The evaluation of the anaerobic baffled reactor for sanitation in dense peri-urban settlements

Report to the Water Research Commission

by

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Evaluation of the Anaerobic Baffled Reactor for Sanitation in Dense Peri-urban Settlements

EXECUTIVE SUMMARY

This report presents the results and findings of Water Research Commission project K5/1248 ***the anaerobic baffled reactor for sanitation in dense peri-urban areas***. In this study, the performance of an anaerobic baffled reactor (ABR) in the treatment of domestic wastewater was assessed by extrapolating existing understanding of the characteristics of laboratory-scale ABRs to design a pilot-scale reactor. The pilot-scale reactor was operated and analysed at Umbilo and Kingsburgh wastewater treatment plants (WWTP) where it was fed municipal wastewater. A parallel study investigated water use and wastewater generation patterns in a peri-urban community. A modelling study was undertaken using data from the pilot-scale ABR from different operating periods and measured community wastewater characteristics, and used to predict effluent characteristics of an ABR treating domestic wastewater from a dense peri-urban area under specified operating conditions. The cumulative experiences and information gained in all the parts of this project were brought together to propose improvements in ABR dimensions and installation in ***Guidelines for the design, operation and maintenance of an ABR*** treating domestic wastewater. Recommendations regarding the appropriateness of the system for different applications and suggestions for management and maintenance strategies were made.

1 INTRODUCTION

The provision of water and sanitation services to previously unserved communities is a South African development priority. No single technological solution is universally applicable to solve this backlog and a solution for a particular community requires that a range of technologies to be available for consideration. This report describes the performance of the anaerobic baffled reactor (ABR) as a possible technology for the treatment of water-borne sewage. This system was originally developed for high-strength organic loads as found, for example, in agro-industrial effluent. The ABR's particular attributes are that it provides for efficient COD removal, does not require external power and has been shown to be resilient to shock loads (hydraulic and organic loading).

The motivation for this project was that, in Durban, it could take approximately 20 years for water-borne sewage to be provided to some of the dense peri-urban communities of the Metro. Because of the lack of availability of water, both for consumption and household use, the wastewater produced from these areas is concentrated. Moreover, the ambient temperatures in KwaZulu-Natal are relatively high. In this context, it was hypothesised that the application of the ABR could provide an immediate solution to the sanitation problem in dense peri-urban areas, where it could be used to treat the domestic wastewater of a small community. The density of dwelling and the topography of these settlements negate the possibility of implementing treatment options such as anaerobic ponds or wetlands.

eThekwini Municipality has been divided into areas where waterborne sanitation exists, and where it does not. Within the sewered area, the aim is to have 100% waterborne sanitation. To the sea-ward side of the area, where possible, sewers will be built or extended where appropriate. On the inland side of the area, however, on-site treatment or decentralised options will be necessary.

eThekwini Municipality has adopted a policy of supplying dry sanitation options to low-income households outside of the water-borne edge (Macleod, 2005) However, many householders aspire to water-borne sanitation, and there is a technology gap in water-borne sanitation options that are sustainable, affordable and practical for these conditions.

The ABR meets several critical requirements, namely, it does not require energy for operation; requires low maintenance; is compact and could be mass-produced. Several ABRs could service small sub-groups within an area and eventually connect to a sewer system for further treatment at a WWTP. Some limitations of the ABR are: no nutrient removal; and insufficient pathogen removal.

1.1 Objectives of the study

The aims of this project were:

- To provide an appropriate sanitation system for application in peri-urban areas through scientific and engineering support to the KwaZulu-Natal Business Partners for Development water and sanitation project.
- To develop an anaerobic baffled reactor for use in pre-treating sewage from peri-urban areas.
- To monitor the performance of the anaerobic baffled reactor in a peri-urban area.
- To undertake pilot studies of the anaerobic baffled reactor at a WWTP.
- To gain scientific knowledge on the fluid mechanics and microbiology of the anaerobic baffled reactor for the pre-treatment of sewage from peri-urban areas.
- To contribute to the development and validation of a computer model for anaerobic digestion.

These objectives were not materially altered during the course of the project. This project has been a scoping study on many of the issues relating to the feasibility of implementing the ABR in peri-urban, rural or densely populated or informal communities, focussing on the microbiological and biochemical performance of the reactor, but also investigating community and institutional issues associated with the project.

It was not considered appropriate to implement a field ABR in a community situation within this project since there were several process issues that required further experimentation before the technology could be considered ripe for implementation in the field. The extra experimentation generated a considerable body of scientific

information which has greatly enhanced the understanding of the dynamics of anaerobic digestion within the ABR.

2 THE PILOT ABR DESIGN CONSTRUCTION AND INSTALLATIONS

The ABR is similar in design and application to the up-flow anaerobic sludge blanket (UASB) but requires no special granule formation for its operation. The ABR has alternately hanging and standing baffles, which divide it into compartments. The liquid flow is alternately upward and downward between the partitions. A sludge blanket accumulates by settling in the bottom of each compartment, and the liquid flow is forced through this blanket as it passes under each hanging baffle. Good contact between wastewater flow and active biomass is ensured by this design. In principle, all phases of the anaerobic degradation process can proceed simultaneously in each compartment. However, the sludge in each compartment will differ depending on the specific environmental conditions prevailing and the compounds or intermediates to be degraded.

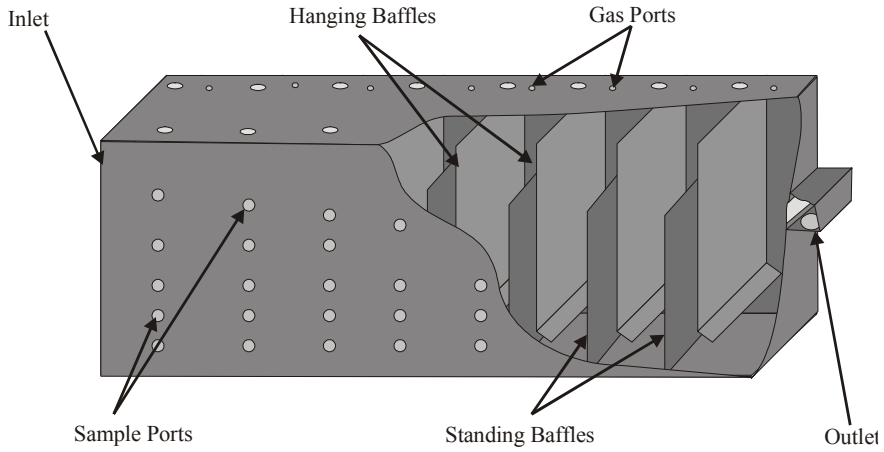

Figure 1.1: Diagram of the pilot-scale ABR with a cut-away to give an indication of the baffle configuration.

A pilot-scale ABR was designed to have a similar structure to 10 ℓ 8-compartment ABRs used in WRC project K5/853 *The assessment of a baffled compartmentalised anaerobic digester for the treatment of high-strength or toxic organic industrial effluents* as a guide. Computational fluid dynamics was used to select a baffle spacing and construction. The pilot-scale ABR had a working volume of 3 000 ℓ, had 8 compartments and was constructed from laser cut sheets of mild steel. A diagram of the pilot ABR is presented in Figure 1.1, with a cut-away showing the internal baffle configuration.

The 3 000 ℓ pilot ABR was initially seeded with 10 ℓ anaerobic digester sludge and installed at Umbilo WWTP for a period of 18 months from July 2000. In January 2002, it was moved to Kingsburgh WWTP. It was fed wastewater pumped out of the influent streams at the head of works at each of these WWTP by a submersible pump. A number of submersible pumps were used during the project, all of which delivered at least 10 times more flow than was required for the ABR to operate. More than 90% of the flow was diverted back to the WWTP influent stream via a splitter box with a 90% overflow side and a controlled bypass on the feed side. The effluent flow rate was recorded by a programmable logic controller (PLC) which calculated the flow bypass

requirements at the feed splitter box. The pneumatic control valve on the bypass line of the splitter box was supplied air by a compressor installed on the top of the pilot ABR.

The feeding system, including flow meter, pump, splitter box, control valve, compressor and PLC, were required to sample wastewater from a much larger flow. In a community or household installation, the ABR would be gravity fed an entire wastewater flow, and therefore all these components will not be required.

Virtually all problems in operation of the pilot ABR were associated with elements of the feeding system. Rags and string, particularly at Umbilo WWTP caused regular jamming of the submersible pump, resulting in down-time. Other problems included mechanical problems with the compressor, blockages in the small bore (25 mm) tube through the flow meter, problems with electrics and the PLC program. None of these problems are expected in a community or household installation.

3 RESULTS OF PILOT ABR OPERATION

The pilot ABR was operated for 409 d at Umbilo WWTP from 18 July 2000 to 31 August 2001 fed municipal wastewater that is comprised of approximately 50 % domestic and 50 % industrial wastewater. In January 2002, the ABR was moved to Kingsburgh WWTP where it treated municipal wastewater that had no formal industrial component, in three operating periods of 4.5, 4 and 6 months in 2002, 2003 and 2004 respectively.

3.1 Summary of results from operation at Umbilo and Kingsburgh WWTP

The 3 000 ℓ pilot ABR was operated over a 5 year period at Umbilo and Kingsburgh WWTP. The reactor was initially seeded with a small amount of anaerobic digester sludge (approximately 10 ℓ) and consequently had a lengthy start-up period in which biomass built up in each of the compartments. Analysis of solids concentrations 200 mm above the bottom of the reactor in the first year of operation showed compartment sludges developing sequentially, i.e. the rate of accumulation in any compartment was faster than in the subsequent compartment.

Amounts of sludge in each compartment continued to change through most of the operating periods, and it cannot be stated with any certainty that a steady state with respect to solid load in each compartment was reached. The rate of sludge build-up was also dependent on wastewater feed flow rate. In the first operating periods at Umbilo WWTP, development of appreciable sludge levels only occurred after the target hydraulic retention time had been decreased to 20 h.

