Comparison of the performance of an external nitrification biological nutrient removal activated sludge system with a UCT biological nutrient removal activated sludge system

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Abstract

A laboratory scale external nitrification (EN) biological nutrient removal activated sludge (BNRAS) system and a UCT BNRAS system with similar design and operating parameters receiving the same influent wastewater were operated in parallel for 234 d to compare their N and P removal performance and to establish the advantages and disadvantages of the ENBNRAS configuration.

For both systems, the COD mass balances, COD removals and filtered and unfiltered effluent COD concentrations were virtually identical, i.e. 78%, 93%, 40 and 50 mgCOD/*l* respectively. However, the oxygen demand in the ENBNRAS system was only 23% of that in the UCT system.

The N mass balances and TKN removals also were virtually the same, i.e. 87% and 94% respectively. The filtered and unfiltered effluent TKN concentrations were somewhat higher for the ENBNRAS system (4.0 and 4.8 mgN/*l*) than the UCT system (2.6 and 4.2 mgN/*l*) due to a slightly higher effluent free and saline ammonia (FSA) concentration (3.5 vs 1.8 mgN/*l*) because nitrification of FSA from (i) the EN part of the system; (ii) that released in the anoxic reactor; and (iii) the internal settling tank underflow, was not complete in the aerobic reactor. The effluent nitrate concentration was significantly lower for the ENBNRAS system (4.0 mgN/*l*) than the UCT system (12.3 mgN/*l*) due to (i) the larger anoxic mass fraction and (ii) nitrification preceding denitrification so that 100% denitrification is possible without nitrate recycling. Consequently, the overall N removal performance of the ENBNRAS system was significantly better [88%, effluent total N (TN= TKN + NO_x) = 9.3 mgN/*l*] than the UCT system (78%, effluent TN = 16.7 mgN/*l*).

The biological excess P removal (BEPR) in the ENBNRAS system was associated with 55-65% anoxic P uptake throughout the investigation and was around 9.7 mgP/*l*. In the UCT system, the BEPR was predominantly aerobic (>95%) for the first half of the investigation and the P removal was 3.6 mgP/*l* higher (13.5 mgP/*l*) than in the ENBNRAS system (9.9 mgP/*l*). When anoxic P uptake (~20%) BEPR was induced in the UCT system by dosing FSA to the influent, the P removal declined and was 1.7 mgP/*l* lower (8.3 mgP/*l*) than the ENBNRAS system (9.9 mgP/*l*).

Dosing FSA to both systems caused the nitrate concentration in the outflow from the main anoxic reactors to increase above 2 mgN/*l*. This stimulated a deterioration in sludge settleability in the UCT system (DSVI from 120 to 200 ml/g), while that in the ENBNRAS system remained very good (80 to 100 ml/g).

Anoxic/aerobic P uptake BEPR appears to be stimulated in BNRAS systems with (i) small aerobic and large anoxic mass fractions and (ii) anoxic reactor nitrate loads greater than the denitrification potential. However, associated with this is a decrease P removal compared with predominantly aerobic (>90%) P uptake BEPR. While anoxic P uptake BEPR often occurs in ENBNRAS systems because conditions (i) and (ii) are usually met, the system can be designed for aerobic P uptake BEPR by countering conditions (i) and (ii) above. However, the more conditions conducive for aerobic P uptake BEPR are created in the ENBNRAS system, the more sensitive its sludge settleability becomes to the nitrate concentration at the anoxic-aerobic transition like in conventional (internal nitrification) BNR systems.

Introduction

The external nitrification (EN) biological nutrient removal activated sludge (BNRAS) system has been investigated in the Wastewater Research Laboratory at the University of Cape Town (UCT). In the ENBNRAS system configuration, the nitrification process is separated from the BNRAS mixed liquor and achieved externally by means of a trickling filter, thereby significantly intensifying the BNRAS system because:

- the sludge age can be significantly reduced since it is no longer governed by nitrification;
- the unaerated mass fraction can be increased to 70% to increase denitrification;
- the oxygen demand of the BNRAS system can be decreased by about 60 to 70%, as the nitrifiers are now located externally; and
- the sludge settleability is significantly improved.

The reduction in sludge age and improvement in settleability increase the treatment capacity of an existing works by ~50% or, alternatively, decrease the required biological reactor volume per M*l* wastewater treated by about $^1/_3$, without impacting negatively on either biological N or P removal. An increase in unaerated mass fraction results in a higher denitrification potential and complete denitrification can be achieved, depending on the TKN/COD ratio of the influent wastewater (<0.12 mgN/mgCOD). Furthermore, a fraction of the additionally available unaerated mass fraction can be added to the anaerobic reactor zone, thereby improving BEPR.

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TABLE 1
UCT and ENBNRAS systems design and operating parameters

Parameter	UCT system	ENBNRAS system
Influent flow (*l*/d)	20	20
Sludge age (d)	10	10
Temperature (^{O}C)	20	20
DO in aerobic zone (mgO/*l*)	2 to 5	2 to 5
Total system volume (*l*)	20	20
Pre-anoxic reactor (*l*)	N/A	2*
Anaerobic reactor (*l*)	3**	5
Main anoxic reactor (*l*)	7	9
Main aerobic reactor (*l*)	10	4
Aerobic mass fraction	0.5	0.2
Anoxic mass fraction	0.35	0.55
Anaerobic mass fraction	0.15	0.25
a - Recycle (w.r.t. influent flow)	1 : 1	2 : 1 (sewage batches 1 to 7) 0 : 1 (sewage batches 8 to 17)
s - Recycle (w.r.t. influent flow)	1 : 1	1 : 1
r - Recycle (w.r.t. influent flow)	1 : 1	N/A

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

Further details on the ENBNRAS system investigations are given by Moodley et al. (1999), Hu et al. (2000, 2001, 2002a, b, 2003), Sötemann et al. (2000) and Vermande et al. (2000 and 2002).

In order to directly compare the performance of an ENBNRAS system with the performance of a 'conventional' (internal nitrification) BNRAS system and to confirm the advantages of a ENBNRAS configuration, laboratory scale UCT BNRAS and ENBNRAS systems, with similar design and operation parameters, were run at the same time in the Wastewater Treatment Laboratory at UCT.

