EXTERNAL NITRIFICATION WITH THE AID OF FIXED MEDIA TRICKLING FILTERS (TF) TO INCREASE THE CAPACITY OF BIOLOGICAL NUTRIENT REMOVAL (BNR) SUSPENDED MEDIUM ACTIVATED SLUDGE (AS) SYSTEMS

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EXTERNAL NITRIFICATION WITH THE AID OF FIXED MEDIA TRICKLING FILTERS TO INCREASE THE CAPACITY OF BIOLOGICAL NUTRIENT REMOVAL (BNR) SUSPENDED MEDIUM ACTIVATED SLUDGE (AS) SYSTEMS

by

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EXECUTIVE SUMMARY

1.   MOTIVATION

Biological nutrient removal (BNR) activated sludge (AS) systems have become the preferred treatment system for advanced municipal wastewater treatment in South Africa. The widespread implementation of BNRAS systems has drawn attention to some weaknesses of the system, predominantly (i) long sludge ages required for nitrification, (ii) filamentous organism bulking and (iii) the treatment/disposal of liquors/supernatants generated from sludge and solids handling. This research project focusses on the first two weaknesses, namely nitrification and bulking.

The requirement to nitrify governs the sludge age and aerated mass fraction of BNRAS systems. To achieve both nitrogen (N) and phosphorus (P) removal, a high unaerated sludge mass fraction is usually required (>40%), resulting in a reduced (<60%) aerated mass fraction. To compensate for the reduction in aerobic mass fraction, long sludge ages (around 20 to 25 days) must be selected to guarantee nitrification at the lowest expected temperature. Such long sludge ages result in large biological reactor volumes per Mℓ wastewater treated. If nitrification can be made independent of the suspended solids sludge age, then the N and P removal processes rather than nitrification will govern the system sludge age, and indications are that sludge ages can be reduced to less than half, from about 20 to 25 days to about 8 to 10 days. Such a reduction in sludge age will result in a reduction in the biological reactor volume requirement of approximately 1/3rd or alternatively, for an existing plant, in an increase in treatment capacity of some 50%, provided the secondary settling tank area requirements are accommodated.

To uncouple the suspended solids sludge age from the requirement to nitrify, the activated sludge system can be modified in two ways, by including (i) internal or (ii) external fixed media for attached nitrifier growth. The nitrifiers grow on the fixed media establishing a permanently resident population that is not subjected to either the unaerated-aerated reactor interchanges or to the suspended solids sludge age, with the result that the latter can be reduced significantly without losing nitrification. Since the effectiveness of internal fixed media has not been as good as expected and the effect of internal media on the BNRAS process has not been investigated, a BNRAS scheme in which nitrification is removed from the BNRAS system to an external fixed media system is proposed. This proposed system will be termed the external nitrification biological nutrient removal activated sludge (ENBNRAS) system.

In the proposed system, nitrification takes place outside the suspended sludge BNRAS system in a fixed media trickling filter or a similar type of system. This is achieved by including an internal settling tank (IST) following the anaerobic reactor. The mixed liquor from the anaerobic reactor enters the IST and the solids and supernatant are separated. The solids in the IST underflow are discharged to the following anoxic reactor, while the ammonia rich supernatant is discharged to the nitrifying fixed media system. The resultant nitrate rich flow is then discharged to the anoxic reactor, to recombine with the IST underflow for denitrification. The anoxic reactor discharges to the aerobic reactor, then to the secondary settling tank (SST). The SST supernatant is the system effluent, and the underflow discharges via an underflow anoxic reactor to the anaerobic reactor. In this manner, nitrification is achieved external to the activated

sludge system.

Including external nitrification has the potential to reduce the sludge age of the BNRAS system significantly. Furthermore, removing nitrification from the aerobic reactor of the BNRAS system has a number of additional benefits: (i) Minimum aerobic mass fraction for nitrification not required, (ii) the aerobic reactor volume does not have to be maintained to accommodate internal fixed media and (iii) aeration in the aerobic reactor for nitrification is no longer required, only for COD utilization. As the aerated mass fraction in the proposed system is no longer controlled by nitrification, the unaerated mass fraction can be enlarged to >60%. As a result, the anaerobic mass fraction may be increased, which should improve biological excess P removal (BEPR), and the anoxic mass fraction can be increased to improve denitrification to the point where complete denitrification may be possible. The possibility of complete denitrification at the short sludge age, together with the possibility of an aerobic mass fraction of <30%, holds promise to ameliorate anoxic/aerobic (AA) filament bulking (Casey et al., 1994). This will be particularly beneficial as it will significantly increase the treatment capacity of an existing system, or reduce secondary settling tank surface area for a proposed new system. Thus, by uncoupling the sludge age from nitrification by implementing the proposed ENBNRAS system configuration, the treatment capacity of existing BNRAS systems can be significantly increased, or bioreactor volumes and secondary settling tank surface areas in proposed systems can be significantly decreased, resulting in a major increase in system intensification with considerable economic benefit. External nitrification can be implemented at existing wastewater treatment plants (WWTPs) where old trickling filters have been extended with BNRAS systems. There are many such WWTPs, particularly in South Africa, and therefore considerable opportunities exist in South Africa for implementation of the proposed scheme and accordingly the system merits further investigation. This report summarizes this investigation: detailed reports are Moodley et al. (1999), Sötemann et al. (2000), Vermande et al. (2000), Little et al. (2001) and Hu et al. (2001).

2.   PRINCIPLE AIM AND MAIN TASKS

Before the ENBNRAS system can be implemented at full scale, it needs to be comprehensively investigated at laboratory scale, and this was the principle objective of this research contract:

- *To evaluate, investigate, develop and refine at laboratory scale the external nitrification biological nutrient removal activated sludge system.*

To achieve this objective, the following aims were identified:
(i)   To examine the effect of external nitrification on BNRAS systems.
(ii)   To examine N and P removal performance and sludge settleability of non- or partially nitrifying BNRAS systems at short sludge ages (3 to 10 days) and over a range of temperatures from 12 to 20°C at laboratory scale.
(iii) To examine BEPR performance and sludge settleability in anaerobic-anoxic (i.e. no aerobic zone) BNRAS systems at laboratory scale.
(iv)   To extend BNR simulation models to include anoxic P uptake/denitrification kinetics.

To address the aims above, a number of specific tasks were identified:

Task 1:   Experimental evaluation of the ENBNRAS system.

3.   TASK 1:   EXPERIMENTAL EVALUATION OF THE ENBNRAS SYSTEM

In this task the performance of the ENBNRAS system was evaluated at laboratory scale under strictly controlled conditions. In particular, nutrient removal (N and P) and sludge settleability were examined. A series of three separate laboratory scale experimental investigations were undertaken, collectively aimed at determining the performance of this proposed system. The experimental investigations are reported in detail by Moodley et al. (1999), Sötemann et al. (2000) and Hu et al. (2001).

3.1   Introduction

System 1:   The first experimental investigation was on the proposed ENBNRAS system with a small aerobic mass fraction (19%) and is reported in detail by Hu et al. (2001). This investigation was a feasibility study on the proposed system, and after it was shown to operate successfully, the second investigation was initiated.

System 2:   The second investigation examined the effect of varying aerobic (and consequently anoxic) mass fractions and shorter sludge age on the ENBNRAS system nutrient removal performance, and is reported in detail by Moodley et al. (2000). The ENBNRAS system was the same as that used in the first investigation, and this investigation covered two phases:
In Phase I the aerobic mass fraction was reduced from 19 to 15% and the sludge age was reduced from an initial 10 days to 8 days. Then, in Phase II, the aerobic mass fraction was increased to 30% at the expense of a reduced anoxic mass fraction.

System 3:   The third investigation was initiated to optimize N and P removal in the ENBNRAS system, and is reported in detail by Sötemann et al. (2000). This investigation covered three phases:
In Phase I the aerobic and anoxic mass fractions were equal at 32.5%, and a mixed liquor recycle between the aerobic and anoxic reactors was included because the nitrification efficiency of the external nitrification system was initially poor, and the large aerobic mass fraction resulted in significant nitrification in the aerobic reactor. In Phase II, efficient nitrification in the external nitrification system was restored and the aerobic and anoxic mass fractions were changed to 20 and 45% respectively. In Phase III the mixed liquor recycle between the aerobic and the anoxic reactors was removed.

3.2   System performance

From the investigations:

- COD removals in excess of 90% can be expected.
- A considerable reduction in oxygen demand will be obtained, approximately 60% less than in an equivalent conventional BNRAS system.
- A nitrification efficiency of approximately 85 - 90% can be expected in the external nitrification system, but this requires confirmation at full scale.
- Total N removal of 80 - 90% is possible, with an effluent TKN of about 5 mgN/ℓ, effluent nitrate between 2.5 - 5 mgN/ℓ and effluent total N between 7 and 10 mgN/ℓ. However, this will be influenced by the nitrification efficiency of the external nitrification system as well as by the influent wastewater characteristics.
- BEPR of approximately 9 - 11 mgP/ℓ can be expected, but again this depends on the wastewater characteristics. Anoxic/aerobic P uptake BEPR was consistently observed, with anoxic P uptake at 52 - 63% of the total P uptake. It seems that anoxic/aerobic P uptake BEPR is inherent to the ENBNRAS system, because the system configuration provides the conditions which favour the development of denitrifying phosphate accumulating organisms (PAOs). These conditions are (i) large anoxic mass fractions and small aerobic mass fractions, (ii) sufficiently large nitrate load on the anoxic reactor and (iii) no mixed liquor recycle between the aerobic and anoxic reactors (a - recycle). The nitrate load on the anoxic reactor appears to be the most important factor influencing the magnitude of anoxic P uptake.
- The contribution of PAOs to denitrification was found to be low, at less than 26% for all systems investigated.
- A good settling sludge can be expected, with a dilute sludge volume index (DSVI) of about 100 mℓ/gTSS.

3.3   System design and operation

From the investigations on the laboratory scale ENBNRAS systems, a number of design and operating parameters can be identified:

- The successful implementation of the ENBNRAS system hinges around the successful operation of the external nitrification system. If the nitrification efficiency of the external nitrification system is reduced, denitrification in the anoxic reactor will be reduced because of a lower nitrate load and the absence of an a-recycle. The free and saline ammonia (FSA) that is not nitrified in the external nitrification system will be nitrified in the aerobic reactor to a greater or lesser extent (depending on the aerobic mass fraction and nitrifier seeding from the external nitrification system), and this will cause a deterioration in the effluent nitrate quality. Once the nitrate concentration in the underflow exceeds the denitrification potential of the underflow anoxic reactor, nitrate will enter the anaerobic reactor and hence reduce BEPR.
- Anaerobic mass fractions of approximately 25%. However, this can be reduced if larger anoxic mass fractions are required, with a resultant reduction in BEPR (Wentzel et al., 1990).
- A minimum aerobic mass fraction of 20%. Smaller aerobic mass fractions appear to

stimulate pin-point floc, which causes a deterioration in sludge retention and effluent quality. Such aerobic mass fractions also stimulate aerobic P uptake BEPR. Inclusion and maximization of aerobic P uptake BEPR appears desirable to maximize BEPR, because aerobic P uptake BEPR is more efficient than anoxic P uptake BEPR. However, conditions that promote aerobic P uptake BEPR are also conducive to nitrifier growth. Although complete exclusion of nitrifiers from the activated sludge of the ENBNRAS system may not prove possible, it is not necessary as long as virtually complete nitrification occurs in the external nitrification system to limit nitrification in the aerobic reactor of the main system. Accepting nitrification in the aerobic reactor of the BNRAS system, this nitrification will be limited to the ammonia bypassing the external nitrification system in the internal settler underflow. This nitrate is prevented from entering the anaerobic reactor by denitrification in the underflow anoxic reactor.

- The requirements above for anaerobic and aerobic mass fractions of 25 and 20% respectively provide 55% of the total mass fraction for the anoxic reactors. Allowing 10% for the underflow anoxic reactor, 45% of the total mass fraction remains for the main anoxic reactor.

- The proportion of the internal settling tank inflow that can pass to the external nitrification system is approximately 85%, due to the constraint of sludge underflow to the anoxic reactor.

- A permanent a-recycle from the aerobic to the anoxic reactor should not be included, as this appears detrimental to BEPR and denitrification. However, provision of an a-recycle for 'emergency use' in case nitrification in the external nitrification system deteriorates may prove useful.

4. TASK 2: ENBNRAS SYSTEM PERFORMANCE AT SHORT SLUDGE AGES

4.1 Introduction

In Task 1 extensive experimental data on ENBNRAS systems with sludge ages of 10 and 8 days was collected. Towards the end of the practical laboratory investigation on System 3 (Phase III), it was decided to increase the influent sewage flow from 20 ℓ/d to 30 ℓ/d in order to observe the systems response to this 50% increase in load. However, shortly after the influent flow had been increased to 30 ℓ/d, the systems internal settling tanks began to fail hydraulically. Thus, instead of allowing the system to fail completely as a result of this hydraulic failure, it was decided to reduce the influent flow to 25 ℓ/d, and instead of implementing a gradual increase in feed, a gradual reduction in sludge age was implemented and the effect of the reduced sludge age on system performance evaluated. The system was operated at a 8 day sludge age for 49 days and a 5 day sludge age for a further 13 days. Details of this investigation are given by Sötemann *et al.* (2000).

4.2 System performance

- The overall average COD removals of the 8 and 5 day sludge age configurations of the ENBNRAS system were 93 and 90% respectively. The COD removal performances of the short sludge age systems were very good and close to COD removal observed in the systems of Task 1.

- 88 and 96% of the FSA flowing into the external nitrification system was nitrified and 93 and 95% of the total ENBNRAS system nitrification occurred externally for the 8 and 5 day sludge age configurations respectively. This demonstrates one of the main benefits of the ENBNRAS system configuration - virtually complete nitrification at 8 and 5 days sludge age with a aerobic mass fraction of only 20%. This will be attainable also at temperatures lower than 20°C.
- The TKN removals for the 8 and 5 day sludge age configurations were 94 and 92% respectively. The total N removal was 92 and 76% respectively. The discrepancies in the total N removals for the 8 and 5 day sludge age configurations were due to the difference in the influent TKN/COD ratios of the sewage batches fed to the two configurations; for the 8 day sludge age configuration this was 0.096 mgN/mgCOD, while for the 5 day sludge age configuration it was much higher at 0.120 mgN/mgCOD. The higher influent TKN/COD ratio combined with a lower denitrification potential of the 5 day sludge age configuration led to more nitrate in the effluent and therefore a lower total N removal.
- The P removal for the 8 day sludge age configuration was 14.0 mgP/ℓ influent. For the 5 day sludge age configuration this was 8.6 mgP/ℓ influent. The 8 day sludge age configuration removed 4.2 mgP/ℓ influent more than the System 3 10 day sludge age configuration, showing that there is indeed an improvement in BEPR as the sludge age is decreased, as reflected in the BEPR model of Wentzel et al. (1990). It would therefore be expected that the 5 day sludge age configuration would show a further improvement in BEPR performance. However, for the 5 day sludge age configuration, high nitrate concentrations were recycled to the underflow anoxic reactor, which was consequently overloaded, causing nitrate to enter the anaerobic reactor, which in turn caused a decrease in P removal.
- The percentage anoxic P uptake for the 8 day sludge age configuration was 47% (with a nitrate load of 20.7mgN/ℓ on the main anoxic reactor), and the overall average percentage anoxic P uptake for the 5 day sludge age configuration was 58% (with a nitrate load of 34.9 mgN/ℓ on the main anoxic reactor). This shows clearly that as the nitrate load on the main anoxic reactor increases, the percentage anoxic P uptake increases.
- The DSVI for the 8 and 5 day sludge age configurations was about 90 and 93 mℓ/gTSS respectively.

4.3   Closure

A detailed analysis of the results of the 8 and 5 day sludge age configuration was not intended, as the system performance evaluation lasted for only 62 days. However, the results show that the ENBNRAS system BNR performance in no way deteriorated at the shorter sludge ages, in fact a reduction in sludge age tends to increase N and P removal per mass of organic load (Wentzel et al., 1990), provided that it is not reduced below a lower limit of about 5 days for operational reasons (sludge flocculation, effluent turbidity); in terms of 'wash out' of PAOs the sludge age can be as low as 3 days (Wentzel et al., 1990).

5.   TASK 3:   BNR AT LOW TEMPERATURES

The ENBNRAS system is expected to be particularly beneficial at low temperatures, due to sensitivity of the nitrifiers to temperature. Accordingly, information on BNR behaviour at low temperatures needs to be evaluated. Information on this is available in the literature and from

previous investigations at UCT on filamentous organism bulking (Pilson *et al.*, 1995). Therefore this was not investigated experimentally in this research project.

6.   TASK 4:   COMPARISON OF THE ENBNRAS SYSTEM WITH A CONVENTIONAL BNRAS SYSTEM

6.1   Introduction

In implementing the ENBNRAS system, its performance with respect to nutrient removal should not be significantly reduced compared to the conventional BNRAS system. With regard to N removal via denitrification, the investigations in the previous tasks indicated that the ENBNRAS system is superior in that near complete denitrification can be achieved for influent wastewater TKN/COD ratios that are considerably higher than those for which it can be achieved in conventional BNRAS systems. However, with regard to P removal, the ENBNRAS system stimulates anoxic P uptake in the system which would appear to be reduced compared to aerobic P uptake. This indicates that the P removal performance of the ENBNRAS system may be reduced compared to the conventional BNRAS system. In this task this aspect is examined in more detail, to compare more precisely the N and P removal performance in the external nitrification and conventional BNR systems. Also, the factors stimulating anoxic P uptake in the conventional BNRAS system are investigated. This investigation is reported in detail by Vermande *et al.* (2000).

In this investigation, a laboratory scale BNRAS system (UCT configuration) with similar design and operating parameters to the ENBNRAS system (System 3, Phases II and III) was run in parallel with the System 3 ENBNRAS system. In order to compare the performance of the two systems, they were fed the same influent sewage (prepared in the same container and split equally between the two systems) for 254 days.

6.2   System performance

From the investigations:

- The COD removal achieved for the UCT and ENBNRAS systems were 93 and 94% respectively. In terms of carbonaceous material removal, the two systems performed identically.
- The overall average total oxygen demand (including nitrification) of the UCT system was 7625 mgO/d while that of the ENBNRAS system was 1798 mgO/d. By nitrifying externally, the ENBNRAS system required 76% less oxygen per day; this is a significant reduction.
- The TKN removals for the UCT and ENBNRAS systems were 86 and 87% respectively and the total N removals 78 and 88% respectively. On average, the effluent total N of the UCT system was 16.8 mgN/ℓ, of which 12.8 mgN/ℓ was NO_x and 4.0 mgN/ℓ was TKN. Of the 4.0 mgN/ℓ TKN, 1.8 mgN/ℓ was FSA. For the ENBNRAS system the effluent total N was on average 9.8 mgN/ℓ, of which 4.6 mgN/ℓ was NO_x and 5.2 mgN/ℓ was TKN. Of the 5.2 mgN/ℓ TKN, 3.5 mgN/ℓ was FSA. The ENBNRAS system achieved an average effluent total N concentration <10 mgN/ℓ while the UCT system did not.
- In the UCT system, on average, 21.3 mgP/ℓ influent P was released in the anaerobic

reactor and 34.0 mgP/ℓ uptake occurred mainly in the aerobic reactor. In the ENBNRAS system, on average, 18.3 mgP/ℓ P was released in the anaerobic reactor and internal settling tank with an additional P release (which also had to be taken up in the anoxic and aerobic reactors) of 4.5 mgP/ℓ influent in the external nitrification system, and 32.8 mgP/ℓ P uptake occurred in the anoxic and aerobic reactors.

- The average P removal for the UCT system was 12.7 mgP/ℓ influent while that of the ENBNRAS system was 9.8 mgP/ℓ influent. The UCT system showed only 9.8% anoxic P uptake on overall average, showing that predominantly aerobic P uptake BEPR occurred in the UCT system. In the ENBNRAS system, 60% of the P uptake occurred in the anoxic reactor. During 7 (out of 28) sewage batches, anoxic P uptake was induced in the UCT system by feeding sewage with high TKN/COD ratios (leading to high nitrate loads on the anoxic reactor). During this period the UCT system showed about 18% anoxic P uptake and the P removal decreased to the same level as measured in the ENBNRAS system.

- The overall average DSVI for the UCT and ENBNRAS systems were 138 mℓ/gTSS and 103 mℓ/gTSS respectively. During the period where sewage with a high TKN/COD ratio was fed to the systems to induce anoxic P uptake in the UCT system, the DSVI of the UCT system increased significantly from around 110 mℓ/gTSS to over 200 mℓ/gTSS, while the DSVI of the ENBNRAS system increased only slightly from around 90 mℓ/gTSS to around 105 mℓ/gTSS. This shows that the conventional BNRAS system reacts much more strongly to significant (>2 mgN/ℓ) nitrate concentrations in the outflow of the main anoxic reactor because its aerobic mass fraction was higher (0.5) than that of the ENBNRAS system (0.2).

6.3   Closure

In terms of carbonaceous material removal the UCT and ENBNRAS systems achieved almost identical results. For the nitrogenous material removal, the ENBNRAS system produced an effluent of better quality with an effluent total N concentration of nearly half that of the UCT system on overall average. The UCT system, which exhibited predominantly aerobic P uptake BEPR, removed about 3 mgP/ℓ influent more P than the ENBNRAS system with anoxic/aerobic P uptake BEPR did. BEPR is the only process where the UCT system achieved superior results to that of the ENBNRAS system. The ENBNRAS system BNR is effected using approximately 75% less oxygen than was required by the UCT system. The ENBNRAS system consistently produced a better settling sludge than the UCT system did, and furthermore, the ENBNRAS system did not produce a bulking sludge when high nitrate concentrations flowed from the anoxic reactor, as was observed in the UCT system.

7.   TASK 5:   IMPLEMENTATION OF THE ENBNRAS SYSTEM AT FULL SCALE (ECONOMIC EVALUATION)

7.1   Introduction

The experimental investigations into the ENBNRAS system at laboratory scale have demonstrated that this system holds considerable promise for practical implementation. However, successful implementation of the ENBNRAS system at full scale will depend largely on its cost compared to the conventional BNR activated sludge systems and its ability to meet required

effluent quality standards. Accordingly, the system should be assessed with respect to the economic cost and effluent quality standards, and these compared to those for an equivalent conventional BNRAS system. In this task such a practical case study is demonstrated based on a specific plant (Potsdam WWTP at Milnerton in Cape Town, South Africa) and the effluent quality standards of South African law. Details of the economic evaluation are reported by Little *et al.* (2001).

The economic cost will be influenced to a large extent by site specific factors. For the purpose of the evaluation, the Potsdam WWTP at Milnerton in Cape Town, South Africa was selected since considerable data are available for this plant. The existing Potsdam WWTP comprises two streams which are operated in parallel:

- The old rock media trickling filters with a capacity of 18 Ml/d. These trickling filters are low loaded and achieve good COD removal and nitrification. However, no N and P removal are obtained. A chemical P removal plant has been provided but currently (2001) this is not operated.
- The newer conventional BNRAS system (Modified UCT process) with a capacity of 17 Ml/d with full biological nutrient removal.

Therefore, the plant has a combined capacity of 35 Ml/d. The trickling filter effluent can be treated chemically with aluminium sulphate or ferric chloride for P removal and with lime for alkalinity and pH correction.

7.2   Economic evaluation

The engineering and economic evaluation by Little *et al.* (2001) provides system schemes and capital, operation and maintenance costs for 2 scenarios, (i) a 35 Ml/d greenfields scenario (assuming that no plant exists at Milnerton) and (ii) scenarios where the existing Potsdam plant is modified to treat the entire existing flow for nutrient removal, and expanded to cope with an increased flow of 49Ml/d expected in the year 2020. The costs for each scenario are given in net present values (NPV) at 6% in the year 2000 in South African Rands and include capital and operational costs for the indicated period:

Scenario 1 (35 Ml/d greenfields plant):

		Cost(million ZAR)	Period
1a)	Conventional BNRAS system (UCT process)	147.44	2000-2015
1b)	ENBNRAS system	138.25	2000-2015

Scenario 2 (Modifying the existing plant):

2a)	35 Ml/d conventional BNRAS system	93.55	2000-2015
2b)	35 Ml/d ENBNRAS system	76.50	2000-2015
2a)	49 Ml/d conventional BNRAS system	164.80	2000-2020
2b)	49 Ml/d ENBNRAS system	154.27	2000-2020

7.3 Discussion

When the project was initiated, it was thought that a significant saving in capital cost would be achieved for the ENBNRAS system as a result of the increase in treatment capacity or smaller biological reactors, the reduction in oxygen demand and a better settling sludge. These appeared to make the ENBNRAS system an attractive and viable alternative as a full scale wastewater treatment plant. However, the economic evaluation indicated that the savings may not be as large as initially expected.

Although the ENBNRAS system does provide a cost saving in construction costs of about 30% compared to a conventional BNR activated sludge system, the operating and maintenance costs in the long term mask this saving, because these costs are around 70% of the overall cost and virtually the same for both ENBNRAS systems and conventional BNR activated sludge systems. While significant savings in operation costs are made from the very low oxygen demand, the increased sludge production at a shorter sludge age and the associated increased cost in sludge treatment, transport and disposal offsets this saving. Although the total NPV (capital, operation and maintenance) for the ENBNRAS system is 5 to 10 % lower than that of a conventional BNR activated sludge system, this difference may not be large enough to favour selection of the new and untested ENBNRAS system over the tried and trusted conventional BNR activated sludge system. It must be remembered, however, that the sludge treatment route is specific to the Potsdam WWTP (disposal to landfill) and is a relatively costly alternative and contributes significantly to the total cost. For other WWTPs, if sludge treatment and disposal costs can be reduced, then this will make the ENBNRAS system more economically attractive.

The most significant advantage of the ENBNRAS system is that it offers biological N and P removal for wastewater treatment plants with existing trickling filters and activated sludge units without an increase in existing process units, enabling these plants to meet the stricter new effluent quality standards proposed under the National Water Act of 1998 by the Department of Water Affairs and Forestry. The ENBNRAS system will provide a feasible and economical upgrading option. Although the ENBNRAS system does not provide a large enough savings in monetary terms to make it as an attractive system, the new effluent quality standards may favour implementation of the ENBNRAS system, because the ENBNRAS system is capable of producing effluents with a quality comparable with (and for effluent N better than) conventional BNR activated sludge plants. Therefore, it is anticipated that the proposed new effluent quality standards will be the driving force that promotes application of the ENBNRAS system at full-scale, i.e. implementation probably will be legislation driven.

8. TASK 6: MODEL DEVELOPMENT FOR THE ENBNRAS SYSTEM

8.1 Introduction

For the design and operation of, and research into, conventional biological wastewater treatment systems, mathematical simulation models have proved to be invaluable as a process evaluation tool. Mathematical simulation models provide quantitative descriptions of the dynamic behaviour of the wastewater treatment system. By providing quantitative descriptions, they allow predictions of the system response and performance to be made. From these predictions, design and operational criteria can be identified for optimization of system performance. Also,

mathematical models are very useful as research tools. By evaluating model predictions, it is possible to test hypotheses on the behaviour of the wastewater treatment system in a consistent and integrated fashion. This may direct attention to issues not obvious from the physical system and lead to deeper understanding of the fundamental behaviour patterns controlling the system response. Recognising the usefulness of mathematical models, it was decided to develop a kinetic simulation model that includes the processes that can be expected to be operative in ENBNRAS systems. For the full detail of this development, the reader is referred to Hu *et al.* (2001).

The approach taken to develop the kinetic simulation model for the ENBNRAS system was to: (i) evaluate and compare the available existing kinetic models for BNR activated sludge systems; (ii) identify difficulties and deficiencies in the models when applied to BNR activated sludge systems in general, and to the proposed ENBNRAS system in particular; (iii) gather information from the literature and experiments to overcome the difficulties and deficiencies to develop the kinetic simulation model; (iv) gather data from the literature to calibrate and validate the resultant kinetic model; and (v) apply the model to simulate the behaviour of the proposed system. State of the art models identified for evaluation were: UCTPHO (Wentzel *et al.*, 1992), IAWQ Activated Sludge Model (ASM) No2 (Henze *et al.*, 1995) and ASMNo2d (Henze *et al.*, 1998). BIOWIN model of Barker and Dold (1997) and the Delft group of models (Smolders *et al.*,1994 ; Kuba *et al.*,1997). In the initial assessment of these models and the ENBNRAS system, one process immediately apparent to be critically important was anoxic P uptake and denitrification by PAOs, and attention was focussed on this aspect.

8.2 Model development

The UCTPHO kinetic model was modified to extend application to conventional BNRAS systems where anoxic P uptake occurs and particularly to the ENBNRAS system. For these modifications information was drawn from experimental data in the literature and the other available kinetic models (e.g. ASM2d and BIOWIN). The approach followed was to include a single PAO population with reduced rates of substrate utilization/growth under anoxic conditions compared to aerobic conditions. Within this basic approach, the main modifications made to UCTPHO were:

- Processes for anoxic growth of PAOs with associated anoxic P uptake and denitrification were included, with separate yield coefficients for anoxic and aerobic PAO growth and separate ratios for P uptake per unit PHB substrate utilized. The anoxic rates were reduced compared to the equivalent aerobic rates, by multiplying the aerobic rates by a reduction factor under anoxic conditions.
- Processes for anoxic death/maintenance of the PAOs were included, recognizing that under anoxic conditions PAOs that can denitrify will behave differently from those that cannot.
- Separate anoxic and aerobic yield coefficients were introduced for ordinary heterotrophic organism growth.
- The organic N and P fractions were linked to the corresponding COD fractions, and the transformations between the different forms of organic N and P fractions were linked to the corresponding COD transformations (Henze *et al.*, 1995).

In order to calibrate and validate this kinetic model, it was applied to a wide variety of

conventional BNRAS systems operated at laboratory scale in the UCT laboratory over a range of influent concentrations, system configurations, sludge ages, mass fractions and recycle ratios. The main objectives of this application were to evaluate the predictive capacity of the model, evaluate values for the model parameters and calibrate those parameters for which values were not available. With essentially a single set of constants, close correlation between predicted and measured performance was obtained for most of the systems, thereby validating the model.

8.3   Application of the model to ENBNRAS systems

The kinetic model was applied to the laboratory scale ENBNRAS systems operated for the previous tasks and predicted behaviour compared to the experimental observations. This showed that the model is capable of modelling ENBNRAS system behaviour with the default values for kinetic and stoichiometric parameters proposed, except for $f_{p,rel}$, η_H, η_G and K_{mp}. This provides substantive evidence supporting the validity of the proposed kinetic model. In the model application $f_{p,rel}$ had to be reduced from the default value of 0.5 to 0.4. The η_G value needs to be determined by simulation for each system; however, as the PAOs contribution to denitrification is small, a default value of $\eta_G = 0.5$ can be used for ENBNRAS systems with little error. The K_{mp} value had to be increased significantly in the application to ENBNRAS systems, in order to achieve improved OUR and VSS predictions. The η_H value had to be calibrated in association with the K_{mp} value, and the values for η_H appeared to be influenced significantly by the nitrate load on the anoxic reactor.

8.4   Closure

A kinetic model describing the behaviour of ENBNRAS systems (and BNRAS systems in general) has been developed, including ordinary heterotrophic organisms (OHOs), autotrophic organisms (AOs) and phosphorus accumulating organisms (PAOs). The process kinetics and stoichiometry for the OHOs and AOs were taken largely unmodified from the UCTPHO kinetic simulation model (Wentzel et al., 1992), except that the transformations in N and P compounds were linked to the equivalent transformations in the COD compounds (Henze et al., 1995). For the PAOs, the anaerobic and aerobic kinetic and stoichiometric behaviour were also taken from UCTPHO, but additionally kinetics and stoichiometry for anoxic PAO growth and death (with associated anoxic P uptake and PAO denitrification) were included, based on information available in the literature.

The model was calibrated and applied to an extensive data set from conventional and external nitrification BNR activated sludge systems collected in the UCT laboratory in this and previous WRC funded contracts. The correlation between observed and predicted results was good for the wide variety of systems (including ENBNRAS systems), except for conventional BNR activated sludge systems with small aerobic mass fractions.

This model provides a useful tool for the design and operation of ENBNRAS systems.

9.   CLOSURE AND RECOMMENDATIONS

The investigations on the three laboratory scale ENBNRAS systems provide a comprehensive framework for the understanding of the ENBNRAS system operation and performance, and any further laboratory investigations would not provide more knowledge and understanding. The next step would be to begin full scale trials of an ENBNRAS system. To begin with, a full scale trickling filter would have to be converted into a nitrifying trickling filter to ascertain its performance as a nitrifying trickling filter at full scale. Once it has been proven that existing full scale trickling filters can successfully be converted to nitrifying trickling filters and their capacity determined, the trickling filters can be integrated into a BNR activated sludge system in an ENBNRAS system configuration to obtain BNR on the full influent wastewater flow.

Initially it was thought that the savings in capital cost brought about by an increased capacity or smaller biological reactors, reduced oxygen demand and better settling sludge would make the ENBNRAS system an attractive and viable alternative as a full scale plant. However, Task 5 (economic evaluation) indicates that this may not be the case. While the ENBNRAS system alternative does provide a saving in construction costs of about 30% when compared to a conventional BNR activated sludge system, the operating costs in the long run overshadow this saving. The operating costs of a WWTP, whether ENBNRAS or conventional BNR activated sludge system, account for the bulk of the NPV. While significant savings in operation costs are made from the very low oxygen demand, the increased sludge production at the shorter sludge ages and the associated increase in sludge treatment, transport and disposal costs reduce these savings (However, it should be remembered that the sludge disposal costs for the case study selected WWTP are high).While the total NPV (capital, operation and maintenance) for the ENBNRAS system option is 5 to 10% lower than that of a conventional BNR activated sludge system, this difference may not be large enough for a definite choice of the ENBNRAS system over the conventional BNR system. However, the most significant advantage is that the ENBNRAS system offers biological N and P removal for the full wastewater flow without an increase in existing process units. In South Africa, if the Department of Water Affairs and Forestry implement the proposed new effluent quality standards proposed under the National Water Act of 1998, the ENBNRAS system will provide a feasible and economical plant upgrade option. The ENBNRAS system is capable of producing effluents with a quality that are within the new effluent quality standards, especially with regards to nitrogen. Thus, the proposed new effluent quality standards rather than economics may well be the driving force that will see the ENBNRAS system implemented at full scale.

From the discussion above, the next step in development of the ENBNRAS system is implementation and evaluation at full scale. In particular the behaviour of nitrifying trickling filters needs to be assessed, and this is best done at full scale. This will form the basis for a new research contract between UCT and the WRC and WSSA.

10. ADDITIONAL PRODUCTS

In addition to the research results described above, the following were produced.

10.1 Publications

- Articles in refereed journals: 3 of direct relevance, and 9 indirect.
- Articles in conference proceedings: 4 of direct relevance, and 9 indirect.
- Research reports: 5 of direct relevance, and 3 indirect.

10.2 Postgraduates

- 1 PhD student, completed.
- 2 MSc students, completed.
- 1 MSc student, in preparation.

10.3 Undergraduates

- 3 BSc (Eng) undergraduate students completed their 9 week final year thesis in this research project.

10.4 Foreign students

- 1 BSc student from Toulouse (France) completed 6 month practical training on the research project.

ACKNOWLEDGEMENTS

TABLE OF CONTENTS

LIST OF SYMBOLS AND ABBREVIATIONS

Symbol/abbreviation	Description
AO	Autotrophic organisms
ASM	Activated sludge model
BEPR	Biological excess phosphorus removal
BNR	Biological nutrient removal
COD	Chemical oxygen demand
DPAO	Denitrifying phosphate accumulating organisms
DSVI	Diluted sludge volume index
DWAF	Department of Water Affairs and Forestry
EN	External nitrification
ENBNR	External nitrification biological nutrient removal
ENBNRAS	External nitrification biological nutrient removal activated sludge
f_{CV}	Mixed liquor COD/VSS ratio
f_N	Mixed liquor TKN/VSS ratio
FSA	Free and saline ammonia
IAWPRC	International Association for Water Pollution Research and Control
IAWQ	International Association on Water Quality (formerly IAWPRC)
IST	Internal settling tank
OHO	"Ordinary" heterotrophic organisms
MLOSS	Mixed liquor organic suspended solids
MLSS	Mixed liquor suspended solids
N	Nitrogen
ND	Nitrification denitrification
NDBEPR	Nitrification denitrification biological excess phosphorus removal
NNBEPR	Non nitrifying biological excess phosphorus removal
NPV	Net present value
OHO	Ordinary heterotrophic organisms (non-P removal)
OUR	Oxygen utilization rate
P	Phosphorus
PAO	Phosphate accumulating organisms
PHA	Polyhydroxyalkanoate
PST	Primary settling tank
RBCOD	Readily biodegradable COD
SBCOD	Slowly biodegradable COD
SBR	Sequencing batch reactor
SCFA	Short-chain fatty acids
SVI	Sludge volume index
TF	Trickling filter
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TSS	Total suspended solids
UCT	University of Cape Town
VSS	Volatile suspended solids

WAS	Waste activated sludge
WRC	Water Research Commission
WWTP	Wastewater treatment plant

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CHAPTER 1

INTRODUCTION

1.1 MOTIVATION FOR THE RESEARCH

The biological nutrient removal (BNR) activated sludge (AS) system has become an established technology in wastewater treatment practice. This development has been facilitated by an improved understanding of the nitrification denitrification (ND) and biological excess phosphorus removal (BEPR) processes. However, implementation of BNRAS systems has brought with it a new set of difficulties (Ekama and Wentzel, 1997), the main ones being the long sludge age required for nitrification, filamentous organism bulking and the treatment/disposal of liquors/supernatants generated from sludge and solids handling. The last named received attention under a previous Water Research Commission contract (K5/692); this research project focusses on the first two, namely nitrification and bulking.

In the BNRAS system, the requirement to nitrify governs selection of the two linked design parameters, sludge age and aerated mass fraction. The need for nitrogen (N) and phosphorus (P) removal sets a requirement for an unaerated sludge mass fraction (anaerobic for P removal and anoxic for N removal). In N and P removal plants, the unaerated sludge mass fraction usually needs to be high, i.e. > 40%, causing the aerated mass fraction to be reduced, i.e. < 60%. To compensate, long sludge ages need to be selected to ensure nitrification. For example, with maximum specific growth rates for the nitrifiers at $20\,^{\circ}C$ (μ_{nm20}) of around 0.45/d, to guarantee nitrification at the minimum temperature of $14\,^{\circ}C$, the sludge age of the single sludge system must be around 20 to 25 days if 50 to 60% of the sludge mass in the system is unaerated. Such long sludge ages result in large biological reactors per Mℓ wastewater (WW) treated. If nitrification can be made independent of the suspended solids sludge age, then selection of the sludge age no longer will be governed by the requirement to nitrify, but rather by the N removal (denitrification) and P removal (BEPR) processes. For both these biological processes, a reduction in sludge age increases respectively the N and P removal per mass organic load (WRC, 1984; Wentzel et al., 1990), provided the sludge age remains longer than some lower limit to prevent "wash-out" of P removal and denitrifying organisms. Indications are that, if nitrification and the sludge age are uncoupled, then the sludge age can be reduced to less than half, from about 20 to 25 days to about 8 to 10 days. This will result in a reduction in the biological reactor volume requirement per Mℓ WW treated of about 1/3rd, or alternatively, in an increase in WW treatment capacity of some 50% (provided secondary settling tank area requirements are accommodated).

BNRAS systems have been found to promote the growth of a specific group of filamentous microorganisms, previously called low F/M (Jenkins et al., 1984) but renamed anoxic/aerobic (AA) (Casey et al., 1994). These filamentous organisms cause sludges to settle poorly, resulting in an increase in the required surface area of secondary settling tanks. If the settleability of BNRAS sludges can be improved, then the flow through existing secondary settling tanks can be increased considerably, or alternatively for a proposed system, the secondary settling tank surface area can be considerably reduced, both options providing considerable economic benefit. Casey et al. (1994) have identified two main causes for AA filament proliferation in BNRAS

systems:

- Aerated sludge mass fractions in the range 30 to 50%.
- Incomplete denitrification in the anoxic reactor preceding the aerobic reactor.

The uncoupling of nitrification and the sludge age will introduce greater flexibility into the BNR activated sludge system configuration and should allow elimination of the two conditions identified above to promote AA filament growth.

Thus, uncoupling the sludge age from nitrification will significantly increase the treatment capacity of existing BNRAS systems, or reduce bioreactor volumes and secondary settling tanks surface area in proposed systems, resulting in a major step increase in system intensification with considerable economic benefit - this is the main research focus in this project.

1.2   THE PROPOSED SYSTEM

To uncouple the suspended solids sludge age from the requirement to nitrify, the activated sludge system can be modified in two ways, by including (1) internal or (2) external fixed media for attached nitrifier growth. Internal fixed media such as Ringlace[R] and floating tubes or small sponges have been placed in the aerobic reactor of the BNRAS system (Sen et al., 1994, 1995; Randall and Sen, 1996). The nitrifiers grow on the fixed media, establishing a population permanently resident in the aerobic reactor. These nitrifiers are not subjected to either the unaerated-aerated reactor interchanges or to the suspended solids sludge age, with the result that the latter can be reduced significantly without losing nitrification. However, the effectiveness of the internal fixed media has not been as good as expected, it yields a low cost/benefit ratio and the effect of the internal fixed media on BEPR has not been investigated. To resolve these difficulties, a BNRAS scheme in which nitrification is removed from the BNRAS system to an external fixed media system is proposed. This proposed system will be termed the *external nitrification biological nutrient removal activated sludge (ENBNRAS)* system.

In the proposed system, nitrification takes place outside the suspended sludge BNRAS system in a fixed media trickling filter or similar type of system. This has the potential to reduce the sludge age of the BNRAS system significantly. Furthermore, removing nitrification from the aerobic reactor of the BNRAS system has a number of additional benefits:

1. Minimum aerobic mass fraction for nitrification not required.
2. Aerobic reactor volume (and accordingly mass fraction) does not have to be maintained to accommodate internal fixed media.
3. Aeration for nitrification in the aerobic reactor is not required, only for COD utilization.

As a consequence of 1 and 2 above, the aerated mass fraction in the proposed system is no longer controlled by nitrification and therefore the unaerated mass fraction can be large (> 60%). This would have two main benefits:

1. The anaerobic mass fraction can be increased, which should improve BEPR (Wentzel et al., 1990).
2. The anoxic mass fraction can be increased, to improve denitrification and thus N removal (WRC, 1984) to the point where complete denitrification may be possible.

1.3

The possibility of complete denitrification with the short sludge age holds promise to ameliorate AA filament bulking (Casey *et al.*, 1994). As noted above, this will be particularly beneficial as it will significantly increase the treatment capacity of an existing system, or reduce secondary settling tank surface area for a proposed system.

The DEPHANOX system (Fig 1.1) is one in which nitrification takes place externally to the BNR activated sludge system (Bortone *et al.*, 1996; Sorm *et al.*, 1996). In this system the influent wastewater is discharged to the anaerobic reactor to maximize BEPR. After the anaerobic reactor, the sludge mass is separated from the liquid in an internal settling tank and discharged to the anoxic reactor. The internal settling tank overflow, which has a high ammonia concentration, is passed through a fixed medium reactor wherein nitrification takes place. The nitrified outflow from the fixed medium reactor is discharged to the anoxic reactor for denitrification. From the anoxic reactor, the mixed liquor passes to the last reactor which is aerobic. After the aerobic reactor, the activated sludge is separated from the treated wastewater in a final secondary settling tank. The final settling tank overflow is the effluent from the system and the settled sludge is returned to the anaerobic reactor.

Fig 1.1: The DEPHANOX biological nutrient removal system (after Bortone *et al.*, 1996; Sorm *et al.*, 1996).

It appears that the DEPHANOX system was developed with the specific objective of stimulating denitrification by BEPR organisms, generically called phosphate accumulating organisms (PAO). Using PAOs for denitrification has the advantage that the influent wastewater substrate sequestered by the PAO in the anaerobic reactor (and therefore implicated in BEPR) also is used for denitrification (and therefore N removal). Some laboratory and pilot scale experimental work has been done on the DEPHANOX system by Bortone *et al.* (1996) and Sorm *et al.* (1996). They found considerable P uptake in the anoxic reactor, indicating that denitrifying PAOs (DPAOs) did participate in the denitrification process. Also, improved sludge settleability (SVIs ~ 50 mℓ/g) have been consistently observed in a laboratory scale DEPHANOX system by Sorm *et al.* (1996).

Thus, it would appear that the DEPHANOX system holds considerable promise, producing nutrient (N and P) removal and a sludge that settles well. However, system intensification does not appear to have been a consideration in the development of and investigations into this system.

Clearly this aspect is of major benefit and requires extensive investigation.

1.3 OPPORTUNITIES FOR IMPLEMENTATION OF THE PROPOSED SYSTEM

External nitrification could be achieved at wastewater treatment plants (WWTPs) where old trickling filter (TF) plants have been extended with a BNRAS system. There are many such WWTPs, particularly in South Africa. For example, East Rand Water (ERWAT) treats in excess of 500 Mℓ/d, more than 50% (270 Mℓ/d) of which is treated in old TF plants. Some 200 Mℓ/d is treated in 3 combined TF/BNR activated sludge plants, of which 102 Mℓ/d is treated in old TFs with chemical P removal, the balance in BNRAS systems. Also, 168 Mℓ/d is treated in TF only plants, which at some time in the future need to be extended. In the SE Highveld region, some 10 TF/BNRAS plants treat more than 100 Mℓ/d, 25% of which is treated in old TFs. Then there are Krugersdorp (Gauteng), Daspoort (Gauteng), Rooiwal (Gauteng), Darvill (Kwazulu/Natal), and Athlone and Milnerton (W Cape) treating around 450 Mℓ/d with a TF treatment capacity in excess of 200 Mℓ/d. At these and similar WWTPs, to retain the benefit of the old TFs, a proportion of influent WW is passed through the TFs and the effluents are (Fig 1.2):

(1)   Discharged to the BNRAS system for biological N and P removal (e.g. van Huyssteen *et al.*, 1990). This in effect increases the TKN/COD and P/COD ratios of the WW discharged to the BNRAS system and hence increases the effluent nitrate and phosphate concentration.

(2)   Chemically treated to precipitate the P before discharge to the BNRAS system. This is not only costly, but also reduces the alkalinity of the water and only reduces the effective P/COD ratio of the wastewater on the BNRAS system, the nitrogen returned is not reduced.

(3)   Irrigated on land at the WWTP. This practice is being closely monitored by Department of Water Affairs and Forestry in South Africa and is expected to be substantially disallowed because it leads to a significant loss of valuable surface water.

If, instead of the above three strategies, the nitrification process is removed from the BNR activated sludge system and transferred to the TFs, all the WW flow can be discharged to the BNRAS system (Fig 1.3): A side-stream of mixed liquor is taken from the end of the anaerobic zone and passed through the TF 'humus' tanks (upgraded to internal secondary settling tanks) to remove the activated sludge solids. The underflow sludge is discharged to the beginning of the anoxic zone and the overflow is passed onto the TF for nitrification. The nitrified TF effluent is then discharged to the anoxic zone for denitrification. In this way the TF assists the BNR activated sludge system in its area of weakness, i.e. nitrification, rather than taking away from its strength, i.e. biological N and P removal with influent organics. Furthermore, the oxygen demand in the aerobic reactor is markedly reduced because nitrification no longer takes place there. Indeed, not only is the nitrification oxygen demand obtained "free" outside the BNR activated sludge system, but the oxygen equivalent of the nitrate generated in the trickling filter also helps to reduce the carbonaceous oxygen demand in the BNRAS system by about a 1/3rd. In fact, with external nitrification, the reduction in oxygen demand in the BNRAS system is much greater than when 1 3rd of the wastewater is bypassed to the trickling filter as in existing TF BNRAS systems. Therefore, by changing the TF to a nitrifying system, the treatment capacity of the BNRAS plant is increased without having to increase aeration capacity. If a TF plant is not available, it is possible to construct an artificial external fixed media system, the cost of which may be offset by the increase in WW treatment capacity.

Fig 1.2: Conventional integration of trickling filters with biological nutrient removal activated sludge systems.

Fig 1.3: Proposed integration of trickling filters with biological nutrient removal activated sludge systems: Nitrification is achieved externally on nitrifying trickling filters.

1.6

At short sludge ages and small aerobic mass fractions, nitrifiers would not ordinarily be supported in the BNRAS system. However, it will not be possible to completely exclude nitrifiers from the system because nitrifiers are likely to be seeded into the system from the TF effluent. Therefore, the potential for nitrification in the aerobic reactor will always exist in the system, and the potential nitrate concentration in the aerobic reactor will be governed by the ammonia concentration that enters it. Provided the TF nitrifies well, this nitrate concentration will be mainly from the ammonia which bypasses the TF via the internal settling tank underflow and therefore will be relatively low. If the TF does not nitrify well and the residual ammonia concentration from it is high, then, if sufficient nitrifiers are present in the aerobic reactor, the nitrate concentration will be high, with the result that a significant nitrate concentration will be present in the underflow from the final settling tank. To protect the BEPR against this potential nitrate ingress to the anaerobic reactor, a pre-anoxic (underflow anoxic) reactor is placed in the underflow to denitrify the nitrate (Fig 1.3). If sufficient nitrifiers are not present in the aerobic reactor, then the ammonia concentration in the aerobic reactor will only be partially nitrified, with the result that return sludge nitrate concentration will be relatively low, but the effluent TKN concentration will be high - the concentration depending on the nitrification efficiency of the TF.

Tertiary nitrifying trickling filters, which are employed for nitrification only and negligible organic material removal, are fairly common in USA (Lutz et al., 1990). While certain problems with macro fauna (snails, worms, larvae and flies), which reduce nitrification rates, have been encountered, high removals of ammonia have been economically achieved in tertiary nitrifying trickling filters (Parker et al.,1989;1995;1996). Therefore, while some full scale trials would need to be done to determine how rock media trickling filters can be adapted to fulfill the external nitrification function, from the USA experience this is not expected to be a major problem in implementing the external nitrification scheme. Depending on the TF treatment capacity at the WWTP, it may be necessary to replace the rock media with plastic media to achieve the required nitrification capacity.

Thus, considerable opportunities exist in South Africa for implementation of the proposed scheme, and accordingly the system merits further investigation.

1.4   OBJECTIVES AND AIMS OF RESEARCH

Before the ENBNRAS system can be implemented at full-scale, it needs to be comprehensively investigated at laboratory-scale and this is the principle objective of this research contract:
- To evaluate, investigate, develop and refine at laboratory-scale the external nitrification biological nutrient removal activated sludge (ENBNRAS) system.

To achieve this objective, the following aims were identified:
1. To examine the effect of external nitrification on BNRAS systems.
2. To examine N & P removal performance and sludge settleability of non- or partially nitrifying BNR activated sludge systems at short sludge ages (5 to 10 days) and over temperature range 12 to 20°C at laboratory-scale.
3. To examine BEPR performance and sludge settleability in anaerobic-anoxic zone (i.e. no aerobic zone) BNRAS systems at laboratory-scale.
4. To extend BNR simulation models to include anoxic phosphate uptake denitrification kinetics.

1.5   SPECIFIC TASKS

To address the aims above, a number of specific tasks have been identified for completion.

Task 1:   Experimental evaluation of the external nitrification BNR system

The behaviour and performance of the external nitrification BNR activated sludge system need to be assessed. In this task the performance of the system is evaluated at laboratory-scale under strictly controlled conditions. In particular, nutrient removal (both N and P) and sludge settleability are examined. The experimental investigation is summarized in Chapter 3, and reported in detail by Moodley et al. (1999), Sötemann et al. (2000) and Hu et al. (2001). In this investigation a number of sub-tasks were identified for completion.

Task 1.1:   External nitrification BNR system with small aerobic mass fractions

By removing nitrification from the aerobic reactor of the BNR activated sludge system, the mass fraction of this reactor can be considerably reduced, perhaps even to the extreme of eliminating this reactor altogether except for a small reaeration tank before final settling. In this task the external nitrification BNR system is operated with a small aerobic mass fraction and nutrient removal and sludge settleability monitored, to assess system behaviour and performance.

Task 1.2:   Aerobic P uptake in external nitrification BNR systems

In evaluating the performance of the external nitrification BNR system with small aerobic mass fractions (Task 1.1), it appeared that anoxic P uptake caused BEPR to be reduced compared to aerobic P uptake. Also, denitrification in the anoxic reactor was complete, with an "excess" denitrification potential for the influent wastewater TKN; from a denitrification point of view the anoxic reactor was in effect over designed. Furthermore, in attempting to develop anoxic P uptake enhanced cultures (Tasks 1.4 and 6), it was not found possible to develop a stable culture and the anoxic P uptake process by itself was considered unreliable and unstable. This lead to a re-evaluation of the system configuration, with a realization that over emphasis had been placed on anoxic P uptake. If significant aerobic P uptake could be stimulated in the system, BEPR would be improved. In this task, aerobic P uptake in the external nitrification system is investigated.

Task 1.3:   Nutrient removal in the external nitrification BNR system with variable aerobic and anoxic mass fractions

From the investigation into aerobic P uptake in the external nitrification BNR activated sludge system, it was evident that the aerobic P uptake stabilized P removal. However, initially the investigation was hindered by the difficulties experienced with nitrification in the fixed media stone column, particularly due to infestation of the trickling filter fly *Psychoda*. To resolve this difficulty, the stone column was replaced with a suspended media activated sludge system for nitrification. This proved successful and enabled consistent nitrification to be achieved, with the result that nutrient removal performance could be investigated more consistently and comprehensively. Following the successful resolution of the external nitrification, a more extensive investigation has been conducted to examine the effect of variable aerobic and anoxic

mass fractions on the nutrient removal performance in the external nitrification BNR system. In particular, the denitrification performance in the system and its interrelationship with P removal are investigated.

Task 1.4:   Anoxic P uptake behaviour

Central to the external nitrification system is the behaviour in the anoxic reactor. In the system the mass faction of this reactor is increased at the "expense" of the aerobic mass fraction. From experimental work on the DEPHANOX system, this is expected to stimulate denitrification by PAOs, with anoxic P uptake. However, little quantitative information exists on anoxic P uptake. Qualitative indications are that anoxic P uptake is not as efficient as aerobic P uptake (Ekama and Wentzel, 1997) and causes reduced BEPR. It was hoped to collect information on anoxic P uptake processes by developing enhanced cultures of anoxic P uptake PAOs. However, the enhanced cultures did not prove successful and were abandoned (for details see Hu *et al.*, 2001). Accordingly, in this task behaviour of anoxic P uptake is investigated as part of the investigation into the external nitrification BNR activated sludge system (see Task 4 also).

Task 2:   External nitrification BNR systems at short sludge ages

With external nitrification, the BNR activated sludge system can be operated at short sludge ages. Information on BNR behaviour at sludge ages of 10 and 8 days has been collected in Task 1 above. Information on external nitrification BNR behaviour at short sludge ages in the range 5 to 8 days has been collected, and is summarised in Chapter 4, and reported in detail by Sötemann *et al.* (2000). Information on BNR at very short sludge ages in the range 3 to 6 days is available from a previous investigation at UCT (Burke *et al.*, 1984).

Task 3:   BNR at low temperatures

The external nitrification system is expected to be particularly beneficial at low temperatures, due to sensitivity of the nitrifiers to temperature. Accordingly, information on BNR behaviour at low temperatures needs to be evaluated. Information on this is available in the literature, and from previous investigations at UCT on filamentous organism bulking (Pilson *et al.*, 1995). Accordingly, this was not investigated experimentally in this research project.

Task 4:   Comparison of the external nitrification BNR with a conventional BNR activated sludge system

In implementing the external nitrification BNR activated sludge system, its performance with respect to nutrient removal should not be significantly reduced compared to the conventional BNR activated sludge system. With regard to nitrogen removal via denitrification, the investigations above indicate that the external nitrification BNR system is superior in that complete denitrification can be achieved for influent wastewater TKN/COD ratios that are considerably higher than those for which it can be achieved in conventional BNR systems. However, with regard to P removal, the external nitrification BNR system stimulates anoxic P uptake in the system which would appear to be reduced compared to aerobic P uptake (Task 1.4 above). This indicated that the P removal performance of the external nitrification BNR system may be reduced compared to the conventional system. In this task this aspect is examined in more

detail, to compare more precisely the N and P removal performance in the external nitrification and conventional BNR activated sludge systems. Also, the factors stimulating anoxic P uptake in the conventional BNR activated sludge system are investigated.

This task is summarised in Chapter 5, and reported in detail by Vermande *et al.* (2000).

Task 5: Implementation of external nitrification BNR systems at full-scale

Successful implementation of the external nitrification BNR activated sludge system will depend largely on its cost compared to conventional BNR activated sludge systems and its ability to meet proposed new effluent quality standards. In this task an economic evaluation of implementing the external nitrification BNR system is undertaken, and the costs compared to those for an equivalent conventional BNR system. The Milnerton (Potsdam) Wastewater Treatment Plant is selected for an economic evaluation of the external nitrification system. Also, the expected effluent quality of conventional and external nitrification BNR systems are evaluated against the proposed new effluent quality standards.

This task is summarised in Chapter 6, and reported in detail by Little *et al.* (2001) and Hu *et al.* (2001).

Task 6: Model development for external nitrification systems

For the design and operation of, and research into conventional biological wastewater treatment systems, mathematical models have proved to be invaluable as a process evaluation tool (e.g. Dold *et al.*, 1980, 1991; Henze *et al.*, 1987). From model predictions, design and operational criteria can be identified for optimization of system performance. Also, mathematical models are very useful as research tools. By evaluating model predictions, it is possible to test hypotheses on the behaviour of the wastewater treatment system (e.g. biological processes, their response to system constraints, etc.) in a consistent and integrated fashion. This may direct attention to issues not obvious from the physical system and lead to deeper understanding of the fundamental behavioural patterns controlling the system response. In essence, mathematical models can provide a defined framework which can direct thinking (design, operation or research).

Recognising the usefulness of mathematical models, it was decided to develop a model that will include the processes that can be expected to be operative in external nitrification BNR activated sludge systems.

As noted above (Task 1.4), it was hoped to collect information on anoxic P uptake processes for the model by developing enhanced cultures of anoxic P uptake PAOs. However, the enhanced cultures did not prove successful and had to be abandoned (for details, see Hu *et al.*, 2001). Accordingly, information on anoxic P uptake processes has been collected from the literature and from previous investigations in the UCT laboratory on BNR. Existing models for BNR are evaluated and the most well established model selected and modified to incorporate anoxic P uptake and other processes of importance. This model is calibrated and validated against the literature data, and then applied to the external nitrification BNR systems described in Task 1 above.

This task is summarised in Chapter 7, and reported in deta¹ by Hu *et al.* (2001).

CHAPTER 2

LITERATURE REVIEW

2.1 INTRODUCTION

In Chapter 1, an external nitrification BNRAS system (ENBNRAS) has been proposed. To provide guidelines for the research into the proposed system, a brief literature survey will be presented in this chapter, to review aspects relevant to the proposed ENBNRAS system. The review is focussed on two areas:

- The first concerns the biological processes, i.e. nitrification, denitrification, and BEPR as well as AA filamentous bulking.

- The second is on modelling BNRAS systems.

2.2 NITRIFICATION

Nitrification is the biological process whereby ammonia is oxidised to nitrate in the presence of oxygen, and is mediated by the organism group conventionally termed nitrifiers, or more recently autotrophic organisms (AO). Nitrifiers can grow in either suspended media, i.e. suspended activated sludge system, or fixed media such as trickling filters. In this section, nitrification behaviour in the different media systems will be evaluated and compared, based on information in the literature.

2.2.1 Nitrification in suspended media BNRAS systems

Nitrification is the biological process whereby free and saline ammonia is oxidized to nitrate by nitrifying organisms. These organisms are chemical autotrophs and have characteristics that differ significantly from the heterotrophs. In particular, they are slow growing and have a maximum specific growth rate only about 1/10th of that of heterotrophs. They utilize ammonia to obtain energy for cell synthesis, and dissolved inorganic carbon for their carbon requirements. The ammonia required for cell synthesis is negligible (<2%) compared to the ammonia converted to nitrate for obtaining energy. This allows the nitrifiers to be considered simply as a catalyst in the stoichiometric reactions for nitrification.

A number of factors influence nitrification, viz. influent source, temperature, pH and alkalinity, unaerated zones, dissolved oxygen (DO) concentration and cyclic flow and load conditions. All these factors, except for the last two, affect the maximum specific growth rate (μ_{nm}), which in turn affects the minimum sludge age for nitrification and the residual effluent ammonia concentration. The μ_{nm} value for different sewages varies considerably, between 0.2/d and 1.0/d, and is so unpredictable that it should be considered a sewage characteristic rather than a kinetic constant. The μ_{nm} (and half saturation coefficients) is quite sensitive to temperature, halving for every 6^0C temperature decrease. The μ_{nm} is extremely sensitive to pH, halving for every pH unit decrease below 7. Nitrification itself releases hydrogen ions and consumes 7.14 mg/ℓ alkalinity

(as $CaCO_3$) for every nitrate generated. This is particularly important for low alkalinity influents in which nitrification can be self inhibitory, by lowing the pH below a critical value (<6).

Introduction of unaerated zones in the activated sludge system to accommodate denitrification and BEPR reduces the nitrifier mass held under aerobic conditions, and this has an effect equivalent to reducing the μ_{nm} in proportion to the aerated sludge mass fraction of the system. This aspect is important in the design of N and P removal activated sludge plants, because if nitrification is jeopardized by having an unaerated fraction that is too large, denitrification (N removal) is no longer possible.

Because the concentration of ammonia in the influent is very low compared with the COD concentration (i.e. TKN/COD ratio = 0.1 to 0.12) and the nitrifier yield coefficient is very low compared to that of heterotrophic microorganisms, the mass of nitrifiers can be considered almost negligible in BNRAS systems. This small population requires favourable conditions to survive in biocenosis of the activated sludge. Therefore, in single sludge BNRAS systems, because the maximum specific growth rate of nitrifiers is very low and variable, the nitrification process governs selection of the two linked parameters in design:

- Aerobic mass fraction, and
- System sludge age.

To ensure complete nitrification throughout the year, in particular during winter, there are two options in design: Increasing aerobic mass fraction or extending sludge age. However, in biological N and P removal systems, the maximum aerobic mass fraction is limited by the requirement for anaerobic mass fraction for P removal and anoxic mass fraction for denitrification. Usually the aerobic mass fraction is limited to within the range 40-50%. In this case, for example, with the maximum specific growth rate of nitrifiers of around 0.45/d at 20°C, the system sludge age must be around 20 to 25 days to guarantee near complete nitrification all year round, if the minimum temperature is around 14°C. Such long sludge ages result in large biological reactor volumes per Mℓ wastewater treated. Furthermore, the maximum specific growth rate of nitrifiers varies from one municipal wastewater to another, even from one sewage batch to another for the same wastewater. Clearly, nitrification in the suspended media activated sludge system results in large and costly biological reactors. To remedy this shortcoming, it is proposed to uncouple the suspended solids sludge age of the activated sludge system from the requirement to nitrify. To achieve this, the activated sludge system can be modified in two ways, by including (1) internal or (2) external fixed media for attached nitrifier growth.

2.2.2 Nitrification with internal fixed media

Internal fixed media such as Ringlace[5] and floating tubes or small sponges have been placed in the aerobic reactor of the BNRAS system (Sen et al., 1994, 1995; Randall and Sen, 1996). The nitrifiers grow on the fixed media, establishing a population permanently resident in the aerobic reactor. These nitrifiers are not subjected to either the unaerated-aerated reactor interchanges or to the suspended solids sludge age, with the result that the latter can be reduced significantly without losing nitrification. However, the effectiveness of the internal fixed media has not been as good as expected, it yields a rather low cost/benefit ratio and the effect of the internal fixed media on BEPR has not been investigated.

2.2.3   Nitrification in fixed media trickling filter systems

In trickling filters, nitrifiers grow in a slime layer, or film, attached to a fixed media such as rocks, horizontal wood slats, random plastic rings, polyethylene strips and corrugated plastic sheets. Corrugated plastic sheets are very popular media in modern trickling filters, which can be divided into two types: vertical and cross-flow. The cross-flow trickling filters have a higher oxygen transfer efficiency as well as a higher contact time between the biofilm and the bulk liquid and therefore are favourable for nitrification (Parker and Merrill, 1984).

According to their usage, trickling filters can be classified into two groups:

- Single-stage trickling filters;

- Two-stage trickling filters.

In the single-stage trickling filters, both carbon oxidation and nitrification are accomplished in a single unit, often termed the combined carbon-oxidation-nitrification trickling filter. In the two-stage trickling filters, the effluent from the first stage (for carbon oxidation) is irrigated over the second stage trickling filters, in which mainly nitrification takes place, often termed tertiary trickling filters or nitrifying trickling filters (NTFs).

For the most part, the trickling filters for nitrification are used as a separate-stage tertiary treatment system. According to Boller and Gujer (1986), since a tertiary nitrification system produces only 2-3 mgTSS/ℓ of total suspended solids, no additional clarifier is required and generally this system yields a higher quality effluent than single-stage systems.

2.2.3.1   Single-stage trickling filters

According to Parker and Richards (1986), in the single-stage trickling filter system, nitrification begins only when the bulk solution soluble BOD_5 concentration is 20 mg/ℓ or less. Therefore, in single-stage systems, nitrifiers can only become established in the lower portion of the trickling filters where BOD concentrations have been reduced to low values. The degree of nitrification achieved in the trickling filters depends on a number of factors, including the BOD loading rate, temperature, dissolved oxygen concentration, ammonia concentration, and the pH and alkalinity of the bulk liquid.

At high BOD concentrations, factors which affect nitrification in trickling filters include hydraulic loading, hydraulic pattern and retention time on the filter media, the dissolved oxygen concentration in the liquid, pH, temperature, influent Total Kjeldahl Nitrogen (TKN) concentration, and influent BOD concentration (USEPA, 1993). From Gullicks and Cleasby (1986), empirical design curves for nitrification in trickling filters should incorporate the parameters for hydraulic loading, influent ammonia concentration, the effects of recycling and wastewater temperature.

High BOD loading rates tend to suppress nitrification, since autotrophic nitrifying bacteria are out-competed by fast-growing heterotrophs. The presence of organic matter inhibits the growth of nitrifiers due to the increased competition for dissolved oxygen by heterotrophs (Halling-

Sorensen and Jorgensen, 1993). In a study of the single-stage system in the United States, USEPA (1991) reported hydraulic loading in the range 0.13-0.24 $\ell/(m^2.s)$, total BOD loading in the range 1.17-1.91 $g/(m^2.d)$ (soluble BOD loading not reported), and ammonia loading in the range 0.19-0.28 $gN/(m^2.d)$.

From the discussions above, the single-stage trickling filter appears not to be favourable for nitrification.

2.2.3.2　　　Two-stage trickling filters (Nitrifying trickling filters)

Tertiary tricking filters, which are employed for nitrification only and with negligible organic material removal, are fairly common in the USA (Lutz *et al.*, 1990). In the tertiary trickling filters, increasing the ammonia loading rate (by increasing either the ammonia concentration or the hydraulic loading rate) generally increases the nitrification rate. However, the percent removal efficiency is reduced (Gullicks and Cleasby, 1986). Mass invasions of higher organisms which graze on the nitrifying biomass in the trickling filters (e.g. filter fly larvae, worms and snails) can result in reduced nitrification rates.

Most of the available design information for nitrification in trickling filters is based on tertiary systems where the media is irrigated with secondary clarifier overflow. Complications in modelling trickling filter performance include the influence of particulate organics and nonhomogeneous biofilm thickness and density. Mechanistic biofilm models, which usually include both microbial kinetics and substrate diffusion through the biofilm, consist of a set of partial differential equations, the solution of which can be difficult even with the best numerical techniques. Accordingly, the design of attached growth nitrification processes is primarily based on empirical data from pilot-scale and full-scale systems, and it has been found that empirical data are generally consistent with mechanistic biofilm models (USEPA, 1993).

While certain problems with macro fauna (snails, worms, larvae and flies), which reduce nitrification rates, have been encountered, high removals of ammonia have been economically achieved in tertiary NTFs (Parker *et al.*, 1989; 1995; 1996). Therefore, from the USA experience it appears to be completely feasible to use trickling filters for external nitrification in the proposed external nitrification BNR activated sludge system. External nitrification coupled to BNR activated sludge systems is investigated in detail in this research project.

2.3　　DENITRIFICATION

Denitrification in activated sludge systems requires carbon source input; carbon sources may be provided either by the endogenous decay of the activated sludge biomass, by adding a supplemental carbon source (usually methanol), or by the organic material present in the influent wastewater. In this section, the focus is on denitrification by using the organic materials present in wastewater influent, i.e. influent readily biodegradable COD (RBCOD) and slowly biodegradable COD (SBCOD), since in the proposed system the influent wastewater will be the substrate source for denitrification.

