UNIVERSITY OF CAPE TOWN
Department of Civil Engineering

FINAL REPORT
to the
Water Research Commission
on the contract

CONSOLIDATION OF ACTIVATED SLUDGE RESEARCH II

(January 1991 - December 1994)

by

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EXECUTIVE SUMMARY

1. AIMS AND OBJECTIVES

In 1991 a 3 year contract "Consolidation of activated sludge research II" (K5/356) was set up between the Water Research Commission (WRC) and the University of Cape Town (UCT). Subsequently the contract has been extended by one year without additional funds. For the contract, the following principal aims were identified.

- Complete the calibration of, and the user manual for the nitrification denitrification (ND) kinetic simulation model.

- Continue the development of a kinetic simulation model for nitrification denitrification biological excess phosphorus removal (NDBEPR) systems.

- Revise the 1984 NDBEPR system design manual to incorporate the research findings on ND and BEPR since 1984.

- Write papers and reports on experimental work completed under preceding contracts, in particular on aspects relating to NDBEPR systems.

- Conduct experimental work to evaluate the effect of incorporation of fixed media in the aerobic zone in NDBEPR systems, with the view to reducing the system sludge age yet maintaining virtually complete nitrification.

At the Steering Committee meeting in June 1992, the members noted that the last aim, namely incorporation of fixed media in NDBEPR systems, might lead to duplication of effort since Professor Pretorius at the University of Pretoria is conducting research in this area. Accordingly, the aim was modified to:

- Develop and refine methods for characterization of municipal wastewaters.

Progress in achieving these aims is summarized in this report. Also, progress in four additional areas of research is reported:

- Dynamic simulation of secondary settling tanks and evaluation of secondary settling tank design procedures based on various sludge settleability parameters.

- Treatment of landfill leachates.

- The effect of alternative detergent builders on NDBEPR systems.

- The effect of thermophilic heat treatment on the anaerobic digestibility of sewage sludge.

Research in the above four areas does not fall directly under the ambit of the current consolidation contract, but progress is reported to keep the WRC informed of the work being

done in these areas.

This final report is a summary of the research undertaken to address the aims identified above. As specific research topics have been completed, detailed reports and papers have been prepared and published. These detailed reports and papers have been referenced at the appropriate place in this final report and are also listed in the section "**PUBLICATIONS DURING CONTRACT PERIOD**". For greater detail on the research, the reader is referred to these publications.

2. NITRIFICATION DENITRIFICATION (ND) KINETIC SIMULATION MODEL

With the increased complexity in the activated sludge system, it is no longer possible to make a reliable quantitative, or sometimes even qualitative prediction as to the effluent quality to be expected from a design, or to assess the effect of a system or operational modification, without some model that simulates the system behaviour accurately. Over the past 20 years, the Water Research Group at UCT has made concerted efforts to model the behaviour of the activated sludge system. The result of this research endeavour was a powerful kinetic model that gives reliable description of the nitrification and nitrification/denitrification (ND) system response over wide ranges of system configurations wastewater characteristics and operational parameters.

To make the kinetic model more widely available, computer programs of the model were written that are (1) suitable for PCs, (2) "user friendly", (3) flexible and (4) provide rapid solutions. During the preceding WRC consolidation contract (K5/251), the computerized models were extensively calibrated, and a user manual written and tabled at the final steering Committee meeting.

The Water Research Group at UCT received considerable feedback from Steering Committee Members who identified errors and omissions in both the computer programs and user manual. During the current consolidation contract, these problems in the computer programs and the user manual have been corrected. The user manual has been printed and is being distributed by WRC together with the computer programs (Dold *et al.*, 1991). This completes this section of the work.

3. NITRIFICATION DENITRIFICATION BIOLOGICAL EXCESS PHOSPHORUS REMOVAL (NDBEPR) KINETIC SIMULATION MODEL

Following the development of the kinetic model for nitrification and nitrification/denitrification (ND) systems, in 1983 the UCT group set the objective to develop a general kinetic model that describes the dynamic behaviour of an activated sludge system which incorporates also biological excess phosphorus removal (BEPR). It was recognized that such a model would require inclusion of the kinetic and stoichiometric behaviour of three organism groups and their interactions: (1) heterotrophic organisms able to accumulate polyP, termed polyP organisms, (2) heterotrophic organisms unable to accumulate polyP, termed non-polyP organisms and (3) autotrophic organisms mediating nitrification, termed autotrophs. With regard to the non-polyP heterotrophs and autotrophs:

- The ND kinetic model (Section 2 above) was available

- Under the preceding Consolidation Contract (K5/251), the kinetics of denitrification in NDBEPR systems had been investigated and quantified.

With regard to the polyP organisms, under preceding WRC contracts:

- Biochemical and mechanistic models for BEPR had been developed (K5/148).

- The kinetic and stoichiometric characteristics of the polyP organisms in enhanced cultures receiving artificial substrates had been delineated, and incorporated into a kinetic model for the enhanced cultures (K5/148).

- A steady state design model for mixed culture BEPR systems receiving municipal wastewaters had been developed (K5/251).

Under the current consolidation contract, attention has been focused on developing a general activated sludge NDBEPR kinetic model, by integrating the enhanced culture BEPR kinetic model with the mixed culture ND kinetic model taking due account of the denitrification kinetics in NDBEPR systems, and any interactions between the population groups introduced by the integration. Work on this model is nearing completion. The processes, their kinetic rate formulations and their stoichiometric interactions with the compounds have been published by Wentzel *et al.* (1992), together with preliminary estimates of values for the stoichiometric and kinetic constants.

The general NDBEPR kinetic model includes 25 process and 19 compounds. Thirteen processes reflect non-polyP organism behaviour, two processes autotrophic organism behaviour and nine process polyP organism behaviour.

With regard to the non-polyP and autotrophic organisms, the processes, their rate formulations and stoichiometric interactions with the compounds remain the same as for the ND kinetic model (Section 2 above, Dold *et al.*, 1991), i.e. death regeneration-bisubstrate synthesis model, with the exception that the interactions between these processes and the compound phosphorus are added. From initial simulations (see later), it would appear that the associated kinetic and stoichiometric constants also remain the same as for the ND model, with one exception.

In addition to processes obtained from the ND kinetic model, an extra process mediated by non-polyP organisms (i.e. conversion of readily biodegradable COD, RBCOD to short-chain fatty acids, SCFA) under anaerobic conditions is included in the NDBEPR kinetic model, to expand description of non-polyP organism behaviour under anaerobic conditions. Formulation of this process rate and stoichiometry was taken unmodified from the enhanced culture kinetic model (where it was included even though it served no kinetic function) for the general NDBEPR kinetic model.

With regard to the polyP organism description, the processes, the process rate formulations and kinetic and stoichiometric constants remain the same as in the enhanced culture kinetic

model.

At present the NDBEPR kinetic model is being evaluated; the evaluation still is restricted to steady state conditions for reason that extensive steady state experimental data are available (Wentzel *et al.*, 1990). From the simulations undertaken thus far, it appears that the values for constants from the ND and enhanced culture kinetic models can be largely retained. Good correlation between predicted and measured P removal for both mixed and enhanced culture systems has been obtained.

A number of aspects of the NDBEPR kinetic model have been identified to require further attention: For example, evaluation of the model under cyclic flow and load conditions; denitrification by polyP organisms; temperature dependency. It is envisaged that work on these aspects will continue, either under parallel WRC contracts (e.g. K5/542, where the effect temperature on bulking is being studied and the measured data can be used also to evaluate the effect of temperature on P release, uptake and removal) or by using information published in the literature.

One aspect of the kinetic model that has been investigated experimentally is the importance of slowly biodegradable COD (SBCOD) in BEPR. This aspect arose from the Water Research Group's participation in an IAWQ Task Group on Mathematical Modelling of Wastewater Treatment Systems. The IAWQ Task Group has finalized a kinetic model for NDBEPR systems (Gujer *et al.*, 1994; Henze *et al.*, 1994). In development of this model, contrary to the model developed at UCT, it was proposed that in the anaerobic reactor of the NDBEPR system SBCOD is hydrolysed to readily biodegradable COD (RBCOD) which then can contribute to the BEPR. If the SBCOD hydrolysis is significant, the SBCOD will make a substantial contribution to BEPR and this will influence the approach to both the design and operation of BEPR systems. Due to the importance of this aspect, an experimental investigation into the role of SBCOD in BEPR was undertaken (Wentzel *et al.*, 1994). From a series of batch test experiments in which the anaerobic P release response to different biodegradable COD fractions was monitored, it was concluded that the SBCOD does not induce P release and therefore will not make a significant contribution to BEPR. Accordingly, unless evidence to the contrary for the particular wastewater is available, anaerobic SBCOD hydrolysis should be excluded from the steady state design models. Also for dynamic models including an anaerobic hydrolysis process, the rate should be allocated a zero or very low value so that it makes a negligible contribution to the BEPR. In the absence of conclusive evidence indicating a high rate, a high rate would distort BEPR unrealistically. This observation is in agreement with experience at laboratory- and full-scale with South African municipal wastewaters.

In conclusion, by consolidating information on NDBEPR system behaviour generated under preceding WRC contracts, and experimentally investigating aspects where information was lacking, a general kinetic model for simulating the dynamic behaviour of the NDBEPR activated sludge system has been developed (Wentzel *et al.*, 1992). A number of aspects in this model have been identified to require investigation, and these will receive attention in the future. However, it is unlikely that a final solution ever will be attained - the nutrient removal plant is a complicated system in which the different system elements, processes and compounds interact in a complex manner; resolution of one problem may bring into focus, or create, another problem.

4. REVISION OF NDBEPR SYSTEM DESIGN MANUAL

With the introduction in South Africa of legislation in 1980 limiting P concentrations in effluents discharged from municipal wastewater treatment plants, intensive efforts were directed by the South African research community towards biological excess phosphorus removal (BEPR) in the activated sludge system. By 1984 the information gained was sufficient to enable a manual to be written in collaboration between the Water Research Group at UCT, the Johannesburg City Council and the National Institute for Water Research, for the design of aerobic, anoxic/aerobic and anaerobic/anoxic/aerobic activated sludge systems (WRC, 1984).

Since 1984 considerable new information on the nutrient (N & P) removal activated sludge system has become available, particularly on denitrification and BEPR. In this task, it was proposed to update the 1984 design manual to incorporate this new information. In both the previous (K5/251) and current (K5/356) consolidation contracts, revision of the 1984 design manual has been considered a priority. However, before the revision could commence, it was found necessary to undertake experimental investigations into areas where the information required was incomplete or not available, for example, into denitrification kinetics in NDBEPR systems (Clayton *et al.* 1989, 1991), importance of SBCOD in BEPR (Wentzel *et al.*, 1994). Under the previous and current consolidation contracts, these investigations have essentially been completed, and work has commenced on writing the revised design manual: Chapters 1, 3 and 4 have been completed; drafts have been prepared for Chapters 6, 7, 8, 10 and 11; drafts are in preparation for Chapters 2 and 9. Although the current consolidation contract has ended, completing revision of the design manual is a priority for the Water Research Group, and work on this aspect will continue.

5. CHARACTERIZATION OF MUNICIPAL WASTEWATER

The objectives for the activated sludge system have expanded to progressively include COD removal, nitrification, denitrification and BEPR, all mediated biologically. Concomitantly, to provide reliable designs and predictions of expected system performances, design procedures and kinetic simulation models of increasing complexity have been proposed. In terms of the framework for these design procedures and models, it is necessary to divide the influent COD, TKN and more recently P into a number of fractions. In the literature, procedures to quantify these fractions are either too difficult or elaborate, or simply not available. In this task, attention was focused on developing simple reliable techniques to quantify the influent carbonaceous material fractions (measured in terms of the COD parameter) required in the design and simulation models, i.e. readily biodegradable COD, slowly biodegradable COD, unbiodegradable soluble COD, unbiodegradable particulate COD, and heterotrophic active biomass (Mbewe *et al.*, 1995; Wentzel *et al.*, 1995).

A simple batch test procedure has been developed to quantify the influent COD fractions. In the batch test a sample of wastewater is placed in a batch reactor (no mixed liquor seed is added) and the oxygen utilization rate (OUR) response monitored with time for about 12 hours; from the OUR-time profile the heterotrophic active biomass and readily biodegradable COD concentration can be determined. After a further 12 hours aeration, a sample is drawn from the batch reactor, flocculated then filtered (0,45μm or glass fibre) and the COD concentration of the filtrate measured; this gives the unbiodegradable soluble COD. The

batch test is continued for at least a further 24 hours, then flocculated filtered raw sewage is added and OUR monitored for about 12 hours; from the OUR-time profile the unbiodegradable particulate COD and slowly biodegradable COD can be determined.

In evaluating the batch test procedure, it was concluded that the batch test method has advantages over previous methods in that:

- The experimental procedure is relatively simple.
- No mixed liquor acclimatized to the wastewater is required.
- The only independent constants required for calculation are the heterotrophic yield (Y_{ZH}), endogenous residue fraction for heterotrophic active biomass (f), and specific death rate (b_H): Dosing the batch test with known concentrations of acetate showed that the standard value for Y_{ZH} in the literature (Y_{ZH} = 0,666 mgCOD/mgCOD; Dold and Marais, 1986) can be accepted; the batch test procedure is relatively insensitive to the values for b_H and f. All other constants required for calculations are obtained from the experimental data. However, it is unlikely that these constants (i.e. maximum specific growth rate of heterotrophs of RBCOD, μ_H, and maximum specific growth rate of heterotrophs on SBCOD, K_{MP}) will be of much value in modelling and design of activated sludge systems - most probably a population will develop in the activated sludge system that differs appreciably from that in the wastewater since the conditions in the wastewater (high COD, low active mass) differ significantly from those in the activated sludge system (low COD, high active mass).

The batch test method was evaluated by comparing its results with those from conventional flow through activated sludge system methods accepted as the standard in the literature. Results from a number of batch tests on municipal wastewater from Mitchell's Plain and Borcherds Quarry (Cape Town, South Africa) indicate that:

- Autotrophic biomass is not present in either wastewater.
- Measured RBCOD concentrations correlate closely with those from the conventional square-wave flow through method (WRC, 1984; Ekama *et al.*, 1986).
- Although the values for wastewater heterotrophic active biomass could not be compared to conventional methods (none are available), the batch test was able to detect correctly variations in heterotrophic active biomass caused by changes in plant operational procedures.
- Values for unbiodegradable soluble COD derived from the batch test compared reasonably with those derived from the effluent of a long sludge age activated sludge system (Ekama *et al.*, 1986).
- Values for unbiodegradable particulate COD derived from the batch test fall in the same range as estimates from the conventional completely aerobic activated sludge system method (Ekama *et al.*, 1986). However, the direct correlation between the

values from the two tests is poor. For the present, the batch test does not provide estimates for unbiodegradable particulate COD that are sufficiently accurate and precise for use in design and simulation of activated sludge systems. For design and simulation, unbiodegradable particulate COD as a fraction of total COD should at least be able to be quantified into the ranges 0-0,05; 0,05-0,10; 0,10-0,15, etc. As yet, there is not sufficient surety that the estimate from the batch test will meet this requirement; more data are required.

- The errors in unbiodegradable particulate COD are reflected in the estimate from the batch test for slowly biodegradable COD. However, because the absolute value for the slowly biodegradable COD concentration is very much larger than that for the unbiodegradable particulate COD concentration, the relative error in the estimate for slowly biodegradable COD is very much less. The estimate for slowly biodegradable COD can be accepted for design and simulation.

From this investigation the following recommendations can be made:

- The batch test can be used successfully to determine the heterotrophic active biomass, RBCOD and the soluble unbiodegradable COD concentrations in the influent wastewater. In this investigation, the estimates for RBCOD and unbiodegradable soluble COD concentrations from the batch test could be compared to results from conventional test methods. However, the heterotrophic active biomass concentration could not be evaluated against other tests because no such tests are available. To evaluate estimates for heterotrophic active biomass, it is recommended that an inoculum of activated sludge mixed liquor from a defined continuous flow steady state system is introduced into the batch test. From the steady state model (WRC, 1984) the concentration of the heterotrophic active biomass in continuous flow system and therefore added into the batch test can be calculated, and compared to the concentration obtained from the batch test. If similar results are obtained then a powerful verification of the basis of present activated sludge models will have been provided. This is currently under investigation by the UCT group.

- For the batch test method, a technique has been developed to quantify the unbiodegradable particulate COD and slowly biodegradable COD fractions. However, direct correlation of estimates for these parameters from the batch test and conventional tests were poor. Also, no discernible trends could be identified in the relationship between values from the two tests. To identify clear trends, a more extensive experimental investigation is required, so that more data are available.

- This investigation has been restricted to quantifying the influent carbonaceous material fractions. Similar studies should be undertaken on the influent nitrogenous and phosphorous materials.

6. ANCILLARY INFORMATION

In this report progress is described on four research projects that did not fall under the ambit of the current consolidation contract, or any other WRC funded contract. This information

is included to keep the WRC and readers informed of progress at UCT on non-WRC sponsored contracts:

6.1 Dynamic simulation of secondary settling tanks and evaluation of secondary settling tank design procedures.

Theory and design of secondary settling tanks has progressed along two parallel directions with little cross-linking between them. At UCT, research is being conducted in both parallel directions seeking to bring them closer. This is being done by attempting to establish relationships between the settleability measures on which the flux theory and two empirical design procedures (German ATV and English WRc) are based, using data available in the literature. With the aid of these relationships, the designs produced by the flux theory and the two empirical procedures can be compared. With the relationships established to date, in the main a great deal of overlap between the procedures has been found.

Also, work has been conducted to evaluate the flux theory for modelling the dynamic behaviour of secondary settling tanks. A one dimensional model which includes turbulent diffusion has been developed. The model gives very good predictions compared with full-scale results measured by STORA in Netherlands for blanket rise rate, effluent and underflow concentration with time and sludge concentration depth profiles (Ozinsky *et al.*, 1994).

To make this and related information more widely available, a separate contract with the WRC has been set up to produce an IAWQ Scientific and Technical Report on design, simulation and operation of secondary settling tanks with the cooperation of an international team of experts.

6.2 Treatability of stabilized landfill leachate in a nitrogen removal activated sludge system

This study was undertaken to provide information on the feasibility of an integrated approach to municipal waste management, by operating sewage treatment plants and sanitary landfills in conjunction with each other: The leachate stream produced in the landfill is treated in the sewage treatment plant and the solid sludge stream produced in the sewage treatment plant is disposed of to the landfill.

From an experimental investigation on the effect of addition of stabilized leachate to a N removal activated sludge system, it was found that (1) the leachate is about 90% biodegradable, and (2) due to its high short-chain fatty acid (SCFA) and readily biodegradable COD content, it not only is able to denitrify all the nitrate generated from its own TKN, but also contributes significantly to the removal of N from the TKN in the sewage. The high SCFA content of the leachate may also stimulate BEPR when added to a nutrient removal system, an aspect not tested.

This preliminary study indicated that there is merit in pursuing leachate treatability studies further. This research will form part of a new contract between UCT and WRC on treatment of low organic (COD) high nutrient (N & P) wastewaters.

6.3 The effect of alternative detergent builders on NDBEPR systems

There is an argument for a ban on phosphate in detergents to reduce the phosphate load on sewage treatment plants. In South Africa, two possible replacements for phosphate are zeolite 4A and high surface area (HSA) calcite. In this research project the effect of these detergent builders on the NDBEPR activated sludge system was experimentally investigated (Kashula *et al.*, 1993). From the results, it would appear that the substitution of phosphates in detergent formulations with zeolite or HSA calcite will not have any adverse effects on the biological excess phosphorus removal sewage treatment process: No effect of zeolite or HSA calcite on COD removal, nitrification, volatile solids production, pH and denitrification could be established. The mass of sludge production (as total suspended solids) would increase (which is to be expected from the addition of inorganic material to the sewage), but this increase is likely to be very small - only 56% and 32% of the respective zeolite and HSA calcite dose was recovered in the sludge. The presence of zeolite and HSA calcite are not likely to adversely affect sludge settleability - indeed it appears that HSA calcite may have a small beneficial effect. Zeolite also appears to improve biological phosphate removal, but the reason for this is unclear.

6.4 The effect of thermophilic heat treatment on the anaerobic digestibility of sewage sludge

This research project followed from a WRC sponsored full-scale investigation into the dual digestion process by Milnerton Municipality, Afrox, Division of Water Technology and UCT. The full-scale investigation terminated due to structural failure of the aerobic reactor. As a result, one of the claimed advantages of the dual digestion system could not be convincingly confirmed, namely that the hot aerobic stage pretreats or conditions the sludge in such a way that the retention time of the anaerobic stage can be reduced compared to the conventional system. It was decided to investigate this claim at laboratory-scale (Izzet *et al.*, 1992).

From the results it was found that there was no discernable difference in anaerobic digester performance between systems receiving artificially thermophilically pretreated sludge (70°C for 1 day) and not pretreated, right down to seven days retention time. Unfortunately, due to the scale of operation, a laboratory autothermal thermophilic aerobic reactor could not be operated successfully. Thus, the influence of biological autothermal heat generation and not just the heat itself, on the anaerobic digester could not be tested. To test this conclusively, it will be necessary to set up a pilot- or full-scale aerobic reactor and repeat the experiments.

With regard to the overall status of research on dual digestion, work has been proceeding at Athlone sewage treatment works on dual digestion with pure oxygen enriched air (Pitt and Ekama, 1993). This essentially has completed the full spectrum of technical feasibility of the dual digestion at full-scale; pure oxygen (Milnerton), air (Athlone) and oxygen enriched air (Athlone).

7. RECOMMENDATIONS/FUTURE WORK

The research work reported in the previous and current consolidation contracts has provided all the information necessary to revise the 1984 manual for the design and operation of nutrient removal activated sludge systems. Work has commenced on this task and will be

completed in the near future. This design manual, together with the NDBEPR kinetic simulation model that has been developed, will essentially complete the Water Research Group's major research contribution to BEPR. It is clear that BEPR technology is now firmly established and widely exploited at full-scale. Essentially the future of BEPR no longer depends so much on a better understanding of the phenomenon, but on a better understanding of how to deal with the problems that can arise in operation of NDBEPR plants. Two such operational problems are currently being investigated by the Water Research Group under WRC sponsored contracts, namely filamentous organism bulking (K5/542) and treatment of anaerobic digester liquors (K5/692).

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Mr F P Marais	- Water Research Commission (Committee Secretary 1993 - 1994).

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LIST OF SYMBOLS AND ABBREVIATIONS

AA	Anoxic aerobic
ATV	Abwassertechnik Vereinigung
b_H	Heterotroph specific death rate (/d)
BEPR	Biological excess phosphorus removal
COD	Chemical oxygen demand
DSVI	Diluted sludge volume index
F/M	Food to microorganism ratio
F-RBCOD	Fermentable readily biodegradable COD
f_{up}	Fraction of total COD that is unbiodegradable particulate
f_{us}	Fraction of total COD that is unbiodegradable soluble
HSA	High surface area (calcite)
IAWPRC	International Association on Water Pollution Research and Control
IAWQ	International Association on Water Quality
K_{MP}	Heterotroph maximum specific growth rate on slowly biodegradable COD (/d)
N	Nitrogen
ND	Nitrification denitrification
NDBEPR	Nitrification denitrification biological excess phosphorus removal
NH_4^+	Ammonium
NO_2^-	Nitrite
NO_3^-	Nitrate
OUR	Oxygen utilization rate
P	Phosphorus
PHA	Polyhydroxyalkanoate
PHB	Polyhyroxybutyrate
polyP	Polyphosphate
RBCOD	Readily biodegradable COD
SBCOD	Slowly biodegradable COD
SCFA	Short-chain fatty acids
SSVI	Stirred specific volume index
STORA	Stichting Toegepast Onderzoek Reiniging Afvalwater
STR	Scientific and Technical Report
SVI	Sludge volume index
S_{bi}	Influent biodegradable COD concentration (mgCOD/ℓ)
S_{bpi}	Influent slowly biodegradable COD concentration (mgCOD/ℓ)
S_{bsi}	Influent readily biodegradable COD concentration (mgCOD/ℓ)
S_{ti}	Influent total COD concentration (mgCOD/ℓ)
S_{ui}	Influent unbiodegradable COD concentration (mgCOD/ℓ)
S_{upi}	Influent biodegradable particulate COD concentration (mgCOD/ℓ)
S_{usi}	Influent biodegradable soluble COD concentration (mgCOD/ℓ)
TKN	Total Kjeldahl Nitrogen
TSS	Total suspended solids
UCT	University of Cape Town
USCOD	Unbiodegradable soluble COD

$\hat{\mu}_H$	Heterotroph maximum specific growth rate on readily biodegradable COD (/d)
VSS	Volatile suspended solids
V_s	Solids zone settling velocity (m/h)
WRC	Water Research Commission (RSA)
WRc	Water Research Centre (England)
Y_{ZH}	Heterotroph specific yield (mgCOD/mgCOD)

PUBLICATIONS DURING CONTRACT PERIOD (JANUARY 1991 TO DECEMBER 1994)

An important objective of this contract was the transfer of technology generated under previous contracts, and of research results obtained during the contract period. In this regard the Water Research Group participated in a number of seminars and conferences (local and international), submitted a number of papers for publication in refereed journals, and published a number of books and reports. These contributions are listed below:

A. BOOKS

Published

1. Dold P L, Wentzel M C, Billing A E, Ekama G A and Marais GvR (1991). Activated sludge system simulation programs: Version 1.0 Nitrification and nitrification/denitrification systems. Water Research Commission, P O Box 824, Pretoria 0001, South Africa.

2. Moosbrugger R E, Wentzel M C, Ekama G A and Marais GvR (1992). Simple titration procedures to determine H_2CO_3*alkalinity and short-chain fatty acids in aqueous solutions containing known concentrations of ammonium, phosphate and sulphide weak acid/bases. Water Research Commission, P O Box 824, Pretoria 0001, South Africa.

Submitted

1. Henze M, Gujer W, Mino T, Matsuo T, Wentzel M C and Marais GvR (1994). Activated sludge model No.2. IAWQ Scientific and Technical Report, IAWQ, London.

