -93	C2	Tue.	15.8	0.0	2.7	2.7	10.3	1.12	1.88	0.75	1.2		0	11.9	13.1												
31/3 -93	C2	Wed.	15.8	0.0	2.7	2.7	10.3	1.07	1.93	0.75	1.6		0	12.1	13.1	3.04	4.12	14.24	0.034	41	325	141	66	46			
1/4 -93	C2	Thu.	15.8	0.0	2.7	2.7	10.3	1.28	1.71	0.75	1.51		0	12.1	13.3	3.11	4.66	15.06	0.019	41	340	140	62	49		47	68
2/4 -93	C2	Fri.	15.8	0.0	2.7	2.7	10.3	1.12	1.85	0.75			0	12.0	13.3	3.11	4.24	13.98								38	
3/4 -93	C2	Sat	15.8	0.0	2.7	2.7	10.3	1.04	1.96	0.75			0														
4/4 -93	C2	Sun.	15.8	0.0	2.7	2.7	10.3	1.10	1.90	0.75			0			1.84	2.54	8.94	0.016	39	286	89	63	38			36
5/4 -93	C3	Mon.	15.8	0.0	2.7	2.7	10.3	1.09	1.91	0.75	1.35		0	12.5	13.6	1.84	2.56	8.94		39							
6/4 -93	C3	Tue.	15.8	0.0	0.0	5.5	10.3	1.29	1.51	0.75	1.23		0	12.1	13.3												
7/4 -93	C3	Wed.	15.8	0.0	0.0	5.5	10.3	1.15	1.85	0.75	1.18		0	12.6	13.4	2.08		10.00		37							
8/4 -93	C3	Thu.	15.8	0.0	0.0	5.5	10.3	1.15	1.85	0.75		96	0	12.5	13.5												
9/4 -93	C3	Fri.	15.8	0.0	0.0	5.5	10.3	1.03	1.97	0.75			0														
10/4 -93	C3	Sat.	15.8	0.0	0.0	5.5	10.3	0.99	2.01	0.75			0														
11/4 -93	C3	Sun.	15.8	0.0	0.0	5.5	10.3	0.94	2.06	0.75			0														
12/4 -93	C3	Mon.	15.8	0.0	0.0	5.5	10.3	0.91	2.09	0.75			0			2.05		9.60	0.014	38	271	125	77	31			
13/4 -93	C3	Tue.	15.8	0.0	0.0	5.5	10.3	1.23	1.77	0.75	0.93		0	12.1	13.1	2.05		9.60	0.024	38	299	117	314	49			
14/4 -93	C5	Wed.	16.0	0.0	0.0	8.2	7.7	1.01	1.99	0.75	0.68		0	12.3	13.2												
15/4 -93	C5	Thu.	16.0	0.0	0.0	8.2	7.7	1.14	1.86	0.75	0.82		0	12.8	13.7	2.46		11.80	0.019	39	348	179	102	50			
16/4 -93	C5	Fri.	16.0	0.0	0.0	8.2	7.7	1.15	1.85	0.75			0	12.4	13.6												
17/4 -93	C5	Sat.	16.0	0.0	0.0	8.2	7.7	1.59	1.40	0.75			0														
18/4 -93	C5	Sun.	16.0	0.0	0.0	8.2	7.7	1.12	1.87	0.75			0			2.41		12.04	0.016	42	232	86	205	50			
19/4 -93	C5	Mon.	16.0	0.0	0.0	8.2	7.7	1.15	1.85	0.75	1.15		0	12.3	13.4	2.41		12.04	0.011	42	281	117	183	55			
20/4 -93	C5	Tue.	16.0	0.0	0.0	8.2	7.7	1.14	1.86	0.75	0.97	92	0	12.7	13.7												
21/4 -93	C5	Wed.	16.0	0.0	0.0	8.2	7.7	0.90	0.84	0.75			0	12.9	13.8												
22/4 -93	C5	Thu.	16.0	0.0	0.0	8.2	7.7	1.17	1.66	0.75	0.93		0	12.8	13.8	3.33		10.16	0.010	42	303	143	86	48			
23/4 -93	C2	Fri.	15.8	0.0	2.7	2.7	10.3	1.46	1.54	0.75			0	12.7	14.0												
24/4 -93	C2	Sat.	15.8	0.0	2.7	2.7	10.3	1.11	1.89	0.75			0														
25/4 -93	C2	Sun.	15.8	0.0	2.7	2.7	10.3	0.74	2.26	0.75			0			3.03		14.16	0.030	42	229.5	100.5	46.5	36			
26/4 -93	C2	Mon.	15.8	0.0	2.7	2.7	10.3	1.30	1.50	0.75	1.1		0	13.7	15.4	3.03	4.75	14.16		68							
27/4 -93	C2	Tue	15.8	0.0	2.7	2.7	10.3	1.42	1.45	0.75	1.1		0	13.8	14.6	3.32	4.29	9.16	0.011	42	335	167.5	69.5	41			
28/4 -93	C2	Wed.	15.8	0.0	2.7	2.7	10.3	1.10	1.90	0.75	1.06		0	13.9	15.4	2.88	3.86	12.88	0.008	41	316.5	150	55	44			
29/4 -93	C2	Thu.	15.8	0.0	2.7	2.7	10.3	1.08	1.92	0.75	1.11		0	14.1	15.6	2.73	3.61	12.06	0.111	40	356.5	188.5	68.5	43			
30/4 -93	C4	Fri.	16.0	0.0	2.7	5.5	7.7	1.11	1.89	0.75			0	14.2	15.3												
1/5 -93	C4	Sat.	16.0	0.0	2.7	5.5	7.7	0.95	1.09	0.75			0														
2/5 -93	C4	Sun.	16.0	0.0	2.7	5.5	7.7	0.94		0.75			0														

Date	PROCESS	Day of the week	Volume: Total activated sludge	Return sludge stab.	Deoxygenation tank	Anoxic tank	Aerated tanks	Flow: To pilot plant (Q1)	Recirculated water (Q9)	Return activated sludge (Q5)	Waste activated sludge (Q11)	Addition of FeSO4	Addition of NaNO3	Temperature: Primary settled WW	Activated sludge tanks	Suspended solids: Activated sludge tanks	Deoxygenation tank (estimated)	RAS	Effluent	Ash content, activated sludge	COD: Primary settled WW	COD filtered sample: Primary settled WW	Anoxic tank	Effluent	RAS	Deoxygenation tank	RBC
Unit			m^3					l/s			m^3/d	gFe/h	g N/d	°C		kg SS/m^3				%ofSS		gO_2/m^3					
3/5 -93	C4	Mon.	16.0	0.0	2.7	5.5	7.7	1.38	1.49	0.72	1.12		0	14.5	15.2	4.24	3.63	10.60		42							
4/5 -93	C4	Tue.	16.0	0.0	2.7	5.5	7.7	1.10	1.89	0.72	1	100	0	14.1	15.0	2.58	3.52	12.20	0.015	42	323	163	42	46			
5/5 -93	C4	Wed.	16.0	0.0	2.7	5.5	7.7	1.39	1.61	0.72	1.38		0	14.7	15.4	2.78	4.18	12.66	0.011	41	300	135	74	44			
6/5 -93	C4	Thu.	16.0	0.0	2.7	5.5	7.7	1.19	1.81	0.72	1.62		0	14.4	14.7	2.82	3.77	11.90	0.012	40	366	170	46.5	39			
7/5 -93	C5	Fri.	16.0	0.0	0.0	8.2	7.7	1.07	1.92	0.72			0	14.3	15.0												
8/5 -93	C5	Sat.	16.0	0.0	0.0	8.2	7.7	0.97	2.02	0.72			0														
9/5 -93	C5	Sun.	16.0	0.0	0.0	8.2	7.7	0.95	2.04	0.72			0			2.91		12.20	0.042	42	352	118	60	40			
10/5 -93	C5	Mon.	16.0	0.0	0.0	8.2	7.7	1.09	1.91	0.72	1		0	14.7	15.7	2.91		12.20	0.011	42	313	150	51	38			
11/5 -93	C5	Tue.	16.0	0.0	0.0	8.2	7.7	0.92	2.07	0.72	1.18	96	0	15.1	16.0	2.78		12.80	0.012	41	360	182	54	43			
12/5 -93	C5	Wed.	16.0	0.0	0.0	8.2	7.7	0.92	2.07	0.72	0.97		0	15.1	16.3	2.74		12.64	0.014	42	318	170	56	43			
13/5 -93	C5	Thu.	16.0	0.0	0.0	8.2	7.7	1.17	1.83	0.72			0	15.1	16.1	2.73		12.56	0.018	40	362	178	52	44			
14/5 -93	C5	Fri.	16.0	0.0	0.0	8.2	7.7	1.13	1.86	0.72			0	15.2	16.4	2.78		12.40		40							
15/5 -93	C5	Sat.	16.0	0.0	0.0	8.2	7.7	0.99	2.01	0.72			0														
16/5 -93	C5	Sun.	16.0	0.0	0.0	8.2	7.7	1.00	1.99	0.72			0														
17/5 -93	C3	Mon.	15.8	0.0	0.0	5.5	10.3	1.07	1.87	0.75			0	15.4	16.2	2.79			0.034	41	418	136	48	42			
18/5 -93	C3	Tue.	15.8	0.0	0.0	5.5	10.3	1.03	1.97	0.75			0	15.7	16.3	2.59		11.28	0.011	43	560	156	56	48			
19/5 -93	C3	Wed.	15.8	0.0	0.0	5.5	10.3	1.03	1.91	0.75			0	15.9	16.7	2.75				41							
20/5 -93	C3	Thu.	15.8	0.0	0.0	5.5	10.3	1.14	1.86	0.75			0														
21/5 -93	C3	Fri.	15.8	0.0	0.0	5.5	10.3	1.28	1.72	0.75			0														
22/5 -93	C3	Sat.	15.8	0.0	0.0	5.5	10.3	1.51	1.48	0.75			0														
23/5 -93	C3	Sun.	15.8	0.0	0.0	5.5	10.3	1.09	1.91	0.75			0			2.90		13.48	0.016	48	154	77	44	29			
24/5 -93	C3	Mon.	15.8	0.0	0.0	5.5	10.3	1.15	1.84	0.71	1.3		0	16.2	17.0	2.90		13.48	0.011	48	278	134	42	32			
25/5 -93	C3	Tue.	15.8	0.0	0.0	5.5	10.3	1.04	1.96	0.71			0	16.2	17.1	2.78		12.04	0.010	42	333	170	57	48			
26/5 -93	C3	Wed.	15.8	0.0	0.0	5.5	10.3	1.08	1.92	0.71	1.2		0	16.4	16.8	2.58		11.46	0.010	39	325	176	61	51			
27/5 -93	C3	Thu.	15.8	0.0	0.0	5.5	10.3	1.07	1.93	0.71	1.3		0	16.5	16.7	2.62		11.56	0.010	42	356	186	58	48			
28/5 -93	C	Fri.		0.0				0.96	2.04	0.71			0	16.7	17.2												
31/5 -93	C2	Mon.	15.8	0.0	2.7	2.7	10.3	1.08	1.70	0.77			0			2.85		9.14	0.005	42	255	101	38				28
1/6 -93	C2	Tue.	15.8	0.0	2.7	2.7	10.3	1.74	1.26	0.77	1.4	100	0	15.6	16.5	2.85	4.38	9.14	0.006	42	255	96	44	26			
2/6 -93	C2	Wed.	15.8	0.0	2.7	2.7	10.3	1.13	1.87	0.77	1.3		0	16.5	16.7	2.73	3.61	11.36	0.008	41	306	133	51	35			
3/6 -93	C2	Thu.	15.8	0.0	2.7	2.7	10.3	0.99	2.01	0.77	1.25		0	16.6	17.3	2.78	3.36	10.62	0.008	43	343	170	46	32		38	
4/6 -93	C4	Fri.	16.0	0.0	2.7	5.5	7.7			0.77			0	16.8	17.6												
6/6 -93	C4	Sun.	16.0	0.0	2.7	5.5	7.7	1.09	1.91	0.77			0			2.37	3.04	9.54	0.010	40	282	88	55	35			
7/6 -93	C4	Mon.	16.0	0.0	2.7	5.5	7.7	0.92	2.08	0.75	1.27		0	16.6	17.3	2.37	2.83	9.54	0.009	40	309	115	47	34		43	
8/6 -93	C4	Tue.	16.0	0.0	2.7	5.5	7.7	1.05	1.95	0.75	1.34		0	17.0	17.7	2.44	3.12	10.30	0.008	42	362	173	53	33		38	
9/6 -93	C4	Wed.	16.0	0.0	2.7	5.5	7.7	0.99	2.01	0.75	1.13		0	16.1	17.8	2.49	3.19	11.04	0.008	40	335	140	50	41		39	
10/6 -93	C4	Thu.	16.0	0.0	2.7	5.5	7.7	0.95	2.05	0.75	1.41		0	17.2	18.1	2.71	3.37	11.60	0.008	40	332	157	57	47	96	50	
11/6 -93	C5	Fri.	16.0	0.0	2.7	5.5	7.7		3.00	0.75		113	0	17.6	18.5											60	
13/6 -93	C5	Sun.	16.0	0.0	0.0	8.2	7.7	0.82	2.18	0.75			0			2.36		10.26	0.008	43	275	142	48	33			
14/6 -93	C5	Mon.	16.0	0.0	0.0	8.2	7.7	0.96	2.04	0.75	1.44		0	17.3	17.9	2.36		10.26	0.009	43	301	160	50	39	111	56	
15/6 -93	C5	Tue.	16.0	0.0	0.0	8.2	7.7	1.25	1.29	0.75	1.55		0	17.4	17.9												
16/6 -93	C5	Wed.	16.0	0.0	0.0	8.2	7.7	0.86	2.14	0.75			0	17.5	17.9	2.89		12.94	0.009	44	364	141	47	27	58		
17/6 -93	C5	Thu.	16.0	0.0	0.0	8.2	7.7	1.08	1.92	0.75			0	17.8	18.2	2.77		12.17	0.008	45	385	182	57	44	112	48	
18/6 -93	C3	Fri.	16.0	0.0	0.0	8.2	7.7	0.86	2.14	0.75			0	17.6	17.6											68	
19/6 -93	C3	Sat.	15.8	0.0	0.0	5.5	10.3	1.14	1.85	0.75			0														
20/6 -93	C3	Sun.	15.8	0.0	0.0	5.5	10.3	0.97	2.02	0.75			0			2.35		10.30	0.008	44	232	123	70	36			
21/6 -93	C3	Mon.	15.8	0.0	0.0	5.5	10.3	1.29	1.71	0.75	1.53		0	17.0	17.4	2.35		10.30	0.010	44	326	148	59	32	77	42	
22/6 -93	C3	Tue.	15.8	0.0	0.0	5.5	10.3	0.88	2.12	0.75	0.61		0	17.0	17.5	2.77		10.57	0.009	42	354	165	55	40	75	52	
23/6 -93	C3	Wed.	15.8	0.0	0.0	5.5	10.3	1.01	1.99	0.75	1.03	108	0	18.0	17.9	2.55		10.52	0.008	44	343	186	61	42	124	64	
24/6 -93	C2	Thu.	15.8	0.0	0.0	5.5	10.3	0.93	1.95	0.75			0	17.8	18.0												67
25/6 -93	C2	Fri.	15.8	0.0	2.7	2.7	10.3	1.20		0.75			0														
26/6 -93	C2	Sat.	15.8	0.0	2.7	2.7	10.3	0.72		0.75			0														
26/6 -93	C2	Sat.	15.8	0.0	2.7	2.7	10.3	0.72		0.75			0														
27/6 -93	C2	Sun.	15.8	0.0	2.7	2.7	10.3	1.29		0.75			0														
28/6 -93	C2	Mon.	15.8	0.0	2.7	2.7	10.3	0.94	1.97	0.75	1.45	113	0	17.2	17.8	4.63	2.72	9.38	0.148	45	351	123	45	32			
29/6 -93	C2	Tue.	15.8	0.0	2.7	2.7	10.3	0.85	2.14	0.75	1.69		0	17.4	18.4	2.32	2.72	9.36	0.013	43	382	159	60	45	38	72	
30/6 -93	C2	Wed.	15.8	0.0	2.7	2.7	10.3	0.95	2.02	0.75	1.88		0	18.2	18.7	2.16	2.68	9.12	0.006	43	463	175	54	39	102	55	
1/7 -93	C2	Thu.	15.8	0.0	2.7	2.7	10.3	0.80	2.20	0.75	2.08		0	18.3	19.0	2.15	2.47	8.70	0.012	43	350	160	65	35	91	40	
2/7 -93	C4	Fri.	15.8	0.0	2.7	2.7	10.3	0.94	2.06	0.75			0	18.5	19.3	2.28										39	
3/7 -93	C4	Sat.	16.0	0.0	2.7	5.5	7.7	1.44	1.56	0.75			0														
4/7 -93	C4	Sun.	16.0	0.0	2.7	5.5	7.7	0.93	2.07	0.75			0			2.06	2.56	8.92	0.017	46	211	86	38	27			
5/7 -93	C4	Mon.	16.0	0.0	2.7	5.5	7.7	0.94	2.06	0.75	1.42		0	17.7	18.5	2.06	2.56	8.92	0.012	46	284	131	39	37	80	34	
6/7 -93	C4	Tue.	16.0	0.0	2.7	5.5	7.7	0.92	2.08	0.75	1.12		0	18.2	18.6	2.06	2.53	8.80	0.035	45	301	150	40	41	97	47	
7/7 -93	C4	Wed.	16.0	0.0	2.7	5.5	7.7	0.97	2.00	0.75			0	18.2	18.6	2.03	2.50	8.26	0.041	46	295	139	41	37	86	50	
8/7 -93	C4	Thu.	16.0	0.0	2.7	5.5	7.7	1.02	1.94	0.75			0	18.2	18.7	1.93	2.52	8.54	0.020	43	298	152	54	35	102	45	
9/7 -93	C4	Fri.	16.0	0.0	2.7	5.5	7.7	1.78	1.22	0.75		113	0	17.7	18.7	1.89										43	
10/7 -93	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.61	1.38	0.75			0														
11/7 -93	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.09	1.91	0.75			0			2.00	2.15	5.28	0.014	44	210	74	39	24			
12/7 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	2.79	0.28	0.75	stängd		0	15.5	16.7	2.00	5.63	5.28	0.087	44	102	40	27	16	123	46	
13/7 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	2.59	0.45	0.75	0.74		0	15.2	15.5	1.75	5.33	8.22	0.058	47	139	56	38	27	42	35	
14/7 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	1.44	1.56	0.75			0	15.8	16.3	2.08	3.56	11.52	0.010	47	165	76	39	25	91	36	
15/7 -93	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.26	1.74	0.75	0.75		0	16.8	16.2	2.39	3.34	10.20	0.008	44	136	76	49	28	74	33	
16/7 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	1.21	1.78	0.75	0.66		0	16.7	17.4												