In nitrification and denitrification (ND) activated sludge systems, the only heterotrophic

organisms recognized for modelling purposes are the ordinary heterotrophic organisms (OHOs), and the denitrification, therefore, is ascribed to them only. The system nitrate removal achieved is the sum of the denitrification obtained from the utilization of influent RBCOD and SBCOD by OHOs.

However, in systems that include additionally BEPR, i.e. nitrification denitrification BEPR (NDBEPR) systems, the inclusion of *polyphosphate accumulating organisms* (PAOs) complicates the denitrification behaviour, in particular, where the PAOs are involved in denitrification. In this section, denitrification by OHOs and PAOs will be reviewed.

2.3.1 Denitrification by OHOs

2.3.1.1 Denitrification in ND activated sludge systems

Stern and Marais (1974) observed in ND systems under constant flow and load conditions, that denitrification in a plugflow primary anoxic reactor took place in two linear phases: a rapid first phase which persisted for a short period then terminated, and a second slow phase which continued for the rest of the retention time in the reactor; in a plugflow secondary anoxic reactor only one linear denitrification phase was operative at a slow rate, about two thirds of the slow second rate in the primary anoxic reactor. Ekama *et al.* (1979) hypothesized that the two linear phases in the primary anoxic reactor arose from the utilization of the two biodegradable COD fractions in the influent, namely RBCOD and SBCOD:

- The first rate denitrification which is connected to the RBCOD, and
- The second to the SBCOD.

With regard to the slow single denitrification rate observed in the secondary anoxic reactor, they proposed that this is due to SBCOD from endogenous mass loss.

To incorporate denitrification in the synthesis-death-generation aerobic model of Dold *et al.* (1980) to develop a general kinetic model for the ND activated sludge systems, Van Haandel *et al.* (1981) showed that the denitrification kinetic behaviour could be modelled in terms of RBCOD and SBCOD, and that the same formulations proposed by Dold *et al.* (1980) for RBCOD and SBCOD utilization under aerobic conditions can be used to model their utilization under anoxic conditions, except that the rate of SBCOD hydrolysis/utilization under anoxic conditions needed to be reduced to about 1/3rd of that under aerobic conditions. This reduction was incorporated in the model as a constant η_h = anoxic rate/aerobic rate = 0.33, representing either the population of OHOs that are facultative or a reduction in the aerobic rate for anoxic conditions. Using the general nitrification denitrification kinetic model, simulations of the denitrification response satisfactorily predicted two near linear phase denitrification behaviour in the primary anoxic reactor, and a single near linear phase in the secondary anoxic reactor, as observed by Stern and Marais (1974).

Accepting linear denitrification phases, Van Haandel *et al.* (1982) and Ekama *et al.* (1983) developed two simplified steady state mathematical models for ND activated sludge systems. The steady state models provided guidelines for designing ND activated sludge systems, sizing anoxic reactors, and estimating the denitrification potential (see WRC, 1984).

2.3.1.2 Denitrification in the NDBEPR activated sludge systems

In the late 70s and early 80s, the denitrification kinetics in the ND activated sludge systems was applied without correction to NDBEPR activated sludge systems, because it was repeatedly observed at both lab-and -full scale, that placing an anaerobic reactor ahead of the ND activated sludge system to form a NDBEPR activated sludge system, did not significantly change the magnitude of N removal (Siebritz et al., 1983; Nicholls, 1982). This approach gave predictions that were reasonably close to those observed experimentally.

Although apparently satisfactory, with the development of BEPR theory, an inconsistency in this approach became evident:

- The enhanced culture studies (Wentzel et al., 1989a and b) showed that PAOs in these systems did not denitrify. This implied that the RBCOD, converted to SCFA (short chain fatty acid) by OHOs and taken up by PAOs in the anaerobic reactor, no longer was available for denitrification in the primary anoxic reactor of NDBEPR activated sludge systems.

This in turn implied that the magnitude of denitrification in the primary anoxic reactor of the NDBEPR system should be significantly smaller than that in the primary anoxic reactor of the ND system. However, the experimental observations indicated that denitrification in both the ND and NDBEPR systems is approximately the same. The implication of this was that the denitrification kinetics in the NDBEPR system was different from that in the ND system.

Therefore, the denitrification kinetics in the ND activated sludge system needed to be modified for the application to NDBEPR activated sludge systems. To describe the denitrification kinetics in NDBEPR activated sludge systems, Clayton et al. (1989, 1991) undertook an extensive experimental investigation into the denitrification kinetics in NDBEPR systems by using plugflow anoxic reactors and batch tests. They observed that:

(1) The rapid rate of denitrification associated with RBCOD was much reduced or absent in the primary anoxic reactor;

(2) The slow rate of denitrification [in $mgNO_3\text{-}N$ (mgAVSS.d)] associated with SBCOD in the primary anoxic reactor in NDBEPR systems was approximately 2.5 times greater than the rate measured in the primary anoxic reactor in ND systems;

(3) The specific denitrification rate in the secondary anoxic reactor of NDBEPR systems was about 1.5 times greater than the rate measured in secondary anoxic reactors of ND systems.

From an in-depth enquiry into possible causes for these differences, Clayton et al. (1991) concluded that:

(1)   The increased denitrification rates were not due to:

  (a)   denitrification by PAOs, because in their systems the PHB and P measurements
          indicated that PAOs did not denitrify;

  (b)   the modification of the sewage organics in the anaerobic zone because sewage
          organics that had not passed through an anaerobic zone induced the same
          denitrification response as sewage organics that had passed through the anaerobic
          zone.

(2)   The increased denitrification rates are due to a stimulation in the activated sludge mass
        of an increased rate of hydrolysis of SBCOD in the anoxic reactor of NDBEPR systems,
        apparently induced by the presence of the anaerobic reactor in these systems.

The modified denitrification kinetics has been incorporated in the general NDBEPR mixed
culture kinetic simulation model by Wentzel *et al.* (1992). Wentzel *et al.* (1992) investigated the
η_H value (representing the population of OHOs that can denitrify or the reduction in the aerobic
SBCOD hydrolysis/synthesis rate for anoxic conditions) over a wide range of configurations and
conditions for NDBEPR systems by using the general NDBEPR kinetic simulation model. The
simulations for a particular system were repeated using a series of η_H values until the
experimentally measured nitrate concentrations were closely predicted. From the set of η_H values
obtained for the different NDBEPR systems, the "best" η_H value was estimated. From the
simulations completed (70 of which 57 could be used to evaluate η_H) a mean η_H value of 0.6 was
obtained. This value was significantly increased compared to the η_H value of 0.33 in ND systems.

With the proposed ENBNR activated sludge system, the anoxic mass fraction is substantially
increased compared to conventional BNR activated sludge systems. It is not known what the
effect of this on OHO denitrification will be. Accordingly, this aspect is investigated in this
research project.

2.3.2   Denitrification by both OHOs and PAOs

Since 1990, significant anoxic P uptake has been increasingly reported in lab-scale BNRAS
systems (Kerrn-Jespersen and Henze, 1993; Kuba *et al.*, 1993; Bortone *et al.*, 1996; Sorm *et al.*,
1996), and full-scale BNRAS systems (Kuba *et al.*, 1997). Ekama and Wentzel (1999b)
demonstrated that two types of BEPR behaviour have been observed in conventional single
sludge NDBEPR systems:

(1)   P uptake predominantly in the aerobic reactor (aerobic P uptake);

(2)   P uptake in both aerobic and anoxic reactors (anoxic/aerobic P uptake).

With predominantly aerobic P uptake, there is a minimal PAO activity in the anoxic reactor, i.e.
PAOs are not involved in the denitrification process and therefore the denitrification is mediated
by OHOs only, as observed by Clayton *et al.* (1990) above in mixed culture NDBEPR systems,
and Wentzel *et al.* (1989a and b) in enhanced culture BEPR systems. With anoxic/aerobic P
uptake, this implies that PAOs are also involved in the denitrification process, and therefore the
denitrification is mediated by both OHOs and PAOs.

Very little is known about denitrification in BNR activated sludge systems when it is mediated by both OHOs and denitrifying PAOs (DPAOs). From the previous investigations into DEPHANOX systems (one kind of ENBNRAS system, Bortone *et al.*, 1996; Sorm *et al.*, 1996), it was found that significant anoxic P uptake and associated denitrification by PAOs have been consistently observed in the systems. Actually, it seems that the DEPHANOX system was developed with the specific objective of fully exploiting DPAO behaviour. The researchers reasoned that denitrification by DPAOs would be advantageous in that the same substrate source (influent RBCOD) can be used for both denitrification and P removal, i.e. influent RBCOD is converted to SCFA that is sequestered and stored as PHB by the DPAOs in the anaerobic reactor (and therefore implicated in BEPR), and in the subsequent anoxic reactor, the stored PHB is utilized for denitrification and P uptake. In this way the influent RBCOD would be recaptured for denitrification in much the same way as in ND systems with the OHOs.

It is expected that denitrification by PAOs will also take place in the proposed external nitrification (EN) BNR activated sludge scheme (Chapter 1). It should be noted that stimulating DPAOs in the proposed system is not the main focus of this research. The research focus is on system intensification through uncoupling the suspended media sludge age from the nitrification requirement. However, since DPAO activity is expected in the proposed ENBNRAS system, the following issues need to be addressed:

- To quantify denitrification kinetics of OHOs in such systems (see above).

- To quantify denitrification kinetics of DPAOs in such mixed culture systems, i.e. in the presence of OHOs in the system; in other words, how do OHOs and PAOs compete for limited nitrate in mixed culture systems?

- To quantify the contribution of DPAOs to denitrification compared with that of OHOs.

As noted above, OHO denitrification kinetics in ENBNR activated sludge systems are not defined and this requires investigation (see Chapters 3, 4 and 5). With respect to denitrification kinetics of DPAOs, some experimental data are available from enhanced culture studies of DPAOs at Delft (Kuba *et al.*, 1993, 1996). However, in these studies BEPR was limited by the influent P concentration (effluent P concentration is zero), not by the biological processes, and thus some doubt exists on the applicability of the data to carbon-limited systems (the usual case in WWTPs). Furthermore, in enhanced culture systems, nitrate is always supplied sufficiently to maintain DPAO growth; the DPAOs therefore do not experience nitrate limitation as often occurs in mixed culture systems in which DPAOs have to compete with OHOs for limited nitrate. Therefore, the DPAO denitrification kinetics in the "Delft" enhanced culture systems may be different from that in mixed culture systems. This issue will be investigated as part of this research (see Chapter 3, 4 and 5).

Regarding the third issue, the relative contribution of DPAOs to denitrification has not been quantified. From biochemical studies (Wentzel *et al.* 1986, Comeau *et al.* 1986) and experimental observations (Ekama and Wentzel, 1999b), it was found that with anoxic-aerobic P uptake BEPR the system P removal was reduced compared to aerobic P uptake BEPR (see below). This indicates that denitrification by PAOs comes at a cost to P removal. Therefore it is necessary to know the contribution of DPAOs to denitrification, to assess whether this is

advantageous and outweighs the associated reduction in BEPR. This aspect will be addressed in this research project (see Chapters 3, 4, 5 and 7).

2.4   BIOLOGICAL EXCESS P REMOVAL (BEPR)

Biological excess P removal (BEPR) is mediated by a group of heterotrophic organisms called polyphosphate accumulating organisms (PAOs) that exhibit the propensity to store P internally as polyphosphate. Polyphosphate serves as an energy source enabling the organisms to store substrates and simultaneously release phosphate into the bulk solution during anaerobic conditions with influent substrate present. When sufficient PAOs are accumulated in the sludge of an activated sludge system, most or all phosphate is taken up during anoxic and/or aerobic conditions and a low effluent phosphate concentration is achieved. The stored phosphate is removed via sludge wasted from the system or stripped from the biomass and precipitated with chemicals. The question to achieve a good BEPR can be simply formulated as:

- How to create conditions in the system that specifically favour PAO growth.

Proposed explanations of the biochemical behavioural patterns associated with P release and P uptake that lead to BEPR have been presented in a number of mechanistic biochemical models such as those of Comeau et al. (1986), Wentzel et al. (1986, 1992) and Mino et al. (1987). At the time that these biochemical models were developed, the observed BEPR was principally associated with aerobic P uptake BEPR; despite this, the biochemical model of Wentzel et al. (1986) recognizes and describes PAO denitrification with anoxic P uptake BEPR.

2.4.1   Aerobic P uptake BEPR

The aerobic P uptake process is reasonably well understood. Both steady state design (e.g. Wentzel et al., 1990) and dynamic simulation models (e.g. Wentzel et al., 1992; Henze et al., 1995) incorporating the process have been developed. To obtain quantitative kinetic information for the mathematical BEPR models, enhanced PAO cultures were developed by Wentzel et al. (1989) in continuous-flow activated sludge systems (modified Bardenpho and UCT configurations), with acetate as the only organic substrate. Based on observations on these systems and batch tests on mixed liquor harvested from them, Wentzel et al. (1989a, b) developed a kinetic simulation model for enhanced PAO culture BEPR systems. With a single set of kinetic and stoichiometric constants, this model provided a very good description of the observed responses in the enhanced culture constant flow and load continuous systems and the batch tests on sludges drawn from these systems (Wentzel et al., 1989b). The enhanced PAO cultures exhibited minimal anoxic P uptake and denitrification behaviour, and hence the kinetics only describe the anaerobic P release and aerobic P uptake BEPR processes; anoxic P uptake and PAO denitrification were therefore not included in this kinetic simulation model.

Wentzel et al. (1990) simplified and extended the enhanced culture kinetic model to develop a steady state design model for mixed culture BEPR systems receiving municipal wastewaters as influent. Again, this model was based on aerobic P uptake only. Wentzel et al. (1992) integrated the enhanced culture kinetic model with the kinetic model for nitrification and denitrification systems, to develop a kinetic simulation model for BNR activated sludge systems (called

UCTPHO). This model formed the basis for the IWA Task Group model ASM No. 2 (Henze *et al.*, 1995). Since both these models drew their kinetics for BEPR from the enhanced culture kinetic model, they do not include anoxic P uptake and associated processes.

2.4.2 Anoxic/aerobic P uptake BEPR

Since 1990, anoxic/aerobic P uptake has been reported more frequently in lab-scale (Kerrn-Jespersen and Henze, 1993; Kuba *et al.*, 1993), and full-scale systems (Kuba *et al.*, 1997). Ekama and Wentzel (1999b) observed this type of BEPR behaviour in a number of long term (>500 d) laboratory studies with M/UCT type BNR activated sludge systems at 10 and 20 d sludge ages and at 12, 20 and 30°C (see Section 2.3.2). This type of BEPR has been also observed consistently in the DEPHANOX systems (Bortone *et al.*, 1996; and Sorm *et al.*, 1996).

There seem to be some major differences in P removal performance between the two types of BEPR behaviour, namely aerobic and anoxic/aerobic P uptake. Ekama and Wentzel (1999b) observed that in conventional NDBEPR systems:

- With only aerobic P uptake, the P release/P removal ratio, P removal/infl. RBCOD ratio and the P removal/Infl. COD ratio are around 3.0, 0.11 and 0.021 respectively and are in conformity with the steady state (Wentzel *et al.*, 1990) and dynamic state (Wentzel *et al.*, 1990 and Henze *et al.*, 1995) BEPR models.

- With the anoxic/aerobic P uptake, these ratios decrease to 1.5-2.0, 0.06-0.08 and 0.012-0.015 respectively, and the BEPR is depressed to around 60% of that with only aerobic P uptake. In order to match the P removal calculated with the steady state model of Wentzel *et al.* (1990) to that observed, the P content of the PAOs in the system ($f_{XBG,P}$) needs to be reduced from 0.38 mgP/mgPAOAVSS for aerobic P uptake to 0.10-0.28 mgP/mgPAOAVSS for anoxic/aerobic P uptake.

It seems that, given the appropriate conditions, different species of PAOs which accomplish anoxic P uptake find a niche in the system, but which have a significantly lower BEPR performance and use the influent RBCOD less "efficiently" compared with the aerobic P uptake PAOs.

As discussed above, it is expected that significant anoxic P uptake will take place in ENBNRAS systems, and this probably will result in reduced P removal. Thus, DPAO behaviour and the conditions that induce it need to be understood if the BEPR in these systems is to be defined. Also, DPAOs need to be included in the BNR activated sludge steady state design and kinetic simulation models. This requires investigation of the following aspects:

- The reduced anoxic P uptake BEPR in ENBNRAS systems;

- The conditions which stimulate DPAOs (anoxic P uptake and associated denitrification) in both the conventional BNRAS and ENBNRAS systems.

These two aspects have been investigated in this research and reported in Chapters 3, 4, 5 and 7.

2.5 ANOXIC-AEROBIC (AA) FILAMENT BULKING

It has been found that in BNRAS systems growth of a specific group of filamentous microorganisms is usually promoted, termed low F/M filaments by Jenkins *et al.* (1984) and renamed anoxic-aerobic (AA) filaments by Casey *et al.* (1994). These filamentous organisms, such as *M. parvicella*, type 0092, type 0041, type 1841 and type 0675, lead to sludge bulking in BNRAS systems not only in South Africa (Blackbeard *et al.*, 1988), but all over the world (Seviour *et* al., 1994; Rossetti *et al.*, 1994 and many others).

Historically, the control of AA (low F/M) filaments has been to increase the F/M ratio by incorporating selector reactors (Chudoba *et al.*, 1973). However, in evaluating the efficacy of anoxic and aerobic selectors, Gabb *et al.* (1991) and Ekama *et al.* (1996) concluded that the selection of AA filaments is not controlled by the selector effect, i.e. by kinetic aerobic or anoxic selectors in intermittently aerated ND systems or by metabolic selection with anaerobic reactors in NDBEPR systems.

From the investigations into the DEPHANOX system, it has been found that the system consistently produced a very good settling activated sludge by suppressing the sludge bulking caused by AA filaments, in particular *M. parvicella* (Bortone *et al.*, 1996; Sorm *et al.*, 1996). No explanation for this observation was given. However, a possible explanation may be formed from the AA filamentous bulking hypothesis of Casey *et al.* (1994).

2.5.1 The AA filamentous bulking hypothesis of Casey *et al.* (1994)

Casey *et al.* (1994) established that:

- Low F/M filaments proliferate under low F/M (>10 day sludge age) intermittent aeration conditions [as low F/M conditions *per se* did not appear to influence the sludge bulking, the filaments were renamed anoxic-aerobic (AA) filaments], but not under fully aerobic or fully anoxic conditions.

With intermittently aerated nitrification/denitrification (IAND) systems, Casey *et al.* (1994) observed that:

- Maximum filamentous organism proliferation occurred with an aerobic mass fraction between 30 and 35%.

- The DSVI appeared to be linked to incomplete denitrification, i.e. the nitrate and nitrite (NO_x) concentration at the end of the anoxic period when the conditions switched from anoxic to aerobic.

A relationship between the DSVI and aerobic mass fraction for IAND systems fed artificial wastewater was found and is shown in Fig 2.1.

Fig 2.1: Relationship between DSVI and aerobic mass fraction, as found by Casey et al. (1994).

The AA filamentous bulking hypothesis of Casey *et al.* (1994) explained in detail by Lakay *et al.* (1999), Musvoto *et al.* (1999) and Casey *et al.* (1999a, b, c) was based on the biochemical model for aerobic-anoxic facultative behaviour of heterotrophic organisms proposed by Payne *et al.* (1973). Payne *et al.* (1973) proposed the following denitrification pathway in their biochemical model:

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

(Nitrate → nitrite → nitric oxide → nitrous oxide → nitrogen gas).

Each of the nitrogen oxides is reduced at separate and specific enzyme complexes. One or more of the gaseous denitrification intermediates (NO, N_2O) that are generated under anoxic conditions have an inhibitory effect on the utilization of a substrate under subsequent aerobic conditions, as they interact with the enzyme complexes responsible for oxygen reduction. In particular, NO (nitric oxide) has been found to accumulate intra-cellularly during denitrification, and this causes a measured and prolonged inhibition of oxygen utilization under subsequent aerobic conditions.

Based on this model and their own experimental work, Casey *et al.* (1994, 1999c) proposed the following hypothesis for AA filamentous bulking:

- In BNRAS systems, heterotrophic organisms can be classified by their morphology as:
  - Floc formers or
  - Filamentous organisms.

- Floc formers are hypothesized to reduce nitrate or nitrite to nitrogen gas under anoxic conditions, while filamentous organisms are hypothesized to be nitrate reducers, reducing nitrate to nitrite only. If nitrate or nitrite are present throughout the anoxic period, the floc formers reduce the nitrate to nitrite to nitrogen gas through each of the denitrification intermediates, resulting in the presence at some level of intra-cellular NO. When these floc formers are exposed to subsequent aerobic conditions, the intra-cellular NO inhibits the utilization of oxygen, so that the floc formers continue to respire with nitrate or nitrite (i.e. aerobic denitrification), but at much reduced rates to that under anoxic conditions. In contrast, the filamentous organisms would not have any intra-cellular NO, because they perform only the first step of the denitrification pathway, and are therefore not inhibited in utilizing oxygen as an electron acceptor in the subsequent aerobic zone. This places the filamentous organisms at a competitive advantage in the subsequent aerobic zone, because they are able to utilize a greater portion of the available substrate under aerobic conditions than they would if the floc formers were not inhibited by intra-cellular NO. The filamentous organisms are thereby able to increase their relative mass in the mixed liquor, resulting in a bulking sludge. When nitrate is not present for some time before termination of the anoxic period, viz. complete denitrification occurs, the denitrification intermediates including NO are denitrified in the floc formers' cytochromes before aerobic conditions commence. Hence, the floc formers are not inhibited in using oxygen under subsequent aerobic conditions. Thus, when denitrification is complete (< 0.5 mg NO/ℓ in the anoxic reactor outflow), the floc formers are not at a disadvantage in the utilization of substrate with oxygen in the subsequent aerobic zone.

- The concentration of NO_x flowing from the anoxic reactor is therefore an indication as to whether the filamentous organisms are at an advantage in the aerobic zone, or not.

  - High NO_x concentrations flowing from the anoxic reactor are conditions conductive to AA filament proliferation and bulking, while

  - Near zero NO_x concentrations in the anoxic reactor outflow are indicative of an uninhibited floc forming organism population and better settling sludge.

These mechanisms are considered to be operative to the extent of causing poor sludge settleability at long sludge age (> 8 to 10 days) and aerobic mass fractions between ~15% and 60%.

2.5.2 The relationship between DSVI and aerobic mass fraction

The relationship between DSVI and aerobic mass fraction is very useful for designing BNRAS systems. Stewart Scott Consulting Engineers (Casey, 1998) investigated the applicability in practice of the relationship between the DSVI and the aerobic mass fraction as shown in Fig 2.1. The investigation included seven full-scale BNRAS plants, viz. Daspoort (operated by Pretoria Municipality), Rynfield and Vlakplaas (operated by ERWAT), and Goudkoppies, Bushkoppie, Olifantsvlei and Northern Works (operated by the Greater Johannesburg Metropolitan Council). These plants were chosen because they were the only plants at which historic experimental performance data were available. The historical sludge settleability data were converted to DSVI units where applicable, and an overall average DSVI value was calculated for each of the seven full-scale plants. The results obtained are shown in Table 2.1.

Table 2.1: DSVI values for seven full scale BNRAS plants in Gauteng, South Africa.

Plants	DSVI (mℓ/g)	aerobic mass fractions (%)	period of analysis
Goundkoppies	93	60	6/1997-6/1998
Bushkoppie (Unit 1)	61	76	1/1998-12/1998
Olifantsvlei (Unit 3)	61	58	1/1998-11/1998
Northern Works (Unit 4)	74	63	1/1998-11/1998
Rynfield (South)	113	50	1/1996-2/1997
Vlakplaas (Module D)	104	54	1/1997-12/1997
Daspoort (Module 9)	162	44	1/1995-3/1997

The aerobic mass fraction and DSVI data for seven full-scale treatment plants are superimposed on the artificial wastewater IAND systems DSVI versus aerobic mass fraction diagram of Casey *et al.* (1994) and plotted in Fig 2.2. It was found that a clear relationship exists between the two parameters, with improving sludge settleability for an aerobic mass fraction increase from 40% upwards.

However, it was not possible to comment on aerobic mass fractions below 40%, as data from full-scale BNRAS systems in this range were not available, and full scale systems tend not to be designed and operated with less than 40% aerobic mass fraction, due to the nitrification requirement. The information on this also is not available in lab-scale BNRAS systems because of the same reason as in full-scale plants, i.e. in conventional BNRAS systems, the aerobic mass fraction usually is larger than 40% due to the requirement for nitrification. The ENBNRAS system will provide an opportunity to observe the effect of very low (20 to 40%) aerobic mass fractions on DSVI. It is expected from the DEPHANOX experiments, that a good sludge settleability would be obtained. This aspect is reported in Chapters 3 and 4.

2.15

Fig 2.2: DSVI and aerobic mass fraction of seven full-scale BNRAS plants superimposed on the DSVI/aerobic mass fraction relationship developed for artificial wastewaters by Casey et al. (1994).

2.6 MODELLING BNR ACTIVATED SLUDGE SYSTEMS

For the design and operation of, and research into biological activated sludge systems, mathematical simulation models have proved to be invaluable as a process evaluation tool (e.g. Dold *et al.*, 1980, 1991; Henze *et al*, 1987). Mathematical simulation models provide quantitative descriptions of the dynamic behaviour of the wastewater treatment system. By providing quantitative descriptions, they allow predictions of the system response and performance to be made. From the predictions, design and operational criteria can be identified for optimization of system performance. Also, mathematical models are very useful as research tools. By evaluating model predictions, it is possible to test hypotheses on the behaviour of the wastewater treatment system (e.g. biological processes, their response to system constraints, etc.) in a consistent and integrated fashion. This may direct attention to issues not obvious from the physical system and lead to deeper understanding of the fundamental behavioural patterns controlling the system

response. In essence, mathematical models can provide a defined framework which can guide thinking (design, operation or research). Recognising the usefulness of mathematical models, it was proposed to develop such a model for the ENBNR activated sludge system.

For the proposed ENBNR activated sludge system (Chapter 1), from the previous investigations into DEPHANOX systems (Bortone *et al.*, 1996; and Sorm *et al.*, 1996), it will be expected that significant anoxic P uptake BEPR would take place in the ENBNR activated sludge system. A model which can simulate this system should take this into account. This phenomenon has also been observed in conventional BNRAS systems (Kerrn-Jespersen and Henze, 1993; Kuba *et al.*, 1993, 1997; Ekama and Wentzel, 1999a and b), and thus a model that can simulate the ENBNR activated sludge system also can be applied to conventional BNRAS systems. Thus, anoxic P uptake behaviour and associated denitrification are essential processes for inclusion in the model. Development of the mathematical model and incorporation of anoxic P uptake behaviour are described in Chapter 7.

2.7 CLOSURE

From the literature review on the relevant aspects of the proposed ENBNRAS systems in this chapter it is evident that a number of aspects require detailed investigation, for example:
- denitrification performance
- OHO and PAO denitrification kinetics
- PAO contribution to denitrification
- BEPR performance
- anoxic P uptake and the factors that stimulate it
- development of a simulation kinetic model for ENBNRAS systems
- occurrence of AA filament bulking at small aerobic mass fractions

These aspects amongst others are addressed in the body of the research report.

CHAPTER 3

EXPERIMENTAL INVESTIGATION INTO EXTERNAL NITRIFICATION BIOLOGICAL NUTRIENT REMOVAL (ENBNR) ACTIVATED SLUDGE SYSTEMS

3.1 INTRODUCTION

A series of three separate laboratory-scale experimental investigations on the external nitrification biological nutrient removal (ENBNR) activated sludge system were undertaken, aimed at determining the performance of this proposed system. This Chapter summarises the results from these investigations. For the details, the reader is referred to Moodley et al. (1999), Sötemann et al. (2000) and Hu et al. (2001).

3.2 EXPERIMENTAL INVESTIGATIONS

The first experimental investigation was on the proposed ENBNR activated sludge system with a small aerobic mass fraction (19%) and is reported in detail by Hu et al. (2001); for convenience in comparison and discussion, this system is referred to as System 1. This investigation actually was a feasibility study on the proposed ENBNR activated sludge system. After the proposed ENBNR activated sludge system was shown to operate successfully, the second investigation was initiated to examine the effect of varying aerobic mass fractions (and consequently anoxic mass fractions) and shorter sludge age on the ENBNR activated sludge system N and P removal performance, and is reported in detail by Moodley et al. (2000). This investigation covered two phases, and the ENBNR activated sludge system was the same as that used in the first investigation, except that in Phase I the aerobic mass fraction was reduced from 19 to 15% and the sludge age was reduced from the initial 10 days (sewage batches 1 to 3) to 8 days (sewage batches 4 to 16); this system will be referred to System 2a. Then, in Phase II, the aerobic mass fraction was increased to 30% at the expense of a reduced anoxic mass fraction (30%); this system will be referred to System 2b, which included sewage batches 17 to 33 of the second experimental investigation. The third investigation was initiated to optimize N and P removal in the ENBNR activated sludge system, and is reported in detail by Sötemann et al. (2000). This third investigation also comprised two phases and covered three systems; these will be referred to as System 3a which included sewage batches 1 to 13, System 3b which included sewage batches 14 to 20 and System 3c which included sewage batches 21 to 30. System 3a had aerobic and anoxic mass fractions of 32.5% each and included a mixed liquor (a) recycle of 2:1 between them. This was necessary because the nitrification efficiency of the external nitrification system initially was poor and the large aerobic mass fraction resulted in significant nitrification in the aerobic reactor. The a-recycle was included to (i) increase the nitrate load on the anoxic reactor and (ii) reduce the nitrate recycle to the anaerobic reactor. In System 3b, efficient nitrification in the external nitrification system was restored, and the anoxic and aerobic mass fractions were changed to 45% and 20% respectively. In System 3c the a-recycle removed. The system setup and design and operating parameters for the different investigations are summarized in Table 3.1.

The data for evaluation in this chapter were the averaged results of all sewage batches for each system. Detailed day-by-day and sewage batch average results can be found in the references above.

Table 3.1: Design and operating parameters for the ENBNRAS systems during the three investigations

Parameters	Investigation							
	1 (Hu et al., 2001)	2 (Moodley et al., 1999)				3 (Sötemann et al., 2000)		
System	System 1	System 2a		System 2b		System 3a	System 3b	System 3c
Sewage batch	1 to 13	1 to 3	4 to 16	17 to 21	22 to 33	1 to 13	14 to 20	21 to 30
Operating conditions								
Sludge age (d)	10	10	8	8		10		
Temps (°C)	20							
pH (Anaerobic)	7.2-8.2							
pH (Aerobic)	7.2-8.2							
DO (mg/ℓ)	2.0 - 5.0							
Influent (Raw sewage) from Mitchell's Plain WWTP								
Flow (ℓ/d)	20	20	40	30		20		
COD (mg/ℓ)	750							
RBCOD (mg/ℓ)	70-110							
TKN/COD	0.06-0.11[1]							
Total P (mgP/ℓ)	13.6-28.9[2]							
Reactor volume (ℓ) and mass fractions (%)								
Anaerobic	5; 23.8%	5; 25%						
Pre-Anoxic	[3]2; 9.5%	[3]2; 10%	[3]3; 15%	[3]3; 15%		[3]2; 10%		
Main Anoxic	10; 47.6%	10; 50%	9; 45%	6; 30%		6.5; 32.5%	9; 45%	
Aerobic	4; 19%	3; 15%	3; 15%	6; 30%		6.5; 32.5%	4; 20%	
Unaerated	17; 81%	17; 85%	17; 85%	14; 70%		13.5; 67.5%	16; 80%	
[4]EN system	Stone column fixed media			Suspended media activated sludge				
Recycle ratios with respect to influent flow								
s-recycle	1	1	0.5	0.5		1	1	1
To EN system	1.75	1.75	1.25	1.07		1.88	1.70	1.68
Bypass flow	0.25	0.25	0.25	0.43		0.12	0.30	0.32
a-recycle	0	0			1	2	2	0

[1] Excludes N contribution of nitrate dose to the main anoxic reactor

[2] Includes influent P supplement with K_2HPO_4 at 5-15 mgP/ℓ influent

[3] Actual volume is 1ℓ but with s = 1:1, the equivalent volume at system VSS conc. is 2ℓ; with s = 0.5:1, the equivalent volume at system VSS conc. is 3ℓ.

[4] EN External nitrification system.

3.3   EVALUATION OF THE ENBNR ACTIVATED SLUDGE SYSTEMS

3.3.1   Carbonaceous material removal

3.3.1.1   COD mass balance

The COD mass balances for each system are shown in Table 3.2. It can be seen that COD mass balances were less than 90%, varying between 76% and 88%. These low COD mass balances were not likely to be due to the system operating and sampling procedures, because similar low COD mass balance results have also been observed in other conventional BNR activated sludge experimental systems operated in the same laboratory, see Table 3.3. This would indicate that in BNR activated sludge systems (including external nitrification or not), COD is lost to some sink that is not taken into account in the COD mass balance equation. Therefore, the COD mass balances are not suitable for confirming the system operating and sample analytical procedures and the accuracy of the experimental data. Nitrogen mass balance results are more suitable for these purposes (see below).

Table 3.2:   COD mass balance components for each system during the three experimental investigations.

System	COD mass balance components (%)					Overall Balance (%)	Unaccounted for (%)	Number of days (d)
	Oxygen	Nitrate denitrified	COD used in EN	COD in wastage	COD in Effluent			
System 1	19	16	15	30	8	88	12	250
System 2a	8.5	13.5	13.8	35.5	10.8	82.1	17.9	154
System 2b	17.4	12.7	10	38	8.5	86.6	13.4	219
System 3a	18.3	13.5	21.6	22.8	7	83.2	16.8	186
System 3b	11.6	11.4	21.1	26.2	5.7	76	24	98
System 3c	10.9	12.8	16.7	31.1	5.8	77.3	22.7	137

The contributions of each COD mass balance components to the overall COD mass balance are also shown in Table 3.2 and summarised in Fig 3.1. From Table 3.2 and Fig 3.1 it can be seen that:

- 8.5% to 19% influent COD mass was removed via oxygen utilized;
- 11.4% to 16% via nitrate denitrified;
- 10% to 21.6% used in external nitrification systems;
- 22.8% to 38% via sludge wasted and
- 5.7% to 10.8% via the effluent flow, leaving
- 12% to 24% influent COD mass unaccounted for.

Table 3.3: COD and N mass balances obtained from conventional BNR activated sludge systems (see Ekama and Wentzel 1999b).

Researchers	System	COD mass balance (%)	N mass balance (%)
Clayton et al., (1989)	UCT	92	91
Kaschula et al., (1993)	MUCT	84	89
Musvoto et al. (1992)	MUCT	107	98
Pilson et al., (1995)	MUCT	84	97
Mellin et al., (1998)	MUCT	84	82
Sneyders et al., (1997)	UCT	90	92

Fig 3.1: COD mass balance components for each system during the three experimental investigations.

These results indicate that the COD utilized with nitrate as electron acceptor was close to or larger than that with oxygen as electron acceptor. This will depend on the aerobic mass fraction and nitrate available; for example, in System 2a with 15% aerobic mass fraction (Table 3.1) the COD removal via nitrate denitrified was greater than that via oxygen, but in System 2b with 30% aerobic mass fraction (Table 3.1), the COD removal via nitrate denitrified was less than that via oxygen, although in both cases nitrate was dosed into the main anoxic reactor. However, for

3.5

Systems 3a and 3b, because no nitrate was dosed into the anoxic reactor, the COD removal via nitrate denitrified was always less than that via oxygen, regardless of whether the aerobic mass fraction was large (System 3a, 30%) or small (System 3b, 20%).

The low oxygen demand in all systems was reflected in the low OUR in the aerobic reactor; for example, in System 1 the OUR was only about 29 mg/(ℓ.h) in the 19% aerobic mass fraction of the system. The OUR in an equivalent conventional internal nitrification BNR activated sludge system with 90% COD mass balance, 10 day's sludge age, complete nitrification, 90% nitrate denitrification leading to 50% recovery in nitrification, the OUR would be about 2.5 times higher, i.e. 75 mg/(ℓ.h). Clearly not nitrifying in the ENBNR activated sludge system and utilizing the nitrate generated in the external nitrification system results in a major decrease in OUR. The proportion of COD utilized with nitrate indicates that the denitrification potential of the system is very high, due to the large anoxic mass fraction (in this example 47.6%). This allows the system to treat very high influent TKN/COD ratios without jeopardizing BEPR and achieving complete denitrification, provided near complete nitrification is achieved in the external fixed media system.

The COD component of the sludge wasted was very high compared to the other COD components. This is because the sludge ages in all systems were reduced significantly, from the usual 20 - 25 days in conventional BNR activated sludge systems to 8 to 10 days. In the external nitrification system this is a prerequisite to achieve the objective of system intensification.

3.3.1.2 COD removal

From Table 3.2, the COD component in the effluent flow was very low and varied from 5.7% to 10.8%. This is reflected in system effluent COD concentrations. From Table 3.4 and Fig 3.2, the unfiltered effluent COD concentration ranged between 46 and 74 mgCOD/ℓ. The 0.45μm membrane filtered effluent COD concentration ranged between 38 and 52 mg COD/ℓ. These latter values were accepted to correspond to the unbiodegradable soluble COD concentrations in the influent, and gave the unbiodegradable soluble COD fractions ($f_{s,us}$) ranging from 0.052 to 0.084. The COD removal efficiencies were very good and ranged from 89 to 94% (see Table 3.4).

Table 3.4: Unfiltered and filtered effluent COD concentrations, and fraction of the total influent COD that is unbiodegradable soluble in each system.

Parameter	System					
	1	2a	2b	3a	3b	3c
Influent COD (mg COD/ℓ)	717	683	698	741	724	730
Unfiltered Effluent COD (mg COD/ℓ)	60	74	59	57	46	47
Filtered Effluent (0.45μm) (mg COD/ℓ)	51	52	52	48	39	38
Unbiod. soluble COD fraction ($f_{s,us}$)	0.071	0.084	0.075	0.065	0.054	0.052
COD removal (%)	92	89	92	92	94	94

Fig 3.2:   Influent COD and filtered and unfiltered effluent COD for the three investigations.

It should be noted that in System 2a, the unfiltered effluent COD concentration ranged between 49 and 101 with a mean of 74 mgCOD/ℓ, and the 0.45μm membrane filtered effluent COD concentration ranged between 33 and 73 mg COD/ℓ, with a mean of 52 mgCOD/ℓ (Table 3.4 and Fig 3.2). Thus for this system, there was a large difference between filtered and unfiltered effluent COD concentrations. The reason for this was an effluent turbidity from carryover of sludge in the settling tank. This was not due to settling tank failure caused by sludge bulking, as the sludge settleability during this period was very good (DSVI<80 mℓ/g except for the first three sewage batches, see Moodley et al., 1999), rather the sludge carryover seemed to be due to pin-point floc.

The pin-point floc was observed only in System 2a, in which the aerobic mass fraction was 15%. In the other ENBNR activated sludge systems (System 1, System 2b, System 3a, 3b and 3c where aerobic mass fractions were 19%, 30%, 30%, 20% and 20% respectively), the pin-point floc behaviour was not observed, but another parallel ENBNR activated sludge system with the same aerobic mass fraction (15%) (see Hu et al., 2001) showed similar pin-point floc behaviour. Thus, the pin-point floc causing the turbid effluent seemed to be associated with a small aerobic mass fraction (<19%). This would suggest that for good floc formation, a minimum aerobic mass fraction of about 19% is required.

3.3.2 Nitrogenous material removal

3.3.2.1   Nitrogen mass balance

The nitrogen (N) mass balances for each system are shown in Table 3.5 and summarised diagrammatically in Fig 3.3. It can be seen from Table 3.5 and Fig 3.3 that the N mass balances

ranged from 87% to 96%. These results are comparable to those in other experimental investigations performed on conventional BNR activated sludge systems in the same laboratory (see Table 3.3). Therefore, the experimental data can be accepted for the system performance evaluation for each system.

Table 3.5: N mass balance components for each system during three phase investigations.

System	NOx denitrified %	N used in EN %	N in wasted sludge %	N in Effluent %		Nitrogen balance %	Unaccounted for %
				NOx	TKN		
System 1	52	12	13	5	7	89	11
System 2a	36.2	8.7	17.4	6.3	27.2	96	4
System 2b	45.3	7.6	23.4	9.9	10.6	97	3
System 3a	50.6	2.1	16.4	11.9	7.4	88	12
System 3b	37.8	17.8	16.8	8.1	6.4	87	13
System 3c	42.8	16.7	19.4	3.8	6.1	89	11

Fig 3.3: N mass balance components for the three investigations.

The contribution of the N mass balance components to the overall N mass balances is also shown in Table 3.5 and Fig 3.3. It can be seen from Table 3.5 and Fig 3.3 that:
- 36.2% to 52% influent nitrogen mass (including nitrate dose where applicable) was removed via nitrogen gas (denitrification);
- 6% to 17.8% was used in external nitrification systems;
- 13% to 23.4% was removed via sludge wasted and
- 9.9% to 33.5% exited the system via the effluent flow, leaving
- 3% to 15% influent nitrogen mass unaccounted for.

The N removal in the external nitrification system should be low. However, in Systems 1, 3b and 3c these components were high (12, 17.8 and 16.7% respectively), and almost equal to the N removed in the sludge wasted in the BNR activated sludge part of the system. This implied that denitrification probably took place in the external nitrification system due to insufficient air supply.

In all systems, the N removed via N_2 gas (denitrification) were very high and varied from 36.2% to 52%. Most of this denitrification took place in the main anoxic reactors. The high N removal via denitrification in the main anoxic reactors is due to the large anoxic mass fractions at the expense of aerobic mass fraction in the ENBNR activated sl..dge system and the dosing of nitrate in some systems (Systems 1, 2a and 2b) to realize these reactors full denitrification potential.

The influent N mass that exited the system via the effluent flow varied from 9.9% to 33.5% depending on the system nitrification efficiency and denitrification potential. The effluent TKN concentrations depend on nitrification in both the external nitrification system and the BNR activated sludge system, and the effluent nitrate and nitrite concentrations depend on both system nitrification performance and denitrification potential (see below).

3.3.2.2   Nitrification

In the ENBNR activated sludge system, nitrification is designed to take place externally in the external nitrification system. However, nitrification may take place in the BNR suspended media part of the system under some conditions. The experimental observations on nitrification in the ENBNR activated sludge system are listed in Table 3.6.

Table 3.6:   Nitrification in external nitrification (EN) and BNR activated sludge systems.

Parameter	System					
	1	2a	2b	3a	3b	3c
External nitrification systems (EN)						
Flow / total flow from Anaerobic to EN (%)	87.5	85.4	62.9	94	85	84
FSA concentration into EN (mg N/ℓ)	27.1	32.2	27.4	26.1	29.2	29.5
FSA concentration out of EN (mg N/ℓ)	4.4	16.4	10	7.7	3.3	3.3
NOx generated in EN/total NOx in System (%)	88.0	77.6	57	79.4	92.6	91.4
Nitrification efficiency in EN (%)	84	49	64	71	89	89
BNR activated sludge systems (BNRAS)						
Sludge underflow/total flow from Anaerobic (%)	12.5	14.6	37.1	6	15	16
NOx generated in BNRAS/total NOx in System (%)	12.0	22.4	43	20.6	7.4	8.6
Aerobic mass fraction (%)	19	15	50	32.5	20	20
Sludge age (days)	10	10.8	8	10	10	10
Final effluent FSA concentration (mg N/ℓ)	3.8	16.7	6.3	3.6	3.6	3.6
Final effluent NOx concentration (mgN/ℓ)	3.0	4.6	7.8	8.9	4.8	2.8

Only for those sewage batches with the suspended medium activated sludge external nitrification system.

Nitrification in external nitrification system

In the experimental investigations reported here, the type of external nitrification system was not the focus of the research. Therefore, the requirement for the external nitrification system was only to achieve as high a nitrification efficiency as possible.

To achieve as much nitrification as possible in the external nitrification system, the maximum of ammonia-rich supernatant from the internal settling tank overflow should be pumped to the external nitrification system. However, this is limited by the sludge underflow which should ensure that it transfers as much as possible of the sludge from the internal settling tank to the main anoxic zone to prevent sludge from accumulating in the internal settling tank and overflowing to the external nitrification system. From Table 3.6, at least 6% but usually about 15% of the internal settling tank inflow was required to transfer the sludge from the internal settling tank to the main anoxic zone. This implies that at best only 90% (but more likely 85%) of the system ammonia can be nitrified in the external nitrification system, and 10% (more likely 15%) of the system ammonia is expected to be nitrified in the activated sludge system if complete external nitrification is achieved (neglecting the N requirement for sludge production in the system), i.e. if the effluent ammonia concentration from the external nitrification system is zero.

From Table 3.6, nitrification efficiency in the external nitrification system was 89% at best (Systems 3b and 3c) and only 49% and 64% for Systems 2a and 2b respectively. This is because the performance of the laboratory-scale stone column external nitrification system used in the initial investigations deteriorated due to *Psycoda* fly and larvae infestation, resulting in high ammonia concentrations from the external nitrification system. If the aerobic mass fraction is small, this ammonia is not nitrified in the aerobic reactor of the BNR activated sludge system. From a BEPR point of view, low nitrification in the aerobic reactor is an advantage because it avoids a high nitrate concentration return to the anaerobic reactor. However, from a N removal point of view it is a disadvantage because it (i) increases the effluent ammonia and (ii) reduces the nitrate load on the anoxic reactor and hence the denitrification. Good N removal performance requires the external nitrification system to nitrify virtually completely. If the aerobic mass fraction is "large" (e.g. System 2b), the ammonia will be nitrified in the aerobic reactor of the BNR activated sludge system. This will also be disadvantageous because (i) from a BEPR point of view the nitrate will overload the underflow anoxic reactor and enter the anaerobic reactor, (ii) the nitrate will not be available in the main anoxic reactor for denitrification and (iii) the effluent nitrate concentration will increase.

Nitrification in BNRAS activated sludge system

Nitrification in the BNR activated sludge system depends on the system sludge age, aerobic mass fraction and ammonia available (i.e. external nitrification efficiency and the flow pumped into the external nitrification system, see above). The ENBNR activated sludge system is designed to operate at a short sludge age and hence nitrification mainly depends on the aerobic mass fraction and available ammonia.

When the ENBNR activated sludge system is operated at small aerobic mass fraction, the nitrifiers are not sustained in the suspended media liquor and ammonia will exit the system with the final effluent. However, nitrifiers are seeded from the external nitrification system to the BNR suspended media system, and thus some nitrification can be expected in the aerobic zone; for example, in Systems 1, 2a, 3b and 3c with aerobic mass fractions of 19, 15, 20 and 20%

respectively, the nitrifiers were not supported in the suspended media BNR activated sludge system, but 12, 22.4, 7.4 and 8.6% of the total nitrate generated was in the BNR activated sludge system.

When the ENBNR activated sludge system is operated at relatively larger aerobic mass fractions (e.g. Systems 2b and 3a), nitrifiers are sustained in the suspended media liquor and ammonia entering the aerobic zone will be nitrified virtually completely. When this happens, complete nitrification in the external nitrification systems is vitally important to maintain BEPR, because the underflow nitrate load may exceed the denitrification potential of the pre-anoxic reactor resulting in nitrate entering the anaerobic reactor, and reducing P release and hence P removal. To reduce the nitrate concentration in the aerobic reactor, an a-recycle from the aerobic zone to the main anoxic zone can be added. However, recycling between the aerobic and anoxic reactors does appear to be detrimental to the BEPR performance of the system (see Sötemann *et al.*, 2000). When the a-recycle was removed from System 3b to System 3c, the denitrification, anoxic P uptake and P removal all improved significantly.

3.3.2.3 Denitrification

Denitrification potential
From nitrate and nitrite mass balances around each reactor and settling tank, the measured denitrification for each system were obtained and are listed in Table 3.7. It can be seen that denitrification occurred principally in the main anoxic reactor and a small part of denitrification occurred in the pre-anoxic reactor. Therefore, the system denitrification performance depends on the main anoxic reactor denitrification potential and the nitrate load on it.

It should be noted that for Systems 3a and 3b, denitrification also took place in the anaerobic reactor (5.1 and 3.0 mg N/ℓ respectively). This meant that nitrate was recycled to the anaerobic zone and thereby interfered with P release and hence P removal behaviour. Also it meant that the pre-anoxic zone had been loaded in excess of its denitrification potential. In System 3a, the pre-anoxic denitrification was 4.8 mg N/ℓ influent, but in System 3b it was only 2.9 mg N/ℓ influent for the same anoxic mass fraction (10%). It is not clear why the denitrification potentials of the pre-anoxic reactor were so different for these two systems.

For the main and pre-anoxic reactors, the denitrification potentials could be determined only in those cases where the nitrate concentrations from the main anoxic reactor were greater than 1 mg NO_3-N/ℓ. Thus, the measured denitrification values of the main anoxic reactor for those sewage batches in which nitrate concentrations from the main anoxic reactor were larger than 1 mgNO_3-N/ℓ were averaged for each system and are shown in Table 3.7. These measured total denitrification values can be considered as the respective denitrification potentials of the main anoxic reactors. These values varied from 19 to 49 mg NO_3-N/ℓ influent, mainly because the anoxic mass fraction was varied between 30% and 50% (see Table 3.1), but also partly because of the natural variation in biological systems. From these denitrification potential values, the system denitrification rates [K'$_2$ mg NO_3-N/(mgAVSS.d)] can be calculated. However, because significant anoxic P uptake took place, both OHO and PAOs participated in the denitrification. Therefore, the BEPR behaviour will be discussed first (see below) before considering the denitrification performance further (see Section 3.3.4 below).

Table 3.7: Measured denitrification in anaerobic, pre-anoxic and main anoxic and settling tanks and denitrification potentials (mg N/ℓ influent).

Reactor	System					
	1	2a	2b	3a	3b	3c
Reactor denitrification (mgN/ℓ influent)						
Anaerobic reactor	0.5	0	0.3	5.1	3.0	0.8
Pre-anoxic reactor	2.8	2.0	3.7	4.8	2.9	2.2
Main anoxic reactor	40.1	32.2	26.9	22.0	20.4	28.4
Internal settling tank	0	0	0	0	0	0
Final settling tank	0	0	0	1.4	0.9	0.6
Denitrification potentials (mgN/ℓ influent)						
Main anoxic	4.9	34.3	25.5	22.0	19.0	31.1
Pre-anoxic	4.5	5.8	4.0	5.1	3.8	3.7

3.3.2.4   Nitrogen removal

The influent and effluent TKN and nitrate concentrations as well as N removal efficiencies for each system are given in Table 3.8 and Fig 3.4. From Table 3.8 and Fig 3.4 it can be seen that the N removal efficiencies were greater than 80%, which is very good, except for Systems 2a and 2b when the external nitrification system nitrified poorly (49% to 64%, Table 3.6). For both these systems, high ammonia concentrations (Table 3.6) entered the aerobic reactor of the BNR activated sludge part of the plant. In System 2a, this residual ammonia could not be completely nitrified in the aerobic reactor because the aerobic mass fraction was small (15%), resulting in a high TKN concentration in the effluent (19.8 mg/ℓ) and thus low N removal. In System 2b, the aerobic mass fraction was 30%, so some nitrification took place in the aerobic reactor, resulting in a lower TKN concentration in the effluent (9.2 mgN/ℓ) which increased the overall nitrification efficiency. However, the low nitrification efficiency in the external nitrification system resulted in a low nitrate load on the main anoxic reactor and hence low denitrification and low N removal. In general, the system can achieve an effluent total N (TN) less than 10 mgN/ℓ as demonstrated by Systems 1, 3b and 3c, but this requires virtually complete nitrification in the external nitrification system. The effluent nitrate concentrations were low and less than 9 mgN/ℓ for all systems.

Table 3.8: Influent TKN, and effluent TKN and Nitrate concentrations, as well as N removal efficiency for each system.

Parameter	System					
	1	2a	2b	3a	3b	3c
Influent TKN	70	72	66	69	79	78
Influent TKN/COD	0.112	0.111	0.095	0.092	0.109	0.107
Effluent TKN	5.1	19.8	9.2	5.1	5.0	4.8
Effluent NOx	3	4.6	7.8	8.9	4.8	2.5
Effluent total N	8.1	24.4	17	14.0	9.8	7.3
% N removal	88	66	74	80	88	91

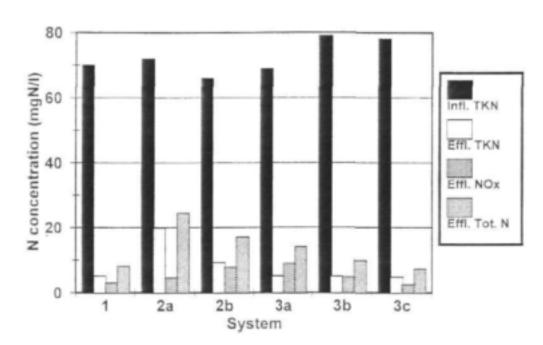[1] *Including nitrate dose.*

Fig 3.4: Influent TKN and effluent TKN, NOx and total nitrogen for the three investigations.

3.3.3   Biological excess P removal (BEPR)

3.3.3.1 Total phosphorus (TP) mass balance

To assess the BEPR behaviour, total phosphorus (TP) mass balances were conducted around each reactor, internal and final settling tanks and the external nitrification system from the TP concentrations measured in the influent and soluble ($< 0.45 \mu m$) TP concentrations measured in the reactors, internal and final settling tanks, effluent and external nitrification system outflow for all sewage batches of the investigations. The TP mass balances were calculated by subtracting the outflow P mass from the inflow P mass, so a negative result indicates P release (-ve) and a positive result indicates P uptake (+ve). The calculated P release and P uptake masses were divided by the influent flow, to give the P release and P uptake in mgP/ℓ influent. The mean P uptake and P release for all sewage batches for each system are given in Table 3.9.

3.3.3.2   Phosphorus release

From Table 3.9 and Fig 3.5, it can be seen that, as expected, most of the P release occurred in the anaerobic reactor. A small amount of P release also occurred in the internal settling tank. This latter P release is probably due to leakage of some RBCOD out of the anaerobic reactor, stimulating P release in the sludge blanket that formed in the bottom of the internal settling tank due to the low underflow pumping rate to the main anoxic reactor. This was not unexpected as similar observations had been made in previous investigations into the DEPHANOX system, another kind of external nitrification BNR activated sludge system (Bortone et al., 1996; Sorm et al., 1996). This additional P release is beneficial because it augments the anaerobic P release. In the modified DEPHANOX system, the anaerobic reactor and internal setting tank were combined in a single reactor allowing sludge accumulation, P release and settlement simultaneously (Bortone et al., 1997).

Table 3.9:   Phosphorus release (-ve) and uptake (+ve) in each reactor, internal settling tank and final settling tank as well as external nitrification system (mgP/ℓ influent), and % anoxic P uptake.

Reactor	P release (-ve) or P uptake (+ve) (mgP/ℓ influent)					
	System 1	System 2a	System 2b	System 3a	System 3b	System 3c
Anaerobic reactor	-25.7	-11	-21.4	-4.9	-11.0	-16.7
Internal settling tank	-4.5	-0.9	-2.3	-5.9	-4.8	-4.1
External nitrification system	-0.9	-1.1	-1.8	-4.3	-4.0	-5.1
Pre-anoxic reactor	0.2	-2.8	-2.7	0.8	1.2	0
Main anoxic reactor	22.2	13.5	15	22.5	15.1	22.3
Aerobic reactor	20.2	10.9	21.1	13.9	11.5	12.9
Final settling tank	0.1	-1.1	1.8	0.5	1.1	1.2
% anoxic P uptake	52	55	42	62	57	63

Fig 3.5: P release (rel., -ve) and P uptake (upt., +ve) for anaerobic (anaer.), internal settling tank (Int. settl.), external nitrification (EN), pre-anoxic (Preanx.), main anoxic (main anx.), aerobic and final settling tank (Final settl.) for the three investigations.

With regard to P release in the external nitrification system of Systems 3a, 3b and 3c, this was not expected and it is not beneficial to the system BEPR. This phenomenon was not observed in the other experimental systems to the same extent. This may be due to the external nitrification system itself, because in Systems 3a, 3b and 3c, the external nitrification system was a suspended media activated sludge system, not a fixed media stone column as used in Systems 1, 2a and 2b. This P release is not beneficial to the BEPR because (i) it is not likely to be associated with substrate uptake by PAOs and hence (ii) this is "extra" P that needs to be taken up subsequently. Sotemann et al. (2000) speculated on the cause of this P release as (a) PAOs that do not settle in the internal settling tank and undergo endogenous P release in the external nitrification system or (b) filterable but non-settleable P containing organics that are hydrolyzed in the external nitrification system, but both explanations do not seem very plausible.

3.3.3.3 Phosphorus uptake

From Table 3.9 and Fig 3.5, it can be seen that significant P uptake took place in the main anoxic reactor for all systems. The % anoxic P uptake varied from 42% to 63% of the total P uptake. From these results, together with the results from the previous investigations into DEPHANOX systems (Bortone et al., 1996; Sorm et al., 1996), it can be concluded that anoxic P uptake BEPR is an endemic characteristic of ENBNR activated sludge systems, and thus information on the factors that stimulate anoxic P uptake BEPR in general may be obtained from these investigations.

The factors that stimulate anoxic P uptake BEPR in BNRAS systems

One of the objectives in the first investigation (System 1, see Hu *et al.*, 2001) was to investigate and establish the conditions that stimulate anoxic P uptake BEPR. In the course of the investigation, initially both aerobic and anoxic P uptake were low, and most of the P uptake took place in the aerobic reactor. As the investigation continued, the total P uptake increased and gradually relatively more took place in the anoxic reactor and less in the aerobic reactor. Examining this data in more detail, for the first 6 sewage batches, the % anoxic P uptake remained approximately constant at about 25%. At that stage, ways to increase this anoxic P uptake were explored.

It was noted from earlier investigations on BEPR in conventional BNR activated sludge systems (Ekama and Wentzel, 1999b), that when the main anoxic reactor was underloaded with nitrate, the P uptake tended to be confined to the aerobic reactor, whereas if the nitrate load exceeded the denitrification potential so that nitrate was present in the anoxic reactor outflow, then significant anoxic P uptake was observed. Accordingly, this proposal was applied to the ENBNR activated sludge system to stimulate anoxic P uptake in the ENBNR activated sludge system: From sewage batch 6, nitrate was dosed into the main anoxic reactor and the amount dosed increased stepwise with succeeding sewage batches; the stepwise increase was to avoid nitrate overload to the main anoxic reactor and hence a high nitrate recycle to the anaerobic reactor. This nitrate dosing appeared to stimulate significant anoxic P uptake (see Hu *et al.*, 2001). These experimental results confirmed the observations from conventional BNR activated sludge systems (above). In the subsequent investigations into ENBNR activated sludge systems (Systems 2a, 2b, 3a, 3b and 3c), this observation was further substantiated.

The observations above suggest that with limited nitrate available, the OHOs tend to outcompete the PAOs for nitrate. The greater affinity of OHOs than PAOs for nitrate is reflected in the calculated specific denitrification rates by OHOs ($K_2'_{OHO}$) and PAOs ($K_2'_{PAO}$) (see below): It was found that the specific denitrification rates of the OHOs ($K_2'_{OHO}$) are significantly higher than those of the PAOs ($K_2'_{PAO}$). In terms of this competition, if the nitrate load into the main anoxic reactor is less than the denitrification potential of the OHOs, then the PAOs have limited opportunity for use of the limited nitrate. In contrast, if the nitrate load on the main anoxic reactor exceeds the denitrification potential of the OHOs, then the denitrifying PAOs (DPAOs) would have a greater opportunity to use the "excess" nitrate and so develop in the system. The development of the DPAOs in BNR activated sludge systems appears very slow, because in the experimental investigation anoxic P uptake appeared still to be increasing in System 1 after some 150 days of operation (see Hu *et al.*, 2001). This probably is a further contributing reason for the variable (and rather limited) occurrence of DPAOs in conventional BNR activated sludge systems.

In the ENBNR activated sludge system, there appear to be three further conditions that appear to favour DPAOs, viz.

- Small aerobic mass fraction;

- Sequence of reactors;

- The low frequency of alternation between anoxic and aerobic states.

3.16

The first condition limits development of aerobic PAOs in the system and ensures PAOs grow anoxically.

The sequence of reactors in the ENBNR activated sludge system is such that nitrification precedes the main anoxic reactor, and hence all the nitrate generated is available for denitrification; this ensures a substantial nitrate load to the main anoxic reactor which stimulates anoxic P uptake. Therefore, for Systems 3a, 3b and 3c, even though no additional nitrate was dosed into the main anoxic reactor, the nitrate load from the external nitrification system to the main anoxic reactor was large, and hence significant anoxic P uptake was still observed. In contrast, in conventional BNR activated sludge systems nitrification usually follows the anoxic reactor so that the system requires a mixed liquor recycle to bring nitrate generated in the aerobic zone to the anoxic zone for denitrification. Thus, in these systems, the nitrate generated in the system that can be denitrified in the anoxic reactor is limited by the mixed liquor recycle (a recycle) from aerobic to anoxic reactor, and it is not possible to bring all nitrate generated in the system into the anoxic zone for denitrification. From the observations above, this will tend to disadvantage the DPAOs, unless the nitrate load via the a- recycle is substantial, requiring high a-recycle ratios, but such ratios tend also to disadvantage DPAOs (see below).

It seems that frequent alternation of the sludge between the anoxic and aerobic zones adversely affects the DPAOs. This was apparent from Systems 3b and 3c where removing the a -recycle (while ensuring a sufficient high nitrate load on the main anoxic reactor) reduced the sludge alternation between anoxic and aerobic conditions, and this improved the anoxic P uptake and denitrification (Tables 3.9 and 3.7 respectively). The sludge alternation frequency between anoxic and aerobic conditions in ENBNR activated sludge systems usually is much lower than that in a conventional BNR activated sludge system (like the UCT system) for the same TKN/COD ratio and nitrification efficiency, due to the requirement of the a-recycle in the latter system.

The combination of the above may explain the reason why anoxic P uptake BEPR was consistently observed in the ENBNR activated sludge system, but only occasionally in conventional BNR activated sludge systems.

3.3.3.4 Phosphorus removal performance

The biological P removals achieved in the five ENBNRAS systems are given in Table 3.10. The system conditions that influence this, viz, influent COD and RBCOD concentration, anaerobic mass fraction, sludge age and nitrate entering the anaerobic reactor, are also listed in Table 3.10. Compared with the BEPR obtained with predominantly aerobic P uptake, the BEPR achieved in the ENBNR activated sludge systems with predominantly anoxic P uptake is significantly lower, at about 2/3rds that expected for the influent RBCOD, sludge age and anaerobic mass fraction from the BEPR model of Wentzel et al. (1990), which is based on aerobic P uptake (see below).

Table 3.11: Observed BEPR and calculated P contents of PAOs ($f_{XBG,P}$) with the steady state model (Wentzel et al., 1990) in conventional BNR (Ekama and Wentzel, 1999b) and ENBNR activated sludge systems.

Systems	Anoxic P uptake (%)	P removal (mgP/ℓ)	Prel/Prem ratio	Prem/infl. RBCOD ratio	Prem/Total infl COD ratio	$f_{XBG,P}$ (mgP/mg PAOAVSS)
Conventional BNR activated sludge systems (UCT and MUCT systems)					
Clayton et al., 1991	5	21	3.3	0.105	0.021	0.388
Sneyders et al., 1998 (C)	0	13.1	2.42	0.116	0.023	0.471
Sneyders et al., 1998 (E)	0	16.8	2.74	0.116	0.024	0.471
Musvoto et al., 1992 (1)	27	12.2	2.44	0.063	0.012	0.144
Musvoto et al., 1992 (2)	47	11.3	2.7	0.060	0.012	0.113
Pilson et al., 1995 (1)	47	12	1.34	0.069	0.012	0.136
Pilson et al., 1995 (2)	16	10.9	1.24	0.063	0.011	0.098
Mellin et al., 1998	29	11.4	1.8	0.082	0.016	0.260
ENBNR activated sludge systems					
System 1	52	10.6	2.9	0.081	0.015	0.235
System 2a	55	8.5	1.4	0.080	0.012	0.167
System 2b	42	11.8	2	0.116	0.017	0.250
System 3a	62	9.7	1.1	0.080	0.013	0.240
System 3b	57	9.1	1.7	0.062	0.013	0.170
System 3c	63	10.5	2.0	0.070	0.014	0.188

3.3.3.5 Comparison of the measured and calculated P removal

The BEPR performance in the BNR activated sludge system can be assessed by comparing the observed P removal with that theoretically calculated from the steady state BEPR model of Wentzel et al. (1990). This model requires as input all the system design parameters (see Table 3.1) and influent wastewater characteristics including the influent RBCOD concentration. The procedure for calculating the theoretical BEPR is summarized by Ekama and Wentzel (1999b) and in detail by Hu et al. (2001). Following this procedure[1] the OHO and PAO masses and the calculated P contents of PAOs ($f_{XBG,P}$) in conventional BNRAS systems and ENBNRAS systems were obtained and are listed in Table 3.11. It can be seen that in the systems with predominantly (>95%) aerobic P uptake BEPR behaviour, the $f_{XBG,P}$ values are the same as that in the steady state model of Wentzel et al. (1990) (0.38 mgP mgPAOAVSS). Comparing the $f_{XBG,P}$ values of

[1] In the calculation procedure, the influent SBCOD was decreased to take account of the unaccounted for COD in the COD mass balances for all the system and for the ENBNRAS system the SBCOD was reduced further to take account of the COD lost in the external nitrification system, but the influent RBCOD was kept at the measured values. Thus was necessary to reduce the sludge variation in the OHO and PAO active fraction ($f_{x,OHO}$ and $f_{x,PAO}$) of the VSS and specific OHO denitrification rates obtained by Ekama and Wentzel (1999b).

systems with aerobic and anoxic/aerobic P uptake BEPR, it seems that the latter value is only about one quarter to two thirds of the 0.38 mgP/mgPAOAVSS aerobic value.

3.3.4 Calculation of specific denitrification rate

3.3.4.1 Relative contribution of OHOs and PAOs to denitrification

From Table 3.11, in the ENBNR activated sludge systems significant P uptake took place in the main anoxic reactors. This implicated PAO participation in the denitrification process, i.e. PAOs contributed to the system denitrification. It appears that anoxic P uptake and associated denitrification took place consistently in all ENBNR activated sludge systems investigated.

To calculate the denitrification potential in the ENBNR activated sludge systems, the contribution of OHOs and PAOs to denitrification needs to be known. In the main anoxic zone, if PAOs are involved in denitrification, PAOs and OHOs will compete for nitrate by using different substrates. PAOs use internally stored PHA, and OHOs use slowly biodegradable COD (SBCOD). This difference in substrate source makes it possible to estimate the contribution of OHOs and PAOs to the denitrification. The procedure detailed by Hu *et al.* (2001) was followed for calculating the specific denitrification rates of the OHOs and PAOs. The specific denitrification rates, denoted $K_2'_{OHO}$ [mg NO_3-N/(mgOHOAVSS.d)] and $K_2'_{PAO}$ [mg NO_3-N/(mgPAOVSS.d)] respectively, are listed in Table 3.12.

The results were obtained by averaging those sewage batches only in which RBCOD was measured and nitrate concentrations in the main anoxic reactor was larger than 1 mg N/ℓ. Furthermore, in fractioning the VSS mass into OHO and PAO active masses and the other three inert masses, the unbiodegradable particular COD fraction ($f_{s,up}$) needs to be determined. All the investigations received raw (unsettled) municipal wastewater from the same source, i.e. Mitchell's Plain Wastewater Treatment Plant (Cape Town, South Africa). Thus, since $f_{s,up}$ is a wastewater characteristic, there is a reasonable expectation that $f_{s,up}$ should be constant. This was the case for 4 long term parallel investigations on aerobic and anoxic/aerobic activated sludge systems receiving the same wastewater, $f_{s,up} = 0.10$ to 0.16 (Warburton et al., 1991; Mbewe et al., 1995; Ubisi et al., 1997; Mellin et al., 1998). Accordingly, a constant $f_{s,up}$ of 0.12 was accepted for the Mitchell's Plain wastewater. Also, while the influent RBCOD concentrations were kept at the measured values, the influent SBCOD was reduced to take account of (i) the COD loss in the external nitrification system and (ii) the COD unaccounted for in the COD balances.

Table 3.12: Calculated specific denitrification rates for PAOs and OHOs and the contribution of PAOs and OHOs to denitrification in the main anoxic reactor for each system.