B. CHAPTERS IN BOOKS

Submitted

1. Wentzel M C and Ekama G A (1994). Principles in the modelling of biological wastewater treatment plants. Chapter 7, IAWQ Scientific and Technical Report: Community analysis - the key to the design of biological wastewater treatment plants. Ed. Cloete T E, IAWQ, London.

C. PAPERS PUBLISHED

Refereed journals

1. Wentzel M C, Ekama G A and Marais GvR (1992). Processes and modelling of nitrification denitrification biological excess phosphorus removal systems - A review. Presented at IAWPRC conference on "Interactions between wastewater, biomass and reactor configurations in biological treatment plants", Copenhagen, August 1991. Wat.Sci.Tech., 25(6), 59-82.

2. Moosbrugger R E, Wentzel M C, Ekama G A and Marais GvR (1993). Treatment of wine distillery waste in UASB systems - feasibility, alkalinity requirements and pH control. Presented at IAWPRC 2nd Int. Symp.

on Waste Management problems in Agro-industries, Turkey, Sept. 1992. Wat.Sci.Tech., 28(2), 45-54.

3. Moosbrugger R E, Wentzel M C, Ekama G A and Marais GvR (1993). A 5 pH point titration method for determining the carbonate and SCFA weak acid/bases in anaerobic systems. Presented at IAWPRC 2nd Int. Symp. on Waste Management problems in Agro-Industries, Turkey, Sept. 1992. Wat.Sci.Tech., 28(2), 237-246.

4. Wentzel M C, Moosbrugger R E, Sam-Soon P A L N S, Ekama G A and Marais GvR (1994). Tentative guidelines for waste selection, process design, operation and control of upflow anaerobic sludge bed reactors. Presented at AD 94, Cape Town, Jan. 1994. Wat.Sci.Tech., 30(12), 31-42.

5. Wentzel M C, Mbewe A and Ekama G A (1995). Batch test for measurement of readily biodegradable COD and active organism concentrations in municipal wastewaters. Water SA, 21(2), 117-124.

Conference papers (refereed in abstract and published in proceedings)

1. Lilley I D, Wentzel M C, Loewenthal R E, Ekama G A and Marais GvR (1991). Acid fermentation of primary sludge at 20° C. Presented at WISA biennial conference, Kempton Park, May.

2. Ekama G A, Wentzel M C, Loewenthal R E and Marais GvR (1993). Some scientific and technological developments in biological wastewater treatment. Presented at Biotech. SA 93 Conference, Grahamstown, January.

3. Wentzel M C, Ekama G A and Marais GvR (1993). Modelling and design of nitrification denitrification biological excess phosphorus removal systems. Presented at Triennial EWPCA Conference/IFAT Exhibition, Munich, May.

4. Wentzel M C, Fourie L and Ekama G A (1994). Influence of wastewater biodegradable COD fractions on biological excess phosphorus removal. Presented at IAWQ Specialized Seminar on "Modelling and Control of Activated Sludge Processes", Copenhagen, August.

5. Gujer W, Henze M, Mino T, Matsuo T, Wentzel MC and Marais GvR (1994). The activated sludge model No.2: Biological phosphorus removal. Presented at IAWQ Specialized Seminar on "Modelling and Control of Activated Sludge Processes", Copenhagen, August.

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CHAPTER 1

INTRODUCTION

1.1 BACKGROUND TO CONSOLIDATION CONTRACTS

Since 1968 the Water Research Group in the Department of Civil Engineering at the University of Cape Town has conducted extensive research into

- wastewater treatment by means of the activated sludge system, and
- water chemistry.

The wastewater treatment research covered the spectrum of processes in the activated sludge system, such as organic energy (COD) removal, nitrification, denitrification and biological excess phosphorus removal (Marais and Ekama, 1976; Dold *et al.*, 1980; Van Haandel *et al.*, 1981; Siebritz *et al.*, 1983). By 1984 the information gained on the activated sludge system was sufficient to enable a manual to be written, in collaboration with the Johannesburg City Council and the National Institute for Water Research, for the design of aerobic (COD removal and nitrification), anoxic/aerobic (COD removal, nitrification and denitrification) and anaerobic/anoxic/aerobic (COD removal, nitrification, denitrification and biological excess phosphorus removal) systems, the last one being designated as the nutrient (nitrogen + phosphorus) removal system (WRC 1984). This design manual, disseminated the Water Research Commission, has received a very favourable response from the research and engineering communities. In addition to the design manual, a general kinetic model describing the behaviour of aerobic and anoxic/aerobic systems had been developed (Van Haandel *et al.*, 1981). This kinetic model also has been received favourably - an international task group, constituted by the International Association on Water Quality (IAWQ) to enquire into the best kinetic model, accepted the UCT model with some minor modifications (Henze *et al.*, 1987).

The water chemistry research covered principally the behaviour of the calcium/magnesium/carbonate weak acid/base in low salinity (total dissolved solids, TDS < 1 000 mg/ℓ) waters (Loewenthal and Marais, 1976; 1983). By the end of 1987 a manual had been written to assist in the chemical treatment of low salinity waters for domestic use, incorporating characterization and dosing estimation for softening, stabilization, 3-phase equilibria, etc. (Loewenthal *et al.*, 1986). This manual made extensive use of graphical procedures to reduce the labour of calculation, to such a level that routine application was achieved. To assist application even further, a user-friendly computer program was developed that replaced the graphical procedures with numerical algorithms (Loewenthal *et al.*, 1988). The design manual and the computer program (with its user manual) have been distributed through the Water Research Commission and also have been favourably received.

With regard to the activated sludge system design manual, for anaerobic/anoxic/aerobic nutrient removal systems, phosphorus removal was formulated empirically in terms of some of the system parameters such as anaerobic mass fraction, available readily biodegradable COD concentration and active mass concentration. Organisms directly implicated in biological excess phosphorus removal (BEPR) did not feature and the BEPR phenomenon was not linked to any basic biological or biochemical behaviour. As a consequence designs based on the manual had a measure of reliability only within the range of conditions from which

the empirical BEPR model in the manual had been developed; indeed there was a measure of uncertainty even with designs within this range because the basic mechanisms underlying the behaviour were not understood. Furthermore, BEPR did not feature in the general kinetic model so that no model to estimate the dynamic behaviour of BEPR was available for the purposes of design, operation and control of nutrient removal plants.

Accordingly, in 1984 a 3-year contract with the Water Research Commission (Contract No. K5/148) was undertaken to investigate BEPR with the principal objective to develop a microbiologically/biochemically based kinetic model for BEPR. From the research work under this contract the main development in the understanding of BEPR can be summarized as follows (Wentzel *et al.*, 1988a; 1988b):

(1) A biochemical model for BEPR was developed describing the pathways and their regulation. This model explains most of the observations relating to BEPR in the nutrient removal system.

(2) Enhanced cultures of polyP organisms were successfully developed in the Modified Bardenpho and UCT systems. By conducting batch tests on sludge harvested from the enhanced culture systems, the processes and compounds associated with BEPR could be delineated.

(3) With the processes and compounds defined, a mathematical model was developed in which the kinetics of the process rates and their stoichiometry with the compounds could be quantitatively expressed. This model described the BEPR response closely, in batch tests on sludge harvested from the enhanced culture systems, and in enhanced culture systems under constant flow and load conditions.

The research tasks above in essence completed the basic enquiry into the biochemical mechanisms and kinetics of BEPR. However, by the end of this contract (December 1987) it was apparent that a number of tasks related to application of the basic information still had to be completed, viz. the information supplied by the enhanced culture study had not been adequately disseminated, nor had it been incorporated in the mixed culture situations present in "real" nutrient removal systems treating domestic sewages, for the purposes of design, operation and control of such plants.

With regard to the manual and computer program on water treatment, application of these are restricted to waters containing only the calcium/magnesium/carbonate weak acid/base system in solution. In municipal wastewaters, however, in addition to the calcium/ magnesium/carbonate system, the phosphate and ammonia systems are present, and in anaerobic wastewater treatment processes sulphides and short chain fatty acids (e.g. acetate and propionate) are generated. All these weak acid/base systems may be present in such large concentrations relative to the calcium/magnesium/carbonate weak acid/base system, that they exert significant influence on the pH and therefore on the treatment of the water to achieve a desired condition. No information was available on the chemistry of waters containing mixtures of weak acid/base systems and no guidelines for their treatment were available.

1.2 PREVIOUS CONSOLIDATION CONTRACT (K5/251)

To address the deficiencies highlighted above, in 1988 a 3 year contract (K5/251) "Consolidation of activated sludge and water chemistry research" was set up between the Water Research Commission (WRC) and University of Cape Town (UCT) to:

- provide opportunity to publish the research work undertaken so far,
- fill in the areas of research perceived to require attention in order to
  - update the activated sludge system design manual,
  - extend the general kinetic model to incorporate BEPR,
  - extend the theory on water chemistry to include mixtures of weak acid/bases,
- promote technology transfer by means of seminars, updating of design manuals and development of user-friendly computer programs.

To meet these objectives the following tasks were identified:

- Re-evaluation of the technology available on completely aerobic nitrification and on anoxic/aerobic nitrification/denitrification (ND) systems.
- Consolidation of research into nitrification/denitrification/biological excess phosphorus removal (NDBEPR) systems.
- Extension of the weak acid/base chemistry of the carbonate system in aqueous solution to include a mixture of weak acid/base systems.

During the course of the contract a further task was identified:

- Chemical phosphorus removal from municipal wastewater by the disposal of waste alum sludge to the activated sludge system.

Progress achieved in completing these tasks by the end of the contract period (December 1990) has been reported in detail in the Final Report (Wentzel *et al.*, 1991). This can be summarized as follows:

- From a re-evaluation of the design procedures set out in the design manual (WRC, 1984), it was concluded that the procedures are still valid for ND systems. With regard to the general ND kinetic model to simulate the dynamic behaviour of ND systems, a computer program was developed to make the model more widely available. A user manual for this program still needed to be finalized for distribution with the program.
- Research on NDBEPR systems focused on the following areas:
  - Acid fermentation of primary sludge.
  - Kinetics of denitrification in NDBEPR systems.

- Modelling of BEPR.

Research on the first two areas above was completed (Lilley *et al.*, 1990; Clayton *et al.*, 1989, 1991). For modelling of BEPR, a steady state design model for BEPR was developed (Wentzel *et al*, 1990). In this model the nitrate recycled to the anaerobic reactor had to be known; in design it has been the practice to use the denitrification theory set out in the design manual (WRC, 1984). However, from the research into denitrification in NDBEPR systems, the denitrification kinetics in NDBEPR systems are not the same as accepted in the design manual - the denitrification kinetics measured in NDBEPR system needed to be incorporated into the steady state BEPR design model. Also a general dynamic model for NDBEPR systems still needed to be developed.

- The weak acid/base chemistry of the carbonate system in aqueous solutions was extended to include mixtures of weak acid/base systems. This was achieved by:

  - generalizing the concepts of alkalinity and acidity and developing a consistent nomenclature,
  - extending the Gran function to include the carbonate subsystem in mixtures of weak acid/bases (Loewenthal *et al.*, 1989),
  - making use of the concept of weak acid/base subsystem and water subsystem alkalinities.

  Using the concepts above, relatively simple algorithms were developed to determine the chemical type and dosage to achieve a desired change in the chemical state of a solution consisting of a mixture of weak acid/bases (Loewenthal *et al.*, 1991).

- An extensive experimental investigation was completed on chemical phosphorus removal from municipal wastewater by disposal of waste alum sludge to the activated sludge system (Power *et al.*, 1991).

1.3 CURRENT CONSOLIDATION CONTRACT (K5/356)

From the brief summary above, it is evident that not all the tasks had been completed by the end of the contract period (December, 1990). Accordingly, in 1991 a new 3 year contract "Consolidation of activated sludge research II" (K5/356) was set up between the WRC and UCT to continue the work not completed under the preceding contract. Subsequently, the contract has been extended by one year without additional funds. For the contract, the following principal aims were identified:

- Complete the calibration of, and the user manual for the nitrification denitrification (ND) kinetic simulation model.

- Continue the development of a kinetic simulation model for nitrification denitrification biological excess phosphorus removal (NDBEPR) systems.

- Revise the 1984 NDBEPR system design manual to incorporate the research findings

on ND and BEPR since 1984.

- Write papers and reports on experimental work completed under preceding contracts, in particular on aspects relating to NDBEPR systems.

- Conduct experimental work to evaluate the effect of incorporation of fixed media in the aerobic zone in NDBEPR systems, with the view to reducing the system sludge age yet maintaining virtually complete nitrification.

At the Steering Committee meeting in June 1992, the members noted that the last aim, namely incorporation of fixed media in NDBEPR systems, might lead to duplication of effort since Professor Pretorius at the University of Pretoria is conducting research in this area. Accordingly, the aim was modified to:

- Develop and refine methods for characterization of municipal wastewaters.

Progress in achieving these aims is summarized in this report. Also, progress in four additional areas of research is reported:

- Dynamic simulation of secondary settling tanks and evaluation of secondary settling tank design procedures based on various sludge settleability parameters.

- Treatment of landfill leachates.

- The effect of alternative detergent builders on NDBEPR systems.

- The effect of thermophilic heat treatment on the anaerobic digestibility of sewage sludge.

Research in the above four areas does not fall directly under the ambit of the current consolidation contract, but progress is reported to keep the WRC informed of the work being done in these areas.

This final report is a summary of the research undertaken to address the aims identified above. As specific research topics have been completed, detailed reports and papers have been prepared and published. These detailed reports and papers have been referenced at the appropriate place in this final report and are also listed in the section "**PUBLICATIONS DURING CONTRACT PERIOD**". For greater detail on the research, the reader is referred to these publications.

CHAPTER 2

NITRIFICATION DENITRIFICATION (ND) KINETIC SIMULATION MODEL

Worldwide, increasing awareness of the adverse impact of wastewater discharges on aquatic environments has led to the introduction of more stringent legislation controlling the quality of wastewater treatment plant effluents (e.g. South Africa, 1984 amendment to Section 21 of the 1956 Water Act, Government Gazette, 1984). To comply with the new legislations, over the past 20 years there have been extensive developments in the activated sludge method of treating wastewater. The functions of the single sludge system have expanded from carbonaceous energy removal to include progressively nitrification, denitrification and phosphorus removal, all mediated biologically. These extensions have been accommodated through manipulation of the system configuration - incorporation of multiple in-series reactors, some aerated and others not, with various inter-reactor recycles. Not only has the system configuration and its operation increased in complexity, but concomitantly the number of biological processes influencing the system performance and the number of compounds involved in these processes have increased. With such complexity, designs based on experience or semi-empirical methods no longer will give optimal performance; design procedures based on more fundamental behavioural patterns are required. Also, it is no longer possible to make a reliable quantitative or sometimes even qualitative prediction as to the effluent quality to be expected from a design, or to assess the effect of a system or operational modification, without some model that simulates the system behaviour accurately.

To design the system optimally and to operate it effectively, concerted efforts were made over the past 15 to 20 years to model the behaviour of these systems. This was also the case in South Africa, where development of a reliable mathematical model was given intensive attention by the Water Research Group at the University of Cape Town (UCT). The result of this research endeavour was a very powerful kinetic model that gives a reliable description of the nitrification and nitrification/denitrification (ND) system response over wide ranges of system configuration (single and in-series reactor systems, aerated and non-aerated reactors, inter-reactor recycles), wastewater characteristics (COD, TKN, flow pattern) and operational parameters (sludge age, temperature, dissolved oxygen concentration) (Dold *et al.*, 1980; Van Haandel *et al.*, 1981). Subsequently, an international task group was constituted by the IAWQ (formally IAWPRC) to enquire into the best kinetic model. This task group accepted the UCT model, but incorporated some minor modifications (Henze *et al.*, 1987).

The UCT model has had a significant impact on design and operational procedures of single sludge nitrification and ND activated sludge systems. With regard to design, the model led to the identification of procedures to estimate the optimal or near optimal design configuration, reactor sizes and operation parameters (e.g. sludge age) and to estimate the expected response (WRC, 1984).

Once the system had been designed, the time response under dynamic flow and load conditions could be estimated using the kinetic model. Thereupon the design could be modified if necessary to achieve improved performance, or, the sensitivity of the design to changes in flow and load conditions or to operational modifications could be assessed. In full-scale plant operation it also found application in assessing the effects of changes in waste

flows and loads, operational modifications (e.g. changes in recycles), and proposed modifications to plant configuration. Furthermore, it proved valuable in operator training; through simulation exercises using the model the operator acquired "instant" experience in the behaviour to be expected with changes in inputs, system configuration and operational strategies. A difficulty in making routine use of the model was that it required the availability of a main-frame computer.

To make the UCT and IAWQ Task Group kinetic models more widely available at the different levels of application, computer programs of the model were written that are (1) suitable for personal computers, (2) "user friendly", (3) flexible, and (4) provide rapid solutions, numerical or graphical. During the preceding consolidation contract, the computerised models were extensively calibrated against sets of experimental data accumulated over a lengthy period by the UCT group. A user manual to accompany the two computer simulation programs was also written and tabled at the final Steering Committee meeting of the preceding consolidation contract.

The Water Research Group at UCT received considerable feedback from Steering Committee Members who identified errors and omissions in both the computer programs and user manual. During the current consolidation contract, these problems in the computer programs and the user manual have been corrected. The user manual has been printed and is being distributed by WRC together with the computer programs (Dold *et al.*, 1991). This completes this section of the work.

CHAPTER 3

NITRIFICATION DENITRIFICATION BIOLOGICAL EXCESS PHOSPHORUS REMOVAL (NDBEPR) KINETIC SIMULATION MODEL

3.1 BACKGROUND

Following the development of the kinetic model for nitrification and nitrification denitrification (ND) systems (Van Haandel *et al.*, 1981; see Chapter 2), in 1983 the UCT group set the objective to develop a general kinetic model that describes the dynamic behaviour of an activated sludge system treating municipal wastewater which incorporates also biological excess phosphorus removal (BEPR), i.e. a nitrification denitrification biological excess phosphorus removal (NDBEPR) system. In such a system, a mixed culture of organisms would develop which can be categorized into three groups; (1) heterotrophic organisms able to accumulate polyP, termed polyP organisms, (2) heterotrophic organisms unable to accumulate polyP, termed non-polyP organisms, and (3) autotrophic organisms mediating nitrification, termed autotrophs. It was recognized that development of a kinetic model for NDBEPR systems would require inclusion of all three organism groups, and their interactions. With regard to the non-polyP and autotrophic organisms, the ND kinetic model (Dold *et al.*, 1980; Van Haandel *et al.*, 1981) was available. This model needed to be extended to incorporate polyP organism behaviour, to give a NDBEPR kinetic model. To achieve this objective, the kinetic and stoichiometric characteristics of the polyP organisms in the activated sludge environment needed to be established. Under preceding WRC contracts considerable progress had been made in achieving this objective:

> From attempts to obtain information on the characteristics of the polyP organisms using mixed liquor from NDBEPR systems treating municipal wastewaters, Wentzel *et al.* (1988a) concluded that, in these mixed culture systems, the non-polyP organism behaviour tends to dominate and mask the polyP organism behaviour. Accordingly, it was endeavoured to isolate the polyP organism characteristics, by developing enhanced cultures of these organisms in activated sludge systems. Using the UCT and 3-stage Modified Bardenpho systems, with system sludge ages ranging from 7,5-20d, enhanced cultures of polyP organism were developed (Wentzel *et al.*, 1988c). From experimental observations on the enhanced culture steady state systems, and on batch tests in which mixed liquors drawn from the steady state systems were subjected to a wide variety of conditions, Wentzel *et al.* (1989a) elucidated the characteristics and kinetic response of the polyP organism mass. Wentzel *et al.* (1989b) formulated mathematically the process rates and their stoichiometric interactions with the compounds, to develop a *kinetic model for the enhanced cultures of polyP organisms.* Using the enhanced cultures, the kinetic and stoichiometric constants were quantified by a variety of experimental procedures. With these constants, application of the kinetic model to the various test responses observed with the enhanced cultures gave good correlation between observation and simulations (Wentzel *et al.*, 1989b).

Having developed the kinetic model for enhanced cultures of polyP organisms, the intention was to extend this model to the mixed cultures that develop in NDBEPR systems treating municipal wastewaters. This required integration of the enhanced culture BEPR kinetic model with the ND kinetic model (see Chapter 2), to give a NDBEPR kinetic model. Before the

integration could be commenced, however, attention had to be given to the denitrification kinetics in NDBEPR systems.

3.2 DENITRIFICATION KINETICS IN NDBEPR SYSTEMS

In the initial integration of the enhanced culture BEPR and ND kinetic models, the denitrification kinetics in the ND model were retained unmodified. With this approach an inconsistency became evident:

> The enhanced culture studies have indicated that polyP organisms do not denitrify. This implies that the readily biodegradable COD (RBCOD), converted to short chain fatty acids (SCFA) by the non-polyP organisms and sequestered by the polyP organisms in the anaerobic reactor, no longer are available for denitrification in the primary anoxic reactor of a NDBEPR system. This in turn would imply that the magnitude of the denitrification in the primary anoxic reactor of the NDBEPR system should be significantly smaller than that in the primary anoxic reactor of the ND system. However, experimental observations on NDBEPR systems indicate that this is not so, that approximately the same magnitude of denitrification is achieved. The implications are that the denitrification kinetics for ND systems need to be adapted, or modified, for application in NDBEPR systems.

As reported in the final report for the previous consolidation contract, (Wentzel *et al.*, 1991), experimental investigations have been conducted into the kinetics of denitrification in NDBEPR systems (Clayton *et al.*, 1989; 1991) to resolve this inconsistency. These have indicated that in NDBEPR systems:

(i) In the primary anoxic reactor the rapid rate of denitrification (K_1, WRC, 1984) associated with RBCOD was much reduced or absent (in agreement with the understanding of the mechanisms for BEPR, see above) and

(ii) in the primary and secondary anoxic reactors, the specific denitrification rates associated with adsorbed slowly biodegradable COD (SBCOD) are up to 2½ times (K_2, WRC, 1984) and 1½ times (K_3, WRC, 1984) higher respectively than in ND systems.

These two observations explain the apparent equality in the denitrification achieved in NDBEPR and ND systems. From an extensive enquiry into causes, Clayton *et al.* concluded that the increased denitrification rates *were not due to*:

(i) denitrification by polyP organisms - for the systems investigated, polyhydroxybutyrate (PHB) and P measurements indicated that the polyP organisms did not denitrify,

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

The above observations led Clayton *et al.* to conclude that the increased denitrification rate

is due to a stimulation in the active sludge mass of an increased specific rate of hydrolysis of SBCOD in the anoxic reactor of NDBEPR systems, apparently induced by the presence of the anaerobic reactor.

In this work, a connection between the degree of denitrification and bulking was noted. In follow up work under the Bulking Contract (K5/286) the NDBEPR denitrification kinetics observed by Clayton *et al.* were confirmed by Musvoto *et al.* (1992).

This represents the state of kinetic modelling of NDBEPR systems on termination of the previous consolidation contract.

3.3 INTEGRATED NDBEPR KINETIC MODEL

Under the current consolidation contract attention has focused on integration of the enhanced culture BEPR *kinetic* model with the general activated sludge ND *kinetic* model, to give a general activated sludge NDBEPR *kinetic* model (Ekama *et al.*, 1990).

Work on the integration is well advanced and is reported in detail by Wentzel *et al.* (1992). The integrated NDBEPR kinetic model (Table 3.1) includes 25 processes and 19 compounds. Processes 1-12 and process 25 reflect non-polyP organism behaviour, processes 13-14 autotrophic organism behaviour and processes 15-24 polyP organism behaviour. Kinetic and stoichiometric constants are listed in Table 3.2, and the symbol system used is set out in Table 3.3.

With regard to the non-polyP and autotrophic organism, the processes (processes 1-14), their rate formulations and stoichiometric interactions with the compounds remain the same as for the ND kinetic model, UCT version (Dold *et al.*, 1991), i.e. death regeneration-bisubstrate synthesis model, with the exception that the interactions between these processes and the compound phosphorus are added. From initial simulations (see later), it would appear that the associated kinetic and stoichiometric constants also remain the same as for the ND model, with one exception - the value for η_G which is the ratio of the hydrolysis/utilization rate for SBCOD under anoxic and aerobic conditions, i.e. η_G = anoxic rate/aerobic rate. The value for η_G had to be increased, from 0,33 to approximately 0,6 to reflect the experimentally observed denitrification by non-polyP organisms in NDBEPR systems (see Section 3.2 above). Simulations using the NDBEPR kinetic model with the increased value for η_G showed that the increased denitrification rates in primary and secondary anoxic reactors of NDBEPR systems are, as noted earlier, about 2½ times and 1½ times respectively the rates for ND systems.

In addition to processes obtained from the ND kinetic model, an extra process mediated by non-polyP organisms (i.e. process 25 conversion of RBCOD to SCFA under anaerobic conditions) is included in the NDBEPR kinetic model, to expand description of non-polyP organism behaviour to anaerobic conditions. Formulation of this process rate and stoichiometry was taken unmodified from the enhanced culture kinetic model (see Wentzel et al., 1985; 1989b) (where it was included even though it served no kinetic function) for the general NDBEPR kinetic model.

Table 3.1: Process kinetics and stoichiometry for nitrification denitrification biological excess phosphorus removal in NDBEPR systems.

