FULL-SCALE DEMONSTRATION OF FILAMENTOUS BULKING CONTROL AT A BIOLOGICAL NUTRIENT REMOVAL ACTIVATED SLUDGE PLANT

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FULL-SCALE DEMONSTRATION OF FILAMENTOUS BULKING CONTROL AT A BIOLOGICAL NUTRIENT REMOVAL ACTIVATED SLUDGE PLANT

(Mitchells Plain Wastewater Treatment Works, South Africa)

Report to the

Water Research Commission

by

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EXECUTIVE SUMMARY

1 INTRODUCTION

Specific control of filamentous bulking in biological N and N & P removal plants has been undertaken principally at laboratory scale by the Water Research Group (WRG) at UCT under previous contracts with the Water Research Commission (WRC) (K5/542 and K5/286 respectively). This research has shown that the promoted specific control method of selectors, which stimulate removal of influent readily biodegradable COD in anaerobic, anoxic or aerobic selectors by metabolic or kinetic selection, are not successful for controlling bulking in biological N and N&P removal plants. Rather, it was found that the conditions that stimulate biological N removal are conducive to bulking in biological nutrient removal (BNR) plants - stated simply (but not completely) that if denitrification is not complete (nitrate and nitrate concentrations >2 mgN/) at the time conditions switch from anoxic to aerobic, then proliferation of low F/M (renamed Anoxic-Aerobic to more accurately describe the conditions under which they proliferate) filaments takes place.

The filaments most frequently dominant in BNR plants are types 0092, 0041, *Microthrix parvicella* and type 1851. Other types such as 0675 and *Haliscomenobacter hydrossis* were also observed but not at dominant levels. These filaments are classified as typical of the low F/M (now AA) category (Jenkins *et al.*, 1984) and are almost always observed in laboratory (Ekama *et al.*, 1996) and full scale (Blackbeard *et al.*, 1986, 1988) ND and NDBEPR systems. During changes in sludge settleability (DSVI) in the systems, the filament types did not change significantly and therefore the different DSVIs were caused by essentially the same filament types at different levels of abundance. From the earlier research, and the AA filament bulking hypothesis that was developed from it, the nitrate plus nitrite concentrations (NO_x, mgN/) at the anoxic aerobic transition need to be low (< 0.5 mgN/) to obtain reasonably good settling sludges.

A large body of laboratory scale evidence of an indirect nature supporting this alternative explanation for bulking in nutrient removal plants has been accumulated in the previous two research contracts. Some direct support for the hypothesis has recently been provided by Tandoi *et al.* (1998), who found that an isolate of *M. parvicella* from an Italian activated sludge (AS) plant, which was identical to Australian isolates, could reduce nitrate to nitrite but could not denitrify nitrate to nitrogen gas. Also, increasingly more indirect supportive evidence from full scale nutrient removal plants in Europe has been appearing in the literature. The observations of Eikelboom (1994) and Foot *et al* (1994) summarize this anecdotal evidence particularly well (see Section 1 below): The problem being worse in spring, worse with settled wastewater, an inverse relationship between total filament length and effluent total oxidized nitrogen, the efficacy of selectors decreasing if denitrification is permitted to take place in the main reactor (due to underaeration or plant load increase) are all factors that influence the denitrification performance of the plant and increase the likelihood of significant nitrate and nitrate concentrations being present at the transition from anoxic to aerobic conditions and therefore provides indirect support for the bulking hypothesis. Although not unequivocally, this was shown to be the case also in several laboratory investigations undertaken in the Water Research laboratory at UCT over the past 10 years.

2. FULL SCALE DEMONSTRATION OF FILAMENTOUS BULKING CONTROL IN THE MITCHELLS PLAIN BNR PILOT PLANTS

2.1 Objectives

Specific control of filamentous bulking in biological N and N & P removal plants has been undertaken principally at laboratory scale. This research has shown that the promoted specific control method of selectors, which stimulate removal of influent readily biodegradable COD in anaerobic, anoxic or aerobic selectors by metabolic or kinetic selection, is not successful in controlling bulking by biological N and N & P removal plants (Ekama *et al.*, 1996). It was found that the conditions that stimulate biological N removal are conducive to bulking, *viz.* sequencing of anoxic and aerobic conditions. Hence, the low F/M filaments were renamed anoxic-aerobic (AA) (Casey *et al.*, 1994), to more accurately reflect the conditions under which they proliferate. From a review of the experimental results collected at laboratory scale, a hypothesis on AA filamentous organism bulking was developed - stated simply (but not completely), if denitrification is not complete (nitrate and/or nitrite concentrations > 2 mgN/) at the time conditions switch from anoxic to aerobic, then proliferation of AA filaments takes place. Application of this specific method to control AA filamentous organism bulking derived from the hypothesis of Casey *et al.* (1994) needs to be demonstrated at full scale to check the validity of this new specific bulking control approach. The main aim of this research project was to demonstrate at full-scale this specific control methodology.

A second important phenomenon observed repeatedly in the laboratory-scale investigations on NDBEPR systems is that in these systems low COD mass balances (80 to 90 %) were observed. In contrast, in nitrification-denitrification (ND) systems (*i.e.* no anaerobic reactor) invariably the COD mass balances are much higher (90-95 %) and closer to the theoretically expected 100 %. The mechanism whereby this COD loss has taken place has not been identified scientifically yet but is assumed to be via the fermentation process which transforms the readily biodegradable (RB) COD to volatile fatty acids (VFA) in the anaerobic reactor - instead of an equal concentration of VFA being generated from the RBCOD, only a fraction (~50 %) of the RBCOD becomes VFA, the balance is COD that is lost. The fermentation process has been selected for this COD loss because it is one of the few biological processes confined to the anaerobic reactor. The second aim of this investigation was to establish whether or not this inexplicable COD loss observed in laboratory scale systems also takes place in large scale systems.

2.2 Pilot plant set-up, operation and monitoring.

For this purpose, two parallel pilot plants at the Mitchell's Plain Wastewater Treatment Plant were made available by the Cape Metropolitan Council (CMC). The two pilot plants (Modules A and B) were redesigned as N & P removal UCT configuration systems, such that the two systems had the same design and operating parameters. The sludge age was 15 days and the anaerobic, anoxic and aerobic mass fractions 0.091, 0.350 and 0.559 respectively. The treatment capacity of the two modules was estimated to be 2.0 and 2.7 M /d for Modules A and B respectively but due to aeration capacity limitations of the old fine bubble ceramic dome aeration system, the flows were reduced to 1.4 and 2.1 M /d. The idiosyncrasies of this old aeration system and persistent problems with the sludge back flow into the pilot plants from the sludge treatment facilities resulted in the anoxic and aerobic reactors not functioning as such (leading to poor anoxic denitrification and significant aerobic simultaneous nitrification-denitrification) and poor sludge age control. These problems compromised achieving the two research objectives and ultimately forced cessation of the investigation after 589 days.

To test the AA filament bulking hypothesis at large scale, in the one system (Module B), the nitrate/nitrite concentration in the anoxic reactor would be controlled to be > 2 mgN/ and hence in terms of the hypothesis should bulk, while in the other parallel system (Module A) it would be controlled to < 1 mg and hence in terms of the hypothesis should not bulk. To control the nitrate/nitrite concentrations at the transition from anoxic to aerobic conditions, the 'a-recycles' (aerobic to anoxic) were varied to under and overload with nitrate the anoxic reactors of Modules A and B respectively.

In the 589 day investigation, the pilot plants were sampled routinely once weekly by the CMC and an additional 3 times weekly specifically for this project. From an examination of the influent, operational parameters and measured system performance, eight 'steady state' long term periods where these remained approximately constant were identified. An identical laboratory scale system (~1:200000) was operated for 163 days from day 470 to day 632 (43 days after the termination of the pilot plant investigation) to evaluate the magnitude of the aeration and sludge wastage problems on the results of the pilot plant.

2.3 Pilot plant performance

2.3.1 COD removal

Over the 589 day investigation the average (for the eight 'steady state' periods) percentage COD removal was 87 and 89 % for Modules A and B respectively. The COD mass balance of the two modules over the investigation could not be determined because steady state conditions were not achieved due to waste sludge ingress from the main treatment plant and therefore the oxygen utilisation rates (OUR) were not measured.

2.3.2 Nitrification, denitrification and N removal

For the first two periods (332 days), the nitrification performance was reasonable, with effluent FSA concentrations at about 1 mgN/ . For all the subsequent periods nitrification was partial only, indicated by the relatively high effluent TKN and FSA concentrations (5 - 30 mgN/). For most periods, the effluent nitrate concentrations were low (10 to 17 mgN/) in Module A and B and indicates that denitrification in the system was very good (~50 - 55 mgN/), except for Periods IV and V Module A where nitrification stopped (< 2 mgN/) due to aeration problems. The nitrate concentrations in the anoxic reactors of both modules were higher than expected (5 - 8 mgN/) which resulted in (i) nitrate feed back to the anaerobic reactor and (ii) not achieving low nitrate at the anoxic-aerobic transition in Module A. Reducing the 'a-recycle' ratios on Module A and B from 2:1 and 5:1 to 0:1 and 3:1 respectively did not solve this problem. Later in the investigation it became apparent that these high anoxic reactor nitrate concentrations were due to air leaks in the air distribution network and back mixing from the aerobic reactor, in particular in Module A.

To examine the nitrification and denitrification performance, nitrate and nitrite mass balances were calculated around each reactor and SST from the results. Net denitrification in the anoxic zones was very low, on average only 2 and 23 mgN/ influent for Modules A and B respectively. Compared with the influent TKN of 95 to 98 mgN/ , with such low denitrification it is impossible to achieve effluent nitrate concentrations between 10 and 17 mgN/ unless significant simultaneous nitrification-denitrification was taking place in the anoxic and aerobic reactors. This was confirmed during Periods VII and VIII when the air supply to Module A was increased and resulted in a net nitrate production (nitrification) in the anoxic reactor. While such

simultaneous N removal can have advantages, it comes at considerable risk to nitrification, indicated by the relatively incomplete nitrification. The overall average N removal was 77 % for Modules A and B. The average N balances calculated for Modules A and B were 57 % and 76 % respectively due to the simultaneous nitrification-denitrification in the anoxic and aerobic reactors (in the N balance calculations it is assumed that no denitrification takes place in the aerobic reactor and no nitrification in the anoxic reactor). Operation of the laboratory scale UCT system also confirmed simultaneous ND - this system yielded a good (99 %) N balance (verifying that no simultaneous ND took place in it) and its effluent nitrate concentration was 38 mgN/ , double that of the pilot plants. Furthermore, the lab system VSS and TSS concentrations were ~40 % lower than those in the pilot plants indicating that sludge ingress was substantial.

2.3.3 Biological P removal

With the exception of Module A Period III, the filtered (< 0.45 m) effluent TP concentration > 2 mgP/ , indicating that the P removal was not limited by the influent P concentration. This allowed the overall average system P removal capacity for Modules A and B to be measured, which based on unfiltered influent and filtered effluent TP was 10.0 and 9.3 mgP/ influent respectively. The P removal attained was considerably reduced below that potentially achievable due to the high concentration of nitrate recycled to the anaerobic reactors; on average at 5.6 and 5.0 mgN/ for Modules A and B respectively. This reduced the P removal by an estimated 3 mgP/ . Nevertheless, consistently good biological P removal was obtained at 0.015 and 0.014 mgP/mg influent COD for Modules A and B respectively. In contrast to the N mass balance, the overall average P mass balance for Modules A and B were excellent, at 103 and 97 % respectively.

The average anaerobic P release was very similar in both modules at 26 and 29 mgP/ influent for Modules A and B respectively. The average aerobic P uptake in Module B is significantly higher than in Module A at 45 compared with 19 mgP/ influent respectively. In the anoxic reactors the two modules appeared to exhibit divergent behaviour. Throughout the investigation P uptake occurred in the anoxic reactor of Module A, while in Module B P release occurred in the anoxic reactor. For Module A, over the investigation 48 % of the P uptake occurred in the anoxic reactor. Initially it was thought this was anoxic P uptake in Module A. Hu *et al.* (2001) noted that anoxic P uptake tends to be stimulated by an overload of nitrate on the anoxic reactor and has been observed quite often in laboratory-scale and full-scale plants. When it takes place, the BEPR is only two thirds to three quarters of BEPR with predominantly (>90 %) aerobic P uptake (Ekama and Wentzel, 1999). So its occurrence in Module A is apparent and not real because (i) both modules were overloaded with nitrate, (ii) had similar average anoxic nitrate concentrations and (iii) aerobic P uptake took place in Module B and (iv) the BEPR was the same in both modules. Moreover, when the air supply to Module A was increased (at the expense of Module B), the anoxic reactor switched from poor denitrification (2 mgN/) to nitrification (5 mgN/) confirming that the reactor was more aerobic than anoxic. Although not apparent during the investigation, the occurrence of P uptake in the anoxic reactor therefore is best explained as aerobic P uptake as a result of excessive DO ingress into the Module A anoxic reactor. Thus for Module A the experimental evidence suggested P uptake by PAOs under aerobic conditions rather than under anoxic conditions. This was later confirmed after termination of the investigation - many air leaks were found in the air distribution network passing through the anoxic reactor.

To determine the unbiodegradable particulate COD fraction ($f_{S,up}$) of the sewage fed to the pilot plants, the appropriate $f_{S,up}$ value was selected so that the system VSS mass calculated with the BEPR model of Wentzel *et al.* (1990) was equal to the measured VSS mass using the measured influent readily biodegradable COD (RBCOD) concentration, the influent characteristics of the sewage (*i.e.* $f_{S,us}$ and total influent COD) and the known system parameters (anaerobic mass fraction and sludge age) as input taking due account of the nitrate recycled to the anaerobic reactor. An overall average $f_{S,up}$ value of 0.239 was estimated. The P content of the PAOs ($f_{XBG,P}$) was estimated as that value which set the calculated P removal equal to the measured P removal. An overall average $f_{XBG,P}$ value of 0.322 mgP/mgPAOAVSS was estimated for the pilot plants. A $f_{S,up}$ value of 0.239 for the settled wastewater is far too high and is the result of the sludge ingress from the main treatment plant. Accordingly a more realistic $f_{S,up}$ of 0.05 was selected for the settled wastewater (the lab-scale system yielded a $f_{S,up}$ of 0.03) and the calculations repeated. The $f_{S,up}$ of 0.05 yielded a PAO P content ($f_{XBG,P}$) of 0.39 mgP/mgPAOVSS which is very close to the Wentzel *et al.* (1990) model standard of 0.38 for 100 % aerobic P uptake BEPR, confirming that the BEPR in the pilot plants was normal aerobic P uptake BEPR, and as high as can be expected.

2.3.4 Sludge settleability and filamentous organisms

The filament most frequently dominant in both Module A and B was *M. parvicella* (41 and 50 % respectively). The next most frequently dominant filaments were type 0092 and type 1851. All of these filaments are classified as typical of the low F/M category (Jenkins *et al.*, 1984) later renamed Anoxic-Aerobic(AA) (Casey *et al.*, 1994) and are almost always observed in full scale NDBEPR systems whether bulking or not (Blackbeard *et al.*, 1986, 1988). In both modules the anoxic nitrite concentrations were very low throughout the investigation with an overall average of 0.17 and 0.25 mgN/ for Modules A and B respectively. In both modules anoxic nitrate concentrations were high, with an overall average of 5.36 and 5.17 mgN/ for Modules A and B respectively. The high nitrate concentrations in the anoxic reactors indicated that the proposed control of nitrate in the Module A anoxic reactor to low concentrations had not been achieved in practise, even though the 'a-recycle' was set to zero from Period III onwards. However, the proposed control of nitrate in the Module B anoxic reactor to high concentrations had been achieved in practise. Thus, while it was not possible to demonstrate the effect of complete anoxic denitrification on sludge settleability, the effect of incomplete anoxic denitrification could be demonstrated. However, because the intended anoxic and aerobic conditions were not achieved in the anoxic and aerobic reactors, and these reactors were a continuous quasi anoxic quasi aerobic reactor, so it is difficult to apply the AA filament bulking hypothesis to the results. In terms of the hypothesis and the bulking control strategy, since both modules have high anoxic nitrate concentrations both should produce a bulking sludge due to AA filament proliferation. In contrast both modules produced very good settling sludges with low overall DSVIs of 97 and 102 m /gTSS and at no time during the investigation did the 7 day moving average DSVI exceed 120 m /gTSS in both modules. The quasi anoxic quasi aerobic conditions in the pilot plants would be similar to those in Carousel, Orbal and ditch type ND systems, which often produce poor settling sludges, yet the sludge in the pilot plants settled well. To compound the matter further, the laboratory scale system fed the same wastewater with 0:1 'a-recycle' like Module A but with defined anoxic and aerobic conditions and high anoxic to aerobic transition nitrate concentration (2.1 mgN/) produced a poor settling sludge (DSVI > 250 m /gTSS), as expected from the AA filament bulking hypothesis.

2.3.5 Simulation of pilot plant performance

By giving the same influent flow and concentrations and system design parameters as input, the system performance was modeled with 2 NDBEPR simulation programmes, *UCTPHO* (Wentzel *et al.*, 1992) and *BIOWIN* (Envirosim & Associates, Canada, 2001 release), the first representing models based on 100% COD balance and the latter representing models that have included the inexplicable COD loss. Because the modules could not be operated with well defined aerobic and anoxic conditions and without sporadic sludge ingress, the measured results cannot be used to comment on the model predictions. Therefore, the second objective, that of establishing whether or not the unexplained COD loss observed in laboratory scale NDBEPR systems also takes place in large scale systems, it could not be validated.

2.3 Conclusions

Despite the significant expenditure by the CMC to modify and refurbished the pilot plants to UCT NDBEPR systems and major effort in operating, monitoring and analysis of the pilot plants, the persistent problems with the aged fine bubble aeration system, back-mixing of mixed liquor across the anoxic-aerobic reactor baffle and the backflow of sludge from the main treatment plant, prevented achieving the project objectives.

(1) COD mass balances could not be done so no additional information could be obtained on the to date unexplained COD loss in NDBEPR systems.

(2) From the way the pilot plants actually operated, i.e. significant simultaneous ND in the quasi-anoxic quasi-aerobic conditions of the anoxic and aerobic reactors, in terms of the AA filament bulking hypothesis, AA filament bulking should have taken place in both modules, but didn't. Orbal, Carousel and other ditch type ND systems also have such simultaneous ND conditions and often have bulking sludge. Curiously, the parallel UCT system did produce an AA filament bulking sludge. Clearly finding a cure for AA filament bulking in NDBEPR systems remains elusive despite the considerable South African and international research attention it has received over the past 20 years.

3 BULKING CONTROL WITH REDOX POTENTIAL

3.1 Objectives

Casey *et al.* (1992, 1994) proposed an hypothesis for the cause of bulking by the low F/M (Food/Microorganism) filamentous organism group (after Jenkins *et al.*, 1984) in biological nutrient removal (BNR) activated sludge (AS) systems. In terms of this hypothesis, it is proposed that the floc forming organisms are able to reduce nitrate to di-nitrogen gas through the denitrification intermediates NO_2, NO and N_2O. If denitrification is incomplete with the onset of subsequent aerobic conditions (i.e. nitrate and nitrite are observed to be present during the anoxic to aerobic transition), then the intracellular gaseous denitrification intermediates NO and N_2O (particularly NO) inhibit the aerobic cytochrome o in the flocformers, and hence inhibit their aerobic substrate utilisation (Casey *et al.*, 1999). In contrast, the filamentous organisms, in particular *Microthrix parvicella*, but also some others in this low F/M group (renamed Anoxic Aerobic or AA as more descriptive of the conditions under which they proliferate), are hypothesized to reduce nitrate only to nitrite and therefore do not accumulate NO and N_2O under denitrification conditions. Therefore, these filaments are not inhibited upon commencement of aerobic conditions when the nitrate and nitrate concentrations have not been reduced to zero. This provides a competitive advantage to the AA filaments, enabling them to proliferate in the

system. If denitrification is essentially complete at the anoxic to aerobic transition, the denitrification intermediates will not be present intracellularly within the flocformers, the flocformers will not be inhibited in their subsequent aerobic substrate utilisation, and hence the AA filaments will not have a competitive advantage and will not proliferate.

Experimental evidence supporting this hypothesis comes from *inter alia* (i) observed inhibition of OUR upon commencement of aeration after anoxic exposure to elevated (1 to 25 mgN/) nitrite concentrations (Casey *et al.*, 1999), (ii) amelioration and inducement of AA filament bulking in laboratory nitrification denitrification (ND) biological excess phosphorus removal (BEPR) UCT systems with low (<1 mgNO_x-N/) and high (>5 mgNO_x-N/) concentrations respectively of nitrate or nitrite at the transition from anoxic to aerobic conditions (Musvoto *et al.*, 1994, 1999) and (iii) a finding by Tandoi *et al.* (1998) that a common strain of *M. parvicella* from a wastewater treatment plant (WWTP) denitrifies nitrate only to nitrite.

A consequence of this hypothesis is that there should be a strong association between nitrate and/or nitrite present at the time of transition from anoxic to aerobic conditions and the proliferation of AA filamentous organisms. In several subsequent investigations with ND (Ubisi *et al.*, 1997; Cronje *et al.*, 2000, Beeharry *et al.*, 2001) and NDBEPR systems (Pilson *et al.*, 1995, Mellin *et al.*, 1998; Sneyders *et al.*, 1998), episodes of poor sludge settleability have been observed several times. In most cases, there was an association between the nitrate/nitrite concentration at the transition from anoxic to aerobic conditions and the DSVI. This would suggest that controlling the nitrate and nitrite concentrations at the anoxic to aerobic transition may be a strategy that can be implemented to control AA filament bulking.

3.2 Development of a redox controller

To positively control the anoxic-aerobic nitrate/nitrite concentration to <1 mgNO_x-N/ , an online redox potential controller (called "redox box") was developed for single reactor intermittently aerated ND (IAND) systems by modifying the OUR meter of Randall *et al.* (1991). This redox box allows anoxic conditions to continue until a redox potential set point value is reached, whereafter the aeration period is commenced. Selecting a low redox potential set point value (about -45 mV), ensures that the nitrate and nitrite concentrations are below detectable limits at the anoxic to aerobic transition and *vice versa* (substantiated by nitrate and nitrite profile measurements). During the aerobic period, which was set at a fixed time interval (usually 3h), the redox box controlled the DO between a low (1 mgO/) and high (3 mgO/) set point and automatically measured and logged the OUR.

3.3 Testing the AA filament bulking hypothesis with redox control

To test the redox box's potential to control AA filament bulking, a long sludge age (15d) single reactor (15) IAND system (Experimental, ES) was set up and fed 15 /d real unsettled wastewater from the Mitchells Plain WWTP (Cape Town, South Africa). The influent TKN/COD ratio was fixed at ~0.10 by adding ammonia as required. The aerobic period was fixed at 3h. The redox set point was at - 45 mV, resulting in negligible NO_x at the anoxic to aerobic transition (confirmed with NO_X measurements). This gave an anoxic time between 3.6 and 5.6h, with the result that the aerobic mass fraction varied between 35 and 45%. The system was operated in this mode for 250 days. During this time, it appeared as if validation of the AA filament bulking hypothesis was achieved, because the AA filaments (identified to be predominantly *M. parvicella*) behaved as expected from the hypothesis (Tsai *et al.*, 2002) - with

a low redox set point (i.e. low NO_X at the anoxic to aerobic transition) low DSVI was recorded. To demonstrate repeatability, from day 250 an identical non-redox controlled IAND (3h anoxic - 3h aerobic) system (Control, CS) was operated in parallel, in which the AA filaments should proliferate since the anoxic period was inadequate to allow complete denitrification. In contrast, a low redox set point (- 45 mV) was selected in the redox controlled (ES) system to ensure complete denitrification, and hence control AA filament proliferation. From day 250 to 859, behaviour mostly contrary to that expected from the AA filament bulking hypothesis was observed in both systems, in that both experienced bulking (DSVI > 150 m /g) due to *M. parvicella*, and the bulking tended to be more severe in ES than CS. Many different possible causes for this contrary behaviour were examined experimentally (including changing the time for the total aeration/non-aeration cycle, selecting different redox set points, switching sludges between the two systems), but none reversed the contrary behaviour in the two systems. Confounding the matter was that observations from two other ND (MLE) systems in the laboratory operated on the same wastewater during this time, showed a close association between the anoxic to aerobic transition nitrate concentration and (mostly) *M. parvicella* proliferation, indicating that the AA filament hypothesis still has validity (Beeharry *et al.*, 2001). Thus, it appeared that some factor(s) outside of the AA filament bulking hypothesis was influencing *M. parvicella* proliferation in the IAND systems.

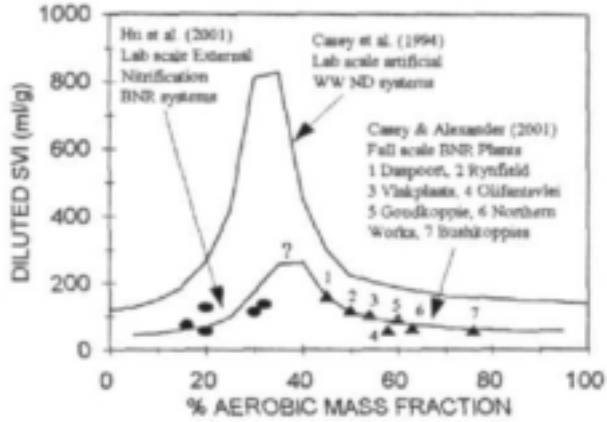

Fig 1: *DSVI versus aerobic mass fraction for lab-scale IAND systems fed artificial wastewater (Casey et al., 1994), and lab-scale external nitrification BNRAS systems fed real wastewater (, Hu et al., 2001) and fullscale BNR systems (, Casey and Alexander, 2001).*

3.4 Different cause for *M. parvicella* proliferation

Kruit *et al.* (2001), from an in-depth performance evaluation of 4 Dutch full-scale BNR (UCT) plants, noted that increasing aerobic reactor residual free and saline ammonia (FSA) concentration resulted in increasing DSVIs caused by *M. parvicella*. Earlier, Pitman (1982) and Barnard and Hoffmann (1986) noted at full-scale plants that the oxygen input to the aerobic reactor affected AA filament proliferation, especially *M. parvicella* - when the oxygen supply was reduced (by switching off aerators), nitrification became partial, the residual FSA concentration increased and the SVI increased and *vice versa*. Thus, it would appear that FSA may play a role in *M. parvicella* proliferation. Slijkhuis (1983) and Slijkhuis and Deinema (1984) showed that *M. parvicella* requires long chain fatty acids (C_{14} - C_{18}), low DO and ammonia for growth. It is therefore possible that the residual FSA is required to provide the N source for *M. parvicella* growth.

Casey and Alexander (2001) undertook a survey of full-scale BNR plants in South Africa to see if the AA filament bulking hypothesis could be validated with full-scale plant data. While the hypothesis could not be validated due to the difficulty of measuring the anoxic to aerobic NO_x concentration under full-scale operating conditions, they did note an association between aerobic mass fraction and DSVI (Fig 1). This observation appears to support the FSA requirement for

M. parvicella above: In BNR systems with large aerobic mass fractions, nitrification usually is rapid so that the FSA reaches very low concentrations before significant growth of the slow growing *M. parvicella* can take place. As the aerobic mass fraction decreases, increasing pressure is placed on the nitrifiers causing the FSA to increase, perhaps enabling *M. parvicella* to proliferate.

3.5 Testing the effect of the residual ammonia concentration on *M. parvicella* proliferation

The possible role of FSA in the proliferation of the AA filament *M. parvicella* noted above prompted an experimental enquiry into this aspect. It was proposed to examine the effect of aerobic FSA on AA filament growth in one system with a low DSVI (ES above), and in the other with a high DSVI (CS above). To reduce the DSVI in the ES, on day 832 it was changed to fully aerobic, which in the past had invariably ameliorated AA filament bulking and resulted in low (<60 m /g) DSVIs (Ekama *et al.*, 1996). As expected, this caused the DSVI to decrease from 250 to 67 m /g by day 860 (28d). The CS[1] was maintained anoxic aerobic (3h, 3h) and had a DSVI of 283 m /g on day 860 (Fig 2). On day 860 the CS was converted to fully aerobic also. Thereafter, allylthiourea (ATU) was slug dosed to both systems directly into the reactors (10 to 80 mg/ reactor volume) every 3 to 4 days to partially inhibit the autotrophic ammonia oxidizers (AAOs) and increase the residual FSA concentration. In the CS, ATU was dosed for 36 days to day 896, during which time the DSVI increased to 400 m /g. After stopping ATU dosing on day 896, nitrification was again complete (< 1mgNH_4^+-N/) by day 907 (11d), but the DSVI continued to increase to 591 m /g by day 914 (18d). From day 914, the DSVI rapidly decreased, reaching 50 m /g on day 962 (28d). A similar response was observed in the ES. ATU was dosed from day 860 to day 914 (56d) over which time the DSVI increased steadily from 67 to 314 m /g. On day 914, ATU dosing was stopped. Complete nitrification recovered over the next 10 days, but the DSVI continued to increase to 356 m /g on day 935 (21d). After day 935, the DSVI rapidly decreased, reaching 55 m /g on day 962 (Fig 2). Thus, irrespective of whether the DSVI initially was low (ES) or high (CS), dosing ATU, which increased the FSA concentration by inhibiting nitrification, caused the DSVI to increase and removing the ATU dose, which restored nitrification and reduced the FSA, caused the DSVI to decrease after a lag period. Microscopic examination during this period implicated *M. parvicella* as the dominant filament in both systems.

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[1]Note that the system called Control System (CS) here is actually the Warburton System (WS) in the summary report (Part 3 of the full report of which this is the executive summary) and detailed report of Tsai *et al.* (2002).

3.6 Conclusions

In the last 100 days of this 962 day investigation, it was found that the aerobic reactor free and saline ammonia (FSA) concentration had the strongest controllable influence yet observed on the

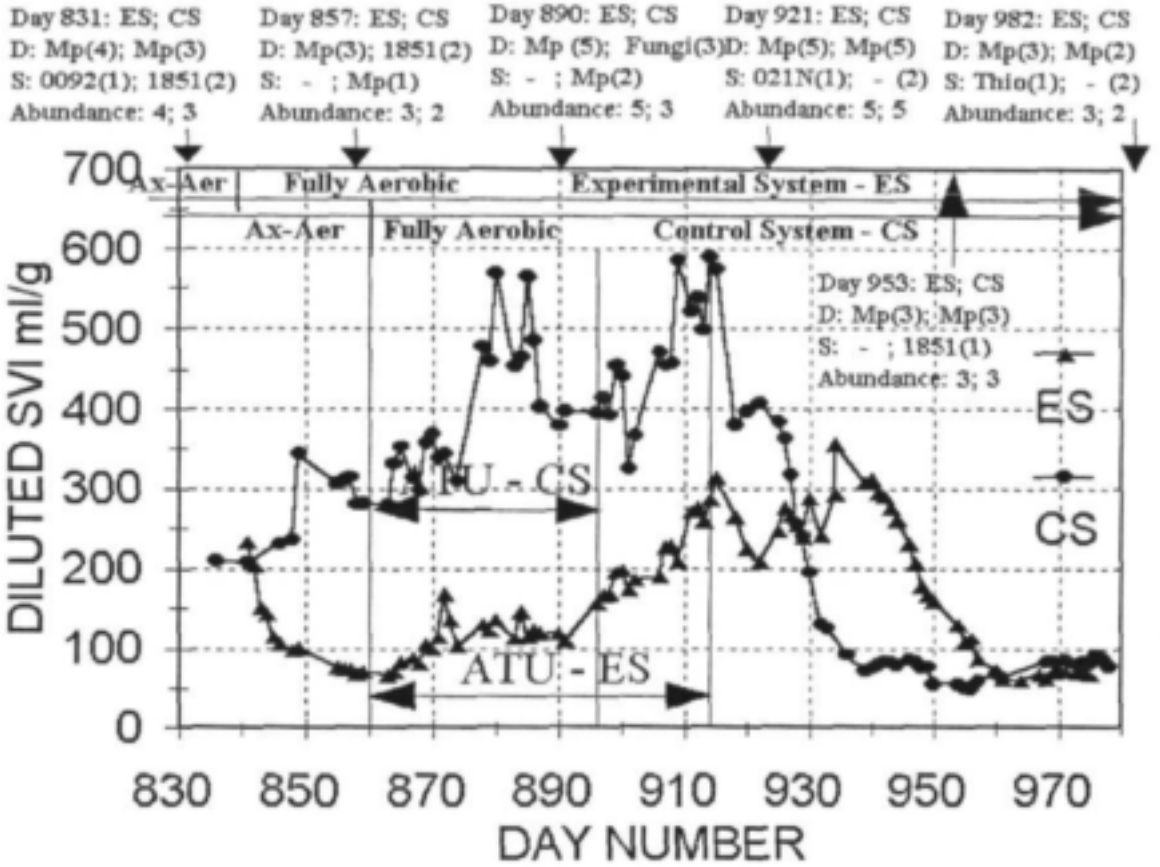

Fig 2: *Diluted SVI versus time (day number) for the Experimental (ES) and Control (CS) systems with and without allylthiourea (ATU) dosing. Filament identifications shown above where D = Dominant, S = Secondary, Mp = Microthrix parvicella and Thio= Thiothrix spp.*

AA (low F/M) filament proliferation, in particular *M. parvicella*. If the FSA was high (by inhibiting the nitrifiers) under completely aerobic conditions (which in the past invariably cured AA filament bulking, Ekama *et al.*, 1996), bulking (i) continued if the DSVI was high (>250 m /g) and (ii) was stimulated if the DSVI was low (<60 m /g). Removal of the nitrifier inhibitor restored complete nitrification and after a 20 to 28 day delay, cured the bulking and reduced the DSVI from 500 and 300 m /g respectively to 60 m /g in 30 days. The hypothesized explanation for this observation is that *M. parvicella* requires FSA as a N source for growth and cannot use nitrate or nitrite as an alternative. In terms of this hypothesis, if nitrification is rapid and complete, then FSA is not available and the slow growing *M. parvicella* are limited in their growth. In contrast, if nitrification is slow or incomplete, then the *M. parvicella* have FSA available as an N source for growth and so proliferate.

The above hypothesis for *M. parvicella* growth explains several observations on full-scale BNR plants: (i) the seasonal proliferation of *M. parvicella* during winter, and (ii) the increasing DSVI with decreasing aerobic mass fraction (Fig1). Both temperature and reduced aerobic mass

fraction place pressure on the nitrifiers resulting in increasing aerobic reactor FSA concentration which favours *M. parvicella* growth. Being capable of nitrate reduction (Tandoi *et al.*, 1998) it is recognised that *M. parvicella* could grow in the anoxic reactor since both nitrate and ammonia are available. However, *M. parvicella* is slow growing and obtains only a limited amount of energy under anoxic conditions, since it can only reduce NO_3 to NO_2. Thus, the anoxic growth of *M. parvicella* is probably too small to significantly affect the DSVI.

The above hypothesis for *M. parvicella* growth does not overturn the AA filament bulking hypothesis of Casey *et al.* (1999) - the association between the anoxic to aerobic transition nitrate concentration and the DSVI has been observed too frequently to discard it. Both hypotheses are regarded as relevant, and future research will focus on how elements of these two hypotheses superimpose on the conditions in different BNR systems.

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2. Refereed Papers

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6. Papers submitted for publication

Tsai M-W, Wentzel MC and Ekama GA (2002) The effect of residual ammonia concentration under aerobic conditions on the growth of *Microthrix parvicella* in biological nutrient removal plants. Submitted to *Water Research*

7. Higher degrees

Ubisi MF	MSc	1997	Organic and inorganic components of activated sludge mixed liquor.
[2]Tsai M-W	MSc	2001	Anoxic-aerobic (AA, low F/M) filament bulking control with

[2]Tsai did not submit his MSc in 2002; he upgraded to PhD to pursue investigating the effect of residual ammonia concentration under aerobic conditions on the growth of *Microthrix parvicella*.

			redox potential.
Hercules S	MSc	2001	Full scale demonstration of filamentous bulking control at a biological nutrient removal activated sludge plant.

CAPACITY BUILDING

The Table 1 below gives the number, race, gender and nationality of students who did BSc and MSc thesis topics associated with these projects. Table 2 gives a summary of these outputs.

Table 1: *Year, degree, name, thesis title, race, gender and nationality of students who undertook practical research work in the Water Research Laboratory associated with these projects.*

Year	Deg	Name	Thesis Title	Race	Gender	Nationality
1997	BSc	Sotemann, SW	Nitrification in trickling filters	W	M	SAPR
1997	BSc	Plaatjies, B	Impact of septic tank/small bore sewer discharges into sewers on BNR	B	M	SA
1997	BSc	Dlokweni, T	Effect of slowly biodegradable COD on biological excess P removal	B	M	SA
1998	BSc	Rhode, P	Application of Aquasim to water and wastewater treatment	B	M	SA
1998	BSc	Tsai, M-W	Sludge bulking control with redox potential	Asian	M	SAPR
1999	BSc	Deelchand R	The influence of pH on heterotrophic and autotrophic growth rates	Asian	M	Maur
1999	BSc	Beeharry, AO	Measurement of nitrifier maximum growth rates	Asian	M	Maur
1999	BSc	Nongogo, MC	Treatment of septic tank influents	B	M	SA
1999	BSc	Pitse, W	Flocculation characteristics of activated sludge	B	M	SA
1999	BSc	Simelane, ME	Assessment of anoxic and aerobic P uptake in full-scale plants	B	M	Swazi
2000	BSc	Modipa, MJ	Performance assessment of Mitchells Plain pilot plants	B	F	SA
2000	BSc	Diale, TV	Performance assessment of Mitchells Plain pilot plants	B	F	SA
2000	BSc	Bokako, C	The influence of pH on heterotrophic and autotrophic growth rates	B	M	SA
2000	BSc	Cheng, N	Anaerobic processes in biological excess P removal	Asian	F	China
2000	BSc	Woldemariam, E	Effect of bio-augmentation in activated sludge systems	B	M	Zim
2001	BSc	Patel, C	Measurement of nitrifier maximum growth rates	Asian	F	Zim
2001	BSc	Mbedle, N	Modelling hydrogen sulphide emission in the sewer environment	B	M	SA
2001	BSc	Ndoloshe, M	Modelling bio-yield in wastewater treatment	B	M	SA
1998	Dipl Eng	Swartzkop, H	Sludge bulking control with redox potential	W	F	Other
1997	MSc	Ubisi, MF	Organic and inorganic components of activated sludge mixed liquor.	B	M	SA
2001	MSc	[1]Tsai, M-W	AA (Low F/M) filament bulking control with redox potential	Asian	M	SAPR
2002	MSc	Hercules, SM	Full scale demonstration of filamentous bulking control at a biological nutrient removal activated sludge plant	B	M	SA

B= Black and for South Africans (SA) includes African, Coloured and Indian.

SAPR= SA permanent resident.

[1] Was registered for MSc 2000-2001 but did not submit MSc thesis; upgraded to PhD in 2002.

Table 2: Summary of BSc, MSc and PhD degree outputs over the project period.

Year	Deg	Total	Race			Gender		Nationality		
			B	W	Other	Men	Women	SA	SADC	Other
1997 - 2001	BSc	18	12	1	5	14	4	12	5	1
1997 - 2001	DipEng	1	0	1	0	0	1	0	0	1
1997 - 2001	MSc	2	2	0	0	2	0	2	0	0
1997 - 2001	PhD	0	0	0	0	0	0	0	0	0

From the Tables, 12 Black, 1 White and 5 Asian students did their BSc theses research in the area of wastewater treatment associated with the two WRC projects K5/823 and K8/373. One foreign research associate from the Technical University of Hannover (with whom the WRG has good collaboration), Germany did her practical research at UCT for her Diploma Engineer degree. Two black South Africans completed their masters degrees. The BSc thesis students Sotemann (1997), Tsai (1998) and Beeharry (1999) continued their studies in the Water Research Group and registered for MSc degrees but on different WRC projects. Sotemann worked on the External Nitrification project (K5/970) and graduated MSc in 2000; he is currently registered for PhD and working on the Material mass balances project (K5/1338). Beeharry worked on the Measurement of Active Biomass project (K5/1179) and graduated MSc in 2001. Tsai worked on the Bulking Control with Redox Potential project (K8/373) for his MSc research. He wrote a report on this work (see detailed report Part 3) but did not submit a MSc thesis. Instead he upgraded to PhD in 2002 and will continue investigating AA filament bulking control.
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TABLE OF CONTENTS

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Messrs *Ming-Wei (Andy) Tsai* of the Dept of Civil Engineering and *Selwyn Hercules* of the CMC, both postgraduate students in the Water Research Group of the Dept reading for their MSc degrees.

It is the dedication and effort of these students that produced the large body of useful experimental data and information.

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Acknowledgement is due to the members of the Steering Committee of the project who guided the research work of both projects during the 4 year period:

Dr NP (Nozi) Mjoli	- Water Research Commission (Chair)
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Ms M (Manda) Hinsch	- Department of Water Affairs and Forestry
Mr GB (Gert) Saayman	- City Council of Pretoria
Mr D (Dave) Nosaic	- Umgeni Water (alt Mr PD (Paul) Gaydon)

Mr DN (Danie) Klopper - Cape Metropolitan Council
Mr K (Kevin) Fawcett - Cape Metropolitan Council
Mr R Moolman - Cape Metropolitan Council
Mr OCE (Otto) Hempel - Water Research Commission (Committee Secretary)

Finally the writers express their appreciation to all their colleagues and associates in the field for their willingness to inform us of their experiences and observations on full-scale plant behaviour. We value this contact with practice, not only for the information it provides, but also for the sobering reminders of the magnitude and urgency of the bulking problem.

Causes and control of AA (low F/M) filament bulking in biological nutrient removal activated sludge systems

Final reports on the

(1) Four year research contract K5/823 (1997-2001) full scale demonstration of filamentous bulking control at a biological nutrient removal activated sludge plant (Mitchells Plain Wastewater Treatment Works, Cape, South Africa)

and

(2) Two year research consultancy K8/373 (2000-2001) bulking control with redox potential.

1. BACKGROUND TO RESEARCH

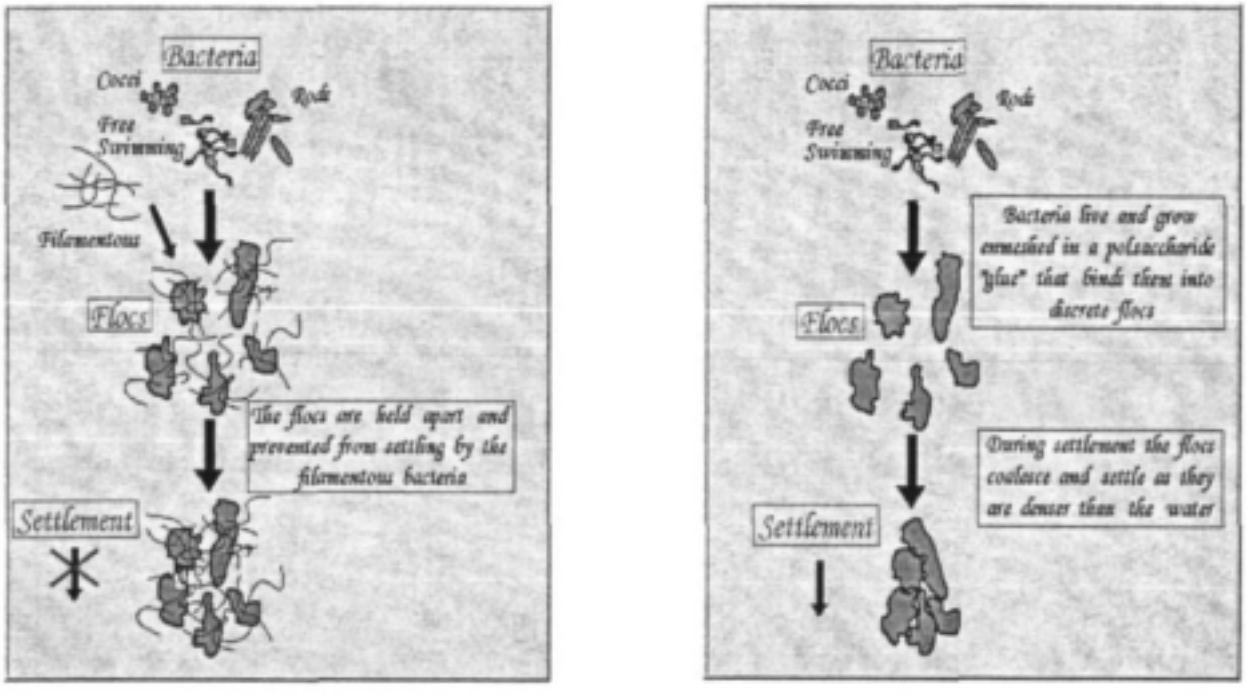
1.1 The effect of Bulking on the activated sludge system

The activated sludge process is an efficient means whereby the nutrients nitrogen (N) and phosphorus (P) can be reduced biologically to low concentrations in municipal wastewater effluents. However, a major disadvantage of the process is that in the sludge mass which develops, filamentous organisms can proliferate which results in poor settleability of the sludge in the secondary settling tank. From a survey of nutrient removal plants in South Africa (Blackbeard *et al.*, 1988) it was found that out of 45 nutrient removal activated sludge plants, 33 had bulking problems of considerable proportions which severely reduced the treatment capacity of these plants.

It has been shown by Ekama and Marais (1986) that an important factor limiting the treatment capacity of an activated sludge plant is the inefficient separation of solids from the liquid phase in the secondary settling tank caused by poor sludge settleability. For a mixed liquor with a suspended solids concentration of 3.5 g/ℓ and a Diluted Sludge Volume Index (DSVI) of 150 mℓ/g a maximum overflow rate of 1 m/h can be achieved without settling tank failure (i.e. solids carry over). If the DSVI is reduced from 150 mℓ/g to 100 mℓ/g the maximum overflow rate can be increased to 1.8 m/h, thereby increasing the treatment capacity by 2/3. However, if the DSVI deteriorates from 150 mℓ/g to 200 mℓ/g then a reduction in maximum overflow rate to 0.6 m/h can be expected, effectively reducing the treatment capacity by 1/3. These findings demonstrate the importance of developing and maintaining a good settling sludge. The large potential savings from increasing the treatment capacity of activated sludge plants through improvement in sludge

settleability have motivated considerable research into the causes of filamentous bulking.

