Treatment of sludge return liquors: Experiences from the operation of full-scale plants

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ABSTRACT

More stringent effluent criteria with regard to nitrogen call for improved nutrient removal techniques in wastewater treatment plants (WWTPs). Besides optimisation of the liquid treatment train of the plants, over the last years, attention has increasinly centred on the problem of return flows from sludge treatment. Depending on sludge handling and treatment, some 15 to 25 % of the influent nitrogen load are usually returned from the sludge dewatering facility to the inlet of the WWTP. By minimising this extra nitrogen load, it can be expected to substantially improve the effluent quality. On a full-scale basis, mainly ammonia stripping and different biological processes have been applied, in Europe, for the treatment of process water streams with the overall goal to reduce the return nitrogen load. A recently performed survey on full-scale plants shows that only eight plants use ammonia stripping. Whereas the majority of WWTPs have been upgraded by implementation of biological measures for the treatment of return flows. Most of these biological systems use classical nitrification and denitrification or – to reduce the consumption of energy and organic substrate – nitritation and denitritation. One of the most recent developments in this field is deammonification which has so far been applied in three full-scale plants. Based on the experience gained from operation of two of these plants, it can be said that a stable nitrogen elimination of no less than 80 % is possible irrespective of the process configuration used, like the fixed film system at the Hattingen WWTP or the Sequential Batch Reactor (SBR) process at the Strass WWTP. While, in both cases, the operating costs are relatively low, the investment costs vary significantly as these strongly depend on the specific site conditions, i.e. the possibility to use existing reactors and machinery of the WWTP.

KEYWORDS

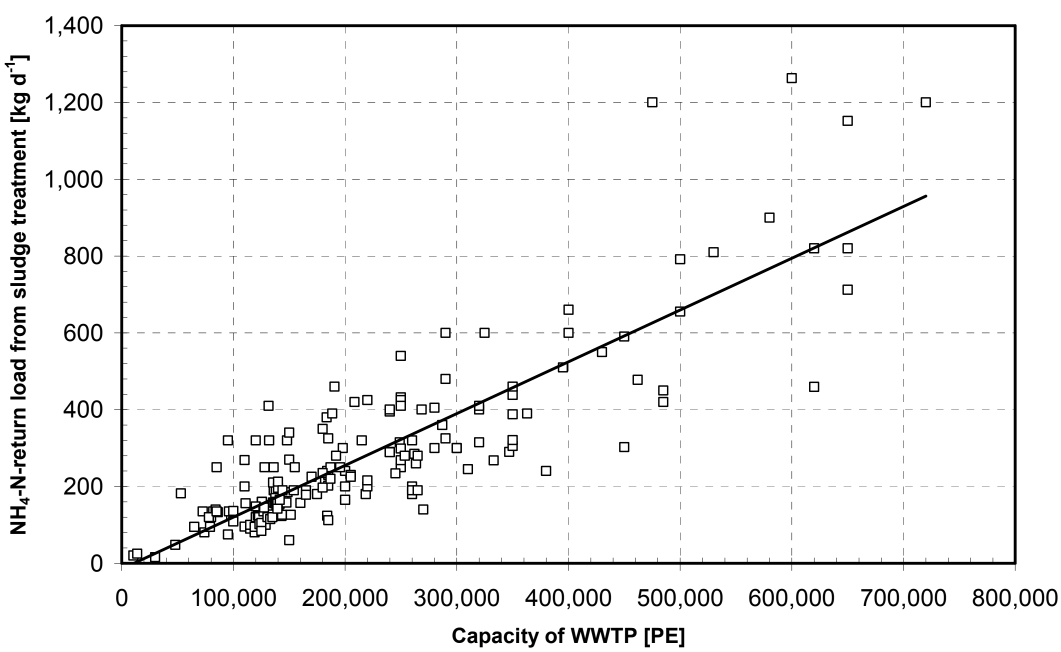
Return flows, reject water, sludge water treatment, deammonification, Kaldnes, SBR, Demon

INTRODUCTION

With adoption of the European Council Directive on urban wastewater treatment in 1991 (EU, 1991), a number of new emission standards were set for wastewater discharge. In sensitive areas, nutrient removal is mandatory for all plants with more than 10,000 population equivalents (PE). To ensure compliance with European legislation, the German effluent requirements for municipal wastewater treatment plants were laid down in the German Wastewater Ordinance (2002). For treatment plants with more than 10,000 PE, effluent criteria for nitrogen and phosphorus have been set to 18 mg/l N and 2 mg/l P respectively. While more stringent standards (13 mg/l N and 1 mg/l P) are applicable for treatment plants with a capacity of more than 100,000 PE. In contrast to most other European states, effluent standards in Germany are monitored on the basis of the so-called qualified grab samples or 2-h composite samples. This results in more stringent requirements with regard to process stability. As implementation of the Urban Wastewater Directive (UWWD) had to be finalized by the end of 2005, nearly all existing wastewater treatment plants (WWTPs), for which nutrient removal now is mandatory, had to be extended or at least optimized by that deadline. Athough today a number of European countries are being able to fulfill their obligations under the Directive, there is still a substantial backlog demand when taking into account all EU member states.

To improve treatment quality, either conventional measures, like e.g. the construction of new tanks, or approaches geared to process optimization can be used. As far as nitrogen elimination is concerned, a separate treatment of return flows from sludge treatment is a promising option because substantial nitrogen loads are usually recycled from the sludge treatment facility when sludge digestion and dewatering are being used. Figure 1 summarizes a Germany-wide survey on the nitrogen loads recycled to the inflow of the treatment plant. Calculated average return loads, under normal conditions, are in the range of 10 to 15% of the inflow nitrogen load. Therefore, it can be expected that with a treatment technology utilizing separate elimination of this specific nitrogen load it is possible to substantially improve the loading situation for the main or central wastewater treatment process.

Figure 1 – Return load of nitrogen in German WWTPs (Grömping 1998)



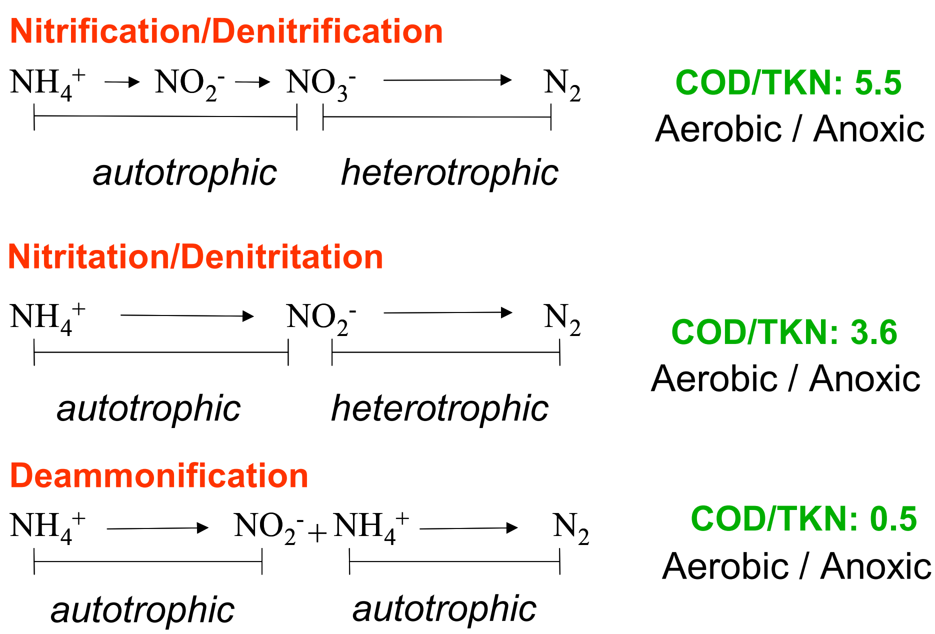
Several different options are available to establish a separate treatment of process water. This paper gives an overview of suitable concepts and of processes realized that target nitrogen elimination. Included is an inventory of full-scale plants for separate or sidestream treatment currently being operated in central Europe. Deammonification is one of the most promising routes in the field of separate treatment of nitrogen-rich process waters, which has received particular attention through recent years. The biological principles involved and full-scale realization, based on two different process variants, are discussed in greater detail.