Date	PROCESS	Day of the week	Volume: Total activated sludge	Return sludge stab.	Deoxygenation tank	Anoxic tank	Aerated tanks	Flow: To pilot plant (Q1)	Recirculated water (Q9)	Return activated sludge (Q5)	Waste activated sludge (Q11)	Addition of FeSO4	Addition of NaNO3	Temperature: Primary settled WW	Activated sludge tanks	Suspended solids: Activated sludge tanks	Deoxygenation tank (estimated)	RAS	Effluent	Ash content, activated sludge	COD: Primary settled WW	COD filtered sample: Primary settled WW	Anoxic tank	Effluent	RAS	Deoxygenation tank	RBC
Unit			m³					l/s			m³/d	gFe/h	g N/d	°C		kg SS/m³				%ofSS		gO₂/m³					
17/7 -93	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.12	1.88	0.75			0														
18/7 -93	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.11	1.88	0.75			0			2.38	3.20	10.55	0.010	43	163	64	48	31			
19/7 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.08	1.92	0.75			0	15.9	19.8	2.38	3.15	10.55	0.009	43	230	107	43	25	51	31	
20/7 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	1.16	1.84	0.75	1.18		0	17.3	18.0	3.02	4.11	13.29	0.009	47	256	133	53	43	71	42	
21/7 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	1.72	1.28	0.75	1.3	113	0	17.1	18.0	2.86	5.04	12.97		46					68	43	
22/7 -93	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.56	1.44	0.75	1.24		0	17.3	18.1	3.03	4.86	13.25	0.009	45	191	92	39	30	65	40	
23/7 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	1.31	1.69	0.75	1.15		0	16.8	17.6												
24/7 -93	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.08	1.92	0.75			0														
25/7 -93	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.02	1.98	0.75			0			2.51	3.17	10.55	0.009	45	178	78	40	24			
26/7 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.14	1.86	0.75			0	17.2	17.8	2.51	3.32	10.55		45					51	50	
27/7 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	1.12	1.88	0.75	0.88		0	17.4	18.0	3.00	4.05	13.41	0.010	48	240	111		25	77	45	
28/7 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	0.92	2.08	0.75	1.38		0	17.4	18.1												
29/7 -93	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.07	1.93	0.75	1.17		0	17.7	18.5	3.00	3.93	13.09		48	275	123	69	36			
30/7 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	1.85	1.14	0.75	1.26		0	17.6	18.5												
31/7 -93	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.39	1.61	0.75			0														
1/8 -93	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.09	1.91	0.75			0			2.38	3.23	11.01	0.010	46	170	81	46	28			
2/8 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.04	1.96	0.75			0	17.4	17.9	2.38	3.17	11.01		46					76	34	
3/8 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	1.66	1.34	0.75	1.21	102	0	17.5	18.4	2.40	4.08	10.71	0.011	46	182	87		35	66	44	31
4/8 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	1.51	1.48	0.75	1.33		0	17.2	17.9												
5/8 -93	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.19	1.80	0.75	1.26		0	17.5	18.2	2.96	3.98	12.34	0.011	44	214	104	51	31	55	40	
6/8 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	2.13	0.87	0.75			0	17.4	18.3			7.00									
7/8 -93	D4	Sat	16.0	0.0	2.7	5.5	7.7	1.86	1.14	0.75			0														
8/8 -93	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.40	1.60	0.75			0														
9/8 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.87	1.15	0.75		108	0														
10/8 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	2.83	0.33	0.75	1.1		0														
11/8 -93	D5	Wed.	16.0	0.0	0.0	8.2	7.7	2.14	0.30	0.75	1.38		0														
12/8 -93	D5	Thu.	16.0	0.0	0.0	8.2	7.7	2.38	0.30	0.75	stängt		0														
13/8 -93	D5	Fri.	16.0	0.0	0.0	8.2	7.7	2.78	0.36	0.75			0														
14/8 -93	D5	Sat.	16.0	0.0	0.0	8.2	7.7	1.94	1.06	0.75			0														
15/8 -93	D5	Sun.	16.0	0.0	0.0	8.2	7.7	1.59	1.41	0.75			0														
16/8 -93	D5	Mon.	16.0	0.0	0.0	8.2	7.7	1.72	0.97	0.75			0														
17/8 -93	D5	Tue.	16.0	0.0	0.0	8.2	7.7	1.53	1.47	0.75			0														
18/8 -93	D5	Wed.	16.0	0.0	0.0	8.2	7.7	1.30	1.70	0.75			0														
19/8 -93	D5	Thu.	16.0	0.0	0.0	8.2	7.7	1.20	1.80	0.75			0	17.2	17.8	3.00		13.60	0.032	47	234	128	58	46	92		
20/8 -93	D5	Fri.	16.0	0.0	0.0	8.2	7.7	1.22	1.78	0.75			0	17.5	18.2												
21/8 -93	D5	Sat.	16.0	0.0	0.0	8.2	7.7	1.40	1.60	0.75			0														
22/8 -93	D5	Sun.	16.0	0.0	0.0	8.2	7.7	1.02	1.98	0.75			0														
23/8 -93	D5	Mon.	16.0	0.0	0.0	8.2	7.7	1.41	1.59	0.75	1		0	17.2	17.8	3.00		13.75	0.063	58	222	101	56	36	64		
24/8 -93	D5	Tue.	16.0	0.0	0.0	8.2	7.7	1.28	1.72	0.75	1.7	112	0	17.4	18.0	2.96		13.34	0.036	48	249	116	49	34	79		
25/8 -93	D5	Wed.	16.0	0.0	0.0	8.2	7.7	1.36	1.34	0.75			0	16.8	17.8												
26/8 -93	D5	Thu.	16.0	0.0	0.0	8.2	7.7	1.43	1.56	0.75	1.74		0	17.2	18.0	2.97		12.46	0.018	49	208	98	48	32	80		
27/8 -93	D5	Fri.	16.0	0.0	0.0	8.2	7.7	1.28	1.71	0.75	1.32		0	16.8	17.3	2.84			0.018	51							
28/8 -93	D5	Sat.	16.0	0.0	0.0	8.2	7.7	1.14	1.86	0.75			0														
29/8 -93	D5	Sun.	16.0	0.0	0.0	8.2	7.7	1.26	1.74	0.75			0			2.83		13.06	0.013	50							
30/8 -93	D5	Mon.	16.0	0.0	0.0	8.2	7.7	1.28	1.72	0.75		98	0	16.9	17.5	2.83		13.06		50		90	55	33	66		
31/8 -93	D5	Tue.	16.0	0.0	0.0	8.2	7.7	1.17	1.82	0.75	2.17		0	17.3	18.0	2.88		12.80	0.012	50	281	130	58	43	66		
1/9 -93	D5	Wed.	15.9	0.0	0.0	8.2	7.7	1.20	1.83	0.75			0	17.8	18.4												
2/9 -93	D5	Thu.	15.9	0.0	0.0	8.2	7.7	1.20	1.83	0.75			0	18.8	18.6	2.63		11.60	0.010	45	270	120	52	43	69		
3/9 -93	D5	Fri.	15.9	0.0	0.0	8.2	7.7	1.30	1.57	0.75			0	17.5	18.6												
4/9 -93	D5	Sat	15.9	0.0	0.0	8.2	7.7	1.10	1.91	0.75			0														
5/9 -93	D5	Sun.	15.9	0.0	0.0	8.2	7.7	1.00	1.98	0.75			0			1.78		7.74	0.008	44	230	102	42	29			
6/9 -93	D5	Mon.	15.9	0.0	0.0	8.2	7.7	1.10	1.86	0.75	2.14		0	17.5	18.3	1.78		7.74		44					48		
7/9 -93	D5	Tue.	15.9	0.0	0.0	8.2	7.7	1.00	1.02	0.75	2.04		0	17.8	18.7	0.74		7.80	0.010	85	280	125	62	42	55		
8/9 -93	D5	Wed.	15.9	0.0	0.0	8.2	7.7	0.70	1.83	0.75	0.63		0	17.8	18.6												
9/9 -93	D5	Thu.	15.9	0.0	0.0	8.2	7.7	1.40	1.59	0.75		102	0	18.3	18.9	2.15		9.82	0.013	63	309	173	51	70	49		
10/9 -93	D5	Fri.	15.9	0.0	0.0	8.2	7.7	1.00	1.98	0.75			0	17.8	18.7												
11/9 -93	D5	Sat.	15.9	0.0	0.0	8.2	7.7	1.20	1.85	0.75			0														
12/9 -93	D5	Sun.	15.9	0.0	0.0	8.2	7.7	0.90	2.05	0.75			0			2.43		11.32	0.023	43	294	127	34	29			
13/9 -93	D5	Mon.	15.9	0.0	0.0	8.2	7.7	1.10	1.90	0.75	1.03		0	17.7	18.3	2.43		11.32		43					55		
14/9 -93	D5	Tue.	15.9	0.0	0.0	8.2	7.7	1.20	1.81	0.75	0.78		0	17.4	17.8	2.50		10.84	0.004	44	282	134	42	24	62		
15/9 -93	D5	Wed.	15.9	0.0	0.0	8.2	7.7	1.10	1.86	0.75	0.99		0	16.4	16.9	3.44				54							
16/9 -93	D5	Thu.	15.9	0.0	0.0	8.2	7.7	1.00	1.99	0.75	1		0	17.3	17.8	2.51		11.16	0.064	54	375	124	39	57	62		
17/9 -93	D5	Fri.	15.9	0.0	0.0	8.2	7.7	1.40	1.57	0.75			0	16.5	17.5												
18/9 -93	D5	Sat.	15.9	0.0	0.0	8.2	7.7	0.90	2.09	0.75			0														
19/9 -93	D5	Sun.	15.9	0.0	0.0	8.2	7.7	1.00	2.04	0.75			0			1.67		8.49	0.103	53	965	125	31	15			
20/9 -93	D5	Mon.	15.9	0.0	0.0	8.2	7.7	1.10	1.95	0.75	1.03		0	16.6	17.3	1.67		8.49		53					58		
21/9 -93	D5	Tue.	15.9	0.0	0.0	8.2	7.7	1.10	1.89	0.75	1.05	108	0	16.8	17.6	2.16		9.49	0.053	43	304	125	49	74	54		
22/9 -93	D4	Wed.	15.9	0.0	2.7	5.5	7.7	1.30	1.66	0.75	0.98		0	16.8	17.6												
23/9 -93	D4	Thu	15.9	0.0	2.7	5.5	7.7	1.10	1.68	0.75	1.03		0	17.2	18.2	2.75	3.77	11.49	0.011	43	319	142	46	39	74		
24/9 -93	D4	Fri.	15.9	0.0	2.7	5.5	7.7	1.00	1.88	0.75		104	0	17.0	18.1												
25/9 -93	D4	Sat.	15.9	0.0	2.7	5.5	7.7	1.10	1.90	0.75			0														
26/9 -93	D4	Sun.	15.9	0.0	2.7	5.5	7.7	1.60	1.45	0.75			0			2.82	4.57	12.51	0.023	43	206	167	22.5	23			

Date	PROCESS	Day of the week	Volume: Total activated sludge	Return sludge slab.	Deoxygenation tank	Anoxic tank	Aerated tanks	Flow: To pilot plant (Q1)	Recirculated water (Q9)	Return activated sludge (Q5)	Waste activated sludge (Q11)	Addition of FeSO4	Addition of NaNO3	Temperature: Primary settled WW	Activated sludge tanks	Suspended solids: Activated sludge tanks	Deoxygenation tank (estimated)	RAS	Effluent	Ash content, activated sludge	COD: Primary settled WW	COD filtered sample: Primary settled WW	Anoxic tank	Effluent	RAS	Deoxygenation tank	RBC
Unit			m³					l/s			m³/d	gFe/h	g N/d	°C		kg SS/m³				%ofSS		gO_2/m^3					
27/9 -93	D4	Mon.	15.9	0.0	2.7	5.5	7.7	1.30	1.54	0.75	0.91		0	16.1	16.6	2.82	4.26	12.51		43					52	34	
28/9 -93	D4	Tue.	15.9	0.0	2.7	5.5	7.7	1.00	1.96	0.75	1.14		0	16.5	17.1	2.82	3.98	12.71	0.011	44	312	143	48	39	74	42	
29/9 -93	D4	Wed.	15.9	0.0	2.7	5.5	7.7	1.20	1.82	0.75	1.59		0	16.7	17.4	2.88				44							
30/9 -93	D4	Thu.	15.9	0.0	2.7	5.5	7.7	1.00	1.97	0.75	1.58		0	16.7	17.2	2.88	3.69	12.53	0.018	44	317	150	41	44	80	50	
1/10 -93	D4	Fri.	15.9	0.0	2.7	5.5	7.7	0.90	2.07	0.75			0	16.0	17.3												
2/10 -93	D4	Sat.	15.9	0.0	2.7	5.5	7.7	1.00	2.02	0.75			0														
3/10 -93	D4	Sun.	15.9	0.0	2.7	5.5	7.7	1.00	1.98	0.75			0			2.40	3.32	12.23	0.016	42	241	92	33.5	46			
4/10 -93	D4	Mon.	15.9	0.0	2.7	5.5	7.7	1.40	1.61	0.75	1.59		0	16.3	16.9	2.40	3.86	12.23		42					56	40	
5/10 -93	D4	Tue.	15.9	0.0	2.7	5.5	7.7	1.50	1.50	0.75	1.5		0	15.6	16.6	2.44	3.92	11.33	0.016	41	247	107	24	33	49	35	
6/10 -93	D4	Wed.	15.9	0.0	2.7	5.5	7.7	1.30	1.74	0.70	1.53	92	0	15.9	17.0												
7/10 -93	D4	Thu.	15.9	0.0	2.7	5.5	7.7	0.90	1.46	0.70	1.59		0	16.2	17.7	2.56	4.00	13.49		41					56		
8/10 -93	D4	Fri.	15.9	0.0	2.7	5.5	7.7	1.00	1.74	0.70			0														
9/10 -93	D4	Sat.	15.9	0.0	2.7	5.5	7.7	1.60	1.40	0.70			0														
10/10 -93	D4	Sun.	15.9	0.0	2.7	5.5	7.7	2.60	0.45	0.70			0														
11/10 -93	D4	Mon.	15.9	0.0	2.7	5.5	7.7	1.40	0.58	0.70	1.59		0														
12/10 -93	D4	Tue.	15.9	0.0	2.7	5.5	7.7	1.60	1.35	0.70	1.53		0	15.3	16.2	2.59	4.51	12.93	0.014	42	187	55	19	16	68		
13/10 -93	D4	Wed.	15.9	0.0	2.7	5.5	7.7	2.40	0.48	0.70	1.53		0	14.2	15.5												
14/10 -93	D4	Thu.	15.9	0.0	2.7	5.5	7.7	2.30	0.74	0.70	1.59	121	0	14.4	14.7	1.89	4.86	9.87	0.014	46	176	71	32.5	34	46		
15/10 -93	D4	Fri.	15.9	0.0	2.7	5.5	7.7	1.70	1.33	0.70			0	14.9	14.9												
16/10 -93	D4	Sat.	15.9	0.0	2.7	5.5	7.7	1.50	1.48	0.70			0														
17/10 -93	D4	Sun.	15.9	0.0	2.7	5.5	7.7	1.40	1.56	0.70			0			2.33	3.46	10.15	0.014	41	198	66	31	29			
18/10 -93	D4	Mon.	15.9	0.0	2.7	5.5	7.7	1.30	1.68	0.70	1.6		0	14.6	15.1	2.33	3.29	10.15		41					43		
19/10 -93	D4	Tue.	15.9	0.0	2.7	5.5	7.7	1.20	1.82	0.70	1.5		0	15.2	15.8	2.29	3.09	10.09	0.016	42	277	96	32.5	37	55		
20/10 -93	D4	Wed.	15.9	0.0	2.7	5.5	7.7	1.30	1.65	0.70	1.6		0	15.2	15.8												
21/10 -93	D4	Thu.	15.9	0.0	2.7	5.5	7.7	1.20	1.53	0.70	1.46	104	0	14.7	15.5	2.37	3.60	11.31	0.016	41	332	98	40	33	56	40	
22/10 -93	D4	Fri.	15.9	0.0	2.7	5.5	7.7	1.10	1.85	0.70			0	14.5	15.5												
23/10 -93	D4	Sat.	15.9	0.0	2.7	5.5	7.7	1.00	2.00	0.70			0														
24/10 -93	D4	Sun.	15.9	0.0	2.7	5.5	7.7	1.10	1.93	0.70			0			2.06	2.57	8.33	0.015	39	273	87	32	33			
25/10 -93	D4	Mon.	15.9	0.0	2.7	5.5	7.7	1.20	1.79	0.70	1.59		0	14.5	15.2	2.06	2.70	8.33		39					45	45	
26/10 -93	D4	Tue.	15.9	0.0	2.7	5.5	7.7	1.20	1.84	0.70	1.59	104	0	14.7	15.3	2.23	3.39	12.69	0.014	39	352	128	44	36	74	43	
27/10 -93	D4	Wed.	15.9	0.0	2.7	5.5	7.7	1.10	1.90	0.70	1.5		0	14.5	15.4												
28/10 -93	D4	Thu.	15.9	0.0	2.7	5.5	7.7	1.10	1.90	0.70	1.51	108	0	14.8	15.5	2.29	2.98	10.07	0.014	38	335	129	42	40	76	54	
29/10 -93	D4	Fri.	15.9	0.0	2.7	5.5	7.7	1.08	1.87	0.70			0	14.5	15.4												
30/10 -93	D4	Sat.	15.9	0.0	2.7	5.5	7.7	0.94	2.02	0.70			0														
31/10 -93	D4	Sun.	15.9	0.0	2.7	5.5	7.7	0.97	2.03	0.70			0			2.31	2.66	8.55	0.010	40	295	91	49	43			
1/11 -93	D4	Mon.	15.9	0.0	2.7	5.5	7.7	1.10	1.90	0.70	1.54		0	14.7	15.2	2.31	2.79	8.55		40					63	58	
2/11 -93	D4	Tue.	15.9	0.0	2.7	5.5	7.7	1.00	1.78	0.70	1.56		0	14.8	15.2	2.32	2.94	9.31	0.014	41	372	156	58	44	65	53	
3/11 -93	D4	Wed.	15.9	0.0	2.7	5.5	7.7	1.07	1.93	0.70	1.5		0	14.8	15.2												
4/11 -93	D4	Thu.	15.9	0.0	2.7	5.5	7.7	1.05	1.95	0.70	1.58	121	0	14.6	15.4	2.45	3.37	12.55	0.012	41	365	125	37	28	98	47	
5/11 -93	D4	Fri.	15.9	0.0	2.7	5.5	7.7	0.90	1.99	0.70			0	14.5	15.5												
6/11 -93	D4	Sat.	15.9	0.0	2.7	5.5	7.7	0.93	2.07	0.70			0														
7/11 -93	D4	Sun	15.9	0.0	2.7	5.5	7.7	0.90	2.09	0.70			0			2.61	3.55	14.51	0.013	42	325	114	56	37			
8/11 -93	D4	Mon.	15.9	0.0	2.7	5.5	7.7	1.17	1.83	0.70	1.56		0	14.5	15.5	2.61	3.92	14.51		42					87	55	
9/11 -93	D4	Tue.	15.9	0.0	2.7	5.5	7.7	1.17	1.83	0.70	1.59		0	14.4	15.2	2.71	3.35	9.85	0.095	42	364	136	49	95	63	51	
10/11 -93	D4	Wed.	15.9	0.0	2.7	5.5	7.7	1.07	1.82	0.70	1.57		0	14.1	15.2	2.64	3.94	14.79	0.013	42	343	136	65	37	95	55	
11/11 -93	D4	Thu.	15.9	0.0	2.7	5.5	7.7	0.85	1.63	0.70	1.51		0			2.41	3.54	12.62		41					93	51	
12/11 -93	D4	Fri.	15.9	0.0	2.7	5.5	7.7	1.50	1.46	0.70		96	0														
13/11 -93	D4	Sat.	15.9	0.0	2.7	5.5	7.7	1.24	1.76	0.70			0														
14/11 -93	D4	Sun	15.9	0.0	2.7	5.5	7.7	1.89	1.11	0.70			0			2.08	4.02	9.78	0.016	41	153	62	32	27			
15/11 -93	D4	Mon.	15.9	0.0	2.7	5.5	7.7	1.55	1.45	0.70	1		0	14.4	13.5	2.08	3.38	9.78							44	41	
16/11 -93	D4	Tue.	15.9	0.0	2.7	5.5	7.7	1.26	1.73	0.70	1		0														
17/11 -93	D4	Wed.	15.9	0.0	2.7	5.5	7.7	1.26	1.74	0.70	1.04		0	13.5	14.1	2.67	3.29	8.80		40			39		59	43	
18/11 -93	D4	Thu.	15.9	0.0	2.7	5.5	7.7	1.12	1.87	0.70	1.03	104	0	13.5	14.6	2.68	2.90	7.20	0.017	40	305	120	46	37		43	
19/11 -93	D4	Fri.	15.9	0.0	2.7	5.5	7.7	1.20	1.80	0.70			0	13.9	14.8												
20/11 -93	D4	Sat.	15.9	0.0	2.7	5.5	7.7	1.02	1.97	0.70			0														
21/11 -93	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.11	1.89	0.70			0			1.69	2.61	10.42	0.015	40	308	103	45	43			
22/11 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.00	2.00	0.70	1.11		0	13.4	14.0	1.69	2.51	10.42	0.016	40			43	31	70	44	
23/11 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	1.04	1.95	0.70	1.06	88	0	13.3	14.1	2.20	2.38	13.99	0.019	41	186	103	43	39	64	44	
24/11 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	1.18	1.82	0.70	1.06		0	13.1	13.4	2.62	4.04	15.33	0.019	41	140	90	43	35	91	50	
25/11 -93	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.07	1.77	0.70	1.35		0	13.3	13.3	2.57	4.10	15.93	0.020	39	297	115	32	28	74	44	
26/11 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	1.11	1.76	0.75			0	13.2	13.7												
27/11 -93	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.22	1.78	0.75			0														
28/11 -93	D4	Sun	16.0	0.0	2.7	5.5	7.7	1.12	1.87	0.75			0			2.54	3.96	14.91	0.015	39	275	101	33	33			
29/11 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	0.88	1.80	0.75	1.4		0	12.8	13.7			14.91							87	40	
30/11 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	0.94	2.06	0.75	1.33		0	13.0	13.7	2.54	3.61	14.35	0.016	39	268	134	38	36	74	47	
1/12 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	0.94	2.06	0.75	1.38		0	13.3	13.9												
2/12 -93	D4	Thu	16.0	0.0	2.7	5.5	7.7	1.35	1.65	0.70	1.31	88	0	12.9	13.3	2.64	3.59	10.11		41					61		
3/12 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	1.99	1.01	0.70			0														
4/12 -93	D4	Sat.	16.0	0.0	2.7	5.5	7.7	2.84	0.38	0.70			0														
5/12 -93	D4	Sun	16.0	0.0	2.7	5.5	7.7	2.81	0.42	0.70			0			2.81	10.20	16.73	0.045	39	116	49	34	25			
6/12 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	2.13	0.86	0.70	1.57		0	10.7	11.0	2.81	7.08	16.73		39					58	38	
7/12 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	2.40	0.74	0.70	1.56		0	11.2	11.7	3.08	7.91	15.63	0.025	39	198	71	48	36	84	50	

