Organizing and the maintenance of on-line measurements in a biological P and N removal study

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1 Introduction

In 1995 a project entitled Biological nutrient removal from municipal wastewater was started at the Suomenoja research station to study biological nutrient removal in two parallel activated-sludge treatment pilot plants. The project is being carried out jointly by the Finnish Environment Institute, Helsinki University of Technology and Helsinki University, and consists of three subprojects. The project has 19 financial supporters, including the waterworks of four cities. It is due for completion in December 1997.

2 Process description and measurements

The research was done in two parallel pilot plants, one of which is described by Rantanen and Valve elsewhere in this conference; the other was an identical plant without carriers. Both plants were equipped with several on-line sensors (see Fig. 1) and were also observed with standardized laboratory analyses. Special attention was paid to ascertaining the reliability of the data produced by the on-line sensors.

The data produced by the on-line sensors were collected with a PC-based Onspec Program into monthly files, each measurement in its own file. Every two weeks the instantaneous values with 15-min intervals from these files were extracted into text files including six measurements each. These files were imported into Excel and all data known to be faulty (peaks due to calibration, cleaning etc.) were removed before further calculations.

To ensure appropriate operation, all the analyzers specially purchased for the present project (2 PO_4-P, 1 NO_3-N, 1 MLSS and 1 pH) were required to undergo a 1.5 - 2 month trial period before they were actually paid for. If measuring devices do not function as expected, a considerable proportion of the research time may be spent in establishing faulty measurement data and in repairing the defects in the devices. Some measuring devices may even turn out to be completely unsuitable for use under the conditions prevailing in a wastewater treatment process, as was earlier demonstrated in Finland by Sikow and Pursiainen (1995) in a full-scale nitrogen-removal study.

During the trial period the functioning of each measuring device was controlled by laboratory analyses. The laboratory samples were taken from the same place as the samples for the on-line measurement, for example, from the side of the pH probe or the ultrafiltered sample after a flow-through cell of a nitrate analyzer. The reading of the measuring device was recorded at the sampling time. The process sample was analyzed in the laboratory according to standardized routines and the results compared with the readings of the measuring device.

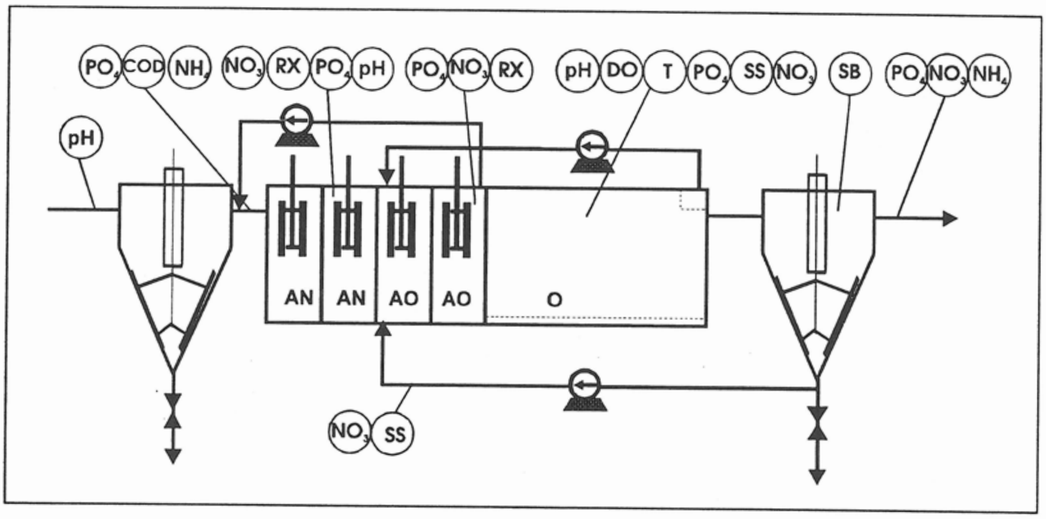

Figure 1. Process scheme of the Suomenoja pilot plants including on-line measurements.

3 Orthophosphate and nitrate analyses from presedimented waters and sludge compartments

3.1 Sampling

Orthophosphate and nitrate analyses from presedimented waters and anaerobic, anoxic, and aerobic sludge compartments from the two pilot plants were performed on eight ultrafiltered samples (Fig. 2). Water or sludge was pumped into an ultrafiltration unit with a submersible pump of 10 $m^3 h^{-1}$ capacity. The yield of the ultrafilter was controlled by choking the valve in the upper end of the filter. The pore size was 20 µm and the openings of the filter outlets were blocked to prevent air entering the anaerobic and anoxic compartments through the ejector effect.

The samples were led via a 6-mm-diameter tube from the filters to two multiplexer units that alternated the samples. For each sample a sidestream was arranged, which was continually running except when the sample in question was taken for analysis. A fresh sample was thus always available. From the multiplexer valves the sample was first led through the nitrate analyzer and then to the orthophosphate analyzer according to the scheme in Figure 2. One 4 - 20-mA signal from each nitrate and orthophosphate analyzer was led to a PC, in which the results of each sample were separated from the other results.

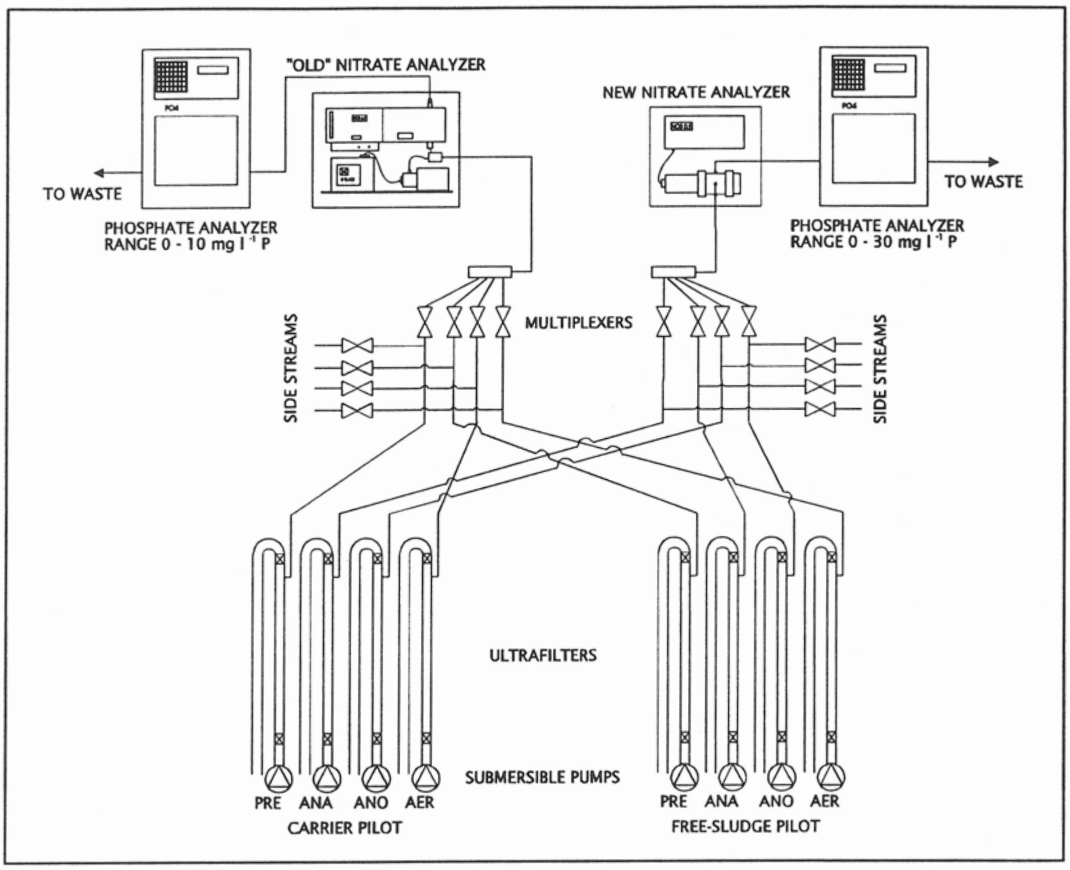

Figure 2. Sampling scheme of the on-line nitrate and orthophosphate analyses from both pilot plants at the Suomenoja research station. Analyses from the presedimented waters and sludge compartments are included.

Operation of the two multiplexer units that came with the orthophosphate analyzers was checked with two alternating sample streams, before the analyzers were checked. The trial period lasted 2 weeks during which the multiplexers operated as required.

During the normal run it was noted that the samples filtered from the presedimented waters and the anaerobic sludge compartments caused more settling of dirt inside the sample tubes and also inside the analyzers. Some samples even clogged valves in the sampling line. Regular cleaning of the sample tubes with 5% hydrochloric acid was found to be necessary. This was done every 2 weeks (see also Chapter 3.3). For comparison, cleaning of the effluent sample tubes was necessary only twice a year. In addition the ultrafilters were brushed 2 - 3 times a week.

3.2 Orthophosphate analyzers

Two orthophosphate analyzers, one for the region of 0 - 10 mg l^{-1} PO_4-P (device 1) and one for 0 - 30 mg l^{-1} (device 2), were put through a trial run. Both used the vanado-molybdophosphoric acid colorimetric method, in which a yellow color forms in the presence of orthophosphate and the absorbance of the color is measured at 400-nm wavelength. The analyzers operated batchwise and were able to do one analysis in 10 min although in practice the analysis period was set to 20 min. After a set of 16 sample analyses, both devices made one blank analysis. Device 1 analyzed the sample without dilution and device 2 diluted the sample in the proportion of 6:40.

Device 1 operated faultlessly. A comparison of the on-line and laboratory analyses is presented in Figure 3. During the trial period the on-line results differed from the laboratory results by a mean of +0.04 ±0.03 mg P l^{-1} (median + 0.04 mg P l^{-1}, standard deviation 0,13). The correlation curve of the laboratory and the continuous analyses is in Fig. 4.

Device 1 was accepted after these results; later results are also shown in Figure 3 (mean difference -0.009±0.04 mg P l^{-1}, median -0.03 mg P l^{-1}, correlation 0.995, n = 21).

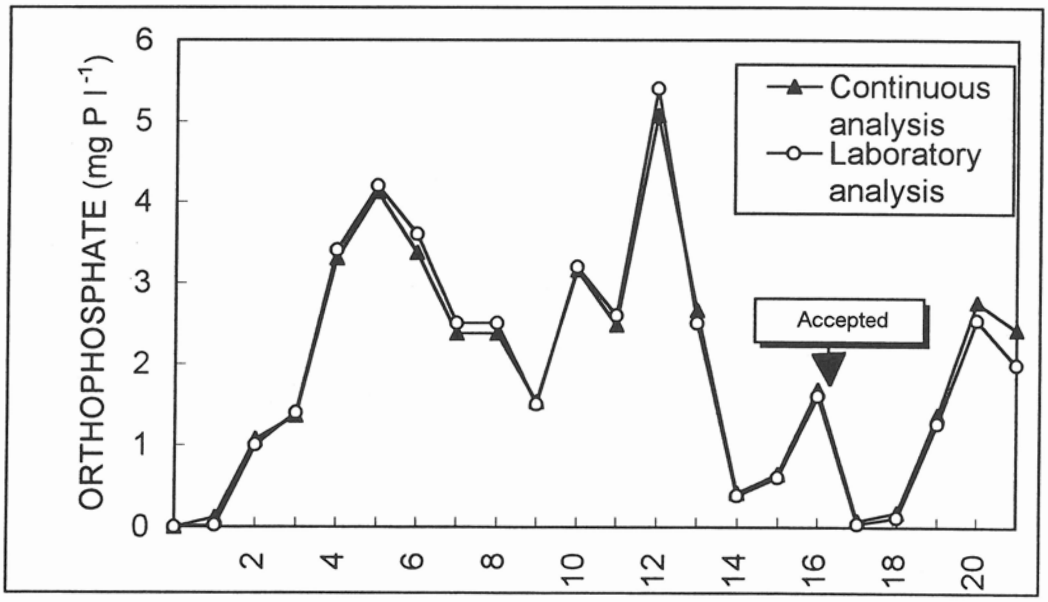

Figure 3. The results the of on-line PO_4 -P analyses (device 1, range 0 - 10 mg P l^{-1}) compared with results of the laboratory analyses.

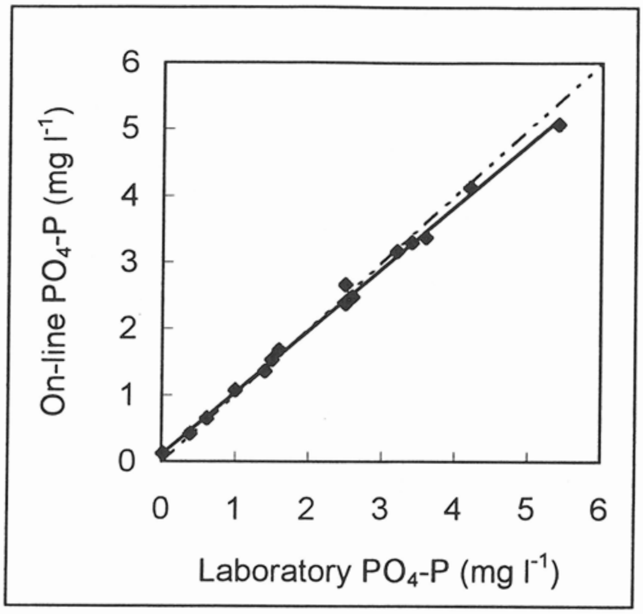

Figure 4. Correlation of the laboratory and on-line PO_4-p analyses (device 1, range 0 - 10 mg P l^{-1}).

Device 2 did not give acceptable results from the start (Fig. 5), as laboratory results differed from the on-line results by a mean of +1.4 mg P l^{-1}. The differences ranged from -0.4 to +5.0 mg P l^{-1} (median +0.95 mg P l^{-1}); even so the correlation coefficient was as high as 0.96 (see Fig. 6). It was noted that the correlation coefficient alone was not a good basis for ascertaining the reliability of on-line analyzers.

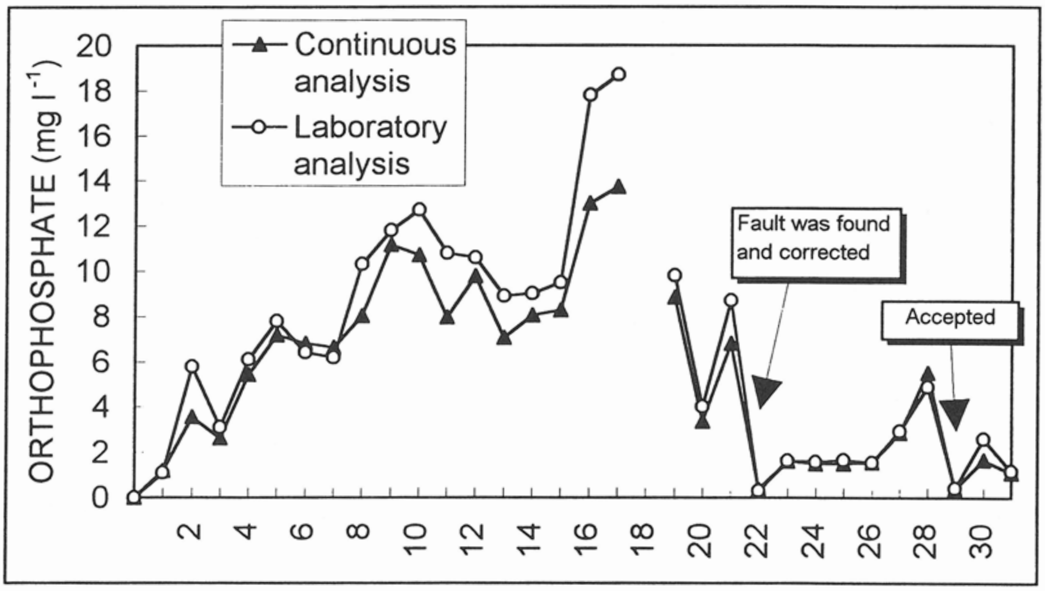

Figure 5. Results of the on-line PO_4 -P analyses (device 2, range 0 - 30 mg P l^{-1}) compared with results of the laboratory analyses.