Design and operating parameters

The ENBNRAS and UCT systems were run in parallel on the same wastewater, thereby enabling a direct quantitative comparison of the systems' performances. Both systems were operated in a temperature-controlled laboratory (20^{o}C) at a sludge age of 10 d and had a total system volume of 20 *l*. The dissolved oxygen (DO) concentration was controlled between 2 and 5 mgO/*l* (Randall et al., 1991) and the underflow (s) recycle was set at 1:1 with respect to influent flow for both systems. The aerated and unaerated mass fractions were dictated by the two system configurations. The ENBNRAS system allows for very large unaerated mass fractions, while for the UCT system configuration the unaerated mass fraction is dependent on the nitrification process. Figures 1a and 1b show the layout of the two systems and their design parameters are listed in Table 1. The external nitrification (EN) part of the ENBNRAS system was a suspended medium activated sludge system with its own clarifier (Fig. 1b) because in previous laboratory investigations (Hu et al., 2000; Moodley et al., 1999) the fixed media stone column EN system was plagued with *Psychoda* fly infestations leading to poor nitrification.

The two systems were fed the same Mitchell's Plain wastewater treatment plant influent wastewater, which is the usual source for the UCT activated sludge research. The wastewater was collected in 1.5 m^3 batches, macerated and stored in 400 *l* stainless steel tanks at 4^{o}C. During the 234 d investigation, 17 batches of wastewater were fed, each batch lasting approximately 14 d. The influent was prepared daily by drawing the required volume for both systems from the storage tanks and diluting the raw wastewater with tap water to the target COD concentration of 750 mgCOD/*l*. The daily 40 *l* influent required for both systems (20 *l*/d each) was prepared in the same container to which was added:

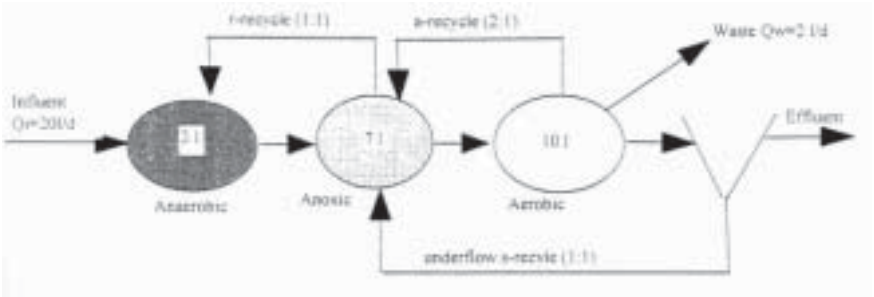

Figure 1a
Schematic layout of the laboratory scale UCT system run in parallel with the laboratory scale ENBNRAS system

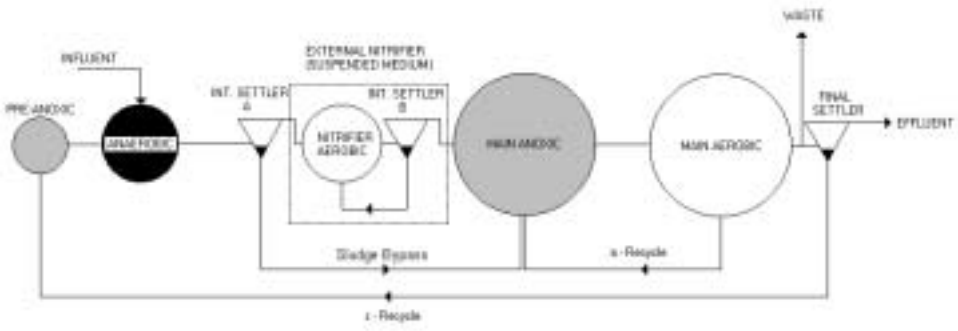

Figure 1b
Schematic layout of the laboratory scale external nitrification biological nutrient removal activated sludge (ENBNRAS) system run in parallel with the laboratory scale UCT system

TABLE 2
Sampling positions and parameter measurement

Test	UCT	ENBNRAS	COD	TKN	FSA	NO_3	NO_2	Tot. P	OUR	DSVI	VSS/TSS	pH
Influent	✓	✓	✪⛛	✪	✪			✪				
Pre Anoxic	N/A	✓				✞	✞	✞			■	
Anaerobic	✓	✓				✞	✞	✞				■
Int. Set. A	N/A	✓	✪'		✪'	✞	✞	✞				
Int. Set. B	N/A	✓	✪'		✪'	✞	✞	✞				
Main Anoxic	✓	✓				✞	✞	✞				
Main Aerobic	✓	✓	◆	◆		✞	✞	✞	■	■	■	■
Final Effluen	✓	✓	✪✞	✪✞	✪	✞	✞	✪✞				

✪ = unfiltered sample
✪' = unfiltered supernatant
⛛ = floc filtered sample
(1 *l* sample subjected to 10 m*l* 0.25 M aluminium sulphate and allowed to settle for a minimum of 10 min)
✞ = filtered sample
(filtered through Schleicher & Schuell 0.45 µm glassfibre filter membrane)
◆ = unfiltered, macerated mixed liquor sample
■ = measurement taken, filtration not applicable

- approximately 10 mgP/*l* potassium di-hydrogen phosphate (KH_2PO_4) to avoid P limitation and to ensure an effluent P concentration of >5 mgP/*l*;
- 1 to 2 teaspoons of sodium hydrogen carbonate ($NaHCO_3$) as a buffer to control the pH values to between 7.0 and 8.0; and
- if the influent TKN/COD ratio was too low, predetermined volumes of 20 gN/*l* ammonium chloride (NH_4Cl) stock solution to achieve the required TKN/COD ratio (~0.11 mgN/mgCOD).

Before division between the two systems, the influent was thoroughly mixed and a sample was taken for analysis. To monitor the steady-state performance of both systems, samples were drawn virtually daily from each of the reactors (and internal settling tanks for the ENBNRAS system) and final effluent for analysis. The parameters measured and the analytical procedures followed are listed in Table 2.