Bulking is caused by the excessive growth of filamentous organisms which leads to a deterioration in the separation of solids from the liquid phase in the secondary settling tank (SST). If a sludge contains very small quantities of filamentous organisms and is dominated by floc-formers, then pin-point flocs result, which while providing a good settling sludge, tends to generate a poorly clarified effluent. Conversely, a sludge which is largely made up of filamentous organisms will generate a well clarified effluent (if carry-over of solids is avoided) but which settles poorly. Clearly, the correct combination of these two extremes is desirable such that a good settling sludge is produced together with a well clarified effluent. Figures 1.1 and 1.2 shows simple graphic representation of the relationship between the floc-formers and filamentous organisms, and its effects on the overall settleability. Figure 1.1 shows the filamentous organisms have out numbered floc-formers which results bad settleability, while Fig 1.2 shows that the floc-formers are dominant in this particular sludge and it results better settleability.



Figs 1.1 and 1.2: *Graphic representations of bad and better settling sludge with respect to the relationship between the floc-formers and filamentous organisms. Dominance of filamentous organisms over floc-formers results in poor settleability, and conversely dominance of floc-formers over the filamentous organisms results in better settleability (www.geocities.com/~htoprak/cokrak.html, 29July 1998).*

Lee *et al.* (1983) investigated the effect of the presence of different quantities of filamentous organisms - measured by Total Extended Filament Length (TEFL, km/g) - on the sludge settleability parameters Sludge Volume Index (SVI) and Diluted Sludge Volume Index (DSVI).

They found that the DSVI was much better correlated to the TEFL than the SVI and when the DSVI increased above 150 mℓ/g, filamentous organisms began to dominate the settling characteristics of the sludge (Fig 1.3). From this finding, a sludge with a DSVI of greater than 150 mℓ/g is regarded as a bulking sludge. DSVI values between 80 and 100mℓ/g are regarded as ideal because these have sufficient filaments to enable good flocculation and clarification but insufficient filaments to cause poor settleability.

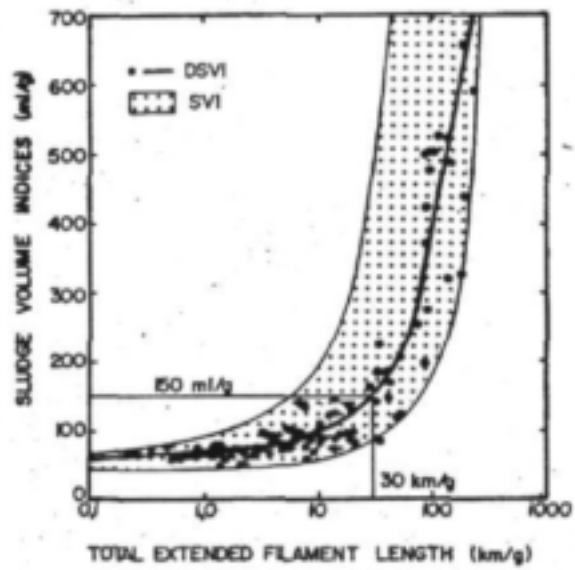


Fig 3: DSVI (and SVI) versus TEFL in km/g showing effect of filaments on DSVI for TEFL > 30 km/g for DSVI > ~150mℓ/g.

1.2 Control of filamentous bulking

The current methodology for analysis and control of filamentous bulking sludges is to use the filament categorization method of Jenkins *et al.* (1984) in which the presence of a specific filament types causing bulking is associated with a causative wastewater characteristic or a process operating condition (see Table 1.1). Eliminating the causative condition will result in cessation of the proliferation of the specific filament type and hence amelioration of the bulking problem. This method of controlling filament bulking is termed "specific" and contrasts with "non-specific" control methods such as chlorination which impairs the growth of filamentous organisms to a greater extent than floc-formers because of their high surface area to volume ratio. The use of non-specific methods for the control of bulking does not address the causes of filament proliferation but merely treats the symptoms. Consequently, it was desirable to develop specific methods which control the causative filamentous organisms and to do this a more fundamental understanding of the interaction between floc-formers and filaments was necessary.

Table 1.1: *Categorisation of filaments according to suggested causative conditions (Jenkins et al. 1984).*

Suggested causative conditions	Indicative filament types
Low F/M ratio	*M. parvicella*, Types 0041, 0675, 0092, 0581, 0961, 0803, 021N, *H. hydrossis*, *Norcardia* spp.
Low dissolved oxygen	Type 1701, *S. natans*, *H. hydrossis*
Presence of sulphide / septic sewage	*Thiothrix* spp., *Beggiatoa* spp., Type 021N
Low pH	Fungi
Nutrient deficiencies	*S. natans*, *Thiothrix* spp., Type 021N, and possibly *H. hydrossis*, Types 0041, 0675

Filament identification surveys of South African N&P removal activated sludge plants (Blackbeard *et al.*, 1988) indicated that the six most commonly dominant filamentous organisms were types 0092, 0675, 0041, *M. parvicella*, 0914 and 1851. From Table 1.1 the first 4 of these are low Food to Micro-organism ratio (F/M) filaments. Therefore in order to improve sludge settleability in low F/M or long sludge age nutrient removal activated sludge plants in South Africa, control strategies need to be investigated to minimize low F/M filament proliferation.

1.3 The selector affect and low F/M filamentous bulking

Chudoba *et al.* (1973) proposed a selection criterion to explain the occurrence and non-occurrence of filament bulking in low F/M systems, which was based on the differences in growth kinetics between floc-formers and filamentous organisms at different substrate concentrations. In the Monod formulation for specific growth rates it was found that filamentous organisms generally have lower values for both the half saturation coefficient (K_S) and the maximum specific growth rate (μ_H) making them more responsive to low bulk liquid substrate concentrations. Hence, in low F/M or long sludge age plants where the bulk liquid substrate concentrations are low, filamentous organisms have a selective advantage and tend to dominate over the floc-forming organisms, thereby producing a poor settling sludge. Resulting from this, it was proposed that if a "selector" reactor was incorporated in a system such that the substrate concentration in this reactor was maintained at a high level, then floc-formers would be selected because of their higher maximum specific growth rate at high substrate (soluble readily biodegradable organics - RBCOD) concentrations. The mixed liquor would then tend to contain fewer filamentous organisms, producing a sludge with better settling characteristics. This idea stimulated considerable research into the use of selectors for low F/M bulking control, but by 1984 (Still *et al.*, 1985, 1996) it became apparent that Chudoba's selection criterion does not completely account for the suppression of filamentous organism proliferation in either aerobic or anoxic selectors (see Ekama *et al.*, 1996).

From the literature it appeared that *S. natans*, *Thiothrix* and 021N are possibly controlled by inducing a selector effect (Gabb *et al.*, 1989) but there was no conclusive evidence that all low F/M filaments, in particular *M. parvicella* - are controlled by this method. Consequently a four year investigation (1985-1988) was initiated at UCT to consolidate and in some cases repeat experiments reported in the literature so as to provide conclusive evidence for the conditions under which filamentous organisms are controlled by selectors. A number of different experiments were carried out by Gabb *et al.* (1989), in which the effect filament seeding and of anoxic, aerobic and anaerobic selectors on low F/M bulking was examined in fully aerobic, anoxic-aerobic and anaerobic-anoxic systems. The following conclusions were drawn from this work:

1. High RBCOD uptake rates were stimulated in sludges under alternating feed-starve conditions (in batch and/or selector reactor systems) compared with sludges from continuously fed completely mixed systems; if the conditions were aerobic or anoxic the high RBCOD uptake rate resulted in high oxygen or nitrate utilization rates respectively; a sludge with a high RBCOD uptake rate had acquired a "selector" effect.
2. Seeding of *S. natans* and *Thiothrix*, which grew on the walls of the influent feed lines, into the activated sludge systems is a cause of the bulking by those filaments in laboratory (and possibly pilot) scale low F/M systems, but a selector reactor can control their proliferation.
3. Fully aerobic and anoxic conditions in low F/M systems started up with poorly settling sludge with low F/M filaments from full scale plants invariably developed good settling sludges within 1 month.
4. Alternating anoxic-aerobic conditions in single reactor intermittent aeration nitrification and denitrification (IAND) low F/M systems stimulated low F/M filament proliferation in full scale and lab scale systems.
5. Aerobic selectors ahead of the IAND systems did not control low F/M filament proliferation.
6. The observation that aerobic selectors did not control bulking by low F/M filaments resolved the apparent inconsistency in behaviour of anaerobic reactors (metabolic selection) and aerobic (and anoxic) selectors (kinetic selection). Because the anaerobic, anoxic and aerobic selectors stimulate preferential RBCOD uptake by floc-formers (in anaerobic reactors by the poly phosphate accumulating organisms PAOs) and this did not control low F/M filament proliferation, the selector effect did not play an important role in low F/M filament bulking.

The finding that metabolic and kinetic selection did not control bulking by low F/M filaments placed the research back by into an exploratory phase. The endeavour of the research that followed was to establish a new framework (paradigm) for understanding the fundamental causes of low F/M bulking from which specific control methods could be derived.

1.4 The AA filament bulking hypothesis

A second wide-ranging follow-up research investigation began in 1989 to study the effect of various sewage characteristics and system operating parameters on low F/M filament bulking, in particularly the low F/M filament bulking response to:

1. Readily biodegradable or slowly biodegradable COD from artificial substrate and real sewage.

2. Plant configurations (i.e. fully aerobic; fully anoxic; intermittent aeration; pre- and post-denitrification; MUCT and JHB N and P removal systems),
3. Sludge age [i.e. short (5 days) or long (20 days)],
4. Magnitude of anoxic/aerobic mass fraction,
5. Frequency of alternation between aerobic and anoxic conditions,
6. Dissolved oxygen in the aerobic zone,
7. Nitrate and nitrite concentrations in the anoxic zone(s).

Some of the above parameters were investigated using artificial substrate by Gabb *et al.* (1989); Ketley *et al.* (1991); Hulsman *et al.* (1992); de Villiers *et al.*(1994) and Casey *et al.* (1994a). In many instances it was necessary (either for the purposes of confirmation or because the filaments found were not those generally found in full-scale plants), to study the effects of the above parameters with real sewage [Warburton *et al.* (1991); Ketley *et al.* (1991); Hulsman *et al.* (1992); de Villiers *et al.*(1994); Casey *et al.* (1994a)] so that additional uncertainties using artificial substrate did not unnecessarily complicate the objective of finding methods for ameliorating low F/M filament bulking.

From this research it was established that the following conditions have a significant influence on low F/M filament bulking (after Casey *et al.* 1994b, Musvoto *et al.*, 1999 and Lakay *et al.*, 1999):

1. Continuous anoxic and continuous aerobic conditions controlled low F/M filament proliferation to low DSVI values (< 100 mℓ/g),
2. An aerobic mass fraction of between 30 and 40 % were observed to coincide with highest DSVI values, aerobic fractions greater or less than this were associated with progressively lower DSVIs until fully aerobic or fully anoxic conditions were present,
3. Low F/M filament proliferation was observed in single reactor intermittent aeration systems irrespective of the biodegradability of the available substrate (i.e. RBCOD or SBCOD) for both artificial substrate and real sewage separated into soluble and particulate fraction by micro-filtration,
4. Low dissolved oxygen concentration in the aerobic reactor did not significantly influence low F/M filament proliferation,
5. The presence of nitrate and/or nitrite concentrations at the time the conditions in the various N and N&P removal systems become aerobic (having been anoxic) promoted low F/M filament bulking. In this respect, it appeared that nitrite had a greater influence than nitrate.

From research results, summarized by Lakay *et al.* (1999) and Musvoto *et al.* (1999), Casey *et al.* (1992; 1994ab, 1999abc) developed a hypothesis for the proliferation of low F/M filaments in N and N&P removal plants: Accepting that the filamentous organisms reduce nitrate to nitrite only and the floc-formers denitrify nitrate to nitrogen gas, when denitrification is not complete in the anoxic reactor, gaseous denitrification intermediates (NO, N_2O) in the floc-formers inhibit their oxygen uptake enzymes when conditions become aerobic. The filaments, which do not accumulate the inhibitory denitrification intermediates, are not inhibited in their oxygen uptake ability and therefore have an advantage over the floc-formers. As a consequence Casey *et al.* (1994ab) renamed the low F/M filaments that tend to proliferate in N and N&P removal plants as Anoxic-Aerobic (AA) filaments, a name more descriptive of the apparent cause for their proliferation.

1.5 Experimental evidence supporting the hypothesis

1.5.1 Demonstration of inhibition

To determine whether or not inhibition of oxygen utilization takes place in activated sludge which is subjected to alternating anoxic-aerobic conditions, a series of batch tests were conducted on sludge drawn from the anoxic reactor of the 2RND system operated by de Villiers *et al.* (1994). To assess oxygen utilization the maximum specific OUR was measured (Ekama *et al.*, 1986) upon sewage addition with both anoxic and aerobic pretreatment conditions.

Anoxic denitrification

From Fig 1.4, it is demonstrated that inhibition of OUR was induced in the sludge after a 2 hour anoxic period with NO_2^- present during both the anoxic and subsequent aerobic periods. The addition of ≈ 25,0 mgNO_2^--N/ℓ at the start of the aerobic period exhibited dramatic inhibition while less marked inhibition was noted on addition of ≈ 5,5 mgNO_2^--N/ℓ, and almost no inhibition was measured on addition of 0,1 mgNO_2^--N/ℓ . Two conclusions were drawn from the observation (1) inhibition of OUR in the presence of NO_2^- is observed, and (2) the degree of inhibition is directly related to the concentration of NO_2^- at the beginning of the aerobic period.

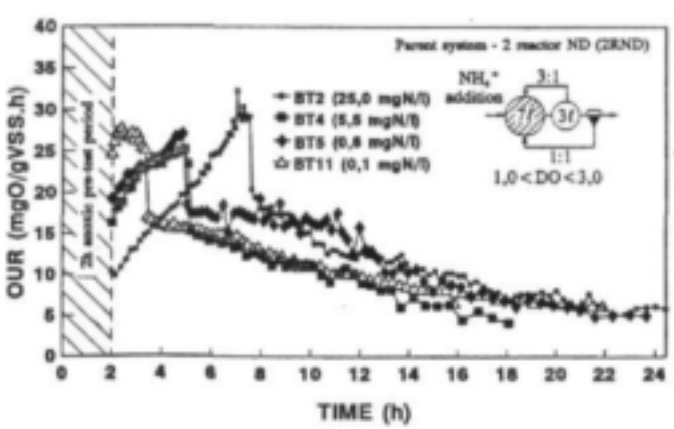


Fig 1.4: Specific oxygen utilization rate OUR, in mgO/(gVSS.h), with time (in h) for batch tests (BT) 2, 4, 5, and 11 with different concentrations of nitrite present at the onset of aerobic conditions conducted on sludge harvested from System 15 of Lakay et al. (1999) (see their Fig 7) on days 227 (BT2), 233 (BT4), 240 (BT5) and 280 (BT11).

It could not be determined from these tests however, whether the inhibition results from the NO generated by NO_2^- denitrification under anoxic conditions or under aerobic conditions.

Aerobic denitrification

To determine whether activated sludge from the 2RND system exhibited aerobic denitrification, aerobic batch tests were conducted on specially prepared sludge samples. In the preparation of these samples virtually all the NO_3^- and NO_2^- were removed from the sludge by dilution with tap water, settling and decanting the supernatant three times. The sludge was then held anoxic in the presence of 120 mg COD/ℓ sewage in order to denitrify any remaining NO that might be present within the organism. After 2 hours, during which thiourea was added (10mg/ℓ) to inhibit NO_2^- formation by *Nitrosomonas*, aeration was commenced (2,0 < DO < 4,0 mg O/ℓ). After 1hr aeration, 20 mg NO_2^-- N/ℓ of nitrate was dosed. After a further 1 hour aeration, 360 mg COD/ℓ (final batch volume) sewage was added and the OUR nitrate and nitrite concentrations measured with time. Figure 1.5 shows that OUR inhibition is exhibited. In a similar test but with NO_3^- addition (20mg N/ℓ) instead of NO_2^-, no inhibition was exhibited. These observations suggested that NO inhibition does take place with NO_2^- (the NO apparently produced by aerobic denitrification of NO_2^-), but not with NO_3^-. In a control batch test, in which no NO_2^- or NO_3^- was added, no inhibition was exhibited. These results were reproducible with sludges from IAND and MUCT systems.

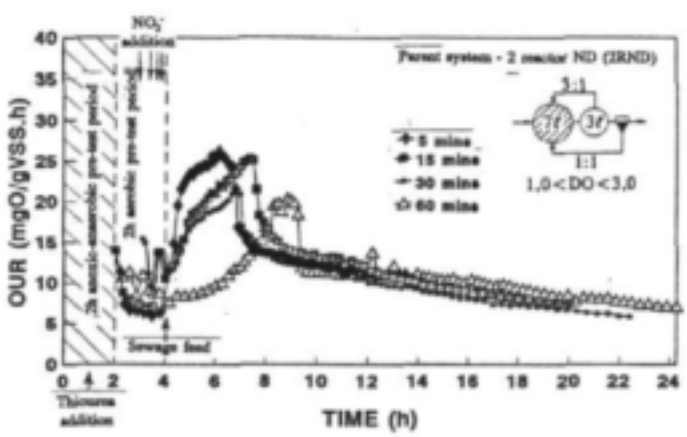


Fig 1.5: Specific oxygen utilization rate OUR, in mgO/(gVSS.h), with time (in h) for batch tests (BT) 13 to 16 with nitrite addition under aerobic conditions at different time periods before the addition of substrate conducted on sludge harvested from System 5 of Lakay et al. (1999) (see their Fig 7) on days 285 (BT13), 330 (BT14), 333 (BT15) and 334 (BT16).

In the batch tests presented so far, it appears that during the aerobic period after sewage addition the inhibition is relieved, reflected in a steadily increasing maximum specific OUR, in some cases levelling off at a constant value before the precipitous decrease in OUR when the RBCOD has been depleted. The relief of OUR inhibition possibly arises because the presence of significant quantities of RBCOD under aerobic conditions accelerates the $NO \rightarrow N_2O \rightarrow N_2$ part of the denitrification pathway so that the NO produced from NO_2^- denitrification does not accumulate.

1.5.2 Effect of RBCOD on OUR inhibition by NO

To check if OUR inhibition takes place in the presence of significant quantities of RBCOD, an aerobic batch test was conducted in which NO_2^- was added after the sewage addition but while RBCOD was still present, rather than before sewage addition when only SBCOD (principally generated from organism death and lysis) is present as in the previous batch experiments. In this test no inhibition was noted, and it was concluded that the presence of RBCOD (in sufficient quantity) prevented or relieved the inhibition. From this it seemed reasonable to accept the suggestion above that the RBCOD accelerates the $NO \rightarrow N_2$ steps of the pathway in such a way that NO no longer is accumulated.

1.5.3 Determination of the extent of NO_3^- reduction and denitrification under anoxic conditions by filaments and floc-formers

With the experiments above, it was demonstrated that OUR inhibition hypothesized to be by NO, takes place in the presence of NO_2^- in switching from anoxic to aerobic conditions. For the proposed explanation to be acceptable, it needed to be shown even superficially that floc-formers denitrify from NO_3^- to N_2 gas, and so are susceptible to OUR inhibition by accumulated NO, whereas the low F/M filaments reduce NO_3^- to NO_2^- only, and therefore do not accumulate NO and so are not susceptible to this inhibition. Clearly this is an experiment that needs to be taken up by microbiologists and biochemists, but for the purposes of testing the hypothesis, sludge samples from a fully anoxic (FX) system (low DSVI) and the 2RND system on which the batch tests above were done (high DSVI), both fed real sewage, were subjected to a nitrate reduction test, a test which allows the generation of NO_2^- and/or N_2 gas to be determined. The sample with the high DSVI (many AA filaments) showed an accumulation of NO_2^- with no N_2 gas being detected in 8 out of 10 tests. The sample with the low DSVI (few AA filaments) accumulated N_2 gas, but no NO_2^- accumulated in 8 out of 10 tests. From this it is reasonable to accept that qualitatively, filaments tend to reduce NO_3^- to NO_2^- only, where floc-formers denitrify NO_3^- to N_2 gas. This observation lends credibility to the proposed hypothesis for low F/M filament proliferation. With a reasonable hypothesis for low F/M filament proliferation in N and N & P removal systems, attention was directed at devising strategies for the control of these filaments in the systems.

1.5.4 The effect of incomplete denitrification on sludge settleability in MUCT systems

Having found some credibility for the AA filament bulking hypothesis of Casey *et al.* (1992) by demonstration of OUR inhibition and correspondingly substrate utilization in batch tests, Musvoto *et al.* (1992) set up two MUCT systems in order to demonstrate the effect of floc-formers inhibition on sludge settleability, in nutrient removal activated sludge plants running at steady state.

In these experiments the anoxic zones comprised 65% of the system mass fraction (i.e. large to enable complete denitrification) with 15% anaerobic and 20% aerobic mass fractions. It was found that while no nitrate or nitrite was dosed to the 2nd anoxic reactor of these systems, and thus nitrate and nitrite concentrations entering the aerobic reactor were < 1 .0 mg NO_3^--N/ℓ and < 0.2 mgNO_2^--N/ℓ respectively, low DSVIs were observed (Fig 1.6). Conversely, when nitrate was dosed to the second anoxic reactor of one system to provide an equivalent TKN/COD ratio of 0.16mgN/mgCOD the DSVI increased from 80ml/g to 176ml/g (bulking by AA filaments 0092, *M. parivcella*, 0041, 0803 and *H. hydrossis*) in 111 days. Separately, when nitrite was dosed to the second anoxic reactor of the other system to provide an equivalent TKN/COD ratio of 0,18mgN/mgCOD, the DSVI increased rapidly from 90 to 174ml/g (also bulking by AA filaments 0092, *M. parivcella*, 0675 and 021N) in 55 days.

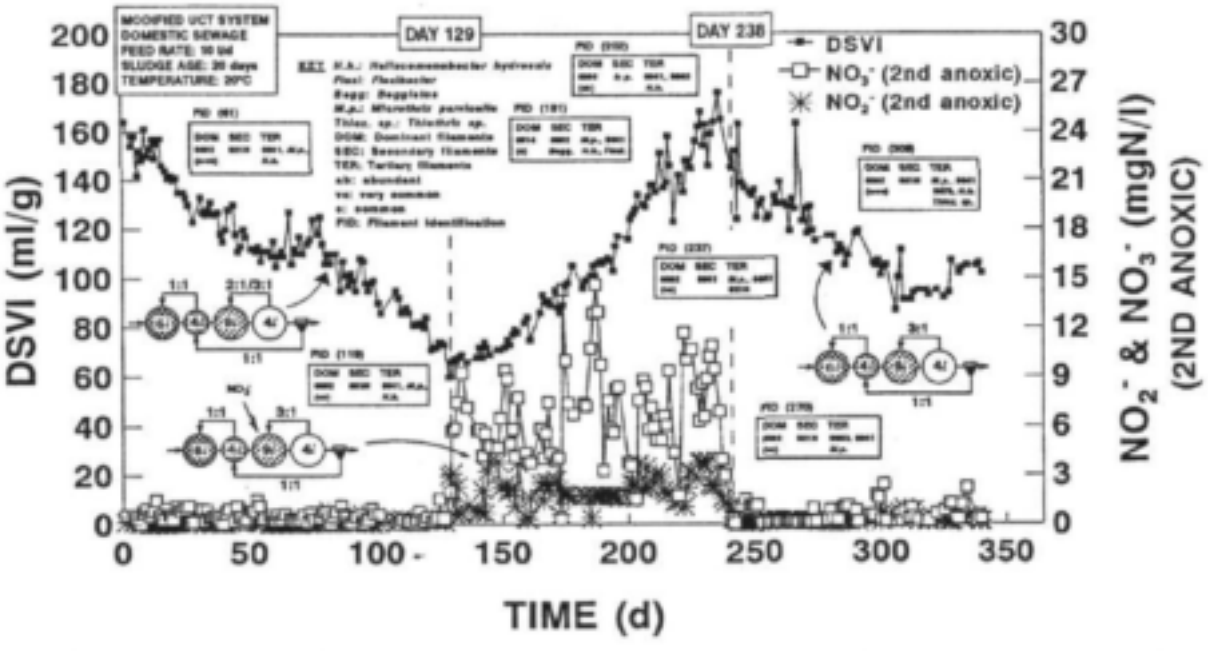


Fig 1.6: *Sludge settleability as DSVI (ml/g) and concentrations of nitrate and nitrite in the 2nd anoxic reactor of MUCT system 3 of Musvoto et al. (1999) with time in response to dosing (Day 129 to 239) or not dosing (Day 1 to 128 and day 240 to 340) nitrate to the 2nd anoxic reactor.*

The above results provided considerable support for the AA bulking hypothesis of Casey *et al.* (1992) and from it was concluded that it would be desirable to ensure that denitrification is complete (i.e. nitrate < 0.5 and nitrite < 0.2 mgN/ℓ respectively) in the anoxic reactor prior to the aerobic reactor to avoid proliferation of AA filaments. This behaviour has been repeatedly found in subsequent investigations with laboratory N and N&P removal systems (Kaschula - 1993, Pilson -1995, Sneyders - 1998, Mellin-1998 and Vermande-2000) and provided further evidence for the bulking hypothesis of Casey *et al.* (1992, 1994b) which attempts to explain the cause of AA (low F/M) filament bulking in N and N&P removal plants.

1.6 Full-scale plant support for the AA new hypothesis

The above hypothesis, as an alternative to the selector approach for explaining the causes and control of low F/M (AA) filament bulking, is a relatively recent development and therefore still needs thorough evaluation and validation. However, a significant body of information providing indirect support for this alternative conceptualization of the low F/M filament bulking problem is emerging not only from laboratory scale systems (cited above) but also from full scale plant observations. In this respect the conference proceedings of the 1st ASPD Specialist Group make interesting reading (summarized by Ekama, 1994) e.g.

1. From a survey of Danish plants, Andreasen and Sigvardsen (1993) concluded that the "first survey at nutrient removal plants show a higher percentage with SVI >150 mℓ/g and especially at some plants which include BEPR which seem to have constant high SVI with filaments *M. parvicella*, 0041 and 0803."
2. From a survey of Rome's plants, Rossetti *et al.* (1994) found that "systems operating with alternating aeration conditions (like Carousel type) with short anoxic-aerobic cycles show permanent high levels of low F/M filaments with the main filament being *M. parvicella*."
3. From a survey of German plants, Kunst and Reins (1994) state that "in the last years a lot of treatment plants with bio-P removal went into operation and people thought that anaerobic reactors in the treatment plants would be able to prevent sludge bulking - the experience in technical practice shows this didn't happen."
4. From practical experience in operating long sludge age plants in South England, Foot *et al.* (1994) concluded that (i) there is an inverse relationship between the total filament length (TFL) and the total oxidized nitrogen (TON) concentration in the effluent - the higher the TON concentration, the lower the TFL and *vice versa*, and (ii) "the widespread occurrence of this species (*M. parvicella*) in WWTPs would tend to indicate that it is not so much the substrate (sewage characteristics) which are important as the configuration and operation of the plant."
5. From 10 years experience with bulking in Dutch full scale plants, Eikelboom (1993,1994) concludes that (i) "development of *M. parvicella* shows a distinct seasonal pattern with highest DSVIs in spring and lowest in autumn", (ii) "*M. parvicella* grows better in Carousel type systems than in other extended aeration plants and it is worse with settled sewage", (iii) "the usefulness of selectors for controlling *M. parvicella* decreases as the overall load on the plant increases", (iv) "after introduction of nutrient removal conditions, the DSVI increased in 60% of these plants and *M. parvicella* was dominant in 87% of them" so that (v) "the application of BNR methods will even increase the dominating position of this organism", and (vi) "the ultimate effect of selectors for control of *M. parvicella* is insufficient and unpredictable so far. In Holland over 80

selectors have been incorporated in full-scale plants. Comparing the results with 15 years ago, it seems that the percentage of plants with bulking has not significantly changed with application of selectors".

While most of these statements and conclusions indirectly support the research on low F/M (AA) filament bulking reviewed above, those of Foot *et al.* (1994) and Eikelboom (1994) are particularly pertinent; (i) the problem being worst in spring (also noted by Kunst and Reins, 1994), (ii) worse with settled wastewater, (iii) the inverse relationship between the TFL and effluent TON, and (iv) the efficacy of aerobic selectors decreasing if denitrification is permitted to take place in the main aeration reactor (due to either under aeration or increased plant load), all influence the denitrification performance of the plant and increase the likelihood of significant nitrate and nitrite concentrations being present at the transition from anoxic to aerobic conditions.

1.7 Microbiological support for the AA new hypothesis

M. parvicella, which is invariably dominant in bulking activated sludges from N and N&P removal plants, has, due to its morphology, a dramatic effect on sludge settleability and therefore is the most troublesome filament in full-scale N and N&P removal plants. Tandoi *et al.* (1998) showed that *M. parvicella* strain RN1, which was isolated on agar R2A plates using Skerman micro-manipulation from a 10 day sludge age domestic wastewater activated plant in Rome and had an identical 16S rDNA sequence to *M. parvicella* strains isolated in Australia, was unable to denitrify nitrate to N_2 but only to nitrate to nitrite.

1.8 Objectives of the two research projects

The hypothesis of Casey *et al.* (1992, 1994b) and the results of Musvoto *et al.* (1992, 1994, 1999) indicate that AA filament bulking possibly can be controlled by careful design and operation of the denitrification zones of N and N&P removal systems, viz.

(1) In multi-reactor N and N&P removal systems, to control the mixed liquor (a) recycle ratio from the aerobic to the anoxic reactor such that the anoxic reactor is underloaded with nitrate so that the denitrification in it can be complete, and
(2) in single reactor intermittently aerated systems, to commence aeration only when nitrate and nitrite concentrations have reached low values.

Both these approaches were evaluated in the two research projects presented in this report. The first was evaluated at full-scale (2Mℓ/d) in the two parallel modules A and B at the Mitchells Plain WWTP and the second at laboratory scale in a single reactor intermittently aerated activated sludge system with redox potential control.

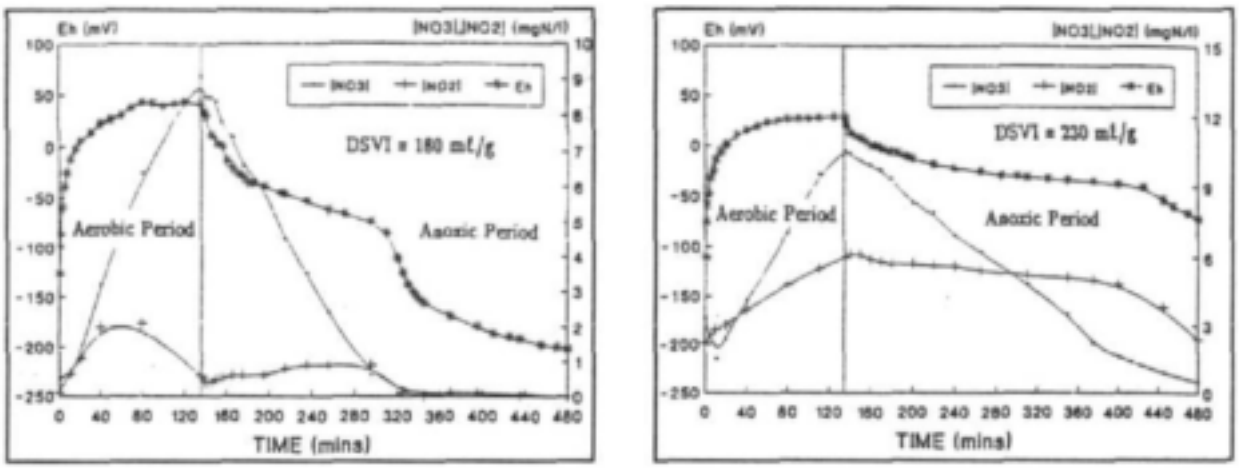
1.8.1 Full scale demonstration of filamentous bulking control at a biological nutrient removal activated sludge plant (Mitchells Plain Wastewater Treatment Works, Cape, South Africa)

Modules A and B at the Mitchell's Plain WWTP were the old Modified Ludzack Ettinger (MLE) anoxic aerobic (two reactors) and four stage Bardenpho anoxic-aerobic-anoxic-aerobic (four reactors) ND systems. The Modules A and B have settled sewage capacities of about 1.5 and 2.0 Mℓ/d and have reactor volumes of 1450 and 2084 m^3 respectively i.e. ~24 h nominal hydraulic retention time at 15 days sludge age. The two modules were built in 1975 by the Cape Metropolitan Council (CMC) to generated design data for the main plant (Modules C to F) built a few years later. The Modules A and B were converted in 1997 by the CMC to two similar UCT type NDBEPR systems for this project and to evaluated biological P removal at Mitchells Plain WWTP for possible future upgrading of the main plant to bioP removal.

After considerable delays and teething problems, the two modules were operated continuously for 589 days from 31 October, 1999 to 10 June, 2001. They were operated identically except for the mixed liquor (a) recycle ratio. In Module A this was low or zero to ensure a low nitrate /nitrite concentrations at the anoxic-aerobic reactor transition, and in Module B, this was high (>3:1) to ensure a high nitrate concentration at the anoxic-aerobic reactor transition. Following the observations of Musvoto *et al.* (1994, 1999), these different a-recycle ratios in Modules A and B should result in a good settling sludge in Module A and a poor settling sludge in Module B. The results of this research project are summarized in Section 2 in this report and a detailed report is given by Hercules *et al.* (2002).

1.8.2 AA filament bulking control with redox potential

In one of their experiments, Lakay *et al.* (1999) measured redox potential in an intermittently aerated nitrification denitrification (IAND) system. This is illustrated in Figs 1.7 and 1.8, which show the nitrate and nitrite concentrations and redox potential measured during the aerated (aerobic, which was 2.5 hours long) and unaerated (anoxic, which was 5.5 h long) periods of the single reactor IAND system with an aeration cycle of 8 hours. During the aerobic period the nitrate (and nitrite) concentration and redox potential increase due to nitrification and aeration and during the anoxic period the nitrate (and nitrite) concentration and redox potential decrease due to denitrification on non-aeration. In Fig 1.7, the nitrate and nitrite concentrations are to below detectable values during the anoxic period which in turn causes the redox potential to decline to below -200 mV. In Fig 1.8 this is not the case; the nitrate and nitrite concentrations are not reduced to below detectable values which in turn causes the redox potential to decline to only -75 mV. In these experiments, the sludge settleability for the conditions in Fig 1.7 was fair (DSVI = 180 mℓ/g) and improving with few AA filamentous organisms present, whereas sludge settleability for the conditions in Fig 1.8 was poor (DSVI = 230 mℓ/g) and deteriorating with many AA filamentous organisms present.



Figs 1.7 and 1.8: *Redox potential (Eh in mV) and nitrate and nitrite concentrations (in MgN/ℓ) for one 8h aerated-unaerated cycle on day 189 in System 5 of Lakay et al. (1999).*

From the above it was considered that if the duration of the anoxic period can be controlled with on-line redox potential monitoring in such a way that aeration only commences again when the redox potential is low (implying low nitrate and nitrite concentrations at the time that conditions become aerobic), the indications are from the AA filament bulking hypothesis of Casey *et al.* (1999c) that a good sludge settleability should be maintained in the system.

In this research project, the prospects of AA filament bulking control with redox potential are investigated. For the project a redox/DO/OUR meter was developed to control the DO and measure the OUR during the aerobic period and monitor the redox potential during the anoxic period. The investigation spanned over 960 days and during this time three laboratory scale IAND systems were operated, (i) a 3h aerobic period Experimental system (ES) from Day 1 to Day 999 controlled by redox potential to ensure a low NO_x concentration at the anoxic→aerobic transition, (ii) a similar Control system (CS) from day 251 to day 861 without redox control to ensure a high NO_x at the anoxic→aerobic transition, and (iii) from Day 832 to Day 999 a 3min aerobic 7 min anoxic period IAND system (WS) similar to those previously operated by Warburton *et al.* (1991) and Gabb *et al.* (1996) as a check against results obtained in earlier investigations. The result of this investigation are summarized in Section 3 of this report and detailed report is given by Tsai *et al.* (2002).

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SUMMARY REPORT ON RESEARCH CONTRACT K5/823 (1997-2001): FULL SCALE DEMONSTRATION OF FILAMENTOUS BULKING CONTROL AT A BIOLOGICAL NUTRIENT REMOVAL ACTIVATED SLUDGE PLANT (Mitchells Plain wastewater treatment works, Cape Town, South Africa)

This is a summary of the detailed report: Hercules *et al.* (2002), Research Report No. W91, Dept. of Civil Eng., Univ of Cape Town, Rondebosch, 7701, RSA.

2.1 MOTIVATION

From Section 1, extensive research on low F/M (Jenkins *et al.*, 1984) filamentous organism bulking in biological N and N & P removal plants has been undertaken by the Water Research Group at UCT, under previous WRC contracts (K5/286; K5/542). This research has shown that the promoted specific control method of selectors, which stimulate removal of influent readily biodegradable COD in anaerobic, anoxic or aerobic selectors by metabolic or kinetic selection (Section 1.3), is not successful in controlling bulking in biological N and N & P removal plants. It was found that the conditions that stimulate biological N removal are conducive to bulking, namely sequencing of anoxic and aerobic conditions. Hence, the low F/M filaments were renamed anoxic-aerobic (AA) to more accurately reflect the conditions under which they proliferate. From a review of the experimental results collected in the WRG laboratory, an hypothesis on AA filamentous organism bulking was developed (Section 1.4) - stated simply (but not completely), if denitrification is not complete (nitrate and/or nitrite concentrations > 2 mgN/ℓ) at the time conditions switch from anoxic to aerobic, then proliferation of AA filaments takes place (see Section 1.4 for a more detailed statement of the hypothesis).

A large body of laboratory-scale evidence supporting this alternative explanation for bulking in nutrient removal plants was accumulated in the previous WRC research contracts (Section 1.5). Also, indirect evidence from full-scale plants supporting the hypothesis is appearing in the literature (Section 1.6). However, application of specific methods to control AA filamentous organism bulking derived from the hypothesis needs to be demonstrated, both at full-scale and at laboratory-scale. Accordingly, two parallel research contracts were set up with the WRC: In Section 3, the results of the WRC research consultancy (K8/373) on application of specific control at laboratory-scale is described. In this Section a WRC research contract (K5/823) on specific control at full-scale is briefly described (for details the reader is referred to Hercules *et al.*, 2002).

A second important phenomenon observed repeatedly in the laboratory-scale investigations on NDBEPR systems as well as on external nitrification (EN) biological nutrient removal (BNR) activated sludge (AS) systems (Moodley *et al.*, 1999; Sotemann *et al.*, 2000; and Hu *et al.*,

2001), is that in these systems low COD mass balances (80 to 90 %) were observed. In contrast, on ND systems (*i.e.* no anaerobic reactor) invariably the COD mass balances are much higher (90-95 %) and closer to the theoretically expected 100 %. This inexplicable loss of COD in NDBEPR systems has also been observed in other research laboratories (*e.g.* Virginia Tech, Randall *et al.*, 1988). Because the COD mass balance is based on reconciling the COD mass exiting the system via effluent flow, sludge VSS wastage and carbonaceous oxygen demand, with the influent wastewater COD mass, the lower COD mass balance results in lower sludge production and carbonaceous oxygen demand. Indeed, so often and repeated has this low COD mass balance been observed over the past decade, that certain BNR models, coded into commercially available computer simulation packages, include this COD loss (~15 %) and so calculate reduced sludge production and oxygen demand. The mechanism whereby this COD loss takes place is via the fermentation process which transforms the readily biodegradable (RB) COD to volatile fatty acids (VFA) in the anaerobic reactor - instead of an equal concentration of VFA being generated from the RBCOD, only a fraction (~50 %) of the RBCOD becomes VFA, the balance is COD that is lost. The fermentation process has been selected for this COD loss because it is one of the few biological processes confined to the anaerobic reactor. The COD loss cannot be due to a lower anoxic heterotrophic yield coefficient (0.54 mgCOD/mg COD instead of 0.66 mgCOD/mgCOD) because this (i) doesn't affect the COD balance and (ii) happens in both ND and NDBEPR systems. The question that arises from this COD loss phenomenon is - does this COD loss also happen at large or full scale so that it is scientifically defensible to include this COD loss and design full scale plants on this basis? Because the pilot plant is potentially a large well controlled and monitored plant, it provides an opportunity *to check the COD mass balance at a scale 200 000 X larger than laboratory scale.*

2.2 RESEARCH OBJECTIVES

The objectives of the research were:

21. To demonstrate and evaluate at full-scale specific bulking control measures in N & P removal systems.
(2) To transfer research and development in bulking control undertaken at laboratory-scale to full-scale.
(3) To verify at full-scale the specific bulking control hypothesis developed from research over the past 6 years.
(4) To check the COD mass balance at pilot (semi-full) scale.

2.3 RESEARCH APPROACH

From the hypothesis for bulking by AA filaments, it is evident that if the nitrate and nitrite concentrations leaving the anoxic reactor/zone and entering the subsequent aerobic reactor/zone can be controlled to be < about 1 mgN/ℓ, then bulking by AA filaments should be minimised, and *visa versa*. ***The aim of this research project was to demonstrate this control methodology at full-scale.*** For this purpose, the 2 parallel "pilot" plants at the Mitchells Plain Wastewater Treatment Plant (WWTP) were to be used. The 2 pilot plants were refitted as N & P removal UCT configuration systems, such that the 2 systems have the same operating parameters. In one system, the nitrate/nitrite concentration in the anoxic would be controlled to be > 2 mgN/ℓ and hence in terms of the hypothesis should bulk, while in the other parallel system it would be controlled to < 1 mgℓ and hence in terms of the hypothesis should not bulk. To control the

nitrate/nitrite concentrations leaving the anoxic reactors, the a-recycle (aerobic to anoxic) would be varied to under or overload the anoxic reactors as required. In this manner, by monitoring the sludge settleability in the two systems (via the DSVI), the efficacy of the control strategy formulated from the hypothesis could be demonstrated. The pilot plants would be performance tested in sufficient detail to check the COD (and N) mass balances.

2.4 LAYOUT AND REDESIGN OF THE PILOT PLANTS

At the Mitchell Plain WWTP, two parallel modules (A and B) were built in 1975 by the Cape Metropolitan Council (CMC) to generate design data for the main plant (Modules C to F) built a few years later. The Modules A and B, appropriately modified, were used in this investigation. In this section, the layout of the original 2 parallel pilot plants (Modules A and B respectively) and their redesign to the UCT configuration are briefly described (for details, see Hercules *et al.*, 2002).

The position of the pilot plants in relation to the rest of the Mitchell's Plain WWTP is shown in Figure 2.1 (highlighted in box). A schematic layout of the pilot plants before modification is shown in Figure 2.2 with the details of the original infrastructure summarised in Table 2.1. The two pilot plants comprised the following main units:

2.41 Primary settling tank PST

A single 22 m diameter Stage 1 PST serving both Modules A and B. The primary settler had a sloped floor and primary sludge settled on the tank floor was scraped by mechanical scraper arms and collected for removal with the underflow.

2.4.2 Biological reactors

The biological reactors of the two pilot plant were not identical:

Module A: 1361 m^3 two reactor anoxic/aerobic Modified Ludzack Ettinger (MLE) N removal plant, see Figure 2.2 and Table 2.1.

Module B: 2043 m^3 four reactor anoxic/aerobic/anoxic/aerobic 4stage Bardenpho N removal plant, see Figure 2.2 and Table 2.1.

In both modules aeration was achieved via fine bubble aeration through ceramic diffuser domes served by three Roots type blowers. During the investigation this aeration system gave considerable problems due to progressive blocking of the aged ceramic domes and eventually failed irreparably resulting in termination of the project. In all the unaerated zones, sludge was mixed by bridge mounted mixers to prevent settlement. Module A had no mixed liquor 'a-recycle' whereas Module B had an Archimedean screw pump to pump mixed liquor from the ends of the aerobic zones to the head of the primary anoxic zone. At the end of the aeration zone of each module, waste activated sludge (WAS) was drawn off hydraulically to maintain the required sludge age. This WAS flow was discharged to the downstream dissolved air flotation (DAF) and linear screen sludge handling unit operations of the main plant. This sludge wastage facility caused considerable difficulties in the investigation and will be discussed further in this report.

Figure 2.1: Stage 1 pilot plants in relation to the rest of the Mitchell's Plain WWTP.

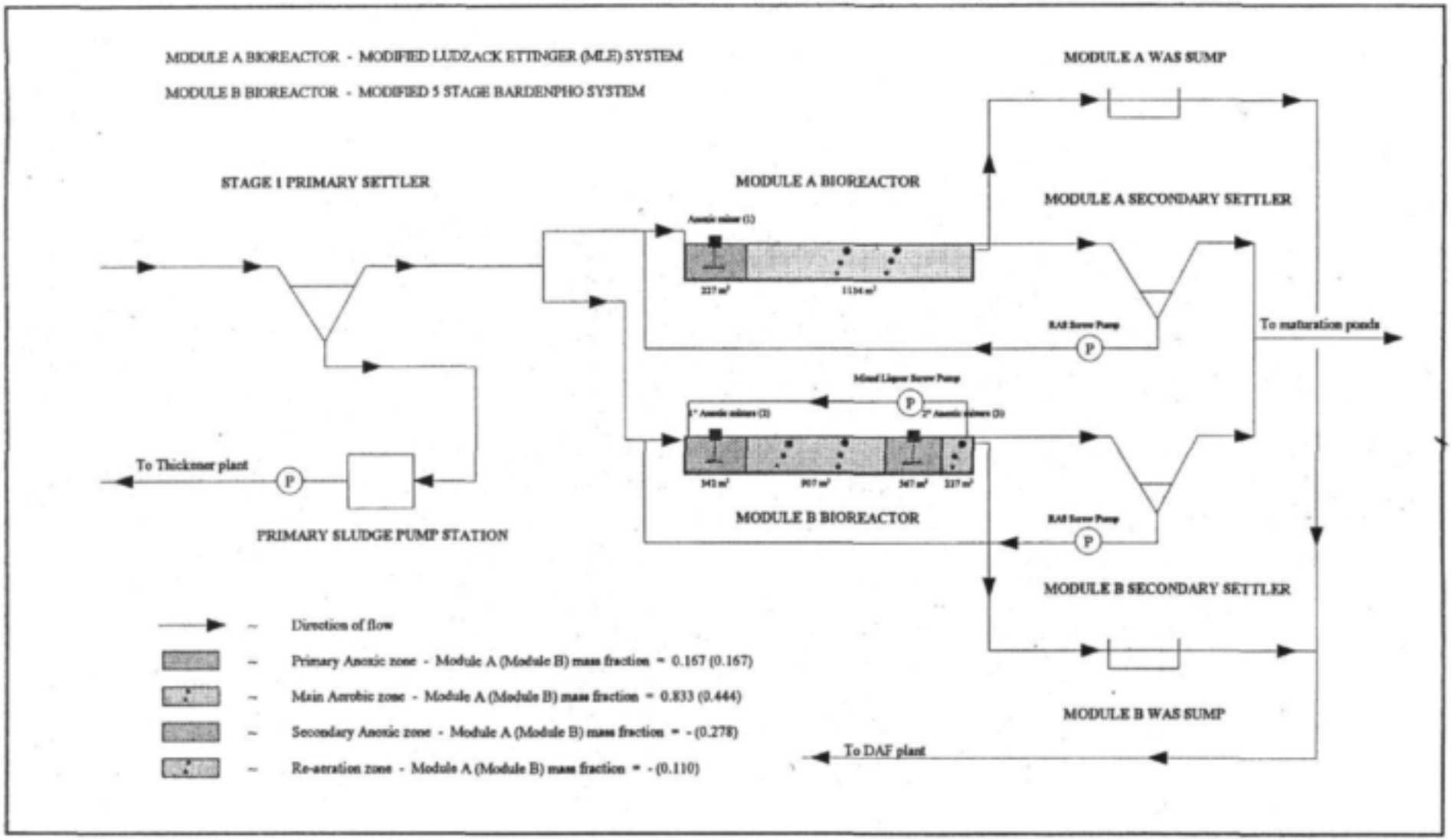

Fig 2.2: Existing Mitchells Plain pilot plants before modifications (see Table 2.1 for dimensions).