Analyzer faults were sought by checking the analysis program. A couple of faults originating from factory settings were found, but the device still did not operate reliably, even when these faults had been corrected. Checking was made with a standard solution of 10 mg P l^{-1} and the results varied in the range 3.8 - 10.3 mg P l^{-1}; the variation was therefore quite large.

The fault resulted from air bubbles leaking into the sample tube between the overflow pipette and the reaction vessel. The repair job, raising the end of the tube inside the reaction vessel enough to prevent siphon flow, was extremely easy, although it required an expert to diagnose the fault. The siphon flow had caused a backflow after the tube was emptied and air was sucked into the tube, thus causing error in the next sample volume. Once repaired the analyzer worked flawlessly as shown in Figure 5. The mean difference of the most recent results was +0.07 mg P l^{-1} and the median + 0.06 mg P l^{-1} (n = 10).

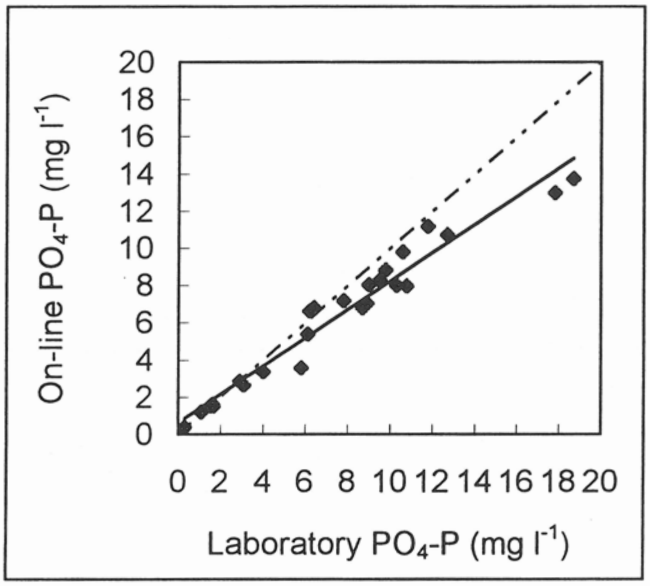

Figure 6. Correlation of the laboratory and on-line PO_4-p analyses (device 2, range 0 - 30 mg P l^{-1}).

3.3 Nitrate analyzer

Before being connected to the system described in Chapter 3.1 and Figure 2 with the orthophosphate analyzers and multiplexers, the nitrate analyzer obtained for the present project was tested alone in an anaerobic compartment with an ultrafiltered sample. The range of the measurement was set at 0 - 10 mg N l^{-1}. The nitrate measurement was a direct spectrophotometric measurement at a UV wavelength. The sample was led into a flow-through cell, inside of which a rubber wiper kept the measuring surfaces clean. There was no possibility of automatic blank-calibration or washing of the flow-through cell. The two nitrate analyzers that had already been used for 3 years at the research station were not tested at this time.

During the trial period the mean difference between the laboratory results and the on-line results was +0.64 mg N l^{-1}, and the range -0.16 - -1.27 mg N l^{-1} and the median -0.58 mg N l^{-1} (Fig. 5). The difference was so systematic that its influence was easily eliminated. The correlation coefficient between the on-line results and the laboratory results was 0.99 (see Fig. 7).

In the light of these results it was decided to approve and purchase the device. Including the later results until the break shown in Figure 5 the mean difference was -0.55 mg N l^{-1} and the median -0.53 mg N l^{-1} (n = 42).

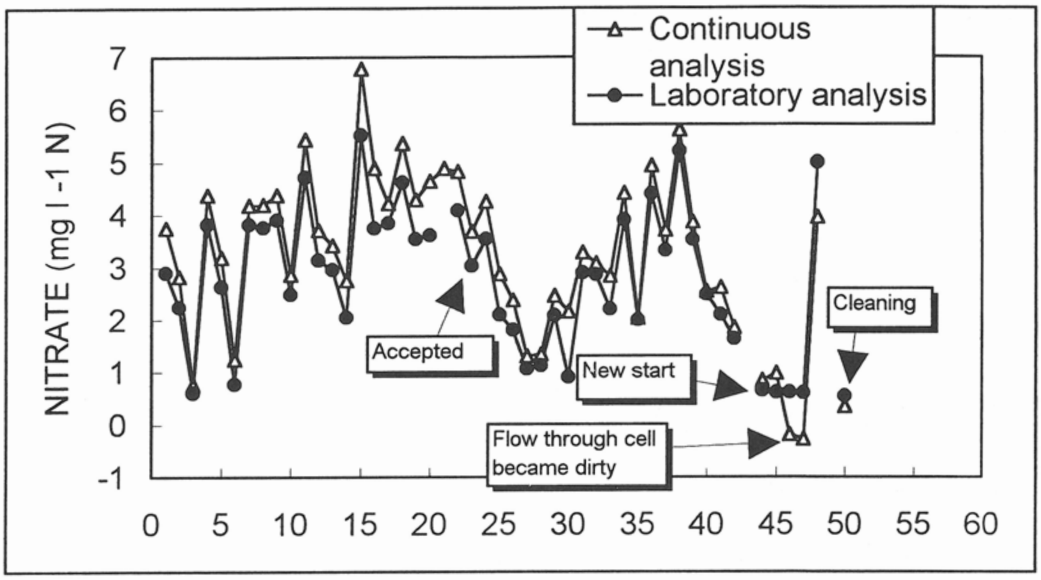

Figure 7. Results of the on-line NO_3-N analyses (range 0 - 10 mg N l^{-1}) compared with results of the laboratory analyses.

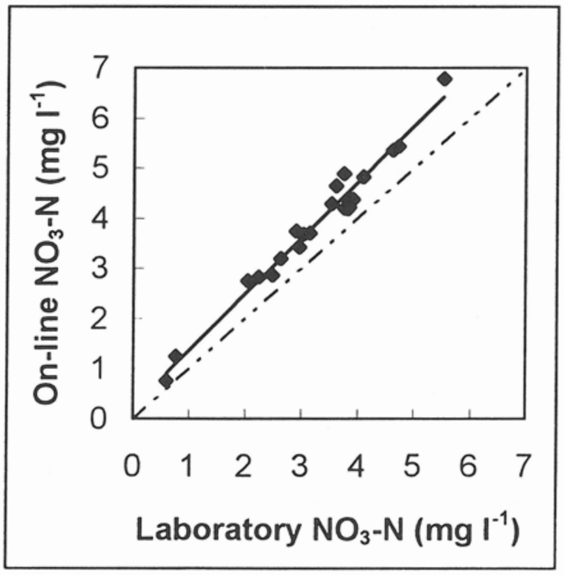

Figure 8. Correlation of the laboratory and on-line NO_3-N analyses (range 0 - 10 mg N l^{-1}).

The device then remained unused for about 2 months. After it had been restarted, it easily became dirty, and the installed rubber wiper was not sufficient to maintain clean measuring surfaces. The flow-through cell was therefore washed with 5% hydrochloric acid. However, since it had to be washed at 2-day intervals, it did not operate reliably during the weekend. A trial period of 2 months was not sufficient to ensure reliability of the device.

It was later noted that the problem was air inside the sample tube. Unlike in Figure 2, the orthophosphate analyzer was first connected in front of the nitrate analyzer, and on the way through the former the sample was in continual contact with air as a result of the overflow pipette. Air was thus led also inside the nitrate analyzer, causing enhanced microbiological growth. The situation was corrected when the sample was first led through the nitrate and then through the orthophosphate analyzer as shown in Figure 2.

4 MLSS analyzer

The suspended solids analyzer was tested in the aeration basin. The measuring probe was installed inside a float situated on one end of a shaft. The range of the measurement was 0 - 10 g l^{-1}.

During the trial period the on-line results differed from the laboratory results by a mean of -0.20 g l^{-1} (-5%) (see Fig. 9). The median was -0.17 g l^{-1} (-3%) and the range -0.89 - +0.61 g l^{-1} (-23 - +11%). The probe had to be cleaned weekly. The correlation coefficient between the laboratory results and the on-line results was 0.92 (see Fig. 10) . The device was approved and purchased.

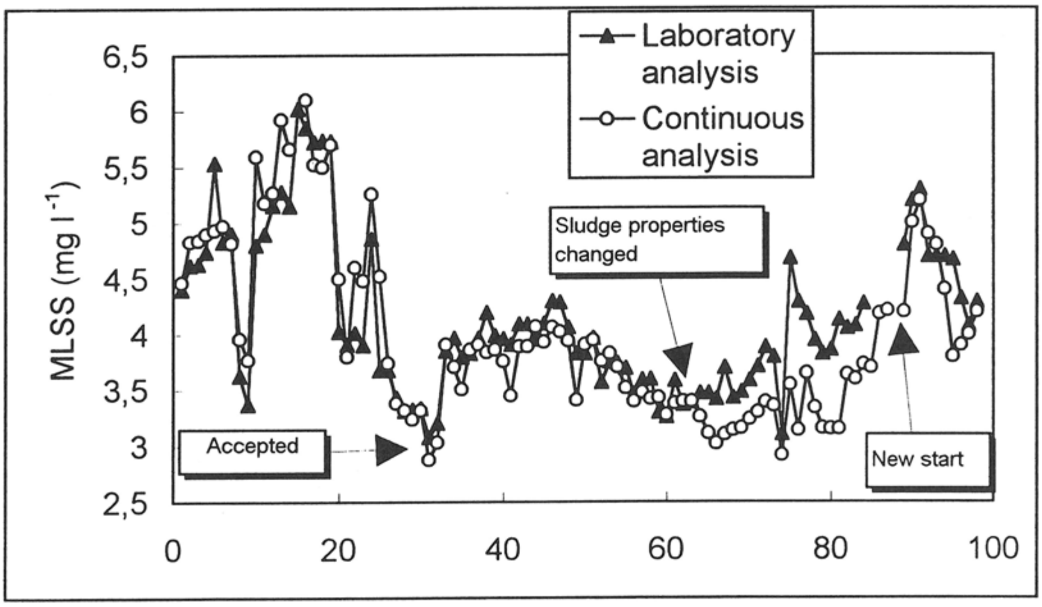

Figure 9. Results of the on-line MLSS analyses (range 0 - 10 g l^{-1}) compared with results of the laboratory analyses.

It was noted that the calibration settings were valid for 2.5 months. Later results are also shown in Fig. 9. For the first 2.5 months, while the calibration was valid, the mean difference of the results was +0.11 mg l^{-1} (-3 %) and the median +0.13 mg l^{-1} (-3 %).

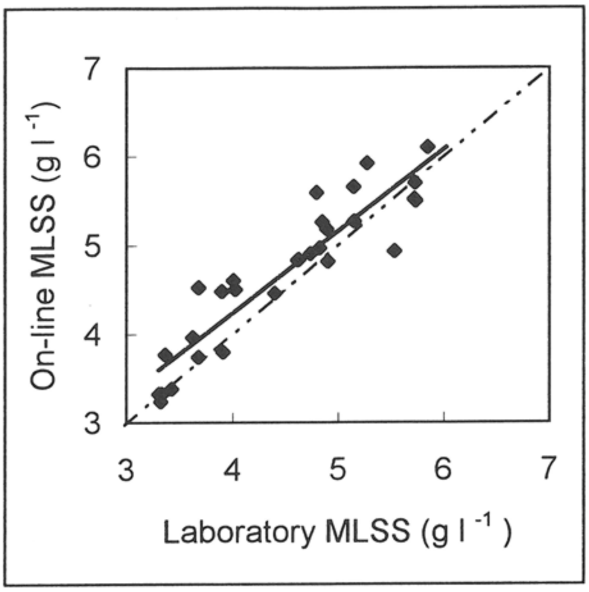

Figure 10. Correlation of the laboratory and on-line MMLSS analyses (range 0 - 10 g l^{-1}).

5 pH meter

The pH meter was tested in the aeration basin. The probe was a gel electrode surrounded by a 5-cm-diameter plastic shield. The gap between the shield and the probe itself was too narrow and a thick layer of sludge collected inside the gap. This resulted in lower on-line results than with laboratory analysis; for example 6 days after cleaning the on-line reading was 5.4 whereas the laboratory result was 6.6 (Fig. 11). The maximum cleaning interval for the probe was 2 days, which meant that the device did not operate reliably during the weekend.

The meter proved to be unsuitable for this purpose and was returned without payment. The project decided to use over 10-year-old, but still reliable, liquid electrodes and not to purchase any new pH meters. During the trial period of the new gel electrode, one of the old liquid electrodes was also immersed in the aeration basin; comparison of the results of the liquid electrode and laboratory results is shown in Figure 7. As can be seen, the liquid electrode functioned flawlessly.

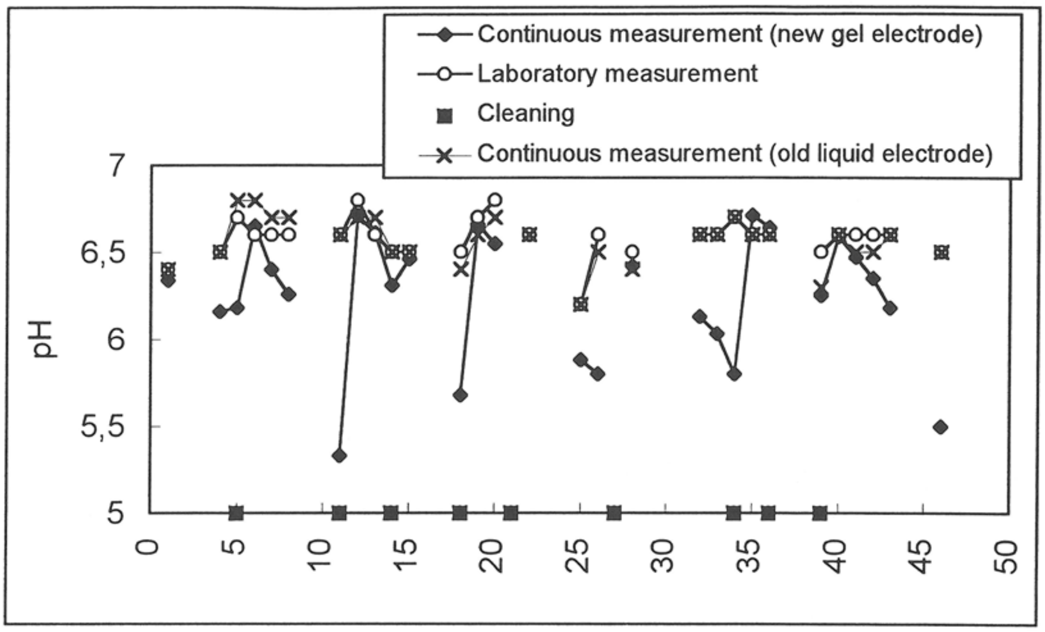

Figure 11. Results of the two on-line pH analyses compared with results of the laboratory analyses. .

6 Conclusions

The orthophosphate, nitrate and MLSS analyzers were found to be reliable; despite some technical difficulties with one of the orthophosphate analyzers and nitrate analyzer. The gel electrode proved unsuitable for pH measurement in the aeration basin. All analyzers require continuous laboratory control with process samples. The sample lines for orthophosphate and nitrate analyzers require cleaning once in 2 weeks with 5% hydrochloric acid.