Experimental results

Carbonaceous material removal

Figure 2 shows the overall COD mass balances achieved for the 17 sewage batches for the UCT and the ENBNRAS systems.

The overall average COD mass balances achieved for the UCT and ENBNRAS systems were 78.5% and 77.1% respectively. These values are within 2%, indicating that while the overall average COD mass balance for the UCT system is 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(es) which appear to consume a fraction of the influent COD in the ENBNRAS system without detection with the usual analytical procedures, also occur in the UCT system. Also, it confirms that low COD balances are not characteristics of internal nitrification BNRAS systems alone, but are a characteristic of BNRAS systems in general. 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

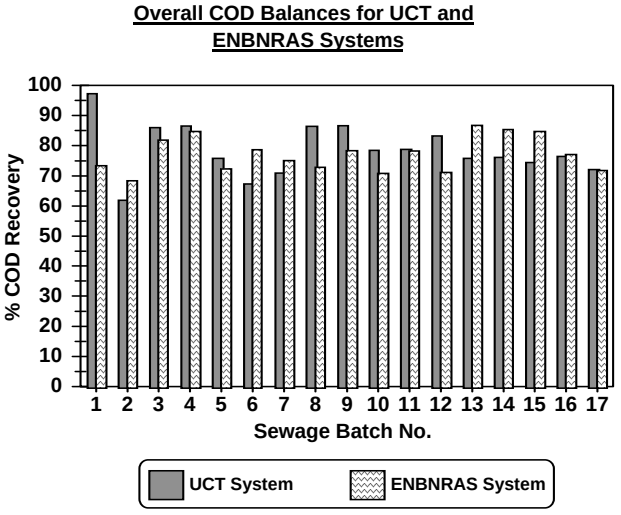

Figure 2
COD mass balances for the UCT and ENBNRAS systems for sewage batches 1 to 17

ENBNRAS system achieving better COD balances for sewage batches with a very high influent TKN/COD ratio (e.g. sewage batches 2, 6, 13, 14 and 15 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 5, 8, 9 and 12 with influent TKN/COD ratios of 0.087, 0.089, 0.107 and 0.085 respectively).

On average over the 17 sewage batches, the UCT and ENBNRAS systems influent COD were 732 mgCOD/*l* and 728 mgCOD/*l* respectively. These are within 1% of each other, confirming that the two systems did indeed receive the same feed even though there were minor variations in the influent COD values for each of the separate sewage batches.

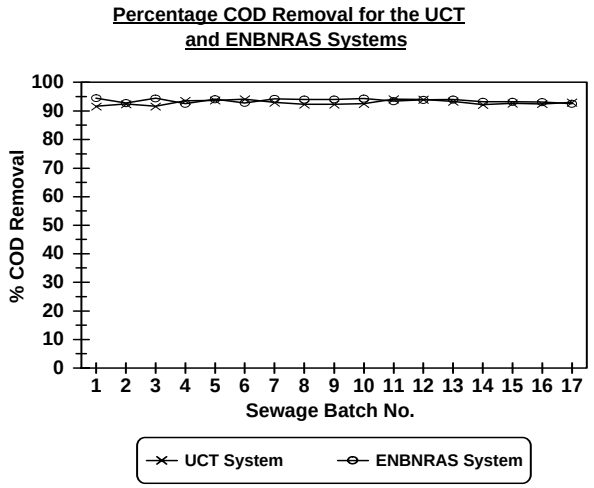

Figure 3
% COD removal by the UCT and ENBNRAS systems for sewage batches 1 to 17

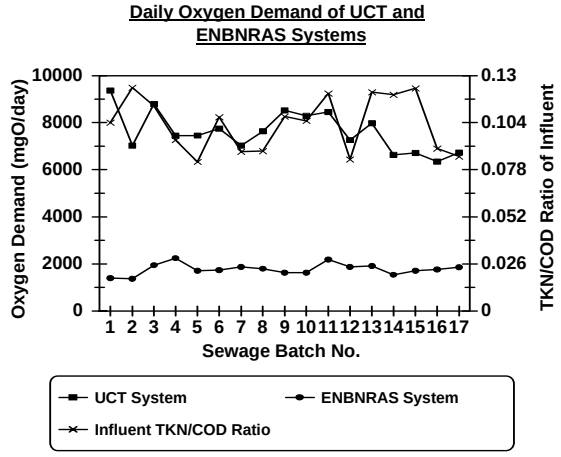

Figure 4
Batch average oxygen demand for the UCT and ENBNRAS systems for sewage batches 1 to 17

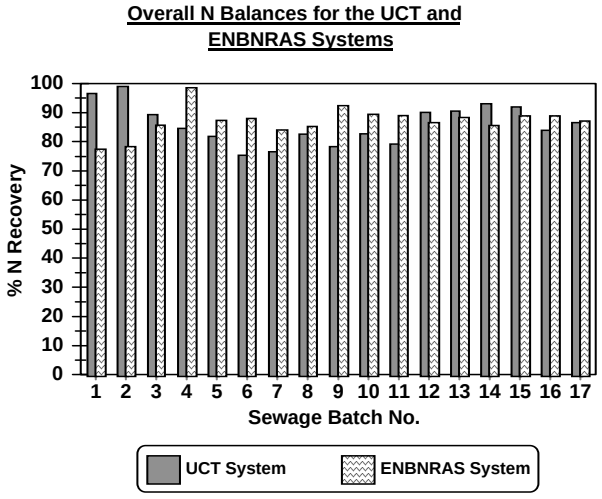

Figure 5
N mass balances for the UCT and ENBNRAS systems for sewage batches 1 to 17

Figure 3 shows the COD removal performance based on unfiltered effluent COD for each of the two systems as a percentage of the influent COD concentration fed to each system. The COD removal performances of the two systems were virtually identical. The UCT and ENBNRAS systems removed an overall average of 92.9% and 93.6% of the influent COD respectively. BNRAS systems generally remove COD virtually completely irrespective of configuration and this is clearly demonstrated here.