System	System					
	1	2a	2b	3a	3b	3c
Anoxic P uptake (%)	53.8	52.7	37.5	33.8	57.2	66.6
Measured total denitrification (mgN/ℓ inf)	53.2	32.2	26.8	18.8	19.0	31.2
% denitrification by PAOs	17.5	25.4	23.6	9.6	24.6	22.0
% denitrification by OHOs	82.5	74.6	76.4	91.4	75.3	78.0
K'_{2PAO} (mg NO_3-N/(mg PAOAVSS.d))	0.042	0.048	0.059	0.029	0.038	0.044
K'_{2OHO} (mg NO_3-N/(mg OHOAVSS.d))	0.184	0.166	0.185	0.114	0.098	0.157

From Table 3.12 it can be seen that the contribution of PAOs to denitrification varied from 9.6% to 25.4% even when the anoxic P uptake was 66%. This indicates that the contribution of PAOs to denitrification was low, and less than 26% for all systems investigated. The specific denitrification rates of PAOs ranged from 0.029 to 0.059 mg NO_3-N/(mg PAOAVSS.d), these values are low compared to those of OHOs varying from 0.098 to 0.185 mg NO_3-N/(mg OHOAVSS.d). This may explain the reason why sufficient nitrate is required to stimulate anoxic P uptake BEPR in BNR activated sludge systems. If nitrate is not sufficient, OHOs will outcompete the PAOs for limited nitrate. The low contribution of PAOs to the denitrification raises a question about explaining denitrifying PAOs (DPAOs), because it also causes the reduction of P removal, i.e. denitrification by PAOs was obtained by sacrificing about 1/3rd of the P removal. However, considering that the concentration of PAOs is only around 1/5th of that of the OHOs, and the specific rate of this small PAO mass is only 1/3rd of that of the OHOs, this denitrification by PAOs does not appear to be very significant, and most likely not worth sacrificing 1/3rd of P removal for.

3.3.5 Sludge settleability

The mean DSVIs observed from all sewage batches for each system are listed in Table 3.13. It is evident that the ENBNR activated sludge systems consistently produced a good settling sludge. The aerobic mass fractions for each system are also listed in Table 3.13. Casey *et al.* (1994) observed that maximum filamentous organism proliferation occurred with an aerobic mass fraction between 30 and 35% and established an association between the DSVI and the aerobic mass fraction from artificial wastewater experiments; this is shown in Chapter 2, Fig 2.1.

Table 3.13: Observed DSVIs and aerobic mass fractions in ENBNR activated sludge systems.

System	System 1	System 2a	System 2b	System 3a	System 3b	System 3c
DSVI (mℓ/g)	69	73.3	97	119	109	96
Aerobic mass fraction (%)	19	15	30	32.5	20	20

To investigate the applicability of the association between the DSVI and the aerobic mass fraction in practice, Stewart Scott Consulting Engineers (Casey, 1998) conducted a survey. The survey included seven full-scale conventional BNRAS plants. These plants were chosen because they were the only plants for which historical data of the sludge settleability performance were available. The historical sludge settleability data were converted to DSVI units where applicable, and an overall average of the DSVI value was calculated for each of the seven full-scale plants. The results obtained are shown in Table 2.1 (Chapter 2). The aerobic mass fraction data for the seven full-scale treatment plants were superimposed on the artificial wastewater laboratory system DSVI/aerobic mass fraction diagram from Casey et al. (1994) and plotted in Fig 2.2 (Chapter 2). For the full-scale plants, it appears that an association also exists between the two parameters, with improving sludge settleability for an aerobic mass fraction increase from 40% upwards. However, Casey et al. could not comment on the effect of aerobic mass fractions on DSVI for aerobic mass fractions below 40% as conventional BNR activated sludge systems are not designed and operated at less than 40% aerobic mass fraction because of the nitrification requirement.

The information from the ENBNR activated sludge systems provides the possibility to assess the association between DSVI and aerobic mass fractions in the less than 40% aerobic mass fraction range. The DSVI results from the ENBNR activated sludge systems were superimposed on the DSVI/aerobic mass fraction diagram from Casey et al. (1994) and plotted in Fig 3.6. It can be seen that the ENBNRAS experiments confirm the artificial wastewater association between the two parameters of Casey et al. (1994), with improving sludge settleability for aerobic mass fractions less than 30%.

Fig 3.6: Relationship between DSVI and aerobic mass fraction from artificial wastewater, full-scale BNR activated sludge plants and lab-scale ENBNR activated sludge systems.

3.4   CLOSURE

From the three laboratory-scale experimental investigations into the proposed ENBNR activated sludge systems, it is evident that BNR activated sludge system intensification by separating the process of nitrification from the main BNR activated sludge system and effecting nitrification externally is possible in practice.

3.4.1   System performance

From the investigations:
- COD removals in excess of 90% can be expected.
- Considerable reduction in oxygen demand will be obtained, about 60% less than in an equivalent conventional BNR activated sludge system.
- The proportion of inflow to the internal settling tank that can be directed to the external nitrification system is about 85% due to the constraint of sludge underflow to the anoxic reactor, and a nitrification efficiency in the external nitrification system of about 85 - 90% can be expected, but this requires confirmation at full-scale.
- Total N removal of 80 - 90% is possible, with an effluent TKN of about 5 mgN/ℓ, effluent NO_3^- of about 2.5 - 5 mgN/ℓ and total nitrogen of about 7 - 10 mgN/ℓ. However, this will be influenced by the nitrification efficiency of the external nitrification system, and the influent wastewater characteristics.
- BEPR of about 9 - 11 mgP/ℓ, but again this will depend on the wastewater characteristics. Anoxic/aerobic P uptake BEPR was consistently observed at 52 - 63% of total P uptake. It seems that such BEPR is inherent to the ENBNR activated sludge system, because the system configuration provides the conditions which favour the development of denitrifying PAOs. These conditions are:
  - (1)   large anoxic mass fraction at the expense of the aerobic mass fraction, i.e. small aerobic mass fraction;
  - (2)   sufficient nitrate load into anoxic zone.
  - (3)   no mixed liquor (a) recycle.
  The nitrate load on the anoxic reactor appears to be the most important.
- The contribution of PAOs to denitrification is low, and less than 26% for all systems investigated.
- A good settling sludge can be expected, with a DSVI of about 100mℓ/gTSS.

3.4.2   Design and operation

From the investigation, a number of design and operation parameters can be identified:
- The successful implementation of the ENBNR activated sludge system hinges around successful operation of the external nitrification system. If the efficiency of this unit process is reduced, denitrification in the anoxic reactor will be reduced (due to the lower nitrate load). Furthermore, FSA not nitrified in the external nitrification system will be nitrified in the aerobic reactor of the BNR activated sludge system to a greater or lesser extent (depending on the aerobic mass fraction and nitrifier seeding from the external nitrification system). This will cause deterioration in the effluent nitrate quality, and the nitrate in the underflow recycle may exceed the denitrification potential of the underflow anoxic reactor and enter the anaerobic reactor and hence reduce BEPR.

- Anaerobic mass fraction of about 25%; this can be reduced if larger anoxic mass fractions are required, with a resultant reduction in BEPR (Wentzel *et al.*, 1990).

- Minimum aerobic mass fraction of 20%. Smaller aerobic mass fractions appear to stimulate pin point floc formation, which cause deterioration in sludge retention and effluent quality. Furthermore, such aerobic mass fractions will stimulate some aerobic P uptake BEPR. Inclusion of anoxic P uptake PAOs in, and exclusion of aerobic P uptake PAOs from, the biocenosis of the BNR activated sludge system mixed liquor are not essential for achieving BNR in the external nitrification scheme. In fact, inclusion and maximization of aerobic P uptake would appear desirable to maximize BEPR (aerobic P uptake BEPR is more efficient than anoxic P uptake BEPR). However, conditions that promote aerobic P uptake BEPR are also conducive to nitrifier growth. Although exclusion of nitrifiers from the mixed liquor of the BNR activated sludge system was originally considered essential, this is not necessary as long as virtual complete nitrification in the fixed media system is obtained, to limit nitrification in the main aerobic reactor. In fact, complete exclusion of nitrifiers may not prove possible. Although conditions in the BNR activated sludge system may not be conducive to nitrifier growth, they are likely to be continually seeded into the system from the external nitrification system. Accepting nitrification in the BNR activated sludge system, with complete nitrification in the external nitrification system, the nitrification in the BNR activated sludge system will be limited by the ammonia bypassing the internal settling tank in its underflow. This nitrate is prevented from entering the anaerobic reactor with the underflow recycle by denitrification in the underflow anoxic reactor. However, should complete nitrification not be obtained in the external nitrification system, then the ammonia from this system will be nitrified in the aerobic reactor of the BNR activated sludge system. If, as a result the aerobic reactor nitrate concentration is too high, the pre-anoxic reactor will become overloaded with nitrate resulting in nitrate discharge to the anaerobic reactor and reduced BEPR. However, if nitrification were not included in the BNR activated sludge system, the ammonia would leave via the effluent, an even less desirable situation.

- The requirements above for an anaerobic mass fraction of 25% and an aerobic mass fraction of 20% provide 55% of the total mass fraction for the anoxic reactors. This has to be divided between the primary and underflow anoxic reactors. An underflow anoxic mass fraction of about 10% appears adequate, leaving 45% for the main anoxic reactor.

- The proportion of the internal settling tank inflow that can be directed to the external nitrification system is about 85%, due to the constraint of sludge underflow to the anoxic reactor (see above).

- An a-recycle from the aerobic to the anoxic reactor should not be included, as this appears detrimental to BEPR and denitrification.

CHAPTER 4

EXTERNAL NITRIFICATION BIOLOGICAL NUTRIENT REMOVAL (ENBNR) ACTIVATED SLUDGE SYSTEM PERFORMANCE AT SLUDGE AGES SHORTER THAN 10 DAYS

4.1 INTRODUCTION

In Chapter 3 extensive experimental data on ENBNR activated sludge systems have been presented. In these systems, sludge ages were 10 (Systems 1 and 3a and 3b) and 8 (Systems 2a and 2b) days (see Table 3.1, Chapter 3). However, as noted in Chapter 3, *Psycoda* fly and larvae infestation in the stone column external nitrification system for the systems at 8 days sludge age (Systems 2a and 2b) caused deterioration in external nitrification performance. This complicated assessment and comparison of system performance at different sludge ages. Initially, evaluation of the laboratory scale ENBNR activated sludge system performance at sludge ages lower than 10 days was not specifically part of the scope of the investigation - data on BNR activated sludge systems at short sludges ages are available from previous investigations in the UCT laboratory (Burke *et al.*, 1984, Wentzel *et al.*, 1990). However, towards the end of the practical laboratory investigation on System 3b (Chapter 3), it was decided to increase the influent sewage flow from 20 ℓ/d to 30 ℓ/d in order to observe the system response to this 50% increase in load. Shortly after the influent flow had been increased to 30 ℓ/d, the system internal settling tanks began to fail hydraulically. Thus, instead of allowing the system to fail completely as a result of this hydraulic failure, it was decided to reduce the influent flow to 25 ℓ/d and instead of implementing a gradual increase in feed, a gradual reduction in sludge age should rather be implemented and the effect of the reduced sludge age on system performance evaluated. This Chapter summarises this investigation; details are given by Sötemann *et al.* (2000).

4.2 SYSTEM OPERATION

At the beginning of sewage batch 31 for System 3b (Chapter 3), the influent was increased from 20 to 30 ℓ/d. Two days later, it was reduced to 25 ℓ/d and the sludge age was decreased from 10 to 8 days (Configuration 4, see Table 4.1). The system was run at the 8 day sludge age for sewage batches 31, 32 and 33 (49 days, 6 sludge ages) after which the sludge age was reduced further to 5 days (Configuration 5, see Table 4.1) at the beginning of sewage batch 34. The system was run at a 5 day sludge age for a further 13 days (3 sludge ages). For the 5 day sludge age 4 ℓ/d were wasted. The 4 ℓ/d could no longer be taken from the 4 ℓ aerobic reactor in one batch, so a small peristaltic pump was installed and calibrated to waste 4 ℓ of mixed liquor from the aerobic reactor over a 24 hour period. System operation is described in detail by Sötemann *et al.* (2000).

Table 4.1: ENBNR activated sludge system design and operating parameters.

System Parameter	Config. 3b	Config. 4	Config. 5
Days	285 to 421	422 to 470	471 to 483
Sewage Batches	21 to 30	31 to 33	34
No. of Days	137	49	13
Dates: From	09/12/99	18/04/00	08/06/00
To	17/04/00	07/06/00	18/06/00
Operating Parameters			
Influent Flow (l/d)	20	**25**	25
Sludge Age (d)	10	**8**	**5**
Waste (l/d)	2	2.5	4
Temperature (oC)	20	20	20
pH - Anaerobic Reactor	7.5 - 7.8	7.1 - 7.9	7.7 - 8.0
pH - Main Aerobic Reactor	7.7 - 8.2	7.4 - 8.4	7.7 - 8.0
D.O. Main Aerobic Reac. (mgO/l)	2 - 5	2 - 5	2 - 5
Reactor Vol. / Mass Frac.			
Total System Volume (l)	20	20	20
Pre-Anoxic Reactor (l)	1*	1*	1*
Anaerobic Reactor (l)	5	5	5
Main Anoxic Reactor (l)	9	9	9
Main Aerobic Reactor (l)	4	4	4
Total Aerobic Mass Fraction	0.20	0.20	0.20
Anoxic Mass Fraction	0.55	0.55	0.55
Anaerobic Mass Fraction	0.25	0.25	0.25
Total Unaerated Mass Fraction	0.80	0.80	0.80
Recycles			
s - Recycle (w.r.t influent flow)	1 : 1	1 : 1	1 : 1
a - Recycle (w.r.t influent flow)	**0 : 1**	0 : 1	0 : 1
Sludge Bypass (w.r.t influent flow)	0.32 : 1	0.34 : 1	0.37 : 1
External Nitrifier Parameters			
Sludge Age (d)	Very long		
Waste (l/d)	As required to maintain sludge level in internal SST		
D.O (mgO/l)	>5		
Reactor Volume (l)	3		

*Actual volume with sludge at double concentration. Effective volume at system sludge concentration = 2 litres.

Denotes changes made to previous configuration.

4.3 SYSTEM PERFORMANCE

Given the comparatively short time that the system was run at these low sludge ages it would be of little practical value to give an as detailed evaluation for the 8 and 5 day sludge age system configurations as was done in Chapter 3 for the 10 and 8 day sludge age configurations. For this reason only a brief comparison of the main nutrient removal performances (COD, N and P) will be given in this Chapter; detailed results are given by Sötemann *et al.* (2000). The sewage batch averages for all measured parameters for sewage batches 31 to 34 (Configurations 4 and 5) are given in Tables 4.2a, b and c.

TABLE 4.2a: Sewage batch averages of measured COD, TKN and FSA parameters for sewage batches 31 to 34 (Configurations 4 and 5).

				Sewage Batch				Overall
				31	32	33	34	
COD	Influent	UF	mgCOD/ℓ	731.6	779.8	-	709.2	740.2
	Influent	FF	mgCOD/ℓ	158.3	191.3	-	186.2	178.6
	Int. Set. A	UF	mgCOD/ℓ	153.6	145.6	-	161.5	153.5
	Int. Set. B	UF	mgCOD/ℓ	77.6	92.2	-	91.4	87.0
	Aerobic M.L.	UF	mgCOD/ℓ	2392	2564	-	1735	2231
	Effluent	UF	mgCOD/ℓ	61.5	62.7	-	83.9	69.4
	Effluent	F	mgCOD/ℓ	36.1	41.0	-	31.6	36.2
TKN	Influent	UF	mgN/ℓ	64.3	79.9	-	87.5	77.3
	Aerobic M.L.	UF	mgN/ℓ	150.4	149.5	-	123.0	141.0
	Effluent	UF	mgN/ℓ	4.6	5.0	-	6.9	5.5
	Effluent	F	mgN/ℓ	3.7	4.1	-	4.1	4.0
FSA	Influent	UF	mgN/ℓ	50.0	64.0	-	71.8	62.0
	Int. Set. A	UF	mgN/ℓ	23.4	28.7	-	33.7	28.6
	Int. Set. B	UF	mgN/ℓ	2.9	3.4	-	2.7	3.0
	Effluent	UF	mgN/ℓ	3.1	3.5	-	2.8	3.1
Influent TKN COD Ratio		-		0.088	0.103	-	0.120	0.104

(UF = Unfiltered, FF = Floc Filteres, F = 0.45µm membrane filtered)

TABLE 4.2b: Sewage batch averages of measured suspended solids, OUR, DSVI and pH for sewage batches 31 to 34 (Configurations 4 and 5).

			Sewage Batch				Overall
			31	32	33	34	
TSS	Pre Anoxic	mgTSS/ℓ	3782	4013	-	2760	3518
	Aerobic	mgTSS/ℓ	1957	2109	-	1410	1825
VSS	Pre Anoxic	mgVSS/ℓ	3095	3264	-	2254	2871
	Aerobic	mgVSS/ℓ	1626	1738	-	1177	1513
ISS[1]	Pre Anoxic	mgVSS/ℓ	688	749	-	506	648
	Aerobic	mgVSS/ℓ	332	371	-	233	312
COD/VSS Ratio[2] (Aerobic Reactor)			1.45	1.45	-	1.44	1.45
TKN/VSS Ratio[2] (Aerobic Reactor)			0.09	0.09	-	0.11	0.10
OUR	Aerobic	mgO/ℓ.h	19.9	25.5	-	17.7	21.1
DSVI		mℓ/g	92.6	86.9	-	92.9	90.8
pH	Anaerobic	-	7.50	7.43	-	7.89	7.61
	Aerobic	-	7.97	7.85	-	7.85	7.89

[1] ISS calculated from TSS - VSS.

[2] Calculated from unfiltered aerobic reactor COD and TKN concentrations divided by the VSS.

TABLE 4.2c: Sewage batch averages for measured nitrite, nitrate, and P concentrations for sewage batches 31 to 34 (Configurations 4 and 5).

				Sewage Batch				Overall
				31	32	33	34	
NO_2	Pre Anoxic	F	mgN/ℓ	0.0	0.0	-	0.8	0.3
	Anaerobic	F	mgN/ℓ	0.0	0.1	-	0.0	0.0
	Int. Set. A	F	mgN/ℓ	0.0	0.1	-	0.0	0.0
	Int. Set. B	F	mgN/ℓ	0.0	0.1	-	0.0	0.0
	Anoxic	F	mgN/ℓ	0.0	0.7	-	0.3	0.4
	Aerobic	F	mgN/ℓ	0.0	0.3	-	1.2	0.5
	Effluent	F	mgN/ℓ	0.0	0.2	-	0.8	0.3
NO_3	Pre Anoxic	F	mgN/ℓ	0.0	0.1	-	10.9	3.7
	Anaerobic	F	mgN/ℓ	0.0	0.1	-	0.2	0.1
	Int. Set. A	F	mgN/ℓ	0.1	0.2	-	0.2	0.2
	Int. Set. B	F	mgN/ℓ	17.6	23.7	-	34.9	25.4
	Anoxic	F	mgN/ℓ	0.1	1.2	-	15.0	5.4
	Aerobic	F	mgN/ℓ	0.1	3.3	-	15.9	6.4
	Effluent	F	mgN/ℓ	0.6	2.9	-	17.0	6.8
P	Influent	F	mgP/ℓ	25.1	26.3	-	26.0	25.8
	Pre Anoxic	F	mgP/ℓ	13.2	14.1	-	15.3	14.2
	Anaerobic	F	mgP/ℓ	28.1	32.0	-	23.8	28.0
	Int. Set. A	F	mgP/ℓ	30.2	33.8	-	25.5	29.8
	Int. Set. B	F	mgP/ℓ	33.0	36.5	-	28.0	32.5
	Anoxic	F	mgP/ℓ	21.5	24.1	-	22.2	22.6
	Aerobic	F	mgP/ℓ	10.8	14.5	-	18.3	14.5
	Effluent	UF	mgP/ℓ	10.4	14.1	-	18.9	14.5
	Effluent	F	mgP/ℓ	9.6	13.8	-	17.4	13.6

(UF = Unfiltered, FF = Floc Filteres, F = 0.45μm membrane filtered)

4.3.1 Carbonaceous Material Removal

Table 4.3 shows the COD mass balances (including their components) and percentage COD removals for sewage batches 31 to 34. Table 4.4 gives a comparison of the most important COD parameters for the 10 (System 3b, Table 3.1, Chapter 3), 8 (System 4) and 5 (System 5) day sludge age configurations: these are summarised in Fig 4.1.

TABLE 4.3: COD mass balances for sewage batches 31 to 34.

Average of Batch	Influent COD mgCOD/d	MOC mgO/d	Denitrification Recovery mgCOD/d	COD used Ext. Nit. mgCOD/d	COD in Waste mgCOD/d	COD in Effluent mgCOD/d	COD out mgCOD/d	% Recovery
31	16290	1790	2174	3322	5981	1383	14655	90.4
32	19496	2054	2962	2284	6412	1411	15123	78.0
33								
Config 4	18893	1924	2568	2603	6197	1397	14889	79.2
34	17729	1336	3224	2990	6941	1762	16253	91.8
Config 5	17729	1336	3224	2990	6941	1762	16253	91.8

TABLE 4.4: Comparison of average COD parameters for 10, 8 and 5 day sludge age system configurations.

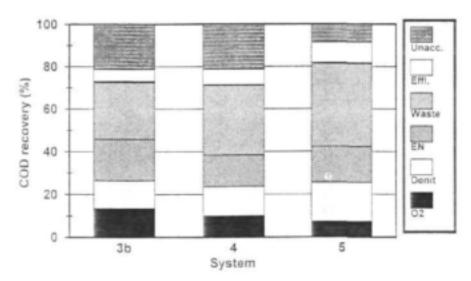|	ENBNRAS System Configurations		
	1,2 and 3	4	5
	10 Day Sludge Age	8 Day Sludge Age	5 Day Sludge Age
% COD Reduction	94%	92%	90%
OUR	19.9 mgO/l.h	22.7 mgO/l.h	17.7 mgO/l.h
COD Balance	79.8%	79.2%	91.8%
% COD to Oxygen	13.8	10.2	7.5
% COD to Denitrification	12.7	13.6	18.2
% COD to EN System	19.7	14.8	16.8
% COD in Waste	26.7	32.8	39.2
% COD in Effluent	6.2	7.4	9.9
% COD Unaccounted	20.8	21.2	8.3

Fig 4.1: COD mass balance components for Systems 3b (Chapter 3), 4 and 5.

The overall average COD mass balance for the 10, 8 and 5 day sludge age system configurations are 79.8%, 79.2% and 91.8% respectively (see Table 4.4 and Fig 4.1). The overall COD mass balances for the 10 and 8 day sludge age configurations are virtually the same, but that for the 5 day sludge age configuration is substantially higher. This is probably because the VSS concentration had not yet reached a steady state value for the 5 day sludge age. The overall average COD mass balance components for the various sludge age configurations do not show any unexpected changes, except that for the 5 day sludge age configuration the % COD in the wasted VSS is probably higher than would be at steady state. The variations in %COD to oxygen and % COD to denitrification can be attributed to variations in the influent sewage characteristics rather than to the reduced sludge ages. A higher influent COD will bring about a slight increase in the oxygen utilisation rate (OUR) and a higher influent TKN/COD ratio will cause a shift from the COD passed to oxygen to the COD utilised for denitrification. This can be clearly noted for the 10 and 5 day sludge age configurations: For the 10 day sludge age configuration which received an average of 736 mgCOD/ℓ influent with a TKN/COD ratio of 0.106 mgN/mgCOD, 13.8% of the COD was passed to oxygen and 12.7% of the COD was utilised for denitrification. For the 5 day sludge age configuration the influent COD was 709 mgCOD/ℓ with a TKN/COD ratio of 0.120 mgN/mgCOD. The combination of the lower COD in the influent and the substantially higher TKN/COD ratio resulted in 10.2% of the COD being passed to oxygen and 18.2% of the COD being utilised for denitrification. There is no reason for the percentage COD 'lost' to the EN system to change with a lowering of the sludge age of the system, and the results for the three sludge ages are similar. The small variations can be attributed to the different number of sewage batches fed rather than to the change in sludge age. The percentage COD in the waste is the only parameter that is expected to change with the sludge age. A shorter sludge age results in more mixed liquor being wasted and hence the percentage COD in the waste will be proportionally more. This can clearly be seen from Table 4.4: The 10 day sludge age configuration has an overall average of 26.7% of the influent COD in the waste flow, while the 8 day sludge age configuration has 32.8% and the 5 day sludge age configuration 39.2% (which is probably somewhat high as noted above) - an increase for each respective reduction in sludge age. The percentage COD in the unfiltered effluent shows a similar trend to the COD in the waste sludge, but this cannot be as a result of the lower sludge age. The influent flow for the 8 and 5 day sludge age configurations was increased from 20 to 25 ℓ/d and this put greater strain on the final settler causing a greater fraction of the dispersed suspended solids to spill over with the effluent. The filtered effluent COD concentration remained essentially unchanged at 36.2 mgCOD/ℓ.

The overall COD reduction for the three sludge age configurations are all within 4% of each other (94% for the 10 day sludge age configuration, 93% for the 8 day and 90% for the 5 day sludge age configuration), indicating that the COD removal of the ENBNR activated sludge system in not affected to any great extent by a reduction in sludge age.

4.3.2 Nitrogenous material removal

Table 4.5a lists the results for the nitrite and nitrate mass balances over each reactor and settler for sewage batches 31 to 34. A negative value indicates nitrification and a positive value denitrification. Table 4.5b and Fig 4.2 give the results for the total N mass balances and all of its components for sewage batches 31 to 34. Table 4.6 lists the denitrification potentials for the pre- and main anoxic reactors for those batches where the nitrate concentration exiting the respective anoxic reactor was >1 mgN/ℓ. Table 4.7 shows a comparison of the main N parameters for the 10, 8 and 5 day sludge age configurations.

TABLE 4.5a: Nitrite and nitrate mass balances across each reactor and settling tank for sewage batches 31 to 34.

Average of Batch	NITRITE						
	Δ PreANO mgN/d	Δ Anaerobic mgN/d	Δ Int. Set. A mgN/d	Δ Int. Set. B+Nit mgN/d	Δ Anoxic mgN/d	Δ Aerobic mgN/d	Δ Fin. SET mgN/d
31	-0.0	-0.2	-0.1	-1.1	0.1	0.8	0.3
32	2.9	-1.4	-2.0	-1.6	-30.2	23.8	4.4
33							
Config 4	1.4	-0.8	-1.0	-1.3	-15.0	12.3	2.3
34	0.9	17.3	0.4	1.0	-15.7	-43.1	18.4
Config 5	0.9	17.3	0.4	1.0	-15.7	-43.1	18.4
Average of Batch	**NITRATE**						
	Δ PreANO mgN/d	Δ Anaerobic mgN/d	Δ Int. Set. A mgN/d	Δ Int. Set. B+Nit mgN/d	Δ Anoxic mgN/d	Δ Aerobic mgN/d	Δ Fin. SET mgN/d
31	14.1	-1.7	-3.4	-738.3	740.5	-26.5	0.7
32	71.8	-2.0	-4.7	-984.2	913.2	-104.2	16.7
33							
Config 4	42.0	-1.9	-4.0	-851.2	826.9	-65.4	8.2
34	150.8	263.3	1.8	-1415.1 *	676.1	-48.0	-46.5
Config 5	150.8	263.3	1.8	-1415.1	676.1	-48.0	-46.5

TABLE 4.5b: N mass balance with all components for sewage batches 31 to 34.

Average of Batch	Sum NO2 denitrified mgN/d	Sum NO3 denitrified mgN/d	N Wasted mgN/d	N in Effluent mgN/d	N loss Nitrifier mgN/d	Sum N Out mgN/d	TKN in mgN/d	% Recovery	% N Removal
31	1.5	799.2	377.4	117.1	131.0	1386.3	1506.3	96.2	92.7
32	34.4	1015.2	382.6	181.2	73.0	1686.3	1997.6	84.4	90.9
33									
Config 4	18.0	887.2	380.0	149.1	102.0	1536.3	1802.9	85.3	91.9
34	39.4	1103.8	560.4	518.0	-152.7	2068.8	2188.0	94.5	76.3
Config 5	39.4	1103.8	560.4	518.0	-152.7	2068.8	2188.0	94.5	76.3

Fig 4.2:   N mass balance components.

TABLE 4.6:   Denitrification potential of the pre- and main anoxic reactors for sewage batches where the outflow NO_x concentration exceeds 1 mgN/ℓ, for sewage batches 31 to 34.

Batch	Main Anoxic		Pre Anoxic		Total Anoxic Mass Fraction of System
	NO(x) Denitrified mgN/d	Denit. Potential mgN/ℓ infl.	NO(x) Denitrified mgN/d	Denit. Potential mgN/ℓ infl.	
31	740.06		14.1		0.55
32	913.2	45.7	74.7		0.55
33					
34	876.1	33.8	151.7	7.6	0.55

The overall average N mass balance for the 10 (Configuration 3b, Chapter 3), 8 and 5 day sludge age configurations are 88.1%, 85.3% and 94.5% respectively. As for the overall COD mass balances, the 10 and 8 day sludge age configurations have a similar overall average N mass balance, while that for the 5 day sludge age configuration is higher because the VSS concentration had not yet reached a steady state value. As was the case with the COD mass balance components, the variations in the N mass balance components are a result of factors that are independent of reducing the sludge age, with the exception of the N in the waste sludge. The overall average nitrite denitrification of 2.1% for the 10 day sludge age configuration is higher than the 1.0% and 1.8% for the 8 and 5 day sludge ages respectively. The percentage of the influent N denitrified via nitrate denitrification was 40.7% for the 10 day sludge age configuration and 49.2 and 50.4% for the 8 and 5 day sludge age configurations respectively. The percentage N leaving the system in the waste sludge increased with decreasing sludge age as expected, from 19.4% for the 10 day sludge age configuration to 21.1 for the 8 day, and 25.6% for the 5 day sludge age configuration. This is the result of a greater volume of mixed liquor wasted, particularly for the 5 day configuration because the VSS concentration had not yet

TABLE 4.7: Comparison of average N parameters for 10, 8 and 5 day sludge age configurations.

	ENBNRAS System Configurations		
	3	4	5
	10 Day Sludge Age	8 Day Sludge Age	5 Day Sludge Age
N Balance	88.1 %	85.3 %	94.5 %
% NO$_2$ Denit.	2.1	1.0	1.8
% NO$_3$ Denit.	40.7	49.2	50.4
% N Wasted	19.4	21.1	25.6
% N in Effluent	9.2	8.3	23.7
% N Loss to EN System	16.7	5.7	-7.0
% N Unaccounted	11.9	14.5	5.4
Nitrification Occurring Externally	91.1 %	92.9%	94.5 %
TKN in Final Effluent (Unfiltered)	4.8 mgN/l	4.1 mgN/l	5.9 mgN/l
FSA in Final Effluent	3.6 mgN/l	3.3 mgN/l	2.8 mgN/l
NOx in Final Effluent	3.2 mgN/l	1.90 mgN/l	17.8 mgN/l
Total N in Final Effluent	8.0 mgN/l	6.7 mgN/l	24.7 mgN/l
Denit. Pot. Pre-Anoxic Reac.	3.7 mgN/l influent	-	7.6 mgN/l influent
Denit. Pot. Main Anoxic Reac.	31.1 mgN/l influent	45.7 mgN/l influent	33.8 mgN/l influent
TKN/COD Ratio of Influent	0.107	0.096	0.12
Total N Reduction	90.8 %	91.8%	76.3%

reached a steady state value. The percentage N in the final effluent of 9.2%, 8.3% and 23.7% for the 10, 8 and 5 day sludge age configurations respectively are difficult to compare as they are a function of the nitrate concentration in the effluent which depends on the TKN/COD ratio of the influent as well as the nitrification and denitrification performance of the system.

One of the main motives in implementing the ENBNR activated sludge system configuration was to uncouple the nitrification process from the main system and hence making nitrification independent of sludge age. Table 4.7 clearly shows the success of this system configuration - nitrification remained completely unaffected by the lowering of the sludge age. For the 8 and 5 day sludge age configurations 92.9 and 94.5% of the system nitrification occurred externally. This is even higher than the 90.8% for the 10 day sludge age configuration. From the final effluent FSA concentrations given in Table 4.7, it can be seen that full nitrification occurred throughout the 8 and 5 day sludge age configurations with only the residual FSA (from the internal settler underflow) appearing in the effluent.

The overall average denitrification potential of the main anoxic reactor was 31.1 mgN/ℓ influent, 45.7 mgN/ℓ influent and 33.8 mgN/ℓ influent for the 10, 8 and 5 day sludge age configurations respectively. The overall average TKN removal was 94.0%, 93.9% and 92.1% for the 10, 8 and 5 day sludge age configurations respectively. The TKN removals are very similar and all above 90%, which is a very good result. The total N (TN) removal (TKN and NO$_x$) was 90.8%, 91.8% and 76.3% for the 10, 8 and 5 day sludge age configurations respectively. The TN removals show more variation, but this is not due to the decrease in sludge age. The criteria that govern the TN removal are the system nitrification and denitrification, as well as the TKN/COD ratio of the influent. For an influent with, for example, a high TKN/COD ratio, full nitrification and poor denitrification the concentration of nitrate in the effluent will be high, leading to a lower overall TN removal. This is reflected in the results for the 10, 8 and 5 day sludge age configurations. The

overall average TKN/COD ratio of the influent for the 10 day sludge age configuration was 0.107 mgN/mgCOD and the denitrification potential of the main anoxic reactor 31.1 mgN/ℓ influent, resulting in an overall average TN removal of 90.8%. The overall average influent TKN/COD ratio for the 8 day sludge age configuration was 0.096 mgN/mgCOD, and the denitrification potential of the main anoxic reactor was 45.7 mgN/ℓ influent, resulting in an overall average TN removal of 91.8% - showing that the lower influent TKN/COD ratio combined with the higher denitrification potential of the main anoxic reactor resulted in a better TN removal performance for the 8 day sludge age configuration. The influent TKN/COD ratio of the 5 day sludge age configuration was a high 0.120 mgN/mgCOD and the denitrification potential of the main anoxic reactor was 33.8 mgN/ℓ influent (higher than for the 10 day, but significantly lower than for the 8 day sludge age configuration), which resulted in a overall average TN removal of only 76.3%, which is lower than that of both the 10 and 8 day sludge age configurations.

Accepting the variations occurring in the N removal parameters because of varying influent sewage characteristics as well as varying denitrification performance, the lowering of the sludge age did not have any marked effect on either nitrification, denitrification or TKN and TN removal. The results achieved for the 10 (Configuration 3b, Chapter 3), 8 and 5 day sludge age configurations are very similar and this shows that the ENBNR activated sludge system configuration is able to attain high N removals at sludge ages up to as low as 5 days.

4.3.3 Biological Excess Phosphorus Removal (BEPR)

Table 4.8 shows the results of the P mass balances over each of the reactors and settling tanks for sewage batches 31 to 34, and these are summarised in Fig 4.3. A negative result indicates P release while a positive result indicates P uptake. Table 4.9 shows a comparison for the main P parameters for the 10, 8 and 5 day sludge age configurations.

FIGURE 4.8: Average P release (-ve) or P uptake (~ve) for each reactor/settler and total P removal for sewage batches 31 to 34.

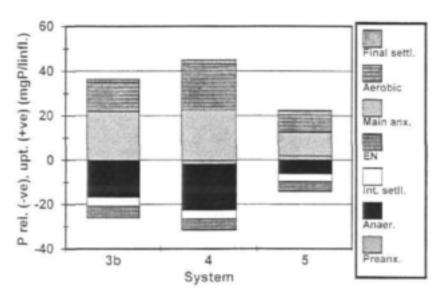| Δ Anaerobic mgP/ℓ inf	Δ in SET A mgP/ℓ inf	Δ in SET B+Nit mgP/ℓ inf	Δ Anoxic mgP/ℓ inf	Δ Aerobic mgP/ℓ inf	Δ Fin SET mgP/ℓ inf	ΣΔ	Total P Removal	% Recovery
-17.9	-4.2	-4.6	22.1	21.3	2.5	18.5	18.5	100
-23.7	-3.5	-4.4	23.7	19.5	1.1	12.5	12.5	100
-20.9	-3.9	-4.5	22.9	20.4	1.8	14.0	14.0	100
-6.25	-3.42	-4.11	10.76	7.82	1.71	8.63	8.63	100
-6.25	-3.42	-4.11	10.76	7.82	1.71	8.63	8.63	100

Fig 4.3: P release (rel., -ve) and P uptake (upt., +ve) for anaerobic (anaer.), internal settling tank (Int. settl.), external nitrification (EN), pre-anoxic (Preanx.), main anoxic (Main anx.), aerobic and final settling tank (Final settl.) for the three investigations.

TABLE 4.9: Comparison of average P parameters for the 10, 8 and 5 day sludge age configurations.

	ENBNRAS System Configurations		
	3	4	5
	10 Day Sludge Age	8 Day Sludge Age	5 Day Sludge Age
NO_3 Flowing into Anaerobic Reac.	0.95 mgN/ℓ	0 mgN/ℓ	10.9 mgN/ℓ
P Release (excl. rel. in EN sys.)	30.8 mgP/ℓ influent	26.6 mgP/ℓ influent	9.67 mgP/ℓ influent
P uptake	36.4 mgP/ℓ influent	49.1 mgP/ℓ influent	22.4 mgP/ℓ influent
Anoxic P uptake	63.2%	47.1%	57.9%
P Removal	10.5 mgP/ℓ influent	14.0 mgP/ℓ influent	8.6 mgP/ℓ influent

A reduction in sludge age increases P removal per mass of organic load (Wentzel et al., 1990), provided that it is not reduced below some lower limit (~ 3 days) to prevent the PAOs from being 'washed' out of the system completely. It would therefore be expected that the 8 and 5 day sludge age configurations should show improved P removal compared to the 10 day sludge age system configuration. From Table 4.9 it can be seen that the overall average P removal was 10.5 mgP/ℓ influent, 14.0 mgP/ℓ influent and 8.6 mgP/ℓ influent for the 10, 8 and 5 day sludge age configurations respectively. The low P removal for the 5 day sludge age configuration occurred because the influent TKN/COD ratio was very high and this led to a high concentration of nitrate (10.9 mg/ℓ, see Table 4.9) being recycled to the anaerobic reactor and limiting the P release, resulting in a low overall P removal performance of 8.6 mgP/ℓ influent. However, this is not linked to the short sludge age, but rather to the main anoxic reactor being overloaded by the high nitrate load that resulted from the high TKN/COD ratio of the influent. The 8 day sludge age

configuration showed the highest P removal (14.0 mgP/ℓ influent), but there was no nitrate leaking into the anaerobic reactor during this configuration and this contributed to the higher overall P removal. The 10 day sludge age configuration achieved an overall average P removal of 10.5 mgP/ℓ influent with an average of 0.95 mgN/ℓ flowing into the anaerobic reactor from the pre-anoxic reactor. Even with zero nitrate flowing into the anaerobic reactor the 10 day sludge age configuration would not have achieved an average P removal of 14.0 mgP/ℓ influent, showing that a reduction in sludge age does improve the P removal performance. Had no nitrate flowed into the anaerobic reactor during the 5 day sludge age configuration, it would probably have achieved higher P removal than the 8 day sludge age configuration did.

For the 10 day sludge age configuration, an overall average of 63.3% of the P uptake occurred in the anoxic reactor. For the 8 and 5 day sludge age configurations, the percentage anoxic P uptake was 47.1 and 57.9% respectively. The average NO_x load on the main anoxic reactor was 18.6 mgN/ℓ, 20.7 mgN/ℓ and 34.9 mgN/ℓ for the 10, 8 and 5 day sludge age configurations respectively. The higher NO_x load on the anoxic reactor for the 5 day sludge age configuration led to the 10.8% higher anoxic P uptake compared to that of the 8 day sludge age configuration. The 8 day sludge age configuration had a 2.4 mgN/ℓ higher NO_x load on the anoxic reactor than the 10 day sludge age configuration, but 14.8% lower anoxic P uptake. This is most likely because the 8 day sludge age configuration result is an average of only two sewage batches, while the result from the 10 day sludge age configuration is the average of 10 sewage batches. Had the 8 day sludge age configuration been run for 10 sewage batches, the result would have been closer to that of the 10 day sludge age configuration.

When the ENBNR activated sludge system is operated at lower sludge ages, an improvement in the overall P removal can be expected. Considerable anoxic P uptake continues to occur at the lower sludge ages, and the percentage anoxic P uptake continues to shift with the NO_x load on the main anoxic reactor.

4.3.4 Sludge settleability

The overall average DSVI was 95.6 mℓ/g, 89.8 mℓ/g and 92.9 mℓ/g for the 10 (Configuration 3b, Chapter 3), 8 and 5 day sludge age configurations respectively. This shows that the decrease in sludge age had no effect on the DSVI performance of the ENBNR activated sludge system. It cannot be said that the lowering of the sludge age produced a better settling sludge, because the values of the 8 and 5 day sludge age configurations are close to those obtained for the 10 day sludge age configuration (Configuration 3b). Even with a nitrate concentration of 15 mgN/ℓ flowing from the main anoxic reactor (see Table 4.2c) for the 5 day sludge age configuration the DSVI deteriorated only very slightly and did not rise above 100 ml/g as it did for the 10 day sludge age system configuration when similarly high nitrate concentrations flowed from the anoxic reactor during the period that the system was recovering from the bad sewage batch 9.

4.4   CONCLUSIONS

The ENBNR activated sludge system configuration was changed to shorter sludge ages and increased influent flow towards the end of the investigation on System Configuration 3b (Chapter 3). The system was operated in Configuration 4 (8 days sludge age and 25 instead of 20 ℓ/d influent flow) for 49 days (sewage batches 31 to 33) and in Configuration 5 (5 days sludge age and 25 ℓ/d influent flow) for a further 13 days (sewage batch 34). The system configuration was changed to the shorter sludge ages to evaluate the response to and performance of the ENBNR activated sludge system to the shorter sludge ages.

1. The overall average COD mass balances for the 8 and 5 day sludge age configurations (4 and 5 respectively) of the ENBNR activated sludge system were 79 and 92% respectively and the overall average COD removals (based on unfiltered COD samples) 93 and 90% respectively. Although lower than the overall average COD removal of 94% achieved by the 10 days sludge age configuration (Configuration 3b, Chapter 3), the COD removal performances of the short sludge age configurations are still very good. The slightly lower values are most likely due to the hydraulic impact on the final settler caused by the increase in influent flow, rather than due to the system removing less COD.

2. The overall average N mass balances attained for the 8 and 5 day sludge age configurations were 85 and 95% respectively. For the 8 and 5 day sludge age configurations, 88 and 96% of the FSA flowing into the EN system was nitrified. On average, 93 and 95% of the ENBNR activated sludge system nitrification was effected externally for the 8 and 5 day sludge age configurations respectively. This demonstrates one of the main benefits of the ENBNR activated sludge system configuration - virtually complete nitrification at 8 and 5 days sludge age with only 0.20 aerobic mass fraction, and this will be attainable also at lower temperatures than 20°C.

3. The overall average denitrification potential of the main anoxic reactor was about 46 and 34 mgN/ℓ influent for the 8 and 5 day sludge age configurations respectively. These are higher than observed at ten days sludge age, even when the system denitrification was greatest (i.e. Configuration 3, 31 mgN/ℓ).

4. The overall average TKN removal for the ENBNR activated sludge system operated at the 8 and 5 day sludge age configurations were 94 and 92% respectively. The TN removal was 92 and 76% respectively. The discrepancies in the TN removals for the 8 and 5 day sludge age configurations are due to the average influent TKN COD ratios of the sewage batches fed to the two configurations; for the 8 day sludge age configuration this was 0.096, while for the 5 day sludge age configuration it was much higher at 0.120 mgN mgCOD. The higher influent TKN COD ratio and lower denitrification potential of the 5 day sludge age configuration led to more nitrate in the effluent, and therefore to a lower TN removal.

5. The overall average P removal for the 8 day sludge age configuration was 14.0 mgP/ℓ influent. For the 5 day sludge age configuration this was 8.6 mgP/ℓ influent. The 8 day sludge age configuration removed 4.2 mgP/ℓ influent more than the 10 days sludge age configuration, showing that there is indeed an improvement in BEPR as the sludge age

is decreased as reflected in the BEPR model of Wentzel *et al.* (1990). It would therefore be expected that the 5 day sludge age would show a further improved BEPR performance; however, for the 5 day sludge age configuration, high nitrate concentrations were recycled to the pre-anoxic reactor, which was consequently overloaded, causing nitrate to enter the anaerobic reactor, which in turn caused a decrease in P release and hence a decrease in P removal. The nitrate recycle was due to the high influent TKN/COD ratio (0.12) and reduced main anoxic reactor performance.

6. The overall average % anoxic P uptake for the 8 day sludge age configuration was 47% (with a nitrate load of 20.7 mgN/ℓ on the main anoxic reactor), and the overall average % anoxic P uptake for the 5 day sludge age configuration was 58% (with a nitrate load of 34.9 mgN/ℓ on the main anoxic reactor). This shows clearly that as the nitrate load on the main anoxic reactor increases, the % anoxic P uptake increases.

7. The overall average DSVI for the 8 and 5 day sludge age configurations were about 90 mℓ/g and 93 mℓ/g respectively. For the 8 day sludge age configuration <1 mgN/ℓ flowed out of the main anoxic reactor, but for the 5 day sludge age configuration about 15 mgN/ℓ flowed out of the main anoxic reactor. Good sludge settleability at sludge ages less than 10 days (8, 6 and 5 days sludge age) has been observed in intermittently aerated ND systems by Warburton *et al.* (1991) and Phoredox, 3 stage Bardenpho, UCT and JHB BEPR systems (Burke *et al.*, 1986). The AA (low F/M) filament bulking hypothesis (Casey *et al.*, 1994) is not considered to be applicable at 3 to 8 days sludge age because the AA filaments are slow growers that tend to proliferate in long sludge age (>8d) systems.

A detailed analysis of the results of the 8 and 5 day sludge age configurations is not intended, as the system performance evaluation at these shorter sludge ages lasted for only 62 days. However, the results show that the ENBNR activated sludge system BNR performance in no way deteriorated at the shorter sludge ages, in fact a reduction in sludge age tends to increase N and P removal per mass of organic load (Wentzel *et al.*, 1990), provided that it is not reduced below a lower limit of about 5 days for operational reasons (sludge flocculation, effluent turbidity); in terms of 'wash out' of PAOs the sludge age can be as low as 3 days (Wentzel *et al.*, 1990).

CHAPTER 5

COMPARISON OF EXTERNAL NITRIFICATION (EN) BIOLOGICAL NUTRIENT REMOVAL (BNR) ACTIVATED SLUDGE SYSTEM WITH A CONVENTIONAL BNR (UCT) ACTIVATED SLUDGE SYSTEM

5.1   INTRODUCTION

In implementing the external nitrification BNR activated sludge (ENBNRAS) system, its performance with respect to nutrient removal should not be significantly reduced compared to the conventional BNR activated sludge system.   With regard to nitrogen removal via denitrification, the investigations in Chapters 3 and 4 indicate that the ENBNRAS system is superior in that complete denitrification can be achieved for influent wastewater TKN/COD ratios that are considerably higher than those for which it can be achieved in conventional BNR systems. However, with regard to P removal, the ENBNRAS system stimulates anoxic P uptake in the system which would appear to be reduced compared to aerobic P uptake. This indicates that the P removal performance of the ENBNRAS system may be reduced compared to the conventional system. In this Chapter this aspect is examined in more detail, to compare more precisely the N and P removal performance in the external nitrification and conventional BNR systems. Also, the factors stimulating anoxic P uptake in the conventional BNR system are investigated. This investigation is summarised in this Chapter, and reported in detail by Vermande *et al*. (2000).

5.2   SYSTEM OPERATION

A laboratory-scale BNRAS system (UCT configuration) with similar design and operating parameters to the ENBNRAS system Configuration 3b (Chapters 3 and 4) was run in parallel with this laboratory- scale ENBNRAS system. Figure 5.1 shows the system layout of the laboratory-scale UCT system and Table 5.1 lists the design and operating parameters for both the ENBNRAS and UCT systems. In order to compare the performance of the two systems, they were fed the same influent sewage for 18 sewage batches (sewage batches 13 to 30, Table 3.1 Configuration 3) - from the 7 August 1999 (day 167) to the 17 April 2000 (day 421). 40 ℓ of influent were prepared in the same container, thoroughly stirred, and 20 ℓ of this prepared sewage was fed to each of the systems respectively. While the ENBNRAS system was operated and analysed by Sötemann *et al*. (2000), the UCT system was operated, tested and reported on by Vermande *et al*. (2000); for details the reader is referred to the relevant report. Detailed comparisons of the two systems are also given in both reports.

FIGURE 5.1: Schematic layout of the laboratory scale UCT system run in parallel with the laboratory scale ENBNRAS system.

TABLE 5.1: ENBNRAS and UCT system design and operating parameters.

Parameter	UCT System	ENBNRAS System
Influent Flow (l/d)	20	20
Sludge Age (d)	10	10
Temperature (°C)	20	20
D.O. Main Aerobic Reactor (mgO/l)	2 to 5	2 to 5
Total System Volume (l)	20	20
Pre-Anoxic Reactor (l)	-	2*
Anaerobic Reactor (l)	3**	5
Main Anoxic Reactor (l)	7	6.5 (for sewage batch 13) / 9 (for sewage batches 14 to 30)
Main Aerobic Reactor (l)	10	6.5 (for sewage batch 13) / 4 (for sewage batches 14 to 30)
Aerobic Mass Fraction	0.5	0.33 (for sewage batch 13) / 0.2 (for sewage batches 14 to 30)
Unaerated Mass Fraction	0.5	0.67 (for sewage batch 13) / 0.8 (for sewage batches 14 to 30)
Anoxic Mass Fraction	0.35	0.42 (for sewage batch 13) / 0.55 (for sewage batches 14 to 30)
Anaerobic Mass Fraction	0.15	0.25
a-recycle (w.r.t infl. flow)	1:1	2:1 (for sewage batches 13 to 20) / 0:1 (for sewage batches 21 to 30)
s- recycle (w.r.t infl. flow)	1:1	1:1
r - recycle (w.r.t infl. flow)	1:1	-

* Actual volume 1l, with sludge at double concentration.
** Actual volume 6l, with sludge diluted to half the normal concentration.

5.3   SYSTEM PERFORMANCE

Tables 5.2a, b and c list the sewage batch averages for all measured parameters for sewage batches 13 to 30 for the UCT system. The sewage batch numbers correspond to the sewage batch numbers used for the ENBNRAS system in Chapters 3 and 4 for Configuration 3, to facilitate the direct comparison of the two systems. Where the overall averages of the two systems are compared in this section, the overall averages refer to the average of the sewage batch averages for sewage batches 13 to 30. This also applies to the ENBNRAS system results - in this section the overall averages refer to the average of the sewage batch averages for sewage batches 13 to 30 calculated from the data in Chapter 3. Therefore, the overall averages may differ from those mentioned in Chapter 3, where the overall averages include sewage batches 1 to 13 also.

It should further be noted that because the chemical tests were performed by two different researchers, the influent sewage characteristics show some minor differences, even though the respective feeds originated from the same feed mixing container. This is as a result of the independent analyses and associated analytical variations, but these do not impact significantly on the comparison, because the overall averages of the influent sewage characteristics over the 18 sewage batches are almost identical. Furthermore, both independent analyses arrived at the same overall average mixed liquor COD/VSS ratio of 1.43 and TKN/VSS ratio of 0.1 over the 18 sewage batches.

TABLE 5.2a: Sewage Batch averages of measured COD and TKN parameters for the UCT system for sewage batches 13 to 30 (UF = unfiltered; F = 0.45µm membrane filtered).

Sewage Batch	mgCOD/l				mgN/l						TKN/COD Ratio
	COD				TKN				FSA		
	Influent	Aerobic ML	Unflt Eff	Flt Eff	Influent	Aerobic ML	Unflt Eff	Flt Eff	Influent	Unflt Eff	Influent
13	790.9	2651.4	67.4	43.2	82.1	150.4	2.6	2.1	61.3	1.9	0.104
14	685.6	2732.2	57.5	37.6	71.3	101.7	2.7	1.3	52.7	1.4	0.104
15	723.1	2170.1	55.3	43.7	89.2	160.5	5.4	1.9	69.8	2.0	0.123
16	732.3	2595.5	61.5	47.8	83.0	168.0	3.2	1.6	62.3	1.1	0.113
17	675.5	2492.9	43.8	31.1	63.8	171.3	3.2	1.8	47.8	1.1	0.094
18	784.3	2505.5	49.7	38.6	64.7	161.2	3.0	1.6	52.0	0.6	0.083
19	788.7	2084.2	46.5	39.7	84.5	156.8	3.1	2.3	68.2	0.6	0.107
20	784.9	2320.7	54.9	43.0	69.1	159.4	3.1	2.8	50.3	1.1	0.088
21	715.5	2538.7	55.2	37.7	63.2	163.5	4.1	3.3	49.9	2.5	0.088
22	718.6	2411.8	55.3	43.2	77.2	163.3	5.1	4.7	60.9	2.9	0.107
23	760.1	2386.3	56.9	39.7	79.9	150.4	5.2	3.8	61.4	2.7	0.105
24	721.1	2224.2	42.6	35.5	86.7	145.6	4.9	3.1	68.3	2.4	0.120
25	694.6	2510.5	42.0	35.8	58.2	160.9	3.7	2.8	46.7	2.1	0.084
26	699.1	2294.6	46.4	38.5	84.5	150.9	4.5	3.1	70.6	1.9	0.121
27	687.9	2775.3	53.9	38.6	82.2	169.3	4.6	3.3	72.6	2.1	0.119
28	716.2	2850.3	52.9	33.0	88.1	188.1	5.1	3.1	71.4	2.6	0.123
29	748.9	2784.8	56.5	41.2	67.3	180.0	4.5	3.2	57.2	2.2	0.090
30	799.1	2761.4	56.3	42.7	68.3	172.8	3.5	2.1	53.9	1.4	0.085
Overall	734.8	2505.0	53.0	39.5	75.7	160.8	4.0	2.6	59.9	1.8	0.103
*	UF	UF	UF	F	UF	UF	UF	F	UF	UF	UF

TABLE 5.2b: Sewage batch averages of measured suspended solids, OUR, DSVI and pH for the UCT system for sewage batches 13 to 30.

Sewage Batch	TSS	VSS	COD/VSS Ratio [1] Aerobic	TKN/VSS Ratio [1] Aerobic	mgO/l.h OUR Aerobic	ml/g DSVI Aerobic	pH Anaerobic	pH Aerobic
13	2226.4	1838.4	1.42	0.082	32.4	116.2	-	-
14	2102.7	1704.3	1.58	0.096	39.0	73.4	-	-
15	2002.7	1624.0	1.31	0.097	29.3	94.3	-	-
16	2066.3	1699.0	1.50	0.098	36.7	114.3	-	-
17	2128.3	1698.0	1.45	0.101	31.0	112.0	-	-
18	1946.3	1567.7	1.57	0.102	31.1	119.8	-	-
19	1777.8	1444.5	1.42	0.109	32.3	121.1	-	-
20	1988.3	1609.3	1.42	0.100	29.3	112.8	-	-
21	2207.3	1780.7	1.40	0.103	31.8	112.1	-	-
22	2137.6	1696.5	1.40	0.096	35.5	145.1	-	7.77
23	1931.3	1580.4	1.48	0.096	34.5	186.8	-	7.81
24	1972.7	1642.4	1.33	0.089	35.3	197.1	7.68	7.87
25	2208.4	1813.6	1.36	0.089	30.3	183.1	7.56	7.91
26	2056.3	1672.5	1.35	0.090	33.3	201.0	7.61	7.82
27	2480.0	1923.1	1.42	0.088	27.6	154.9	7.53	7.83
28	2499.2	2006.7	1.40	0.094	28.0	157.2	7.51	7.80
29	2409.0	1878.0	1.46	0.096	26.5	144.1	7.35	7.91
30	2492.8	1900.3	1.43	0.091	28.0	141.6	6.89	7.14
Overall	2146.3	1726.8	1.43	0.095	31.8	138.1	7.4	7.76

[1] Calculated from unfiltered aerobic reactor COD and TKN concentrations divided by the VSS.

TABLE 5.2c: Sewage batch averages for measured nitrate, nitrite and P concentrations for the UCT system for sewage batches 13 to 30.

Sewage Batch	Nitrite				Nitrate				mgP/l Phosphates					
	Anaerobic	Anoxic	Aerobic	Filt Eff	Anaerobic	Anoxic	Aerobic	Filt Eff	Influent	Anaerobic	Anoxic	Aerobic	Unfilt Eff	Filt Eff
13	0.1	0.5	1.8	1.5	0.5	1.2	13.1	13.4	26.2	31.1	21.2	14.7	-	14.3
14	0.0	0.1	0.6	0.4	0.3	2.4	15.5	13.4	26.2	31.3	17.4	8.3	-	7.3
15	0.0	0.2	0.7	0.4	0.3	5.3	20.0	22.2	30.8	30.2	19.4	14.6	-	13.9
16	0.0	0.1	1.1	1.0	0.1	1.2	13.4	13.6	25.0	33.4	22.6	13.6	-	14.0
17	0.1	0.2	0.3	0.2	0.1	0.3	8.3	8.6	25.4	34.5	21.3	12.6	-	13.0
18	0.0	0.1	0.2	0.2	0.1	0.3	8.4	7.8	26.9	35.9	23.5	14.1	-	14.7
19	0.0	0.3	0.5	0.2	0.1	0.9	11.2	12.2	21.0	30.9	21.3	13.0	-	12.6
20	0.1	0.1	0.2	0.1	0.1	0.3	8.3	8.6	25.8	32.9	21.8	13.5	-	13.5
21	0.1	0.1	0.1	0.1	0.2	0.3	7.0	7.9	25.2	33.6	22.2	14.9	15.1	15.1
22	0.1	0.9	0.5	0.2	0.2	1.0	10.5	11.5	24.6	28.7	19.5	14.6	15.1	14.9
23	0.1	0.8	0.7	0.5	0.2	1.7	12.5	13.4	23.0	25.8	18.9	15.8	15.5	15.7
24	0.1	0.5	0.4	0.6	0.2	1.0	12.5	13.3	25.2	29.2	19.0	15.0	14.9	14.6
25	0.0	0.1	0.2	0.1	0.1	0.2	8.0	8.2	28.7	33.3	23.3	18.0	18.0	17.7
26	0.0	0.5	0.8	0.4	0.1	2.4	15.1	17.2	29.1	35.0	23.8	17.5	16.8	16.6
27	0.0	0.4	0.3	0.2	0.2	2.9	15.7	16.9	25.6	35.8	19.5	11.9	11.8	11.1
28	0.0	0.4	0.4	0.3	0.2	1.6	15.0	16.1	25.3	38.6	22.9	13.1	12.6	12.0
29	0.1	0.1	0.3	0.1	0.1	0.1	8.1	8.3	24.9	41.9	20.0	8.4	9.0	8.2
30	0.0	0.1	0.1	0.2	0.1	0.6	9.8	10.0	26.8	41.7	21.3	9.5	8.8	8.5
Overall	0.1	0.3	0.5	0.4	0.2	1.3	11.8	12.4	25.9	33.5	21.0	13.5	13.8	13.2

5.3.1 Carbonaceous Material Removal

Figure 5.2 shows the overall COD mass balances achieved for sewage batches 13 to 30 for the UCT and the ENBNRAS systems. The overall average COD mass balance achieved for the UCT system is 78.3% and that for the ENBNRAS system is 76.8%. These values are within 2 percent, indicating that while the overall average COD mass balance for the UCT system is 1.5% higher than the overall average COD mass balance achieved for the ENBNRAS system, both are equally low. This indicates that the same as yet unidentified biological process which is thought to consume a fraction of the influent COD without being taken account of in the usual analytical procedures also occurred in the UCT system, and it confirms that the low COD balances are not characteristics of the ENBNRAS system alone, but rather a characteristic of BNRAS systems in general. From Figure 5.2 it can be seen that the COD mass balances achieved for each sewage batch are similar. It seems that the largest discrepancies occur at low and very high influent TKN/COD ratios, with the ENBNRAS system achieving better COD balances for sewage batches with a very high influent TKN/COD ratio (e.g. sewage batches 15, 19, 26, 27 and 28 with influent TKN/COD ratios of 0.124, 0.116, 0.118, 0.111 and 0.123 respectively), and the UCT system achieving higher COD mass balances for sewage batches with lower influent TKN/COD ratios (e.g. sewage batches 18, 21, 22 and 25 with influent TKN/COD ratios of 0.087, 0.089, 0.107 and 0.085 respectively).
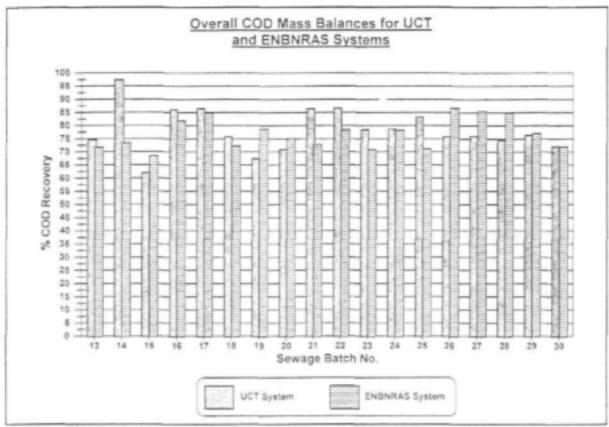
FIGURE 5.2: COD mass balances for the UCT and ENBNRAS systems for sewage batches 13 to 30.

On average over sewage batches 13 to 30, the UCT system influent COD was 735 mgCOD/ℓ and the ENBNRAS system influent COD was 731 mgCOD/ℓ. The overall average influent COD values are within 1% of each other, confirming that the two systems did indeed receive the same feed even though there are variations in the influent COD values for each of the separate sewage batches. Figure 5.3 shows the COD removal performance for each of the two systems, as a percentage of the influent COD concentration fed to each system. From Figure 5.3 it can be seen that the COD removal performances of the two systems are virtually identical. The UCT and ENBNRAS systems removed an overall average of 92.8% and 93.5% of the influent COD respectively; while the ENBNRAS system removed 0.7% more COD on average, this difference is negligible. BNRAS systems generally remove COD virtually completely irrespective of configuration and this is clearly demonstrated here.

Figure 5.4 shows the daily oxygen demand of the main aerobic reactors for the UCT and the ENBNRAS systems. The oxygen demand is given in units of mgO/d because, being independent of the reactor volume, it gives a more accurate reflection of the oxygen demand in the respective systems.
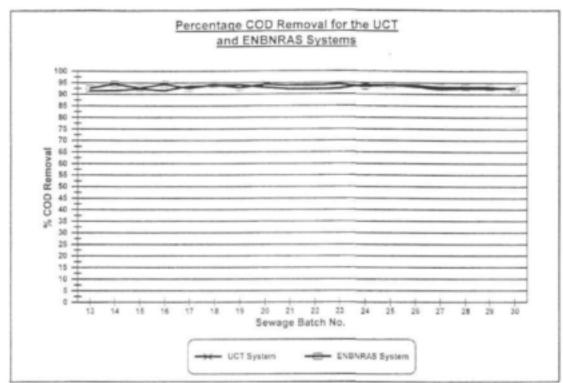
FIGURE 5.3: Percentage COD removal by the UCT and ENBNRAS systems for sewage batches 13 to 30.
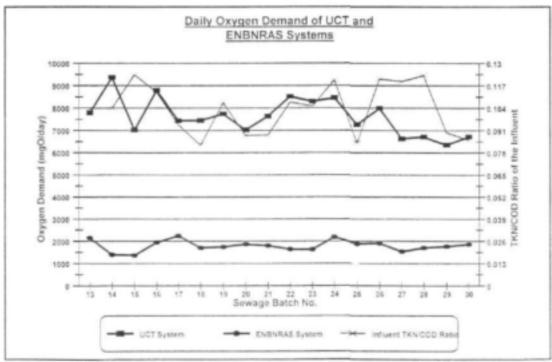

FIGURE 5.4: Daily oxygen demand for the UCT and ENBNRAS systems for sewage batches 13 to 30.

Figure 5.4 shows the advantage of the ENBNRAS system in terms of oxygen demand. The UCT system had an average daily oxygen demand of 7625 mgO/d over the 18 sewage batches, while the ENBNRAS had an average daily oxygen demand of only 1798 mgO/d. By nitrifying externally, the ENBNRAS system requires about 76% less oxygen than the UCT system requires with nitrification taking place internally. The influent TKN/COD ratio is included in Figure 5.4 to illustrate the variation of the daily oxygen demand of the UCT system with the variation of the influent TKN/COD ratio. As the influent TKN/COD ratio increases, more nitrate is produced and the daily oxygen demand of the UCT system rises, and vice versa. The daily oxygen demand of the ENBNRAS system does not show the same variation with varying influent TKN/COD ratios because nitrification occurs externally and is not coupled to the oxygen demand of the system. This results in a more constant daily oxygen demand for the ENBNRAS system, Figure 5.4.

A further interesting comparison can be made regarding the VSS concentrations of the two systems. The UCT system had an overall average VSS concentration of 1727 mgVSS/ℓ while the ENBNRAS systems overall average VSS concentration was 1437 mgVSS/ℓ over sewage batches 13 to 30. The 16.8% lower VSS concentration for the ENBNRAS system almost corresponds to the 18.3% of the influent COD 'lost' to the EN system of the ENBNRAS system. This indicates that the COD that is 'lost' to the EN system is not available to the organisms in the BNRAS system and this will result in the ENBNRAS system containing lower VSS concentrations, roughly in proportion to the fraction of the influent COD that is removed in the EN system and hence 'lost' to the main system.

5.3.2 Nitrogenous Material Removal

Figure 5.5 shows the N mass balances for sewage batches 13 to 30 for the UCT and the ENBNRAS systems. The overall average N mass balance over the 18 sewage batches for the UCT and ENBNRAS systems was 86.1% and 87.0% respectively. As was the case for the COD balances, the results are very close, albeit considerably higher than the respective COD balances, which is usually the case for BNRAS systems in the Water Research Laboratory. From Figure 5.5 it can be seen that the N mass balances for the respective sewage batches are similar with marked differences in the N mass balances only occurring for sewage batches 14, 15, 17 and 22. Figure 5.6 shows the overall TKN reduction achieved by the UCT and ENBNRAS systems, as a percentage reduction of the influent TKN. The TKN reduction achieved by the two systems is very similar. The UCT system achieved an overall average TKN reduction of 94.7% and the ENBNRAS system achieved a slightly lower TKN reduction of 93.8%. The reason for the ENBNRAS system achieving a lower value is the FSA concentration in the final effluent. The effluent FSA of the ENBNRAS system was on average over the 18 sewage batches 3.5 mgN/ℓ FSA, while that of the UCT system final effluent was 1.8 mgN/ℓ FSA. The source of this effluent FSA in the ENBNRAS system is the FSA that bypasses the EN system in the sludge bypass, and because almost no nitrification occurs in the main aerobic reactor, this FSA flows out in the effluent without being nitrified. The concentration of FSA in the EN system outflow of the ENBNRAS system, on average over the 18 sewage batches, was 3.3 mgN/ℓ. This is very similar to the 3.5 mgN/ℓ FSA in the final effluent and indicates that (i) the FSA in the final effluent is approximately equal to the FSA that was not nitrified in the EN system, (ii) the FSA that bypasses the EN system in the EN system sludge bypass is nitrified in the aerobic reactor and (iii) the FSA released in the main anoxic reactor is also nitrified in the aerobic reactor. The nitrifiers are seeded into the activated sludge with the EN system outflow, and this is how the system is

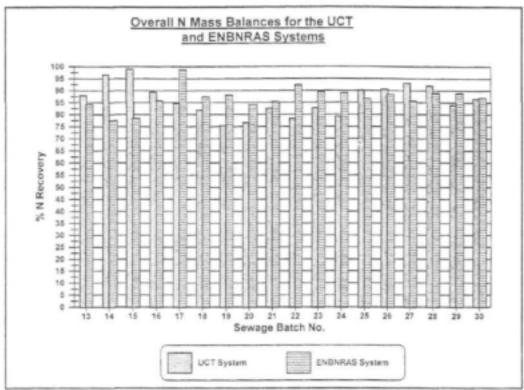intended to operate to maintain a low final effluent FSA.

FIGURE 5.5:   N mass balances for the UCT and ENBNRAS systems for sewage batches 13 to 30.
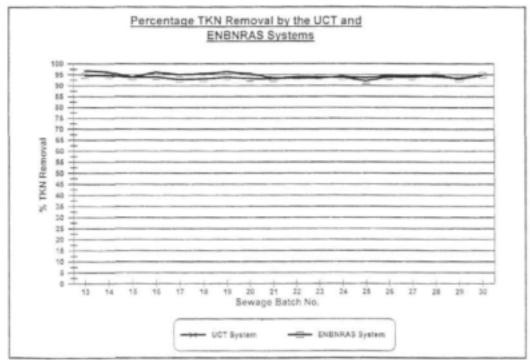

FIGURE 5.6:   Percentage TKN removal for the UCT and ENBNRAS systems for sewage batches 13 to 30.