MEASURES FOR SLUDGE WATER TREATMENT – AN OVERVIEW

Biological treatment

Basically, there are three approaches for eliminating nitrogen from sludge waters in sidestreams: First, the classical method of nitrification/denitrification, second, nitritation/denitritation and, third, deammonification (see Figure 2).

Figure 2 – Classification of process steps of biological nitrogen removal



Nitrification/denitrification techniques neccessitate, in many cases, the use of a supplementary carbon source on account of the mostly unfavourable C/N ratio prevailing in the sludge waters. Both external substrates and internal C-sources – like e.g. raw wastewater, primary sludge, or the filtrate from sludge disintegration – can be used for the purpose. Besides synthetic, external substrates (like acetic acid or methanol), also residual materials from industry and trade can be utilized that meet the quality standards for application in wastewater treatment. As regards nitrification/denitrification technology, particular attention should be given to the operational costs for the oxygen supply of the nitrification process and the carbon demand of the denitrification process which, compared to other options, is in fact much higher.

Technically, nitrification/denitrification can be readily implemented – like it is common practice in main stream processes – by providing a pre or intermittent denitrification step. Whereby frequently methods using suspended biomass and sludge return are employed. Besides, quite a number of SBR-plants for sludge water treatment have been built. And finally, there is also the possibility to use membrane biology or biofilm processes for sludge water treatment.

To save operating costs, it is possible to restrict ammonium oxidation which occurs during nitrification to nitrate, to the first, the nitritation, process step. The nitrite forming thereby can then be reduced to molecular nitrogen in a second, the denitritation, step. With this technology about 25% of the oxygen demand and no less than 40% of the carbon demand may be saved, compared to nitrification/denitrification (Abeling und Seyfried 1992). Generally, a possibly incomplete nitrogen elimination – resulting in an elevated nitrite content in the sludge waters – is not problematic because, normally, the nitrite is completely removed in the biological main stage.

To engineer the process that prevents total oxidation of the ammonium it is either possible to inhibit the nitrite oxidation (nitratation inhibition) (Anthonisen et al. 1976) or to wash out the nitrite oxidizers, which, at higher temperatures, exhibit a growth rate lower than that of the ammonium oxidizers (Knowles et al. 1965). To suppress the nitrite oxidation, it is necessary to raise the ammonia concentration and/or to limit the oxygen supply (Abeling and Seyfried 1992). Technically, to meet the first requirement, the pH-value can be raised, for example by addition of caustic soda, which will shift the dissociation balance between ammonium and ammonia. To wash out the nitrite oxidizers, for example continuous flow reactors without sludge return can be employed (Chemostat). Their hydraulic residence times should be just low enough so as to suppress the growth of nitrite oxidizers, while still maintaining the growth of the ammonium oxidizers. This concept has been successfully applied in the Netherlands in the so-called SHARON® process (Mulder et al. 2001). It works with hydraulic residence times of about 10 to 12 hours which, at reactor temperatures above 30°C, still allow for an effective suppression of the growth of nitrite oxidizers (Hellinga et al. 1998). Technically, also SBR-systems, the concept of which is to control the washing out of nitrite oxidizers by excess sludge removal, have proved to work satisfactorily (Fux et al. 2003). Finally, it is also possible to operate fixed-bed reactors for nitritation. However, the process technology involved makes it more difficult to fine-tune the process of nitritation inhibition. To obtain optimum performance, an additional expenditure of measuring and control instrumentation is necessary that should not be left unconsidered.

By application of the so-called deammonification process it is is even possible to do it without any dosage of carbon for denitrification. In two process steps the ammonium is directly and almost completely converted into molecular nitrogen. In the first step, a fixed portion of the ammonium present in the waters will have to be fully oxidized to nitrite, before, in the second step, the ammonium elimination process takes place, under anoxic conditions, with simultaneous reduction of nitrite and N_2 as the end product (see also Figure 2) (van de Graaf et al. 1995). However, with this process some 10% of the ammonium are converted to nitrate.

Suitable for the technical realization of the deammonification process are all fixed-bed systems, this so on account of the low growth rate of the bacteria accountable for the reaction. Experiences from large-scale facilities using synthetic rings as carriers (Hippen 2001) and the moving-bed process (Thöle et al. 2004) show that it is possible to establish a stable process. Principally, it is also possible to implement the process with suspended biomass (for example SBR) (Böhler et al. 2003, Wett 2005).

Physico-chemical treatment

Ammonia stripping

Ammonia stripping means that the ammonium is first converted to slightly volatile gaseous ammonia that is readily soluble in water. This gas is then physically stripped off from the water (Pfennig 1996). A balanced ammonium-to-ammonia ratio is a function of temperature and pH. At a pH of 10 and a temperature of 70°C, the dissociation equilibrium will be completely on the side of ammonia. That means, there is no ammonium in the water phase. If this status is to be obtained at a temperature of only 20°C, a pH greater than 11 will be required.

Subsequently, the ammonium dissolved in the water is to be transferred to the gas phase (desorption). For this process step, air and steam stripping in packed columns has proved to be the solution of choice in industrial-scale applications. Also desorption is temperature-dependent. Consequently, rising temperatures will accelerate the rate of reaction. That means on the other hand, the volumetric flow rate of the gas required for stripping will decrease if temperatures rise. But this eminent advantage compares with the unfavourably elevated energy demand typical of steam generation. Hence, large-scale application of steam stripping will only be economically viable at sites where sufficient steam is available, like, for example, in sewage sludge incineration or drying plants.

Downstream of desorption, the ammonia stripped into the gas phase has to be converted to a recyclable or disposable product. For ecological reasons, the ammonia should not be directly released to the atmosphere, which as a matter of fact would hardly be permitted by the authorities. For large-scale applications, acidic scrubbing involving the production of ammonium sulphate and rectification to aqueous ammonia have made their way into practice as reliable options. By rectification, a 25- to 35%-aqueous ammonia is produced that can be readily reutilised in flue gas scrubbers.

Precipitation of struvite

For a chemico-physical nitrogen removal from the sludge waters it is also feasible to convert the dissolved ammonium into an insoluble state by means of precipitation and to separate it from the sludge waters as solid matter. This can be achieved by precipitation of struvite ($MgNH_4PO_4$ 6 H_2O). With the concentrations of Mg and PO_4 generally present in the sludge waters, it will nevertheless be necessary to add both magnesium and phosphate to ensure an ammonium elimination worth-mentioning. Moreover, the precipitation performance is related to the given pH-value; the minimum of struvite solubility (maximum precipitation) is achieved at a pH of about 11.