The overall average unfiltered and filtered final effluent COD concentrations also were virtually identical for both systems viz. 47.2 and 53.0 mgCOD/*l* and 38.7 and 39.5 mgCOD/*l* unfiltered and filtered COD from the ENBNRAS and UCT systems respectively. The small difference between the unfiltered and filtered final effluent COD concentrations indicated that the effluent suspended solids (ESS) was very low (~8 mgVSS/*l*).

The daily oxygen demand of the main aerobic reactors for the UCT and the ENBNRAS systems are shown in Fig. 4. 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. The advantage of the ENBNRAS system in terms of oxygen demand is clearly demonstrated. The UCT system had an average daily oxygen demand of 7 615 mgO/d over the 17 sewage batches, while the ENBNRAS had an average daily oxygen demand of only 1 778 mgO/d, 77% less. By nitrifying externally, the ENBNRAS system requires only a quarter of the oxygen that the UCT system requires with nitrification taking place internally. The influent TKN/COD ratio is included in Figure 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 increases, 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. In fact, the more nitrate that is generated in the EN part of the system, the higher the denitrification in the anoxic reactor (provided it is not overloaded) and the lower the oxygen demand in the aerobic reactor. This results in a more constant daily oxygen demand for the ENBNRAS system, which can be seen in Fig. 4.

Nitrogen removal

The N mass balances for the 17 sewage batches for the UCT and the ENBNRAS systems are shown in Fig. 5. The overall average N mass balances over the 17 sewage batches for the UCT and ENBNRAS systems were 86.0% and 87.1% respectively. As was the case for the COD balances, the results are very close, but considerably higher than the respective COD balances, which is usual for NDBEPR systems operated in the Water Research Laboratory. 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 1, 2, 4 and 9.

The overall average unfiltered final effluent TKN concentrations were 4.8 and 4.0 mgN/*l* for the ENBNRAS and UCT systems respectively; the filtered values were 4.2 and 2.6 mgN/*l*. Again the small difference between the unfiltered and filtered values confirms that the ESS concentrations were very low from both systems (<10 mgVSS/*l*). The final effluent FSA concentrations were 3.5 and 1.8 mgN/*l* from the ENBNRAS and UCT systems respectively. The difference between the filtered TKN and FSA concentrations is the unbiodegradable organic N and this was very low in the final effluent from both systems (0.7 mgN/*l*).

Figure 6 shows the overall TKN reduction achieved by the UCT and ENBNRAS systems as a percentage of the influent TKN. The TKN reductions achieved by the two systems are virtually identical. The UCT system achieved an overall average TKN reduction of 94.6% and the ENBNRAS system achieved a slightly lower TKN reduction of 93.7%. The reason for the lower TKN removal in the ENBNRAS is the higher FSA concentration in the final effluent compared with the UCT system. The effluent FSA from the ENBNRAS system is higher because some FSA bypasses the EN system in the sludge bypass and because only partial nitrification occurs in the main aerobic reactor, some of this FSA flows out in the effluent without being nitrified. The EN system outflow FSA concentration was on average over the 17 sewage batches 3.3 mgN/l which is similar to the 3.5 mgN/l FSA in the final effluent. Overall, 14% of the nitrate generated in the ENBNRAS system took place in the aerobic reactor. Although the short sludge age (10 d) and small aerobic mass fraction (20%) precluded nitrifiers being sustained in the BNRAS part of the system, nitrifiers from the EN part of the system were seeded into the BNRAS part resulting in partial nitrification in the aerobic reactor.

The difference in the N removal performance of the UCT and the ENBNRAS systems can be seen more clearly from Fig. 7, which shows the total N (TN=TKN+NO_x) concentrations in the effluents of the UCT and the ENBNRAS systems for sewage batches 1 to 17. While both systems removed TKN equally efficiently (Fig. 6), the ENBNRAS system removed significantly more TN than the UCT system. The ENBNRAS system therefore produced a final effluent with a lower nitrate concentration than the UCT system. On average over the 17 sewage batches, the UCT system effluent nitrate was 12.3 mgN/l, while that from the ENBNRAS system was only 4.0 mgN/l.

The ENBNRAS system therefore has the potential to produce effluents with <10 mg/l total N, while the UCT system was not capable of achieving this. The ENBNRAS system achieved an effluent TN concentration of <10 mgN/l for 10 of the 17 sewage batches, while the UCT system did not achieve this for any sewage batch. The influent TKN/COD ratio is also included in Figure 7 to illustrate the variations in the effluent TN concentrations with the variations in the influent TKN/COD ratio. On average over the 17 sewage batches, the UCT system effluent TN concentration was 16.7 mgN/l, while that for the ENBNRAS system was 9.3 mgN/l. The main reason for this difference is the high denitrification potential of the ENBNRAS system due to:

- its larger anoxic mass fraction;
- the fact that denitrification takes place after nitrification so 100% of the nitrate generated in the EN part of the system is discharged to the main anoxic reactor; and
- the low nitrification in the aerobic reactor.

The UCT system cannot denitrify completely because denitrification takes place before nitrification and 100% of nitrate generated in the aerobic reactor cannot be recycled to the anoxic reactor.

The percentage TN removals for the UCT and the ENBNRAS systems for sewage batches 1 to 17 are given in Fig. 8 which shows that the ENBNRAS system removed a greater percentage N (average 88.4%) from the influent wastewater than the UCT system for all 17 sewage batches (average 78.2%).