As a result of these experiences it was decided to ascertain the reliability of all on-line analyzers and measuring devices used in the present project. These devices were not purchased specifically for the present project. The measuring devices included tests for ammonium (3 devices), nitrate (2), dissolved oxygen (4), redox-potential (4), pH (4), temperature (2), sludge blanket (2) and suspended solids (4). The laboratory control of all on-line sensors is to be continued once a week for each sensor until the end of the project.

References

Sikow, M. and Pursiainen, J. 1995, Use and maintenance of on-line measurements at Suomenoja wastewater plant, Sensors in waste water technology, IAWQ specialized conference, Copenhagen 25 to 27 October 1995, Session 3.

Studie av mikrobiologin som en integrerad del i den dagliga driften

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1 Inledning

Tekniken som används vid avloppsvattenrening utvecklas kontinuerligt. I de olika utvecklingssteg som skett hittills har olika kompetens och kunskapsområden behövts. Från början renades avloppsvattnet endast mekanisk genom silning och sedimentering. Utvecklingen av tekniker för avloppsvattnerening bestod då främst av konstruktioner av bassänger samt pumpning av vatten och slam. När kraven på rening ökande infördes biologisk- och kemisk rening. I samband med detta började utvecklingen allt mer handla om processtyrning och kontroll. En mängd mätinstrument började införas i de olika reningsstegen.

Det biologiska reningssteget behandlades till en början som en svart låda. Slammet syresattes och reningssteget styrdes genom mätning av slamhalten eller slamvolymen i systemet. Tidvis kunde slamegenskaperna vara så dåliga så att slammet orsakade slamsvällning vilket medförde att stora mängder slam följde med det renade avloppsvattnet ut.

Kraven på rening har fortsatt att öka och dessutom har fler anläggningar fått krav på kväverening. För att med säkerhet kunna nå dagens och framtidens reningskrav är det nödvändigt att förståelsen för aktivslamsteget ökar och att man vid konstruktion av bassänger, pumpning av vatten, styrning av olika processparametrar även tar hänsyn till hur olika processtrategier påverkar mikroorganismerna som lever och arbetar i slammet.

I denna dokumentation beskrivs hur aktivt slam kan observeras och dokumenteras och några exempel på hur dessa kunskaper kan användas vid drift av ett reningsverk för att undvika driftstörningar som slamsvällning, skumbildning och höga resthalter av slam i det utgående avloppsvattnet.

2 Mikroskopering av aktivt slam

2.1 Mikroskop

För att mikroskopera slammet behövs ett faskontrastmikroskop med 100 -1000 gångers förstoring. Den högsta förstoringen är nödvändig för att kunna bestämma de olika typerna av filamentbildande bakterier. För att kunna få en dokumentation av slammet är det bra om en kamera kan anslutas till mikroskopet. Det går även att koppla en bildskärm med ansluten printer till mikroskopet.

2.2 Provtagning

För att mikroskopera aktivt slam behövs ca 10 ml slam. Det är praktiskt att använda samma prov som tas ut för bestämning av halten suspenderad substans i luftningsbassängerna. För att kunna följa förändringar som sker på anläggningen bör slammet mikroskoperas 1 gång per vecka.

2.3 Parametrar vid mikroskopering av aktivt slam

I tabell 1 redovisas de parametrar som studeras på Bromma reningsverk, Henriksdals reningsverk och Himmerfjärdsverket vid mikroskopering av aktivt slam. Skalan på ett flertal av parametrarna är subjektiv. För att kunna få en rättvis bedömning av slammet krävs därför en viss träning för att lära sig skalan.

Tabell 1. Analysparametrar vid mikroskopering av aktivt slam.

	Parmeter	**Skala***	**Förstoring**
Beskrivning av flock	Storlek	µm	100x
	Form	Regelbunden/Oregelbunden	100x
	Kompakthet	0-4	100x/400x
Bakterieinnehåll	Filamenthalt total	0-6	100x
	Filamenthalt utstick	0-6	100x
	Zoogloea	% av flock	400x
	Fria celler	0-4	400x
Protozooer	Flagellater	0-4	100x
	Amöba	0-4	100x
	Frisimmande cilliater	0-4	100x
	Fastsittande ciliater	0-4	100x

* 0-4; Inga/få/några/många/väldigt många, 0-6; Enligt Jenkins m fl 1993

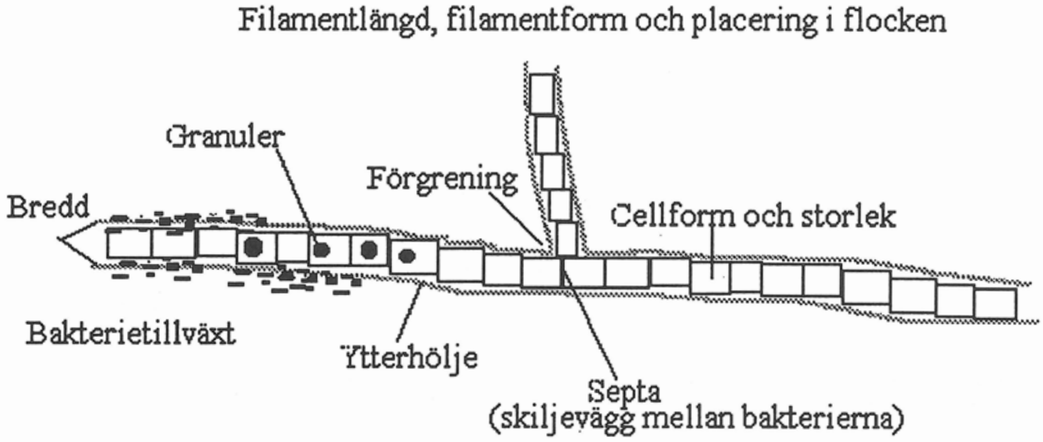
Vid analys av slammet tas en droppe slam (ca 10 µl) och läggs på ett objektglas som sedan täcks med ett täckglas. För att förhindra att slammet torkar ut under täckglaset kan kanterna på täckglaset smörjas in med vaselin. Genom att sedan observera samtliga flockar inom ett synfält på 3-4 platser av objektglaset kan en bedömning av slammet göras. Eftersom slamflockarna kan variera i sammansättning och utseende kan det ibland vara mycket svårt att göra en entydig bedömning av slammet.

De viktigaste parametrarna för att förstå slamsvällning och skumbildning är flockarnas storlek, filamenthalt, typ av filament och andelen zoogleala flockar. Även flockkompaktheten kan ha en avgörande betydelse för slammets sedimenteringshastighet.

En noggranna beskrivning av de olika parametrarna finns beskrivna av Jenkins m fl, (1993) och av Dillner m fl, (1996).

2.4 Identifiering av filamentbildande bakterier

Vid mikroskopering av slammet bedöms filamenthalten i slammet. Vid höga filamenthalter är det även nödvändigt att bestämma vilka typer av filament som dominerar i slammet. Att identifiera olika filamenttyperna kan till en början vara svårt. Efter ganska kort tid är det dock möjligt att känna igen de vanligast förekommande filamenten på sin egna anläggning.



Figur 1. Identifieringsnycklar för filamentbildande bakterier.

De olika filamentbildande bakterierna beskrivs genom att bestämma filamentens längd och placering i flocken samt hur filamenten ser ut (Figur 1). Dessutom bestäms utseendet på de enskilda bakterierna i filamentet i 1000 gångers förstoring. För en slutlig identifiering används infärgningsmetoderna gram- och neisserinfärgning (Jenkins m fl, 1993). Vid infärgning reagerar olika filamenttyper positivt eller negativt. Filamenttypen kan sedan bestämmas genom att jämföra sina egna observationer med hur olika filament beskrivs i litteraturen. Utseende och infärgningsresultat av olika filament finns beskrivna av bl a Jenkins m fl, 1993.

3 Praktisk användning

3.1 Tillväxt av mikroorganismer

För att de resultat som erhålls vid studier av slammet ska kunna användas vid driften av reningsverket behövs en förståelse för vad som påverkar tillväxten av olika slam och olika organismer. För att kunna påverka sammansättningen i slammet krävs därför att alla förutsättningar för det biologiska reningssteget kartläggs. Därefter kan man gå igenom dessa och se vilka som skulle kunna förändras och vilken effekt detta skulle kunna ge. Kunskap om hur olika parmetrar påverkar slammets sammansättning och egenskaper finns bl a sammanställda av Jenkins m fl (1993) och Wanner (1994).

De parametrar som påverkar slammet och mikroorganismernas sammansättning kan delas upp i sammansättningen på avloppsvattnet, alltså hur maten ser ut, och i hur miljön där avloppsvattnet ska brytas ner ser ut. Här kommer en rad processparametrar in som slamålder, slamhalter, syrehalter, inblandningsförhållanden, returslamflöde, uppehållstider i luftningsbassängerna och i sedimenteringsbassängerna mm. För att kunna styra det biologiska reningssteget måste alla dessa kartläggas och dokumenteras. Det är dessutom viktigt att tänka på att avloppsvattnets sammansättning kan förändras i ledningsnätet och genom olika förbehandlingsmetoder på reningsverket.

3.2 Slamsvällning

Vid slamsvällning hinner inte det aktiva slammet sedimentera i sedimenteringsbassängerna utan följer istället med det renade avloppsvattnet ut. Detta uppstår främst i samband med flödesökningar i anläggningar där slammet har dåliga sedimenteringsegenskaper, men sedimenteringsegenskaperna kan även vara så dåliga så att slamflykt även sker vid normalflöden.

Den vanligaste orsak till slamsvällning är att slammet består av höga halter filamentbildande bakterier. Slammet bildar ett stort nätverk som hindra slammet från att packas ihop och sjunka. Vid dessa problem måste den filamentbildande bakterien identifieras. I litteraturen kan man sedan ta reda på vilka faktorer som gynnar eller missgynnar dess tillväxt. Därefter gäller det att skapa en miljö som missgynnar bakterien som orsakar problemen. Första gången detta problem uppstår har ofta den filamentbildande bakterien redan hunnit växa till i ett mycket stort antal. Vid detta tillfälle kan det därför krävas mycket kraftiga åtgärder för att lösa problemet. En möjlighet kan då vara att sänka slamåldern mycket kraftigt till 1-2 dygn och därefter successivt gå upp igen. Den snabba förändringen i slamålder ändrar förutsättningarna i slammet och ökar därigenom möjligheten för de flockbildande bakterierna att konkurrera ut de filamentbildande bakterierna. För att förhindra att samma problem sedan uppstår igen krävs en kontinuerlig bevakning så att bakterien som orsakade problemen aldrig tillåts öka i antal igen. Dosering med klor eller PAX in till aktivslamsteget har också givit resultat vid kraftig tillväxt av vissa filamentbildande bakterier.

En annan orsak till slamsvällning kan vara att vissa av bakterierna producerar stora mängder polymer som förhindrar slammet att packas ihop. Detta problem kallas zoogleal slamsvällning och en orsak till detta är ofta obalans i näringförhållandena.

3.3 Skumbildning

Skumbildning i aktivslamsteget och i rötkamrarna kan orsaka stora säkerhetsproblem samtidigt som de utgör ett sanitetsproblem. Ett antal orsaker och fakta runt detta problem finns sammanställt av Bergman m fl. (1996). Skum orsakad av mikroorganismer är främst kopplat till de trådformiga bakterier

Microthrix parvicella och *Nocardia*. Återigen är det nödvändigt att först ta reda på vilken bakterie som orsakar problemet. Även om skumningen främst sker i rötkamrarna ligger orsaken i att bakterierna har växt till i det biologiska reningssteget. På samma sätt som vid slamsvällning gäller det sedan att hitta orsaken till varför bakterien gynnas och därefter skapa förutsättningar som missgynnar dess tillväxt.

Att mekaniskt slå sönder skummet med toppomrörare i rötkammaren har på flera anläggningar varit en effektiv metod att förhindra skumbildning. För att förhindra skumbildning i biosteget ska ytvattnet i luftningsbassängerna ha en hög flödeshastighet och inte avgränsas med mellanväggar. Utloppet till sedimenteringsbassängen ska ske med överfall.

3.4 Hög resthalt av slam i renat avloppsvatten

Tidvis kan det renade avloppsvattnet vara mycket grumligt och innehålla höga halter suspenderad substans. Detta kan t ex ske om belastningen till det biologiska reningssteget förändrats. På Bromma reningsverk har höga halter amöbor observerats i slammet under dess perioder. Vid mikroskopering kan man observera att slammet består av mycket stora mängder små slamflockar som varierar mellan 10-50 µm. Ofta hjälper det att höja slamåldern i systemet vid dessa tillfällen.

4 Sammanfattning

Att mikroskopera slammet och använda sig av den informationen vid drift av reningsverk ökar kunskapen om de organismer som lever i det biologiska reningssteget samtidigt som driftproblem som slamsvällning, skumbildning och höga slamhalter i det utgående vatten kan motverkas. För att mikroskoperingen ska vara ett bra verktyg är det dock nödvändigt att den utförs av eller sker i samarbete med personal som arbetar nära driften.

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The new sensor development for waste-water treatment plants with nitrogen removal

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Introduction

Overviews of commercial available sensors for waste water treatment plants (Lynggaard-Jensen, 1995) show that many of these sensors (eg. analyzers) require sampling and filtration. Although these sensors/analyzers are improving (Wacheux, 1993, 1996), they still require a good deal of maintenance (Schlegel, 1996; Thomsen, 1996; Londong, 1996; Nyberg, 1996) and the installation costs can be considerable. Furthermore, long response times (10-25 minutes) for these analyzers cause difficulties in incorporating them in standard feed-back process control loops.

Thus the need for new principles for sensors for real-time on-line process control seems obvious, and some of the major demands to the development work has been and are to design sensors, which can be placed directly in activated sludge (no sampling and filtration) and measure within appropriate ranges with a fast response time (less than 5 minutes) and continue to do that with maintenance only once a month.

Several well known measuring principles can be used to comply with these demands if the principles are used together with technology now available (possibilities for fast and big computations, optical standard components and micro-mechanic designs). Some of these principles include:

- Colorimetry combined with semi-micro mechanics and membrane technology
- Absorbance using photodiode-arrays for spectral information
- Fluorescence with multi excitation and scanning for emissions
- Software sensors (or virtual sensors) based on sensors or installed equipment

This paper concentrate on the description of the development of the first of these principles, however, brief descriptions and examples are given for the other 3 mentioned principles.

Colorimetric Sensors

The sensors developed using colorimetric principles include Ammonium, Nitrate and Phosphate sensors. The support technologies used are semi-micro mechanics and membrane technology, defining a concept which now is called µCFA (Micro Continuous Flow Analysis).

The Sensor Design

Basically the principles used are the same for all 3 sensors. Figure 1 shows a diagram of the functions and how these have been built into the prototype construction. Reagents and clean water are stored in bags (1) and transported around the sensor with pumps mounted on a transportation manifold (2). A carrier (clean water) is pumped to the membrane device (3) and flows in a track on one side of the membrane. The other side of the membrane is exposed to the activated sludge/wastewater, and the carrier is enriched with ions passing through the membrane. The enriched carrier is mixed with reagents and pumped to the reaction area (4) and further on to the photometer for detection (5) - both of these are heated to a constant temperature. From the photometer all liquid is pumped to the waste bag (6). All functions are supervised and controlled by a microcomputer (7).