Figure 5.7 shows the total N concentrations in the effluents of the UCT and the ENBNRAS systems for sewage batches 13 to 30. From Figure 5.7 the difference in the N removal performance of the UCT and the ENBNRAS systems can be seen more clearly. While the TKN measurements take account of only organic N and FSA, the total N represents organic N, FSA, nitrite and nitrate. Figure 5.6 shows that both systems remove TKN equally efficiently, but from Figure 5.7 it can be seen that the ENBNRAS system removes more total N than the UCT system. This means that the ENBNRAS system produces a final effluent with a lower nitrate concentration than the UCT system, because it is the varying concentration of nitrate in the effluent that leads to the variation of the total N in the effluent. On average over sewage batches 13 to 30, the UCT system effluent nitrate was 12.4 mgN/ℓ, while that from the ENBNRAS system was only 4.6 mgN/ℓ of nitrate. This results in the ENBNRAS system having the potential of producing effluents containing <10 mg/ℓ total N, while the UCT system is not capable of achieving similar results[1]. This can also be seen from Figure 5.7: The ENBNRAS system achieved a total N (TN) concentration in the effluent of <10 mgN/ℓ for 10 of the 18 sewage batches, while the UCT system did not once achieve TN concentrations of <10 mgN/ℓ in the effluent. The influent TKN/COD ratio has also been included on Figure 5.7 to illustrate how the variations in the effluent TN concentrations mirror the variations in the influent TKN/COD ratio. On average over the 18 sewage batches, the UCT system effluent TN concentration was 16.7 mgN/ℓ, while that for the ENBNRAS system was 9.8 mgN/ℓ. The main reason for this difference is the potential of the ENBNRAS system to denitrify completely with its larger anoxic mass fraction and its low nitrification in the aerobic reactor. The UCT system cannot denitrify completely because all nitrification takes place in the aerobic reactor.

Figure 5.8 shows the percentage TN removals for the UCT and the ENBNRAS systems for sewage batches 13 to 30. From Figure 5.8 it can be seen that the ENBNRAS system removed a greater percentage N from the influent wastewater than the UCT system for all 18 sewage batches. On average over the 18 sewage batches, the UCT system removed 78.2% of the total influent N and the ENBNRAS system removed 87.8% of the total influent N.

[1] The higher effluent nitrate for the UCT system was due to the smaller anoxic mass fraction. The effluent NO_x could have been reduced somewhat by increasing the a-recycle ratio, but high NO_x concentrations (>3 mgN/ℓ) in the outflow of the anoxic reactor leads to (i) deterioration in sludge settleability and (ii) anoxic P uptake BEPR and lower P removal (see Figures 5.12 and 5.13) below.
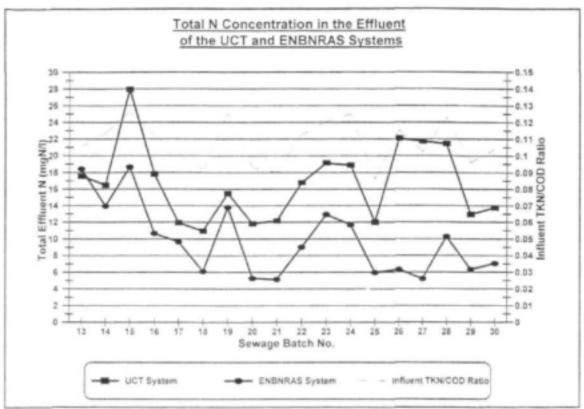
5.13

FIGURE 5.7: Total N concentrations in the effluent of the UCT and ENBNRAS systems and influent TKN/COD ratios for sewage batches 13 to 30.
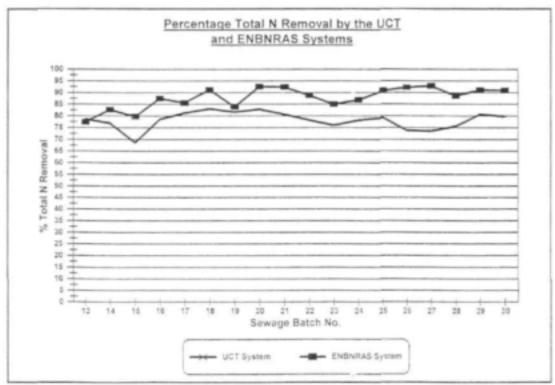

FIGURE 5.8: Percentage total N removal by the UCT and ENBNRAS systems for sewage batches 13 to 30.

5.3.3 Biological Excess Phosphorus Removal (BEPR)

The ENBNRAS system favours anoxic/aerobic P uptake BEPR, while the UCT system favours aerobic P uptake BEPR. However, when the UCT system is fed sewage with a high influent TKN/COD ratio, which results in a high nitrate load on the main anoxic reactor, anoxic P uptake does occur. For sewage batches 21 to 27, the influent TKN/COD ratio was kept consistently high by FSA addition to the influent (>0.100 mgN/mgCOD) to induce anoxic P uptake in the UCT system, so that the BEPR performance of the UCT system with anoxic P uptake as well as with predominantly aerobic P uptake can be compared to the BEPR of the ENBNRAS system. Figure 5.9 shows the percentage anoxic P uptake for both the UCT and the ENBNRAS systems for sewage batches 13 to 30.
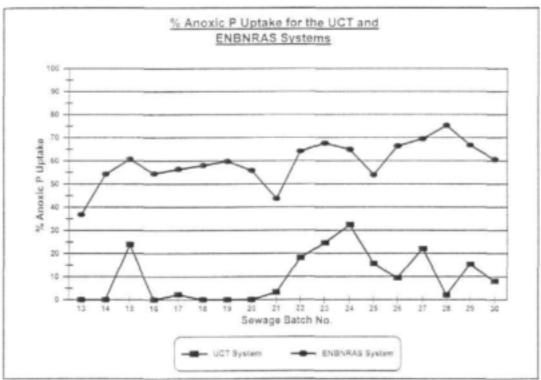

FIGURE 5.9: Percentage anoxic P uptake for the UCT and ENBNRAS systems for sewage batches 13 to 30.

From Figure 5.9 it can be seen that considerable anoxic P uptake (40 to 70%) occurred in the ENBNRAS system throughout the 18 sewage batches, with an overall average over the 18 sewage batches of ~60%. In the UCT system negligible anoxic P uptake occurred for sewage batches 13 to 21, with the exception of sewage batch 15, which had a very high influent TKN/COD ratio of about 0.123. During sewage batches 21 to 27, where the influent TKN/COD ratio was kept consistently above 0.100 (by dosing FSA to the influent), appreciable anoxic P uptake took place in the UCT system (10 to 30%). However, the anoxic P uptake in the UCT system never reached the same magnitude observed in the ENBNRAS system and on overall average over the 6 sewage batches (22 to 27) only 20% anoxic P uptake occurred. This shows that the BEPR in the UCT system was essentially aerobic P uptake BEPR. After sewage batches 21 to 27, the FSA dosing to the influent was stopped which lowered the influent TKN/COD ratio

and underloaded the anoxic reactor with nitrate, and the system returned to predominantly aerobic P uptake.

Figures 5.10 and 5.11 show the P release and P uptake respectively for the UCT and the ENBNRAS systems over the 18 sewage batches. From Figure 5.10 it can be seen that for the sewage batches where there was negligible anoxic P uptake in the UCT system (sewage batches 13,14,16 to 20 and 28 to 30) it released on average ~7 mgP/ℓ influent more P than the ENBNRAS system. However, for the sewage batches where there was anoxic P uptake in the UCT system (sewage batches 15 and 21 to 27) the ENBNRAS system released on average ~3 mgP/ℓ influent more P than the UCT system did. On overall average over all of the 18 sewage batches, the UCT system released 21.3 mgP/ℓ influent and the ENBNRAS system released 18.3 mgP/ℓ influent. When operating with predominantly aerobic P uptake, the UCT system releases more P than the ENBNRAS system does, even though it has a lower anaerobic mass fraction than the ENBNRAS system. However, when anoxic P uptake takes place in the UCT system, the P release drops to lower levels than in the ENBNRAS system. This shows that with anoxic P uptake BEPR in the UCT system (i) less P is released per unit RBCOD than under aerobic P uptake BEPR and (ii) P release decreases also due to the high nitrate load on the anoxic reactor and nitrate recycle to the anaerobic reactor. From Figure 5.11 it can be seen that the P uptake follows exactly the same trend of the P release. The P uptake for the UCT system was 33.2, 26.9 and 50.5 mgP/ℓ influent for sewage batches 13 to 21 (aerobic P uptake), 22 to 27 (anoxic/aerobic P uptake) and 28 to 30 (aerobic P uptake) respectively. That of the ENBNRAS system was 28.1, 35.8 and 41.1 mgP/ℓ influent respectively, with anoxic/aerobic P uptake throughout. For sewage batches 13 to 21, the UCT system P uptake (predominantly aerobic) was about 5 mgP/ℓ influent higher than that of the ENBNRAS system. For sewage batches 22 to 27, when anoxic/aerobic P uptake occurred in the UCT system (20% anoxic P uptake), the P uptake was about 9 mgP/ℓ influent less than that of the ENBNRAS system (64% anoxic P uptake). For sewage batches 28 to 30, when the P uptake in the UCT system had returned to predominantly aerobic P uptake, the P uptake was 9 mgP/ℓ influent higher than that of the ENBNRAS system. On overall average over the 18 sewage batches, the UCT system P uptake was 34.0 mgP/ℓ influent and that of the ENBNRAS system was 32.8 mgP/ℓ.

Figure 5.12 shows the P removal achieved by the UCT and the ENBNRAS systems for sewage batches 13 to 30. In essence the P removal reflects the combination of those tendencies found for the P release and the P uptake. When the UCT system operates with predominantly aerobic P uptake, on average it removes ~4 mgP/ℓ influent more P than the ENBNRAS system. Under conditions where the UCT system does show anoxic P uptake, the ENBNRAS system removes ~2 mgP/ℓ more P than the UCT system. On overall average over the 18 sewage batches, the UCT system removed 12.7 mgP/ℓ influent, while the ENBNRAS system removed 9.8 mgP/ℓ influent. This shows that under normal circumstances the UCT system with predominantly aerobic P uptake BEPR removes ~23% more P than the ENBNRAS with anoxic P uptake BEPR. If however, the UCT system receives an influent that causes a consistent high nitrate load on its anoxic reactor, anoxic P uptake (to a lesser extent than in the ENBNRAS system) occurs, resulting in poorer P removal performance than the ENBNRAS system can achieve when receiving the same influent. A more detailed investigation into the anoxic P uptake BEPR of the ENBNRAS system and the aerobic uptake BEPR of the UCT system is given by Vermande *et al.* (2000).
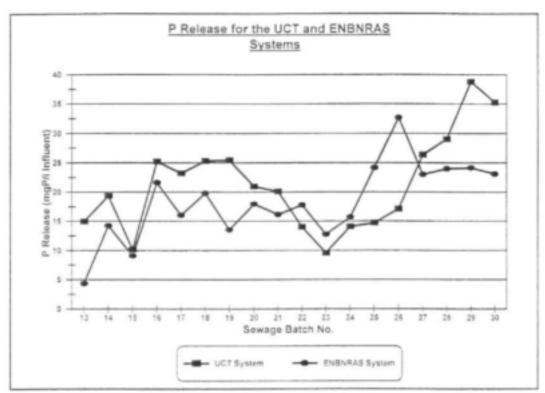
5.16

FIGURE 5.10: P release for the UCT and ENBNRAS systems for sewage batches 13 to 30.
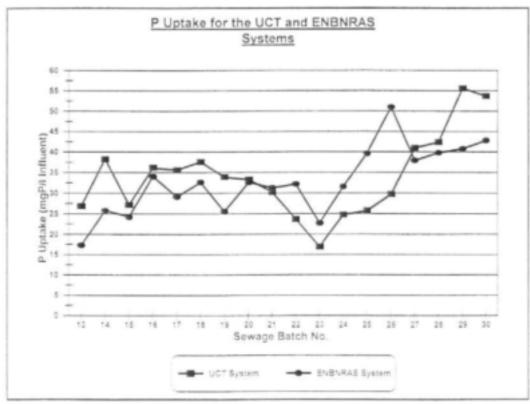

FIGURE 5.11: P uptake for the UCT and ENBNRAS systems for sewage batches 13 to 30.
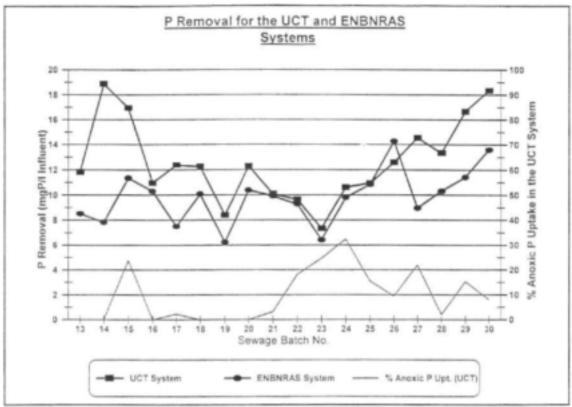
FIGURE 5.12: P removal achieved by the UCT and ENBNRAS systems for sewage batches 13 to 30.

5.3.4 Sludge settleability

Figure 5.13 shows the DSVI for the UCT and the ENBNRAS systems for sewage batches 13 to 30. The % anoxic P uptake for the UCT system has also been included in the Figure 5.13 to illustrate how the DSVI of the UCT system fluctuates with an increase in % anoxic P uptake. The overall average DSVI of the UCT system over the 18 sewage batches was 138 ml/g and that for the ENBNRAS system was 102 ml/g.

From Figure 5.13 it can be seen that the DSVI of the UCT system fluctuates with the % anoxic P uptake. As the % anoxic P uptake increases, the sludge settleability deteriorates rapidly. This can also be seen in another way: As the nitrate load on the anoxic reactor of the UCT system increases, the nitrate concentration flowing from the anoxic reactor also increases, causing the DSVI to increase (see Casey *et al.*, 1994). From the DSVI of the ENBNRAS system it can be seen that this phenomenon does not occur in the ENBNRAS system. The DSVI of the ENBNRAS does not fluctuate as widely as the DSVI of the UCT system, even though it received the same feed as the UCT system. During sewage batches 21 to 27, where the influent TKN/COD ratio was kept very high consistently, the DSVI of the ENBNRAS system increased slightly from around 90 ml/g to around 105 ml/g, while the DSVI of the UCT system responded by increasing sharply from around 110 ml/g to over 200 ml/g. During sewage batches 13, 14 and 15 the UCT system showed a considerably lower DSVI than that of the ENBNRAS system. However, during this period the ENBNRAS system had not yet recovered from the effects of the toxic sewage

5.18

batch 9, and this was the period where the DSVI of the ENBNRAS system was at its highest (i.e. the sludge settleability was at its worst).
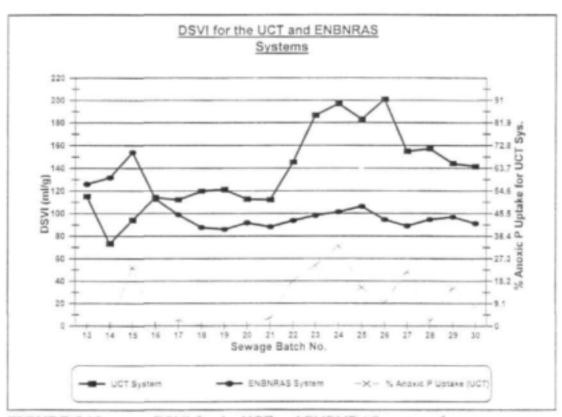

FIGURE 5.13:   DSVI for the UCT and ENBNRAS systems for sewage batches 13 to 30.

5.4 CONCLUSIONS

A laboratory scale 'conventional' BNRAS system (UCT configuration) with similar design and operating parameters to the 10 days sludge age ENBNRAS system Configuration 3 (Chapters 3 and 4) was run in parallel with the laboratory scale ENBNRAS system. To compare the performance of the two systems, both were fed identical influent sewage for 255 days spanning 18 sewage batches (from 13 to 30). For the purpose of comparing the BNR performance of the two systems, the overall averages for the ENBNRAS system are the overall averages of sewage batches 13 to 30, not the overall averages of sewage batches 1 to 30. The overall averages presented in this Chapter therefore differ from those calculated for the entire Configuration as discussed Chapter 3 and 4. A detailed discussion of this comparison is given by Vermande *et al.* (2000).

From the results the following conclusions can be drawn:

1. The overall average COD mass balance achieved for the UCT and ENBNRAS systems were 78 and 77% respectively. The COD removal was 93 and 94% respectively. In terms of carbonaceous material removal, the two systems performed identically.

2. The overall average total oxygen demand (including nitrification) of the UCT system was 7625 mgO/d while that of the ENBNRAS system was 1798 mgO/d. By nitrifying externally, the ENBNRAS system requires 76% less oxygen per day; this is a significant difference.

3. The overall average N mass balance for the UCT and ENBNRAS systems were 86 and 87% respectively and the overall average TKN removal 95 and 94% respectively. The effluent TN of the UCT system was 16.8 mgN/ℓ, of which 12.8 mgN/ℓ was NO_x (filtered sample) and 4.0 mgN/ℓ was TKN (unfiltered sample). Of the 4.0 mgN/ℓ TKN, 1.8 mgN/ℓ was FSA (unfiltered sample). For the ENBNRAS system the effluent TN was 9.8 mgN/ℓ, of which 4.6 mgN/ℓ was nitrate (filtered sample) and 5.2 mgN/ℓ was TKN (unfiltered sample). Of the 5.2 mgN/ℓ TKN, 3.5 mgN/ℓ was FSA (unfiltered sample). The ENBNRAS system achieved effluent TN concentrations <10 mgN/ℓ in 10 out of the 18 sewage batches, while the UCT system did not achieve effluent TN concentrations <10 mgN/ℓ in any of the 18 sewage batches. The overall average TN removal for the UCT and ENBNRAS systems were 78 and 88% respectively.

4. In the UCT system an overall average of 21.3 mgP/ℓ influent P was released in the anaerobic reactor. In the ENBNRAS system an overall average of 18.3 mgP/ℓ influent P was released in the anaerobic reactor and internal settler, with an additional P release of 4.5 mgP/ℓ influent in the EN system (which also has to be taken up in the anoxic and aerobic reactors). On overall average, 34.0 mgP/ℓ influent P uptake occurred in the UCT system, and 32.8 mgP/ℓ influent P uptake occurred in the ENBNRAS system.

5. The overall average P removal for the UCT system was 12.7 mgP/ℓ influent (34.0 - 21.3), and the overall average P removal for the ENBNRAS system was 9.8 mgP/ℓ influent (32.8 - 4.7 - 18.3). The UCT system showed only 9.8% anoxic P uptake on overall average, showing that predominantly aerobic P uptake BEPR occurred in the UCT

system. In the ENBNRAS system, 60% of the P uptake occurred in the main anoxic reactor. During sewage batches 21 to 27 anoxic P uptake was induced in the UCT system by feeding influent sewage with high TKN/COD ratios (leading to a high nitrate load on the anoxic reactor). During this period the UCT system showed about 18% anoxic P uptake and the P removal decreased to the same level as measured in the ENBNRAS system.

6. The overall average DSVI for the UCT and ENBNRAS system were 138 ml/g and 103 ml/g respectively. During sewage batches 21 to 27, where sewage with a high influent TKN/COD ratio were fed to induce anoxic P uptake in the UCT system, the DSVI of the UCT system increased sharply from around 110 ml/g to over 200 ml/g, while the DSVI of the ENBNRAS system increased only slightly from around 90 ml/g to around 105 ml/g. This shows that the 'conventional' BNRAS system reacts much more strongly to significant (>2 mgN/ℓ) nitrate concentrations in the outflow of the main anoxic reactor because its aerobic mass fraction is higher (0.5) than that of the ENBNRAS system (0.2). This response to nitrate in the outflow of the main anoxic reactor was also observed by Moodley *et al.* (1999) in their ENBNRAS system with a higher aerobic mass fraction (0.30), which conforms to the AA filament sludge bulking hypothesis of Casey *et al.* (1994) (see Chapter 2).

In terms of carbonaceous material removal, the UCT and the ENBNRAS system achieve almost identical results. For the nitrogenous material removal, the ENBNRAS system produces an effluent of better quality with an effluent TN concentration of nearly half that of the UCT system on overall average. The ENBNRAS system produced an effluent with a TN content <10 mgN/ℓ for 10 out of the 18 sewage batches, while the UCT system did not achieve this for any sewage batch. The UCT system, which exhibited predominantly aerobic uptake BEPR, removed about 3 mgP/ℓ influent more P than the ENBNRAS system with anoxic/aerobic P uptake BEPR did. P removal is the only process where the UCT system achieves superior results to that of the ENBNRAS system. The ENBNRAS system BNR is effected by using approximately 75% less oxygen than was required by the UCT system to perform the same BNR. The ENBNRAS system produced a better settling sludge than the UCT system did, and the ENBNRAS system DSVI did not produce a bulking sludge when high nitrate concentrations flowed from the anoxic reactor, as was observed in the UCT system.

CHAPTER 6

IMPLEMENTATION OF EXTERNAL NITRIFICATION BIOLOGICAL NUTRIENT REMOVAL (ENBNR) ACTIVATED SLUDGE SYSTEMS AT FULL-SCALE

6.1   INTRODUCTION

The experimental investigations into the external nitrification biological nutrient removal (ENBNR) activated sludge system at laboratory-scale (Chapters 3, 4 and 5) have clearly demonstrated that this system holds considerable promise for practical implementation. However, successful implementation of the ENBNR activated sludge system at full-scale will depend largely on its cost compared to conventional BNR activated sludge systems and its ability to meet required effluent quality standards. Accordingly, the system should be assessed with respect to the economic cost and effluent quality standards, and these compared to those for an equivalent conventional BNR activated sludge system. In this Chapter such a practical case study is demonstrated based on a specific plant and the effluent quality standards of South African law. Details of the economic evaluation are reported by Little *et al.* (2001).

6.2   SYSTEM SELECTED FOR EVALUATION

The economic cost will be influenced to a large extent by site specific factors. For the purpose of the evaluation, the Potsdam Wastewater Treatment Plant (WWTP) at Milnerton in Cape Town, South Africa was selected, since considerable data are available for this plant (see Little *et al.*, 2001).

The existing Potsdam WWTP comprises two streams which are operated in parallel:

- The old rock media tricking filters (TFs) with a capacity of 18 Mℓ/d. These TFs are low loaded and achieve good COD removal and nitrification. However, no N and P removal is obtained. A chemical P removal plant has been provided but currently (2001) this is not operated.

- The newer conventional BNR activated sludge system (Modified UCT process) with a capacity of 17 Mℓ/d with full biological nutrient removal.

Therefore, the plant has a combined capacity of 35 Mℓ/d and is operated in a conventional way as discussed in Chapter 1 (Fig 1.2). The trickling filter effluent can be treated chemically with aluminum sulphate or ferric chloride for P removal and lime dosing facilities for alkalinity and pH correction are also provided if required.

The effluent of the plant must comply with special standards and achieve filtered COD < 75 mg/ℓ, total suspended solids (TSS) <25 mg/ℓ, free and saline ammonia (FSA) < 2 mg N/ℓ, total

N (FSA + NO$_x$) <10 mgN/ℓ and dissolved ortho-phosphate < 1mgP/ℓ, and thus the BNR activated sludge plant was designed for biological N and P removal. The current influent flow is in the order of 26 Mℓ/d. The 35 Mℓ/d design capacity of the existing plant is expected to be reached by 2005. The sewage influent flow is expected to be 49 Mℓ/d in 2020. Currently the activated sludge plant is achieving the required effluent quality standard. However, the TF effluent, without chemical P removal, fails to comply with the FSA, total N and P standards. New effluent quality standards have been promulgated in 2000 and all WWTPs that depend on size, need to comply with these for their full wastewater flow by 2010. For the Milnerton plant these are filtered COD < 50 mg/ℓ, TSS <15 mg/ℓ, FSA <2 mgN/ℓ, NO$_x$ < 7 mgN/ℓ and dissolved ortho-phosphate < 0.8 mgP/ℓ (see Table 6.3). The new effluent standards were taken account of in the design of different schemes for the extension and upgrade of the Milnerton plant for economic evaluation of external nitrification.

6.3 ECONOMIC EVALUATION

6.3.1 The scenarios for economic evaluation

The engineering and economic evaluation by Little *et al.* (2001) provides system schemes and capital, operation and maintenance costs for the following scenarios:

Scenario 1-greenfields
In this scenario, it is assumed that no plant exists at Milnerton and 2 options with the 2005 required capacity of 35 Mℓ/d were designed. The two options selected to compare the capital and operating costs were:

(1a)   A greenfields conventional BNR activated sludge system (UCT) with 13 day sludge age and a capacity of 35 Mℓ/d, and 15 year period from 2000 to 2015;

(1b)   A greenfields ENBNR activated sludge system with 8 day sludge age and a capacity of 35 Mℓ/d, and 15 year period from 2000 to 2015;

Scenario 2:
This scenario aimed to look at the possibility of expanding the existing plant at Potsdam to cope with the flow that is expected in the year 2020. Full nutrient removal for total wastewater flow was designed for. Two options for the extension and effluent quality required were examined: (1) a conventional BNR activated sludge system which would not make use of the existing trickling filters, and (2) an ENBNR activated sludge system which would use the existing trickling filters. This scenario had two design horizons, viz. 2005 and 2020.

First Horizon (2005) for Scenario 2: The flow is expected to reach 35 Mℓ/d in 2005 from the current flow of 26 Mℓ/d in 2000.

Option 2.1a: Doubling the existing activated sludge plant capacity so that all 35 Mℓ/d can be treated in two parallel 13-day sludge age BNR activated sludge systems (i.e. decommissioning the old trickling filters);

Option 2.1b: Retrofitting the existing plant to an ENBNR activated sludge system with 8 day

sludge age, i.e. upgrading the old trickling filters with plastic media to increase their nitrification capacity and combining them with the existing BNR activated sludge system to obtain BNR on the full capacity of 35 Mℓ/d as in (2.1a) for the period from 2000 to 2005;

Second Horizon (2020) for Scenario 2: The flow is expected to reach 49 Mℓ/d in 2020

Option2.2a: Option 2.1a is extended to accommodate 49 Mℓ/d for the period from 2000 to 2020, i.e. keeping the existing plant unchanged until 2005, after which Option 2.1a is implemented, but with a capacity of 49 Mℓ/d for the period 2000 to 2020.

Option 2.2b: Option 2.1b will be extended to accommodate 49 Mℓ/d for the period from 2000 to 2020, i.e. keeping the existing plant unchanged until 2005, after which, Option 2.1b is implemented, but with a capacity of 49 Mℓ/d for the period 2000 to 2020.

Proposed layouts and preliminary designs for all the options were prepared in sufficient detail for costing and tendering purposes, the prices, in South Africa Rands (in 2000), were obtained for the mechanical equipments and civil construction from contractors. Operation and maintenance costs including staffing, sludge handling and disposal were also costed for the Milnerton plant. The sludge treatment included only dewatering to 20% with belt presses and transport of the dewatered cake to a landfill site 10 km away for disposal, which is the current practice.

6.3.2 Results of economic evaluation

The costs for each of the scenarios are given in net present values (NPV) at 6% in the year 2000 SA Rands. They include all civil construction cost, mechanical construction and operational as well as maintenance cost. The overall costs are summarized in Table 6.1.

Table 6.1: NPV at 6% for implementation of a conventional BNR or an ENBNR activated sludge plant at the Potsdam WWTP (Cape Town, South Africa).

Projects		NPV at 6% (in million ZAR)	Period
Scenario 1: *35 Mℓ/d Greenfields plant*			
(1a)	Conventional BNRAS system (UCT)	147.44	2000-2015
(1b)	ENBNRAS system	138.25	2000-2015
Scenario 2: *Modified the existing plant of 35 Mℓ/d and extending the plant with 49 Mℓ/d*			
Option 2.1: Design horizon 2005-35 Mℓ/d capacity			
2.1a	Conventional BNRAS (decommission TFs)	93.55	2000-2006
2.1b	ENBNRAS system, using TFs	76.50	2000-2006
Option 2.2 Extending the existing plant to a 49 Mℓ/d for 2020			
2.2a	as for (2.1a) above	164.80	2000-2020
2.2b)	as for (2.2b) above	154.27	2000-2020

Table 6.1 shows that the overall costs of the ENBNR activated sludge system are lower than those for a conventional BNR activated sludge system. From the economic evaluation, it was noted that:

- The ENBNR activated sludge system uses less electricity (lower oxygen demand) and therefore the power costs are less.
- The ENBNR activated sludge system produces more waste sludge (shorter sludge age) which results in higher costs for sludge dewatering, transport and landfill disposal (the current practice at Potsdam).
- The capital cost for the ENBNR activated sludge system are about 20 % cheaper.
- The operating and maintenance costs dominate the overall costs for both the conventional BNR activated sludge system and the ENBNR activated sludge system. These costs make up about 70 % of the total cost, which are essentially the same for both configurations.
- The only significant saving is therefore in the capital costs which are only 30% of the total cost. The external nitrification option is 6%, 18% and 6% cheaper on total cost for Scenario 1 and Options 2.1 and 2.2 of Scenario 2 respectively; while on a percentage basis this is small, in Rand terms it is R9 million, R17 million and R11 million over 20 years respectively, i.e. around R0.5 and R1.0 million per year, which is a significant saving.

6.4   EVALUATION AGAINST EFFLUENT QUALITY STANDARDS

The system selection for BNR is not only dependent on an economic evaluation, but also needs to consider the effluent quality standards required. The South African Department of Water Affairs and Forestry (DWAF) is currently revising the South African effluent discharge standards in terms of the National Water Act of 1998. A brief review of the development of these standards is given below, before discussing the capacity of conventional and external nitrification BNR activated sludge systems to achieve the new proposed standards.

6.4.1   Development of effluent quality standards in South Africa

The Water Act of 1956 requiring treated effluents to be returned to the water catchments from where the water originates, subject to effluent discharge standards. The effluent discharge standards published in terms of the Water Act of 1956 required > 90 % COD removal to avoid deoxygenation of the receiving water bodies as well as nitrification of the FSA to effluent values below 10 mgN/ℓ to avoid deoxygenation of, and the toxicity effect on the receiving water bodies.

After in-depth research and development of BNR activated sludge systems for the treatment of municipal wastewater between 1975 and 1983, the special standard for P was promulgated in 1985 and required wastewater treatment plants discharging to sensitive catchment areas to remove P to less than 1 mgP/ℓ dissolved ortho-P. Also, research indicated that eutrophication occurring in some water bodies in South Africa was caused by N rather than P, and therefore a special standard for N of < 1.5 mgN/ℓ was also promulgated and enforced from 1985 onwards for these water bodies. The effluent quality concentrations of the general and special standards for N and P are listed in Table 6.2. These standards are enforced as uniform effluent standards, and all sewage treatment plants in South Africa irrespective of size must comply with the relevant standards applicable. These standards are still enforced in South Africa today (2000).

The DWAF is currently in the process of revising the effluent quality standards, as the old general standards are not considered to be sufficient in the framework of the newly developed policy of waste load allocation, receiving water quality objectives and minimum requirements. New effluent quality standards have been proposed under tne new National Water Act of 1998. The proposed new standards differentiate between WWTPs with secondary treatment only (i.e. organic material and N removal) and those with advanced secondary treatment (i.e. organic material, N and P) and stipulate separate effluent quality requirements for each. The WWTPs wishing to discharge effluents that are not within the proposed new standards need to apply to the DWAF for special permission to do so. Table 6.3 lists the proposed new effluent quality standards for WWTPs with secondary treatment only and for WWTPs with advanced secondary treatment.

Table 6.2: Effluent quality standards for the general and special standards for N and P.

Parameters	General Standard	Special Standard	Special Standard for P	Units
Filtered COD	<75	<75	<75	mg COD/ℓ
FSA	<10	<1	<10	mg N/ℓ
Nitrate	-	<1.5	-	mg N/ℓ
Phosphate	-	-	<1	mg P/ℓ
Suspended Solids	<25	<10	<25	mg/ℓ
DO	>75	>75	>75	% Saturation
pH	5.5-9.5	5.5-7.5	5.5-9.5	-

Table 6.3: Effluent quality standards proposed under the new National Water Act of 1998.

Parameters	Secondary Treatment Only	Secondary and Tertiary Treatment	Units
COD	65	50	mg COD/ℓ
FSA	3	2	mg N/ℓ
Nitrate	8	7	mg N/ℓ
Phosphate	-	0.8	mg P/ℓ
Suspended Solids	18	15	mg/ℓ

Table 6.5: Performance potential for the secondary and advanced secondary treatment technologies.

Effluent quality variable	Units	Secondary treatment		Advanced Secondary treatment	
		Median	95% ile	Median	95% ile
COD	mg/ℓ	44	65	32	50
Suspended Solids (SS)	mg/ℓ	10	18	10.	17
FSA	mgN/ℓ	1.5	3.0	1.4	2.9
Nitrate	mgN/ℓ	3.5	6.0	3.2	5.5
Sol PO₃-P	mgP/ℓ	-	-	0.5	1.0

From Table 6.5, the performance potential of advanced secondary treatment is significantly better in effluent quality at the 95% ile only in so far as COD is concerned, i.e. 50 mg/ℓ versus 65 mg/ℓ for secondary treatment; the SS, FSA and nitrate concentrations are virtually the same at about 18 mg/ℓ, 3.0 mgN/ℓ and 6.0 mgN/ℓ respectively. Secondary treatment does not include biological P removal. Advanced secondary treatment, which includes biological P removal, has the potential to achieve at the 95% ile level, an effluent concentration of 1.0 mgP/ℓ dissolved ortho-P.

The survey of operating N and N & P removal activated sludge plants in South Africa indicates that BNR technology, as designed and operated in SA, has the potential to achieve effluent quality concentrations at the 95% ile level of 50 mgCOD/ℓ, 18 mgSS/ℓ, 3.0 mgNH$_4$-N/ℓ, and 6.0 mgNO$_3$-N/ℓ and 1.0 mgPO$_4^{3-}$-N/ℓ in SA. Of these, the FSA and P are outside the new standards, which for FSA is 2 mgNH$_4$-N/ℓ and for P is 0.8 mgP/ℓ. Taking the performance of ENBNR activated sludge Systems 1 and 3c (Chapter 3) as the potential of the ENBNR activated sludge system, the performance of this system is compared with the new effluent standards in Table 6.6.

Table 6.6: Comparison of the performance of ENBNRAS with new standards

Effluent quality parameters	COD mg/ℓ	SS mg/ℓ	FSA mgN/ℓ	NOx mgN/ℓ	P mgP/ℓ
Conventional	50	17	2.9	5.5	1.0
External nitrification					
System 1	51[2]	6.1[3]	3.6	3.9	10.5[1]
System 3b	42[2]	5.4[3]	3.8	3	10.6[1]
New Standard	50	18	2	7	0.8

[1] *P removal-effluent P concentration not meaningful because P was dosed to influent to avoid P limitation.*

[2] *Unfiltered COD*

[3] *Obtained from (unfiltered COD-filtered COD)/1.48.*

From Table 6.6 it can be seen that neither the conventional nor the ENBNR activated sludge system will reach the new FSA standard of 2 mg/ℓ. The ENBNR activated sludge NO$_x$ concentrations are well within the standard. Achievement of P standard with the ENBNR activated sludge system cannot be commented on because it would depend on the influent P concentration. However, considering that the conventional BNR activated sludge systems develop mostly aerobic P uptake BEPR and achieves 1 mgP/ℓ (Table 6.5), then it is unlikely that the ENBNR activated sludge system, when designed to include anoxic P uptake BEPR, will achieve the new standard, because anoxic P uptake BEPR is lower than aerobic P uptake BEPR (see Chapter 3). If the ENBNR activated sludge system is implemented, it is therefore best to design the system with aerobic P uptake to maximize the P removal.

6.5   CLOSURE

When the project was initiated it was thought that a significant saving in capital cost would be achieved for the ENBNR activated sludge system, as a result of the increase in treatment capacity or smaller biological reactors, the reduction in oxygen demand and a better settling sludge. These appeared to make the ENBNR activated sludge system an attractive and viable alternative as a full-scale wastewater treatment plant. However, the economic evaluation indicated that this may not be the case.

Although the ENBNR activated sludge system does provide a cost saving in construction costs of about 30% compared to a conventional BNR activated sludge system, the operating and maintenance costs in the long term mask this saving because these costs are around 70% of the overall cost and virtually the same for both ENBNR activated sludge systems and conventional BNR activated sludge systems. While significant savings in operation costs are made from the very low oxygen demand, the increased sludge production at a shorter sludge age and the associated increased cost in sludge treatment, transport and disposal offsets this saving. Although the total NPV (capital, operation and maintenance) for the ENBNR activated sludge system is 5 to 10 % lower than that of a conventional BNR activated sludge system, this difference may not be large enough to favour selection of the new and untested ENBNR activated sludge system over the tried and trusted conventional BNR activated sludge system. It must be remembered, however, that the sludge treatment route is specific to the Potsdam WWTP and is a relatively costly alternative, and contributes significantly to the total cost. For other WWTPs, if sludge treatment and disposal costs can be reduced, then this will make the ENBNR activated sludge system more economically attractive.

The most significant advantage of the ENBNR activated sludge system is that it offers biological N and P removal for wastewater treatment plants with existing trickling filters and activated sludge units without an increase in existing process units, enabling these plants to meet the stricter new effluent quality standards proposed under the National Water Act of 1998 by the Department of Water Affairs and Forestry. The ENBNR activated sludge system will provide a feasible and economical upgrading option. Although the ENBNR activated sludge system does not provide a large enough savings in monetary terms to make it as an attractive system, the new effluent quality standards may favour implementation of the ENBNR activated sludge system, because the ENBNR activated sludge system is capable of producing effluents with a quality comparable with (and for effluent N better than) conventional BNR activated sludge plants.

Therefore, it is anticipated that the proposed new effluent quality standards will be the driving force that promotes application of the ENBNRAS system at full-scale, i.e. implementation probably will be legislation driven.

CHAPTER 7

DEVELOPMENT OF A MATHEMATICAL SIMULATION MODEL FOR EXTERNAL NITRIFICATION BIOLOGICAL NUTRIENT REMOVAL ACTIVATED SLUDGE SYSTEMS

7.1 INTRODUCTION

For the design and operation of, and research into, conventional biological wastewater treatment systems, mathematical simulation models have proved to be invaluable as a process evaluation tool (e.g. Dold *et al.*, 1980, 1991; Henze *et al.*, 1987). Mathematical simulation models provide quantitative descriptions of the dynamic behaviour of the wastewater treatment system. By providing quantitative descriptions, they allow predictions of the system response and performance to be made. From the predictions, design and operational criteria can be identified for optimization of system performance. Also, mathematical models are very useful as research tools. By evaluating model predictions, it is possible to test hypotheses on the behaviour of the wastewater treatment system (e.g. biological processes, their response to system constraints, etc.) in a consistent and integrated fashion. This may direct attention to issues not obvious from the physical system, and lead to deeper understanding of the fundamental behavioural patterns controlling the system response. In essence, mathematical models can provide a defined framework which can direct thinking (design, operation or research).

Recognising the usefulness of mathematical models, it was decided to develop a kinetic simulation model that will include the processes that can be expected to be operative in external nitrification BNR (ENBNR) activated sludge systems. This Chapter summarises this development; for details, the reader is referred to Hu *et al.* (2001).

7.2 MODELLING APPROACH

The approach taken to develop the kinetic simulation model for the ENBNR activated sludge system was to (1) evaluate and compare the available existing kinetic models for BNR activated sludge systems; (2) identify difficulties and deficiencies in the models when applied to BNR activated sludge systems in general, and to the proposed system in particular; (3) gather information from the literature and experiments to overcome the difficulties and deficiencies to develop the kinetic simulation model; (4) gather data from the literature to calibrate and validate the resultant kinetic model; and (5) apply the model to simulate the behaviour of the proposed system.

State of the art models identified for evaluation were UCTPHO (Wentzel *et al.*, 1992), IAWQ Activated Sludge Model (ASMNo2) (Henze *et al.*, 1995) and ASMNo2d (Henze *et al.*, 1998), BIOWIN model of Barker and Dold (1997) and the Delft based group of models (Smolders *et al.*,1994 ; Kuba *et al.*,1997). In the initial assessment of these models and the ENBNR activated sludge system, one process immediately apparent to be critical important was anoxic P uptake and denitrification by PAOs, and attention was focussed on this aspect.

7.3 DEVELOPMENT OF ENHANCED CULTURE DENITRIFYING PAOs

In evaluating existing kinetic models for BNR activated sludge systems, the importance of including denitrifying PAOs (DPAOs) and associated anoxic P uptake became evident. To achieve this, information on the kinetic and stoichiometric behaviour of DPAOs is essential. To gather the necessary information, it was attempted to develop enhanced cultures of DPAOs, similar to the enhanced cultures of aerobic P uptake PAOs developed by Wentzel *et al.* (1988, 1989a). However, this endeavour did not prove successful.

The experimental investigation on enhanced cultures of DPAOs indicates that:

- For ENBNR activated sludge systems with small aerobic mass fraction (15%) and short sludge age (10 days) at 20°C, receiving influent supplemented with acetate, the typical P behavioural pattern (i.e. anaerobic P release and anoxic/aerobic P uptake) was completely absent. In a similar investigation, Bortone *et al.* (1997) found it was necessary to include a short aerobic period in an SBR cycle to obtain anoxic P uptake. Thus, it would appear that an aerobic period may be essential for stable anoxic P uptake. Recognizing that anoxic P uptake DPAOs are facultative organisms, including an aerobic period would provide opportunity for aerobic metabolisms by these organisms, in addition to the anoxic metabolism. Biochemically, aerobic metabolism provides a higher energy yield for the organism than anoxic metabolism. This would suggest that anoxic P uptake by itself is an unstable and unreliable process. The inability to stimulate this process under the closely controlled conditions in the laboratory implies that this process would probably also prove unreliable at full-scale. This reinforces the conclusions from the investigation on the ENBNR activated sludge systems (Chapter3), that overemphasis has been placed on anoxic P uptake in the system. Rather, anoxic P uptake must be stimulated in conjunction with aerobic P uptake.

The requirement for aerobic P uptake to stabilize the BEPR implies that enhanced cultures of DPAOs cannot be successfully developed in the ENBNR activated sludge systems. Accordingly, this investigation was abandoned. Instead, information on DPAOs was sought in the literature.

7.4 MODEL DEVELOPMENT

To select the most suitable simulation model to serve as a starting point for further development, the existing available kinetic models for BNR activated sludge systems were compared and evaluated against experimental observations in the literature. In this evaluation, initially, it was thought that the most complete model would best serve as a basis for model development. The evaluation showed that:

- The BIOWIN (Barker and Dold, 1997a) model appeared to fit this criteria. However, from the assessment of this model and simulations of available experimental data, it became apparent that the model was inadequately validated and that the highly interactive nature of the processes in the model makes calibration virtually impossible; an infinite number of sets of constants can be obtained that give the same net behaviour. Accordingly, it was decided to revert to the most extensively validated kinetic model as

a starting point, namely UCTPHO (Wentzel et al., 1992) and to extend and modify this model to incorporate those processes omitted, or to resolve deficiencies in the model to develop a general BNR kinetic model.

The UCTPHO kinetic model considers only aerobic growth of PAOs, with associated aerobic P uptake. Anoxic growth of PAOs with associated anoxic P uptake and PAO denitrification are not included. However, experimental data demonstrating these processes in BNR activated sludge systems have been more often reported in the literature recently (Ekama and Wentzel 1999b). Furthermore, in the ENBNR activated sludge system, anoxic P uptake dominates P uptake (see Chapters 3, 4 and 5). Accordingly, UCTPHO was modified to extend its application to conventional BNR activated sludge systems where anoxic P uptake occurs and particularly to the ENBNR activated sludge system. For these modifications, information was drawn from experimental data in the literature and the other available kinetic models (e.g. ASM2d, BIOWIN). This indicated that, most likely, two PAO populations exist, one that can denitrify and one that cannot. However, in modelling, incorporating two PAO populations is not possible, as both essentially compete for the same substrate source and thus one population will be predicted to dominate. Thus, the approach followed was to include a single PAO population, but with reduced rates of substrate utilization/growth under anoxic conditions compared to aerobic conditions. With this basic approach, the main modifications made to UCTPHO were:

(1)   Processes for anoxic growth of PAOs with associated anoxic P uptake and denitrification were included, with separate yield coefficients for anoxic and aerobic PAO growth and separate ratios for P uptake per unit PHB substrate utilized. Also, from the above, the anoxic rates were reduced compared to the equivalent aerobic rates, by multiplying the aerobic rates by the reduction factor η_G under anoxic conditions.

(2)   Processes for anoxic death/maintenance of the PAOs were included, recognizing that under anoxic conditions PAOs that can denitrify will behave differently from those that cannot.

(3)   Separate anoxic and aerobic yield coefficients were introduced for OHO growth.

(4)   The organic N and P fractions were linked to the corresponding COD fractions, and the transformations between the different forms of organic N and P fractions were linked to the corresponding COD transformations (Henze et al., 1995).

In developing the model and from the review of existing models, two aspects were identified as of importance, but were not included in the model, namely:

(1)   Anaerobic slowly biodegradable (SB) COD hydrolysis;

(2)   COD loss mechanisms.

With regard to (1), anaerobic SBCOD hydrolysis, this process hydrolyses SBCOD to fermentable readily biodegradable (F-RB) COD (S_{bsf}) under anaerobic conditions. The resultant S_{bsf} can then be acid fermented by OHOs to short chain fatty acids (SCFA, S_{bsa}) which are available to PAOs for sequestration. The net result, thus, is that:

- This process makes SBCOD available to the PAOs and hence contributes to BEPR.

  - In ASM2 and 2d this increases the substrate for PAOs and hence increases P removal;
  - In BIOWIN, to compensate, in the fermentation of S_{bsc} to S_{bsca} a substantial COD loss occurs (see below) and hence the net substrate obtained by PAOs does not increase.

Information available in the literature on this aspect is contradictory, with some evidence supporting the process and other evidence indicating that the process is not significantly operative under anaerobic conditions. Until this issue is resolved, the process was omitted from the model, with this omission to be evaluated in model application to experimental data.

With regard to (2), COD loss mechanisms, from experimental data available in the literature it does appear that in BNR activated sludge systems some COD is lost that cannot be accounted for in terms of the measured COD mass balance parameters included in the model (e.g. sludge production, oxygen utilized). However, the mechanisms whereby this loss occurs are not known and the amount of COD loss appears to vary considerably between different systems. Accordingly, it was not possible to include this aspect in the model until greater clarity is obtained, hence it was decided to omit this process. This omission, also, was evaluated in model application to experimental data (see below).

7.5   CALIBRATION AND VALIDATION OF THE PROPOSED KINETIC MODEL

The BNR kinetic model was applied to a wide variety of conventional BNR activated sludge systems operated at laboratory-scale in the UCT laboratory over a range of conditions (influent concentrations, system configurations, sludge age, mass fractions and recycle ratios); these systems were selected because complete experimental data sets are available (Wentzel et al., 1989b, 1990; Sneyders et al., 1998; Kashula et al., 1993; Pilson et al.,1995 and Musvoto et al., 1992). The main objectives of this application were to:

- Evaluate the predictive capacities of the model;
- Evaluate values for the model parameters;
- Calibrate those parameters for which values were not available.

In model application to conventional BNR activated sludge systems, a number of issues of interest were identified and are summarized below.

7.5.1 Model constants

From model application it was apparent that:

- For most of the kinetic and stoichiometric parameters, the values derived from the literature can be retained. This included the values proposed for the new constants introduced into the model to more accurately reflect anoxic behaviour (i.e. OHO anoxic yield, PAO anoxic P uptake PHA utilization ratio).

- For OHO maximum specific SBCOD hydrolysis/utilization rate (K_{mp}), for systems with the "usual" range of aerobic mass fraction (50 -60%), the default value can be retained. However, for systems with small aerobic mass fractions (< 40%) (Kashula et al., 1993; Pilson et al.,1995 and Musvoto et al., 1992) the value has to be increased, otherwise significant unbiodegraded SBCOD is predicted to accumulate in the system contrary to experimental observations. The smaller the aerobic mass fraction, the larger the increase required. Most likely, systems with small aerobic mass fractions favour the growth of OHOs with higher substrate utilization rates (i.e. K_{mp}). Such selection has been observed previously in systems that included selectors (Still et al., 1996).

- For OHO and PAO anoxic reduction factors for SBCOD hydrolysis/utilization (η_H and η_G respectively):

    - These were found to vary significantly from system to system; η_H ranged from 0.1 to 0.7, η_G from 0 to 0.6.

    - The value for η_H appeared to be closely linked to the selected K_{mp} value; where K_{mp} needs to be increased, η_H correspondingly was decreased.

In searching for reasons for the variability in η_H and η_G, two factors appeared to be of importance:
    - Anoxic mass fraction
    - Nitrate load to the anoxic reactor.

However, a quantitative relationship between the above two factors (and possibly other design/operational criteria) and the exact values for η_H and η_G could not be established. Accordingly, it is not possible to determine values for η_H and η_G *ab initio*, these will be system specific and require calibration (or measurement) for each situation.

Clearly, this is undesirable and requires further investigation. Where it is not possible to calibrate (or measure) η_H and η_G for example for design of conventional BNR activated sludge systems with the "usual" range of aerobic mass fractions, it is reasonable to accept $\eta_H = 0.34$ and $\eta_G = 0.39$.

- For the PAO ratio P release/acetate taken up, $f_{p,rel}$, for systems with conventional aerobic mass fractions (50 - 60%) a value $f_{p,rel} = 0.45$-0.5 can be accepted. From the $f_{p,rel}$ and pH relationship of Smolders et al. (1995), this corresponds to the anaerobic pH of 6.9 to 7.2, which is reasonable. However, for systems with small aerobic mass fraction, the $f_{p,rel}$ value required significant reduction (<0.4). This reduction concurred with a number of other modifications to the model to correctly predict BEPR in these systems. It was concluded that these systems produced BEPR behaviour that deviated from the "usual" (see below).

7.5.2 Model predictive capacity

For systems with the "usual" aerobic mass fractions of 50 - 60% (Wentzel et al., 1989b, 1990; Sneyders et al., 1998), the BNR activated sludge kinetic model proposed here gave reasonable

predictions. However, due to the deviation in BEPR behaviour in systems with small aerobic mass fractions (Kashula et al., 1993; Pilson et al.,1995 and Musvoto et al., 1992) the predictions were poor. Unfortunately, these system conditions are those that appear to stimulate significant anoxic P uptake. Hence, the kinetic model could not be conclusively validated for BNR activated sludge systems exhibiting predominately anoxic P uptake, by simulation of the response of the Kashula et al. (1993), Pilson et al. (1995) and Musvoto et al. (1992) systems. However, the data set of Wentzel et al. (1990) does include more conventional BNR activated sludge systems in which anoxic P uptake did take place: the kinetic model was capable of correctly predicting the observed BEPR in those systems. Thus, it would appear that the model holds potential for application to the ENBNR activated sludge system (see Section 7.6).

7.5.3 COD loss

In developing the kinetic model here, it was thought that COD loss mechanisms were not sufficiently understood for definitive inclusion. Hence, the model does not take COD loss into account. In application of this model to the variety of systems, the data on COD loss appears contradictory. For some systems, excluding COD loss caused the sludge production (VSS) and oxygen demand (OUR) to be overpredicted, whereas for other systems this was not the case. Thus, there is no consistency in the information on the COD loss phenomenon. Hence, until greater clarity on this phenomenon is obtained, it is not possible to include COD loss in the model.

7.5.4 Contribution of PAOs to system denitrification

The model proposed here includes anoxic P uptake and associated denitrification by PAOs. Hence, the PAOs will contribute to system denitrification. It is this behaviour that is sought to be exploited in the DEPHANOX and similar systems, where the concept is that the same substrate source (i.e. the PHA sequestered by the PAOs) can be used for both P removal and denitrification. However, from the simulations and a sensitivity analysis, it was evident that the contribution of PAOs to denitrification is relatively small (<10% for conventional systems; <25% for systems with larger anoxic mass fractions), with the vast majority of denitrification mediated by OHOs. Furthermore, the denitrification by PAOs comes at the cost of lower P uptake and hence P removal efficiency (compared to aerobic P uptake). This would suggest that denitrification by PAOs in the BNRAS system should be minimized rather than maximized. This is in agreement with the experimental observations on the ENBNR activated sludge system (see Chapter 3).

In conclusion, from extensive model application in conventional BNR activated sludge systems, it was demonstrated that:

- The BNR kinetic model developed here is capable of simulating system response and performance of conventional BNR activated sludge systems with "usual" aerobic mass fractions (>50%), including nitrification and denitrification and the two types of BEPR (aerobic and aerobic-anoxic P uptake).

- However, in application to BNR activated sludge systems with small aerobic mass fractions, considerable model manipulation was required to match predicted to observed

behaviour. It appears that the small aerobic mass fractions induce behavioural patterns that are not adequately taken into account in the model, or that the experimental data for these systems deviated from the "usual".

This was investigated further when the model was applied to the experimental data on the ENBNR activated sludge systems gathered in Chapter 3; these ENBNR activated sludge systems also include small aerobic mass fractions (19%) (see below).

7.6 APPLICATION OF THE MODEL TO ENBNR ACTIVATED SLUDGE SYSTEMS

The BNR activated sludge kinetic model was applied to the laboratory scale ENBNR activated sludge systems operated here (Chapter 3), and predicted behaviour compared to experimental observations. This application shows that:

- The model is capable of modelling the system behaviour, with the default values for the kinetic and stoichiometric parameters proposed, except for $f_{P,rel}$, η_H, η_G and K_{mp}. This provides substantive evidence supporting the validity of the proposed kinetic model. This is in contrast to the simulations of conventional BNR activated sludge systems with small aerobic mass fractions (Kashula *et al.*, 1993; Pilson *et al.*, 1995 and Musvoto *et al.*, 1992), where considerable model manipulation was required to correctly reflect the observed BEPR behaviour. This would suggest that the data of Kaschula *et al.* (1993), Pilson *et al.* (1995) and Musvoto *et al.* (1992) deviates from the "usual".

With regard to those constants that required adjustment:

- For $f_{P,rel}$, in model application this had to be reduced from the default value of 0.5 to 0.4; this corresponds to pH = 6.8 in the anaerobic reactor according to the relationship between $f_{P,rel}$ and pH of Smolders *et al.* (1995), which is not unreasonable. In simulating conventional BNR activated sludge systems with small aerobic mass fractions the same reduction in $f_{P,rel}$ was required. Possibly this change reflects that the P release behaviour of denitrifying PAOs differs from aerobic PAOs, as is evident from the observed change in P release/P removal ratios when anoxic P uptake is significant (see Chapter 3), but this is speculation.

- For the η_G value, as in conventional BNR activated sludge systems, this parameter needs to be determined by simulation for each system, even for each steady state period for the same system. Although it appears that two factors may significantly influence the η_G value, namely the nitrate load on the anoxic zone and the aerobic mass fraction, the quantitative relationship between these factors and η_G is still unknown. However, as noted above, the PAOs contribution to denitrification is small compared to the OHOs contribution and hence, with little error, a default value of $\eta_G = 0.5$ can be used for the ENBNR activated sludge systems where η_G is not known; preferably, if possible, η_G should be determined by calibration as above, or by measurement.

- For K_{mp}, the value for this constant had to be increased significantly in application of the

model to the ENBNR activated sludge systems. In these simulations, it was found that the default value for K_{mp} caused significant accumulation of SBCOD (S_{enm}) in the system, which resulted in predicted low OUR and VSS which were not reflected in the observed data. Increasing K_{mp} improved OUR and VSS predictions. This was also observed in simulations of conventional BNR activated sludge systems with similar small aerobic mass fractions above. Most likely, systems with small aerobic mass fractions select for OHOs with higher substrate utilization rates (i.e. K_{mp}). As noted previously, such selection has been observed in systems that include selectors.

- For the η_H value, as for the simulations of conventional BNR activated sludge systems, this constant has to be calibrated in associated with K_{mp} above. The determined values for η_H show some variability for the different wastewater batches. In particular, the values appear to be influenced by the nitrate load on the anoxic reactor. Accepting this variability, the product of η_H and K_{mp} (which is the anoxic OHO SBCOD hydrolysis/utilization rate) appears reasonably consistent with the products determined for conventional BNR activated sludge systems. This would suggest that the OHO anoxic SBCOD utilization rate should be uncoupled from the equivalent aerobic rate. This requires further investigation.

7.7 APPLICATION FOR DESIGN AND OPERATION

After calibration against the ENBNR activated sludge system, the model was applied for system optimization in design and operation. This application demonstrated that the model can be an extremely valuable aid for system design and operation. Two scenarios were investigated, the effect of varying influent TKN/COD ratio and the effect of sludge age. From this investigation, it would appear that:

- Aerobic P uptake in the ENBNR activated sludge system should be maximized with anoxic P uptake correspondingly minimized. Since aerobic P uptake is more efficient than anoxic P uptake, improved P removal can be obtained. This will not be at much expense to denitrification, since the PAO contribution to denitrification is relatively small, and the denitrification that is achieved is nitrate limited.

- The optimum sludge age for the ENBNR activated sludge system is in the range 6-8 days at 20°C, and probably slightly higher at lower temperatures. This corresponds to the maximum P removal.

7.8 CLOSURE

In this Chapter a kinetic simulation model for conventional BNR and ENBNR activated sludge systems has been developed. This model includes the biological processes mediated by the:

- "Ordinary" heterotrophic organisms (OHOs)
- Autotrophic (nitrifiers) organisms (AOs)
- Phosphorus accumulating organisms (PAOs)

The process kinetics and stoichiometry for the OHOs and AOs were taken largely unmodified from the UCTPHO kinetic simulation model (Wentzel *et al.*, 1992), except that the transformations in N and P compounds were linked to the equivalent transformations in the COD compounds (Henze *et al.*, 1995). For the PAOs, the anaerobic and aerobic kinetic and stoichiometric behaviour were also taken from UCTPHO, but additionally, kinetics and stoichiometry for anoxic PAO growth and death (with associated anoxic P uptake and PAO denitrification) were included, based on information available in the literature.

The model was calibrated and applied to an extensive data set from conventional and external nitrification BNR activated sludge systems collected in the UCT laboratory in this and previous WRC funded contracts. The correlation between observed and predicted results was good for the wide variety of systems (including ENBNR activated sludge systems), except for conventional BNR activated sludge systems with small aerobic mass fractions.

This model provides a useful tool for the design and operation of ENBNR activated sludge systems.

CHAPTER 8

DISCUSSION/FUTURE WORK

In the external nitrification (EN) biological nutrient removal (BNR) activated sludge system, the biologically mediated process of nitrification is removed from the main BNR activated sludge system and effected externally in a fixed media system. This is achieved by including in a BNR activated sludge system an internal settling tank between the anaerobic and the primary anoxic reactors; the overflow from this settler is discharged to a fixed media nitrifying system (the EN) and the underflow to the activated sludge system anoxic reactor. The nitrified effluent from the EN system is then discharged to the anoxic reactor to meet with the internal settling tank underflow, and hence is denitrified. Removing nitrification from the activated sludge system eliminates the requirements of a minimum sludge age and aerobic mass fraction for nitrification, and hence allows a step increase in system intensification. The investigations on laboratory-scale ENBNR activated sludge systems by Hu *et al.* (2001), Moodley *et al.* (1999) and Sötemann *et al.* (2000) (Chapter 3) show that implementation of this system is possible in practice. In the three laboratory scale ENBNR activated sludge system investigations, the EN part of the systems nitrified between about 85 and 90% of the free and saline ammonia (FSA) flowing into them (provided failure of the EN system did not occur). Of the total system nitrification, up to 90% occurred externally. Some residual nitrification of the FSA not nitrified in the EN system and the FSA in the internal settling tank underflow (which bypasses the EN part of the system) did occur in the activated sludge system main aerobic reactor, mediated by nitrifiers seeded to the activated sludge part of the system by the EN. However, this is not undesirable as this nitrification ensures a low FSA final effluent quality.

The laboratory scale ENBNR activated sludge systems removed >90% of the influent carbonaceous material (COD), utilising on average about 60% less oxygen than an equivalent 'conventional' BNR activated sludge system. The ENBNR activated sludge systems have also been shown to give excellent TKN and very good TN removals (TKN removals >90%, TN removals >80%). Furthermore, it has been shown that the ENBNR activated sludge systems are capable of producing effluents with TN concentrations of <10 mgN/l for influent wastewaters with TKN/COD ratios of up to between 0.13 and 0.14.

The P removal in the ENBNR activated sludge system was about 30% less than that expected for a conventional BNR activated sludge system. The biological excess P removal (BEPR) occurring in the ENBNR activated sludge systems was undoubtably anoxic/aerobic P uptake BEPR, with the anoxic reactor effecting up to 60 - 70% of the total system P uptake. Reduced BEPR with anoxic P uptake is in agreement with previous observations on conventional BNR activated sludge systems (Ekama and Wentzel, 1997b). The relative magnitude of the anoxic uptake BEPR appears to be dependant on the nitrate load on the main anoxic reactor and the aerobic mass fraction: If the nitrate load was equal to or below the denitrification potential of the main anoxic reactor, the % anoxic P uptake decreased and the % aerobic P uptake increased provided the aerobic mass fraction was sufficiently large to complete the P uptake process. Conversely, when the nitrate load on the main anoxic reactor was greater than the denitrification potential of the main anoxic reactor, the % anoxic P uptake increased, and the % aerobic P uptake decreased. In practise, it is unlikely that a steady state in terms of anoxic P uptake will be reached; the P

8.2

uptake will shift from anoxic P uptake to aerobic P uptake and *visa versa*, as the nitrate load on the main anoxic reactor decreases or increases respectively due to variations in the influent wastewater composition. However, from the investigation as the P uptake shifts from predominantly anoxic P uptake to increased aerobic P uptake, the total P removal seems to increase provided the aerobic mass fraction is sufficiently large. Thus, it would not be advisable to implement aerobic mass fractions much smaller than 0.20; even though it is theoretically possible to do so, it would be detrimental to the overall BEPR. Further, aerobic mass fractions less than about 20% seem to promote pin point floc formation, leading to deterioration in effluent quality.

One consistent observation made in all three investigations was that the ENBNR activated sludge systems produced sludges that settle very well (from about 70 to 110 mℓ/g). Furthermore, it seems that these systems are not affected to the same extent as 'conventional' BNR activated sludge systems are by high nitrate concentrations flowing from the main anoxic reactor, as stated in the AA filament bulking hypothesis of Casey *et al.* (1994). The good settling characteristics of the sludges produced in these systems will further facilitate system intensification, by reducing the required surface area of the secondary settling tanks. However, this benefit will be somewhat offset by the requirement of internal settling tanks.

It has further been demonstrated that the ENBNR activated sludge systems perform full and uncompromised BNR for short sludge ages down to about 5 days (Chapter 4). Thus, if the system is implemented in an existing conventional BNR activated sludge system, the sludge age can be significantly reduced. The implication of such a reduction in sludge age is that the influent flow can be doubled to an existing system without a negative impact on the BNR, provided the system does not fail hydraulically due to the increased influent sewage flow. Sludge ages below 10 days have an added advantage in that N and P removals increase per mass of organic load (Wentzel *et al.*, 1990) as the sludge age is reduced.

The comparison of the performance of a laboratory scale ENBNR activated sludge system with that of a parallel laboratory scale 'conventional' BNR activated sludge system (UCT configuration) by Vermande *et al.* (2000) and Sötemann *et al.* (2000) (Chapter 5) demonstrated that the carbonaceous material removal performance of both systems was effectively equal. The TN removal performance of the ENBNR activated sludge system was superior to that of the UCT system; the ENBNR activated sludge system produced effluents with half the TN concentrations of the UCT system final effluent and was capable of producing effluents with TN concentrations of <10 mgN/ℓ, which was not possible with the UCT system. Furthermore, the ENBNR activated sludge system was able to perform total denitrification in the main anoxic reactor, while this was not possible for the UCT system because of the limitation imposed by the a-recycle.

However, the UCT system did show higher BEPR than the ENBNR activated sludge system; it removed on average 3 mgP/ℓ influent more P than the ENBNR activated sludge system, equivalent to about 30% more P removal. This may be ascribed to the predominantly aerobic P uptake BEPR occurring in the UCT system versus the anoxic/aerobic P uptake BEPR that occurred in the ENBNR activated sludge system. In agreement with previous observations on conventional BNR activated sludge systems (Ekama and Wentzel, 1997a), anoxic P uptake is less efficient than aerobic P uptake. This would suggest that to increase system P removal, aerobic P uptake should be encouraged which supports the previous suggestion of a minimum 20%

aerobic mass fraction.

The ENBNR activated sludge system produced a sludge with a DSVI of between 90 and 100 mℓ/g, while the DSVI of the UCT system fluctuated between 80 and 200 mℓ/g. This difference in DSVI became particularly apparent when high nitrate concentrations flowed from the anoxic reactors to the aerobic reactors of the two systems: In response to high anoxic reactor nitrate concentrations, the UCT system's DSVI increased sharply, from about 100 mℓ/g to about 200 mℓ/g, while the ENBNR activated sludge system DSVI increased from around 90 mℓ/g to just over 100 mℓ/g. Hence, the UCT system is much more sensitive to AA filament bulking with significant nitrate concentrations in the outflow of the anoxic reactor than the ENBNR activated sludge system. This is because the aerobic mass fraction of the UCT system was 0.50 and within the range of applicability of the AA filament bulking hypothesis of Casey et al. (1994), whereas the ENBNR system's aerobic mass fraction was 0.2 and outside this range.

Initially it was thought that the savings in capital cost brought about by an increased capacity or smaller biological reactors, reduced oxygen demand and better settling sludge would make the ENBNR activated sludge system an economically attractive and viable alternative as a full-scale plant. The economic evaluation of Little et al. (2000) (Chapter 6), however, indicates that this may not necessarily be the case. The ENBNR activated sludge system alternative does provide a substantial saving in construction costs of about 30% when compared to a 'conventional' BNR activated sludge system. With regard to operational costs, while the ENBNR activated sludge system does also offer significant savings due to the very low oxygen demand, these cost savings are offset by the increased sludge production at the shorter sludge ages and the associated increase in sludge treatment, transport and disposal costs (for the particular case study and sludge disposal route). This causes that the ENBNR activated sludge system operating costs are about the same as those for a conventional BNR activated sludge system. Since the operating costs of a sewage treatment works, whether ENBNR activated sludge or 'conventional' BNR activated sludge system, account for the bulk of the net present value (NPV), the total NPV (capital, operation and maintenance) for the ENBNR activated sludge system option is only 5 to 10% lower than that of a 'conventional' BNR activated sludge system. This difference may not be large enough for a definite choice of the ENBNR activated sludge system over the 'conventional' BNR system. However, the choice of system may be significantly influenced by the effluent quality requirements. In South Africa, if the Department of Water Affairs and Forestry implement the new effluent quality standards proposed under the National Water Act of 1998, the ENBNR activated sludge will provide a feasible and economical plant upgrade option, particularly for treatment plants with parallel trickling filters and BNR activated sludge systems, or trickling filters only. For the former, the ENBNR activated sludge system offers biological N and P removal for the full wastewater flow without increase in existing process units. For the latter, the existing trickling filters can be integrated into the ENBNR activated sludge system. Furthermore, the ENBNR activated sludge system is capable of producing effluents with a quality that are within the new effluent quality standards, especially with regards to nitrogen; conventional BNR activated sludge systems may have difficulty meeting these standards. Thus, the proposed new effluent quality standards rather than economics may well be the driving force that will see the ENBNR activated sludge system implemented at full-scale.

From the research summarised in this report and discussed above, it is evident that the ENBNR activated sludge system holds considerable merit for implementation. The investigations on the

three laboratory-scale ENBNR activated sludge systems and the comparison with the conventional UCT system provide a comprehensive framework for the understanding of the ENBNR activated sludge system operation and performance, and further laboratory investigations would not provide more knowledge and understanding. The next step would be to begin full-scale trials of an ENBNR activated sludge system. To begin with, a full-scale trickling filter would have to be converted into a nitrifying trickling filter to ascertain the performance of a nitrifying trickling filter at full-scale. Once it has been proven that existing full-scale trickling filters can successfully be converted to nitrifying trickling filters and their capacity determined, the trickling filters can be integrated into a BNR activated sludge system in an ENBNR activated sludge system configuration to obtain BNR on the full influent wastewater flow.

From the discussion above, the next step in development of the ENBNR activated sludge system is implementation and evaluation at full-scale. In particular the behaviour of nitrifying trickling filters need to be assessed - this is best done at full-scale. This will form the basis for a new research contract between UCT and the WRC and WSSA.