There is some potential for struvite precipitation in facilities with enhanced biological phosphorus elimination, as the polyphosphates forming during wastewater treatment tend to redissolve in the course of digestion. Thereby, both phosphate and magnesium are being released into the sludge water. However, on account of the chemico-physical refixation processes occuring during the stabilization phase, the greater portion of the firstly dissolved phosphorus is bound up again through adsorption and precipitation processes (Jardin 1995). On the other hand, markedly higher magnesium concentrations in the sludge waters have been observed in facilities with enhanced biological phosphorus elimination. So conditions facilitating a targeted struvite precipitation are generally more favourable with this type of plant than with other WWTPs designed for chemical P-precipitation.

However, the large amounts of sludge produced with this method represent a great disadvantage. During the precipitation reaction, at least 17.5 kg of precipitation sludge are obtained per 1 kg of NH_4N. To what extent these sludges can be utilized in agriculture does not only depend on plant availability for the nutrients, but also on the scope of heavy metals or organic pollutants these contain. Industrial scale experiences, gained in Japan with struvite precipitation processes that primarily target phosphorus elimination, show that the pollutant load of the struvite forming in municipal WWTPs is rather low. If available in the form of a customised product, the struvite may well be successfully marketed in agriculture (Ueno and Fujii 2001). All legal requirements of the fertilizer ordinance have of course to be met.

Technically, struvite precipitation can be carried out in different ways. For one thing, there is the "classical" method of precipitation of the ammonium from the dissolved phase with subsequent separation in settling tanks. And for another, there is the fluidized-bed process that involves precipitation on sand particles as seed crystals (crystallization). Both approaches have become well-established methods in particular in large-scale applications. With the "classical" precipitation method, a significantly greater amount of sludges is to be expected due to co-precipitation of particulate substances from the sludge waters. Though the process has so far failed to gain general acceptance as an option for targeted nitrogen elimination, it is now increasingly being reconsidered on account of the current efforts to intensify the recovery of phosphorus from sewage sludge (DWA 2003).

Meanwhile, a variety of sludge water treatment plants have been built in Euopean WWTPs. Table 1 shows a selection of full-scale facilities with indication of the process technology used. A review of the requests for bids published in Germany over the last years confirms that the SBR-process for nitrogen elimination is the currently most frequently applied approach.

Table 1 – Selected full-scale wastewater treatment plants for separate nitrogen elimination from process water in Germany, Austria and the Netherlands

WWTP	capacity	process
Aachen-Süd	60,000	Air stripping
Cuxhaven	400,000	Air stripping
Göttingen	220,000	Air stripping
Heppenheim	80,000	Air stripping
Perfgebiet-Wallau	45,000	Air stripping
RHV Mattig-Hainbach (A)	45,000	Air stripping
Straubing	200,000	Air stripping
Cloppenburg	190,000	Steam stripping
Hattingen	100,000	Deammonification/Moving bed
Strass/Zillertal (A)	167,000	Deammonification/SBR
Dormagen	80,000	Membrane system
Kohlfurth	200,000	Membrane system
Fulda	150,000	Moving bed
Landshut	260,000	Moving bed
Bergisch-Gladbach	200,000	Nitrification only
Goch	122,000	Nitrification only
Gütersloh-Putzhagen	200,000	Nitrification only
Hamburg	2,100,000	Nitrification only
Kamp-Lintfort	75,000	Nitrification only
Krefeld	1,200,000	Nitrification only
München I	2,000,000	Nitrification only
Obere Lutter	380,000	Nitrification only
Rheda-Wiedenbrück	64,000	Nitrification only
Wolfhagen	23,000	Nitrification only
Aachen-Soers	458,000	SBR
Altötting	40,000	SBR
Bad Hersfeld	57,000	SBR
Bern (CH)	300,000	SBR
Bickenbach-Engelskirchen	25,000	SBR
Friedrichshafen	86,000	SBR
Hanau	270,000	SBR
Herbolzheim	15,000	SBR
Ingolstadt	235,000	SBR
Köln-Stammheim	1,450,000	SBR
Memmingen	275,000	SBR
Rosenheim	350,000	SBR
Salzburg (A)	620,000	SBR
Sinzing	115,000	SBR
Tuttlingen	40,000	SBR
Amsterdam-Ost (NL)	600,000	Sharon®

Rotterdam (NL)	470,000	Sharon®
Utrecht (NL)	400,000	Sharon®
Beverwijk (NL)	326,000	Sharon®
Garmerwolde (NL)	300,000	Sharon®
Zwolle (NL)	200,000	Sharon®
Cloppenburg	190,000	Steam stripping
Moers-Gerdt	250,000	Trickling filters
Nürnberg I	1,225,000	Trickling filters
Pforzheim	250,000	Trickling filters

NEW APPROACHES FOR SLUDGE TREATMENT BY DEAMMONIFICATION

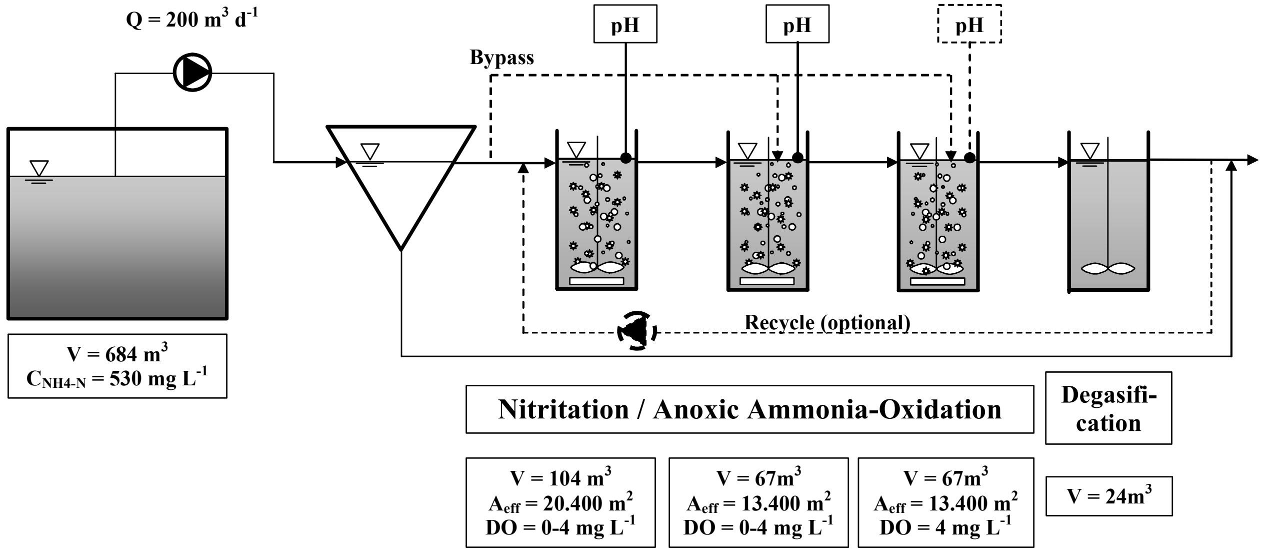
Principle

The deammonification process can be described as a process divided into two different steps and performed by two groups of organisms. In the first aerobic step, the ammonia is converted to nitrite with the help of bacteria of the nitrosomonas group. In the second anoxic step, nitrite and ammonia are converted to elementary nitrogen with the help of organisms of the group of planctomycetales. The process has been investigated in depth over the last decade, which included a variety of different technologies. Besides the use of floating or fixed carriers to build up a biofilm, also cultivation of organisms in suspended systems has undergone large-scale testing. In the following, the results from full-scale operations of deammonification in a biofilm as well as in a SBR system will be considered more closely.