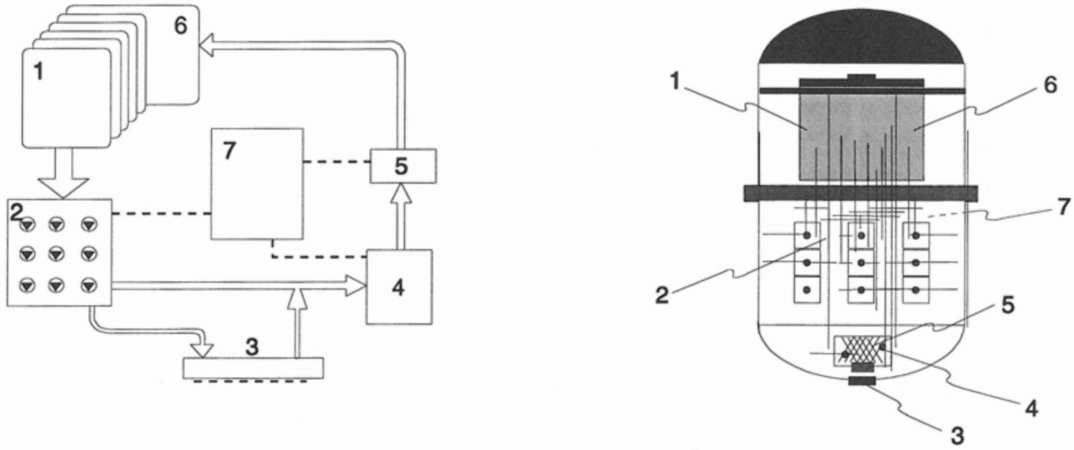

Figure 1: Functionality and construction of the sensors.

Transport manifold: All transportation of liquids inside the sensor takes place in channels with a width of 0.3 to 1 mm milled directly into the manifold material. Pumps delivering 3 µl/turn are integrated with the manifold, and deliver carrier and reagents from the cooled storage bags with a speed of 5 rpm, resulting in a total consumption of liquids within the sensor of less than 3 litres/month. The length of all channels (also on the analysis manifold) have been minimised in order to obtain the smallest possible volume of flowing liquids giving the fastest response time as possible.

Membrane Device: The membrane functions as an ion sieve with in practical no mass transportation of water. This means that the sensors with respect to liquids can be regarded as a closed loop system, with all used liquid collected in the waste bag. As the carrier is flowing continuously on one side of the membrane and the other side is in direct contact with the activated sludge/wastewater , this enables the sensors to deliver a true continuous measurement, without the need for any further sample preparation (pumps, cross flow filters, etc.). The MW cut-off of the membrane is approximately 100 Daltons, and it is approximately 50 μm thick - the same as in many Dissolved Oxygen sensors. The lifetime of the membrane is tested to be well over one month.

Analysis Manifold: The analysis manifold integrates the mixing of carrier and reagents, the colorimetric reactions and the detection by the sensor photometer. Transport inside this manifold is performed as in the transport manifold and together with the functionality of the membrane device this defines a μCFA system. Roughly speaking the milled channels are designed to comply with necessary conditions for the colorimetric reactions to take place and the minimising of the response time. The analysis manifold (together with the wavelength in the photometer) is giving the sensor its identity and 3 different manifolds have been developed for wellknown colorimetric reactions - Indophenol Blue method for Ammonium, Hydrazine reduction followed by formation of Diazo complex for Nitrate and Molybdenum Blue method for Phosphate. However, the high demand to the response time makes it impossible to allow the reactions to come to an end. This problem has been solved by design and construction of a system with a very high repeatability enabling the photometer detection to take place at the exact same time during the development of the detectable colour.

Calibration and measurement: Beside the normal operation mode the sensors are equipped with an auto-calibration system using standard-solutions of the ion to be measured. These standards are stored in bags together with the reagent bags. Furthermore, it is possible to clean the channels inside the analysis manifold - if necessary - with a proper cleaning liquid, which is also stored in a bag together with the reagents. Each bag has its own pump controlled by the microcomputer, which then can operate the sensor in auto-calibration mode or internal cleaning mode (or other modes) according to preset criteria. The auto-calibration has been divided into two steps, calibration of the chemical system (colorimetric reactions inside the analysis manifold) and the determination of the membrane efficiency. Both steps are performed automatically with the sensor placed in the activated sludge/waste water.

For low concentrations of the actual ions the measured transmission follows Beers law, however, Beers law is not applicable for higher concentrations. Therefore the dynamic range is extended by using a calibration involving 3 references and a nonlinear relation between absorbance and concentration at higher concentrations.

$$C^* = \mathrm{Log}(Tr/a)/b \text{ (at low conc.)} \qquad \text{and} \qquad C^* = c/(Tr-d) \text{ (at high conc.)} \qquad (1)$$

where C^* is the resulting concentration measured by the chemical system, Tr is the measured transmission and a, b and c, d are the constants calculated from the measured transmissions with low and medium reference respectively medium and high reference. During the calibration of the chemical system the carrier flow is substituted with the 3 references one by one pumped into the analysis manifold after the membrane device.

As the membrane efficiency, M_{eff}, is lower than 100%, depends on the ion and can variate from membrane to membrane, it is necessary that the sensor automatically can determine the efficiency in order to be able to calculate the correct concentration of the actual ion in the activated sludge/wastewater defined as $C = C^*/M_{eff}$. If standard solution is added to the carrier before the membrane device it can under these measuring conditions be shown (Lynggaard-Jensen (1996)) that:

$$M_{eff} = 1 + [(C^* - C_s^*)/C_s] \qquad (2)$$

where C_s is the concentration of the standard (the actual ion of interest) in the carrier before passage of the membrane device, and C_s^* is the concentration in the carrier after passage of the membrane device.

Signalconditioning: The signal from the photometer undergoes several transformations before it becomes the result of the measurement. The raw analog signal from the photometer is sampled and digitized with a frequency of 10 Hz. This raw digital signal is filtered for peaks caused by noise or passage of air-bubbles in the photometer. As an oscillation in the signal is caused by the pumps, a moving average with a time period of 12 seconds is then calculated (equals one turn of the pumps and 120 sampled values) and used as input to equation (1), which then is divided with the membrane efficiency.

Laboratory Tests

Laboratory tests were designed to give information regarding the sensor performance according to the definitions in EN45001 (1991) of performance characteristics of an instrument. In order to produce calibrationcurves for each sensor standard solutions were produced containing the ions. Each sensor in turn were placed in these for an hour, and the signal logged every 5 seconds giving 720 measurements for each sensor in each standard solution. Experiments with response times were carried out in standard solutions as well, as the sensors were moved rapidly from a blank to a standard solution.

Table 1: Sensor performance according to laboratory tests.

	Unit	**NO_3^- - N**	**NH_4^+ - N**	**PO_4^{3-} - P**
Range	mg/l	0 - 10	0 - 25	0 - 10
Sensitivity	mg/l	0.05	0.05	0.025
Limit of quantification	mg/l	0.05	0.05	0.025
Limit of detection	mg/l	0.075	0.1	0.2
Accuracy*	mg/l	±(0.1 + 10%)	±(0.1 + 5%)	±(0.1 + 5%)
Repeatability*	mg/l	±(0.1 + 5%)	±(0.1 + 3%)	±(0.2 + 3%)
Response time	min.	3	2½	5

*) Percentages is given of the actual measured concentrations.

Table 1 present a summary of the sensors performances according to the well defined terms. The ranges for the 3 sensors are determined from the calibration curves produced. The response times are estimated as the time it takes each sensor to reach 90% of a new concentration caused by a step change.

Tests in Waste Water Treatment Plants

Tests of the sensor prototypes have been performed in a pilot scale recirculation waste water treatment plant loaded with real and/or artificial wastewater and in a full scale biodenitro waste water treatment plant - both with extended nitrogen removal.

Fouling of sensors. Ten of the described membrane devices were placed in two test rigs (5 in each). One of the rigs was then placed in the nitrification tank and the other in the denitrification tank of the pilot scale treatment plant. In a period over half a year laboratory analyses were carried out periodically on the carrier from each membrane device as well as from the two tanks in the pilot plant (nitrate and phosphate in the nitrification tank and ammonium and phosphate in the denitrification tank). At the same time visual inspection of the membranes was carried out and documented with photos. The pilot plant was operated as close to steady state as possible and the feed to the plant in the period consisted of raw wastewater. The mass transfer coefficient for the membranes, has been calculated from the experimental results (Lynggaard-Jensen, 1996) and it was found that the transport of ions through the membranes are practically unaffected during the experimental period. In other words the membrane efficiency is not changed during the period of half a year where the membranes have been placed in the activated sludge. This (rather surprising) result is confirmed by the visual inspection. No fouling of the membranes could be seen during the experimental period although the test rigs themselves showed heavily growth with biomass.

Sensor responses to the treatment processes. The sensors were installed in the aeration tank of the pilot plant and several experiments done over a longer time period with determination of process rates were conducted using the sensor responses and laboratory analyses on grab samples from the tank. The tank has during the experiments with determination of process rates functioned as a batch container, because the inflow of feed has been stopped during the experiments.

All recorded sensor responses are in good agreement with the laboratory analyses, and it can be seen that the sensor response times are very close to follow real time changes in the pilot scale wastewater treatment plant. An example is given in figure 2, which shows that the calibration and the response of the sensors (demonstrated by the ammonium sensor) do not change over several days compared to the la. Analyses (dots), and that the pilot plant normally is operated with a very low concentration of ammonium in the nitrification tank. The figure shows several experiments with adding of ammoniumchloride (first and second peak) or stop/start of the aeration.

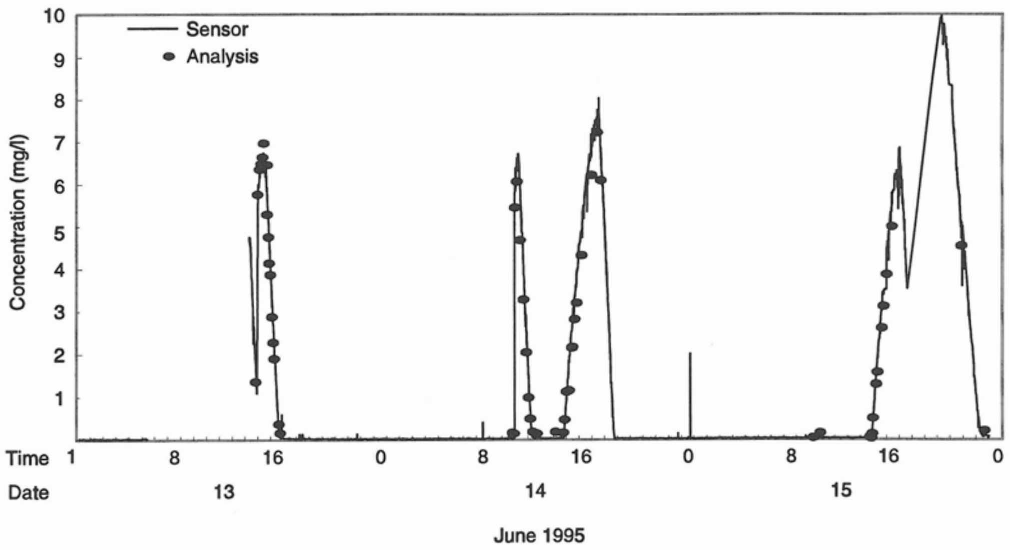

Figure 2: Ammonium measurements in the pilot plant.

Figure 3 shows the ammonium sensor response during 3 days (Friday afternoon to Monday morning) of the test in the full scale treatment plant. Responses are as expected both the the very high load (big slaughterhouse cleaning up for the weekend) and the low load of the weekend. The autocalibration period - which is scheduled to be around midnight in this case - can clearly be seen, and from the pattern of the response it can also be seen that calibration every day might not be needed. Furthermore, it can be seen that the control scheme at the treatment plant easily can be optimised - the aeration period is clearly too long in the weekend and mayby too short at high load.

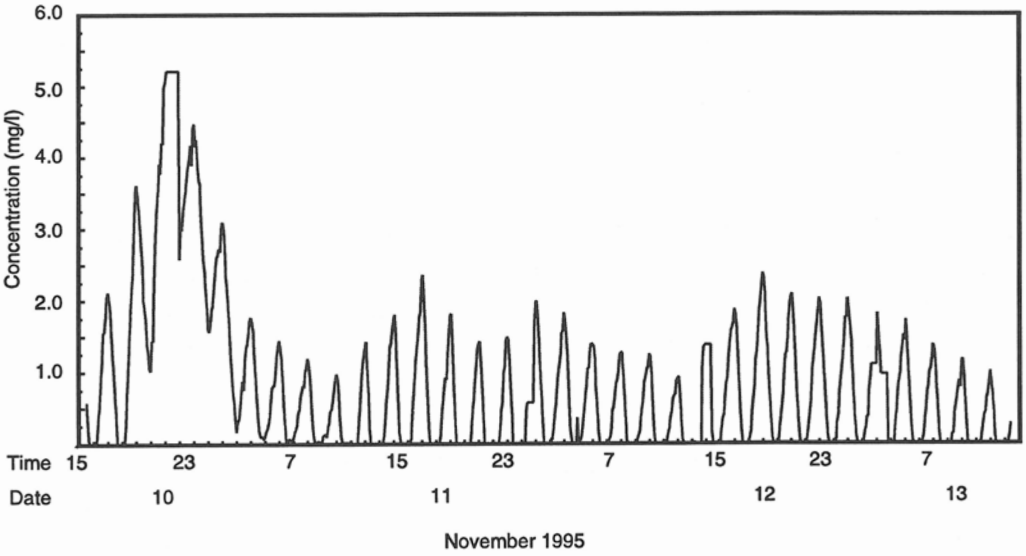

Figure 3: Ammonium measurements in the full scale plant.

Conclusions

The tests of the prototypes of the sensors demonstrates that it is possible to design and construct real-time on-line in-situ closed loop sensors (without the need for a preceding cross flow filtration and without spill of chemicals) with response times less than 5 minutes for ammonium, nitrate and phosphate for wastewater treatment plant applications. Furthermore, the results suggests that a maintenance period of a month is possible due to a very low consumption of chemicals (less than 3 litres/month), an effective auto-calibration system and no fouling of the membranes used. The sensors has now been brought from the prototypes to commercially available sensors and are available on the market.

The µCFA and membrane technology developed for the prototype sensors are believed to have a big potential for further refinement such as extension of dynamic ranges and even faster response times, and it should also be possible to extend the types of ions, which can be measured, without too much difficulty.

It has been demonstrated (Balslev, 1996) that this type of sensors supports the use of set-point control by simple rule based control schemes, and it is believed that there is a great potential for optimisation of treatment processes even with these simple control schemes.

Absorbance Sensors

Whereas the sensors presented in the previous section measures absorbance at a specific wavelength determined by the developed colour, it is also possible to measure the absorbance spectrum of a sample. Light is sent through the sample and the absorbance is measured at different wavelengths. Due to the content of different substances the absorbance will be different at different wavelengths, thus giving a spectrum. The spectrum therefore represents a composite and complex picture of information of the absorbance of all substances which have molecular energy levels which are able to absorb the light used.

These energy levels (eg. wavelengths) are specific for the specific substance, and therefore it is possible to detect a substance using the absorbance at the specific wavelength for this. However, when many different substances are present at the same time, these will somehow cover for each other - interfere with the absorbance of the substance of interest.

Absorbance measurements are usually used in the UV-VIS (Ultraviolet and Visible light) region. The region has a lower limit due to the strong absorbance of water itself below 200 nm. Nitrate is a good example of a strong specific absorbance (205 nm), but unfortunately this is in waste water often hidden in the strong absorbances from organic substances. However, nitrate analysers based on absorbance at 205 nm are available on the market - all of these trying to compensate for the organic substances.

Figure 4 shows absorbance spectra of samples of activated sludge at different times during the cycle of a bio-denitro waste water treatment plant. If the same samples are analysed for different substances (which absorbs in the region shown) it is possible to carry out a multivariate calibration of the spectra to the analysed values. This calibration can then be used with spectra recorded at a later time giving the concentrations of the substances without any need for analyses.