Date	PROCESS	Day of the week	Volume: Total activated sludge	Return sludge stab.	Deoxygenation tank	Anoxic tank	Aerated tanks	Flow: To pilot plant (Q1)	Recirculated water (Q9)	Return activated sludge (Q5)	Waste activated sludge (Q11)	Addition of FeSO4	Addition of NaNO3	Temperature: Primary settled WW	Activated sludge tanks	Suspended solids: Activated sludge tanks	Deoxygenation tank (estimated)	RAS	Effluent	Ash content, activated sludge	COD: Primary settled WW	COD filtered sample: Primary settled WW	Anoxic tank	Effluent	RAS	Deoxygenation tank	RBC
Unit			m³					l/s			m³/d	gFe/h	g N/d	°C		kg SS/m³				%ofSS		gO₂/m³					
8/12 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	2.74	0.42	0.70			0	9.8	10.5												
9/12 -93	D4	Thu.	16.0	0.0	2.7	5.5	7.7	2.10	0.85	0.70			0	10.6	11.1	2.72	6.07	12.67	0.020	40	217	103	53	32	55	51	
10/12 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	1.70	1.28	0.70			0	11.2	11.9												
11/12 -93	D4	Sat.	16.0	0.0	2.7	5.5	7.7	2.10	0.91	0.70			0														
12/12 -93	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.80	1.18	0.70			0			2.69	4.50	10.00	0.017	40	159	66	39	27			
13/12 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.40	1.57	0.70	1.67		0	10.5	11.3	2.69	3.72	10.00		40					34	32	
14/12 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	1.60	1.44	0.70	1.64	92	0	11.2	11.7	2.58	4.21	11.98	0.018	40	237	107	48	38	51	43	
15/12 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	1.50	1.45	0.70	1.59		0	11.0	11.7												
16/12 -93	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.40	1.48	0.70	1.63		0														
17/12 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	1.90	1.11	0.70			0	8.2	8.9												
18/12 -93	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.80	1.20	0.70			0														
19/12 -93	D4	Sun.	16.0	0.0	2.7	5.5	7.7	2.60	0.54	0.70			0														
20/12 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	2.80	0.46	0.70	1.61		0			2.53											
21/12 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	2.90	0.46	0.70	1.68	88	0			1.95	6.98	11.82	0.100	41	165	85	46	37	56	41	
22/12 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	2.50	0.53	0.70	1.15		0	9.4	9.7												
23/12 -93	D4	Thu.	16.0	0.0	2.7	5.5	7.7	2.60	0.44	0.70			0														
24/12 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	2.20	0.85	0.70			0														
25/12 -93	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.40	1.61	0.70			0														
26/12 -93	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.40	1.40	0.70			0														
27/12 -93	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.30	1.70	0.70	1.13		0	9.7	10.9	2.31	3.56	12.22		40							
28/12 -93	D4	Tue.	16.0	0.0	2.7	5.5	7.7	1.20	1.76	0.70	1.13		0			2.45				41							
29/12 -93	D4	Wed.	16.0	0.0	2.7	5.5	7.7	1.50	1.51	0.70	1.18		0			2.53				40							
30/12 -93	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.90	1.12	0.70		92	0														
31/12 -93	D4	Fri.	16.0	0.0	2.7	5.5	7.7	2.40	0.66	0.70			0														
1/1 -94	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.90	1.14	0.70			0														
2/1 -94	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.70	1.27	0.70			0														
3/1 -94	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.90	1.13	0.70			0	10.3	11.0	2.59	5.01	10.85	0.043	38	251	89	51	40	61		
4/1 -94	D4	Tue.	16.0	0.0	2.7	5.5	7.7	1.50	1.49	0.70			0	10.3	11.1												
5/1 -94	D5	Wed.	16.0	0.0	0.0	8.2	7.7	2.00	1.04	0.70			0	8.7	10.4	2.62	5.34	12.63		38							
6/1 -94	D5	Thu.	16.0	0.0	0.0	8.2	7.7	2.20	0.88	0.70			0														
7/1 -94	D5	Fri.	16.0	0.0	0.0	8.2	7.7	2.40	0.68	0.70			0														
8/1 -94	D5	Sat.	16.0	0.0	0.0	8.2	7.7	2.50	0.49	0.70			0														
9/1 -94	D5	Sun.	16.0	0.0	0.0	8.2	7.7	2.50	0.99	0.70			0														
10/1 -94	D5	Mon.	16.0	0.0	0.0	8.2	7.7	1.80	1.21	0.70	0.57		0			2.74	5.05	8.23	0.041	39	231	74	39	36	42		
11/1 -94	D5	Tue.	16.0	0.0	0.0	8.2	7.7	1.60	1.41	0.70			0	9.6	10.4												
12/1 -94	D5	Wed.	16.0	0.0	0.0	8.2	7.7	1.50	1.45	0.70			0	9.8	10.8	2.95	4.76	17.80	0.038	38	271	93	50	40	103		
13/1 -94	D5	Thu.	16.0	0.0	0.0	8.2	7.7	1.20	0.48	0.70			0	7.2	9.7												
14/1 -94	D5	Fri.	16.0	0.0	0.0	8.2	7.7	2.70	0.41	0.70			0	8.6	8.9	2.75	8.97	11.38		69							
15/1 -94	D5	Sat.	16.0	0.0	0.0	8.2	7.7	2.20	0.88	0.70			0														
16/1 -94	D5	Sun.	16.0	0.0	0.0	8.2	7.7	1.90	1.14	0.70			0						0.192		160	65	53	48			
17/1 -94	D5	Mon.	16.0	0.0	0.0	8.2	7.7	1.80	1.23	0.70			0	9.4	9.9										61		
18/1 -94	D5	Tue.	16.0	0.0	0.0	8.2	7.7	1.40	1.67	0.70			0	9.4	10.3	2.31	3.49	11.70		40					64		
19/1 -94	D4	Wed.	16.0	0.0	2.7	5.5	7.7	1.90	1.21	0.70	1.19		0	8.7	10.7	2.26	4.28	14.03		40	157	84	39		94		
20/1 -94	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.60	1.33	0.70	1.28		0	9.5	10.2	2.22	3.77	11.60		41					75		
21/1 -94	D4	Fri.	16.0	0.0	2.7	5.5	7.7	1.50	1.49	0.70		138	0	9.9	10.9	2.13	3.41	5.62		41							
22/1 -94	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.60	1.45	0.70			0														
23/1 -94	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.60	1.40	0.70			0														
24/1 -94	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.80	1.27	0.70	1.28		0	9.2	10.1	1.90	3.45	8.86		41							
25/1 -94	D4	Tue.	16.0	0.0	2.7	5.5	7.7	1.60	1.38	0.70	1.12	42	0	9.5	10.3	2.10	3.53	14.12	0.153	41	341	78	44	47	80	42	
26/1 -94	D4	Wed.	16.0	0.0	2.7	5.5	7.7	1.50	1.48	0.70	1.07		0	9.3	10.6												
27/1 -94	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.80	1.26	0.70	1.21		0	9.3	10.2	2.52	4.59	6.66	0.060	40	205	62	33	41	55	74	
28/1 -94	D4	Fri.	16.0	0.0	2.7	5.5	7.7	1.80	1.22	0.70		100	0	9.0	9.5												
29/1 -94	D4	Sat.	16.0	0.0	2.7	5.5	7.7	1.80	1.23	0.70			0														
30/1 -94	D4	Sun.	16.0	0.0	2.7	5.5	7.7	1.40	1.59	0.70			0	8.0	8.9	2.15	3.40	10.40	0.042	36	261	63		34			
31/1 -94	D4	Mon.	16.0	0.0	2.7	5.5	7.7	1.80	1.26	0.70	1.35		0			2.15	3.92	10.40		36					32	29	
1/2 -94	D4	Tue.	16.0	0.0	2.7	5.5	7.7	1.70	1.37	0.70	1.16		0			2.29	3.96	10.90	0.048	37	354	111	41	36	57	52	
2/2 -94	D5	Wed.	16.0	0.0	0.0	8.2	7.7	1.10	1.56	0.70	1.41		0	9.8	10.0												
3/2 -94	D5	Thu.	16.0	0.0	0.0	8.2	7.7	1.30	1.77	0.70	1.41		0	10.2	11.2	2.75	3.99	9.78		38					52		
4/2 -94	D5	Fri.	16.0	0.0	0.0	8.2	7.7	1.20	1.86	0.70		117	0	10.3	11.0												
5/2 -94	D5	Sat.	16.0	0.0	0.0	8.2	7.7	1.10	1.84	0.70			0														
6/2 -94	D5	Sun.	16.0	0.0	0.0	8.2	7.7	0.80	1.22	0.70			0														
7/2 -94	D5	Mon.	16.0	0.0	0.0	8.2	7.7	1.00	1.33	0.70	1.41		0	10.3	10.9	4.19	5.94	11.14	0.031	36	365	119	39	29	58		
8/2 -94	D5	Tue.	16.0	0.0	0.0	8.2	7.7	1.00	1.65	0.70	1.7		0	10.8	11.3	2.61	3.54	12.25	0.034	36	296	105	49	39	71		
9/2 -94	D4	Wed.	16.0	0.0	2.7	5.5	7.7	1.00	1.45	0.70	1.36		0	10.8	11.5												
10/2 -94	D4	Thu.	16.0	0.0	2.7	5.5	7.7	1.30	1.67	0.70	1.35		0	9.5	11.0	2.81	4.14	20.81	0.020	37	499	98	54	38	335	109	

Date	PROCESS	Day of the week	BOD: Primary settled WW	Effluent	Phosphate: Effluent	Anoxic tank	Total phosphorus: Effluent	RBC	RAS	Ammonium: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrate: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrite: RBC	Anoxic tank	Effluent	Deoxygenation tank
Unit			g O$_2$/m^3		g P/m^3		g P/m^3			g N/m^3					g N/m^3					g N/m^3			
14/6 -90	A	Thu.			0.29					18.2		18.3			0.10								
15/6 -90	A	Fri.																					
18/6 -90	A	Mon.			0.23					17.7		16.9			0.10								
19/6 -90	A	Tue.																					
20/6 -90	A	Wed.			0.04					6.2		6.5			0.10		0.4						
4/7 -90	A	Wed.			0.15					12.1		11.5			0.10		0.1						
5/7 -90	A	Thu.																					
6/7 -90	A	Fri.																					
9/7 -90	A	Mon.			0.14																		
10/7 -90	A	Tue.			0.18					11.1		10.3			0.10		1.1						
11/7 -90	A	Wed.			0.17																		
12/7 -90	A	Thu.			0.24					13.1		12.9			0.10		0.3						
13/7 -90	A	Fri.																					
16/7 -90	A	Mon.			0.26					15.4		15.5			0.10		0.2						
17/7 -90	A	Tue.																					
18/7 -90	A	Wed.														0.1							
19/7 -90	A	Thu.			0.19					16.3		15.2			0.10		0.4						
20/7 -90	A	Fri.																					
23/7 -90	A	Mon.														0.7							
24/7 -90	A	Tue.			0.12					18.0		17.6			0.10		0.3						
25/7 -90	A	Wed.																					
26/7 -90	A	Thu			0.17					19.1		18.7			0.10		0.5						
27/7 -90	A	Fri.																					
30/7 -90	A	Mon.			0.25					16.1		16.1			0.10	0.2	1.8						
31/7 -90	A	Tue.														2.2	3.5						
1/8 -90	A	Wed.			0.19					17.4		16.9			0.10	2.5	1.5						
2/8 -90	A	Thu.														1.0	3.4						
3/8 -90	A	Fri.														5.1							
6/8 -90	A	Mon.			0.26					17.7		17.9			0.10	10.6	4.3						
7/8 -90	A	Tue.														0.4	5.1						
8/8 -90	A	Wed.														5.9	8.0						
9/8 -90	A	Thu.			0.08					15.7		15.1			0.08	7.8	7.4						
10/8 -90	A	Fri.														6.1							
13/8 -90	A	Mon.			0.11					19.4		19.1			0.03	10.1	8.2						
14/8 -90	A	Tue.														5.9	4.1						
15/8 -90	A	Wed.			0.20											2.6	2.4						
16/8 -90	A	Thu.			0.14					15.2		17.1			0.09	2.8	2.7						
17/8 -90	A	Fri														4.8							
20/8 -90	A	Mon.														11.2	10.1						
21/8 -90	A	Tue.			0.11					16.7		15.2			0.17	10.3	8.8						
22/8 -90	A	Wed.														7.7	6.0						
23/8 -90	A	Thu.			0.19					17.6		16.9			0.03	4.8	4.2						
24/8 -90	A	Fri.														3.2							
27/8 -90	A	Mon.			0.23					19.9		19.5			0.06	11.6	6.9						
28/8 -90	A	Tue.			0.35											4.5	3.1						
29/8 -90	A	Wed			1.14					19.7		19.4			0.01	4.3	2.2						
30/8 -90	A	Thu.														3.0	1.2						
31/8 -90	A	Fri.														2.5							
6/9 -90	A	Thu			0.31					20.0		18.0			0.10	2.0	1.1						
7/9 -90	A	Fri.														0.5							
10/9 -90	A	Mon.			0.07					19.6		18.4			0.10	0.4	1.0						
11/9 -90	A	Tue.														4.0							
12/9 -90	A	Wed.														2.1							
13/9 -90	A	Thu.			0.23										0.10	1.3	1.1						
18/9 -90	A	Tue.			1.06					13.1		12.0			0.10		0.7						
20/11 -90	A	Tue.			0.82					17.2		18.8				2.5	0.5						
21/11 -90	A	Wed.			1.24											0.9	0.1						
22/11 -90	A	Thu			2.16	2.52			219	19.6		18.0			0.13	1.0	0.2						
23/11 -90	A	Fri.																					
25/11 -90	A	Sun.			1.79											2.0	0.2						
26/11 -90	A	Mon.			2.29											1.9	0.1						
27/11 -90	A	Tue.			1.84	2.60			168	21.1		19.5			0.00	1.5	0.1						
28/11 -90	A	Wed.			1.96																		
29/11 -90	A	Thu.			1.56	2.76			162	20.7		18.5				2.4	0.2						
30/11 -90	A	Fri																					
3/12 -90	A	Mon.			1.61	2.21			181							3.3	0.6						
4/12 -90	A	Tue.			1.79	2.22				21.0		20.1				9.5	6.1						
5/12 -90	A	Wed.			1.87	2.40			188	25.3		24.0				13.2	10.0						
6/12 -90	A	Thu.			2.28	2.68									0.10	13.8	11.3						
7/12 -90	A	Fri.																					
9/12 -90	A	Sun.			2.09	1.63										27.2	21.4						
10/12 -90	A	Mon.			1.46	1.78			143	18.3		17.8			0.10	21.4	17.7						

Date	PROCESS	Day of the week	BOD: Primary settled WW	Effluent	Phosphate: Effluent	Anoxic tank	Total phosphorus: Effluent	RBC	RAS	Ammonium: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrate: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrite: RBC	Anoxic tank	Effluent	Deoxygenation tank
Unit			g O_2/m^3		g P/m^3		g P/m^3			g N/m^3					g N/m^3					g N/m^3			
11/12 -90	A	Tue.			1.67	1.69									0.10	19.9	14.1						
12/12 -90	A	Wed.			1.52	1.44										19.7	14.0						
13/12 -90	A	Thu.			1.64	1.76			142	17.8		15.8			0.14	20.5	14.0						
14/12 -90	A	Fri.																					
17/12 -90	A	Mon.			2.34	2.56			162	23.5		22.8			0.10	8.0	4.5						
18/12 -90	A	Tue.			2.04	1.98										4.8	0.7						
19/12 -90	A	Wed.			2.45	2.40			156	23.1		21.5				3.7	0.6						
20/12 -90	A	Thu.			1.98	1.56										2.2	0.8						
7/1 -91	A	Mon.			0.02	0.06				7.5		7.2			1.37	9.1	7.9						
8/1 -91	A	Tue.			0.00	0.02										12.4	10.9						
9/1 -91	A	Wed.			0.01	0.02			146	4.8		5.6				12.3	11.0						
10/1 -91	A	Thu.			0.01	0.03										11.7	11.5						
11/1 -91	A	Fri.										6.5											
12/1 -91	A	Sat.																					
13/1 -91	A	Sun.																					
14/1 -91	A	Mon.			0.02	0.15										7.6	7.4						
15/1 -91	A	Tue.			0.03	0.21				13.5		12.9			0.28	3.4	3.8						
16/1 -91	A	Wed.			0.09	0.30											2.9						
17/1 -91	A	Thu.			0.08	0.44				16.1		15.4				1.5	1.1						
18/1 -91	A	Fri.																					
19/1 -91	A	Sat.																					
20/1 -91	A	Sun.																					
21/1 -91	A	Mon.				0.11										4.2							
22/1 -91	A	Tue.			0.03					16.6		15.3				1.5	0.7						
23/1 -91	A	Wed.														1.0	0.6						
24/1 -91	A	Thu.			0.03					15.6		15.6				0.7	0.5						
25/1 -91	A	Fri.																					
26/1 -91	A	Sat.																					
27/1 -91	A	Sun.																					
28/1 -91	A	Mon.			0.04					18.1		19.1			0.16	2.4	1.9						
29/1 -91	A	Tue.			0.02											0.8	0.5						
30/1 -91	A	Wed.			0.02					21.7		19.1				1.1	0.5						
31/1 -91	A	Thu.			0.02											2.2	0.7						
1/2 -91	A	Fri.																					
2/2 -91	A	Sat.																					
3/2 -91	A	Sun.																					
4/2 -91	A	Mon.														10.8							
5/2 -91	A	Tue.			0.11					23.4		23.5			0.04	10.2	7.3						
6/2 -91	A	Wed.														9.7	6.6						
4/2 -92	B	Mon			0.02					21.8					0.32	0.7	0.1						
6/2 -92	B	Wed.			0.01					21.3		18.9				5.8	4.1						
10/2 -92	B	Mon.			0.01					13.2		13.9			0.57	4.0	3.7						
13/2 -92	B	Thu.								8.7		9.4				15.1	14.8						
17/2 -92	B	Mon.			0.01							19.5				2.5	1.6						
18/2 -92	B	Tue.			0.02					21.3		20.6			0.32	7.2	6.1					1.54	
24/2 -92	B	Mon.			0.03					12.6		13.2			1.20	3.9	1.4					0.31	
27/2 -92	B	Thu.			0.01					21.0		18.8				2.0	0.6					0.14	
2/3 -92	B	Mon.			0.04							13.8				5.9	3.0					1.86	
9/3 -92	B	Mon.			0.04							20.4				3.1	0.9						
12/3 -92	B	Thu.			0.03					9.6		10.4				5.1	2.7						
16/3 -92	B	Mon.			0.04							17.1				6.5	4.2					1.95	
19/3 -92	B	Thu.			0.04					16.4		15.8			0.46	6.3	3.3						
25/3 -92	B	Wed.			0.05					14.3		13.9			0.35	3.0	0.8					0.24	
26/3 -92	B	Thu.			0.06					14.8		13.8			0.30	6.7	3.3						
31/3 -92	B	Tue.			0.05					24.8		22.0			0.05	1.9	0.9						
1/4 -92	B	Wed.			0.07					20.9		25.5			0.35	5.9	2.1					0.41	
2/4 -92	B	Thu.			0.07					23.3		21.9			0.15		2.0						
6/4 -92	B	Mon.			0.08					16.5		16.6			0.32	2.1	0.7					0.07	
7/4 -92	B	Tue.			0.05					20.7		17.6			0.04		0.8						
9/4 -92	B	Thu.			0.03					22.6		21.5			0.08	2.5	0.6						
13/4 -92	B	Mon.			0.08					14.2		17.3			0.62		5.9						
14/4 -92	B	Tue.			0.06					15.9		16.9			0.51		1.0						
15/4 -92	B	Wed.			0.01					14.6		14.3			0.51		0.4						
5/5 -92	B	Tue.			0.10					16.3		15.7			0.32		0.6					0.03	
10/5 -92	B	Sun.			0.06					19.5		19.5			0.13		0.3					0.01	
11/2 -93																							
14/2 -93	C5	Sun.		6.2	0.06		0.35			25.6		1.1			0.07		1.7					0.34	
15/2 -93	C5	Mon.											4.0			1.8		9.7		0.38			
16/2 -93	C5	Tue.																					
17/2 -93	C5	Wed		4.0	0.04		0.35			13.4		9.0	2.2		0.37	0.1	0.1	7.1		0.32		0.05	
18/2 -93	C3	Thu.																					
19/2 -93	C3	Fri.																					