Full-scale deammonification in a fixed-film system (Hattingen WWTP)

Full scale plant In 2000, a full-scale plant for the treatment of process water, using a fixed film system, was built by the Ruhrverband at the WWTP of Hattingen, Germany. The Hattingen WWTP was chosen as an ideally suited site for investigating the deammonification process due to the elevated nitrogen loads originating from different types of water (process water from a central drying facility, landfill leachate and process water from a central sludge treatment facility). The additional nitrogen load stemming from recycle streams accounts for more than 30% of the influent nitrogen load. After an intensive research phase focusing on nitritation and denitritation, deammonification was established on a full scale in 2003. During the last years, attention has been concentrated on process stabilization and optimization. The flow scheme of the full-scale plant is shown in Figure 3.

Figure 3 - Flow sheet of the full-scale deammonification plant at Hattingen WWTP



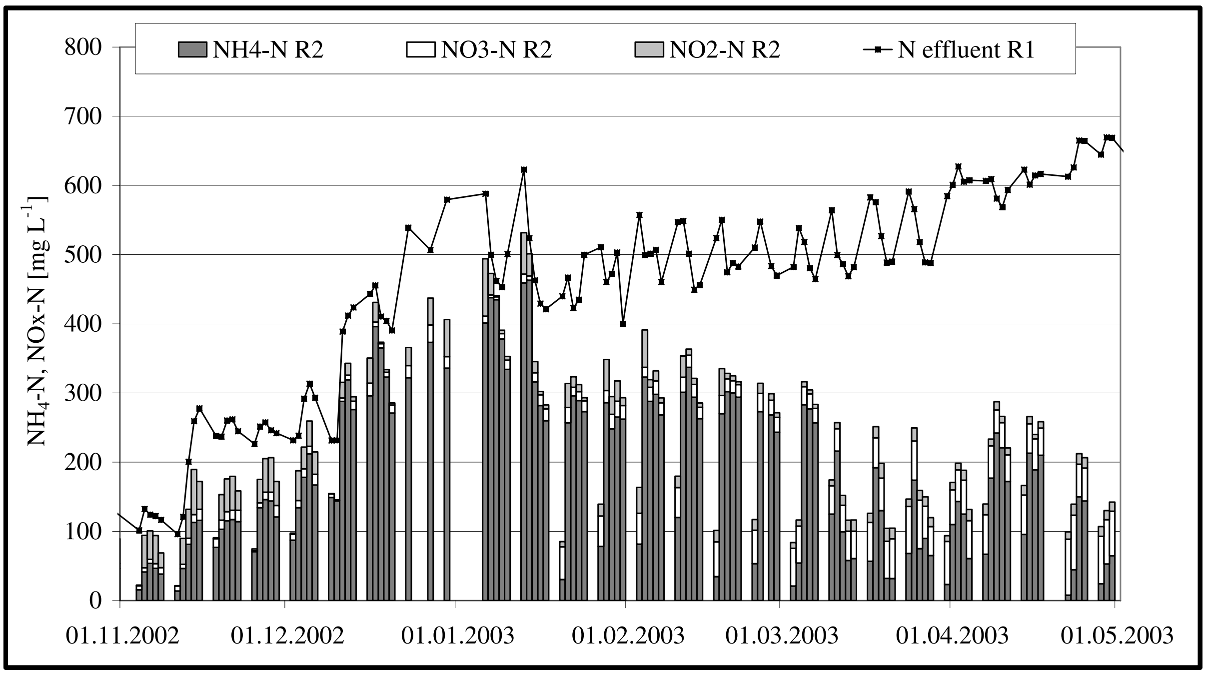
The total volume of the full-scale pilot plant, consisting of five tanks, is 319 m^3. One settling tank with a volume of 58 m^3 can be used for primary, secondary or intermediate clarification. The three biological reactors, the first holding 104 m^3, the second and third holding 67 m^3 each, may be either aerated or, using agitators, operated under anoxic conditions. A 24- m^3-degassing basin is available to prevent the dispersal of oxygen in the recycle stream (optional). The plant is charged by external feed pumps from the sludge-liquor storage tank with a capacity of 684 m^3 and, via external by-pass pipelines, the influent can be allocated to the three reactors.

As carriers needed to establish a biofilm, Kaldnes® Moving Bed material was used. It consists of polyethylene rings with a stable internal cross and a density slightly below that of water, which, therefore, can be suspended in the wastewater to allow a perfect circulation. The plastic elements provide a specific surface area of no less than 750 m^2 m^{-3}. With the biofilm mainly forming at the 'sheltered' inside, a value of 500 m^2 m^{-3} bulk volume can be assumed. To ensure adequate mixing of the Kaldnes® carrier elements also during deammonification under low oxygen concentrations, all three reactors are provided with both bubble aeration systems and agitators. Further, each reactor is equipped with a pH-measuring and a dosing system for acid and caustic soda addition to control the pH-value, as well as with an oxygen probe to monitor the oxygen concentration in the biological processes.

The plant is fed from the front side, with a bypass of approx. 30% directed to the third reactor. As there is no need for denitritation, neither a recycle stream, nor an external carbon source are required. The sludge liquor treatment plant has been designed to treat a partial stream of the daily sludge liquor influent, corresponding to an average wastewater flow rate of 200 m^3 d-1 and a nitrogen load of 120 kg d^{-1}. To meet the elimination target of 80%, the plant was sized on the basis of a surface elimination capacity of 5g NO_2-N m^{-2} d^{-1} for the denitritation and of 4.5 g NH_4-N m^{-2} d^{-1} for the nitritation process. The required specific capacity for the deammonification was assumed to be 2g NH_4-N m^{-2} d^{-1}, based on the selected reactor volumes in conjunction with the actual filling level and the design data for nitritation/denitritation.

Results Following a first nitritation/denitritation operating phase, the full-scale plant was modified to allow deammonification by using the denitritation section (reactor 1), formerly operated downstream, for the purpose of aerobic nitritation. In the then following two reactors, the oxygen concentration was lowered to boost the growth of planctomycetes in the anoxic zones of the biofilm. In the nitritation stage in reactor 1, a substrate was obtained in the form of a mixture of ammonium and nitrite after a period of about four weeks. To prevent the suppression of planctomycetes development by too high nitrite values, the influent concentration was lowered by addition of process-water. The resulting drop in temperature was compensated by the addition of warm water from the sludge drying facilities. In spite of the given almost optimal preconditions, no losses in nitrogen were registered over a period of several months. In a laboratory unit, using an operational setting comparable to that of the full-scale plant, nitrogen losses by deammonification were registered after three months.