The extent of nitrification in the aerobic reactor of the ENBNRAS system is governed by the FSA load on it and the concentration of nitrifiers in it, which are influenced by the efficiency of nitrification in the EN part of the system and the system sludge age and aerobic mass fraction. If the sludge age and aerobic mass fraction are such

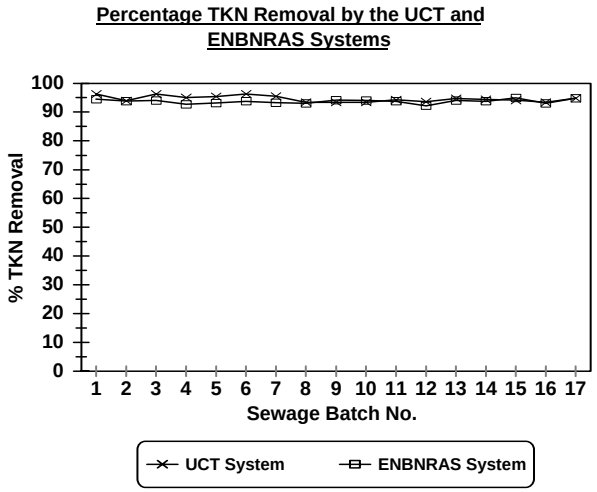

Figure 6
% TKN removal for the UCT and ENBNRAS systems for sewage batches 1 to 17

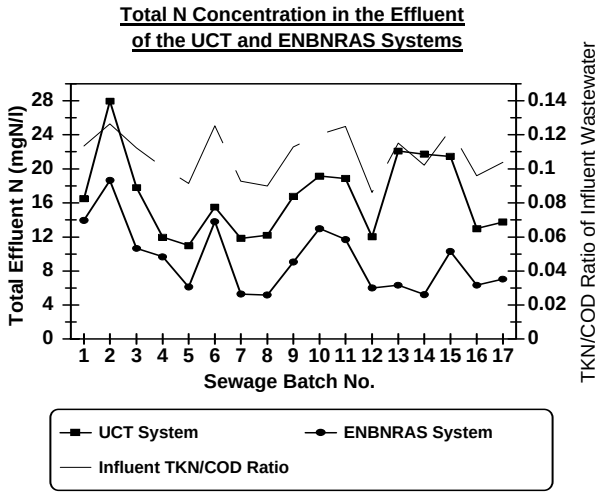

Figure 7
Effluent total N concentrations and influent TKN/COD ratios for the UCT and ENBNRAS systems for sewage batches 1 to 17

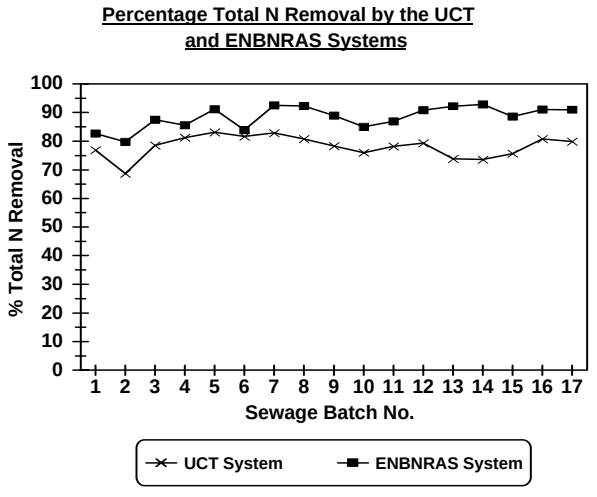

Figure 8
% Total N removal by the UCT and ENBNRAS system for sewage batches 1 to 17

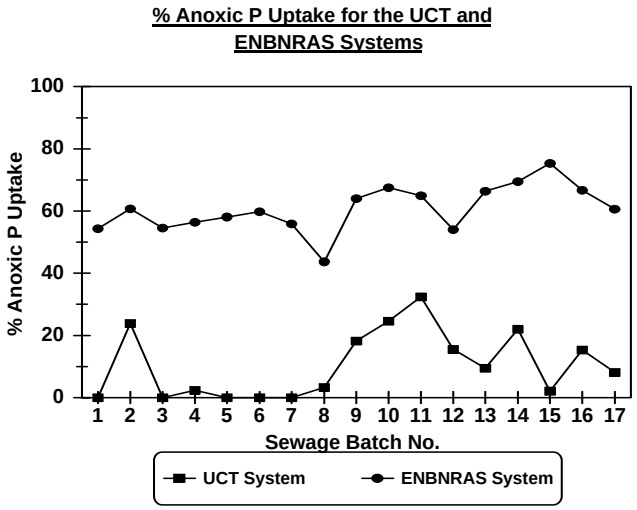

Figure 9
% Anoxic P uptake for the UCT and ENBNRAS systems for sewage batches 1 to 17