Date	PROCESS	Day of the week	BOD: Primary settled WW	Effluent	Phosphate: Effluent	Anoxic tank	Total phosphorus: Effluent	RBC	RAS	Ammonium: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrate: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrite: RBC	Anoxic tank	Effluent	Deoxygenation tank
Unit			g O_2/m^3		g P/m^3		g P/m^3			g N/m^3					g N/m^3					g N/m^3			
20/2 -93	C3	Sat.																					
21/2 -93	C3	Sun.																					
22/2 -93	C3	Mon.																					
23/2 -93	C3	Tue.																					
24/2 -93	C3	Wed.																					
25/2 -93	C3	Thu.		9.7	0.03		0.47			24.0		13.4	4.7		0.02	0.7	2.1	10.3		0.56		0.80	
26/2 -93	C3	Fri.																					
27/2 -93	C3	Sat.																					
28/2 -93	C3	Sun.		11.4	0.05		0.62			35.9		12.0	5.9		0.04	3.0	2.1	9.7		0.37		0.27	
1/3 -93	C1	Mon.																					
2/3 -93	C1	Tue.																					
3/3 -93	C1	Wed.																					
4/3 -93	C1	Thu.		9.6	0.01		0.39			24.6		11.0	2.0		0.02	2.4	4.0	11.3		0.51		0.64	
5/3 -93	C1	Fri.																					
6/3 -93	C1	Sat.																					
7/3 -93	C1	Sun.		8.4	0.04		0.53			24.6		11.3	2.7		0.02	2.3	4.7	13.3		0.43		0.42	
8/3 -93	C2	Mon.																					
9/3 -93	C2	Tue.																					
10/3 -93	C2	Wed.																					
11/3 -93	C2	Thu.																					
12/3 -93	C2	Fri.																					
13/3 -93	C2	Sat.																					
14/3 -93	C	Sun.		11.1	0.03		0.34			26.7		11.4	4.5		0.02	2.3	5.1	12.6		0.84		0.88	
15/3 -93	C	Mon.																					
16/3 -93	C	Tue.		11.2	0.02		0.37			22.8		12.5	3.5		0.02	2.7	5.0	13.1		0.51		0.82	
17/3 -93	C4	Wed.																					
18/3 -93	C4	Thu.																					
19/3 -93	C4	Fri.																					
20/3 -93	C4	Sat.																					
21/3 -93	C4	Sun.			0.12		0.44			22.5		11.3	2.5		0.06	0.6	0.6	9.3		0.50		0.39	
22/3 -93	C4	Mon.																					
23/3 -93	C4	Tue.																					
24/3 -93	C4	Wed.		12.9	0.03		0.64	0.05		20.7		11.1	2.5		0.13	0.3	0.1	8.9		0.48		0.02	
25/3 -93	C4	Thu.	105	9.7	0.03		0.35	0.08		22.2		10.9	1.8		0.08	0.4	0.1	9.7		0.49		0.03	
26/3 -93	C4	Fri.																					
27/3 -93	C4	Sat.																					
28/3 -93	C4	Sun.		9.5	0.04		0.46	0.10		22.9		10.3	1.1		0.02	1.6	0.7	10.4		0.66		0.27	
29/3 -93	C2	Mon.		10.7	0.04		0.45	0.11		28.3		11.7	1.8		0.01	0.7	1.3	10.5		0.73		0.43	
30/3 -93	C2	Tue.																					
31/3 -93	C2	Wed.		12.6	0.02		0.57	0.11		23.3		10.4	0.7		0.01	0.6	0.7	10.3		0.68		0.35	
1/4 -93	C2	Thu.		9.8	0.30		0.40	0.08		24.2		13.0	1.3		0.01	0.3	0.4	11.3		1.04		0.19	
2/4 -93	C2	Fri.																					
3/4 -93	C2	Sat.																					
4/4 -93	C2	Sun.		8.6	0.03		0.39	0.09		23.5		10.2	1.8		0.02	3.5	7.5	17.0		0.98		0.86	
5/4 -93	C3	Mon.																					
6/4 -93	C3	Tue.																					
7/4 -93	C3	Wed.																					
8/4 -93	C3	Thu.																					
9/4 -93	C3	Fri.																					
10/4 -93	C3	Sat.																					
11/4 -93	C3	Sun.																					
12/4 -93	C3	Mon.		8.6	0.02		0.38	0.09		28.3		10.8	4.0		0.01	5.0	5.8	19.3		1.31		0.80	
13/4 -93	C3	Tue.		17.4	0.05		0.65	0.19		22.3		14.9	3.2		0.01	0.5	5.4	15.3		1.40		2.04	
14/4 -93	C5	Wed.																					
15/4 -93	C5	Thu.		14.2	0.04		0.49	0.19		24.1		10.4	0.8		0.01	1.5	0.2	10.9		0.50		0.13	
16/4 -93	C5	Fri.																					
17/4 -93	C5	Sat.																					
18/4 -93	C5	Sun.		14.2	0.02		0.49			21.0		11.4	1.2		0.02	2.2	0.6	12.2		0.65		0.41	
19/4 -93	C5	Mon.		12.3	0.05		0.45	0.17		21.0		10.9	0.9		0.01	0.3	1.6	14.3		0.73		1.36	
20/4 -93	C5	Tue.																					
21/4 -93	C5	Wed.																					
22/4 -93	C5	Thu.		12.0	0.18		0.41	0.13		21.9		10.4	0.4		0.01	0.0	1.2	11.6				0.74	
23/4 -93	C2	Fri.																					
24/4 -93	C2	Sat.																					
25/4 -93	C2	Sun.		9.2	0.06		0.48	0.13		21.7		8.5	0.5		0.01	3.7	4.0	13.8		0.45		0.95	
26/4 -93	C2	Mon.																					
27/4 -93	C2	Tue.		7.6	0.10		0.41	0.20		23.9		14.1	1.2		0.01	0.2	1.5	13.9		0.66		0.71	
28/4 -93	C2	Wed.			0.15		0.37			21.0		8.2	0.1		0.01	1.7	2.3	10.5		0.30		0.85	
29/4 -93	C2	Thu.	126	27.0	0.08		1.44			24.6		9.4	0.2		0.01	1.9	2.5	11.5		0.40		1.01	
30/4 -93	C4	Fri.																					
1/5 -93	C4	Sat.																					
2/5 -93	C4	Sun.																					

Date	PROCESS	Day of the week	BOD: Primary settled WW	Effluent	Phosphate: Effluent	Anoxic tank	Total phosphorus: Effluent	RBC	RAS	Ammonium: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrate: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrite: RBC	Anoxic tank	Effluent	Deoxygenation tank
Unit			g $\mathrm{O_2/m^3}$		g $\mathrm{P/m^3}$		g $\mathrm{P/m^3}$			g $\mathrm{N/m^3}$					g $\mathrm{N/m^3}$					g $\mathrm{N/m^3}$			
3/5 -93	C4	Mon.																					
4/5 -93	C4	Tue.		8.0	0.04		0.38			27.2		10.0			0.01	1.4	0.9	11.2		0.54		0.78	
5/5 -93	C4	Wed.		8.7	0.03		0.38			21.4		10.3	0.6		0.01	1.0	0.5	9.3		0.37		0.35	
6/5 -93	C4	Thu.		9.3	0.03		0.34			38.8		11.8	0.8		0.01	1.0	0.7	10.8		0.45		0.54	
7/5 -93	C5	Fri.																					
8/5 -93	C5	Sat.																					
9/5 -93	C5	Sun.		13.0	0.03		0.67			27.8		9.8	0.3		0.01	4.4	4.0	13.1		0.42		0.93	
10/5 -93	C5	Mon.		6.6	0.04		0.36			25.7		10.6	0.3		0.01	1.8	2.0	11.3		0.38		0.61	
11/5 -93	C5	Tue.		7.2	0.03		0.36			23.8		8.7	0.2		0.01	0.9	0.7	9.0		0.22		0.34	
12/5 -93	C5	Wed.		7.4	0.07		0.41			23.4		10.2	0.2		0.01	0.8	0.3	8.8		0.21		0.14	
13/5 -93	C5	Thu.		8.7	0.05		0.46			24.3		9.6	0.2		0.01	0.9	0.1	8.5		0.28		0.03	
14/5 -93	C5	Fri.																					
15/5 -93	C5	Sat.																					
16/5 -93	C5	Sun.																					
17/5 -93	C3	Mon.		9.9	0.04		0.78			26.8		9.9	0.2		0.01	3.0	2.3	11.3		0.41		0.90	
18/5 -93	C3	Tue.		6.5	0.02		0.34			28.2		9.4	0.2		0.02	1.2	1.9	10.0				0.77	
19/5 -93	C3	Wed.																					
20/5 -93	C3	Thu.																					
21/5 -93	C3	Fri.																					
22/5 -93	C3	Sat.																					
23/5 -93	C3	Sun.		6.1	0.02		0.44			21.0		9.1	0.1		0.36	2.7	4.1	11.3		0.24		0.89	
24/5 -93	C3	Mon.		7.4	0.04		0.28			23.8		8.7	0.1		0.01	1.9	2.8	10.5		0.32		0.71	
25/5 -93	C3	Tue.		7.9	0.04		0.30			26.4		8.5	0.1		0.01	1.3	1.9	9.5		0.25		0.52	
26/5 -93	C3	Wed.			0.04		0.30			26.9		9.3	0.2		0.01	1.9	1.5	10.0		0.38		0.65	
27/5 -93	C3	Thu.		5.5	0.05		0.33			33.2		10.4	0.2		0.01	1.5	2.4	11.5		0.45		0.99	
28/5 -93	C	Fri.																					
31/5 -93	C2	Mon.		3.5	0.09		0.31			21.9		7.8	0.2		0.01	4.2	7.2	13.3		0.43		0.63	
1/6 -93	C2	Tue.		10.3	0.09		0.34			18.3		9.5	0.3		0.07	0.8	1.6	10.4		0.56		0.43	
2/6 -93	C2	Wed.		8.6	0.07		0.29			25.1		8.6	0.2		0.01	1.6	2.4	10.3		0.47		0.68	
3/6 -93	C2	Thu.		7.4	0.09		0.32			27.8		8.2	0.1		0.01	1.7	2.3	9.8	4.8	0.29		0.54	
4/6 -93	C4	Fri.																					
6/6 -93	C4	Sun.		8.1	0.10		0.38			28.1		9.9	1.1		0.01		2.8	11.0		0.56		0.60	
7/6 -93	C4	Mon.		7.0	0.09		0.35			29.0		8.9	0.4		0.08	2.3	1.9	9.5	6.4			0.52	
8/6 -93	C4	Tue.		8.8	0.10		0.35			26.9		9.3	0.1		0.01	0.8	0.6	7.9	3.3			0.14	
9/6 -93	C4	Wed.		7.8	0.07		0.32			28.5		8.8	0.2		0.01	0.4	0.4	8.6	3.4	0.23		0.09	
10/6 -93	C4	Thu.		8.3	0.06		0.30			29.2		8.5	0.1		0.01	0.8	0.5	9.0	0.0	0.21		0.09	
11/6 -93	C5	Fri.																	3.4				
13/6 -93	C5	Sun.		6.0	0.05		0.25			27.4		7.6	0.5		0.01	2.2	3.2	8.0		0.28		0.60	
14/6 -93	C5	Mon.		8.1	0.06		0.30			24.5		8.4	0.3		0.01	1.4	1.7	8.8	0.9	0.26		0.39	
15/6 -93	C5	Tue.																					
16/6 -93	C5	Wed.		8.1	0.04		0.30			21.9		6.7	0.2		0.01	0.7	0.1	7.8	0.7	0.50		0.02	
17/6 -93	C5	Thu.		7.0	0.05		0.31			27.3		9.4	0.3		0.01	0.5	0.1	7.6	0.2	0.24		0.02	
18/6 -93	C3	Fri.																	0.4				
19/6 -93	C3	Sat.																					
20/6 -93	C3	Sun.		3.6	0.03		0.20			21.0		6.3	0.3		0.02	4.3	2.8					0.34	
21/6 -93	C3	Mon.		5.6	0.05		0.30			26.6		12.0	0.8		0.02	0.0	0.7	11.2	1.3	0.70		0.21	
22/6 -93	C3	Tue.		5.9	0.02		0.29			26.4		8.6	0.3		0.01	1.0	0.8	8.4	3.1	0.29		0.29	
23/6 -93	C3	Wed.		5.9	0.06		0.27			28.5		10.7	0.4		0.02	1.0	0.3	9.6	0.2	0.49		0.11	
24/6 -93	C2	Thu.																	0.1				
25/6 -93	C2	Fri.																					
26/6 -93	C2	Sat.																					
26/6 -93	C2	Sat.																					
27/6 -93	C2	Sun.																					
28/6 -93	C2	Mon.		18.9	0.04		3.20			20.8		8.0	0.8		0.10	2.0	3.0	13.0		0.46		0.48	
29/6 -93	C2	Tue.		5.5	0.09		0.38			25.4		7.8	0.6		0.03	3.0	2.9	10.7	7.5	0.69		0.69	
30/6 -93	C2	Wed.		10.5	0.08		1.08			28.0		9.5	0.8		0.10	1.6	0.4	9.7	4.9	0.56		0.17	
1/7 -93	C2	Thu.			0.05		0.35			23.7		6.8	0.6		0.09	1.6	0.4	7.3	5.3	0.22		0.17	
2/7 -93	C4	Fri.																	3.2				
3/7 -93	C4	Sat.																					
4/7 -93	C4	Sun.		4.5	0.04		0.40			18.3	6.9	6.6	0.8		0.18	2.2	1.4	7.9		0.24	0.20	0.50	
5/7 -93	C4	Mon.		4.1	0.04		0.34			20.5		6.9	1.3	3.0	0.15	1.7	0.9	7.6	5.9	0.25		0.33	0.36
6/7 -93	C4	Tue.		9.8	0.02		0.71			23.2		7.5	1.2	3.6	0.10	0.9	0.4	8.6	4.2	0.37		0.27	
7/7 -93	C4	Wed.		11.1	0.01		0.72			23.2		9.0		4.0	0.13	0.6	0.3		3.6		0.15	0.22	
8/7 -93	C4	Thu.		6.0	0.01		0.38			23.0		8.9	1.2	5.2	0.02	1.1	0.4	7.9	5.2	0.28		0.32	0.60
9/7 -93	C4	Fri.												4.2				3.2					
10/7 -93	D4	Sat.																					
11/7 -93	D4	Sun.		4.4	0.01		0.23			16.8	5.3	5.4	1.1		0.10	3.1	2.9	8.0		0.24	0.58	1.00	
12/7 -93	D4	Mon.			0.02		1.10			4.4	4.1	3.9	0.6	6.8	1.22	0.7	0.9	5.6	0.1	0.18		0.32	0.10
13/7 -93	D4	Tue.		11.8	0.01		0.74			6.5	4.7	5.0	0.5	3.8	1.06	0.8	0.9	5.7	0.1	0.21	0.19	0.29	
14/7 -93	D4	Wed		4.2	0.01		0.21			11.6	5.4	5.2	0.7	4.2	0.46	0.4	0.7	5.9	0.1	0.18	0.10	0.22	0.10
15/7 -93	D4	Thu.		4.0	0.02		0.20			12.8	5.2	5.9	0.6	2.5	0.41	1.6	1.3	6.9	3.5	0.21		0.41	0.52
16/7 -93	D4	Fri.																					