J_i COMPONENT i → PROCESS	1 Z_{BH}	2 Z_{BA}	3 Z_{BC}	4 Z_E	5 Z_I	6 S_{acba}	7 S_{amm}	8 S_{PHB}	9 X_{cbp}	10 P_{polyP}	11 $S_{NO,N}$	12 $S_{NO,N}$
1 Aerobic growth of Z_{BH} on S_{NO} with η_A	1									u	$-1/Y_{ZH}$	
2 Aerobic growth of Z_{BH} on S_{NO} with η_{AS}	1										$-1/Y_{ZH}$	
3 Anoxic growth of Z_{BH} on S_{NO} with η_g	1										$-1/Y_{ZH}$	
4 Anoxic growth of Z_{BH} on S_{NO} with η_{AS}	1										$-1/Y_{ZH}$	
5 Aerobic growth of Z_{BH} on S_{acba} with η_A	1					$-1/Y_{ZH}$						
6 Aerobic growth of Z_{BA} on S_{acba} with η_{AS}	1					$-1/Y_{ZA}$						
7 Anoxic growth of Z_{BH} on S_{acba} with η_g	1					$-1/Y_{ZH}$						
8 Anoxic growth of Z_{BA} on S_{acba} with η_{AS}	1					$-1/Y_{ZA}$						
9 Death of Z_{BH}	-1			$f_{EP,H}$			$1-f_{EP,H}$		$f_{ZBH,N}-f_{EPH}f_{ZEH,N}$			
10 Adsorption of S_{amm}						1	-1					
11 Hydrolysis of X_{cbp}									$-1/Y_{ZH}$			
12 Ammonification of N_{cbp}												
13 Aerobic growth of Z_{BA}		1										
14 Death of Z_{BA}		-1		$f_{EP,A}$			$1-f_{EP,A}$		$f_{ZBA,N}-f_{EPA}f_{ZEA,N}$			
15 Aerobic growth of Z_{BC} on S_{PHB} with η_A			1					$-1/Y_{ZB}$		$f_{P,HBC}/Y_{ZB}$		
16 Aerobic growth of Z_{BC} on S_{PHB} with η_{AS}			1					$-1/Y_{ZB}$		$f_{P,HBC}/Y_{ZB}$		
17 J_{13} if P_s limiting			1					$-1/Y_{ZB}$		$-f_{ZBH,P}$		
18 I_{14} if ϕ_s limiting			1					$-1/Y_{ZB}$		$-f_{ZBH,P}$		
19 Aerobic death of Z_{BC}			-1	$f_{EP,C}$					X^*			
20 Anaerobic/anoxic death of Z_{BC}			-1	$f_{EP,C}$			$1-f_{EP,C}-f_{EN,C}$		X^*			
21 Lysis of P_{polyP} for J_{19} - J_{20}										-1		
22 Lysis of S_{PHB} for J_{19} - J_{20}			1				1	-1				
23 Anaerobic/anoxic maintenance P_{polyP} cleavage										-1		
24 Sequestration of $S_{NO,N}$ by Z_{BC}								1		$-f_{P,Pol}$		-1
25 Conversion of $S_{NO,C}$, Z_{BH} to $S_{NO,N}$ by											-1	1
	Biological (active) heterotrophic mass - M (COD) L^{-3}	Biological (active) autotrophic mass - M (COD) L^{-3}	Biological (active) polyP^- mass - M (COD) L^{-3}	Endogenous mass - M (COD) L^{-3}	Inert mass - M (COD) L^{-3}	Adsorbed slowly biodegradable substrate - M (COD) L^{-3}	Unadsorbed slowly biodegradable substrate - M (COD) L^{-3}	Stored polyhydroxybutyrate - M (COD) L^{-3}	Biodegradable particulate organic nitrogen - M (N) L^{-3}	Stored polyphosphate - M (P) L^{-3}	Readily biodegradable "complex" substrate - M (COD) L^{-3}	Readily biodegradable "acetate" substrate - M (COD) L^{-3}

Table 3.1: Continued.

13 S_o	14 S_{ndbs}	15 S_{n3}	16 P_s	17 S_{ss}	18 Alk	19 O	PROCESS RATES, ρ_j
$-i_{ZBA,N}$			$-i_{ZBA,P}$			$-\frac{1-Y_{ZH}}{Y_{ZH}}$	$\hat{\mu}_H \left[\frac{S_{bs,c}}{K_{BH}+S_{bs,c}}\right]\left[\begin{matrix}\text{O Air}\\ \text{on}\end{matrix}\right]\left[\begin{matrix}S_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BH}$
		$-i_{ZBN,N}$	$-i_{ZBA,P}$			$-\frac{1-Y_{ZH}}{Y_{ZH}}$	$\hat{\mu}_H \left[\frac{S_{bs,c}}{K_{BH}+S_{bs,c}}\right]\left[\begin{matrix}\text{O Air}\\ \text{on}\end{matrix}\right]\left[\begin{matrix}1-N_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}N_{n3}\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BH}$
$-i_{ZBA,N}$		$-\frac{1-Y_{ZH}}{2.86Y_{ZH}}$	$-i_{ZBA,P}$				$\hat{\mu}_H \left[\frac{S_{bs,c}}{K_{BH}+S_{bs,c}}\right]\left[\begin{matrix}\text{O Air}\\ \text{Off}\end{matrix}\right]\left[\begin{matrix}N_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}N_{n3}\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BH}$
		0*	$-i_{ZBA,P}$				$\hat{\mu}_H \left[\frac{S_{bs,c}}{K_{BH}+S_{bs,c}}\right]\left[\begin{matrix}\text{O Air}\\ \text{Off}\end{matrix}\right]\left[\begin{matrix}1-N_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}N_{n3}\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BH}$
$-i_{ZBA,N}$			$-i_{ZBA,P}$			$-\frac{1-Y_{ZH}}{Y_{ZH}}$	$\hat{\mu}_{AH}\left[\frac{(S_{ndbs}/Z_{BH})}{K_{SH}+(S_{ndbs}/Z_{BH})}\right]\left[\begin{matrix}\text{O Air}\\ \text{On}\end{matrix}\right]\left[\begin{matrix}S_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BA}$
		$-i_{ZBA,N}$	$-i_{ZBA,P}$			$-\frac{1-Y_{ZH}}{Y_{ZH}}$	$\hat{\mu}_{AH}\left[\frac{(S_{ndbs}/Z_{BH})}{K_{SH}+(S_{ndbs}/Z_{BH})}\right]\left[\begin{matrix}\text{O Air}\\ \text{On}\end{matrix}\right]\left[\begin{matrix}1-N_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}N_{n3}\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BA}\,\eta_G$
$-i_{ZBA,N}$		$-\frac{1-Y_{ZH}}{2.86Y_{ZH}}$	$-i_{ZBA,P}$				$\hat{\mu}_{AH}\left[\frac{(S_{ndbs}/Z_{BH})}{K_{SH}+(S_{ndbs}/Z_{BH})}\right]\left[\begin{matrix}\text{O Air}\\ \text{Off}\end{matrix}\right]\left[\begin{matrix}S_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}N_{n3}\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BA}\,\eta_G$
		0*	$-i_{ZBA,P}$				$\hat{\mu}_{AH}\left[\frac{(S_{ndbs}/Z_{BH})}{K_{SH}+(S_{ndbs}/Z_{BH})}\right]\left[\begin{matrix}\text{O Air}\\ \text{Off}\end{matrix}\right]\left[\begin{matrix}1-N_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}N_{n3}\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BA}\,\eta_G$
			$i_{ZBA,P}$ $f_{ECH}i_{ZEH,P}$				$b_H Z_{BH}$
							$\eta_A S_{amm} Z_{BH}\left(S_{NH}-S_{ndbs}/Z_{BH}\right)$
	$1/Y_{ZH}$						$(\rho_5+\rho_6+\rho_7+\rho_8)\,(S_{pbo}/S_{ndbs})$
1	-1						$\eta_H N_{nbio} Z_{BH}$
$-i_{ZBA,N}$ $-1/Y_{ZA}$		$1/Y_A$	$-i_{ZBA,P}$			$-\frac{4.57-Y_{ZA}}{Y_{ZA}}$	$\hat{\mu}_A\left[\frac{S_o}{K_{SA}+S_o}\right]\left[\begin{matrix}\text{O Air}\\ \text{On}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BA}$
			$i_{ZBA,P}$ $f_{ECH}i_{ZEA,P}$				$b_A Z_{BA}$
$-i_{ZBG,N}$			$i_{P,upt}/Y_{ZG}$ $-i_{ZBG,P}$			$-\frac{1-Y_{ZG}}{Y_{ZG}}$	$\hat{\mu}_{G1}\left[\frac{(S_{PHB}/Z_{BG})}{K_{SG1}+(S_{PHB}/Z_{BG})}\right]\left[\begin{matrix}\text{O Air}\\ \text{On}\end{matrix}\right]\left[\begin{matrix}S_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right] Z_{BG}$
		$-i_{ZBG,N}$	$i_{P,upt}/Y_{ZG}$ $-i_{ZBG,P}$			$-\frac{1-Y_{ZG}}{Y_{ZG}}$	$\hat{\mu}_{G1}\left[\frac{(S_{PHB}/Z_{BG})}{K_{SG1}+(S_{PHB}/Z_{BG})}\right]\left[\begin{matrix}\text{O Air}\\ \text{On}\end{matrix}\right]\left[\begin{matrix}1-N_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_s\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}N_{n3}\\ \text{Limit}\end{matrix}\right] Z_{BG}$
$-i_{ZBG,N}$						$-\frac{1-Y_{ZG}}{Y_{ZG}}$	$\hat{\mu}_{G2}\left[\frac{(S_{PHB}/Z_{BG})}{K_{SG2}+(S_{PHB}/Z_{BG})}\right]\left[\begin{matrix}\text{O Air}\\ \text{On}\end{matrix}\right]\left[\begin{matrix}S_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}1-P_s\\ \text{Limit}\end{matrix}\right] Z_{BG}$
		$-i_{ZBG,N}$				$-\frac{1-Y_{ZG}}{Y_{ZG}}$	$\hat{\mu}_{G2}\left[\frac{(S_{PHB}/Z_{BG})}{K_{SG2}+(S_{PHB}/Z_{BG})}\right]\left[\begin{matrix}\text{O Air}\\ \text{On}\end{matrix}\right]\left[\begin{matrix}1-N_o\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}1-P_s\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}N_{n3}\\ \text{Limit}\end{matrix}\right] Z_{BG}$
			$i_{ZBG,P}$ $f_{ECH}i_{ZEG,P}$	$f_{Fe,G}$		$-(1-i_{Ep,G}$ $-i_{Fe,G})$	$b_G Z_{BG}\left[\begin{matrix}\text{O Air}\\ \text{On}\end{matrix}\right]$
			$i_{ZBG,P}$ $f_{ECH}i_{ZEG,P}$	$f_{Fe,G}$			$b_G Z_{BG}\left[\begin{matrix}\text{O Air}\\ \text{Off}\end{matrix}\right]$
			1				$(P_{polye}/Z_{BG})\,((P19+P20))$
							$(S_{PHB}/Z_{BG})\,((P19+P20))$
			1				$b_{pp} Z_{BG}\left[\begin{matrix}\text{O Air}\\ \text{Off}\end{matrix}\right]\left[\begin{matrix}P_{polye}\\ \text{Limit}\end{matrix}\right]$
			$f_{P,rel}$				$K_P Z_{BG}\left[\begin{matrix}\text{NAc}\\ \text{Limit}\end{matrix}\right]\left[\begin{matrix}P_{polye}\\ \text{Limit}\end{matrix}\right]$
							$\eta_C S_{bs,c} Z_{BH}\left[\begin{matrix}\text{N Air}\\ \text{Off}\end{matrix}\right]\left[\begin{matrix}1-N_{n3}\\ \text{Limit}\end{matrix}\right]$
Ammonia nitrogen - N (S_{NH}) L^{-3}	Biodegradable soluble organic nitrogen - N (S_{ND}) L^{-3}	Nitrate nitrogen - N (S_{NO}) L^{-3}	Soluble phosphate - M (S_{PO4}) L^{-3}	Unbiodegradable soluble substrate - M (COD) L^{-3}	Alkalinity - molar units	Oxygen - M ($-O_2$) L^{-3}	

KEY
$A^* = i_{ZBG,N} - i_{Ep,G}\, i_{ZEG,N} - i_{Fe,G}\, i_{ECH,N}$
$B^* = \frac{1-Y_{ZH}}{2.86\,Y_{ZH}} - i_{ZBA,N}$

Table 3.2(a): Values for the kinetic parameters in the NDBEPR kinetic model (Table 3.1), and their temperature dependency.

Kinetic parameters (20°C)		
SYMBOL	**VALUE**	**UNITS**
Non-polyP heterotrophs		
μ_H	1,5 - 3,5	d^{-1}
K_{SH}	5,0	g COD m^{-3}
K_{MP}	1,35	g COD (g cell COD.d)$^{-1}$
K_{SP}	0,027	g COD (g cell COD)$^{-1}$
b_H	0,62	d^{-1}
K_A	0,17	g COD (g cell COD.d)$^{-1}$
K_R	0,032	m^3 (g cell COD.d)$^{-1}$
K_C	0,04	m^3 (g cell COD.d)$^{-1}$
η_G	0,33	-
Autotrophs		
μ_A	0,2 - 0,75	d^{-1}
K_{SA}	1,0	g NH_3-N m^{-3}
b_A	0,04	d^{-1}
PolyP organisms		
μ_{G1}	1,2	d^{-1}
K_{SG1}	0,18	g COD m^{-3}
μ_{G2}	0,42	d^{-1}
K_{SG2}	0,18	g COD m^{-3}
b_G	0,04	d^{-1}
b_{pp}	0,03	g P (g cell COD.d)$^{-1}$
K_p	6,0	g COD (g cell COD.d)$^{-1}$

Arrhenius temperature dependency constants

$K_{i,T} = K_{i,20}\ \theta^{(T-20)}$

where $K_{i,T}$ = K_i at T°C

Θ = Arrhenius constant

SYMBOL	**VALUE**	**UNITS**
Non-polyP heterotrophs		
μ_H	1,200	-
K_{SH}	1,000	-
K_{MP}	1,080	-
K_{SP}	0,910	-
b_H	1,029	-
K_A	1,029	-
K_R	1,029	-
K_C	?	-
Autotrophs		
μ_A	1,123	-
K_{SA}	1,123	-
b_A	1,029	-
PolyP organisms		
?	?	-

Table 3.2(b): Values for the stoichiometric and switching function parameters in the NDBEPR kinetic model (Table 3.1).

Stoichiometric parameters		
SYMBOL	**VALUE**	**UNITS**
Non-polyP heterotrophs		
Y_{ZH}	0,666	g cell COD (g COD utilized)$^{-1}$
f_{MA}	1,0	g COD (g cell COD)$^{-1}$
$f_{Ep,H}$	0,08	-
$f_{ZBH,N}$	0,068	g N (g COD)$^{-1}$ in active mass
$f_{ZEH,N}$	0,068	g N (g COD)$^{-1}$ in endog. mass
$f_{ZBH,P}$	0,02	g P (g COD)$^{-1}$ in active mass
$f_{ZEH,P}$	0,02	g P (g COD)$^{-1}$ in endog. mass
f_{cv}	1,48	g COD (gVSS)$^{-1}$
Autotrophs		
Y_{ZA}	0,15	g cell COD (g N utilized)$^{-1}$
$f_{Ep,A}$	0,08	-
$f_{ZBA,N}$	0,068	g N (g COD)$^{-1}$ in active mass
$f_{ZEA,N}$	0,068	g N (g COD)$^{-1}$ in endog. mass
$f_{ZBA,P}$	0,02	g P (g COD)$^{-1}$ in active mass
$f_{ZEA,P}$	0,02	g P (g COD)$^{-1}$ in endog. mass
PolyP organisms		
Y_{ZG}	0,666	g cell COD (g COD utilized)$^{-1}$
$f_{Ep,G}$	0,25	-
$f_{Es,G}$	0,20	-
$f_{ZBG,N}$	0,068	g N (g COD)$^{-1}$ in active mass
$f_{ZEG,N}$	0,068	g N (g COD)$^{-1}$ in endog. mass
$f_{EsG,N}$	0,068	g N (g COD)$^{-1}$ in sol inert COD
$f_{ZBG,P}$	0,02	g P (g COD)$^{-1}$ in active mass
$f_{ZEG,P}$	0,02	g N (g COD)$^{-1}$ in endog. mass
$f_{P,rel}$	0,5	g P (g COD)$^{-1}$
$f_{P,upt}$	0,75	g P (g COD)$^{-1}$

Switching function parameters		
SYMBOL	**VALUE**	**UNITS**
K_{OH}	0,002	g O_2 m^{-3}
K_{OA}	0,002	g O_2 m^{-3}
K_{OG}	0,002	g O_2 m^{-3}
K_{HA}	0,01	g NH_3-N m^{-3}
K_{NO}	0,1	g NO_3-N m^{-3}
K_{Ps}	0,1	g PO_4-P m^{-3}
K_{pp}	1,0	g P m^{-3}
K_{ac}	1,0	g COD m^{-3}

Table 3.2(c): Switching functions used in the NDBEPR kinetic model (Table 3.1).

SWITCHING FUNCTION	FORMULATION	SWITCHING FUNCTION	FORMULATION
$\left[\frac{\text{H Air}}{\text{On}}\right]$	$\frac{O}{K_{OH} + O}$	$\left[\frac{\text{G Air}}{\text{On}}\right]$	$\frac{O}{K_{OG} + O}$
$\left[\frac{\text{H Air}}{\text{Off}}\right]$	$\frac{K_{OH}}{K_{OH} + O}$	$\left[\frac{\text{G Air}}{\text{Off}}\right]$	$\frac{K_{OG}}{K_{OG} + O}$
$\left[\frac{\text{A Air}}{\text{On}}\right]$	$\frac{O}{K_{OA} + O}$	$\left[\frac{P_s}{\text{Limit}}\right]$	$\frac{P_s}{K_{Ps} + P_s}$
$\left[\frac{N_s}{\text{Limit}}\right]$	$\frac{N_s}{K_{HA} + N_s}$	$\left[\frac{P_{polyP}}{\text{Limit}}\right]$	$\frac{P_{polyP}}{K_{pp} + P_{polyP}}$
$\left[\frac{1-N_s}{\text{Limit}}\right]$	$\frac{K_{HA}}{K_{HA} + N_s}$	$\left[\frac{\text{HAc}}{\text{Limit}}\right]$	$\frac{S_{bs,s}}{K_{ac} + S_{bs,s}}$
$\left[\frac{N_{o3}}{\text{Limit}}\right]$	$\frac{N_{o3}}{K_{NO} + N_{o3}}$	$\left[\frac{1-N_{o3}}{\text{Limit}}\right]$	$\frac{K_{NO}}{K_{NO} + N_{o3}}$

Table 3.3: Symbol system used in matrix (Table 3.1).

SYMBOL SYSTEM		IAWPRC
Symbol	Description	EQUIVALENT
	MAIN SYMBOLS	
S	Substate	X or S
X	Volatile solids in VSS units	X
Z	Volatile solids in COD units	X
N	Nitrogen	X or S
P	Phosphorus	X or S
O	Oxygen	S_o
f	Fractional contents	f
M	Mass	-
	SUBSCRIPTS	
B	Biological (active) mass	B
E	Endogenous mass	E
I	Inert mass	I
H	Heterotrophs	H
A	Autotrophs (nitrifiers)	A
G	PolyP organisms	-
b	Biodegradable	-
u	Unbiodegradable	-
p	Particulate	-
s	Soluble	-
o	Organic (nitrogen)	-
a	Ammonia	-
i	Influent	-
e	Effluent	-
t	Total	-
	FRACTIONAL CONTENTS	
$f_{X,YZ}$	Fraction of X which is YZ	i
$f_{XY,Z}$	Fraction of XY which is Z	i

With regard to the polyP organism description (processes 15-24), the processes, the process rate formulations and kinetic constants remain the same as in the enhanced culture BEPR kinetic model.

In the NDBEPR kinetic model, interactions between the organism populations are accommodated through the stoichiometric relationships between processes and compounds - no direct *interaction processes* have been identified. To accommodate the interactions between polyP and non-polyP organisms the stoichiometric relationships between processes and compounds carried over from the enhanced culture kinetic model required relatively minor changes.

At present the NDBEPR kinetic model is being evaluated; the evaluation still is restricted to steady state conditions for reason that extensive steady state experimental data are available (Wentzel *et al.*, 1990). This evaluation allows verification or modification of constants carried over from the ND and enhanced culture kinetic models. From the simulations undertaken thus far, it appears that the values for constants from the ND and enhanced culture kinetic models can be retained, with the exception of n_G. An important aspect of the evaluation has been to investigate the value for n_G over the wide range of configurations and conditions: The simulations for a particular system are repeated using a series of η_G values until the experimentally measured nitrate concentrations are closely predicted. From the set of η_G values obtained for the different systems, the "best" η_G value is estimated. From the simulations completed (70 of which 57 could be used to evaluate η_G), a mean value for n_G of 0,6 has been obtained, with standard deviation of the mean ± 0,03. Using the individual n_G values that give the correct nitrate concentrations for a particular system, the correlation between predicted and measured P removal for both the mixed and enhanced culture systems is shown in Fig 3.1.

A number of aspects of the NDBEPR kinetic model require further attention:

- Completion of the model evaluation under steady state conditions, in particular quantification of η_G (total data sets available 111).

- Evaluation of the model under cyclic flow and load conditions - a problem here is the lack of comprehensive experimental data for NDBEPR systems under cyclic flow and load.

- Denitrification by polyP organisms - in the NDBEPR kinetic model, from experimental observations (Wentzel *et al.*, 1989a; Clayton *et al.*, 1989; 1991) denitrification by polyP organisms is not included. However, in some instances significant P uptake in the anoxic reactors of NDBEPR systems has been observed, implying that on occasion the polyP organisms do denitrify. Biochemical assays have indicated that some *Acinetobacter* species/strains have the ability to denitrify (Lötter, 1985). At present, no conclusive judgement can be made as to what conditions will induce the presence of denitrifying polyP organisms in the NDBEPR system, and for a specified system, what proportion of the polyP organisms will have the ability to denitrify.

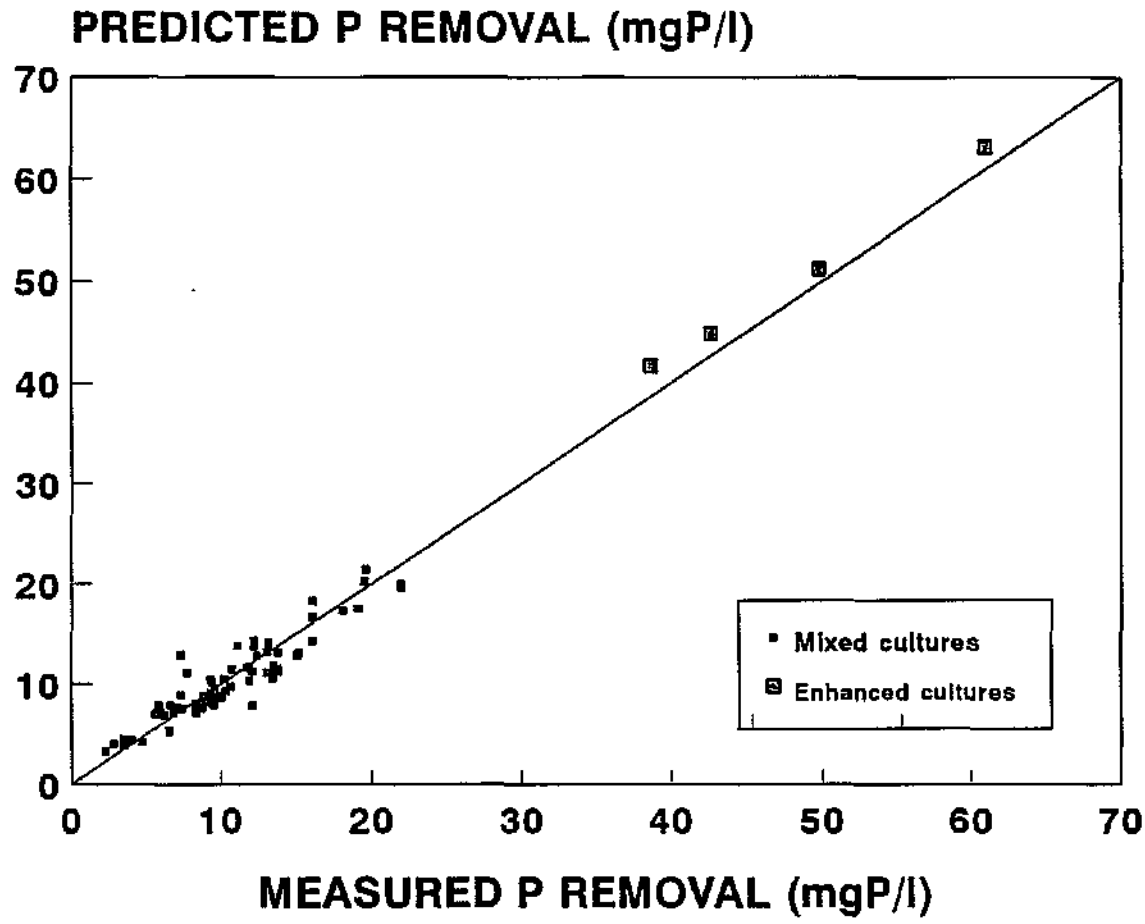

Fig 3.1: Predicted versus measured P removal; predictions using NDBEPR kinetic model, data from steady state laboratory scale systems (Wentzel *et al.*, 1990).

- Temperature dependency - no quantitative information is available on the temperature dependency of the reactions mediated by the polyP organisms (e.g. maximum specific growth rates, etc.). Qualitative indications are that the mass of P removal is relatively temperature insensitive. It is proposed to obtain more quantitative information on this aspect from research being conducted under a new parallel Water Research Commission contract on bulking in NDBEPR systems (K5/542, Jan.1993-Dec. 1995). In this research the effect of temperature on bulking in NDBEPR systems will be investigated. The measured data can be used also to evaluate the effect of temperature on P release, uptake and removal.

- Behaviour of non-polyP and autotrophic organisms under P limitation - in the model, following the experimental observations and conclusions of Wentzel *et al.* (1989a; 1989b), should soluble P become limiting the polyP organisms utilize polyP as an alternative P source for cell growth. No information is available as to the behaviour of the non-polyP and autotrophic organisms under these conditions - in the model it is assumed that growth of these organisms ceases under P limiting conditions.

- The model does not include the storage compound glycogen (Matsuo *et al.*, 1992), the effect of "G" bacteria (Cech et al., 1990), nor the effect of possible potassium or magnesium limitation (Lindrea et al., 1994).