In 2002, the pilot ABR was moved to Kingsburgh WWTP where it operated on a feed of domestic wastewater from middle-income suburbs south of Durban. Three operating periods, in 2002, 2003 and 2004 were achieved. During operation at Kingsburgh WWTP, when fairly well established sludge blankets were present in all compartments, sludge levels were not high in the operating period characterised by repeated high flow incidents that resulted in sludge washout (2002). Sludge levels also seemed to be higher in the 2004 period (40 to 44 h hydraulic retention time) than the 2003 period (22 h hydraulic retention time). This could be due either to lower pseudo-steady-state sludge levels establishing at the higher washout rate of the 2003 period, or simply that sludge was still accumulating during the 2004 period.

Tables 1.1 and 1.2 presents a summary of measured effluent characteristics from the ABR in each of the three operating periods at Umbilo WWTP and the three periods at Kingsburgh WWTP.

Significant COD removal was observed in all operating periods, except immediately after commissioning. Fairly constant effluent COD concentrations were observed except during process upsets such as a souring incident in November 2002. Effluent COD concentration decreased with decreasing hydraulic retention time at Umbilo. This decrease is attributed to improving reactor performance as a result of establishing sludge populations, rather than a function of loading. Effluent COD concentrations decreased significantly when the hydraulic retention time was increased from 22 h to over 40 h between the 2003 and 2004 operating periods at Kingsburgh WWTP as a result of increased contact time in the reactor at the higher retention time.

Table 1.1: Summary of influent and effluent characteristics from the pilot ABR at Umbilo WWTP, July 2000 to August 2001, showing average influent characteristics from the entire period, and average effluent characteristics from each of the 3 operating periods.

	Unit	Influent	Effluent 60 h HRT	Effluent 32 h HRT	Effluent 20 h HRT
COD	mgCOD/ℓ	712 ± 243 (n = 265)	379 ± 124 (n = 16)	170 ± 77 (n = 8)	272 ± 101 (n = 24)
Alkalinity	mgCaCO3/ℓ	215 ± 52 (n = 271)	396± 74 (n = 17)	286 ± 47 (n = 11)	371 ± 129 (n = 20)
NH_3	mgN/ℓ	23 ± 5 (n = 271)	33 ± 19 (n = 16)	33 ± 9 (n = 11)	44 ± 19 (n = 21)
PO_4	mgP/ℓ	6.3 ± 3.0 (n = 96)	2.4 ± 2.6 (n = 16)	1.1 ± 1.5 (n = 10)	7.0 ± 4.4 (n = 23)
Total solids	mgTS/ℓ	1 256 ± 1 086 (n = 52)	2 177 ± 1 893 (n = 16)	1 080 ± 580 (n = 10)	13 782 ± 16 3 20 (n = 24)
pH	range	6.0 – 9.2 (n = 272)	6.3 – 7.2 (n = 17)	6.8 – 7.6 (n = 11)	6.5 - 7.5 (n = 24)

During operation at Umbilo WWTP, higher pH values were observed than during operation at Kingsburgh WWTP. The reason for this difference is not clear, although it may be attributable to generally lower treatment rates at Umbilo WWTP as a result of low biomass populations, resulting in low acidification rates, or some function of the semi-industrial nature of the Umbilo wastewater.

Alkalinity values during all of the experimental periods were low relative to standard anaerobic digestion applications. Consequently the pilot ABR was poorly buffered and therefore susceptible to pH inhibition. In general pH values in the ABR, especially during operation at Kingsburgh WWTP were low, and consequently reduced micro-organism activities, particularly of methanogens could be inferred. A net generation of alkalinity as a result of anaerobic digestion in the pilot ABR was observed in all operating periods.

The shape of the pH profile (i.e. the relative changes in pH value between compartments) showed different trends in all of the 4 operating periods. Examination of each of the profiles provided clues to the relative rates of acid producing and consuming processes in each of the compartments.

Enumeration of pathogen indicator organisms (total coliforms, *E. coli*, coliphage and helminth eggs) in the influent and effluent of the pilot ABR in the 2004 operating period in each case showed significant pathogen removal as a result of anaerobic digestion and sedimentation. However significant counts of all indicator organisms were observed in all effluent samples indicating that further disinfection is required before ABR effluent can be reused.

Table 1.2: Summary of influent and effluent characteristics from the pilot ABR at Kingsburgh WWTP, 2002 to 2004, showing average influent characteristics from the entire period, and average effluent characteristics from each of the 3 operating periods.

	Unit	Influent	Effluent 2002 20 h HRT	Effluent 2003 22 h HRT	Effluent 2004 40 - 44 h HRT
Total COD	mgCOD/ℓ	698 ± 220 (n = 280)	299 ± 131 (n = 16)	212 ± 143 (n = 57)	130 ± 64 (n = 202)
Soluble COD		154 ± 85 (n = 20)	204 ± 53 (n = 8)	71 ± 21 (n = 26)	157 ± 89 (n = 18)
Alkalinity	mgCaCO3/ℓ	248 ± 45 (n = 217)	226± 70 (n = 15)	268 ± 38 (n = 13)	246 ± 53 (n = 4)
NH_3	mgN/ℓ	39 ± 11 (n = 196)		34 ± 3 (n = 7)	51 ± 23 (n = 10)
PO_4	mgP/ℓ	13 ± 4.5 (n = 170)		5.5 ± 0.5 (n = 5)	20.3 ± 5.6 (n = 7)
Total solids	mgTS/ℓ	667 ± 215 (n = 44)	475 ± 138 (n = 15)	225 ± 96 (n = 14)	368 ± 114 (n = 13)
pH	range	4.4-7.9 (n = 211)	4.5 – 7.1 (n = 7)	6.2 – 6.7 (n = 9)	6.2 - 7.4 (n = 6)
Total coliforms	Log[cfu/100mℓ]	7.3 (n=25)			6.6 (n=25)
E. Coli	Log[cfu/100mℓ]	7.7 (n=25)			6.8 (n=25)
Coliphage	Log[pfu/100mℓ]	4.1 (n=24)			3.5 (n=24)
***Ascaris* spp.**	No. eggs/ℓ	772 ± 341 (n=13)			17 ± 15 (n=13)

4 MICROBIAL COMMUNITY CHARACTERISATION STUDIES OF THE ABR

Two studies of microbial community dynamics were undertaken during operation of the pilot anaerobic baffled reactor at Kingsburgh WWTP. The first study in 2003 studied the dynamics of a sample of micro-organism classes and genera using a number of molecular techniques. The second study was performed in 2004 using Scanning Electron Microscopy (SEM) to provide visual evidence to support FISH results, and to gain an understanding of the granulation process that appeared to be occurring in the pilot ABR.

Both the FISH / DNA sequencing study and the SEM study demonstrated that a diverse community of micro-organisms exist in the pilot ABR treating domestic wastewater. The FISH / DNA sequencing study positively identified and enumerated specific micro-organism types, while the SEM study provided insight into the mechanisms of anaerobic digestion and granule formation.

The two studies presented conflicting evidence on the presence of acetoclastic methanogens, particularly those in the genus *Methanosaeta*, in the pilot ABR; The FISH study probed this genus but did not detect any, while micro-organisms with morphologies similar to *Methanosaeta* were observed in abundance in the SEM study. It is hypothesised that the binding of *Methanosaeta* in granules may have resulted in poor permeability to oligonucleotide probes in the FISH study, resulting in undetectable hybridisation. This hypothesis is supported by the fact that significant populations of acetoclastic methanogens would be required to achieve the COD removal obtained by treatment of domestic wastewater by the pilot ABR in these operating periods since acetoclastic methanogenesis is responsible for most of the conversion of COD to CH_4 gas in anaerobic digestion.

The FISH study concluded that little differentiation in population characteristics occurred among compartments. The SEM study was not able to quantify micro-organisms of different classes, but noted that *Methanosarcina*-like species were observed in the first compartment, but not in later compartments. From these observations, it would appear that phase separation, as originally expected, did not occur. In other words, spatial separation of hydrolysis, acidogenesis, acetogenesis and methanogenesis into different compartments did not occur in the ABR treating a relatively low strength (in terms of conventional anaerobic digestion applications), particulate wastewater.

The ratio of hydrolytic, acidogenic, and acetogenic micro-organisms to each other appeared to be relatively constant throughout the ABR, but a change in concentration and dominant genus of acetoclastic methanogens was observed, particularly between the first and subsequent compartments. It is hypothesised that hydrolysis was the overall rate-limiting step in treatment of domestic wastewater: hydrolysable material in the ABR feed was carried through the reactor, undergoing continuous hydrolysis from the surface of the waste material inwards. The exception to this theory is compartment 1 where acid production caused by readily hydrolysable material in the influent resulted in a decrease in pH value, which subsequently inhibited methanogenesis. Here, higher concentrations of soluble intermediates could be expected. This hypothesis is borne out by the results of the chemical analyses performed on the pilot ABR.

5 COMMUNITY WATER USE AND WASTEWATER GENERATION STUDY

A study was undertaken to quantify water use and wastewater generation in a low-income peri-urban community and to characterise the wastewater in terms of chemical contaminants and pathogen indicator organisms. These data are intended to facilitate model-based predictions of the performance of an ABR or similar on-site or decentralised technology under conditions similar to those encountered in a South African low-income peri-urban community. This study was performed in the KwaMashu-Newlands Interface Housing Development, a low-income peri-urban community 20 km from the Durban CBD.

This study had three components:

- Community water use habits were investigated by means of a household questionnaire. The questionnaire was designed to gather information about water use habits, daily quantity of water used and the daily amount of wastewater generated.

- A water meter data survey was undertaken in which eThekwini Municipality water meter records were studied to identify trends in water consumption in communities using semi-pressure (roof-tank) water delivery systems.

- Samples were obtained from sewers in the area in which the household questionnaire study was conducted. The samples were analysed for various chemical and microbial determinands.

5.1 Results of the water use questionnaire study

A questionnaire was administered to households within the study area. The questionnaire consisted of several sections each designed to gather information about water use habits, daily water use and daily wastewater generation. The survey was conducted verbally with the aid of a translator. A total of 81 households were interviewed.

Each householder was asked questions relating to the amount of water they *believed* that they used, and how much was used for specific daily functions. Where householders were unable to guess volumes of water used (which was true in most instances) estimates were made in terms of the numbers of 5 ℓ bucketfuls used for a task per day.