Date	PROCESS	Day of the week	BOD: Primary settled WW	Effluent	Phosphate: Effluent	Anoxic tank	Total phosphorus: Effluent	RBC	RAS	Ammonium: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrate: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrite: RBC	Anoxic tank	Effluent	Deoxygenation tank
Unit			g O$_2$/m^3		g P/m^3		g P/m^3			g N/m^3					g N/m^3					g N/m^3			
17/7 -93	D4	Sat																					
18/7 -93	D4	Sun.		3.4	0.03		0.22			13.2	5.2	4.4	0.5		0.15	2.5	2.2	6.7		0.17	0.40	0.48	
19/7 -93	D4	Mon		3.4	0.03		0.21			16.6	5.6	5.7	0.6	2.9	0.10	1.0	1.5	7.0	4.5	0.17		0.27	0.33
20/7 -93	D4	Tue.		3.8	0.04		0.25			18.6	7.5	7.3	0.8	4.2	0.10	0.9	0.9	7.9	3.0	0.31	0.16	0.10	
21/7 -93	D4	Wed												3.9					0.1				0.10
22/7 -93	D4	Thu.		4.2	0.03		0.21			13.4	6.4	6.7	0.6	3.2	0.10	0.4	0.9	7.1	2.0	0.24		0.09	
23/7 -93	D4	Fri.																					
24/7 -93	D4	Sat																					
25/7 -93	D4	Sun.			0.04		0.24			14.3	5.7	4.3	0.5		0.22	1.9	3.3	7.8					
26/7 -93	D4	Mon.												2.6					4.9				
27/7 -93	D4	Tue.			0.05		0.24			17.4	6.4	5.5	0.5	3.5	0.04	0.9	1.6	7.3	2.9	0.23	0.14	0.12	
28/7 -93	D4	Wed.																					
29/7 -93	D4	Thu.			0.04		0.24			18.6	6.8	6.4	0.7	2.6	0.03	0.5	1.0	7.0	2.6	0.20		0.09	
30/7 -93	D4	Fri.																					
31/7 -93	D4	Sat.																					
1/8 -93	D4	Sun.			0.03		0.22			15.2	6.2	5.2	0.6		0.30	1.5	3.0	8.1		0.20	0.23	0.33	
2/8 -93	D4	Mon.												3.2					3.9				0.58
3/8 -93	D4	Tue.			0.16		0.40			14.8	8.3	8.1	0.9	3.6	0.06	0.6	0.8	8.0	3.5	0.33		0.09	0.37
4/8 -93	D4	Wed.																					
5/8 -93	D4	Thu.			0.06		0.29			16.0	6.5	6.1	0.7	2.7	0.08	0.7	0.7	6.9	1.9	0.20		0.08	0.46
6/8 -93	D4	Fri																					
7/8 -93	D4	Sat.																					
8/8 -93	D4	Sun.																					
9/8 -93	D4	Mon																					
10/8 -93	D4	Tue.																					
11/8 -93	D5	Wed.																					
12/8 -93	D5	Thu.																					
13/8 -93	D5	Fri.																					
14/8 -93	D5	Sat.																					
15/8 -93	D5	Sun.																					
16/8 -93	D5	Mon																					
17/8 -93	D5	Tue.																					
18/8 -93	D5	Wed.																					
19/8 -93	D5	Thu.			0.03		0.65			17.7	8.1	7.6	1.0		0.10	0.9	0.6	7.9		0.25		0.33	
20/8 -93	D5	Fri.																					
21/8 -93	D5	Sat																					
22/8 -93	D5	Sun.																					
23/8 -93	D5	Mon.					1.12			19.3	9.7	10.0	1.6		0.10	0.7	0.8	9.2		0.58		0.19	
24/8 -93	D5	Tue.			0.02		0.70			19.4	8.8	9.3	1.2		0.10	0.5	0.8	9.1		0.48		0.19	
25/8 -93	D5	Wed.																					
26/8 -93	D5	Thu			0.02		0.40			21.8	8.0	8.3			0.01	1.2	0.8			0.38		0.38	
27/8 -93	D5	Fri.																					
28/8 -93	D5	Sat.																					
29/8 -93	D5	Sun.			0.03		0.29			20.7	9.0	8.1	1.1		0.10	2.5	3.3			0.47		0.66	
30/8 -93	D5	Mon.																					
31/8 -93	D5	Tue			0.04		0.28			20.0	6.7	7.3	0.7		0.10	1.0	0.8	7.9		0.25		0.10	
1/9 -93	D5	Wed.																					
2/9 -93	D5	Thu.			0.04		0.31			22.2		8.2	0.9		0.09	0.6	1.0	8.4		0.10		0.33	
3/9 -93	D5	Fri.																					
4/9 -93	D5	Sat.																					
5/9 -93	D5	Sun.			0.06		0.35			17.0		4.3	0.4		0.10	0.8	2.3	7.5		0.35		0.18	
6/9 -93	D5	Mon.																					
7/9 -93	D5	Tue.			0.02		0.33			33.6		15.5	1.0		0.10	1.2	1.2	13.7		0.21		0.21	
8/9 -93	D5	Wed.																					
9/9 -93	D5	Thu.			0.03		0.50			25.4		14.4	3.6		0.05	1.0	1.2	9.4		0.24		0.42	
10/9 -93	D5	Fri.																					
11/9 -93	D5	Sat																					
12/9 -93	D5	Sun			0.05		0.31			27.6		6.6	0.7		0.37	2.8	2.9	9.6		0.50		0.35	
13/9 -93	D5	Mon.																					
14/9 -93	D5	Tue.			0.05		0.37			27.6		9.6	1.5		0.10	1.0	1.5	11.0		0.27		0.50	
15/9 -93	D5	Wed.																					
16/9 -93	D5	Thu.			0.02		1.36			26.0		8.8	1.4		0.05	0.8	0.7	9.3		0.17		0.44	
17/9 -93	D5	Fri.																					
18/9 -93	D5	Sat.																					
19/9 -93	D5	Sun.			0.03		1.62			29.2		7.3	2.7		0.20	6.8	2.4	7.8		0.21		0.49	
20/9 -93	D5	Mon.																					
21/9 -93	D5	Tue.			0.01		0.80			23.7		10.6	2.1		0.09	1.3	1.1			0.24		0.66	
22/9 -93	D4	Wed.																					
23/9 -93	D4	Thu.			0.02		0.31			23.2		8.0	0.9		0.10	1.3	0.1			0.05		0.51	
24/9 -93	D4	Fri.																					
25/9 -93	D4	Sat.																					
26/9 -93	D4	Sun			0.04		0.41			17.0		17.0	0.6		0.09	2.1	0.4			0.04		0.41	

Date	PROCESS	Day of the week	BOD: Primary settled WW	Effluent	Phosphate: Effluent	Anoxic tank	Total phosphorus: Effluent	RBC	RAS	Ammonium: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrate: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrite: RBC	Anoxic tank	Effluent	Deoxygenation tank
Unit			g O_2/m^3		g P/m^3		g P/m^3			g N/m^3					g N/m^3					g N/m^3			
27/9 -93	D4	Mon.																	3.7				
28/9 -93	D4	Tue.			0.03		0.34			29.3		9.6	0.8		0.10	1.0	0.1	9.8	4.1	0.03		0.39	
29/9 -93	D4	Wed.																					
30/9 -93	D4	Thu.			0.02		0.40			28.4		10.5	1.1		0.28	1.3	0.4	10.8	4.7	0.06		0.47	
1/10 -93	D4	Fri.																					
2/10 -93	D4	Sat.																					
3/10 -93	D4	Sun.			0.05		0.46			27.4		8.9	1.1		0.10	3.1	1.9	11.5		0.31		0.45	
4/10 -93	D4	Mon.																	1.0				
5/10 -93	D4	Tue.			0.03		0.44			18.4		9.6	0.5		0.03	0.7	0.3	8.5	0.6	0.06		0.31	
6/10 -93	D4	Wed.																					
7/10 -93	D4	Thu.																					
8/10 -93	D4	Fri.																					
9/10 -93	D4	Sat.																					
10/10 -93	D4	Sun.																					
11/10 -93	D4	Mon.																					
12/10 -93	D4	Tue.			0.02		0.30			12.2		6.9	0.7		0.17	0.4	1.4	8.5		0.24		0.35	
13/10 -93	D4	Wed.																					
14/10 -93	D4	Thu.					0.29			13.9		8.3	0.8		0.12	1.0	1.2	8.2		0.13		0.34	
15/10 -93	D4	Fri.																					
16/10 -93	D4	Sat.																					
17/10 -93	D4	Sun.			0.01		0.29			13.9		5.6	0.5		0.15	1.0	2.5	7.9		0.35		0.24	
18/10 -93	D4	Mon.																					
19/10 -93	D4	Tue.					0.34			18.6		6.8	0.6		0.05	0.7	1.2	7.8		0.12		0.27	
20/10 -93	D4	Wed.																					
21/10 -93	D4	Thu.			0.01		0.36			22.5		8.2	0.7		0.10	0.6	1.3	9.0		0.12		0.35	
22/10 -93	D4	Fri.																					
23/10 -93	D4	Sat.																					
24/10 -93	D4	Sun.			0.04		0.37			26.6		7.6	1.2		0.05	1.7	4.2	11.7		0.57		0.53	
25/10 -93	D4	Mon.																	1.1				
26/10 -93	D4	Tue.			0.02					23.4		9.8	1.5		0.19	0.9	1.4	10.2	1.2	0.11		0.55	
27/10 -93	D4	Wed.																					
28/10 -93	D4	Thu.			0.06		0.46			27.0		9.5	1.1		0.16	0.7	1.5	10.5	4.5	0.18		0.56	
29/10 -93	D4	Fri.																					
30/10 -93	D4	Sat.																					
31/10 -93	D4	Sun.			0.02		0.30			24.6		6.6	0.5		0.09	1.5	3.4	10.3		0.41		0.33	
1/11 -93	D4	Mon.																	3.8				
2/11 -93	D4	Tue.			0.02		0.33			24.0		9.0	0.5		0.32	0.9	0.4	9.7	0.7	0.10		0.34	
3/11 -93	D4	Wed.																					
4/11 -93	D4	Thu.			0.01		0.28			29.6	10.7	11.2	0.8		0.10	0.7	0.3	10.0	4.9	0.08		0.47	
5/11 -93	D4	Fri.																					
6/11 -93	D4	Sat.																					
7/11 -93	D4	Sun.			0.01		0.31			29.2	9.0	10.2	0.7		0.72	3.6	2.5	11.7					
8/11 -93	D4	Mon.																	3.1				
9/11 -93	D4	Tue.			0.01		0.32			22.7	6.7	9.6	0.5		0.10	2.3	0.3	9.0	5.1	0.67		0.32	
10/11 -93	D4	Wed.			0.01		0.32					9.4	0.7				0.3	9.0	0.7	0.04		0.27	
11/11 -93	D4	Thu.																					
12/11 -93	D4	Fri.																					
13/11 -93	D4	Sat.																					
14/11 -93	D4	Sun.					0.36				8.0	7.9				1.2	1.2			0.35		0.25	
15/11 -93	D4	Mon.																					
16/11 -93	D4	Tue.																					
17/11 -93	D4	Wed.											0.7						3.0			0.25	
18/11 -93	D4	Thu.			0.01		0.39			19.7	7.5	7.8			0.10	0.4	0.3	8.7	1.7	0.05		0.21	
19/11 -93	D4	Fri.																					
20/11 -93	D4	Sat.																					
21/11 -93	D4	Sun.			0.02		0.39			22.4	9.1	8.9	1.0		0.10	2.5	2.9	11.8		0.29		0.41	
22/11 -93	D4	Mon.			0.01		0.34				7.6					4.4			7.8	0.22		0.59	
23/11 -93	D4	Tue.			0.01		0.38			18.9	7.0	6.9	0.6		0.10	2.5	2.3	9.3		0.18		0.34	
24/11 -93	D4	Wed.			0.00		0.41			23.2	9.4	8.7	1.2		0.11	0.4	1.3	9.1	3.2	0.24		0.15	
25/11 -93	D4	Thu.			0.01		0.03			26.0	10.5	10.2	1.4		0.10	0.7	1.4	10.5	4.9	0.30		0.22	
26/11 -93	D4	Fri.																					
27/11 -93	D4	Sat.																					
28/11 -93	D4	Sun.			0.03		0.03			22.8	8.7	8.7	0.1		0.10	0.7	1.4	9.3		0.29		0.33	
29/11 -93	D4	Mon.																	5.0				
30/11 -93	D4	Tue.			0.02		0.42			23.4	7.5	7.3	0.1		0.10	0.3	0.4	7.5	3.6	0.18		0.06	
1/12 -93	D4	Wed.																					
2/12 -93	D4	Thu.																					
3/12 -93	D4	Fri.																					
4/12 -93	D4	Sat.																					
5/12 -93	D4	Sun.			0.01		0.62			6.9		5.5	1.0		1.68	0.3	0.9	5.9		0.16		0.17	
6/12 -93	D4	Mon.																	0.1				
7/12 -93	D4	Tue.			0.01		0.49			11.2	8.8	8.7	1.9		0.77	0.2	0.3	8.3	0.1	0.45		0.04	

Date	PROCESS	Day of the week	BOD: Primary settled WW	Effluent	Phosphate: Effluent	Anoxic tank	Total phosphorus: Effluent	RBC	RAS	Ammonium: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrate: Primary settled WW	Anoxic tank	Effluent	RBC	Deoxygenation tank	Nitrite: RBC	Anoxic tank	Effluent	Deoxygenation tank
Unit			$g\ O_2/m^3$		$g\ P/m^3$		$g\ P/m^3$			$g\ N/m^3$					$g\ N/m^3$					$g\ N/m^3$			
8/12 -93	D4	Wed.																					
9/12 -93	D4	Thu.			0.01		0.42			14.6	9.5	9.2	2.6		0.55	0.1	0.2	7.7	0.1	0.40		0.03	
10/12 -93	D4	Fri.																					
11/12 -93	D4	Sat.																					
12/12 -93	D4	Sun.			0.01		0.36			11.4	6.4	6.4	1.0		0.59	0.1	0.9	6.8		0.24		0.18	
13/12 -93	D4	Mon.																	4.1				
14/12 -93	D4	Tue.			0.01		0.42			13.2	7.7	7.6	1.1		0.10	0.1	0.4	7.5	2.2	0.28		0.03	
15/12 -93	D4	Wed.																					
16/12 -93	D4	Thu.																					
17/12 -93	D4	Fri.																					
18/12 -93	D4	Sat.																					
19/12 -93	D4	Sun.																					
20/12 -93	D4	Mon.																					
21/12 -93	D4	Tue.			0.00		0.71			8.8	7.2	8.2	2.2		0.75	0.1	0.2	7.3	0.1	0.23		0.03	
22/12 -93	D4	Wed.																					
23/12 -93	D4	Thu.																					
24/12 -93	D4	Fri.																					
25/12 -93	D4	Sat.																					
26/12 -93	D4	Sun.																					
27/12 -93	D4	Mon.																					
28/12 -93	D4	Tue.																					
29/12 -93	D4	Wed.																					
30/12 -93	D4	Thu.																					
31/12 -93	D4	Fri.																					
1/1 -94	D4	Sat.																					
2/1 -94	D4	Sun.																					
3/1 -94	D4	Mon.			0.01		0.82			17.4	12.4	12.2	4.2		0.34	0.2	0.5			0.49		0.10	
4/1 -94	D4	Tue.																					
5/1 -94	D5	Wed.																					
6/1 -94	D5	Thu.																					
7/1 -94	D5	Fri.																					
8/1 -94	D5	Sat.																					
9/1 -94	D5	Sun.																					
10/1 -94	D5	Mon.			0.01		0.57			10.7	6.1	6.6	1.0		0.64	0.3	1.0	8.3		0.23		0.28	
11/1 -94	D5	Tue.																					
12/1 -94	D5	Wed.			0.00		0.54			10.2	5.7	5.5	1.2		0.58	0.2	0.4	7.3		0.20		0.14	
13/1 -94	D5	Thu.																					
14/1 -94	D5	Fri.																					
15/1 -94	D5	Sat.																					
16/1 -94	D5	Sun.			0.16		2.88			11.8	8.1	7.9	1.4		1.01	3.9	4.1	10.9		0.33		0.32	
17/1 -94	D5	Mon.																					
18/1 -94	D5	Tue.																					
19/1 -94	D4	Wed.								9.3	5.3		0.8		0.63	0.5		6.1		0.18			
20/1 -94	D4	Thu.																					
21/1 -94	D4	Fri.																					
22/1 -94	D4	Sat.																					
23/1 -94	D4	Sun.																					
24/1 -94	D4	Mon.																					
25/1 -94	D4	Tue.			0.01		2.25			13.8	8.2	8.5	2.0		0.10	0.4	0.1	7.2	3.4	0.37		0.24	
26/1 -94	D4	Wed.																					
27/1 -94	D4	Thu.			0.01		0.72			9.0	6.4	5.7	0.7		0.50	0.2	0.2	5.5	0.1	0.18		0.07	
28/1 -94	D4	Fri.																					
29/1 -94	D4	Sat.																					
30/1 -94	D4	Sun.			0.01		0.78			11.8		6.2	0.7		0.47		2.1	8.0		0.21		0.40	
31/1 -94	D4	Mon.																	2.7				
1/2 -94	D4	Tue.			0.01		0.74			15.3	8.7	8.9	2.2		0.04	0.1	0.2	7.4	0.6	0.37		0.09	
2/2 -94	D5	Wed.																					
3/2 -94	D5	Thu.																					
4/2 -94	D5	Fri.																					
5/2 -94	D5	Sat.																					
6/2 -94	D5	Sun.																					
7/2 -94	D5	Mon.			0.02		0.67			19.8	10.3	10.0	2.9		0.10	0.5	1.7	4.7		0.46		0.35	
8/2 -94	D5	Tue.			0.02		0.67			20.3	9.7	9.3	1.7		0.25	0.1	0.4	8.6		0.38		0.13	
9/2 -94	D4	Wed.																					
10/2 -94	D4	Thu.			0.01		0.42			16.0	7.2	8.2	1.3		0.10	0.2	0.5	7.8	0.1	0.32		0.13	

Date	PROCESS	Day of the week	Total nitrogen: Primary settled WW	Effluent	RBC	Return activated sludge	Alkalinity: Primary settled WW	Effluent	RBC	Oxygen concentration: Aeration tank	Effluent from RBC(estimated)	Effluent pH	Ox. red. potential: Anoxic tank	Aerated tank	Sett. of mixed liquor: Sludge volume (30 min)	Stirred sl. vol(30min)	Initial settling velocity	Turbidity	Flow to main WWTP
Unit			g N/m^3				ekv HCO$_3$/m^3			g O$_2$/m^3			mV		ml/l	ml/l	m/h	NTU	m^3/s
14/6 -90	A	Thu.	27.0	24.4		272									180	240	2.6	2.4	3.7
15/6 -90	A	Fri.																	
18/6 -90	A	Mon.	25.8	23.2		256									310	230	3.8	5.0	2.7
19/6 -90	A	Tue.																	
20/6 -90	A	Wed.	12.9	14.2		184									290	200	4.1	2.1	5.1
4/7 -90	A	Wed.	16.9	14.3		123									270	220	2.6	2.7	3.6
5/7 -90	A	Thu.																	
6/7 -90	A	Fri.																	
9/7 -90	A	Mon.																	
10/7 -90	A	Tue.	14.8	12.8		181									260	200	2.3		4.0
11/7 -90	A	Wed.																	3.2
12/7 -90	A	Thu.	17.4	15.5		188									250	200	2.6	3.4	3.3
13/7 -90	A	Fri.																	
16/7 -90	A	Mon.	20.2	18.5		169									250	220	2.5	2.0	2.6
17/7 -90	A	Tue.																	
18/7 -90	A	Wed.																	
19/7 -90	A	Thu.	23.2	20.4		213									255	230	2.5	2.1	2.4
20/7 -90	A	Fri.																	
23/7 -90	A	Mon.																	
24/7 -90	A	Tue.	26.6	23.2		256									280	240	2.2	5.0	2.4
25/7 -90	A	Wed.																	
26/7 -90	A	Thu.	25.3	22.7		231									290	240	2.8	2.9	2.3
27/7 -90	A	Fri.																	
30/7 -90	A	Mon.	22.6	23.1		212									285	240	2.5	3.0	2.5
31/7 -90	A	Tue.																	2.5
1/8 -90	A	Wed.	24.0	22.4		270									280	220	2.8	2.9	2.4
2/8 -90	A	Thu.																	2.4
3/8 -90	A	Fri.																	
6/8 -90	A	Mon.	26.4	25.4		274									310	200	2.7	4.1	2.5
7/8 -90	A	Tue.																	2.7
8/8 -90	A	Wed.																	3.4
9/8 -90	A	Thu.	24.2	26.2		259									410	220	2.8	4.0	2.9
10/8 -90	A	Fri.																	
13/8 -90	A	Mon.	25.2	30.6		288									310	220	2.7	4.0	2.7
14/8 -90	A	Tue.																	2.7
15/8 -90	A	Wed.																	2.5
16/8 -90	A	Thu.	22.8	28.4		264									310	220	2.9	3.7	2.6
17/8 -90	A	Fri.																	
20/8 -90	A	Mon.																	3.5
21/8 -90	A	Tue.	24.8	27.0		232									290	200	3.2	3.7	3.2
22/8 -90	A	Wed.																	3.0
23/8 -90	A	Thu	25.2	25.4		284									325	230	2.7	4.4	2.9
24/8 -90	A	Fri.																	
27/8 -90	A	Mon.	27.8	29.8		280									290	210	3.0	3.6	2.8
28/8 -90	A	Tue.																	2.8
29/8 -90	A	Wed.	29.8	28.4		294									310	220	3.1	3.5	2.8
30/8 -90	A	Thu.																	2.6
31/8 -90	A	Fri.																	
6/9 -90	A	Thu.	28.8	24.1		255									350	210	2.8	3.5	2.7
7/9 -90	A	Fri.																	
10/9 -90	A	Mon.	28.8	25.2		275									265	204	2.7	6.3	2.9
11/9 -90	A	Tue.																	
12/9 -90	A	Wed.																	
13/9 -90	A	Thu	29.0	27.2		268									270	220	2.4	8.3	2.5
18/9 -90	A	Tue	20.4	17.8		297									300	194	2.6		4.1
20/11 -90	A	Tue																	5.0
21/11 -90	A	Wed.																	4.1
22/11 -90	A	Thu	27.2	28.0		344									280	166	3.4	7.1	3.7
23/11 -90	A	Fri.																	
25/11 -90	A	Sun.																	3.5
26/11 -90	A	Mon.																	3.4
27/11 -90	A	Tue	31.2	25.8		342									220	130	3.7	7.3	3.3
28/11 -90	A	Wed.																	
29/11 -90	A	Thu.													230	120	4.2	8.4	3.4
30/11 -90	A	Fri.	27.8	23.6		370													
3/12 -90	A	Mon.	28.4	23.8		416									200	114	4.2	8.4	3.2
4/12 -90	A	Tue																	3.2
5/12 -90	A	Wed.	31.0	27.8		478									250	130	3.9	6.3	2.7
6/12 -90	A	Thu.																	3.3
7/12 -90	A	Fri.																	
9/12 -90	A	Sun.																	3.7
10/12 -90	A	Mon.	26.8	39.8		364									160	100	4.5	7.2	3.5