2.4.3 Secondary Settling Tanks

Each module had it's own 25m diameter flat-bottomed secondary settling tank (SST). Activated sludge settled to the tank floor was collected and returned (by suction lift) with the underflow ('s-recycle'[2]) to the sump of the return activated sludge (RAS) pumpstation from where it was pumped separately by Archimedean screws to the head of the appropriate module. Thus, the mixed liquor in the two modules was totally separated which meant that the two modules could be operated independently.

Table 2.1 : *Summary of the original pilot plant infrastructure, before modification.*

Primary Settling Tank	**Stage 1**	
Diameter (m)	22	
Area (m^2)	380	
Circumference (m)	69.12	
Side Wall Depth (m)	3	
Floor Slope (°)	15	
Biological Reactors	**Module A**	**Module B**
Total Process Volume (m^3)	1361	2043
Side Wall Depth (m)	4	4
Width (m)	7	7
Primary Anoxic Mass Fraction	0.167	0.167
Main Aerobic Mass Fraction	0.833	0.444
Secondary Anoxic Mass Fraction	None	0.278
Re-aeration Mass Fraction	None	0.110
Max. 's-recycle' (Mℓ/d)	6.2	6.2
Max. Mixed Liquor 'a- recycle' (Mℓ/d)	None	11
Primary Anoxic Zone Stirrers	One 5.5kW, 30rpm	Two 5.5kW, 30rpm
Secondary Anoxic Zone Stirrers	None	Three 4.0kW, 20rpm
Max. aeration - Aerobic Zone ($gO_2/m^3/h$)	62	77
Max. aeration - Re-aeration Zone ($gO/m^3/h$)	None	62
Secondary Settling Tank	**Module A**	**Module B**
Diameter (m)	25	25
Area (m^2)	491	491
Circumference (m)	78.54	78.54
Side Wall Depth (m)	3	3
Floor Slope (°)	0	0

[2]In this investigation, the 's-recycle' refers to the activated sludge return from the SST to the beginning of the anaerobic or anoxic reactors.

2.5 INFLUENT WASTEWATER CHARACTERISTICS

Because the influent wastewater characteristics govern to a large extent both the selection of the system and the removals of nitrogen and phosphorus attainable in the system , the characteristics of the wastewater are of prime importance for the design of the new configurations.

Two data sources were used to characterise the influent wastewater to the pilot plants:

- Weekly records of the Scientific Services Department of the Cape Metropolitan Council (CMC), and
- Two 24 hour composite sample analyses by Leballo and Sibuyi (1992).

From these two sources of data, projections of future wastewater concentrations and fractions were made and used as input to the *UCTPHO* (Wentzel *et al.*, 1992) NDBEPR simulation programme.

2.5.1 Estimated Influent Diurnal Concentrations

The estimated COD, TKN and Total Phosphate diurnal variations in concentration for the settled wastewater entering Modules A and B are given in Table 2.2. The data in Table 2.2 is based on 24hr observations of the diurnal variation in influent concentration by Leballo and Sibuyi (1992). By means of extrapolation of annual trends determined from the CMC weekly records and applying these to the diurnal variation, the expected concentrations at the estimated time for commissioning of the UCT configured pilot plants in mid. 1997, were determined.

Daily flow weighted mean values of 861 mgCOD/ℓ, 96 mgN/ℓ, 79 mgN/ℓ and 19.2 mgP/ℓ for the settled sewage were used as steady state inputs to the *UCTPHO* simulation programme for the influent COD, TKN, FSA and Total P concentrations respectively. An average influent TKN/COD ratio of 0.11 was estimated for the settled wastewater entering Modules A and B from these values.

Table 2.2 : *Estimated diurnal influent COD, TKN and Total P concentrations.*

Time of Day	COD (mgCOD/ℓ)	TKN (mgN/ℓ)	Total P (mgP/ℓ)
14 : 00	898.45	98.49	20.81
16 : 00	1014.73	96.7	20.50
18 : 00	1098.08	54.32	18.60
20 : 00	1098.06	106.84	17.66
22 : 00	1014.89	97.89	19.86
24 : 00	1014.89	87.74	15.13
02 : 00	831.89	90.73	20.01
04 : 00	449.22	60.29	18.37
06 : 00	382.66	50.14	21.98
08 : 00	798.61	121.17	17.58
10 : 00	798.61	148.03	20.87
12 : 00	831.89	139.67	19.04
Average	**861**	**96**	**19.2**

2.5.2 Estimated Influent COD, TKN and Total P Fractions

As input to the current mathematical models for activated sludge systems, it is necessary to quantitatively characterise the influent C, N & P components. Leballo and Sibuyi (1992) determined various fractions for the influent COD, TKN and Total P from sewage samples taken at the Mitchell's Plain WWTP. Table 2.3 below gives a summary of the fractions which were applied to the estimated average influent concentrations for settled wastewater given in Table 2.2. Detailed characterisation for the redesign calculations is given by Hercules *et al.* (2002).

Table 2.3 : *Estimated fractions and predicted influent concentrations.*

COD fraction	**Settled wastewater fraction**	**Settled COD concentration**	**Estimated concentration (mgCOD/l)**
f_{usi}	0.095	S_{usi}	81.62
f_{upi}	0.040	S_{upi}	34.44
f_{bsi}	0.321	S_{bsi}	238.94
f_{bpi}	0.588	S_{bpi}	506.00
TKN fraction	**Settled wastewater fraction**	**Settled TKN concentration**	**Estimated concentration (mgN/l)**
f_{nai}	0.823	N_{ai}	79.00
f_{Nousi}	0.037	N_{ousi}	3.55
f_{Noupi}	0.024	N_{oupi}	2.33
f_{Nobsi}	0.058	N_{obsi}	5.56
f_{Nobpi}	0.058	N_{obpi}	5.56
Total P fraction	**Settled wastewater fraction**	**Settled Soluble P concentration**	**Estimated concentration (mgP/l)**
f_{Psi}	0.775	P_{si}	14.89

2.6 Pilot Plant Treatment Capacity and System Operational Parameters

The UCT NDBEPR configuration was considered to be the most appropriate for this research as this configuration allows biological P removal to be relatively independent of the effluent nitrate concentration at high influent TKN/COD ratios (>0.10). Hence, it was proposed to modify the two pilot plants to the UCT N & P removal configuration, with the same anaerobic, anoxic and aerobic mass fractions.

2.6.1 Treatment Capacity Estimation

From the existing pilot plants, the process volume of the bioreactor and the area of the SST were fixed as well as the influent wastewater concentrations. This left the influent flow rate and reactor mixed liquor concentration to be determined. These were calculated with the aid of the steady state activated sludge and SST models accepting, (i) PWWF/ADWF = 1, (ii) sludge age = 15 days satisfying the requirement of the minimum sludge age for nitrification for a maximum specific growth ate of nitrifiers of 0.45/d at (iii) minimum temperature = 13 °C and (iv) a typical

bulking sludge DSVI of 250 mℓ/g (V_0 = 2.90 m/h, n = 0.615 m^3/kg). This gave an influent flow rate of 2.04 and 2.72 Mℓ/d for Modules A and B respectively, and reactor TSS concentration of 4.2 and 3.8 kgTSS/m^3 respectively. Because NDBEPR systems produce about 10 % more TSS mass and 10 % less oxygen demand per kg COD load compared with ND systems (for which the above capacity analysis is valid), the estimated influent flows were reduced to 18.75 and 2.50 Mℓ/d. Simulations of the pilot plants (see Section 2.7 below) indicated that the capacity of the pilot plants was limited by the air supply of the aeration system, rather than the SSTs. Accordingly the influent flow was reduced to 1.4 and 2.1 Mℓ/d for Modules A and B respectively. Table 2.4 gives a summary of the estimated flows and design anaerobic, anoxic and aerobic mass and volume fractions. These fractions are good for NDBEPR but were selected not so much for the optimum UCT configuration but more to suit existing baffle walls in the modules and minimal reshuffling of aeration pipework and domes.

In the proposed layouts, the 's-recycles' and 'r-recycles'[3] were each set at 1:1 with respect to the influent flow. To assess the bulking control strategy, one module (B) was operated such that the nitrate concentration leaving the anoxic reactor and entering the subsequent aerobic reactor would be > 2 mgN/ℓ, whereas for the other module (A) it would be < 1 mgN/ℓ. By monitoring the sludge settleability in the two modules and comparing these, the efficacy of the proposed bulking control strategy would be evaluated. It was proposed to control the nitrate concentrations in the anoxic reactor through the 'a-recycle' ratio: Module B would have a high 'a-recycle' ratio to overload the anoxic reactor with nitrate, and Module A would have a low 'a-recycle' to underload the anoxic reactor with nitrate. Accordingly, the Module A 'a-recycle' ratio was set at 2:1 with respect to influent flow and Module B at 5:1. Simulations with *UCTPHO* (Wentzel *et al.*, 1992, see below) indicated that these 'a-recycle' ratios would achieve the desired effect at the higher temperatures, but may require to be decreased at the lower temperatures. In practice, the 'a-recycle' ratios were adjusted depending on the measured anoxic nitrate concentrations. A system flow diagram showing the proposed operational layout is given in Figure 2.3 below.

2.7 Simulations of the Proposed UCT Systems

Swanepoel (1996) did steady state and dynamic simulations of the proposed UCT configurations with the *UCTPHO* (Wentzel *et al.*, 1992) computer programme. His results showed that at a sludge age of 23 days maximum nitrogen removal could be attained provided that sufficient 'a-recycle' pumping capacity was available. However, from a research point of view, a shorter sludge age is desirable, so that (i) the influent load could be higher and reactor TSS concentration lower and (ii) steady state can be achieved more rapidly following a system operation change. Accordingly, the sludge age of 15 days was accepted for the pilot plants.

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[3]In this investigation, the 'r-recycle' refers to the inter-reactor mixed liquor recycle pumped from the end of the anoxic reactor to the beginning of the anaerobic reactor.

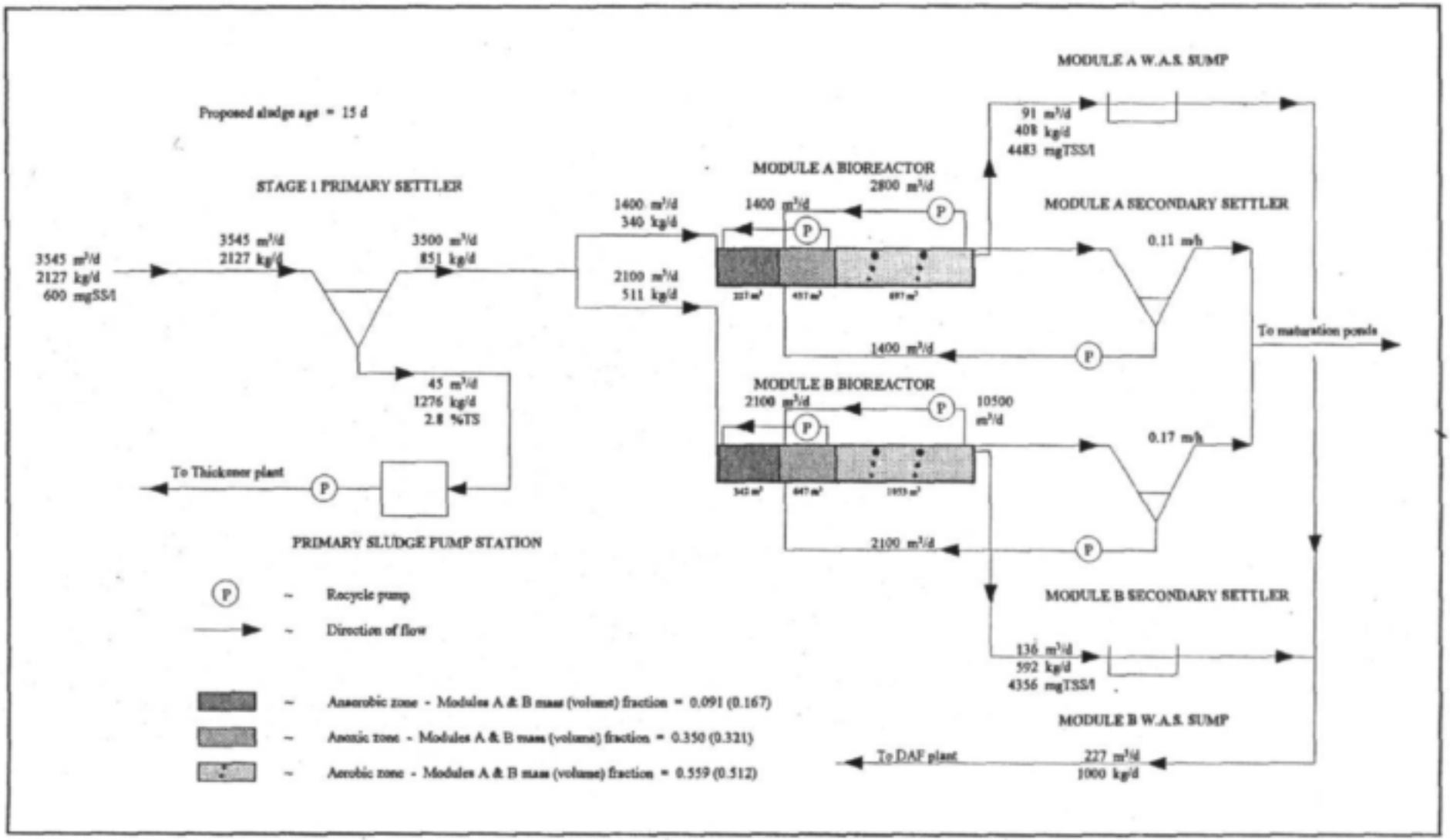

Fig 2.3: Layout of modified Mitchells Plain pilot plants as modified UCT systems.

Table 2.4 : *Predicted performance of the UCT NDBEPR pilot plants.*

Parameter	Units	Module A		Module B	
		13 °C	20 °C	13 °C	20 °C
Reactor VSS (X_v)	kgVSS/m^3	3.209	3.062	3.151	3.029
Estimated Influent flow (Q_i)	Mℓ/d	1.4	1.4	2.1	2.1
Sludge Age (R_s)	d	15	15	15	15
Anaerobic Volume Fraction	-	0.167	0.167	0.167	0.167
Anoxic Volume Fraction	-	0.321	0.321	0.317	0.317
Aerobic Volume Fraction	-	0.512	0.512	0.516	0.516
'a-recycle' Ratio*	-	2.0	2.0	5.0	5.0
'r-recycle' Ratio*	-	1.0	1.0	1.0	1.0
's-recycle' Ratio*	-	1.0	1.0	1.0	1.0
Waste Flow Rate (WAS)	m^3/d	91	91	136	136
SST Overflow Rate	m/h	0.12	0.18	0.12	0.18
DSVI	mℓ/g	250	250	250	250
Total Oxygen Demand (OUR_t)	gO/m^3/h	54.7	55.8	54.5	54.3
Peak Total Oxygen Demand	gO/m^3/h	-	64.7	-	65.1
Effluent Ammonia	mgN/ℓ	3.7	0.9	3.5	0.7
Anoxic Nitrate	mgN/ℓ	4.0	0.4	8.6	2.7
Effluent Nitrate	mgN/ℓ	20.9	18.2	18.3	12.9
Effluent Phosphate (soluble)	mgP/ℓ	1.1	0.9	2.1	1.1
Effluent COD (soluble)	mgCOD/ℓ	87.2	89.1	86.3	88.6

* All recycle ratios are with respect to the influent.

In rearranging the diffused air pipework in Modules A and B, it was found that the aeration capacity would limit the influent flows to Modules A and B at about 1.7 Mℓ/d and 2.1 Mℓ/d respectively at the influent COD and TKN concentration of 861 mgCOD/ℓ and 96 mgN/ℓ (see Section 2.6.1 above). Further, it was proposed to operate the two modules as identically as possible. Thus, the load to the two modules should be proportional to their reactor volumes, so that the reactor mixed liquor concentrations are equal. To facilitate this, the influent flow to Module B was set at 2.1 Mℓ/d. The influent flow to Module A was calculated from that for Module B as being in proportion to the reactor volumes, to give 1.4 Mℓ/d. These influent flow rates were implemented from March 1999 to February 2000 (see Section 2.11).

Accepting the layout for the pilot plants above, the expected response of the systems could be simulated with the *UCTPHO* (Wentzel *et al.*, 1992) computer programme. Results are summarised in Table 2.4 below, for the higher expected temperature of 20 °C, and the lower expected temperature of 13 °C.

From the simulations the following can be noted:

- Nitrification is achieved at all temperatures, as required, indicated by the low effluent

ammonia concentrations.

- At 20 °C, Module A anoxic nitrate concentration = 0.4 mgN/ℓ, while Module B = 2.7 mgN/ℓ - the desired difference in nitrate concentration is achieved.
- At 13 °C, Module A anoxic nitrate concentration = 4.0 mgN/ℓ, while Module B = 8.6 mgN/ℓ - while the concentration for Module B is acceptable, the concentration for Module A is too high; the 'a-recycle' ratio for Module A may require to be reduced at the lower temperatures.
- The predicted biological P removal is reasonable (effluent P = about 1 mgP/ℓ), but is reduced in Module A at the lower temperatures due to the recycling on nitrate from the anoxic to anaerobic reactors.
- The predicted VSS concentrations in Modules A and B are near identical (3062 and 3029 mgVSS/ℓ respectively at 20 °C); this indicates that the required equivalence for the two Modules has been achieved with the selected influent flow rates.
- The average and peak OURs for the two modules are virtually identical at about 55 and 65 mgO/ℓ/h respectively. Again the behaviour in the two systems is the same, as required.

2.8 Modifications to the Existing Pilot Plants to convert to UCT Systems

For the period July 1997 to February 1999 the following physical modifications, additions and alterations were made to the pilot plants to upgrade them from ND to UCT NDBEPR systems.

- Construction of recycle channels. This was achieved by using existing channels and diverting the recycle flows to the appropriate UCT configured reactors.
- The required recycles were achieved by the installation of three submersible pumps (one 7.5 kW pump for the Module A 'a-recycle' and one 3.0 kW pump each for Modules A and B 'r-recycles') together with uPVC pipework, controls and power supply.
- Variable speed control of the new submersible recycle pumps was achieved by the installation of frequency inverters.
- Installation of two new Roots type blowers with an air flow capacity of 40 m^3/min each.
- Rearrangement of the diffused air aeration pipework in Reactors A and B. The new arrangements would provide sufficient air to the aerobic zones of the new UCT configurations for maximum influent flows of 1.73 Mℓ/d for Module A and 2.1 Mℓ/d for Module B at the estimated influent COD and TKN concentrations for the settled sewage.
- Installation of two new structural steel bridges (one in each module) together with mixers in the new anoxic compartments of Reactors A and B.
- Installation of baffle walls in both reactors to separate the new anoxic and aerobic zones.
- Installation of a gate valve in the WAS pipeline. This gate had two functions viz. (i) to prevent back-flow of waste sludge from the rest of the treatment plant and (ii) to allow a sludge flow from the rest of the works for sludge seeding during start-up.
- Installation of a fine-tune sluice gate control at the main splitter box to control the flow to the primary settling tank (PST) ahead of the pilot plants.
- Installation of two fine-tune sluice gate controls at the PST splitter box to achieve the correct flow split to Modules A and B.
- Installation of influent and effluent channel screens. The former acted as a back up to the screens at the main inlet works and the latter prevented possible blockage of the pipes leading to the secondary settling tanks by floating objects in the bioreactors.
- Installation of ultrasonic flow transducers in the influent channels of Modules A and B

and dataloggers to record any variations in the influent flows.

- Installation of flow measuring plates (v-notches and rectangular weir plates) to measure all inter-reactor mixed liquor and RAS recycles.
- Installation of two dissolved oxygen probes with a recording facility.
- Installation of scum traps to aid in containing the scum.
- Installation of new waste flow meters and measuring plates.
- Installation of new guardrails on the pilot plants.
- Installation of new scum scraper arms on all the settling tanks.
- Repairs to broken diffuser pipework and the installation of new condensation lines at various points along the diffuser pipework.
- Repairs to the primary and secondary settling tanks.

2.9 Process Control and System Calibration

In the operation of the pilot plants, control of the influent flows and the various recycles is essential, to achieve the required conditions in the two modules. Also, adequate aeration is required to ensure nitrification. Control of these parameters is described below.

2.9.1 Influent flow control and measurement

The raw sewage to the Stage 1 (Modules A and B) PST was controlled at the main flow splitter of the Mitchell's Plain WWTP. Each overflow weir at the splitter was fitted with an adjustable sluice gate to achieve flow control to the pilot plants and the rest of the Mitchell's Plain WWTP. The overflow (settled sewage) from the Stage 1 PST was controlled by means of adjustable sluice gates installed in the distribution box at the end of the PST which allowed appropriate flow distribution between Modules A and B. The influent to each module was measured by means of ultrasonic flow transducers located at measuring points upstream of v-notch plates in their respective inlet channels ahead of the anaerobic reactor. Flows to Modules A and B were recorded for monthly downloading by means of electronic loggers and totalisers located in the electrical switchroom at the Stage 1 blower house.

2.9.2 Dissolved oxygen control and measurement

Control of the dissolved oxygen (DO) concentration was achieved by means of valves on the main air manifolds and a pressure-relief valve outside the blower house. These valves could be adjusted to increase or decrease the aeration in the aerobic reactors of Modules A and B. The DO concentrations in the aeration basins were recorded by DO meters located in the middle length of the aeration basin on the side wall. The DO concentrations were continually recorded on paper reams kept for record purposes. The DO probes were supposed to be regularly checked and calibrated by CMC maintenance staff with portable YSI DO meters but this did not happen.

2.9.3 Mixed liquor recycle control and measurement

In both Modules A and B variable speed control of the mixed liquor 'r-recycle' and 'a-recycle' of Module A was achieved by frequency inversion. The potentiometer of the frequency inverter allowed for the selection of any frequency in the range of 20 to 60 H_z thereby giving flexibility in the choice of flows. The Module B 'a-recycle' was controlled by a penstock at the end of the aerobic zone. By manually adjusting the penstock the flow to the Archimedean screw pump could be adjusted thereby achieving control of the recycle. Both Modules A and B 's-recycle' (RAS) flows did not have flow control and their flows were determined by the rate of sludge withdrawal from the SSTs. Each RAS Archimedean screw pump was independent and

discharged the sludge return into separate channels. All the recycle flows were measured by rectangular or v-notch weir plates where the height above the weir was recorded and used in the calculation the flow.

2.9.4 Waste flow control and measurement

Sludge age in the modules was maintained by hydraulic control, *i.e.* wasting the required sludge directly from the aeration basin at a rate per day of 1/15 of the effective system volume. The waste activated sludge (WAS) flow from each reactor was taken directly from the end of the aerobic reactor via a valve and discharged into a sump where the flows were measured across rectangular weir plates, before discharge to the WAS pipeline.

2.10 Commissioning and Operational Adjustments to the Pilot Plants

The pilot plants were commissioned on the 15 March 1999 with influent flows to Modules A and B initially set at 1.4 Mℓ/d and 2.1 Mℓ/d respectively which is in proportion to their respective total system volumes and satisfied the limitation imposed by the diffused air system. During the commissioning stage (March 1999 to October 1999) various problems were encountered which delayed the start of the system performance monitoring by a year. The main problem was the fragile/poor condition of the diffused air system mainly due to its already extended life span. Numerous holes appeared along the PVC pipework at each start-up causing poor air distribution in the aerobic reactors of both modules and air leaks into the anoxic reactors (parts of which were formerly part of the aerobic reactors). Both modules had to be completely drained before remedial work (fibreglassing) could proceed and the rapid rate of infiltration of underground water into the drained tanks made this repair work difficult. Despite these difficulties both modules received adequate air supply at their selected influent flows after the repairs and external surface cleaning had been done.

Another related problem experienced during commissioning was the considerable electrical problems with both new blowers. At the desired influent flows with the pressure-relief valve shut, both blower circuits drew high current and consequently tripped on thermal overload as the power absorbed at the blower shaft exceeded the rated power of the motor. This led to insufficient aeration in the two reactors ultimately causing Module A to be decommissioned. In order to remedy the problem with the new blowers, the motor pulley size was reduced which in turn reduced the torque and power absorbed by the blower. This resulted in a lower current drawn, together with a decrease in air flow. The initial air flow capacity of the blowers was 40 m^3/min each, but this was reduced to just over 37 m^3/min. This slightly lower air flow would have little effect on the two pilot plants because the treatment load were set lower than the oxygen supply limit of the system. However, the aeration system continued to give problems in the investigation mainly due to a progressive decline in oxygen supply due to progressive blocking of the ceramic domes. This resulted in reduced oxygen supply to the aerobic reactors stimulating simultaneous nitrification-denitrification in them.

Another problem was the excessive accumulation of scum in both the unaerated and aerated zones. On investigation it was discovered that the scum formed in the aeration zone and reached the unaerated zones mainly with the surface back-flow across the submerged aerobic-anoxic transition baffles and also via the 'a-recycle'. During this period the whole Mitchell's Plain WWTP experienced problems with scum. It was generally accepted that the high fat content of the influent wastewater to the Mitchell's Plain WWTP was the cause of the high level of scum

on the surface of all the bioreactors. In order to prevent the accumulation of scum with its aesthetic and odour nuisances, it was decided to explore ways in which to encourage forward flow of the surface liquid to prevent back-flow from one zone to another. In this regard the appropriate baffle wall submergence and underflow drainage area based on the total flow passing the baffle was calculated and the anoxic-aerobic baffles were modified accordingly. In hindsight, while this effectively dealt with the foaming problem, the reduction in 'a-recycle' flows in both modules caused an increased back mixing of mixed liquor from the aerobic to the anoxic reactors, which adversely affected their denitrification efficiency.

Remedial measures taken for other minor problems encountered after start-up included :

- Repairs to the Rotork pressure-relief valve on the main air manifold at the blower house.
- Overhauling of Module B mixers.
- Repairs to all Module B penstocks to achieve adjustable control.
- Electrical repairs to the Module B mixed liquor screw pump and Module A mixed liquor submersible pumps.

2.11 System Operating Conditions and Changes during Experimental Periods

For the period October 1999 to June 2001, sustained operation of the UCT configured pilot plants was achieved with no significant system downtime experienced. This Section describes the operational changes and experimental results of the investigation which lasted 589 days. At the end of the Section a brief description of the experimental results of a parallel laboratory-scale UCT system is presented. This lab system was operated to assess the magnitude of the operation problems on the performance results pilot plants.

In the operation of the pilot plants, from an examination of the influent, operational parameters and measured system performance, eight periods were identified where these remained approximately constant during the 589 day investigation, see Table 2.5.

Table 2.5: 'Steady state' long term periods of the investigation.

Long Term Period No.	Period	Day No.		No. of days	No. of sampling days
		From	To		
I	30/10/99 - 11/06/00	1	224	224	30
II	12/06/00 - 27/09/00	225	332	108	57
III	28/09/00 - 21/11/00	333	387	55	29
IV	22/11/00 - 14/12/00	388	410	23	12
V	15/12/00 - 26/01/01	411	453	43	15
VI	27/01/01 - 07/03/01	454	493	40	23
VII	08/03/01 - 04/05/01	494	551	58	26
VIII	05/05/01 - 11/06/01	552	589	38	20

The operating parameters for Modules A and B for these long term periods are listed in Table 2.6. During the investigation a number of incidents and changes to the operating parameters were made and are recorded in Table 2.7.

Table 2.6 : *Pilot plant operating parameters.*

System Parameter	Mod.	Long Term Period No.							
		I	II	III	IV	V	VI	VII	VIII
Sludge age (d)[4]	A	14.2	13.3	11.5	14.5	14.2	14.1	14.2	14.1
	B	14.1	13.1	11.5	14.4	14.2	14.1	14.1	14.1
Influent (Mℓ/d)	A	1.49	1.47	1.52	1.13	1.63	1.91	1.70	1.90
	B	2.24	2.24	2.14	1.57	2.12	2.28	2.20	2.23
'a-recycle' (Mℓ/d)	A	3.20	1.80	0.00	0.00	0.00	0.00	0.00	0.00
	B	9.60	8.76	5.90	7.73	6.51	6.45	5.54	4.61
'a-recycle' ratio*	A	2.3	1.2	0.0	0.0	0.0	0.0	0.0	0.0
	B	4.6	3.9	2.8	4.9	3.1	2.8	2.5	2.1
's-recycle' (Mℓ/d)	A	5.20	4.75	1.77	3.20	5.20	3.44	2.24	2.18
	B	5.20	4.73	1.93	2.05	5.05	4.02	2.52	2.75
's-recycle' ratio*	A	3.7	3.2	1.2	2.8	3.2	1.8	1.3	1.1
	B	2.5	2.1	0.9	1.3	2.4	1.8	1.1	1.2
'r-recycle' (Mℓ/d)	A	1.40	1.58	1.50	1.41	1.45	1.41	1.51	1.61
	B	2.40	2.40	2.41	2.40	2.40	2.15	2.16	2.16
'r-recycle' ratio*	A	1.0	1.1	1.0	1.2	0.9	0.7	0.9	0.8
	B	1.1	1.1	1.1	1.5	1.1	0.9	1.0	1.0
Waste flow (m^3/d)	A	87.58	87.58	87.58	87.58	87.58	87.58	87.58	87.58
	B	132.67	132.67	132.67	132.67	132.67	132.67	132.67	132.67
Aerobic DO' (mgO/ℓ)	A	2.00	2.00	1.80	0.15	0.35	1.70	1.95	1.90
	B	2.10	2.00	2.10	2.00	0.90	2.05	0.50	1.70
Temperature of sludge (°C)		17 - 25	17 - 25	17 - 25	17 - 25	17 - 25	17 - 25	17 - 25	17- 25
Influent TKN / COD ratio		0.11-0.19	0.10-0.24	0.13-0.19	0.10-0.18	0.13-0.17	0.14-0.20	0.14-0.19	0.14-0.19

* All recycle ratios with respect to the influent.' Average aerobic DO concentration for the 'steady state' period.

[4]From Section 3.5, the proposed design sludge age was 15 days. However in the UCT NDBEPR configuration the sludge age needs adjustment as the sludge concentration in the anaerobic reactor is about half that of the anoxic and aerobic reactors due to a dilution effect of the influent entering the anaerobic reactor. Therefore the effective anaerobic volume is a factor R/(1+R) multiplied by the actual volume, where R is the 'r-recycle' ratio.

Table 2.7 : *Record of incidents and system operational changes to the pilot plants.*

Description of incidents or changes	Day No.		Action Taken
	From	To	
Blower's non-return valve faulty causing poor aeration.	120	121	Air switched off. Non-return valve repaired.
Influent re-split to Modules A & B.	122	589	Flows set at 1.6 & 2.4 Mℓ/d.
Module B SST rotating bridge stuck causing solids up-flow.	140	141	Obstruction removed.
Excessive scum build-up in aerobic zone of Reactor A.	193	201	Scum gate opened to allow scum to leave via waste flow.
Local power failure.	212	213	Electricity supply reinstated.
Blower 1 no longer in operation.	262	262	Standby blower switched on.
Reactor A anaerobic mixer down.	264	276	Motor repaired.
NO_3 load on Module A anoxic reactor too high.	306	589	Module A 'a-recycle' set to 0, 's-recycle' reduced.
Very low flow into Stage 1 PST.	328	331	Blockage at splitter cleared.
Module A RAS screw pump down.	369	370	Motor repaired.
Module B mixed liquor 'a-recycle' screw pump down.	370	375	Motor replaced. 'S-recycle' increased to compensate.
Low MLSS in Module A - no 's- recycle'. Flow and air stopped on days 386 & 387.	383	397	Blockage in SST A pipework cleared. Flow and air reinstated.
Blockage in Module B 's- recycle'.	432	435	Blockage removed.
Flow to Module A stopped ?	450	451	Flow reinstated.
Increased aeration to Module A.	458	589	Control valves gradually adjusted.
Air leakage at Module A air manifold downpipe.	497	589	Situation monitored.
Solids carryover at Module B SST.	513	516	Samples treated.
Blower No. 2 tripping on thermal over-load causing poor aeration.	532	549	Blower restarted and reset each time to prevent tripping.
Module B mixed liquor 'a-recycle' screw pump down.	540	563	Screw pump's lower bearing replaced.
Modules A and B diffused air system failure.	588	589	Modules A & B decommissioned.

From Tables 2.6 and 2.7:

- A sludge age of approximately 14 days was achieved in both modules. However, difficulties were experienced in maintaining a constant sludge age (see below).
- From the design data for the influent (Section 2.6), the design influent flow for Module A was 1.4 Mℓ/d and for Module B 2.1 Mℓ/d. During Period I, it was found that the influent COD concentration was much lower than the expected 861 mgCOD/ℓ (see Table 2.2), at about 600 mgCOD/ℓ. Thus, from day 122, Period I the influent flow rates were increased to 1.6 and 2.4 Mℓ/d for Modules A and B respectively. The recorded influent flow rates are reasonably close to these values. Difficulty was experienced in attaining the exact flow rates as construction work at the inlet works and the equalisation basins caused unusual fluctuations in the influent flow.
- In the design, the 'a-recycle' ratios for Modules A and B were set at 2:1 and 5:1 respectively, but it was noted that these probably would have to be decreased at the lower temperatures. In operation, these recycle ratios were closely achieved for Period I (2.3 and 4.6:1 for Modules A and B respectively). However, from plant performance monitoring it was noted that the nitrate concentrations in the anoxic reactors of both modules were excessively high (~5 mgN/ℓ), which caused significant nitrate to be recycled to the anaerobic reactor, adversely influencing biological P removal. These high anoxic nitrate concentrations were due in part to the high 's-recycle' ratios (see below), and the high TKN/COD ratio of the settled wastewater. Accordingly on day 306, the 'a-recycle' ratios for both modules were reduced, as recorded in Table 2.7.
- In the design, the 's-recycle' ratio was set at 1:1. In operation of the modules, due to the suction lift draw-off from the SSTs, it was found not possible to control the 's-recycles' at the proposed 1:1 as this was too low to maintain suction lift. The 's-recycle' flow rates were measured in both modules, as recorded in Table 2.6. These flow rates were accepted for operation.
- For Periods I to III, the aerobic DO concentration was adequate at about 2 mgO/ℓ. These three periods which represent the longest experimental periods of the investigation (up to day 387 of 589) with the most stable plant operation (spanning about 27 sludge ages) gave good effluent quality results. However, from period IV difficulty in maintaining adequate aerobic DO was experienced in both modules due to the deteriorating condition of the diffused air system (see below).

During the first half of the investigation, difficulty in maintaining the desired sludge age was experienced due to considerable difficulties with activated sludge wastage from the pilot plants. The underlying cause was the sludge handling facilities (DAF, linear screen etc.) downstream which had inadequate capacity to deal effectively with sludge wastage from the pilot plants and the rest of the treatment plant. As the pilot plants shared a common waste activated sludge pipeline with the rest of the treatment plant, waste sludge back flows into the pilot plant were often experienced. The back flow of sludge into the pilot plants from the main plant was substantial and resulted in reactor sludge concentrations at times in excess of 60 % of those that would normally accumulate in the reactor from the influent wastewater. The seriousness and significance of this problem and its potential to compromise the objectives of the investigation were communicated to the Cape Metropolitan Council (CMC). The construction of an independent sludge wastage pipeline and pump was requested. However, the CMC stated it did not have funds to install such a system and instead put a plant operator's sludge wastage protocol in place which if adhered to would have limited sludge ingress into the pilot plants. The results

indicate that this protocol did not successfully eliminate the problem, because if it did, the pilot plants would not have accumulated so much more sludge than can reasonably be expected. To alleviate this problem a gate valve was installed in the existing sludge wastage pipeline but this also proved to be ineffective as the gate needed to be opened when wasting from the pilot plants. Often when this gate was opened the problem was encountered due to simultaneous wastage from the pilot plants and the rest of the plant. Finally, a new waste schedule for the entire treatment plant was set up which allowed wasting from the pilot plants without simultaneous wasting from the rest of the plant. This strategy had a fair amount of success but sporadic large fluctuations in reactor solids concentrations were still noted in the pilot plants. This persistent problem throughout the investigation compromised the second objective of the investigation *i.e.*, checking COD balances for possible anaerobic reactor COD losses. The COD balance requires the oxygen utilisation rate (OUR) to be measured on both pilot plants. Measurement of OUR is very labour intensive - it has to be measured hourly at 3 different places in the aerobic reactor over 24-30 hour period in both pilot plants while all the other parameters were also measured. It was planned to do this only when reasonable surety of "steady state" conditions in the pilot plants was achieved, which due to this sporadic ingress of sludge did not happen. Nevertheless, 2 x 24 hour tests were done on the pilot plants by Modipa and Diale (2000) for their BSc theses. The COD and N balances obtained in these two tests were 150 & 44 % and 83 & 74 % respectively, too deviant to be useful for the second project objective and simulation.

In operation the most serious problem experienced was with the diffused air system. The ceramic diffuser domes became increasingly blocked as the investigation continued. To attempt to maintain aeration to ensure complete nitrification and avoid simultaneous denitrification in the aerobic reactor, the pressure relief valves on the blowers were shut. This increased the back pressure in the system to the extent that the blowers often tripped on thermal overload, despite adjustment to the blower pulleys made earlier. On day 588 (10 June 2001) the back pressure in the diffused air system became so excessive that the end caps on the PVC piping blew out causing structural damage to the pilot plants and the investigation had to be terminated.

2.12 Data Acquisition and Sample Analysis

System performance monitoring commenced on 31 October 1999. Initially the plant was monitored once a week on Sundays by the Scientific Services Department of the CMC. The sampling on Sundays was done independently of this project for the CMC's own weekly laboratory records. Table 2.8 shows the samples taken and typical analyses which will be termed "CMC" data.

From June 2000 more extensive sampling and analysis was carried out on the pilot plants every second day, excluding Sundays. These analyses will be termed "site" data. Table 2.9 shows the sampling and analysis schedule. The influent and effluent samples were 24 hour composites, and the reactor samples were grab samples. The results from the 2 x 24 hour tests showed that the daily variation in reactor concentrations were small due to constant flow and long retention time ~ 8 hr (see Section 5.6) so that the grab samples can be accepted as composite equivalent. All grab samples from the various locations in the reactors were filtered immediately on-site and further prepared if necessary (e.g. flocculation and filtration) before being stored in a refrigerator for analysis the following day. Chemical and physical analyses of these samples were done at the Mitchells Plain WWTP laboratory and the Scientific Services laboratory. Table 2.10 below shows typical results for the "site" data. Filament identifications were done once monthly.

Table 2.8: *Typical sample sheet for "CMC" sampling and analysis. Sampling done weekly every Sunday.*

MITCHELL'S PLAIN WWTW - PILOT PLANT : SCIENTIFIC SERVICES WEEKLY LAB. REPORT

Date : 10 April 2000

SAMPLE	UNITS	MODULES A & B INFLUENT	MODULE A MIXED LIQUOR / ACTIVATED SLUDGE							MODULE B MIXED LIQUOR / ACTIVATED SLUDGE							MODULE A EFFLUENT	MODULE B EFFLUENT
			Anaerobic	Anoxic	Anox./Aer. Transition	Aerobic	R-recycle	A-recycle	S-recycle	Anaerobic	Anoxic	Anox./Aer. Transition	Aerobic	R-recycle	A-recycle	S-recycle		
Unfiltered COD	mg COD/l	673.00															58.00	34.00
Filtered COD	mg COD/l	313.00															48.00	34.00
Unfiltered TKN	mg N/l	118.20																
Filtered TKN	mg N/l																	
Flocculated/Filtered TKN	mg N/l																	
Filtered FSA	mg N/l	73.20															0.10	0.10
Filtered NO_3	mg N/l																	
Filtered NO_2	mg N/l																	
Filtered NO_3 + NO_2	mg N/l																13.80	13.50
Unfiltered Total P	mg P/l	16.10																
Filtered Total P	mg P/l																	
Filtered Ortho P	mg P/l																11.60	9.60
pH		7.30				6.30							6.30				6.70	6.60
Alkalinity	mg/l as $CaCO_3$	420.00															39.00	30.00
TSS	mg MLSS/l	158.00				2540.00							3680.00				5.00	1.00
VSS	mg MLVSS/l					2250.00							3090.00					
SVI	ml/g					83.00							98.00					
DSVI	ml/g					79.00							82.00					

Table 2.9: *Sampling procedure and sample analysis implemented every second day - "site" samples.*

MITCHELL'S PLAIN WTW PILOT PLANT : SAMPLING & LABORATORY TESTING											
SAMPLE	**MEASUREMENT PARAMETER**										
	COD	**TKN**	**TP**	**NO_3**	**NO_2**	**FSA**	**OrthoP**	**VSS & TSS**	**DO**	**SVI & DSVI**	**pH & ALK**
Influent[1]	i.) Unfiltered ii.) Floc./filt. (0.45µm)	i.) Unfiltered ii.) Filtered (0.45µm)	i.) Unfiltered ii.) Filtered (0.45µm)	i.) Filtered (0.45µm)	i.) Filtered (0.45µm)	i.) Filtered (Whatman's No. 1)	i.) Filtered (Whatman's No. 1)	i.) Influent samples			i.) Influent pH ii.) Influent alkalinity
Anaerobic			i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]						
Anoxic			i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]						
Aerobic			i.) Unfiltered ii.) Filtered (What's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]			i.) Reactor samples	i.) Reactor sample	i.) Reactor sample	i.) Reactor pH
Effluent[1]	i.) Unfiltered ii.) Floc./filt. (0.45µm)	i.) Unfiltered ii.) Floc./filt. (0.45µm)	i.) Unfiltered ii.) Filtered (0.45µm)	i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)	i.) Filtered (Whatman's No. 1)				i.) Effluent pH ii.) Effluent alkalinity
R - recycle[4]				i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]						
A - recycle[4]				i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]				i.) Channel sample		
S - recycle[4]				i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]				i.) Channel sample		
Anoxic-Aerobic Transition[3]				i.) Filtered (Whatman's No. 1)[2]	i.) Filtered (Whatman's No. 1)[2]						

Notes :

1. Influent and effluent samples to be composite samples. All other samples to be grab samples.
2. All Whatman's No. 1 paper filtering, with the exception of the FSA & OrthoP samples, to be done at time of sampling to prevent denitrification.
3. The anoxic - aerobic transition sample to be a composite filtrate of grab samples taken at the upper, middle & lower sections on aerobic side of baffle.
4. The A, R & S - recycle samples to be taken in the channels close to the end of the pipework.
5. Shaded areas are CRITICAL i.t.o. sampling & testing.
6. Filament identification to be done once a month.
7. Other measurement parameters to be taken are all flows, wasting, temperature & rainfall.

Table 2.10: *Typical sample sheet for "site" sample and analysis. Sampling done every second day, excluding Sundays.*

MITCHELL'S PLAIN WWTW - PILOT PLANT : LABORATORY REPORT

Date : 29-08-2000

SAMPLE	UNITS	MODULES A & B INFLUENT	MODULE A MIXED LIQUOR / ACTIVATED SLUDGE							MODULE B MIXED LIQUOR / ACTIVATED SLUDGE							MODULE A EFFLUENT	MODULE B EFFLUENT
			Anaerobic	Anoxic	Anox./Aer Transition	Aerobic	R-recycle	A-recycle	S-recycle	Anaerobic	Anoxic	Anox./Aer Transition	Aerobic	R-recycle	A-recycle	S-recycle		
Unfiltered COD	mg COD/l	541.00															41.00	37.00
Flocculated/Filtered COD	mg COD/l	220.00															37.00	32.00
Unfiltered TKN	mg N/l	95.50															3.30	1.00
Filtered TKN	mg N/l	86.00																
Flocculated/Filtered TKN	mg N/l																2.70	0.70
Filtered FSA	mg N/l	65.00															2.00	0.50
Filtered NO_3	mg N/l	0.182	0.167	4.312	4.849	11.539	4.198	10.474	8.913	0.087	6.457	5.788	16.527	7.665	16.634	15.644	10.910	16.815
Filtered NO_2	mg N/l	0.059	0.202	0.154	0.110	0.082	0.101	0.058	0.079	0.000	0.447	0.142	0.042	0.225	0.029	0.021	0.135	0.064
Filtered NO_3 + NO_2	mg N/l	0.100															10.200	15.800
Unfiltered Total P	mg P/l	12.50				225.00							167.00				0.20	0.50
Filtered Total P	mg P/l	10.30	33.90	9.00		0.10				27.80	7.30		0.10				0.10	0.30
Filtered Ortho P	mg P/l	8.90															0.10	0.20
pH		7.24				6.63							6.51				6.65	6.55
Alkalinity	mg/l as $CaCO_3$	300.00															67.00	35.00
TSS	mg ML/SS/l	134.00				3782.00							3917.00					
VSS	mg ML/VSS/l	115.00				2876.00							3053.00					
SVI	ml/g					120.00							105.00					
DSVI	ml/g					95.00							87.00					

Notes :
1. The influent and effluent samples are composite samples. All other samples are grab samples.