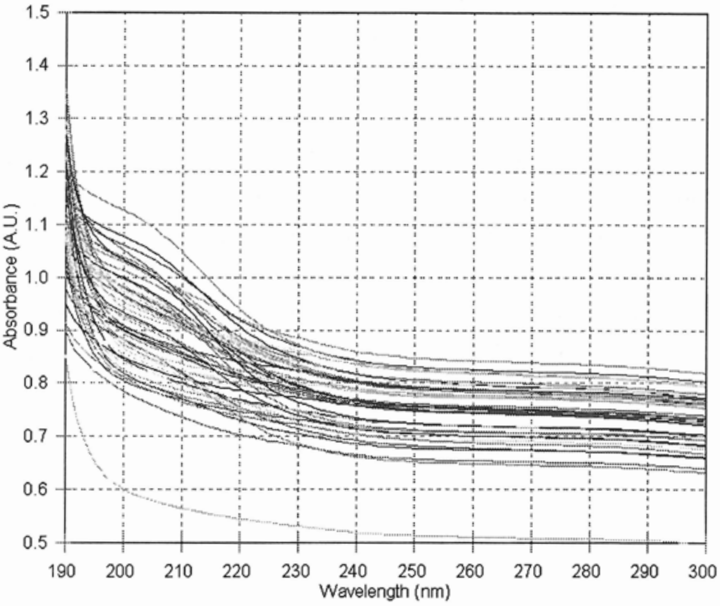

Figure 4: Activated sludge absorbance spectra in the UV-region.

The principle is clearly demonstrated for nitrate by Karlsson (1995), who also is able to measure total-P, total-N, ammonium and iron simultaneously with the nitrate. The methodology used is also known as chemometrics, and together with the development of micro-scale spectrophotometers (diodearrays) it will be possible to develop small integrated multiparameter absorbance sensors.

Fluorescence Sensors

As described above, absorbance is caused by substances absorbing the light - or more correct the energy represented by the light with the wavelength where the absorbance occur. The substance which is now in a higher - and unstable - energetic state normally releases this energy surplus through a series of molecular processes like vibration and rotation or transfer of the energy to other molecules.

However, in some substances some of these pathways of energy release are not allowed, and therefore the energy release happens through emission of light - the energy release "jumps over" the forbidden pathways. This light is the fluorescence and it has a specific wavelength (corresponding to the energy in the "jump") for a given substance. The fluorescence will always have a longer wavelength than the light absorbed, because some af the energy release always will happen through vibration and rotation.

The oldest and best known sensors based on fluorescence are fluorimeters for chlorophyll and oil respectively. Most of these are single channel fluorimeters, meaning that a specific wavelength of light is used for excitation of the molecules and the resulting fluorescence is measured at one specific and somewhat longer wavelength. The advantages of fluorescence to absorbance are the more specific measurement and lower detection limits.

A typical design of a single channel fluorimeter is described in Nørgaard,1996 and shown in figure 5. This fluorimeter measures NADH, which is an important proton transporter for the processes inside the bacteria. NADH fluorescence is measured at 460 nm (excitation at 340 nm). Because the corresponding oxidised state, NAD$^+$, is non-fluorescent at 460 (does not absorp at 340), it is possible to measure biological activity expressed by the concentration of NADH, a fact that has been well known within fermentation industry for several years.

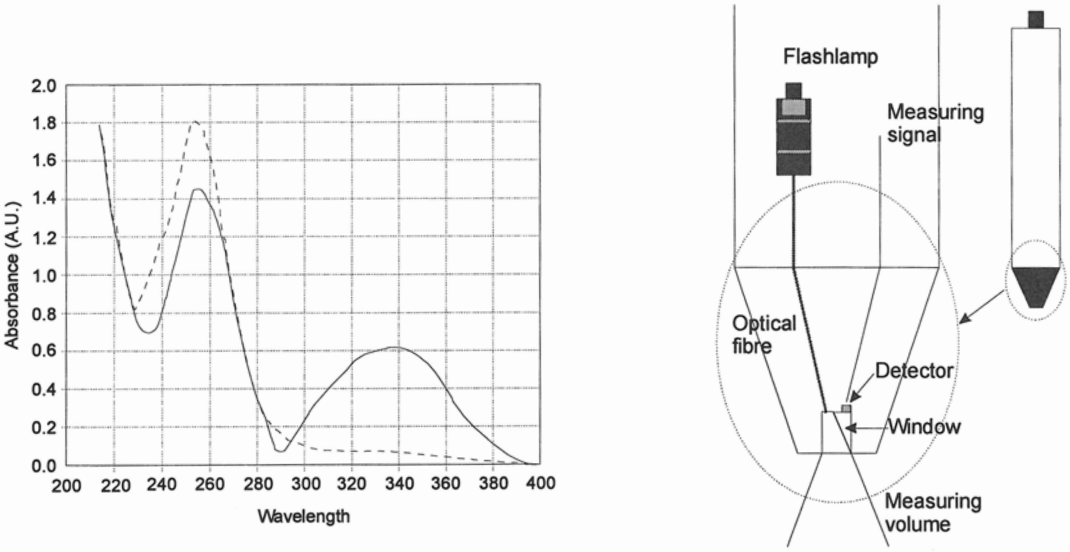

Figure 5: Absorbance (NAD$^+$and NADH (full line)) and sensor, Nørgaard, 1996

Isaacs (1994) shows that the NADH fluorescence can be used to detect the end of the denitrification in an alternating activated sludge plant and also reports some correlation between fluorimeter output and the peak oxygen uptake rate. These results are confirmed by Nørgaard (1996), who also describe a theory for the fluorimeter output and how this can be used for a process control strategy for nitrogen removal.

As with absorbance also fluorescence can be measured as a spectrum. This spectrum is 3-dimensional as the fluorescence is a function of both the wavelengths of the excitation and the emission. Multichannel fluorimeters for detection of different PAHs have been developed, but very little has been done on waste water treatment plants on measurement of real spectras combined with chemometrics.

Software Sensors

Based on reliable available sensors in combination with other signals like on/off indications and timecounters which also are available in the SCADA systems, it is possible to design real-time calculations giving important process information to the operator of the treatment plant. The nature of the "signals" from these calculations are the same as for real sensors, which has caused the term virtual sensors or software sensors to be used for these.

It is also possible to design different types of filters to be used in real time in order to avoid outliers in the signals from the real sensors or to design different types of real time quality control/sensor validation procedures on the real sensors. These software sensors are in fact sensors on the sensors, which are used to follow if the real sensor information is reliable.

Below is given an example of software sensors, which - although very simple - gives important information regarding the process in a pilotscale waste water treatment plant run in biodenitro mode. The real measurements used are the dissolved oxygen concentration, the temperature and the airflow.

The rate of change of the dissolved oxygen concentration, $d(C_{Ox})/dt$, can be calculated from the oxygen concentration measurements, C_{Ox}, done at time n and n-1 (in this case is used a logging interval of 2 minutes)

$$d(C_{Ox})/dt = (C_{Ox,n} - C_{Ox,n-1})/(t_n - t_{n-1}) \tag{3}$$

Furthermore, the rate of change of the dissolved oxygen concentration is given by the normally used expression for respiration:

$$d(C_{Ox})/dt = K_L a(C_{Ox,m} - C_{Ox}) - OUR \tag{4}$$

considering that the oxygen transfer coefficient, K_La, can be written as a new constant multiplied with the airflow, Q_{air}, to the aeration tank and that the oxygen saturation concentration, $C_{Ox,m}$, can be calculated as:

$$C_{Ox,m} = C_{Ox,20}\ 1.02^{(20-T)} \tag{5}$$

the oxygen uptake rate, OUR, can be written as:

$$OUR = K \cdot Q_{air}(\ 9.07 \cdot 1.02^{(20-T)} - C_{Ox}\ ) - d(C_{Ox})/dt \tag{6}$$

K is determined by setting $Q_{air} = 0$, ($OUR = d(C_{Ox})/dt$), and following this by trial and error setting a Q_{air} giving a constant oxygen concentration, ($d(C_{Ox})/dt = 0$). Hereafter OUR can be calculated from measurements done by the oxygen sensor, the temperature sensor and the airflowmeter.

The results from the OUR sensor are shown in figure 6 together with results of the $d(C_{Ox})/dt$ sensor, the oxygen sensor and airflowmeter. However, as can be seen from the figure the best results are obtained from the $d(C_{Ox})/dt$ sensor, whereas the OUR sensor gives a rather oscillating signal - especially the unexpected peaks in the start and end of the aeration period. This is mainly caused by the different time responses of the oxygen sensor and the airflowmeter, most clearly expressed by the peak in the OUR calculation at the end of the aeration period.

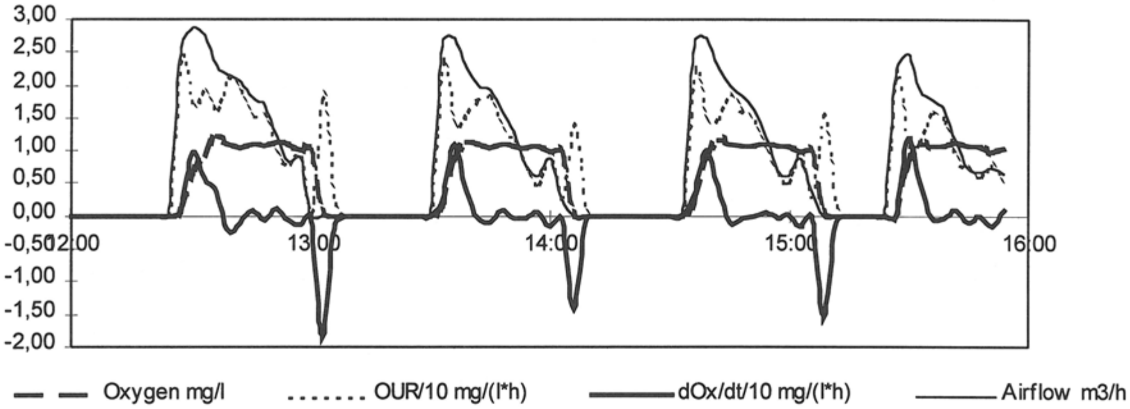

Figure 6: On-line real time calculation of the rate of change of the oxygen concentration and the oxygen uptake rate

However, two different levels in the OUR are expected - with and without nitrification. After the first peak the first level including nitrification can be estimated to approximately 20 mg/l/h and after about 2/3 of the aeration period the second level without nitrification can be estimated to 7 - 8 mg/l/h. As the oxygen concentration is nearly constant during the aeration period the OUR corresponds to the respiration, and the rather low values correspond to the low load of the treatment plant in the actual period.

The $d(C_{Ox})/dt$ sensor represents the 1st derivative of the oxygen concentration signal and therefore the peak value at the end of the period should give the slope of the concentration signal. When aeration is off the slope is equal to the respiration, so a value of 7 - 8 mg/l/h would have been expected in stead of the approximately 15 mg/l/h.

Despite the very typical problem of adjusting response times of real sensors before using the results in software sensors, it is believed that with more different types of real sensors and more computing power available, software sensors will become more and more important and eventually these will develop into real time models based on statistical principles.

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Monitoring of Activated Sludge Settleability and Characteristics

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Introduction

Sludge settleability is an important factor affecting designs and capital costs of municipal wastewater treatment plants. Sizing of secondary clarifiers depends largely on settleability of activated sludge. Well settling sludge requires small tank volumes resulting in good reductions. On the other hand, most of operational problems at wastewater treatment plants are related to settler operation and sludge settleability. By maintaining well settling sludge many operational problems could be avoided. Early and accurate detection of sludge settleability problems in daily monitoring allows operators of wastewater treatment plants to take measures to avoid and reduce operational problems.

Monitoring provides not only means for detecting operational problems but also a tool for optimizing the process. Both optimization and control (as a final goal) require reliable information about the treatment process that can be collected by monitoring. In addition, long-term monitoring can also be utilized for finding the interactions in the process and for understanding the process, i.e. a basis for the optimization and control algorithms. For example, when factors affecting sludge settleability are not very precisely known, monitoring can then be used in discerning the correlations.

At present, monitoring is under active research and quickly developing in the field of engineering. However, many unsolved problems still exist. The answers to the questions what and how we are going to monitor, are still open. For instance, when assessing settleability, do we use SVI, DSVI, ZSV or something else. Besides, a common problem in monitoring wastewater treatment process is scarcity of reliable on-line measurement instruments. Manual monitoring is expensive and continuous monitoring can hardly be maintained. In such a case, important data may not be collected.

The various settleability indexes are mechanical ones and they do not bear information about the nature and chemical characteristics of the flocs. This type of information is needed, i.e. sludge composition and surface properties, and amount and type of filamentous bacteria, for describing the factors affecting changes in settleability.

In this study the sludge settleability is monitored on a higher (on-line) level and new characterization methods are used for identifying the interactions. Some background for the selection of methods used here is presented.

Determination methods

Settleability

Settling properties of activated sludge are traditionally assessed by sludge volume index (SVI). Despite its popularity SVI has been criticized already for decades (e.g. Dick and Vesilind 1969). For example, SVI depends largely on initial sludge concentration, and two differently settling sludges might have identical SVIs (Figure 1.) In addition, SVI has been criticized for its limitations in comparison to settleability of sludges from different plants. Due to these constraints, several alternative methods for settleability description have been suggested and investigated (e.g. Dick and Vesilind 1969; Fitch and Kos 1976: Wilson 1983; Catunda and van Haandel 1992).

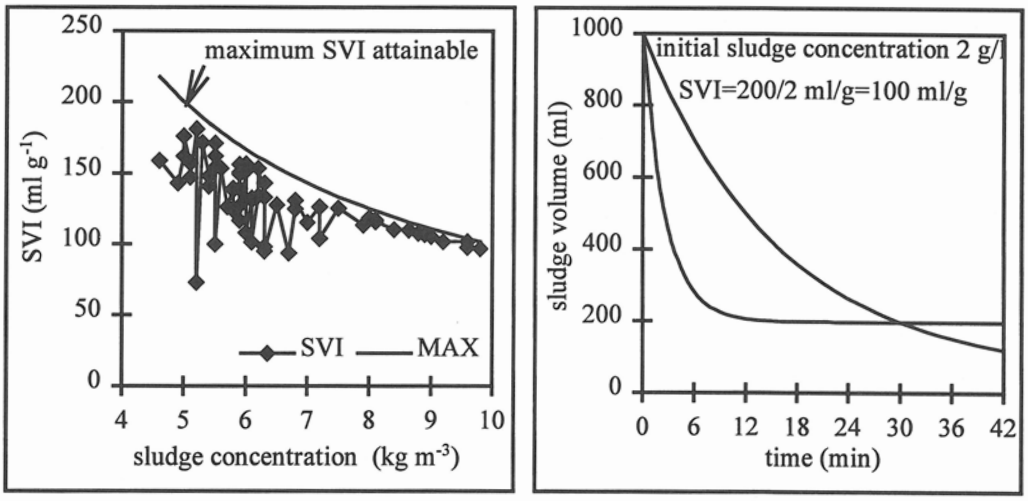

Figure 1. Shortcomings of sludge volume index: SVI depends on initial sludge concentration (Kiuru et al. 1996) and two sludges with different settling characteristics have identical SVI values.

One of the experimental methods used for describing sludge settling instead of SVI is diluted sludge volume index (DSVI). In determining DSVI the sludge sample is diluted in order to have a sludge volume which is after 30 minutes of settling less than 250 ml/l. Because the effect of the initial sludge concentration is reduced DSVI has been considered more accurate than SVI in describing sludge settling properties. However, DSVI is not always as superior to SVI as it has usually been thought of. Figure 2 shows the result of the research by Kiuru et al. (1996) where DSVI is plotted as a function of SVI. The Figure shows that SVI and DSVI give practically equal results as an indicator of variation even if SVI was systematically bigger than DSVI. It is easy to interpret and understand this result since both the indexes basically describe the same property of sludge: sludge volume after 30 minutes of settling.