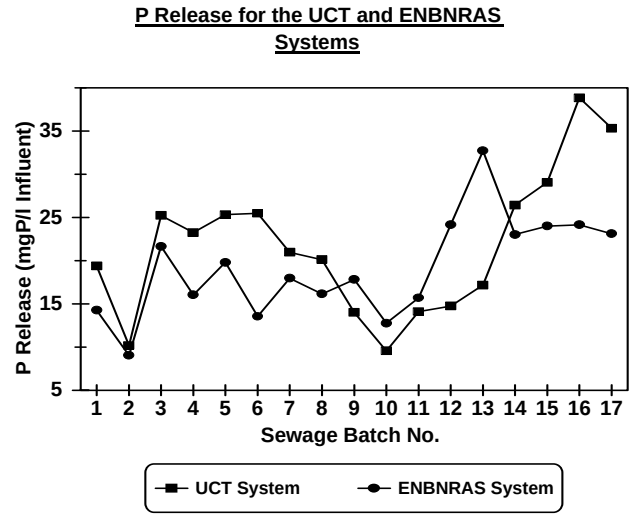

Figure 10
P release for the UCT and ENBNRAS systems for sewage batches 1 to 17

that nitrifiers are not sustained in the BNRAS part of the system, then nitrification in the aerobic reactor will be low because the only nitrifiers in it are those that are seeded from the EN part of the system. If the EN part does not nitrify well, then the FSA load on the aerobic reactor is high and the nitrate load on the anoxic reactor is low. Hence the final effluent FSA will be high and nitrate low. If the aerobic mass fraction is larger (~30%) and/or the sludge age longer, then nitrifiers may be sustained in the BNRAS part of the system and the potential for greater nitrification in the aerobic reactor exists. If the EN system nitrifies well, then the nitrate load on the anoxic reactor is high and the FSA load on the aerobic reactor is low with the result that the final effluent FSA and nitrate will both be low. If the EN system does not nitrify well, then the nitrate load on the anoxic reactor is low, but the FSA load on the aerobic reactor is high. Nitrification in the aerobic reactor will be significant and the effluent FSA will be low but nitrate high. The high nitrate concentration results in nitrate in the return sludge flow. If the nitrate load in the return sludge flow is higher than the denitrification potential of the pre-anoxic reactor, nitrate will enter the anaerobic reactor and reduce the BEPR. In order to counter this effect in the ENBNRAS system during the first two sewage batches when the EN part of the system was not nitrifying well, a 2:1 mixed liquor (a) recycle from the aerobic reactor to the main anoxic reactor was installed (Table 1). This increased the nitrate load on the main anoxic reactor and reduced the nitrate in the final effluent and in the return sludge flow. Although no longer required after the second sewage batch, the a-recycle was maintained until the end of sewage batch 7 to investigate its effect on the system. It was noted during this period that the denitrification potential of the main anoxic reactor was not as high (~50 mgN/l) as observed before in a previous investigation (Hu et al., 2000). So from sewage batch 8, the a-recycle was stopped. This significantly increased the denitrification potential of the main anoxic reactor and had a beneficial effect on BEPR also. It therefore seems that the a-recycle had a negative effect on the N and P removal performance of the system and should be avoided if possible.

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 was fed sewage with a high influent TKN/COD ratio, which resulted in a high nitrate load on the main anoxic reactor, some anoxic P uptake BEPR did occur. For sewage batches 8 to 14, the influent TKN/COD ratio to both systems was kept consistently high (>0.100) with FSA addition to the influent 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 9 shows the percentage anoxic P uptake for both the UCT and the ENBNRAS systems for sewage batches 1 to 17.

From Fig. 9 it can be seen that considerable anoxic P uptake (40 to 70%) occurred in the ENBNRAS system throughout the 17 sewage batches, with an overall average over the 17 sewage batches of ~60%. In the UCT system negligible anoxic P uptake occurred for sewage batches 1 to 8, with the exception of sewage batch 2, which had a very high influent TKN/COD ratio of about 0.123. During sewage batches 8 to 14, where the influent TKN/COD ratio was kept consistently above 0.100, 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 (9 to 14) only 20% anoxic P uptake occurred in the UCT system. This shows that the BEPR in the UCT system was essentially aerobic P uptake BEPR. After sewage batches 8 to 14, 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 10 and 11 show the P release and P uptake respectively for the UCT and the ENBNRAS systems over the 17 sewage batches.

On average over all of the 17 sewage batches, the UCT system released 21.7 mgP/l influent and the ENBNRAS system released 19.2 mgP/l influent. From Fig. 10 it can be seen that for the sewage batches where there was negligible anoxic P uptake in the UCT system (sewage batches 1,3 to 8 and 15 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 2 and 9 to 14) the ENBNRAS system released on average ~3 mgP/l influent more P than the UCT system

did. Thus, 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 took place in the UCT system, the P release dropped to lower levels than in the ENBNRAS system. This shows that with anoxic P uptake BEPR in the UCT system less P is released per unit RBCOD than under aerobic P uptake BEPR; and P release decreases also due to the high nitrate load on the anoxic reactor and nitrate recycle to the anaerobic reactor.

From Fig. 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 35.0, 26.9 and 50.5 mgP/l influent for sewage batches 1 and 3 to 8 (aerobic P uptake), 9 to 14 (anoxic/aerobic P uptake) and 15 to 17 (aerobic P uptake) respectively. That of the ENBNRAS system was 30.1, 35.8 and 41.1 mgP/l influent respectively, with anoxic/aerobic P uptake throughout. For sewage batches 1 and 2 to 8, the UCT system P uptake (predominantly aerobic) was about 5 mgP/l influent higher than that of the ENBNRAS system. For sewage batches 2 and 9 to 14, 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 15 to 17, 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 17 sewage batches, the UCT system P uptake was 34.4 mgP/l influent and that of the ENBNRAS system was 33.7 mgP/l.

Figure 12 shows the P removal achieved by the UCT and the ENBNRAS systems for the 17 sewage batches. In essence the P removal reflects the combination of those tendencies found for the P release and the P uptake. When the UCT system operated with predominantly aerobic P uptake, on average, it removed 3.6 mgP/l influent more P than the ENBNRAS system. Under conditions where the UCT system did show anoxic P uptake, the ENBNRAS system removed 1.7 mgP/l more P than the UCT system. On overall average over the 17 sewage batches, the UCT system removed 12.7 mgP/l influent, while the ENBNRAS system removed 9.9 mgP/l influent. This shows that under normal circumstances the UCT system with predominantly aerobic P uptake BEPR removed 22% more P than the ENBNRAS with anoxic P uptake BEPR. If, however, the UCT system received an influent that causes a consistently high nitrate load on its anoxic reactor, anoxic P uptake (to a lesser extent than in the ENBNRAS system) occured, resulting in poorer P removal performance than the ENBNRAS system.

Sludge settleabiliy

Figure 13 shows the DSVI for the UCT and the ENBNRAS systems for the 17 sewage batches. The percentage anoxic P uptake for the UCT system has also been included in the Fig. 13 to illustrate the increase in DSVI of the UCT system with an increase in percentage anoxic P uptake. The overall average DSVI of the UCT system over the 17 sewage batches was 139 ml/g and that for the ENBNRAS system was 101 ml/g. From Figure 13 it can be seen that the DSVI of the UCT system increases and decreases as the percentage anoxic P uptake increases and decreases.