Graphs and tables of the bi-daily "site" data and weekly "CMC" data are given by Hercules *et al.* (2002). Table 2.11 below shows the analytical methods applied top obtain the measured parameters.

Table 2.11 : *Analytical methods applied to test parameters.*

Test Parameter	Analytical Method
COD	Standard Methods (1985)
TKN	Standard Methods (1985)
Total P	Lachat 8000 Flow Injection Analysis (FIA) colorimeter - QuikChem® Method 10-115-01-1-E10-115-01-1-E (Acid-Persulphate Digestion)
Ortho P	FIA colorimeter - QuikChem® Method 10-115-01-1-A
NO_3, NO_2	HP LC Auto Analyser
FSA - Low/High	FIA colorimeter - QuikChem® Method 10-107-06-1-A/H
VSS, TSS	Solids separation by centrifugation, drying in a crucible at 105 °C, incinerating at 600 °C
SVI, DSVI	Lee *et al.* (1983) or Ekama and Marais (1984)
pH	Schott Gerate Digital Lab. pH meter CG 825
Alkalinity	FIA colorimeter - QuikChem® Method 10-303-31-1-A
Flocculation - influent and effluent samples	1l sample subjected to 10ml of 0.25M Aluminium Sulphate flocculant concentration and allowed to settle for 1hr.
Filtration - reactor samples	Filtered through Whatman® GF/C glass microfibre filters.
Filtration - effluent samples	Filtered through Millipore 0.45µm sterile membrane filters.
DO - aerobic zone	Zullig Model DO94 and GLI Int. model D53 fixed DO meters.
DO - various	Yellow Springs Int.(YSI) Model 55 portable DO meter

2.13 Measured System Performance

For each "steady state" period given in Table 2.5, the appropriate data were averaged and are presented below. For Period I (the first 224 days of the investigation) only the "CMC" data were examined to monitor the system performance of the pilot plants, because the "site" data monitoring programme started only in Long Term Period II.

2.13.1 Influent Data

The influent data averaged for each of the long term periods in Table 2.5 are given in Table 2.12 below, for both the weekly "CMC" analysis and the more detailed bi-daily "site" analysis.

From Table 2.12 :

- The influent COD concentrations (averages 653 and 614 mg/ℓ) are much lower than the expected 861 mg/ℓ (Section 3.3.1) determined for the design. For this reason the influent flow rates were to the pilot plants were increased from 1.4 and 2.1 Mℓ/d for Modules A and B respectively, to 1.6 and 2.4 Mℓ/d respectively from Period II (Section 4.1 above).
- The average flow rates (1.60 and 2.16 Mℓ/d for Modules A and B respectively) indicate that the desired flow rates above were closely achieved.
- The influent TKN (averages 95.3 and 98.0 mgN/ℓ) is very similar to that determined for the design (96 mgN/ℓ).
- The high influent TKN concentration together with the reduced COD concentration, indicates that the PST removals were better than expected (COD removal in primary sedimentation is higher than TKN removal, due to the large soluble free and saline ammonia (FSA) fraction in the TKN). This caused the influent TKN/COD ratio to be considerably higher than expected, 0.15 to 0.16 mgN/mgCOD compared to 0.11 mgN/mgCOD.
- The influent total phosphorus (P) concentrations (averages 14.4 and 12.7 mgP/ℓ) are significantly lower than the concentration accepted for the design (19.2 mgP/ℓ). This is unexpected since a large proportion of the P is soluble and should not be influenced by the better than expected primary sedimentation.
- Influent nitrate + nitrite measurements (not shown here) were < 0.5mgN/ℓ indicating these to be present in the influent at negligible concentrations, as expected.

Table 2.12 : *Averaged influent data for steady state periods; averages are for "CMC" (weekly) and "site" (bi-daily) data.*

Long Term Period No.	Flow Rate (Mℓ/d)		COD (mgCOD/ℓ)			TKN (mgN/ℓ)			FSA (mgN/ℓ)		Total P (mgP/ℓ)	
	Mod. A	Mod. B	CMC data	Site data	Floc/ Filt	CMC data	Site data	Filt	CMC data	Site data	CMC data	Site data
I	1.40	2.10	666	N/A	N/A	90.4	N/A	N/A	72.3	N/A	14.8	N/A
II	1.47	2.24	664	567	254	96.4	94.8	81.6	75.3	68.8	14.8	13.1
III	1.52	2.14	634	612	268	98.8	98.0	85.9	76.1	67.8	13.8	13.0
IV	1.13	1.57	649	653	302	87.8	92.2	80.1	67.6	65.4	14.0	12.8
V	1.63	2.12	640	676	427	92.9	97.6	86.0	75.2	67.6	13.4	12.7
VI	1.91	2.28	630	653	292	100.8	99.9	85.7	77.2	70.0	13.9	12.4
VII	1.70	2.20	631	612	230	103.4	101.2	88.9	82.8	69.6	13.8	12.2
VIII	1.90	2.22	648	616	239	105.3	104.2	90.5	84.5	66.7	14.1	12.3
Ave.	1.61	2.17	653	614	278	95.3	98.0	83.2	75.4	68.3	14.4	12.7

Note : Influent COD, TKN, FSA and Total P = unfiltered. FSA = free and saline ammonia. CMC averages include Period I, whereas site averages do not (N/A). Flocculated/filtered influent TKN was not measured, but 0.45 μm filtered was (see Table 2.10).

2.13.2 Reactor Parameters

The various averaged reactor concentrations for the steady state periods are listed in Tables 2.13 and 2.14 below. In Table 2.14 only the "site" data is listed as the "CMC" data did not include these measurements.

The averaged reactor concentrations will be discussed in more detail in Section 5, but immediately apparent from Tables 4.10 & 4.11 and Figures 4.1 to 4.14 below, are:

- Nitrite concentrations were negligible throughout the plant.
- Anoxic nitrate concentrations were high for both modules, except when nitrification failed, in particular Module A Periods IV and V.
- An apparent anomaly in the nitrate data exists; anoxic reactor nitrate concentrations were high (~5mgN/ℓ), yet effluent nitrate concentrations were relatively low (~13 mgN/ℓ). It became evident during the investigation that the main cause for this was inadequate aeration and significant back mixing of aerobic reactor mixed liquor into the anoxic reactors resulting in quasi anoxic quasi aerobic conditions in both the anoxic and aerobic reactors leading to simultaneous nitrification-denitrification in these reactors, particularly in Module A.
- The reactor P concentrations follow the typical biological P removal pattern, with anaerobic P release followed by subsequent P uptake in the anoxic and/or aerobic reactors.
- The overall average TSS and VSS concentrations in the 2 modules are similar, as required (~3600 mgTSS/ℓ and 2700 mgVSS/ℓ).
- The DSVI is consistently low (~100 mℓ/g).

Table 2.13 : *Averaged reactor mixed liquor data for steady state periods; averages are for "site" data which excludes Period I (N/A) and "CMC" data.*

Long Term Period No.	Site TSS (mgTSS/ℓ)		CMC TSS (mgTSS/ℓ)		Site VSS (mgVSS/ℓ)		CMC VSS (mgVSS/ℓ)		Site DSVI (mℓ/gTSS)		CMC DSVI (mℓ/gTSS)	
	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B
I	N/A	N/A	4257	5473	N/A	N/A	3824	5142	N/A	N/A	97.4	116.1
II	3604	4082	3760	4236	2621	3146	N/A	N/A	99.8	89.2	103.6	87.9
III	3150	3829	3580	3719	2322	2893	N/A	N/A	86.8	87.0	93.0	89.7
IV	2503	4321	3217	4345	2085	3204	N/A	N/A	109.2	93.7	106.0	91.3
V	5006	2819	3240	3374	3787	2123	N/A	N/A	98.7	101.7	126.2	109.4
VI	3878	3914	3964	2228	2860	2958	N/A	N/A	96.6	98.6	94.3	95.3
VII	3343	3330	3161	3110	2593	2580	N/A	N/A	88.6	92.4	89.3	61.5
VIII	3626	3012	3445	2877	2702	2249	2825	2390	105.3	96.8	112.8	91.8
Average	3622	3661	3808	4265	2710	2787	3049	3826	97.1	92.5	97.8	88.7

Note : CMC average VSS for Periods I and VIII only.

Table 2.14 : *Averaged filtered concentrations for steady state periods; averages are for "site data" (see also Figs 2.5 to 2.17 below).*

Parameter (mg/l)	Module	Long Term Period No.						
		II	III	IV	V	VI	VII	VIII
Influent NO_3	A & B	0.22	0.17	0.05	0.15	0.51	0.08	0.24
Influent NO_2	A & B	0.04	0.11	0.03	0.22	0.17	0.14	0.13
Influent Total P	A & B	10.75	11.09	11.09	10.66	10.64	10.26	9.86
Anaerobic NO_3	A	1.40	0.21	0.11	0.21	0.51	0.48	0.18
	B	0.39	0.18	0.13	0.10	0.27	0.14	0.26
Anaerobic NO_2	A	0.08	0.14	0.16	0.13	0.15	0.53	0.19
	B	0.06	0.06	0.08	0.10	0.11	0.20	0.16
Anaerobic Total P	A	18.60	29.99	13.03	34.10	25.69	23.92	24.48
	B	22.22	31.49	27.85	28.25	31.33	20.39	31.02
Anoxic NO_3	A	7.22	3.39	0.07	0.13	6.94	6.73	8.32
	B	8.01	5.17	5.01	3.29	5.34	1.67	4.44
Anoxic NO_2	A	0.13	0.25	0.03	0.04	0.28	0.20	0.22
	B	0.20	0.21	0.07	0.46	0.42	0.18	0.24
Anoxic Total P	A	6.04	11.14	9.24	12.66	10.26	10.05	7.25
	B	9.38	13.33	15.05	10.21	13.05	13.03	16.19
Aerobic NO_3	A	11.75	10.00	2.00	1.59	11.73	10.09	13.51
	B	14.78	14.91	10.57	11.07	15.03	9.51	13.58
Aerobic NO_2	A	0.08	0.23	0.15	1.81	0.12	0.15	0.09
	B	0.05	0.04	0.06	0.37	0.09	1.01	0.58
Aerobic Total P	A	2.20	1.55	0.44	3.78	3.61	5.87	2.17
	B	4.45	1.26	7.56	1.74	3.43	2.06	5.06
Effluent NO_3	A	12.50	8.69	0.28	1.27	12.61	11.25	13.58
	B	16.12	16.09	12.42	11.05	15.00	8.78	14.77
Effluent NO_2	A	0.13	0.22	0.21	1.38	0.13	0.23	0.16
	B	0.05	0.08	0.09	0.36	0.12	0.97	0.70
Effluent Total P	A	1.71	0.45	7.18	1.50	3.87	4.90	2.32
	B	4.67	1.52	7.90	3.34	3.51	2.61	0.99

Note : Influent, anaerobic, anoxic, aerobic & effluent Total P = filtered Total P concentration.

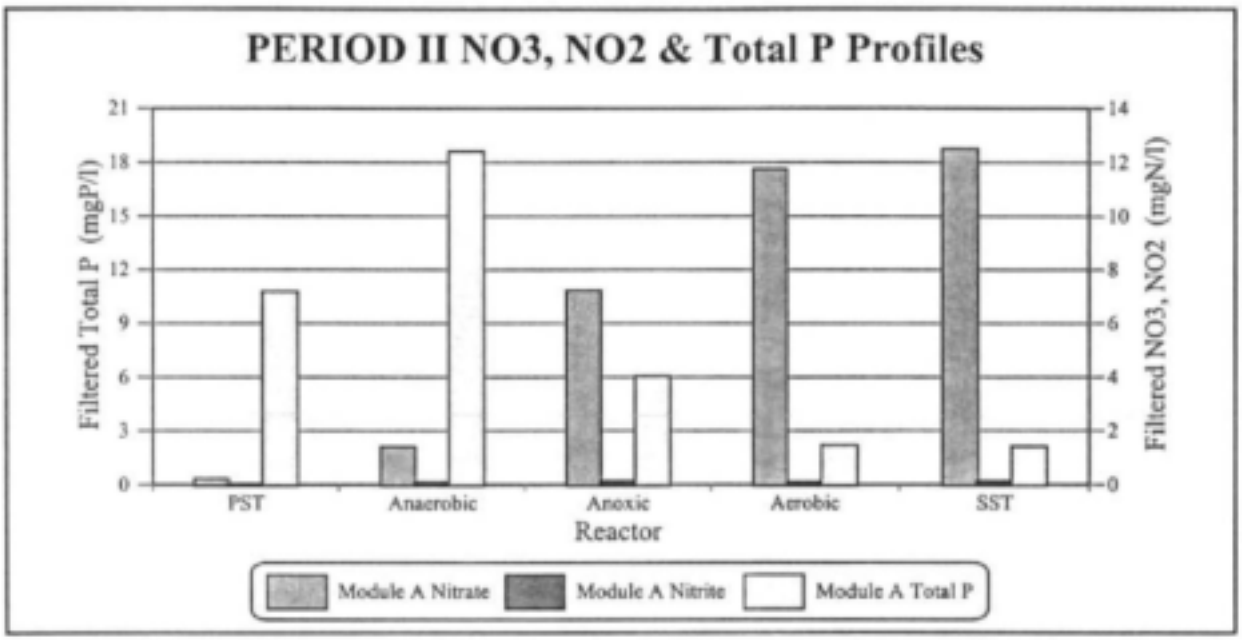

Fig 2.4 : *Long Term Period No. II nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module A.*

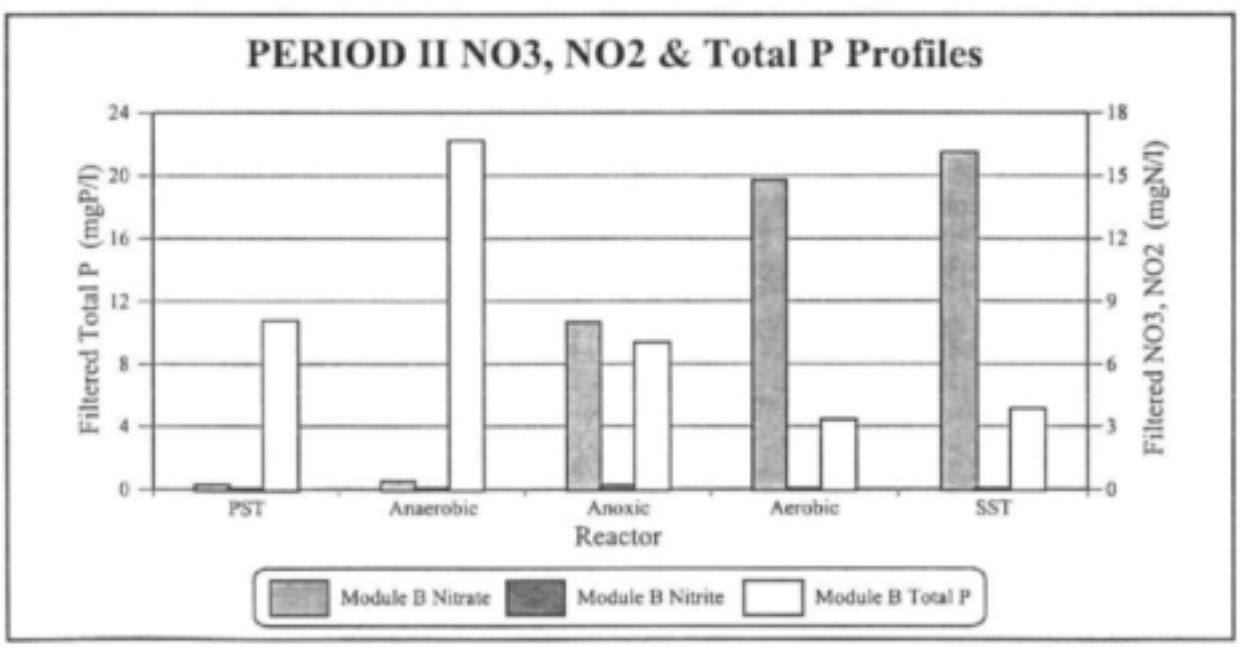

Fig 2.5: *Long Term Period No. II nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module B.*

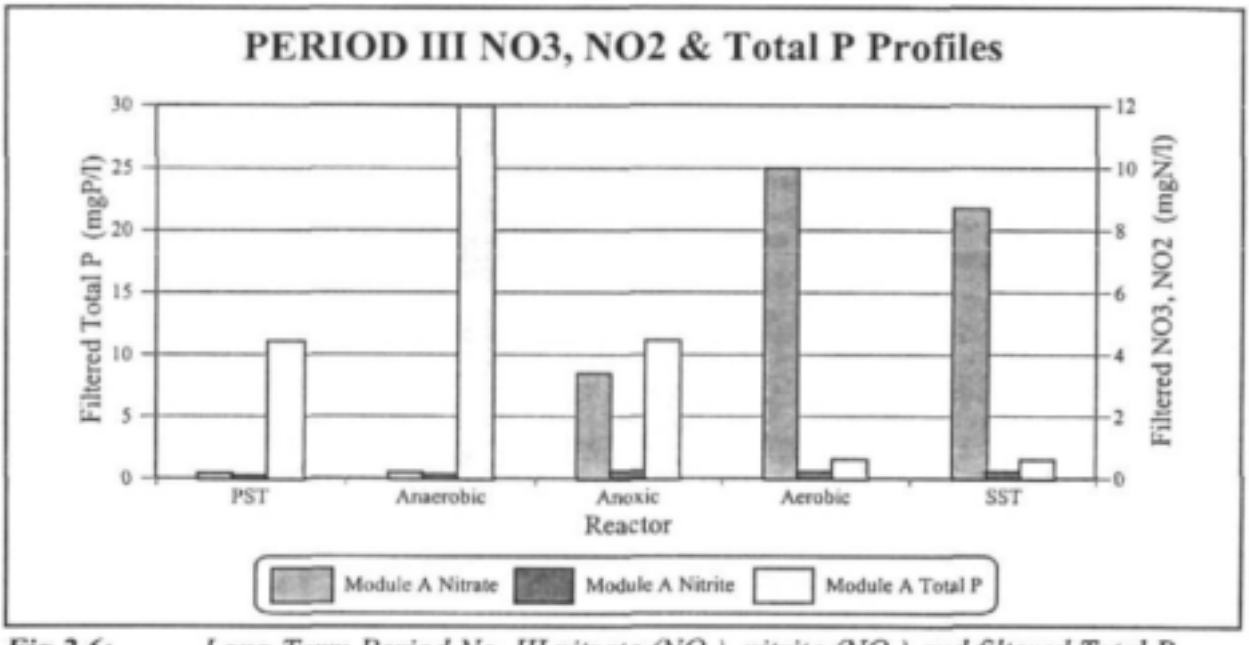

Fig 2.6: *Long Term Period No. III nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module A.*

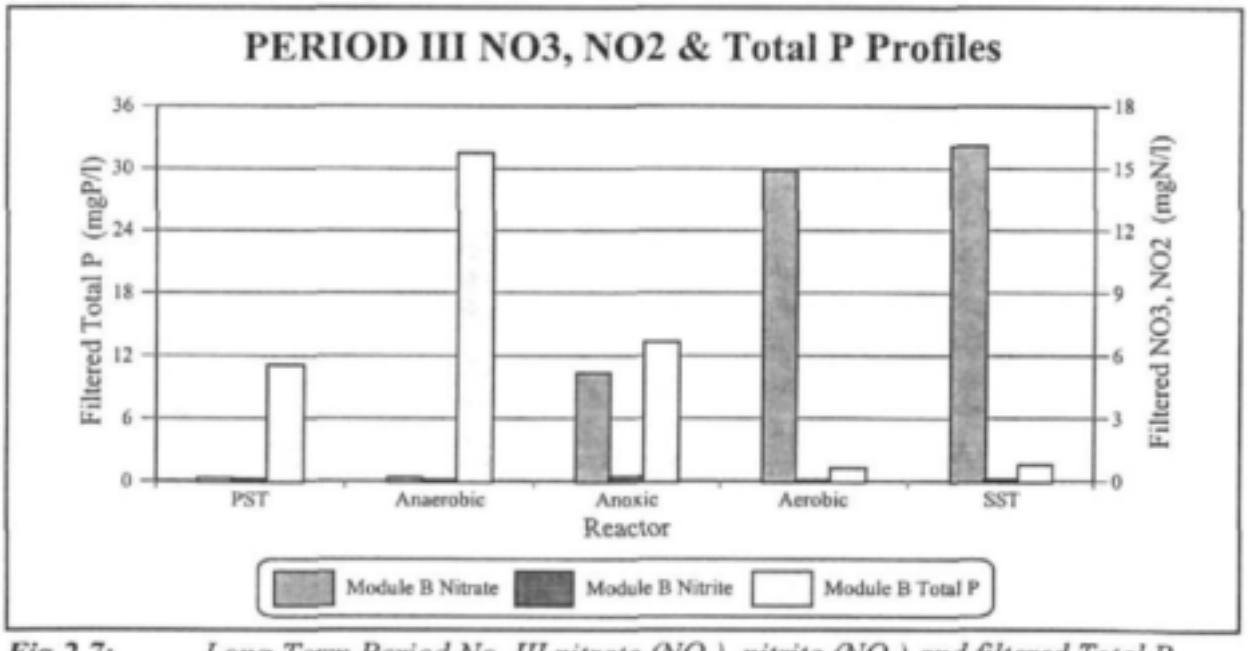

Fig 2.7: *Long Term Period No. III nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module B.*

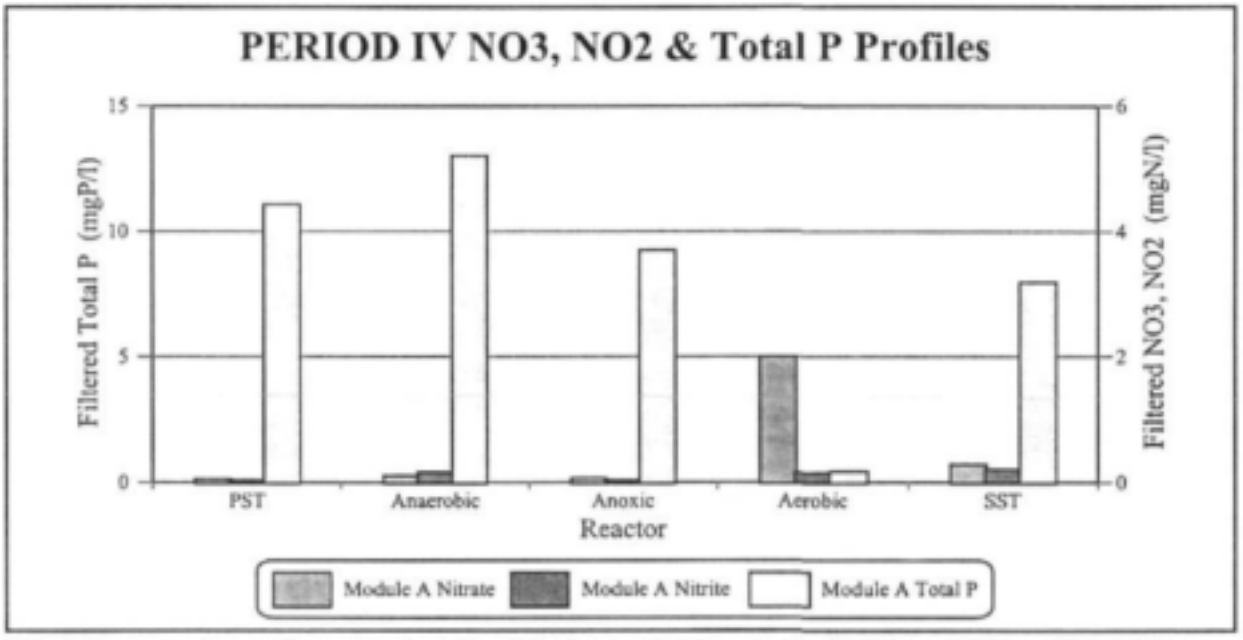

Fig 2.8: *Long Term Period No. IV nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module A.*

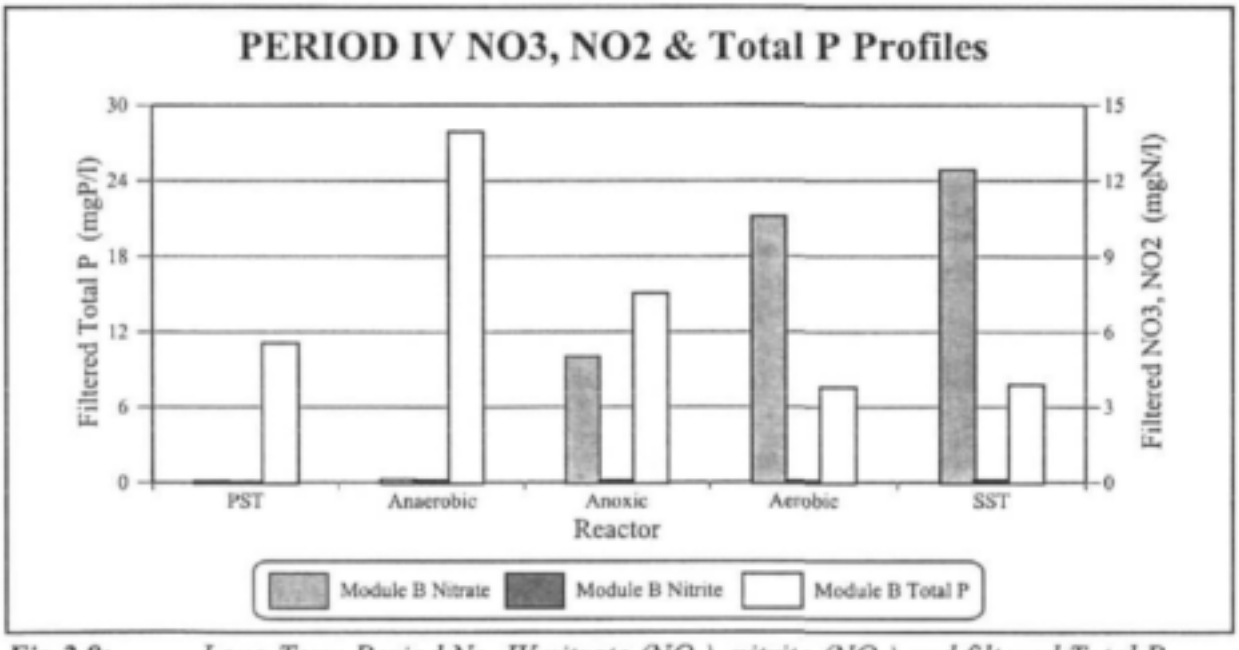

Fig 2.9: *Long Term Period No. IV nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module B.*

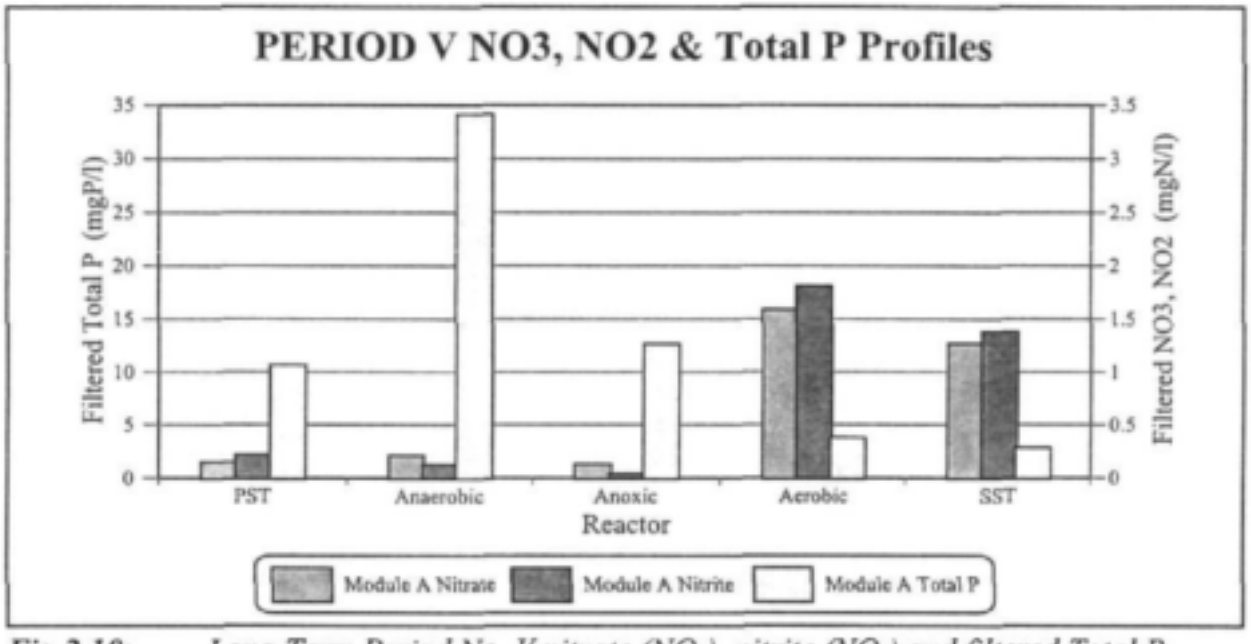

Fig 2.10: *Long Term Period No. V nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module A.*

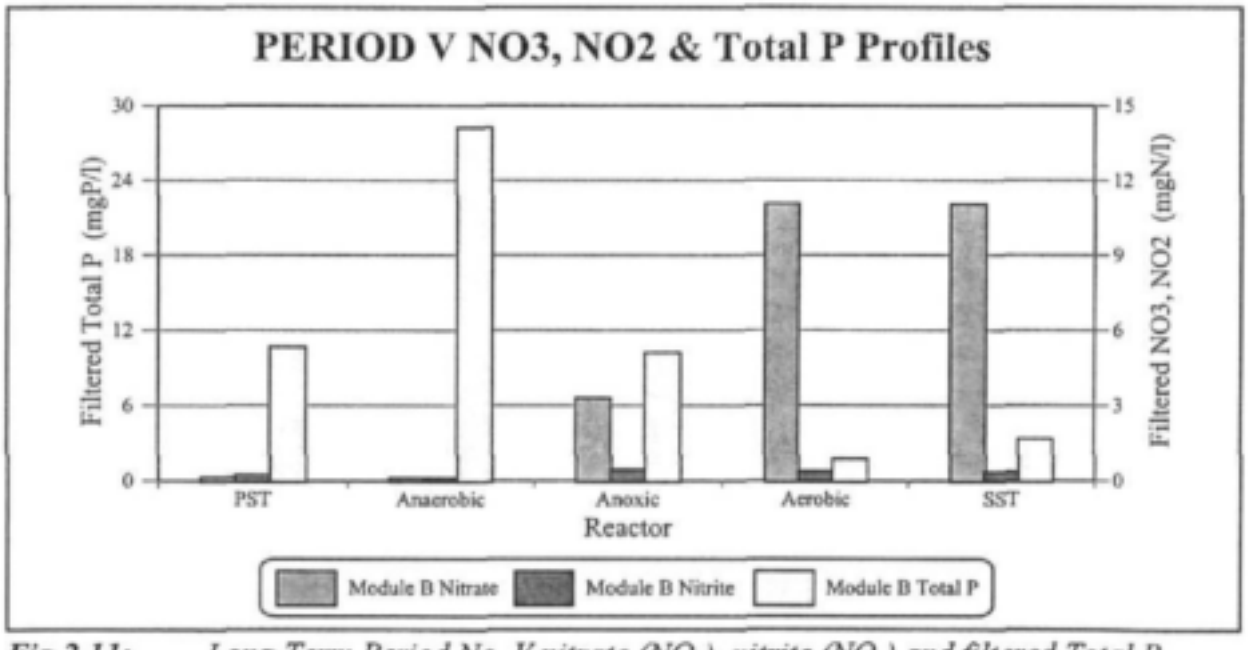

Fig 2.11: *Long Term Period No. V nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module B.*

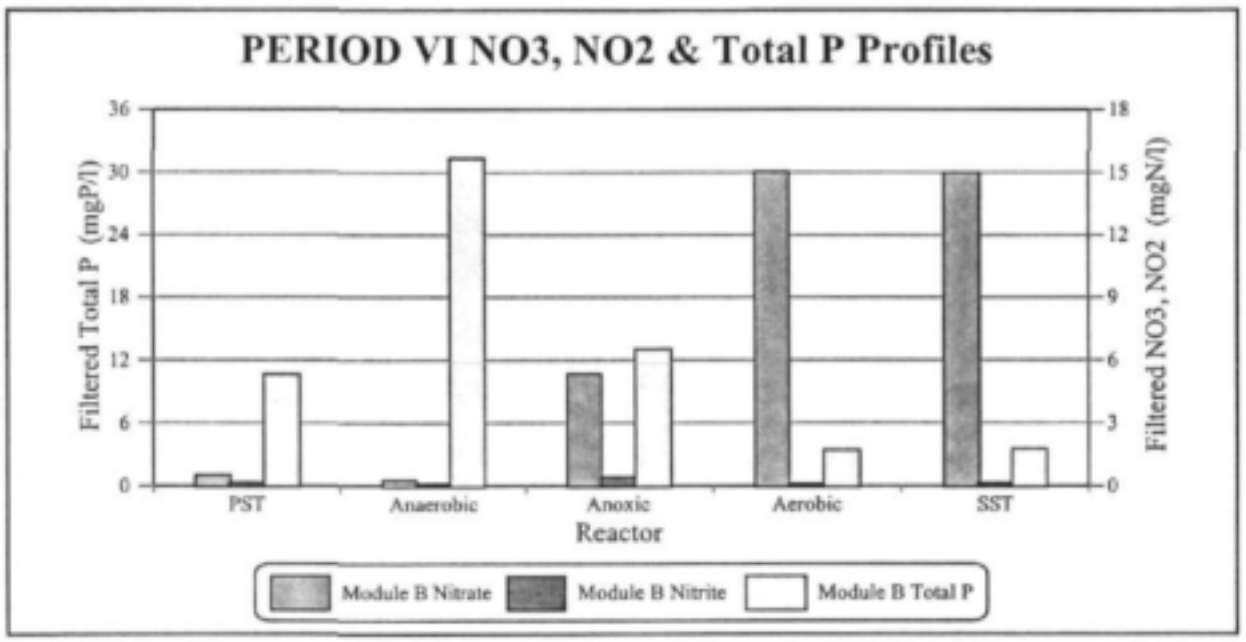

Fig 2.12: *Long Term Period No. VI nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module A.*

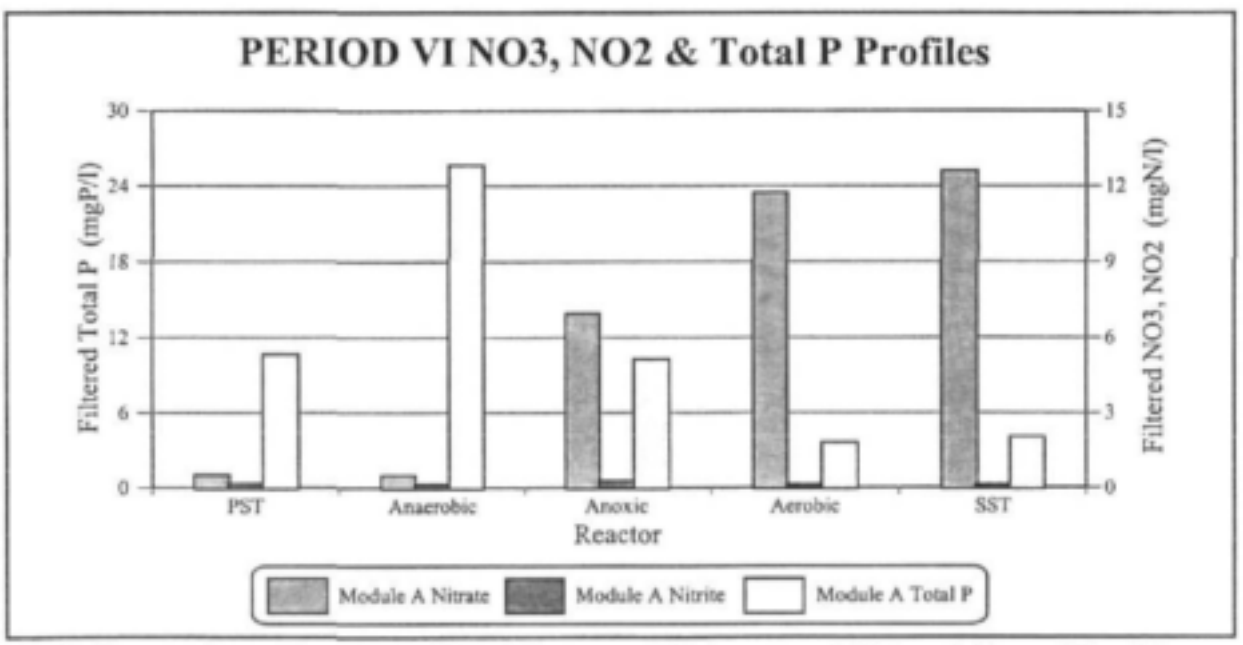

Fig 2.13: *Long Term Period No. VI nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module B.*

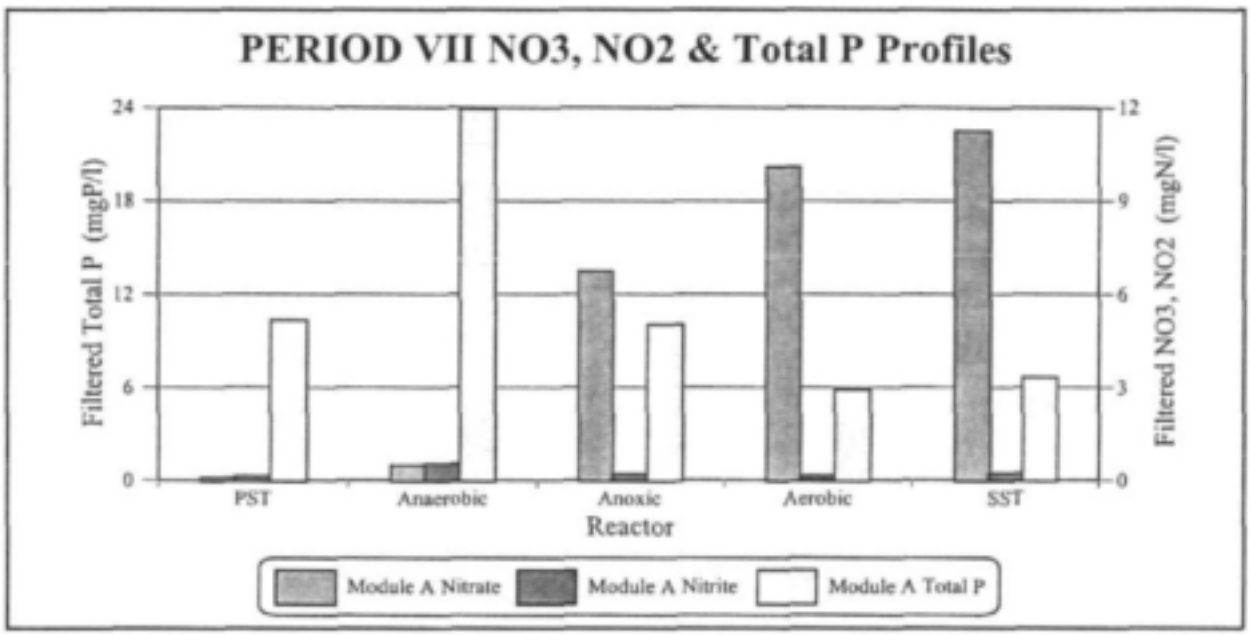

Fig 2.14: *Long Term Period No. VII nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module A.*

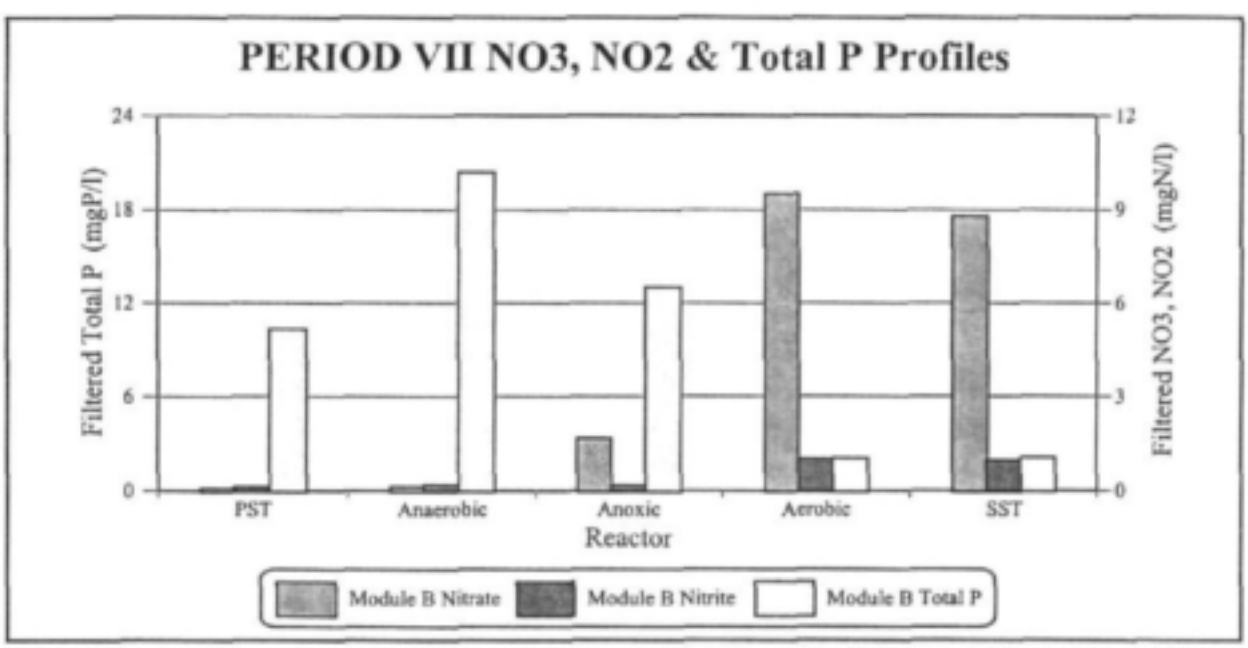

Fig 2.15: *Long Term Period No. VII nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module B.*

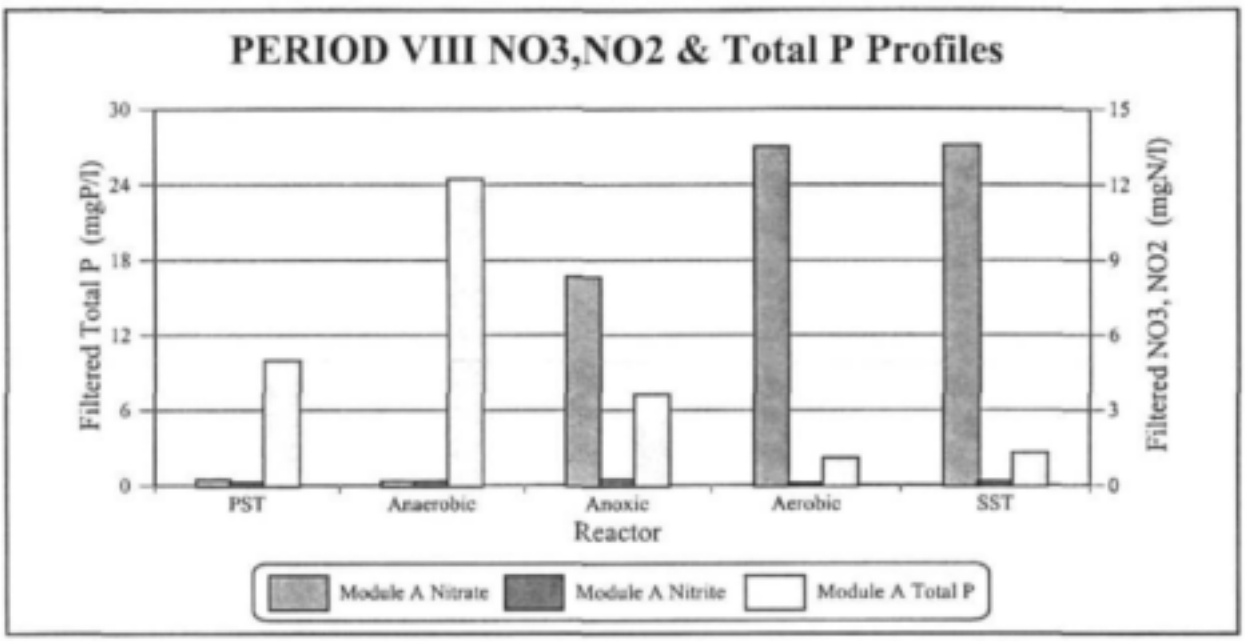

Fig 2.16: *Long Term Period No. VIII nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module A.*

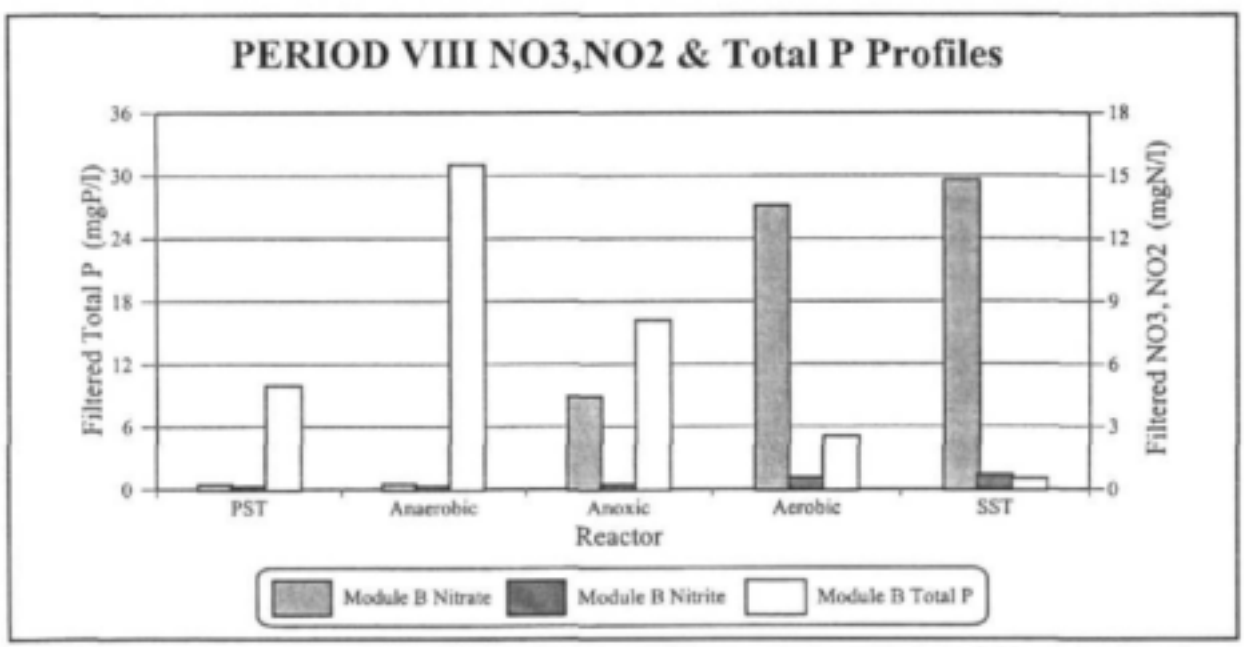

Fig 2.17: *Long Term Period No. VIII nitrate (NO_3), nitrite (NO_2) and filtered Total P concentrations as measured in the various reactors and SST of Module B.*

2.13.3 Effluent Quality

The averaged effluent concentrations for the steady state periods are listed in Table 2.15 below. For the effluent data the averages are weighted averages taking into account the number of days of operation for a particular period and excludes all obvious outliers within the data sets.