3.4 IAWQ TASK GROUP ON MATHEMATICAL MODELLING

The Water Research Group is participating in an IAWQ Task Group on Mathematical Modelling of Wastewater Treatment Systems. The IAWQ Task Group has finalized a dynamic model for NDBEPR systems. The IAWQ Task Group presented the model in the form of two papers at the IAWQ Specialized Seminar on "Modelling and Control of Activated Sludge Systems" held in Denmark, August 1994 (Gujer *et al.*, 1994; Henze *et al.*, 1994). The Task Group is also preparing a "Scientific and Technical Report" (STR) on the model. In application of the IAWQ model by the Water Research Group at UCT, it was found that a number of the constants proposed by the Task Group had to be adjusted to reflect the behaviour observed in systems operated by the Water Research Group. These adjustments have caused that the predictions of the Task Group and UCT models are very similar. Although the UCT group are not in complete agreement with all aspects of the Task Group's proposed NDBEPR model, two suggestions emanating from the Task Group merit consideration for inclusion in the model under development at UCT:

3.4.1. Slowly biodegradable COD utilization

In the UCT NDBEPR model, slowly biodegradable COD (SBCOD) is hydrolysed/utilized under aerobic conditions and at a reduced rate under anoxic conditions. At present, hydrolysis of SBCOD under anaerobic conditions is not included in the UCT NDBEPR model - since under anoxic conditions the rate of SBCOD hydrolysis/utilization is an order of magnitude less than RBCOD substrate utilization, we were of the opinion that anaerobic SBCOD hydrolysis would be negligible. The Task Group are of the opinion that anaerobic SBCOD hydrolysis may be significant and have included this process in their model. In this

anaerobic hydrolysis process, SBCOD is hydrolysed to RBCOD, which then can be fermented/converted to short-chain fatty acids (SCFA) which are available to the polyP organisms. From the research under previous WRC contracts indirect evidence appeared to confirm that anaerobic SBCOD hydrolysis should be included in the NDBEPR kinetic models:

> Mixed liquor from the aerobic reactor of an NDBEPR system was placed into anaerobic batch reactors without addition of external substrate - P release was observed. On subsequent aeration P uptake was observed. This indicated that during the anaerobic phase SBCOD hydrolysis and fermentation were taking place resulting in SCFA sequestration and P release; in the subsequent aerobic phase the sequestered SCFA could be used for P uptake.

SBCOD hydrolysis is of crucial importance in the development of design and simulation models for NDBEPR systems. If SBCOD hydrolysis is significant in the anaerobic reactor, a substantial amount of the influent wastewater SBCOD will be hydrolysed to RBCOD which in turn will be fermented to SCFA to become available to the polyP organisms; accordingly the SBCOD will make a substantial contribution to the magnitude of BEPR. This will influence both the design and operation of BEPR systems, for example, in the effect of wastewater characteristics of BEPR, sizing and number of anaerobic reactors, inclusion of primary sedimentation, maximum BEPR achievable, etc. Due to the importance of anaerobic SBCOD hydrolysis, a preliminary experimental investigation into this aspect has been completed (Wentzel *et al.*, 1994).

The influence of the various biodegradable COD fractions of BEPR was evaluated. This was achieved by monitoring the ability of the various biodegradable COD fractions to stimulate P release in anaerobic batch tests. The P release is proportional to the amount of SCFA taken up by the polyP organisms and stored as polyhydroxyalkanoate (PHA) (Fukase *et al.*, 1982), and the subsequent aerobic P uptake (and thus P removal) is proportional to the P release (Wentzel *et al.*, 1985). Thus, monitoring P release serves as an indicator of the amount of substrate stored by the polyP organisms and the amount of P removal that will be obtained.

Four anaerobic batch tests were run in parallel in which selected biodegradable COD substrates were added to mixed liquor drawn from the 1st primary anoxic reactor of a modified UCT system (see Fig 3.2 for system layout). Selected COD substrates were: (1) tap water, (2) the short-chain fatty acid acetate, (3) raw municipal wastewater RBCOD, and (4) raw municipal wastewater SBCOD. The raw municipal wastewater RBCOD and SBCOD were prepared by centrifuging the wastewater and collecting a supernatant (predominantly RBCOD) and a pellet (only SBCOD) fraction.

Each batch test was repeated three times, typical results are shown plotted in Figs 3.3 to 3.9. From this investigation initial indications are:

- For the wastewater tested, the SBCOD does not induce P release under anaerobic conditions (Fig 3.9) and therefore will not make a significant contribution to BEPR. Accordingly, unless evidence to the contrary for the particular wastewater is available, anaerobic SBCOD hydrolysis/fermentation should be ignored in the steady state design models. Also for dynamic models including an anaerobic SBCOD hydrolysis/fermentation process, the rate should be allocated a zero or very low value

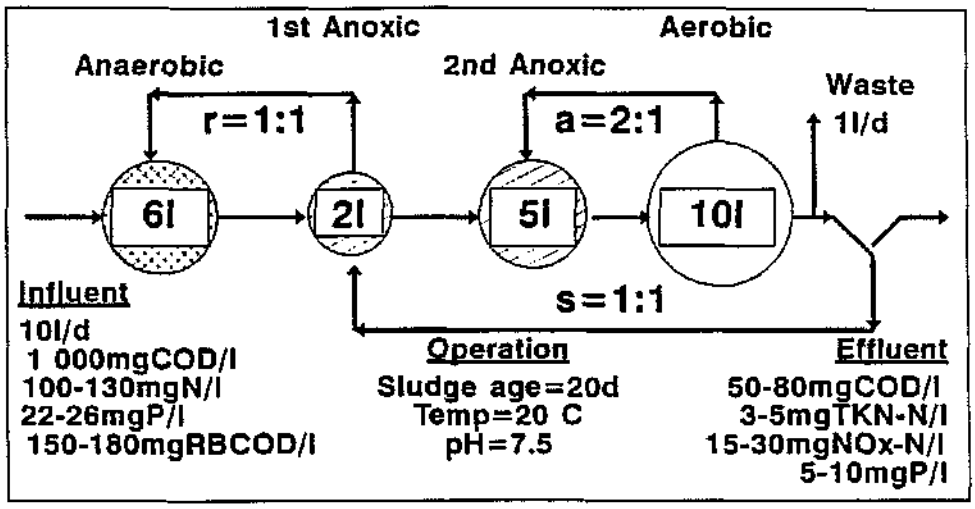

Fig 3.2: Schematic layout and operational data for parent laboratory-scale MUCT NDBEPR system.

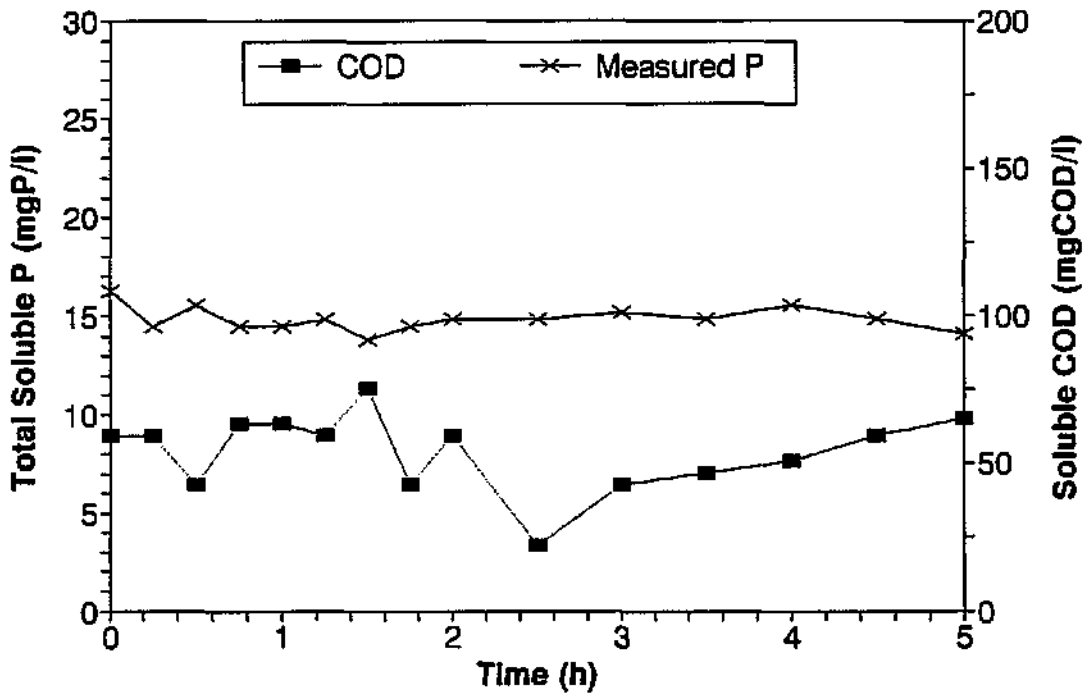

Fig 3.3: Total soluble P and soluble COD concentration time profiles for anaerobic batch test with addition of tap water to mixed liquor drawn from 1st anoxic reactor of parent system (Fig 3.2).

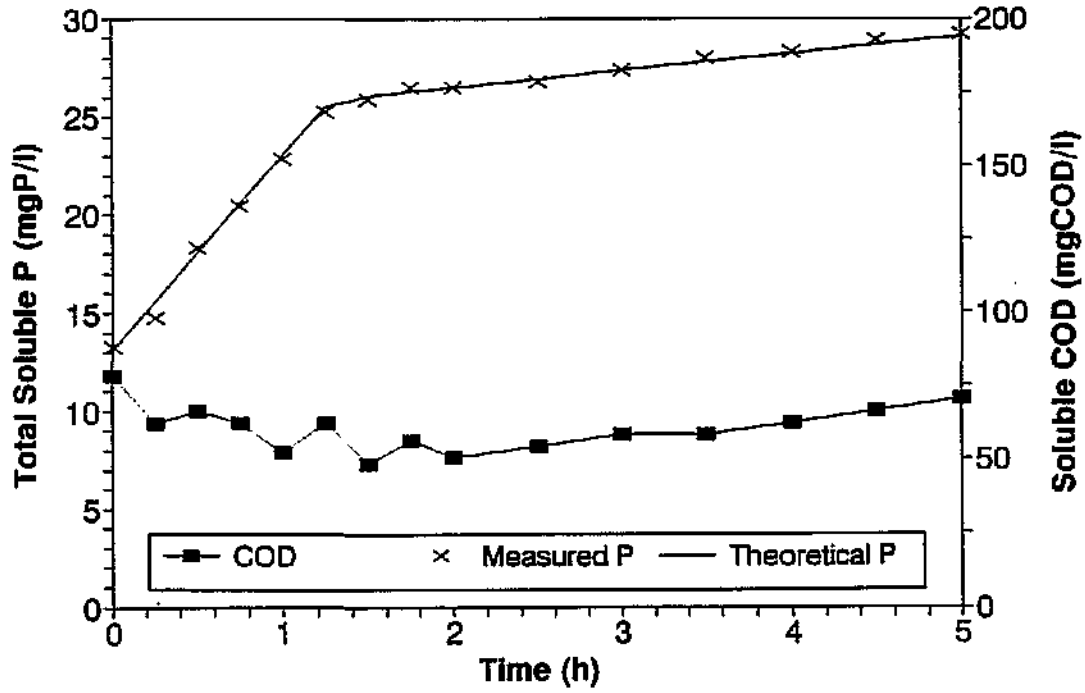

Fig 3.4: Total soluble P and soluble COD concentration time profiles for anaerobic batch test with addition of sodium acetate (30 mgCOD/ℓ) to mixed liquor drawn from 1st anoxic reactor of parent system (Fig 3.2).

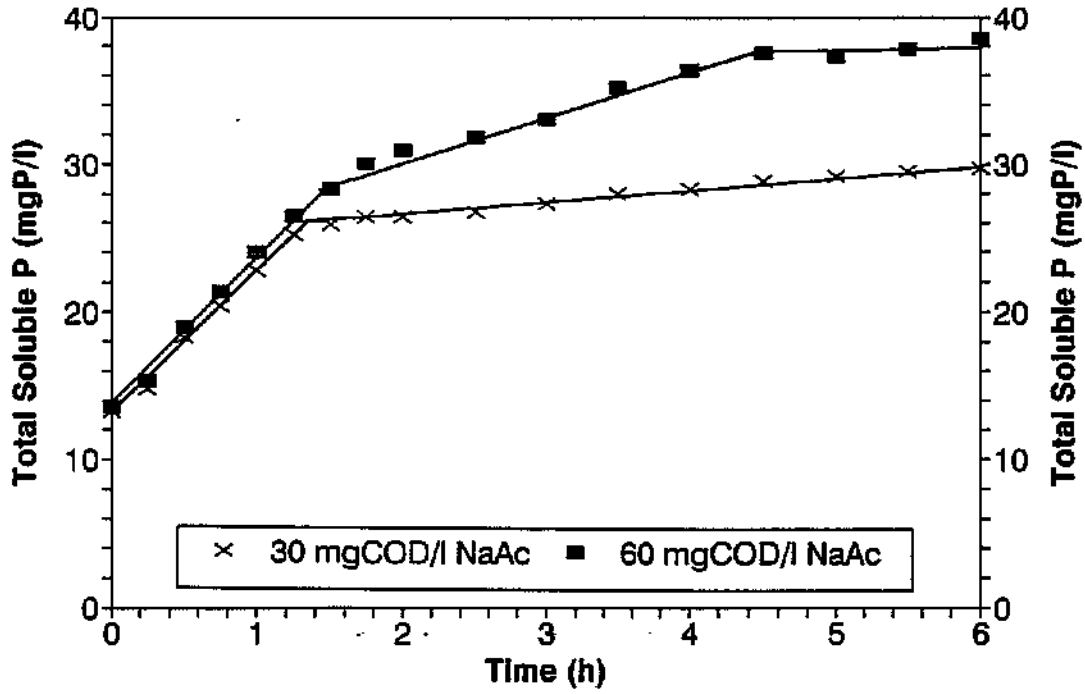

Fig 3.5: Total soluble P concentration time profiles for anaerobic batch tests with addition of sodium acetate (30 & 60 mgCOD/ℓ) to mixed liquor drawn from 1st anoxic reactor of parent system (Fig 3.2).

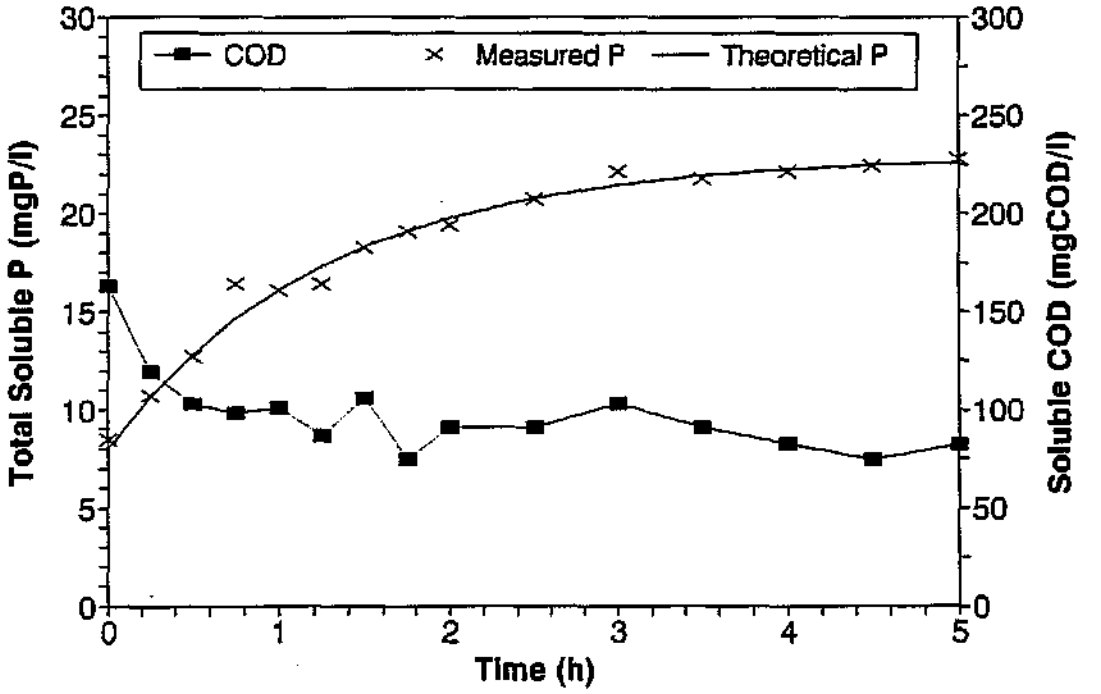

Fig 3.6: Total soluble P and soluble COD concentration time profiles for anaerobic batch test with addition of raw municipal wastewater centrifuge supernatant to mixed liquor drawn from 1st anoxic reactor of parent system (Fig 3.2). Fit to experimental data using first order kinetics.

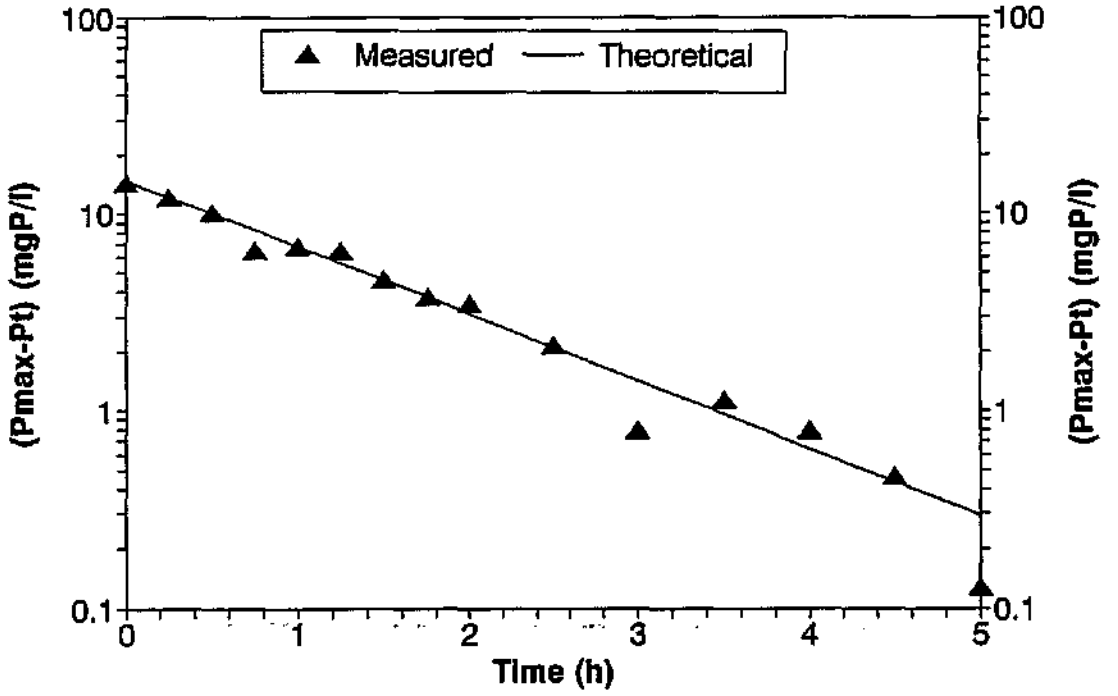

Fig 3.7: Semi-log plot of total soluble P concentration time profile in Fig 3.6. Fit to experimental data using first order kinetics.

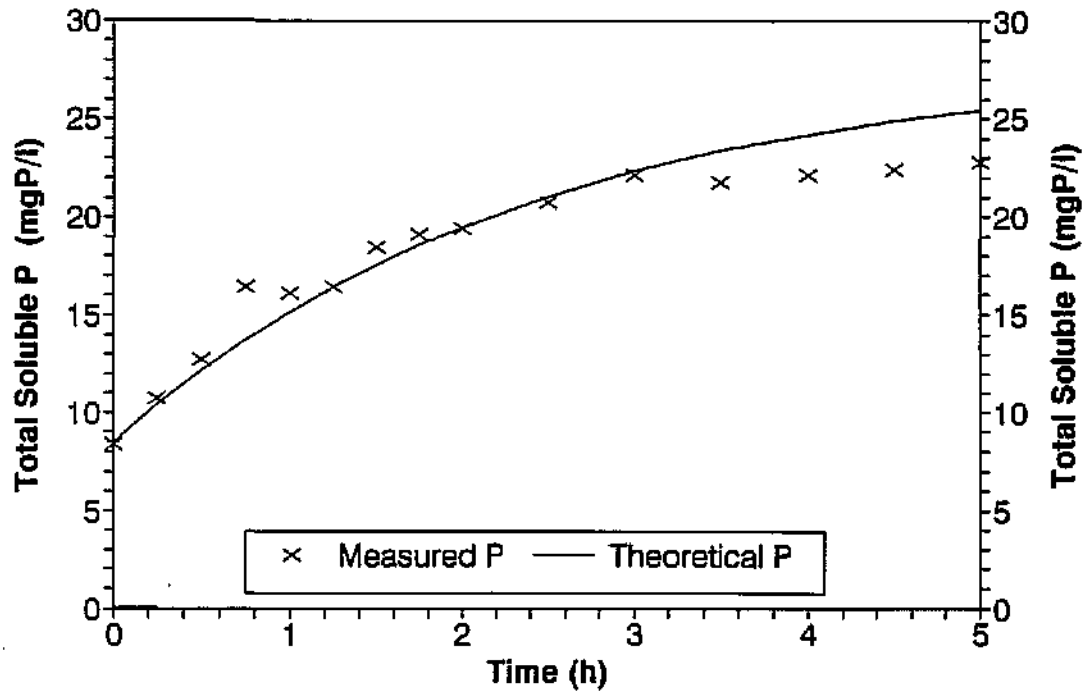

Fig 3.8: Measured and theoretical total soluble P concentration time profiles for anaerobic batch test with addition of raw municipal wastewater centrifuge supernatant to mixed liquor drawn from 1st anoxic reactor of parent system (Fig 3.2). Measured data from Fig 3.6; theoretical fit using model of Wentzel *et al.* (1985; 1990).

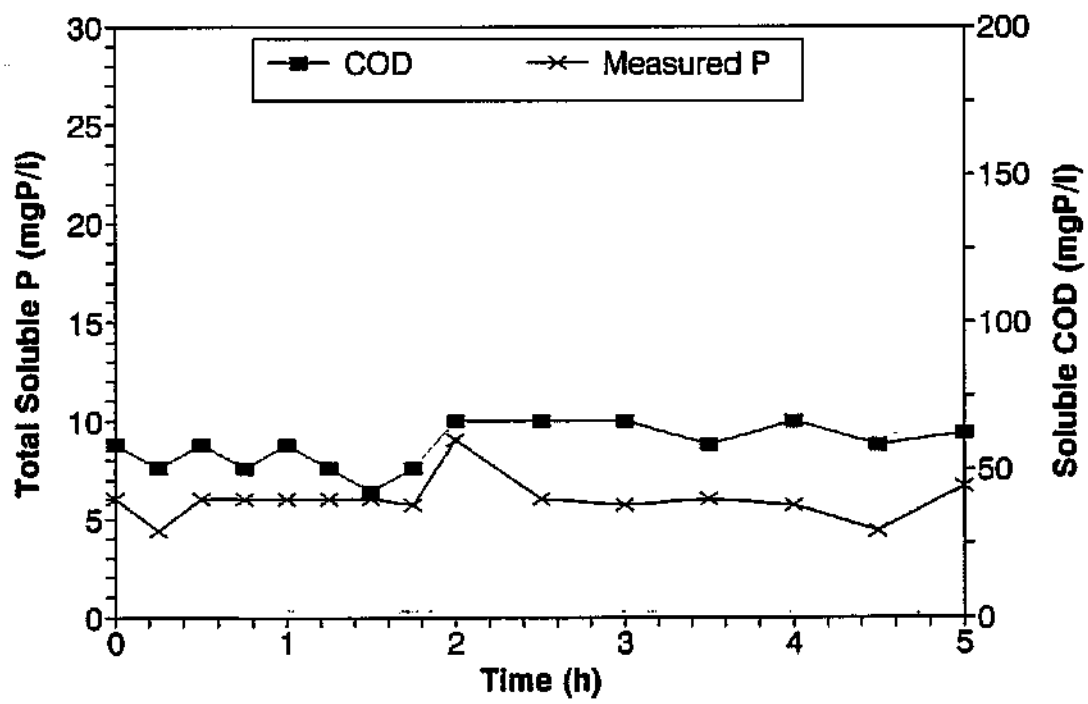

Fig 3.9: Total soluble P and soluble COD concentration time profiles for anaerobic batch test with addition of raw municipal wastewater centrifuge pellet to mixed liquor drawn from 1st anoxic reactor of parent system (Fig 3.2).

so that it makes a negligible contribution to the BEPR; in the absence of conclusive evidence indicating a high rate, a high rate would distort the BEPR to unrealistically high values. This observation is in agreement with the experience at laboratory- and full-scale with South African municipal wastewaters, where the magnitude of BEPR has been strongly linked to the RBCOD fraction of the wastewater (e.g. Siebritz *et al.*, 1983; Nicholls *et al.*, 1985; Wentzel *et al.*, 1985; Pitman, 1991).

- For modelling of BEPR, the wastewater RBCOD fraction must be subdivided into two subfractions, short-chain fatty acids (SCFA) and fermentable RBCOD (F-RBCOD). Under anaerobic conditions, the SCFA subfraction can be taken up directly by the polyP organisms and stored as polyhydroxyalkanoate (PHA), whereas the F-RBCOD subfraction undergoes fermentation to SCFA, the fermentation mediated by the "normal" (non-polyP) heterotrophs.

- For SCFA, the rates of SCFA uptake and P release are independent of SCFA concentration, that is zero order with respect to SCFA concentration (Figs 3.4 and 3.5). The kinetics of these processes can be modelled using the proposals of Wentzel *et al.* (1990; 1992).

- For F-RBCOD, the rate of fermentation to SCFA is first order with respect to F-RBCOD concentration (Figs 3.6, 3.7 and 3.8). The kinetics for this process proposed by Wentzel *et al.* (1985; 1990; 1992) can be retained.

This investigation has been based on a typical raw (unsettled) municipal wastewater from one source in South Africa. Similar studies will have to be undertaken with other wastewaters to determine whether the results can be applied generally. Past experience in characterization of wastewaters indicates that considerable caution should be exercised in simply transferring results linked to wastewater composition from one situation to another.