The average daily water use estimated by householders in the study area was 342 ℓ.

5.2 Results of water meter database survey

The *geographical information systems* (GIS) databases of eThekwini Water Services were mined for water consumption data from water meter readings from communities serviced with semi-pressure (roof tank) water supply using ArcGIS™ software. Data was categorised into residential area and number of houses within an area.

eThekwini Municipality was found to have the 15 major low-income housing developments supplied by roof tank water systems. 3 of these areas (Durban, Pinetown and New Germany) are regarded as urban, whilst the other 11 are regarded as peri-urban.

There was found to be a significant difference between average water consumption in established urban developments 994 /(d.household) and that in peri-urban developments 473 ℓ/(d.household).

Records were not available for the Newlands-KwaMashu Interface housing development where the water use questionnaire study and wastewater characterisation studies were undertaken since this was a fairly new development at the time of the study, and water meters had not been installed; however, a neighbouring area, Melkhout was supplied with metered roof-tanks. This community has similar house designs as the Newlands-KwaMashu Interface community. The average water consumption for the Melkhout area was 351 ℓ/d per household, a value almost identical to that estimated by the Newlands-KwaMashu Interface community.

5.3 Results of community wastewater characterisation study

Three sewers within Section 1 of the Newlands-KwaMashu Interface community were sampled over a number of days, and at different times of day, and were analysed for total and soluble COD, total and soluble protein and carbohydrate content, total Kjeldahl nitrogen, total solids, total coliforms, *E. Coli*, and coliphage, in each of a winter and summer campaign.

Table 1.3: Summary of 80th percentile values calculated from winter (worst case) study of wastewater characteristics from sewers in the Newlands-KwaMashu Interface housing development

	Units	**80th percentile**	
Total COD	mgCOD/ℓ	1089	(n=90)
Soluble COD	mgCOD/ℓ	169	(n=135)
pH	-	8.05	(n=45)
Total protein	mg/ℓ	80	(n=137)
Total carbohydrate	mg/ℓ	46	(n=135)
TKN	mgN/ℓ	176	(n=90)
Total solids	mg/ℓ	153	(n=92)
T. Coli	log(cfu/100 mℓ)	6.75	(n=135)
E. Coli	log(cfu/100 mℓ)	6.70	(n=135)
Coliphage	log(pfu/100 mℓ)	4.20	(n=135)

Most components measured in the wastewater showed a decreasing trend, from the early morning to the late afternoon, although the trend is not statistically significant as a result of large standard deviations calculated from the concentration data. COD

concentrations are significantly higher in winter than in summer samples, as are measured pH values. This is attributed to the development of a biofilm in the sewers during the summer season which caused acidification of organic material in the wastewater. This resulted in a decreased pH value, which was measured at the sampling site. However organic acids are metabolised before samples can be analysed in the laboratory, resulting in the measurement of lower COD values.

Total coliforms and coliphage concentrations in the wastewater are higher in summer than in winter, while there is little difference in measured *E. Coli* concentrations. The reasons for these differences are not certain.

It was not possible to calculate mass loads of contaminant from the available measurements. However, since large variations in concentration measurements were obtained and a large number of measurements were made (between 45 and 137 for each analyte), the value of the 80th percentile concentration was chosen as a representative measure of wastewater characteristics for design purposes since this amount excludes extreme data values, but allows for *worse than average* characteristics for predicting wastewater treatment requirements. 80th percentile values from the winter study are presented in Table 1.3.

6 MODELLING OF THE ABR

Modelling studies of the pilot ABR were undertaken to simulate performance on municipal wastewater and to predict performance of an ABR treating a low-income community wastewater.

Two modelling exercises were undertaken. In the first, a Siegrist biochemical model structure was implemented in WEST® simulation software, describing an 8-compartment reactor. In the second exercise, a steady-state (mass balance) model was used to simulate ABR effluent characteristics and to predict ABR effluent characteristics for a low-income community wastewater. A third step, the implementation of an Anaerobic Digestion Model No. 1 (Batstone et al., 2002) structure in an ABR configuration is continuing.

6.1 Siegrist model of the ABR

The Siegrist model of the ABR implemented in WEST® was able to produce reasonable predictions of pH, particulate COD, alkalinity and ammonia in the compartments and effluent of the pilot ABR for the 2003 operating period, but was not able to describe the soluble COD profile without substantial modification.

It was concluded that both the model structure and the experimental measurements made needed to be altered to improve the model's ability to predict ABR performance.

- A subdivision of particulate COD is required in order that more than one hydrolysis rate is applied.

- Measurements of organic nitrogen, inert COD, VFA and biomass seeding rates in the feed.

- Measurements of compartment gas production.

Where possible additional measurements to supply the required information were made in the 2004 operating period.

It was also seen that, in the biochemical model, the rate limiting step in all but the first few compartments was hydrolysis, and significant inhibition of hydrolysis and methanogenesis as a result of low pH values was calculated in all compartments.

6.2 Steady-state modelling of the ABR

The data obtained from 22 and 42 h steady-state operation were incorporated into a steady-state model modified by the differentiation of feed into carbohydrate, lipid and protein from the steady-state model presented by Sötemann et al. (2005). A good match between measured and calculated output conditions was obtained, despite the fundamental model structure being inappropriate for the plug-flow-like behaviour of the ABR. The kinetic parameters obtained from the model are not expected to describe the reactor response well, particularly as only two operating points were used in the regression. Consequently, although the model was able to provide good insight into how changes in feed characteristics affect reactor performance, the prediction of performance at different retention times is probably not accurate.

A scenario analysis was performed in which the effect of organic strength, alkalinity, pH and composition of the wastewater, and retention time of the reactor were varied. It was seen that the feed alkalinity had the largest effect on calculated pH values, while organic strength and feed composition affected pH less. The calibration used indicated that retention time did not have a significant effect on the calculated pH values. It can be seen that for a constant feed composition, the pH values found in the reactor (under conditions where hydrolysis is the rate-limiting step) will be a function of alkalinity production defined by the extent of COD reduction.

It is concluded that for the hydrolysis-limited case, *the alkalinity, and alkalinity generation potential are the most important variables for maintaining reactor stability*. Furthermore, where low pH values may be resulting in pH inhibition of methanogenesis, increasing alkalinity will also result in improved COD reduction by causing an increase in the rate of methanogenesis.

7 DISCUSSION

Based on the findings of the chemical, microbiological and modelling studies, a theory was developed that described the processes of anaerobic digestion in an ABR based on the following premises:

- In the first compartment, acid production causes a drop in pH value that inhibits methanogenesis, resulting in a net accumulation of acid. Here, the overall rate of anaerobic digestion is limited by the rate of methanogenesis.

- In subsequent compartments, products of hydrolysis and acidogenesis are consumed at the same rate or faster than they are produced, resulting in a gradual

increase in pH value. Here, the overall rate of anaerobic digestion is limited by the rate of hydrolysis.

- There is some differentiation between methanogenic populations in the first and subsequent compartments as a result of the significantly larger concentration of organic acids in the first compartment. However in subsequent compartments, the ratios between the microbial groups that are responsible for different sub-processes in anaerobic digestion do not change significantly, indicating that the development of highly specialised microbial communities in the different compartments of the ABR, as seen in high strength, soluble feed applications, does not occur in an ABR treating domestic wastewater.

7.1 Factors affecting effluent quality

The quality of the effluent will depend on two factors; (i) the *amount of time* that the wastewater is in contact with the biomass; and (ii) *the amount of biodegradable solids and biomass retention* as a result of settling within each compartment. The first will depend on the *average hydraulic retention time* of the system, a function of the volumetric flow and reactor volume, and the second, on the *mean up-flow velocity* in each compartment, which in turn depends on the dimensions and number of the compartments.

7.2 Advantages of an ABR over a septic tank

Despite the lack of compartment microbial community differentiation, the ABR has considerable advantages over a simpler reactor configuration such as a septic tank. The ABR works under similar conditions to a septic tank but it *increases contact between biomass and wastewater* by forcing liquid to flow through biomass beds with each pass under the hanging baffles. In this way there is a biological filtering effect in which solid components are physically retained by settling, and liquid components are removed by adsorption and consumption. As a result, an ABR will produce a far superior effluent to a septic tank operating with a similar hydraulic retention time.

7.3 Effluent quality and discharge/reuse options

An ABR treating domestic wastewater will convert a large amount of wastewater COD to methane gas, and will reduce pathogen loads in the wastewater. However, there is no nutrient removal, and the amount of pathogen removal obtained is insufficient to render the effluent safe for human contact. The presence of significant amounts of ammonia and phosphorus in the effluent mean that it cannot be discharged to surface or ground water, but theoretically can be used in irrigation of agricultural land, or disposed of in a soak-away. The pathogen indicator organism load measured in the pilot ABR effluent indicates that secondary treatment is required before any conventional irrigation methods may be used.

Therefore, except in the case where sufficient area and infrastructure is available to build a sub-surface soak-away system, some post-treatment of the effluent is required before it can be reused. It has been recommended that the use of membrane biofilters in conjunction with the ABR be considered since a biofilter would remove virtually all COD and pathogens, while allowing nutrients, which have a real economic value as a

fertiliser, to be retained for use in agriculture. Research in this area is continuing. Another post-treatment option is a constructed wetland.

7.4 Application of an ABR in community sanitation

A six-stage sanitation system has been presented in which the ABR is the central treatment unit. The six stages include: (i) wastewater generation system (toilet superstructure and flushing mechanism, greywater separation etc.; (ii) wastewater collection, e.g. by a shallow sewer,; (iii) solids screens and the ABR; (iv) polishing or pathogen removal using a membrane or constructed wetland; (v) Effluent reuse in agriculture; and (vi) monitoring and maintenance of the system.

Community education and participation are essential at all of the six stages of the system to protect it from negligence or abuse. Similarly, the application of this kind of system depends on the availability of sufficient land to absorb the generated water so that nutrients do not entire natural water systems.