Table 2.15 : *Averaged effluent data for steady state periods; averages from "site data" and "CMC" data.*

Long Term Period No.	Site COD (mgCOD/ℓ)		CMC COD (mgCOD/ℓ)		Floc/filt COD (mgCOD/ℓ)		Site TKN (mgN/ℓ)		Floc/filt TKN (mgN/ℓ)		Site Total P (mgP/ℓ)	
	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B
I	N/A	N/A	43.6	39.0	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
II	37.3	40.4	39.8	37.4	32.1	33.4	3.3	2.1	2.5	1.5	1.8	5.1
III	46.3	54.0	46.5	35.5	42.2	47.0	13.8	4.8	11.3	3.3	0.5	1.9
IV	260.7	54.7	238.0	53.3	120.9	48.7	90.1	2.3	74.7	1.6	8.5	8.4
V	74.6	70.4	48.2	50.8	64.6	58.3	30.6	8.0	27.5	7.4	1.7	3.8
VI	51.1	60.4	36.0	37.6	47.4	57.0	7.0	5.2	4.3	3.9	4.1	3.7
VII	64.6	102.1	71.2	48.3	59.1	79.5	7.8	38.0	7.6	35.3	5.2	3.4
VIII	70.6	112.5	53.7	78.6	65.6	83.5	14.1	21.1	11.9	13.4	2.5	2.5
Ave.	66.4	66.4	54.0	43.2	52.5	54.5	15.8	11.2	13.3	9.4	2.9	4.0

Long Term Period No.	Site FSA (mgN/ℓ)		CMC FSA (mgN/ℓ)		Site NO_x (mgN/ℓ)		CMC NO_x (mgN/ℓ)		Site Ortho P (mgP/ℓ)		CMC Ortho P (mgP/ℓ)	
	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B	Mod A	Mod B
I	N/A	N/A	0.3	0.6	N/A	N/A	13.0	14.6	N/A	N/A	6.1	6.3
II	1.9	0.7	1.1	1.1	12.5	15.8	13.4	15.4	1.2	3.8	1.3	3.5
III	10.4	2.4	15.8	5.4	10.0	16.2	9.9	11.1	0.4	1.3	0.5	0.9
IV	61.8	1.0	54.9	1.2	0.5	13.6	0.2	14.1	6.9	7.5	7.6	7.1
V	24.2	5.2	23.1	10.2	1.6	13.2	3.6	10.8	1.5	3.3	0.8	0.3
VI	5.2	1.2	2.7	1.9	14.1	17.0	12.3	16.0	3.9	3.4	5.4	4.5
VII	3.1	28.0	1.9	24.7	15.3	7.3	15.8	7.9	4.9	2.0	4.1	3.7
VIII	5.0	5.7	4.2	5.4	12.5	17.2	13.5	16.4	2.0	0.8	3.3	1.5
Ave.	10.5	6.4	6.3	4.6	10.7	14.3	11.9	13.7	2.4	3.0	3.9	4.2

Note : Effluent COD, TKN & Total P = unfiltered. FSA = free and saline ammonia; NO_x = nitrate + nitrite. The floc/filt COD and TKN values are "site" values. "CMC" averages include Period I, whereas "site" averages do not (N/A).

Detailed evaluation of the results ais given by Hercules *et al.* (2002). The conclusions are summarized in Section 2.16 below. However, immediately apparent is that:

- With the exception of Module A Period IV, effluent CODs were consistently low.
- Nitrification failed for Module A Period IV, and partially failed for Module A Period V and Module B Period VII.
- With the exception of Module A Period III, effluent ortho P > 0.5 mgP/ℓ, indicating P removal was not limited by the influent P concentration. However, effluent P concentrations are higher than those predicted with UCTPHO, indicating lower biological P removal than predicted probably due to nitrate recycle to the anaerobic reactors from the anoxic reactors.
- Even with the lower implemented 'a-recycle' ratios, effluent nitrate + nitrite (NO_X) concentrations are lower than predicted with UCTPHO, indicating possibly better denitrification than predicted or error in nitrate measurement and/or simultaneous nitrification-denitrification.
- The sum of the "site" effluent nitrate (NO_3) and "site" effluent nitrite (NO_2) in Table 2.14 is reasonably close to the "site" effluent nitrate + nitrite (NO_X) given in Table 2.15. The NO_X concentrations (Table 2.15) were measured independently as a cross-check on the sum of the separately measured NO_3 and NO_2 concentrations (Table 2.14).

From the influent, reactor and effluent data a reasonable correlation is achieved between the "CMC" and "site" tested data. Similar correlations were achieved for the other parameters duplicated in the two sets of results. This indicated that the sampling and analytical procedures were reasonable. Because the "CMC" data are once weekly, and the "site" tested data three times weekly, the "site" tested data are accepted to be more reliable. Thus, the "site" tested data are evaluated in greater depth and presented in Section 2.16 below. However, where possible, the "CMC" data will be used as a cross check on the "site" data.

2.14 Laboratory-Scale UCT System

From February 2001 (day 470) a parallel laboratory-scale UCT system with the same design and operating parameters as the pilot plants was set up in the Water Research Laboratory at the University of Cape Town. The principal aim of this laboratory system was to investigate the cause(s) of the poor N mass balances achieved with the pilot plants and to characterise the settled wastewater feed, in particular measure the unbiodegradable particulate COD concentration which affects the VSS concentration in the pilot plants.

2.14.1 Experimental Set-up and Control

The laboratory system was set-up and operated in the same way as the Module A pilot plant *i.e.* to limit the anoxic nitrate/nitrite concentrations to < 1 mgN/ℓ in order to achieve a good settling sludge. Hence, the 'a-recycle' ratio was set to 0:1 with respect to influent flow to ensure a nitrate load on the anoxic reactor below its denitrification potential thus controlling the nitrate concentration in the anoxic reactor immediately prior to the aerobic reactor to < 1 mgN/ℓ.

A schematic layout of the laboratory-scale UCT system is given in Figure 2.18 and the design and operating parameters given in Table 2.16. The system consisted of three completely mixed reactors (anaerobic, anoxic and aerobic) with a total physical volume of 20.4 litres (ℓ) and an inclined tubular secondary settling tank of 1.5 ℓ in series. These reactor dimensions were in

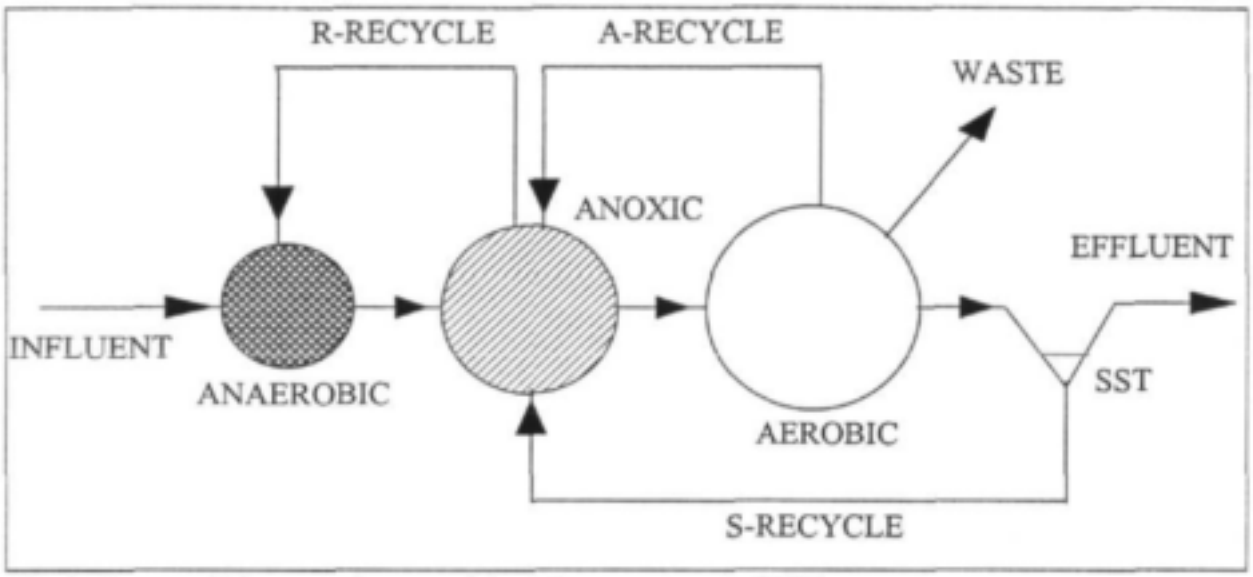

Fig 2.18 : *Schematic layout of the laboratory-scale UCT system.*

proportion to the that of the Module A pilot plant. The system had a 's-recycle' (RAS) ratio and 'r-recycle' ratio, both at 1:1 with respect to influent flow.

The influent sewage fed to the laboratory-scale UCT system was settled sewage collected after the Stage 1 PST at the Mitchell's Plain WWTP. Thus the laboratory-scale system received the same wastewater as influent as the pilot plants. During this investigation, which lasted 163 days from day 470 to day 632 (the pilot plants were decommissioned on day 589), the system received four batches of sewage. These batches of settled sewage collected approximately every four to six weeks, were first macerated and then stored in stainless steel tanks in the laboratory cold room at 4 °C. The system received a constant influent flow of 24 ℓ/d and the sludge age was controlled hydraulically at 15 days by wasting 1.25 ℓ/d from the aerobic reactor, taking due account of the volume of the samples taken for analysis. The system was operated at a constant laboratory controlled ambient temperature of 20 °C.

2.14.2 System Performance Monitoring

System performance monitoring commenced on 12 February 2001 (day 470). To monitor the performance and operation, samples were taken virtually daily from the reactors and SST for analysis. A sampling and analysis schedule is given in Table 2.17; these analyses will be termed "UCT" data. All samples taken from the influent, reactor and effluent were grab samples. All mixed liquor grab samples from the various reactors were filtered immediately to prevent further biological activity. Tables and graphs of the daily "UCT" data are given by Hercules *et al.* (2002).

2.14.3 Experimental Results

For each sewage batch, the appropriate data were averaged and are presented below. A comparison with the Module A pilot plant data is given in Section 2.16 below.

Influent data

The influent data averaged for each batch are given in Table 2.18 below. From Table 2.18 :

- The influent COD concentrations were low; overall average 522 mgCOD/ℓ compared with 621 mgCOD/ℓ fed to the pilot plants over roughly the same period.
- The influent TKN concentrations were consistent for each batch at around 85 mgN/ℓ compared to 102 mgN/ℓ fed to the pilot plants, giving an average influent TKN/COD ratio of 0.164 mgN/mgCOD for the settled sewage, the same as for the pilot plants.
- The influent Total P concentrations were consistent for each batch at around 13 mgP/ℓ compared to 12 mgP/ℓ fed to the pilot plants, giving an average influent TP/COD ratio of 0.026 mgP/mgCOD for the settled sewage, compared to 0.019 for the pilot plants.
- The average influent FSA concentration of 65 mgN/ℓ compared to 69 mgN/ℓ fed to the pilot plants over the same period, gave an FSA/TKN ratio of 0.76 compared to 0.68 for the pilot plants.

Table 2.16 : *Laboratory-scale UCT system design and operating parameters.*

System Parameter	Sewage Batch 1	Sewage Batch 2	Sewage Batch 3	Sewage Batch 4
Period (2001)	12/02-19/03	20/03-19/04	20/04-04/06	05/06-24/07
Day numbers	470-497	498-528	529-574	575-632
Number of days (d)	28	31	45	49
Number of sampling days (d)	26	17	28	37
Total system volume (ℓ)	20.4	20.4	20.4	20.4
Total effective volume (ℓ)*	18.7	18.7	18.7	18.7
Anaerobic volume (ℓ)	3.4	3.4	3.4	3.4
Anaerobic effective volume (ℓ)*	1.7	1.7	1.7	1.7
Anoxic volume (ℓ)	6.5	6.5	6.5	6.5
Aerobic volume (ℓ)	10.5	10.5	10.5	10.5
Anaerobic Mass Fraction	0.091	0.091	0.091	0.091
Anoxic Mass Fraction	0.350	0.350	0.350	0.350
Aerobic Mass Fraction	0.559	0.559	0.559	0.559
Sludge age (d)	15	15	15	15
Temperature (°C)	20	20	20	20
pH of mixed liquor	7.0-7.5	7.0-7.5	7.0-7.5	7.0-7.5
Aerobic reactor DO (mgO/ℓ)	2.0-5.0	2.0-5.0	2.0-5.0	2.0-5.0
Influent TKN/COD ratio	0.12-0.21	0.12-0.22	0.12-0.23	0.14-0.24
Influent TP/COD ratio	0.02-0.03	0.02-0.04	0.02-0.04	0.02-0.06
Influent (ℓ/d)	24.0	24.0	24.0	24.0
Waste (ℓ/d)	1.25	1.25	1.25	1.25
'S-recycle' ratio	1:1	1:1	1:1	1:1
'A-recycle' ratio	0:1	0:1	0:1	0:1
'R-recycle' ratio	1:1	1:1	1:1	1:1

* Volume adjusted to account for the dilution of the anaerobic reactor mixed liquor in the UCT NDBEPR system.

Table 2.15 : *Daily sampling procedure and sample analysis; "UCT" samples.*

SAMPLE	MEASUREMENT PARAMETER								
	COD[1]	TKN[2]	FSA[3]	NO_x[4]	TP[5]	OUR[6]	DSVI[7]	VSS[8]	TSS[9]
Influent	◆	◆	❖		◆				
Anaerobic				❖	❖				
Anoxic				❖	❖				
Aerobic	◆	◆		❖	❖	✓	✓	✓	✓
Effluent	◆❖	◆❖	❖	❖	❖				

✓ Measurement taken.
◆ Unfiltered sample.
❖ Filtered through Whatman® GF/C glass microfibre filters.
[1,2,3] Method according to Standard Methods (1985).
[4] According to Technicon Auto Analyser Industrial Method No 33.69W.
[5] Sulphuric acid/persulphate digestion at 100 °C followed by molybdate-vanadate colour.
[6] With Yellow Springs DO probe and Randall *et al.* (1991) OUR box. The daily OUR is the mean of ~150 OUR readings over a 24h period.
[7] According to Lee *et al.* (1983) or Ekama and Marais (1984).
[8,9] By separation with centrifugation, drying in a crucible at 105 °C and incineration at 600 °C .

Table 2.18 : *Averaged influent data for sewage batches; averages are for "UCT" data (daily).*

Sewage Batch No.	Influent COD (mgCOD/ℓ)	Influent TKN (mgN/ℓ)	Influent FSA (mgN/ℓ)	Influent Total P (mgP/ℓ)
1	523.3	82.1	61.4	14.2
2	540.9	86.5	65.7	14.3
3	513.3	82.5	63.9	12.3
4	520.3	89.3	68.3	14.0
Average	522.3	85.5	65.3	13.7

Reactor mixed liquor data

The averaged reactor concentrations for each sewage batch are given in Table 2.19 below.

Table 2.19 : *Averaged reactor data for sewage batches; averages are for "UCT" data.*

Sewage Batch No.	VSS (mg/ℓ)	TSS (mg/ℓ)	Anaer NO_x (mgN/ℓ)	Anox NO_x (mgN/ℓ)	Aero NO_x (mgN/ℓ)	Anaer Total P (mgP/ℓ)	Anox Total P (mgP/ℓ)	Aero Total P (mgP/ℓ)	OUR (mgO/ℓ/h)	DSVI (ml/g)
1	1930	2312	0.1	0.6	25.2	23.5	17.1	6.7	39.4	168
2	2095	2411	0.6	3.3	30.4	20.7	14.1	9.4	42.7	266
3	1852	2109	0.2	2.1	39.5	14.7	10.6	7.6	29.5	346
4	1592	1841	0.2	2.7	37.3	15.1	12.0	9.1	36.0	181
Ave.	1817	2111	0.2	2.1	34.0	17.9	13.2	8.2	37.5	234

Note : Anaerobic, anoxic & aerobic Total P = filtered Total P concentration. NO_x = nitrate + nitrite.

Immediately apparent from Table 2.19 :

- The VSS and TSS concentrations were fairly consistent throughout the investigation except for Sewage Batch 4 in which it was considerably lower.
- The nitrate + nitrite (NO_x) concentrations in the anaerobic zone were consistently low at < 1 mgN/ℓ.
- With the exception of Sewage Batch 1, NO_x concentrations in the anoxic zone were > 1 mgN/ℓ.
- The NO_x concentrations in the aerobic zone were consistently high (25 to 40 mgN/ℓ).
- The DSVIs were consistently high (> 150 mℓ/g) throughout the investigation.
- The OUR was around 37 mgO/ℓ/h except for Sewage Batch 3, for which it was considerably lower.

Effluent quality

The averaged effluent concentrations for each sewage batch is given in Table 2.20 below. These data are compared with the pilot plant results in Section 2.16. However, it can be seen that :

- The effluent COD concentrations were consistently low (~55 mg/ℓ).
- With the exception of Sewage Batch 4, the effluent FSA concentrations are < 2 mgN/ℓ.
- The effluent NO_x concentrations were consistently high (~40 mgN/ℓ).
- The effluent soluble P concentrations are high throughout (> 5 mgP/ℓ).

Table 2.20 : *Averaged effluent data for sewage batches; averages are for "UCT" data.*

Sewage Batch No.	Effluent COD (mgCOD/ℓ)	Effluent TKN (mgN/ℓ)	Effluent FSA (mgN/ℓ)	Effluent NO_x (mgN/ℓ)	Effluent Soluble P (mgP/ℓ)
1	51.6	2.4	0.8	33.2	5.9
2	49.7	1.8	0.4	36.3	8.8
3	61.1	4.4	1.7	44.9	7.1
4	58.8	5.1	2.5	37.0	8.7
Average	56.4	3.7	1.6	38.1	7.6

2.15 Filament Identifications

Throughout the 589 day investigation filament identification were done once monthly on the pilot plants and the UCT laboratory-scale unit using the microscopic techniques of Eikelboom and van Buijsen (1981) and Jenkins *et al.* (1984). Samples for identification were collected at the end of the aerobic reactor and examined on the same day. Detailed identification results are listed by Hercules *et al.* (2002). The AA (low F/M) filaments *M. parvicella*, types 0092 and 1851 were dominant in both modules as well as the laboratory-scale system throughout the investigation.

2.16 CONCLUSIONS

2.16.1 OPERATION PROBLEMS

1. Progressive blocking of the ceramic diffuser domes increased the back pressure in the aeration system which caused thermal trip-out of the compressor. The compressor pulley wheel diameters were reduced, which solved the trip-out problem but reduced the air supply by about 10 %. On 11 June 2001 (day 589) the pipe distribution system irreparably ruptured and forced cessation of the investigation.

2. Even though the influent flow was reduced to match the aeration capacity, the air supply was too low for the aerobic reactors to be properly aerobic resulting in significant simultaneous denitrification. Also, leaks in the distribution pipework in the anoxic reactors (part of which were aerobic reactors in the former configuration) as well as considerable back mixing from the aerobic to anoxic reactor in Module A, resulted in poor denitrification in the anoxic reactors. As a consequence the anoxic and aerobic reactors did not operate as such and were both quasi anoxic quasi aerobic to different degrees in the two modules.

3. Sporadic ingress of main treatment plant waste sludge via the sludge wastage drain resulted in significantly higher (30 to 50 %) sludge concentrations in the modules than would develop from the settled wastewater itself. As a consequence "steady state conditions" could not be achieved. Due to its labour intensiveness, it was planned to measure the oxygen utilisation rates only when this problem was eliminated but the investigation had to be terminated before this could be achieved. As a consequence, COD balances could not be done.

4. An identical laboratory scale system (~1:200000) was operated for 163 days from day 470 to day 632 (43 days after the termination of the pilot plant investigation) to evaluate the magnitude of the above problems on the pilot plant results.

2.16.2 COD REMOVAL PERFORMANCE

5. Over the 589 day investigation which was divided into eight periods ranging from 40 to 224 days during which influent and operating conditions were relatively unchanged, the average (for the eight steady state periods) percentage COD removal was 87 and 89 % for Modules A and B respectively. With the exceptions of Period IV Module A and Periods VII and VIII Module B, consistently good COD removals were achieved with unfiltered effluent COD concentrations < 75 mgCOD/ℓ. The membrane filtered effluent COD concentration was 53 mg/ℓ giving an unbiodegradable soluble COD fraction of 0.086 for the settled wastewater.

6. The COD mass balance of the two modules over the investigation could not be determined because steady state conditions were not achieved and therefore the oxygen utilisation rates (OUR) were not measured.

7. The overall average TSS and VSS concentrations were 3622 mgTSS/ℓ and 2710 mgVSS/ℓ

and 3661 mgTSS/ℓ and 2787 mgVSS/ℓ for Modules A & B respectively. This is ~10 % higher than expected from 100 % COD mass balance models (*UCTPHO* - see 24 below) and ~50 % higher than measured in the lab-scale system, indicating that sludge ingress was substantial.

2.16.4 NITROGEN REMOVAL PERFORMANCE

8. The influent TKN/COD ratio varied between 0.10 to 0.24 with an average of 0.16 mgN/mgCOD which is above the estimated upper limit (0.14 mgN/mgCOD) to avoid nitrate recycle to the anaerobic reactor.

9. For Periods I and II (332 days), the nitrification performance was reasonable, with effluent FSA concentrations at about 1 mgN/ℓ. For all the other periods nitrification was partial only, indicated by the relatively high effluent TKN and FSA concentrations (5 - 30 mgN/ℓ). From Period III to midway Period VI (140 days), nitrification was particularly poor in Module A (effluent FSA > 5 mgN/ℓ), which indicated inadequate aeration. From Period VI, more air was directed to Module A at the expense of Module B. This reduced the effluent FSA from Module A to < 5 mgN/ℓ but increased it from Module B to > 20 mgN/ℓ.

10. The nitrite (NO_2) concentrations were negligible (<0.5 mgN/ℓ) in the anoxic and aerobic reactors throughout the investigation. For most periods, the effluent nitrate concentrations were low (10 to 17 mgN/ℓ) in Module A and B compared with the very high influent TKN/COD ratio and indicates that denitrification in the system was very good (~50 - 55 mgN/ℓ), except for Periods IV and V Module A where nitrification failed (< 2 mgN/ℓ). The nitrate concentrations in the anoxic zones of both Modules A and B were higher than expected (5 - 8 mgN/ℓ) which resulted in (i) nitrate feed back to the anaerobic reactor and (ii) not achieving low nitrate at the anoxic-aerobic transition in Module A. Reducing the 'a-recycle' ratios on Module A and B from 2:1 and 5:1 to 0:1 and 3:1 respectively did not solve this problem. Later in the investigation it became apparent that this high anoxic reactor nitrate concentrations were due to back mixing from the aerobic reactor and air leaks in the air distribution network.

11. To examine the nitrification and denitrification performance, nitrate and nitrite mass balances were calculated around each reactor and the SST. Net denitrification in the anoxic zones was very low, on average only 2 and 23 mgN/ℓ influent for Modules A and B respectively. Compared with the influent TKN of 95 to 98 mgN/ℓ, with such low denitrification it is impossible to achieve effluent nitrate concentrations between 10 and 17 mgN/ℓ unless significant simultaneous nitrification-denitrification was taking place in the anoxic and aerobic reactors. This was confirmed during Periods VII and VIII when the air supply to Module A was increased (at the expense of Module B) and resulted in a net nitrate production (nitrification) in the anoxic reactor. This simultaneous nitrification-denitrification in the anoxic and aerobic reactors invalidates the N balance calculations which assumes that no denitrification takes place in the aerobic reactor and no nitrification in the anoxic reactor. The average N balances calculated for Modules A and B were 57 % and 76 % respectively indicating that Module B was closer to the desired operation conditions (*i.e.* sufficient air in the aerobic reactor) than Module A.

12. The overall average N removal was 77 % for Modules A and B. Accepting a 100 % N balance in the system for the eight steady state periods, the total N concentration removed (via denitrification + N in the waste sludge) was exceptionally large, on average at about 74 mgN/ℓ influent for both modules. Of this N, about 21 and 25 mgN/ℓ influent for Modules A and B respectively was removed with the waste sludge leaving about 53 and 49 mgN/ℓ influent for Modules A and B respectively as nitrate denitrified. From the nitrate mass balance around the anoxic and aerobic reactors, of this nitrate, only about 3 and 24 mgN/ℓ influent for Modules A and B respectively was denitrified in the anoxic reactor and 17 and 42 mgN/ℓ nitrified in the aerobic reactor, confirming simultaneous nitrification-denitrification in the anoxic and aerobic reactors. While such simultaneous N removal can have advantages, it comes at considerable risk to nitrification, indicated by the relatively incomplete nitrification (see 9 above).

13. On average over the 589 day investigation the N mass balances for both modules were poor - 57 and 76 % for Modules A and B respectively. Two possibilities for this were identified, (i) the simultaneous nitrification-denitrification already mentioned and (ii) error in nitrate measurements. While supporting evidence for the latter was identified - separate independent analysis of nitrate samples gave about 10 % higher concentrations - by far the greatest contribution to the poor N balances was simultaneous N removal in the anoxic and aerobic reactors. Furthermore, a parallel laboratory-scale UCT system with the same design parameters as the Module A pilot plant and fed the same wastewater as influent but with none of the operating problems gave a good N balance (99 %) and a much higher effluent nitrate concentration (38 mgN/ℓ).

2.16.4 Phosphorus Removal Performance

14. The average influent Total P (TP) concentration was 12.7 mgP/ℓ making the influent P/COD ratio 0.021 mgP/mgCOD. This is not a high value, and with the influent readily biodegradable (RB) COD of 217 mgCOD/ℓ (measured as the difference between floc/filtered influent and effluent COD concentrations) has the potential to achieve very good P removal with effluent TP < 1 mgP/ℓ.

15. With the exception of Module A Period III, the filtered (< 0.45 μm) effluent TP concentration > 2 mgP/ℓ, indicating that the P removal was not limited by the influent P concentration. This allowed the overall average system P removal capacity for Modules A and B to be measured, which based on unfiltered influent and filtered effluent TP was 10.0 and 9.3 respectively. The unfiltered effluent TP was 2.9 and 4.0 mgP/ℓ making the P/VSS ratio of the ESS 0.06 mgP/mgVSS (similar to that in the reactor).

16. The percentage P removal of Modules A and B were similar at 69 and 64 % respectively based on unfiltered influent and effluent samples. The P removal attained was considerably reduced below that potentially achievable due to the high concentration of nitrate recycled to the anaerobic reactors; on average at 5.6 and 5.0 mgN/ℓ for Modules A and B respectively. This reduced the P removal by an estimated 3 mgP/ℓ. Nevertheless, consistently good biological P removal was obtained at 0.015 and 0.014 mgP/mg influent COD for Modules A and B respectively.

17. In contrast to the N mass balance, the overall average P mass balance for Modules A and B were excellent, at 103 and 97 % respectively. The good overall averages indicate that the data relating to P was reliable. The P/VSS ratio of the sludge mass in the reactors was 0.065 mgP/mgVSS, which is significantly above the 0.025 mgP/mgVSS for non-BEPR activated sludge. However, it should be noted that had sludge from the non-BEPR main treatment plant not sporadically entered the pilot plants, the P/VSS ratio of the sludge mass would have been significantly higher than 0.065 mgPmgVSS.

18. The average anaerobic P release was very similar in both modules at 26 and 29 mgP/ℓ influent for Modules A and B respectively. The average aerobic P uptake in Module B is significantly higher than in Module A at 45 compared to 19 mgP/ℓ influent respectively. In the anoxic zones the two modules appeared to exhibit divergent behaviour. Throughout the investigation P uptake occurred in the anoxic reactor of Module A, while in Module B, P release occurred in the anoxic reactor. For Module A, over the investigation 48 % of the P uptake occurred in the anoxic reactor. Initially it was thought this was anoxic P uptake in Module A. Hu *et al.* (2001) noted that anoxic P uptake tends to be stimulated by an overload of nitrate on the anoxic reactor and has been observed quite often in laboratory-scale and full-scale plants. When it takes place, the BEPR is only two thirds to three quarters of BEPR with predominantly (>90 %) aerobic P uptake (Ekama and Wentzel, 1997). However, its occurrence in Module A is apparent not real because (i) both modules were overloaded with nitrate, (ii) had similar average anoxic nitrate concentrations and (iii) aerobic P uptake took place in Module B and (iv) the BEPR was the same in both modules. Although not apparent during the investigation, the occurrence of P uptake in the anoxic reactor therefore is best explained as aerobic P uptake as a result of excessive DO ingress into the Module A anoxic reactor. In Module B, DO ingress into the anoxic reactor was much less, and air supply to the aerobic reactor greater than Module A, with the result that the anoxic and aerobic reactors were closer to these conditions than the anoxic and aerobic reactors of Module A. Thus for Module A the circumstantial (and experimental) evidence suggested P uptake by PAOs under aerobic conditions rather than under anoxic conditions.

19. The BEPR performance of the pilot plants (Modules A and B) was assessed by comparing the measured P removal with the theoretical P removal calculated by the steady state model of Wentzel *et al.* (1990). In doing this calculation procedure, the unbiodegradable particulate COD fraction ($f_{S,up}$), the two active heterotrophic organism fractions of the VSS [*i.e.* polyphosphate accumulating organisms (PAOs), $f_{av,PAO}$ and ordinary heterotrophic organisms (OHOs), $f_{av,OHO}$] and the P content of the PAOs ($f_{XBG,P}$) are determined. To determine the unbiodegradable particulate COD fraction ($f_{S,up}$) of the sewage fed to the pilot plants, the appropriate $f_{S,up}$ value was selected so that the system VSS mass calculated with the BEPR model of Wentzel *et al.* (1990) was equal to the measured VSS mass using the measured influent readily biodegradable COD (RBCOD) concentration, and the influent characteristics of the sewage (*i.e.* fraction of unbiodegradable soluble COD/total influent COD, $f_{S,us}$ and total influent COD, S_{ti}) and the known system parameters (anaerobic mass fraction, sludge age and nitrate recycled) as input. An overall average $f_{S,up}$ value of 0.239 was estimated. The OHO and PAO active VSS mass fractions were determined from the ratio of the masses of OHO and PAO VSS to the total VSS giving overall average $f_{av,OHO}$ and $f_{av,PAO}$ values of 0.210 and

0.140 respectively. The P content of the PAOs ($f_{XBG,P}$) was estimated as that value which set the calculated P removal equal to the measured P removal. An overall average $f_{XBG,P}$ value of 0.322 mgP/mgPAOAVSS was estimated for the pilot plants. A $f_{S,up}$ value of 0.239 for the settled wastewater is far too high and is the result of the sludge ingress from the main treatment plant. Accordingly a more realistic $f_{S,up}$ of 0.05 was selected for the settled wastewater (the lab-scale system yielded a $f_{S,up}$ of 0.03) and the calculations repeated. The $f_{S,up}$ of 0.05 yielded a PAO P content ($f_{XBG,P}$) of 0.39 mgP/mgPAOVSS which is very close to the Wentzel *et al.* (1990) model standard of 0.38 for 100 % aerobic P uptake BEPR, confirming that the BEPR in the pilot plants was normal aerobic uptake BEPR.

2.16.5 Sludge Settleability and Filament Identification

20. In both modules the anoxic nitrite concentrations were very low throughout the investigation with an overall average of 0.17 and 0.25 mgN/ℓ for Modules A and B respectively. In both modules anoxic nitrate concentrations were high, except for Module A Periods IV and V where nitrification failed, with an overall average of 5.36 and 5.17 mgN/ℓ for Modules A and B respectively. The high nitrate concentrations in the anoxic zones indicated that the proposed control of nitrate in the Module A anoxic zone to low concentrations had not been achieved in practise, even though the 'a-recycle' was set to zero from Period III onwards. However, the proposed control of nitrate in the Module B anoxic zone to high concentrations had been achieved in practise. Thus, it would not be possible to demonstrate the effect of complete anoxic denitrification on sludge settleability, though the effect of incomplete anoxic denitrification could be demonstrated. However, because the intended anoxic and aerobic conditions were not achieved in the anoxic and aerobic reactors, and these reactors were a continuous quasi anoxic quasi aerobic reactor, it is difficult to apply the AA filament bulking hypothesis to the results. Nevertheless, in single reactor Carousel, Orbal and other intermittently aerated ND plants in which substantial simultaneous ND takes place, sludge settleability often is poor (DSVI > 200 mℓ/g) caused by AA filaments like *M. Parvicella* and Type 1851. So apart from the fact that the pilot plants include BEPR, from the quasi anoxic quasi aerobic conditions in the anoxic and aerobic reactors of the pilot plants, the expectation is that the sludge should bulk due to AA filament proliferation.

21. The sludge settleability of the two systems were monitored by means of the Diluted Sludge Volume Index (DSVI). In terms of the hypothesis and the bulking control strategy, since both modules have high anoxic nitrate concentrations both should produce a bulking sludge due to AA filament proliferation. In contrast both modules produced good settling sludges with low overall DSVIs of 97 and 102 mℓ/gTSS. At no time during the 589 day investigation did the 7 day moving average exceed 120 mℓ/gTSS.

22. From 17 monthly microscopic identifications the filament most frequently dominant in both Module A and B was *M. parvicella* (41 and 50 % respectively). The next most frequently dominant filaments were type 0092 and type 1851. All of these filaments are classified as typical of the low F/M category (Jenkins *et al.*, 1984) later renamed Anoxic-Aerobic(AA) (Casey *et al.*, 1994) and are almost always observed in full scale NDBEPR systems whether bulking or not (Blackbeard *et al.*, 1986, 1988). The five most frequently occurring filament types in Modules A and B in descending order of frequency were, *M.*

parvicella present in 82 and 94 % of samples, type 0092 present in 77 and 69 %, type 1851 in 53 and 63 %, type 1701 in 6 and 13 % and *N. limicola* in 6 and 6 % respectively.

2.16.6 Simulation of the Full-Scale Pilot Plants

23. The 24h intensive monitoring tests were conducted by Diale and Modipa (2000) for their BSc thesis projects on the two pilot plants during which all the influent, reactor and effluent concentrations were measured every 2 hours and the OUR every hour mid-way along the length of the aerobic reactor. These indicated that diurnal variation in all these concentrations was not significant and the grab samples taken for the "CMC" and "site" data sets could be accepted as daily average values. Owing to the operation problems mentioned above and difficulty obtaining representative OUR samples, poor COD and N balances were obtained in these tests.

24. By giving the same influent flow and concentrations and system design parameters as input, the system performance was modeled with two NDBEPR simulation programmes, *UCTPHO* (Wentzel *et al.*, 1992) and *BIOWIN* (Envirosim Associates, 2001). These two programmes were selected because of their distinct model differences (i) *UCTPHO* is COD conservative (*i.e.* 100 % COD balance) whereas *BIOWIN* includes COD losses in the anaerobic processes and (ii) the BEPR in *UCTPHO* is based on 100 % aerobic P uptake whereas *BIOWIN* includes anoxic-aerobic P uptake BEPR. The measured effluent and reactor concentrations of Modules A and B were compared to the predicted values of the *BIOWIN* and *UCTPHO* kinetic models. Because the modules could not be operated with well defined aerobic and anoxic conditions and without sporadic sludge ingress, the measured results cannot be used to comment on the model predictions.

25. The *BIOWIN* simulations with the default kinetic and stoichiometric constants produced results which were at an unacceptably large variance with steady state ND and BEPR models and *UCTPHO* predictions, particularly in regard to denitrification rates in the anoxic reactor. This was in part due anoxic P uptake BEPR and in part due to too high kinetic rate constants. Also the predicted VSS and OUR were significantly below (~10 %) those of *UCTPHO* due to anaerobic COD loss. Hence, adjusted *BIOWIN* model parameters (de Haas *et al.*, 2001) were adopted in the simulation of the pilot plants to closely emulate the *UCTPHO* model parameters *i.e.* (i) eliminated the COD loss, (ii) reduced the denitrification rate and (iii) stopped anoxic P uptake and its associated denitrification. It was found that in general for Modules A and B, the combination of results obtained from the *BIOWIN* simulations using the adjusted model parameters were closer to the measured data than the default *BIOWIN* simulation results. However, the pilot plant data cannot be accepted as a basis for model validation.

26. A reasonably good correlation between the measured and predicted effluent unbiodegradable soluble COD (S_{us}) and free and saline ammonia (FSA) concentrations was achieved. The predicted effluent and reactor nitrate (NO_3) concentrations were consistently higher than the measured values. This can be attributed to the significant denitrification in the aerobic reactors of both Modules A and B which is not modelled in *BIOWIN* and *UCTPHO*. As the effluent and reactor Ortho P concentrations are greatly affected by the predicted NO_3 recycled from the anoxic to the anaerobic reactor, a poor

correlation between the measured and predicted values was achieved.

27. For some long term periods, the measured VSS and TSS are significantly greater than the predicted values of *BIOWIN* and *UCTPHO*. This can be attributed to the inadequate sludge wasting from the pilot plants particularly in the earlier stages of the investigation.

2.16.7 CLOSURE

The principle aim in this research project was to investigate at full-scale the implementation of the proposed strategy to control bulking by AA filaments, namely that by limiting the anoxic nitrate/nitrite concentrations to < 1mgN/ℓ at the anoxic to aerobic transition by controlling the 'a-recycle' ratio, a good settling sludge could be achieved. In the investigation, the two pilot plant modules were run in parallel, one (Module A) with a low 'a-recycle' ratio to give low anoxic nitrate concentrations and the other (Module B) with high 'a-recycle' ratio to give high anoxic nitrate concentrations. In terms of the AA filament bulking hypothesis and the control strategy derived from it, this should cause the module with high anoxic nitrate concentrations to bulk due to AA filament proliferation (Module B) and the module with low anoxic nitrate concentrations (Module A) not to bulk.

In operation of the two pilot plant modules, it was found that despite the difference in 'a-recycle' ratio, both modules had high anoxic nitrate concentrations. This was caused by the poor anoxic denitrification performance in the two modules due to air ingress and back mixing from the aerobic reactor, particularly in Module A. Thus, the effect of low anoxic nitrate concentration on sludge settleability could not be examined. However, since both modules had high anoxic nitrate concentrations, the effect of this on sludge settleability could. In terms of the AA bulking hypothesis this should stimulate AA filament proliferation and hence bulking. However, this did not occur in practise: In both modules, throughout the investigation (589 days where monitoring took place) relatively low DSVIs (< 110mℓ/gTSS) were measured. Clearly, this is contrary to the AA bulking hypothesis. Furthermore, because of inadequate aeration in the aerobic reactors, and air and DO ingress into the anoxic reactors, these were quasi anoxic quasi aerobic reactors in which substantial simultaneous ND took place (~50 % of N removal). These conditions are similar to Orbal and Carousel ND systems which frequently experience high DSVI due to AA filament proliferation. Even so the pilot plants had very good settling sludge. Curiously, the lab-scale system had a poor settling sludge (DSVI > 200 mℓ/gTSS) throughout its 163 day operation. Clearly finding the cure for AA filament bulking in NDBEPR systems remains elusive despite the international research attention it has received over the past twenty years.

With regard to the second objective, that of establishing whether or not the unexplained COD loss observed in laboratory scale NDBEPR systems also takes place in large scale systems, could not be validated also. This was due to the sporadic sludge ingress into the pilot plants from the main ND activated sludge plant via the waste sludge drainage system leading to significantly greater masses of sludge in the modules than would accumulate from the influent flow.

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SUMMARY REPORT ON RESEARCH CONTRACT K3/373 (2000-2001): FILAMENTOUS BULKING CONTROL WITH REDOX POTENTIAL

This is a summary of the detailed report: Tsai *et al.* (2002), Research Report No. W116, Dept. of Civil Eng., Univ of Cape Town, Rondebosch, 7707, RSA.

3.1 MOTIVATION

From the review in Section 1, it was speculated that if the duration of the anoxic period can be controlled in such a way that aeration only commences again when the redox potential is low (implying low nitrate and nitrite concentrations at the time that conditions become aerobic), then from the AA filament bulking hypothesis of Casey *et al.* (1999), a good sludge settleability should be maintained in the system. Therefore, if the nitrate and nitrite concentrations at the end of the anoxic period can be controlled to low values by means of on-line redox potential measurement, it would appear that AA filament bulking should be controlled, but this needs to be investigated at laboratory scale before it can serve as further support for the bulking hypothesis of Casey *et al.* (1994, 1999). So in this research project, the prospects of AA filament bulking control with redox potential are investigated. Details of Casey's bulking hypothesis and the research information associated with it are reviewed in Section 1 above. The experimental system setup and development of the on-line redox potential and aeration cycle controller are described in Section 3.2 below. The experimental results are presented in Section 3.3 and the conclusions in Section 3.4.

3.2 EXPERIMENTAL SYSTEMS AND INVESTIGATION METHODS

3.2.1 System set-up and operation

During the experimental investigation three laboratory-scale intermittently aerated nitrification-denitrification (IAND) systems were operated. IAND systems were named Experimental System ES, Control System CS and Warburton System WS. All three systems consisted of a single completely mixed reactor with a volume of 15 litres and a inclined tubular secondary settling tank of 1.5 litres in series with an underflow recycle ratio of 1:1 (Fig 3.1, Table 3.1). Three such systems called ES, CS and WS were operated at different stages of the investigation.

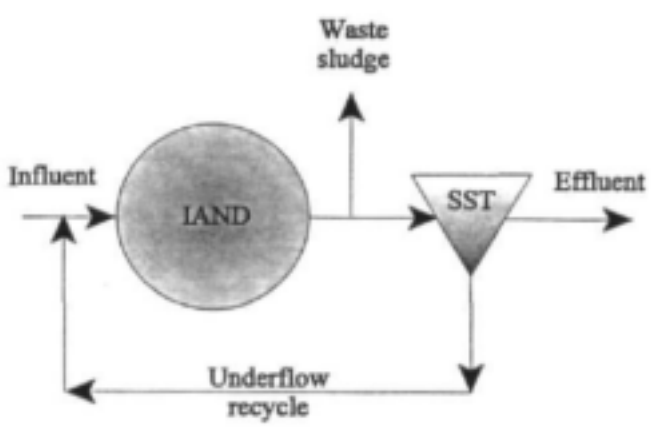


Fig 3.1: Single reactor IAND systems.

Table 3.1: *Initial operating conditions and parameters for the 3 single reactor, continuously fed, completely mixed, and intermittently aerated nitrification-denitrification systems.*

Parameter (unit)	Value	Value	Value
Reactor:	ES	CS	WS
Operation duration	Day 1-1000	Day 250-866	Day 724-1000
Experimental Periods	Periods 1-9	Periods 2-9	Periods 7-9
System configuration:			
Volume (ℓ)	15	15	15
Underflow recycle ratio	1 : 1	1 : 1	1 : 1
Hydraulic retention time (d)	1	1	1
Sludge age <Rs> (d)	15	15	15
Temperature (°C)	20	20	20
pH of the mixed liquor	7.0-7.5	7.0-7.5	7.0-7.5
MLSS (mg/ℓ)	1700-2900 [1]	1700-2900 [1]	1700-2900 [1]
MLVSS (mg/ℓ)	1400-2400 [1]	1400-2400 [1]	1400-2400 [1]
Influent raw sewage:			
Source	Mitchell's Plain sewage works	Mitchell's Plain sewage works	Mitchell's Plain sewage works
Flow rate (ℓ/d)	15	15	15
COD <target> (mg/ℓ)	750	750	750
Mass of influent COD <target> (g/d)	11.25	11.25	11.25
TKN (mg/ℓ)	60-145 1	60-145 1	60-145 1
FSA (mg N/ℓ)	45-120 1	45-120 1	45-120 1
F/M-ratio (mg COD/mg VSS)	0.3-0.54	0.3-0.54	0.3-0.54
TKN/COD Ratio	0.08-0.19	0.08-0.19	0.08-0.19
Aeration:	Intermittent	Intermittent	Intermittent
DO <aerobic period> (mg O/ℓ)	2-5 [2]	2-5 [5]	2-3 [6]
Means of control of aerobic period	Redox	Timer	Timer
Duration of aerobic period	3 hours [2]	3 hours [3]	~ 4 minutes [4]
Means of control of aerobic period	Redox	Timer	Timer
Duration of anoxic period	Variable	3 hours [3]	~ 6 minutes [4]

1 Values varied between sewage batches and as a result of sludge spillage due to pipe blockages.

[2] Controlled by Redox/DO controller- see text for description.

[3] Controlled by timer connected to a solenoid value in the air supply pipe.

4 Aeration was on for 1 minute and stopped. This raised the DO to 2-3 mgO/ℓ which decreased to zero about 3 minutes later, which was repeated every 10 minutes.

5 The DO/Redox meter of Randall *et al.* (1991) controlled the DO between 2 and 5 mgO/ℓ during the 3 hours aerobic period measured the OUR; the DO/OUR meter was switched on and off every 3 hours by a timer.

[6] Only a timer controlled solenoid value controlled the aeration cycle; the timer opened the solenoid value for 1 minute every 10 minutes. During the 1 min aeration, the DO increased to 2-3 mgO/ℓ which was reduced to zero in the following 2-3 minutes, leaving the system anoxic for the final 6-7 minutes of the cycle.

