Evaluation of separate urine collection and treatment to augment existing wastewater treatment works

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Abstract
Simultaneous increases in wastewater loads and effluent quality demand improved nutrient removal techniques. A simple technique for nitrogen removal is post-denitrification with methanol. The trade-off between better effluent vs. methanol consumption is debatable. Methanol dosing is not only unsustainable in the long term, but the cost of methanol is also becoming increasingly important. Urine contains 80% of the total nitrogen (N) and 50% of the phosphate in wastewater. Separate collection and treatment of urine could improve existing treatment works and diminish the need for post-denitrification. In this paper, a nitritation-denitrification process is proposed where 20–30% of the N in urine is removed with the COD available in urine. The treated urine consists of ammonium-nitrite, which is to be introduced to the anoxic zone of a conventional treatment plant. Optimal denitrification via nitrite is possible with COD from other wastewater sources. The case study presented here shows that 40–50% urine separation and improvement of the flow scheme would improve effluent quality from 19 to 10 g N/m³, which would eliminate the need for post-denitrification.

Keywords
Nitritation; post-denitrification; source control; urine; wastewater

Introduction
Many municipal wastewater treatment works are under stress to improve effluent quality while waste loads are slowly increasing. Requirements for better performance almost invariably lead to modifications or upgrades of plants. For example, the modifications required for incorporating biological nutrient removal in plants not originally designed for this purpose often prove to be both disruptive and expensive. Where possible, modifications are best avoided or minimised.

Most nutrients in municipal wastewater originate from urine. Urine contains around 80% of the nitrogen (N) and up to 50% of the phosphorus (P) in wastewater (Larsen and Gujer, 1996; Larsen and Udert, 1999). Source separation of urine is gaining interest and could be an alternative for upgrading existing wastewater treatment plants. In two previous studies, we have shown the advantages of partial urine separation on central wastewater treatment plants in general terms:

(i) State-of-the-art wastewater treatment works (modified UCT process) would gain 30% capacity at around 30–50% urine separation. Primary sedimentation could be included in such a process without limiting the N-removal performance (Wilsenach and Van Loosdrecht, 2004).

(ii) Treatment of wastewater and urine could be integrated and optimised in a compact system. Nutrients in the wastewater could be removed through high sludge production, which allows higher methane production. Supernatant from the sludge digestion could be treated with separated urine in a combined Sharon/Anammox process. Consequently, net energy could be generated at the treatment plant (Wilsenach and Van Loosdrecht, 2003).
However, the first of these two studies represents a system that already operates well and produces good effluent quality. This kind of system does not require urgent (or any) upgrade. The second study represents a system that has never been implemented as described, and therefore presently operates well on a conceptual level only. Many real systems, on the other hand, require immediate and/or extensive upgrade.

Direct re-use of urine, e.g. as a fertiliser, could be a problematic option in Western Europe. The amount of nutrients available from the waste of animal husbandry is so high that, combined with the lack of available farmland close to cities, nutrients from urine are available in excess (Wilsenach et al., 2003). If urine were collected separately, it would have to be treated within the scope of urban wastewater management. A simple technique, in which urine is treated in a partial nitrification and denitrification process, is described here. This technique would fit well within a transitional phase of moving from conventional technology to wastewater management based on source control.

This paper shows to what extent separate collection and treatment of urine could augment and improve the performance of an existing conventional wastewater treatment system. This case study was mainly focussed on wastewater treatment plant De Groote Lucht, in Vlaardingen, The Netherlands.

**Case study: wastewater treatment plant De Groote Lucht**

The water board Delfland operates three large wastewater treatment plants and two smaller plants. A new treatment plant is currently being constructed for the region of Delft and The Hague. This region has the equivalent wastewater production of around 1.5 million people. Wastewater treatment plant De Groote Lucht treats wastewater from 250,000–280,000 people, some of which is from offices and working environments. De Groote Lucht has to conform to nitrogen effluent standards of 10 g N\textsubscript{tot}/m\textsuperscript{3}, while the region as a whole has to remove 75% of the total influent phosphate load. However, this situation is expected to change in future, with effluent quality standards becoming stricter for all treatment plants.

The treatment works consist of primary sedimentation, followed by a high loaded trickling filter with intermediate sedimentation, and an activated sludge basin (20,000 m\textsuperscript{3}) followed by final clarification (Figure 1). Some of the settled wastewater bypasses the trickling filter to allow some denitrification in the first compartment (anoxic) of the activated sludge basin. Sludge from the primary sedimentation, trickling filter and excess sludge are anaerobically digested (Figure 1).