8 GUIDELINES FOR THE DESIGN, OPERATION AND MAINTENANCE OF AN ABR TREATING DOMESTIC WASTEWATER

The accumulated experiences of this project have been incorporated into guidelines for design, operation and maintenance of an ABR. In this section, a detailed description of the effect on the overall performance of the ABR of factors including operating hydraulic retention time, number of compartments, up-flow velocity in each compartment and specific compartments is given. Recommendations for each of these factors are given, and a sample calculation is given for dimensions for an ABR treating a cluster of 10 low-income houses with wastewater characteristics equivalent to 80^{th} percentile concentrations measured in the community wastewater characterisation study. The selected design hydraulic retention time is 36 h. A description of how this design is expected to perform is presented, based on predictions using the steady-state model.

Guidelines on operation and maintenance are also presented.

9 CONCLUSIONS

This project was undertaken to determine the appropriateness of an anaerobic baffled reactor in treatment of domestic wastewater in low-income communities. A pilot ABR was built and operated at two municipal WWTPs and operation in terms of chemical and microbial performance was characterised under a number of different operating conditions. A study was performed in which water use patterns and wastewater characteristics in a low-income community were measured. These data were incorporated in a model to predict the performance of the ABR would perform in a low-income community. Based on experiences with pilot ABR, a series of design, operating and maintenance guidelines were developed for future installations.

The pilot ABR operated fairly smoothly, showing good biological activity in all of the operating periods. Almost all the problems associated with operation of the system were related to the feeding system and peripheral equipment required to sample wastewater from a much larger flow. These included pump blockages, wear and tear on the compressor and pneumatic valve, limitations of the programmable logic controller (PLC)

algorithm and blockages of the effluent pipe at the magnetic flow meter. In a community installation, none of these problems will occur since the ABR unit would be gravity fed, and would treat the entire wastewater flow generated.

The ABR was found to be a robust treatment system, with biological and hydraulic advantages over septic tank systems, and with considerably reduced installation, operation and maintenance costs compared to aerobic or centralised systems. It also provides an option for communities with dry sanitation that aspire to waterborne sanitation.

However, the ABR was not able to treat wastewater to an acceptable chemical and microbiological standard alone. There must be some post-treatment step and appropriate reuse or discharge method implemented with the ABR as an integrated sanitation system, since unpolished ABR effluent is not fit for discharge to surface or groundwater or for direct use in agriculture.

As with septic tank systems, the ABR has no intrinsic mechanism for managing build-up of inert solids. Therefore an installation treating domestic wastewater must include a screening and grit removal pre-treatment step, or a maintenance plan for regular degritting of the first compartment should be in place. A key factor in the management of inert solids in the ABR is to educate system users to avoid disposing of unsuitable substances into the wastewater treatment system.

10 RECOMMENDATIONS

There are many factors relating to the implementation of a decentralised wastewater treatment system that have not been addressed directly in this project. However, it is believed that a sufficient understanding of the process mechanisms of the ABR have been gained in this project to consider the technology ripe for application in certain situations.

- The ABR is able to provide better and more efficient treatment of wastewater than a septic tank. Therefore it is recommended that an ABR system can be used in any situation that is considered appropriate for a septic tank.

- Further research into post-treatment options is required for implementation of an ABR in a community setting where water cannot be disposed of in a soak-away.

- The application of an ABR in an institutional setting such as for schools, clinics or community toilet blocks should be thoroughly investigated.

11 PUBLICATIONS EMANATING FROM THIS PROJECT

There has been a large *technology transfer* element to this project, including 3 peer-reviewed papers, 3 dissertations and 25 conference and workshop papers and posters. A full technology transfer report is presented in Appendix 4.

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The anaerobic baffled reactor for sanitation in dense peri-urban settlements

The Steering Committee responsible for this project consisted of the following persons:

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LIST OF SYMBOLS AND ABBREVIATIONS

1 LIST OF SYMBOLS

Symbol	Meaning	Units
K_M	maximum reaction rate	1/d
K_S	half saturation constant	mgCOD/ℓ
T	temperature	º C
κ	Arrhenius activation energy	
μ	biological growth rate	mgCOD/(ℓ.d)
μ_{max}	maximum biological growth rate	mgCOD/(ℓ.d)

2 LIST OF ABBREVIATIONS

ABR	anaerobic baffled reactor
ADM1	anaerobic digestion model No 1
ANANOX	anaerobic-anoxic-oxic
BOD	biological oxygen demand
CBD	central business district
CFD	computational fluid dynamics
cfu	colony forming units
COD	chemical oxygen demand
CSTR	continuous stirred tank reactor
DAPI	4’6-diamidino-2-phenylindole
DEWATS	decentralised wastewater treatment systems
DNA	deoxy ribonucleic acid
FISH	fluorescent in-situ hybridisation
GC	gas chromatograph
GIS	geographic information system
HPLC	high performance liquid chromatograph
HRT	hydraulic retention time
LOFLOS	low flow on-site systems
OUR	oxygen uptake rate
PCR	polymerase chain reaction
PE	population equivalent
pfu	plaque forming units
PI	proportional + integral (control)

PLC	programmable logic controller
PVC	poly vinyl chloride
RBCOD	readily biodegradable COD
RNA	ribonucleic acid
SEM	scanning electron microscopy
SFS	solids free sewers
SS	suspended solids
TKN	total Kjeldahl nitrogen
TP	total phosphorous
TSS	total suspended solids
TS	total solids
UASB	up-flow anaerobic sludge blanket (reactor)
USEPA	United States Environmental Protection Agency
VFA	volatile fatty acids
VIP	ventilated improved pit (latrine)
WRC	Water Research Commission
WWTP	wastewater treatment plant

1 INTRODUCTION

The provision of water and sanitation services to previously unserved communities is a South African development priority. No single technological solution is universally applicable to solve this backlog and a solution for a particular community requires that a range of technologies to be available for consideration. This report describes the performance of the anaerobic baffled reactor (ABR) as a possible technology for the treatment of water-borne sewage. This system was originally developed for high-strength organic loads as found, for example, in agro-industrial effluent. The ABR's particular attributes are that it provides for efficient COD removal, does not require external power and has been shown to be resilient to shock loads (hydraulic and organic loading).

This chapter describes the world-wide need for improved sanitation, background to the project, the objectives of the project and the project methodology.

1.1 MILLENNIUM DEVELOPMENT GOALS: APPLICATION TO SANITATION

Since the Millennium Development Goals were adopted at the United Nations Millennium Summit in 2000, they have become the reference for measuring and tracking improvements in the human condition in developing countries. The Goals are backed by a political mandate agreed to by leaders of the United Nations member states. They offer a comprehensive and multidimensional development framework and set clear quantifiable targets to be achieved by 2015. Goal 7 relates to ensuring environmental sustainability. Target 10, within Goal 7, aims … *to halve by 2015, the proportion of people without sustainable access to safe drinking water and* ***basic sanitation*** *…*

1.2 SANITATION IN SOUTH AFRICA

The South African Constitution states that ...*Everyone has the right - to … an environment that is not harmful to their health or well-being;…Everyone has the right to have access to - … sufficient food and water*;

South Africa is committed to eradicating its water supply and sanitation backlogs by 2008 and 2010 respectively. According to the 2005 Millennium Development Goals Report, only 26% of rural dwellers and 55% of urban dwellers in Sub-Saharan Africa have access to acceptable levels of sanitation. In 2001, the Statistics South Africa 2001 Census (Statistics SA, 2005) reported that 13.6% of households did not have a toilet (chemical, flush or pit toilet). The South African Minister of Water Affairs and Forestry in her 2005 budget speech stated that 16 million South Africans still do not have basic sanitation and must be serviced by March 2010. There is thus both a national and international drive to provide sustainable water and sanitation services to millions of South Africans in the course of this decade, and an unsurpassed opportunity for innovation in the sanitation sector. In the KwaZulu-Natal region, there are large communities of semiformal and informal inhabitants of peri-urban and rural areas that are unserved, and the proposed time frame in which formal sanitation services will be provided to them by the local metropolitan council is 10 to 20 y.

Some of the key external factors that would influence the selection of a water supply and sanitation system are:

- Settlement density and pattern: there has been a movement of people from rural areas to urban areas as people search for greater economic opportunities and a more sophisticated lifestyle.

- Disposable income among recipients of the service: the level of income of the recipients will determine the form of service provided.

- The institutional or governmental environment.

- Aspirations, expectations and perceptions of the service: it is not feasible in the short or medium term for the living standards of the poorest to reach the standards which the rich currently hold.

- Technology: there is a need to develop technologies which are appropriate to the social and economic circumstances of the recipients.

1.2.1 Effluent discharge standards

The effluent from any water-borne sanitation system needs to be discharged to the environment. Standards for the discharge of effluents are set by the Department of Water Affairs and Forestry.

Target values are obtained from the General Authorisations (DWAF, 1999) for discharge of waste or water containing waste into a water resource through a pipe, canal, sewer or other conduit and irrigation of any land with waste or water containing waste generated through any industrial activity or by a waterworks.

The authorisation for discharge allows a person who owns or lawfully occupies property registered in the Deeds Office or lawfully occupies or uses land that is not registered or surveyed outside of certain listed sensitive areas may on that property or land discharge up to 2 000 kℓ of wastewater on any given day into a water resource that is not a listed water resource provided that the discharge does not alter the natural ambient water temperature of the receiving water by more than 3Cº.

The authorisation for irrigation allows a person who owns or lawfully occupies property registered in the Deeds Office or lawfully occupies or uses land that is not registered or surveyed outside of certain listed sensitive areas may on that property or land irrigate up to 500 kℓ of domestic wastewater on any given day.

1.3 THE BUSINESS PARTNERS FOR DEVELOPMENT PROJECT: IMPROVING WATER AND SANITATION IN KWAZULU-NATAL

Business Partners for Development (BPD) Water and Sanitation cluster is an informal network of partners who seek to demonstrate that strategic partnerships involving business, government and civil society can achieve more at the local level to improve access to safe water and effective sanitation for the poor than any of the groups acting individually.

A Business Partners for Development study into improving the supply of water and sanitation services to low income communities in KwaZulu-Natal identified a need for an interim sanitation technology that requires minimal maintenance and has no energy requirements since there is currently insufficient infrastructure to provide and service such technology. For biological treatment, excess sludge production is undesirable because of the cost of removing, transporting and disposing of the sludge. Preferably, members of the community should be involved in the construction as well as the operation of the system. Community acceptance and education are key components to the long term success of a sanitation system.