The 3 systems were operated in the usual way in the Water Research Laboratory. Details of the initial design and operation parameters of the three IAND systems are given in Table 3.1. The operating conditions and parameters of the systems were changed at strategic times during the investigation as the observations of the system performance suggested new directions for investigation. These are discussed in Section 3.3 below, where the results of the investigation are presented chronologically.

3.2.2 Control of aeration cycle

The control of the aeration cycle for ES, CS and WS was by means of Redox/DO controller, timer only and timer only respectively. Because the timer controlled aeration cycle has been used often before (Warburton *et al.*, 1991) and is set out in sufficient detail on Table 3.1, the aeration cycle control for the CS and WS is not explained. The development of the redox/DO controller for the Experimental System ES is presented below.

In order to control the duration of the anoxic and aerobic periods for ES, the DO/OUR meter of Randall *et al.* (1991) was modified to include online monitoring of Redox potential, nitrate, ammonia and temperature with probes. It operated as follows starting from the beginning of the aerobic period (see Fig. 3.3)

The Redox/DO controller measured the redox potential and DO concentration in the reactor on-line and showed both graphs separately on a computer screen as output. The minimum set point redox potential (usually -45 mV) marked the initiation of the aeration period (at time = 0 minute in Fig. 3.3). During the aerobic period the DO/OUR monitoring part of the system controlled the DO between the high and low set point DO concentrations (in this case, 5 and 2 mgO/ℓ respectively Fig. 3.3) and measured the OUR during the air-off period from the DO vs time line. This alternating air-on and air-off operation repeated itself until the preset aerobic period expired (typically 180 minutes, see Fig. 3.3). When the preset aerobic period expired, the air-supply was shut off and the Redox monitoring part of the system took over the control from the DO. When the low set point redox potential value (E_{hmin} = -45mV) was reached which typically took 3 to 4 hours depending on the nitrate concentration at the end of the aerobic period, the aerobic period commenced again, and the DO/OUR monitoring part of the system resume the control again for the next 3 hours.

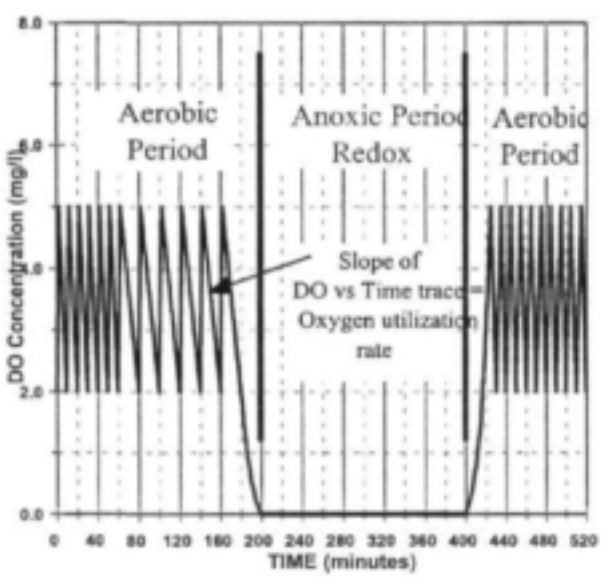


Fig 3.2: Operation of the Redox/DO controller on the ES.

At the end of the aeration period, the aeration switched off and *DO* rapidly decreased to zero and remain at zero throughout the anoxic period. During the anoxic period, the redox potential was measured every 15 seconds and recorded as an average over 8 to 10 minutes (same as the OUR).

The OUR values determined in this manner together with the average Redox Potential measurements over the same air off time were stored in a file, (see Table 3.2) which could be transferred to a spreadsheet. A graph of OUR and Redox potential versus time is plotted daily to monitor changes of the OUR and Redox potential and calculate average duration of anoxic period, for later calculations and analysis (e.g. COD and N mass balance and comparison of aerobic mass fraction with CS).

The average proportion of the aerobic period as a fraction of the aeration cycle ($f_{X,aer}$) over the day was determined from the controller output information (see Table 3.2) as:

$$f_{X,aer} = \frac{\text{Duration of the aerobic period (minutes) (Time 0 to Time 200)}}{\text{Mean duration of the aeration cycle* per day (minutes)}}$$

* Aeration cycle = aerobic + anoxic period, between Time = 0 to 400 minutes (Fig. 3.3)

Aerobic period was fixed (initially at 3 hours) but anoxic period was variable, depending on how quickly the low redox set point value was reached. The duration of the anoxic period varied from sewage batch to sewage batch depending on how much nitrate and nitrite were nitrified during aerobic period and how fast these were denitrified to reach the low redox set point. The higher the nitrate concentration, which depended on the influent TKN concentration compared with the influent COD concentration the longer the anoxic period.

For high influent TKN/COD ratios, the anoxic period was long (4 to 6h) and for low influent TKN/COD ratios the anoxic period was much shorter (1.5 to 3 h). Therefore due to the variation of the sewage characteristics from batch to batch, the number of aeration cycles (aerobic + anoxic periods) varied from batch to batch, causing aerobic time per 24 hours or aerobic mass fraction ($f_{X,aer}$) to vary from sewage batch to sewage batch. The anoxic mass fraction ($1 - f_{X,aer}$).

3.2.3 Daily sampling and testing of the systems

On-line data collection for analysis was done daily before the re-start of the computer program so that the data were a good representation of the past 24-h period. Fig. 3.4 is an example of the graphs obtained from the computer program controlling the ES. The Redox/DO controller system has the capacity for on-line ammonia, nitrate and temperature measurements. The redox potential probe was cleaned every 2nd or 3rd day with toothpaste using a toothbrush to overcome

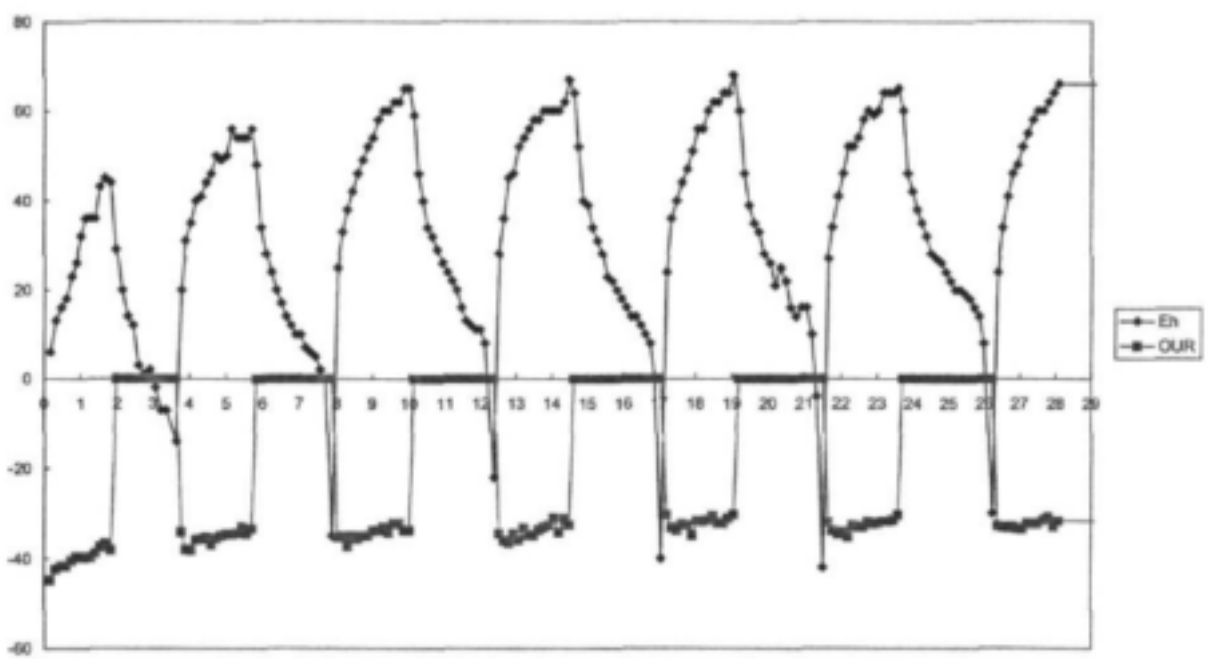


Fig 3.3: An example of daily measured redox potential and OUR (mgO/l/h) values collected by the Redox/DO controller system.

coating problems expected on the titanum tip of the probe to maintain good sensibility. Calibration of the DO probe was done at least once weekly by checking the probe's measuring range with the maximum of 9.07 mgO/ℓ in the DO saturated water (at 20°C and 1 atm. pressure) and minimum of 0 mgO/ℓ in the sodium sulphite solution.

Samples were taken from the reactor and effluent while the systems were still feeding on previous day's influent feed. The parameters listed in Table 3.3 were analyzed 3 to 5 times per week, where filtration was through 0.45µm filter membrane. Additionally, two mixed liquor samples were taken for nitrate and nitrite concentration determination, one at the end of aerobic and the other at the end of the anoxic period of all systems. Redox potential (E_h) was measured on-line automatically by the Redox controller and the length of time the ES system was aerobic and anoxic over a 24 h period ($f_{X,aer}$) was calculated for ES. Nitrate and nitrite concentration profiles during the aerobic and anoxic periods were conducted regularly on the systems to check the progress of nitrification and denitrification respectively. Also filament identifications in accordance with the methods outlined by Jenkins *et al.* (1984) were done monthly.

Table 3.2: *Sampling position and parameter measurement for ES, CS and WS.*

Test	COD[1]	TKN[2]	FSA[3]	NO_3[4]	NO_2[4]	OUR[5]	DSVI[6]	V/TSS[7]	pH[8]
Influent	✶ †[9]	✶	†						
Reactor	✶[9]	✶		†	†	✓	✓	✓	✓
Final Eff	✶ †[9]	✶ †	†	†	†				

✓ Measurement taken (filtering not applicable).
✶ Unfiltered sample.
† Filtered through Schleicher & Schüll 0.45 µm glass fibre membrane.
1,2,3 Method according to Standard Methods (1985).
4 According to Technicon AutoAnalyser Industrial Method No 33.69W.
5 With Yellow Springs DO probe and the automated method of Randall *et al.* (1991) built into the Redox/DO controller box
6 According to Lee *et al.* (1983) or Ekama and Marais (1984).
7 By separation with centrifugation, drying in a crucible 105°C and incineration at 600°C.
8 With Hanna Instruments pH meter No HI 9023.
9 For the influent readily biodegradable (RB)COD concentration, the samples were subjected to an alum flocculation step prior to filtration (Mamais *et al.*, 1993; Wentzel *et al.*, 1999).

The results of the nitrate and nitrite concentration-time profiles were plotted with the corresponding redox potential measurements. The redox profile showed the "end-point" of the denitrification process, which was observed as a sharp decline in redox potential curve, also called the redox "knee" during anoxic period (see Fig. 3.4). At the redox "knee", the nitrate and nitrite concentrations were very low (<0.1 mgN/ℓ) which led to the low set redox potential value (-45 mV) and the system became anaerobic. Therefore the redox potential measurements were used as an indication for the depletion of the nitrate and nitrite. The effect of a redox "knee" had been observed before and described as a control strategy for nitrogen removal systems by Chang and Hao (1996), Plisson-Saune *et al.* (1996) and Bernardes and Klapwijk *et al.* (1996).

3.3 EXPERIMENTAL RESULTS AND DISCUSSION

The experimental investigation was divided into ten periods according to the operating conditions imposed on the systems. These are listed in Table 3.3 below.

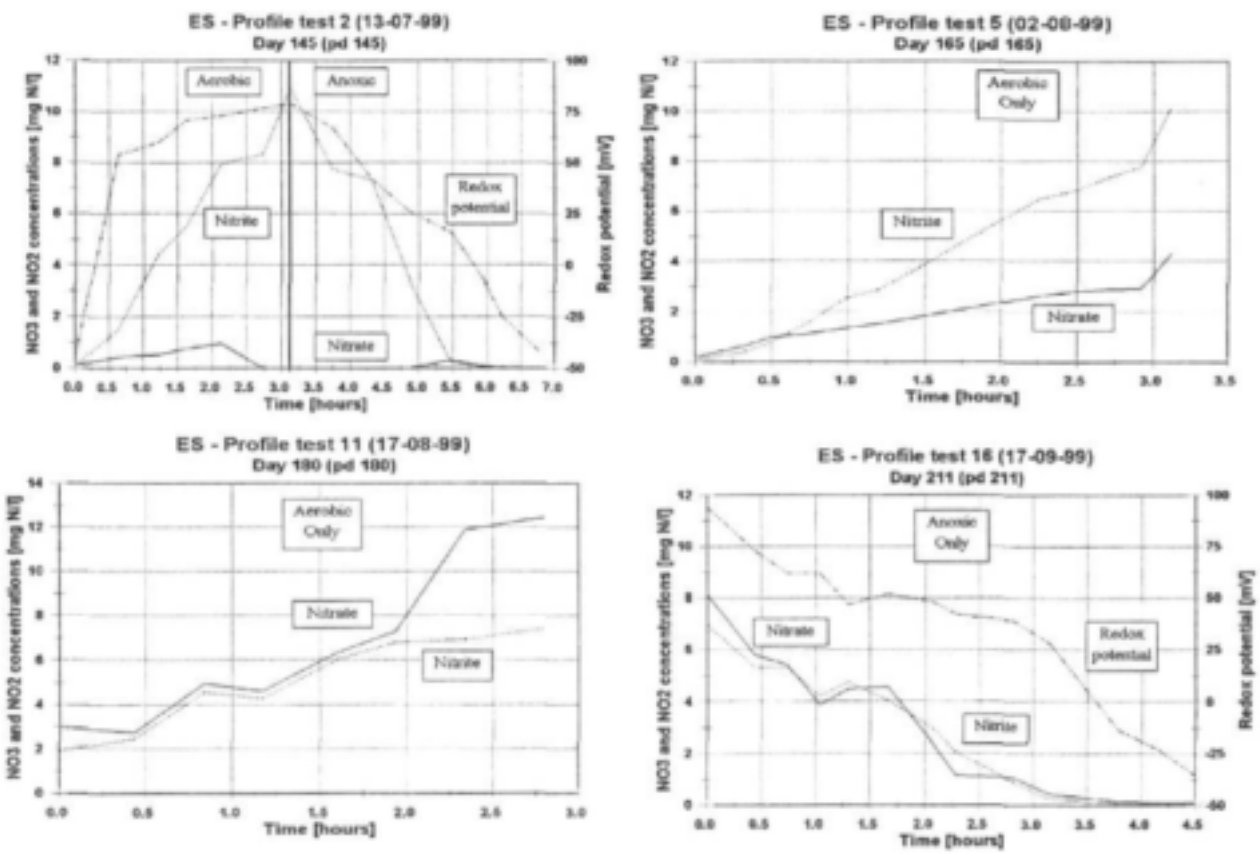
Table 3.3: Summary of the Periods with relevant system parameters.

Periods	Day (day - day)	Systems	Aerobic duration	Anoxic duration	Notes
1	1-245	Only ES No CS	3 hours -	Redox controlled	-45mV redox
2	250-384	ES CS	3 hours 3 hours	Redox controlled 3 hours	-45mV redox
3	385-426	ES CS	3 hours 3 hours	Redox controlled 3 hours	Period initiated with switched sludge
4	427-459	ES CS	30 minutes 30 minutes	Redox controlled 30 minutes	-45mV redox but changed to +100mv on day 200
5	460-497	ES CS	20 minutes 20 minutes	Redox controlled 40 minutes	+100 for ES
6	498-582	ES CS	3 hours 3 hours	Redox controlled 3 hours	-45mV redox
7	583-831	ES CS WS	3 hours 3 hours 1 minute	Redox controlled 3 hours 9 minutes	-45mV redox WS started at day 729
8	832-859	ES CS WS	Fully aerobic 0 1 minute	0 Fully anoxic 9 minutes	Fully aerobic condition Fully anoxic condition Ended with fully aerobic
9	860-947	ES CS WS	Fully aerobic 0 Fully aerobic	0 Fully anoxic 0	ATU dosage/ stopped at Day 914 Terminated at day 866 ATU dosage/ stopped at Day 896

Note: Each profile test conducted during the investigation is recorded with the date of the profile test and the numbers of the day from the beginning of the investigation and the particular Period, for example, Day 600 (pd 200) donates to the 600th day from the start of the investigation and the 200th day from the start of the Period in concern.

3.3.1 System performance - Period 1 (Day 1 to Day 245)

Only the ES was operated in this period. For this system, the aerobic time was set at 3 hours and the anoxic time was controlled by redox potential, where the low redox set value was -45 mV. Upon reaching the low set value of redox potential at -45 mV, the anoxic time ended and initiated the aerobic time. The start of aeration triggered an initial rapid increase in redox potential, see Figs 3.4 and 3.9 to 3.10. In less than 30 minutes, the redox potential increased rapidly from -45 mV to about +50 mV. After the initial rapid increase in redox, the profile exhibited a slower increase in redox potential. During the aerobic phase, ammonia was converted to nitrite by Ammonia Oxidizers and nitrite to nitrate by Nitrite Oxidizers. Hence, during the aerobic phase the nitrate and nitrite concentrations increased, at rates depending on the conversion rates of the ammonia to nitrite and nitrite to nitrate by the above two nitrifying organism groups. The anoxic phase followed the aerobic phase, and its duration was controlled by the low redox potential set value. During the anoxic condition, nitrate and nitrite concentrations decreased due to denitrification with the conversion of the nitrate to nitrite, nitrite to nitrous oxide, nitrous oxide to nitric oxide and nitric oxide to dinitrogen gas which escaped into the atmosphere.



Figs 3.4 to 3.7: *Profile tests on Experimental System (ES) on Days 145 (top right), 165 (top left), 180 (bottom right) and 211 (bottom left) respectively. The profile tests consists of nitrate and nitrite concentrations and redox potential with time over the profile. (The pd next to the day number is the period day number, i.e., number of days from start of period, nitrate - solid line, nitrite - dotted line and redox - broken line)*

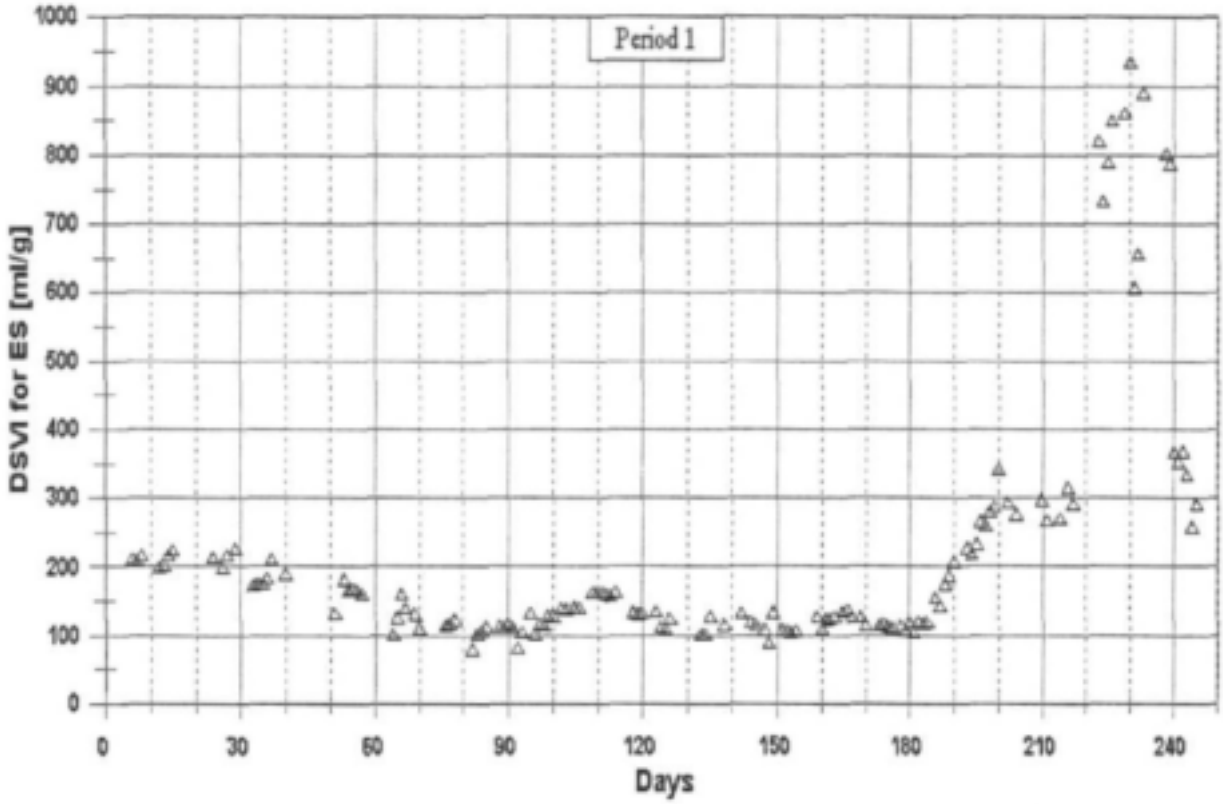
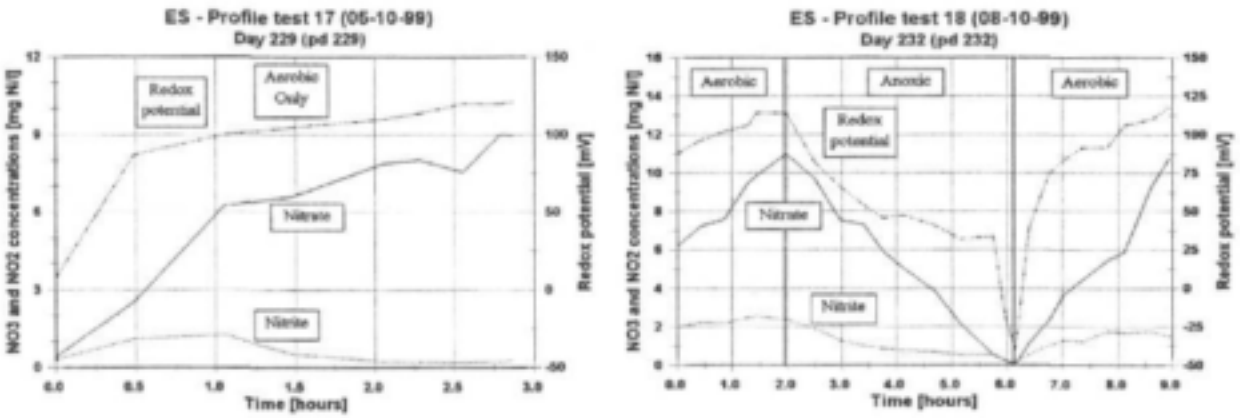


Figure 3.8: *Diluted Sludge Volume Index (DSVI) for Experimental System (ES, Δ) during the entire Period 1.*

Complete nitrification was indicated by low nitrite concentrations and significant increase in nitrate concentrations during the aerobic phase. Similarly, complete denitrification was indicated by low nitrite concentrations and significant decrease in nitrate concentrations during anoxic phases, but additionally zero nitrite and nitrate concentrations present at the end of the anoxic phase. Initially, profiles of nitrate and nitrite concentrations through an aeration cycle indicated that these conditions were satisfied - for example, a profile on Day 145 is shown in Fig 3.4. According to the bulking hypothesis of Casey *et al.* (1994), these conditions should suppress the proliferation of AA filaments so that a good settling sludge is produced. Settleability is shown plotted in Fig 3.8; DSVI of the starting mixed liquor >200 mℓ/g, but decreased to DSVI = *ca.* 120 mℓ/g by Day 150. Microscopic examination of the mixed liquor during this period indicated that filaments present in decreasing order of abundance were *M. parvicella* and type 1851; these are filaments of the AA group. These observations appeared to substantiate the AA filament bulking hypothesis, and indicated that redox potential could be successfully implemented as a parameter to control bulking by AA filaments.

However, in contrast to the behaviour above, a profile test on Day 161 (data not shown) showed significant and increasing nitrite concentrations during the aerobic phase, which suggested that nitrification was incomplete. Furthermore, subsequent profile tests carried out substantiated this observation; the second stage of nitrification in the system (nitrite oxidized to nitrate) appeared to have failed (Figs 3.5 to 3.7). These profiles also indicated that the failure of the second stage in nitrification during the aerobic phase caused significant nitrite concentrations to be present during the subsequent anoxic phase (Fig. 3.7). However, both nitrate and nitrite were

successfully denitrified to near zero by the end of the anoxic phase (Fig 3.7). The failure in nitrite oxidation was also observed in parallel NDBEPR laboratory-scale systems operated in the UCT laboratory at the time. Since all the laboratory-scale systems were fed the same influent wastewater, nitrite oxidation failure was attributed to the wastewater. During this period, on Day 180 the DSVI started to increase, to reach DSVI = *ca.* 295 mℓ/g by Day 200. Microscopic examination indicated that the increase in DSVI was caused by AA filament proliferation. The AA filament bulking hypothesis of Casey *et al.* (1994, 1999) does not specifically address the situation observed with significant nitrite concentrations present in the anoxic and aerobic phases, but zero nitrate and nitrite present at the transition from anoxic to aerobic. Observations here indicate that these conditions also promote AA filament bulking.



Figs 3.9 and 3.10: *Profile tests on Experimental System (ES) on Days 229 and 232 (nitrate - solid line, nitrite - dotted line and redox - broken line).*

On Day 212 the redox potential probe malfunctioned causing failure of redox control of the system. This caused an immediate rapid increase in DSVI, to reach DSVI = *ca.* 820 mℓ/g by Day 220. This confirmed that redox control was crucial to suppress AA filament bulking in the system.

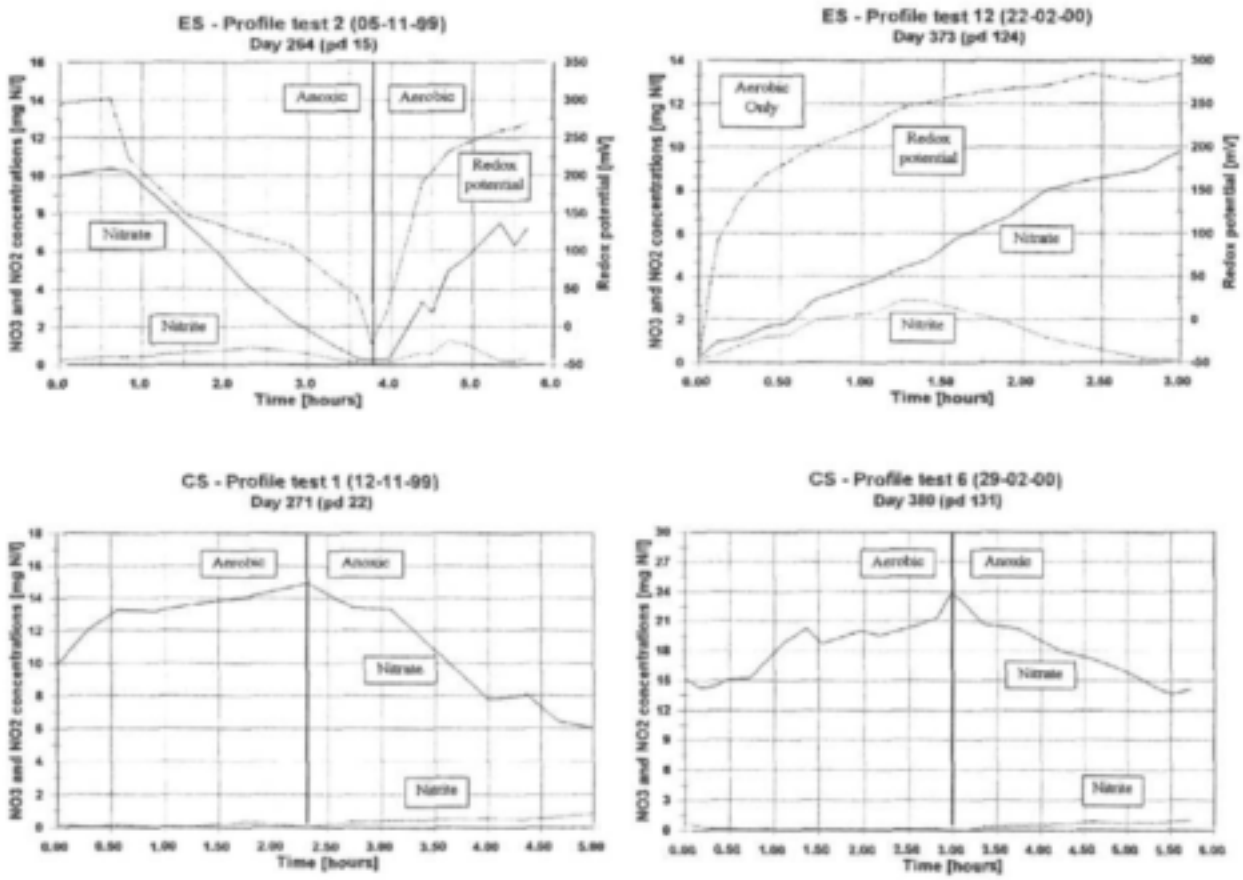
After 10 days, the problem of the redox potential probe was solved by replacing the probe (Day 222). Profiles were done on Days 229 (Fig 3.9) and 232 (Fig 3.10). These indicated that redox control had been successfully re-installed and that nitrite oxidation had been re-established in the system, so that both nitrification and denitrification were essentially complete at the end of the aerobic and anoxic phases respectively. Similarly, it was observed that nitrite oxidation was also re-established in the parallel laboratory-scale systems. This confirmed the observation that the failure of nitrite oxidation was wastewater related. Following the re-installation of redox control and the re-establishment of nitrite oxidation, the DSVI decreased rapidly to reach DSVI = *ca.* 260 mℓ/g by Day 242.

From Period 1, it can be concluded that in the intermittently aerated system, (i) high nitrite concentrations in the anoxic and aerobic phases promote AA filament bulking, and (ii) redox control is crucial to suppress AA filament proliferation and consequent bulking in this system. To confirm that redox control was successful in suppressing AA filament proliferation, it was decided to conduct an investigation with two parallel systems, identical except that one system would have redox control and the other not.

3.3.2 System performance - Period 2 (Day 250 to Day 384)

In Period 2, a Control System (CS) without redox control was started in parallel to the Experimental System (ES) from Period 1 above with redox control. The objective was to confirm the effect of redox control on AA filament bulking, by allowing the CS to bulk, but the ES not and then changing the sludges between the two systems. If the high DSVI sludge from the CS (now in ES) decreased in DSVI, this would demonstrate that redox control could not only prevent AA filament bulking, but also cure it.

The CS was set-up with same operating parameters (volume, influent flow rate, influent feed composition and sludge age) as the ES (Table 3.1). At the end of the Period 1, the ES sludge was divided equally between the ES and CS, to give rise to identical sludge characteristics in the two systems at startup. Aerobic and anoxic parameters of the ES were identical to that of Period 1, where aerobic time was fixed at 3 hours and termination of the aerobic time was controlled by a timer. The anoxic time continued after the termination of the aerobic time, and the termination of the anoxic time was controlled by the low set-value of the redox potential at -45 mV. The aerobic time of the CS system was also set to 3 hours with the termination of the aerobic time controlled by a timer. The anoxic time was not controlled by redox in this system, but was controlled by a timer at 3 hours.



Figs 3.11 to 3.14: *Profile tests on Experimental System ES and Control System CS on Days 264 (top left) and 373 (top right) and Days 271 (bottom left) and 380 (bottom right) respectively; profiles show entire nitrate and nitrite concentrations for both systems and redox potential for the ES (nitrate - solid line, nitrite - dotted line and redox - broken line).*

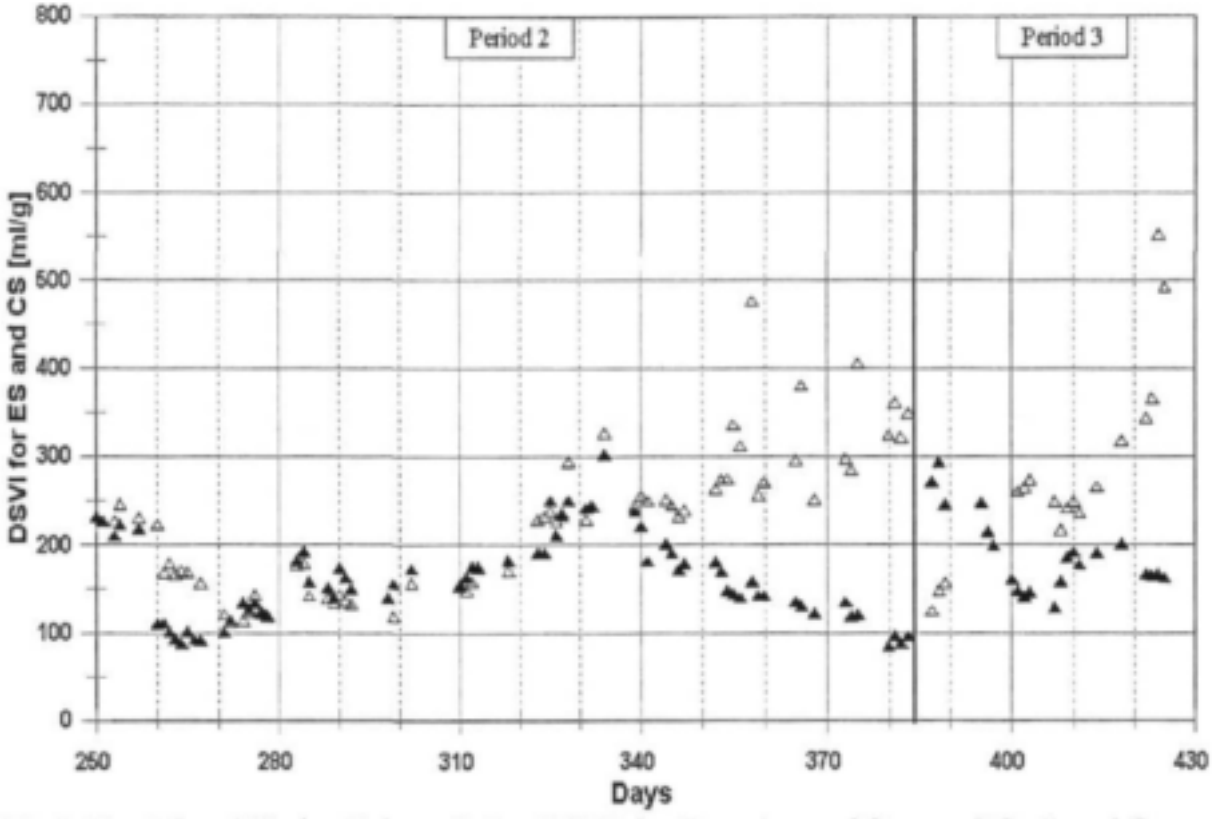


Fig 3.15: Diluted Sludge Volume Index (DSVI) for Experimental System (ES, Δ) and Control System (CS, ◢) from Period 2 to 3.

The investigation for Period 2 ran from Day 250 to Day 384, and the systems received 9 $^{1}/_{2}$ sewage batches as influent. During this period, 13 and 7 profile tests were carried out on ES and CS respectively, the purpose of the profile tests being to observe the nitrification and denitrification processes during the aerobic and anoxic phases respectively.

From ES profile test nos. 2, 3, 4, 5, 6, 7, 10, 11, 12, 13 and 14, it was evident that low set-value of redox potential at -45 mV resulted in nitrate and nitrite concentrations of less than 0.5 mgN/ℓ at the transition from anoxic to aerobic. For CS, profile tests no. 1, 3, 4, 5 and 7 showed that the nitrate concentrations at the anoxic-aerobic transition were high, at an average of 8.0 mgN/ℓ.

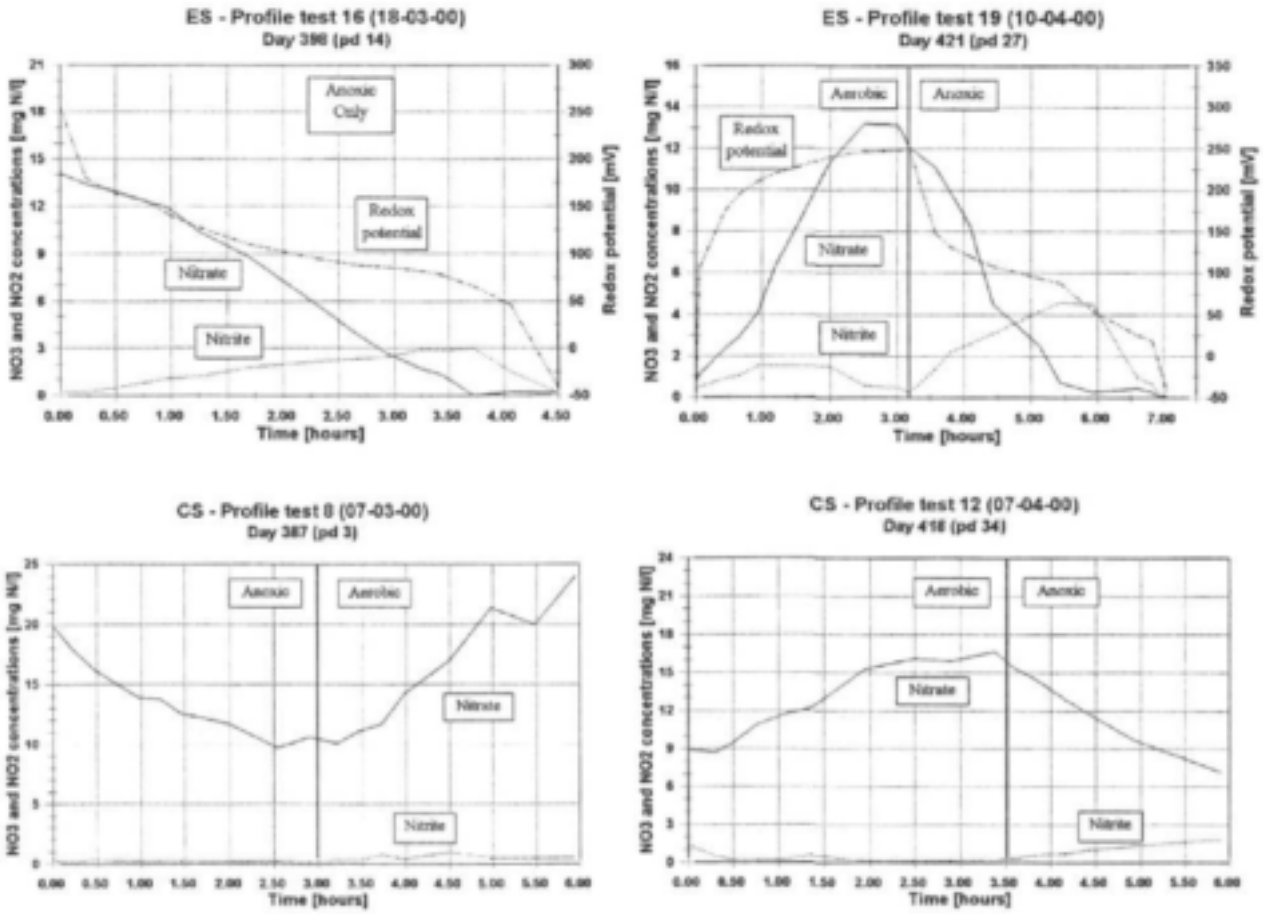
For example, profile test no. 2 (Day 264) and 12 (Day 373) for ES (Figs 3.11 and 3.12) confirmed low nitrate and nitrite concentrations at the anoxic aerobic transition, just before the subsequent aerobic phase. This confirmed the assumption that the low set-value of redox potential of -45 mV is an indirect indication of low nitrate and nitrite concentrations. Upon reaching the low set-value of the redox potential, the aerobic phase begins and terminates in 3 hours, by closure of the air-supply valve on the air supply. ES showed near-complete nitrification indicated by low nitrite concentrations and the plateau of the nitrate concentration; also it demonstrated complete denitrification indicated by low nitrate and nitrite concentrations at the end of the anoxic phase. Profile test nos. 1 (Day 271) and 6 (Day 380) for CS (Figs 3.13 and 3.14) demonstrate that nitrification and denitrification took place, but that the denitrification was not complete, indicated by high nitrate and nitrite concentrations at the end of the anoxic phase.

Interpreting the profile results in terms of the AA filament bulking hypothesis, in the ES the nitrate and nitrite concentrations are negligible at the transition from anoxic to aerobic whereas these are significant in the CS. According to the hypothesis, AA filament bulking should be suppressed in ES and promoted in CS. However, this hypothesized behaviour was not observed (Fig. 3.15). The DSVI in both systems started at DSVI = *ca.* 230 mℓ/g and both showed an initial decline to reach DSVI = *ca.* 110 mℓ/g by Day 270. Thereafter, the DSVI in both systems increased at about the same rate, to reach DSVI = *ca.* 250 mℓ/g by Day 339. The DSVI in ES continued to increase to reach DSVI = *ca.* 349 mℓ/g by Day 383. Microscopic examination indicated that the increase in DSVI was caused by AA filament proliferation, with *M. parvicella* and type 1851 dominant in both ES and CS. In contrast, the DSVI in CS decreased to DSVI = *ca.* 90 mℓ/g by Day 383. This observed behaviour is exactly contrary to that expected from the AA filament bulking hypothesis.

The cause for the deviation in observed behaviour from that expected in terms of the bulking hypothesis could not be determined. Accordingly, the decision was made to switch the sludges between the ES and CS. The objective of the switch was to identify any possible physical system parameter(s) that could have influenced the biological performance of the ES and CS sludges.

3.3.3 System performance - Period 3 (Day 385 to Day 426)

Period 3 started with the switch of the sludge contents of the two reactors ES and CS; the sludge previously in the ES was now in the CS and visa versa. Apart from this switch, conditions for the two systems remained identical to those in Period 2.



Figs 3.16 to 3.19: *Profile tests on Experimental System on Days 398 (top left) and 421 (top right) and on the Control System on Days 387 (bottom left) and 418 (bottom right) (nitrate - solid line, nitrite - dotted line and redox - broken line).*

Profiles conducted on ES on Days 398 and 421 (Figs 3.16 and 3.17) indicated that this system was operating as required, with negligible nitrate and nitrite concentrations at the transition from anoxic to aerobic. Profiles on CS on Days 387 and 418 (Figs 3.18 and 3.19) indicated that this system also was operating as required, with high nitrate and nitrite concentrations present at the transition from anoxic to aerobic. However, despite correct system operation, the settleability again was contrary to the behaviour expected in terms of the AA bulking hypothesis: from Fig 3.20 (Day 384 to 426), the DSVI in ES continually increased from DSVI = *ca.* 120 mℓ/g on Day 375 to DSVI = *ca.* 360 mℓ/g by Day 424 and to over 500 mℓ/g by Day 426, while the DSVI in CS decreased from DSVI = *ca.* 300 mℓ/g on Day 375 to DSVI = *ca.* 150 mℓ/g by Day 424. Microscopic examination showed that the predominant filaments in both systems were of the AA group, with *M. parvicella* dominant for ES and with *M. parvicella* and type 1851 dominant for CS. This behaviour conforms to that observed during Period 2, and indicates that, contrary to expectation, the CS suppressed AA filament proliferation while ES promoted it.

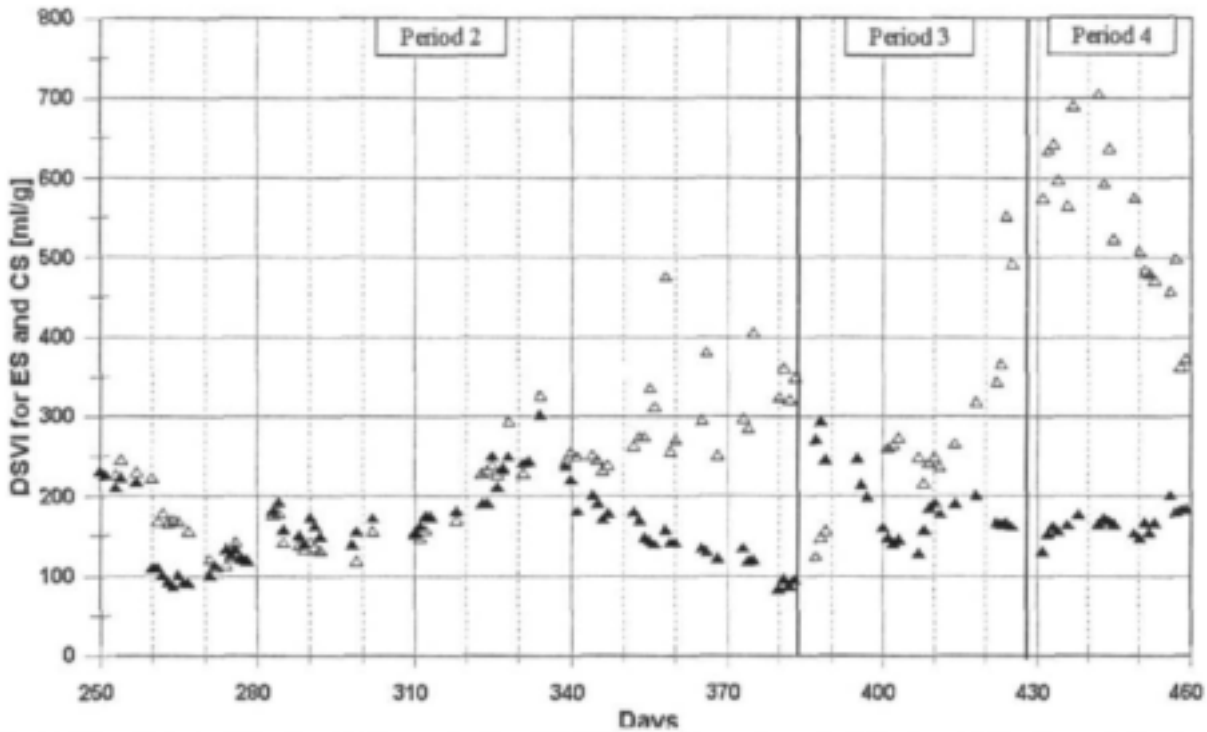


Fig 3.20: *Diluted Sludge Volume Index (DSVI) for Experimental System (ES, Δ) and Control System (CS, ◢) from Period 2 to 4.*

Warburton *et al.* (1991) and Gabb *et al.* (1989, 1996) had no difficulty developing AA filament bulking sludges (in particular *M. parvicella* and type 1851) in intermittently aerated nitrification denitrification systems. These systems generally had 10 to 20 minute cycle times commencing with a two minute aeration period raising the DO to 2 to 3 mgO/ℓ, after which the aeration was switched off for the remainder of the cycle time. The systems generally had aerobic mass fractions of 30 to 50 % and at sludge ages longer than 10 days and nitrification was usually complete.