Figure 4 - Startup of Deammonification in Reactor 2 (R2)

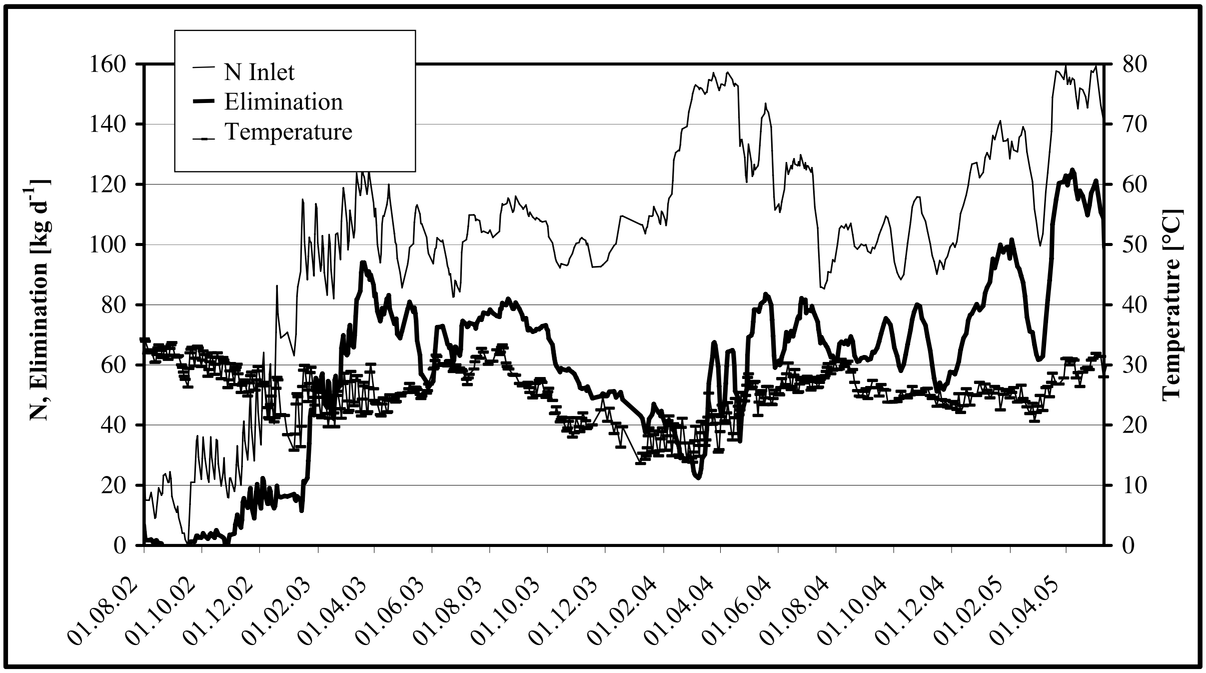


A comparison with the laboratory units showed that the biofilm in the full-scale plant was significantly thinner, so that different changes in operation, like optimization of the substrate supply by increasing the feed volume and reducing the agitator energy input, had to be implemented. Only by rearrangement of the agitator in reactor 2, it was possible to reduce the shear forces, which allowed the biofilm to build up to an adequate thickness in a relatively short period of time. Corresponding to that rise of biofilm thickness, first losses of nitrogen were registered.

Figure 4 shows the concentration of inorganic nitrogen compounds in the effluent of reactor 2 together with the summarized nitrogen concentrations in the effluent of reactor 1. The 'gap' between this line and the bars displays the nitrogen removed. By performing a gene-specific probe test it was possible to provide evidence for the presence of an adequate population level of aerobic and anoxic ammonia oxidizers, produced by deammonification.

In Figure 5, the nitrogen feed for the plant is shown together with the temperature of the wastewater and the load converted to elementary nitrogen. Maximum performance is at approx. 70 to 80% converted at a load between 100-160 kg N d^{-1} during normal operation.

Figure 5 - Overall performance of the full-scale deammonification process



At the beginning of 2004, a reduced nitrogen elimination of down to 25 kg N d^{-1} was detected, caused by very low wastewater temperature and followed by biofilm loss. Meanwhile, by rising both the nitrogen load and the oxygen supply, it was possible to redevelop the biofilm thickness and to enhance the nitrogen elimination performance to a satisfactory level. During the winter months of 2004/2005, plant performance could be upheld, because the water temperature did not drop below 20°C due to installation of a cover on the sludge water storage tank. Nevertheless, monitoring of the biomass cultivated on the Kaldnes®-carriers remains of great importance, this being a reliable indicator for an efficient performance of the deammonification plant. Due to ongoing problems in ensuring stable nitritation in the first reactor by using selective inhibition of nitrate build-up caused by free ammonia or oxygen deficiency, reactor 1 has been aerated intermittently since July 2002. Since then, nitritation was found to be stable for several months until April 2003, when unscheduled deammonification occurred in this reactor. The nitrite built-up in reactor 1 during the aerobic phases was completely used up during the following anoxic periods. As a consequence, a few weeks later, the operation mode of reactor 2 had also to be changed to intermittent aeration, because the nitritation reactor 1 completely used up the available nitrite. On account of the good performance of reactors 1 and 2, reactor 3 showed only a thin biofilm due to substrate deficiency in the outlet of reactor 2. To support biomass growth, reactor 3 was provided with an additional ammonia load via inlet water, bypassing reactors 1 and 2. Furthermore, this reactor was constantly aerated at a DO level of 4 mg L^{-1} since November 2003. In April 2004, another unscheduled effect was observed in reactor 3. Regardless of the high oxygen concentration, deammonification started also in this reactor. A dense biofilm with high activity and oxygen consumption had developed, comprising both, aerobic and anoxic ammonia oxidizers. Until today, deammonification has been running stably in all three reactors, with intermittent aeration in the first and second and constant aeration in the third reactor.

Costs Table 2 shows the yearly costs for the deammonification plant calculated on the basis of a period of stable operation of several months in 2003. Considering capital and operating costs, specific costs of 3.32 € kg N^{-1} removed can be calculated. Due to the special process layout of the full-scale pilot plant to allow a maximum of flexibility during the research phase, there is still considerable savings potential as regards investment costs. This especially so if existing plant structures, e.g. shut down reactors, can be reintegrated in the layout of the deammonification plant.

Table 2 - Yearly costs for full-scale deammonification at the Hattingen WWTP

	$L_{d,NH4-N,0}$ ŋ N_{eli}	= 103 kgd^{-1} = 72% = 74 kg d^{-1}
	Annual Costs	Specific Costs
	[€ a^{-1}]	[€ kg N_{eli}^{-1}]
Investment costs	52,000	1.93
Maintenance, constructions (0.5%/a)	900	0.03
Maintenance, machinery & electrical equip. (2 %/a)	7,400	0.27
Maintenance, instrumentation & control	2,500	0.09
Personnel (0.25 man year)	12,000	0.44
Power consumption	418 kWh/d	0.50
Sulphuric acid	4.41 €/d	0.06
Total		**3.32 € kg N_{eli}^{-1}**

Outlook Experiences from two years of operation of the full-scale deammonification plant at Hattingen demonstrate that the Kaldnes®-Moving Bed-Process is suitable for a completely autotrophic ammonia removal. To ensure the efficiency of elementary nitritation in the biofilm system, intermittent aeration has been used successfully to inhibit further nitrification. Over the years it was found that the biofilm structure has to be protected from the shear forces caused by intense mixing. A balance is built up between biofilm removal and growth that is influenced by substrate load, oxygen supply, temperature-sensitive growth rate and mixing energy. The understanding of these relationships helped to establish an efficient deammonification process, capable of removing up to 80% of the ammonia load from sludge dewatering at the Hattingen WWTP.