**Figure 1** Simplified process flow diagram of wastewater treatment plant, De Groote Lucht, with nitrogen mass balance.
activated sludge are all thickened before anaerobic digestion. Digested sludge is centrifuged and removed by truck for incineration off site. Centrate is returned to the plant inlet with thickener overflow.

Most of the effluent is routed via a post-denitrification filter (DNF), as described by Kramer et al. (2000). Backwash water from the DNF is returned to the plant inlet. The carbon source added for biological denitrification in this process is methanol (950 ton/year). With the DNF, which accounts for almost a third of the nitrogen removal at this plant, a total nitrogen removal efficiency of 75–80% is achieved. In terms of the direct operational cost as well as the sustainability aspects of using methanol, Delfland would benefit much from a decrease in the methanol consumption. Phosphate removal is not a main priority at this plant. Although only 50% of total phosphate load is removed at De Groote Lucht, the regional removal efficiency is presently above 75%.

Materials and method

Data and analysis

Annual data including daily measurements of influent and effluent flow rates and concentrations (N_{Kjeldahl}, N_{nitrate}, P_{tot}, COD, BOD, and total solids) were available from internal reports. Additional data from the plant included monthly measurements of the same parameters at different points in the treatment process (downstream of primary sedimentation and influent to activated sludge basin) and the return flows (centrate and DNF washwater). Furthermore, the sludge loads from the three different thickeners to the anaerobic digester were known, as well as the yearly load of dewatered sludge. Annual average wastewater influent and effluent characteristics are shown in Table 1.

Apart from recorded data, a 1-litre sample of activated sludge was collected and analysed on the day of collection. A Dr Lange spectrophotometer was used to determine the N, P and COD content of the sludge. The value of total suspended solids was determined by filtering over vacuum and drying samples at 100°C for 24 h before weighing. Averages from three sets of measurements are shown in Table 2.

Mass balances

Flow rates, and total solids were balanced over all sedimentation tanks and sludge treatment (digestion and dewatering) before a nitrogen balance was calculated. Kjeldahl nitrogen and nitrate mass balances were calculated, based on the flow rates, total solids combined with sludge composition (Table 2) and the measured Kjeldahl and nitrate concentrations. Nitrogen removal over the DNF was calculated from daily on-line measurements and monthly laboratory measurements. The nutrient content of primary sludge was calculated to close the balance over the primary sedimentation tanks; N = 3.7% and P = 0.7% of TSS, which is reasonable based on literature (e.g. Tchobanoglous, 1991). Nutrient removal through sludge production was based on the actual mass of dewatered sludge removal, the measured nutrient content of activated sludge (Table 2) and the actual nitrogen load in reject water. Denitrification in the activated sludge basin was calculated from the difference between influent nitrogen and the sum of the effluent loads.

<table>
<thead>
<tr>
<th>Flow rate m³/d</th>
<th>Chemical oxygen demand g COD/m³</th>
<th>Biochem. oxygen demand g BOD/m³</th>
<th>Total suspend solids g TSS/m³</th>
<th>Kjeldahl Nitrogen g N/m³</th>
<th>Nitrate Nitrogen g N/m³</th>
<th>Total Phosphate g P/m³</th>
</tr>
</thead>
<tbody>
<tr>
<td>Influent</td>
<td>64,015</td>
<td>367</td>
<td>133</td>
<td>172</td>
<td>38.3</td>
<td>0</td>
</tr>
<tr>
<td>Effluent</td>
<td>64,009</td>
<td>41</td>
<td>5</td>
<td>5</td>
<td>2.5</td>
<td>5.9</td>
</tr>
</tbody>
</table>

Table 1 Composition of raw wastewater and final effluent from De Groote Lucht
The most important nitrogen loads are shown in Figure 1. The influent nitrogen load was 2,500 kg/d (100%), which divided into final effluent (21%), denitrification in the anoxic zone (28%), post-denitrification (29%) and nitrogen in dewatered sludge (22%).