This behaviour conforms to the anoxic-aerobic (AA or low F/M) filament bulking hypothesis of Casey et al. (1994, 1999). As the nitrate load on the anoxic reactor of the UCT system increases and exceeds the denitrification potential of this reactor, the nitrate concentration in the anoxic reactor increases. With nitrate not limited

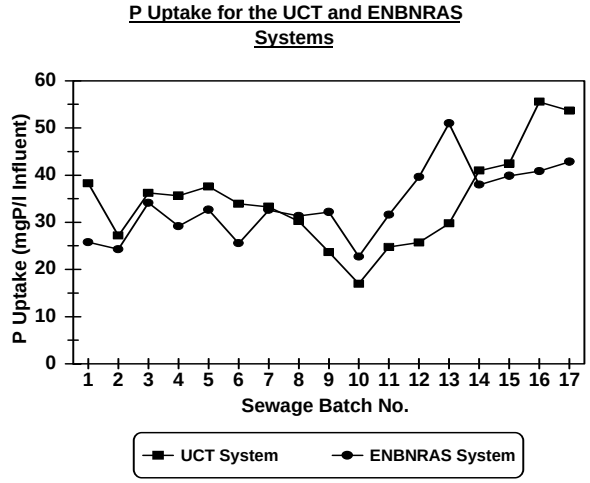

Figure 11
P uptake for the UCT and ENBNRAS systems for sewage batches 1 to 17

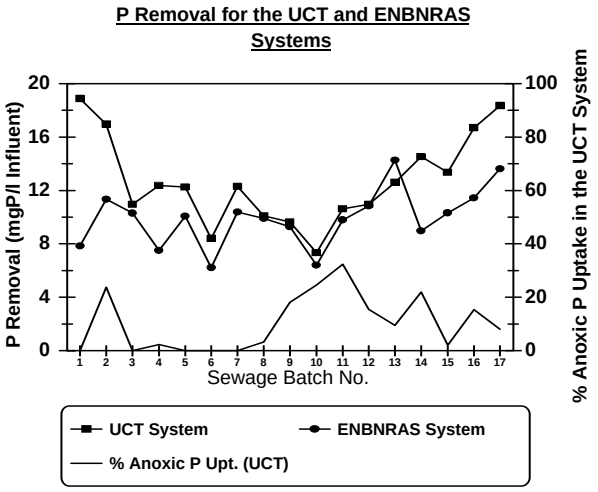

Figure 12
P removal achieved by the UCT and ENBNRAS systems for sewage batches 1 to 17

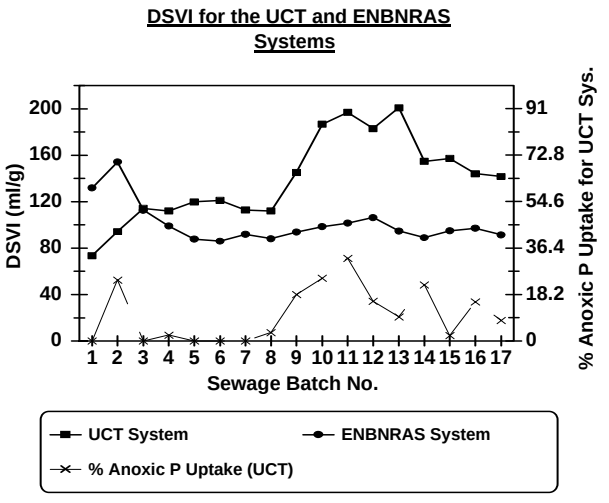

Figure 13
DSVI for the UCT and ENBNRAS systems for sewage batches 1 to 17

in the anoxic reactor, the denitrifying polyphosphate accumulating organisms (DPAOs) find a niche in the system and anoxic P uptake BEPR commences in the system and denitrification is not complete leaving elevated nitrate and nitrite concentrations (>2 mgN/*l*) in the anoxic reactor outflow which stimulates AA filament growth and causes the DSVI to increase (see Musvoto et al., 1994, 1999). From the DSVI of the ENBNRAS system it can be seen that this phenomenon did not occur in the ENBNRAS system. During the period of FSA dosing to the influent (sewage batches 9 to 14), the anoxic reactor outflow nitrate and nitrite concentration from the ENBNRAS and UCT systems were 4.5 mgNO_3-N/*l* and 1.9 mgNO_3-N/*l* and 0.9 mgNO_2-N/*l* and 0.4 mgNO_2-N/*l* respectively. Clearly both systems had elevated NO_X concentrations in the outflow of the anoxic reactor, yet only the DSVI of the sludge in the UCT system increased significantly. Cessation of the FSA dosing to the influent from sewage batch 15 resulted in a decrease in nitrate load on the anoxic reactor of both systems and therefore a decrease in anoxic reactor nitrate and nitrite concentrations viz. 0.8 mgNO_3-N/*l* and 1.0 mgNO_3-N/*l* and 0.7 mgNO_2-N/*l* and 0.3 mgNO_2-N/*l* for the ENBNRAS and UCT systems respectively. The decrease in anoxic reactor NO_X concentration resulted in a decrease in AA filament growth and DSVI (Fig. 13). The DSVI of the ENBNRAS did not vary as widely as the DSVI of the UCT system, even though it received the same feed as the UCT system. During sewage batches 8 to 14, where the influent TKN/COD ratio was kept consistently high, the DSVI of the ENBNRAS system increased slightly from around 90 m*l*/g to around 105 m*l*/g, while the DSVI of the UCT system increased sharply from around 110 m*l*/g to over 200 m*l*/g. During sewage batches 1 and 2 the UCT system showed a considerably lower DSVI than that of the ENBNRAS system, but this was a consequence of the previous experimental conditions for which the systems were used.

Conclusions

A comparison of the ENBNRAS system with a 'conventional' BNRAS system (UCT configuration), both at laboratory scale, demonstrated that the organic material (COD) removal performance of both systems was essentially the same (93%). The filtered and unfiltered final effluent COD concentrations were similar at around 40 and 50 mgCOD/*l* respectively. With nitrification taking place externally, and the denitrification reducing the oxygen demand for COD removal, the oxygen demand in the ENBNRAS system was only 23% of that in the UCT system. Provided the EN part of the system nitrifies completely, the higher the influent TKN/COD ratio, the lower the oxygen demand in the ENBNRAS system relative to that in the UCT system.

Both systems nitrified almost equally efficiently - TKN removal was 93.7% and 94.6% for the ENBRAS and UCT systems respectively. The slightly lower TKN removal in the ENBNRAS system (amounting to 1.8 mgFSA-N/*l*) is a consequence of incomplete nitrification in the aerobic reactor of (i) the FSA in the internal settling tank underflow which bypasses the EN part of the system, (ii) FSA release in the main anoxic reactor during denitrification and (iii) residual FSA from the EN part of the system. However, the TN removal performance of the ENBNRAS system (88%) was higher than that of the UCT system (78%). The ENBNRAS system produced a final effluent with about half the TN concentration (9.3 mgN/*l*) compared with the UCT system (16.7 mgN/*l*). The main reason for this difference is significantly better denitrification in the ENBNRAS system due to its larger anoxic mass fraction; and potential for complete denitrification because nitrification precedes denitrification so that no nitrate recycling is required. The ENBNRAS is capable of producing effluents with TN concentrations of <10 mgN/*l*, while this is not the case for the UCT system.