3.4.2 Link between COD, N and P fractions

At present, in the UCT ND and NDBEPR models, and in the Task Group ND model, the link between COD, N and P is not firmly established. Separate processes have been included for the transformations between different forms of these compounds. For example, one process is included for RBCOD utilization and a separate process for conversion of biodegradable soluble organic nitrogen to ammonia. This has resulted in inconsistencies in the current models. For example, soluble biodegradable organic nitrogen may be present without a corresponding soluble biodegradable COD. The Task Group has proposed that the transformations between the different N and P forms be linked directly to the transformations of the COD as follows:

> Each COD fraction has associated with it a certain N and P content. When the COD fractions are transformed from one form to another, the N and P will undergo corresponding transformations.

This proposal has considerable merit and needs to be investigated for inclusion in the UCT models. This will require that N and P content of each of the COD fractions be determined.

3.5 CLOSURE

A general kinetic model for simulation of the dynamic behaviour of the NDBEPR activated sludge system has been developed by the UCT group and published in the literature (Wentzel *et al.*, 1992). A number of aspects in the model have been identified to require further attention. It is envisaged that work on these aspects will continue, either under parallel WRC contracts (e.g. K5/542 where the effect of temperature on bulking is being studied, and the measured data can be used also to evaluate the effect of temperature on P release, uptake and removal) or by using information published in the literature. However, it is unlikely that a final solution ever will be attained - the nutrient removal plant is a complex system in which the different system elements, processes and compounds often exhibit interaction in a complex manner; resolution of one problem may bring into focus, or create, another problem.

CHAPTER 4

REVISION OF NDBEPR SYSTEM DESIGN MANUAL

4.1 INTRODUCTION

With the introduction of legislation in 1980 limiting P concentrations in effluents discharges from municipal wastewater treatment works, intensive research efforts were directed by the South African research community towards biological P removal in the activated sludge system. By 1984 the information gained on the activated sludge system was sufficient to enable a manual to be written in collaboration between the Water Research Group at UCT, the Johannesburg City Council and the National Institute for Water Research, for the design of aerobic (COD removal and nitrification), anoxic/aerobic (COD removal, nitrification and denitrification) and anaerobic/anoxic/aerobic (COD removal, nitrification and denitrification and biological excess P removal) activated sludge systems, the last one being designated as the nutrient (N & P) removal system (WRC, 1984).

The design manual of 1984 provided credible guidelines and recommendations for COD removal, nitrification and denitrification based on sound microbiological/biochemical and kinetic behavioural patterns. Design guidelines for BEPR, however, still contained an appreciable empirical content. This empirical content brought a measure of uncertainty and insecurity to the design of BEPR systems. Since 1984, research at UCT under WRC sponsored contracts, together with research by the Johannesburg City Council and other research groups (both National and International), has significantly broadened understanding of BEPR and of its microbiology/biochemistry and kinetics. This information is sufficient to warrant revision and updating of the 1984 design manual, to provide improved guidelines for the design of nutrient removal plants.

In both the previous (K5/251, Wentzel *et al.*, 1991) and current (K5/356) consolidation contracts, revision of the 1984 design manual (WRC, 1984) has been considered to be a priority. Revision of the design manual is being undertaken in cooperation with Mr A Pitman and Dr L Lötter of the Johannesburg City Council Group, and any persons delegated by them. Before the revision could be commenced, it was found necessary to undertake experimental investigations into areas where information was incomplete or not available, for example, denitrification in NDBEPR systems (Clayton *et al.*, 1989, 1991), importance of SBCOD in BEPR (Wentzel *et al.*, 1994). Under the previous and current consolidation contracts, these investigations have essentially been completed, and work has commenced on writing the revised design manual. Progress in this revision, and work still to be completed, is set out briefly below.

4.2 STRUCTURE FOR REVISED DESIGN MANUAL

Chapter 1: Fundamentals of Biological Behaviour

This Chapter has been expanded to include bioenergetic concepts. The bioenergetics will promote increased understanding of activated sludge system behaviour amongst users of the manual, and provide a theoretical framework for development of design equations that will complement the existing kinetic based development.

Chapter 2: Microbiological Aspects

This was Chapter 9 in the previous manual, which is to be updated to include the latest information on polyP and filamentous organisms.

Chapter 3: Characterization of Municipal Wastewaters

This Chapter has been updated to include information on this topic that has become available locally and overseas.

Chapter 4: Influence of Biological Behaviour and Wastewater Characteristics on Activated Sludge System Design

This Chapter has been revised to incorporate the latest information on denitrification and BEPR.

Chapter 5: Wastewater Modification to Enhance Nutrient Removal

This is a new Chapter to be included in the manual and will be written jointly by the UCT and Johannesburg groups.

Chapter 6: Carbonaceous Material Removal

The Chapter can be extracted from the previous design manual with relatively minor revision.

Chapter 7: Nitrification

This Chapter can be extracted from the previous design manual with relatively minor revision.

Chapter 8: Biological Denitrification in Nitrogen Removal Systems

On evaluation of Chapter 8, it was concluded that for design of ND systems, the procedures set out the 1984 design manual remain valid. However, the procedures make extensive use of graphical methods for solution of design problems. With the widespread availability of personal computers, it was considered essential that numerical methods for solution be developed (to complement the graphical methods) as these lend themselves readily to incorporation in computer programs and spreadsheets. Considerable effort has been expended in developing the numerical methods, and these now have been completed. The methods have been extensively evaluated and tested, and found to be satisfactory.

Chapter 9: Biological Excess Phosphorus Removal

The design procedures for P removal in the 1984 design manual, are empirically based:

> The active sludge is considered as a whole, to constitute a surrogate sludge with a propensity for P removal; variation in BEPR between different systems is hypothesized to be due to changes in the propensity for P removal of this surrogate sludge, caused by changes in influent RBCOD concentration, anaerobic mass fraction

> and/or nitrate discharge to the anaerobic reactor. The propensity for P removal is formulated empirically in terms of the three key parameters, and the mass of sludge wasted per day.

As noted in the 1984 design manual, the empirical nature of the design procedures causes that,

> "its use must be limited strictly to within the ranges of process parameters and wastewater characteristics listed",

and that,

> "the parametric model is unsatisfactory from a scientific point of view because it is independent of any formal hypothesis on the biological mechanisms driving the process."

Accordingly, under the previous and current consolidation contracts intensive research effort has been directed to developing steady state models and design procedures for BEPR that have a more fundamental basis. A fundamentally based steady state design model has been developed that incorporates mixed cultures of polyP and non-polyP organisms, i.e. cultures that develop in BEPR systems treating municipal wastewaters. This model incorporates the latest information on denitrification in NDBEPR systems (Clayton *et al.*, 1991). Still to be completed is to incorporate this model in a design procedure for NDBEPR systems. This task is currently receiving attention. Rewriting of the design manual Chapter 9 has commenced, even though the NDBEPR design procedures are incomplete. As the NDBEPR design procedures evolve, Chapter 9 will be appropriately expanded.

Chapter 10: Filamentous Organism Bulking in Nutrient Removal Plants

It is proposed that this be included as a new Chapter. The latest hypothesis for causes of filamentous bulking in nutrient removal plants (Casey *et al.*, 1993) will be outlined, and the hypothesis used to identify operational procedures to minimize bulking.

Chapter 11: Practical Design and Operation of Nutrient Removal Systems

Johannesburg City Council Group, with their wealth of experience in this field, will be principal authors on this Chapter. The Chapter will incorporate the latest research findings of the Johannesburg City Council Group.

4.3 CURRENT STATUS OF DESIGN MANUAL REVISION

Chapters 1, 3 and 4 have been completed. Drafts have been prepared of Chapters 6, 7, 8, 10 and 11. Drafts are in preparation for Chapters 2 and 9. Although the current consolidation contract has ended, work on completing the design manual will continue.

CHAPTER 5

CHARACTERIZATION OF MUNICIPAL WASTEWATERS

5.1 INTRODUCTION

In the original proposal for this Consolidation contract, one of the principal aims was to conduct experimental work to evaluate the effect of incorporating fixed media in the aerobic zone of NDBEPR systems, with the view to reducing the system sludge age yet maintaining virtually complete nitrification. During the course of the contract it was noted that this might lead to a duplication of research effort since Professor Pretorius at the University of Pretoria is conducting research in this area. Accordingly, in consultation with the members of the Steering Committee it was decided to change this aim to

- Develop and refine methods for characterization of municipal wastewaters.

The need for this project arose from two sources:

- The objectives for the activated sludge system have expanded to include progressively COD removal, nitrification, denitrification and biological excess P removal (BEPR). Concomitantly, to provide reliable designs and predictions of expected system performances, design procedures and mathematical models of increasing complexity have been proposed (Dold *et al.*, 1980; Van Haandel *et al.*, 1981; WRC, 1984; Henze *et al.*, 1987; Dold *et al.*, 1991; Wentzel *et al.*, 1990; Wentzel *et al.*, 1992). In terms of the framework of these design procedures and models, it is necessary to divide the influent COD, TKN and more recently P into a number of fractions. At present, procedures to quantify these fractions are either too difficult or elaborate, or simply not available.

- The original UCT model (Dold *et al.*, 1980; Van Haandel *et al.*, 1981) did not consider heterotroph or autotroph active biomass to be present in the influent and these fractions were therefore not included in the wastewater characterization; for municipal wastewaters in South Africa, the sewers generally are short (retention < 6 hours) and anaerobic, and were considered unlikely to support active biomass generation. Further, simulations with the UCT model appeared to support this supposition. However, investigators in Europe have indicated that European municipal wastewaters can contain a significant heterotroph active biomass fraction (Henze, 1989), up to 20% of the total COD (Kappelar and Gujer, 1992). Seeding of this influent biomass to the activated sludge system can have a significant influence on modelling and design. Also, the presence of autotroph active biomass in the influent will have a significant influence on modelling and design of nitrification.

  From simulation studies, quantifying the influent active biomass fractions by simulation of observed activated sludge system responses leads to the situation that it is not possible to separate efficiently the kinetics and the influent active biomass - these tend to act in a compensatory fashion, for each selected influent active biomass a set of kinetic constants (or new kinetic behaviour) can be obtained to give consistency between observed and simulated results. It would appear essential, therefore, that the active biomass present in wastewaters be quantified independently.

In this project, the issues raised above are being addressed. Attention has been focused of measurement of influent carbonaceous material fractions including the heterotrophic active biomass.

5.2 CARBONACEOUS MATERIAL FRACTIONS

Characterization of the carbonaceous material in the influent is in terms of the chemical oxygen demand (COD). Research by the UCT Group has indicated a division shown diagrammatically in Fig 5.1.

Biodegradable and unbiodegradable fractions
The first division of the influent COD (S_{ti}) is into biodegradable COD (S_{bi}) and unbiodegradable COD (S_{ui}). Each of these fractions is subdivided further into two sub-fractions:

Unbiodegradable sub-fractions: The influent unbiodegradable COD (S_{ui}) is divided into two fractions, unbiodegradable soluble COD (S_{usi}) and unbiodegradable particulate COD (S_{upi}). Both are hypothesized to be unaffected by biological action in the system. The S_{us} passes out in the secondary settling tank overflow; the S_{up} is enmeshed in the sludge mass and accumulated in the system as inert volatile suspended solids (VSS). At steady state the mass of S_{up} entering the system in the influent will be balanced by the mass of inert VSS leaving via the sludge wastage stream. From a mass balance, the mass of inert VSS in the system will equal the influent mass of S_{up} per day multiplied by the system sludge age.

Biodegradable sub-fractions: The influent biodegradable COD (S_{bi}) is divided into two fractions, readily biodegradable COD (RBCOD, S_{bsi}) and slowly biodegradable COD (SBCOD, S_{bpi}). The RBCOD is hypothesized to consist of simple soluble molecules that can be absorbed readily by the organism and metabolized for energy and synthesis, whereas the SBCOD is assumed to be made up of particulate/colloidal/complex organic molecules that require extracellular enzymatic breakdown prior to absorption and utilization.

It has become apparent from recent research that active biomass in the wastewater also should be included in the characterization of the COD.

5.3 EXPERIMENTAL INVESTIGATION INTO QUANTIFYING INFLUENT COD FRACTIONS

5.3.1 Background

In a review of the literature on methods available for quantifying the influent wastewater COD fractions it was apparent that existing procedures are either too elaborate or approximate, or are sometimes not even available. Of the methods available, two appeared to hold potential for development, the flocculation-filtration method of Mamais *et al.* (1993) to quantify RBCOD and the batch test method of Kappelar and Gujer (1992) to quantify heterotrophic active biomass.

In this research project the intention is to evaluate these two methods and to develop new methods to quantify wastewater COD fractions. In this regard a number of tasks have been

identified:

- Development of batch test methods to determine wastewater RBCOD and heterotroph active biomass.

- Evaluation of results from the batch test method against those from conventional bioassay methods.

- Evaluation of the flocculation/filtration method of Mamais *et al.* (1993) to measure RBCOD.

- Extension of the batch test method to quantify the remaining COD fractions; unbiodegradable soluble, unbiodegradable particulate and slowly biodegradable.

A detailed report on progress in completing these tasks is in preparation (Mbewe *et al.*, 1995). A brief summary is given below.

5.3.2 Batch test for measurement of RBCOD and heterotroph active biomass

A batch test method has been developed to determine the two influent wastewater COD fractions, heterotroph active biomass and readily biodegradable COD (RBCOD). Details of this batch test method procedure are presented in Mbewe *et al* (1995) and Wentzel *et al.* (1995). In the batch test method, a sample of raw municipal wastewater is placed in a stirred batch reactor - no mixed liquor seed is added to the batch test. The batch test is operated aerobically at a constant temperature (20° C) and pH (7,5), and the oxygen utilization rate (OUR) response monitored with time, for approximately 12 hours. A typical OUR-time profile is shown plotted in Fig 5.2. From the OUR-time response the influent wastewater heterotroph active biomass and RBCOD concentrations can be determined. The method has advantages over previous methods in that:

- The experimental procedure is relatively simple.

- No mixed liquor acclimatized to the wastewater is required.

- The only independent constants required for calculation are the heterotrophic yield (Y_{ZH}) endogenous residue fraction for heterotrophic active biomass (f), and specific death rate (b_H): Dosing the batch test with known concentrations of acetate showed that the standard value for Y_{ZH} in the literature Y_{ZH} = 0,666 mgCOD/mgCOD; Dold and Marais, 1986) can be accepted; the batch test procedure is relatively insensitive to the value for b_H and f. All other constants required for calculations are obtained from the experimental data.

5.3.3 Evaluation of batch test method for RBCOD and heterotroph active biomass

Several batch tests were conducted on wastewaters from two sources, Borchard's Quarry and

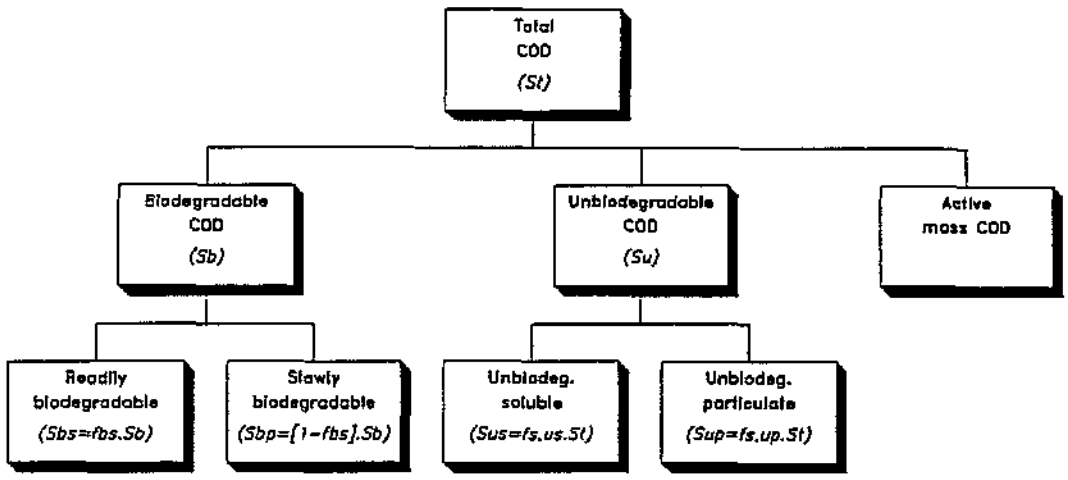

Fig 5.1: Division of the influent COD into its constituent fractions (Dold *et al.*, 1991).

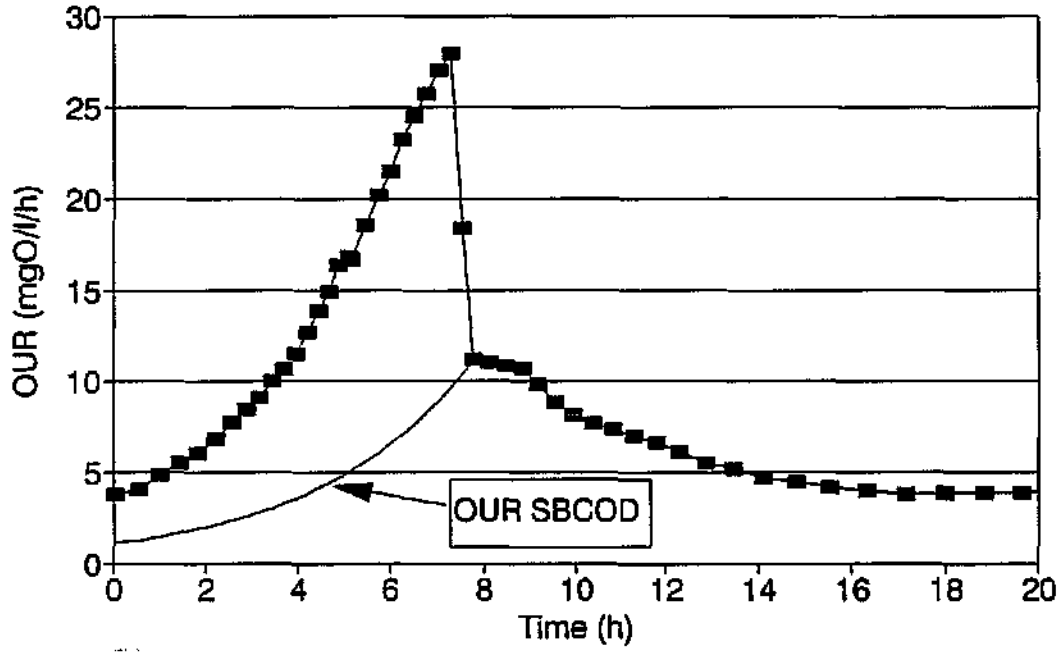

Fig 5.2: Typical oxygen utilization rate (OUR) response with time for aerobic batch test on raw municipal wastewater from Mitchell's Plain (Cape Town).

Mitchell's Plain Treatment Plants (Cape Town, South Africa). Results from these tests indicate that:

- COD recoveries in the batch test generally are good, the majority failing in the range of 90 - 108%, indicating the reliability of the measurements.

- The RBCOD concentrations measured in the batch test correlate closely with those measured using the conventional flow-through square wave method of WRC (1984) and Ekama *et al.* (1986): For each batch of sewage, probability statistic plots of the data were constructed for the batch test and square wave methods, to test that the data were normally distributed and to determine the mean and standard deviation; for example see Fig 5.3 (a and b). For the different sewage batches, the means, number of tests and the standard deviation of the means are listed in Table 5.1 for the two methods. In Fig 5.4 (a and b), for each batch of sewage the means of the RBCOD derived from the batch test are plotted against the means derived from the conventional test; good correlation is obtained.

- Although heterotroph active biomass concentration obtained from the batch test could not be compared to conventional tests (no such tests are available), the values measured in the batch test correctly reflect changes in heterotroph active biomass arising from Wastewater Treatment Plant operation.

- The values for the kinetic constants derived from the batch test (i.e. maximum specific growth rate of heterotrophs of RBCOD, $\hat{\mu}_H$, and maximum specific growth rate of heterotrophs on SBCOD, K_{MF}) differ from those in the literature for activated sludge. Most probably a population develops in the activated sludge system that differs appreciably from that in the wastewater since the conditions in the wastewater (high COD, low active biomass) differ significantly from those in the activated sludge system (low COD, high active biomass). Accordingly, it is unlikely that the values for the constants derived from the batch test will be of much value in modelling and design of activated sludge systems - their use is restricted to the batch test to derive estimates for RBCOD.

- For wastewaters from both Borchard's Quarry and Mitchell's Plain, autotroph active biomass could not be detected in the batch test.

5.3.4 Evaluation of flocculation filtration method to determine RBCOD

A flocculation-filtration method has been developed by Mamais *et al.* (1993) to quantify the influent RBCOD concentration. In this method, the inclusion of the flocculation step prior to filtration appears to overcome the problem of correct selection of filter pore size inherent in other physical methods (Mbewe *et al.*, 1995). In the Mamais *et al.* method, raw wastewater and long sludge age activated sludge system effluent are flocculated using zinc sulphate with the pH adjusted to 10,5. The flocculated wastewater and effluent are then filtered through 0,45μm filter papers. The difference between the COD of the filtrates of the raw wastewater and the effluent gives the RBCOD concentration.

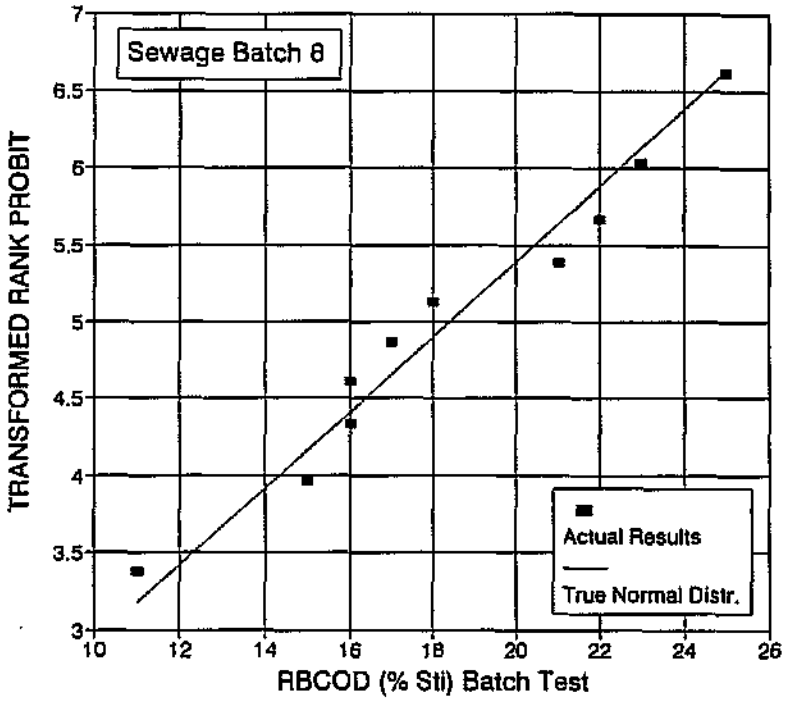

Fig 5.3a: Probability statistic plot of RBCOD as percentage of total COD (S_{ti}) for values derived from the batch test for one sewage batch from Borcherds Quarry Treatment Plant.

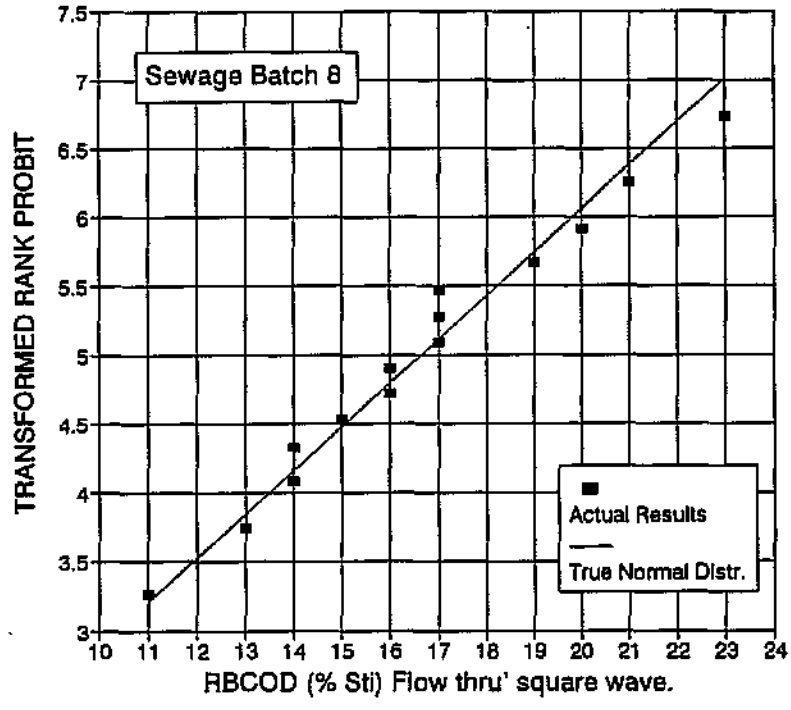

Fig 5.3b: Probability statistic plot of RBCOD as percentage of total COD (S_{ti}) for values derived from the flow through square wave test for one sewage batch from Borcherds Quarry Treatment Plant.

Table 5.1: Mean RBCOD, number of tests and standard deviation of the mean for the batch test and flow through square wave methods for the different sewage batches. Wastewater obtained from Borcherds Quarry and Mitchell's Plain Treatment Plants.