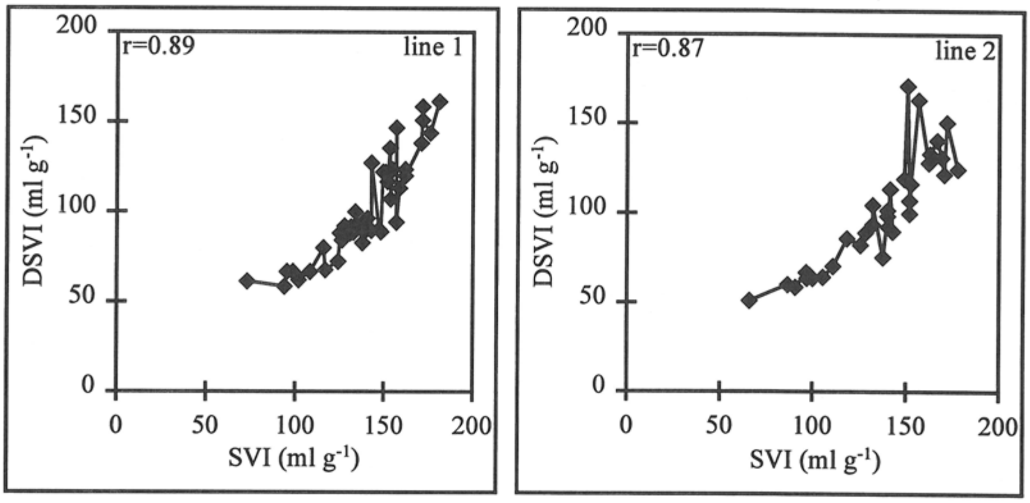

Figure 2. Correlation between SVI and DSVI with the solids concentration under 7 kg m^{-3} (Kiuru et al. 1996).

One well known method used for describing sludge settling is zone settling velocity (ZSV) which is not much used in daily monitoring of treatment plants and it has been restricted to research circles (Ozinsky and Ekama 1995). Measuring for ZSV is laborious and probably for this reason it has not become a standard method at wastewater treatment plants.

On the other hand, SVI and ZSV can be criticized because they interpret sludge settleability as one parameter property, and they ignore parts of the settling curve. SVI (DSVI) takes into account only one point, t=30 min, and ZSV only in beginning of the settling curve.

One possibility to avoid the shortcomings of SVI and ZSV is to use models which take into account the whole settling curve including zone settling, transition zone and compression zone. One such is a model for batch settling curve proposed by Renko (1996). This model interprets the settling as a time dependent phenomenon and takes into account the settling curve as a whole. The model can be written as

$$h_t = \frac{CXh_0}{\alpha} + (h_0 - \frac{CXh_0}{\alpha})e^{-t\alpha/(Xh_0)} \tag{1}$$

where

t is time (h),

h_t is the sludge blanket interface level at time t (m),

h_0 is the initial sludge blanket interface level (m),

X is activated sludge concentration, (kg m^{-3}) and

α and C are the parameters describing sludge settling having the units of kg m^{-2} h^{-1} and m h^{-1}, respectively.

When the model is understood as a procedure to compress information its parameters are indexes for describing the phenomenon. Thus, parameters α and C can be used as indexes characterizing settleability. High value of α indicates well settling and low value poorly settling sludge. Respectively, the high value of C indicates poorly settling and the low value well settling sludge. Since the parameters correlate considerably and the parameter α has stronger influence in the model, it can also be used alone as an index describing settleability. Then, the limitations of this approximation could be understood: probably, sludge settleability is not one parameter property, and cannot be described accurately with one parameter. Thus, for the purposes of accurate settleability analysis, both parameters should be applied. For more detailed description and discussion about the model see Renko (1996), and Renko and Sirviö (1997).

In this study the above-mentioned model is used for assessing sludge settleability.

Characteristics

Sludge exocellular polymers have an important role in floc formation. Eriksson and Alm (1991) showed that charged polymers are important for the properties of sludge surfaces and building up the sludge macroflocs. Polymers interact also by hydrophobic bonding, in which has been paid quite little attention (Urbain et al 1993). Presence of hydrophobic bacteria (or their surfaces) provides conditions for good floc sedimentation (Jorand et al 1994). However, excess hydrophobic properties can cause foaming (Wanner 1994).

Sludge composition and exocellular polymers have been analysed with many different methods. In a recent study, in which various methods were compared, the sludges from two plants (nitrifying/ biological N- and P- removal) consisted of mainly proteins, approx. 50 % of VS (Frolund et al 1996). Also in extracellular polymers extracted from sludge (Frolund et al 1996) or from two activated sludge bacterial strains (Jorand et al 1994) the proteins were the major constituent. In this study a different method was used to evaluate the role of proteins, namely protease sensitivity test, which has earlier been used in pure culture studies (Hantula and Bamford 1991).

The hydrophobic properties of activated sludge were tested using a hydrocarbon adhesion test modified from Jorand et al (1994) in a following way. The sludge was diluted with distilled water to give an absorbance of approx. 0.300 at 600 nm. 1 ml of n-octane (Aldrich 0-325-7) was added to 4 ml of diluted sludge in a glass tube of 10 mm diameter. The suspension was mixed vigorously for 2 minutes with a vibratory agitator (Heidolph Reax 2000) to achieve a certain degree if disruption of flocs and contact of the two phases. After 5 minutes of sedimentation supernatant was sampled with a Pasteur pipette and absorbance at 600 nm was measured (Abs1). A reference test was made without hydrocarbon addition and Abs2-value measured. The more hydrophobic the sludge is, the stronger adhesion of flocs to the hydrocarbon phase will happen. A hydrophobicity index was calculated as: HI = (1- Abs1/Abs2)*100. The test result is evaluated as *strongly hydrophobic, rather hydrophobic* or *less hydrophobic*.

The protease sensitivity test was made in a small glass tube (as the hydrophobicity test) by adding different amounts of protease (proteinase K, Boeringer, Mannheim) to the diluted sludge. The sludge was mixed periodically (different enzyme/sludge contact time periods were used) and allowed to settle for 5 minutes, and a supernatant sample was taken and absorbance at 600 nm (Abs3) was measured. A reference test was made without enzyme addition (Abs4 measured). Deflocculation can be seen as an increasing supernatant turbidity as a function of enzyme concentration and contact time. The protease sensitivity index was calculated as PS = Abs3 - Abs 4, in which Abs3 represents the maximum absorbance obtained with the enzyme concentration and contact time range used.

The amount of filaments was examined with microscope and calculated as an index indicating filament length per gram suspended solids.

Batch settling tests were used to study the impact of surface charge of polyelectrolytes and a model hydrophobic compound (surfactant) on flocculation efficiency. Tests were made in 1 L graduated cylinder by diluting the sludge with effluent wastewater to give a sludge volume approx. 200 ml after 30 minutes of settling. A series of batches were prepared, and two different polyelectrolytes (cationic and nonionic ones consisting of at least 80 % of polyacrylamide with the same molecular weight) were added in the beginning of the test (concentration 1 mg/l). In one batch nonionic fatty alcohol ethoxylate (C_{13} - C_{15} + in average 7 ethoxylate groups) was used in concentration of 10 mg/l. A reference batch without additions was also made. After 10 minutes since the start a small supernatant sample was taken the density of which was measured with a spectrophotometer (600 nm).

Experimental

The experimental data were collected at the Suomenoja reseach station between 3rd October and 31st December 1996. The wastewater was treated in a technical scale plant with UCT -type activated sludge process (Figure 3) (for further information see Rantanen and Valve 1997).

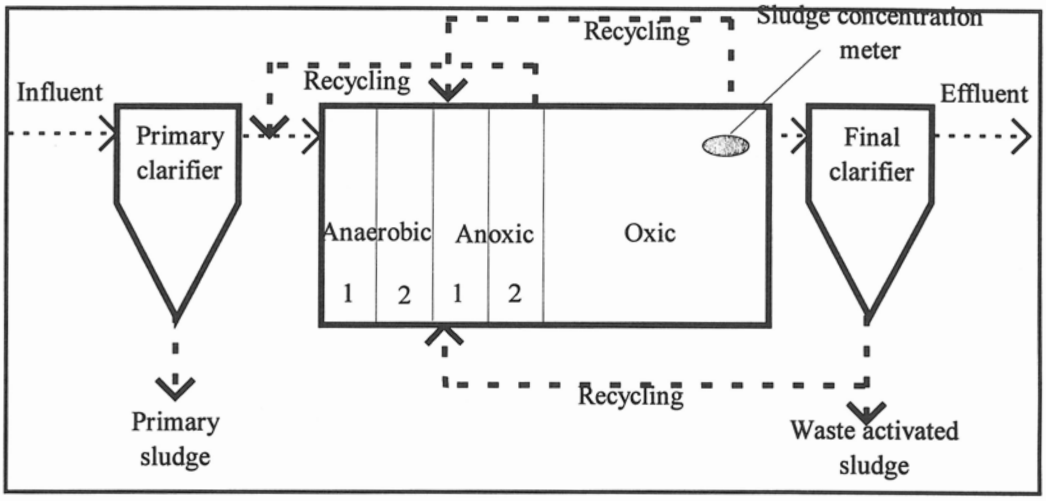

Figure 3. UCT -type wastewater treatment process at Suomenoja research station.

The data needed for the parameter estimation are activated sludge concentration in the aerobic unit and sludge blanket interface level as a function of time (see Equation 1). Activated sludge concentration was measured with an on-line sludge concentration sensor. Sludge blanket interface settling was measured in a transparent cylinder with a sludge depth meter at 30 mm intervals. The inner diameter of the test cylinder was 13 cm and the initial sludge level 145 cm. The automatic system operated during three-hour periods, and the entire procedure consisted of the following stages:

1. Filling the test column with the activated sludge sample from the biological unit into the test cylinder.
2. Measuring. The observation data from the sludge interface level meter was registered at six-minute intervals and sludge concentration meter was registered in the beginning of every test-period.
3. Emptying. At the end of the test period, the sludge in the test cylinder was emptied.

Some parts of protease sensitivity and hydrophobicity tests were made in this plant with sludge from a parallel process with carrier material in the aerobic zone. The surveys of sludge hydrophobicity in other plants were made in 1995-6.

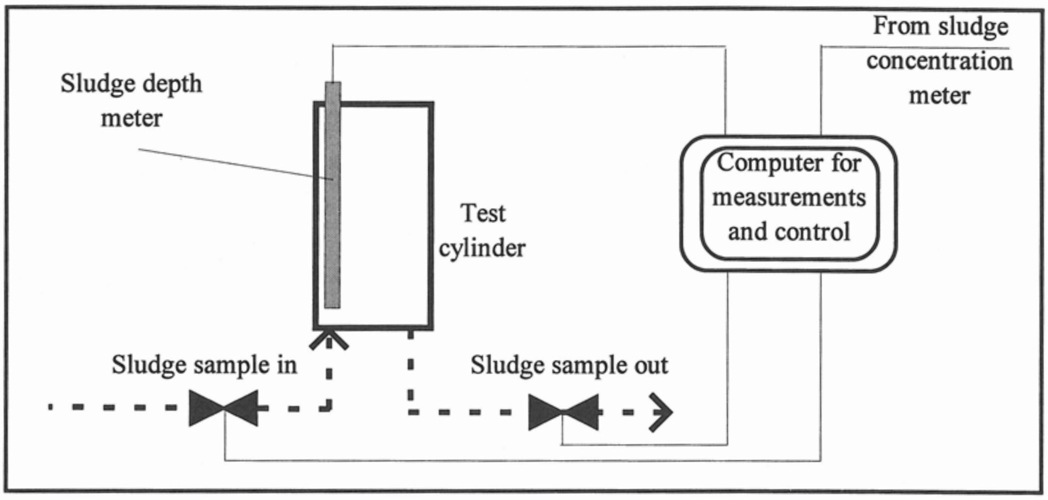

Figure 4. Measurement system of sludge settling at Suomenoja research station.

Results

Settleability

The variations of the parameter α, MLSS and soluble COD load are depicted in Figure 5. Figure 6 shows measured and computed (by Equation 1) sludge settling curves as a function of time and Figure 7 correlation between α and temperature, and between α and recycling flow rate.

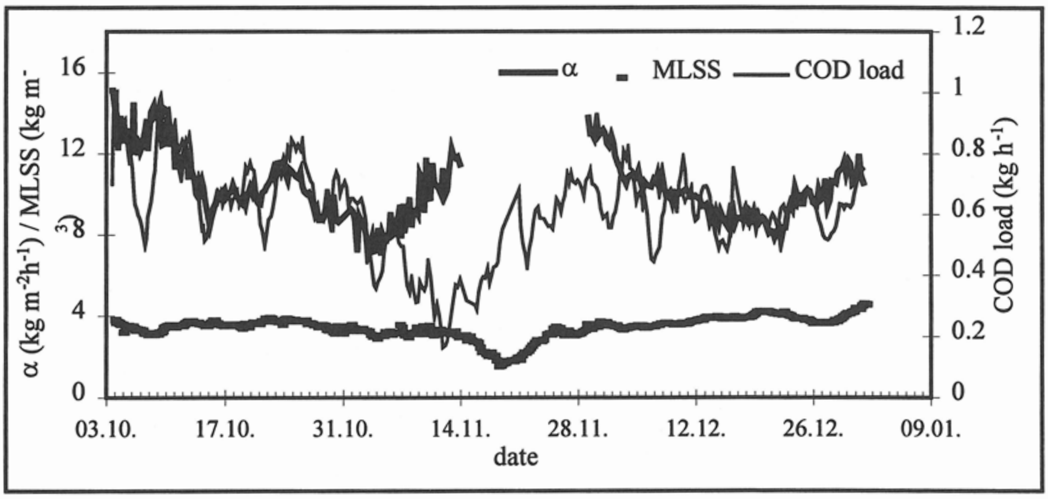

Figure 5. Variations of the parameter α, sludge concentration and soluble COD load

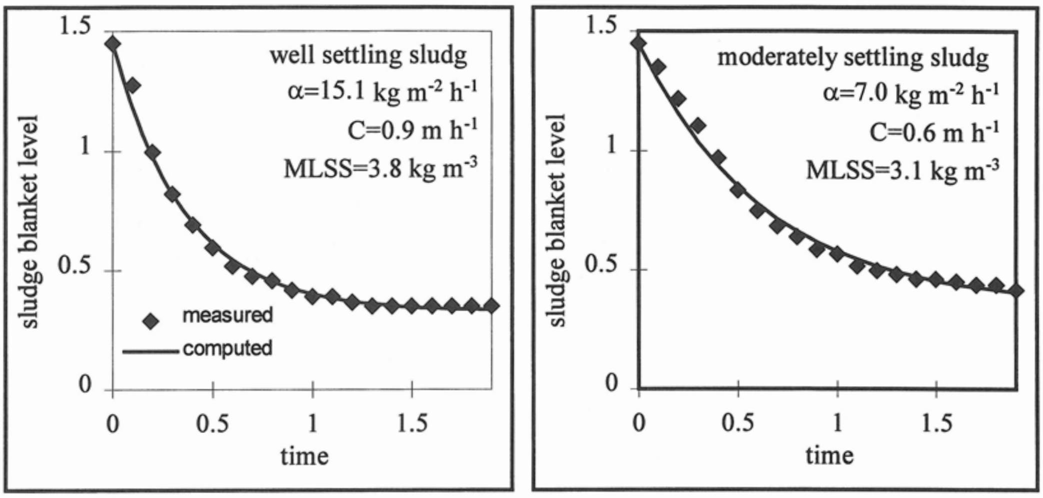

Figure 6. Measured and computed sludge settling curves.