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EXTERNAL NITRIFICATION WITH THE AID OF FIXED MEDIA TRICKLING FILTERS (TF) TO INCREASE THE CAPACITY OF BIOLOGICAL NUTRIENT REMOVAL (BNR) SUSPENDED MEDIUM ACTIVATED SLUDGE (AS) SYSTEMS

Z Hu · SW Sötemann · SM Vermande · R Moodley ·
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Water Research Commission

FINAL REPORT
to the
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by

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EXECUTIVE SUMMARY

1.   MOTIVATION

Biological nutrient removal (BNR) activated sludge (AS) systems have become the preferred treatment system for advanced municipal wastewater treatment in South Africa. The widespread implementation of BNRAS systems has drawn attention to some weaknesses of the system, predominantly (i) long sludge ages required for nitrification, (ii) filamentous organism bulking and (iii) the treatment/disposal of liquors/supernatants generated from sludge and solids handling. This research project focusses on the first two weaknesses, namely nitrification and bulking.

The requirement to nitrify governs the sludge age and aerated mass fraction of BNRAS systems. To achieve both nitrogen (N) and phosphorus (P) removal, a high unaerated sludge mass fraction is usually required (>40%), resulting in a reduced (<60%) aerated mass fraction. To compensate for the reduction in aerobic mass fraction, long sludge ages (around 20 to 25 days) must be selected to guarantee nitrification at the lowest expected temperature. Such long sludge ages result in large biological reactor volumes per Mℓ wastewater treated. If nitrification can be made independent of the suspended solids sludge age, then the N and P removal processes rather than nitrification will govern the system sludge age, and indications are that sludge ages can be reduced to less than half, from about 20 to 25 days to about 8 to 10 days. Such a reduction in sludge age will result in a reduction in the biological reactor volume requirement of approximately 1/3rd or alternatively, for an existing plant, in an increase in treatment capacity of some 50%, provided the secondary settling tank area requirements are accommodated.

To uncouple the suspended solids sludge age from the requirement to nitrify, the activated sludge system can be modified in two ways, by including (i) internal or (ii) external fixed media for attached nitrifier growth. The nitrifiers grow on the fixed media establishing a permanently resident population that is not subjected to either the unaerated-aerated reactor interchanges or to the suspended solids sludge age, with the result that the latter can be reduced significantly without losing nitrification. Since the effectiveness of internal fixed media has not been as good as expected and the effect of internal media on the BNRAS process has not been investigated, a BNRAS scheme in which nitrification is removed from the BNRAS system to an external fixed media system is proposed. This proposed system will be termed the external nitrification biological nutrient removal activated sludge (ENBNRAS) system.

In the proposed system, nitrification takes place outside the suspended sludge BNRAS system in a fixed media trickling filter or a similar type of system. This is achieved by including an internal settling tank (IST) following the anaerobic reactor. The mixed liquor from the anaerobic reactor enters the IST and the solids and supernatant are separated. The solids in the IST underflow are discharged to the following anoxic reactor, while the ammonia rich supernatant is discharged to the nitrifying fixed media system. The resultant nitrate rich flow is then discharged to the anoxic reactor, to recombine with the IST underflow for denitrification. The anoxic reactor discharges to the aerobic reactor, then to the secondary settling tank (SST). The SST supernatant is the system effluent, and the underflow discharges via an underflow anoxic reactor to the anaerobic reactor. In this manner, nitrification is achieved external to the activated

sludge system.

Including external nitrification has the potential to reduce the sludge age of the BNRAS system significantly. Furthermore, removing nitrification from the aerobic reactor of the BNRAS system has a number of additional benefits: (i) Minimum aerobic mass fraction for nitrification not required, (ii) the aerobic reactor volume does not have to be maintained to accommodate internal fixed media and (iii) aeration in the aerobic reactor for nitrification is no longer required, only for COD utilization. As the aerated mass fraction in the proposed system is no longer controlled by nitrification, the unaerated mass fraction can be enlarged to >60%. As a result, the anaerobic mass fraction may be increased, which should improve biological excess P removal (BEPR), and the anoxic mass fraction can be increased to improve denitrification to the point where complete denitrification may be possible. The possibility of complete denitrification at the short sludge age, together with the possibility of an aerobic mass fraction of <30%, holds promise to ameliorate anoxic/aerobic (AA) filament bulking (Casey et al., 1994). This will be particularly beneficial as it will significantly increase the treatment capacity of an existing system, or reduce secondary settling tank surface area for a proposed new system. Thus, by uncoupling the sludge age from nitrification by implementing the proposed ENBNRAS system configuration, the treatment capacity of existing BNRAS systems can be significantly increased, or bioreactor volumes and secondary settling tank surface areas in proposed systems can be significantly decreased, resulting in a major increase in system intensification with considerable economic benefit. External nitrification can be implemented at existing wastewater treatment plants (WWTPs) where old trickling filters have been extended with BNRAS systems. There are many such WWTPs, particularly in South Africa, and therefore considerable opportunities exist in South Africa for implementation of the proposed scheme and accordingly the system merits further investigation. This report summarizes this investigation: detailed reports are Moodley et al. (1999), Sötemann et al. (2000), Vermande et al. (2000), Little et al. (2001) and Hu et al. (2001).

2.   PRINCIPLE AIM AND MAIN TASKS

Before the ENBNRAS system can be implemented at full scale, it needs to be comprehensively investigated at laboratory scale, and this was the principle objective of this research contract:
* *To evaluate, investigate, develop and refine at laboratory scale the external nitrification biological nutrient removal activated sludge system.*

To achieve this objective, the following aims were identified:
(i)   To examine the effect of external nitrification on BNRAS systems.
(ii)   To examine N and P removal performance and sludge settleability of non- or partially nitrifying BNRAS systems at short sludge ages (3 to 10 days) and over a range of temperatures from 12 to 20°C at laboratory scale.
(iii)   To examine BEPR performance and sludge settleability in anaerobic-anoxic (i.e. no aerobic zone) BNRAS systems at laboratory scale.
(iv)   To extend BNR simulation models to include anoxic P uptake denitrification kinetics.

To address the aims above, a number of specific tasks were identified:

Task 1:   Experimental evaluation of the ENBNRAS system.

Task 2:   ENBNRAS systems at short sludge ages
Task 3:   BNR at low temperatures.
Task 4:   Comparison of the external nitrification BNR with a conventional BNR activated sludge system.
Task 5:   Implementation of ENBNRAS systems at full scale (economic evaluation).
Task 6:   Model development for external nitrification systems.

3.   TASK 1:   EXPERIMENTAL EVALUATION OF THE ENBNRAS SYSTEM

In this task the performance of the ENBNRAS system was evaluated at laboratory scale under strictly controlled conditions. In particular, nutrient removal (N and P) and sludge settleability were examined. A series of three separate laboratory scale experimental investigations were undertaken, collectively aimed at determining the performance of this proposed system. The experimental investigations are reported in detail by Moodley *et al.* (1999), Sötemann *et al.* (2000) and Hu *et al.* (2001).

3.1   Introduction

System 1:   The first experimental investigation was on the proposed ENBNRAS system with a small aerobic mass fraction (19%) and is reported in detail by Hu *et al.* (2001). This investigation was a feasibility study on the proposed system, and after it was shown to operate successfully, the second investigation was initiated.

System 2:   The second investigation examined the effect of varying aerobic (and consequently anoxic) mass fractions and shorter sludge age on the ENBNRAS system nutrient removal performance, and is reported in detail by Moodley *et al.* (2000). The ENBNRAS system was the same as that used in the first investigation, and this investigation covered two phases:
In Phase I the aerobic mass fraction was reduced from 19 to 15% and the sludge age was reduced from an initial 10 days to 8 days. Then, in Phase II, the aerobic mass fraction was increased to 30% at the expense of a reduced anoxic mass fraction.

System 3:   The third investigation was initiated to optimize N and P removal in the ENBNRAS system, and is reported in detail by Sötemann *et al.* (2000). This investigation covered three phases:
In Phase I the aerobic and anoxic mass fractions were equal at 32.5%, and a mixed liquor recycle between the aerobic and anoxic reactors was included because the nitrification efficiency of the external nitrification system was initially poor, and the large aerobic mass fraction resulted in significant nitrification in the aerobic reactor. In Phase II, efficient nitrification in the external nitrification system was restored and the aerobic and anoxic mass fractions were changed to 20 and 45% respectively. In Phase III the mixed liquor recycle between the aerobic and the anoxic reactors was removed.

stimulate pin-point floc, which causes a deterioration in sludge retention and effluent quality. Such aerobic mass fractions also stimulate aerobic P uptake BEPR. Inclusion and maximization of aerobic P uptake BEPR appears desirable to maximize BEPR, because aerobic P uptake BEPR is more efficient than anoxic P uptake BEPR. However, conditions that promote aerobic P uptake BEPR are also conducive to nitrifier growth. Although complete exclusion of nitrifiers from the activated sludge of the ENBNRAS system may not prove possible, it is not necessary as long as virtually complete nitrification occurs in the external nitrification system to limit nitrification in the aerobic reactor of the main system. Accepting nitrification in the aerobic reactor of the BNRAS system, this nitrification will be limited to the ammonia bypassing the external nitrification system in the internal settler underflow. This nitrate is prevented from entering the anaerobic reactor by denitrification in the underflow anoxic reactor.

- The requirements above for anaerobic and aerobic mass fractions of 25 and 20% respectively provide 55% of the total mass fraction for the anoxic reactors. Allowing 10% for the underflow anoxic reactor, 45% of the total mass fraction remains for the main anoxic reactor.

- The proportion of the internal settling tank inflow that can pass to the external nitrification system is approximately 85%, due to the constraint of sludge underflow to the anoxic reactor.

- A permanent a-recycle from the aerobic to the anoxic reactor should not be included, as this appears detrimental to BEPR and denitrification. However, provision of an a-recycle for 'emergency use' in case nitrification in the external nitrification system deteriorates may prove useful.

4.   TASK 2:   ENBNRAS SYSTEM PERFORMANCE AT SHORT SLUDGE AGES

4.1   Introduction

In Task 1 extensive experimental data on ENBNRAS systems with sludge ages of 10 and 8 days was collected. Towards the end of the practical laboratory investigation on System 3 (Phase III), it was decided to increase the influent sewage flow from 20 ℓ/d to 30 ℓ/d in order to observe the systems response to this 50% increase in load. However, shortly after the influent flow had been increased to 30 ℓ/d, the systems internal settling tanks began to fail hydraulically. Thus, instead of allowing the system to fail completely as a result of this hydraulic failure, it was decided to reduce the influent flow to 25 ℓ/d, and instead of implementing a gradual increase in feed, a gradual reduction in sludge age was implemented and the effect of the reduced sludge age on system performance evaluated. The system was operated at a 8 day sludge age for 49 days and a 5 day sludge age for a further 13 days. Details of this investigation are given by Sötemann et al. (2000).

4.2   System performance

- The overall average COD removals of the 8 and 5 day sludge age configurations of the ENBNRAS system were 93 and 90% respectively. The COD removal performances of the short sludge age systems were very good and close to COD removal observed in the systems of Task 1.

- 88 and 96% of the FSA flowing into the external nitrification system was nitrified and 93 and 95% of the total ENBNRAS system nitrification occurred externally for the 8 and 5 day sludge age configurations respectively. This demonstrates one of the main benefits of the ENBNRAS system configuration - virtually complete nitrification at 8 and 5 days sludge age with a aerobic mass fraction of only 20%. This will be attainable also at temperatures lower than 20 °C.

- The TKN removals for the 8 and 5 day sludge age configurations were 94 and 92% respectively. The total N removal was 92 and 76% respectively. The discrepancies in the total N removals for the 8 and 5 day sludge age configurations were due to the difference in the influent TKN COD ratios of the sewage batches fed to the two configurations; for the 8 day sludge age configuration this was 0.096 mgN/mgCOD, while for the 5 day sludge age configuration it was much higher at 0.120 mgN/mgCOD. The higher influent TKN COD ratio combined with a lower denitrification potential of the 5 day sludge age configuration led to more nitrate in the effluent and therefore a lower total N removal.

- The P removal for the 8 day sludge age configuration was 14.0 mgP/ℓ influent. For the 5 day sludge age configuration this was 8.6 mgP/ℓ influent. The 8 day sludge age configuration removed 4.2 mgP/ℓ influent more than the System 3 10 day sludge age configuration, showing that there is indeed an improvement in BEPR as the sludge age is decreased, as reflected in the BEPR model of Wentzel et al. (1990). It would therefore be expected that the 5 day sludge age configuration would show a further improvement in BEPR performance. However, for the 5 day sludge age configuration, high nitrate concentrations were recycled to the underflow anoxic reactor, which was consequently overloaded, causing nitrate to enter the anaerobic reactor, which in turn caused a decrease in P removal.

- The percentage anoxic P uptake for the 8 day sludge age configuration was 47% (with a nitrate load of 20.7mgN/ℓ on the main anoxic reactor), and the overall average percentage anoxic P uptake for the 5 day sludge age configuration was 58% (with a nitrate load of 34.9 mgN/ℓ on the main anoxic reactor). This shows clearly that as the nitrate load on the main anoxic reactor increases, the percentage anoxic P uptake increases.

- The DSVI for the 8 and 5 day sludge age configurations was about 90 and 93 mℓ/gTSS respectively.

4.3   Closure

A detailed analysis of the results of the 8 and 5 day sludge age configuration was not intended, as the system performance evaluation lasted for only 62 days. However, the results show that the ENBNRAS system BNR performance in no way deteriorated at the shorter sludge ages, in fact a reduction in sludge age tends to increase N and P removal per mass of organic load (Wentzel et al., 1990), provided that it is not reduced below a lower limit of about 5 days for operational reasons (sludge flocculation, effluent turbidity); in terms of 'wash out' of PAOs the sludge age can be as low as 3 days (Wentzel et al., 1990).

5.   TASK 3:   BNR AT LOW TEMPERATURES

The ENBNRAS system is expected to be particularly beneficial at low temperatures, due to sensitivity of the nitrifiers to temperature. Accordingly, information on BNR behaviour at low temperatures needs to be evaluated. Information on this is available in the literature and from

previous investigations at UCT on filamentous organism bulking (Pilson *et al.*, 1995). Therefore this was not investigated experimentally in this research project.

6. TASK 4: COMPARISON OF THE ENBNRAS SYSTEM WITH A CONVENTIONAL BNRAS SYSTEM

6.1 Introduction

In implementing the ENBNRAS system, its performance with respect to nutrient removal should not be significantly reduced compared to the conventional BNRAS system. With regard to N removal via denitrification, the investigations in the previous tasks indicated that the ENBNRAS system is superior in that near complete denitrification can be achieved for influent wastewater TKN/COD ratios that are considerably higher than those for which it can be achieved in conventional BNRAS systems. However, with regard to P removal, the ENBNRAS system stimulates anoxic P uptake in the system which would appear to be reduced compared to aerobic P uptake. This indicates that the P removal performance of the ENBNRAS system may be reduced compared to the conventional BNRAS system. In this task this aspect is examined in more detail, to compare more precisely the N and P removal performance in the external nitrification and conventional BNR systems. Also, the factors stimulating anoxic P uptake in the conventional BNRAS system are investigated. This investigation is reported in detail by Vermande *et al.* (2000).

In this investigation, a laboratory scale BNRAS system (UCT configuration) with similar design and operating parameters to the ENBNRAS system (System 3, Phases II and III) was run in parallel with the System 3 ENBNRAS system. In order to compare the performance of the two systems, they were fed the same influent sewage (prepared in the same container and split equally between the two systems) for 254 days.

6.2 System performance

From the investigations:

- The COD removal achieved for the UCT and ENBNRAS systems were 93 and 94% respectively. In terms of carbonaceous material removal, the two systems performed identically.
- The overall average total oxygen demand (including nitrification) of the UCT system was 7625 mgO/d while that of the ENBNRAS system was 1798 mgO/d. By nitrifying externally, the ENBNRAS system required 76% less oxygen per day; this is a significant reduction.
- The TKN removals for the UCT and ENBNRAS systems were 86 and 87% respectively and the total N removals 78 and 88% respectively. On average, the effluent total N of the UCT system was 16.8 mgN/ℓ, of which 12.8 mgN/ℓ was NO_x and 4.0 mgN/ℓ was TKN. Of the 4.0 mgN/ℓ TKN, 1.8 mgN/ℓ was FSA. For the ENBNRAS system the effluent total N was on average 9.8 mgN/ℓ, of which 4.6 mgN/ℓ was NO_x and 5.2 mgN/ℓ was TKN. Of the 5.2 mgN/ℓ TKN, 3.5 mgN/ℓ was FSA. The ENBNRAS system achieved an average effluent total N concentration <10 mgN/ℓ while the UCT system did not.
- In the UCT system, on average, 21.3 mgP/ℓ influent P was released in the anaerobic

reactor and 34.0 mgP/ℓ uptake occurred mainly in the aerobic reactor. In the ENBNRAS system. on average. 18.3 mgP/ℓ P was released in the anaerobic reactor and internal settling tank with an additional P release (which also had to be taken up in the anoxic and aerobic reactors) of 4.5 mgP/ℓ influent in the external nitrification system, and 32.8 mgP/ℓ P uptake occurred in the anoxic and aerobic reactors.

- The average P removal for the UCT system was 12.7 mgP/ℓ influent while that of the ENBNRAS system was 9.8 mgP/ℓ influent. The UCT system showed only 9.8% anoxic P uptake on overall average. showing that predominantly aerobic P uptake BEPR occurred in the UCT system. In the ENBNRAS system. 60% of the P uptake occurred in the anoxic reactor. During 7 (out of 28) sewage batches. anoxic P uptake was induced in the UCT system by feeding sewage with high TKN COD ratios (leading to high nitrate loads on the anoxic reactor). During this period the UCT system showed about 18% anoxic P uptake and the P removal decreased to the same level as measured in the ENBNRAS system.

- The overall average DSVI for the UCT and ENBNRAS systems were 138 mℓ/gTSS and 103 mℓ/gTSS respectively. During the period where sewage with a high TKN COD ratio was fed to the systems to induce anoxic P uptake in the UCT system. the DSVI of the UCT system increased significantly from around 110 mℓ/gTSS to over 200 mℓ/gTSS, while the DSVI of the ENBNRAS system increased only slightly from around 90 mℓ/gTSS to around 105 mℓ/gTSS. This shows that the conventional BNRAS system reacts much more strongly to significant (>2 mgN/ℓ) nitrate concentrations in the outflow of the main anoxic reactor because its aerobic mass fraction was higher (0.5) than that of the ENBNRAS system (0.2).

6.3   Closure

In terms of carbonaceous material removal the UCT and ENBNRAS systems achieved almost identical results. For the nitrogenous material removal, the ENBNRAS system produced an effluent of better quality with an effluent total N concentration of nearly half that of the UCT system on overall average. The UCT system, which exhibited predominantly aerobic P uptake BEPR, removed about 3 mgP/ℓ influent more P than the ENBNRAS system with anoxic/aerobic P uptake BEPR did. BEPR is the only process where the UCT system achieved superior results to that of the ENBNRAS system. The ENBNRAS system BNR is effected using approximately 75% less oxygen than was required by the UCT system. The ENBNRAS system consistently produced a better settling sludge than the UCT system did, and furthermore, the ENBNRAS system did not produce a bulking sludge when high nitrate concentrations flowed from the anoxic reactor, as was observed in the UCT system.

7.   TASK 5:   IMPLEMENTATION OF THE ENBNRAS SYSTEM AT FULL SCALE (ECONOMIC EVALUATION)

7.1   Introduction

The experimental investigations into the ENBNRAS system at laboratory scale have demonstrated that this system holds considerable promise for practical implementation. However, successful implementation of the ENBNRAS system at full scale will depend largely on its cost compared to the conventional BNR activated sludge systems and its ability to meet required

effluent quality standards. Accordingly, the system should be assessed with respect to the economic cost and effluent quality standards, and these compared to those for an equivalent conventional BNRAS system. In this task such a practical case study is demonstrated based on a specific plant (Potsdam WWTP at Milnerton in Cape Town, South Africa) and the effluent quality standards of South African law. Details of the economic evaluation are reported by Little *et al.* (2001).

The economic cost will be influenced to a large extent by site specific factors. For the purpose of the evaluation, the Potsdam WWTP at Milnerton in Cape Town, South Africa was selected since considerable data are available for this plant. The existing Potsdam WWTP comprises two streams which are operated in parallel:

- The old rock media trickling filters with a capacity of 18 Mℓ/d. These trickling filters are low loaded and achieve good COD removal and nitrification. However, no N and P removal are obtained. A chemical P removal plant has been provided but currently (2001) this is not operated.
- The newer conventional BNRAS system (Modified UCT process) with a capacity of 17 Mℓ/d with full biological nutrient removal.

Therefore, the plant has a combined capacity of 35 Mℓ/d. The trickling filter effluent can be treated chemically with aluminium sulphate or ferric chloride for P removal and with lime for alkalinity and pH correction.

7.2   Economic evaluation

The engineering and economic evaluation by Little *et al.* (2001) provides system schemes and capital, operation and maintenance costs for 2 scenarios, (i) a 35 Mℓ/d greenfields scenario (assuming that no plant exists at Milnerton) and (ii) scenarios where the existing Potsdam plant is modified to treat the entire existing flow for nutrient removal, and expanded to cope with an increased flow of 49Mℓ/d expected in the year 2020. The costs for each scenario are given in net present values (NPV) at 6% in the year 2000 in South African Rands and include capital and operational costs for the indicated period:

Scenario 1 (35 Mℓ/d greenfields plant):

		Cost(million ZAR)	Period
1a)	Conventional BNRAS system (UCT process)	147.44	2000-2015
1b)	ENBNRAS system	138.25	2000-2015

Scenario 2 (Modifying the existing plant):

2a)	35 Mℓ/d conventional BNRAS system	93.55	2000-2015
2b)	35 Mℓ/d ENBNRAS system	76.50	2000-2015
2a)	49 Mℓ/d conventional BNRAS system	164.80	2000-2020
2b)	49 Mℓ/d ENBNRAS system	154.27	2000-2020

7.3 Discussion

When the project was initiated, it was thought that a significant saving in capital cost would be achieved for the ENBNRAS system as a result of the increase in treatment capacity or smaller biological reactors, the reduction in oxygen demand and a better settling sludge. These appeared to make the ENBNRAS system an attractive and viable alternative as a full scale wastewater treatment plant. However, the economic evaluation indicated that the savings may not be as large as initially expected.

Although the ENBNRAS system does provide a cost saving in construction costs of about 30% compared to a conventional BNR activated sludge system, the operating and maintenance costs in the long term mask this saving, because these costs are around 70% of the overall cost and virtually the same for both ENBNRAS systems and conventional BNR activated sludge systems. While significant savings in operation costs are made from the very low oxygen demand, the increased sludge production at a shorter sludge age and the associated increased cost in sludge treatment, transport and disposal offsets this saving. Although the total NPV (capital, operation and maintenance) for the ENBNRAS system is 5 to 10 % lower than that of a conventional BNR activated sludge system, this difference may not be large enough to favour selection of the new and untested ENBNRAS system over the tried and trusted conventional BNR activated sludge system. It must be remembered, however, that the sludge treatment route is specific to the Potsdam WWTP (disposal to landfill) and is a relatively costly alternative and contributes significantly to the total cost. For other WWTPs, if sludge treatment and disposal costs can be reduced, then this will make the ENBNRAS system more economically attractive.

The most significant advantage of the ENBNRAS system is that it offers biological N and P removal for wastewater treatment plants with existing trickling filters and activated sludge units without an increase in existing process units, enabling these plants to meet the stricter new effluent quality standards proposed under the National Water Act of 1998 by the Department of Water Affairs and Forestry. The ENBNRAS system will provide a feasible and economical upgrading option. Although the ENBNRAS system does not provide a large enough savings in monetary terms to make it as an attractive system, the new effluent quality standards may favour implementation of the ENBNRAS system, because the ENBNRAS system is capable of producing effluents with a quality comparable with (and for effluent N better than) conventional BNR activated sludge plants. Therefore, it is anticipated that the proposed new effluent quality standards will be the driving force that promotes application of the ENBNRAS system at full-scale, i.e. implementation probably will be legislation driven.

8. TASK 6: MODEL DEVELOPMENT FOR THE ENBNRAS SYSTEM

8.1 Introduction

For the design and operation of, and research into, conventional biological wastewater treatment systems, mathematical simulation models have proved to be invaluable as a process evaluation tool. Mathematical simulation models provide quantitative descriptions of the dynamic behaviour of the wastewater treatment system. By providing quantitative descriptions, they allow predictions of the system response and performance to be made. From these predictions, design and operational criteria can be identified for optimization of system performance. Also,

mathematical models are very useful as research tools. By evaluating model predictions, it is possible to test hypotheses on the behaviour of the wastewater treatment system in a consistent and integrated fashion. This may direct attention to issues not obvious from the physical system and lead to deeper understanding of the fundamental behaviour patterns controlling the system response. Recognising the usefulness of mathematical models, it was decided to develop a kinetic simulation model that includes the processes that can be expected to be operative in ENBNRAS systems. For the full detail of this development, the reader is referred to Hu *et al.* (2001).

The approach taken to develop the kinetic simulation model for the ENBNRAS system was to: (i) evaluate and compare the available existing kinetic models for BNR activated sludge systems; (ii) identify difficulties and deficiencies in the models when applied to BNR activated sludge systems in general, and to the proposed ENBNRAS system in particular; (iii) gather information from the literature and experiments to overcome the difficulties and deficiencies to develop the kinetic simulation model; (iv) gather data from the literature to calibrate and validate the resultant kinetic model; and (v) apply the model to simulate the behaviour of the proposed system. State of the art models identified for evaluation were: UCTPHO (Wentzel *et al.*, 1992), IAWQ Activated Sludge Model (ASM) No2 (Henze *et al.*, 1995) and ASMNo2d (Henze *et al.*, 1998), BIOWIN model of Barker and Dold (1997) and the Delft group of models (Smolders *et al.*, 1994 ; Kuba *et al.*,1997). In the initial assessment of these models and the ENBNRAS system, one process immediately apparent to be critically important was anoxic P uptake and denitrification by PAOs, and attention was focussed on this aspect.

8.2   Model development

The UCTPHO kinetic model was modified to extend application to conventional BNRAS systems where anoxic P uptake occurs and particularly to the ENBNRAS system. For these modifications information was drawn from experimental data in the literature and the other available kinetic models (e.g. ASM2d and BIOWIN). The approach followed was to include a single PAO population with reduced rates of substrate utilization/growth under anoxic conditions compared to aerobic conditions. Within this basic approach, the main modifications made to UCTPHO were:

- Processes for anoxic growth of PAOs with associated anoxic P uptake and denitrification were included, with separate yield coefficients for anoxic and aerobic PAO growth and separate ratios for P uptake per unit PHB substrate utilized. The anoxic rates were reduced compared to the equivalent aerobic rates, by multiplying the aerobic rates by a reduction factor under anoxic conditions.
- Processes for anoxic death/maintenance of the PAOs were included, recognizing that under anoxic conditions PAOs that can denitrify will behave differently from those that cannot.
- Separate anoxic and aerobic yield coefficients were introduced for ordinary heterotrophic organism growth.
- The organic N and P fractions were linked to the corresponding COD fractions, and the transformations between the different forms of organic N and P fractions were linked to the corresponding COD transformations (Henze *et al.*, 1995).

In order to calibrate and validate this kinetic model, it was applied to a wide variety of

conventional BNRAS systems operated at laboratory scale in the UCT laboratory over a range of influent concentrations, system configurations, sludge ages, mass fractions and recycle ratios. The main objectives of this application were to evaluate the predictive capacity of the model, evaluate values for the model parameters and calibrate those parameters for which values were not available. With essentially a single set of constants, close correlation between predicted and measured performance was obtained for most of the systems, thereby validating the model.

8.3   Application of the model to ENBNRAS systems

The kinetic model was applied to the laboratory scale ENBNRAS systems operated for the previous tasks and predicted behaviour compared to the experimental observations. This showed that the model is capable of modelling ENBNRAS system behaviour with the default values for kinetic and stoichiometric parameters proposed, except for $f_{P,rel}$, η_H, η_G and K_{mp}. This provides substantive evidence supporting the validity of the proposed kinetic model. In the model application $f_{P,rel}$ had to be reduced from the default value of 0.5 to 0.4. The η_G value needs to be determined by simulation for each system; however, as the PAOs contribution to denitrification is small, a default value of $\eta_G = 0.5$ can be used for ENBNRAS systems with little error. The K_{mp} value had to be increased significantly in the application to ENBNRAS systems, in order to achieve improved OUR and VSS predictions. The η_H value had to be calibrated in association with the K_{mp} value, and the values for η_H appeared to be influenced significantly by the nitrate load on the anoxic reactor.

8.4   Closure

A kinetic model describing the behaviour of ENBNRAS systems (and BNRAS systems in general) has been developed, including ordinary heterotrophic organisms (OHOs), autotrophic organisms (AOs) and phosphorus accumulating organisms (PAOs). The process kinetics and stoichiometry for the OHOs and AOs were taken largely unmodified from the UCTPHO kinetic simulation model (Wentzel et al., 1992), except that the transformations in N and P compounds were linked to the equivalent transformations in the COD compounds (Henze et al., 1995). For the PAOs, the anaerobic and aerobic kinetic and stoichiometric behaviour were also taken from UCTPHO, but additionally kinetics and stoichiometry for anoxic PAO growth and death (with associated anoxic P uptake and PAO denitrification) were included, based on information available in the literature.

The model was calibrated and applied to an extensive data set from conventional and external nitrification BNR activated sludge systems collected in the UCT laboratory in this and previous WRC funded contracts. The correlation between observed and predicted results was good for the wide variety of systems (including ENBNRAS systems), except for conventional BNR activated sludge systems with small aerobic mass fractions.

This model provides a useful tool for the design and operation of ENBNRAS systems.

9.   CLOSURE AND RECOMMENDATIONS

The investigations on the three laboratory scale ENBNRAS systems provide a comprehensive framework for the understanding of the ENBNRAS system operation and performance, and any further laboratory investigations would not provide more knowledge and understanding. The next step would be to begin full scale trials of an ENBNRAS system. To begin with, a full scale trickling filter would have to be converted into a nitrifying trickling filter to ascertain its performance as a nitrifying trickling filter at full scale. Once it has been proven that existing full scale trickling filters can successfully be converted to nitrifying trickling filters and their capacity determined, the trickling filters can be integrated into a BNR activated sludge system in an ENBNRAS system configuration to obtain BNR on the full influent wastewater flow.

Initially it was thought that the savings in capital cost brought about by an increased capacity or smaller biological reactors, reduced oxygen demand and better settling sludge would make the ENBNRAS system an attractive and viable alternative as a full scale plant. However, Task 5 (economic evaluation) indicates that this may not be the case. While the ENBNRAS system alternative does provide a saving in construction costs of about 30% when compared to a conventional BNR activated sludge system, the operating costs in the long run overshadow this saving. The operating costs of a WWTP, whether ENBNRAS or conventional BNR activated sludge system, account for the bulk of the NPV. While significant savings in operation costs are made from the very low oxygen demand, the increased sludge production at the shorter sludge ages and the associated increase in sludge treatment, transport and disposal costs reduce these savings (However, it should be remembered that the sludge disposal costs for the case study selected WWTP are high).While the total NPV (capital, operation and maintenance) for the ENBNRAS system option is 5 to 10% lower than that of a conventional BNR activated sludge system, this difference may not be large enough for a definite choice of the ENBNRAS system over the conventional BNR system. However, the most significant advantage is that the ENBNRAS system offers biological N and P removal for the full wastewater flow without an increase in existing process units. In South Africa, if the Department of Water Affairs and Forestry implement the proposed new effluent quality standards proposed under the National Water Act of 1998, the ENBNRAS system will provide a feasible and economical plant upgrade option. The ENBNRAS system is capable of producing effluents with a quality that are within the new effluent quality standards, especially with regards to nitrogen. Thus, the proposed new effluent quality standards rather than economics may well be the driving force that will see the ENBNRAS system implemented at full scale.

From the discussion above, the next step in development of the ENBNRAS system is implementation and evaluation at full scale. In particular the behaviour of nitrifying trickling filters needs to be assessed, and this is best done at full scale. This will form the basis for a new research contract between UCT and the WRC and WSSA.

10. ADDITIONAL PRODUCTS

In addition to the research results described above, the following were produced.

10.1 Publications

- Articles in refereed journals: 3 of direct relevance, and 9 indirect.
- Articles in conference proceedings: 4 of direct relevance, and 9 indirect.
- Research reports: 5 of direct relevance, and 3 indirect.

10.2 Postgraduates

- 1 PhD student, completed.
- 2 MSc students, completed.
- 1 MSc student, in preparation.

10.3 Undergraduates

- 3 BSc (Eng) undergraduate students completed their 9 week final year thesis in this research project.

10.4 Foreign students

- 1 BSc student from Toulouse (France) completed 6 month practical training on the research project.

ACKNOWLEDGEMENTS

TABLE OF CONTENTS

LIST OF SYMBOLS AND ABBREVIATIONS

Symbol/abbreviation	Description
AO	Autotrophic organisms
ASM	Activated sludge model
BEPR	Biological excess phosphorus removal
BNR	Biological nutrient removal
COD	Chemical oxygen demand
DPAO	Denitrifying phosphate accumulating organisms
DSVI	Diluted sludge volume index
DWAF	Department of Water Affairs and Forestry
EN	External nitrification
ENBNR	External nitrification biological nutrient removal
ENBNRAS	External nitrification biological nutrient removal activated sludge
f_{CV}	Mixed liquor COD/VSS ratio
f_N	Mixed liquor TKN/VSS ratio
FSA	Free and saline ammonia
IAWPRC	International Association for Water Pollution Research and Control
IAWQ	International Association on Water Quality (formerly IAWPRC)
IST	Internal settling tank
OHO	"Ordinary" heterotrophic organisms
MLOSS	Mixed liquor organic suspended solids
MLSS	Mixed liquor suspended solids
N	Nitrogen
ND	Nitrification denitrification
NDBEPR	Nitrification denitrification biological excess phosphorus removal
NNBEPR	Non nitrifying biological excess phosphorus removal
NPV	Net present value
OHO	Ordinary heterotrophic organisms (non-P removal)
OUR	Oxygen utilization rate
P	Phosphorus
PAO	Phosphate accumulating organisms
PHA	Polyhydroxyalkanoate
PST	Primary settling tank
RBCOD	Readily biodegradable COD
SBCOD	Slowly biodegradable COD
SBR	Sequencing batch reactor
SCFA	Short-chain fatty acids
SVI	Sludge volume index
TF	Trickling filter
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TSS	Total suspended solids
UCT	University of Cape Town
VSS	Volatile suspended solids

WAS	Waste activated sludge
WRC	Water Research Commission
WWTP	Wastewater treatment plant

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CHAPTER 1

INTRODUCTION

1.1   MOTIVATION FOR THE RESEARCH

The biological nutrient removal (BNR) activated sludge (AS) system has become an established technology in wastewater treatment practice. This development has been facilitated by an improved understanding of the nitrification denitrification (ND) and biological excess phosphorus removal (BEPR) processes. However, implementation of BNRAS systems has brought with it a new set of difficulties (Ekama and Wentzel, 1997), the main ones being the long sludge age required for nitrification, filamentous organism bulking and the treatment/disposal of liquors/supernatants generated from sludge and solids handling. The last named received attention under a previous Water Research Commission contract (K5/692); this research project focusses on the first two, namely nitrification and bulking.

In the BNRAS system, the requirement to nitrify governs selection of the two linked design parameters, sludge age and aerated mass fraction. The need for nitrogen (N) and phosphorus (P) removal sets a requirement for an unaerated sludge mass fraction (anaerobic for P removal and anoxic for N removal). In N and P removal plants, the unaerated sludge mass fraction usually needs to be high, i.e. > 40%, causing the aerated mass fraction to be reduced, i.e. < 60%. To compensate, long sludge ages need to be selected to ensure nitrification. For example, with maximum specific growth rates for the nitrifiers at $20\,°C$ (μ_{nm20}) of around 0.45/d, to guarantee nitrification at the minimum temperature of $14\,°C$, the sludge age of the single sludge system must be around 20 to 25 days if 50 to 60% of the sludge mass in the system is unaerated. Such long sludge ages result in large biological reactors per Mℓ wastewater (WW) treated. If nitrification can be made independent of the suspended solids sludge age, then selection of the sludge age no longer will be governed by the requirement to nitrify, but rather by the N removal (denitrification) and P removal (BEPR) processes. For both these biological processes, a reduction in sludge age increases respectively the N and P removal per mass organic load (WRC, 1984; Wentzel et al., 1990), provided the sludge age remains longer than some lower limit to prevent "wash-out" of P removal and denitrifying organisms. Indications are that, if nitrification and the sludge age are uncoupled, then the sludge age can be reduced to less than half, from about 20 to 25 days to about 8 to 10 days. This will result in a reduction in the biological reactor volume requirement per Mℓ WW treated of about 1/3rd, or alternatively, in an increase in WW treatment capacity of some 50% (provided secondary settling tank area requirements are accommodated).

BNRAS systems have been found to promote the growth of a specific group of filamentous microorganisms, previously called low F/M (Jenkins et al., 1984) but renamed anoxic/aerobic (AA) (Casey et al., 1994). These filamentous organisms cause sludges to settle poorly, resulting in an increase in the required surface area of secondary settling tanks. If the settleability of BNRAS sludges can be improved, then the flow through existing secondary settling tanks can be increased considerably, or alternatively for a proposed system, the secondary settling tank surface area can be considerably reduced, both options providing considerable economic benefit. Casey et al. (1994) have identified two main causes for AA filament proliferation in BNRAS

systems:

- Aerated sludge mass fractions in the range 30 to 50%.
- Incomplete denitrification in the anoxic reactor preceding the aerobic reactor.

The uncoupling of nitrification and the sludge age will introduce greater flexibility into the BNR activated sludge system configuration and should allow elimination of the two conditions identified above to promote AA filament growth.

Thus, uncoupling the sludge age from nitrification will significantly increase the treatment capacity of existing BNRAS systems, or reduce bioreactor volumes and secondary settling tanks surface area in proposed systems, resulting in a major step increase in system intensification with considerable economic benefit - this is the main research focus in this project.

1.2 THE PROPOSED SYSTEM

To uncouple the suspended solids sludge age from the requirement to nitrify, the activated sludge system can be modified in two ways, by including (1) internal or (2) external fixed media for attached nitrifier growth. Internal fixed media such as Ringlace[R] and floating tubes or small sponges have been placed in the aerobic reactor of the BNRAS system (Sen et al., 1994, 1995; Randall and Sen, 1996). The nitrifiers grow on the fixed media, establishing a population permanently resident in the aerobic reactor. These nitrifiers are not subjected to either the unaerated-aerated reactor interchanges or to the suspended solids sludge age, with the result that the latter can be reduced significantly without losing nitrification. However, the effectiveness of the internal fixed media has not been as good as expected, it yields a low cost/benefit ratio and the effect of the internal fixed media on BEPR has not been investigated. To resolve these difficulties, a BNRAS scheme in which nitrification is removed from the BNRAS system to an external fixed media system is proposed. This proposed system will be termed the *external nitrification biological nutrient removal activated sludge (ENBNRAS)* system.

In the proposed system, nitrification takes place outside the suspended sludge BNRAS system in a fixed media trickling filter or similar type of system. This has the potential to reduce the sludge age of the BNRAS system significantly. Furthermore, removing nitrification from the aerobic reactor of the BNRAS system has a number of additional benefits:
1. Minimum aerobic mass fraction for nitrification not required.
2. Aerobic reactor volume (and accordingly mass fraction) does not have to be maintained to accommodate internal fixed media.
3. Aeration for nitrification in the aerobic reactor is not required, only for COD utilization.

As a consequence of 1 and 2 above, the aerated mass fraction in the proposed system is no longer controlled by nitrification and therefore the unaerated mass fraction can be large (> 60%). This would have two main benefits:
1. The anaerobic mass fraction can be increased, which should improve BEPR (Wentzel et al., 1990).
2. The anoxic mass fraction can be increased, to improve denitrification and thus N removal (WRC, 1984) to the point where complete denitrification may be possible.

The possibility of complete denitrification with the short sludge age holds promise to ameliorate AA filament bulking (Casey *et al.*, 1994). As noted above, this will be particularly beneficial as it will significantly increase the treatment capacity of an existing system, or reduce secondary settling tank surface area for a proposed system.

The DEPHANOX system (Fig 1.1) is one in which nitrification takes place externally to the BNR activated sludge system (Bortone *et al.*, 1996; Sorm *et al.*, 1996). In this system the influent wastewater is discharged to the anaerobic reactor to maximize BEPR. After the anaerobic reactor, the sludge mass is separated from the liquid in an internal settling tank and discharged to the anoxic reactor. The internal settling tank overflow, which has a high ammonia concentration, is passed through a fixed medium reactor wherein nitrification takes place. The nitrified outflow from the fixed medium reactor is discharged to the anoxic reactor for denitrification. From the anoxic reactor, the mixed liquor passes to the last reactor which is aerobic. After the aerobic reactor, the activated sludge is separated from the treated wastewater in a final secondary settling tank. The final settling tank overflow is the effluent from the system and the settled sludge is returned to the anaerobic reactor.

Fig 1.1:   The DEPHANOX biological nutrient removal system (after Bortone *et al.*, 1996; Sorm *et al.*, 1996).

It appears that the DEPHANOX system was developed with the specific objective of stimulating denitrification by BEPR organisms, generically called phosphate accumulating organisms (PAO). Using PAOs for denitrification has the advantage that the influent wastewater substrate sequestered by the PAO in the anaerobic reactor (and therefore implicated in BEPR) also is used for denitrification (and therefore N removal). Some laboratory and pilot scale experimental work has been done on the DEPHANOX system by Bortone *et al.* (1996) and Sorm *et al.* (1996). They found considerable P uptake in the anoxic reactor, indicating that denitrifying PAOs (DPAOs) did participate in the denitrification process. Also, improved sludge settleability (SVIs ~ 50 mℓ/g) have been consistently observed in a laboratory scale DEPHANOX system by Sorm *et al.* (1996).

Thus, it would appear that the DEPHANOX system holds considerable promise, producing nutrient (N and P) removal and a sludge that settles well. However, system intensification does not appear to have been a consideration in the development of and investigations into this system.

Clearly this aspect is of major benefit and requires extensive investigation.

1.3   OPPORTUNITIES FOR IMPLEMENTATION OF THE PROPOSED SYSTEM

External nitrification could be achieved at wastewater treatment plants (WWTPs) where old trickling filter (TF) plants have been extended with a BNRAS system. There are many such WWTPs, particularly in South Africa. For example, East Rand Water (ERWAT) treats in excess of 500 Ml/d, more than 50% (270 Ml/d) of which is treated in old TF plants. Some 200 Ml/d is treated in 3 combined TF/BNR activated sludge plants, of which 102 Ml/d is treated in old TFs with chemical P removal, the balance in BNRAS systems. Also, 168 Ml/d is treated in TF only plants, which at some time in the future need to be extended. In the SE Highveld region, some 10 TF/BNRAS plants treat more than 100 Ml/d, 25% of which is treated in old TFs. Then there are Krugersdorp (Gauteng), Daspoort (Gauteng), Rooiwal (Gauteng), Darvill (Kwazulu/Natal), and Athlone and Milnerton (W Cape) treating around 450 Ml/d with a TF treatment capacity in excess of 200 Ml/d. At these and similar WWTPs, to retain the benefit of the old TFs, a proportion of influent WW is passed through the TFs and the effluents are (Fig 1.2):

(1)   Discharged to the BNRAS system for biological N and P removal (e.g. van Huyssteen *et al.*, 1990). This in effect increases the TKN/COD and P/COD ratios of the WW discharged to the BNRAS system and hence increases the effluent nitrate and phosphate concentration.

(2)   Chemically treated to precipitate the P before discharge to the BNRAS system. This is not only costly, but also reduces the alkalinity of the water and only reduces the effective P/COD ratio of the wastewater on the BNRAS system, the nitrogen returned is not reduced.

(3)   Irrigated on land at the WWTP. This practice is being closely monitored by Department of Water Affairs and Forestry in South Africa and is expected to be substantially disallowed because it leads to a significant loss of valuable surface water.

If, instead of the above three strategies, the nitrification process is removed from the BNR activated sludge system and transferred to the TFs, all the WW flow can be discharged to the BNRAS system (Fig 1.3): A side-stream of mixed liquor is taken from the end of the anaerobic zone and passed through the TF 'humus' tanks (upgraded to internal secondary settling tanks) to remove the activated sludge solids. The underflow sludge is discharged to the beginning of the anoxic zone and the overflow is passed onto the TF for nitrification. The nitrified TF effluent is then discharged to the anoxic zone for denitrification. In this way the TF assists the BNR activated sludge system in its area of weakness, i.e. nitrification, rather than taking away from its strength, i.e. biological N and P removal with influent organics. Furthermore, the oxygen demand in the aerobic reactor is markedly reduced because nitrification no longer takes place there. Indeed, not only is the nitrification oxygen demand obtained "free" outside the BNR activated sludge system, but the oxygen equivalent of the nitrate generated in the trickling filter also helps to reduce the carbonaceous oxygen demand in the BNRAS system by about a 1/3rd. In fact, with external nitrification, the reduction in oxygen demand in the BNRAS system is much greater than when 1/3rd of the wastewater is bypassed to the trickling filter as in existing TF/BNRAS systems. Therefore, by changing the TF to a nitrifying system, the treatment capacity of the BNRAS plant is increased without having to increase aeration capacity. If a TF plant is not available, it is possible to construct an artificial external fixed media system, the cost of which may be offset by the increase in WW treatment capacity.

Fig 1.2:   Conventional integration of trickling filters with biological nutrient removal activated sludge systems.

Fig 1.3:   Proposed integration of trickling filters with biological nutrient removal activated sludge systems: Nitrification is achieved externally on nitrifying trickling filters.

At short sludge ages and small aerobic mass fractions, nitrifiers would not ordinarily be supported in the BNRAS system. However, it will not be possible to completely exclude nitrifiers from the system because nitrifiers are likely to be seeded into the system from the TF effluent. Therefore, the potential for nitrification in the aerobic reactor will always exist in the system, and the potential nitrate concentration in the aerobic reactor will be governed by the ammonia concentration that enters it. Provided the TF nitrifies well, this nitrate concentration will be mainly from the ammonia which bypasses the TF via the internal settling tank underflow and therefore will be relatively low. If the TF does not nitrify well and the residual ammonia concentration from it is high, then, if sufficient nitrifiers are present in the aerobic reactor, the nitrate concentration will be high, with the result that a significant nitrate concentration will be present in the underflow from the final settling tank. To protect the BEPR against this potential nitrate ingress to the anaerobic reactor, a pre-anoxic (underflow anoxic) reactor is placed in the underflow to denitrify the nitrate (Fig 1.3). If sufficient nitrifiers are not present in the aerobic reactor, then the ammonia concentration in the aerobic reactor will only be partially nitrified, with the result that return sludge nitrate concentration will be relatively low, but the effluent TKN concentration will be high - the concentration depending on the nitrification efficiency of the TF.

Tertiary nitrifying trickling filters, which are employed for nitrification only and negligible organic material removal, are fairly common in USA (Lutz et al., 1990). While certain problems with macro fauna (snails, worms, larvae and flies), which reduce nitrification rates, have been encountered, high removals of ammonia have been economically achieved in tertiary nitrifying trickling filters (Parker et al.,1989;1995;1996). Therefore, while some full scale trials would need to be done to determine how rock media trickling filters can be adapted to fulfill the external nitrification function, from the USA experience this is not expected to be a major problem in implementing the external nitrification scheme. Depending on the TF treatment capacity at the WWTP, it may be necessary to replace the rock media with plastic media to achieve the required nitrification capacity.

Thus, considerable opportunities exist in South Africa for implementation of the proposed scheme, and accordingly the system merits further investigation.

1.4   OBJECTIVES AND AIMS OF RESEARCH

Before the ENBNRAS system can be implemented at full-scale, it needs to be comprehensively investigated at laboratory-scale and this is the principle objective of this research contract:
- To evaluate, investigate, develop and refine at laboratory-scale the external nitrification biological nutrient removal activated sludge (ENBNRAS) system.

To achieve this objective, the following aims were identified:
1. To examine the effect of external nitrification on BNRAS systems.
2. To examine N & P removal performance and sludge settleability of non- or partially nitrifying BNR activated sludge systems at short sludge ages (5 to 10 days) and over temperature range 12 to 20°C at laboratory-scale.
3. To examine BEPR performance and sludge settleability in anaerobic-anoxic zone (i.e. no aerobic zone) BNRAS systems at laboratory-scale.
4. To extend BNR simulation models to include anoxic phosphate uptake denitrification kinetics.

1.5   SPECIFIC TASKS

To address the aims above, a number of specific tasks have been identified for completion.

Task 1:   Experimental evaluation of the external nitrification BNR system

The behaviour and performance of the external nitrification BNR activated sludge system need to be assessed. In this task the performance of the system is evaluated at laboratory-scale under strictly controlled conditions. In particular, nutrient removal (both N and P) and sludge settleability are examined. The experimental investigation is summarized in Chapter 3, and reported in detail by Moodley et al. (1999), Sötemann et al. (2000) and Hu et al. (2001). In this investigation a number of sub-tasks were identified for completion.

Task 1.1:   External nitrification BNR system with small aerobic mass fractions

By removing nitrification from the aerobic reactor of the BNR activated sludge system, the mass fraction of this reactor can be considerably reduced, perhaps even to the extreme of eliminating this reactor altogether except for a small reaeration tank before final settling. In this task the external nitrification BNR system is operated with a small aerobic mass fraction and nutrient removal and sludge settleability monitored, to assess system behaviour and performance.

Task 1.2:   Aerobic P uptake in external nitrification BNR systems

In evaluating the performance of the external nitrification BNR system with small aerobic mass fractions (Task 1.1), it appeared that anoxic P uptake caused BEPR to be reduced compared to aerobic P uptake. Also, denitrification in the anoxic reactor was complete, with an "excess" denitrification potential for the influent wastewater TKN; from a denitrification point of view the anoxic reactor was in effect over designed. Furthermore, in attempting to develop anoxic P uptake enhanced cultures (Tasks 1.4 and 6), it was not found possible to develop a stable culture and the anoxic P uptake process by itself was considered unreliable and unstable. This lead to a re-evaluation of the system configuration, with a realization that over emphasis had been placed on anoxic P uptake. If significant aerobic P uptake could be stimulated in the system, BEPR would be improved. In this task, aerobic P uptake in the external nitrification system is investigated.

Task 1.3:   Nutrient removal in the external nitrification BNR system with variable aerobic and anoxic mass fractions

From the investigation into aerobic P uptake in the external nitrification BNR activated sludge system, it was evident that the aerobic P uptake stabilized P removal. However, initially the investigation was hindered by the difficulties experienced with nitrification in the fixed media stone column, particularly due to infestation of the trickling filter fly *Psychoda*. To resolve this difficulty, the stone column was replaced with a suspended media activated sludge system for nitrification. This proved successful and enabled consistent nitrification to be achieved, with the result that nutrient removal performance could be investigated more consistently and comprehensively. Following the successful resolution of the external nitrification, a more extensive investigation has been conducted to examine the effect of variable aerobic and anoxic

mass fractions on the nutrient removal performance in the external nitrification BNR system. In particular, the denitrification performance in the system and its interrelationship with P removal are investigated.

Task 1.4:   Anoxic P uptake behaviour

Central to the external nitrification system is the behaviour in the anoxic reactor. In the system the mass faction of this reactor is increased at the "expense" of the aerobic mass fraction. From experimental work on the DEPHANOX system, this is expected to stimulate denitrification by PAOs, with anoxic P uptake. However, little quantitative information exists on anoxic P uptake. Qualitative indications are that anoxic P uptake is not as efficient as aerobic P uptake (Ekama and Wentzel, 1997) and causes reduced BEPR. It was hoped to collect information on anoxic P uptake processes by developing enhanced cultures of anoxic P uptake PAOs. However, the enhanced cultures did not prove successful and were abandoned (for details see Hu *et al.*, 2001). Accordingly, in this task behaviour of anoxic P uptake is investigated as part of the investigation into the external nitrification BNR activated sludge system (see Task 4 also).

Task 2:   External nitrification BNR systems at short sludge ages

With external nitrification, the BNR activated sludge system can be operated at short sludge ages. Information on BNR behaviour at sludge ages of 10 and 8 days has been collected in Task 1 above. Information on external nitrification BNR behaviour at short sludge ages in the range 5 to 8 days has been collected, and is summarised in Chapter 4, and reported in detail by Sötemann *et al.* (2000). Information on BNR at very short sludge ages in the range 3 to 6 days is available from a previous investigation at UCT (Burke *et al.*, 1984).

Task 3:   BNR at low temperatures

The external nitrification system is expected to be particularly beneficial at low temperatures, due to sensitivity of the nitrifiers to temperature. Accordingly, information on BNR behaviour at low temperatures needs to be evaluated. Information on this is available in the literature, and from previous investigations at UCT on filamentous organism bulking (Pilson *et al.*, 1995). Accordingly, this was not investigated experimentally in this research project.

Task 4:   Comparison of the external nitrification BNR with a conventional BNR activated sludge system

In implementing the external nitrification BNR activated sludge system, its performance with respect to nutrient removal should not be significantly reduced compared to the conventional BNR activated sludge system. With regard to nitrogen removal via denitrification, the investigations above indicate that the external nitrification BNR system is superior in that complete denitrification can be achieved for influent wastewater TKN/COD ratios that are considerably higher than those for which it can be achieved in conventional BNR systems. However, with regard to P removal, the external nitrification BNR system stimulates anoxic P uptake in the system which would appear to be reduced compared to aerobic P uptake (Task 1.4 above). This indicated that the P removal performance of the external nitrification BNR system may be reduced compared to the conventional system. In this task this aspect is examined in more

1.9

detail, to compare more precisely the N and P removal performance in the external nitrification and conventional BNR activated sludge systems. Also, the factors stimulating anoxic P uptake in the conventional BNR activated sludge system are investigated.

This task is summarised in Chapter 5, and reported in detail by Vermande *et al.* (2000).

Task 5: Implementation of external nitrification BNR systems at full-scale

Successful implementation of the external nitrification BNR activated sludge system will depend largely on its cost compared to conventional BNR activated sludge systems and its ability to meet proposed new effluent quality standards. In this task an economic evaluation of implementing the external nitrification BNR system is undertaken, and the costs compared to those for an equivalent conventional BNR system. The Milnerton (Potsdam) Wastewater Treatment Plant is selected for an economic evaluation of the external nitrification system. Also, the expected effluent quality of conventional and external nitrification BNR systems are evaluated against the proposed new effluent quality standards.

This task is summarised in Chapter 6, and reported in detail by Little *et al.* (2001) and Hu *et al.* (2001).

Task 6: Model development for external nitrification systems

For the design and operation of, and research into conventional biological wastewater treatment systems, mathematical models have proved to be invaluable as a process evaluation tool (e.g. Dold *et al.*, 1980, 1991; Henze *et al.*, 1987). From model predictions, design and operational criteria can be identified for optimization of system performance. Also, mathematical models are very useful as research tools. By evaluating model predictions, it is possible to test hypotheses on the behaviour of the wastewater treatment system (e.g. biological processes, their response to system constraints, etc.) in a consistent and integrated fashion. This may direct attention to issues not obvious from the physical system and lead to deeper understanding of the fundamental behavioural patterns controlling the system response. In essence, mathematical models can provide a defined framework which can direct thinking (design, operation or research).

Recognising the usefulness of mathematical models, it was decided to develop a model that will include the processes that can be expected to be operative in external nitrification BNR activated sludge systems.

As noted above (Task 1.4), it was hoped to collect information on anoxic P uptake processes for the model by developing enhanced cultures of anoxic P uptake PAOs. However, the enhanced cultures did not prove successful and had to be abandoned (for details, see Hu *et al.*, 2001). Accordingly, information on anoxic P uptake processes has been collected from the literature and from previous investigations in the UCT laboratory on BNR. Existing models for BNR are evaluated and the most well established model selected and modified to incorporate anoxic P uptake and other processes of importance. This model is calibrated and validated against the literature data, and then applied to the external nitrification BNR systems described in Task 1 above.

This task is summarised in Chapter 7, and reported in deta'l by Hu *et al.* (2001).

CHAPTER 2

LITERATURE REVIEW

2.1 INTRODUCTION

In Chapter 1, an external nitrification BNRAS system (ENBNRAS) has been proposed. To provide guidelines for the research into the proposed system, a brief literature survey will be presented in this chapter, to review aspects relevant to the proposed ENBNRAS system. The review is focussed on two areas:

- The first concerns the biological processes, i.e. nitrification, denitrification, and BEPR as well as AA filamentous bulking.

- The second is on modelling BNRAS systems.

2.2 NITRIFICATION

Nitrification is the biological process whereby ammonia is oxidised to nitrate in the presence of oxygen, and is mediated by the organism group conventionally termed nitrifiers, or more recently autotrophic organisms (AO). Nitrifiers can grow in either suspended media, i.e. suspended activated sludge system, or fixed media such as trickling filters. In this section, nitrification behaviour in the different media systems will be evaluated and compared, based on information in the literature.

2.2.1 Nitrification in suspended media BNRAS systems

Nitrification is the biological process whereby free and saline ammonia is oxidized to nitrate by nitrifying organisms. These organisms are chemical autotrophs and have characteristics that differ significantly from the heterotrophs. In particular, they are slow growing and have a maximum specific growth rate only about 1/10th of that of heterotrophs. They utilize ammonia to obtain energy for cell synthesis, and dissolved inorganic carbon for their carbon requirements. The ammonia required for cell synthesis is negligible (<2%) compared to the ammonia converted to nitrate for obtaining energy. This allows the nitrifiers to be considered simply as a catalyst in the stoichiometric reactions for nitrification.

A number of factors influence nitrification, viz. influent source, temperature, pH and alkalinity, unaerated zones, dissolved oxygen (DO) concentration and cyclic flow and load conditions. All these factors, except for the last two, affect the maximum specific growth rate (μ_{nm}), which in turn affects the minimum sludge age for nitrification and the residual effluent ammonia concentration. The μ_{nm} value for different sewages varies considerably, between 0.2/d and 1.0/d, and is so unpredictable that it should be considered a sewage characteristic rather than a kinetic constant. The μ_{nm} (and half saturation coefficients) is quite sensitive to temperature, halving for every $6°C$ temperature decrease. The μ_{nm} is extremely sensitive to pH, halving for every pH unit decrease below 7. Nitrification itself releases hydrogen ions and consumes 7.14 mg/ℓ alkalinity

(as $CaCO_3$) for every nitrate generated. This is particularly important for low alkalinity influents in which nitrification can be self inhibitory, by lowing the pH below a critical value (<6).

Introduction of unaerated zones in the activated sludge system to accommodate denitrification and BEPR reduces the nitrifier mass held under aerobic conditions, and this has an effect equivalent to reducing the μ_m in proportion to the aerated sludge mass fraction of the system. This aspect is important in the design of N and P removal activated sludge plants, because if nitrification is jeopardized by having an unaerated fraction that is too large, denitrification (N removal) is no longer possible.

Because the concentration of ammonia in the influent is very low compared with the COD concentration (i.e. TKN/COD ratio = 0.1 to 0.12) and the nitrifier yield coefficient is very low compared to that of heterotrophic microorganisms, the mass of nitrifiers can be considered almost negligible in BNRAS systems. This small population requires favourable conditions to survive in biocenosis of the activated sludge. Therefore, in single sludge BNRAS systems, because the maximum specific growth rate of nitrifiers is very low and variable, the nitrification process governs selection of the two linked parameters in design:

- Aerobic mass fraction, and
- System sludge age.

To ensure complete nitrification throughout the year, in particular during winter, there are two options in design: Increasing aerobic mass fraction or extending sludge age. However, in biological N and P removal systems, the maximum aerobic mass fraction is limited by the requirement for anaerobic mass fraction for P removal and anoxic mass fraction for denitrification. Usually the aerobic mass fraction is limited to within the range 40-50%. In this case, for example, with the maximum specific growth rate of nitrifiers of around 0.45/d at 20°C, the system sludge age must be around 20 to 25 days to guarantee near complete nitrification all year round, if the minimum temperature is around 14°C. Such long sludge ages result in large biological reactor volumes per Mℓ wastewater treated. Furthermore, the maximum specific growth rate of nitrifiers varies from one municipal wastewater to another, even from one sewage batch to another for the same wastewater. Clearly, nitrification in the suspended media activated sludge system results in large and costly biological reactors. To remedy this shortcoming, it is proposed to uncouple the suspended solids sludge age of the activated sludge system from the requirement to nitrify. To achieve this, the activated sludge system can be modified in two ways, by including (1) internal or (2) external fixed media for attached nitrifier growth.

2.2.2   Nitrification with internal fixed media

Internal fixed media such as Ringlace[8] and floating tubes or small sponges have been placed in the aerobic reactor of the BNRAS system (Sen et al., 1994, 1995; Randall and Sen, 1996). The nitrifiers grow on the fixed media, establishing a population permanently resident in the aerobic reactor. These nitrifiers are not subjected to either the unaerated-aerated reactor interchanges or to the suspended solids sludge age, with the result that the latter can be reduced significantly without losing nitrification. However, the effectiveness of the internal fixed media has not been as good as expected, it yields a rather low cost benefit ratio and the effect of the internal fixed media on BEPR has not been investigated.

2.2.3 Nitrification in fixed media trickling filter systems

In trickling filters, nitrifiers grow in a slime layer, or film, attached to a fixed media such as rocks, horizontal wood slats, random plastic rings, polyethylene strips and corrugated plastic sheets. Corrugated plastic sheets are very popular media in modern trickling filters, which can be divided into two types: vertical and cross-flow. The cross-flow trickling filters have a higher oxygen transfer efficiency as well as a higher contact time between the biofilm and the bulk liquid and therefore are favourable for nitrification (Parker and Merrill, 1984).

According to their usage, trickling filters can be classified into two groups:

- Single-stage trickling filters;

- Two-stage trickling filters.

In the single-stage trickling filters, both carbon oxidation and nitrification are accomplished in a single unit, often termed the combined carbon-oxidation-nitrification trickling filter. In the two-stage trickling filters, the effluent from the first stage (for carbon oxidation) is irrigated over the second stage trickling filters, in which mainly nitrification takes place, often termed tertiary trickling filters or nitrifying trickling filters (NTFs).

For the most part, the trickling filters for nitrification are used as a separate-stage tertiary treatment system. According to Boller and Gujer (1986), since a tertiary nitrification system produces only 2-3 mgTSS/ℓ of total suspended solids, no additional clarifier is required and generally this system yields a higher quality effluent than single-stage systems.

2.2.3.1 Single-stage trickling filters

According to Parker and Richards (1986), in the single-stage trickling filter system, nitrification begins only when the bulk solution soluble BOD_5 concentration is 20 mg/ℓ or less. Therefore, in single-stage systems, nitrifiers can only become established in the lower portion of the trickling filters where BOD concentrations have been reduced to low values. The degree of nitrification achieved in the trickling filters depends on a number of factors, including the BOD loading rate, temperature, dissolved oxygen concentration, ammonia concentration, and the pH and alkalinity of the bulk liquid.

At high BOD concentrations, factors which affect nitrification in trickling filters include hydraulic loading, hydraulic pattern and retention time on the filter media, the dissolved oxygen concentration in the liquid, pH, temperature, influent Total Kjeldahl Nitrogen (TKN) concentration, and influent BOD concentration (USEPA, 1993). From Gullicks and Cleasby (1986), empirical design curves for nitrification in trickling filters should incorporate the parameters for hydraulic loading, influent ammonia concentration, the effects of recycling and wastewater temperature.

High BOD loading rates tend to suppress nitrification, since autotrophic nitrifying bacteria are out-competed by fast-growing heterotrophs. The presence of organic matter inhibits the growth of nitrifiers due to the increased competition for dissolved oxygen by heterotrophs (Halling-

Sorensen and Jorgensen, 1993). In a study of the single-stage system in the United States, USEPA (1991) reported hydraulic loading in the range $0.13-0.24$ $\ell/(m^2.s)$, total BOD loading in the range $1.17-1.91$ $g/(m^2.d)$ (soluble BOD loading not reported), and ammonia loading in the range $0.19-0.28$ $gN/(m^2.d)$.

From the discussions above, the single-stage trickling filter appears not to be favourable for nitrification.

2.2.3.2 Two-stage trickling filters (Nitrifying trickling filters)

Tertiary tricking filters, which are employed for nitrification only and with negligible organic material removal, are fairly common in the USA (Lutz et al., 1990). In the tertiary trickling filters, increasing the ammonia loading rate (by increasing either the ammonia concentration or the hydraulic loading rate) generally increases the nitrification rate. However, the percent removal efficiency is reduced (Gullicks and Cleasby, 1986). Mass invasions of higher organisms which graze on the nitrifying biomass in the trickling filters (e.g. filter fly larvae, worms and snails) can result in reduced nitrification rates.

Most of the available design information for nitrification in trickling filters is based on tertiary systems where the media is irrigated with secondary clarifier overflow. Complications in modelling trickling filter performance include the influence of particulate organics and nonhomogeneous biofilm thickness and density. Mechanistic biofilm models, which usually include both microbial kinetics and substrate diffusion through the biofilm, consist of a set of partial differential equations, the solution of which can be difficult even with the best numerical techniques. Accordingly, the design of attached growth nitrification processes is primarily based on empirical data from pilot-scale and full-scale systems, and it has been found that empirical data are generally consistent with mechanistic biofilm models (USEPA, 1993).

While certain problems with macro fauna (snails, worms, larvae and flies), which reduce nitrification rates, have been encountered, high removals of ammonia have been economically achieved in tertiary NTFs (Parker et al., 1989; 1995; 1996). Therefore, from the USA experience it appears to be completely feasible to use trickling filters for external nitrification in the proposed external nitrification BNR activated sludge system. External nitrification coupled to BNR activated sludge systems is investigated in detail in this research project.

2.3 DENITRIFICATION

Denitrification in activated sludge systems requires carbon source input: carbon sources may be provided either by the endogenous decay of the activated sludge biomass, by adding a supplemental carbon source (usually methanol), or by the organic material present in the influent wastewater. In this section, the focus is on denitrification by using the organic materials present in wastewater influent, i.e. influent readily biodegradable COD (RBCOD) and slowly biodegradable COD (SBCOD), since in the proposed system the influent wastewater will be the substrate source for denitrification.

In nitrification and denitrification (ND) activated sludge systems, the only heterotrophic

organisms recognized for modelling purposes are the ordinary heterotrophic organisms (OHOs), and the denitrification, therefore, is ascribed to them only. The system nitrate removal achieved is the sum of the denitrification obtained from the utilization of influent RBCOD and SBCOD by OHOs.

However, in systems that include additionally BEPR, i.e. nitrification denitrification BEPR (NDBEPR) systems, the inclusion of *polyphosphate accumulating organisms* (PAOs) complicates the denitrification behaviour, in particular, where the PAOs are involved in denitrification. In this section, denitrification by OHOs and PAOs will be reviewed.

2.3.1 Denitrification by OHOs

2.3.1.1 Denitrification in ND activated sludge systems

Stern and Marais (1974) observed in ND systems under constant flow and load conditions, that denitrification in a plugflow primary anoxic reactor took place in two linear phases: a rapid first phase which persisted for a short period then terminated, and a second slow phase which continued for the rest of the retention time in the reactor; in a plugflow secondary anoxic reactor only one linear denitrification phase was operative at a slow rate, about two thirds of the slow second rate in the primary anoxic reactor. Ekama *et al.* (1979) hypothesized that the two linear phases in the primary anoxic reactor arose from the utilization of the two biodegradable COD fractions in the influent, namely RBCOD and SBCOD:

- The first rate denitrification which is connected to the RBCOD, and
- The second to the SBCOD.

With regard to the slow single denitrification rate observed in the secondary anoxic reactor, they proposed that this is due to SBCOD from endogenous mass loss.

To incorporate denitrification in the synthesis-death-generation aerobic model of Dold *et al.* (1980) to develop a general kinetic model for the ND activated sludge systems, Van Haandel *et al.* (1981) showed that the denitrification kinetic behaviour could be modelled in terms of RBCOD and SBCOD, and that the same formulations proposed by Dold *et al.* (1980) for RBCOD and SBCOD utilization under aerobic conditions can be used to model their utilization under anoxic conditions, except that the rate of SBCOD hydrolysis/utilization under anoxic conditions needed to be reduced to about 1/3rd of that under aerobic conditions. This reduction was incorporated in the model as a constant η_H = anoxic rate/aerobic rate = 0.33, representing either the population of OHOs that are facultative or a reduction in the aerobic rate for anoxic conditions. Using the general nitrification/denitrification kinetic model, simulations of the denitrification response satisfactorily predicted two near linear phase denitrification behaviour in the primary anoxic reactor, and a single near linear phase in the secondary anoxic reactor, as observed by Stern and Marais (1974).