The BPD, through an active association involving Durban Metro Water Service (now eThekwini Water and Sanitation)), Vivendi Water (now Veolia Water), Mvula Trust, the Water Research Commission (WRC) and Umgeni Water (UW), developed a project based in the greater Durban area to provide a sustainable community-focused service in the identified pilot zones of Bhambayi, Amatikwe and Ntuzuma G. The proposal involved the setting up of pilot projects in the Inanda-Ntuzuma area, building on and co-ordinated with the work that was underway in the pilot zones, with the aim that, after a two to three year trial period, the schemes would be sustainable. The objective was also to demonstrate, through the success of this project, an approach which might be adapted on a larger scale within this urban area and elsewhere, nationally and internationally.

In the co-operation agreement for the Inanda-Ntuzuma project, the partners agreed on a proposal from Mvula Trust to implement a sanitation project in the pilot area of Bhambayi. However, soon after the implementation of the BPD, the partners were informed that a housing development was about to start in Bhambayi, providing waterborne sewerage. It was thus decided to look at another sanitation project.

At this time, the anaerobic baffled reactor (ABR) has been identified as a possible on-site treatment option in peri-urban settlements. The results obtained from a WRC Project 853 – *The assessment of a baffled compartmentalised anaerobic digester for the treatment of high-strength or toxic organic industrial effluents* identified a need for further research on the ABR as a possible on-site treatment option. Following discussion between Vivendi Water and the Pollution Research Group, University of KwaZulu-Natal, a project on the use of the ABR to treat wastewater from peri-urban settlements was conceived.

The ABR project was approved by the BPD Steering Committee on 14 October 1999. The motivation for this project was that, in Durban, it could take approximately 20 years for water-borne sewage to be provided to some of the dense peri-urban communities of the Metro. Because of the lack of availability of water, both for consumption and household use, the wastewater produced from these areas is concentrated. Moreover, the ambient temperatures in KwaZulu-Natal are relatively high. In this context, it was hypothesised that the application of the ABR could provide an immediate solution to the sanitation problem in dense peri-urban areas, where it could be used to treat the domestic wastewater of a small community. The density of dwelling and the topography of these settlements negate the possibility of implementing treatment options such as anaerobic ponds or wetlands.

At the beginning of the ABR project, the local council predicted a population growth of 3.5 % that would result in the eThekwini Municipal region being fully populated within 35 years. Water-borne sanitation for the entire Metro region was planned, to be phased in over the next 50 years. However in following few years, the traffic department performed a detailed study to predict the impact of HIV/AIDS on the population in order to realign their policies with a more probable growth scenario. The middle scenario generated by this study indicated that by 2020, there would be no population growth from the figures that were current at the time of the study. If the population remains static, it was predicted that 16 000 houses would need to be built, but that the development would be limited to areas where existing services are available or can be reasonably extended.

With this in mind, eThekwini Municipality has been divided into areas where waterborne sanitation exists, and where it does not. Within the sewered area, the aim is to have 100% waterborne sanitation. To the sea-ward side of the area, where possible, sewers will be built or extended where appropriate. On the inland side of the area, however, on-site treatment or decentralised options will be necessary.

eThekwini Municipality has adopted a policy of supplying dry sanitation options to low-income households outside of the water-borne edge (Macleod, 2005) However, many householders aspire to water-borne sanitation, and there is a technology gap in water-borne sanitation options that are sustainable, affordable and practical for these conditions.

The ABR meets several critical requirements, namely, it does not require energy for operation; requires low maintenance; is compact and could be mass-produced. Several ABRs could service small sub-groups within an area and eventually connect to a sewer system for further treatment at a WWTP. Some limitations of the ABR are: no nutrient removal; and insufficient pathogen removal.

1.4 APPLICATION OF THE ANAEROBIC BAFFLED REACTOR FOR SANITATION

The ABR was considered to have an application as a technology for treating domestic wastewater for the following reasons:

- Water conservation is addressed by the ability the potential reuse of effluent.

- From an environmental life cycle assessment perspective, no electricity is expended on removing nutrients. Further, these nutrients could enhance the reuse value of the effluent. This provides a two-fold advantage, no adverse environmental impacts from the use of electricity in the sewage treatment processes, and credits for the environmental impacts associated with displacing the need for all the inputs associated with the production of food (provided the effluent is used for productive horticulture).

- Implementing an ABR for sanitation would reduce the cost of delivery of essential services, which would be of benefit in poverty alleviation.

- The national focus on food security coupled with the realisation that the use of potable water for horticulture is not sustainable implies that the nutrient rich effluent

should be regarded as a valuable resource provided sufficient pathogen removal is obtained.

- The simple on-site construction of the ABR has the potential for job creation within communities.

1.5 OBJECTIVES OF THE STUDY

The aims of this project were:

- To provide an appropriate sanitation system for application in peri-urban areas through scientific and engineering support to the KwaZulu-Natal Business Partners for Development water and sanitation project.

- To develop an anaerobic baffled reactor for use in pre-treating sewage from peri-urban areas.

- To monitor the performance of the anaerobic baffled reactor in a peri-urban area.

- To undertake pilot studies of the anaerobic baffled reactor at a WWTP.

- To gain scientific knowledge on the fluid mechanics and microbiology of the anaerobic baffled reactor for the pre-treatment of sewage from peri-urban areas.

- To contribute to the development and validation of a computer model for anaerobic digestion.

These objectives were not materially altered during the course of the project. This project has been a scoping study on many of the issues relating to the feasibility of implementing the ABR in peri-urban, rural or densely populated or informal communities, focussing on the microbiological and biochemical performance of the reactor, but also investigating community and institutional issues associated with the project.

It was not considered appropriate to implement a field ABR in a community situation within this project since there were several process issues that required further experimentation before the technology could be considered ripe for implementation in the field. The extra experimentation generated a considerable body of scientific information which has greatly enhanced the understanding of the dynamics of anaerobic digestion within the ABR.

1.6 PRODUCTS OF THE STUDY

The product from the study was to be information for water authorities, consultants, planners and designers on the design and operation of an ABR for the treatment of sewage from peri-urban communities.

1.7 PROJECT METHODOLOGY

The project was undertaken in a series of phases described in Sections 1.7.1 to 1.7.5.

1.7.1 Literature review

A review of literature pertinent to sanitation in South Africa (and developing countries in general), and anaerobic digestion (and anaerobic baffled reactors in particular) was undertaken.

1.7.2 Pilot-scale study

Based on previous experience with laboratory-scale ABRs, a pilot-scale reactor was designed and built and operated intermittently over a 4-year period.

1.7.2.1 Chemical characterisation

The performance of the pilot-scale reactor in terms of COD and pathogen removal, effluent characteristics and internal particulate and soluble component dynamics was investigated using a range of chemical techniques.

1.7.2.2 Microbial characterisation

The types, concentrations and biological activities of different micro-organisms in each compartment were determined using a range of microbiological and molecular techniques.

1.7.3 Community water use and wastewater characterisation study

Community water use and wastewater characterisation was studied using three approaches

- A study using, in part, the *geographical information system* (GIS) database at eThekwini Municipality attempted to identify the amount of water used by different communities.
- A household survey study attempted to quantify water use habits of low-income households.
- A wastewater characterisation study investigated the composition and flow of wastewater from a low-income peri-urban community.

1.7.4 Modelling study

A range of modelling techniques were used to simulate the performance of the pilot-scale ABR and to predict its performance on different wastewater feed characteristics so as to guide the design of future systems and to identify priority research areas where there is the greatest lack of knowledge.

1.7.5 Design guidelines

The combined experiences of all of the portions of this study were incorporated into a set of preliminary design guidelines for implementation of the ABR system in community sanitation applications.

2 LITERATURE REVIEW

This chapter is presented in two parts. The first is a literature review on sanitation options in South Africa, and the second presents a review on the anaerobic baffled reactor (ABR), including an introduction to anaerobic digestion.

2.1 ON-SITE AND DECENTRALISED DOMESTIC WASTEWATER TREATMENT

The primary function of a sanitation system is to create a physical barrier between humans and human excrement to prevent the transmission of pathogens via the faecal oral route. In densely populated communities, a further and equally important objective is to prevent contamination of the environment with large amounts of pollutants including organics, nitrogen and phosphorus, which lead to eutrophication of water resources and disruption of natural eco-systems. Water-borne sanitation where diluted toilet contents are collected and transported to an activated sludge treatment system is a high technology solution, however, it has large infrastructural, operational, maintenance and environmental costs which are economically, environmentally and socially unsustainable in many communities (Foxon et al., 2005).

The goal of sanitation can therefore be summarised as protection of human health and environment, with the added objective of protecting human dignity.

The South African national sanitation policy (DWAF, 2001) defines the minimum acceptable level of sanitation to be *a ventilated improved pit (VIP) toilet in a variety of forms, or equivalent, as long as it meets certain criteria in terms of cost, sturdiness, health benefits and environmental impact*. Officially, adequate service level options include:

- *various improved latrines*
- *septic tanks*
- *composting latrines*
- *full water-borne flush toilets*

These may be categorised as either dry or wet systems, where dry systems do not require the addition of water to dilute or transport human excrement during normal operation.

2.1.1 Dry on-site sanitation systems

There are two categories of dry sanitation systems, namely pit latrines and composting toilets. Rudimentary pits, i.e. those without improvements for fly and odour control and structural stability are not regarded as an acceptable level of sanitation and are therefore not discussed here.

2.1.1.1 Ventilated Improved Pit (VIP) latrines

A VIP toilet is a partly or fully-lined pit covered by a concrete slab, topped by a superstructure equipped with a safe pedestal, walls and a roof (Figure 2.1). The VIP is characterised by a vent pipe and fitted with a fly screen: movement of air across the top of the vent pipe creates a suction that draws air into the pit via the pedestal, and out via the vent pipe, releasing odours above the pit. Solar radiation enhances ventilation by heating air in the vent pipe, causing it to rise and be replaced by cooler air from the pit (Bester and Austin, 2000). The fly screen on the top of the vent pipe prevents flies entering the pit, and those that may have gained access via the pedestal from emerging from the pit, thereby limiting the spread of pathogens from toilets to people and food via flies.