Modelling

**Modelling activated sludge basin and model calibration.** Activated sludge model 2d (ASM2d) was used as a basis for simulating activated sludge (Henze et al., 1999), but without the processes for biological phosphate removal. The model was implemented in Aquasim software (Reichert, 1994). Five compartments were defined; one anoxic compartment (3,300 m³) followed by four aerobic compartments (4,150 m³ each) to simulate a semi plug flow operation. At De Groote Lucht, return activated sludge from the clarifiers is pumped with an Archimedean screw pump. We simulated this as a small aerobic compartment (30 m³) in the return sludge line. Influent COD fractions were determined according to guidelines by Roeleveld and van Loosdrecht (2002), who also reported a high particulate inorganic fraction in the raw wastewater at De Groote Lucht (\(X_S = 102\), \(X_I = 187\), \(S_A + S_F = 129\) and \(S_I = 29\) g/m³). We adjusted the model parameters according to this fractionation and the corresponding annual measurements for TSS, COD and BOD at different points in the treatment, i.e. after primary sedimentation and after the trickling filter (Table 3).

Because of the relatively high load of inert particulate material in the influent, we differentiated between inert particulates in the influent and inert particulates produced through sludge hydrolysis. We defined a new model state variable, \(X_{IW}\), to describe particulate material in the influent as well as a process whereby all \(X_{IW}\) is converted to \(X_I\), with exchange of soluble phosphate and ammonia between solids and the bulk liquid. An arbitrary process rate was defined so that \(X_{IW}\) was zero in the final aerobic reactor. In this way, the nutrient content of excess sludge could be defined precisely without affecting the influent soluble nutrient concentrations, which is important when urine separation is considered. We assumed the following COD fractionation after primary sedimentation: \(X_S = 32\), \(X_{IW} = 72\), \(X_H = 5\), \(S_A = 36\), \(S_F = 39\) and \(S_I = 30\) g/m³; and for wastewater entering the activated sludge basin: \(X_S = 25\), \(X_{IW} = 40\), \(X_H = 18\), \(S_A = 18\), \(S_F = 19\) and \(S_I = 30\) g/m³.

With this approach, the default reaction rates, kinetics and yields gave a close representation of the activated sludge process. The predicted excess sludge production as

<table>
<thead>
<tr>
<th>Table 2 Average composition of activated sludge from De Groote Lucht</th>
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<tbody>
<tr>
<td>Chemical oxygen demand</td>
</tr>
<tr>
<td>(g COD/m³)</td>
</tr>
<tr>
<td>4,200</td>
</tr>
</tbody>
</table>

(final effluent, DNF and dewatered sludge). The most important nitrogen loads are shown in Figure 1. The influent nitrogen load was 2,500 kg/d (100%), which divided into final effluent (21%), denitrification in the anoxic zone (28%), post-denitrification (29%) and nitrogen in dewatered sludge (22%).

<table>
<thead>
<tr>
<th>Table 3 Organic substrate in raw wastewater as well as activated sludge influent at wastewater treatment plant De Groote Lucht (all values in g/m³)</th>
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</thead>
<tbody>
<tr>
<td>COD&lt;sub&gt;tot&lt;/sub&gt;</td>
</tr>
<tr>
<td>BOD&lt;sub&gt;5&lt;/sub&gt;</td>
</tr>
<tr>
<td>TSS</td>
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<td>BCOD</td>
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well as the total suspended solids and total COD concentration in the reactor equalled the real data (Table 2 and mass balance) without further adjustment.

We further followed the protocol suggested by Hulsbeek et al. (2002) to calibrate nitrification and denitrification. With default values \( b_{AUT} = 0.09 \) (1/d) at 15°C, \( K_{NH4} = 1 \text{ g N/m}^3 \), \( K_{NH4,ox} = 0.5 \text{ g O}_2/\text{m}^3 \) the model is not sensitive to changes in actual dissolved oxygen concentration between 1 and 2 g O\(_2\)/m\(^3\) and predicts ammonium effluent concentrations around 1 g N/m\(^3\). The autotrophic decay rate as well as both affinity constants for ammonium and oxygen were increased to \( b_{AUT} = 0.14 \) (1/d) at 15°C, \( K_{NH4} = 1.28 \text{ g N/m}^3 \) and \( K_{NH4,ox} = 1 \text{ g O}_2/\text{m}^3 \) to simulate the ammonium effluent concentration correctly (Table 1). The reduction factor for anoxic activity was lowered from 0.8 to 0.6, which then gave the correct nitrate effluent concentration. In this way, denitrification was also modelled correctly (based on the mass balance data).