Accepting linear denitrification phases, Van Haandel *et al.* (1982) and Ekama *et al.* (1983) developed two simplified steady state mathematical models for ND activated sludge systems. The steady state models provided guidelines for designing ND activated sludge systems, sizing anoxic reactors, and estimating the denitrification potential (see WRC, 1984).

2.3.1.2   Denitrification in the NDBEPR activated sludge systems

In the late 70s and early 80s, the denitrification kinetics in the ND activated sludge systems was applied without correction to NDBEPR activated sludge systems, because it was repeatedly observed at both lab-and -full scale, that placing an anaerobic reactor ahead of the ND activated sludge system to form a NDBEPR activated sludge system, did not significantly change the magnitude of N removal (Siebritz et al., 1983; Nicholls, 1982). This approach gave predictions that were reasonably close to those observed experimentally.

Although apparently satisfactory, with the development of BEPR theory, an inconsistency in this approach became evident:

• The enhanced culture studies (Wentzel et al., 1989a and b) showed that PAOs in these systems did not denitrify. This implied that the RBCOD, converted to SCFA (short chain fatty acid) by OHOs and taken up by PAOs in the anaerobic reactor, no longer was available for denitrification in the primary anoxic reactor of NDBEPR activated sludge systems.

This in turn implied that the magnitude of denitrification in the primary anoxic reactor of the NDBEPR system should be significantly smaller than that in the primary anoxic reactor of the ND system. However, the experimental observations indicated that denitrification in both the ND and NDBEPR systems is approximately the same. The implication of this was that the denitrification kinetics in the NDBEPR system was different from that in the ND system.

Therefore, the denitrification kinetics in the ND activated sludge system needed to be modified for the application to NDBEPR activated sludge systems. To describe the denitrification kinetics in NDBEPR activated sludge systems, Clayton et al. (1989, 1991) undertook an extensive experimental investigation into the denitrification kinetics in NDBEPR systems by using plugflow anoxic reactors and batch tests. They observed that:

(1)   The rapid rate of denitrification associated with RBCOD was much reduced or absent in the primary anoxic reactor;

(2)   The slow rate of denitrification [in $mgNO_3$-N $(mgAVSS.d)$] associated with SBCOD in the primary anoxic reactor in NDBEPR systems was approximately 2.5 times greater than the rate measured in the primary anoxic reactor in ND systems;

(3)   The specific denitrification rate in the secondary anoxic reactor of NDBEPR systems was about 1.5 times greater than the rate measured in secondary anoxic reactors of ND systems.

From an in-depth enquiry into possible causes for these differences, Clayton et al. (1991) concluded that:

(1)   The increased denitrification rates were not due to:

(a)   denitrification by PAOs, because in their systems the PHB and P measurements indicated that PAOs did not denitrify;

(b)   the modification of the sewage organics in the anaerobic zone because sewage organics that had not passed through an anaerobic zone induced the same denitrification response as sewage organics that had passed through the anaerobic zone.

(2)   The increased denitrification rates are due to a stimulation in the activated sludge mass of an increased rate of hydrolysis of SBCOD in the anoxic reactor of NDBEPR systems, apparently induced by the presence of the anaerobic reactor in these systems.

The modified denitrification kinetics has been incorporated in the general NDBEPR mixed culture kinetic simulation model by Wentzel et al. (1992). Wentzel et al. (1992) investigated the η_H value (representing the population of OHOs that can denitrify or the reduction in the aerobic SBCOD hydrolysis/synthesis rate for anoxic conditions) over a wide range of configurations and conditions for NDBEPR systems by using the general NDBEPR kinetic simulation model. The simulations for a particular system were repeated using a series of η_H values until the experimentally measured nitrate concentrations were closely predicted. From the set of η_H values obtained for the different NDBEPR systems, the "best" η_H value was estimated. From the simulations completed (70 of which 57 could be used to evaluate η_H) a mean η_H value of 0.6 was obtained. This value was significantly increased compared to the η_H value of 0.33 in ND systems.

With the proposed ENBNR activated sludge system, the anoxic mass fraction is substantially increased compared to conventional BNR activated sludge systems. It is not known what the effect of this on OHO denitrification will be. Accordingly, this aspect is investigated in this research project.

2.3.2   Denitrification by both OHOs and PAOs

Since 1990, significant anoxic P uptake has been increasingly reported in lab-scale BNRAS systems (Kerrn-Jespersen and Henze, 1993; Kuba et al., 1993; Bortone et al., 1996; Sorm et al., 1996), and full-scale BNRAS systems (Kuba et al., 1997). Ekama and Wentzel (1999b) demonstrated that two types of BEPR behaviour have been observed in conventional single sludge NDBEPR systems:

(1)   P uptake predominantly in the aerobic reactor (aerobic P uptake);

(2)   P uptake in both aerobic and anoxic reactors (anoxic/aerobic P uptake).

With predominantly aerobic P uptake, there is a minimal PAO activity in the anoxic reactor, i.e. PAOs are not involved in the denitrification process and therefore the denitrification is mediated by OHOs only, as observed by Clayton et al. (1990) above in mixed culture NDBEPR systems, and Wentzel et al. (1989a and b) in enhanced culture BEPR systems. With anoxic/aerobic P uptake, this implies that PAOs are also involved in the denitrification process, and therefore the denitrification is mediated by both OHOs and PAOs.

Very little is known about denitrification in BNR activated sludge systems when it is mediated by both OHOs and denitrifying PAOs (DPAOs). From the previous investigations into DEPHANOX systems (one kind of ENBNRAS system, Bortone *et al.*, 1996; Sorm *et al.*, 1996), it was found that significant anoxic P uptake and associated denitrification by PAOs have been consistently observed in the systems. Actually, it seems that the DEPHANOX system was developed with the specific objective of fully exploiting DPAO behaviour. The researchers reasoned that denitrification by DPAOs would be advantageous in that the same substrate source (influent RBCOD) can be used for both denitrification and P removal, i.e. influent RBCOD is converted to SCFA that is sequestered and stored as PHB by the DPAOs in the anaerobic reactor (and therefore implicated in BEPR), and in the subsequent anoxic reactor, the stored PHB is utilized for denitrification and P uptake. In this way the influent RBCOD would be recaptured for denitrification in much the same way as in ND systems with the OHOs.

It is expected that denitrification by PAOs will also take place in the proposed external nitrification (EN) BNR activated sludge scheme (Chapter 1). It should be noted that stimulating DPAOs in the proposed system is not the main focus of this research. The research focus is on system intensification through uncoupling the suspended media sludge age from the nitrification requirement. However, since DPAO activity is expected in the proposed ENBNRAS system, the following issues need to be addressed:

- To quantify denitrification kinetics of OHOs in such systems (see above).

- To quantify denitrification kinetics of DPAOs in such mixed culture systems, i.e. in the presence of OHOs in the system; in other words, how do OHOs and PAOs compete for limited nitrate in mixed culture systems?

- To quantify the contribution of DPAOs to denitrification compared with that of OHOs.

As noted above, OHO denitrification kinetics in ENBNR activated sludge systems are not defined and this requires investigation (see Chapters 3, 4 and 5). With respect to denitrification kinetics of DPAOs, some experimental data are available from enhanced culture studies of DPAOs at Delft (Kuba *et al.*, 1993, 1996). However, in these studies BEPR was limited by the influent P concentration (effluent P concentration is zero), not by the biological processes, and thus some doubt exists on the applicability of the data to carbon-limited systems (the usual case in WWTPs). Furthermore, in enhanced culture systems, nitrate is always supplied sufficiently to maintain DPAO growth; the DPAOs therefore do not experience nitrate limitation as often occurs in mixed culture systems in which DPAOs have to compete with OHOs for limited nitrate. Therefore, the DPAO denitrification kinetics in the "Delft" enhanced culture systems may be different from that in mixed culture systems. This issue will be investigated as part of this research (see Chapter 3, 4 and 5).

Regarding the third issue, the relative contribution of DPAOs to denitrification has not been quantified. From biochemical studies (Wentzel *et al.*, 1986, Comeau *et al.*, 1986) and experimental observations (Ekama and Wentzel, 1999b), it was found that with anoxic aerobic P uptake BEPR the system P removal was reduced compared to aerobic P uptake BEPR (see below). This indicates that denitrification by PAOs comes at a cost to P removal. Therefore it is necessary to know the contribution of DPAOs to denitrification, to assess whether this is

advantageous and outweighs the associated reduction in BEPR. This aspect will be addressed in this research project (see Chapters 3, 4, 5 and 7).

2.4 BIOLOGICAL EXCESS P REMOVAL (BEPR)

Biological excess P removal (BEPR) is mediated by a group of heterotrophic organisms called polyphosphate accumulating organisms (PAOs) that exhibit the propensity to store P internally as polyphosphate. Polyphosphate serves as an energy source enabling the organisms to store substrates and simultaneously release phosphate into the bulk solution during anaerobic conditions with influent substrate present. When sufficient PAOs are accumulated in the sludge of an activated sludge system, most or all phosphate is taken up during anoxic and/or aerobic conditions and a low effluent phosphate concentration is achieved. The stored phosphate is removed via sludge wasted from the system or stripped from the biomass and precipitated with chemicals. The question to achieve a good BEPR can be simply formulated as:

• How to create conditions in the system that specifically favour PAO growth.

Proposed explanations of the biochemical behavioural patterns associated with P release and P uptake that lead to BEPR have been presented in a number of mechanistic biochemical models such as those of Comeau *et al.* (1986), Wentzel *et al.* (1986, 1992) and Mino *et al.* (1987). At the time that these biochemical models were developed, the observed BEPR was principally associated with aerobic P uptake BEPR; despite this, the biochemical model of Wentzel *et al.* (1986) recognizes and describes PAO denitrification with anoxic P uptake BEPR.

2.4.1 Aerobic P uptake BEPR

The aerobic P uptake process is reasonably well understood. Both steady state design (e.g. Wentzel *et al.*, 1990) and dynamic simulation models (e.g. Wentzel *et al.*, 1992; Henze *et al.*, 1995) incorporating the process have been developed. To obtain quantitative kinetic information for the mathematical BEPR models, enhanced PAO cultures were developed by Wentzel *et al.* (1989) in continuous-flow activated sludge systems (modified Bardenpho and UCT configurations), with acetate as the only organic substrate. Based on observations on these systems and batch tests on mixed liquor harvested from them, Wentzel *et al.* (1989a, b) developed a kinetic simulation model for enhanced PAO culture BEPR systems. With a single set of kinetic and stoichiometric constants, this model provided a very good description of the observed responses in the enhanced culture constant flow and load continuous systems and the batch tests on sludges drawn from these systems (Wentzel *et al.*, 1989b). The enhanced PAO cultures exhibited minimal anoxic P uptake and denitrification behaviour, and hence the kinetics only describe the anaerobic P release and aerobic P uptake BEPR processes: anoxic P uptake and PAO denitrification were therefore not included in this kinetic simulation model.

Wentzel *et al.* (1990) simplified and extended the enhanced culture kinetic model to develop a steady state design model for mixed culture BEPR systems receiving municipal wastewaters as influent. Again, this model was based on aerobic P uptake only. Wentzel *et al.* (1992) integrated the enhanced culture kinetic model with the kinetic model for nitrification and denitrification systems, to develop a kinetic simulation model for BNR activated sludge systems (called

2.10

UCTPHO). This model formed the basis for the IWA Task Group model ASM No. 2 (Henze *et al.*, 1995). Since both these models drew their kinetics for BEPR from the enhanced culture kinetic model, they do not include anoxic P uptake and associated processes.

2.4.2 Anoxic/aerobic P uptake BEPR

Since 1990, anoxic/aerobic P uptake has been reported more frequently in lab-scale (Kerrn-Jespersen and Henze, 1993; Kuba *et al.*, 1993), and full-scale systems (Kuba *et al.*, 1997). Ekama and Wentzel (1999b) observed this type of BEPR behaviour in a number of long term (>500 d) laboratory studies with MUCT type BNR activated sludge systems at 10 and 20 d sludge ages and at 12, 20 and 30°C (see Section 2.3.2). This type of BEPR has been also observed consistently in the DEPHANOX systems (Bortone *et al.*, 1996; and Sorm *et al.*, 1996).

There seem to be some major differences in P removal performance between the two types of BEPR behaviour, namely aerobic and anoxic/aerobic P uptake. Ekama and Wentzel (1999b) observed that in conventional NDBEPR systems:

- With only aerobic P uptake, the P release/P removal ratio, P removal/infl. RBCOD ratio and the P removal/Infl. COD ratio are around 3.0, 0.11 and 0.021 respectively and are in conformity with the steady state (Wentzel *et al.*, 1990) and dynamic state (Wentzel *et al.*, 1990 and Henze *et al.*, 1995) BEPR models.

- With the anoxic/aerobic P uptake, these ratios decrease to 1.5-2.0, 0.06-0.08 and 0.012-0.015 respectively, and the BEPR is depressed to around 60% of that with only aerobic P uptake. In order to match the P removal calculated with the steady state model of Wentzel *et al.* (1990) to that observed, the P content of the PAOs in the system ($f_{XBG,P}$) needs to be reduced from 0.38 mgP/mgPAOAVSS for aerobic P uptake to 0.10-0.28 mgP/mgPAOAVSS for anoxic/aerobic P uptake.

It seems that, given the appropriate conditions, different species of PAOs which accomplish anoxic P uptake find a niche in the system, but which have a significantly lower BEPR performance and use the influent RBCOD less "efficiently" compared with the aerobic P uptake PAOs.

As discussed above, it is expected that significant anoxic P uptake will take place in ENBNRAS systems, and this probably will result in reduced P removal. Thus, DPAO behaviour and the conditions that induce it need to be understood if the BEPR in these systems is to be defined. Also, DPAOs need to be included in the BNR activated sludge steady state design and kinetic simulation models. This requires investigation of the following aspects:

- The reduced anoxic P uptake BEPR in ENBNRAS systems;

- The conditions which stimulate DPAOs (anoxic P uptake and associated denitrification) in both the conventional BNRAS and ENBNRAS systems.

These two aspects have been investigated in this research and reported in Chapters 3, 4, 5 and 7.

2.11

2.5   ANOXIC-AEROBIC (AA) FILAMENT BULKING

It has been found that in BNRAS systems growth of a specific group of filamentous microorganisms is usually promoted, termed low F/M filaments by Jenkins et al. (1984) and renamed anoxic-aerobic (AA) filaments by Casey et al. (1994). These filamentous organisms, such as M. parvicella, type 0092, type 0041, type 1841 and type 0675, lead to sludge bulking in BNRAS systems not only in South Africa (Blackbeard et al., 1988), but all over the world (Seviour et al., 1994; Rossetti et al., 1994 and many others).

Historically, the control of AA (low F/M) filaments has been to increase the F/M ratio by incorporating selector reactors (Chudoba et al., 1973). However, in evaluating the efficacy of anoxic and aerobic selectors, Gabb et al. (1991) and Ekama et al. (1996) concluded that the selection of AA filaments is not controlled by the selector effect, i.e. by kinetic aerobic or anoxic selectors in intermittently aerated ND systems or by metabolic selection with anaerobic reactors in NDBEPR systems.

From the investigations into the DEPHANOX system, it has been found that the system consistently produced a very good settling activated sludge by suppressing the sludge bulking caused by AA filaments, in particular M. parvicella (Bortone et al., 1996; Sorm et al., 1996). No explanation for this observation was given. However, a possible explanation may be formed from the AA filamentous bulking hypothesis of Casey et al. (1994).

2.5.1   The AA filamentous bulking hypothesis of Casey et al. (1994)

Casey et al. (1994) established that:

- Low F/M filaments proliferate under low F/M (>10 day sludge age) intermittent aeration conditions [as low F/M conditions per se did not appear to influence the sludge bulking, the filaments were renamed anoxic-aerobic (AA) filaments], but not under fully aerobic or fully anoxic conditions.

With intermittently aerated nitrification/denitrification (IAND) systems, Casey et al. (1994) observed that:

- Maximum filamentous organism proliferation occurred with an aerobic mass fraction between 30 and 35%.

- The DSVI appeared to be linked to incomplete denitrification, i.e. the nitrate and nitrite (NO_3) concentration at the end of the anoxic period when the conditions switched from anoxic to aerobic.

A relationship between the DSVI and aerobic mass fraction for IAND systems fed artificial wastewater was found and is shown in Fig 2.1.

Fig 2.1: Relationship between DSVI and aerobic mass fraction, as found by Casey et al. (1994).

The AA filamentous bulking hypothesis of Casey *et al.* (1994) explained in detail by Lakay *et al.* (1999), Musvoto *et al.* (1999) and Casey *et al.* (1999a, b, c) was based on the biochemical model for aerobic-anoxic facultative behaviour of heterotrophic organisms proposed by Payne *et al.* (1973). Payne *et al.* (1973) proposed the following denitrification pathway in their biochemical model:

- $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$
(Nitrate → nitrite → nitric oxide → nitrous oxide → nitrogen gas).

Each of the nitrogen oxides is reduced at separate and specific enzyme complexes. One or more of the gaseous denitrification intermediates (NO, N_2O) that are generated under anoxic conditions have an inhibitory effect on the utilization of a substrate under subsequent aerobic conditions, as they interact with the enzyme complexes responsible for oxygen reduction. In particular, NO (nitric oxide) has been found to accumulate intra-cellularly during denitrification, and this causes a measured and prolonged inhibition of oxygen utilization under subsequent aerobic conditions.

Based on this model and their own experimental work, Casey *et al.* (1994, 1999c) proposed the following hypothesis for AA filamentous bulking:

- In BNRAS systems, heterotrophic organisms can be classified by their morphology as:
    - Floc formers or
    - Filamentous organisms.

- Floc formers are hypothesized to reduce nitrate or nitrite to nitrogen gas under anoxic conditions, while filamentous organisms are hypothesized to be nitrate reducers, reducing nitrate to nitrite only. If nitrate or nitrite are present throughout the anoxic period, the floc formers reduce the nitrate to nitrite to nitrogen gas through each of the denitrification intermediates, resulting in the presence at some level of intra-cellular NO. When these floc formers are exposed to subsequent aerobic conditions, the intra-cellular NO inhibits the utilization of oxygen, so that the floc formers continue to respire with nitrate or nitrite (i.e. aerobic denitrification), but at much reduced rates to that under anoxic conditions. In contrast, the filamentous organisms would not have any intra-cellular NO, because they perform only the first step of the denitrification pathway, and are therefore not inhibited in utilizing oxygen as an electron acceptor in the subsequent aerobic zone. This places the filamentous organisms at a competitive advantage in the subsequent aerobic zone, because they are able to utilize a greater portion of the available substrate under aerobic conditions than they would if the floc formers were not inhibited by intra-cellular NO. The filamentous organisms are thereby able to increase their relative mass in the mixed liquor, resulting in a bulking sludge. When nitrate is not present for some time before termination of the anoxic period, viz. complete denitrification occurs, the denitrification intermediates including NO are denitrified in the floc formers' cytochromes before aerobic conditions commence. Hence, the floc formers are not inhibited in using oxygen under subsequent aerobic conditions. Thus, when denitrification is complete (< 0.5 mg NO/ℓ in the anoxic reactor outflow), the floc formers are not at a disadvantage in the utilization of substrate with oxygen in the subsequent aerobic zone.

- The concentration of NO_x flowing from the anoxic reactor is therefore an indication as to whether the filamentous organisms are at an advantage in the aerobic zone, or not.

    - High NO_x concentrations flowing from the anoxic reactor are conditions conductive to AA filament proliferation and bulking, while

    - Near zero NO_x concentrations in the anoxic reactor outflow are indicative of an uninhibited floc forming organism population and better settling sludge.

These mechanisms are considered to be operative to the extent of causing poor sludge settleability at long sludge age (> 8 to 10 days) and aerobic mass fractions between ~15% and 60%.

2.5.2 The relationship between DSVI and aerobic mass fraction

The relationship between DSVI and aerobic mass fraction is very useful for designing BNRAS systems. Stewart Scott Consulting Engineers (Casey, 1998) investigated the applicability in practice of the relationship between the DSVI and the aerobic mass fraction as shown in Fig 2.1. The investigation included seven full-scale BNRAS plants, viz. Daspoort (operated by Pretoria Municipality), Rynfield and Vlakplaas (operated by ERWAT), and Goudkoppies, Bushkoppie, Olifantsvlei and Northern Works (operated by the Greater Johannesburg Metropolitan Council). These plants were chosen because they were the only plants at which historic experimental performance data were available. The historical sludge settleability data were converted to DSVI units where applicable, and an overall average DSVI value was calculated for each of the seven full-scale plants. The results obtained are shown in Table 2.1.

Table 2.1: DSVI values for seven full scale BNRAS plants in Gauteng, South Africa.

Plants	DSVI (mℓ/g)	aerobic mass fractions (%)	period of analysis
Goundkoppies	93	60	6/1997-6/1998
Bushkoppie (Unit 1)	61	76	1/1998-12/1998
Olifantsvlei (Unit 3)	61	58	1/1998-11/1998
Northern Works (Unit 4)	74	63	1/1998-11/1998
Rynfield (South)	113	50	1/1996-2/1997
Vlakplaas (Module D)	104	54	1/1997-12/1997
Daspoort (Module 9)	162	44	1/1995-3/1997

The aerobic mass fraction and DSVI data for seven full-scale treatment plants are superimposed on the artificial wastewater IAND systems DSVI versus aerobic mass fraction diagram of Casey *et al.* (1994) and plotted in Fig 2.2. It was found that a clear relationship exists between the two parameters, with improving sludge settleability for an aerobic mass fraction increase from 40% upwards.

However, it was not possible to comment on aerobic mass fractions below 40%, as data from full-scale BNRAS systems in this range were not available, and full scale systems tend not to be designed and operated with less than 40% aerobic mass fraction, due to the nitrification requirement. The information on this also is not available in lab-scale BNRAS systems because of the same reason as in full-scale plants, i.e. in conventional BNRAS systems, the aerobic mass fraction usually is larger than 40% due to the requirement for nitrification. The ENBNRAS system will provide an opportunity to observe the effect of very low (20 to 40%) aerobic mass fractions on DSVI. It is expected from the DEPHANOX experiments, that a good sludge settleability would be obtained. This aspect is reported in Chapters 3 and 4.

Fig 2.2: DSVI and aerobic mass fraction of seven full-scale BNRAS plants superimposed on the DSVI/aerobic mass fraction relationship developed for artificial wastewaters by Casey et al. (1994).

2.6   MODELLING BNR ACTIVATED SLUDGE SYSTEMS

For the design and operation of, and research into biological activated sludge systems, mathematical simulation models have proved to be invaluable as a process evaluation tool (e.g. Dold et al., 1980, 1991; Henze et al, 1987). Mathematical simulation models provide quantitative descriptions of the dynamic behaviour of the wastewater treatment system. By providing quantitative descriptions, they allow predictions of the system response and performance to be made. From the predictions, design and operational criteria can be identified for optimization of system performance. Also, mathematical models are very useful as research tools. By evaluating model predictions, it is possible to test hypotheses on the behaviour of the wastewater treatment system (e.g. biological processes, their response to system constraints, etc.) in a consistent and integrated fashion. This may direct attention to issues not obvious from the physical system and lead to deeper understanding of the fundamental behavioural patterns controlling the system

response. In essence, mathematical models can provide a defined framework which can guide thinking (design, operation or research). Recognising the usefulness of mathematical models, it was proposed to develop such a model for the ENBNR activated sludge system.

For the proposed ENBNR activated sludge system (Chapter 1), from the previous investigations into DEPHANOX systems (Bortone *et al.*, 1996; and Sorm *et al.*, 1996), it will be expected that significant anoxic P uptake BEPR would take place in the ENBNR activated sludge system. A model which can simulate this system should take this into account. This phenomenon has also been observed in conventional BNRAS systems (Kerrn-Jespersen and Henze, 1993; Kuba *et al.*, 1993, 1997; Ekama and Wentzel, 1999a and b), and thus a model that can simulate the ENBNR activated sludge system also can be applied to conventional BNRAS systems. Thus, anoxic P uptake behaviour and associated denitrification are essential processes for inclusion in the model. Development of the mathematical model and incorporation of anoxic P uptake behaviour are described in Chapter 7.

2.7   CLOSURE

From the literature review on the relevant aspects of the proposed ENBNRAS systems in this chapter it is evident that a number of aspects require detailed investigation, for example:
*   denitrification performance
*   OHO and PAO denitrification kinetics
*   PAO contribution to denitrification
*   BEPR performance
*   anoxic P uptake and the factors that stimulate it
*   development of a simulation kinetic model for ENBNRAS systems
*   occurrence of AA filament bulking at small aerobic mass fractions

These aspects amongst others are addressed in the body of the research report.

CHAPTER 3

EXPERIMENTAL INVESTIGATION INTO EXTERNAL NITRIFICATION BIOLOGICAL NUTRIENT REMOVAL (ENBNR) ACTIVATED SLUDGE SYSTEMS

3.1   INTRODUCTION

A series of three separate laboratory-scale experimental investigations on the external nitrification biological nutrient removal (ENBNR) activated sludge system were undertaken, aimed at determining the performance of this proposed system. This Chapter summarises the results from these investigations. For the details, the reader is referred to Moodley et al. (1999), Sötemann et al. (2000) and Hu et al. (2001).

3.2   EXPERIMENTAL INVESTIGATIONS

The first experimental investigation was on the proposed ENBNR activated sludge system with a small aerobic mass fraction (19%) and is reported in detail by Hu et al. (2001); for convenience in comparison and discussion, this system is referred to as System 1. This investigation actually was a feasibility study on the proposed ENBNR activated sludge system. After the proposed ENBNR activated sludge system was shown to operate successfully, the second investigation was initiated to examine the effect of varying aerobic mass fractions (and consequently anoxic mass fractions) and shorter sludge age on the ENBNR activated sludge system N and P removal performance, and is reported in detail by Moodley et al. (2000). This investigation covered two phases, and the ENBNR activated sludge system was the same as that used in the first investigation, except that in Phase I the aerobic mass fraction was reduced from 19 to 15% and the sludge age was reduced from the initial 10 days (sewage batches 1 to 3) to 8 days (sewage batches 4 to 16); this system will be referred to System 2a. Then, in Phase II, the aerobic mass fraction was increased to 30% at the expense of a reduced anoxic mass fraction (30%); this system will be referred to System 2b, which included sewage batches 17 to 33 of the second experimental investigation. The third investigation was initiated to optimize N and P removal in the ENBNR activated sludge system, and is reported in detail by Sötemann et al. (2000). This third investigation also comprised two phases and covered three systems; these will be referred to as System 3a which included sewage batches 1 to 13, System 3b which included sewage batches 14 to 20 and System 3c which included sewage batches 21 to 30. System 3a had aerobic and anoxic mass fractions of 32.5% each and included a mixed liquor (a) recycle of 2:1 between them. This was necessary because the nitrification efficiency of the external nitrification system initially was poor and the large aerobic mass fraction resulted in significant nitrification in the aerobic reactor. The a-recycle was included to (i) increase the nitrate load on the anoxic reactor and (ii) reduce the nitrate recycle to the anaerobic reactor. In System 3b, efficient nitrification in the external nitrification system was restored, and the anoxic and aerobic mass fractions were changed to 45% and 20% respectively. In System 3c the a-recycle removed. The system setup and design and operating parameters for the different investigations are summarized in Table 3.1.

3.2

The data for evaluation in this chapter were the averaged results of all sewage batches for each system. Detailed day-by-day and sewage batch average results can be found in the references above.

Table 3.1: Design and operating parameters for the ENBNRAS systems during the three investigations

Parameters	Investigation							
	1 (Hu et al., 2001)	2 (Moodley et al., 1999)				3 (Sötemann et al., 2000)		
System	System 1	System 2a		System 2b		System 3a	System 3b	System 3c
Sewage batch	1 to 13	1 to 3	4 to 16	17 to 21	22 to 33	1 to 13	14 to 20	21 to 30
Operating conditions								
Sludge age (d)	10	10	8	8		10		
Temps (°C)	20							
pH (Anaerobic)	7.2-8.2							
pH (Aerobic)	7.2-8.2							
DO (mg/ℓ)	2.0 - 5.0							
Influent (Raw sewage) from Mitchell's Plain WWTP								
Flow (ℓ/d)	20	20	40	30		20		
COD (mg/ℓ)	750							
RBCOD (mg/ℓ)	70-110							
TKN/COD	0.06-0.11[1]							
Total P (mgP/ℓ)	13.6-28.9[2]							
Reactor volume (ℓ) and mass fractions (%)								
Anaerobic	5; 23.8%	5; 25%						
Pre-Anoxic	[3]2; 9.5%	[3]2; 10%	[3]3; 15%	[3]3; 15%		[3]2; 10%		
Main Anoxic	10; 47.6%	10; 50%	9; 45%	6; 30%		6.5; 32.5%	9; 45%	
Aerobic	4; 19%	3; 15%	3; 15%	6; 30%		6.5; 32.5%	4; 20%	
Unaerated	17; 81%	17; 85%	17; 85%	14; 70%		13.5; 67.5%	16; 80%	
[4]EN system	Stone column fixed media			Suspended media activated sludge				
Recycle ratios with respect to influent flow								
s-recycle	1	1	0.5	0.5		1	1	1
To EN system	1.75	1.75	1.25	1.07		1.88	1.70	1.68
Bypass flow	0.25	0.25	0.25	0.43		0.12	0.30	0.32
a-recycle	0	0		1		2	2	0

[1] Excludes N contribution of nitrate dose to the main anoxic reactor

[2] Includes influent P supplement with K_2HPO_4 at 5-15 mgP/ℓ influent

[3] Actual volume is 1ℓ, but with s = 1:1, the equivalent volume at system VSS conc. is 2ℓ; with s = 0.5:1, the equivalent volume at system VSS conc. is 3ℓ.

[4] EN External nitrification system.

3.3 EVALUATION OF THE ENBNR ACTIVATED SLUDGE SYSTEMS

3.3.1 Carbonaceous material removal

3.3.1.1 COD mass balance

The COD mass balances for each system are shown in Table 3.2. It can be seen that COD mass balances were less than 90%, varying between 76% and 88%. These low COD mass balances were not likely to be due to the system operating and sampling procedures, because similar low COD mass balance results have also been observed in other conventional BNR activated sludge experimental systems operated in the same laboratory, see Table 3.3. This would indicate that in BNR activated sludge systems (including external nitrification or not), COD is lost to some sink that is not taken into account in the COD mass balance equation. Therefore, the COD mass balances are not suitable for confirming the system operating and sample analytical procedures and the accuracy of the experimental data. Nitrogen mass balance results are more suitable for these purposes (see below).

Table 3.2: COD mass balance components for each system during the three experimental investigations.

System	COD mass balance components (%)					Overall Balance (%)	Unaccounted for (%)	Number of days (d)
	Oxygen	Nitrate denitrified	COD used in EN	COD in wastage	COD in Effluent			
System 1	19	16	15	30	8	88	12	250
System 2a	8.5	13.5	13.8	35.5	10.8	82.1	17.9	154
System 2b	17.4	12.7	10	38	8.5	86.6	13.4	219
System 3a	18.3	13.5	21.6	22.8	7	83.2	16.8	186
System 3b	11.6	11.4	21.1	26.2	5.7	76	24	98
System 3c	10.9	12.8	16.7	31.1	5.8	77.3	22.7	137

The contributions of each COD mass balance components to the overall COD mass balance are also shown in Table 3.2 and summarised in Fig 3.1. From Table 3.2 and Fig 3.1 it can be seen that:

- 8.5% to 19% influent COD mass was removed via oxygen utilized;
- 11.4% to 16% via nitrate denitrified;
- 10% to 21.6% used in external nitrification systems;
- 22.8% to 38% via sludge wasted and
- 5.7% to 10.8% via the effluent flow, leaving
- 12% to 24% influent COD mass unaccounted for.

Table 3.3: COD and N mass balances obtained from conventional BNR activated sludge systems (see Ekama and Wentzel 1999b).

Researchers	System	COD mass balance (%)	N mass balance (%)
Clayton et al., (1989)	UCT	92	91
Kaschula et al., (1993)	MUCT	84	89
Musvoto et al. (1992)	MUCT	107	98
Pilson et al., (1995)	MUCT	84	97
Mellin et al., (1998)	MUCT	84	82
Sneyders et al., (1997)	UCT	90	92

Fig 3.1: COD mass balance components for each system during the three experimental investigations.

These results indicate that the COD utilized with nitrate as electron acceptor was close to or larger than that with oxygen as electron acceptor. This will depend on the aerobic mass fraction and nitrate available: for example, in System 2a with 15% aerobic mass fraction (Table 3.1) the COD removal via nitrate denitrified was greater than that via oxygen, but in System 2b with 30% aerobic mass fraction (Table 3.1), the COD removal via nitrate denitrified was less than that via oxygen, although in both cases nitrate was dosed into the main anoxic reactor. However, for

3.5

Systems 3a and 3b, because no nitrate was dosed into the anoxic reactor, the COD removal via nitrate denitrified was always less than that via oxygen, regardless of whether the aerobic mass fraction was large (System 3a, 30%) or small (System 3b, 20%).

The low oxygen demand in all systems was reflected in the low OUR in the aerobic reactor; for example, in System 1 the OUR was only about 29 mg/(ℓ.h) in the 19% aerobic mass fraction of the system. The OUR in an equivalent conventional internal nitrification BNR activated sludge system with 90% COD mass balance, 10 day's sludge age, complete nitrification, 90% nitrate denitrification leading to 50% recovery in nitrification, the OUR would be about 2.5 times higher, i.e. 75 mg/(ℓ.h). Clearly not nitrifying in the ENBNR activated sludge system and utilizing the nitrate generated in the external nitrification system results in a major decrease in OUR. The proportion of COD utilized with nitrate indicates that the denitrification potential of the system is very high, due to the large anoxic mass fraction (in this example 47.6%). This allows the system to treat very high influent TKN/COD ratios without jeopardizing BEPR and achieving complete denitrification, provided near complete nitrification is achieved in the external fixed media system.

The COD component of the sludge wasted was very high compared to the other COD components. This is because the sludge ages in all systems were reduced significantly, from the usual 20 - 25 days in conventional BNR activated sludge systems to 8 to 10 days. In the external nitrification system this is a prerequisite to achieve the objective of system intensification.

3.3.1.2 COD removal

From Table 3.2, the COD component in the effluent flow was very low and varied from 5.7% to 10.8%. This is reflected in system effluent COD concentrations. From Table 3.4 and Fig 3.2, the unfiltered effluent COD concentration ranged between 46 and 74 mgCOD/ℓ. The 0.45μm membrane filtered effluent COD concentration ranged between 38 and 52 mg COD/ℓ. These latter values were accepted to correspond to the unbiodegradable soluble COD concentrations in the influent, and gave the unbiodegradable soluble COD fractions ($f_{s,us}$) ranging from 0.052 to 0.084. The COD removal efficiencies were very good and ranged from 89 to 94% (see Table 3.4).

Table 3.4: Unfiltered and filtered effluent COD concentrations, and fraction of the total influent COD that is unbiodegradable soluble in each system.

Parameter	System					
	1	2a	2b	3a	3b	3c
Influent COD (mg COD/ℓ)	717	683	698	741	724	730
Unfiltered Effluent COD (mg COD/ℓ)	60	74	59	57	46	47
Filtered Effluent (0.45μm) (mg COD/ℓ)	51	52	52	48	39	38
Unbiod. soluble COD fraction ($f_{s,us}$)	0.071	0.084	0.075	0.065	0.054	0.052
COD removal (%)	92	89	92	92	94	94

Fig 3.2: Influent COD and filtered and unfiltered effluent COD for the three investigations.

It should be noted that in System 2a, the unfiltered effluent COD concentration ranged between 49 and 101 with a mean of 74 mgCOD/ℓ, and the 0.45μm membrane filtered effluent COD concentration ranged between 33 and 73 mg COD/ℓ, with a mean of 52 mgCOD/ℓ (Table 3.4 and Fig 3.2). Thus for this system, there was a large difference between filtered and unfiltered effluent COD concentrations. The reason for this was an effluent turbidity from carryover of sludge in the settling tank. This was not due to settling tank failure caused by sludge bulking, as the sludge settleability during this period was very good (DSVI<80 mℓ/g except for the first three sewage batches, see Moodley et al., 1999), rather the sludge carryover seemed to be due to pin-point floc.

The pin-point floc was observed only in System 2a, in which the aerobic mass fraction was 15%. In the other ENBNR activated sludge systems (System 1, System 2b, System 3a, 3b and 3c where aerobic mass fractions were 19%, 30%, 30%, 20% and 20% respectively), the pin-point floc behaviour was not observed, but another parallel ENBNR activated sludge system with the same aerobic mass fraction (15%) (see Hu et al., 2001) showed similar pin-point floc behaviour. Thus, the pin-point floc causing the turbid effluent seemed to be associated with a small aerobic mass fraction (<19%). This would suggest that for good floc formation, a minimum aerobic mass fraction of about 19% is required.

3.3.2 Nitrogenous material removal

3.3.2.1 Nitrogen mass balance

The nitrogen (N) mass balances for each system are shown in Table 3.5 and summarised diagrammatically in Fig 3.3. It can be seen from Table 3.5 and Fig 3.3 that the N mass balances

ranged from 87% to 96%. These results are comparable to those in other experimental investigations performed on conventional BNR activated sludge systems in the same laboratory (see Table 3.3). Therefore, the experimental data can be accepted for the system performance evaluation for each system.

Table 3.5:   N mass balance components for each system during three phase investigations.

System	NOx denitrified %	N used in EN %	N in wasted sludge %	N in Effluent %		Nitrogen balance %	Unaccounted for %
				NOx	TKN		
System 1	52	12	13	5	7	89	11
System 2a	36.2	8.7	17.4	6.3	27.2	96	4
System 2b	45.3	7.6	23.4	9.9	10.6	97	3
System 3a	50.6	2.1	16.4	11.9	7.4	88	12
System 3b	37.8	17.8	16.8	8.1	6.4	87	13
System 3c	42.8	16.7	19.4	3.8	6.1	89	11

Fig 3.3:   N mass balance components for the three investigations.

The contribution of the N mass balance components to the overall N mass balances is also shown in Table 3.5 and Fig 3.3. It can be seen from Table 3.5 and Fig 3.3 that:
- 36.2% to 52% influent nitrogen mass (including nitrate dose where applicable) was removed via nitrogen gas (denitrification);
- 6% to 17.8% was used in external nitrification systems;
- 13% to 23.4% was removed via sludge wasted and
- 9.9% to 33.5% exited the system via the effluent flow, leaving
- 3% to 15% influent nitrogen mass unaccounted for.

The N removal in the external nitrification system should be low. However, in Systems 1, 3b and 3c these components were high (12, 17.8 and 16.7% respectively), and almost equal to the N removed in the sludge wasted in the BNR activated sludge part of the system. This implied that denitrification probably took place in the external nitrification system due to insufficient air supply.

In all systems, the N removed via N₂ gas (denitrification) were very high and varied from 36.2% to 52%. Most of this denitrification took place in the main anoxic reactors. The high N removal via denitrification in the main anoxic reactors is due to the large anoxic mass fractions at the expense of aerobic mass fraction in the ENBNR activated sludge system and the dosing of nitrate in some systems (Systems 1, 2a and 2b) to realize these reactors full denitrification potential.

The influent N mass that exited the system via the effluent flow varied from 9.9% to 33.5% depending on the system nitrification efficiency and denitrification potential. The effluent TKN concentrations depend on nitrification in both the external nitrification system and the BNR activated sludge system, and the effluent nitrate and nitrite concentrations depend on both system nitrification performance and denitrification potential (see below).

3.3.2.2 Nitrification

In the ENBNR activated sludge system, nitrification is designed to take place externally in the external nitrification system. However, nitrification may take place in the BNR suspended media part of the system under some conditions. The experimental observations on nitrification in the ENBNR activated sludge system are listed in Table 3.6.

Table 3.6: Nitrification in external nitrification (EN) and BNR activated sludge systems.

Parameter	System					
	1	2a	2b	3a	3b	3c
External nitrification systems (EN)						
Flow / total flow from Anaerobic to EN (%)	87.5	85.4	62.9	94	85	84
FSA concentration into EN (mg N/ℓ)	27.1	32.2	27.4	26.1	29.2	29.5
FSA concentration out of EN (mg N/ℓ)	4.4	16.4	10	7.7	3.3	3.3
NOx generated in EN/total NOx in System (%)	88.0	77.6	57	79.4	92.6	91.4
Nitrification efficiency in EN (%)	84	49	64	71	89	89
BNR activated sludge systems (BNRAS)						
Sludge underflow / total flow from Anaerobic (%)	12.5	14.6	37.1	6	15	16
NOx generated in BNRAS/total NOx in System (%)	12.0	22.4	43	20.6	7.4	8.6
Aerobic mass fraction (%)	19	15	30	32.5	20	20
Sludge age (days)	10	10.8	8	10	10	10
Final effluent FSA concentration (mg N/ℓ)	3.8	16.7	6.3	3.6	3.6	3.6
Final effluent NOx concentration (mgN/ℓ)	3.0	4.6	7.8	8.9	4.8	2.8

Only for those sewage batches with the suspended medium activated sludge external nitrification system.

Nitrification in external nitrification system

In the experimental investigations reported here, the type of external nitrification system was not the focus of the research. Therefore, the requirement for the external nitrification system was only to achieve as high a nitrification efficiency as possible.

To achieve as much nitrification as possible in the external nitrification system, the maximum of ammonia-rich supernatant from the internal settling tank overflow should be pumped to the external nitrification system. However, this is limited by the sludge underflow which should ensure that it transfers as much as possible of the sludge from the internal settling tank to the main anoxic zone to prevent sludge from accumulating in the internal settling tank and overflowing to the external nitrification system. From Table 3.6, at least 6% but usually about 15% of the internal settling tank inflow was required to transfer the sludge from the internal settling tank to the main anoxic zone. This implies that at best only 90% (but more likely 85%) of the system ammonia can be nitrified in the external nitrification system, and 10% (more likely 15%) of the system ammonia is expected to be nitrified in the activated sludge system if complete external nitrification is achieved (neglecting the N requirement for sludge production in the system), i.e. if the effluent ammonia concentration from the external nitrification system is zero.

From Table 3.6, nitrification efficiency in the external nitrification system was 89% at best (Systems 3b and 3c) and only 49% and 64% for Systems 2a and 2b respectively. This is because the performance of the laboratory-scale stone column external nitrification system used in the initial investigations deteriorated due to *Psycoda* fly and larvae infestation, resulting in high ammonia concentrations from the external nitrification system. If the aerobic mass fraction is small, this ammonia is not nitrified in the aerobic reactor of the BNR activated sludge system. From a BEPR point of view, low nitrification in the aerobic reactor is an advantage because it avoids a high nitrate concentration return to the anaerobic reactor. However, from a N removal point of view it is a disadvantage because it (i) increases the effluent ammonia and (ii) reduces the nitrate load on the anoxic reactor and hence the denitrification. Good N removal performance requires the external nitrification system to nitrify virtually completely. If the aerobic mass fraction is "large" (e.g. System 2b), the ammonia will be nitrified in the aerobic reactor of the BNR activated sludge system. This will also be disadvantageous because (i) from a BEPR point of view the nitrate will overload the underflow anoxic reactor and enter the anaerobic reactor, (ii) the nitrate will not be available in the main anoxic reactor for denitrification and (iii) the effluent nitrate concentration will increase.

Nitrification in BNRAS activated sludge system

Nitrification in the BNR activated sludge system depends on the system sludge age, aerobic mass fraction and ammonia available (i.e. external nitrification efficiency and the flow pumped into the external nitrification system, see above). The ENBNR activated sludge system is designed to operate at a short sludge age and hence nitrification mainly depends on the aerobic mass fraction and available ammonia.

When the ENBNR activated sludge system is operated at small aerobic mass fraction, the nitrifiers are not sustained in the suspended media liquor and ammonia will exit the system with the final effluent. However, nitrifiers are seeded from the external nitrification system to the BNR suspended media system, and thus some nitrification can be expected in the aerobic zone: for example, in Systems 1, 2a, 3b and 3c with aerobic mass fractions of 19, 15, 20 and 20%

respectively, the nitrifiers were not supported in the suspended media BNR activated sludge system, but 12, 22.4, 7.4 and 8.6% of the total nitrate generated was in the BNR activated sludge system.

When the ENBNR activated sludge system is operated at relatively larger aerobic mass fractions (e.g. Systems 2b and 3a), nitrifiers are sustained in the suspended media liquor and ammonia entering the aerobic zone will be nitrified virtually completely. When this happens, complete nitrification in the external nitrification systems is vitally important to maintain BEPR, because the underflow nitrate load may exceed the denitrification potential of the pre-anoxic reactor resulting in nitrate entering the anaerobic reactor, and reducing P release and hence P removal. To reduce the nitrate concentration in the aerobic reactor, an a-recycle from the aerobic zone to the main anoxic zone can be added. However, recycling between the aerobic and anoxic reactors does appear to be detrimental to the BEPR performance of the system (see Sötemann *et al.*, 2000). When the a-recycle was removed from System 3b to System 3c, the denitrification, anoxic P uptake and P removal all improved significantly.

3.3.2.3 Denitrification

Denitrification potential

From nitrate and nitrite mass balances around each reactor and settling tank, the measured denitrification for each system were obtained and are listed in Table 3.7. It can be seen that denitrification occurred principally in the main anoxic reactor and a small part of denitrification occurred in the pre-anoxic reactor. Therefore, the system denitrification performance depends on the main anoxic reactor denitrification potential and the nitrate load on it.

It should be noted that for Systems 3a and 3b, denitrification also took place in the anaerobic reactor (5.1 and 3.0 mg N/ℓ respectively). This meant that nitrate was recycled to the anaerobic zone and thereby interfered with P release and hence P removal behaviour. Also it meant that the pre-anoxic zone had been loaded in excess of its denitrification potential. In System 3a, the pre-anoxic denitrification was 4.8 mg N/ℓ influent, but in System 3b it was only 2.9 mg N/ℓ influent for the same anoxic mass fraction (10%). It is not clear why the denitrification potentials of the pre-anoxic reactor were so different for these two systems.

For the main and pre-anoxic reactors, the denitrification potentials could be determined only in those cases where the nitrate concentrations from the main anoxic reactor were greater than 1 mg NO$_3$-N ℓ. Thus, the measured denitrification values of the main anoxic reactor for those sewage batches in which nitrate concentrations from the main anoxic reactor were larger than 1 mgNO$_3$-N/ℓ were averaged for each system and are shown in Table 3.7. These measured total denitrification values can be considered as the respective denitrification potentials of the main anoxic reactors. These values varied from 19 to 49 mg NO$_3$-N ℓ influent, mainly because the anoxic mass fraction was varied between 30% and 50% (see Table 3.1), but also partly because of the natural variation in biological systems. From these denitrification potential values, the system denitrification rates [K ; mg NO$_3$-N (mgAVSS.d)] can be calculated. However, because significant anoxic P uptake took place, both OHO and PAOs participated in the denitrification. Therefore, the BEPR behaviour will be discussed first (see below) before considering the denitrification performance further (see Section 3.3.4 below).

Table 3.7: Measured denitrification in anaerobic, pre-anoxic and main anoxic and settling tanks and denitrification potentials (mg N/ℓ influent).

Reactor	System					
	1	2a	2b	3a	3b	3c
Reactor denitrification (mgN/ℓ influent)						
Anaerobic reactor	0.5	0	0.3	5.1	3.0	0.8
Pre-anoxic reactor	2.8	2.0	3.7	4.8	2.9	2.2
Main anoxic reactor	40.1	32.2	26.9	22.0	20.4	28.4
Internal settling tank	0	0	0	0	0	0
Final settling tank	0	0	0	1.4	0.9	0.6
Denitrification potentials (mgN/ℓ influent)						
Main anoxic	4.9	34.3	25.5	22.0	19.0	31.1
Pre-anoxic	4.5	5.8	4.0	5.1	3.8	3.7

3.3.2.4 Nitrogen removal

The influent and effluent TKN and nitrate concentrations as well as N removal efficiencies for each system are given in Table 3.8 and Fig 3.4. From Table 3.8 and Fig 3.4 it can be seen that the N removal efficiencies were greater than 80%, which is very good, except for Systems 2a and 2b when the external nitrification system nitrified poorly (49% to 64%, Table 3.6). For both these systems, high ammonia concentrations (Table 3.6) entered the aerobic reactor of the BNR activated sludge part of the plant. In System 2a, this residual ammonia could not be completely nitrified in the aerobic reactor because the aerobic mass fraction was small (15%), resulting in a high TKN concentration in the effluent (19.8 mg/ℓ) and thus low N removal. In System 2b, the aerobic mass fraction was 30%, so some nitrification took place in the aerobic reactor, resulting in a lower TKN concentration in the effluent (9.2 mgN/ℓ) which increased the overall nitrification efficiency. However, the low nitrification efficiency in the external nitrification system resulted in a low nitrate load on the main anoxic reactor and hence low denitrification and low N removal. In general, the system can achieve an effluent total N (TN) less than 10 mgN/ℓ as demonstrated by Systems 1, 3b and 3c, but this requires virtually complete nitrification in the external nitrification system. The effluent nitrate concentrations were low and less than 9 mgN/ℓ for all systems.

3.12

Table 3.8: Influent TKN, and effluent TKN and Nitrate concentrations, as well as N removal efficiency for each system.

Parameter	System					
	1	2a	2b	3a	3b	3c
Influent TKN	70	72	66	69	79	78
Influent TKN/COD	0.112	0.111	0.095	0.092	0.109	0.107
Effluent TKN	5.1	19.8	9.2	5.1	5.0	4.8
Effluent NOx	3	4.6	7.8	8.9	4.8	2.5
Effluent total N	8.1	24.4	17	14.0	9.8	7.3
% N removal	88	66	74	80	88	91

[1] *Including nitrate dose.*

Fig 3.4: Influent TKN and effluent TKN, NOx and total nitrogen for the three investigations.

3.3.3 Biological excess P removal (BEPR)

3.3.3.1 Total phosphorus (TP) mass balance

To assess the BEPR behaviour, total phosphorus (TP) mass balances were conducted around each reactor, internal and final settling tanks and the external nitrification system from the TP concentrations measured in the influent and soluble (< 0.45μm) TP concentrations measured in the reactors, internal and final settling tanks, effluent and external nitrification system outflow for all sewage batches of the investigations. The TP mass balances were calculated by subtracting the outflow P mass from the inflow P mass, so a negative result indicates P release (-ve) and a positive result indicates P uptake (+ve). The calculated P release and P uptake masses were divided by the influent flow, to give the P release and P uptake in mgP/ℓ influent. The mean P uptake and P release for all sewage batches for each system are given in Table 3.9.

3.3.3.2 Phosphorus release

From Table 3.9 and Fig 3.5, it can be seen that, as expected, most of the P release occurred in the anaerobic reactor. A small amount of P release also occurred in the internal settling tank. This latter P release is probably due to leakage of some RBCOD out of the anaerobic reactor, stimulating P release in the sludge blanket that formed in the bottom of the internal settling tank due to the low underflow pumping rate to the main anoxic reactor. This was not unexpected as similar observations had been made in previous investigations into the DEPHANOX system, another kind of external nitrification BNR activated sludge system (Bortone et al., 1996; Sorm et al., 1996). This additional P release is beneficial because it augments the anaerobic P release. In the modified DEPHANOX system, the anaerobic reactor and internal setting tank were combined in a single reactor allowing sludge accumulation, P release and settlement simultaneously (Bortone et al., 1997).

Table 3.9: Phosphorus release (-ve) and uptake (+ve) in each reactor, internal settling tank and final settling tank as well as external nitrification system (mgP/ℓ influent), and % anoxic P uptake.

Reactor	P release (-ve) or P uptake (+ve) (mgP/ℓ influent)					
	System 1	System 2a	System 2b	System 3a	System 3b	System 3c
Anaerobic reactor	-25.7	-11	-21.4	-4.9	-11.0	-16.7
Internal settling tank	-4.5	-0.9	-2.3	-5.9	-4.8	-4.1
External nitrification system	-0.9	-1.1	-1.8	-4.3	-4.0	-5.1
Pre-anoxic reactor	0.2	-2.8	-2.7	0.8	1.2	0
Main anoxic reactor	22.2	13.5	15	22.5	15.1	22.3
Aerobic reactor	20.2	10.9	21.1	13.9	11.5	12.9
Final settling tank	0.1	-1.1	1.8	0.5	1.1	1.2
% anoxic P uptake	52	55	42	62	57	63

Fig 3.5: P release (rel., -ve) and P uptake (upt., +ve) for anaerobic (anaer.), internal settling tank (Int. settl.), external nitrification (EN), pre-anoxic (Preanx.), main anoxic (main anx.), aerobic and final settling tank (Final settl.) for the three investigations.

With regard to P release in the external nitrification system of Systems 3a, 3b and 3c, this was not expected and it is not beneficial to the system BEPR. This phenomenon was not observed in the other experimental systems to the same extent. This may be due to the external nitrification system itself, because in Systems 3a, 3b and 3c, the external nitrification system was a suspended media activated sludge system, not a fixed media stone column as used in Systems 1, 2a and 2b. This P release is not beneficial to the BEPR because (i) it is not likely to be associated with substrate uptake by PAOs and hence (ii) this is "extra" P that needs to be taken up subsequently. Sotemann *et al.* (2000) speculated on the cause of this P release as (a) PAOs that do not settle in the internal settling tank and undergo endogenous P release in the external nitrification system or (b) filterable but non-settleable P containing organics that are hydrolyzed in the external nitrification system, but both explanations do not seem very plausible.

3.3.3.3   Phosphorus uptake

From Table 3.9 and Fig 3.5, it can be seen that significant P uptake took place in the main anoxic reactor for all systems. The % anoxic P uptake varied from 42% to 63% of the total P uptake. From these results, together with the results from the previous investigations into DEPHANOX systems (Bortone *et al.*, 1996; Sorm *et al.*, 1996), it can be concluded that anoxic P uptake BEPR is an endemic characteristic of ENBNR activated sludge systems, and thus information on the factors that stimulate anoxic P uptake BEPR in general may be obtained from these investigations.

The factors that stimulate anoxic P uptake BEPR in BNRAS systems

One of the objectives in the first investigation (System 1, see Hu *et al.*, 2001) was to investigate and establish the conditions that stimulate anoxic P uptake BEPR. In the course of the investigation, initially both aerobic and anoxic P uptake were low, and most of the P uptake took place in the aerobic reactor. As the investigation continued, the total P uptake increased and gradually relatively more took place in the anoxic reactor and less in the aerobic reactor. Examining this data in more detail, for the first 6 sewage batches, the % anoxic P uptake remained approximately constant at about 25%. At that stage, ways to increase this anoxic P uptake were explored.

It was noted from earlier investigations on BEPR in conventional BNR activated sludge systems (Ekama and Wentzel, 1999b), that when the main anoxic reactor was underloaded with nitrate, the P uptake tended to be confined to the aerobic reactor, whereas if the nitrate load exceeded the denitrification potential so that nitrate was present in the anoxic reactor outflow, then significant anoxic P uptake was observed. Accordingly, this proposal was applied to the ENBNR activated sludge system to stimulate anoxic P uptake in the ENBNR activated sludge system: From sewage batch 6, nitrate was dosed into the main anoxic reactor and the amount dosed increased stepwise with succeeding sewage batches; the stepwise increase was to avoid nitrate overload to the main anoxic reactor and hence a high nitrate recycle to the anaerobic reactor. This nitrate dosing appeared to stimulate significant anoxic P uptake (see Hu *et al.*, 2001). These experimental results confirmed the observations from conventional BNR activated sludge systems (above). In the subsequent investigations into ENBNR activated sludge systems (Systems 2a, 2b, 3a, 3b and 3c), this observation was further substantiated.

The observations above suggest that with limited nitrate available, the OHOs tend to outcompete the PAOs for nitrate. The greater affinity of OHOs than PAOs for nitrate is reflected in the calculated specific denitrification rates by OHOs ($K_3'_{OHO}$) and PAOs ($K_2'_{PAO}$) (see below): It was found that the specific denitrification rates of the OHOs ($K_2'_{OHO}$) are significantly higher than those of the PAOs ($K_2'_{PAO}$). In terms of this competition, if the nitrate load into the main anoxic reactor is less than the denitrification potential of the OHOs, then the PAOs have limited opportunity for use of the limited nitrate. In contrast, if the nitrate load on the main anoxic reactor exceeds the denitrification potential of the OHOs, then the denitrifying PAOs (DPAOs) would have a greater opportunity to use the "excess" nitrate and so develop in the system. The development of the DPAOs in BNR activated sludge systems appears very slow, because in the experimental investigation anoxic P uptake appeared still to be increasing in System 1 after some 150 days of operation (see Hu *et al.*, 2001). This probably is a further contributing reason for the variable (and rather limited) occurrence of DPAOs in conventional BNR activated sludge systems.

In the ENBNR activated sludge system, there appear to be three further conditions that appear to favour DPAOs, viz.

- Small aerobic mass fraction;

- Sequence of reactors;

- The low frequency of alternation between anoxic and aerobic states.

The first condition limits development of aerobic PAOs in the system and ensures PAOs grow anoxically.

The sequence of reactors in the ENBNR activated sludge system is such that nitrification precedes the main anoxic reactor, and hence all the nitrate generated is available for denitrification; this ensures a substantial nitrate load to the main anoxic reactor which stimulates anoxic P uptake. Therefore, for Systems 3a, 3b and 3c, even though no additional nitrate was dosed into the main anoxic reactor, the nitrate load from the external nitrification system to the main anoxic reactor was large, and hence significant anoxic P uptake was still observed. In contrast, in conventional BNR activated sludge systems nitrification usually follows the anoxic reactor so that the system requires a mixed liquor recycle to bring nitrate generated in the aerobic zone to the anoxic zone for denitrification. Thus, in these systems, the nitrate generated in the system that can be denitrified in the anoxic reactor is limited by the mixed liquor recycle (a recycle) from aerobic to anoxic reactor, and it is not possible to bring all nitrate generated in the system into the anoxic zone for denitrification. From the observations above, this will tend to disadvantage the DPAOs, unless the nitrate load via the a- recycle is substantial, requiring high a-recycle ratios, but such ratios tend also to disadvantage DPAOs (see below).

It seems that frequent alternation of the sludge between the anoxic and aerobic zones adversely affects the DPAOs. This was apparent from Systems 3b and 3c where removing the a -recycle (while ensuring a sufficient high nitrate load on the main anoxic reactor) reduced the sludge alternation between anoxic and aerobic conditions, and this improved the anoxic P uptake and denitrification (Tables 3.9 and 3.7 respectively). The sludge alternation frequency between anoxic and aerobic conditions in ENBNR activated sludge systems usually is much lower than that in a conventional BNR activated sludge system (like the UCT system) for the same TKN/COD ratio and nitrification efficiency, due to the requirement of the a-recycle in the latter system.

The combination of the above may explain the reason why anoxic P uptake BEPR was consistently observed in the ENBNR activated sludge system, but only occasionally in conventional BNR activated sludge systems.

3.3.3.4   Phosphorus removal performance

The biological P removals achieved in the five ENBNRAS systems are given in Table 3.10. The system conditions that influence this, viz. influent COD and RBCOD concentration, anaerobic mass fraction, sludge age and nitrate entering the anaerobic reactor, are also listed in Table 3.10. Compared with the BEPR obtained with predominantly aerobic P uptake, the BEPR achieved in the ENBNR activated sludge systems with predominantly anoxic P uptake is significantly lower, at about 2/3rds that expected for the influent RBCOD, sludge age and anaerobic mass fraction from the BEPR model of Wentzel *et al.* (1990), which is based on aerobic P uptake (see below).

Table 3.10: P removal, influent COD, RBCOD and system operating parameters for the three investigations.

Parameter	System					
	1	2a	2b	3a	3b	3c
P removal (mgP/ℓ)	10.6	8.5	11.8	9.7	9.1	10.5
Influent COD (mgCOD/ℓ)	717	683	699	741	724	730
Influent RBCOD (mgCOD/ℓ)	131	109	102	120	146	149
Anaerobic mass fraction (%)	23.8	25	25	25	25	25
Sludge age (d)	10	10/8	8	10	10	10
Nitrate in Pre-anoxic (mgN/ℓ)	0.7	0.5	0.4	4.7	3.4	0.7

Anoxic P uptake has also been observed in conventional BNR activated sludge systems (Ekama and Wentzel 1999b). In the UCT and modified UCT systems of Musvoto *et al.* (1992), Kaschula *et al.* (1993), Pilson *et al.* (1995) and Mellin *et al.* (1998), significant anoxic P uptake (>40%) was observed, which was confirmed with anoxic batch tests on sludge harvested from these systems. The observed features of aerobic P uptake BEPR and anoxic/aerobic P uptake BEPR in conventional BNR activated sludge systems and the anoxic/aerobic P uptake BEPR in the ENBNR activated sludge systems are listed in Table 3.11. While anoxic/aerobic P uptake between the conventional and external nitrification BNR activated sludge systems appears similar, there seem to be some major differences in P removal performance between aerobic P uptake BEPR and anoxic/aerobic uptake BEPR. In all the investigations which were all long term (>500 days), where anoxic/aerobic P uptake BEPR was observed not only was the excess P removal lower compared with that expected from the model of Wentzel *et al.* (1990), but also the P release to removal ratio was decreased (see Table 3.11). In conventional BNR activated sludge systems:

- With only aerobic P uptake, the P release/P removal ratio, P removal/Influent RBCOD ratio and the P removal/Influent COD ratio are around 3.0, 0.11 and 0.021 respectively (Wentzel *et al.* 1985, 1989; Clayton *et al.* 1991, and more recently Sneyders *et al.*, 1997 - see Table 3.11), and are in conformity with the steady state BEPR model of Wentzel *et al.* (1990) and the dynamic state BEPR model of Wentzel *et al.* (1992).

- With anoxic/aerobic P uptake these ratios decrease to 1.5-2.0, 0.06-0.08 and 0.012-0.015 respectively and the BEPR is depressed to around ²⁄₃rds of that with only aerobic P uptake (see Table 3.11).

Comparing the anoxic/aerobic BEPR performance results obtained in ENBNR activated sludge systems with those observed in conventional BNR activated sludge systems, similarly low values are obtained for the P release/P removal, P removal/influent RBCOD and the P removal/influent COD ratios, i.e. 1.1 - 2.9, 0.06 - 0.12 and 0.012 - 0.017. It seems that, given the appropriate conditions, different species of PAOs which accomplish anoxic P uptake find a niche in the system, but which have a significantly lower BEPR performance and use their internally stored PHA (obtained from anaerobic uptake of VFA which are produced by the OHOs via fermentation of the influent RBCOD) less "efficiently" compared with the aerobic P uptake PAOs. This aspect is explained further in Chapter 7 and in detail by Hu *et al.* (2001).

Table 3.11: Observed BEPR and calculated P contents of PAOs ($f_{XBG,P}$) with the steady state model (Wentzel et al., 1990) in conventional BNR (Ekama and Wentzel, 1999b) and ENBNR activated sludge systems.

Systems	Anoxic P uptake (%)	P removal (mgP/ℓ)	Prel/Prem ratio	Prem/infl. RBCOD ratio	Prem/Total infl COD ratio	$f_{XBG,P}$ (mgP/mg PAOAVSS)
Conventional BNR activated sludge systems (UCT and MUCT systems)						
Clayton et al., 1991	5	21	3.3	0.105	0.021	0.388
Sneyders et al., 1998 (C)	0	13.1	2.42	0.116	0.023	0.471
Sneyders et al., 1998 (E)	0	16.8	2.74	0.116	0.024	0.471
Musvoto et al., 1992 (1)	27	12.2	2.44	0.063	0.012	0.144
Musvoto et al., 1992 (2)	47	11.3	2.7	0.060	0.012	0.113
Pilson et al., 1995 (1)	47	12	1.34	0.069	0.012	0.136
Pilson et al., 1995 (2)	16	10.9	1.24	0.063	0.011	0.098
Mellin et al., 1998	29	11.4	1.8	0.082	0.016	0.260
ENBNR activated sludge systems						
System 1	52	10.6	2.9	0.081	0.015	0.235
System 2a	55	8.5	1.4	0.080	0.012	0.167
System 2b	42	11.8	2	0.116	0.017	0.250
System 3a	62	9.7	1.1	0.080	0.013	0.240
System 3b	57	9.1	1.7	0.062	0.013	0.170
System 3c	63	10.5	2.0	0.070	0.014	0.188

3.3.3.5 Comparison of the measured and calculated P removal

The BEPR performance in the BNR activated sludge system can be assessed by comparing the observed P removal with that theoretically calculated from the steady state BEPR model of Wentzel et al. (1990). This model requires as input all the system design parameters (see Table 3.1) and influent wastewater characteristics including the influent RBCOD concentration. The procedure for calculating the theoretical BEPR is summarized by Ekama and Wentzel (1999b) and in detail by Hu et al. (2001). Following this procedure[1] the OHO and PAO masses and the calculated P contents of PAOs ($f_{XBG,P}$) in conventional BNRAS systems and ENBNRAS systems were obtained and are listed in Table 3.11. It can be seen that in the systems with predominantly (>95%) aerobic P uptake BEPR behaviour, the $f_{XBG,P}$ values are the same as that in the steady state model of Wentzel et al. (1990) (0.38 mgP/mgPAOAVSS). Comparing the $f_{XBG,P}$ values of

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[1] In the calculation procedure, the influent SBCOD was decreased to take account of the unaccounted for COD in the COD mass balances for all the system and for the ENBNRAS system the SBCOD was reduced further to take account of the COD lost in the external nitrification system, but the influent RBCOD was kept at the measured values. Thus was necessary to reduce the sludge variation in the OHO and PAO active fraction ($f_{av,OHO}$ and $f_{av,PAO}$) of the VSS and specific OHO denitrification rates obtained by Ekama and Wentzel (1999b).

systems with aerobic and anoxic/aerobic P uptake BEPR, it seems that the latter value is only about one quarter to two thirds of the 0.38 mgP/mgPAOAVSS aerobic value.

3.3.4   Calculation of specific denitrification rate

3.3.4.1   Relative contribution of OHOs and PAOs to denitrification

From Table 3.11, in the ENBNR activated sludge systems significant P uptake took place in the main anoxic reactors. This implicated PAO participation in the denitrification process, i.e. PAOs contributed to the system denitrification. It appears that anoxic P uptake and associated denitrification took place consistently in all ENBNR activated sludge systems investigated.

To calculate the denitrification potential in the ENBNR activated sludge systems, the contribution of OHOs and PAOs to denitrification needs to be known. In the main anoxic zone, if PAOs are involved in denitrification, PAOs and OHOs will compete for nitrate by using different substrates. PAOs use internally stored PHA, and OHOs use slowly biodegradable COD (SBCOD). This difference in substrate source makes it possible to estimate the contribution of OHOs and PAOs to the denitrification. The procedure detailed by Hu *et al.* (2001) was followed for calculating the specific denitrification rates of the OHOs and PAOs. The specific denitrification rates, denoted $K_2{'}_{OHO}$ [mg NO$_3$-N/(mgOHOAVSS.d)] and $K_2{'}_{PAO}$ [mg NO$_3$-N/(mgPAOVSS.d)] respectively, are listed in Table 3.12.

The results were obtained by averaging those sewage batches only in which RBCOD was measured and nitrate concentrations in the main anoxic reactor was larger than 1 mg N/ℓ. Furthermore, in fractioning the VSS mass into OHO and PAO active masses and the other three inert masses, the unbiodegradable particular COD fraction ($f_{S,up}$) needs to be determined. All the investigations received raw (unsettled) municipal wastewater from the same source, i.e. Mitchell's Plain Wastewater Treatment Plant (Cape Town, South Africa). Thus, since $f_{S,up}$ is a wastewater characteristic, there is a reasonable expectation that $f_{S,up}$ should be constant. This was the case for 4 long term parallel investigations on aerobic and anoxic/aerobic activated sludge systems receiving the same wastewater, $f_{S,up}$ = 0.10 to 0.16 (Warburton et al., 1991; Mbewe et al., 1995; Ubisi et al., 1997; Mellin et al., 1998). Accordingly, a constant $f_{S,up}$ of 0.12 was accepted for the Mitchell's Plain wastewater. Also, while the influent RBCOD concentrations were kept at the measured values, the influent SBCOD was reduced to take account of (i) the COD loss in the external nitrification system and (ii) the COD unaccounted for in the COD balances.

Table 3.12: Calculated specific denitrification rates for PAOs and OHOs and the contribution of PAOs and OHOs to denitrification in the main anoxic reactor for each system.