Human excreta and anal cleansing material are dropped into the pit via the pedestal. The mechanism of treatment within the pit is not well understood, but is generally accepted to be anaerobic digestion (Still, 2002). The rate and stability of the anaerobic digestion process is strongly affected by the amount of water and alkalinity in the system, and these in turn depend on the geohydrological conditions of the pit location.

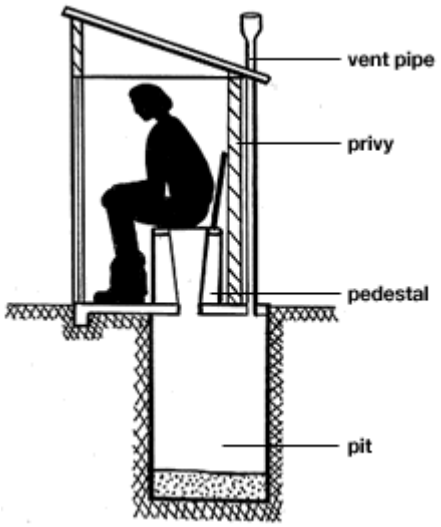


Figure 2.1: Example of VIP toilet showing vent pipe

2.1.1.2 Composting toilets

Composting toilets are designed to receive faeces and urine and to render them innocuous through dehydration (EcoSanRes, 2005). Ecological Sanitation (EcoSan) is the name often applied to sanitation systems that contain, treat and recycle excreta, usually without the addition of water and is the term commonly applied to urine diverting systems. In these systems, urine is separated from the faeces in urinals (for men) and urine diverting pedestals (for women) (Figure 2.2) for collection and reuse or discharge via a soak-away.

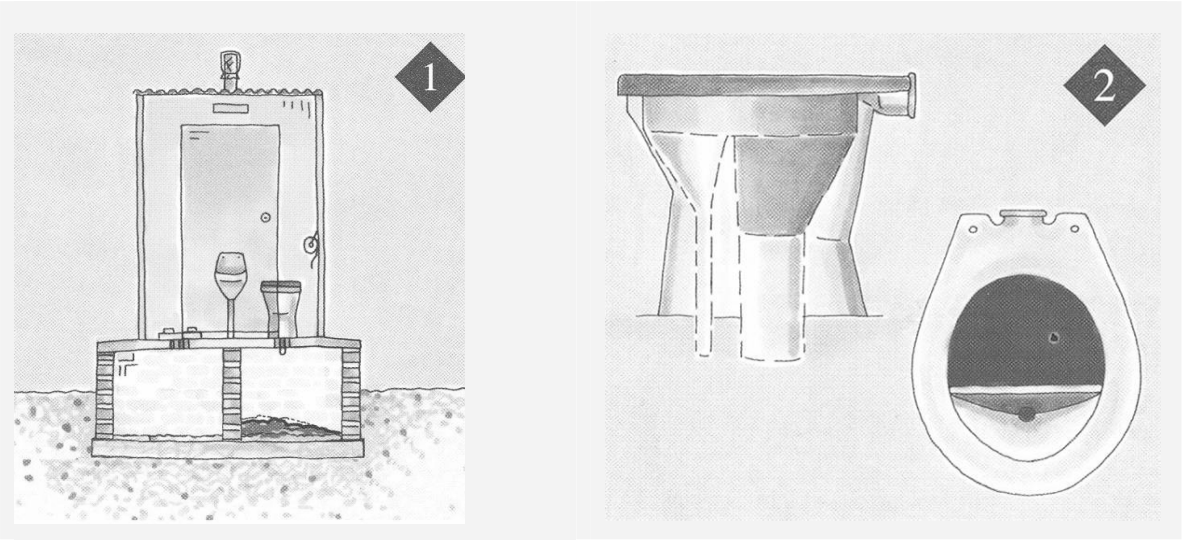


Figure 2.2: Double-pit composting urine-diverting toilet design implemented in eThekwini Municipality from 2003 showing (1) double vault with movable pedestal and (2) urine-diverting pedestal

Faeces are retained in a composting vault below the chamber and dehydration is assisted by the addition of drying materials such as wood ash, lime or soil. Sanitisation is achieved by increasing pH, dehydration or thermal degradation, and sanitised waste is removed from the vault after an appropriate sanitisation period. In many instances, two vaults are supplied, and the pedestal is switched from one vault to the next when the first vault is full. A standing period ensues while the second pit fills. When the second pit is full, the first pit is emptied and reused while the second pit stands full. Air circulation in the pit is achieved with the assistance of a vent pipe fitted with mesh to prevent ingress/egress of flies. Figure 2.2 shows the double-vault composting urine-diversion toilet system implemented in eThekwini Municipality from 2003.

There is no clear diagnosis on the mechanism of stabilisation of organic material in the vaults of composting toilets. Aerobic composting may occur to a certain extent at the air-solid interface, but it is expected that diffusion into the mass of solids is limited. Solid-state anaerobic digestion inside the mass of solids may occur, but the mechanisms and rate of this process have not yet been elucidated. Understanding of the processes in urine-diversion toilets is the subject of current research in WRC project K5/1629.

2.1.2 Wet on-site sanitation systems

Wet sanitation systems include any system that uses water to assist in the transport of excreta. The most common form of on-site wet sanitation is the septic tank, although several variations and improvements the septic tank concept are marketed.

2.1.2.1 Septic tank and soak-away systems

Septic tanks are the most commonly used unit for pre-treatment of domestic wastewater in on-site applications. A septic tank system consists of two units. The first unit is the

septic tank itself, which pre-treats wastewater by solids and scum retention and partial anaerobic digestion. The second part of treatment occurs in a subsurface wastewater infiltration system, a French drain or evapo-transpiration area where septic tank effluent is infiltrated into the ground via a series of gravel-filled trenches. From here water percolates through the ground, or is removed by evapo-transpiration. Micro-organisms associated with the soil and plant roots, as well as specific plants are able to effect significant nutrient removal from the wastewater. Effluent from these systems is rarely collected for reuse (USEPA, 2002).

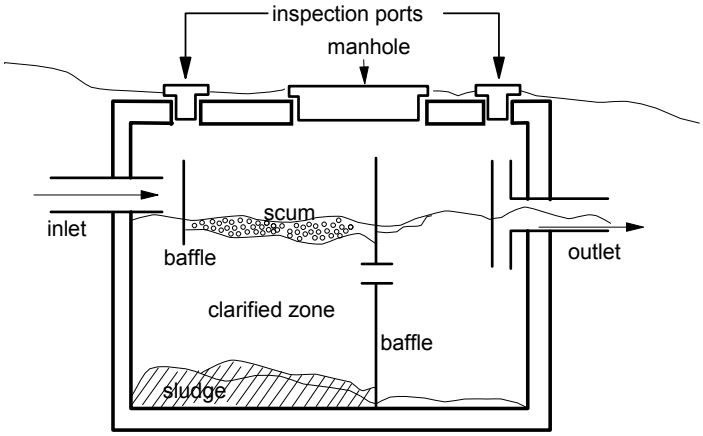


Figure 2.3: Example of septic tank construction showing internal baffle, inlet baffle, outlet tee piece, inspection ports and manhole

The United States Environmental Protection Agency (USEPA) has published an *Onsite Wastewater Treatment Systems Manual* which covers the subject of septic tank and soak-away design and performance. The tank is a covered, watertight rectangular, oval or cylindrical vessel that is usually buried. Dimensions vary, but it has been shown that the tank should be longer than it is wide or high. Primary treatment in the tank is due to wastewater retention under quiescent conditions. Solids and scum from the influent wastewater are separated in the tank by settling or floating. A population of anaerobic micro-organisms develops in the tank which partially digest solids and scum, and to a lesser extent, soluble organic material in the liquid phase. Digestion of scum and solids can result in reduction of up to 40% of retained material, however a slow accumulation of sludge is observed in the tank over a period of between 2 and 20 y, depending on loading (USEPA, 2002).

Figure 2.3 shows an example of a septic tank design. Anaerobic digestion in the tank generates methane (CH_4) and carbon dioxide (CO_2) gases that are commonly vented back through the building sewer and out of the house plumbing stack vent. Wastewater inlet structures in the tank are designed to reduce short-circuiting of incoming wastewater across the tank to the outlet. Outlet structures are designed to retain sludge and scum layers by drawing effluent from the clarified zone between the sludge and scum layers. The outlet should be fitted with an effluent screen (commonly called a septic tank filter) to retain larger solids that would otherwise be carried out in the effluent to the soak-away, where it could contribute to clogging and eventual system failure.

Inspection ports and manholes are provided in the tank cover to allow access for the periodic removal of tank contents, including the accumulated scum and sludge.

Compartmentalised tanks such as that shown in Figure 2.3, or tanks placed in series are reported to provide better suspended solids removal than single-compartment tanks alone, although results from different studies vary (Baumann and Babbitt, 1953; Boyer and Rock, 1992; Weibel et al., 1949, 1954; University of Wisconsin, 1978).

Septic tanks are reported to remove 60 to 80% of non-soluble material in domestic wastewater (USEPA, 2002). Solid and colloidal material is hydrolysed and acidified producing volatile fatty acids that are only partially converted to methane, and exit in the effluent stream. Biological oxygen demand (BOD) removal is typically in the order of 30 to 50% for a septic tank operating at a 48 h retention time (Boyer and Rock, 1992). Actual performance of the septic tank will depend on the ambient temperature, operating hydraulic retention time and presence of inert or micro-organism inhibiting chemicals in the influent. Table 2.2 presents septic tank effluent characteristics from a number of systems, before secondary treatment in a soak-away or other system.

Soak-away systems provide a degree of wastewater treatment and dispersal through soil purification processes and ground water recharge. The performance is dependent on the treatment efficiency of the septic tank, the method of wastewater distribution and loading to the soil infiltrative surface, and the properties of the vadose and saturated zones underlying the infiltrative surface. Considerable data on the treatment efficiency of soak-away systems are available in the literature (USEPA, 2002). High removal rates of BOD, suspended solids, faecal coliforms and surfactants have been observed within a few metres of unsaturated, aerobic soil. Phosphorus and metals are removed through adsorption, ion-exchange and precipitation reactions, although the retention capacity of the soil has a limit that depends on specific soil properties including soil mineralogy, organic content, pH, redox potential, and cation exchange capacity. Pathogen survival rates have been found to vary with a number of factors including initial pathogen load, temperature, humidity and solar radiation.