Date	PROCESS	Day of the week	Total nitrogen: Primary settled WW	Effluent	RBC	Return activated sludge	Alkalinity: Primary settled WW	Effluent	RBC	Oxygen concentration: Aeration tank	Effluent from RBC(estimated)	Effluent pH	Ox. red. potential: Anoxic tank	Aerated tank	Sett. of mixed liquor: Sludge volume (30 min)	Stirred sl. vol(30min)	Initial settling velocity	Turbidity	Flow to main WWTP
Unit			g N/m^3				ekv HCO_3/m^3			g O_2/m^3			mV		ml/l	ml/l	m/h	NTU	m^3/s
11/12 -90	A	Tue.																	3.4
12/12 -90	A	Wed.																	4.6
13/12 -90	A	Thu.	29.0	29.9		444									190	110	4.2	8.5	3.7
14/12 -90	A	Fri.																	
17/12 -90	A	Mon.	28.2	27.8		376									200	110	4.4	7.4	3.2
18/12 -90	A	Tue.																	3.3
19/12 -90	A	Wed.	26.4	23.7		352									205	134	3.8	9.8	3.2
20/12 -90	A	Thu																	3.4
7/1 -91	A	Mon.	12.9	14.6		219									125	90	5.2	6.3	6.8
8/1 -91	A	Tue.																	6.5
9/1 -91	A	Wed	11.5	20.4		262									115	86	5.8	4.2	8.9
10/1 -91	A	Thu																	8.5
11/1 -91	A	Fri.																	
12/1 -91	A	Sat.																	
13/1 -91	A	Sun.																	
14/1 -91	A	Mon.																	4.4
15/1 -91	A	Tue.	18.7	19.1		292									120	90	5.5	6.2	4.0
16/1 -91	A	Wed.																	3.7
17/1 -91	A	Thu.	19.4	17.8		252									160	110	4.1	3.1	3.5
18/1 -91	A	Fri.																	
19/1 -91	A	Sat																	
20/1 -91	A	Sun.																	
21/1 -91	A	Mon.				246									150	110	4.7	5.5	
22/1 -91	A	Tue.	24.0	20.3															4.5
23/1 -91	A	Wed.																	4.1
24/1 -91	A	Thu.	23.9	20.8		292									160	126	3.9	6.2	3.8
25/1 -91	A	Fri.																	
26/1 -91	A	Sat.																	
27/1 -91	A	Sun.																	
28/1 -91	A	Mon.	25.9	24.1		422									170	120	4.5	6.1	3.3
29/1 -91	A	Tue.																	3.3
30/1 -91	A	Wed.	32.0	24.4		408									180	140	4.2	8.8	3.3
31/1 -91	A	Thu.																	3.2
1/2 -91	A	Fri.																	
2/2 -91	A	Sat.																	
3/2 -91	A	Sun.																	
4/2 -91	A	Mon.																	
5/2 -91	A	Tue.	31.4	24.2		438									225	150	4.1	7.6	2.8
6/2 -91	A	Wed.																	2.0
4/2 -92	B	Mon.	26.7				4.12	4.12		3.6		7.1	-207	2					4.0
6/2 -92	B	Wed.	28.1	37.2		224	3.58	3.76		3.4		7.2	-173	47					3.7
10/2 -92	B	Mon.	25.3	27.1		136				3.2		7.0	-137	66					3.6
13/2 -92	B	Thu.	16.3	34.1		210	2.10	1.70		5.6		7.0	-47	22					4.1
17/2 -92	B	Mon.		24.0						1.8		7.2	-173	-28					3.8
18/2 -92	B	Tue.	29.0	29.6		130	3.84	3.72		1.9		7.2	-155	-12					3.5
24/2 -92	B	Mon.	17.9	17.1		171	2.94	2.82		2.2		6.9	-96	-3					6.2
27/2 -92	B	Thu.	25.2	20.8		123	3.54	3.44		2.0		6.8	-137	9					4.2
2/3 -92	B	Mon.		20.3		108	2.72	3.14		2.0		6.8	-106	18					4.8
9/3 -92	B	Mon.		28.5		215	4.30	4.14		1.9		7.0	-216	-19					3.4
12/3 -92	B	Thu.	13.7	17.6		233	2.38	2.26		2.3		6.6	-93	-19					6.0
16/3 -92	B	Mon.		19.5		146	3.58	3.38		1.9		6.8	-131	-79					4.0
19/3 -92	B	Thu.	19.8	23.4		301	2.98	2.80		2.1		7.0	-95	10					4.6
25/3 -92	B	Wed.	21.2	19.1		108	3.42	3.38		1.8		7.2	-146	-21					4.3
26/3 -92	B	Thu	22.4	19.3		109	3.60	3.92		1.8		7.2	-146	-23					4.2
31/3 -92	B	Tue.	35.1	31.2		210	4.46	5.28		1.9		7.3	-202	4					3.5
1/4 -92	B	Wed.	36.4	33.9		160	4.52	5.32		1.9		7.3	-164	-16					3.4
2/4 -92	B	Thu.	31.1	28.8		159	4.38	5.08		1.8		7.3	-160	-13					3.5
6/4 -92	B	Mon.	21.9	28.7		215	3.58	3.82		1.7		7.2	-212	-3					4.4
7/4 -92	B	Tue.	27.1	19.4		215	3.90	3.84		1.7		7.1	-204	-9					3.9
9/4 -92	B	Thu.	29.5	25.4		180	3.90	4.24		1.9		7.2	-204	6					3.4
13/4 -92	B	Mon.	17.4	26.1		113	2.82	3.66		3.3		7.2	-56	-11					5.1
14/4 -92	B	Tue.	20.4	27.0		96	3.58	3.60		2.5		7.2	-165	-93					4.8
15/4 -92	B	Wed.	21.0	16.2		77	3.46	3.36		1.9		7.1	-173	-42					4.4
5/5 -92	B	Tue.	23.8	17.6		88	3.58	3.58		2.9		7.1	-171	-65					4.4
10/5 -92	B	Sun.	25.1	22.7		107	3.88	3.42		0.4		7.1	-292	-34					3.2
11/2 -93												7.0	-196	47					3.3
14/2 -93	C5	Sun.	28.3	14.4			3.92	2.30				7.0	-228	90					3.1
15/2 -93	C5	Mon.			15.1	308			0.46			7.0	-224	75	280	160	4.5		3.3
16/2 -93	C5	Tue.										7.0	-178	62					3.7
17/2 -93	C5	Wed.	17.4	9.3	9.5	427	2.96	1.14	1.06		9.4	6.9	-129	98					5.7
18/2 -93	C3	Thu.										6.9	-139	79					4.3
19/2 -93	C3	Fri.										6.9	-125	87					5.8

Date	PROCESS	Day of the week	Total nitrogen: Primary settled WW	Effluent	RBC	Return activated sludge	Alkalinity: Primary settled WW	Effluent	RBC	Oxygen concentration: Aeration tank	Effluent from RBC(estimated)	Effluent pH	Ox. red. potential: Anoxic tank	Aerated tank	Sett. of mixed liquor: Sludge volume (30 min)	Stirred sl. vol(30min)	Initial settling velocity	Turbidity	Flow to main WWTP
Unit			g N/m^3				ekv HCO$_3$/m^3			g O$_2$/m^3			mV		ml/l	ml/l	m/h	NTU	m^3/s
20/2 -93	C3	Sat.										7.0	-115	115					4.5
21/2 -93	C3	Sun.										7.0	-99	114					4.5
22/2 -93	C3	Mon.										6.9	-92	112					3.9
23/2 -93	C3	Tue.										6.9	-113	24					3.6
24/2 -93	C3	Wed.										7.0	-127	3					3.6
25/2 -93	C3	Thu.	29.6	15.9	16.3	570	3.98	2.22	1.12		9.2	7.0	-145	-206	400	150	3.5		3.4
26/2 -93	C3	Fri.										7.0	-181	-35					3.5
27/2 -93	C3	Sat.										6.9	-181	-2					3.7
28/2 -93	C3	Sun.	41.0	16.2	17.3	410	4.36	2.08	0.98		9.7	6.9	-179	7					3.3
1/3 -93	C1	Mon.				410									320	124	4.1		3.1
2/3 -93	C1	Tue.										6.9	-251	-38					2.9
3/3 -93	C1	Wed.				484						7.0	-238	-51					3.3
4/3 -93	C1	Thu.	33.2	15.8	15.1	372	4.12	2.10	0.82		9.2	6.9	-238	-37					3.1
5/3 -93	C1	Fri.										6.9	-364	-22					3.5
6/3 -93	C1	Sat.										6.8	-458	-11					3.2
7/3 -93	C1	Sun.	33.6	17.5	17.9		4.14	1.98	0.70		9.2	6.8	-416	-10					3.0
8/3 -93	C2	Mon.				613						6.9	-348	-17					3.0
9/3 -93	C2	Tue.										6.9	-317	-23					3.0
10/3 -93	C2	Wed.										7.0	-342	-23					3.3
11/3 -93	C2	Thu.										6.9	-446	-13					3.2
12/3 -93	C2	Fri.										7.0	-476	5					3.0
13/3 -93	C2	Sat.										6.9	-475	-11					3.1
14/3 -93	C	Sun.	34.7	18.4	19.3		4.16	1.82	0.66		9.2	6.9	-475	22	220	150	2.8		3.2
15/3 -93	C	Mon.				361						6.9	-493	15					3.4
16/3 -93	C	Tue.	32.2	18.6	17.7	393	4.06	2.08	0.82		9.2	7.0	-496	3					3.2
17/3 -93	C4	Wed.										7.0	-497	-17					3.9
18/3 -93	C4	Thu.										7.1	-405	-31					3.6
19/3 -93	C4	Fri.										7.1	-232	-31					4.0
20/3 -93	C4	Sat.										7.0	-135	-26					3.5
21/3 -93	C4	Sun.	29.3	13.4	12.5		3.68	2.18	0.90		9.2	7.1	-125	-29					3.2
22/3 -93	C4	Mon.				355						6.9	-183	-50	460	170	3.1		4.2
23/3 -93	C4	Tue.										6.9	-168	-40					4.7
24/3 -93	C4	Wed.	27.7	12.9	12.4		3.46	2.26	1.04		9.2	7.0	-100	-1					4.1
25/3 -93	C4	Thu.	29.5	11.7	11.9	418	3.54	2.10	0.90		9.2	7.0	-105	21	275	160	2.8		3.3
26/3 -93	C4	Fri.										7.0	-108	36					3.2
27/3 -93	C4	Sat.										7.0	-95	63					3.2
28/3 -93	C4	Sun.	31.3	12.6	12.9		3.74	2.16			9.0	7.0	-87	89					2.9
29/3 -93	C2	Mon.	33.2	14.4	13.1		4.06	2.26			9.0	7.0	-98	80	420	190	3.4		3.1
30/3 -93	C2	Tue.										7.0	-114	70					2.9
31/3 -93	C2	Wed.	32.7	12.8	11.9	485	3.98	2.30	1.02		9.0	7.1	-114	69	820	190	3.0		3.0
1/4 -93	C2	Thu.	34.2	14.3	13.5	414					8.0	7.1	-115	61					3.3
2/4 -93	C2	Fri.				530						7.0	-91	63					3.0
3/4 -93	C2	Sat.										6.9	16	49					2.8
4/4 -93	C2	Sun.	34.2	18.4	20.2						8.8	6.9	66	10					2.9
5/4 -93	C3	Mon.				316						6.9	-125	-19					3.0
6/4 -93	C3	Tue.										7.0	-170	-31					3.1
7/4 -93	C3	Wed.				353						6.9	-179	-26	180	100	6.2		3.1
8/4 -93	C3	Thu.										6.9	-251	-30					3.1
9/4 -93	C3	Fri.										6.8	-192	-20					2.8
10/4 -93	C3	Sat.										6.7	-114	4					2.7
11/4 -93	C3	Sun.										6.6	-63	17					2.5
12/4 -93	C3	Mon.	34.6	17.6	24.1		3.68	1.82			9.0	6.6	-18	17					2.5
13/4 -93	C3	Tue.	32.9	21.5	19.6	484	3.98	2.26	0.74		8.0	6.8	-112	-28	160	100	6.3		3.1
14/4 -93	C5	Wed.										6.8	-262	-12					2.8
15/4 -93	C5	Thu.	32.2	11.8	12.8	388	3.84	2.26	0.94		8.8	6.9	-394	-36					3.0
16/4 -93	C5	Fri.										6.9	-190	-10					2.9
17/4 -93	C5	Sat.										6.9	-250	-16					4.3
18/4 -93	C5	Sun.	27.2	14.5	14.5		3.36	2.22	0.50		9.0	6.8	-263	23					3.0
19/4 -93	C5	Mon.	32.1	14.6	15.9	396	4.00	2.00	0.38		9.0	6.9	-362	11	220	120	5.1		3.2
20/4 -93	C5	Tue.										6.9	-137	40					3.0
21/4 -93	C5	Wed.										7.0	-84	76					3.4
22/4 -93	C5	Thu.	30.6	13.6	14.2	374	3.50	1.76			8.8	6.9	-95	92					3.6
23/4 -93	C2	Fri.										7.0	-83	46					3.6
24/4 -93	C2	Sat.										6.9	-44	51					3.4
25/4 -93	C2	Sun.	30.8	14.5	15.6		3.80	1.68			8.7	6.8	-5	64					2.3
26/4 -93	C2	Mon.				630						6.9	-21	51	520	150	3.7		3.0
27/4 -93	C2	Tue.	32.8	17.0	16.7	398	3.72	2.54	0.74		7.7	6.9	-105	36					4.0
28/4 -93	C2	Wed.	30.1	11.5	11.8	428	3.60	2.16	0.90		8.7	6.8	-72	101					2.9
29/4 -93	C2	Thu	33.4	16.2	12.8	411	4.06	2.16	0.80		8.4	6.9	-56	49					2.9
30/4 -93	C4	Fri.										7.0	-91	20					3.0
1/5 -93	C4	Sat.										7.1	-78	32					2.6
2/5 -93	C4	Sun.										7.2	-105	57					2.5

Date	PROCESS	Day of the week	Total nitrogen: Primary settled WW	Effluent	RBC	Return activated sludge	Alkalinity: Primary settled WW	Effluent	RBC	Oxygen concentration Aeration tank	Effluent from RBC(estimated)	Effluent pH	Ox. red. potential: Anoxic tank	Aerated tank	Sett. of mixed liquor Sludge volume (30 min)	Stirred sl. vol(30min)	Initial settling velocity	Turbidity	Flow to main WWTP
Unit			g N/m^3				ekv HCO$_3$/m^3			g O$_2$/m^3			mV		ml/l	ml/l	m/h	NTU	m^3/s
3/5 -93	C4	Mon				321						7.2	7	49					3.4
4/5 -93	C4	Tue.	36.7	13.6	16.8		4.04	2.40	0.86		8.7	7.1	21	49					3.1
5/5 -93	C4	Wed.	32.4	13.8	12.5	367	3.62	2.30	0.94		8.7	7.0	21	-14					3.4
6/5 -93	C4	Thu.	46.5	14.9	13.3	348	4.20	2.48	1.04		7.7	7.2	-117	15	320	200	2.5		3.3
7/5 -93	C5	Fri.										7.3	-146	76					3.1
8/5 -93	C5	Sat.										7.2	-116	127					2.6
9/5 -93	C5	Sun.	35.6	15.4	14.6		4.08	1.90	0.50		8.4	7.2	-83	175					2.6
10/5 -93	C5	Mon.	33.6	13.4	12.9	436	4.06	2.00	0.62		8.4	7.1	-94	133					2.8
11/5 -93	C5	Tue.	33.0	10.6	10.6	404	3.82	2.16	0.98		8.4	7.1	-90	153	280	190	3.2		2.6
12/5 -93	C5	Wed.	32.0	11.7	10.4	324	3.80	2.38			8.4	7.1	-90	153					3.1
13/5 -93	C5	Thu.	32.1	11.1	10.4	484	3.82	2.15	1.30		8.4	7.2	-130	94					3.2
14/5 -93	C5	Fri.										7.2	-152	94					2.9
15/5 -93	C5	Sat.										7.2	-90	101					2.8
16/5 -93	C5	Sun.										7.1	-53	116					2.7
17/5 -93	C3	Mon.	36.0	14.7	13.4	370	3.96	2.00	0.72		8.4	7.1	-66						2.8
18/5 -93	C3	Tue.	32.9	12.7	12.6	514	4.08	2.14			8.4	7.1	-97	-78					2.7
19/5 -93	C3	Wed.										7.1	-102	-120	280	220	2.6		2.8
20/5 -93	C3	Thu.										7.0	-99	-119					2.8
21/5 -93	C3	Fri.										7.0	-90	-136					3.2
22/5 -93	C3	Sat.										7.0	-79	-170					3.9
23/5 -93	C3	Sun.	27.3	15.2	13.3		3.36	1.74	0.46		8.2	6.8	-47	-280					3.4
24/5 -93	C3	Mon.	29.1	12.4	12.0		3.90	1.98	0.86		8.2	7.0	-70	-283					3.1
25/5 -93	C3	Tue.	31.8	11.1	10.3	335	4.20	2.38	1.34		8.2	7.1	-88	-115	315	200	2.9		2.8
26/5 -93	C3	Wed.	33.7	11.8	11.2	320	4.06	2.48	1.30		8.2	7.1	-93	-74					3.0
27/5 -93	C3	Thu.	39.6	14.1	13.3	451	4.46	2.48	1.12		8.2	7.1	43	-3					2.9
28/5 -93	C	Fri.										7.1	61	-16					2.6
31/5 -93	C2	Mon.	28.1	15.6	14.3		3.42	1.36	0.32		8.4	6.8	59	12					2.6
1/6 -93	C2	Tue.	24.7	12.2	11.7	301					8.4	6.9	18	5	395	210	3.1		4.8
2/6 -93	C2	Wed.	32.0	11.7	11.5	328	3.98	2.06	0.90		8.2	6.9	32	-46					3.1
3/6 -93	C2	Thu.	34.0	11.3	11.3	262	4.14	2.14	1.02		8.2	6.9	-15	2					2.6
4/6 -93	C4	Fri.																	2.7
6/6 -93	C4	Sun.	37.1	14.4	14.1		4.06	1.92	0.64		8.2	6.9	7	18					2.8
7/6 -93	C4	Mon.	35.2	12.4	11.7	280	4.08	1.90			8.2	6.9	20	21					2.7
8/6 -93	C4	Tue.	32.6	11.6	9.5	318	4.00	2.22			8.1	7.0	17	1	480	200	3.9		2.7
9/6 -93	C4	Wed.	33.6	10.3	8.8	235	4.04	2.24	1.18		8.1	7.0	19	-18					2.6
10/6 -93	C4	Thu.	30.1	8.9	8.9	224	4.04	2.06	0.94		8.1	7.7	-19	30					2.6
11/6 -93	C5	Fri.																	2.5
13/6 -93	C5	Sun.	29.1	10.8	9.9		4.12	1.58	0.64		8.1	7.0	-47	45					2.1
14/6 -93	C5	Mon.	30.8	10.8	10.5	261	3.88	1.78	0.74		8.1	7.0	-72	33					2.4
15/6 -93	C5	Tue.										7.0	-91	19					2.6
16/6 -93	C5	Wed.	29.9	16.8	9.4	271	4.04	2.14	1.22		8.1	7.3	-76	25	510	260	2.1		2.5
17/6 -93	C5	Thu.	36.7	10.9	18.4	258	4.08	2.22	0.96		8.1	7.2	-85	-30					2.7
18/6 -93	C3	Fri.										7.2	-92	-83					2.4
19/6 -93	C3	Sat.										7.1	-66	-136					3.2
20/6 -93	C3	Sun.	26.5	10.5	9.9		3.44	1.36				7.0	-19	-230					2.3
21/6 -93	C3	Mon.	33.6	12.7	13.3	227	4.04	2.16			7.3	7.1	64						3.5
22/6 -93	C3	Tue.	34.4	10.2	10.2	549	4.06	2.22	1.12		8.1	7.1	-182		600	200	2.3		2.3
23/6 -93	C3	Wed.	35.4	12.5	11.8	445	4.16	1.48	1.12		8.1	7.1	-178						2.7
24/6 -93	C2	Thu.										7.2	-127						2.5
25/6 -93	C2	Fri.										7.0	-293	-20					2.3
26/6 -93	C2	Sat.										7.0	-272	-19					2.1
26/6 -93	C2	Sat.										7.0	-272	-19					2.1
27/6 -93	C2	Sun.										6.9	-276	13					3.3
28/6 -93	C2	Mon.	31.0	16.1			4.14	1.94			8.1	6.9	26	15					2.4
29/6 -93	C2	Tue.	34.6	12.1		187	4.04	2.32			8.1	7.1	-3	97					2.4
30/6 -93	C2	Wed.	36.7	12.8	11.6	259	4.22	2.80	1.38		7.9	7.1	-78	-24	300	200	2.2		2.6
1/7 -93	C2	Thu.	33.5	9.2	9.0	182	4.24	2.38	1.34		8.4	7.0	-64	-49					2.1
2/7 -93	C4	Fri.				274						7.1	-172	-72					2.6
3/7 -93	C4	Sat.										7.0	-173	-50					3.7
4/7 -93	C4	Sun.	26.1	9.8	8.8		3.44	1.78	0.78		8.6	7.0	22	-29					2.5
5/7 -93	C4	Mon.	28.2	9.2	9.2		4.08	2.16	1.22		8.6	7.0	-22	-53					2.5
6/7 -93	C4	Tue.	31.2	10.3	11.0	171	4.12	2.22	1.22		7.9	6.9	-79	-57					2.4
7/7 -93	C4	Wed.	31.6	12.3		193	4.04	2.24				6.9	-86	-53	210	260	2.3		2.6
8/7 -93	C4	Thu.	30.4	11.0	9.7	178	3.82	2.10	0.90		7.9	6.9	-108	-66					2.4
9/7 -93	C4	Fri.										6.9	-194	-61					4.4
10/7 -93	D4	Sat.																	4.5
11/7 -93	D4	Sun.	23.2	9.1	9.1	158	3.76	1.54	0.80		8.7								2.8
12/7 -93	D4	Mon.	8.9	9.1	6.3		1.60	1.26	0.58		8.7								8.6
13/7 -93	D4	Tue.	11.8	8.6	6.8	136	2.50	1.90	0.98		8.9								8.3
14/7 -93	D4	Wed.	17.9	7.3	7.2	260	3.34	1.98	1.18		8.9								4.8
15/7 -93	D4	Thu.	17.6	8.8	8.3	184	3.28	2.00	1.18		8.9				250	200	2.6		3.0
16/7 -93	D4	Fri.																	3.4