Sewage Batch	Dates of Test	MEAN RBCOD (% of TOTAL COD)					
		SQUARE WAVE			BATCH TEST		
		mean RBCOD	no.of tests	Std.dev of mean	mean RBCOD	no.of tests	Std.dev of mean
1	Jul 12-28	21	11	1.2	20	5	2.0
2	6-17 August	-	-	-	11	5	0.7
3	Aug 25-Sep 3	15	5	1.6	15	8	0.9
4	Sep 8-13	17	7	1.6	20	8	0.9
5	16-23 Sept	20	6	0.9	21	6	0.9
6	Sep 26-Oct 3	21	7	1.1	23	6	0.4
7	Oct 5-14	18	7	1.4	18	9	1.0
8	Oct 16 -30	17	14	0.8	18	10	1.4
9	Oct 31-Nov 8	18	7	1.1	17	7	0.9
10a	Jan 21-Feb 2	22	6	0.5	-	-	-
10	Feb 4-13	18	10	1.1	17	9	1.2
11	feb17-mar 2	18	12	1.4	17	7	0.7
12	Mar 18-28	19	9	1.6	19	5	1.1
13	Apr 1- 13	20	12	1.1	25	10	0.8
14	Apr 19-May 15	17	10	1.8	19	9	0.9
15	May 19-31	-	-	-	19	8	1.0
16	12 June-1 July	21	17	0.6	22	7	1.9
17	july2-21	23	17	0.7	27	10	0.6
18	jul25-aug9	20	8	1.2	26	11	1.5
19	aug12-30	24	10	1.1	24	9	1.5
20	sept1-15	21	6	1.9	22	10	1.1
21	sept 16-2 oct	21	10	1.1	20	8	0.8
22	12-20 Oct	-	-	-	20	4	0.5
23	21 Oct-3 Nov.	19	9	1.9	18	8	1.3

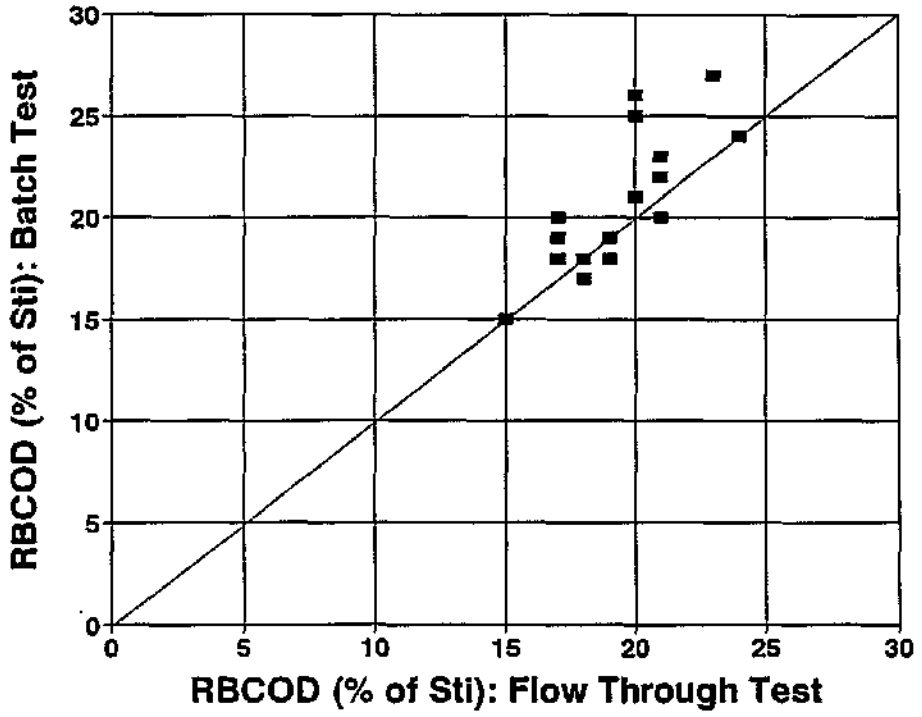

Fig 5.4a: RBCOD (as % of total COD, S_{ti}) derived from the batch test versus that derived from the flow through square-wave method. Each data point is the mean of a number (see Table 5.1) of tests on one batch of wastewater. Wastewater obtained from Borcherds Quarry and Mitchell's Plain Treatment Plants.

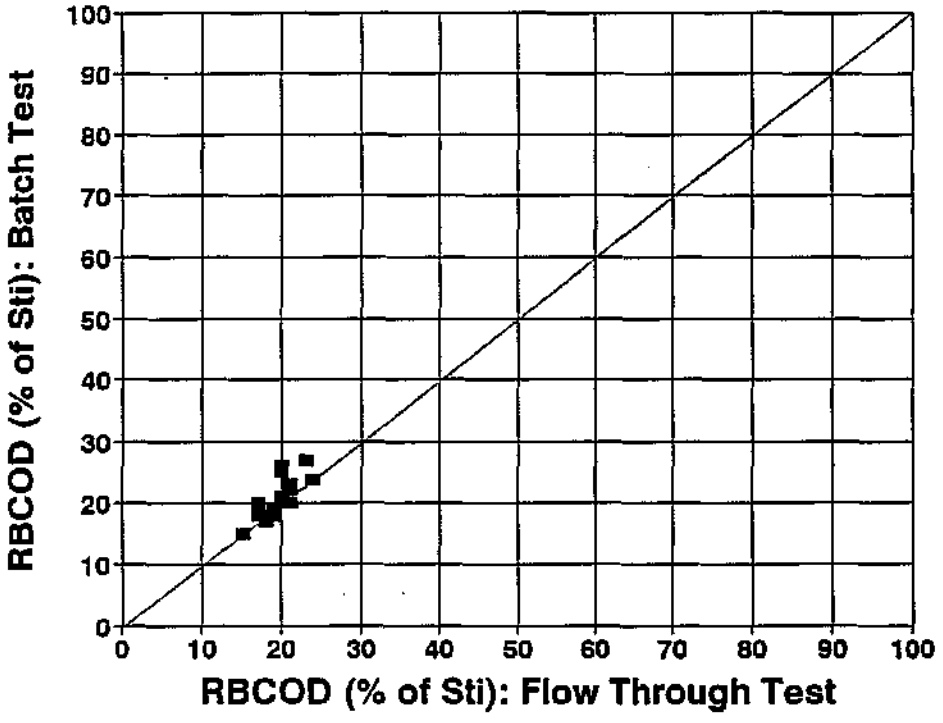

Fig 5.4b: Duplicate plot of Fig 5.4a but with axes extended to 100%.

In preliminary tests it was found that the zinc sulphate flocculant recommended by Mamais *et al.* could be replaced with aluminium sulphate - this has the advantage that no pH adjustment is necessary. The physical flocculation-filtration method was evaluated by comparing the RBCOD concentration measured with this method with those from the batch test and "standard" flow-through square wave methods.

Several batches of wastewater from two sources were tested, Borchard's Quarry and Mitchell's Plain (Cape Town). For each batch of sewage, probability statistic plots of data were constructed for the flocculation-filtration and square wave methods, to test that the data were normally distributed and to determine the mean and standard deviation; for example see Fig 5.5 (a and b). For the different sewage batches, the means, number of tests and the standard deviation of the means are listed in Table 5.2 for the two methods. For each batch, in Fig 5.6a the mean RBCOD from the flocculation-filtration method are plotted against those from the square wave test; results correlate reasonably. In Fig 5.6b the mean RBCOD from the flocculation-filtration method are plotted against those from the batch test; again results correlate closely.

For the flocculation-filtration test, Mamais *et al.* recommend 0,45μm filter papers; replacement of these filter papers with glass fibre filter papers was also evaluated. Results for RBCOD from flocculation glass fibre filtration are listed in Table 5.2 also. In Fig 5.7, filtrate COD concentrations for both influent and effluent samples with 0,45μm and glass fibre filters are shown plotted against each other; close correlation is obtained. In Fig 5.8 RBCOD is a percentage of total COD determined using glass fibre filter papers is plotted against that using 0,45μm filter papers; close correlation is obtained.

From this series of tests it can be concluded that:

- The zinc sulphate flocculant recommended by Mamais *et al.* (1993) can be replaced with aluminium sulphate. This has the advantage that pH adjustment after flocculation is not required.

- Measured RBCOD correlate closely with those from the conventional flow-through square-wave method (WRC, 1984; Ekama *et al.*, 1986) and the batch test method.

- The method is relatively simple and easy to apply but requires independent determination of unbiodegradable soluble COD, from effluent samples which may not always be available.

- The 0,45μm filters recommended by Mamais *et al.* (1993) can be replaced with glass fibre filters (Whatman's GF/C) to reduce costs, without any loss in accuracy.

5.3.5 Extension of batch test to determine unbiodegradable soluble COD

The success with the Mamais *et al.* (1993) method to determine RBCOD indicated that perhaps the method could be applied to the batch test to determine unbiodegradable soluble COD: From the OUR-time profiles in the batch test it is evident that the RBCOD is depleted after ± 10 hours, and after this time the only soluble COD remaining should be

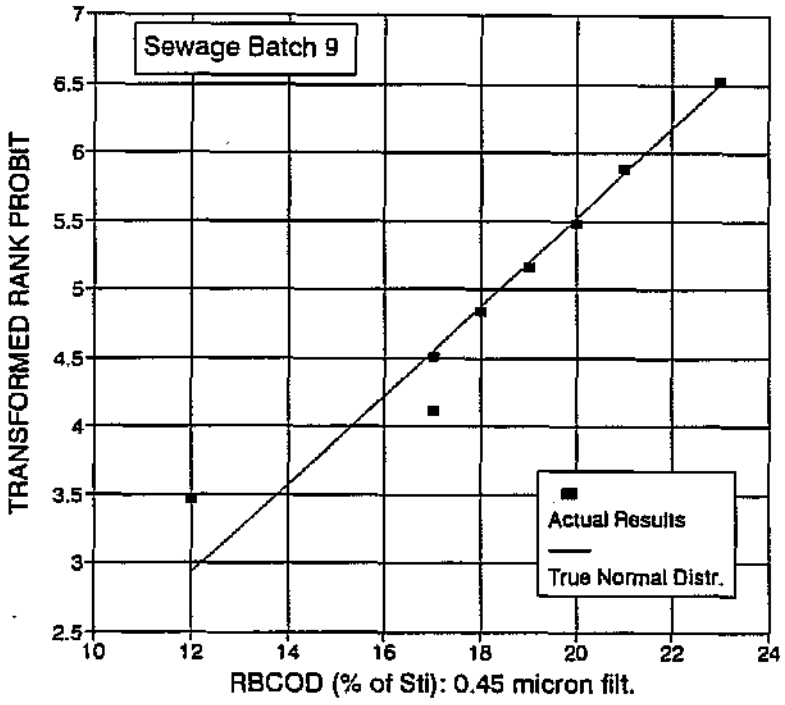

Fig 5.5a: Probability statistic plot of RBCOD as percentage of total COD (S_{ti}) for values derived from the 0,45μm flocculation-filtration test for one sewage batch from Borcherds Quarry Treatment Plant.

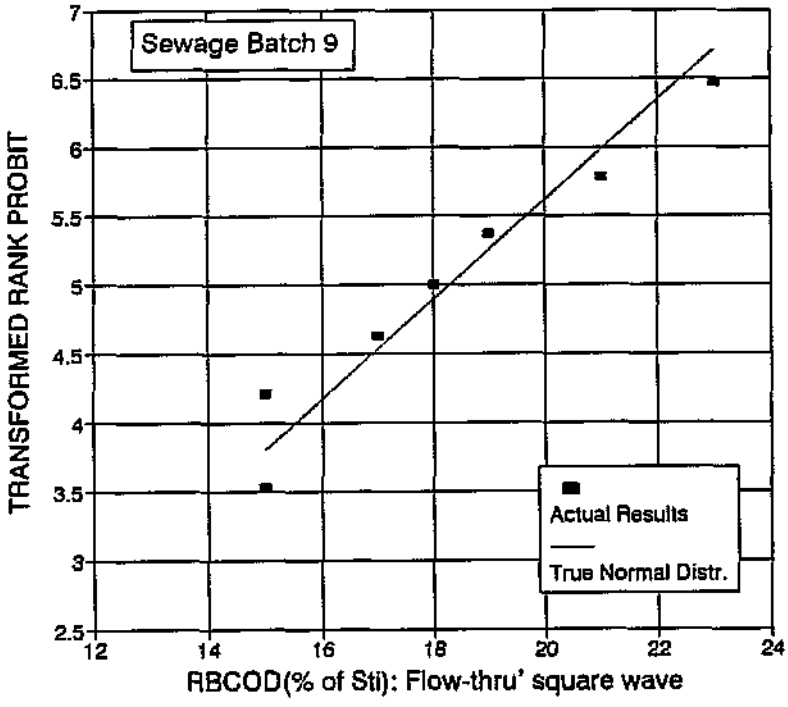

Fig 5.5b: Probability statistic plot of RBCOD as percentage of total COD (S_{ti}) for values derived from the flow-through square-wave test for one sewage batch from Borcherds Quarry Treatment Plant.

Table 5.2: Mean RBCOD, number of tests and standard deviation of the means for the flow-through square-wave test, batch test and the glass fibre and 0,45μm flocculation-filtration methods for the different sewage batches. Wastewater obtained from Borcherds Quarry and Mitchell's Plain Treatment Plants.

Sewage	Dates of	MEAN RBCOD (as % of TOTAL COD)											
		SQUARE WAVE			BATCH TEST			GLASS FIBRE FILTRATION			0.45 FILTRATION		
Batch	Tests	mean of RBCOD	No. of tests	Std. dev. of mean	mean of RBCOD	No. of tests	Std. dev. of mean	mean of RBCOD	No. of tests	Std. dev. of mean	mean of RBCOD	No. of tests	Std. dev. of mean
1	Jul 12-28	21	11	1.2	20	5	2.0	-	-	-	-	-	-
2	6-17 August	-	-	-	11	5	0.7	-	-	-	-	-	-
3	Aug 25-Sep 3	15	5	1.6	15	8	0.9	-	-	-	17	7	1.9
4	Sep 8-13	17	7	1.6	20	8	0.9	-	-	-	17	6	1.7
5	16-23 Sept	20	6	0.9	21	5	0.9	19	7	1.1	18	7	1.4
6	Sep 26-Oct 3	21	7	1.1	23	5	0.4	19	6	1.1	18	6	1.6
7	Oct 5-14	18	7	1.4	18	6	1.0	17	6	0.9	17	7	0.9
8	Oct 16 -30	17	14	0.8	18	10	1.4	14	5	1.0	14	5	1.0
9	Oct 31-Nov 8	18	7	1.1	17	7	0.9	18	8	1.3	18	8	1.2
10a	Jan 21-Feb 2	22	6	0.5	-	-	-	17	11	1.4	18	11	1.0
10	Feb 4-13	18	10	1.1	17	9	1.2	18	11	1.1	19	11	1.0
11	Feb17-Mar 2	18	12	1.4	17	7	0.7	17	11	0.9	17	11	0.7
12	Mar 18-28	19	9	1.6	19	5	1.1	18	10	1.6	20	10	1.5
13	Apr 1- 13	20	12	1.1	25	10	0.8	22	11	0.8	23	11	0.7
14	Apr 19-May 15	17	10	1.8	19	9	0.9	21	15	0.7	21	15	0.6
15	May 19-31	-	-	-	19	8	1.0	19	17	0.9	19	17	0.7
16	12 June-1 July	21	17	0.6	22	7	1.9	19	8	1.0	18	8	0.6
17	July 2-21	23	17	0.7	27	10	0.6	19	11	1.1	19	11	0.7
18	Jul 25-Aug 9	20	8	1.2	26	11	1.5	19	10	0.6	20	10	0.5
19	Aug 12-30	24	10	1.1	24	9	1.5	25	5	1.4	27	6	1.8
20	Sept 1-15	21	6	1.9	22	10	1.1	23	8	0.8	23	9	1.1
21	Sept 16-2 Oct	21	10	1.1	20	8	0.8	16	10	0.6	16	10	0.6
22	12-20 Oct	-	-	-	20	4	0.5	-	-	-	-	-	-
23	21 Oct-3 Nov	19	9	1.9	18	8	1.3	-	-	-	-	-	-

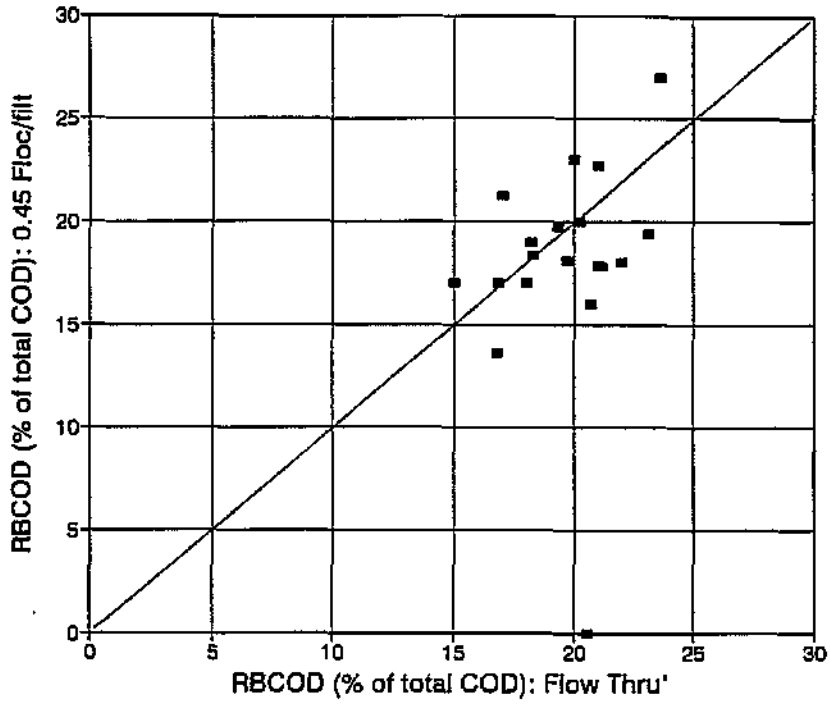

Fig 5.6a: RBCOD as percentage of total COD (S_{ti}) derived from the 0,45μm flocculation-filtration test versus that derived from flow through square wave method. Each data point is the mean of a number (see Table 5.2) of tests on one batch of wastewater.

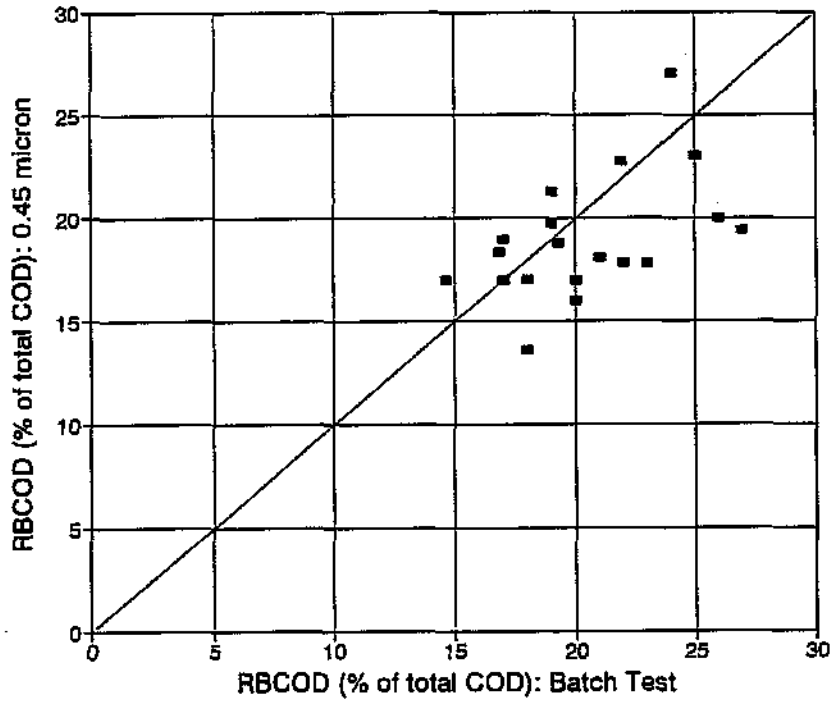

Fig 5.6b: RBCOD as percentage of total COD (S_{ti}) derived from the 0,45μm flocculation-filtration test versus that derived from batch test method. Each data point is the mean of a number (see Table 5.2) of tests on one batch of wastewater.

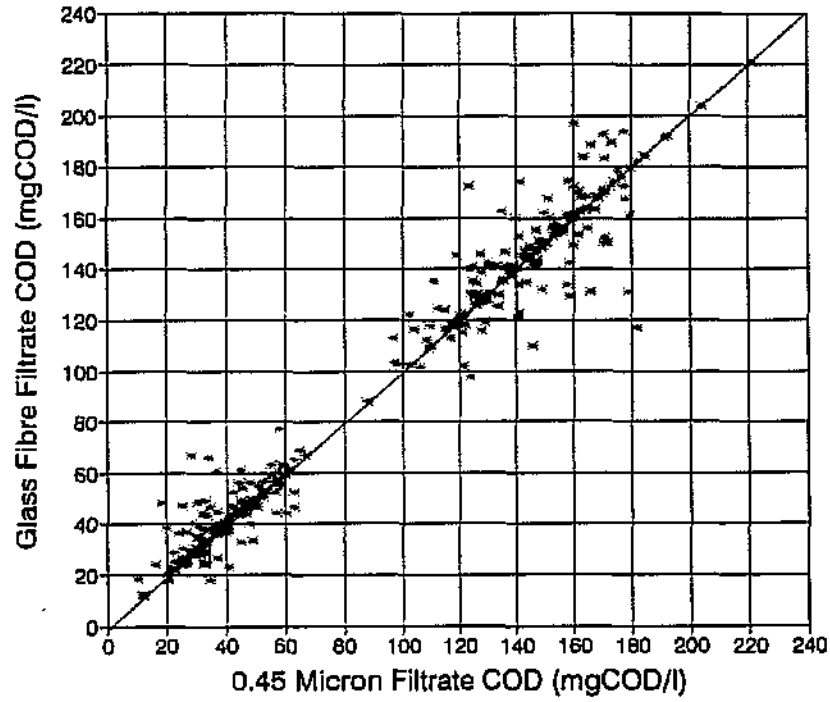

Fig 5.7 COD concentration following flocculation then filtration through glass fibre filters (Whatman's GF/C) versus those following flocculation then filtration through 0,45μm filters (millipore HVLP). Both influent and effluent samples are plotted.

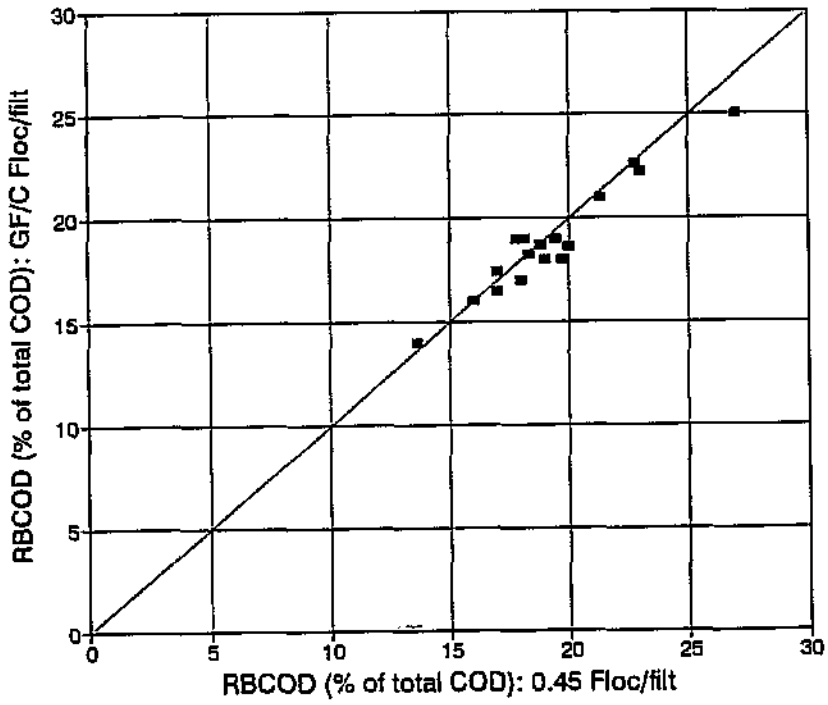

Fig 5.8: RBCOD as percentage of total COD derived from the glass fibre flocculation-filtration test versus those from 0,45μm flocculation-filtration test. Each data point is the mean of a number (see Table 5.2) of tests on one batch of wastewater.

unbiodegradable. Accordingly at the end of the test (the batch test was run for an extended period, 1 or 2 days) samples were drawn from the batch reactor and subject to the flocculation-filtration method. Results from the batch test were compared to those from the effluent of a laboratory-scale long sludge age activated sludge system (Ekama *et al.*, 1986), see Fig 5.9 (a and b); reasonable correlation is obtained. For each batch of sewage, probability statistic plots were constructed for the batch test and effluent methods, to test that the data were normally distributed and determine the means and standard deviations, for example see Fig 5.10 (a and b). For the different sewage batches, means, number of tests and standard deviations of the means are listed in Table 5.3. For each batch, in Fig 5.11 the mean of unbiodegradable soluble COD (S_{us}) from the batch tests is plotted against that from the aerobic activated sludge plant effluent.

From this investigation:

- The batch test method gives values for unbiodegradable soluble COD (USCOD) that tend to be slightly higher than those from the activated sludge system methods; this may be due to the inability of the organisms within the batch test to degrade some of the soluble biodegradable material. However, the differences in USCOD between the two methods are relatively small (< 10%) - the estimates provided by the batch tests are acceptable for design and modelling purposes. Furthermore, values for USCOD as a fraction of the total COD from the batch test (f_{us} = 0,07 to 0,10) fall within the range of values to be expected for a South African raw municipal wastewater (f_{us} = 0,04 to 0,10; WRC, 1984).

- Glass fibre filters can replace the 0,45µm filters without any loss in accuracy.

5.3.6 Extension of batch test to determine unbiodegrable particulate and slowly biodegradable COD

Having developed the batch test method to quantify three of the five influent COD fractions, namely RBCOD, heterotrophic active biomass and unbiodegradable soluble COD, various extensions to the batch test to provide estimates for unbiodegradable particulate COD and slowly biodegradable COD were proposed and evaluated:

- Division of OUR.
- Pasteurization of influent.
- Extended aeration.
- OUR at the end of batch test.
- Acetate addition.
- Raw sewage filtration addition.