Table 2.1: Typical pathogen survival times in water, sewage and soil at 20 ºC to 30 ºC

	Typical survival times in days	
Pathogen	**in fresh water and sewage**	**in unsaturated soil**
Viruses		
Enteroviruses	<120 but usually <50	<100 but usually <20
Bacteria		
Faecal coliforms	<60 but usually <30	<70 but usually <20
Salmonella spp.	<60 but usually <30	<70 but usually <20

Shigella spp.	<30 but usually <10	
Protozoa		
Entamoeba histolitica cysts	<30 but usually <15	<20 but usually <10
Helminths		
Ascaris lumbricoides eggs	many months	many months

Table 2.1 shows typical pathogen survival times in fresh water, sewage and unsaturated soil at 20 ºC to 30 ºC as presented in USEPA (2002). Bacterial and protozoan pathogens have relatively short survival times (less than a month), but viruses and helminth eggs (e.g. *Ascaris*) can survive for considerably longer periods.

Table 2.2 Effluent characteristics from septic tanks (before being discharged e.g. to soak-away)

Study ref	Type	Location	No. of tanks/ homes	No. of samples	BOD mg/ℓ	COD mg/ℓ	TSS mg/ℓ	TKN mgN/ℓ	TP mgP/ℓ	Oil and grease mg/ℓ	*E. Coli* log (cfu/100 mℓ)	pH	Flow ℓ/d
1	cluster	Texas, USA	9			266		29.5	8.2		5.0	7.4	
2	domestic	Wisconsin, USA	7	150	138	327	49	45	13		3.6		
3	cluster	Wisconsin, USA	90		168	338	85	63.4	8.1		6.3	6.8-7.4	140
4	domestic	Wisconsin, USA	33	140-215	132	445	87	82	21.8		5.5		
5	domestic	Oregon, USA	8	56	217		146	57.1			5.4		
6a	cluster	Oregon, USA	11		157	276	36	41		65		6.4-7.2	150-230
6b	cluster	Oregon, USA	Small community		118	228	52	50		16		6.4-7.2	180
6c	cluster	California, USA	330		189	284	75			22		6.5-7.8	150-220
7	domestic	Florida	8	36	141		161	39	11	36	4.1-7.2		
8	domestic	Florida	1	3	179		59	66	17	37	6.0		
9	domestic	SW Cape, RSA			26 (DOC)				14.2		6.6	6.8	
10[1]	domestic	Australia			330		660	250	36		6		

[1] 80th percentile values are reported, i.e. 80% of systems sampled had measured values equal to or less than reported value.

1. Brown et al., 1977
2. University of Wisconsin 1978
3. Otis, 1978
4. Harkin et al., 1979
5. Ronayne et al., 1982
6. Bowne, 1982
7. Ayres Associates, 1993
8. Ayres Associates, 1996
9. Wright, 1999
10. Charles, 2004

2.1.3 Decentralised sanitation

Decentralised sanitation of any kind will fall into the category of wet sanitation since water is required to achieve the transport of waste. Decentralised sanitation describes any system where raw or pre-treated wastewater is collected and transported to a treatment site located off the plot on which the wastewater arises via small-bore sewers.

2.1.3.1 Solids-free sewers

A solids-free sewer (SFS) system uses an on-site tank to settle out solids from sewerage, allowing liquid effluent to overflow to a low-bore sewer along which it flows to a treatment and/or disposal point. Solids remain in the tank where they partially undergo anaerobic digestion, and accumulate. Periodically, the on-site tank must be emptied; the contents are withdrawn by a vacuum tanker and discharged at a wastewater treatment facility (Du Pisani, 1998). These systems can be operated with low flows since solids do not have not be transported far, and the reticulation system is constructed of smaller bore piping than used in conventional sewer systems. Consequently, the infrastructural costs of this system are low, but participation by both users and municipalities is required to ensure that the system functions correctly.

2.1.3.2 Package plants

Laas and Botha (2004) defined package plants as …*any privately owned on-site sewage treatment plant discharging less than 2000 cubic metres of effluent per day....* .Package plants are small-scale aerobic systems that usually collect and treat wastewater from a number of homes. They usually include an aerobic stage that oxidises COD and nitrogen before disinfection and discharge to a nearby river, stream or wetland. Some systems are technically able to achieve sufficient nutrient removal (nitrogen and phosphorus) through incorporation of anaerobic/anoxic zones. These systems are usually prefabricated according to a standard design with a specific loading capacity.

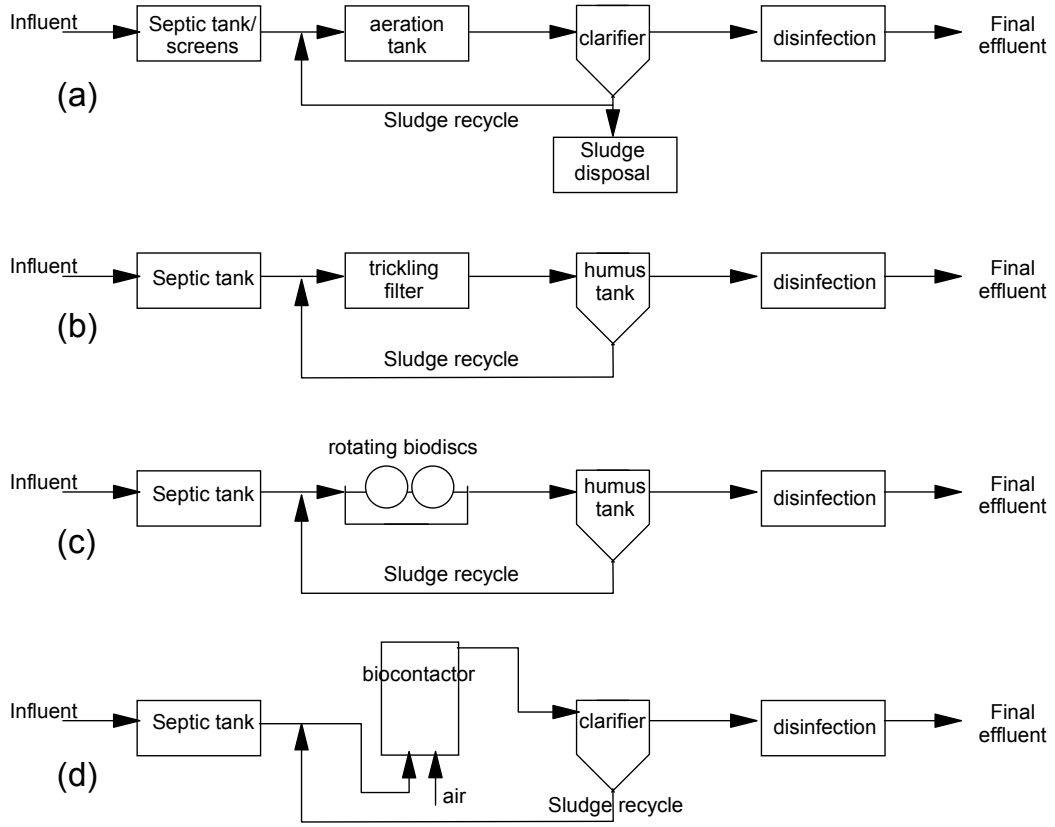


Figure 2.4: Typical configurations of package plants implemented in eThekwini Municipality (a) activated sludge plant; (b) trickling filter plant; (c) rotating biocontactor plants; and (d) submerged biocontactor plant

A package plant operates according to the same principles as a full-scale activated sludge WWTP. Depending on the complexity of the system, it may include sludge and/or nitrate recycles, sludge digestion and thickening, flocculation, filtration and disinfection. Laas and Botha (2004) describe four common package plants that are implemented in the eThekwini Municipal region:

- *Activated sludge plants* (Figure 2.4a) consist of screens or a septic tank followed by an aeration tank, clarification and disinfection. Clarifier underflow is partially recycled to the aeration tank and partially wasted.

- *Trickling filter plants* (Figure 2.4b) consist of a septic tank followed by a trickling filter in which wastewater is trickled through a bed of stones or packing media on which a biofilm develops. The biofilm aerobically treats the wastewater as it passes through. Trickling filter effluent passes through a humus tank where sludge removed is returned to the septic tank, before disinfection and discharge.

- *Rotating biocontactor (biodisc) plants* (Figure 2.4c) have a similar configuration to the trickling filter plants, but have rotating discs partially submerged in a trough

containing the septic tank effluent in the place of the trickling filter. In these systems, a biofilm develops on the rotating discs that is exposed to the air during rotation. When the film becomes too thick, it sloughs off.

- *Submerged biocontactor plants* (Figure 2.4d) are similar to activated sludge plants, but instead of an aeration tank, a packed biocontactor follows a septic tank; septic tank effluent is fed to the bottom of the biocontactor, where an air supply is also introduced. Effluent passes out of the top of the biocontactor to a clarifier.

All package plants have power requirements for aeration and/or pumping, and although they are marketed as requiring little maintenance, they must be regularly monitored. Package plants are often implemented in middle and high income cluster housing developments in South Africa, and a large proportion of these have not been able to achieve an effluent quality that complies with the General Limit Values for the discharge of domestic wastewater as required by the General Authorisations promulgation in terms of section 39 of the National Water Act, 36 of 1998 (Laas and Botha, 2004). Problems observed with these systems are mostly related to bad odours and poor effluent quality, and these can usually be attributed to poor maintenance, overloading or inappropriate use.

2.2 THE ANAEROBIC BAFFLED REACTOR

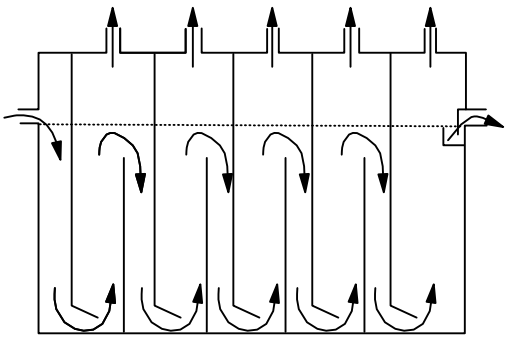


Figure 2.5: Diagram of an anaerobic baffled reactor (ABR) showing hanging and standing baffles. Curved arrows indicate liquid flow, while straight arrows represent gas production.