Full-scale deammonification in a SBR (DEMON process at Strass WWTP)

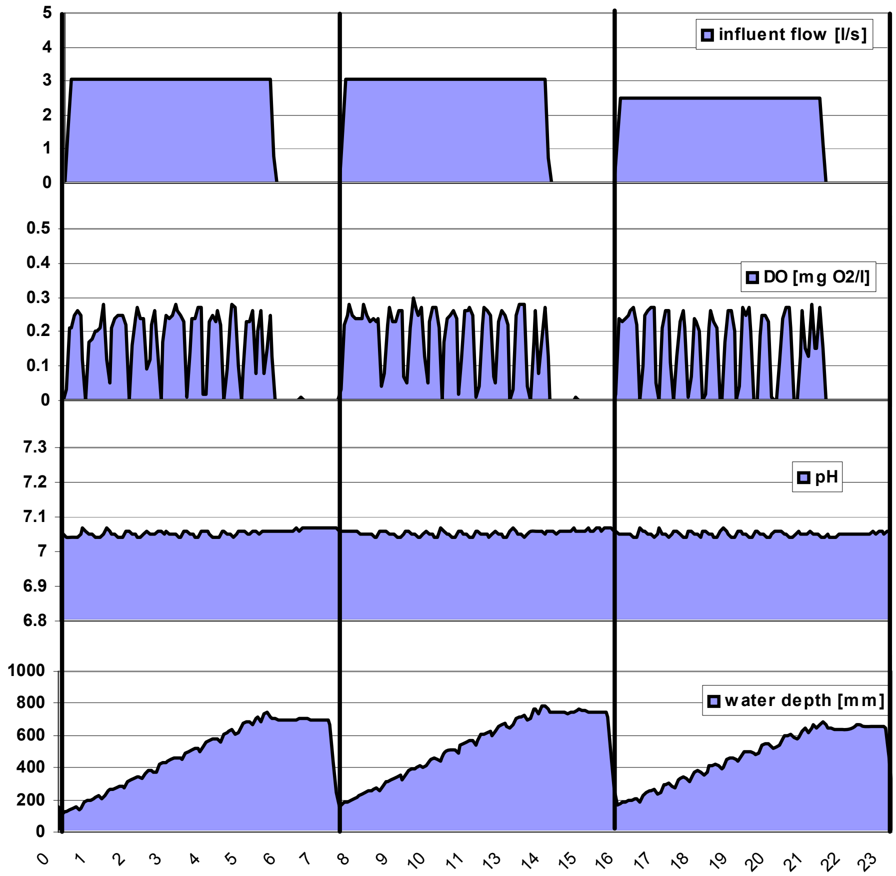
Full-scale plant Since 1996 at the Strass WWTP (200000 pe) in western Austria rejection water produced from dewatering digested sludge is treated separately in a full-scale single sludge system. The process is characterized by a SBR with an intermittent aeration system. Provided process conditions – explicitly low DO values – successfully repress the second oxidation step from nitrite to nitrate despite high sludge retention times of about 30 days.

The aim of a new process development was to decrease the demand for resources without endangering the proven robustness of the treatment system. The main challenge was the shift from a nitritation/denitritation system – that is a system substantially supplied with external organic carbon (primary sludge addition) and enriched with heterotrophic biomass – to a dominantly slowly growing autotrophic biomass. In order to shorten the transition period in the full-scale reactor a stepwise strategy for enrichment of the slowly growing biomass was applied. Starting from 4 litres of inoculum taken from the pilot-plant operated by the EAWAG in Zuerich a 300 l reactor was seeded. Then reactor size was gradually increased at steps of 1 and 2 orders of magnitude to a volume of 2.4 m^3 and finally to 500 m^3. At each upscaling stage biomass enrichment was a vulnerable process until robustness due to a critical mass was achieved which then served as an inoculum for the next reactor. The enrichment period took 2 years and the actual start-up of the full-scale reactor another half a year until the end of 2004 (Wett 2005), when the colour of sludge granules changed from brownish to the characteristic red.

The set-point of dissolved oxygen (DO) control was specified at a low range close to 0.3 mg/l in order to prevent rapid nitrite accumulation and to maintain a continuous repression of the second oxidation step of nitrite to nitrate. During the aeration period of about 6 hours of a SBR cycle (Figure 6) both deammonification processes – partial nitritation and anaerobic ammonia oxidation – are operated. These two processes show converse impacts on the pH-value.

The aeration system is activated only within a very tight pH-control interval of 0.01. Due to oxygen input nitritation runs at a higher rate than anaerobic ammonia oxidation and H^+ production drives the pH-value to the lower set-point and aeration stops. While dissolved oxygen is depleted all the nitrite that has been accumulated during the aeration interval is used for oxidizing ammonia. In the course of this biochemical process some alkalinity recovers and additionally alkaline rejection water is fed continuously to the reactor until the pH-value reaches the upper set-point and aeration is switched on again. The crucial feature of the pH-controlled aeration system is the robust prevention of a built-up of toxic nitrite concentrations which might poison the very sensitive biomass.

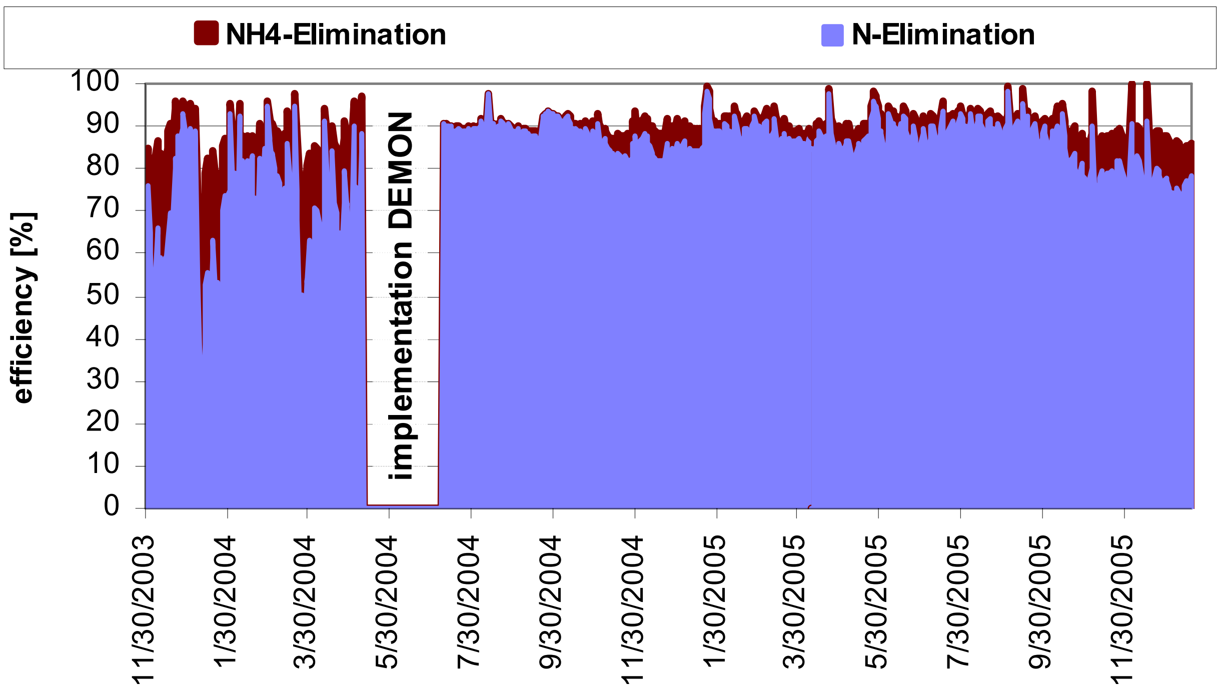
Figure 6 – Monitored on-line profiles of flow rate, DO, pH and water table displaying the control strategy of intermittent aeration within a tight band of pH-setpoints