Of all the proposals above, only the last (namely addition of flocculated-filtered raw sewage) appeared to hold promise for development. In this proposed method, raw wastewater is flocculated with aluminium sulphate and filtered through 0,45µm filter papers. The filtrate is added to the batch test after about 2 days. From the exponential increase in OUR after sewage filtrate addition, the heterotrophic active biomass concentration in the batch test at the time of adding the sewage filtrate can be determined, from which the remaining two COD

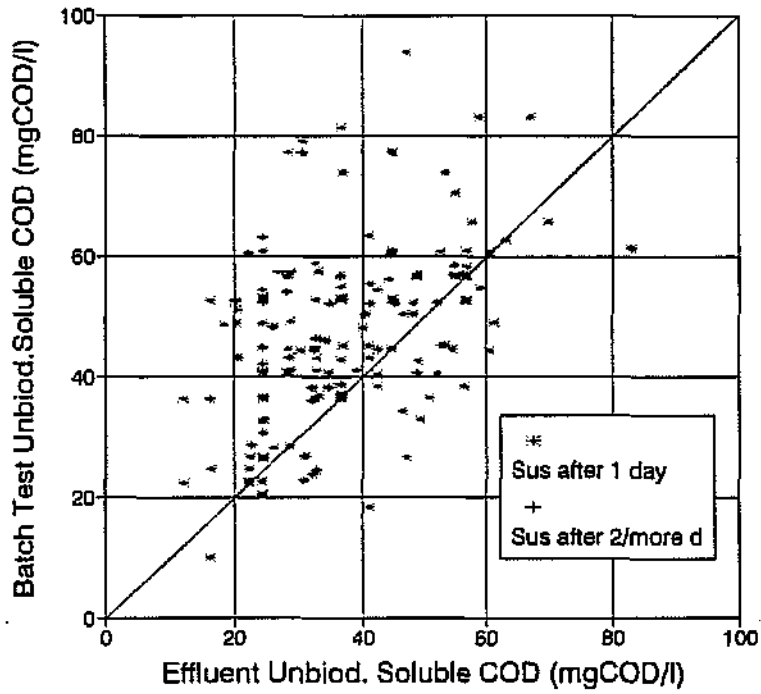

Fig 5.9a: Unbiodegradable soluble COD concentrations determined from the batch test versus those determined from the effluent of laboratory-scale completely aerobic system operated at a sludge age of 12 days. Each data point represents an individual measurement.

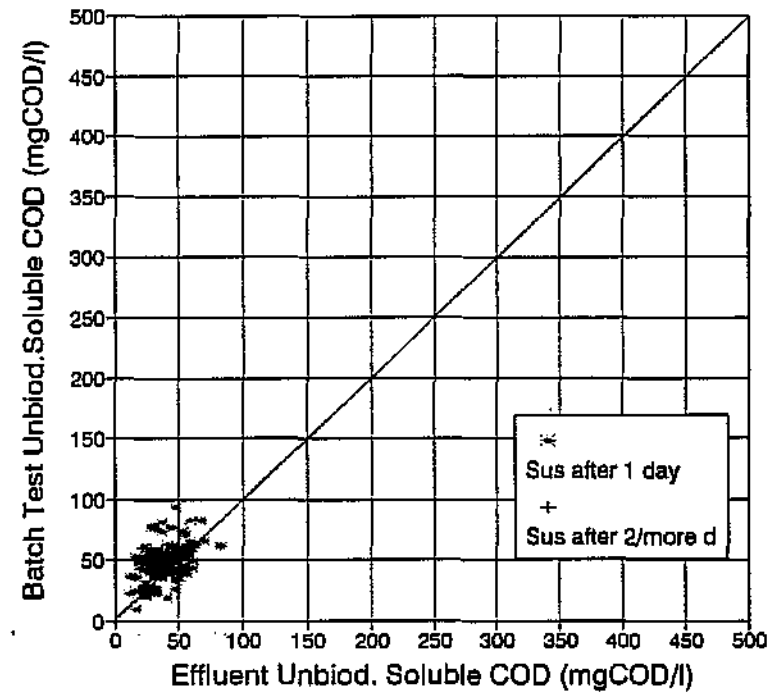

Fig 5.9b: Duplicate plot of Fig 5.9a, but with axes extended to 500 mgCOD/ℓ, the approximate total COD concentration of the sewage in the tests.

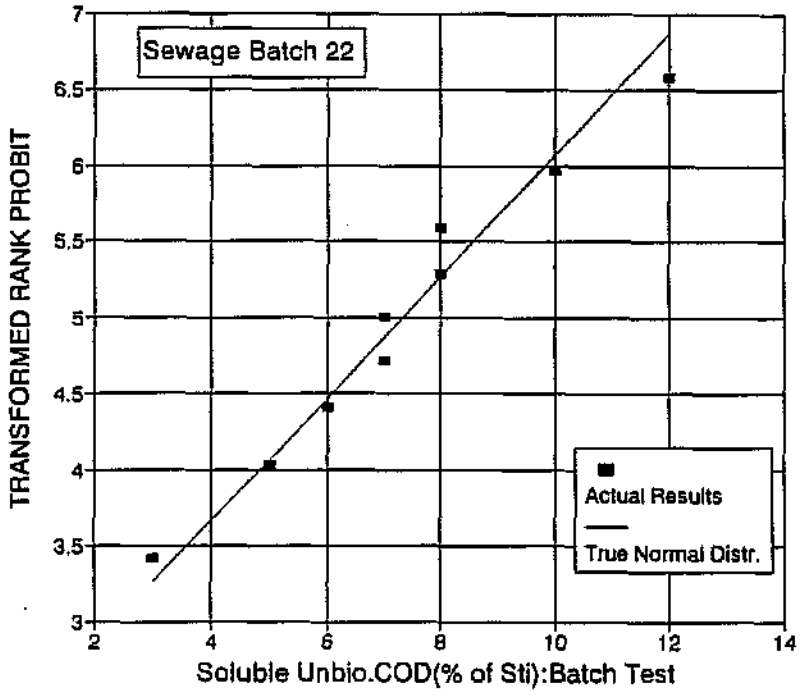

Fig 5.10a: Probability plot of soluble unbiodegradable COD (as % of total COD, S_u) derived from batch tests for one batch of sewage from Mitchell's Plain Treatment Plant.

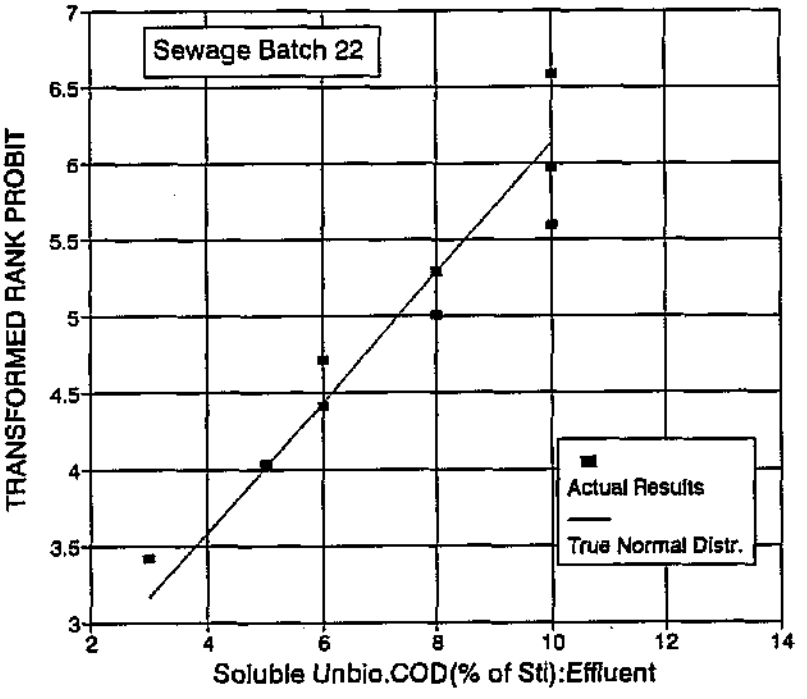

Fig 5.10b: Probability plot of soluble unbiodegradable COD (as % of total COD, S_u) derived from the aerobic unit effluent for one batch from Mitchell's Plain Treatment Plant.

Table 5.3: Mean unbiodegradable soluble COD (as % of total COD, S_{ti}), number of tests and standard deviations of the means for the batch test and aerobic activated sludge system effluent methods, for the different sewage batches. Wastewater obtained from Borcherds Quarry and Mitchell's Plain Treatment Plants.

Sewage	Mean Soluble Unbiodegradable COD (% of Sti)					
	Batch Test			Activated Sludge System		
Batch	Mean Sus	No of Tests	Std.dev. of mean	Mean Sus	No of Tests	Std.dev. of mean
5	10	6	0.8	9	6	0.7
6	10	6	1.0	9	6	0.8
7	8	6	0.5	8	6	0.5
8	11	4	0.3	10	4	0.8
9	10	8	0.4	8	8	0.6
10	8	9	0.9	8	9	0.6
11	9	5	0.9	8	5	0.9
12	9	11	0.7	8	11	0.7
13	9	12	0.7	7	12	0.5
14	8	12	0.4	7	12	0.6
15	7	12	0.5	7	12	0.6
16	9	7	0.8	5	7	0.5
17	8	14	0.7	8	14	0.4
18	8	10	0.7	7	10	0.8
19	8	6	0.6	7	6	0.7
20	8	10	0.6	7	10	0.4
21	8	9	0.6	6	9	0.5
22	8	9	0.9	7	9	0.8

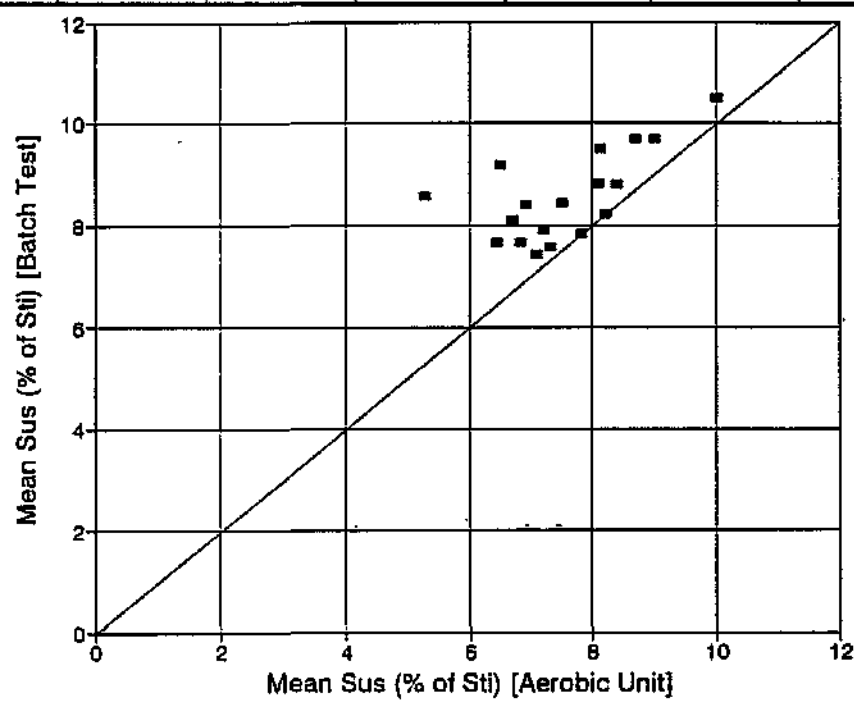

Fig 5.11: Unbiodegradable soluble COD (S_{us}) as a percentage of total COD (S_{ti}) from the batch test versus that from the aerobic activated sludge system effluent. Each data point is the mean of a number of tests on one batch of sewage (see Table 5.3).

fractions can be quantified (i.e. slowly biodegradable and unbiodegradable particulate COD), see Mbewe *et al.* (1995). This method was evaluated by comparing estimates for unbiodegradable particulate COD and slowly biodegradable COD with those from the conventional activated sludge method (Ekama *et al.*, 1986), see Fig 5.12 and 5.13 respectively.

From this series of tests:

- Values for unbiodegradable particulate COD derived from the batch test fall in the same range as estimates from the conventional completely aerobic activated sludge system method (Ekama *et al.*, 1986). However, the direct correlation between the values from the two tests is poor (see Fig 5.12). For the present, the batch test does not provide estimates for unbiodegradable particulate COD that are sufficiently accurate and precise for use in design and simulation of activated sludge systems. For design and simulation, unbiodegradable particulate COD as a fraction of total COD (f_{up}) should at least be able to be quantified into the ranges 0-0,05; 0,05-0,10; 0,10-0,15; etc. As yet, there is not sufficient surety that the estimate for f_{up} from the batch test will meet this requirement; more data are required.

- The errors in unbiodegradable particulate COD are reflected in the estimate from the batch test for slowly biodegradable COD (see Fig 5.13). However, because the absolute value for the slowly biodegradable COD concentration is very much larger than that for the unbiodegradable particulate COD concentration, the relative error in the estimate for slowly biodegradable COD is very much less. The estimate for slowly biodegradable COD can be accepted for design and simulation.

5.4 CONCLUSIONS AND RECOMMENDATIONS

To quantify the influent COD fractions, a simple batch test method has been developed. This batch test has advantages over previous methods in that,

- the experimental procedure is relatively simple,
- no mixed liquor acclimatized to the wastewater is required; this also eliminates the problems in conventional batch tests associated with using mixed liquors from NDBEPR systems,
- for RBCOD, independent determination of unbiodegradable COD is not necessary,
- only three independent constants are required for the calculations; all other constants are derived from the batch test data.

The batch test method was evaluated by comparing its results with those from conventional flow through activated sludge system methods accepted as the standard in the literature. Results from a number of batch tests on municipal wastewater from Mitchell's Plain and Borcherds Quarry (Cape Town, South Africa) indicate that the batch test method supplies estimates for readily biodegradable COD, unbiodegradable soluble COD, heterotrophic active biomass and slowly biodegradable COD that are acceptable for design and simulation of activated sludge systems. However, the batch test does not provide estimates for unbiodegradable particulate COD that are sufficiently accurate and precise for use in design

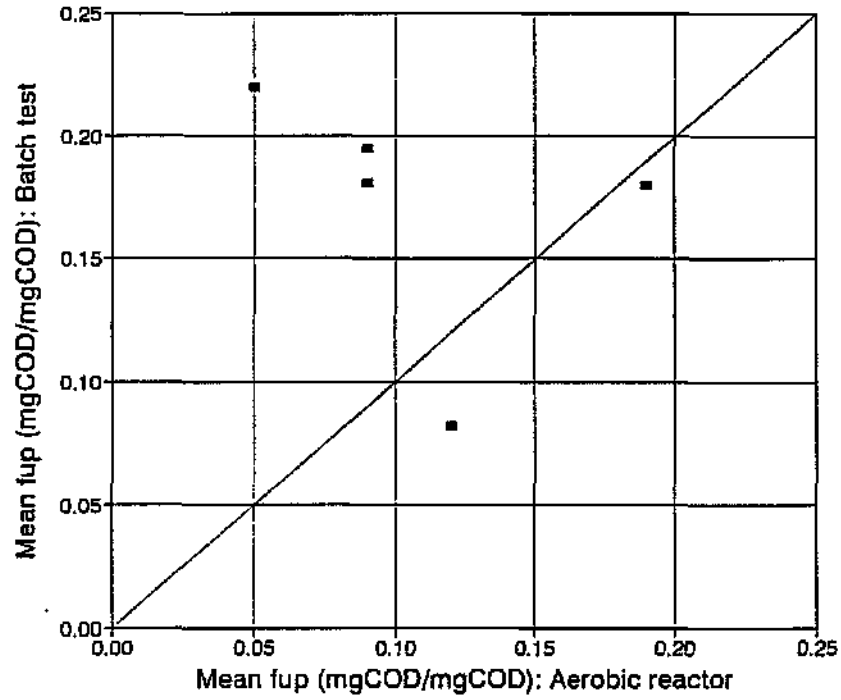

Fig 5.12: Unbiodegradable particulate COD as a fraction of total COD (f_{up}) from the batch test versus those from the aerobic activated sludge system. Each data point represents the mean of a number of tests on one batch of sewage.

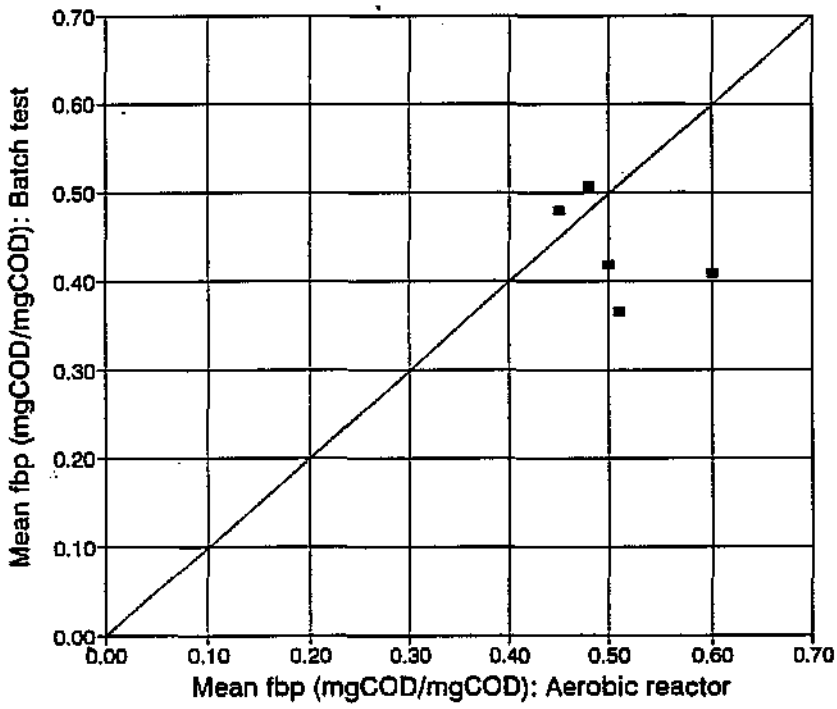

Fig 5.13: Biodegradable particulate COD (i.e. slowly biodegradable COD) as a fraction of total COD (f_{bp}) from the batch test versus those from the aerobic activated sludge system. Each data point represents the mean of a number of tests on one batch of sewage.

and simulation of activated sludge systems. For design and simulation, unbiodegradable particulate COD as a fraction of total COD (f_{up}) should at least be able to be quantified into the ranges 0-0,05; 0,05-0,10; 0,10-0,15; etc. As yet, there is not sufficient surety that the estimate for f_{up} from the batch test will meet this requirement; more data are required.

For the batch test method the following recommendations can be made:

- The batch test can be used successfully to determine the heterotrophic active biomass, RBCOD and the soluble unbiodegradable COD concentrations in the influent wastewater. In this investigation, the estimates for RBCOD and unbiodegradable soluble COD concentrations from the batch test could be compared to results from conventional test methods. However, the heterotrophic active biomass concentration could not be evaluated against other tests because no such tests are available. To evaluate estimates for heterotrophic active biomass, it is recommended that an inoculum of activated sludge mixed liquor from a defined continuous flow steady state system is introduced into the batch test. From the steady state model (WRC, 1984) the concentration of the heterotrophic active biomass in the continuous flow system and therefore added into the batch test can be calculated, and compared to the concentration obtained from the batch test. If similar results are obtained then a powerful verification of the basis of present activated sludge models will have been provided. This is currently under investigation by the UCT group.

- For the batch test method, a technique has been developed to quantify the unbiodegradable particulate COD and slowly biodegradable COD fractions. However, direct correlation of estimates for unbiodegradable particulate COD from the batch test and conventional tests was poor. Also, no discernible trends could be identified in the relationship between values from the two tests. To identify clear trends, a more extensive experimental investigation is required, so that more data are available.

- This investigation has been restricted to quantifying the influent carbonaceous material fractions. Similar studies should be undertaken on the influent nitrogenous and phosphorous materials.

CHAPTER 6

ANCILLARY INFORMATION

6.1 INTRODUCTION

In this Chapter progress is reported on research projects that did not fall under the ambit of the current consolidation contract or any other WRC funded contract. This information is included to keep readers informed of progress at UCT on non-WRC sponsored contracts.

6.2 PROJECT 1: DYNAMIC SIMULATION OF SECONDARY SETTLING TANKS AND EVALUATION OF SECONDARY SETTLING TANK DESIGN PROCEDURES

6.2.1 Theory versus practice

Theory and design of secondary settling tanks has progressed along two parallel but distinct paths. One path has focussed mainly on emperically based procedures for secondary settling tank design and operation. The other path has focussed on developing the flux theory and incorporating it into models of varying complexity for steady state, transient and dynamic simulation of secondary settling tanks. Until recently, not much integration between the two paths has taken place, with the result that theoretical developments in secondary settling tank modelling have not been well integrated into design and operating practice.

While the flux theory has been accepted as a rational and integrated theory for secondary settling tank behaviour, the major problem in applying the flux theory to secondary settling tank design is that the zone settling velocity of the sludge (V_s) on which the flux theory is based is difficult and tedious to measure. Consequently, design of secondary settling tanks has been based primarily on empirical procedures and experience. In its crudest form the design is based purely on hydraulic criteria such as maximum permissible overflow rate. The deficiency of this approach is that it takes no cognizance of sludge settleability. To a large degree this omission has been due to the difficulties with the popular conventional test to measure sludge settleability, the Sludge Volume Index (SVI). To overcome this deficiency two empirical procedures have been developed over the years that are based on improved tests for measuring sludge settleability. The first is the German ATV procedure based on the diluted SVI (DSVI) test and the second is English WRc procedure based on the stirred specific volume index (SSVI) tests, each of these tests being a refinement of the SVI test. Even though both design procedures find their origin in the flux theory, they have developed largely in isolation from each other and, due to their empirical nature and independent verification and calibration, they have deviated considerably from the flux theory. Comparison and integration of design procedures, and of the procedures and the flux theory has been hindered by a lack of clarity and dependability in the relationships between the different settleability measures on which the various procedures/theories are based.

6.2.2 Research at University of Cape Town

The research in this area has been conducted in both parallel directions (theory and practical design) seeking to bring them closer (Ozinsky *et al.*, 1994). This is being done by attempting to establish relationships between the different sludge settleability measures on which the

different procedures are based, using as much data available in the literature as possible. With the aid of these relationships, the designs produced by the flux theory and the two empirical procedures (ATV and WRc) can be compared. It has been found that the precision of the relationships between the sludge settleability measures is poor due to the variability in the data. Nevertheless, with the relationships established to date, in the main a great deal of overlap between the procedures was found. For design, it has emerged that elements not considered in one procedure can be adopted from the other and with a combination of both, the design is superior to that with each individually. This information is to be collated into a secondary settling tank design and information document commissioned under a consultancy by the Water Research Commission, to be published by the IAWQ as a Scientific and Technical Report. This document will serve the purpose of demonstrating the usefulness of the empirical design procedures, familiarizing the English speaking countries with the German ATV procedure and experience (which at present is not well known), thereby to provide design information that will enable design to move away from the present hydraulic criteria to procedures that recognize sludge settleability.

On the theoretical side, work has been conducted to evaluate the flux theory for modelling the dynamic behaviour of secondary settling tanks. Algorithms for solving the settling behaviour equations are complex and difficult because of the hyperbolic (with no turbulent diffusion)/parabolic (with turbulent diffusion) nature of the partial differential equation that governs the settling phenomenon. A solution procedure has been adopted from the literature and after two years of testing and refinement yields good results. The model is valid for sludge movement in one dimension only (i.e. vertical), but allows modelling of large diameter to depth ratio settling tanks because it includes turbulent diffusion which, depending on its magnitude may cause mixing between the different horizontal layers in the tank. A large set of well documented full-scale continuous settling tank operation data is available (STORA) (30-48 mØ and 2-3 m deep) in which two dimensional effects are very likely to have played a role, and the single dimension model with turbulent diffusion has been tested against this data. Provided the appropriate measured V_0 and n values (i.e. the flux theory constants which define the semilog zone settling velocity-sludge concentration relationship) are used, and the diffusivity appropriately calibrated, the model has been found to give very good predictions compared with the full-scale results measured by STORA (Holland) for sludge blanket rise rate, effluent and underflow concentration with time and sludge concentration depth profiles. The sludge theory model with diffusivity also yielded some important additional results insofar as design is concerned: e.g. (1) confirmation of Ekama and Marais' (1986) conclusion that the permissible solids loading should only be 80% of that calculated with the state flux theory without diffusivity, this being a consequence of non ideal (two dimensional) flows of liquids and solids in large scale tanks; (2) the underflow concentration is limited at some maximum value, a feature recognized in the ATV and STORA design procedures (Ozinsky *et al.*, 1994). Both these effects are a consequence of including diffusivity in the flux model. As much information as possible will be gleaned from the research in the theoretical modelling and practical empirical design directions with the view of cross linking between these two parallel lines of work as much as possible, this with the objectives of (i) placing the empirical design procedures on a more theoretically sound footing, and (ii) bringing the advantages of simulation models into the design of secondary settling tanks.

6.2.3 Current status

To make this and related information more widely available, a separate contract with the WRC has been set up to produce an IAWQ Scientific and Technical Report on design, simulation and operation of secondary settling tanks with the cooperation of an international team of experts.

6.3 PROJECT 2: TREATABILITY OF STABILIZED LANDFILL LEACHATE IN A NITROGEN REMOVAL ACTIVATED SLUDGE SYSTEM

6.3.1 Introduction

This treatability study was undertaken to provide additional information on the feasibility of adopting an integrated approach to municipal waste management by operating sewage treatment plants (liquid waste treatment) and sanitary landfills (solid waste treatment) in conjunction with each other. This allows the liquid leachate stream produced in the landfill to be treated in the sewage treatment plant and to dispose of the solids sludge stream generated in the sewage treatment plant to the landfill.

6.3.2 Experimental set-up and testing

To determine the effect of stabilised leachate on the nitrogen removal activated sludge system, two anoxic (4ℓ)-aerobic (8ℓ) reactor systems at 15 days sludge age were operated for 80 days at 20°C. Both, one experimental and one control, received 15ℓ/d real raw sewage diluted to a concentration of 600 mgCOD/ℓ, but the experimental system received additionally 300 mℓ/d stabilized leachate at 5 300 mgCOD/ℓ, representing 18% of the sewage organic load. This is a leachate organic load about 3 times higher than would roughly be expected on a *per capita* basis.

For the final 47 days, daily testing of influent, reactor and effluent COD, TKN, NH_4^+, NO_3^- and NO_2^- concentrations were conducted on both experimental and control systems. Also daily the VSS, TSS, sludge settleability in terms of DSVI and oxygen utilization rate were measured on the aerobic reactors of both systems.

6.3.3 Results

Analysis of the results indicated that

- The N and COD mass balances were generally good (> 90%).
- 11,5% of the leachate COD contributed to unbiodegradable soluble COD of the effluent, which increased the effluent COD concentration from the treatment system by 13 mgCOD/ℓ (25%).
- 5,7% of the leachate was unbiodegradable particulate COD increasing the VSS concentration of the solids in the reactor by 75 mgVSS/ℓ (3%).