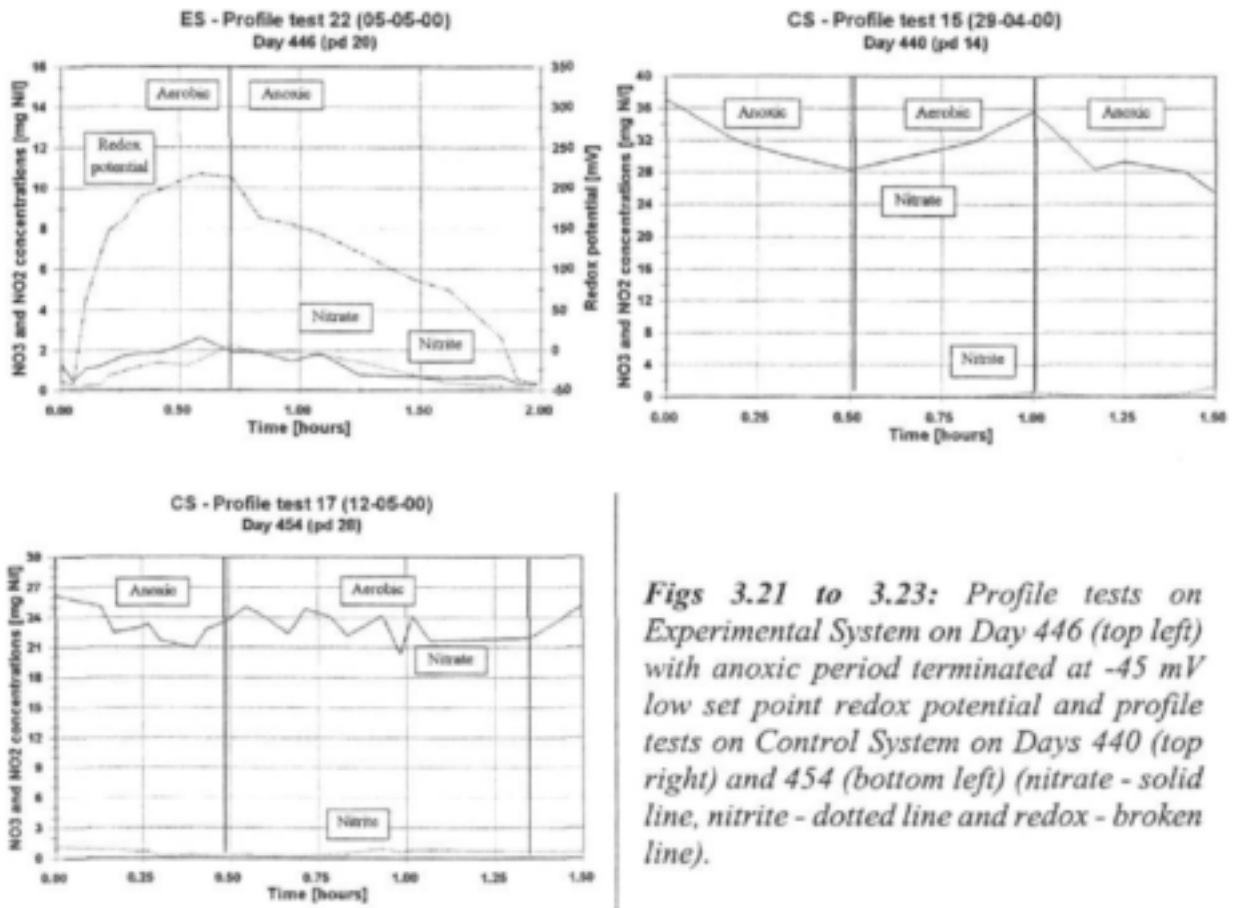
Lakay *et al.* (1999) examined the effect of the aerobic cycle time, which influences the frequency of alternation between anoxic and aerobic conditions, on AA filament bulking. They found that cycle times of 20 minutes to 3 days did not change significantly the AA filament proliferation - bulking was not ameliorated by lengthening the aeration cycle time. FSA was dosed into the

influent to keep the TKN/COD ratio at about 0.10 so that the anoxic time to redox potential -45 mV did not vary much from sewage batch to sewage batch which would keep the aerobic mass fraction at around 0.60 (60 % of the total aeration cycle).

Despite these earlier observations, and not knowing why the ES and CS were exhibiting behaviour completely contrary to the behaviour expected from the AA filament bulking hypothesis, it was thought that possibly the aeration cycle was too long which may influence the sludge settleability, and accordingly this aspect was investigated. Hence Period 3 was terminated by changing aeration cycles for ES and CS.

3.3.4 System performance - Period 4 (Day 427 to Day 459)

Period 4 continued from Period 3, but with changed aeration cycle times. For the CS, the aerobic mass fraction was kept constant at 0.5, with the aerobic time being half of the aeration cycle time. However, the total aeration cycle time was reduced from 6 hours (3 hours aerobic and 3 hours anoxic) to 60 minutes (30 minutes aerobic and 30 minutes anoxic). For ES, the aerobic time was also changed from 3 hours to 30 minutes, but the determining factor for the end of the ES anoxic time was maintained at the -45 mV redox potential. These changes were made to evaluate the effects of aeration cycle time on the performance of the systems.



Figs 3.21 to 3.23: *Profile tests on Experimental System on Day 446 (top left) with anoxic period terminated at -45 mV low set point redox potential and profile tests on Control System on Days 440 (top right) and 454 (bottom left) (nitrate - solid line, nitrite - dotted line and redox - broken line).*

For the ES system, profile tests indicated that this system was operating as required (for example, see Fig. 3.21), with low nitrate and nitrite concentrations at the end of the anoxic phase. Similarly, profiles on the CS system indicated that this system was operating as required (for example, see Figs 3.22 and 3.23), with high nitrate and nitrite at the end of the anoxic phase. However, again settleability did not conform to the AA filament hypothesis. The ES had high and erratic DSVI between *ca.* 580 to 710 mℓ/g and the CS had a much lower DSVI = *ca.* 160 mℓ/g (Fig 3.24).

From the results obtained thus far, it appeared that the CS was more successful at suppressing AA filament growth than the ES. To establish whether this observation may have resulted from some unidentified physical system condition, it was decided to change the redox control in the ES to ensure significant nitrate and nitrite concentrations at the transition from anoxic to aerobic.

Accordingly, on Day 449 the low redox set point on the ES was changed from -45 mV to +100 mV. From observations on previous profiles, this value would occur when significant nitrate and nitrite are still present. With the ES on +100mV low set point redox potential, both CS and ES now switched from anoxic to aerobic conditions at high nitrate/nitrite concentrations. This was done to see if the high DSVI in the ES would become similarly low as that in the CS.

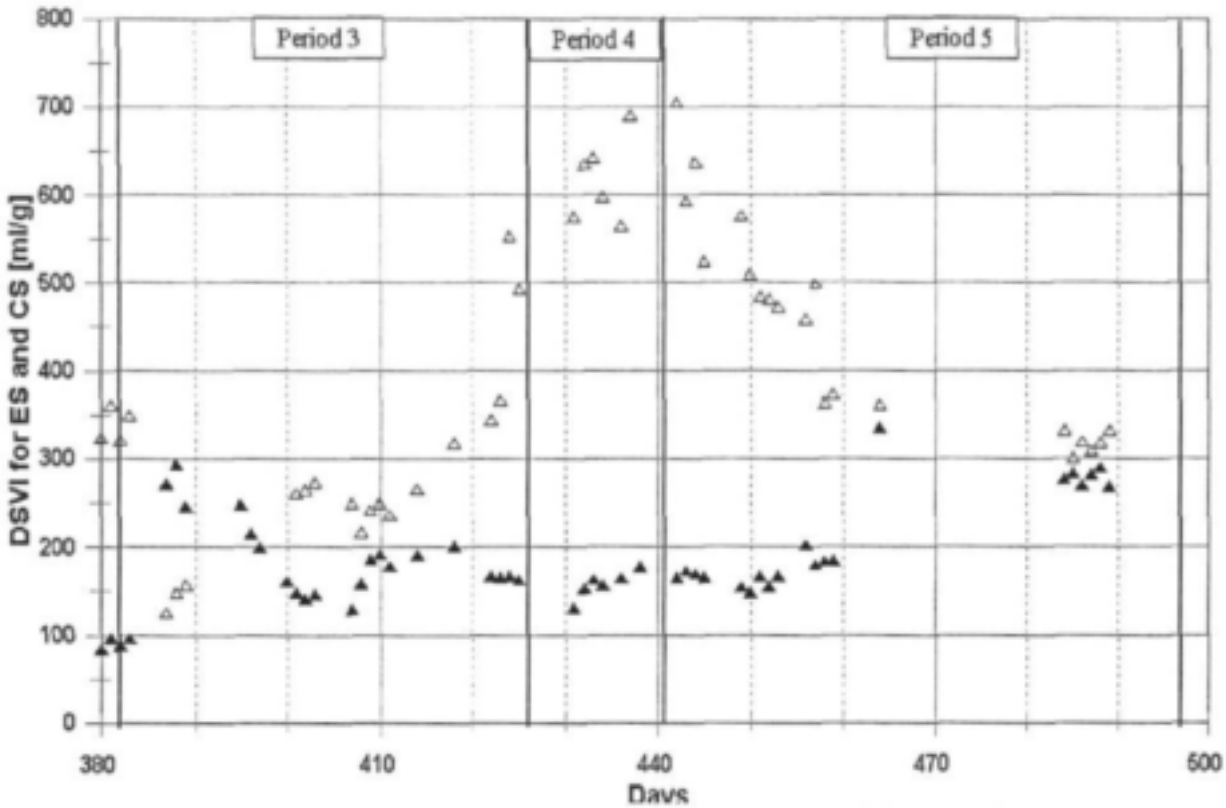


Fig 3.24: Diluted Sludge Volume Index (DSVI) for Experimental System (ES, Δ) and Control System (CS, ◢) from Period 3 to 5.

After changing the low set value of redox potential to +100 mV, as expected the anoxic periods were shorter than previously. Profiles conducted on ES during this period showed that significant nitrate and nitrite were present at the transition from anoxic to aerobic (Fig 3.25). In response to this change, the DSVI did show some improvement, decreasing from DSVI = *ca.* 580 mℓ/g on Day 449 to DSVI = *ca.* 375 mℓ/g by Day 459 (Fig 3.26). Again, this result is contradictory to the AA filament bulking hypothesis, and confirmed that the observation of lower DSVI in the CS system was not due to some unidentified physical condition - the lower DSVI could be reproduced in the ES system if it were operated with nitrate and nitrite present at the transition from anoxic to aerobic.

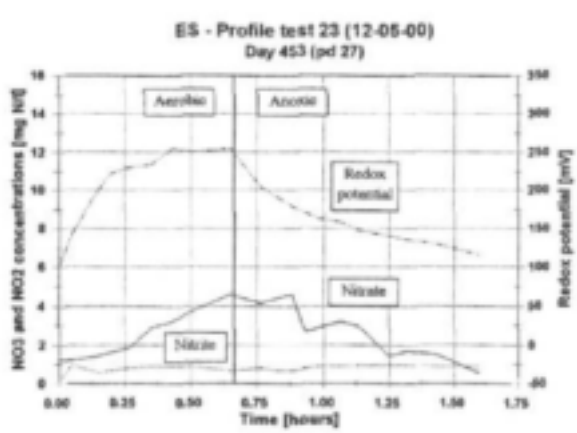


Fig 3.25: Profile test on Experimental System on Day 453, with minimum redox potential set at +100 mV (nitrate - solid line, nitrite - dotted line and redox - broken line).

In the CS, from Day 449 to 459 the DSVI remained stable at about DSVI = *ca*. 150 to 180 mℓ/g, despite high nitrate concentrations (> 20 mgN/ℓ) at the transition from anoxic to aerobic. However, profiles on this system showed that the denitrification performance in this system was poor and deteriorating, (Figs 3.22 and 3.23). Further, nitrification also showed deterioration (Fig 3.23), with little increase in nitrate during the aerobic phase. The lack of denitrification during the anoxic phase was thought to be due to insufficient anoxic time. Accordingly, on Day 460 the Period ended with decision made to increase the anoxic time.

3.3.5 System performance - Period 5 (Day 460 to Day 497)

This period continued from the previous Period 4, but on Day 460 the aerobic/anoxic times of 30 minutes/30 minutes were changed to 20 minutes/40 minutes, to increase the anoxic mass fraction to ensure sufficient time for denitrification process to take place. At the same time, the aerobic time for ES was also reduced from 30 minutes to 20 minutes, but the low set value of redox potential was kept constant at +100 mV.

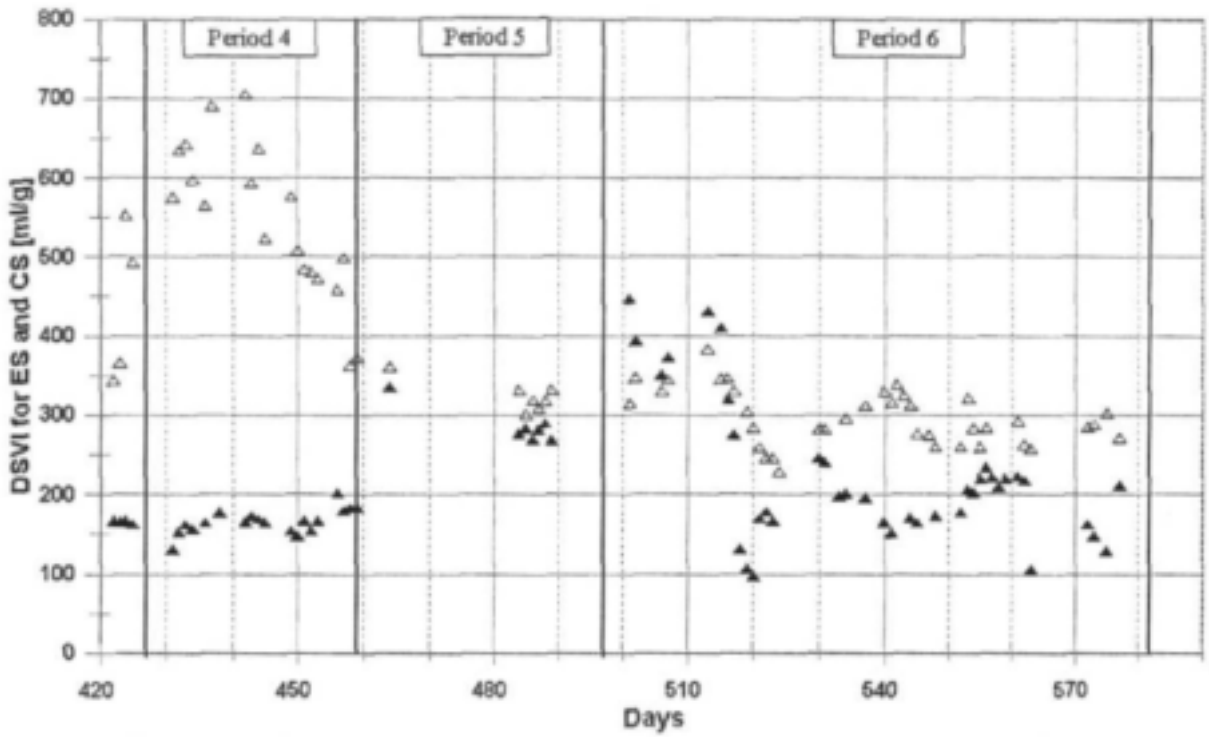
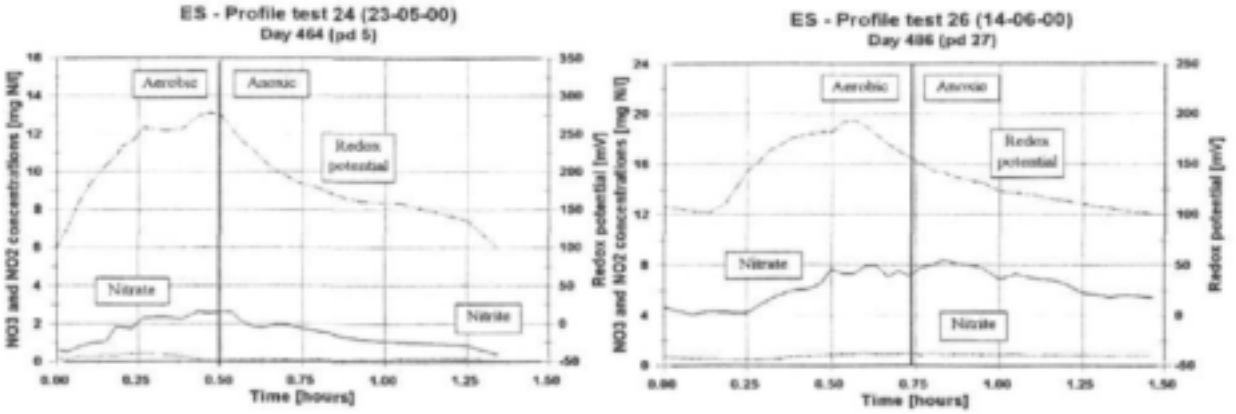


Figure 4.23: *Diluted Sludge Volume Index (DSVI) for Experimental System (ES, Δ) and Control System (CS, ◢) from Period 4 to 6.*

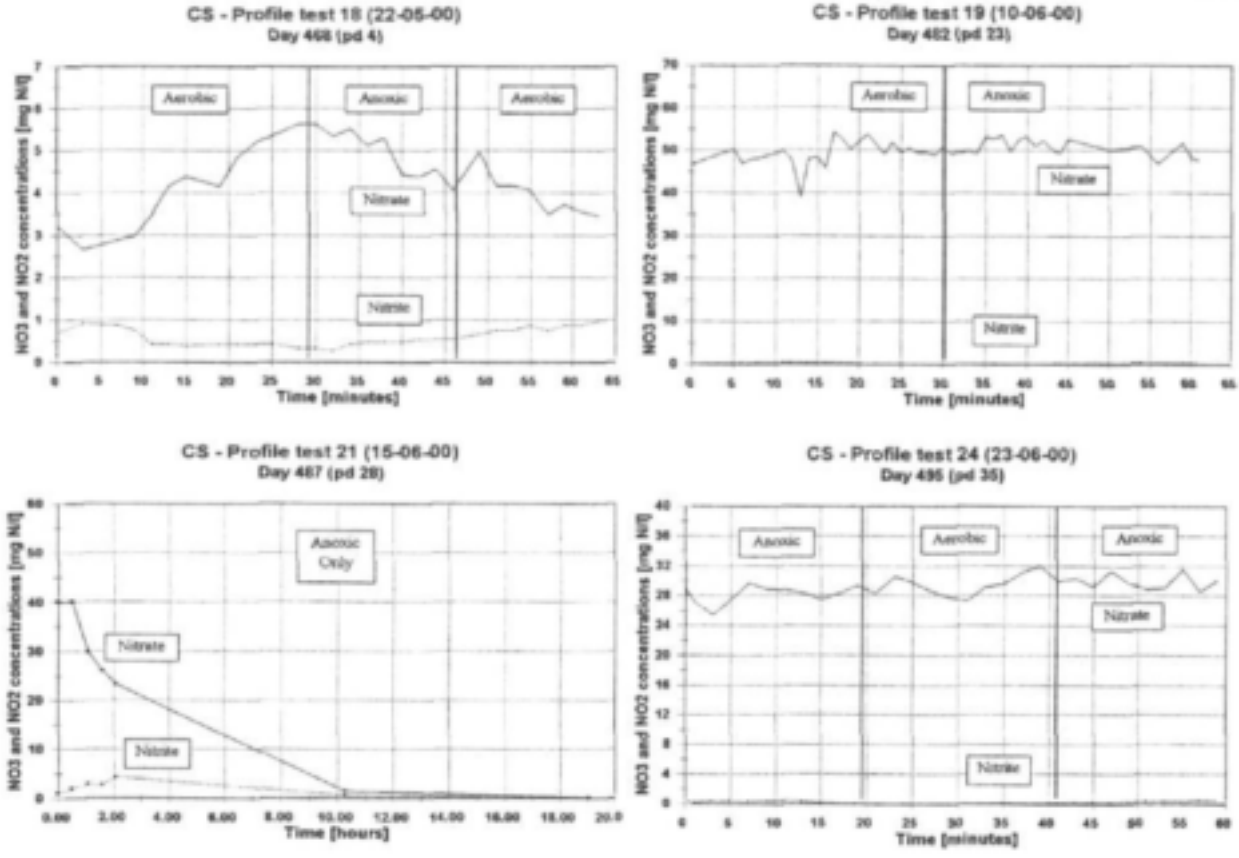
As noted in Period 4, the DSVI did show some decrease, from DSVI = *ca.* 375 mℓ/g at the start of the period (Day 460) to level off at DSVI = *ca.* 320 mℓ/g by the end of the period (Day 497) see Fig 3.26. However, this DSVI indicates that the sludge still is a bulking one (DSVI ≮ 150mℓ/g). As earlier, microscopic examination of the sludge indicated that this bulking was due to AA filaments, with *M. parvicella* dominant. Thus, the redox set value of +100 mV was also not successful in controlling bulking by AA filaments in ES. For ES, profiles indicated that this system was still nitrifying and denitrifying efficiently (Figs 3.27 and 3.28 on Days 464 and 486). Further, the profiles showed significant nitrate and nitrite concentrations present at the transition from anoxic to aerobic, indicating the system was operating as required.



Figs 3.27 and 3.28: *Profile tests on Experimental System Days 464 and 486 (nitrate - solid line, nitrite - dotted line and redox - broken line).*

For CS, after changing the aerobic/anoxic times from 30 minutes/30 minutes to 20 minutes/40 minutes, the processes of nitrification and denitrification initially improved (Fig 3.29). The nitrite concentration was low at all times and increasing and decreasing nitrate concentrations during the aerobic and anoxic phases respectively indicated that efficient nitrification and denitrification was taking place. However, this efficient nitrification and denitrification did not persist. Later profile tests indicated failure of these processes, see e.g. Fig 4.27 (Day 482) which shows high and approximately constant nitrate and nitrite concentrations during both aerobic and anoxic phases. Hence, on Day 487 the decision was made to close the air-supply to the CS until the nitrate and nitrite were fully denitrified, see Fig 3.31. This profile test on Day 487 consisted of a 19-hours denitrification period where denitrifying organisms were given sufficient time to denitrify all available nitrate and nitrite. That denitrification took place indicated that the system retained denitrification capability, but this was not happening during the anoxic phase of the cycle. At the end of the profile test the system was subjected to an aerobic phase where nitrification took place. This 19-hours denitrification period did not stimulate the nitrification and denitrification as expected. During subsequent aeration cycles the CS still exhibited failed nitrification and denitrification, for example see Fig 3.32 on Day 495.

It was concluded that the aerobic and anoxic times were too short for efficient nitrification and denitrification respectively. According, at the end of the Period 5 (Day 497) the aerobic/anoxic times were changed from 20 minutes/40 minutes to 3 hours/3 hours, the same times implemented during Periods 2 and 3, and the low redox set value was set to -45mV.

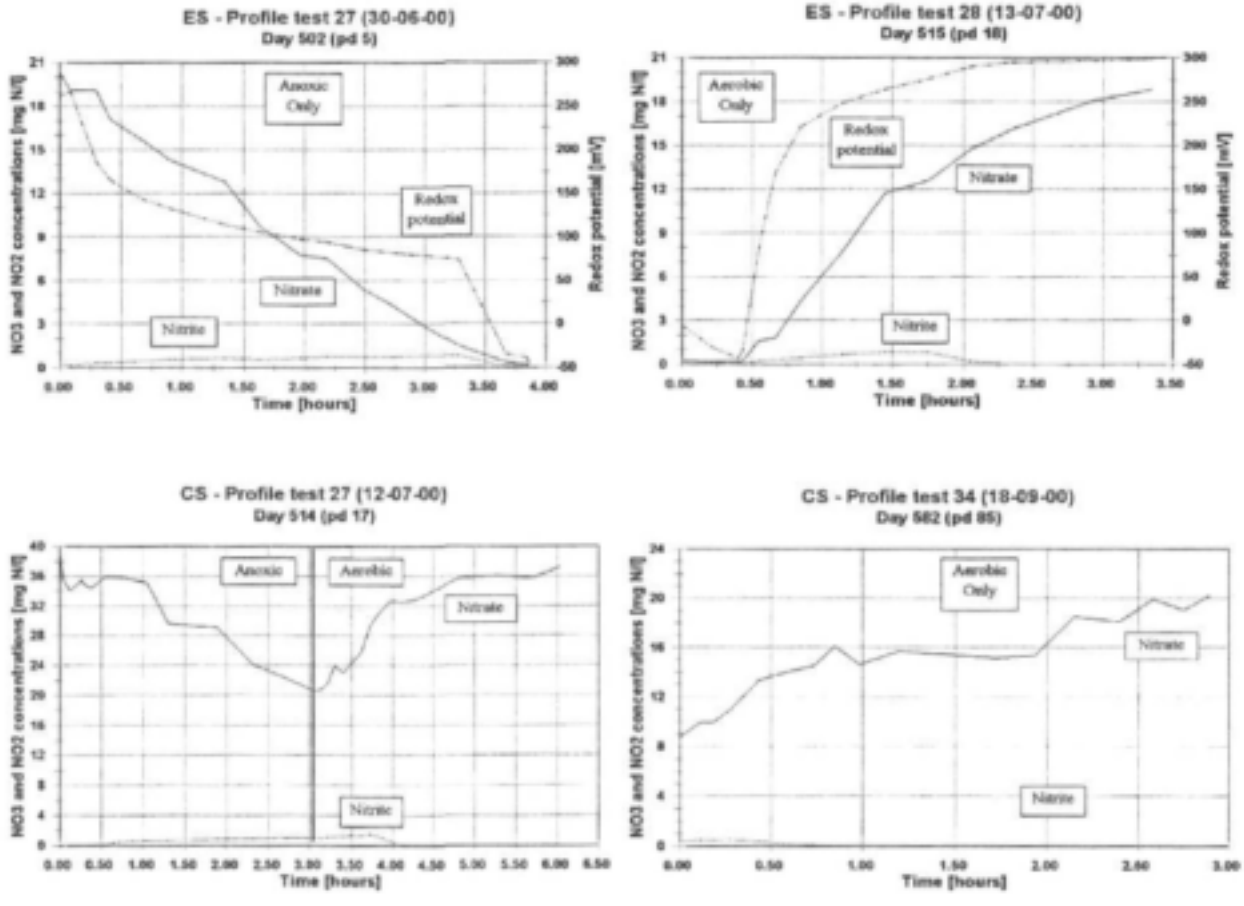


Figs 3.29 to 3.32: *Profile tests on Control System on Days 463 (top left), 482 (top right), 487 (bottom left) and 495 (bottom right) (nitrate - solid line, nitrite - dotted line and redox - broken line).*

3.3.6 System performance - Period 6 (Day 498 to Day 582)

Period 6 continued from Period 5, but with changed aeration cycle times. For the CS, the aerobic mass fraction was changed to 0.5, with the aerobic time being half of the aeration cycle time, the total aeration cycle time was increased from 60 minutes (20 minutes aerobic and 40 minutes anoxic) to 6 hours (3 hours aerobic and 3 hours anoxic). For ES, the aerobic time was also changed from 20 minutes to 3 hours, and the determining factor for the end of the ES anoxic time was also reduced to -45 mV redox potential. These changes were made to evaluate the effects of aeration cycle time on the performance of the systems.

For the ES system, profile tests indicated that this system was operating as required (Figs 3.33 on Day 502 and 3.34 on Day 515), with low nitrate and nitrite concentrations at the end of the anoxic phase. Similarly, profiles on the CS system indicated that this system was operating as required (Figs 3.35 on Day 514 and 3.36 on Day 582), with high nitrate and nitrite at the end of the anoxic phase. However, again settleability did not conform to the AA filament hypothesis. The ES had DSVI between *ca.* 220 to 350 mℓ/g and the CS had a DSVI reduced from *ca.* 450 to 200 mℓ/g (Fig 3.37).



Figs 3.33 to 3.36: *Profile tests on Experimental System on Days 502 and 515 and on Control System on Days 514 and 582 (nitrate - solid line, nitrite - dotted line and redox - broken line).*

At the beginning of the period, the DSVI of the CS was considerably higher than ES. From the previous period, it was found that short aerobic and anoxic times caused inefficient nitrification and denitrification respectively, therefore it was perceived that at the beginning of this period of longer aerobic duration, the organisms within CS quickly adapted to the longer aerobic and anoxic durations giving them ample time for nitrification and denitrification. Therefore after one sludge age of 15 days, the DSVI for CS soon decreased from *ca.* 450 to 200 mℓ/g. Whereas ES, the DSVI stayed between *ca.* 220 to 350 mℓ/g. Hence from the results obtained thus far, it appeared that the CS was more successful at suppressing AA filament growth than the ES.

From the CS profile test nos. 25 through to 34, it was evident that at the end of the transition from anoxic to aerobic, nitrate and nitrite concentrations were indeed higher than 10 mgN/ℓ, yet the DSVI did not increase and AA filaments did not proliferate in these conditions. In contrast, the ES profile test nos. 27, 28 and 30 showed the expected low nitrate and nitrite concentrations at the transition from anoxic to aerobic with low set-value of redox potential of -45 mV. Yet the environment of low nitrate and nitrite concentrations did not discourage the growth of the AA filaments, which was *M. parvicella* dominant. At the end of the of the period, the DSVI for ES and CS was 300 and 150 mℓ/g respectively.

From Period 2 to 6, the observed results of DSVI in ES and CS have been contrary to the AA filament hypothesis and the approach adopted in the investigation no longer seemed worthwhile to pursue any longer. Accordingly a different strategy was adopted from Period 7. Therefore it was decided to switch ES to fully aerobic to try and reduce the proliferation of the AA filaments and to dose azide (NaN_3) to ES. But prior to the dosage of azide, the effect of azide need to be verified. This period ended with the continuation of CS and introduction of fully aerobic conditions for ES.

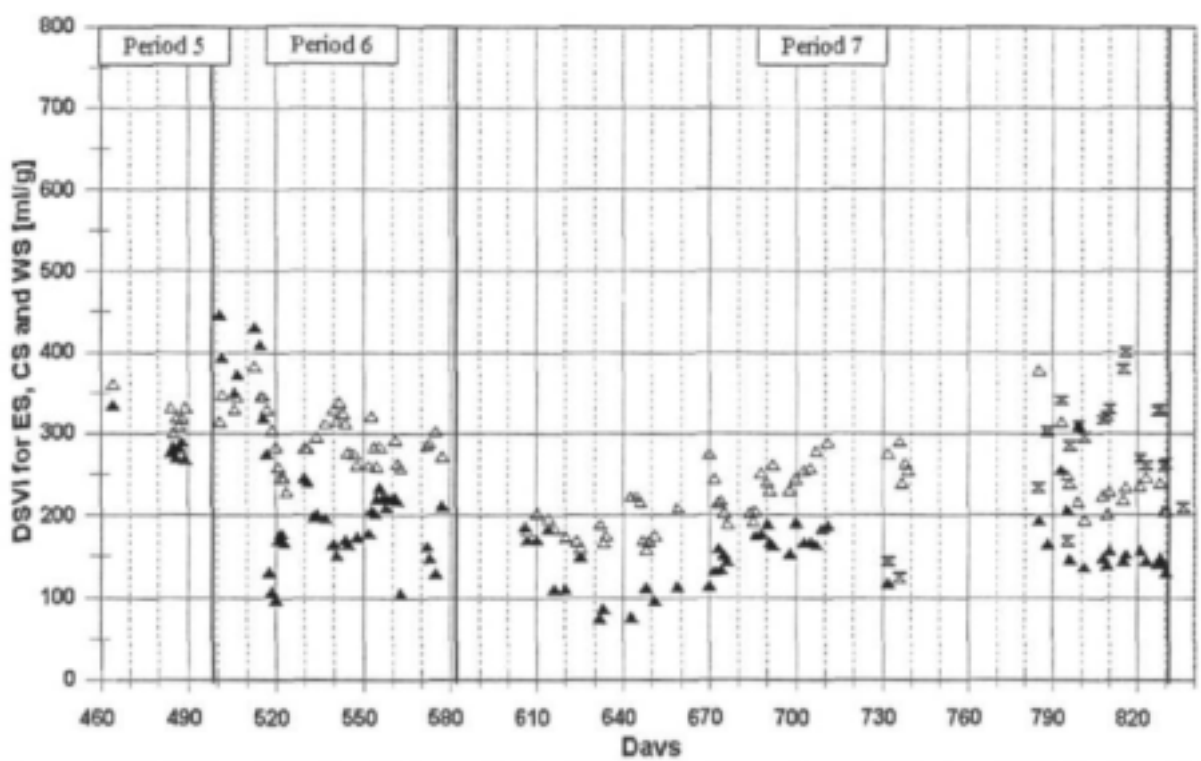


Figure 4.34: *Diluted Sludge Volume Index (DSVI) for Experimental System (ES, Δ), Control System (CS, ◢) and Warburton System (WS, ◪) from Periods 5 to 7.*

3.3.7 System performance - Period 7 (Day 583 to Day 831)

Period 7 continued from Period 6 without any changes made ES and CS. For the CS, the aerobic mass fraction was kept constant at 50%, with the aerobic time being half of the total cycle time, i.e. 3h aerobic and 3h anoxic in a 6h cycle time. For ES, the aerobic time was changed at 3h and the determining factor for the end of the anoxic time was maintained at -45mV. However, due to the high DSVIs for ES, it was decided to change it to fully aerobic to reduce the DSVI - in the past fully aerobic conditions invariably lead to a decrease in DSVI and a reduction in AA filament bulking (Gabb *et al.*, 1996). This change was made to evaluate the effects of fully aerobic conditions (100% aerobic mass fraction) on the settleability of the ES. During this period, batch tests were carried out to check the feasibility of azide addition to ES. Azide inhibits the nitrite oxidizers preventing the formation of nitrate.

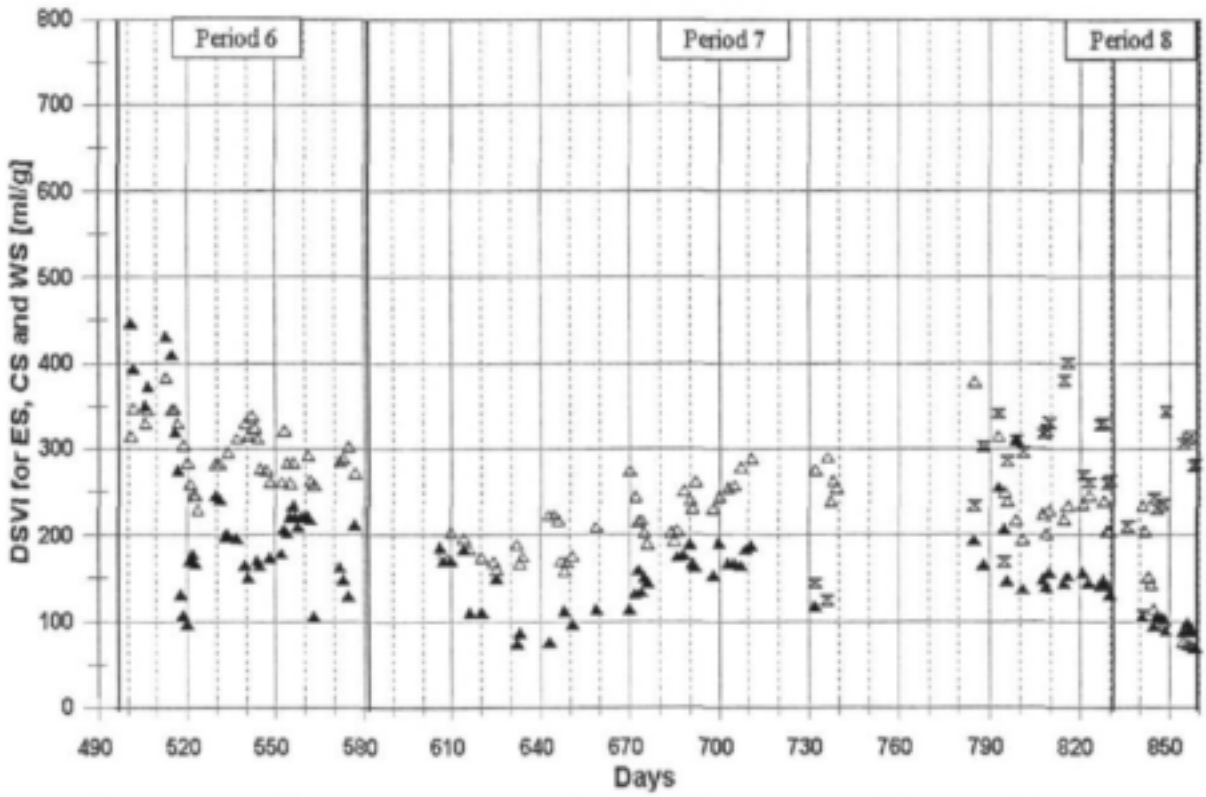
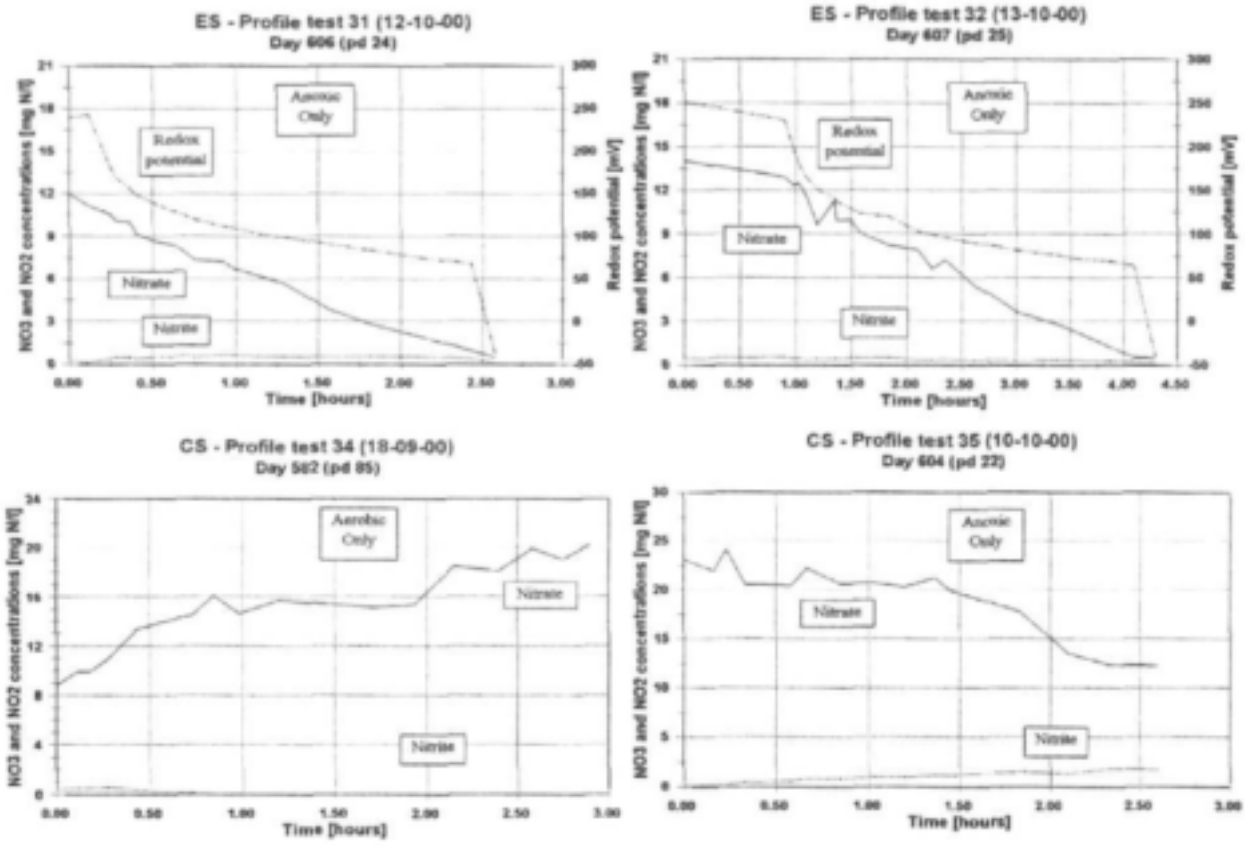


Fig 3.38: Diluted SVI for the Experimental System (ES, Δ), Control System (CS, ◢) and Warburton System (WS, ⨍) for Periods 6 to 8.

Period 7 started with the 7 days continuous aeration in ES from Days 583 to 590, which caused the DSVI to decrease rapidly from 300 to 180 mℓ/g as previously observed by Gabb *et al.* (1996) (Fig 3.38). On Day 590, a 3h aerobic period with anoxic period controlled by the redox potential at -45mV was reinstated on ES and on Day 607 the DSVI was 180 mℓ/g. Profile tests on CS on Days 606 and 607 showed complete denitrification during the anoxic period with nitrate and nitrite concentrations at the anoxic to aerobic transition < 0.5 mgN/ℓ (Figs 3.39 to 3.40). For CS, the profile tests on Days 604 and 605 showed incomplete denitrification during the anoxic period with high nitrate and nitrite concentrations (8 mgN/ℓ) at the anoxic to aerobic transition (Figs 3.41 and 3.42). Yet the settleability of CS was consistently lower than that of ES. For the AA filament bulking hypothesis, an better settleability is expected in an environment of low nitrate and nitrite concentrations at the anoxic to aerobic transition than in an environment of high



Figs 3.39 to 3.42: *Profile tests on the Experimental System (ES) on Days 606 and 607 and on the Control System on Days 581 and 604 (nitrate - solid line, nitrite - dotted line, redox - broken line).*

nitrate and nitrite concentrations at the anoxic to aerobic transition, yet the observed behaviour is exactly contrary to this. The cause for this deviation in observed behaviour from that expected in terms of the AA filament bulking hypothesis could not be determined.

As settleability continued to improve for CS and settling better than ES throughout the period, to further examine the AA filament bulking hypothesis, it was decisions to investigate the feasibility dosing azide (NaN_3). In Period 1 of the investigation, the second step of nitrification (oxidation of nitrite to nitrate) partially failed due to wastewater toxicity, and thus, in addition to nitrate, significant concentrations of nitrite were measured during both the anoxic and aerobic periods. Further it was noted that, even though the ES system operation ensured low nitrate and nitrite at the anoxic to aerobic transition, the presence of nitrite appeared to cause the DSVI to increase substantially due to AA filament proliferation. It was thus concluded that high nitrite concentrations during the aerobic and anoxic periods promote AA filament bulking. Since the AA filament bulking hypothesis was not specifically formulated to address this situation, it was decided to investigate this aspect more rigorously, by dosing azide to the ES. Azide inhibits the second step of nitrification, namely nitrite oxidation to nitrate. Thus, azide dosing would ensure no nitrate is generated in the ES, only nitrite (Fig 3.43).

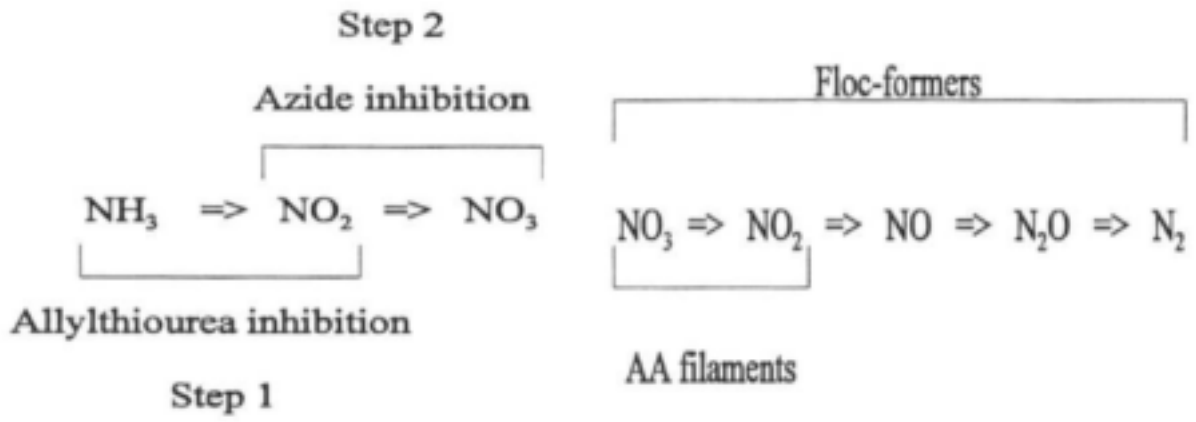


Fig 3.43a and b: *Chemical inhibition of the 2 nitrification steps mediated by ammonia and nitrite oxidizing organisms (Fig 4.43a, left) and the steps of denitrification by floc-formers and AA filaments as hypothesized by Casey et al. (1994, 1999) (Fig 4.43b, right).*

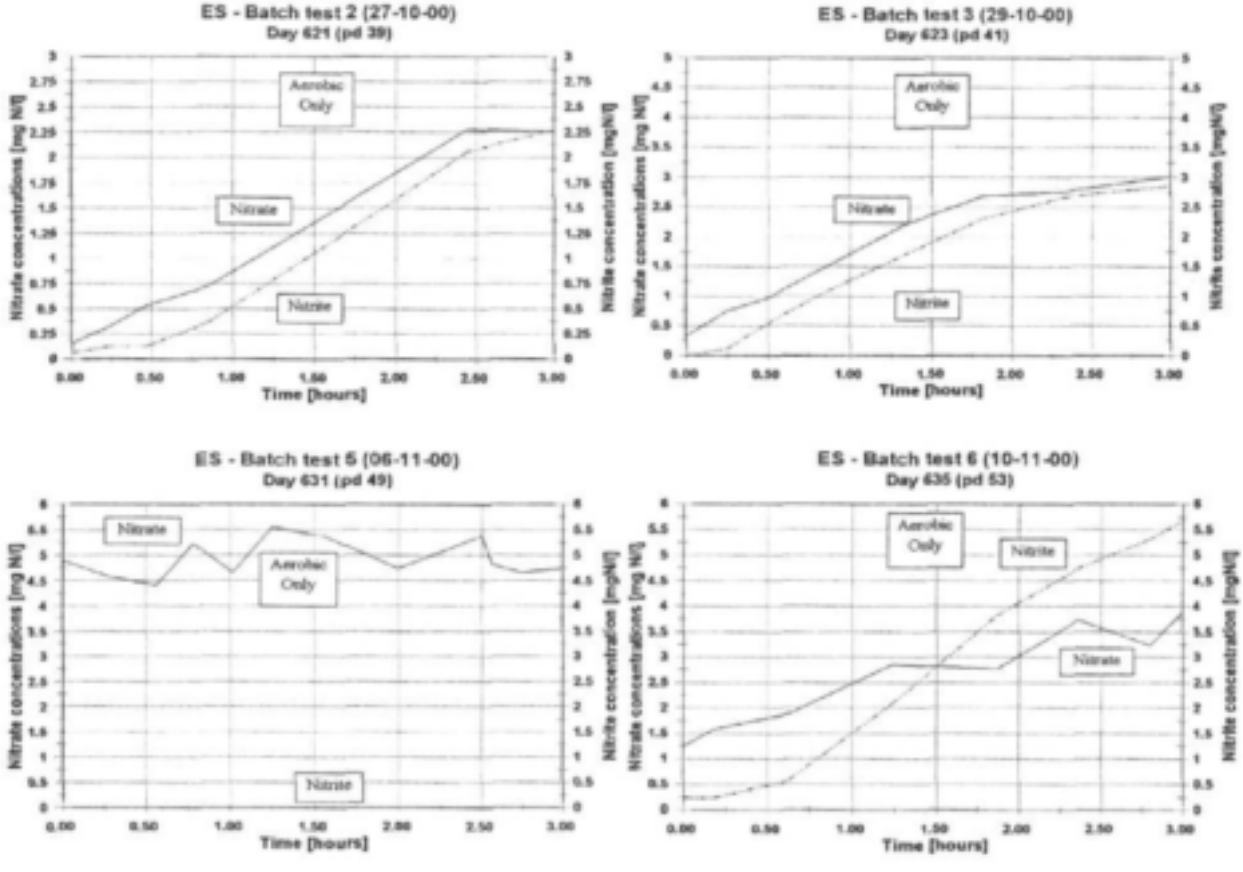
The expected behaviour with no nitrate but significant nitrite present can be interpreted in terms of the AA filament bulking hypothesis. In the hypothesis, it is proposed that the AA filaments reduce nitrate to nitrite only, whereas the floc- formers denitrify nitrate to nitrite and further to nitrogen gas (Fig 4.43b). Thus with no nitrate present in the system, the AA filaments would not be able to denitrify whereas the floc-formers will be able to denitrify nitrite to N_2 gas. This should place the floc-formers at a competitive advantage over the filaments in the anoxic phase. Further, if the system is operated such that no nitrite is present at the anoxic to aerobic transition, then the floc-formers will not be inhibited in their aerobic metabolism, and AA filaments will have a competitive advantage in the aerobic period also. Thus, the AA filament bulking hypothesis predicts that the system should not bulk by AA filaments. However, the observations during Period 1 indicate that these conditions may promote AA filament bulking rather than ameliorate it.