System	System					
	1	2a	2b	3a	3b	3c
Anoxic P uptake (%)	53.8	52.7	37.5	33.8	57.2	66.6
Measured total denitrification (mgN/ℓ inf)	53.2	32.2	26.8	18.8	19.0	31.2
% denitrification by PAOs	17.5	25.4	23.6	9.6	24.6	22.0
% denitrification by OHOs	82.5	74.6	76.4	91.4	75.3	78.0
K_{2PAO} (mg NO_3-N/(mg PAOAVSS.d))	0.042	0.048	0.059	0.029	0.038	0.044
K_{2OHO} (mg NO_3-N/(mg OHOAVSS.d))	0.184	0.166	0.185	0.114	0.098	0.157

From Table 3.12 it can be seen that the contribution of PAOs to denitrification varied from 9.6% to 25.4% even when the anoxic P uptake was 66%. This indicates that the contribution of PAOs to denitrification was low, and less than 26% for all systems investigated. The specific denitrification rates of PAOs ranged from 0.029 to 0.059 mg NO_3-N/(mg PAOAVSS.d), these values are low compared to those of OHOs varying from 0.098 to 0.185 mg NO_3-N/(mg OHOAVSS.d). This may explain the reason why sufficient nitrate is required to stimulate anoxic P uptake BEPR in BNR activated sludge systems. If nitrate is not sufficient, OHOs will outcompete the PAOs for limited nitrate. The low contribution of PAOs to the denitrification raises a question about explaining denitrifying PAOs (DPAOs), because it also causes the reduction of P removal, i.e. denitrification by PAOs was obtained by sacrificing about 1/3rd of the P removal. However, considering that the concentration of PAOs is only around 1/5th of that of the OHOs, and the specific rate of this small PAO mass is only 1/3rd of that of the OHOs, this denitrification by PAOs does not appear to be very significant, and most likely not worth sacrificing 1/3rd of P removal for.

3.3.5 Sludge settleability

The mean DSVIs observed from all sewage batches for each system are listed in Table 3.13. It is evident that the ENBNR activated sludge systems consistently produced a good settling sludge. The aerobic mass fractions for each system are also listed in Table 3.13. Casey *et al.* (1994) observed that maximum filamentous organism proliferation occurred with an aerobic mass fraction between 30 and 35% and established an association between the DSVI and the aerobic mass fraction from artificial wastewater experiments; this is shown in Chapter 2, Fig 2.1.

Table 3.13: Observed DSVIs and aerobic mass fractions in ENBNR activated sludge systems.

System	System 1	System 2a	System 2b	System 3a	System 3b	System 3c
DSVI (mℓ/g)	69	73.3	97	119	109	96
Aerobic mass fraction (%)	19	15	30	32.5	20	20

To investigate the applicability of the association between the DSVI and the aerobic mass fraction in practice, Stewart Scott Consulting Engineers (Casey, 1998) conducted a survey. The survey included seven full-scale conventional BNRAS plants. These plants were chosen because they were the only plants for which historical data of the sludge settleability performance were available. The historical sludge settleability data were converted to DSVI units where applicable, and an overall average of the DSVI value was calculated for each of the seven full-scale plants. The results obtained are shown in Table 2.1 (Chapter 2). The aerobic mass fraction data for the seven full-scale treatment plants were superimposed on the artificial wastewater laboratory system DSVI/aerobic mass fraction diagram from Casey et al. (1994) and plotted in Fig 2.2 (Chapter 2). For the full-scale plants, it appears that an association also exists between the two parameters, with improving sludge settleability for an aerobic mass fraction increase from 40% upwards. However, Casey et al. could not comment on the effect of aerobic mass fractions on DSVI for aerobic mass fractions below 40% as conventional BNR activated sludge systems are not designed and operated at less than 40% aerobic mass fraction because of the nitrification requirement.

The information from the ENBNR activated sludge systems provides the possibility to assess the association between DSVI and aerobic mass fractions in the less than 40% aerobic mass fraction range. The DSVI results from the ENBNR activated sludge systems were superimposed on the DSVI/aerobic mass fraction diagram from Casey et al. (1994) and plotted in Fig 3.6. It can be seen that the ENBNRAS experiments confirm the artificial wastewater association between the two parameters of Casey et al. (1994), with improving sludge settleability for aerobic mass fractions less than 30%.

Fig 3.6:   Relationship between DSVI and aerobic mass fraction from artificial wastewater, full-scale BNR activated sludge plants and lab-scale ENBNR activated sludge systems.

3.4   CLOSURE

From the three laboratory-scale experimental investigations into the proposed ENBNR activated sludge systems, it is evident that BNR activated sludge system intensification by separating the process of nitrification from the main BNR activated sludge system and effecting nitrification externally is possible in practice.

3.4.1   System performance

From the investigations:
* COD removals in excess of 90% can be expected.
* Considerable reduction in oxygen demand will be obtained, about 60% less than in an equivalent conventional BNR activated sludge system.
* The proportion of inflow to the internal settling tank that can be directed to the external nitrification system is about 85% due to the constraint of sludge underflow to the anoxic reactor, and a nitrification efficiency in the external nitrification system of about 85 - 90% can be expected, but this requires confirmation at full-scale.
* Total N removal of 80 - 90% is possible, with an effluent TKN of about 5 mgN/ℓ, effluent NO_3^- of about 2.5 - 5 mgN/ℓ and total nitrogen of about 7 - 10 mgN/ℓ. However, this will be influenced by the nitrification efficiency of the external nitrification system, and the influent wastewater characteristics.
* BEPR of about 9 - 11 mgP/ℓ, but again this will depend on the wastewater characteristics. Anoxic/aerobic P uptake BEPR was consistently observed at 52 - 63% of total P uptake. It seems that such BEPR is inherent to the ENBNR activated sludge system, because the system configuration provides the conditions which favour the development of denitrifying PAOs. These conditions are:
  (1)   large anoxic mass fraction at the expense of the aerobic mass fraction, i.e. small aerobic mass fraction;
  (2)   sufficient nitrate load into anoxic zone.
  (3)   no mixed liquor (a) recycle.
  The nitrate load on the anoxic reactor appears to be the most important.
* The contribution of PAOs to denitrification is low, and less than 26% for all systems investigated.
* A good settling sludge can be expected, with a DSVI of about 100mℓ/gTSS.

3.4.2   Design and operation

From the investigation, a number of design and operation parameters can be identified:
* The successful implementation of the ENBNR activated sludge system hinges around successful operation of the external nitrification system. If the efficiency of this unit process is reduced, denitrification in the anoxic reactor will be reduced (due to the lower nitrate load). Furthermore, FSA not nitrified in the external nitrification system will be nitrified in the aerobic reactor of the BNR activated sludge system to a greater or lesser extent (depending on the aerobic mass fraction and nitrifier seeding from the external nitrification system). This will cause deterioration in the effluent nitrate quality, and the nitrate in the underflow recycle may exceed the denitrification potential of the underflow anoxic reactor and enter the anaerobic reactor and hence reduce BEPR.

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- Anaerobic mass fraction of about 25%; this can be reduced if larger anoxic mass fractions are required, with a resultant reduction in BEPR (Wentzel *et al.*, 1990).
- Minimum aerobic mass fraction of 20%. Smaller aerobic mass fractions appear to stimulate pin point floc formation, which cause deterioration in sludge retention and effluent quality. Furthermore, such aerobic mass fractions will stimulate some aerobic P uptake BEPR. Inclusion of anoxic P uptake PAOs in, and exclusion of aerobic P uptake PAOs from, the biocenosis of the BNR activated sludge system mixed liquor are not essential for achieving BNR in the external nitrification scheme. In fact, inclusion and maximization of aerobic P uptake would appear desirable to maximize BEPR (aerobic P uptake BEPR is more efficient than anoxic P uptake BEPR). However, conditions that promote aerobic P uptake BEPR are also conducive to nitrifier growth. Although exclusion of nitrifiers from the mixed liquor of the BNR activated sludge system was originally considered essential, this is not necessary as long as virtual complete nitrification in the fixed media system is obtained, to limit nitrification in the main aerobic reactor. In fact, complete exclusion of nitrifiers may not prove possible. Although conditions in the BNR activated sludge system may not be conducive to nitrifier growth, they are likely to be continually seeded into the system from the external nitrification system. Accepting nitrification in the BNR activated sludge system, with complete nitrification in the external nitrification system, the nitrification in the BNR activated sludge system will be limited by the ammonia bypassing the internal settling tank in its underflow. This nitrate is prevented from entering the anaerobic reactor with the underflow recycle by denitrification in the underflow anoxic reactor. However, should complete nitrification not be obtained in the external nitrification system, then the ammonia from this system will be nitrified in the aerobic reactor of the BNR activated sludge system. If, as a result the aerobic reactor nitrate concentration is too high, the pre-anoxic reactor will become overloaded with nitrate resulting in nitrate discharge to the anaerobic reactor and reduced BEPR. However, if nitrification were not included in the BNR activated sludge system, the ammonia would leave via the effluent, an even less desirable situation.
- The requirements above for an anaerobic mass fraction of 25% and an aerobic mass fraction of 20% provide 55% of the total mass fraction for the anoxic reactors. This has to be divided between the primary and underflow anoxic reactors. An underflow anoxic mass fraction of about 10% appears adequate, leaving 45% for the main anoxic reactor.
- The proportion of the internal settling tank inflow that can be directed to the external nitrification system is about 85%, due to the constraint of sludge underflow to the anoxic reactor (see above).
- An a-recycle from the aerobic to the anoxic reactor should not be included, as this appears detrimental to BEPR and denitrification.

CHAPTER 4

EXTERNAL NITRIFICATION BIOLOGICAL NUTRIENT REMOVAL (ENBNR) ACTIVATED SLUDGE SYSTEM PERFORMANCE AT SLUDGE AGES SHORTER THAN 10 DAYS

4.1 INTRODUCTION

In Chapter 3 extensive experimental data on ENBNR activated sludge systems have been presented. In these systems, sludge ages were 10 (Systems 1 and 3a and 3b) and 8 (Systems 2a and 2b) days (see Table 3.1, Chapter 3). However, as noted in Chapter 3, *Psycoda* fly and larvae infestation in the stone column external nitrification system for the systems at 8 days sludge age (Systems 2a and 2b) caused deterioration in external nitrification performance. This complicated assessment and comparison of system performance at different sludge ages. Initially, evaluation of the laboratory scale ENBNR activated sludge system performance at sludge ages lower than 10 days was not specifically part of the scope of the investigation - data on BNR activated sludge systems at short sludges ages are available from previous investigations in the UCT laboratory (Burke *et al.*, 1984, Wentzel *et al.*, 1990). However, towards the end of the practical laboratory investigation on System 3b (Chapter 3), it was decided to increase the influent sewage flow from 20 ℓ/d to 30 ℓ/d in order to observe the system response to this 50% increase in load. Shortly after the influent flow had been increased to 30 ℓ/d, the system internal settling tanks began to fail hydraulically. Thus, instead of allowing the system to fail completely as a result of this hydraulic failure, it was decided to reduce the influent flow to 25 ℓ/d and instead of implementing a gradual increase in feed, a gradual reduction in sludge age should rather be implemented and the effect of the reduced sludge age on system performance evaluated. This Chapter summarises this investigation; details are given by Sötemann *et al.* (2000).

4.2 SYSTEM OPERATION

At the beginning of sewage batch 31 for System 3b (Chapter 3), the influent was increased from 20 to 30 ℓ/d. Two days later, it was reduced to 25 ℓ/d and the sludge age was decreased from 10 to 8 days (Configuration 4, see Table 4.1). The system was run at the 8 day sludge age for sewage batches 31, 32 and 33 (49 days, 6 sludge ages) after which the sludge age was reduced further to 5 days (Configuration 5, see Table 4.1) at the beginning of sewage batch 34. The system was run at a 5 day sludge age for a further 13 days (3 sludge ages). For the 5 day sludge age 4 ℓ/d were wasted. The 4 ℓ/d could no longer be taken from the 4 ℓ aerobic reactor in one batch, so a small peristaltic pump was installed and calibrated to waste 4 ℓ of mixed liquor from the aerobic reactor over a 24 hour period. System operation is described in detail by Sötemann *et al.* (2000).

Table 4.1: ENBNR activated sludge system design and operating parameters.

System Parameter	Config. 3b	Config. 4	Config. 5
Days	285 to 421	422 to 470	471 to 483
Sewage Batches	21 to 30	31 to 33	34
No. of Days	137	49	13
Dates: From	09/12/99	18/04/00	08/06/00
To	17/04/00	07/06/00	18/06/00
Operating Parameters			
Influent Flow (l/d)	20	**25**	25
Sludge Age (d)	10	**8**	**5**
Waste (l/d)	2	2.5	4
Temperature (oC)	20	20	20
pH - Anaerobic Reactor	7.5 - 7.8	7.1 - 7.9	7.7 - 8.0
pH - Main Aerobic Reactor	7.7 - 8.2	7.4 - 8.4	7.7 - 8.0
D.O. Main Aerobic Reac. (mgO/l)	2 - 5	2 - 5	2 - 5
Reactor Vol. / Mass Frac.			
Total System Volume (l)	20	20	20
Pre-Anoxic Reactor (l)	1*	1*	1*
Anaerobic Reactor (l)	5	5	5
Main Anoxic Reactor (l)	9	9	9
Main Aerobic Reactor (l)	4	4	4
Total Aerobic Mass Fraction	0.20	0.20	0.20
Anoxic Mass Fraction	0.55	0.55	0.55
Anaerobic Mass Fraction	0.25	0.25	0.25
Total Unaerated Mass Fraction	0.80	0.80	0.80
Recycles			
s - Recycle (w.r.t influent flow)	1 : 1	1 : 1	1 : 1
a - Recycle (w.r.t influent flow)	**0 : 1**	0 : 1	0 : 1
Sludge Bypass (w.r.t influent flow)	0.32 : 1	0.34 : 1	0.37 : 1
External Nitrifier Parameters			
Sludge Age (d)	Very long		
Waste (l/d)	As required to maintain sludge level in internal SST		
D.O (mgO/l)	>5		
Reactor Volume (l)	3		

*Actual volume, with sludge at double concentration. Effective volume at system sludge concentration = 2 litres.

Denotes changes made to previous configuration.

4.3 SYSTEM PERFORMANCE

Given the comparatively short time that the system was run at these low sludge ages it would be of little practical value to give an as detailed evaluation for the 8 and 5 day sludge age system configurations as was done in Chapter 3 for the 10 and 8 day sludge age configurations. For this reason only a brief comparison of the main nutrient removal performances (COD, N and P) will be given in this Chapter; detailed results are given by Sötemann *et al.* (2000). The sewage batch averages for all measured parameters for sewage batches 31 to 34 (Configurations 4 and 5) are given in Tables 4.2a, b and c.

TABLE 4.2a: Sewage batch averages of measured COD, TKN and FSA parameters for sewage batches 31 to 34 (Configurations 4 and 5).

				Sewage Batch				Overall
				31	32	33	34	
COD	Influent	UF	mgCOD/ℓ	731.6	779.8	-	709.2	740.2
	Influent	FF	mgCOD/ℓ	158.3	191.3	-	186.2	178.6
	Int. Set. A	UF	mgCOD/ℓ	153.6	145.6	-	161.5	153.5
	Int. Set. B	UF	mgCOD/ℓ	77.6	92.2	-	91.4	87.0
	Aerobic M.L.	UF	mgCOD/ℓ	2392	2564	-	1735	2231
	Effluent	UF	mgCOD/ℓ	61.5	62.7	-	83.9	69.4
	Effluent	F	mgCOD/ℓ	36.1	41.0	-	31.6	36.2
TKN	Influent	UF	mgN/ℓ	64.3	79.9	-	87.5	77.3
	Aerobic M.L.	UF	mgN/ℓ	150.4	149.5	-	123.0	141.0
	Effluent	UF	mgN/ℓ	4.6	5.0	-	6.9	5.5
	Effluent	F	mgN/ℓ	3.7	4.1	-	4.1	4.0
FSA	Influent	UF	mgN/ℓ	50.0	64.0	-	71.8	62.0
	Int. Set. A	UF	mgN/ℓ	23.4	28.7	-	33.7	28.6
	Int. Set. B	UF	mgN/ℓ	2.9	3.4	-	2.7	3.0
	Effluent	UF	mgN/ℓ	3.1	3.5	-	2.8	3.1
Influent TKN/COD Ratio			-	0.088	0.103	-	0.120	0.104

(UF = Unfiltered, FF = Floc Filteres, F = 0.45µm membrane filtered)

TABLE 4.2b: Sewage batch averages of measured suspended solids, OUR, DSVI and pH for sewage batches 31 to 34 (Configurations 4 and 5).

			Sewage Batch				Overall
			31	32	33	34	
TSS	Pre Anoxic	mgTSS/ℓ	3782	4013	-	2760	3518
	Aerobic	mgTSS/ℓ	1957	2109	-	1410	1825
VSS	Pre Anoxic	mgVSS/ℓ	3095	3264	-	2254	2871
	Aerobic	mgVSS/ℓ	1626	1738	-	1177	1513
ISS[1]	Pre Anoxic	mgVSS/ℓ	688	749	-	506	648
	Aerobic	mgVSS/ℓ	332	371	-	233	312
COD/VSS Ratio[2] (Aerobic Reactor)			1.45	1.45	-	1.44	1.45
TKN/VSS Ratio[2] (Aerobic Reactor)			0.09	0.09	-	0.11	0.10
OUR	Aerobic	mgO/ℓ.h	19.9	25.5	-	17.7	21.1
DSVI		mℓ/g	92.6	86.9	-	92.9	90.8
pH	Anaerobic	-	7.50	7.43	-	7.89	7.61
	Aerobic	-	7.97	7.85	-	7.85	7.89

[1] ISS calculated from TSS - VSS.

[2] Calculated from unfiltered aerobic reactor COD and TKN concentrations divided by the VSS.

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4.5

TABLE 4.2c: Sewage batch averages for measured nitrite, nitrate, and P concentrations for sewage batches 31 to 34 (Configurations 4 and 5).

				Sewage Batch				Overall
				31	32	33	34	
NO₂	Pre Anoxic	F	mgN/ℓ	0.0	0.0	-	0.8	0.3
	Anaerobic	F	mgN/ℓ	0.0	0.1	-	0.0	0.0
	Int. Set. A	F	mgN/ℓ	0.0	0.1	-	0.0	0.0
	Int. Set. B	F	mgN/ℓ	0.0	0.1	-	0.0	0.0
	Anoxic	F	mgN/ℓ	0.0	0.7	-	0.3	0.4
	Aerobic	F	mgN/ℓ	0.0	0.3	-	1.2	0.5
	Effluent	F	mgN/ℓ	0.0	0.2	-	0.8	0.3
NO₃	Pre Anoxic	F	mgN/ℓ	0.0	0.1	-	10.9	3.7
	Anaerobic	F	mgN/ℓ	0.0	0.1	-	0.2	0.1
	Int. Set. A	F	mgN/ℓ	0.1	0.2	-	0.2	0.2
	Int. Set. B	F	mgN/ℓ	17.6	23.7	-	34.9	25.4
	Anoxic	F	mgN/ℓ	0.1	1.2	-	15.0	5.4
	Aerobic	F	mgN/ℓ	0.1	3.3	-	15.9	6.4
	Effluent	F	mgN/ℓ	0.6	2.9	-	17.0	6.8
P	Influent	F	mgP/ℓ	25.1	26.3	-	26.0	25.8
	Pre Anoxic	F	mgP/ℓ	13.2	14.1	-	15.3	14.2
	Anaerobic	F	mgP/ℓ	28.1	32.0	-	23.8	28.0
	Int. Set. A	F	mgP/ℓ	30.2	33.8	-	25.5	29.8
	Int. Set. B	F	mgP/ℓ	33.0	36.5	-	28.0	32.5
	Anoxic	F	mgP/ℓ	21.5	24.1	-	22.2	22.6
	Aerobic	F	mgP/ℓ	10.8	14.5	-	18.3	14.5
	Effluent	UF	mgP/ℓ	10.4	14.1	-	18.9	14.5
	Effluent	F	mgP/ℓ	9.6	13.8	-	17.4	13.6

(UF = Unfiltered, FF = Floc Filteres, F = 0.45μm membrane filtered)

4.3.1   Carbonaceous Material Removal

Table 4.3 shows the COD mass balances (including their components) and percentage COD removals for sewage batches 31 to 34. Table 4.4 gives a comparison of the most important COD parameters for the 10 (System 3b, Table 3.1, Chapter 3), 8 (System 4) and 5 (System 5) day sludge age configurations: these are summarised in Fig 4.1.

TABLE 4.3:   COD mass balances for sewage batches 31 to 34.

Average of Batch	Influent COD mgCOD/d	MOC mgO/d	Denitrification Recovery mgCOD/d	COD used Ext. Nit. mgCOD/d	COD in Waste mgCOD/d	COD in Effluent mgCOD/d	COD out mgCOD/d	% Recovery
31	18290	1795	2174	3322	5981	1383	14655	80.4
32	19496	2054	2962	2284	6412	1411	15123	78.0
33								
Config 4	18893	1924	2568	2803	6197	1397	14969	79.2
34	17729	1336	3224	2090	8841	1762	16253	91.8
Config 5	17729	1336	3224	2090	8841	1762	16253	91.8

TABLE 4.4:   Comparison of average COD parameters for 10, 8 and 5 day sludge age system configurations.

	ENBNRAS System Configurations		
	1, 2 and 3	4	5
	10 Day Sludge Age	8 Day Sludge Age	5 Day Sludge Age
% COD Reduction	94%	93%	90%
OUR	19.9 mgO/l.h	22.7 mgO/l.h	17.7 mgO/l.h
COD Balance	79.8%	79.2%	91.8%
% COD to Oxygen	13.8	10.2	7.5
% COD to Denitrification	12.7	13.6	18.2
% COD to EN System	19.7	14.8	15.8
% COD in Waste	26.7	32.8	39.2
% COD in Effluent	8.2	7.4	9.9
% COD Unaccounted	20.8	21.2	8.3

Fig 4.1:   COD mass balance components for Systems 3b (Chapter 3), 4 and 5.

The overall average COD mass balance for the 10, 8 and 5 day sludge age system configurations are 79.8%, 79.2% and 91.8% respectively (see Table 4.4 and Fig 4.1). The overall COD mass balances for the 10 and 8 day sludge age configurations are virtually the same, but that for the 5 day sludge age configuration is substantially higher. This is probably because the VSS concentration had not yet reached a steady state value for the 5 day sludge age. The overall average COD mass balance components for the various sludge age configurations do not show any unexpected changes, except that for the 5 day sludge age configuration the % COD in the wasted VSS is probably higher than would be at steady state. The variations in %COD to oxygen and % COD to denitrification can be attributed to variations in the influent sewage characteristics rather than to the reduced sludge ages. A higher influent COD will bring about a slight increase in the oxygen utilisation rate (OUR) and a higher influent TKN/COD ratio will cause a shift from the COD passed to oxygen to the COD utilised for denitrification. This can be clearly noted for the 10 and 5 day sludge age configurations: For the 10 day sludge age configuration which received an average of 736 mgCOD/ℓ influent with a TKN/COD ratio of 0.106 mgN/mgCOD, 13.8% of the COD was passed to oxygen and 12.7% of the COD was utilised for denitrification. For the 5 day sludge age configuration the influent COD was 709 mgCOD/ℓ with a TKN/COD ratio of 0.120 mgN/mgCOD. The combination of the lower COD in the influent and the substantially higher TKN/COD ratio resulted in 10.2% of the COD being passed to oxygen and 18.2% of the COD being utilised for denitrification. There is no reason for the percentage COD 'lost' to the EN system to change with a lowering of the sludge age of the system, and the results for the three sludge ages are similar. The small variations can be attributed to the different number of sewage batches fed rather than to the change in sludge age. The percentage COD in the waste is the only parameter that is expected to change with the sludge age. A shorter sludge age results in more mixed liquor being wasted and hence the percentage COD in the waste will be proportionally more. This can clearly be seen from Table 4.4: The 10 day sludge age configuration has an overall average of 26.7% of the influent COD in the waste flow, while the 8 day sludge age configuration has 32.8% and the 5 day sludge age configuration 39.2% (which is probably somewhat high as noted above) - an increase for each respective reduction in sludge age. The percentage COD in the unfiltered effluent shows a similar trend to the COD in the waste sludge, but this cannot be as a result of the lower sludge age. The influent flow for the 8 and 5 day sludge age configurations was increased from 20 to 25 ℓ/d and this put greater strain on the final settler causing a greater fraction of the dispersed suspended solids to spill over with the effluent. The filtered effluent COD concentration remained essentially unchanged at 36.2 mgCOD/ℓ.

The overall COD reduction for the three sludge age configurations are all within 4% of each other (94% for the 10 day sludge age configuration, 93% for the 8 day and 90% for the 5 day sludge age configuration), indicating that the COD removal of the ENBNR activated sludge system in not affected to any great extent by a reduction in sludge age.

4.3.2 Nitrogenous material removal

Table 4.5a lists the results for the nitrite and nitrate mass balances over each reactor and settler for sewage batches 31 to 34. A negative value indicates nitrification and a positive value denitrification. Table 4.5b and Fig 4.2 give the results for the total N mass balances and all of its components for sewage batches 31 to 34. Table 4.6 lists the denitrification potentials for the pre- and main anoxic reactors for those batches where the nitrate concentration exiting the respective anoxic reactor was >1 mgN/ℓ. Table 4.7 shows a comparison of the main N parameters for the 10, 8 and 5 day sludge age configurations.

TABLE 4.5a: Nitrite and nitrate mass balances across each reactor and settling tank for sewage batches 31 to 34.

Average of Batch	NITRITE						
	Δ PreANO mgN/d	Δ Anaerobic mgN/d	Δ Int. Set A mgN/d	Δ Int. Set B+Nit mgN/d	Δ Anoxic mgN/d	Δ Aerobic mgN/d	Δ Fin. SET mgN/d
31	-0.0	-0.2	-0.1	-1.1	0.1	0.8	0.3
32	2.9	-1.4	-2.0	-1.6	-30.2	23.8	4.4
33							
Config. 4	1.4	-0.8	-1.0	-1.3	-15.0	12.3	2.3
34	0.9	17.3	0.4	1.0	-15.7	-43.1	18.4
Config. 5	0.9	17.3	0.4	1.0	-15.7	-43.1	18.4
Average of Batch	NITRATE						
	Δ PreANO mgN/d	Δ Anaerobic mgN/d	Δ Int. Set A mgN/d	Δ Int. Set B+Nit mgN/d	Δ Anoxic mgN/d	Δ Aerobic mgN/d	Δ Fin. SET mgN/d
31	14.1	-1.7	-3.4	-738.3	740.6	-28.6	0.7
32	71.8	-2.0	-4.7	-964.2	913.2	-104.2	15.7
33							
Config. 4	43.0	-1.9	-4.0	-851.2	826.9	-65.4	8.2
34	150.8	263.3	1.8	-1415.1	676.1	-48.0	-46.5
Config. 5	150.8	263.3	1.8	-1415.1	676.1	-48.0	-46.5

TABLE 4.5b: N mass balance with all components for sewage batches 31 to 34.

Average of Batch	Sum NO2 denitrified mgN/d	Sum NO3 denitrified mgN/d	N Wasted mgN/d	N in Effluent mgN/d	N loss Nitrifier mgN/d	Sum N Out mgN/d	TKN in mgN/d	% Recovery	% N Removal
31	1.5	759.2	577.4	117.1	131.0	1386.3	1608.3	86.2	92.7
32	34.4	1015.2	382.6	191.2	73.0	1686.3	1997.5	84.4	90.9
33									
Config. 4	18.0	887.2	380.0	149.1	102.0	1536.3	1802.9	85.3	91.8
34	39.4	1103.8	560.4	518.0	-152.7	2068.9	2188.0	94.5	76.3
Config. 5	39.4	1103.8	560.4	518.0	-152.7	2068.9	2188.0	94.5	76.3

Fig 4.2:   N mass balance components.
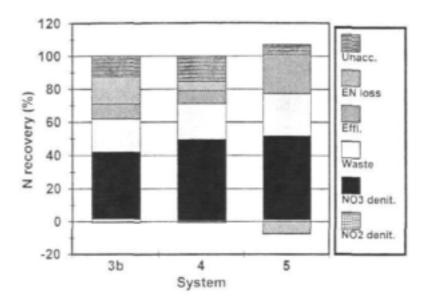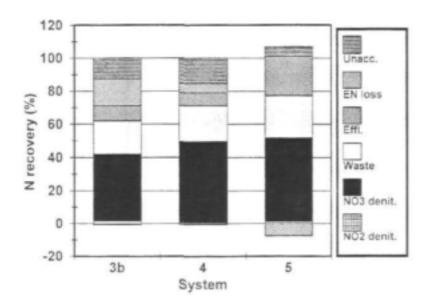
TABLE 4.6:   Denitrification potential of the pre- and main anoxic reactors for sewage batches where the outflow NO_x concentration exceeds 1 mgN/ℓ, for sewage batches 31 to 34.

	Main Anoxic		Pre Anoxic		Total Anoxic
Batch	NO(x) Denitrified mgN/d	Denit. Potential mgN/l infl.	NO(x) Denitrified mgN/d	Denit. Potential mgN/l infl.	Mass Fraction of System
31	740.06		14.1		0.55
32	913.2	45.7	74.7		0.55
33					
34	676.1	33.8	151.7	7.6	0.55

The overall average N mass balance for the 10 (Configuration 3b, Chapter 3), 8 and 5 day sludge age configurations are 88.1%, 85.3% and 94.5% respectively. As for the overall COD mass balances, the 10 and 8 day sludge age configurations have a similar overall average N mass balance, while that for the 5 day sludge age configuration is higher because the VSS concentration had not yet reached a steady state value. As was the case with the COD mass balance components, the variations in the N mass balance components are a result of factors that are independent of reducing the sludge age, with the exception of the N in the waste sludge. The overall average nitrite denitrification of 2.1% for the 10 day sludge age configuration is higher than the 1.0% and 1.8% for the 8 and 5 day sludge ages respectively. The percentage of the influent N denitrified via nitrate denitrification was 40.7% for the 10 day sludge age configuration and 49.2 and 50.4% for the 8 and 5 day sludge age configurations respectively. The percentage N leaving the system in the waste sludge increased with decreasing sludge age as expected, from 19.4% for the 10 day sludge age configuration to 21.1 for the 8 day, and 25.6% for the 5 day sludge age configuration. This is the result of a greater volume of mixed liquor wasted, particularly for the 5 day configuration because the VSS concentration had not yet

TABLE 4.7: Comparison of average N parameters for 10, 8 and 5 day sludge age configurations.

	ENBNRAS System Configurations		
	3	4	5
	10 Day Sludge Age	8 Day Sludge Age	5 Day Sludge Age
N Balance	88.1 %	85.3 %	94.5 %
% NO_2 Denit.	2.1	1.0	1.8
% NO_3 Denit.	40.7	49.2	50.4
% N Wasted	19.4	21.1	25.6
% N in Effluent	9.2	8.3	23.7
% N Loss to BN System	16.7	5.7	-7.0
% N Unaccounted	11.9	14.8	5.4
Nitrification Occurring Externally	91.1 %	92.9%	94.5 %
TKN in Final Effluent (Unfiltered)	4.8 mgN/l	4.8 mgN/l	6.0 mgN/l
FSA in Final Effluent	3.6 mgN/l	3.3 mgN/l	2.8 mgN/l
NO_x in Final Effluent	3.2 mgN/l	1.90 mgN/l	17.8 mgN/l
Total N in Final Effluent	8.0 mgN/l	6.7 mgN/l	24.7 mgN/l
Denit. Pot. Pre-Anoxic Reac.	3.7 mgN/l influent	-	7.0 mgN/l influent
Denit. Pot. Main Anoxic Reac.	31.1 mgN/l influent	45.7 mgN/l influent	33.8 mgN/l influent
TKN/COD Ratio of Influent	0.107	0.096	0.12
Total N Reduction	90.8 %	91.8%	76.3%

reached a steady state value. The percentage N in the final effluent of 9.2%, 8.3% and 23.7% for the 10, 8 and 5 day sludge age configurations respectively are difficult to compare as they are a function of the nitrate concentration in the effluent which depends on the TKN/COD ratio of the influent as well as the nitrification and denitrification performance of the system.

One of the main motives in implementing the ENBNR activated sludge system configuration was to uncouple the nitrification process from the main system and hence making nitrification independent of sludge age. Table 4.7 clearly shows the success of this system configuration - nitrification remained completely unaffected by the lowering of the sludge age. For the 8 and 5 day sludge age configurations 92.9 and 94.5% of the system nitrification occurred externally. This is even higher than the 90.8% for the 10 day sludge age configuration. From the final effluent FSA concentrations given in Table 4.7, it can be seen that full nitrification occurred throughout the 8 and 5 day sludge age configurations with only the residual FSA (from the internal settler underflow) appearing in the effluent.

The overall average denitrification potential of the main anoxic reactor was 31.1 mgN/ℓ influent, 45.7 mgN/ℓ influent and 33.8 mgN/ℓ influent for the 10, 8 and 5 day sludge age configurations respectively. The overall average TKN removal was 94.0%, 93.9% and 92.1% for the 10, 8 and 5 day sludge age configurations respectively. The TKN removals are very similar and all above 90%, which is a very good result. The total N (TN) removal (TKN and NO_x) was 90.8%, 91.8% and 76.3% for the 10, 8 and 5 day sludge age configurations respectively. The TN removals show more variation, but this is not due to the decrease in sludge age. The criteria that govern the TN removal are the system nitrification and denitrification, as well as the TKN/COD ratio of the influent. For an influent with, for example, a high TKN/COD ratio, full nitrification and poor denitrification the concentration of nitrate in the effluent will be high, leading to a lower overall TN removal. This is reflected in the results for the 10, 8 and 5 day sludge age configurations. The

overall average TKN/COD ratio of the influent for the 10 day sludge age configuration was 0.107 mgN/mgCOD and the denitrification potential of the main anoxic reactor 31.1 mgN/ℓ influent, resulting in an overall average TN removal of 90.8%. The overall average influent TKN/COD ratio for the 8 day sludge age configuration was 0.096 mgN/mgCOD, and the denitrification potential of the main anoxic reactor was 45.7 mgN/ℓ influent, resulting in an overall average TN removal of 91.8% - showing that the lower influent TKN/COD ratio combined with the higher denitrification potential of the main anoxic reactor resulted in a better TN removal performance for the 8 day sludge age configuration. The influent TKN/COD ratio of the 5 day sludge age configuration was a high 0.120 mgN/mgCOD and the denitrification potential of the main anoxic reactor was 33.8 mgN/ℓ influent (higher than for the 10 day, but significantly lower than for the 8 day sludge age configuration), which resulted in a overall average TN removal of only 76.3%, which is lower than that of both the 10 and 8 day sludge age configurations.

Accepting the variations occurring in the N removal parameters because of varying influent sewage characteristics as well as varying denitrification performance, the lowering of the sludge age did not have any marked effect on either nitrification, denitrification or TKN and TN removal. The results achieved for the 10 (Configuration 3b, Chapter 3), 8 and 5 day sludge age configurations are very similar and this shows that the ENBNR activated sludge system configuration is able to attain high N removals at sludge ages up to as low as 5 days.

4.3.3 Biological Excess Phosphorus Removal (BEPR)

Table 4.8 shows the results of the P mass balances over each of the reactors and settling tanks for sewage batches 31 to 34, and these are summarised in Fig 4.3. A negative result indicates P release while a positive result indicates P uptake. Table 4.9 shows a comparison for the main P parameters for the 10, 8 and 5 day sludge age configurations.

FIGURE 4.8: Average P release (-ve) or P uptake (+ve) for each reactor/settler and total P removal for sewage batches 31 to 34.

Δ Anaerobic mgP/ℓ inf.	Δ in SET A mgP/ℓ inf.	Δ in SET B+Nit. mgP/ℓ inf.	Δ Anoxic mgP/ℓ inf.	Δ Aerobic mgP/ℓ inf.	Δ Fin. SET mgP/ℓ inf.	ΣΔ	Total P Removal	% Recovery
-17.9	-4.2	-4.9	22.1	21.3	2.5	15.5	15.5	100
-23.7	-3.5	-4.4	23.7	19.5	1.1	12.5	12.5	100
-20.8	-3.9	-4.5	22.9	20.4	1.8	14.0	14.0	100
-6.25	-3.42	-4.11	10.76	7.82	1.71	8.63	8.63	100
-6.25	-3.42	-4.11	10.76	7.83	1.71	8.63	8.63	100

Fig 4.3: P release (rel., -ve) and P uptake (upt., +ve) for anaerobic (anaer.), internal settling tank (Int. settl.), external nitrification (EN), pre-anoxic (Preanx.), main anoxic (Main anx.), aerobic and final settling tank (Final settl.) for the three investigations.

TABLE 4.9: Comparison of average P parameters for the 10, 8 and 5 day sludge age configurations.

| | ENBNRAS System Configurations | | |
| | 3 | 4 | 5 |
	10 Day Sludge Age	8 Day Sludge Age	5 Day Sludge Age
NO₃ Flowing into Anaerobic Reac	0.98 mg/ℓ	0 mg/ℓ	10.9 mg/ℓ
P Release (excl. rel. in EN sys.)	20.8 mgP/ℓ influent	26.6 mgP/ℓ influent	9.67 mgP/ℓ influent
P Uptake	36.4 mgP/ℓ influent	45.1 mgP/ℓ influent	22.4 mgP/ℓ influent
Anoxic P Uptake	62.3%	47.1%	57.9%
P Removal	10.5 mgP/ℓ influent	14.0 mgP/ℓ influent	8.6 mgP/ℓ influent

A reduction in sludge age increases P removal per mass of organic load (Wentzel et al., 1990), provided that it is not reduced below some lower limit (~3 days) to prevent the PAOs from being 'washed' out of the system completely. It would therefore be expected that the 8 and 5 day sludge age configurations should show improved P removal compared to the 10 day sludge age system configuration. From Table 4.9 it can be seen that the overall average P removal was 10.5 mgP/ℓ influent, 14.0 mgP/ℓ influent and 8.6 mgP/ℓ influent for the 10, 8 and 5 day sludge age configurations respectively. The low P removal for the 5 day sludge age configuration occurred because the influent TKN COD ratio was very high and this led to a high concentration of nitrate (10.9 mg/ℓ, see Table 4.9) being recycled to the anaerobic reactor and limiting the P release, resulting in a low overall P removal performance of 8.6 mgP/ℓ influent. However, this is not linked to the short sludge age, but rather to the main anoxic reactor being overloaded by the high nitrate load that resulted from the high TKN COD ratio of the influent. The 8 day sludge age

configuration showed the highest P removal (14.0 mgP/ℓ influent), but there was no nitrate leaking into the anaerobic reactor during this configuration and this contributed to the higher overall P removal. The 10 day sludge age configuration achieved an overall average P removal of 10.5 mgP/ℓ influent with an average of 0.95 mgN/ℓ flowing into the anaerobic reactor from the pre-anoxic reactor. Even with zero nitrate flowing into the anaerobic reactor the 10 day sludge age configuration would not have achieved an average P removal of 14.0 mgP/ℓ influent, showing that a reduction in sludge age does improve the P removal performance. Had no nitrate flowed into the anaerobic reactor during the 5 day sludge age configuration, it would probably have achieved higher P removal than the 8 day sludge age configuration did.

For the 10 day sludge age configuration, an overall average of 63.3% of the P uptake occurred in the anoxic reactor. For the 8 and 5 day sludge age configurations, the percentage anoxic P uptake was 47.1 and 57.9% respectively. The average NO_x load on the main anoxic reactor was 18.6 mgN/ℓ, 20.7 mgN/ℓ and 34.9 mgN/ℓ for the 10, 8 and 5 day sludge age configurations respectively. The higher NO_x load on the anoxic reactor for the 5 day sludge age configuration led to the 10.8% higher anoxic P uptake compared to that of the 8 day sludge age configuration. The 8 day sludge age configuration had a 2.4 mgN/ℓ higher NO_x load on the anoxic reactor than the 10 day sludge age configuration, but 14.8% lower anoxic P uptake. This is most likely because the 8 day sludge age configuration result is an average of only two sewage batches, while the result from the 10 day sludge age configuration is the average of 10 sewage batches. Had the 8 day sludge age configuration been run for 10 sewage batches, the result would have been closer to that of the 10 day sludge age configuration.

When the ENBNR activated sludge system is operated at lower sludge ages, an improvement in the overall P removal can be expected. Considerable anoxic P uptake continues to occur at the lower sludge ages, and the percentage anoxic P uptake continues to shift with the NO_x load on the main anoxic reactor.

4.3.4 Sludge settleability

The overall average DSVI was 95.6 mℓ/g, 89.8 mℓ/g and 92.9 mℓ/g for the 10 (Configuration 3b, Chapter 3), 8 and 5 day sludge age configurations respectively. This shows that the decrease in sludge age had no effect on the DSVI performance of the ENBNR activated sludge system. It cannot be said that the lowering of the sludge age produced a better settling sludge, because the values of the 8 and 5 day sludge age configurations are close to those obtained for the 10 day sludge age configuration (Configuration 3b). Even with a nitrate concentration of 15 mgN/ℓ flowing from the main anoxic reactor (see Table 4.2c) for the 5 day sludge age configuration the DSVI deteriorated only very slightly and did not rise above 100 ml/g as it did for the 10 day sludge age system configuration when similarly high nitrate concentrations flowed from the anoxic reactor during the period that the system was recovering from the bad sewage batch 9.

4.4   CONCLUSIONS

The ENBNR activated sludge system configuration was changed to shorter sludge ages and increased influent flow towards the end of the investigation on System Configuration 3b (Chapter 3). The system was operated in Configuration 4 (8 days sludge age and 25 instead of 20 ℓ/d influent flow) for 49 days (sewage batches 31 to 33) and in Configuration 5 (5 days sludge age and 25 ℓ/d influent flow) for a further 13 days (sewage batch 34). The system configuration was changed to the shorter sludge ages to evaluate the response to and performance of the ENBNR activated sludge system to the shorter sludge ages.

1.   The overall average COD mass balances for the 8 and 5 day sludge age configurations (4 and 5 respectively) of the ENBNR activated sludge system were 79 and 92% respectively and the overall average COD removals (based on unfiltered COD samples) 93 and 90% respectively. Although lower than the overall average COD removal of 94% achieved by the 10 days sludge age configuration (Configuration 3b, Chapter 3), the COD removal performances of the short sludge age configurations are still very good. The slightly lower values are most likely due to the hydraulic impact on the final settler caused by the increase in influent flow, rather than due to the system removing less COD.

2.   The overall average N mass balances attained for the 8 and 5 day sludge age configurations were 85 and 95% respectively. For the 8 and 5 day sludge age configurations, 88 and 96% of the FSA flowing into the EN system was nitrified. On average, 93 and 95% of the ENBNR activated sludge system nitrification was effected externally for the 8 and 5 day sludge age configurations respectively. This demonstrates one of the main benefits of the ENBNR activated sludge system configuration - virtually complete nitrification at 8 and 5 days sludge age with only 0.20 aerobic mass fraction, and this will be attainable also at lower temperatures than 20°C.

3.   The overall average denitrification potential of the main anoxic reactor was about 46 and 34 mgN/ℓ influent for the 8 and 5 day sludge age configurations respectively. These are higher than observed at ten days sludge age, even when the system denitrification was greatest (i.e. Configuration 3, 31 mgN/ℓ).

4.   The overall average TKN removal for the ENBNR activated sludge system operated at the 8 and 5 day sludge age configurations were 94 and 92% respectively. The TN removal was 92 and 76% respectively. The discrepancies in the TN removals for the 8 and 5 day sludge age configurations are due to the average influent TKN/COD ratios of the sewage batches fed to the two configurations: for the 8 day sludge age configuration this was 0.096, while for the 5 day sludge age configuration it was much higher at 0.120 mgN/mgCOD. The higher influent TKN/COD ratio and lower denitrification potential of the 5 day sludge age configuration led to more nitrate in the effluent, and therefore to a lower TN removal.

5.   The overall average P removal for the 8 day sludge age configuration was 14.0 mgP/ℓ influent. For the 5 day sludge age configuration this was 8.6 mgP/ℓ influent. The 8 day sludge age configuration removed 4.2 mgP/ℓ influent more than the 10 days sludge age configuration, showing that there is indeed an improvement in BEPR as the sludge age

is decreased as reflected in the BEPR model of Wentzel *et al.* (1990). It would therefore be expected that the 5 day sludge age would show a further improved BEPR performance; however, for the 5 day sludge age configuration, high nitrate concentrations were recycled to the pre-anoxic reactor, which was consequently overloaded, causing nitrate to enter the anaerobic reactor, which in turn caused a decrease in P release and hence a decrease in P removal. The nitrate recycle was due to the high influent TKN/COD ratio (0.12) and reduced main anoxic reactor performance.

6. The overall average % anoxic P uptake for the 8 day sludge age configuration was 47% (with a nitrate load of 20.7 mgN/ℓ on the main anoxic reactor), and the overall average % anoxic P uptake for the 5 day sludge age configuration was 58% (with a nitrate load of 34.9 mgN/ℓ on the main anoxic reactor). This shows clearly that as the nitrate load on the main anoxic reactor increases, the % anoxic P uptake increases.

7. The overall average DSVI for the 8 and 5 day sludge age configurations were about 90 mℓ/g and 93 mℓ/g respectively. For the 8 day sludge age configuration <1 mgN/ℓ flowed out of the main anoxic reactor, but for the 5 day sludge age configuration about 15 mgN/ℓ flowed out of the main anoxic reactor. Good sludge settleability at sludge ages less than 10 days (8, 6 and 5 days sludge age) has been observed in intermittently aerated ND systems by Warburton *et al.* (1991) and Phoredox, 3 stage Bardenpho, UCT and JHB BEPR systems (Burke *et al.*, 1986). The AA (low F/M) filament bulking hypothesis (Casey *et al.*, 1994) is not considered to be applicable at 3 to 8 days sludge age because the AA filaments are slow growers that tend to proliferate in long sludge age (>8d) systems.

A detailed analysis of the results of the 8 and 5 day sludge age configurations is not intended, as the system performance evaluation at these shorter sludge ages lasted for only 62 days. However, the results show that the ENBNR activated sludge system BNR performance in no way deteriorated at the shorter sludge ages, in fact a reduction in sludge age tends to increase N and P removal per mass of organic load (Wentzel *et al.*, 1990), provided that it is not reduced below a lower limit of about 5 days for operational reasons (sludge flocculation, effluent turbidity); in terms of 'wash out' of PAOs the sludge age can be as low as 3 days (Wentzel *et al.*, 1990).

CHAPTER 5

COMPARISON OF EXTERNAL NITRIFICATION (EN) BIOLOGICAL NUTRIENT REMOVAL (BNR) ACTIVATED SLUDGE SYSTEM WITH A CONVENTIONAL BNR (UCT) ACTIVATED SLUDGE SYSTEM

5.1 INTRODUCTION

In implementing the external nitrification BNR activated sludge (ENBNRAS) system, its performance with respect to nutrient removal should not be significantly reduced compared to the conventional BNR activated sludge system. With regard to nitrogen removal via denitrification, the investigations in Chapters 3 and 4 indicate that the ENBNRAS system is superior in that complete denitrification can be achieved for influent wastewater TKN/COD ratios that are considerably higher than those for which it can be achieved in conventional BNR systems. However, with regard to P removal, the ENBNRAS system stimulates anoxic P uptake in the system which would appear to be reduced compared to aerobic P uptake. This indicates that the P removal performance of the ENBNRAS system may be reduced compared to the conventional system. In this Chapter this aspect is examined in more detail, to compare more precisely the N and P removal performance in the external nitrification and conventional BNR systems. Also, the factors stimulating anoxic P uptake in the conventional BNR system are investigated. This investigation is summarised in this Chapter, and reported in detail by Vermande et al. (2000).

5.2 SYSTEM OPERATION

A laboratory-scale BNRAS system (UCT configuration) with similar design and operating parameters to the ENBNRAS system Configuration 3b (Chapters 3 and 4) was run in parallel with this laboratory- scale ENBNRAS system. Figure 5.1 shows the system layout of the laboratory-scale UCT system and Table 5.1 lists the design and operating parameters for both the ENBNRAS and UCT systems. In order to compare the performance of the two systems, they were fed the same influent sewage for 18 sewage batches (sewage batches 13 to 30, Table 3.1 Configuration 3) - from the 7 August 1999 (day 167) to the 17 April 2000 (day 421). 40 ℓ of influent were prepared in the same container, thoroughly stirred, and 20 ℓ of this prepared sewage was fed to each of the systems respectively. While the ENBNRAS system was operated and analysed by Sötemann et al. (2000), the UCT system was operated, tested and reported on by Vermande et al. (2000); for details the reader is referred to the relevant report. Detailed comparisons of the two systems are also given in both reports.

FIGURE 5.1: Schematic layout of the laboratory scale UCT system run in parallel with the laboratory scale ENBNRAS system.

TABLE 5.1: ENBNRAS and UCT system design and operating parameters.

Parameter	UCT System	ENBNRAS System
Influent Flow (l/d)	20	20
Sludge Age (d)	10	10
Temperature (°C)	20	20
D.O. Main Aerobic Reactor (mgO/l)	2 to 5	2 to 5
Total System Volume (l)	20	20
Pre-Anoxic Reactor (l)	-	2*
Anaerobic Reactor (l)	3**	5
Main Anoxic Reactor (l)	7	6.5 (for sewage batch 13) 9 (for sewage batches 14 to 30)
Main Aerobic Reactor (l)	10	6.5 (for sewage batch 13) 4 (for sewage batches 14 to 30)
Aerobic Mass Fraction	0.5	0.33 (for sewage batch 13) 0.2 (for sewage batches 14 to 30)
Unaerated Mass Fraction	0.5	0.67 (for sewage batch 13) 0.8 (for sewage batches 14 to 30)
Anoxic Mass Fraction	0.35	0.42 (for sewage batch 13) 0.58 (for sewage batches 14 to 30)
Anaerobic Mass Fraction	0.15	0.25
a-recycle (w.r.t infl. flow)	1:1	2:1 (for sewage batches 13 to 20) 0:1 (for sewage batches 21 to 30)
s- recycle (w.r.t infl. flow)	1:1	1:1
r - recycle (w.r.t infl. flow)	1:1	-

* Actual volume 1l, with sludge at double concentration.
** Actual volume 8l, with sludge diluted to half the normal concentration.

5.3 SYSTEM PERFORMANCE

Tables 5.2a, b and c list the sewage batch averages for all measured parameters for sewage batches 13 to 30 for the UCT system. The sewage batch numbers correspond to the sewage batch numbers used for the ENBNRAS system in Chapters 3 and 4 for Configuration 3, to facilitate the direct comparison of the two systems. Where the overall averages of the two systems are compared in this section, the overall averages refer to the average of the sewage batch averages for sewage batches 13 to 30. This also applies to the ENBNRAS system results - in this section the overall averages refer to the average of the sewage batch averages for sewage batches 13 to 30 calculated from the data in Chapter 3. Therefore, the overall averages may differ from those mentioned in Chapter 3, where the overall averages include sewage batches 1 to 13 also.

It should further be noted that because the chemical tests were performed by two different researchers, the influent sewage characteristics show some minor differences, even though the respective feeds originated from the same feed mixing container. This is as a result of the independent analyses and associated analytical variations, but these do not impact significantly on the comparison, because the overall averages of the influent sewage characteristics over the 18 sewage batches are almost identical. Furthermore, both independent analyses arrived at the same overall average mixed liquor COD/VSS ratio of 1.43 and TKN/VSS ratio of 0.1 over the 18 sewage batches.

TABLE 5.2a: Sewage Batch averages of measured COD and TKN parameters for the UCT system for sewage batches 13 to 30 (UF = unfiltered; F = 0.45μm membrane filtered).

| Sewage Batch | mgCOD/l | | | | mgN/l | | | | | | TKN/COD Ratio |
| | COD | | | | TKN | | | | FSA | | |
	Influent	Aerobic ML	Unflt Eff	Filt Eff	Influent	Aerobic ML	Unflt Eff	Filt Eff	Influent	Unflt Eff	Influent
13	790.9	2651.4	67.4	43.2	82.1	150.4	2.6	2.1	61.1	1.9	0.104
14	689.6	2732.2	67.5	37.6	71.3	161.7	2.7	1.3	52.7	1.4	0.104
15	723.1	2470.1	55.3	43.7	89.2	160.5	5.4	1.9	69.8	2.0	0.123
16	732.3	2565.5	61.5	47.8	83.0	168.0	3.2	1.6	62.3	1.1	0.113
17	675.5	2492.9	43.8	31.1	63.8	171.3	3.2	1.8	47.8	1.1	0.094
18	764.3	2505.5	49.7	38.6	84.7	161.2	3.0	1.6	52.0	0.6	0.083
19	748.7	2694.2	46.5	39.7	84.5	150.8	3.1	2.3	68.2	0.6	0.107
20	784.9	2320.7	54.9	43.0	69.1	159.4	3.1	2.8	50.3	1.1	0.088
21	716.5	2539.7	55.2	37.7	63.2	163.5	4.1	3.3	49.9	2.5	0.088
22	718.6	2411.8	55.3	43.2	77.2	163.9	5.1	4.7	68.9	2.9	0.107
23	663.1	2380.3	56.9	39.7	70.9	150.4	5.2	3.8	61.4	2.7	0.105
24	721.1	2224.2	42.6	35.5	86.7	145.6	4.9	3.1	68.3	2.4	0.120
25	694.6	2510.5	42.0	35.8	58.2	160.9	3.7	2.8	46.7	2.1	0.084
26	669.1	2294.6	46.4	38.5	84.5	150.9	4.5	3.1	70.6	1.9	0.121
27	687.9	2276.3	53.9	38.8	82.2	169.3	4.6	3.3	72.6	2.1	0.119
28	716.2	2860.3	52.9	33.0	86.1	166.1	5.1	3.1	71.4	2.6	0.123
29	748.9	2704.8	56.5	41.2	67.3	180.0	4.5	3.2	57.2	2.2	0.090
30	799.1	2261.4	56.3	42.7	68.3	172.8	3.5	2.1	53.9	1.4	0.085
Overall	734.8	2606.0	53.0	39.5	75.7	160.8	4.0	2.6	59.9	1.8	0.103
-	UF	UF	UF	F	UF	UF	UF	F	UF	UF	UF

TABLE 5.2b: Sewage batch averages of measured suspended solids, OUR, DSVI and pH for the UCT system for sewage batches 13 to 30.

Sewage Batch	TSS	VSS	COD/VSS Ratio [1]	TKN/VSS Ratio [1]	mgO/h OUR	m/g DSVI	pH	
			Aerobic	Aerobic	Aerobic	Aerobic	Anaerobic	Aerobic
13	2226.4	1638.4	1.42	0.082	32.4	115.2	-	-
14	2102.7	1704.3	1.58	0.096	39.0	73.4	-	-
15	2002.7	1624.0	1.31	0.097	29.3	94.3	-	-
16	2066.3	1699.0	1.50	0.098	36.7	114.3	-	-
17	2128.3	1698.0	1.45	0.101	31.0	112.0	-	-
18	1946.3	1567.7	1.57	0.102	31.1	119.8	-	-
19	1777.8	1444.6	1.42	0.109	32.3	121.1	-	-
20	1988.3	1609.3	1.42	0.100	29.3	112.9	-	-
21	2207.3	1780.7	1.40	0.103	31.8	112.1	-	-
22	2137.5	1696.5	1.40	0.096	35.5	145.1	-	7.77
23	1931.3	1580.4	1.48	0.095	34.5	186.8	-	7.81
24	1972.7	1642.4	1.33	0.089	35.3	197.1	7.68	7.87
25	2208.4	1813.6	1.36	0.089	30.3	183.1	7.56	7.91
26	2056.3	1672.5	1.35	0.090	33.3	201.0	7.61	7.82
27	2480.0	1923.1	1.42	0.088	27.6	154.9	7.53	7.83
28	2499.2	2006.7	1.40	0.094	28.0	157.2	7.51	7.80
29	2409.0	1878.0	1.46	0.096	26.5	144.1	7.35	7.91
30	2492.8	1900.3	1.43	0.091	28.0	141.6	6.89	7.14
Overall	2146.3	1726.6	1.43	0.095	31.8	138.1	7.4	7.76

[1] Calculated from unfiltered aerobic reactor COD and TKN concentrations divided by the VSS.

TABLE 5.2c: Sewage batch averages for measured nitrate, nitrite and P concentrations for the UCT system for sewage batches 13 to 30.

Sewage Batch	Nitrite				Nitrate				mgP/l Phosphates					
	Anaerobic	Anoxic	Aerobic	Filt Eff	Anaerobic	Anoxic	Aerobic	Filt Eff	Influent	Anaerobic	Anoxic	Aerobic	Unfilt Eff	Filt Eff
13	0.1	0.5	1.8	1.5	0.5	1.2	13.1	13.4	26.2	31.1	21.2	14.7	-	14.3
14	0.0	0.1	0.6	0.4	0.3	2.4	15.5	13.4	26.2	31.3	17.4	8.3	-	7.3
15	0.0	0.2	0.7	0.4	0.3	5.3	20.0	22.2	30.8	30.2	19.4	14.6	-	13.9
16	0.0	0.1	1.1	1.0	0.1	1.2	13.4	13.6	25.0	33.4	22.6	13.6	-	14.0
17	0.1	0.2	0.3	0.2	0.1	0.3	8.3	8.6	25.4	34.5	21.3	12.6	-	13.0
18	0.0	0.1	0.2	0.2	0.1	0.3	8.4	7.8	26.9	35.9	23.5	14.1	-	14.7
19	0.0	0.3	0.5	0.2	0.1	0.9	11.2	12.2	21.0	30.9	21.3	13.0	-	12.6
20	0.1	0.1	0.2	0.1	0.1	0.3	8.3	8.6	25.8	32.9	21.8	13.5	-	13.5
21	0.1	0.1	0.1	0.1	0.2	0.3	7.0	7.9	25.2	33.6	22.2	14.9	15.1	15.1
22	0.1	0.9	0.5	0.2	0.2	1.0	10.5	11.5	24.0	28.7	19.5	14.6	15.1	14.9
23	0.1	0.8	0.7	0.5	0.2	1.7	12.5	13.4	23.0	25.8	18.9	15.8	15.5	15.7
24	0.1	0.5	0.4	0.6	0.2	1.0	12.5	13.3	25.2	29.2	19.0	15.0	14.9	14.6
25	0.0	0.1	0.2	0.1	0.1	0.2	8.0	8.2	28.7	33.3	23.3	18.0	18.0	17.7
26	0.0	0.5	0.8	0.4	0.1	2.4	15.1	17.2	29.1	36.0	23.8	17.5	16.8	16.8
27	0.0	0.4	0.3	0.2	0.2	2.9	15.7	16.9	25.6	35.8	19.5	11.9	11.8	11.1
28	0.0	0.4	0.4	0.3	0.2	1.6	15.0	16.1	25.3	38.6	22.9	13.1	12.6	12.0
29	0.1	0.1	0.3	0.1	0.1	0.1	8.1	8.3	24.9	41.9	20.0	8.4	9.0	8.2
30	0.0	0.1	0.1	0.2	0.1	0.6	9.8	10.0	26.8	41.7	21.3	9.5	8.8	8.5
Overall	0.1	0.3	0.5	0.4	0.2	1.3	11.8	12.4	25.9	33.5	21.0	13.5	13.8	13.2

5.3.1 Carbonaceous Material Removal

Figure 5.2 shows the overall COD mass balances achieved for sewage batches 13 to 30 for the UCT and the ENBNRAS systems. The overall average COD mass balance achieved for the UCT system is 78.3% and that for the ENBNRAS system is 76.8%. These values are within 2 percent, indicating that while the overall average COD mass balance for the UCT system is 1.5% higher than the overall average COD mass balance achieved for the ENBNRAS system, both are equally low. This indicates that the same as yet unidentified biological process which is thought to consume a fraction of the influent COD without being taken account of in the usual analytical procedures also occurred in the UCT system, and it confirms that the low COD balances are not characteristics of the ENBNRAS system alone, but rather a characteristic of BNRAS systems in general. From Figure 5.2 it can be seen that the COD mass balances achieved for each sewage batch are similar. It seems that the largest discrepancies occur at low and very high influent TKN/COD ratios, with the ENBNRAS system achieving better COD balances for sewage batches with a very high influent TKN/COD ratio (e.g. sewage batches 15, 19, 26, 27 and 28 with influent TKN/COD ratios of 0.124, 0.116, 0.118, 0.111 and 0.123 respectively), and the UCT system achieving higher COD mass balances for sewage batches with lower influent TKN/COD ratios (e.g. sewage batches 18, 21, 22 and 25 with influent TKN/COD ratios of 0.087, 0.089, 0.107 and 0.085 respectively).
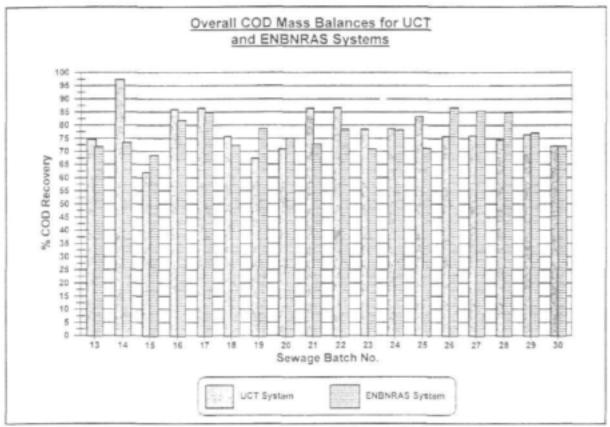
FIGURE 5.2:   COD mass balances for the UCT and ENBNRAS systems for sewage batches 13 to 30.

On average over sewage batches 13 to 30, the UCT system influent COD was 735 mgCOD/ℓ and the ENBNRAS system influent COD was 731 mgCOD/ℓ. The overall average influent COD values are within 1% of each other, confirming that the two systems did indeed receive the same feed even though there are variations in the influent COD values for each of the separate sewage batches. Figure 5.3 shows the COD removal performance for each of the two systems, as a percentage of the influent COD concentration fed to each system. From Figure 5.3 it can be seen that the COD removal performances of the two systems are virtually identical. The UCT and ENBNRAS systems removed an overall average of 92.8% and 93.5% of the influent COD respectively; while the ENBNRAS system removed 0.7% more COD on average, this difference is negligible. BNRAS systems generally remove COD virtually completely irrespective of configuration and this is clearly demonstrated here.

Figure 5.4 shows the daily oxygen demand of the main aerobic reactors for the UCT and the ENBNRAS systems. The oxygen demand is given in units of mgO/d because, being independent of the reactor volume, it gives a more accurate reflection of the oxygen demand in the respective systems.
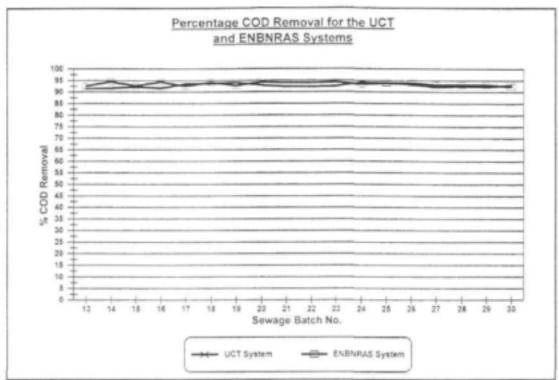
FIGURE 5.3: Percentage COD removal by the UCT and ENBNRAS systems for sewage batches 13 to 30.
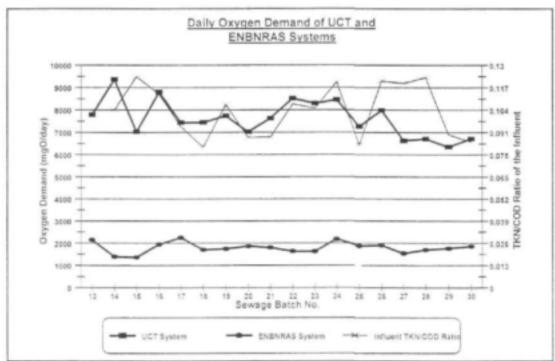

FIGURE 5.4: Daily oxygen demand for the UCT and ENBNRAS systems for sewage batches 13 to 30.

Figure 5.4 shows the advantage of the ENBNRAS system in terms of oxygen demand. The UCT system had an average daily oxygen demand of 7625 mgO/d over the 18 sewage batches, while the ENBNRAS had an average daily oxygen demand of only 1798 mgO/d. By nitrifying externally, the ENBNRAS system requires about 76% less oxygen than the UCT system requires with nitrification taking place internally. The influent TKN/COD ratio is included in Figure 5.4 to illustrate the variation of the daily oxygen demand of the UCT system with the variation of the influent TKN/COD ratio. As the influent TKN/COD ratio increases, more nitrate is produced and the daily oxygen demand of the UCT system rises, and vice versa. The daily oxygen demand of the ENBNRAS system does not show the same variation with varying influent TKN/COD ratios because nitrification occurs externally and is not coupled to the oxygen demand of the system. This results in a more constant daily oxygen demand for the ENBNRAS system, Figure 5.4.

A further interesting comparison can be made regarding the VSS concentrations of the two systems. The UCT system had an overall average VSS concentration of 1727 mgVSS/ℓ while the ENBNRAS systems overall average VSS concentration was 1437 mgVSS/ℓ over sewage batches 13 to 30. The 16.8% lower VSS concentration for the ENBNRAS system almost corresponds to the 18.3% of the influent COD 'lost' to the EN system of the ENBNRAS system. This indicates that the COD that is 'lost' to the EN system is not available to the organisms in the BNRAS system and this will result in the ENBNRAS system containing lower VSS concentrations, roughly in proportion to the fraction of the influent COD that is removed in the EN system and hence 'lost' to the main system.

5.3.2 Nitrogenous Material Removal

Figure 5.5 shows the N mass balances for sewage batches 13 to 30 for the UCT and the ENBNRAS systems. The overall average N mass balance over the 18 sewage batches for the UCT and ENBNRAS systems was 86.1% and 87.0% respectively. As was the case for the COD balances, the results are very close, albeit considerably higher than the respective COD balances, which is usually the case for BNRAS systems in the Water Research Laboratory. From Figure 5.5 it can be seen that the N mass balances for the respective sewage batches are similar with marked differences in the N mass balances only occurring for sewage batches 14, 15, 17 and 22. Figure 5.6 shows the overall TKN reduction achieved by the UCT and ENBNRAS systems, as a percentage reduction of the influent TKN. The TKN reduction achieved by the two systems is very similar. The UCT system achieved an overall average TKN reduction of 94.7% and the ENBNRAS system achieved a slightly lower TKN reduction of 93.8%. The reason for the ENBNRAS system achieving a lower value is the FSA concentration in the final effluent. The effluent FSA of the ENBNRAS system was on average over the 18 sewage batches 3.5 mgN/ℓ FSA, while that of the UCT system final effluent was 1.8 mgN/ℓ FSA. The source of this effluent FSA in the ENBNRAS system is the FSA that bypasses the EN system in the sludge bypass, and because almost no nitrification occurs in the main aerobic reactor, this FSA flows out in the effluent without being nitrified. The concentration of FSA in the EN system outflow of the ENBNRAS system, on average over the 18 sewage batches, was 3.3 mgN/ℓ. This is very similar to the 3.5 mgN/ℓ FSA in the final effluent and indicates that (i) the FSA in the final effluent is approximately equal to the FSA that was not nitrified in the EN system, (ii) the FSA that bypasses the EN system in the EN system sludge bypass is nitrified in the aerobic reactor and (iii) the FSA released in the main anoxic reactor is also nitrified in the aerobic reactor. The nitrifiers are seeded into the activated sludge with the EN system outflow, and this is how the system is

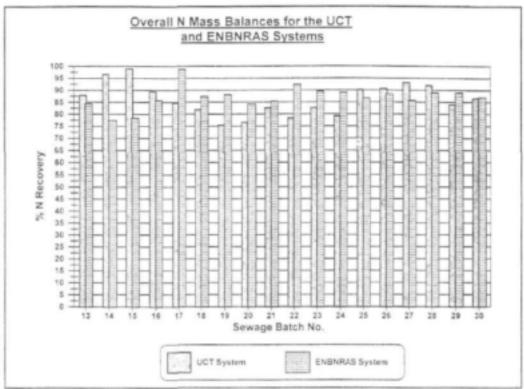intended to operate to maintain a low final effluent FSA.

FIGURE 5.5: N mass balances for the UCT and ENBNRAS systems for sewage batches 13 to 30.
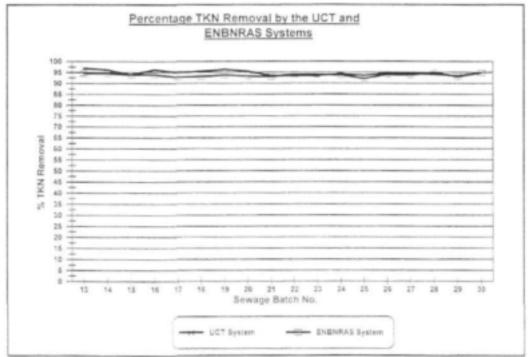

FIGURE 5.6: Percentage TKN removal for the UCT and ENBNRAS systems for sewage batches 13 to 30.