To allow integrated simulation of urine treatment, nitrification was split into two processes, i.e. ammonium oxidation (to nitrite) and nitrite oxidation (to nitrate). The maximum growth rates for both organisms were expressed as functions of temperature (Wiesmann, 1994). At 15°C, ammonia oxidisers had a growth rate of 0.59 and nitrite oxidisers 0.76 (1/d). Arbitrary affinity constants were assumed for nitrite oxidation (\( K_{NO2} = 0.1 \text{ g N/m}^3 \) and \( K_{NO2,ox} = 0.25 \text{ g O}_2/\text{m}^3 \)) to ensure that no nitrite remained in the final effluent, in accordance with observations. Two additional processes were included to allow denitrification (using either \( S_A \) or \( S_F \) as electron donor) with nitrite as the electron acceptor. All changes to affinity constants and rates were checked against the calibrated model (with one-step nitrification and denitrification) to ensure the new model’s validity.

We added an extra term to the standard Monod type equation for the denitrification rate, similar to the terms imported for the two soluble organic substrates (\( S_A \) and \( S_F \)) to equalise the process rates for denitrification. As shown below, one can force the model to give preference to denitrification via nitrite or nitrate:

\[
\frac{f_2 NO_2^-}{f_2 NO_2^- + f_3 NO_3^-} \quad \text{for nitrite denitrification; } \frac{f_3 NO_3^-}{f_3 NO_3^- + f_2 NO_2^-} \quad \text{for nitrate denitrification.}
\]

To evaluate the effect of preferential denitrification via nitrite, the value of the factor \( f_2 \) was set to 10 while \( f_3 \) was set to 1. The values of these factors were reversed (\( f_2 = 1 \) and \( f_3 = 10 \)) to evaluate the effect of preferential denitrification via nitrate.

**Modelling phosphate recovery from urine.** Struvite, or MgNH\(_4\)PO\(_4\)·6H\(_2\)O, can be recovered in a controlled way as demonstrated in the full-scale reactor at a manure treatment installation in Putten, The Netherlands (Schuiling and Andrade, 1999). Struvite recovery from urine is also possible. Urea hydrolysis occurs in urine collection systems, increasing the pH to 9–9.2 (Udert et al., 2003b), where struvite precipitates if all necessary elements are available. Addition of MgCl\(_2\) to a stirred tank is generally sufficient for struvite precipitation and crystallisation. We assumed struvite precipitation as a good process for phosphate removal from urine, with an effluent concentration of 15 g P/m\(^3\) (Figure 2).

**Modelling partial nitritation and denitritation in urine.** Biological treatment of separated urine is proposed in two continuously stirred tank reactors (CSTR) with recycle, as shown in Figure 2.

Biological oxidation of nitrite (NO\(_2^-\)) can be prevented in liquids with elevated temperatures (>20°C) and ammonium concentrations (1 g N/l), such as sludge digester supernatant (e.g. SHARON process, Hellinga et al., 1998). Udert et al. (2003a)

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demonstrated that 50% of the ammonium in urine can be oxidised to nitrite, without any nitrate (NO$_3^-$) production, in a CSTR without pH control. Apart from nutrients, urine also contains soluble organic carbon. This COD could be employed for denitrification via NO$_2^-$, which also serves as a measure for pH correction. We assumed urine production is 2 l/p.d, which includes flush water (Larsen et al., 2001) with NH$_4^+$ = 12 g/p.d, P = 1.0 g/p.d and COD = 10.8 g/p.d (Ciba Geigy, 1977; Udert et al., 2003a). Nitrification in urine lies outside the application range of ASM2d (and similar models). In reality, NH$_3$ and not NH$_4^+$ is the substrate for nitrifying organisms. Concentrations of NH$_3$ + NH$_4^+$ and HNO$_2$ + NO$_2^-$ are very high, and the substrate could be inhibitive in accordance with pH equilibria. Consumption of alkalinity as defined in ASM2d is also not valid for urine, which has a complex pH buffer system. In order to predict the amount of nitrogen gas and biomass produced, we increased the influent alkalinity of urine in ASM2d, until NH$_4^+$:NO$_2^-$ = 1 in the effluent was reached, with no possibility for nitrate production. The use of activated sludge models in this way cannot be encouraged, but we are able to support the assumptions (aerobic effluent of urine treatment, shown in Figure 2) with data from our own laboratory experiments. It is beyond the scope of this study to report on these experiments in more detail, or to improve the modelling of urine treatment according to the proposed process. We can confirm that 20–30% of the influent nitrogen is removed and that NO$_2^-:$NH$_4^+$ = 1 in the effluent is a good assumption. The production of biomass follows from the converted substrates, based on normal yields, with aerobic hydraulic retention times of 4–5 days.