Date	PROCESS	Day of the week	Total nitrogen: Primary settled WW	Effluent	RBC	Return activated sludge	Alkalinity: Primary settled WW	Effluent	RBC	Oxygen concentration: Aeration tank	Effluent from RBC(estimated)	Effluent pH	Ox. red. potential: Anoxic tank	Aerated tank	Sett. of mixed liquor: Sludge volume (30 min)	Stirred sl. vol(30min)	Initial settling velocity	Turbidity	Flow to main WWTP
Unit			g N/m³				ekv HCO₃/m³			g O₂/m³			mV		ml/l	ml/l	m/h	NTU	m³/s
17/7 -93	D4	Sat.																	3.0
18/7 -93	D4	Sun.	18.3	7.5	7.6		3.30	1.62	0.86		8.3								3.0
19/7 -93	D4	Mon.	24.0	8.3	8.2	162	3.66	1.76	0.98		8.3								2.8
20/7 -93	D4	Tue.	26.4	10.1	9.8	253	3.74	2.06	1.06		8.1								3.0
21/7 -93	D4	Wed.				253													4.3
22/7 -93	D4	Thu.	21.1	9.8	8.5	282	3.10	1.82	0.94		8.6								4.0
23/7 -93	D4	Fri.																	3.9
24/7 -93	D4	Sat.																	3.0
25/7 -93	D4	Sun.	21.8	9.1	9.0		3.60	1.74	1.03		8.6								2.8
26/7 -93	D4	Mon.				205													2.9
27/7 -93	D4	Tue.	25.5	8.6	8.6	278	3.84	2.14	1.30		8.6				320	250	2.1		3.1
28/7 -93	D4	Wed.																	2.5
29/7 -93	D4	Thu.	25.8	8.4	7.9	317	3.80	1.94	1.10		8.6								2.8
30/7 -93	D4	Fri.																	4.4
31/7 -93	D4	Sat.																	3.9
1/8 -93	D4	Sun.	22.1	9.4	9.0		3.42	1.52	0.72		8.1								3.1
2/8 -93	D4	Mon.				190													2.9
3/8 -93	D4	Tue.	20.6	10.0	9.9	341	3.20	2.48	1.44		8.1								3.8
4/8 -93	D4	Wed.																	4.6
5/8 -93	D4	Thu.	23.6	8.3	8.4	291	3.54	2.22	1.26		8.6								3.2
6/8 -93	D4	Fri.																	4.9
7/8 -93	D4	Sat.																	5.4
8/8 -93	D4	Sun.																	3.8
9/8 -93	D4	Mon.																	4.1
10/8 -93	D4	Tue.																	8.0
11/8 -93	D5	Wed.																	7.3
12/8 -93	D5	Thu.																	8.0
13/8 -93	D5	Fri.																	8.9
14/8 -93	D5	Sat.																	5.9
15/8 -93	D5	Sun.																	4.1
16/8 -93	D5	Mon.																	4.5
17/8 -93	D5	Tue.																	4.5
18/8 -93	D5	Wed.																	3.6
19/8 -93	D5	Thu.	25.2	10.5	9.9	312	4.14	2.58	1.36		8.1								3.3
20/8 -93	D5	Fri.																	3.3
21/8 -93	D5	Sat.																	3.7
22/8 -93	D5	Sun.																	2.7
23/8 -93	D5	Mon.	24.8	13.5	11.2	352	3.84	2.24	1.06		8.1								3.5
24/8 -93	D5	Tue.	25.6	11.7	11.1	148	3.76	2.32	1.04		8.1								3.3
25/8 -93	D5	Wed.																	4.2
26/8 -93	D5	Thu.	27.2	10.3	10.2	309	4.06	2.30	0.96						260	180	2.5		3.3
27/8 -93	D5	Fri.																	3.8
28/8 -93	D5	Sat.																	3.0
29/8 -93	D5	Sun.	28.6	12.4	12.6		3.34	1.90	0.72										3.1
30/8 -93	D5	Mon.				304													3.6
31/8 -93	D5	Tue.	25.9	9.6	9.3	350	4.00	2.30	1.22		8.1				240	200	2.6		3.1
1/9 -93	D5	Wed.								2.1		7.3	-141	1					3.1
2/9 -93	D5	Thu.	30.9	10.2	10.6	327	3.84	2.08	0.98	2.2	7.9	7.3	-145	-4					3.1
3/9 -93	D5	Fri.								2.1		7.4	-165	-17					3.7
4/9 -93	D5	Sat.								2.3		7.6	-124	-23					2.9
5/9 -93	D5	Sun.	31.0	9.5	11.2		3.46	1.86	1.06	2.8	8.6	7.5	-62	14					2.7
6/9 -93	D5	Mon.				302				2.6		7.4	-118	-42					3.0
7/9 -93	D5	Tue.	41.6	17.8	15.8	285	5.40	2.74	0.86	2.1	7.9	7.4	-193	-45	170	120	3.9		2.8
8/9 -93	D5	Wed.								2.3		7.1	-103	7					2.0
9/9 -93	D5	Thu.	33.2		14.9	272	4.12	2.82	1.38	2.4	7.0	7.0	-136	13					3.7
10/9 -93	D5	Fri.		17.7						2.2		7.1	-208	10					2.7
11/9 -93	D5	Sat.								2.1		7.1	-206	23					3.1
12/9 -93	D5	Sun.	37.0	11.2	11.5		4.40	2.06	1.12	2.3	8.6	7.1	-180	20					2.5
13/9 -93	D5	Mon.				407				2.1		7.1	-231	-43					2.9
14/9 -93	D5	Tue.	35.4	14.2	14.1	349	4.12	2.50	1.30	2.1	8.1	7.1	-274	-89	280	210	2.6		3.1
15/9 -93	D5	Wed.								2.1		7.0	-234	-25					3.0
16/9 -93	D5	Thu.	35.9	14.8	12.4	335	4.04	2.46	1.30	2.1	8.1	7.0	-202	14					2.7
17/9 -93	D5	Fri.								2.1		7.0	-199	14					3.8
18/9 -93	D5	Sat.								2.2		6.9	-135	31					2.4
19/9 -93	D5	Sun.		16.8	13.1		4.52	1.98		2.2	8.7	6.9	-120	33					2.5
20/9 -93	D5	Mon.				290				2.3		7.0	-169	-3					2.8
21/9 -93	D5	Tue.	32.1	14.7	12.9	315	3.98	2.54	1.34	2.5	8.1	7.0	-179	-36	260	180	3.0		3.0
22/9 -93	D4	Wed.								2.2		7.0	-159	-11					3.0
23/9 -93	D4	Thu	32.2	10.3	12.7	274	4.08	2.62	1.36	2.2	8.1	7.1	-126	22					2.9
24/9 -93	D4	Fri.								2.1		7.1	-91	34					2.6
25/9 -93	D4	Sat.								2.1		7.1	-68	32					2.9
26/9 -93	D4	Sun.	24.0	12.4	11.0		3.18	2.24	0.88	2.2	6.8	7.0	-77	43					4.1

Date	PROCESS	Day of the week	Total nitrogen: Primary settled WW	Effluent	RBC	Return activated sludge	Alkalinity Primary settled WW	Effluent	RBC	Oxygen concentration: Aeration tank	Effluent from RBC(estimated)	Effluent pH	Ox. red potential: Anoxic tank	Aerated tank	Sett. of mixed liquor: Sludge volume (30 min)	Stirred sl vol(30min)	Initial settling velocity	Turbidity	Flow to main WWTP
Unit			g N/m^3				ekv HCO$_3$/m^3			g O$_2$/m^3			mV		ml/l	ml/l	m/h	NTU	m^3/s
27/9 -93	D4	Mon.				278				2.1		7.0	-76	44					3.4
28/9 -93	D4	Tue.	38.2	11.8	11.4	465	4.48	2.78	1.46	2.1	8.2	7.1	-68	36	325	224	2.4		2.8
29/9 -93	D4	Wed.								2.1		7.1	-67	28					3.1
30/9 -93	D4	Thu.	36.7	13.0	12.9	274	4.36	2.70	1.50	2.1	8.2	7.1	-60	8					2.7
1/10 -93	D4	Fri.								2.2		6.5	-193	-151					2.5
2/10 -93	D4	Sat.								2.1		5.0	-500	-499					2.6
3/10 -93	D4	Sun.	33.7	12.7	12.8		4.12	2.38	1.10	2.3	8.2	5.0	-500	-499					2.7
4/10 -93	D4	Mon.				298				2.2		5.0	-500	-499					3.7
5/10 -93	D4	Tue.	26.3	12.0	10.0	335	3.46	2.34	1.06	2.1	8.2	6.1	-234	-499	270	190	2.8		4.0
6/10 -93	D4	Wed.								2.3		7.0	-75	-499					3.4
7/10 -93	D4	Thu.				388				2.2		7.1	-114	-499					3.3
8/10 -93	D4	Fri.								2.2		7.1	-101	-499					2.9
9/10 -93	D4	Sat.								2.2		7.0	-96	-499					4.3
10/10 -93	D4	Sun.								3.9		6.9	56	-499					7.0
11/10 -93	D4	Mon.								2.9		7.0	-81	-499					6.0
12/10 -93	D4	Tue.	18.4	10.4	10.1	287	2.90	2.22	1.54	2.2	8.9	7.1	-93	-499	340	200	2.7		4.4
13/10 -93	D4	Wed.								3.1		6.9	44	-499					6.7
14/10 -93	D4	Thu.	19.5	10.8	9.9	265	3.22	2.38	1.28	2.3	8.7	6.9	-27	-498					5.8
15/10 -93	D4	Fri.								2.1		7.0	-81	-499					4.4
16/10 -93	D4	Sat.								2.3		7.0	-66	-499					4.0
17/10 -93	D4	Sun.	21.9	9.8	9.3		3.28	1.94	0.96	3.1	9.2	6.9	-16	-499					3.8
18/10 -93	D4	Mon.				275				3.1		6.9	-26	-196					3.5
19/10 -93	D4	Tue.	27.7	9.9	9.9	356	3.68	2.38	1.12	2.4	8.9	6.9	-71	68	220	140	3.4		3.2
20/10 -93	D4	Wed.								2.2		6.9	-46	178					3.4
21/10 -93	D4	Thu	32.5	11.9	11.1	362	4.08	2.14	1.14	2.2	8.4	7.0	-63	177					3.5
22/10 -93	D4	Fri.								2.2		6.9	-27	182					3.0
23/10 -93	D4	Sat.								2.3		6.9	2	183					2.7
24/10 -93	D4	Sun.	36.2	14.4	14.5		4.04	1.94	0.90	2.9	8.7	7.0	30	183					2.8
25/10 -93	D4	Mon.				337				2.6		7.0	14	170					3.2
26/10 -93	D4	Tue.	33.5	14.2	14.7	464	3.84	2.16	1.22	2.1	8.7	6.9	-34	152	230	140	3.2		3.1
27/10 -93	D4	Wed.								2.2		6.9	-24	133					3.0
28/10 -93	D4	Thu.	39.7	15.0	14.9	397	4.16	2.58	1.30	2.2	8.4	7.0	-34	123					2.9
29/10 -93	D4	Fri.								2.2		7.0	-61	117					2.9
30/10 -93	D4	Sat.								2.6		6.9	2	116					2.4
31/10 -93	D4	Sun.	37.9	14.2	13.5		3.80	1.52	0.56	2.9	9.2	6.8	13	109					2.7
1/11 -93	D4	Mon.				274				2.2		6.9	-68	60					3.0
2/11 -93	D4	Tue	35.9	11.8	11.2	325	3.84	1.94	0.70	2.2	8.7	7.2	-102	83	340	206	2.6		2.8
3/11 -93	D4	Wed.								2.1		7.7	-83	113					2.9
4/11 -93	D4	Thu.	37.0	12.3	12.2	316	3.84	2.30	0.82	2.1	8.7	7.6	-87	130					2.8
5/11 -93	D4	Fri.								2.1		7.4	-90	137					2.6
6/11 -93	D4	Sat.								2.5		7.7	-31	144					2.3
7/11 -93	D4	Sun.	38.3	13.4	13.6		4.24	1.70	0.48	3.2	8.4	8.0	32	149					2.3
8/11 -93	D4	Mon.				392				2.7		8.1	-31	152					3.1
9/11 -93	D4	Tue.	32.3	10.9	12.1	340	3.76	1.92	0.72	2.1	8.7	7.0	-89	152	710	214	2.6		2.8
10/11 -93	D4	Wed.	32.0	10.9	10.2	433	3.92	2.02	0.78	2.1	8.7	6.8	-59	144					3.0
11/11 -93	D4	Thu.				378				2.5		6.8	-34	139					2.8
12/11 -93	D4	Fri.								2.5		8.0	-52	140					3.5
13/11 -93	D4	Sat.								2.6		6.8	-5	139					3.4
14/11 -93	D4	Sun	18.7	10.4	10.1		2.38	1.50	0.50	3.8		6.7	23	140					4.5
15/11 -93	D4	Mon.				322				3.0		6.8	35	138					4.6
16/11 -93	D4	Tue.								2.2		6.8	-47	137					3.6
17/11 -93	D4	Wed.			10.5	287			0.88	2.2		6.8	-72	135	290	186	2.6		3.4
18/11 -93	D4	Thu.	27.2	9.8	9.3	224	3.68	2.02	0.98	2.1 X		6.8	-71	130					3.1
19/11 -93	D4	Fri.								2.2		6.8	-91	121					3.1
20/11 -93	D4	Sat.								2.8		6.7	-86	116					2.8
21/11 -93	D4	Sun.	30.0	13.1	13.4		3.74	1.76	0.56	2.6	8.8	6.7	2	124					2.9
22/11 -93	D4	Mon.		13.8	14.1	316		1.18	0.16	3.6		6.6	63	126					2.9
23/11 -93	D4	Tue	25.5	11.3	11.1	453	3.12	1.34	0.38	3.3	9.4	6.7	30	117					2.5
24/11 -93	D4	Wed.	27.2	12.2	11.6	455	3.14	1.58	0.40	3.8	9.0	6.8	34	117					3.1
25/11 -93	D4	Thu	34.6	13.4	14.2	558	3.46	1.70	0.40	3.3	9.0	6.8	38	118	285	182	2.8		3.0
26/11 -93	D4	Fri.								2.2		6.9	-55	120					2.9
27/11 -93	D4	Sat.								2.4		6.9	-61	118					3.2
28/11 -93	D4	Sun.	29.8	13.2	11.6		3.60	1.92	0.74	2.9	8.8	6.9	1	120					3.1
29/11 -93	D4	Mon.				472				2.4		6.9	-26	123					2.8
30/11 -93	D4	Tue.	30.2	9.6	9.2	516	3.76	1.90	0.90	2.7	8.8	7.0	-9	121	280	200	2.4		2.4
1/12 -93	D4	Wed.								2.7		7.0	-9	121					3.1
2/12 -93	D4	Thu.				232				2.3		7.0	-106	120					3.5
3/12 -93	D4	Fri								2.6		6.8	-143	115					4.8
4/12 -93	D4	Sat.								3.0		6.6	-168	118					8.9
5/12 -93	D4	Sun.	10.6	7.8	6.9		1.86	1.52	0.78	4.3	10.0	6.6	-137	112					8.4
6/12 -93	D4	Mon.				478				2.2		6.8	-137	115					6.3
7/12 -93	D4	Tue	18.5	11.7	11.4	414	2.58	2.08	1.10	2.2	9.2	6.9	-155	119	260	228	1.8		5.8