Figure 5.7 shows the total N concentrations in the effluents of the UCT and the ENBNRAS systems for sewage batches 13 to 30. From Figure 5.7 the difference in the N removal performance of the UCT and the ENBNRAS systems can be seen more clearly. While the TKN measurements take account of only organic N and FSA, the total N represents organic N, FSA, nitrite and nitrate. Figure 5.6 shows that both systems remove TKN equally efficiently, but from Figure 5.7 it can be seen that the ENBNRAS system removes more total N than the UCT system. This means that the ENBNRAS system produces a final effluent with a lower nitrate concentration than the UCT system, because it is the varying concentration of nitrate in the effluent that leads to the variation of the total N in the effluent. On average over sewage batches 13 to 30, the UCT system effluent nitrate was 12.4 mgN/ℓ, while that from the ENBNRAS system was only 4.6 mgN/ℓ of nitrate. This results in the ENBNRAS system having the potential of producing effluents containing <10 mg/ℓ total N, while the UCT system is not capable of achieving similar results[1]. This can also be seen from Figure 5.7: The ENBNRAS system achieved a total N (TN) concentration in the effluent of <10 mgN/ℓ for 10 of the 18 sewage batches, while the UCT system did not once achieve TN concentrations of <10 mgN/ℓ in the effluent. The influent TKN/COD ratio has also been included on Figure 5.7 to illustrate how the variations in the effluent TN concentrations mirror the variations in the influent TKN/COD ratio. On average over the 18 sewage batches, the UCT system effluent TN concentration was 16.7 mgN/ℓ, while that for the ENBNRAS system was 9.8 mgN/ℓ. The main reason for this difference is the potential of the ENBNRAS system to denitrify completely with its larger anoxic mass fraction and its low nitrification in the aerobic reactor. The UCT system cannot denitrify completely because all nitrification takes place in the aerobic reactor.

Figure 5.8 shows the percentage TN removals for the UCT and the ENBNRAS systems for sewage batches 13 to 30. From Figure 5.8 it can be seen that the ENBNRAS system removed a greater percentage N from the influent wastewater than the UCT system for all 18 sewage batches. On average over the 18 sewage batches, the UCT system removed 78.2% of the total influent N and the ENBNRAS system removed 87.8% of the total influent N.

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[1] The higher effluent nitrate for the UCT system was due to the smaller anoxic mass fraction. The effluent NO_x could have been reduced somewhat by increasing the a-recycle ratio, but high NO_x concentrations (>3 mgN/ℓ) in the outflow of the anoxic reactor leads to (i) deterioration in sludge settleability and (ii) anoxic P uptake BEPR and lower P removal (see Figures 5.12 and 5.13) below.

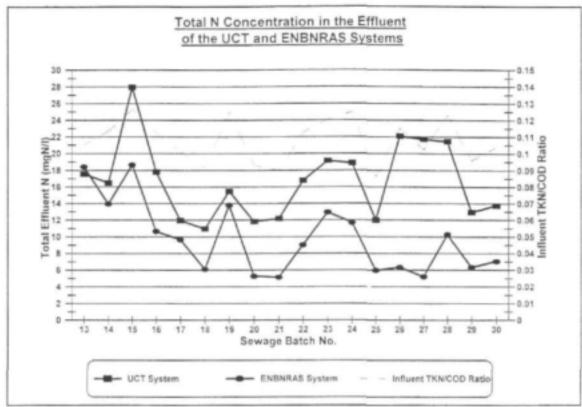**FIGURE 5.7:** Total N concentrations in the effluent of the UCT and ENBNRAS systems and influent TKN/COD ratios for sewage batches 13 to 30.
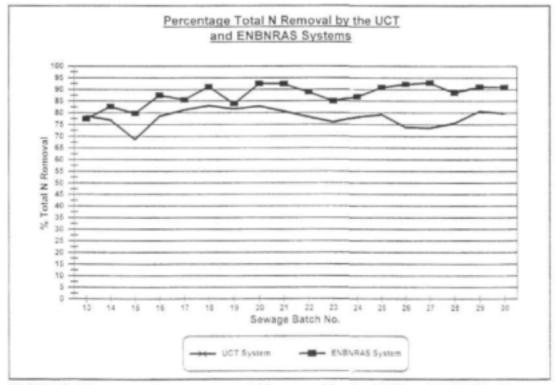

FIGURE 5.8: Percentage total N removal by the UCT and ENBNRAS systems for sewage batches 13 to 30.

5.3.3   Biological Excess Phosphorus Removal (BEPR)

The ENBNRAS system favours anoxic/aerobic P uptake BEPR, while the UCT system favours aerobic P uptake BEPR. However, when the UCT system is fed sewage with a high influent TKN/COD ratio, which results in a high nitrate load on the main anoxic reactor, anoxic P uptake does occur. For sewage batches 21 to 27, the influent TKN/COD ratio was kept consistently high by FSA addition to the influent (>0.100 mgN/mgCOD) to induce anoxic P uptake in the UCT system, so that the BEPR performance of the UCT system with anoxic P uptake as well as with predominantly aerobic P uptake can be compared to the BEPR of the ENBNRAS system. Figure 5.9 shows the percentage anoxic P uptake for both the UCT and the ENBNRAS systems for sewage batches 13 to 30.

FIGURE 5.9:   Percentage anoxic P uptake for the UCT and ENBNRAS systems for sewage batches 13 to 30.

From Figure 5.9 it can be seen that considerable anoxic P uptake (40 to 70%) occurred in the ENBNRAS system throughout the 18 sewage batches, with an overall average over the 18 sewage batches of ~60%. In the UCT system negligible anoxic P uptake occurred for sewage batches 13 to 21, with the exception of sewage batch 15, which had a very high influent TKN/COD ratio of about 0.123. During sewage batches 21 to 27, where the influent TKN/COD ratio was kept consistently above 0.100 (by dosing FSA to the influent), appreciable anoxic P uptake took place in the UCT system (10 to 30%). However, the anoxic P uptake in the UCT system never reached the same magnitude observed in the ENBNRAS system and on overall average over the 6 sewage batches (22 to 27) only 20% anoxic P uptake occurred. This shows that the BEPR in the UCT system was essentially aerobic P uptake BEPR. After sewage batches 21 to 27, the FSA dosing to the influent was stopped which lowered the influent TKN/COD ratio

and underloaded the anoxic reactor with nitrate, and the system returned to predominantly aerobic P uptake.

Figures 5.10 and 5.11 show the P release and P uptake respectively for the UCT and the ENBNRAS systems over the 18 sewage batches. From Figure 5.10 it can be seen that for the sewage batches where there was negligible anoxic P uptake in the UCT system (sewage batches 13,14,16 to 20 and 28 to 30) it released on average ~7 mgP/ℓ influent more P than the ENBNRAS system. However, for the sewage batches where there was anoxic P uptake in the UCT system (sewage batches 15 and 21 to 27) the ENBNRAS system released on average ~3 mgP/ℓ influent more P than the UCT system did. On overall average over all of the 18 sewage batches, the UCT system released 21.3 mgP/ℓ influent and the ENBNRAS system released 18.3 mgP/ℓ influent. When operating with predominantly aerobic P uptake, the UCT system releases more P than the ENBNRAS system does, even though it has a lower anaerobic mass fraction than the ENBNRAS system. However, when anoxic P uptake takes place in the UCT system, the P release drops to lower levels than in the ENBNRAS system. This shows that with anoxic P uptake BEPR in the UCT system (i) less P is released per unit RBCOD than under aerobic P uptake BEPR and (ii) P release decreases also due to the high nitrate load on the anoxic reactor and nitrate recycle to the anaerobic reactor. From Figure 5.11 it can be seen that the P uptake follows exactly the same trend of the P release. The P uptake for the UCT system was 33.2, 26.9 and 50.5 mgP/ℓ influent for sewage batches 13 to 21 (aerobic P uptake), 22 to 27 (anoxic/aerobic P uptake) and 28 to 30 (aerobic P uptake) respectively. That of the ENBNRAS system was 28.1, 35.8 and 41.1 mgP/ℓ influent respectively, with anoxic/aerobic P uptake throughout. For sewage batches 13 to 21, the UCT system P uptake (predominantly aerobic) was about 5 mgP/ℓ influent higher than that of the ENBNRAS system. For sewage batches 22 to 27, when anoxic/aerobic P uptake occurred in the UCT system (20% anoxic P uptake), the P uptake was about 9 mgP/ℓ influent less than that of the ENBNRAS system (64% anoxic P uptake). For sewage batches 28 to 30, when the P uptake in the UCT system had returned to predominantly aerobic P uptake, the P uptake was 9 mgP/ℓ influent higher than that of the ENBNRAS system. On overall average over the 18 sewage batches, the UCT system P uptake was 34.0 mgP/ℓ influent and that of the ENBNRAS system was 32.8 mgP/ℓ.

Figure 5.12 shows the P removal achieved by the UCT and the ENBNRAS systems for sewage batches 13 to 30. In essence the P removal reflects the combination of those tendencies found for the P release and the P uptake. When the UCT system operates with predominantly aerobic P uptake, on average it removes ~4 mgP/ℓ influent more P than the ENBNRAS system. Under conditions where the UCT system does show anoxic P uptake, the ENBNRAS system removes ~2 mgP/ℓ more P than the UCT system. On overall average over the 18 sewage batches, the UCT system removed 12.7 mgP/ℓ influent, while the ENBNRAS system removed 9.8 mgP/ℓ influent. This shows that under normal circumstances the UCT system with predominantly aerobic P uptake BEPR removes ~23% more P than the ENBNRAS with anoxic P uptake BEPR. If however, the UCT system receives an influent that causes a consistent high nitrate load on its anoxic reactor, anoxic P uptake (to a lesser extent than in the ENBNRAS system) occurs, resulting in poorer P removal performance than the ENBNRAS system can achieve when receiving the same influent. A more detailed investigation into the anoxic P uptake BEPR of the ENBNRAS system and the aerobic uptake BEPR of the UCT system is given by Vermande et al. (2000).

FIGURE 5.10:   P release for the UCT and ENBNRAS systems for sewage batches 13 to 30.

FIGURE 5.11:   P uptake for the UCT and ENBNRAS systems for sewage batches 13 to 30.

FIGURE 5.12: P removal achieved by the UCT and ENBNRAS systems for sewage batches 13 to 30.

5.3.4 Sludge settleability

Figure 5.13 shows the DSVI for the UCT and the ENBNRAS systems for sewage batches 13 to 30. The % anoxic P uptake for the UCT system has also been included in the Figure 5.13 to illustrate how the DSVI of the UCT system fluctuates with an increase in % anoxic P uptake. The overall average DSVI of the UCT system over the 18 sewage batches was 138 ml/g and that for the ENBNRAS system was 102 ml/g.

From Figure 5.13 it can be seen that the DSVI of the UCT system fluctuates with the % anoxic P uptake. As the % anoxic P uptake increases, the sludge settleability deteriorates rapidly. This can also be seen in another way: As the nitrate load on the anoxic reactor of the UCT system increases, the nitrate concentration flowing from the anoxic reactor also increases, causing the DSVI to increase (see Casey et al., 1994). From the DSVI of the ENBNRAS system it can be seen that this phenomenon does not occur in the ENBNRAS system. The DSVI of the ENBNRAS does not fluctuate as widely as the DSVI of the UCT system, even though it received the same feed as the UCT system. During sewage batches 21 to 27, where the influent TKN/COD ratio was kept very high consistently, the DSVI of the ENBNRAS system increased slightly from around 90 ml/g to around 105 ml/g, while the DSVI of the UCT system responded by increasing sharply from around 110 ml/g to over 200 ml/g. During sewage batches 13, 14 and 15 the UCT system showed a considerably lower DSVI than that of the ENBNRAS system. However, during this period the ENBNRAS system had not yet recovered from the effects of the toxic sewage

batch 9. and this was the period where the DSVI of the ENBNRAS system was at its highest (i.e. the sludge settleability was at its worst).

FIGURE 5.13:   DSVI for the UCT and ENBNRAS systems for sewage batches 13 to 30.

5.4 CONCLUSIONS

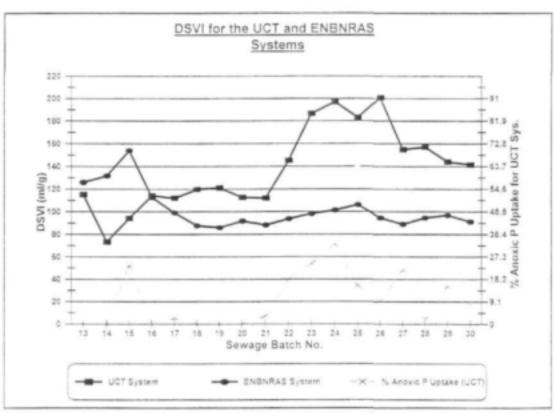A laboratory scale 'conventional' BNRAS system (UCT configuration) with similar design and operating parameters to the 10 days sludge age ENBNRAS system Configuration 3 (Chapters 3 and 4) was run in parallel with the laboratory scale ENBNRAS system. To compare the performance of the two systems, both were fed identical influent sewage for 255 days spanning 18 sewage batches (from 13 to 30). For the purpose of comparing the BNR performance of the two systems, the overall averages for the ENBNRAS system are the overall averages of sewage batches 13 to 30, not the overall averages of sewage batches 1 to 30. The overall averages presented in this Chapter therefore differ from those calculated for the entire Configuration as discussed Chapter 3 and 4. A detailed discussion of this comparison is given by Vermande et al. (2000).

From the results the following conclusions can be drawn:

1. The overall average COD mass balance achieved for the UCT and ENBNRAS systems were 78 and 77% respectively. The COD removal was 93 and 94% respectively. In terms of carbonaceous material removal, the two systems performed identically.

2. The overall average total oxygen demand (including nitrification) of the UCT system was 7625 mgO/d while that of the ENBNRAS system was 1798 mgO/d. By nitrifying externally, the ENBNRAS system requires 76% less oxygen per day; this is a significant difference.

3. The overall average N mass balance for the UCT and ENBNRAS systems were 86 and 87% respectively and the overall average TKN removal 95 and 94% respectively. The effluent TN of the UCT system was 16.8 mgN/ℓ, of which 12.8 mgN/ℓ was NO_x (filtered sample) and 4.0 mgN/ℓ was TKN (unfiltered sample). Of the 4.0 mgN/ℓ TKN, 1.8 mgN/ℓ was FSA (unfiltered sample). For the ENBNRAS system the effluent TN was 9.8 mgN/ℓ, of which 4.6 mgN/ℓ was nitrate (filtered sample) and 5.2 mgN/ℓ was TKN (unfiltered sample). Of the 5.2 mgN/ℓ TKN, 3.5 mgN/ℓ was FSA (unfiltered sample). The ENBNRAS system achieved effluent TN concentrations <10 mgN/ℓ in 10 out of the 18 sewage batches, while the UCT system did not achieve effluent TN concentrations <10 mgN/ℓ in any of the 18 sewage batches. The overall average TN removal for the UCT and ENBNRAS systems were 78 and 88% respectively.

4. In the UCT system an overall average of 21.3 mgP/ℓ influent P was released in the anaerobic reactor. In the ENBNRAS system an overall average of 18.3 mgP/ℓ influent P was released in the anaerobic reactor and internal settler, with an additional P release of 4.5 mgP/ℓ influent in the EN system (which also has to be taken up in the anoxic and aerobic reactors). On overall average, 34.0 mgP/ℓ influent P uptake occurred in the UCT system, and 32.8 mgP/ℓ influent P uptake occurred in the ENBNRAS system.

5. The overall average P removal for the UCT system was 12.7 mgP/ℓ influent (34.0 - 21.3), and the overall average P removal for the ENBNRAS system was 9.8 mgP/ℓ influent (32.8 - 4.7 - 18.3). The UCT system showed only 9.8% anoxic P uptake on overall average, showing that predominantly aerobic P uptake BEPR occurred in the UCT

system. In the ENBNRAS system, 60% of the P uptake occurred in the main anoxic reactor. During sewage batches 21 to 27 anoxic P uptake was induced in the UCT system by feeding influent sewage with high TKN/COD ratios (leading to a high nitrate load on the anoxic reactor). During this period the UCT system showed about 18% anoxic P uptake and the P removal decreased to the same level as measured in the ENBNRAS system.

6. The overall average DSVI for the UCT and ENBNRAS system were 138 mℓ/g and 103 mℓ/g respectively. During sewage batches 21 to 27, where sewage with a high influent TKN/COD ratio were fed to induce anoxic P uptake in the UCT system, the DSVI of the UCT system increased sharply from around 110 mℓ/g to over 200 mℓ/g, while the DSVI of the ENBNRAS system increased only slightly from around 90 mℓ/g to around 105 mℓ/g. This shows that the 'conventional' BNRAS system reacts much more strongly to significant (>2 mgN/ℓ) nitrate concentrations in the outflow of the main anoxic reactor because its aerobic mass fraction is higher (0.5) than that of the ENBNRAS system (0.2). This response to nitrate in the outflow of the main anoxic reactor was also observed by Moodley et al. (1999) in their ENBNRAS system with a higher aerobic mass fraction (0.30), which conforms to the AA filament sludge bulking hypothesis of Casey et al. (1994) (see Chapter 2).

In terms of carbonaceous material removal, the UCT and the ENBNRAS system achieve almost identical results. For the nitrogenous material removal, the ENBNRAS system produces an effluent of better quality with an effluent TN concentration of nearly half that of the UCT system on overall average. The ENBNRAS system produced an effluent with a TN content <10 mgN/ℓ for 10 out of the 18 sewage batches, while the UCT system did not achieve this for any sewage batch. The UCT system, which exhibited predominantly aerobic uptake BEPR, removed about 3 mgP/ℓ influent more P than the ENBNRAS system with anoxic/aerobic P uptake BEPR did. P removal is the only process where the UCT system achieves superior results to that of the ENBNRAS system. The ENBNRAS system BNR is effected by using approximately 75% less oxygen than was required by the UCT system to perform the same BNR. The ENBNRAS system produced a better settling sludge than the UCT system did, and the ENBNRAS system DSVI did not produce a bulking sludge when high nitrate concentrations flowed from the anoxic reactor, as was observed in the UCT system.

CHAPTER 6

IMPLEMENTATION OF EXTERNAL NITRIFICATION BIOLOGICAL NUTRIENT REMOVAL (ENBNR) ACTIVATED SLUDGE SYSTEMS AT FULL-SCALE

6.1 INTRODUCTION

The experimental investigations into the external nitrification biological nutrient removal (ENBNR) activated sludge system at laboratory-scale (Chapters 3, 4 and 5) have clearly demonstrated that this system holds considerable promise for practical implementation. However, successful implementation of the ENBNR activated sludge system at full-scale will depend largely on its cost compared to conventional BNR activated sludge systems and its ability to meet required effluent quality standards. Accordingly, the system should be assessed with respect to the economic cost and effluent quality standards, and these compared to those for an equivalent conventional BNR activated sludge system. In this Chapter such a practical case study is demonstrated based on a specific plant and the effluent quality standards of South African law. Details of the economic evaluation are reported by Little *et al.* (2001).

6.2 SYSTEM SELECTED FOR EVALUATION

The economic cost will be influenced to a large extent by site specific factors. For the purpose of the evaluation, the Potsdam Wastewater Treatment Plant (WWTP) at Milnerton in Cape Town, South Africa was selected, since considerable data are available for this plant (see Little *et al.*, 2001).

The existing Potsdam WWTP comprises two streams which are operated in parallel:

- The old rock media tricking filters (TFs) with a capacity of 18 Mℓ/d. These TFs are low loaded and achieve good COD removal and nitrification. However, no N and P removal is obtained. A chemical P removal plant has been provided but currently (2001) this is not operated.

- The newer conventional BNR activated sludge system (Modified UCT process) with a capacity of 17 Mℓ/d with full biological nutrient removal.

Therefore, the plant has a combined capacity of 35 Mℓ/d and is operated in a conventional way as discussed in Chapter 1 (Fig 1.2). The trickling filter effluent can be treated chemically with aluminum sulphate or ferric chloride for P removal and lime dosing facilities for alkalinity and pH correction are also provided if required.

The effluent of the plant must comply with special standards and achieve filtered COD < 75 mg/ℓ, total suspended solids (TSS) <25 mg/ℓ, free and saline ammonia (FSA) < 2 mg N/ℓ, total

N (FSA – NO$_x$) <10 mgN/ℓ and dissolved ortho-phosphate < 1mgP/ℓ, and thus the BNR activated sludge plant was designed for biological N and P removal. The current influent flow is in the order of 26 Mℓ/d. The 35 Mℓ/d design capacity of the existing plant is expected to be reached by 2005. The sewage influent flow is expected to be 49 Mℓ/d in 2020. Currently the activated sludge plant is achieving the required effluent quality standard. However, the TF effluent, without chemical P removal, fails to comply with the FSA, total N and P standards. New effluent quality standards have been promulgated in 2000 and all WWTPs that depend on size, need to comply with these for their full wastewater flow by 2010. For the Milnerton plant these are filtered COD < 50 mg/ℓ, TSS <15 mg/ℓ, FSA <2 mgN/ℓ, NO$_x$ < 7 mgN/ℓ and dissolved ortho-phosphate < 0.8 mgP/ℓ (see Table 6.3). The new effluent standards were taken account of in the design of different schemes for the extension and upgrade of the Milnerton plant for economic evaluation of external nitrification.

6.3   ECONOMIC EVALUATION

6.3.1   The scenarios for economic evaluation

The engineering and economic evaluation by Little *et al.* (2001) provides system schemes and capital, operation and maintenance costs for the following scenarios:

Scenario 1-greenfields
In this scenario, it is assumed that no plant exists at Milnerton and 2 options with the 2005 required capacity of 35 Mℓ/d were designed. The two options selected to compare the capital and operating costs were:

(1a)   A greenfields conventional BNR activated sludge system (UCT) with 13 day sludge age and a capacity of 35 Mℓ/d, and 15 year period from 2000 to 2015;

(1b)   A greenfields ENBNR activated sludge system with 8 day sludge age and a capacity of 35 Mℓ/d, and 15 year period from 2000 to 2015;

Scenario 2:
This scenario aimed to look at the possibility of expanding the existing plant at Potsdam to cope with the flow that is expected in the year 2020. Full nutrient removal for total wastewater flow was designed for. Two options for the extension and effluent quality required were examined: (1) a conventional BNR activated sludge system which would not make use of the existing trickling filters, and (2) an ENBNR activated sludge system which would use the existing trickling filters. This scenario had two design horizons, viz. 2005 and 2020.

First Horizon (2005) for Scenario 2: The flow is expected to reach 35 Mℓ/d in 2005 from the current flow of 26 Mℓ/d in 2000.

Option 2.1a: Doubling the existing activated sludge plant capacity so that all 35 Mℓ/d can be treated in two parallel 13-day sludge age BNR activated sludge systems (i.e. decommissioning the old trickling filters);

Option 2.1b: Retrofitting the existing plant to an ENBNR activated sludge system with 8 day

sludge age, i.e. upgrading the old trickling filters with plastic media to increase their nitrification capacity and combining them with the existing BNR activated sludge system to obtain BNR on the full capacity of 35 Mℓ/d as in (2.1a) for the period from 2000 to 2005;

Second Horizon (2020) for Scenario 2: The flow is expected to reach 49 Mℓ/d in 2020

Option2.2a: Option 2.1a is extended to accommodate 49 Mℓ/d for the period from 2000 to 2020, i.e. keeping the existing plant unchanged until 2005, after which Option 2.1a is implemented, but with a capacity of 49 Mℓ/d for the period 2000 to 2020.

Option 2.2b: Option 2.1b will be extended to accommodate 49 Mℓ/d for the period from 2000 to 2020, i.e. keeping the existing plant unchanged until 2005, after which, Option 2.1b is implemented, but with a capacity of 49 Mℓ/d for the period 2000 to 2020.

Proposed layouts and preliminary designs for all the options were prepared in sufficient detail for costing and tendering purposes, the prices, in South Africa Rands (in 2000), were obtained for the mechanical equipments and civil construction from contractors. Operation and maintenance costs including staffing, sludge handling and disposal were also costed for the Milnerton plant. The sludge treatment included only dewatering to 20% with belt presses and transport of the dewatered cake to a landfill site 10 km away for disposal, which is the current practice.

6.3.2   Results of economic evaluation

The costs for each of the scenarios are given in net present values (NPV) at 6% in the year 2000 SA Rands. They include all civil construction cost, mechanical construction and operational as well as maintenance cost. The overall costs are summarized in Table 6.1.

Table 6.1:   NPV at 6% for implementation of a conventional BNR or an ENBNR activated sludge plant at the Potsdam WWTP (Cape Town, South Africa).

Projects		NPV at 6% (in million ZAR)	Period
Scenario 1: *35 Mℓ/d Geenfields plant*			
(1a)	Conventional BNRAS system (UCT)	147.44	2000-2015
(1b)	ENBNRAS system	138.25	2000-2015
Scenario 2: *Modified the existing plant of 35 Mℓ/d and extending the plant with 49 Mℓ/d*			
Option 2.1: Design horizon 2005-35 Mℓ/d capacity			
2.1a	Conventional BNRAS (decommission TFs)	93.55	2000-2006
2.1b	ENBNRAS system, using TFs	76.50	2000-2006
Option 2.2 Extending the existing plant to a 49 Mℓ/d for 2020			
2.2a	as for (2.1a) above	164.80	2000-2020
2.2b)	as for (2.2b) above	154.27	2000-2020

Table 6.1 shows that the overall costs of the ENBNR activated sludge system are lower than those for a conventional BNR activated sludge system. From the economic evaluation, it was noted that:

- The ENBNR activated sludge system uses less electricity (lower oxygen demand) and therefore the power costs are less.
- The ENBNR activated sludge system produces more waste sludge (shorter sludge age) which results in higher costs for sludge dewatering, transport and landfill disposal (the current practice at Potsdam).
- The capital cost for the ENBNR activated sludge system are about 20 % cheaper.
- The operating and maintenance costs dominate the overall costs for both the conventional BNR activated sludge system and the ENBNR activated sludge system. These costs make up about 70 % of the total cost, which are essentially the same for both configurations.
- The only significant saving is therefore in the capital costs which are only 30% of the total cost. The external nitrification option is 6%, 18% and 6% cheaper on total cost for Scenario 1 and Options 2.1 and 2.2 of Scenario 2 respectively; while on a percentage basis this is small, in Rand terms it is R9 million, R17 million and R11 million over 20 years respectively, i.e. around R0.5 and R1.0 million per year, which is a significant saving.

6.4 EVALUATION AGAINST EFFLUENT QUALITY STANDARDS

The system selection for BNR is not only dependent on an economic evaluation, but also needs to consider the effluent quality standards required. The South African Department of Water Affairs and Forestry (DWAF) is currently revising the South African effluent discharge standards in terms of the National Water Act of 1998. A brief review of the development of these standards is given below, before discussing the capacity of conventional and external nitrification BNR activated sludge systems to achieve the new proposed standards.

6.4.1 Development of effluent quality standards in South Africa

The Water Act of 1956 requiring treated effluents to be returned to the water catchments from where the water originates, subject to effluent discharge standards. The effluent discharge standards published in terms of the Water Act of 1956 required > 90 % COD removal to avoid deoxygenation of the receiving water bodies as well as nitrification of the FSA to effluent values below 10 mgN ℓ to avoid deoxygenation of, and the toxicity effect on the receiving water bodies.

After in-depth research and development of BNR activated sludge systems for the treatment of municipal wastewater between 1975 and 1983, the special standard for P was promulgated in 1985 and required wastewater treatment plants discharging to sensitive catchment areas to remove P to less than 1 mgP ℓ dissolved ortho-P. Also, research indicated that eutrophication occurring in some water bodies in South Africa was caused by N rather than P, and therefore a special standard for N of < 1.5 mgN ℓ was also promulgated and enforced from 1985 onwards for these water bodies. The effluent quality concentrations of the general and special standards for N and P are listed in Table 6.2. These standards are enforced as uniform effluent standards, and all sewage treatment plants in South Africa irrespective of size must comply with the relevant standards applicable. These standards are still enforced in South Africa today (2000).

The DWAF is currently in the process of revising the effluent quality standards, as the old general standards are not considered to be sufficient in the framework of the newly developed policy of waste load allocation, receiving water quality objectives and minimum requirements. New effluent quality standards have been proposed under tne new National Water Act of 1998. The proposed new standards differentiate between WWTPs with secondary treatment only (i.e. organic material and N removal) and those with advanced secondary treatment (i.e. organic material, N and P) and stipulate separate effluent quality requirements for each. The WWTPs wishing to discharge effluents that are not within the proposed new standards need to apply to the DWAF for special permission to do so. Table 6.3 lists the proposed new effluent quality standards for WWTPs with secondary treatment only and for WWTPs with advanced secondary treatment.

Table 6.2: Effluent quality standards for the general and special standards for N and P.

Parameters	General Standard	Special Standard	Special Standard for P	Units
Filtered COD	<75	<75	<75	mg COD/ℓ
FSA	<10	<1	<10	mg N/ℓ
Nitrate	-	<1.5	-	mg N/ℓ
Phosphate	-	-	<1	mg P/ℓ
Suspended Solids	<25	<10	<25	mg/ℓ
DO	>75	>75	>75	% Saturation
pH	5.5-9.5	5.5-7.5	5.5-9.5	-

Table 6.3: Effluent quality standards proposed under the new National Water Act of 1998.

Parameters	Secondary Treatment Only	Secondary and Tertiary Treatment	Units
COD	65	50	mg COD/ℓ
FSA	3	2	mg N/ℓ
Nitrate	8	7	mg N/ℓ
Phosphate	-	0.8	mg P/ℓ
Suspended Solids	18	15	mg/ℓ

6.4.2 Evaluation of "conventional" BNR activated sludge systems against the effluent quality standards

An evaluation of the existing BNR activated sludge technology (called conventional BNR activated sludge systems to distinguish them from the ENBNR activated sludge system) in South Africa has been performed by the consulting engineering firm, Wates, Meiring and Barnard, Johannesburg, South Africa to assess the measure with which the BNRAS systems as implemented in South Africa over the past 25 years have achieved the existing (Table 6.2) and the new proposed effluent quality standards (Table 6.3).

A total of 17 secondary (2^0) wastewater treatment plants were analysed in terms of the treated effluent quality variables COD, SS, FSA, and nitrate and total of 31 advanced secondary (2^0 +) wastewater treatment were analysed in terms of treated effluent quality variables of COD, SS, FSA, nitrate and dissolved ortho-P. The samples were 24h composites, tested weekly on each plant. The data analysis was based on effluent quality data over a 12 month period which was the same for each plant. Some of the secondary (2^0) wastewater treatment plants have tertiary treatment in the form of chemical P removal, but this aspect of the technology was not evaluated. The number (and %) of plants with median and 95% ile effluent quality variable concentrations exceeding the General Standard effluent discharge concentrations are given in Table 6.4.

Table 6.4: Number (and %) of plants with median and 95% ile effluent quality variable concentrations exceeding the General Standard effluent discharge concentrations.

Effluent quality variable	Standard concentration	Number (%) of plants exceeding stated standard			
		Secondary (2^0) Treatment		Advanced (2^0 -) Treatment	
		Median	95% ile	Median	95% ile
COD	75 mg/ℓ	2 (12%)	10 (59%)	2 (6%)	12 (39%)
SS	25 mg/ℓ	3 (18%)	10 (59%)	3 (10%)	13 (42%)
FSA	10 mgN/ℓ	2 (12%)	9 (53%)	3 (10%)	13 (42%)
Nitrate	²10 mgN/ℓ	4 (24%)	7 (64%)	4 (31%)	54 (6%)
Sol PO₃-P	1.0 mgP/ℓ	-	-	16 (52%)	31 (100%)

*   Because nitrate is not enforced under the existing General Standard requirement, it is not measured for legislative purposes at many plants and therefore the number of the plants in the secondary and advanced secondary treatment samples for which nitrate concentrations are available in the DWAF database are fewer (i.e. 11 and 13 respectively).*

²   *A hypothetical 10 mgN/ℓ nitrate "standard" was assumed for the purpose of this study.*

On the basis that the potential of BNR technology is the performance of the 25ᵗʰ percentile plant (i.e. 25% of plants have better performance and 75% worse), the median and 95 percentile (i.e. achieves this performance 95% of the time) potential performance were calculated, and are listed in Table 6.5.

Table 6.5: Performance potential for the secondary and advanced secondary treatment technologies.

Effluent quality variable	Units	Secondary treatment		Advanced Secondary treatment	
		Median	95% ile	Median	95% ile
COD	mg/ℓ	44	65	32	50
Suspended Solids (SS)	mg/ℓ	10	18	10.	17
FSA	mgN/ℓ	1.5	3.0	1.4	2.9
Nitrate	mgN/ℓ	3.5	6.0	3.2	5.5
Sol PO₃-P	mgP/ℓ	-	-	0.5	1.0

From Table 6.5, the performance potential of advanced secondary treatment is significantly better in effluent quality at the 95% ile only in so far as COD is concerned. i.e. 50 mg/ℓ versus 65 mg/ℓ for secondary treatment; the SS, FSA and nitrate concentrations are virtually the same at about 18 mg/ℓ, 3.0 mgN/ℓ and 6.0 mgN/ℓ respectively. Secondary treatment does not include biological P removal. Advanced secondary treatment, which includes biological P removal, has the potential to achieve at the 95% ile level, an effluent concentration of 1.0 mgP/ℓ dissolved ortho-P.

The survey of operating N and N & P removal activated sludge plants in South Africa indicates that BNR technology, as designed and operated in SA, has the potential to achieve effluent quality concentrations at the 95% ile level of 50 mgCOD/ℓ, 18 mgSS/ℓ, 3.0 mgNH₄-N/ℓ, and 6.0 mgNO₃-N/ℓ and 1.0 mgPO₄³⁻-N/ℓ in SA. Of these, the FSA and P are outside the new standards, which for FSA is 2 mgNH₄-N/ℓ and for P is 0.8 mgP/ℓ. Taking the performance of ENBNR activated sludge Systems 1 and 3c (Chapter 3) as the potential of the ENBNR activated sludge system, the performance of this system is compared with the new effluent standards in Table 6.6.

Table 6.6: Comparison of the performance of ENBNRAS with new standards

Effluent quality parameters	COD mg/ℓ	SS mg/ℓ	FSA mgN/ℓ	NOx mgN/ℓ	P mgP/ℓ
Conventional	50	17	2.9	5.5	1.0
External nitrification					
System 1	51[2]	6.1[2]	3.6	3.9	10.5[1]
System 3b	42[2]	5.4[3]	3.8	3	10.6[1]
New Standard	50	18	2	7	0.8

[1] P removal-effluent P concentration not meaningful because P was dosed to influent to avoid P limitation.
[2] Unfiltered COD
[3] Obtained from (unfiltered COD-filtered COD)/1.48.

From Table 6.6 it can be seen that neither the conventional nor the ENBNR activated sludge system will reach the new FSA standard of 2 mg/ℓ. The ENBNR activated sludge NO₃ concentrations are well within the standard. Achievement of P standard with the ENBNR activated sludge system cannot be commented on because it would depend on the influent P concentration. However, considering that the conventional BNR activated sludge systems develop mostly aerobic P uptake BEPR and achieves 1 mgP/ℓ (Table 6.5), then it is unlikely that the ENBNR activated sludge system, when designed to include anoxic P uptake BEPR, will achieve the new standard, because anoxic P uptake BEPR is lower than aerobic P uptake BEPR (see Chapter 3). If the ENBNR activated sludge system is implemented, it is therefore best to design the system with aerobic P uptake to maximize the P removal.

6.5   CLOSURE

When the project was initiated it was thought that a significant saving in capital cost would be achieved for the ENBNR activated sludge system, as a result of the increase in treatment capacity or smaller biological reactors, the reduction in oxygen demand and a better settling sludge. These appeared to make the ENBNR activated sludge system an attractive and viable alternative as a full-scale wastewater treatment plant. However, the economic evaluation indicated that this may not be the case.

Although the ENBNR activated sludge system does provide a cost saving in construction costs of about 30% compared to a conventional BNR activated sludge system, the operating and maintenance costs in the long term mask this saving because these costs are around 70% of the overall cost and virtually the same for both ENBNR activated sludge systems and conventional BNR activated sludge systems. While significant savings in operation costs are made from the very low oxygen demand, the increased sludge production at a shorter sludge age and the associated increased cost in sludge treatment, transport and disposal offsets this saving. Although the total NPV (capital, operation and maintenance) for the ENBNR activated sludge system is 5 to 10 % lower than that of a conventional BNR activated sludge system, this difference may not be large enough to favour selection of the new and untested ENBNR activated sludge system over the tried and trusted conventional BNR activated sludge system. It must be remembered, however, that the sludge treatment route is specific to the Potsdam WWTP and is a relatively costly alternative, and contributes significantly to the total cost. For other WWTPs, if sludge treatment and disposal costs can be reduced, then this will make the ENBNR activated sludge system more economically attractive.

The most significant advantage of the ENBNR activated sludge system is that it offers biological N and P removal for wastewater treatment plants with existing trickling filters and activated sludge units without an increase in existing process units, enabling these plants to meet the stricter new effluent quality standards proposed under the National Water Act of 1998 by the Department of Water Affairs and Forestry. The ENBNR activated sludge system will provide a feasible and economical upgrading option. Although the ENBNR activated sludge system does not provide a large enough savings in monetary terms to make it as an attractive system, the new effluent quality standards may favour implementation of the ENBNR activated sludge system, because the ENBNR activated sludge system is capable of producing effluents with a quality comparable with (and for effluent N better than) conventional BNR activated sludge plants.

Therefore, it is anticipated that the proposed new effluent quality standards will be the driving force that promotes application of the ENBNRAS system at full-scale, i.e. implementation probably will be legislation driven.

CHAPTER 7

DEVELOPMENT OF A MATHEMATICAL SIMULATION MODEL FOR EXTERNAL NITRIFICATION BIOLOGICAL NUTRIENT REMOVAL ACTIVATED SLUDGE SYSTEMS

7.1 INTRODUCTION

For the design and operation of, and research into, conventional biological wastewater treatment systems, mathematical simulation models have proved to be invaluable as a process evaluation tool (e.g. Dold *et al.*, 1980, 1991; Henze *et al.*, 1987). Mathematical simulation models provide quantitative descriptions of the dynamic behaviour of the wastewater treatment system. By providing quantitative descriptions, they allow predictions of the system response and performance to be made. From the predictions, design and operational criteria can be identified for optimization of system performance. Also, mathematical models are very useful as research tools. By evaluating model predictions, it is possible to test hypotheses on the behaviour of the wastewater treatment system (e.g. biological processes, their response to system constraints, etc.) in a consistent and integrated fashion. This may direct attention to issues not obvious from the physical system, and lead to deeper understanding of the fundamental behavioural patterns controlling the system response. In essence, mathematical models can provide a defined framework which can direct thinking (design, operation or research).

Recognising the usefulness of mathematical models, it was decided to develop a kinetic simulation model that will include the processes that can be expected to be operative in external nitrification BNR (ENBNR) activated sludge systems. This Chapter summarises this development; for details, the reader is referred to Hu *et al.* (2001).

7.2 MODELLING APPROACH

The approach taken to develop the kinetic simulation model for the ENBNR activated sludge system was to (1) evaluate and compare the available existing kinetic models for BNR activated sludge systems; (2) identify difficulties and deficiencies in the models when applied to BNR activated sludge systems in general, and to the proposed system in particular; (3) gather information from the literature and experiments to overcome the difficulties and deficiencies to develop the kinetic simulation model; (4) gather data from the literature to calibrate and validate the resultant kinetic model; and (5) apply the model to simulate the behaviour of the proposed system.

State of the art models identified for evaluation were UCTPHO (Wentzel *et al.*, 1992), IAWQ Activated Sludge Model (ASMNo2) (Henze *et al.*, 1995) and ASMNo2d (Henze *et al.*, 1998), BIOWIN model of Barker and Dold (1997) and the Delft based group of models (Smolders *et al.*,1994 ; Kuba *et al.*,1997). In the initial assessment of these models and the ENBNR activated sludge system, one process immediately apparent to be critical important was anoxic P uptake and denitrification by PAOs, and attention was focussed on this aspect.

7.3 DEVELOPMENT OF ENHANCED CULTURE DENITRIFYING PAOs

In evaluating existing kinetic models for BNR activated sludge systems, the importance of including denitrifying PAOs (DPAOs) and associated anoxic P uptake became evident. To achieve this, information on the kinetic and stoichiometric behaviour of DPAOs is essential. To gather the necessary information, it was attempted to develop enhanced cultures of DPAOs, similar to the enhanced cultures of aerobic P uptake PAOs developed by Wentzel *et al.* (1988, 1989a). However, this endeavour did not prove successful.

The experimental investigation on enhanced cultures of DPAOs indicates that:

• For ENBNR activated sludge systems with small aerobic mass fraction (15%) and short sludge age (10 days) at 20°C, receiving influent supplemented with acetate, the typical P behavioural pattern (i.e. anaerobic P release and anoxic/aerobic P uptake) was completely absent. In a similar investigation, Bortone *et al.* (1997) found it was necessary to include a short aerobic period in an SBR cycle to obtain anoxic P uptake. Thus, it would appear that an aerobic period may be essential for stable anoxic P uptake. Recognizing that anoxic P uptake DPAOs are facultative organisms, including an aerobic period would provide opportunity for aerobic metabolisms by these organisms, in addition to the anoxic metabolism. Biochemically, aerobic metabolism provides a higher energy yield for the organism than anoxic metabolism. This would suggest that anoxic P uptake by itself is an unstable and unreliable process. The inability to stimulate this process under the closely controlled conditions in the laboratory implies that this process would probably also prove unreliable at full-scale. This reinforces the conclusions from the investigation on the ENBNR activated sludge systems (Chapter 3), that overemphasis has been placed on anoxic P uptake in the system. Rather, anoxic P uptake must be stimulated in conjunction with aerobic P uptake.

The requirement for aerobic P uptake to stabilize the BEPR implies that enhanced cultures of DPAOs cannot be successfully developed in the ENBNR activated sludge systems. Accordingly, this investigation was abandoned. Instead, information on DPAOs was sought in the literature.

7.4 MODEL DEVELOPMENT

To select the most suitable simulation model to serve as a starting point for further development, the existing available kinetic models for BNR activated sludge systems were compared and evaluated against experimental observations in the literature. In this evaluation, initially, it was thought that the most complete model would best serve as a basis for model development. The evaluation showed that:

• The BIOWIN (Barker and Dold, 1997a) model appeared to fit this criteria. However, from the assessment of this model and simulations of available experimental data, it became apparent that the model was inadequately validated and that the highly interactive nature of the processes in the model makes calibration virtually impossible; an infinite number of sets of constants can be obtained that give the same net behaviour. Accordingly, it was decided to revert to the most extensively validated kinetic model as

a starting point, namely UCTPHO (Wentzel et al., 1992) and to extend and modify this model to incorporate those processes omitted, or to resolve deficiencies in the model to develop a general BNR kinetic model.

The UCTPHO kinetic model considers only aerobic growth of PAOs, with associated aerobic P uptake. Anoxic growth of PAOs with associated anoxic P uptake and PAO denitrification are not included. However, experimental data demonstrating these processes in BNR activated sludge systems have been more often reported in the literature recently (Ekama and Wentzel 1999b). Furthermore, in the ENBNR activated sludge system, anoxic P uptake dominates P uptake (see Chapters 3, 4 and 5). Accordingly, UCTPHO was modified to extend its application to conventional BNR activated sludge systems where anoxic P uptake occurs and particularly to the ENBNR activated sludge system. For these modifications, information was drawn from experimental data in the literature and the other available kinetic models (e.g. ASM2d, BIOWIN). This indicated that, most likely, two PAO populations exist, one that can denitrify and one that cannot. However, in modelling, incorporating two PAO populations is not possible, as both essentially compete for the same substrate source and thus one population will be predicted to dominate. Thus, the approach followed was to include a single PAO population, but with reduced rates of substrate utilization/growth under anoxic conditions compared to aerobic conditions. With this basic approach, the main modifications made to UCTPHO were:

(1)   Processes for anoxic growth of PAOs with associated anoxic P uptake and denitrification were included, with separate yield coefficients for anoxic and aerobic PAO growth and separate ratios for P uptake per unit PHB substrate utilized. Also, from the above, the anoxic rates were reduced compared to the equivalent aerobic rates, by multiplying the aerobic rates by the reduction factor η_G under anoxic conditions.

(2)   Processes for anoxic death/maintenance of the PAOs were included, recognizing that under anoxic conditions PAOs that can denitrify will behave differently from those that cannot.

(3)   Separate anoxic and aerobic yield coefficients were introduced for OHO growth.

(4)   The organic N and P fractions were linked to the corresponding COD fractions, and the transformations between the different forms of organic N and P fractions were linked to the corresponding COD transformations (Henze et al., 1995).

In developing the model and from the review of existing models, two aspects were identified as of importance, but were not included in the model, namely:

(1)   Anaerobic slowly biodegradable (SB) COD hydrolysis;

(2)   COD loss mechanisms.

With regard to (1), anaerobic SBCOD hydrolysis, this process hydrolyses SBCOD to fermentable readily biodegradable (F-RB) COD (S_{bsf}) under anaerobic conditions. The resultant S_{bsf} can then be acid fermented by OHOs to short chain fatty acids (SCFA, S_{bsa}) which are available to PAOs for sequestration. The net result, thus, is that:

- This process makes SBCOD available to the PAOs and hence contributes to BEPR.

  - In ASM2 and 2d this increases the substrate for PAOs and hence increases P removal;
  - In BIOWIN, to compensate, in the fermentation of S_{bse} to S_{bsca} a substantial COD loss occurs (see below) and hence the net substrate obtained by PAOs does not increase.

Information available in the literature on this aspect is contradictory, with some evidence supporting the process and other evidence indicating that the process is not significantly operative under anaerobic conditions. Until this issue is resolved, the process was omitted from the model, with this omission to be evaluated in model application to experimental data.

With regard to (2), COD loss mechanisms, from experimental data available in the literature it does appear that in BNR activated sludge systems some COD is lost that cannot be accounted for in terms of the measured COD mass balance parameters included in the model (e.g. sludge production, oxygen utilized). However, the mechanisms whereby this loss occurs are not known and the amount of COD loss appears to vary considerably between different systems. Accordingly, it was not possible to include this aspect in the model until greater clarity is obtained, hence it was decided to omit this process. This omission, also, was evaluated in model application to experimental data (see below).

7.5 CALIBRATION AND VALIDATION OF THE PROPOSED KINETIC MODEL

The BNR kinetic model was applied to a wide variety of conventional BNR activated sludge systems operated at laboratory-scale in the UCT laboratory over a range of conditions (influent concentrations, system configurations, sludge age, mass fractions and recycle ratios): these systems were selected because complete experimental data sets are available (Wentzel et al., 1989b, 1990; Sneyders et al., 1998; Kashula et al., 1993; Pilson et al.,1995 and Musvoto et al., 1992). The main objectives of this application were to:

- Evaluate the predictive capacities of the model;
- Evaluate values for the model parameters;
- Calibrate those parameters for which values were not available.

In model application to conventional BNR activated sludge systems, a number of issues of interest were identified and are summarized below.

7.5.1 Model constants

From model application it was apparent that:

- For most of the kinetic and stoichiometric parameters, the values derived from the literature can be retained. This included the values proposed for the new constants introduced into the model to more accurately reflect anoxic behaviour (i.e. OHO anoxic yield, PAO anoxic P uptake PHA utilization ratio).

- For OHO maximum specific SBCOD hydrolysis/utilization rate (K_{mp}), for systems with the "usual" range of aerobic mass fraction (50 -60%), the default value can be retained. However, for systems with small aerobic mass fractions (< 40%) (Kashula *et al.*, 1993; Pilson *et al.*, 1995 and Musvoto *et al.*, 1992) the value has to be increased, otherwise significant unbiodegraded SBCOD is predicted to accumulate in the system contrary to experimental observations. The smaller the aerobic mass fraction, the larger the increase required. Most likely, systems with small aerobic mass fractions favour the growth of OHOs with higher substrate utilization rates (i.e. K_{mp}). Such selection has been observed previously in systems that included selectors (Still *et al.*, 1996).

- For OHO and PAO anoxic reduction factors for SBCOD hydrolysis/utilization (η_H and η_G respectively):

    - These were found to vary significantly from system to system; η_H ranged from 0.1 to 0.7, η_G from 0 to 0.6.

    - The value for η_H appeared to be closely linked to the selected K_{mp} value; where K_{mp} needs to be increased, η_H correspondingly was decreased.

    In searching for reasons for the variability in η_H and η_G, two factors appeared to be of importance:
    - Anoxic mass fraction
    - Nitrate load to the anoxic reactor.

    However, a quantitative relationship between the above two factors (and possibly other design/operational criteria) and the exact values for η_H and η_G could not be established. Accordingly, it is not possible to determine values for η_H and η_G *ab initio*, these will be system specific and require calibration (or measurement) for each situation.

    Clearly, this is undesirable and requires further investigation. Where it is not possible to calibrate (or measure) η_H and η_G for example for design of conventional BNR activated sludge systems with the "usual" range of aerobic mass fractions, it is reasonable to accept $\eta_H = 0.34$ and $\eta_G = 0.39$.

- For the PAO ratio P release/acetate taken up, $f_{P,rel}$, for systems with conventional aerobic mass fractions (50 - 60%) a value $f_{P,rel} = 0.45-0.5$ can be accepted. From the $f_{P,rel}$ and pH relationship of Smolders *et al.* (1995), this corresponds to the anaerobic pH of 6.9 to 7.2, which is reasonable. However, for systems with small aerobic mass fraction, the $f_{P,rel}$ value required significant reduction (<0.4). This reduction concurred with a number of other modifications to the model to correctly predict BEPR in these systems. It was concluded that these systems produced BEPR behaviour that deviated from the "usual" (see below).

7.5.2 Model predictive capacity

For systems with the "usual" aerobic mass fractions of 50 - 60% (Wentzel *et al.*, 1989b, 1990; Sneyders *et al.*, 1998), the BNR activated sludge kinetic model proposed here gave reasonable

predictions. However, due to the deviation in BEPR behaviour in systems with small aerobic mass fractions (Kashula *et al.*, 1993; Pilson *et al.*, 1995 and Musvoto *et al.*, 1992) the predictions were poor. Unfortunately, these system conditions are those that appear to stimulate significant anoxic P uptake. Hence, the kinetic model could not be conclusively validated for BNR activated sludge systems exhibiting predominately anoxic P uptake, by simulation of the response of the Kashula *et al.* (1993), Pilson *et al.* (1995) and Musvoto *et al.* (1992) systems. However, the data set of Wentzel *et al.* (1990) does include more conventional BNR activated sludge systems in which anoxic P uptake did take place; the kinetic model was capable of correctly predicting the observed BEPR in those systems. Thus, it would appear that the model holds potential for application to the ENBNR activated sludge system (see Section 7.6).

7.5.3 COD loss

In developing the kinetic model here, it was thought that COD loss mechanisms were not sufficiently understood for definitive inclusion. Hence, the model does not take COD loss into account. In application of this model to the variety of systems, the data on COD loss appears contradictory. For some systems, excluding COD loss caused the sludge production (VSS) and oxygen demand (OUR) to be overpredicted, whereas for other systems this was not the case. Thus, there is no consistency in the information on the COD loss phenomenon. Hence, until greater clarity on this phenomenon is obtained, it is not possible to include COD loss in the model.

7.5.4 Contribution of PAOs to system denitrification

The model proposed here includes anoxic P uptake and associated denitrification by PAOs. Hence, the PAOs will contribute to system denitrification. It is this behaviour that is sought to be exploited in the DEPHANOX and similar systems, where the concept is that the same substrate source (i.e. the PHA sequestered by the PAOs) can be used for both P removal and denitrification. However, from the simulations and a sensitivity analysis, it was evident that the contribution of PAOs to denitrification is relatively small (<10% for conventional systems; <25% for systems with larger anoxic mass fractions), with the vast majority of denitrification mediated by OHOs. Furthermore, the denitrification by PAOs comes at the cost of lower P uptake and hence P removal efficiency (compared to aerobic P uptake). This would suggest that denitrification by PAOs in the BNRAS system should be minimized rather than maximized. This is in agreement with the experimental observations on the ENBNR activated sludge system (see Chapter 3).

In conclusion, from extensive model application in conventional BNR activated sludge systems, it was demonstrated that:

- The BNR kinetic model developed here is capable of simulating system response and performance of conventional BNR activated sludge systems with "usual" aerobic mass fractions (>50%), including nitrification and denitrification and the two types of BEPR (aerobic and aerobic-anoxic P uptake).

- However, in application to BNR activated sludge systems with small aerobic mass fractions, considerable model manipulation was required to match predicted to observed

behaviour. It appears that the small aerobic mass fractions induce behavioural patterns that are not adequately taken into account in the model, or that the experimental data for these systems deviated from the "usual".

This was investigated further when the model was applied to the experimental data on the ENBNR activated sludge systems gathered in Chapter 3; these ENBNR activated sludge systems also include small aerobic mass fractions (19%) (see below).

7.6 APPLICATION OF THE MODEL TO ENBNR ACTIVATED SLUDGE SYSTEMS

The BNR activated sludge kinetic model was applied to the laboratory scale ENBNR activated sludge systems operated here (Chapter 3), and predicted behaviour compared to experimental observations. This application shows that:

- The model is capable of modelling the system behaviour, with the default values for the kinetic and stoichiometric parameters proposed, except for $f_{P,rel}$, η_{BP}, η_G and K_{mp}. This provides substantive evidence supporting the validity of the proposed kinetic model. This is in contrast to the simulations of conventional BNR activated sludge systems with small aerobic mass fractions (Kashula *et al.*, 1993; Pilson *et al.*, 1995 and Musvoto *et al.*, 1992), where considerable model manipulation was required to correctly reflect the observed BEPR behaviour. This would suggest that the data of Kaschula *et al.* (1993), Pilson *et al.* (1995) and Musvoto *et al.* (1992) deviates from the "usual".

With regard to those constants that required adjustment:

- For $f_{P,rel}$, in model application this had to be reduced from the default value of 0.5 to 0.4; this corresponds to pH = 6.8 in the anaerobic reactor according to the relationship between $f_{P,rel}$ and pH of Smolders *et al.* (1995), which is not unreasonable. In simulating conventional BNR activated sludge systems with small aerobic mass fractions the same reduction in $f_{P,rel}$ was required. Possibly this change reflects that the P release behaviour of denitrifying PAOs differs from aerobic PAOs, as is evident from the observed change in P release/P removal ratios when anoxic P uptake is significant (see Chapter 3), but this is speculation.

- For the η_G value, as in conventional BNR activated sludge systems, this parameter needs to be determined by simulation for each system, even for each steady state period for the same system. Although it appears that two factors may significantly influence the η_G value, namely the nitrate load on the anoxic zone and the aerobic mass fraction, the quantitative relationship between these factors and η_G is still unknown. However, as noted above, the PAOs contribution to denitrification is small compared to the OHOs contribution and hence, with little error, a default value of $\eta_G = 0.5$ can be used for the ENBNR activated sludge systems where η_G is not known; preferably, if possible, η_G should be determined by calibration as above, or by measurement.

- For K_{mp}, the value for this constant had to be increased significantly in application of the

model to the ENBNR activated sludge systems. In these simulations, it was found that the default value for K_{mp} caused significant accumulation of SBCOD (S_{enm}) in the system, which resulted in predicted low OUR and VSS which were not reflected in the observed data. Increasing K_{mp} improved OUR and VSS predictions. This was also observed in simulations of conventional BNR activated sludge systems with similar small aerobic mass fractions above. Most likely, systems with small aerobic mass fractions select for OHOs with higher substrate utilization rates (i.e. K_{mp}). As noted previously, such selection has been observed in systems that include selectors.

- For the η_H value, as for the simulations of conventional BNR activated sludge systems, this constant has to be calibrated in associated with K_{mp} above. The determined values for η_H show some variability for the different wastewater batches. In particular, the values appear to be influenced by the nitrate load on the anoxic reactor. Accepting this variability, the product of η_H and K_{mp} (which is the anoxic OHO SBCOD hydrolysis/utilization rate) appears reasonably consistent with the products determined for conventional BNR activated sludge systems. This would suggest that the OHO anoxic SBCOD utilization rate should be uncoupled from the equivalent aerobic rate. This requires further investigation.

7.7   APPLICATION FOR DESIGN AND OPERATION

After calibration against the ENBNR activated sludge system, the model was applied for system optimization in design and operation. This application demonstrated that the model can be an extremely valuable aid for system design and operation. Two scenarios were investigated, the effect of varying influent TKN/COD ratio and the effect of sludge age. From this investigation, it would appear that:

- Aerobic P uptake in the ENBNR activated sludge system should be maximized with anoxic P uptake correspondingly minimized. Since aerobic P uptake is more efficient than anoxic P uptake, improved P removal can be obtained. This will not be at much expense to denitrification, since the PAO contribution to denitrification is relatively small, and the denitrification that is achieved is nitrate limited.

- The optimum sludge age for the ENBNR activated sludge system is in the range 6-8 days at 20°C, and probably slightly higher at lower temperatures. This corresponds to the maximum P removal.

7.8   CLOSURE

In this Chapter a kinetic simulation model for conventional BNR and ENBNR activated sludge systems has been developed. This model includes the biological processes mediated by the:

- "Ordinary" heterotrophic organisms (OHOs)
- Autotrophic (nitrifiers) organisms (AOs)
- Phosphorus accumulating organisms (PAOs)

The process kinetics and stoichiometry for the OHOs and AOs were taken largely unmodified from the UCTPHO kinetic simulation model (Wentzel *et al.*, 1992), except that the transformations in N and P compounds were linked to the equivalent transformations in the COD compounds (Henze *et al.*, 1995). For the PAOs, the anaerobic and aerobic kinetic and stoichiometric behaviour were also taken from UCTPHO, but additionally, kinetics and stoichiometry for anoxic PAO growth and death (with associated anoxic P uptake and PAO denitrification) were included, based on information available in the literature.

The model was calibrated and applied to an extensive data set from conventional and external nitrification BNR activated sludge systems collected in the UCT laboratory in this and previous WRC funded contracts. The correlation between observed and predicted results was good for the wide variety of systems (including ENBNR activated sludge systems), except for conventional BNR activated sludge systems with small aerobic mass fractions.

This model provides a useful tool for the design and operation of ENBNR activated sludge systems.

CHAPTER 8

DISCUSSION/FUTURE WORK

In the external nitrification (EN) biological nutrient removal (BNR) activated sludge system, the biologically mediated process of nitrification is removed from the main BNR activated sludge system and effected externally in a fixed media system. This is achieved by including in a BNR activated sludge system an internal settling tank between the anaerobic and the primary anoxic reactors; the overflow from this settler is discharged to a fixed media nitrifying system (the EN) and the underflow to the activated sludge system anoxic reactor. The nitrified effluent from the EN system is then discharged to the anoxic reactor to meet with the internal settling tank underflow, and hence is denitrified. Removing nitrification from the activated sludge system eliminates the requirements of a minimum sludge age and aerobic mass fraction for nitrification, and hence allows a step increase in system intensification. The investigations on laboratory-scale ENBNR activated sludge systems by Hu *et al.* (2001), Moodley *et al.* (1999) and Sötemann *et al.* (2000) (Chapter 3) show that implementation of this system is possible in practice. In the three laboratory scale ENBNR activated sludge system investigations, the EN part of the systems nitrified between about 85 and 90% of the free and saline ammonia (FSA) flowing into them (provided failure of the EN system did not occur). Of the total system nitrification, up to 90% occurred externally. Some residual nitrification of the FSA not nitrified in the EN system and the FSA in the internal settling tank underflow (which bypasses the EN part of the system) did occur in the activated sludge system main aerobic reactor, mediated by nitrifiers seeded to the activated sludge part of the system by the EN. However, this is not undesirable as this nitrification ensures a low FSA final effluent quality.

The laboratory scale ENBNR activated sludge systems removed >90% of the influent carbonaceous material (COD), utilising on average about 60% less oxygen than an equivalent 'conventional' BNR activated sludge system. The ENBNR activated sludge systems have also been shown to give excellent TKN and very good TN removals (TKN removals >90%, TN removals >80%). Furthermore, it has been shown that the ENBNR activated sludge systems are capable of producing effluents with TN concentrations of <10 mgN/l for influent wastewaters with TKN/COD ratios of up to between 0.13 and 0.14.

The P removal in the ENBNR activated sludge system was about 30% less than that expected for a conventional BNR activated sludge system. The biological excess P removal (BEPR) occurring in the ENBNR activated sludge systems was undoubtably anoxic/aerobic P uptake BEPR, with the anoxic reactor effecting up to 60 - 70% of the total system P uptake. Reduced BEPR with anoxic P uptake is in agreement with previous observations on conventional BNR activated sludge systems (Ekama and Wentzel, 1997b). The relative magnitude of the anoxic uptake BEPR appears to be dependant on the nitrate load on the main anoxic reactor and the aerobic mass fraction: If the nitrate load was equal to or below the denitrification potential of the main anoxic reactor, the % anoxic P uptake decreased and the % aerobic P uptake increased provided the aerobic mass fraction was sufficiently large to complete the P uptake process. Conversely, when the nitrate load on the main anoxic reactor was greater than the denitrification potential of the main anoxic reactor, the % anoxic P uptake increased, and the % aerobic P uptake decreased. In practise, it is unlikely that a steady state in terms of anoxic P uptake will be reached; the P

uptake will shift from anoxic P uptake to aerobic P uptake and *visa versa*, as the nitrate load on the main anoxic reactor decreases or increases respectively due to variations in the influent wastewater composition. However, from the investigation as the P uptake shifts from predominantly anoxic P uptake to increased aerobic P uptake, the total P removal seems to increase provided the aerobic mass fraction is sufficiently large. Thus, it would not be advisable to implement aerobic mass fractions much smaller than 0.20; even though it is theoretically possible to do so, it would be detrimental to the overall BEPR. Further, aerobic mass fractions less than about 20% seem to promote pin point floc formation, leading to deterioration in effluent quality.

One consistent observation made in all three investigations was that the ENBNR activated sludge systems produced sludges that settle very well (from about 70 to 110 mℓ/g). Furthermore, it seems that these systems are not affected to the same extent as 'conventional' BNR activated sludge systems are by high nitrate concentrations flowing from the main anoxic reactor, as stated in the AA filament bulking hypothesis of Casey *et al.* (1994). The good settling characteristics of the sludges produced in these systems will further facilitate system intensification, by reducing the required surface area of the secondary settling tanks. However, this benefit will be somewhat offset by the requirement of internal settling tanks.

It has further been demonstrated that the ENBNR activated sludge systems perform full and uncompromised BNR for short sludge ages down to about 5 days (Chapter 4). Thus, if the system is implemented in an existing conventional BNR activated sludge system, the sludge age can be significantly reduced. The implication of such a reduction in sludge age is that the influent flow can be doubled to an existing system without a negative impact on the BNR, provided the system does not fail hydraulically due to the increased influent sewage flow. Sludge ages below 10 days have an added advantage in that N and P removals increase per mass of organic load (Wentzel *et al.*, 1990) as the sludge age is reduced.

The comparison of the performance of a laboratory scale ENBNR activated sludge system with that of a parallel laboratory scale 'conventional' BNR activated sludge system (UCT configuration) by Vermande *et al.* (2000) and Sötemann *et al.* (2000) (Chapter 5) demonstrated that the carbonaceous material removal performance of both systems was effectively equal. The TN removal performance of the ENBNR activated sludge system was superior to that of the UCT system; the ENBNR activated sludge system produced effluents with half the TN concentrations of the UCT system final effluent and was capable of producing effluents with TN concentrations of <10 mgN/ℓ, which was not possible with the UCT system. Furthermore, the ENBNR activated sludge system was able to perform total denitrification in the main anoxic reactor, while this was not possible for the UCT system because of the limitation imposed by the a-recycle.

However, the UCT system did show higher BEPR than the ENBNR activated sludge system; it removed on average 3 mgP/ℓ influent more P than the ENBNR activated sludge system, equivalent to about 30% more P removal. This may be ascribed to the predominantly aerobic P uptake BEPR occurring in the UCT system versus the anoxic/aerobic P uptake BEPR that occurred in the ENBNR activated sludge system. In agreement with previous observations on conventional BNR activated sludge systems (Ekama and Wentzel, 1997a), anoxic P uptake is less efficient than aerobic P uptake. This would suggest that to increase system P removal, aerobic P uptake should be encouraged which supports the previous suggestion of a minimum 20%

aerobic mass fraction.

The ENBNR activated sludge system produced a sludge with a DSVI of between 90 and 100 mℓ/g, while the DSVI of the UCT system fluctuated between 80 and 200 mℓ/g. This difference in DSVI became particularly apparent when high nitrate concentrations flowed from the anoxic reactors to the aerobic reactors of the two systems: In response to high anoxic reactor nitrate concentrations, the UCT system's DSVI increased sharply, from about 100 mℓ/g to about 200 mℓ/g, while the ENBNR activated sludge system DSVI increased from around 90 mℓ/g to just over 100 mℓ/g. Hence, the UCT system is much more sensitive to AA filament bulking with significant nitrate concentrations in the outflow of the anoxic reactor than the ENBNR activated sludge system. This is because the aerobic mass fraction of the UCT system was 0.50 and within the range of applicability of the AA filament bulking hypothesis of Casey *et al.* (1994), whereas the ENBNR system's aerobic mass fraction was 0.2 and outside this range.

Initially it was thought that the savings in capital cost brought about by an increased capacity or smaller biological reactors, reduced oxygen demand and better settling sludge would make the ENBNR activated sludge system an economically attractive and viable alternative as a full-scale plant. The economic evaluation of Little *et al.* (2000) (Chapter 6), however, indicates that this may not necessarily be the case. The ENBNR activated sludge system alternative does provide a substantial saving in construction costs of about 30% when compared to a 'conventional' BNR activated sludge system. With regard to operational costs, while the ENBNR activated sludge system does also offer significant savings due to the very low oxygen demand, these cost savings are offset by the increased sludge production at the shorter sludge ages and the associated increase in sludge treatment, transport and disposal costs (for the particular case study and sludge disposal route). This causes that the ENBNR activated sludge system operating costs are about the same as those for a conventional BNR activated sludge system. Since the operating costs of a sewage treatment works, whether ENBNR activated sludge or 'conventional' BNR activated sludge system, account for the bulk of the net present value (NPV), the total NPV (capital, operation and maintenance) for the ENBNR activated sludge system option is only 5 to 10% lower than that of a 'conventional' BNR activated sludge system. This difference may not be large enough for a definite choice of the ENBNR activated sludge system over the 'conventional' BNR system. However, the choice of system may be significantly influenced by the effluent quality requirements. In South Africa, if the Department of Water Affairs and Forestry implement the new effluent quality standards proposed under the National Water Act of 1998, the ENBNR activated sludge will provide a feasible and economical plant upgrade option, particularly for treatment plants with parallel trickling filters and BNR activated sludge systems, or trickling filters only. For the former, the ENBNR activated sludge system offers biological N and P removal for the full wastewater flow without increase in existing process units. For the latter, the existing trickling filters can be integrated into the ENBNR activated sludge system. Furthermore, the ENBNR activated sludge system is capable of producing effluents with a quality that are within the new effluent quality standards, especially with regards to nitrogen; conventional BNR activated sludge systems may have difficulty meeting these standards. Thus, the proposed new effluent quality standards rather than economics may well be the driving force that will see the ENBNR activated sludge system implemented at full-scale.

From the research summarised in this report and discussed above, it is evident that the ENBNR activated sludge system holds considerable merit for implementation. The investigations on the

three laboratory-scale ENBNR activated sludge systems and the comparison with the conventional UCT system provide a comprehensive framework for the understanding of the ENBNR activated sludge system operation and performance, and further laboratory investigations would not provide more knowledge and understanding. The next step would be to begin full-scale trials of an ENBNR activated sludge system. To begin with, a full-scale trickling filter would have to be converted into a nitrifying trickling filter to ascertain the performance of a nitrifying trickling filter at full-scale. Once it has been proven that existing full-scale trickling filters can successfully be converted to nitrifying trickling filters and their capacity determined, the trickling filters can be integrated into a BNR activated sludge system in an ENBNR activated sludge system configuration to obtain BNR on the full influent wastewater flow.

From the discussion above, the next step in development of the ENBNR activated sludge system is implementation and evaluation at full-scale. In particular the behaviour of nitrifying trickling filters need to be assessed - this is best done at full-scale. This will form the basis for a new research contract between UCT and the WRC and WSSA.

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