Results The actual start-up period after seeding the full-scale reactor took half a year from July to end of 2004 when the required treatment capacity of 300 kg of nitrogen was reached. The mean ammonia elimination efficiency calculated from daily measurement values during the year 2005 amounted to 90.3 % ± 2.95 % (Figure 7). The total nitrogen removal rate was only slightly less (85.8 % ± 4.93 %) because the nitrate produced in the process was denitrified by the heterotrophic biomass grown on the organic carbon content of the rejection water.

Due to a doubling time of at least 11 days anaerobic ammonia oxidizers require a sludge retention time of more than 20 days to stay established in such a system. Because of this SRT constraint a biofilm system appears more appropriate to ensure sufficient retention of the biomass. The application of suspended growth in a single sludge system meant a question mark in process development – do settling properties deteriorate or not? Before changing over to the DEMON system primary sludge was added as a carbon source which of course also improved settling characteristics. The sludge volume index SVI was kept well below 100 ml g-1 (Figure 8). When primary sludge dosage was stopped a SVI increase was expected – and occurred. Additionally foaming phenomena occurred during aeration periods depending on aeration intensity and TSS concentration (foaming is favoured at low sludge concentrations). Excessive foaming and SVI around 150 ml g-1 caused a severe loss of solids at the beginning of December 2004. The foaming problem was solved by the installation of a circular pipe along the reactor walls spraying the influent flow well distributed on the foam layer. Finally SVI recovered and the activated sludge produced in the deammonification system shows stable settling properties (SVI=73.6 ± 12.4 ml g-1).

Figure 7 – Elimination efficiency in terms of ammonia nitrogen and total nitrogen after the sidestream process at Strass WWTP as been changed over from a nitritation/denitritation system to a deammonification system



The expected stoichiometric benefit in oxygen demand yields almost half the value when the predominant metabolic route is shifted from nitritation/denitritation to deammonification. The potential for energy saving is even higher when the heterotrophic respiration is considered. An external carbon source for denitritation causes additional oxygen requirements for the heterotrophic biomass. Nitrate produced by deammonification shows a stoichiometric carbon demand of only 10 % of the base value which is usually covered by the organic content of rejection water.

The monitored energy demand shows significant fluctuations between workdays and weekends in correspondence with the operation periods of the sludge dewatering units. After inoculation and start-up of the SBR the specific energy demand decreased from a rather variable range between 2 and 3 kWh per kg N to a stable level of 1.16 kWh per kg N eliminated (including pumping and stirring; no heat requirement).

Figure 8 – Development of settling properties after implementation of deammonification at Strass WWTP and corresponding concentration of total suspended solids

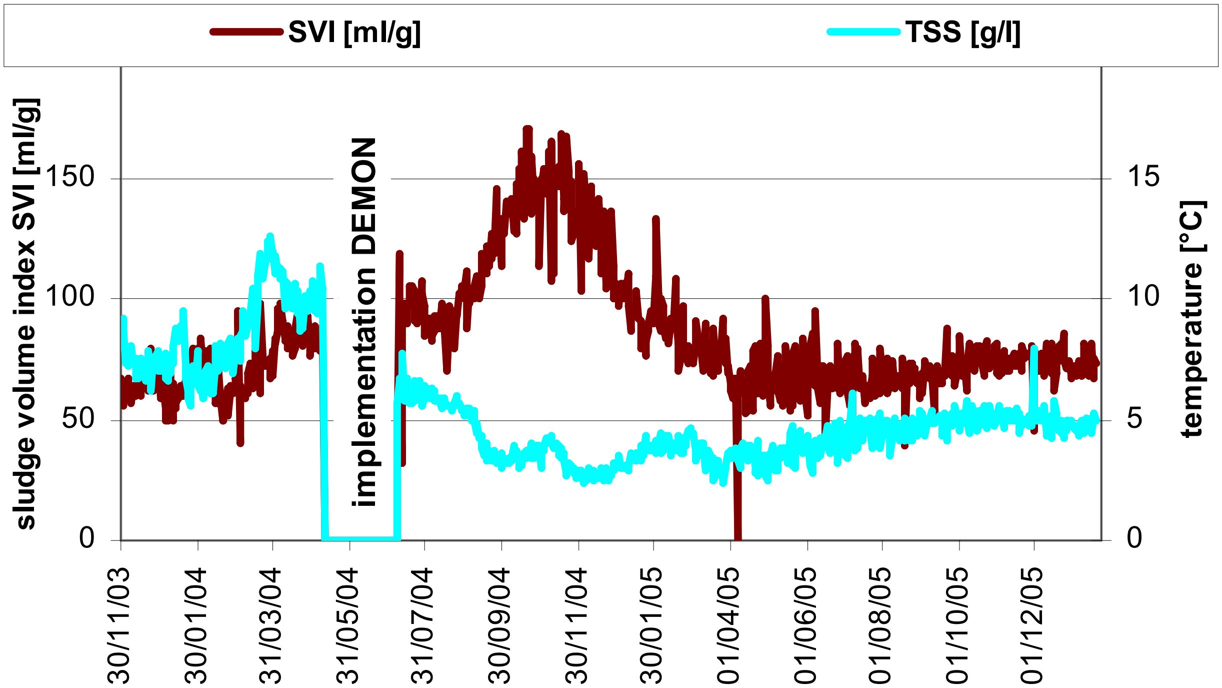


Table 3 – Annual mean values 2005 of main operational variables of the sidestream treatment at Strass WWTP

2005	TSS g m^{-3}	VSS g m^{-3}	SVI ml g^{-1}	temp. ° C	flowrate m^3 d^{-1}	NH_4-removal %	N-removal %
reactor	4.3 ± 0.8	3.0 ± 0.8	73.6 ± 12.4	27.8 ± 1.7	117 ± 39	90.3 ± 2.95	85.8 ± 4.93

	NH_4 g N m^{-3}	NO_2 g N m^{-3}	NO_3 g N m^{-3}	$COD_{soluble}$ g COD m^{-3}	$COD_{particulate}$ g COD m^{-3}		**specif.energy** **[kWh/kg N_{elim}]**
influent	1844 ± 92	0	0	614 ± 27	241 ± 140		1.16 ± 0.21
effluent	179.4 ± 51.7	4.4 ± 6.9	76.8 ± 48.1	344 ± 4	305 ± 61		

Resources Cost calculations are site specific especially when existing resources have been included in the new process scheme. That happened when the initial SBR for nitritation / denitritation was installed in an adapted aeration tank in 1996. No substantial investments were required when the system was switched to deammonification (adaptation of control system, influent distribution and discharge device).