Proposed reconfigured flow scheme

We investigated the option of bypassing the trickling filter altogether and rerouting the settled wastewater directly to the anoxic compartment. The trickling filter should then be
employed for complete nitrification (Figure 2). We investigated the effect of separate urine separation and treatment on both modes of operation, i.e. current situation (Figure 1) and proposed alternative flow scheme (Figure 2) for different percentages of separate urine collection (20%, 40%, 60% and 80% of total urine production).

Results and discussion

Effect of separate urine treatment on nitrogen removal within current flow scheme

The decrease in total effluent nitrogen concentration with increasing percentages of separate urine treatment is shown in Figure 3a. These results show that the effect of separate urine treatment – when treated as described above – would not solve the imminent problems at De Groote Lucht. Even when 80% of the total urine production is collected and treated separately, the total nitrogen effluent concentration would only decrease from 19 g N/m³ to 15.7 g N/m³ (upstream of the DNF).

Figure 3a also shows the increase in N₂ produced with increasing urine separation. This value is the sum of N₂ produced in the separate treatment of urine and in the main wastewater treatment plant (anoxic compartment), expressed as an effluent concentration. When compared to the values assumed for urine treatment (Figure 2), it is revealed that all the additional N₂ production is due to the treatment of urine. The NO₂⁻ added to the anoxic compartment has no effect on overall nitrogen removal. This might be counterintuitive, because the stoichiometry of denitrification via NO₂⁻ shows that less organic carbon is required (compared to denitrification from NO₃⁻) and that more carbon would be available for denitrification, hence more denitrification. However, the anoxic zone always has an excess amount of nitrate (around 7–8 g N/m³). This is higher than the nitrite introduced (1.8 g NO₂⁻-N/m³ for every 20% urine treated separately), which is even further diluted in the return sludge with a flow rate equal to the influent. Although nitrite might be the preferred electron acceptor (above nitrate), the rate of nitrate denitrification is still higher, with the introduction of the saturation terms as discussed above. The COD available for denitrification in the influent is limited and no soluble biodegradable COD remains in the anoxic reactor, while some nitrite remains (0.5–1.5 g NO₂⁻-N/m³).

Where improved denitrification is required, it seems that sufficient soluble organic material is more important than converting ammonium in urine to NO₂⁻. The model was not sensitive to preferential denitrification via either NO₂⁻ or NO₃⁻.

Effect of separate urine treatment on nitrogen removal within proposed flow scheme

With the proposed reconfigured flow scheme (i.e. trickling filter downstream of activated sludge basin), the SRT had to be decreased from 15 to 10 days in order to maintain a
similar concentration of suspended solids in the reactor and prevent solids overload of the final clarifiers. With 10 days, total COD in the aerobic compartment was 4.7 kg/m³ (4.2 kg/m³ under normal operation). The reduced sludge age increases the excess sludge production as well as the nitrogen load removed via this route to 760 kg N/d (460 kg N/d with normal operation). However, the total nitrogen influent load to the activated sludge basin is also increased, to 35 g N/m³ (from 32 g N/m³), due to the bypassing of the trickling filter. The additional nitrogen removed with excess sludge is therefore only a replacement of the nitrogen normally removed with sludge from the trickling filter.

Figure 3a shows that even without any urine separation, the total nitrogen effluent in the reconfigured flow scheme is less than with 80% urine treated separately in the current flow scheme. Figure 3a also shows that the N₂ production is 4 g N/m³ higher in the reconfigured flow scheme, without any urine separation. It is also evident from Figure 3a that the additional N₂ production is more than through separate treatment of urine alone. Although not all the influent NO₂⁻ is denitrified, the situation is improved due to the increased soluble COD load. The nitrate content in the anoxic reactor is also much lower in this case (1.2 g NO₃-N/m³ compared to the current operation of 7 g NO₃-N/m³, as mentioned before). This allows for more denitrification via the more efficient nitrite route. As was observed for the current plant operation, the model was not sensitive to preferential denitrification via either NO₂⁻ or NO₃⁻ (at most 1 g N/m³ difference in effluent quality at 80% urine separation).