- 83% of the leachate is biodegradable and contributed to the biodegradable organics of the sewage.

- No noticeable difference between the effluent TKN concentration of the experimental and control systems could be detected indicating that the leachate contains negligible unbiodegradable soluble organic N. Also nitrification was complete and all the free and saline ammonia of the leachate was nitrified.

- The additional leachate biodegradable COD caused an increase in reactor VSS concentration of about 400 mgVSS/ℓ, i.e. 20% more than in the control. The biodegradable COD and TKN caused the oxygen utilisation rate to increase by 5 mgO/ℓ/h, i.e. also about 20% more than in the control system. These increases seem reasonable considering the leachate increased the organic and nitrogen load by 18%.

- The effluent nitrate concentration from the experimental system was lower than that from the control system. This implied, and detailed comparison on the denitrification performance of the two systems confirmed this, that the leachate not only was capable of denitrifying all the nitrate that was generated from its own TKN content, but also contributed to denitrification of nitrate generated from the sewage's TKN content.

- Sludge settleability in both systems *improved* from over 150 mℓ/g at the start of the investigation to below 60 mℓ/g at the end. The filaments in the sludge were the usual anoxic/aerobic (AA) types (i.e. *M.parvicella*, 0092, 0041, 0675) but were only present at a common level. This indicated that the leachate did not stimulate poor sludge settleability in the N removal system.

6.3.4 Conclusions

Addition of stabilized leachate to a N removal activated sludge system indicated that it (1) is about 90% biodegradable and, (2) due to its high short-chain fatty acid (SCFA) and readily biodegradable COD content it not only is able to denitrify all the nitrate that is generated from its own TKN, but also contributes significantly to the N removal of the TKN in the sewage. The high SCFA content of leachate may also stimulate additional biological excess P removal when it is added to a nutrient removal system, but this was not tested.

6.3.5 Current status

This preliminary study indicated that there is merit in pursuing these kind of leachate treatability studies further. This research will form part of a new WRC contract on treatment of low organic high nutrient (N & P) wastewaters.

6.4 PROJECT 3: THE EFFECT OF ALTERNATIVE DETERGENT BUILDERS ON NDBEPR SYSTEMS

6.4.1 Introduction

Phosphate is an essential nutrient for photosynthetic plant growth. However if over-supplied

in a water body, it leads to excessive algal growth, a condition termed eutrophication. Eutrophication is not only unsightly, but renders the water less usable. In South Africa, most of the highly eutrophied water bodies with a high phosphorus load still receive up to 90% of their phosphorus input from effluents discharged by sewage treatment works (Pretorius, 1983). The phosphorus content of domestic sewage originates from two main sources, namely human waste (± 60%) and detergents (± 40%) (Heynike and Wiechers, 1986).

There is an argument for a ban on phosphate in detergents to reduce the phosphate load on sewage treatment plants and thereby limit phosphate discharges via treated municipal effluent. The removal of phosphate from detergent formulations has been an effective way in many countries of reducing the phosphorus load to receiving waters. In South Africa, detergent manufacturers are anticipating consumer pressure to reformulate their detergent products to eliminate phosphate. The two possible replacements for phosphates are zeolite 4A and high surface area (HSA) calcite.

High surface are a (HSA) calcite is not yet being used as a detergent builder, but is being considered as a potential replacement for phosphate in South African detergent formulations. Zeolite is already widely used in Europe and North America as replacement for phosphate and previous research efforts have not shown any harmful effects in conventional (fully aerated) activated sludge systems.

Unlike in Europe and North America, most activated sludge plants in South Africa are based on the biological excess phosphorus removal (NDBEPR) activated sludge system, which embraces unaerated zones to achieve high removals of nitrogen and phosphate. For this reason it is important to investigate the effects of alternative builders on the NDBEPR process before launching zeolite/HSA calcite detergent formulations in South Africa. Thus, Lever Brothers (South Africa) embarked on a research programme in collaboration with the University of Cape Town, to investigate the impact of zeolite and HSA calcite on the NDBEPR process (Kashula *et al.*, 1993).

6.4.2 Experimental set-up and testing

The original research strategy was to compare the behaviour of two laboratory NDBEPR activated sludge systems; a control supplemented with a phosphate based detergent formulation, and an experimental supplemented with a zeolite based detergent formulation. This approach led to several operational difficulties and the research strategy was modified to compare the alternative detergent builders only as isolated species.

Two laboratory NDBEPR systems were set up in the Modified University of Cape Town (MUCT) configuration; one Experimental and one Control. The Experimental system was dosed with a realistic mass of zeolite or HSA calcite while the Control system was operated on normal sewage. The periods of dosing (first with zeolite and then with HSA calcite) were preceded and succeeded with "baseline" periods when normal sewage was fed to both Control and Experimental systems. The systems were operated for a period of 289 days and the behaviour of the two systems was monitored daily. After evaluating the reliability of the observed data via COD and N mass balances, the effect of the alternative detergent builders zeolite and HSA calcite on the Experimental system was determined by critical and statistical comparison with the Control system.

6.4.3 Results and discussion

In this experimental investigation the following results were observed:

(1) Reasonable nitrogen balances were achieved (weighted average 89.5%).

(2) COD balances were not as good as the nitrogen balances (weighted average 84.3%). This was probably due to a laboratory artifact involving measurement of the oxygen utilization rate (OUR).

(3) Neither zeolite nor HSA calcite had any effect on carbonaceous organic material degradation and there was no statistical difference between the Experimental and Control effluent COD concentration at the 95% confidence level.

(4) Because zeolite is an insoluble inorganic solid it was expected that when zeolite was dosed to the Experimental system, the inorganic suspended solids concentration of the Experimental system relative to the Control would increase by the same amount as the mass dry zeolite dosed. This increase, however was found to be only 180 mg compared with the 320 mg zeolite dosed. No explanation for this discrepancy could be found, i.e. the zeolite did not dissolve and was confirmed not to decompose in the 600°C oven used for VSS determination.

(5) The addition of zeolite and HSA calcite had no inhibitory effect on nitrification and there was no statistical difference between the filtered effluent TKN concentrations of the Control and Experimental systems at the 95% confidence level. Weighted averages of the Control and Experimental systems were 4.62 and 4.56 mg N/ℓ respectively.

Nitrification capacity [i.e. the total mass per day of nitrate and nitrite (NO_3^- + NO_2^- = NO_x) generated by nitrification] of the Control and Experimental system was similar except during the intermediate baseline period, when the nitrification capacity of the Control system was higher than the Experimental system by 12 to 15%. This was the consequence of a comparatively lower N mass balance for the Experimental system during this baseline period.

(6) Because both the first and second anoxic reactors were generally underloaded with respect to nitrate (the first reactor to maximise biological excess P removal and the second to minimise AA or low F/M filament bulking, Casey *et al.*, 1993), it was not possible to determine the denitrification rate directly. The apparent rate (varying from 0,024 mgNO_3-N/mgAVSS.d to 0,109 mgNO_3-N/mgAVSS.d) was always lower than the actual rate (0,224 to 0,296 mgNO_3-N/mgAVSS.d, Clayton *et al.*, 1989; Musvoto *et al.*, 1992) and depended more on the influent TKN and the a-recycle than on the denitrification rate, because these two parameters control the nitrate load on the anoxic reactors.

(7) During and after zeolite dosing, both the Control and Experimental systems showed signs of AA filament bulking, with more severe bulking in the Experimental system (DSVI ±150 mℓ/g) than in the Control system (135 mℓ/g). It was concluded that this

bulking was the result of incomplete denitrification in the second anoxic reactor and not due to the effect of zeolite (Musvoto *et al.*, 1992).

During HSA calcite dosing, the Experimental system showed a sharp decrease in DSVI relative to the Control (Experimental 100 mℓ/g; Control 150 mℓ/g). Since denitrification was complete during this period, this decrease in DSVI of ± 50 mℓ/g can be attributed to the presence of 20 mg/ℓ HSA calcite in the Experimental system influent sewage feed.

(8) Only 60% of the expected P removal was achieved (12 mgP/ℓ instead of 20 mgP/ℓ). This was not due to experimental error or the effect of zeolite or HSA calcite because the other MUCT systems in the laboratory also yielded only 60% of the expected P removal. No assignable cause for the poor P removal in the laboratory could be identified.

During zeolite dosing to the Experimental system, the Experimental systems showed P removal 1,25 mgP/ℓ higher than the Control. No assignable cause for the increased P removal could be identified.

During HSA calcite dosing to the Experimental system, there was no difference between the P removal of the Control and Experimental systems.

6.4.4. Conclusions

From the observations, it would appear that the substitution of phosphates in detergent formulations with zeolite or HSA calcite will not have any adverse effects on the biological excess removal sewage treatment process. No effect of zeolite or HSA calcite on COD removal, nitrification, volatile solids production, pH and denitrification could be established. The mass of sludge production would increase (which is to be expected from the addition of inorganic material to the sewage), but this increase is likely to very small - only 56% and 32% of the respective zeolite and HSA calcite dose was recovered in the sludge. The presence of zeolite and HSA calcite are not likely to adversely affect sludge settleability - indeed it appears that HSA calcite may have a small beneficial effect. Zeolite also appears to improve biological phosphate removal, but the reason for this is unclear.

6.4.5 Current status

Research on this project has been completed and a detailed report published (Kashula et al., 1993).

6.5 PROJECT 4: THE EFFECT OF THERMOPHILIC HEAT TREATMENT ON THE ANAEROBIC DIGESTIBILITY OF SEWAGE SLUDGE

6.5.1 Status of dual digestion technology in South Africa

Thermophilic aerobic digestion, a system popular in West Germany, allows simultaneous stabilization and pasteurization of sludge. In this system temperatures in excess of 60° C can be maintained autothermally (generated by the sludge itself) at 6 to 8 days retention time,

thereby obviating the need for external heating of the sludge. The system however requires a substantial input of energy for oxygenation and mixing. In South Africa, mesophyllic anaerobic digestion of sludge for stabilization is common. This system is not regarded as one that pasteurizes the sludge, but produces considerable energy (through methane generation). So to switch from mesophyllic anaerobic digestion to thermophilic aerobic digestion to pasteurize the sludge as well as stabilize it, is to change from an energy producing system (through methane generation) to an energy consuming system (through oxygenation). The dual digestion system overcomes this problem; it combines the advantage of autothermal thermophilic aerobic digestion, by providing a large degree of pasteurization in a first stage autothermal thermophilic aerobic reactor at around 1 to 2 days retention time, with the advantage of mesophyllic anaerobic digestion, by providing energy efficient stabilization in a second stage mesophyllic anaerobic digester at 10 to 20 days retention time.

In combining a short retention time autothermal thermophilic aerobic reactor and a mesophyllic anaerobic digester a number of advantages are claimed to be obtained, viz. in the aerobic reactor:

(1) the thermophilic temperatures pasteurize the sludge making it safer for disposal,

(2) the sludge is "pretreated" through partial solubilization of particulate organic matter allowing short retention times (10 days) in the anaerobic digester,

(3) solubilization produces alkalinity through ammonification of protein lending greater pH stability to the anaerobic digester,

(4) very little sludge stabilization takes place in the aerobic reactor - only to the degree that the heat generated biologically maintains thermophilic temperature; final and full stabilization takes place in the anaerobic digester - methane generation is reduced only marginally by the aerobic pretreatment stage,

(5) the major portion of the heat required to achieve thermophilic temperatures is biologically generated,

and in the anaerobic digester,

(6) the heat required to maintain mesophyllic temperatures is derived solely from the hot aerobic sludge feed with the result that the methane generated can be used for purposes other than digester heating.

If the above claims can be substantiated, dual digestion would be a particularly useful system for upgrading existing anaerobic digesters in South Africa to produce not only a pasteurized sludge, but also to allow increased anaerobic digester loading rates (shorter retention times).

After a preliminary investigation with the Johannesburg City Council which focused on the aerobic reactor and showed very promising results (Trim and McGlashan, 1985), the Water Research Commission in 1987 contracted the Milnerton Municipality to conduct a technical and economic evaluation of a full-scale (45m^3 aerobic reactor and 600 m^3 anaerobic digester) dual digestion system, with particular reference to the claims above. Contributing

organizations were Afrox, the Division of Water Technology and the University of Cape Town for technical, scientific and academic input.

In November 1990, 4 years after commencement of the project, the research was precipitously terminated due to the structural failure of the 45 m^3 fibre glass aerobic reactor. Fortunately by this time most of the aspects requiring research attention had been addressed at least in part; some aspects could have been investigated further, such as minimum anaerobic retention time and final sludge dewaterability, but sufficient information was acquired to meet the objectives of the project, i.e. to make a reliable technical and economic evaluation of the dual digestion system. Some of the results achieved in the project has been presented from time to time as the research progressed (De Villiers and Messenger, 1988; Messenger, 1989; Messenger *et al.*, 1989, 1990 and De Villiers *et al.*, 1987, 1988, 1989, 1990) but all the results have been comprehensively documented in a 3 part final report viz. Evaluation and Optimisation of dual digestion of sewage sludge Part 1: Overall System Performance by De Villiers *et al.* (1991), Part 2: Aerobic Reactor Performance by Messenger *et al.* (1991), and Part 3: Economic Evaluation by Laubscher *et al.* (1991). Also 4 papers have been prepared on this research and submitted to Water SA for publication.

The research project has been very fruitful and an excellent example of co-operative work between Municipality, private enterprise, scientific agency and university research. All the claims made for the dual digestion system were found to be valid except the one of the reduced anaerobic digester retention time - the digester was found to operate stably and produce a stable product at 15 days, but not at 12 days. Nevertheless 15 days is considerably shorter than the common 25 to 30 days in conventional anaerobic digestion. It was concluded that the anaerobic digester performance was not adversely affected by the first stage thermophilic autothermal aerobic reactor compared to conventional digestion. However, it was not conclusively shown that the sludge retention time in the anaerobic sludge of the dual digestion system could be reduced compared to the convention anaerobic digestion system. This project addresses this issue (Izzet *et al.*, 1992).

6.5.2 Laboratory research into the effect of thermophilic heat treatment on the digestibility of sewage sludge

One of the advantages claimed for the dual digestion system is that the hot aerobic stage pretreats or conditions the sludge in such a way that the retention time of the anaerobic stage can be reduced considerably compared to conventional anaerobic digestion (claim 2 above). This claim could not be convincingly confirmed before the research plant at Milnerton failed. Furthermore, should there be substance to the claim, it is not known whether it is the heat or the biological activity in the aerobic stage that conditions the sludge. As a result of the suspension of the project at Milnerton, it was decided to investigate the claim at laboratory-scale.

Experimental set-up

The plan was to run three anaerobic reactors with three different feed sludges, viz:

(1) autothermal thermophilic aerobic sludge

(2) thermophilic heat treated sludge, processed in a unit with the same retention time as

the thermophilic aerobic sludge

(3) non heat treated primary sludge.

The system fed type (3) would be used as a control with which the other two systems could be compared. The anaerobic digesters would be run at a specific retention time and, after operation for three retention times without showing signs of failure, the retention time would be reduced. If, with the reduced retention time, the digesters again showed no signs of failure, the retention time would be reduced further until failure occurred.

For eight months of 1991 numerous attempts were made to start up and operate a laboratory-scale autothermal thermophilic aerobic reactor to simulate the aerobic reactor that was used at Milnerton. Many configurations and operational methods were devised, but none produced sludge with the same characteristics as the sludge produced at Milnerton. The main reason for the lack of success was the difficulty of running a laboratory-scale oxygen deficient autothermal aerobic reactor (15 000 times smaller than the Milnerton plant). Consequently, work on the aerobic reactor had to be abandoned in order to not lose more time and not unduly hold up completion of the project.

The project was continued by operating 4 mesophyllic anaerobic digesters treating feed types 2 and 3 above. Starting at 20 days retention time, these systems were operated at decreasing retention times to establish at which retention time the digester would fail for the feed types 2 and 3 above. The digesters were operated for 3 retention times at each retention time and during this time were monitored daily for gas production and composition, VSS and COD reduction, short chain fatty acid (SCFA) and alkalinity concentrations, pH, conductivity and ammonia and TKN concentrations. From this data it could readily be established whether or not the digesters were operating satisfactorily. If still satisfactory, the retention time was reduced.

Results and Discussion

From the results there was no discernible difference in digester performance for the thermophilically pretreated sludge (70° C for 1 day) and the control sludge (not heated), right down to 7 days retention time. Digesters with both sludge feeds showed failure at 4 days retention time (SCFA/Alk > 0,2), but recovered very quickly (in 12 days) by (1) increasing the retention time to 20 days by cutting back on the feeding rate for 8 days (until the SCFA/Alk was again < 0,2), then operating at 12 days retention time for 4 days. After this a 7 day retention time with stable operation for 3 sludge ages could again be established.

It would appear from the above that the conditioning effect of heating at 70° C for 1 day does not improve the anaerobic digestibility of the sludge (which was a domestic primary sludge). Therefore, in the sense that the heat treatment pretreats or conditions the sludge so that shorter than conventional anaerobic digester retention times can be implemented, the claim is refuted; it appears that the unheated primary sludge also can be digested at much shorter than the 20-30 days retention time. In the sense that the anaerobic digester of a dual digestion system can be operated at a short retention time (10-12 days), the claim is supported, but it is likely that the sludge source would affect the minimum anaerobic retention time.

It was unfortunate that a laboratory autothermal thermophilic aerobic reactor could not be operated successfully because the sludge pretreatment may be dependent on the *biological* autothermal generation of the heat and not just the heat itself. If this is the case, then it follows from the above results that retention times shorter than 7 days could be implemented. In order to test this conclusively, it will be necessary to set up a pilot-scale (say 1 m^3) aerobic reactor and repeat the experiments. From the point of view of understanding the effect of the autothermal aerobic reactor on the sludge, such an investigation is very worthwhile; however, from a practical point of view, its value is doubtful because at present it seems unlikely that full-scale anaerobic digesters will be operated at retention times as low as 7 days.

A detailed report on this investigation has been published (Izzet *et al.*, 1992).

6.5.3 Current status

In a separate project between the City Council of Cape Town and the Water Research Group, a full- scale (185m^3 aerobic reactor and 2000 m^3 anaerobic digester) dual digestion system with an air oxygenated aerobic reactor is being investigated (Pitt and Ekama, 1993). Over the past 4 years satisfactory results have been achieved and the mathematical model developed for the pure oxygen Milnerton system described above, suitably modified for air oxygenation, was found to predict accurately the reactor temperature on the basis of the Milnerton specific heat yield of 13 MJ/kgO consumed. With the aid of this work a scientific and technical comparison between pure oxygen and air oxygenated aerobic reactors in dual digestion could be made.

This essentially has completed the full spectrum of technical feasibility of the dual digestion system at full-scale; pure oxygen (Milnerton), air (Athlone) and oxygen enriched air (Athlone).

From the experience gained through these projects, it is clear that dual digestion is a low level technology for sludge pasteurization and stabilization compared to other heat treatment processes like Zimpro. While the dewaterability of the sludge product of the Zimpro is very much better than that of dual digestion, the level of technology of Zimpro is much more sophisticated and complex. Thus, for the South African situation, dual digestion is eminently suitable despite its poor dewatering sludge product. In fact, while the dewaterability of the dual digestion may be poor, this is so only in comparison to the dewaterability of the Zimpro sludge; the dual digestion sludge dewaters no worse than conventional anaerobically digested sludge. With the research and technology evaluation completed, all that remains is full-scale application of the technology in South Africa. Interest in not only dual digestion, but also other pasteurization sludge treatment methods is bound to increase in South Africa when guidelines stipulating pasteurization of sewage sludge for certain types of land application disposal are officially published and promoted.

CHAPTER 7

CLOSURE

The research in this Consolidation Contract II (K5/356) has focussed on completing both steady state design and dynamic kinetic simulation models for the nitrogen (N) and nutrient (N & P) removal activated sludge systems. It follows from extensive research over the past three decades by the Water Research Group at the University of Cape Town into the spectrum of processes in such activated sludge systems, namely organic energy (COD) removal, nitrification, denitrification and, for the nutrient removal system, biological excess phosphorus removal. To meet this objective, five principle aims were identified.

- Complete the calibration of, and the user manual for the nitrification denitrification (ND) kinetic simulation model.

- Continue the development of a kinetic simulation model for nitrification denitrificationbiological excess phosphorus removal (NDBEPR) systems.

- Revise the 1984 NDBEPR system design manual to incorporate the research findings on ND and BEPR since 1984.

- Write papers and reports on experimental work completed under preceding contracts, in particular on aspects relating to NDBEPR systems.

- Conduct experimental work to evaluate the effect of incorporation of fixed media in the aerobic zone in NDBEPR systems, with the view to reducing the system sludge age yet maintaining virtually complete nitrification.

Aim 1: Nitrification denitrification (ND) kinetic simulation model

Under preceding Water Research Commission (WRC) sponsored contracts a PC based kinetic simulation model describing the dynamic behaviour of the nitrification denitrification (ND) activated sludge system for N removal had been developed. A user manual for this model has been written and is being distributed by the WRC together with the computer programmes (Dold *et al.*, 1991). This completes this section of the work.

With regard to the design procedures for nitrogen removal activated sludge systems, evaluation of the design procedures set out in the 1984 design manual (WRC, 1984) in light of the information that has become available since 1984 has indicated that these procedures remain valid.

Aim 2: Nitrification denitrification biological excess phosphorus removal (NDBEPR) kinetic simulation model

By consolidating information on NDBEPR system behaviour generated under preceding WRC contracts, and experimentally investigating aspects where information was lacking, a kinetic model describing the dynamic behaviour of the NDBEPR activated sludge system for nutrient (N & P) removal has been developed (Wentzel *et al.*, 1992). The model has been found to

simulate accurately the steady state behaviour of a wide range of NDBEPR activated sludge systems. This model will provide a defined framework or structure which can be used to direct research, design and operation. However, the model is not yet completed. A number of aspects have been identified to require investigation, for example, denitrification by polyP organisms, temperature dependency, behaviour under phosphorus limitation. It is envisaged that work of these aspects will continue, either under parallel WRC contracts (eg. K5/542 where the effect of temperature on bulking is being studied, and the measured data possibly could be used to evaluate the temperature dependency of the BEPR processes) or by using information published in the literature. However, it is unlikely that a final solution ever will be attained - the nutrient removal plant is a complex system in which the different system elements, processes and compounds often exhibit interaction in a complex manner; resolution of one problem may bring into focus, or create, another problem.

Aim 3: Revision of NDBEPR system design manual

In this task it was proposed to update the 1984 manual for design of NDBEPR activated sludge systems (WRC, 1984), to incorporate new information that has become available. In both the previous (K5/251) and current (K5/356) Consolidation Contracts revision of the 1984 manual has been considered a priority. For the revision, it was found necessary to first undertake experimental investigations into areas where the information required was incomplete or not available. Under the preceding and current consolidation contracts, these investigations have essentially been completed and work has commenced on writing the revised design manual. Although the current consolidation contract has ended, completing revision of the design manual is a priority for the Water Research Group, and work on this aspect will continue. The updated design manual will assist in a substantial manner in the design and operation of nutrient removal plants to attain improved and more dependable nutrient removal performance.

Aim 4: Write papers and reports

An important objective of this contract was the transfer of technology generated under previous contracts, and of research results obtained during the contract period. In this regard the Water Research Group participated in a number of seminars and conferences (local and international), submitted a number of papers for publication in refereed journals, and published a number of books and reports. A full list of these contributions is included in the section "**PUBLICATIONS DURING CONTRACT PERIOD**".

Aim 5: Characterization of municipal wastewaters

A simple batch test procedure has been developed to quantify the five influent wastewater COD fractions, namely readily biodegradable COD, slowly biodegradable COD, unbiodegradable soluble COD, unbiodegradable particulate COD and heterotroph active biomass. The batch test was extensively evaluated, by comparing the batch test results with those from conventional tests. It would appear that the batch test provides estimates for the wastewater COD fractions that are sufficiently accurate and precise for use in design and simulation of activated sludge systems, except for the unbiodegradable particulate COD fraction. The batch test procedure has considerable advantages over the conventional tests in that the experimental procedure is relatively simple, no mixed liquor acclimatized to the

wastewater is required, relatively few independent constants are required for the calculation and independent determination of unbiodegradable soluble COD is not necessary to quantify the readily biodegradable COD. The batch test should find wide application for wastewater characterization, a requirement for both the design and simulation of activated sludge systems.

RECOMMENDATIONS/FUTURE WORK

The research work reported in the previous and current consolidation contracts has provided all the information necessary to revise the 1984 manual for the design and operation of nutrient removal activated sludge systems. Work has commenced on this task and will be completed in the near future. This design manual, together with the NDBEPR kinetic simulation model that has been developed, will essentially complete the Water Research Group's major research contribution to BEPR. It is clear that BEPR technology is now firmly established and widely exploited at full-scale. Essentially the future of BEPR no longer depends so much on a better understanding of the phenomenon, but on a better understanding of how to deal with the problems that can arise in operation of NDBEPR plants. Two such operational problems are currently being investigated by the Water Research Group under WRC sponsored contracts, namely filamentous organism bulking (K5/542) and treatment of anaerobic digester liquors (K5/692).

From the work on characterization of municipal wastewaters and the development of the batch test method to quantify the COD-fractions, it is apparent that the batch test can be used successfully to determine the heterotrophic active biomass, RBCOD and the soluble unbiodegradable COD concentrations in the influent wastewater. In this investigation, the estimates for RBCOD and unbiodegradable soluble COD concentrations from the batch test could be compared to results from conventional test methods. However, the heterotrophic active biomass concentration could not be evaluated against other tests because no such tests are available. To evaluate estimates for heterotrophic active biomass, it is recommended that an inoculum of activated sludge mixed liquor from a defined continuous flow steady state system is introduced into the batch test. From the steady state model (WRC, 1984) the concentration of the heterotrophic active biomass in continuous flow system and therefore added into the batch test can be calculated, and compared to the concentration obtained from the batch test. If similar results are obtained then a powerful verification of the basis of present activated sludge models will have been provided. This is currently under investigation by the UCT group.

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