To determine the doses of azide, 7 batch tests of 1ℓ each were carried out on sludge harvested from ES to estimate the best azide dosing rate. These tests avoided the possibility of contamination of the systems if incorrectly dosed. Four batch tests were done on Days 621, 623, 631 and 635 with azide concentrations of 100, 50, 500 and 200 μM NaN_3/ℓ (Figs 3.44 to 3.47). From the batch tests it was concluded that the effect of azide dosing was not immediate. Dosing azide into the batch reactor was not sufficient to immediately inhibit the second step of nitrification. Nitrate concentrations in the batch tests were consistently higher than 3 mgN/ℓ and still increasing at the end of 3h aeration. Hence, it was decided not to dose azide to ES and CS. The two systems continued to be operated and monitored to observed their behaviour.

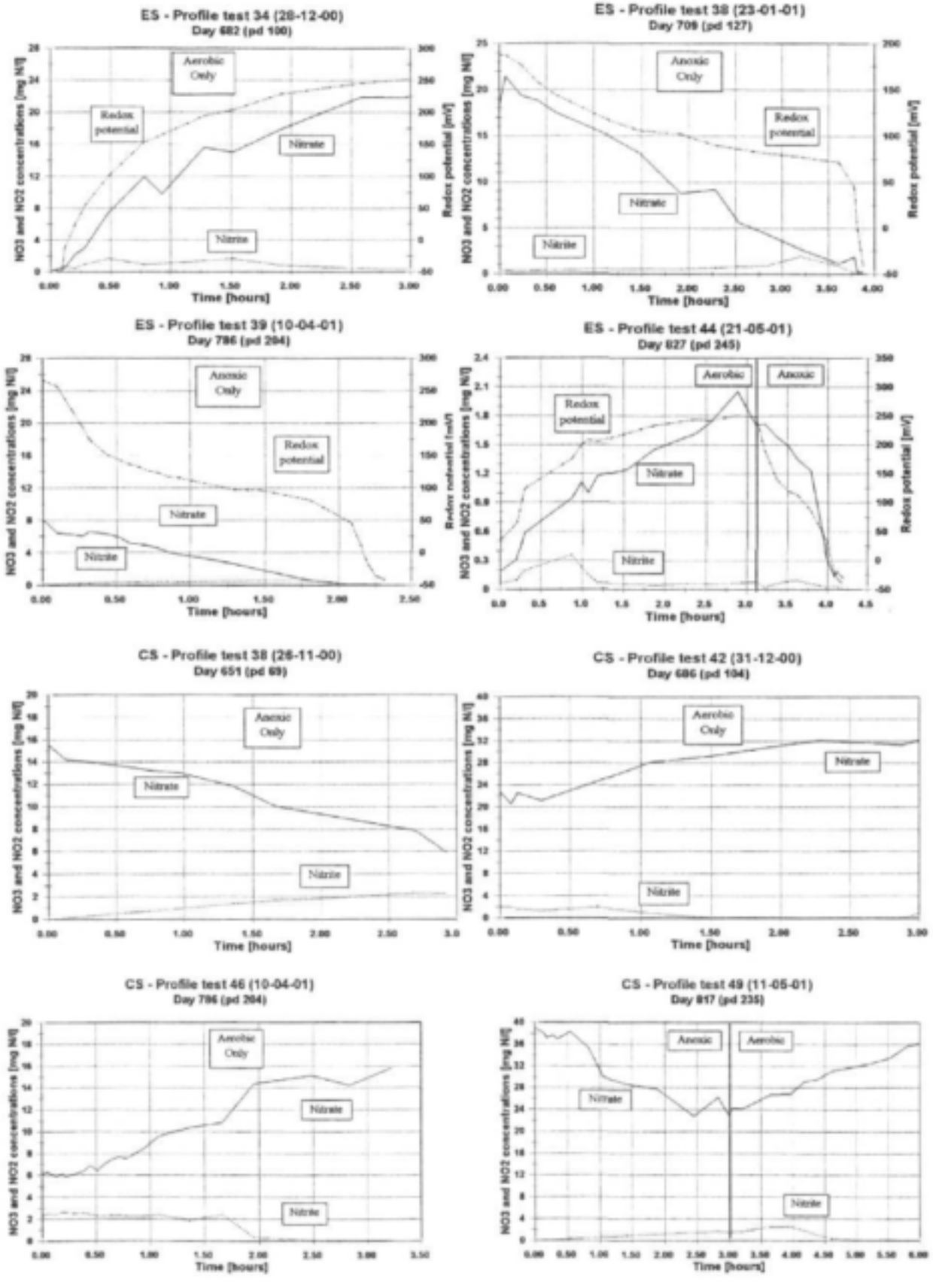
For ES, the profiles indicated that the system was still nitrifying and denitrifying well (Figs 3.48 to 3.51). The profiles show low nitrate and nitrite concentrations at the anoxic to aerobic transition (< 1mgNO_x/ℓ) indicating that the system is operating as required. The DSVI continued fluctuate between and 160 and 300 mℓ/g. In contrast the CS profile tests showed high nitrate and nitrite concentrations at the anoxic to aerobic transition, i.e. > 5mgNO_x-N/ℓ (Figs 3.52 to 3.55). With high nitrate and nitrite concentrations at anoxic to aerobic transition, the CS continued to have lower DSVIs than the ES (between 70 to 200 mℓ/g). Microscopic examination of the ES and CS sludges during this time indicated that *M. parvicella* was dominant. The observed behaviour therefore continued to be contrary to that expected from the AA filament

bulking hypothesis.

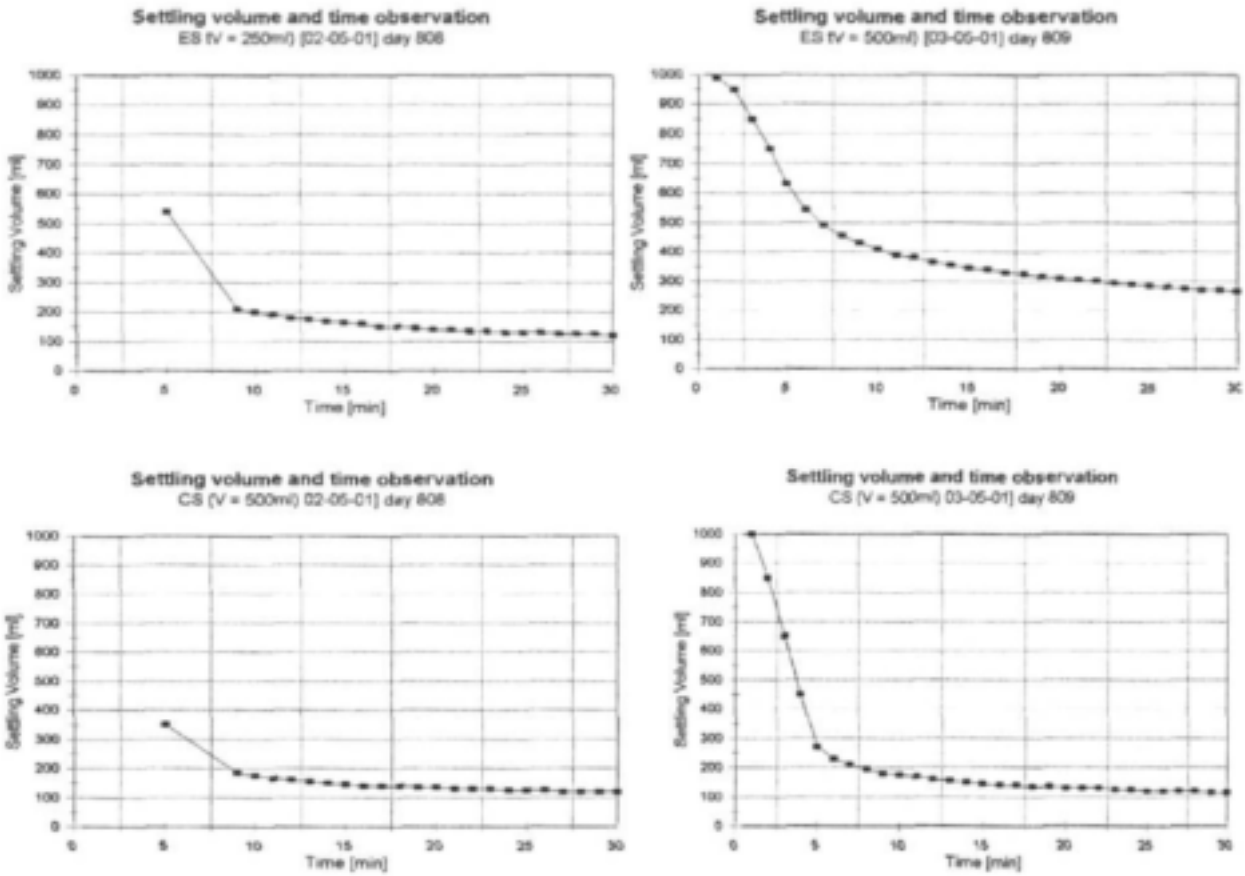
As a result of constantly higher DSVI in ES than CS and the inability to determine the cause for the observed deviation from the AA filament bulking hypothesis, it is decided to carry out some settling tests to examine the settling velocity of the sludges, in case different settling behaviour was manifest in the settling velocity rather than the DSVI. Graphs showing sludge settled volume versus time for the ES and CS are shown for Days 808 and 809 (Figs 3.56 to 3.59). In these tests, 500mℓ mixed liquor was mixed with 500mℓ tapwater in a 1ℓ measuring cylinder allowed to settle without stirring. The graphs indicate that settled volumes observed at the standard 30 minutes were consistent with expectation - the settled volumes were already low after 10 min settling and decreases gradually thereafter. If the settled volume at 30 min was still in the zone settling phase of the sludge, then the DSVI would be strongly influenced by time, but this was not the case.



Figs 3.44 to 3.47: *Batch tests with azide concentrations of 100 (top left), 50 (top right), 500 (bottom left and 200 (bottom right) µM/ℓon Days 621, 623, 631 and 6235 respectively to evaluate the effect of azide dose on the nitrite oxidizers (nitrate - solid line, nitrite - dotted line and redox - broken line).*



Figs 3.48 to 3.55: *Profile tests on the Experimental System (ES) on Days 682, 709, 786 and 827 and on Control System on Days 651, 686, 786 and 817(nitrate- solid line, nitrite - dotted line, redox- broken line).*



Figs 3.57 to 3.60: *Settled volume versus time plots for the Experimental (ES) (top) and Control (CS) Systems (bottom) sludges on Days 808 (left) and 809 (right).*

Because the behaviour of ES and CS was contrary to expectation from the AA filament bulking hypothesis, it was decided to operate the third intermittently aerated (IAND) system identical to those of Gabb *et al.* (1989, 1996) and Warburton *et al.* (1991), to see this type of system would behave similarly as they observed in the past. This third system was named Warburton system (WS) and was started on Day 724 was identical to CS and ES except for the aeration cycle (see Table 3.1). WS had a one minute air-on time, during which the DO increased from 0 to 2-3 mg/ℓ. After one minute aeration, the airflow was stopped causing the DO to decrease to zero over the next 2 to3 min. After 9 min of air-off time, the 1 min air-on time started again. The aerobic time was therefore 3 to 4 minutes out of a cycle time of 10 min, yielding an aerobic period of ~l/3rd hence the aerobic and anoxic mass fractions were about 34% and 66% respectively. Both Gabb *et al.* (1989, 1996) and Warburton *et al.* (1991) found these conditions strongly promoted AA filament bulking, in particular by *M. parvicella*. The nitrate and nitrite profile over a 10 minute aeration cycle on Day 786, 204 days after the beginning of Period 7 and 62 days after starting the WS, (Fig 3.60) showed (i) no nitrite build up (<0.2 mgN/ℓ), (ii) increasing and decreasing nitrate concentration during the aerobic anoxic periods, (iii) good denitrification and (iv) low effluent nitrate concentration (16 - 22 mgN/ℓ). This confirmed that the WS was operating as expected and in conformity with that previously observed. The nitrate concentration at the anoxic aerobic transition was high (> 15 mgN/ℓ) and therefore bulking was expected, which was in fact observed. The WS DSVI was 234 mℓ/g indicating a poor settling sludge. Also, microscopic examination of sludge indicated that this bulking was due to AA filaments with *M. parvicella* dominant. The progress of batch settling in the DSVI test on two consecutive days

(Days 808 and 809) are similar to those observed on ES and CS as expected (Fig 3.61 and 3.62).

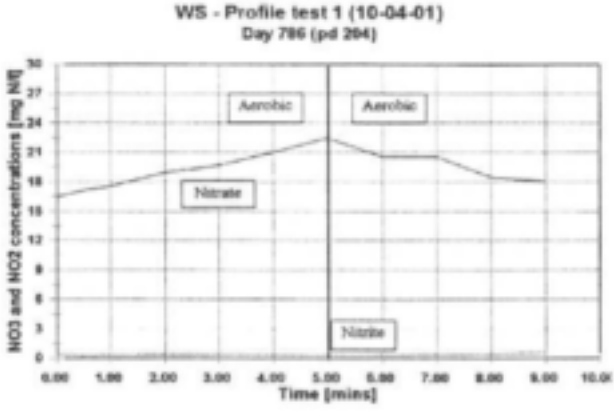
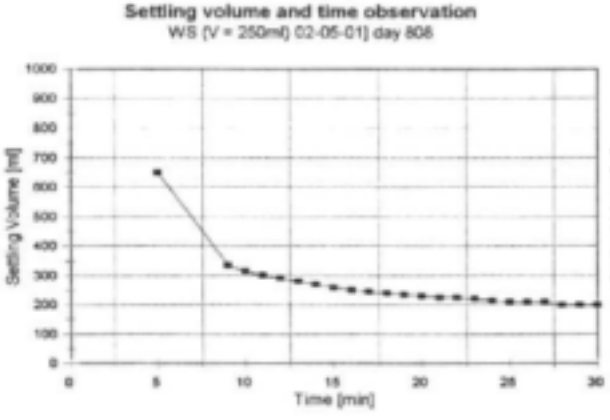
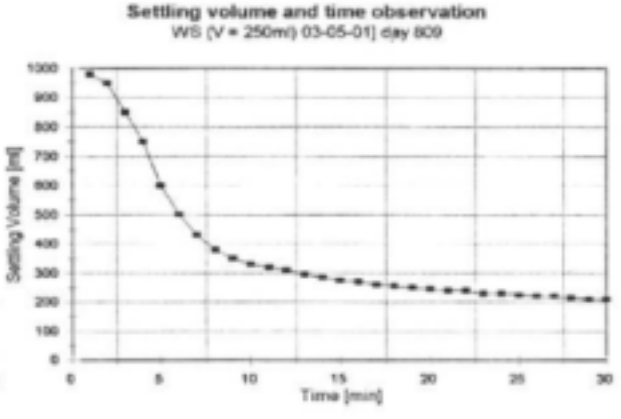


Fig 3.60 (left): *Profile test on the Warburton System (WS) on Day 786, 62 days after it was strarted (nitrate - solid line, nitrite - dotted line).*

Figs 3.61 and 3.61: *Settled volume versus time for the sludge from the Warburton system on Days 808 (bottom left) and 809 (bottom right).*





With the same wastewater, laboratory operation and experimental system set-up and similar operation of WS, CS and ES, it was concluded that the AA filament bulking hypothesis was deficient in explaining the ES and CS system behaviours. The confirm this, the same changes that Gabb *et al.* (1989, 1996), Warburton *et al.* (1991) and Ketley *et al.* (1991) made to their systems were made to the ES and CS systems to see if these changes still stimulated the same behaviour is observed earlier. Hence, the ES and CS were changed to fully aerobic and fully anoxic respectively. Period 7 therefore ended with the ES being operated at maximum aerobic mass fraction (100%) by constant aeration and CS was operated at minimum aerobic mass fraction (0%) i.e. no aeration but dosing nitrate for electron acceptor. The WS was left unchanged to serve as a control system demonstrating continued bulking, because it is expected from the earlier research that AA filament bulking would ceased in the ES and CS.

3.3.8 System performance - Period 8 (Day 832 to Day 859)

Period 8 continued from Period 7, but with changed aerobic mass fractions. For the ES, the aerobic time was changed from 3 hours to 24 hours (100% aerobic), and to ensure the system would be fully aerobic, the determining factor for the end of the ES anoxic time was set at the 250 mV redox potential. For the CS, the aerobic mass fraction was changed to 0%, with the no aerobic time, the system was kept at fully anoxic condition by dosing nitrate as electro acceptor. For the WS, the system operating parameters were kept the same as the previous period, i.e. intermittently aerated with a 10 min aeration cycle and ~30% aerobic mass fraction. The changes made to ES and CS were to evaluate the effects of aeration cycle time on the performance of the systems.

Period 8 extended over about two sludge ages (~30 days) during which time the settleability for both the ES and CS decreased dramatically (Fig. 3.63). For the ES system, the settleability decreased from a DSVI of *ca.* 205 mℓ/g at the last day of the previous period (Day 832) to 70 mℓ/g at the end of the this period (Day 859). For the CS system, the settleability also improved from the DSVI of *ca.* 131 mℓ/g at the last day of the previous period to the DSVI of 70 mℓ/g at the end of the this period. The settleability in WS remained poor with the DSVI increasing, from 265 mℓ/g (Day 830) to 283 mℓ/g (Day 859).

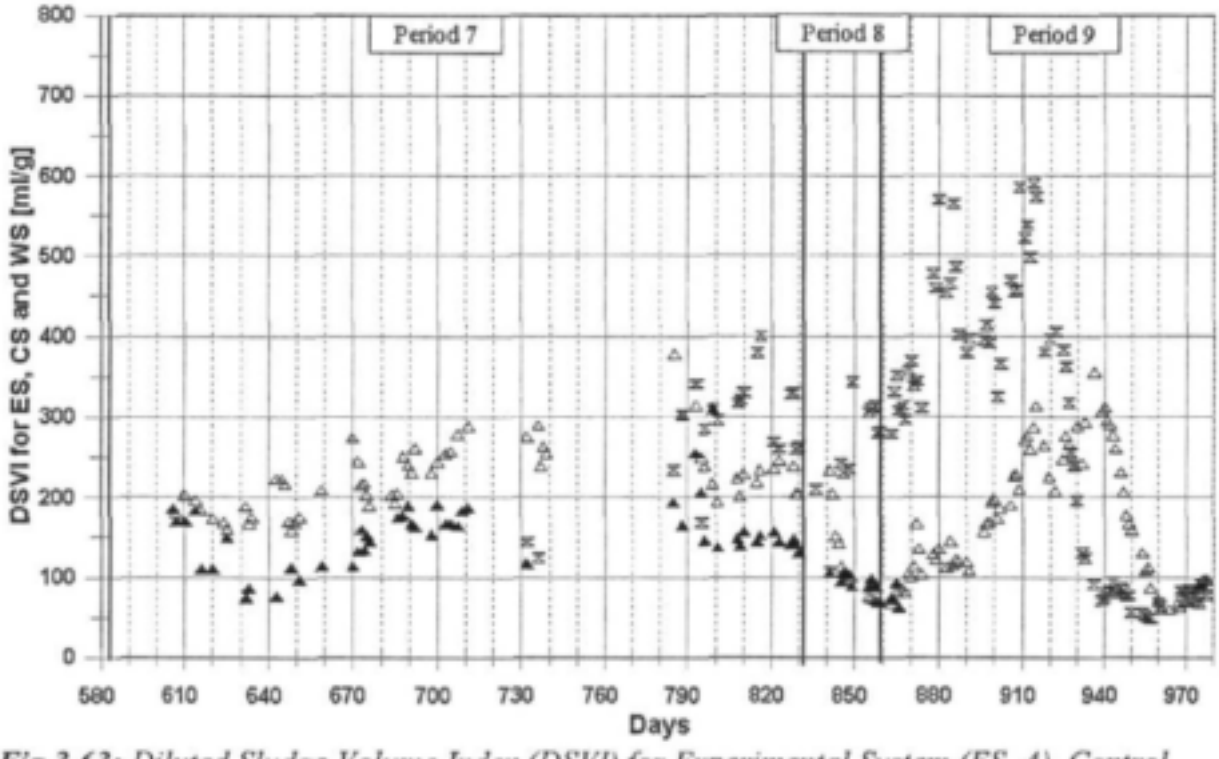


Fig 3.63: Diluted Sludge Volume Index (DSVI) for Experimental System (ES, Δ), Control System (CS, ▲) and Warburton System (WS, ⛵) from Periods 7 to 9.

The WS system was monitored closely by means of profile tests, to check that the nitrate and nitrite concentrations at the anoxic to aerobic transition were high. Figure 3.64 shows a profile test over two 10 minutes aeration cycles. The profile test confirmed the system was operating as required, i.e. no nitrite build up and high nitrate concentration at the anoxic to aerobic transition (> 15 mgNO_3-N/ℓ). The DSVI remained high (200 to 300 mℓ/g). As earlier, microscopic

examination of the sludge indicated that this bulking was due to AA filaments, with *M. parvicella* and type 1851 dominant.

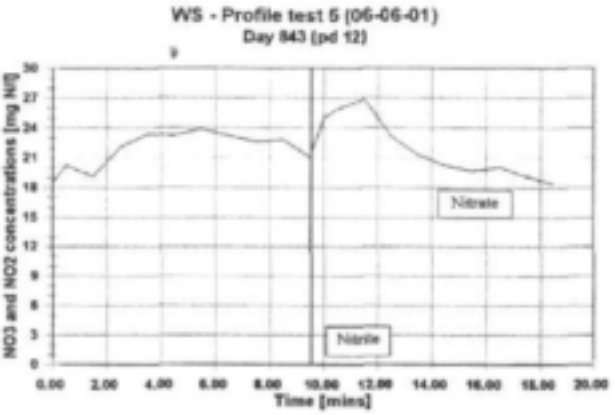


Fig 3.64: *Profile test on Warburton System (WS) on Day 843 over two 10 min cycles (nitrate - solid line, nitrite - dotted line).*

The observations monitored in ES and CS conformed to earlier observations by Gabb *et al.* (1989, 1996), Warburton *et al.* (1991) and Ketley *et al.* (1991) that fully aerobic and fully anoxic conditions ameliorate AA filament bulking, part of the evidence on which the AA filament bulking hypothesis is based. The ES, CS and WS systems therefore conformed to the pattern of the earlier observations by Casey *et al.* (1994) on artificial wastewater IAND systems shown in Fig 3.65, viz. good settling sludge (DSVI ~ 100 mℓ/g) at very low (< 10 %) and high (> 70 %) aerobic mass fraction and poor sludge settleability in between (10 - 70 %) with the poorest settleability (DSVI ~ 800 mℓ/g) between 30 and 40 % aerobic mass fraction. Subsequently Casey and Alexander (2001) amplified this plot with full scale plant data. In this work, Casey attempted to determine the NO_x concentration at the transition from anoxic to aerobic conditions of full scale BNR (N &P) plants to see if this affected the DSVI in the plants. After a year, they unfortunately had to abandon this because of the inconsistency in the nitrate samples - all samples at all plants at all times showed zero NO_x at the anoxic/aerobic transition. The reason turned out to be the failure to filter the samples by the plant operators immediately after taking the samples with the result that denitrification continued in the sample bottles. However, they did obtain an annual average DSVI from the weekly DSVI tests at the full scale plants and plotted these versus aerobic mass fraction in Fig 3,65. The full scale data show the same trend with decreasing aerobic mass fraction in the wastewater IAND systems except the DSVIs are much lower, from 50 mℓ/g at 76 % aerobic mass fraction (Bushkoppie) to 180 mℓ/g at 45 % aerobic mass fraction (Daspoort). On the other end of the scale at low aerobic mass fractions are plotted the laboratory scale data from external nitrification (EN) BNR (N & P) activated sludge systems operated in the UCT laboratory over the past 4 years (Moodley *et al.*, 1999; Sotemann *et al.*, 2000; Hu *et al.*, 2000, 2001). In ENBNRAS systems short sludge age (8 - 10 days) and very low aerobic mass fractions (15-30%) can be expected because nitrification is not accomplished in the suspended sludge but in an external fixed

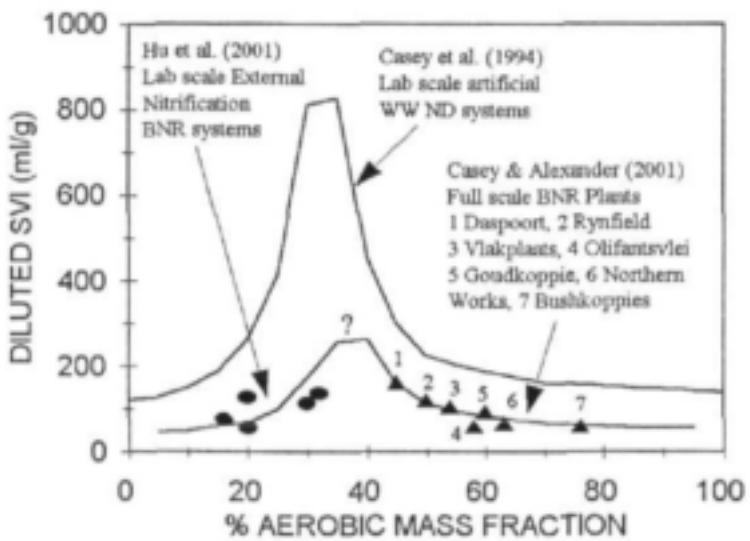


Fig 3.65: *DSVI versus aerobic mass fraction plots for lab scale IAND system fed artificial wastewater (Casey et al., 1994) and external nitrification BNRAS systems fed real wastewater (●, Hu et al., 2001) and full-scale BNR plants (▲, Casey and Alexander, 2001).*

aerobic system. These ENBNRAS systems consistently produce good settling sludges and the average (of 18 months or more) DSVI versus aerobic mass fraction of 5 different ENBNRAS systems are shown in Fig 3.65. The trend in these results are also similar to Casey's artificial wastewater IAND systems (Casey *et al.*, 1994) except that the DSVIs of the real wastewater systems are considerably lower than those of the artificial wastewater systems.

Noting the many factors that influence the sludge settleability and AA filament proliferation, such as aerobic mass fraction as in Fig 3.65, but also temperature, sludge reactor intermittently aerated systems (like Orbal and Carousel). For the same system conditions and wastewater feed, IAND systems bulk worse than other ND systems like MLE. So common factor that could possibly affect AA filament proliferation, in particular with *M. parvicella*, was sought. Increasing aerobic mass fractions and decreasing temperature put pressure to the nitrifiers and result in increasing residual Free and Saline Ammonia (FSA) concentrations (1 - 5 mgN/ℓ). Also IAND systems cannot nitrify completely (effluent exits the system during the anoxic period) and generally have higher effluent FSA concentration than equivalent MLE systems. Furthermore Kruit *et al.* (2001), from an in-depth performance evaluation of 4 Dutch full scale BNR plants, noted that increasing residual FSA concentration results in increasing DSVIs caused by *M. parvicella*. Slijkhuis (1983) and Slijkhuis and Deinema (1983) showed that *M. parvicella* requires long chain fatty acid (C_{14} - C_{18}), low DO and ammonia for growth. It is therefore possible that the residual FSA is providing the N source for *M. parvicella* growth. In fully aerobic systems, where nitrification is rapid, the FSA reaches very low concentration quickly before significant growth of the slow growing *M. parvicella* can take place, and under fully anoxic conditions *M. parvicella* cannot compete with the facultative heterotrophs because it can reduce nitrate to nitrate only, not to nitrogen gas like the facultative heterotrophs.

Following the above reasoning, possible changes were considered that would increase the residual FSA concentrations under circumstances that would otherwise not result in high DSVIs i.e. fully aerobic conditions. Hence it was decided to dose Allylthiourea to fully aerobic systems. This would inhibit the first step in nitrification (Fig 3.43a) and cause high residual FSA and then see if the DSVI increases from a low value (70 mℓ/g) to a high value and from a high value (300 mℓ/g) to a higher value. Consequently the ES (with low DSVI) and WS (with high DSVI) were set to fully aerobic conditions (continuous aeration controlled with the DO box of Randal et al., 1991) while dosing both systems with Allylthiourea to see if this would increase the DSVI. At the same time, the CS was continued under fully anoxic conditions for longer to see if the DSVI would decrease below 100 mℓ/g. This it did and by Day 865 (5 days into period 9) it reached 55 mℓ/g. So Period 8 ended on Day 859 when fully aerobic conditions were applied to ES and WS and allylthiourea dosed to both to inhibit nitrification and increase the residual FSA concentration.

3.3.9 System performance - Period 9 (Day 860 to Day 947)

Period 9 started on Day 860 by imposing fully aerobic conditions on ES and WS and dosing allylthiourea to both to inhibit the first step in nitrification (Fig 3.43a) and see if the DSVI increases due to *M. parvicella* proliferation. At this time the DSVIs of ES and WS were low (~60 mℓ/g) and high (~220 mℓ/g) respectively with ES having been fully aerobic during Period 8 and WS anoxic-aerobic (Fig 3.66)

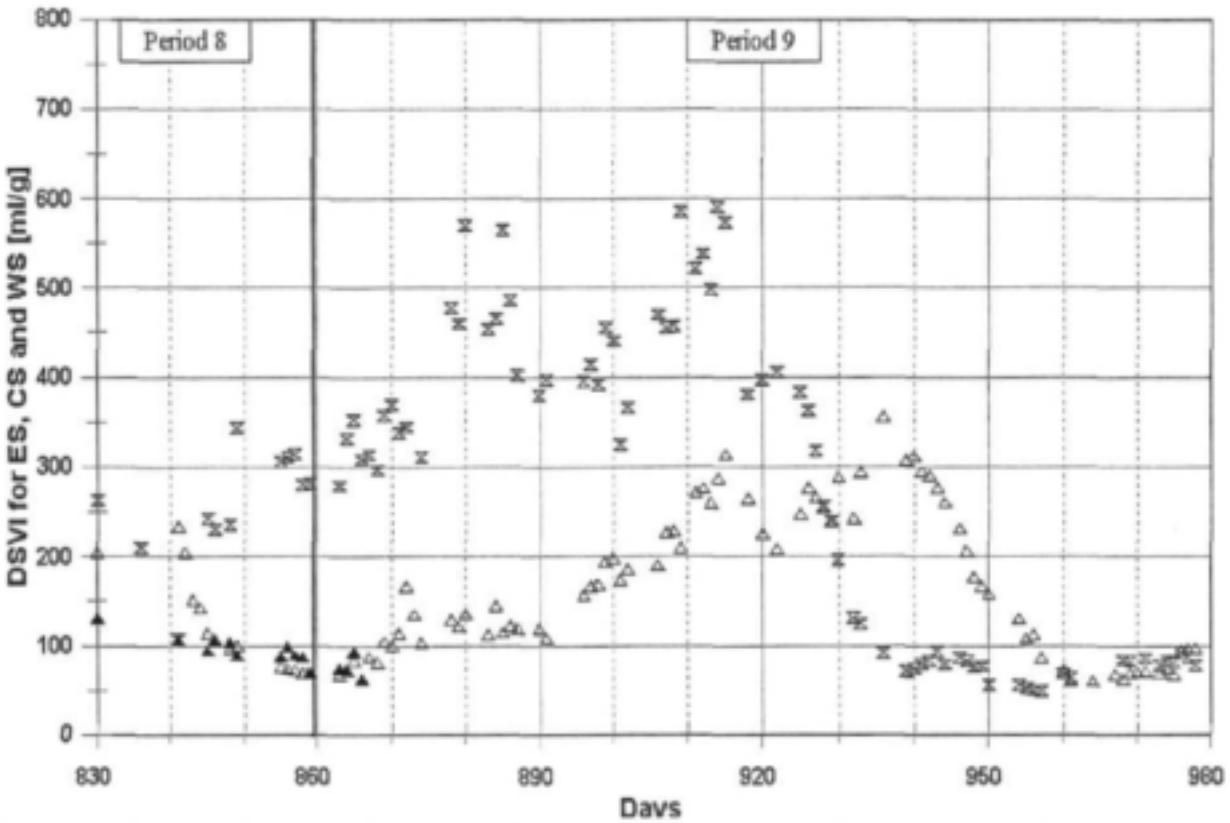


Fig 3.66: Diluted Sludge Volume Index (DSVI) for Experimental System (ES, Δ), Control System (CS, ◢) and Warburton System (WS, ◪) from Periods 8 to 9.

The initial dosage of allylthiourea (ATU) to ES and WS was 10 mg/ℓ $_{inf}$ on Day 860. This dosage was made every three to four days, depending on the recovery in nitrification in the systems. The systems' effluent FSA concentrations were closely monitored daily to see whether the ammonia oxidizers were in fact inhibited by the dosed ATU. It was found that the effluent FSA concentration declined gradually due to the adaptation of the ammonia oxidizers to the dosed ATU, which indicated that the ATU dosage of 10 mg/ℓ was having a diminishing. Consequently the ATU dosage was increased to prevent the residual FSA from become to low (< 10 mgN/ℓ). The ATU dosage had to be increased to as much as 80 mg/ℓ $_{inf}$ on Day 912 to ensure no reduction in FSA and ensure a high effluent FSA concentration.

The settleability for ES deteriorated upon the dosing ATU, the DSVI increased from 67 mℓ/g on Day 866 to 356 mℓ/g on Day 936. A similar effect was observed for WS, in which the settleability continued to be poor with DSVI fluctuating between *ca.* 280 mℓ/g on Day 863 and 591 mℓ/g on Day 914 (Fig 3.66).

On Day 896 ATU dosing to WS (with the very high DSVI) was stopped to restore complete

nitrification and see if this would stimulate the improvement in settleability as usually observed under fully aerobic conditions. Dosing ATU to ES was continued to allow the DSVI to increase further. On Day 914, it was evident that the dosage of ATU had a direct influence on settleability in ES, hence ATU dosing was stopped to see if in this system also the improvement in settleability would take place usually observed under fully aerobic conditions. Microscopic examination of the ES and WS sludges on Days 890 and 921 (whole both were being dosed) indicated that high DSVIs were due to AA filaments, with *M. parvicella* dominant.

For WS, after the cessation of ATU dosing on Day 896, the system took 10 days to recover to complete nitrification indicated by low effluent FSA concentration(< 1 mgN/ℓ). Within the period of recovery in nitrification, the settleability of the WS system continued to deteriorate, with the DSVI increasing from 415 mℓ/g on Day 897 to 455 mℓ/g on Day 907. After cessation of ATU dosing, the settleability continued to deteriorate to as high as 591 mℓ/g on Day 914, but from Day 915, the settleability started to improve reaching 83 mℓ/g on Day 947 (Fig 3.66).

The ES behaved similarly to WS after cessation of ATU dosing on Day 914. It took 6 days to achieve complete nitrification and during this period of the recovery and for 20 days thereafter, the settleability continued to deteriorate with the DSVI increasing from 314 mℓ/g on Day 915 to 356 mℓ/g on Day 935. From Day 935 the settleability improved to reach a DSVI of 60 mℓ/g on Day 960 (Fig 3.66). Thereafter for the next 20 days to Day 980 when the investigation was terminated, the DSVIs remained low (< 60 mℓ/g) in both systems as so often observed in the past in fully aerobic systems with complete nitrification.

3.4 CONCLUSIONS

In the last 100 days of this 980 day investigation, it was found that the aerobic reactor free and saline ammonia (FSA) concentration had the strongest controllable influence yet observed on the AA (low F/M) filament proliferation, in particular *M. parvicella*. If the FSA was high (by inhibiting the nitrifiers) under completely aerobic conditions (which in the past invariably cured AA filament bulking, Ekama *et al.*, 1999), bulking (i) continued if the DSVI was high (>250 mℓ/g) and (ii) was stimulated if the DSVI was low (<60 mℓ/g). Removal of the nitrifier inhibitor restored complete nitrification and after a 20 to 28 day delay, cured the bulking and reduced the DSVI from 500 and 300 mℓ/g respectively to 60 mℓ/g in 30 days. The hypothesized explanation for this observation is that *M. parvicella* requires FSA as a N source for growth and cannot use nitrate or nitrite as an alternative. In terms of this hypothesis, if nitrification is rapid and complete, then FSA is not available and the slow growing *M. parvicella* are limited in their growth. In contrast, if nitrification is slow or incomplete, then the *M. parvicella* have FSA available as an N source for growth and so proliferate.

The above hypothesis for *M. parvicella* growth explains several observations on full-scale BNR plants: (i) the seasonal proliferation of *M. parvicella* during winter, and (ii) the increasing DSVI with decreasing aerobic mass fraction (Fig 3.65). Both temperature and reduced aerobic mass fraction place pressure on the nitrifiers resulting in increasing aerobic reactor FSA concentration which favours *M. parvicella* growth. Being capable of nitrate reduction (Tandoi *et al.*, 1998) it is recognised that *M. parvicella* could grow in the anoxic reactor since both nitrate and ammonia are available. However, *M. parvicella* is slow growing and obtains only a limited amount of energy under anoxic conditions, since it can only reduce NO_3 to NO_2. Thus, the anoxic growth of *M. parvicella* is probably too small to significantly affect the DSVI.

The above hypothesis for *M. parvicella* growth does not overturn the AA filament bulking hypothesis of Casey *et al.* (1999) - the association between the anoxic to aerobic transition nitrate concentration and the DSVI has been observed too frequently to discard it. Both hypotheses are regarded as relevant, and future research should focus on how elements of these two hypotheses superimpose on the conditions in different BNR systems.

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4 GLOSSARY - DEFINITION OF TERMS AND ABBREVIATIONS

AA	Anoxic-aerobic; new group name for most of the low F/M filamentous organisms
AHVSS	Active heterotrophic volatile suspended solids i.e. that part of the Active VSS (AVSS) that is ordinary heterotrophic organisms that do not take part in the BEPR.
ASPD	Activated sludge population dynamics specialist group of the IAWQ
ATV	Abwassertechnische Vereinigung; the German wastewater association.
AVSS	Active Volatile Suspended Solids. The volatile suspended solids (VSS) comprise active organisms and inert organic mass. The active organism mass is the live biological mass which performs the biological reactions; the inert mass originates from two sources (i) from inert organic material in the influent and (ii) endogenous residue. The active fraction of the VSS is a function of the sludge age of the system and sewage characteristics. It is an empirical estimate that has found acceptability because of the consistency it brings to kinetic rates observed in activated sludge systems, e.g. based on active mass the specific endogenous mass/respiration rate and specific denitrification rates are constant with sludge age from 3 to 72 days. Because of this consistency, the readily biodegradable COD (RBCOD) uptake rates also are reduced to specific rates with respect to AVSS so that rates in different sludge age systems can be compared. For details on calculation of the AVSS, see Marais Ekama, 1976 and Ekama, Dold and Marais, 1986. In fully aerobic and anoxic-aerobic systems the AVSS comprises (aside from the autotrophic nitrifiers) only ordinary heterotrophs organisms; in BEPR plants the AVSS includes the polyphosphate accumulating organisms also, and then care needs to be taken that the different kinetic reactions are ascribed to the particular organism group mediating it.
BEPR	Biological excess phosphorus removal
BNR	Biological nutrient removal
COD	Chemical oxygen demand
DO	Dissolved oxygen
DSVI	Diluted sludge volume index, a modified SVI sludge settleability test [see Ekama and Marais (1984). Two improved sludge settleability parameters, *IMIESA*, **9**, 6, 20-25 for method]
d	day
et al.	and others
F/M	Food to Microorganism ratio
FRD	Foundation for Research Development
g	gram
H.hydrossis	*Haliscomenobacter hydrossis*, one of the filamentous organisms in the AA (low F/M) group
h	hour
IAWQ	International Association on Water Quality
K_s	Half saturation coefficient in the Monod equation
K_{2T}, K_{220}	Specific denitrification rate constant in $mgNO_3$-N/(mgAVSS.d) for N removal systems
K'_{2T}, K'_{220}	Specific denitrification rate constant in $mgNO_3$-N/(mgAHVSS.d) for N&P removal systems
low F/M	low food to micro-organism ratio; equivalent to low load factor, or low

	loading rate or long sludge age
ℓ	litre, the unit measure for volume
mℓ	Millilitre
MUCT	Modified University of Cape Town system for biological removal of nitrogen and phosphorus
M. parvicella	*Microthrix parvicella*; one of the most ubiquitous and problematic filamentous organisms in the AA (low F/M) group
MLVSS	mixed liquor volatile suspended solids; same as VSS, the organic part of the suspended solids in activated sludge plants
MLSS	mixed liquor suspended solids; the organic and inorganic suspended solids in activated sludge plants, also referred to as Total Suspended Solids
N	nitrogen; all nitrogen concentrations i.e. nitrate, nitrite or TKN are expressed as mgN/ℓ
NO_3-N; NO_2-N	nitrate and nitrite respectively as N
NO; N_2O	nitric oxide and nitrous oxide respectively, the two gaseous denitrification intermediates between NO_2 and N_2
NUR	Nitrate utilization rate
N_2	Dinitrogen gas, the end product of denitrification
N & P	nitrogen and phosphorus; applied to activated sludge plants incorporating simultaneous biological N and P removal
OUR	oxygen utilization rate; mass oxygen utilized per unit reactor volume per unit time, or per unit VSS mass per unit time, e.g. mgO/(gVSS.h)
P	phosphorus; all phosphorus concentrations are total phosphorus concentrations and expressed as mgP/ℓ
RBCOD	readily biodegradable COD component of the influent COD
RSA	Republic of South Africa
SBCOD	slowly biodegradable COD component of the influent COD
STR	Scientific and Technical Report series of the IAWQ
TFL	Total filament length
TKN	Total Kjeldahl nitrogen
TKN/COD	ratio of the influent TKN and COD concentrations - a useful term comparing the quantity of nitrate that is going to be generated by nitrification from the influent TKN (called nitrification capacity) with the quantity of organic material (influent COD) available for denitrification
TON	Total oxidized nitrogen i.e. nitrate + nitrite concentrations
UCT	University of Cape Town activated sludge system for biological removal of nitrogen and phosphorus
USA	United States of America
WRC	Water Research Commission, a water research co-ordination and funding agency in South Africa. Executive Director Mr P E Odendaal, P O Box 824, Pretoria, 0001
WWTP	Wastewater treatment plant
0092, 0041 0914, 0803 1851, 0675	Seven different filamentous organism types of activated sludge, the first 6 common in biological N&P and N removal plants and therefore sorting into the AA (low F/M) group, the last arising 021N mainly with septic wastewaters (i.e. high sulphides) as happens occasionally in the UCT Water Research Laboratory with long sewage storage periods (> 3 weeks) or refrigeration breakdowns.
µm	micrometers (10^{-6} meters)
μ_H	Maximum specific growth rate of heterotrophs in Mond equation
θ	Arrhenius temperature coefficient.

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Causes and Control of Low F/M Filament bulking in Nutrient Removal Activated Sludge Systems

GA Ekama, MC Wentzel, MT Lakay, RA Pilson, HKO Mellin and TG Casey

Two surveys of filamentous bulking in South African biological nitrogen (N) and nitrogen and phosphorus (N&P) removal plants indicated that bulking caused by low food/micro-organism, (F/M) filaments, is the major solid/liquid separation problem through poor sludge settleability in these plants. The main aim of this research was to establish the cause of low F/M bulking and to develop strategies for the control of the low F/M bulking in N and N&P removal plants. The research has shown that the promoted specific control method of selectors which stimulate removal of influent readily biodegradable COD in anaerobic, anoxic or aerobic selectors by metabolic or kinetic selection is not successful for controlling bulking in biological N&P removal plants. It was found that the conditions that stimulate biological N removal are conducive to bulking on nutrient removal plants, i.e. if denitrification is not complete at the time conditions switch from anoxic to aerobic, then proliferation of low F/M (renamed anoxic-aerobic (A/A) to accurately describe the conditions under which they proliferate) filaments takes place.

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