Date	PROCESS	Day of the week	Total nitrogen: Primary settled WW	Effluent	RBC	Return activated sludge	Alkalinity: Primary settled WW	Effluent	RBC	Oxygen concentration: Aeration tank	Effluent from RBC(estimated)	Effluent pH	Ox. red. potential: Anoxic tank	Aerated tank	Sett. of mixed liquor: Sludge volume (30 min)	Stirred sl. vol(30min)	Initial settling velocity	Turbidity	Flow to main WWTP
Unit			g N/m^3				ekv HCO$_3$/m^3			g O$_2$/m^3			mV		ml/l	ml/l	m/h	NTU	m^3/s
8/12 -93	D4	Wed.								2.5		6.8	-188	117					8.1
9/12 -93	D4	Thu.	22.2	11.3	11.2	312	3.02	2.14	1.12	2.2	9.4	6.8	-166	118					5.5
10/12 -93	D4	Fri.								2.1		6.9	-102	113					4.6
11/12 -93	D4	Sat.								2.7		6.9	-113	110					5.6
12/12 -93	D4	Sun.	17.1	9.2	8.9		2.56	1.74	0.90	3.3	10.0	6.8	-69	101					4.8
13/12 -93	D4	Mon.								3.9		6.9	-64	91					3.8
14/12 -93	D4	Tue.	20.4	9.7	9.4	336	3.02	2.10	1.18	3.0	9.2	6.8	-154	74	250	160	2.7		4.1
15/12 -93	D4	Wed.								2.2		6.8	-297	80					4.1
16/12 -93	D4	Thu.								1.9		6.9	-262	76					3.8
17/12 -93	D4	Fri.								2.1		6.9	-262	1					5.0
18/12 -93	D4	Sat.								2.4		6.7	-95	-86					4.8
19/12 -93	D4	Sun.								3.6		6.7	-81	-40					9.0
20/12 -93	D4	Mon.								5.2		6.7	5	-16					9.3
21/12 -93	D4	Tue.	13.6	10.2	9.0	396	2.16	1.84	0.80	4.3	9.6	6.8	-70	-97	110	140	4.4		7.9
22/12 -93	D4	Wed.								2.5		6.8	-122	-111					6.7
23/12 -93	D4	Thu.								2.9		6.5	-137	-87					7.4
24/12 -93	D4	Fri.								2.5		6.6	-127	-104					5.8
25/12 -93	D4	Sat.								4.2		6.5	-12	-68					3.7
26/12 -93	D4	Sun.								5.1		6.5	37	-50					4.1
27/12 -93	D4	Mon.								5.1		6.6	80	-44					3.5
28/12 -93	D4	Tue.								4.4		6.7	25	-64					3.5
29/12 -93	D4	Wed.								3.2		6.7	-38	-88					4.0
30/12 -93	D4	Thu								2.6		6.5	-96	-85					5.0
31/12 -93	D4	Fri.								3.7		6.2	-104	-42					6.4
1/1 -94	D4	Sat.								4.1		6.3	-36	-17					5.0
2/1 -94	D4	Sun.								4.7		6.4	31	-33					4.6
3/1 -94	D4	Mon.	26.0	15.7	14.6	378	3.36	2.46	0.96	4.0	9.4	7.0	-35	-83					5.0
4/1 -94	D4	Tue.								3.5		6.8	-64	-89					4.0
5/1 -94	D5	Wed.								3.6		6.6	-87	-61	150	230	2.7		6.0
6/1 -94	D5	Thu.								4.1		6.5	-46	-32					7.1
7/1 -94	D5	Fri.								4.8		6.8	-7	-27					7.9
8/1 -94	D5	Sat.								5.1		6.7	20	-2					8.7
9/1 -94	D5	Sun.								4.6		6.8	4	-38					6.7
10/1 -94	D5	Mon.	19.3	11.1	9.7	254	2.90	1.92	0.94	4.7	10.2	6.8	-38	-51					6.0
11/1 -94	D5	Tue.																	5.3
12/1 -94	D5	Wed	18.4	9.3	8.8	662	2.58	1.92	1.06	2.9	10.0	6.8	-93	-108	270	176	2.4		5.1
13/1 -94	D5	Thu.								2.9		6.6	-70	-85					10.0
14/1 -94	D5	Fri								4.3		6.8	-40	-83					9.0
15/1 -94	D5	Sat.								4.3		6.8	-45	-116					7.1
16/1 -94	D5	Sun.	17.8	18.4	13.1		2.56	2.06	0.98	6.8	9.6	7.0	-41	-107					6.2
17/1 -94	D5	Mon				32				4.8		6.8	-128	-144					5.8
18/1 -94	D5	Tue.				355				3.3		6.5	-195	-172					4.5
19/1 -94	D4	Wed.	14.7		7.8	425	2.06		0.13	4.2		6.3	-124	-96	200	130	3.8		6.1
20/1 -94	D4	Thu				299				3.7		6.4	-69	-50					5.3
21/1 -94	D4	Fri								3.3		6.6	-92	-55					5.1
22/1 -94	D4	Sat								3.4		6.7	-69	-42					5.2
23/1 -94	D4	Sun.								4.4		6.6	-32	-11					5.4
24/1 -94	D4	Mon.				287				4.7		6.7	-30	10					5.8
25/1 -94	D4	Tue.	22.9	14.7	10.7	417	3.18	2.26	1.28	4.0	9.6	6.8	-90	-4	180	116	3.3		5.4
26/1 -94	D4	Wed.								3.8		6.8	-105	23					5.1
27/1 -94	D4	Thu.	14.5	8.6	7.4	201	2.24	1.74	1.02	3.7	10.2	6.8	-94	81					5.8
28/1 -94	D4	Fri								4.6		7.0	-24	70					5.9
29/1 -94	D4	Sat.								4.6		7.0	36	-15					5.9
30/1 -94	D4	Sun	18.9	10.9	9.7		2.66	2.02	1.14	4.5	10.4	6.9	22	-22					4.8
31/1 -94	D4	Mon.				283				5.2		6.7	63	13					5.8
1/2 -94	D4	Tue	24.0	12.0	10.9	300	3.44	2.30	1.34	3.5	9.6	7.0	-64	-74	160	96	3.9		5.5
2/2 -94	D5	Wed.								4.0		7.1	-63	-58					3.7
3/2 -94	D5	Thu.				345				4.4		7.2	-54	-24					4.1
4/2 -94	D5	Fri								4.3		7.1	-55	0					3.8
5/2 -94	D5	Sat								4.2		6.9	-59	14					3.6
6/2 -94	D5	Sun.								4.9		7.0	-60	54					3.5
7/2 -94	D5	Mon.	30.4	14.5	14.5	353	3.62	2.18	1.10	3.9	10.0	6.8	-114	43					3.2
8/2 -94	D5	Tue.	29.9	12.5	11.7	396	3.58	2.34	1.22	4.0	9.4	6.8	-106	6					3.4
9/2 -94	D4	Wed.								3.5		6.9	-70	-15					3.6
10/2 -94	D4	Thu.	26.8	10.9	10.7	529	3.22	2.00	1.14	3.5	9.4	6.8	-39	-29	250	150	3.2		4.2

Appendix C Data from pilot plant experiments, profiles

During experimental periods C and D and thereafter "profiles" of ammonium and nitrate concentration and at times COD through the pilot plant. Grab samples were taken from different locations, filtered and analysed. Comments to table of data:

Time of sampling is approximate. The entire sampling may take about 0.5 hours.
Process: refer to appendix B.
Tank 5-Tank 10: location of sampling. The first 4 tanks were never used. When process 5 was used (C5, D5 or 5), primary settled wastewater, RAS and recirculated water were mixed in tank 5. When process 4 was used (C4, D4 or 4), RAS and recirculated water were mixed in tank 5 and primary settled wastewater added in tank 7. Tanks 5-7 were non-aerated and tanks 8-10 aerated.
Analysis: refer to main text.
Estimation of SS in tank 5 when operated as deoxygenation zone: refer to main text.

Date	95 06 09	95 07 07	95 07 14	95 07 20	95 0[illegible]	95 08 03	95 09 07	95 09 09	95 10 01
Time	15:00	14:00	12:30	14:10	15:00	13:15	14:00	14:00	10:30
Process	C4	C4	D4	D4	D4	D4	D5	D5	D4
Day of the week	Wed.	Wed.	Wed.	Tue.	Tue.	Tue.	Thu.	Thu.	Fri.
Temperature, °C	17.8	18.6	16.3	18	18	18.4	18.6	18.9	17.3
Flow to pilot plant, (Q1), l/s	0.97	0.8	2.2	1.09	1.11	1.37	1.1	1.5	0.63
Recirkulated water, (Q9), l/s	2.03	2.2	0.8	1.91	1.88	1.63	1.9	1.5	2.36
RAS, (Q5), l/s	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75
Suspended solids, g SS/l:									
Activated sludge tanks	2.49	2.03	2.08	3.02	3.3	2.4	2.63	2.63	2.883
RAS	11.04	8.26	11.52	13.29	13.41	10.71	11.6	11.6	12.526
Effluent	0.0084	0.041	0.0104	0.0089	0.0103	0.0107	0.01	0.01	0.0177
Ash content mixed liquor, % of SS	40.4	46.3	46.6	46.9	48.4	45.5	45	45	44
Nitrate concentration, g N/m^{3}:									
Primary settled wastewater	0.04	0.1	0.908	0.02	0.02	0.1	0.09	0.18	0.1
From RBC	8.6	9.5	6.2	9.13	7.9	7.22	9.17	9.74	10.34
Tank 5	5.1	5.06	0.1	5	3.76	3.05	2.55	2.16	6.59
Tank 6	0.07	0.765	0.1	0.6	0.33	0.56	1.94	0.23	2.56
Tank 7	1.5	0.904	0.1	0.08	0.67	0.02	0.46	0.1	1.03
Tank 8	0.67	1.22	0.033	0.09	0.82	0.04	0.6	0.05	0.93
Tank 9	0.31	1.38	0.058	0.14	0.84	0.09	0.74	0.11	0.89
Tank 10	0.16	1.52	0.067	0.46	0.82	0.11	0.69	0.14	0.93
Effluent	0.03	1.09	0.078	0.11	0.3	0.03	0.14	0.1	0.14
Effluent (24 h average)	0.4	0.32	0.734	0.88	1.64	0.83	0.97	1.16	
RAS	0.13	0.22	0.1	0.1	0.05	2.44	0.28	0.37	0.05
Ammonium concentration, gN/m^{3}									
Primary settled wastewater	28	24	8.62	20	19	19.1	20.9	24.9	29.71
From RBC	1.2	1.5	0.72	1.22	0.67	0.93	1.36	1.86	1.05
Tank 5	3.3	4.74	4.54	5.28	3.53	3.9	9.32	10.3	3.95
Tank 6	8.7	9.98	7.36	10.2	9.17	9.19	9.32	11.6	8.02
Tank 7	9.7	9.12	7.59	9.75	7.83	9.46	9.32	12.3	8.43
Tank 8	8.6	9.12	7.12	9.12	7.62	8.73	9.32	13	8.14
Tank 9	8.5	9.12	7.09	9.06	7.52	8.64	9.09	13	7.91
Tank 10	8.2	9.33	6.92	9.3	7.68	8.37	8.86	12.9	8.14
Effluent	8.3	9.1	6.34	9.7	7.4	7.46	9.55	12.8	10.17
RAS	10.1	10.75	6.68	11	8.98	8.58	10.2	12.2	12.09
COD, gO_2/m^{3};									
Primary settled wastewater									
COD of filtered sample, g O_2/m^{3}									
From RBC									
Tank 5									
Tank 6									
Tank 7									
Tank 8									
Tank 9									
Tank 10									
RAS									
Effluent									

Date	93 10 12	93 10 26	93 11 30	93 12 14	94 01 12	94 01 25	94 01 27	94 02 10	94 02 22
Time	7:30	0:00	10:30	13:00	10:00	14:00	10:00	12:40	10:00
Process	D5	D4	D4	D4	D5	D4	D5	D4	5
Day of the week	Tue.	Tue.	Tue.	Tue.	Wed.	Tue.	Thu.	Thu.	Tue.
Temperature, ^oC	16.2	15.3	13.7	11.7	10.8	10.3	9.5	11	11.8
Flow to pilot plant, (Q1), l/s	1.6	1.2	0.8	1.73	0.9	1.63	0.9	1.1	0.9
Recirkulated water, (Q9), l/s	1.35	1.84	2.02	1.27	1.75	1.4	1.79	1.9	1.79
RAS, (Q5), l/s	0.7	0.7	0.75	0.7	0.7	0.7	0.7	0.7	0.7
Suspended solids, g SS/l:									
Activated sludge tanks	2.587	2.227	2.537	2.58	2.95	2.1	3.6	2.813	3.607
RAS	12.934	12.693	14.354	11.98	17.8	14.12	16.19	20.806	16.194
Effluent	0.0141	0.0139	0.0161	0.0175	0.038	0.0148	0.0147	0.0197	0.0147
Ash content mixed liquor, % of SS	42	39.4	39.2	39.8	37.9	40.5	3.61	36.8	37.6
Nitrate concentration, g $N/m^{3:}$									
Primary settled wastewater	0.03	0.03	0.1	0.13	0.163	0.151	0.01	0.0779	0.01
From RBC	10.19	10.26	6.8	6.86	7.79	7.07	7.4	8.23	7.396
Tank 5	1.67	5.38	0.688	1.45	3.03	2.97	1.29	4.423	1.285
Tank 6	0.08	1.8	0.1	0.1	2.66	1.19	0.15	1.123	0.149
Tank 7	0.12	0.37	0.1	0.1	1.81	0.879	0.01	0.314	0.01
Tank 8	0.08	0.13	0.1	0.1	1.63	0.724	0.01	0.212	0.01
Tank 9	0.13	0.17	0.1	0.1	1.43	0.661	0.02	0.38	0.0184
Tank 10	0.16	0.19	0.1	0.1	1.18	0.657	0.01	0.554	0.011
Effluent	0.05	0.06	0.1	0.1	0.103	0.578	0.01	1.357	0.01
Effluent (24 h average)	0.354	0.546	0.055	0.029	0.383	0.13	0.19	0.5	0.05
RAS	0.06	0.09	0.1	0.1	0.1	0.1776	0.01	0.169	0.01
Ammonium concentration, gN/m^3									
Primary settled wastewater	16.71	31.34	21.45	12.7	12.3	14.38	22.47	16.28	22.47
From RBC	2.22	1.67	0.959	1.02	1.168	1.95	1.19	1.31	1.191
Tank 5	10.65	5.7	6.37	4.55	6.02	6.28	9.03	3.751	9.03
Tank 6	10.54	12.44	10.4	7.56	5.38	8.9	8.93	7.187	8.927
Tank 7	10.76	14.34	9.75	7.54	4.95	8.89	9.17	7.302	9.172
Tank 8	9.74	14.52	9.48	7.44	4.82	8.69	8.83	7.078	8.825
Tank 9	9.11	14.52	9.13	7.92	5.22	8.49	8.66	6.9	8.655
Tank 10	8.83	14.29	8.62	7.75	4.54	8.45	8.39	6.751	8.388
Effluent	11.96	12.84	7.32	7.26	7.63	8.81	8.66	7.773	8.655
RAS	13.95	13.82	7.75	8	8.48	8.83	9.1	8.825	9.1
COD, gO_2/m^3;									
Primary settled wastewater				192		307	428	332	428
COD of filtered sample, g O_2/m^3			171	100	100	109	220	131	220
From RBC			31	28	31	40	41	53	41
Tank 5			49	42	41	46	71	62	71
Tank 6			61	38	40	53	64	70	64
Tank 7			59	39	36	45	64	70	64
Tank 8									
Tank 9									
Tank 10									
RAS								101	
Effluent			37	28	37	40	45	58	45

Date	94 03 01	94 03 14	94 03 29	94 04 26	94 06 09	94 08 16	94 09 14
Time	13:00	11:00	11:00	10:00	9:00	11:00	11:00
Process	4	4	4	5	5	4	4
Day of the week	Tue.	Mon.	Tue.	Tue.	Thu.	Tue.	Wed.
Temperature, °C	11.5	7.3	11	14.5	17.7	20.5	17
Flow to pilot plant, (Q1), l/s	0.9	2.65	1.87	0.9	0.85	0.65	1.9
Recirkulated water, (Q9), l/s	1.8	0.3	1.16	1.72	1.61	1.26	1.17
RAS, (Q5), l/s	0.7	0.7	0.75	0.75	0.75	0.75	0.75
Suspended solids, g SS/l:							
Activated sludge tanks	3.56	1.917	2.073	1.813	3.64	2.14	3.47
RAS	15.1	14.294	10.506	15.206	15.82	11.55	18.87
Effluent	0.0187	0.0279	0.0317	0.0269	0.0125	0.0143	0.0215
Ash content mixed liquor, % of SS	37	37.4	37.1	39.2	46	42	43.1
Nitrate concentration, g N/m^3:							
Primary settled wastewater	0	1.16	0.71	0.1	0:14	0.05	0.24
From RBC	8.24	5.37	6.94	8.45	9.14	8.57	5.66
Tank 5	3.703	1.75	2.54	2.18	3.55	2.39	1.07
Tank 6	0.1974	1.31	1.32	0.99	0.36	0.33	0.14
Tank 7	0.0691	1.28	0.66	0.1	0.01	0.09	0.01
Tank 8	0.0013	1.27	0.61	0.1	0.05	0.06	0.01
Tank 9	0.0144	1.25	0.61	0.19	0.09	0.11	0.03
Tank 10	0.323	1.25	0.52	0.2	0.16	0.14	0.06
Effluent	0.0061	1.2	0.002	0.1	0.01	0.06	0.01
Effluent (24 h average)	0.01	1.31	0.681	0.17	0.48	0.88	0.48
RAS	0.006	0	0.023	0.1	0.04	0.1	0.02
Ammonium concentration, gN/m^3							
Primary settled wastewater	25.31	4.01	9.28	22.76	21.7	19.76	6.77
From RBC	1.531	0.55	2.43	0.8	0.66	0.01	0.36
Tank 5	4.15	3.4	5.48	8.96	7.68	6.07	3.42
Tank 6	10.49	3.8	7.2	8.72	8.95	7.73	4.76
Tank 7	10.21	3.93	7.34	9.06	9.53	8.37	4.9
Tank 8	10.31	3.99	7.39	9	8.74	8.1	4.8
Tank 9	9.99	4.1	7.35	8.48	8.66	8.05	4.72
Tank 10	10.22	4.06	7.38	8.53	8.36	7.71	4.99
Effluent	9.45	4.31	8.71	9.01	9.33	8.43	5.41
RAS	8.88	4.39	9.21	9.35	10.68	8.76	5.9
COD, gO_2/m^3;							
Primary settled wastewater	473	164	190	356	336	559	158
COD of filtered sample, g O_2/m^3	185	52	66	166	162	201	64
From RBC	45	31	38	34	12	35	23
Tank 5	47	39	51	48	39	44	38
Tank 6	56	46	43	53	36	46	42
Tank 7	57	37	41	50	37	54	42
Tank 8							
Tank 9							
Tank 10							
RAS						72	
Effluent	49	30	39	42	27	43	37

Appendix D Wastewater flow and effluent ammonium concentration

Corresponding values registered from August 1991 to October 1993 of:

24-hour average flow to main WWTP and
ammonium concentration of effluent grab samples from the present non-nitrifying activated sludge system of the main WWTP.

Appendix D

Flow and ammonium concentration

Flow to WWTP, m^3/s	Ammonium concentration from non-nitrifying activated sludge system, g N/m^3
2.1	23.6
2.1	22.0
2.1	16.5
2.1	13.1
2.1	24.4
2.1	25.8
2.4	20.3
2.4	25.2
2.4	13.2
2.4	22.4
2.4	23.7
2.5	23.5
2.5	21.3
2.5	25.7
2.5	20.2
2.5	16.3
2.6	19.9
2.6	19.7
2.6	25.3
2.6	22.1
2.6	23.6
2.6	23.0
2.7	24.1
2.7	25.2
2.7	19.7
2.7	17.5
2.8	20.7
2.8	26.7
2.8	20.8
2.8	25.6
2.8	25.2
2.8	24.5
2.8	14.6
2.8	17.4
2.8	22.2
2.9	23.2
2.9	20.0
2.9	15.9
2.9	16.3
2.9	21.6
2.9	25.1
2.9	15.6
2.9	21.9
2.9	21.5
2.9	21.9
3.0	16.3
3.0	24.3
3.0	29.2
3.0	23.6
3.0	18.8
3.0	18.6
3.0	21.0

Flow to WWTP, m^3/s	Ammonium concentration from non-nitrifying activated sludge system, g N/m^3
3.1	20.7
3.1	21.4
3.1	18.4
3.1	23.0
3.1	25.6
3.1	24.5
3.1	22.5
3.1	23.8
3.1	21.3
3.1	18.6
3.2	23.8
3.2	20.7
3.2	22.7
3.2	23.7
3.2	14.2
3.2	16.1
3.2	16.9
3.2	13.6
3.2	22.6
3.3	27.0
3.3	20.4
3.3	12.7
3.3	21.0
3.3	23.4
3.3	22.8
3.3	17.0
3.3	17.2
3.3	17.5
3.3	17.4
3.3	19.7
3.3	22.0
3.3	23.0
3.3	21.1
3.3	21.6
3.3	17.6
3.4	21.2
3.4	21.6
3.4	21.4
3.4	22.9
3.4	23.0
3.4	15.6
3.4	16.8
3.5	27.3
3.5	20.5
3.5	14.5
3.5	16.6
3.6	19.8
3.6	21.1
3.6	21.7
3.6	16.3
3.6	14.4
3.6	14.4

Flow to WWTP, m^3/s	Ammonium concentration from non-nitrifying activated sludge system, g N/m^3
3.6	14.6
3.7	21.1
3.7	17.4
3.7	16.2
3.8	17.8
3.8	19.2
3.8	18.2
3.9	16.2
3.9	21.9
4.0	16.6
4.0	11.5
4.2	19.9
4.2	17.9
4.2	22.2
4.2	15.8
4.2	14.0
4.2	14.9
4.3	15.0
4.3	11.6
4.3	14.9
4.3	18.0
4.4	20.4
4.4	16.0
4.4	12.4
4.4	21.2
4.4	19.8
4.5	13.7
4.5	14.3
4.6	15.4
4.6	11.4
4.6	13.4
4.8	21.9
4.8	7.6
5.1	22.0
5.4	15.7
5.8	13.3
5.8	8.0
5.9	11.1
6.0	9.2
7.3	7.8
7.3	9.4
7.6	5.8
7.9	7.3
8.3	4.3
8.6	7.9
8.9	4.4
8.9	6.2
9.5	7.0
10.2	6.7