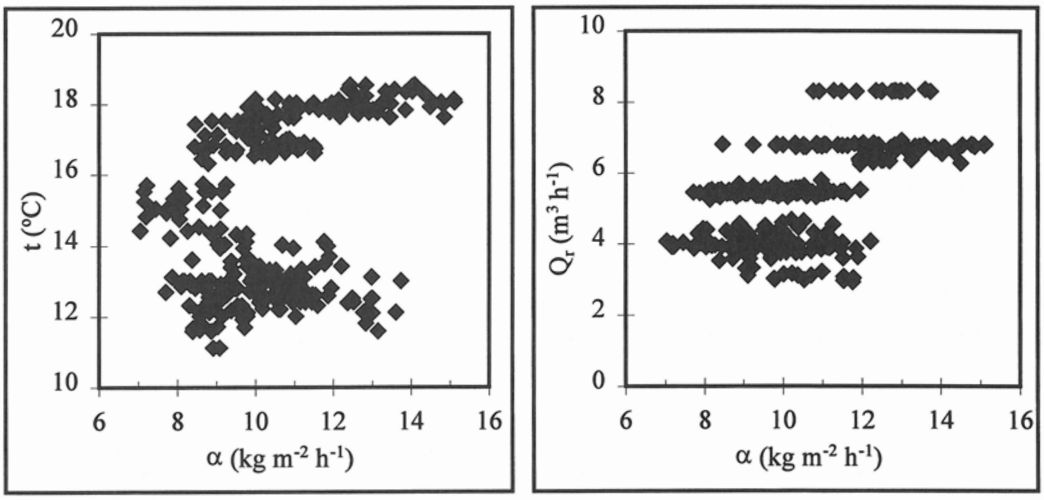

Figure 7. Temperature and recycling flow rate as a function of the parameter α

Characteristics

The quality of sludge is shown in Table 1.

Table 1 . The hydrophobic properties of sludge in different cases

Plant	Period	SRT (d)	Sludge character
Suomenoja tech. scale research (EBPNR)	spring summer autumn	>10	strongly hydrophobic
Suomenoja pilot (CO)	winter	2 - 5	less hydrophobic
Suomenoja full scale (CO)	summer-autumn	2 - 4	less - rather hydrophobic
Savonlinna full scale (EBPNR)	autumn	>20	(rather-) strongly hydrophobic
EBPNR = biol. P and N removal, CO = carbon oxidation			

The impact of protease concentration and contact time on bioflocculation is shown in Figure 8 in two typical samples.

In this examination the following supernatant density comparisons were made to evaluate different strategies or operational conditions:

- reference batch supernatant density / surfactant batch supernatant density,
- reference batch supernatant density / cationic polyelectrolyte batch supernatant density,
- reference batch supernatant density / nonionic polyelectrolyte batch supernatant density, and
- cationic polyelectrolyte batch supernatant density / nonionic polyelectrolyte batch supernatant density .

The comparison is shown in Figure 9. The correlation of hydrophobicity and protease sensitivity is shown in Figure 10.

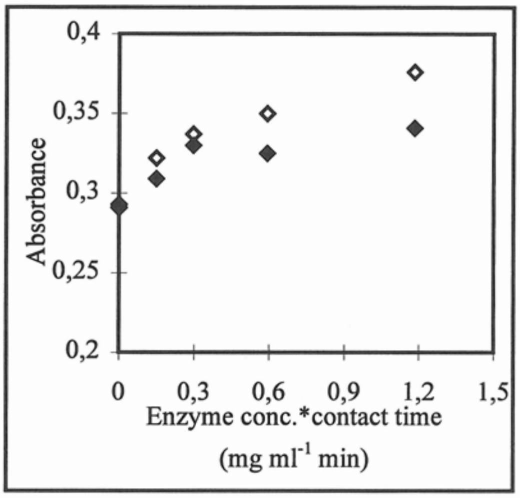

Figure 8. Impact of protease addition on supernatant density (absorbance) in two different sludges

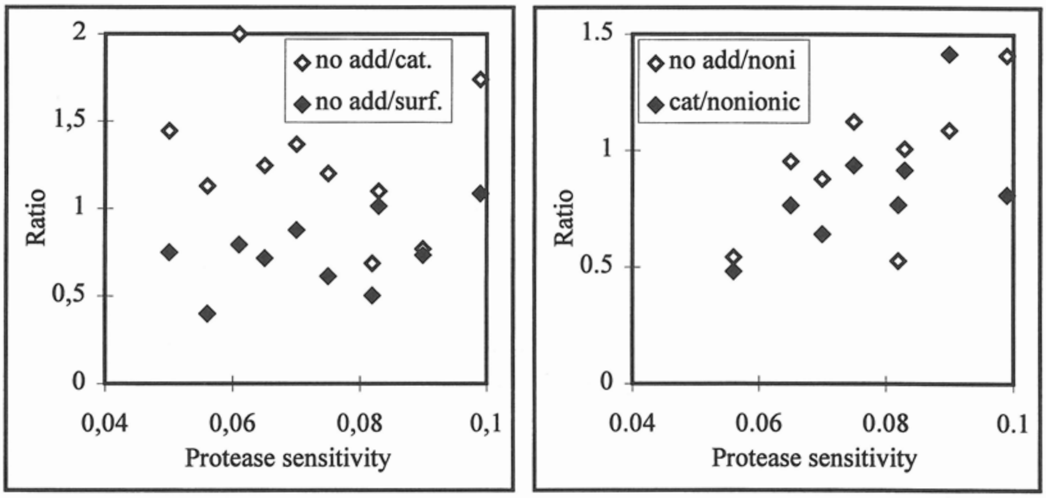

Figure 9. Correlation of batch test supernatant density ratio and protease sensitivity. Impact of surfactant and cationic polyelectrolyte addition (left) and impact of cationic and nonionic polyelectrolyte addition (right).

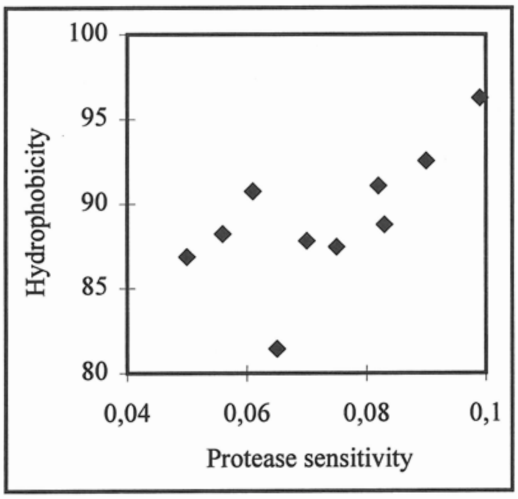

Figure 10. Correlation of protease sensitivity and hydrophobicity

Discussion

Sludge settleability is undoubtedly an important factor in operation of wastewater treatment plants. However, no agreement has been reached for assessing it and consequently monitoring techniques have not been established. For example, prevailing practice is determining sludge settleability from the sludge settling curve but the agreement how to do it in practice is still lacking. It is clear that settleability can be assessed with several different methods which describe the phenomenon under examination. The core question is what method we should use. Because settleability is more abstract than most of parameters at wastewater treatment plants (e.g. MLSS, DO and nutrient concentrations, flow rates, etc. whose interpretation is totally clear), it is obvious that a final agreement cannot be found. Probably, settleability will be assessed with a group of methods unless the definitions for them are exact.

Figure 6 shows the advantage of the model in describing sludge settleability. The model describes accurately the whole settling curve and the parameters compress its information. In this study only the parameter α is utilized for assessing sludge settleability because the interpretation is then easier even though it is not as accurate as when both of the parameters are used.

The sludge was estimated to be from moderately to well settling during the monitoring period. When determining the factors affecting sludge settleability the highest correlations were found between α and recycling flow rate, and α and soluble COD load. The correlations were not very high and for that reason too ambitious conclusions at this stage should be avoided. It is obvious that most of the factors affecting sludge settleability cannot be found with the linear correlation since the relationships are probably far more complicated and the effects might be opposite by cancelling each other and therefore difficult to discover. However, the following deduction of results can be reached: high recycling ration seems to have a positive effect on settleability and high amount of soluble COD seems also to have a similar effect. The former result can simply be interpreted that short detention of the sludge in the final clarifier maintains well settling sludge since the conditions of the sludge can be better controlled in the biological units. The latter result can be interpreted that easily biodegradable substrate (soluble COD) in the combined phosphorus and nitrogen removal process has a positive effect on the process performance and settleability.

Figure 7 reveals that the correlation between α and temperature is low. This is interesting since the higher viscosity due to lower temperature should have a negative effect on settleability. The obtained result can be explained that other factors have had stronger influence on the process than temperature.

Figure 5 shows that the parameter α is independent from the effect of sludge concentration. The relationship between α and MLSS is not relevant in view of finding the factors affecting settleability since MLSS is used in the model. However, this figure tells that the effect of MLSS is successfully eliminated in the model and settleability results at different sludge concentration can be compared (see Figure 1).

The hydrophobic/-philic character of sludge in different types of activated sludge processes seems to be different. In processes without nitrogen removal and with short SRT the sludge does not seem to have as hydrophobic character as in processes with longer SRT and biological nutrient removal.

One part of sludge hydrophobic material was formed by foam-forming microorganisms, which exhibit an increased hydrophobicity in cell walls (Wanner 1994). On the other hand, good flocculation was found with cultivated hydrophobic bacteria isolated from an activated sludge plant (Jorand et al 1994). At the Suomenoja research station process the average level of filaments has not been excessive (measured as filament length mentioned earlier), although in some periods there have been with foaming problems. In spite of the biologically produced material in the flocs mentioned before, also wastewater composition can have effect on the sludge characteristics.

These results can be interpreted that in these EBPNR-plants sludge obviously consists of hydrophobic microorganisms the main parts of which are not filaments or foam-forming microorganisms (like Microthrix etc.). This character can cause ‘a basic sensitivity’ to foaming and has to be taken into consideration in process management and operation.

The role (and proportion) of proteins in sludge extracellular material has not been very well known. One reason has been that methods used to determine the protein and other extracellular material content have not been very reliable or comparable and the yield of extracted material can vary (Frolund et al 1996). When proteins are the main constituent of sludge, their behaviour is of importance.

The use of protease is an alternative method to find out the role of proteins in the floc and the method used here gives an indication of its relation to bioflocculation. The sensitivity of sludge to protease seems to be variable and also correlated with other sludge flocculation parameters, although the number of observations is limited.

The protease test results indicate that
- the less important the proteins are to the floc formation, the poorer is the flocculation with surfactant
- the lowest supernatant turbidity can be achieved with the cationic polyelectrolyte and its effect may increase as the protease sensitivity decreases
- nonionic polyelectrolyte decreases residual turbidity at its best when the sludge indicates from average to high protease sensitivity.
- protease sensitivity and hydrophobicity have a positive correlation, which may indicate that proteins have a hydrophobic character.

These correlations can be interpreted that flocculation of the EBPNR-sludges has varied between two extreme points: one of them is where sludge is less sensitive to protease, not so hydrophobic and surface charge is an important factor in flocculation; and another extreme point, in which the sludge is sensitive to protease, hydrophobic bonding has effect and surface charge is not important. The line connecting these two points is protease sensitivity and more or less hydrophobicity. The observed variation can be due to the changes in growth conditions (as COD-load) as supported by the results of Jorand et al (1994) and Hantula and Bamford (1991) with isolated bacterial strain(s).

Conclusions

- the sludge was estimated to be from moderately to well settling during the monitoring period based on the on-line measurements and model parameters

- soluble COD load seems to have the strongest effect on settleability

- in processes without nitrogen removal and with short SRT the sludge does not have as hydrophobic character as in processes with longer SRT and biological nutrient removal. This hydrophobicity can cause ‘a basic sensitivity’ to foaming

- the exocellular proteins have a hydrophobic character and the sludge sensitivity to protease has varied considerably. This character is affected by growth conditions and gives information of the basic relationships for flocculation management.

In future the settler performance and optimization in connection with process parameters will be studied in more details.

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The use of automatic control in wastewater treatment plants

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Abstract

This paper surveys the use of automatic control strategies applied to wastewater treatment plants. A brief introduction to automatic control is given and control experiments conducted in a novel pilot-scale activated sludge plant are summarised. A strategy for controlling the dissolved oxygen concentration by means of an on-line ammonium meter is also presented.

Keywords: Activated sludge process, automation, instrumentation, nutrient removal, control, operation, supervision, pilot scale plant, wastewater treatment.

1. Introduction

The application of modern control theory, in combination with on-line sensors and the use of appropriate models, has a great potential to improve the effluent water quality, to decrease the use of chemicals and to save energy and money, see Lindberg (1997). However, most wastewater treatment plants (WWTP's), especially old or small ones, use fairly simple control strategies. Reasons for this include: (1) lack of tight effluent standards, (2) wastewater treatment is traditionally regarded as a non productive process, (3) lack of reliable on-line sensors, (4) poor controllability of many treatment processes, (5) the process is complex with nontrivial control objectives, (6) the design of the plants were often made for manual control. See also Olsson (1993) for a discussion of constraints in WWTP's. To some extent the problem may also depend on lack of suitable control methods adapted for WWTP's, poor communication between the automatic control and wastewater communities. Further, very few control engineers devote their work to the wastewater treatment area. It is also important to note that an automatic controller is not a miracle box which solves everything and can be left alone without supervision and maintenance. One may also need to spend several engineering hours to find the appropriate control strategy for a given control problem in a WWTP.

Control techniques commonly in use in todays plants include simple PLC-techniques, time control, flow proportional control and to some extent PID-control. There is, however, a growing interest in applying more efficient control strategies. Main reasons for this include: (1) cost effective solutions are becoming more and more important, (2) increasing demands on the treated wastewater including taxes and fees subject to effluent quality, (3) increasing awareness on environmental issues including sustainability factors, (4) more complex process alternatives which are harder to control manually, (5) development of appropriate dynamic models and adapted control strategies, (6) shortage of land area for building WWTP's. For example, Olsson and Newell (1997) predicts that in ten years time, WWTP's will have as sophisticated control strategies as the oil industry.

In the light of the above discussion, the purpose of this paper is to give an introductory discussion on the use of automatic control in WWTP's. In particular, the control of the activated sludge process is treated. See also Lindberg (1997) and Olsson and Newell (1997) for a thorough discussion.

2. What is automatic control?

Automatic control is the discipline for studying dynamic systems and how to control them. The methodology is general and can be applied to numerous fields including wastewater treatment.

The following simple example will illustrate a typical control problem. Assume that we want to control a level in the tank as shown in Figure 1 by adjusting the effluent flow rate via a control valve. The level is measured by a sensor.

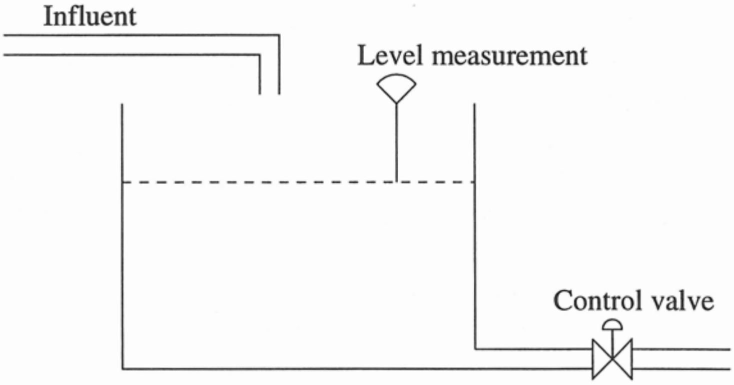

Figure 1: The level in the tank shall be controlled by the control valve.