When the EN part of the system does not nitrify efficiently, the nitrate load on the anoxic reactor is low, and the FSA load on the aerobic reactor is high. Since nitrifiers will be seeded into the BNRAS part of the system, nitrification of ammonia will generally be virtually complete, so the FSA not nitrified in the EN part of the system will be nitrified in the aerobic reactor. To (i) increase denitrification to reduce effluent and sludge return nitrate, (ii) reduce oxygen demand in the aerobic reactor and (iii) prevent nitrate entering the anaerobic reactor, an a-recycle between the aerobic and main anoxic reactors can be installed. While this a-recycle had the above desired effects, it was noted that the system N and P removal performance was not as good as without the a-recycle. It is therefore recommended to instal an a-recycle facility on an ENBNRAS system, but to only operate it when the efficiency of nitrification in the EN part of the system falls below that when the first barrier of protection against nitrate ingress into the anaerobic reactor becomes insufficient, i.e. the nitrate load in the pre-denitrification reactor exceeds its denitrification potential.

The UCT system showed higher BEPR than the ENBNRAS system. The UCT system, when exhibiting >90% aerobic P uptake, removed 3.6 mgP/*l* more (13.5 mgP/*l*) than the ENBNRAS system (9.9 mgP/*l*) which exhibited 55 to 65% anoxic P uptake throughout the investigation. When anoxic P uptake was around 20% in the UCT system (stimulated by dosing FSA to the influent to increase the TKN/COD ratio so that the denitrification potential of the main anoxic reactor was less than its nitrate load), the P removal declined to 8.3 mgP/*l*, and was 1.7 mgP/*l* lower than in the ENBNRAS system (9.9 mgP/*l*). This investigation confirms the observation of Ekama and Wentzel (1999) that anoxic P uptake BEPR yields significantly lower P removal than aerobic uptake BEPR.

The sludge settleability in the UCT system was reasonably good (DSVI ~120 m*l*/g) while the anoxic reactor was underloaded with nitrate and no nitrate (<1 mgN/*l*) was present at the anoxic-aerobic transition (i.e. while exhibiting aerobic P uptake BEPR) but deteriorated to 200 m*l*/g when nitrate (>2 mgN/*l*) was present at the anoxic-aerobic transition (i.e. while exhibiting anoxic/aerobic P uptake BEPR). This behaviour is in conformity with the anoxic-aerobic (AA) filament bulking hypothesis of Casey et al. (1994,1999) and has been observed frequently in UCT systems (Musvoto et al., 1992,1999). Interestingly, while the nitrate concentration at the main anoxic-aerobic transition of the ENBNRAS system also showed similar changes as in the UCT system, the sludge settleability was insensitive to this and remained very good throughout the investigation (DSVI 80 - 100 m*l*/g). Low sensitivity of DSVI at low aerobic mass fractions was speculated by Casey et al. (1994) from zero aerobic mass fraction systems (Hu et al., 2003).

Significant anoxic P uptake (>50%) is a characteristic feature of ENBNRAS systems. The parameters that appear to stimulate it are (i) small aerobic and large anoxic mass fractions which put pressure on the aerobic PAOs, and (ii) nitrate load in excess of the denitrification potential of the anoxic reactor so there is little competition for nitrate between the denitrifying OHOs and PAOs. These features are not exclusive to ENBNRAS systems and hence anoxic P uptake can take place in UCT and other 'internal' nitrification BNR systems.

With aerobic P uptake BEPR the influent RBCOD is lost as a substrate for denitrification because it is taken up in the anaerobic reactor, passes through the anoxic reactor as internally stored polyhydroxyalkanoates (PHAs) in the PAOs and is utilised with oxygen in the aerobic reactor. With anoxic/aerobic P uptake BEPR, part of the influent RBCOD is 'recaptured' as substrate for denitrification-utilisation of PHAs by DPAOs adds to the

denitrification on the slowly biodegradable (SB)COD by the OHOs. In an assessment of the contribution of the DPAOs to denitrification, Hu et al. (2002a) found that the specific denitrification rate of DPAOs on 'RBCOD' [K'_{1PAO} = 0.051 $mgNO_3$-N/(mgPAOVSS.d)] is only 1/3rd of that of the OHOs on SBCOD [K'_{2OHO} = 0.151 $mgNO_3$-N/(mgPAOVSS.d)] and contributed at most 20% to the denitrification even at high anoxic P uptake (>50%). Where P removal has a higher priority than N removal (like in South Africa), anoxic P uptake BEPR therefore is undesirable due to the significant reduction in P removal with which it is associated. When N removal is preferred over BEPR, anoxic P uptake is acceptable because the extra P can be removed by chemical precipitation (De Haas et al., 2001).

So the question is can an ENBNRAS system be designed to exhibit aerobic P uptake BEPR? Yes. To do this requires large aerobic mass fractions (~ 30 to 35%) and a main anoxic reactor underloaded with nitrate, and hence influent TKN/COD ratios not greater than 0.11 mgN/mgCOD to suppress the DPAOs and give sufficient time/space for the aerobic P uptake process to reach completion. However, the more one tries to create aerobic P uptake conditions, the more one moves back to the 'problems' of the conventional internal nitrification NDBEPR system as the conditions become conducive to sustaining nitrifiers and AA filamentous organisms. While the former is not a problem, provided the EN part of the system nitrifies virtually completely, the latter affects the settleability of the sludge. At 30% aerobic mass fraction, Moodley et al. (1999,2000) observed an increased sensitivity of sludge settleability to anoxic-aerobic transition nitrate concentration, which partially undoes some of the advantages of the ENBNRAS system.

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