Outlook Presented experiences from the full-scale system in Strass prove the robustness of the deammonification process once it is established. The main trouble of process implementation was the long period of biomass enrichment of 2 years. In future the start-up period can be substantially reduced by taking a sufficient amount of seed sludge from one facility to the other.

Easy transfer of inoculum is an advantage of suspended growth compared to biofilm systems. The key element of the DEMON process is the pH-controlled intermittent aeration system which keeps nitrite concentration below any toxic range.

CONCLUSIONS

Experiences from over two years of operation of the full-scale deammonification plant at the Hattingen WWTP show that the Kaldnes®-Moving Bed-Process is suitable for establishing a completely autotrophic ammonia removal. To ensure a stable and cost-effective nitrogen elimination it is of crucial importance to minimize the shear stress exerted by the mixing devices and to employ an efficient process strategy to inhibit any further nitrification of the nitrite produced, e.g. by using intermittent aeration.

As an alternative to deammonification in a fixed film system, also the SBR process technology can be applied for the purpose. Due to availability of existing reactor volume at the WWTP in Strass, it was possible to implement the process with negligible investment costs. The energy costs of this plant, which are mainly driven up by the energy demand for aeration, mixing and pumping, come to not more than 1.2 kWh kg N_{eli}^{-1}.

With both process technologies a stable nitrogen elimination of 70 to 90 % can be achieved in full-scale operations. For an economic process comparison, the overall cost situation is mostly dominated by the varying individual site-specific conditions.

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REFERENCES

Abeling, U.; Seyfried, C.F. (1992) Anaerobic-aerobic treatment of high-strength ammonium wastewater – nitrogen removal via nitrite. *Water Sci. Technol.,* **26** (5-6), 1007 – 1015.

Anthonisen, A.C.; Loehr, R.C.; Prakasam, T.B.S.; Srinath, E.G. (1976) Inhibition of nitrification by ammonia and nitrous acid. *Journal WPCF*, **48** (5), 835 – 852.

Böhler, M.; Fux, C.; Egli, K., van der Meer, J.R.; Siegrist, H. (2003) Separate Faulwasserbehandlung durch partielle Nitritation und anaerobe Ammoniumoxidation (ANAMMOX) mit einer Kaskade von zwei SBR-Reaktoren. In: Stickstoffrückbelastung – Stand der Technik 2003, Tagungsband zur 5. Aachener Tagung am 7. und 8.1.2003 in Fulda..

DWA-Arbeitsgruppe AK-1.1 (2003) Phosphorrückgewinnung. *KA-Wasser/Abfall*, **50** (6), 805 – 814.

DWA-Arbeitsgruppe AK-1.3 (2000) Rückbelastung aus der Schlammbehandlung. KA-Wasserwirtschaft, Abwasser, Abfall, **47** (8), 1181 – 1187.

EU (1991) Council directive of 21 May 1991 concerning urban waste water treatment. Official Journal L 135, 30.5.1991, 40.

Fux, C.; Lange, K.; Faessler, A.; Huber, P.; Grueniger, B.; Siegrist, H. (2003) Nitrogen removal from digester supernatant via nitrite – SBR or SHARON? *Water Sci. Technol.*, **48** (8), 9 – 18.

German wastewater ordinance (2001) Verordnung über Anforderungen an das Einleiten von Abwasser in Gewässer (Abwasserverordnung – AbwV) vom 20. September 2001 mit Änderungen vom 2. Juli 2002, BGBl. I, S. 2440.

Grömping, M.; Haeske, J.; Kolisch, G. (1998) Separate Teilstrombehandlung von Prozeßwasser der Schlammentwässerung kommunaler Kläranlagen. *Korrespondenz Abwasser*, **45** (12), 2275-2281.

Hellinga, C.; Schellen, A.A.J.C.; Mulder, J.W.; Loosdrecht, M.C.M., Heijnen, J.J. (1998) The Sharon process as an innovative method for nitrogen removal form ammonium-rich waste water. *Water Sci. Technol.*, **27** (9), 135 – 142.

Hippen, A. (2001) Einsatz der Deammonifikation zur Behandlung hoch stickstoffhaltiger Abwässer. Veröffentlichun-gen des Institutes für Siedlungswasserwirtschaft und Abfalltechnik der Universität Hannover, Heft 166.

Jardin, N. (1995) Untersuchungen zum Einfluß der erhöhten biologischen Phosphorelimination auf die Phosphor-dynamik bei der Schlammbehandlung. Schriftenreihe WAR des Instituts für Wasserversorgung, Abwasserbeseitigung und Raumplanung der Technischen Hochschule Darmstadt, Band 87.

Knowles, G.; Downing, A.L.; Barrett, M.J. (1965) Determination of kinetic constants for nitrifying bacteria in mixed culture. *J. Gen. Microbiol.*, **38**, 263 – 278.

Mulder, J.W. ; van Loosdrecht; M.C.M.; Hellinga, C.; van Kempen, R. (2001) Full-scale application of the SHARON process for treatment of rejection water of digested sludge dewatering. *Water Sci. Technol.*, **43** (11), 127 – 134.

Pfennig, A. (1996) Grundlagen der Strippung und der Rektifikation. In: Stickstoffrückbelastung: Stand der Technik 1996/97, Fachbeitrag 24, TK-Verlag Neuruppin.

Rosenwinkel, K.-H.; Cornelius, A., Thöle, D. (2005) Full scale application of the deammonification process for the treatment of sludge water, IWA Specialised Conference on Nutrient Management in Wastewater Treatmemt Processes and Recycle Streams, Krakow, September 19th to 21st 2005, Conference Proceedings, 483 – 491, ISBN 83-921140-1-9.

Thöle, D.; Cornelius, A.; Rosenwinkel, K.-H. (2004) Deammonifikation von Stickstoff in hochbelasteten Abwasserteilströmen mit Hilfe eines Schwebebettverfahrens. 37. Essener Tagung vom 24.3. bis zum 26.3.2004. In: Gewässerschutz-Wasser-Abwasser, Band 193, 16/1 – 16/15.

Ueno, Y.; Fujii, M. (2001) Three years experience of operating and selling recovered struvite from full-scale plant. *Environmental Technology*, **22**, 1373 – 1381.

van de Graaf, A.A.; Mulder, A.; de Bruijn, P.; Robertson, A.; Jetten, M.S.M.; Kuenen, J.G. (1995) Anaerobic oxidation of ammonium is a biologically mediated process. *Appl. Environ. Mircrobiol.*, **61**, 1246 - 1251.

Wett, B. (2005) Solved scaling problems for implementing deammonification of rejection water, IWA Specialised Conference on Nutrient Management in Wastewater Treatmemt Processes and Recycle Streams, Krakow, September 19th to 21st 2005, Conference Proceedings, 389 – 396, ISBN 83-921140-1-9.