The level is also influenced by a varying influent. It is often illustrative to draw a block diagram as in Figure 2. We can now formulate the control problem as: *construct a controller which, based on level measurements, calculates a control valve positioning so that the level is kept close to a given level (the set point) despite disturbances.* Most control problems can be expressed as variants of the above theme. Note, that the controller utilises *feedback* which is a fundamental concept in automatic control.

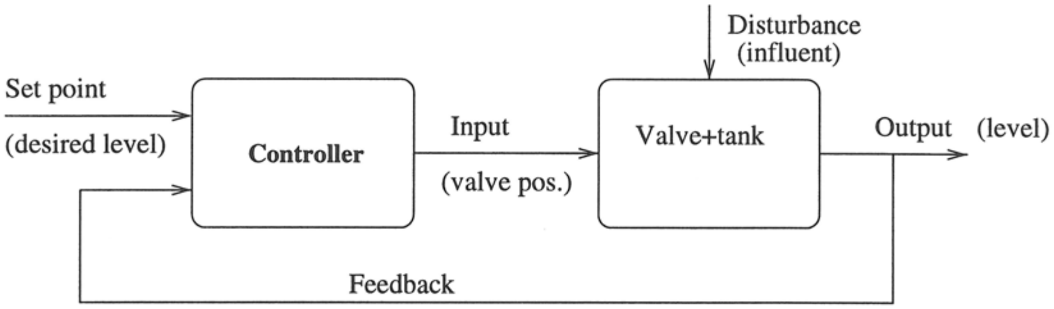

Figure 2: Block diagram of tank level control.

It is important to realize that it does not exist one controller or controller tuning which suits all processes. The controller has to be chosen and tuned depending on the process it will govern, the control objective and available process knowledge. Two common strategies (which often are used in parallel) are:

- Empirical tuning. Here trial and error and rules of thumb are utilised. A common feature in modern controllers is to have automatic tuning rules built in, so called autotuners. An example where an autotuner is used for DO (dissolved oxygen concentration) control in an activated sludge process can be found in Carlsson *et al.* (1994).
- Model based control. A mathematical model of the process to be controlled is used for controller design.

In a WWTP, control problems of very different complexity and specifications exist. Therefore, different control strategies are needed. Many control problems are today solved by *manual control*, i.e. the operator acts as the controller or by simple adjusting the control signal according to a predetermined scheme (time control). Other problems are sequential and can be solved by simple logics and on-off control. A typical example is to switch pumps on and off. These types of controller is implemented in PLC (programmable logic controller) techniques. However, todays PLC equipment can handle more complex control problems as well. Note that performance can often be significantly improved by using more advanced control strategies.

The most common controller is the PID controller, which can solve many basic control problems. Often, the control performance can be improved by utilising more than one sensor. For example, assume that the influent flow rate in the tank level control problem shown in Figure 1 can be measured. This measurements can be used to eliminate or reduce the disturbances arising from changes in the influent. The principle is called *feedforward.* Feedforward control is often useful in WWTP's where, for example, the influent flow rate can be used. Another fruitful strategy is *cascade control.* In the example in Figure 1, cascade control can be used if the effluent flow rate can be measured. A local controller is used for controlling the flow rate by adjusting the valve. A master controller governs the tank level by adjusting the set point to the local controller. Cascade control is often used for DO control in the activated sludge process.

For high control performance or complex processes more advanced controllers may be useful. A difficult control problem may arise if the process to be controlled has a long dead time, oscillations, non linearities or time varying dynamic. It is then often advantageous to use a process model for the controller design. See Lindberg (1997) for several examples.

3. Some control strategies

In this section we briefly survey some control strategies. The survey is by no means complete.

Automatic control techniques have been applied to control the (DO) in the activated sludge process during many years. The reasons include:

- There is much to gain in energy by controlling the DO, since the aeration is very energy demanding.
- On-line sensors have existed for DO measurements for several years.
- The effluent water quality and process efficiency depend largely on a suitable DO.

Several control strategies have been suggested, e.g. Flangan *et al.* (1977), Wells (1979), Ko *et al.* (1982), Rundqwist (1986), Holmberg *et al.* (1989), Carlsson *et al.* (1994), Lindberg and Carlsson (1996*c*) and Lindberg (1997).

Further energy savings can be achieved by minimizing the pressure drop over the air valves in the aerating system . A common strategy is to control the air pressure so that the most opened throttle valve is almost completely opened, see Rundqwist (1986).

On-line ammonium sensors can be used for controlling the set-point of the DO and/or the number of aerated zones. Instead of using a fixed set-point of the DO, the set-point may be determined from the ammonium concentration, see further Section 5. A control strategy for the DO set-point in a Bio-denitro$^{\mathrm{TM}}$ process can be found in Sorensen *et al.* (1994).

On-line nitrate sensors can be used for external carbon dosage control. Note that a too high carbon dosage is expensive, may cause carbon spill and increases the sludge production which affects the nitrification capacity, see Aspegren *et al.* (1992) and Hellström and Bosander (1990). Several approaches exist, see Londong (1992), Vanrolleghem *et al.* (1993), Linde (1993), Isaacs *et al.* (1993), Andersson *et al.* (1995), Yuan *et al.* (1996), and Lindberg and Carlsson (1996*a*).

Also, it is important to control the chemical dosage in a WWTP, for example when phosphorus is removed by chemical precipitation. In many plants no feedback control is used, instead the flow rate of chemical precipitation is made proportional to the influent flow rate or a simple time controller is used. A combination of feedback and feedforward control from on-line sensor, flow rates and an experience data base may be useful to obtain a more effective control of the chemical dosage.

Large variations in the wastewater flow may be significantly reduced by using buffering tanks or using the sewer tunnels as reservoirs. The variations in influent flow is mostly caused by storm water which leaks into the pipes, or by drainage water. These variations force the plant to work with higher flows during short periods; this is less efficient as compared to a constant flow.

The settler has the dual task of clarifying and thickening the sludge. Its function is crucial to the operation of the activated sludge process. Hence, modeling, diagnostics, and control of the sedimentation process is an important topic, see Jeppsson (1996) and Bergh (1996). Furthermore, efficient sludge treatment including control of the digester is crucial.

A key goal in wastewater treatment is to maintain the effluent quality standards at a minimum cost. Obviously it is then not enough to have an efficient control of various subprocesses. One must also consider the total optimization of the plant. In the future once can expect that more integrated control strategies are developed and applied. Optimizing different flow rates in the plant is one important topic. The importance of integrating control and process design to obtain best overall performance is illustrated by Andrews (1994). Cause-effects for relationships in an activated sludge plant have been established in Olsson and Jeppson (1994); these can be of help when designing controllers for new purposes, e.g. optimizing various flow rates. In Vanrolleghem and Verstraete (1993), a survey of new and existing technologies for the monitoring of WWTP's is given. Advanced control strategies have been suggested, see e.g. Bastin and Dochain (1990), Van Impe *et al.* (1992) and Lindberg (1997). Furthermore, new on-line sensors, suitable for use in wastewater treatment are continuously being developed, see the overview in Nielsen and Önnerth (1994).

4. Control experiments in a pilot scale activated sludge process.

A novel pilot-scale activated sludge plant has been used for studying efficient nutrient removal using advanced process technology, applied microbiology and automatic control. The pilot plant is located at the main municipal treatment plant in Uppsala. The plant consists of two separate lines for pre-denitrification. Both lines were extensively monitored by on-line instruments and sampling programs. The plant is controlled by an advanced control and supervision system, see Carlsson and Lindberg (1995). During the period June 1994 to November 1996 a number of experiments have been conducted in the pilot plant. Many of the results have been published in leading journals and conference proceedings. A short summary of control related experiments are given below. Detailed descriptions can be found in the cited references.

- An external carbon source may be needed to improve the denitrification in an activated sludge process. An automatic control strategy for carbon source dosage based on on line meters for nitrate-nitrogen was developed. The objective was to control the flow rate of external carbon so that the nitrate level in the last anoxic zone is kept at a low value despite load changes. Practical tests showed that the control strategy worked well despite a rather noisy nitrate sensor, see further Lindberg and Carlsson (1996*a*).

- A nonlinear dissolved oxygen concentration (DO) controller has been evaluated The controller uses an estimate of the nonlinear oxygen transfer rate (see Carlsson *et al.* (1994)) in order to linearize the DO process. By practical experiments in the pilot plant it was shown that this nonlinear controller outperforms a standard linear controller, see further Lindberg and Carlsson (1996*c*) and Lindberg (1997).

- A supervision DO controller has been suggested and evaluated in the pilot plant, see Lindberg and Carlsson (1996*d*) and Lindberg and Carlsson (1996*b*). The results are outlined in the next section.

- The problem to estimate the respiration rate and oxygen transfer rate from measurements of the DO and air flow rate (or valve positioning) has been studied in the pilot plant. The estimated respiration rate was close to values obtained from batch samples analysis. The estimated oxygen transfer function was nonlinear with respect to the air valve position. The estimation methods and results are reported in Lindberg (1997) and Carlsson *et al.* (1994).

- During the second half of 1996, the pilot plant was used to evaluate how modern detergents affect the oxygen transfer rate. Principal investigator was IVL. The estimation methods presented in Lindberg (1997) and Carlsson *et al.* (1994). were successfully applied.

5. DO set-point control in an activated sludge process

Normally a constant DO, controlled by varying the airflow rate guarantees that enough oxygen is supplied to the aerobic zones in an activated sludge process. An alternative to a constant DO set-point is to use a *time-varying* DO set-point, which is determined by the ammonium concentration in the last aerobic zone. The DO set-point control gives or may give the following advantages, depending on the existing plant performance:

- Better control of effluent ammonium.
- Lower average DO set-point which saves energy.
- Lower nitrate concentration in the effluent, because of improved denitrification due to a lower DO.
- Lower dosage of external carbon (if it is added).

It is, however, important to be aware of the following possible drawbacks:

- Deterioration of sludge properties.
- Risk of nitrous oxide (N_2O) emissions.

The basic idea is to control the DO set-point from on-line measurements of the ammonium concentration, this is illustrated in Figure 3. This structure is referred to as cascade control, where the inner DO loop is much faster than outer, the ammonium loop. The control objective is to maintain a prespecified ammonium concentration by varying the DO set-point. To avoid that the DO set-point becomes too high or too low, it should only be allowed to vary in an interval e.g. 0.5–4 mg O_2/l, which is easily implemented.

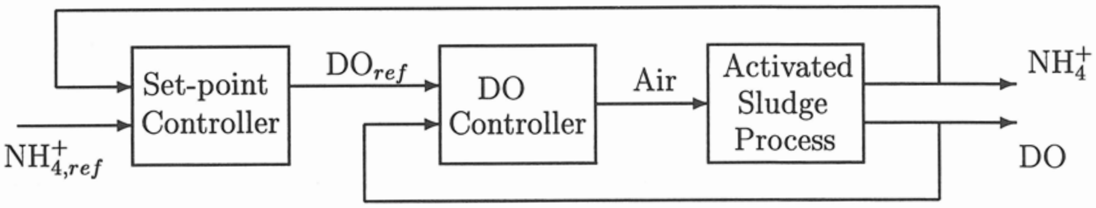

Figure 3: Block diagram of the DO set-point controller

Usually there are several aerated zones in a WWTP, one strategy is to control the DO set-point equally in all aerated zones. Another idea could be to only control the set-point spatially, and switch on and off zones when needed. This latter strategy is probably better than the first but has not been evaluated here.

A practical experiment with the DO set-point controller has been made in the pilot scale plant. The same DO set-point was used in all three aerated zones. The ammonium concentration was measured in the last aerated zone where the highest ammonium concentration is expected to be found. The ammonium sensor gives a new value every 20th minute. The set-point controller was started after approximately 1.5 days with a set-point for the ammonium concentration on 1 mg/l. Since the ammonium level was lower than the ammonium set-point, the DO set-point was decreased from 2 mg/l to below 1 mg/l. The experiment is illustrated in Figure 4.

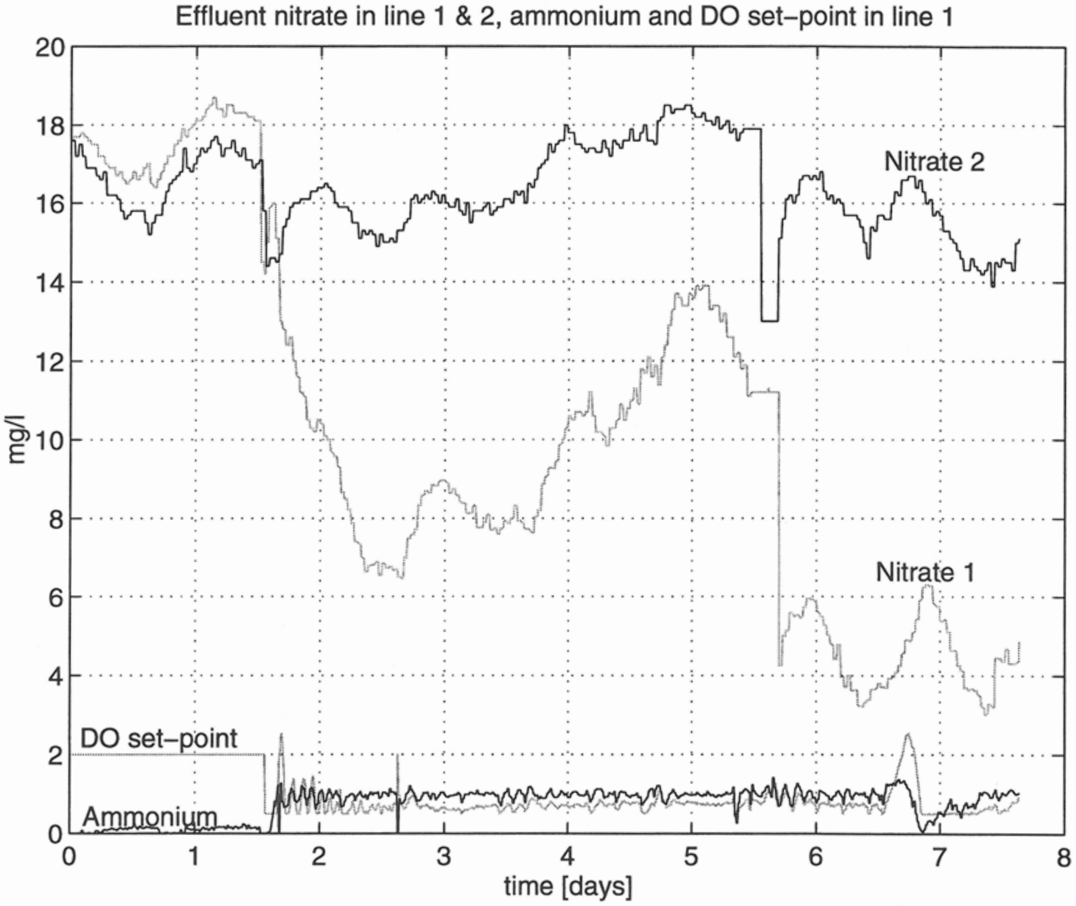

Figure 4: Control of the DO set-point. The ammonium set-point was 1 mg/l.

A positive effect of the lower DO level was that the effluent nitrate level was significantly decreased. Note that effluent nitrate level for the controlled line (Nitrate 1) is less that 50% of that in the reference line (Nitrate 2). Effluent ammonium was increased from about zero to the set-point of 1 mg/l, but this minor increase should be compared with the large reduction in effluent nitrate and energy consumption.

6. Conclusions

This paper has surveyed the use of automatic control in WWTP's. In particular, strategies for the activated sludge process was outlined. The application of advanced control theory, in combination with appropriate on-line sensors and the use of suitable process models, has a great potential to improve the effluent water quality, to decrease the use of chemicals and to save energy and money. However, there are presently several constraints that have to be solved and more full-scale evaluations are certainly needed.

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