We also investigated the possibility of forcing denitrification via NO₂⁻ only by separating the influent and return sludge streams (i.e. a small compartment upstream of return sludge inlet to prevent contact with NO₃⁻). Denitrification of treated urine (via nitrite) over the post-denitrification filter was also investigated. The DNF effluent would in this case contain too high ammonium concentrations for direct discharge into the final effluent, and would require rerouting via the activated sludge basin to the trickling filter. None of these options showed any significant improvement.

The load of methanol required to maintain a nitrogen effluent concentration of 10 g N/m³ is shown in Table 4, for the present and the proposed flow schemes, and within the range of 0–40% urine separation. If reduction in methanol consumption were the primary motive, the proposed flow scheme would be far more beneficial than separate urine collection and treatment. The decrease in methanol consumption with increasing urine separation is bigger in the proposed flow scheme than in the present flow scheme. This observation is in agreement with the results regarding the nitrogen decreases. Furthermore, it seems possible to reach the effluent standard (10 g N/m³) with separate treatment of 40–50% urine, without methanol dosing.

**Effect of separate urine treatment on ammonium oxidation**

In a previous study, we argued that urine separation in itself would not improve the ammonium effluent quality of a nitrifying plant. This is mainly due to a lower autotrophic population (nitrifiers) as a result of a lower ammonium influent load (Wilsenach and Van Loosdrecht, 2004). However, Figure 3b shows that for the treatment option proposed

<table>
<thead>
<tr>
<th>Urine separation</th>
<th>0%</th>
<th>20%</th>
<th>40%</th>
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<tbody>
<tr>
<td>Methanol consumption (ton CH₃OH/year); with present flow configuration</td>
<td>740</td>
<td>670</td>
<td>600</td>
</tr>
<tr>
<td>with proposed new flow configuration</td>
<td>385</td>
<td>215</td>
<td>75</td>
</tr>
</tbody>
</table>
here, ammonium effluent concentration (from the activated sludge basin) decreases with increasing urine separation. This is only due to the constant inoculation with ammonium oxidisers from the treated urine. For the proposed flow scheme, the short sludge age leads to washout of ammonium oxidisers and prevents effective nitrification. Without any separate urine treatment, the effluent ammonium concentration would be more than 5 g N/m³. Figure 3b shows a steep decrease in ammonium concentration with increasing urine separation, where biomass from the urine treatment augments the nitrification capacity. The ammonia oxidisers from the urine treatment system grow under different conditions, e.g. higher temperature, higher ammonium and nitrate concentrations, differences in predation, etc. One could speculate that the bacteria produced with urine treatment would not survive in the activated sludge basin. In that case, the ammonium concentrations shown in Figure 3b would be constant, or even slightly increasing. Nitrification would then have to be completed outside the activated sludge basin. Figure 2 shows the reconfigured flow scheme, with post-nitrification. The trickling filter currently removes 4,200 kg BOD/d. If the activated sludge effluent (64,000 m³/d) still contains 5 g NH₄⁺-N/m³, this would require around 1,500 kg O₂/d for complete nitrification (based on 4.57 g O₂/g NH₄⁺-N), which is clearly much less than the aeration capacity of the trickling filter.

**Effect of separate urine treatment on phosphate removal**

Under normal operating conditions, and without addition of any ferrous salts, the effluent phosphate concentration is around 3.8 g P/m³. With the proposed flow scheme, the higher sludge production and removal decreases the effluent concentration to 2.3 g P/m³. There is a further linear decrease in phosphate effluent concentration to 1.1 g P/m³ at 80% separate treatment of urine. These changes would prevent (or at least postpone) the need for chemical phosphate removal in the future.

**Conclusion**

This specific case shows that the optimal use of available COD is more important than source control techniques, such as separate urine collection and treatment at the plant. Even with almost complete urine separation, COD from the main wastewater stream is required for significant improvements in the effluent nitrate concentration. With a reconfigured flow scheme and 40–50% urine separation, methanol dosing would not be required.

From this study, it seems that denitrification via nitrite is effective only when nitrate is not present in excess of nitrite, in which case denitrification would still proceed mainly via the less efficient nitrate route.

In general, the technique proposed for separate urine treatment (nitritation and denitrification via nitrite) would augment the conventional central wastewater treatment plants, especially in the light of improved effluent quality and reliance on external carbon sources. If the ammonia oxidisers produced in the nitritation of urine were able to survive the activated sludge basin, the treatment of urine could significantly improve the nitrification performance of treatment plants.

**References**