The anaerobic baffled reactor is a high rate anaerobic digester that has been used to pre-treat or co-digest high strength or toxic industrial effluents. Its application in the treatment of low-strength wastewaters has been tested on a laboratory-scale and two full-scale applications for the primary treatment of domestic wastewater exist in Colombia (Orozco, 1997) and Italy (Garuti et al., 2004).

The ABR is similar in design and application to the up-flow anaerobic sludge blanket (UASB) but requires no special granule formation for its operation. Bachmann et al. (1985) developed the ABR, although, baffled reactor units had previously been used to generate methane-rich biogas as an energy source (Chynoweth et al., 1980).

The ABR has alternately hanging and standing baffles (Figure 2.5), which divide it into compartments. The liquid flow is alternately upward and downward between the partitions. The down-flow chamber is narrower than the up-flow chamber so that the up-flow-velocity in the up-flow chamber is lower than the average velocity through the reactor. A sludge blanket accumulates by settling in the bottom of each compartment, and the liquid flow is forced through this blanket as it passes under each hanging baffle. Good contact between wastewater flow and active biomass is ensured by this design. In principle, all phases of the anaerobic degradation process can proceed simultaneously in each compartment. However, the sludge in each compartment will differ depending on the specific environmental conditions prevailing and the compounds or intermediates to be degraded (Nachaiyasit and Stuckey, 1997).

The hydrodynamics and degree of mixing that occur within a reactor of this design strongly influence the extent of contact between substrate and bacteria, thus controlling mass transfer and potential reactor performance. Micro-organisms within the reactor gently rise and settle due to the flow characteristics and gas production, however, their rate of movement along the reactor is slow. The main driving force behind reactor design has been to enhance the solids retention capacity.

The reactor design is simple, with no moving parts or mechanical mixing, making it relatively inexpensive to construct. There is no requirement for biomass with unusual settling properties. Sludge generation is low and solids retention time (SRT) is high; this is achieved without the need for biomass to be fixed to media particles or a solid-settling chamber. Gas separation is not required.

The ABR has been found to be stable to hydraulic and organic shock loads and the reactor configuration provides protection of the biomass to toxic compounds in the influent (Barber and Stuckey, 1999).

In order to be able to understand the mechanisms of wastewater treatment in an ABR, it is necessary to understand the concepts and processes of anaerobic digestion. Sections 2.2.1 to 2.2.5 present a detailed overview of anaerobic digestion.

2.2.1 Anaerobic Digestion: An introduction

Anaerobic digestion converts organic matter to CO_2 and CH_4 gases along a series of interrelated biochemical pathways (Bailey and Ollis, 1986). Traditionally, anaerobic digestion has been used for passive treatment of domestic wastewater in septic tanks, but it is best understood as a process in the pre-treatment of high strength industrial effluents (Speece, 1996). Consequently anaerobic digestion of domestic wastewater is considered to be a *low-strength* application. This should be seen in the context of industrial applications which are often operated at inlet COD concentrations that exceed 5 000 mgCOD/ℓ (Speece, 1996). In contrast, domestic wastewater with a COD value of 1 000 mgCOD/ℓ is considered to be a high-strength domestic wastewater.

In aerobic respiration, molecular oxygen serves as an external electron acceptor, and there is a large flow of electrons and energy associated with these conversions. In the absence of an external oxygen supply, some carbon atoms associated with organic

substrates are reduced (ultimately to CH_4) by accepting electrons from other compounds that are oxidised to carbon dioxide. The conversions are therefore characterised by much smaller energy and electron fluxes resulting in a smaller driving force for the reactions.

Anaerobic conversion to methane gas therefore provides relatively little energy to micro-organisms, resulting in a slow growth rate and only a small portion of the waste being converted to new biomass (i.e. low sludge yields). The conversion of organic material to CH_4 removes COD from the liquid phase. Production of CO_2 gas does not indicate COD reduction; in anaerobic digestion, where there is no external oxygen supply, CO_2 production depends on internally available oxygen in the substrate (such as in the acid group of organic acids i.e. $CH_3(CH_2)_xCOOH$) and therefore does not contribute to the *oxygen demand* of the wastewater measured by the COD analysis.

As much as 80 to 90 % of the degradable organic portion of a waste can be stabilised in anaerobic treatment, even in highly loaded systems (Speece, 1996).

Figure 2.6 presents a conceptual flow of COD in catabolic anaerobic digestion (i.e. ignoring COD converted to biomass), from a hypothetical substrate containing 30% each of carbohydrate, protein and lipid and 10% inert material. For complete digestion of the biodegradable COD (complete stabilisation) all COD is recovered as CH_4 gas.

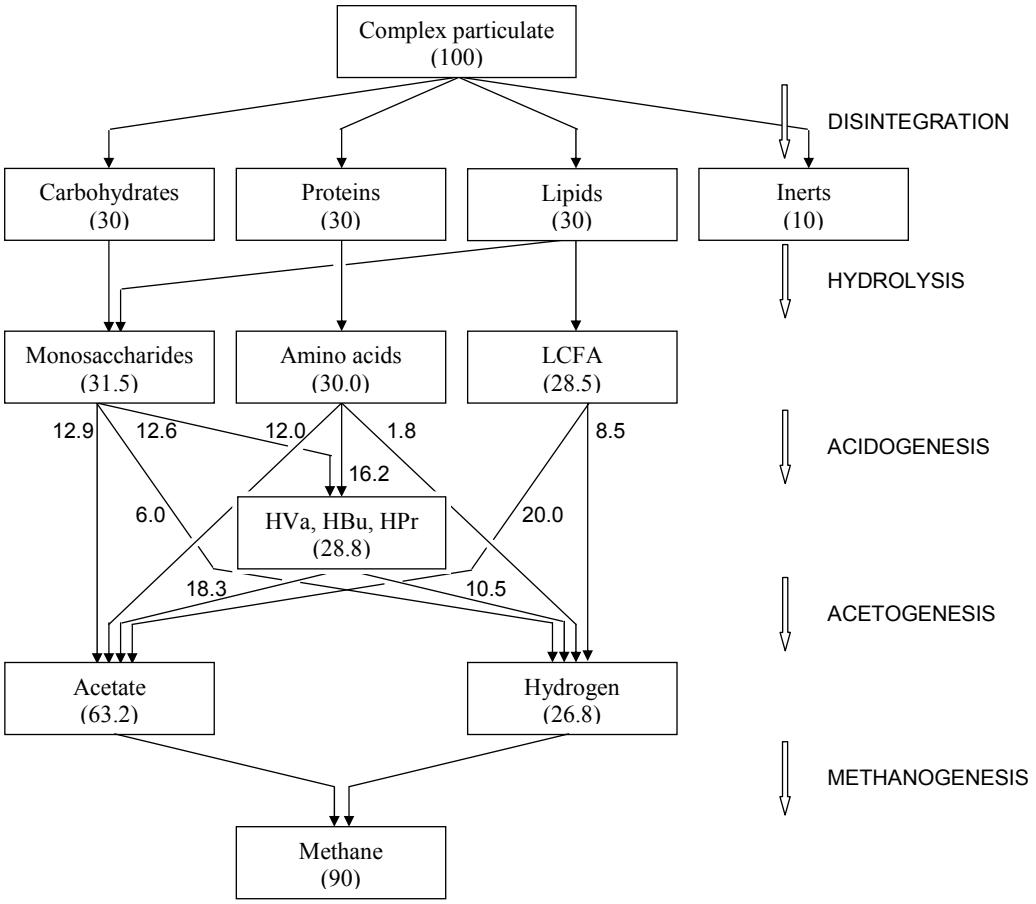


Figure 2.6: Flow-diagram for the anaerobic degradation of a composite particulate material, as implemented in ADM1 (from Batstone et al 2002). Valerate (HVa), Butyrate (HBu) and Propionate (HPr) are grouped for simplicity. Figures in brackets indicate COD fractions

There are 5 major sub-processes within the overall anaerobic digestion process, which are each made up of steps mediated by different microbial groups. Sections 2.2.1.1 to 2.2.1.5 summarise the main features of the sub-processes primarily responsible for conversion of COD in anaerobic digestion, while sections 2.2.1.6 to 2.2.1.9.describe processes that may occur in anaerobic digestion under appropriate conditions.

2.2.1.1 Disintegration and hydrolysis

Complex materials such as high molecular weight proteins, carbohydrates and lipids are hydrolysed extracellularly by enzymes excreted by hydrolytic bacteria, with some release of energy. This process can be divided into the disintegration of composite particulate material into smaller carbohydrate, protein and lipid fractions and the hydrolysis of large molecular weight compounds to long chain fatty acids and the monomers of sugar and protein compounds; to simple sugars; and to amino acids.

2.2.1.2 Anaerobic oxidation

Long chain fatty acids are oxidised to simple organic acids in a process called anaerobic oxidation, in which the carbon chain is sequentially shortened by two carbon atoms at a time. The final product of fatty acids with an even number of carbon atoms is acetate only; when the fatty acid has an odd number of carbon atoms, one mole of propionate is produced per mole of substrate. Relatively large amounts of dissolved hydrogen are released in this process.

2.2.1.3 Acidogenesis

Amino acids and simple sugars are fermented by the next category of bacteria, termed acidogenic or *acid formers* that produce simple organic acids such as acetic, propionic, butyric and lactic acids. The organic acid end product of acidogenesis is determined by the environmental conditions (Mosey, 1983). Different amounts of H_2 are produced during acidogenesis depending on the acidogenesis end product.

2.2.1.4 Acetogenesis

A further category of bacteria (acetogenic) ferment propionic, butyric and lactic acids to acetic acid. In most cases, each group of acetogens can only ferment one type of organic acid. This is considered a separate step to acidogenesis since there is not a large pH affect associated with the conversion of higher acids to acetic acid.

2.2.1.5 Acetoclastic methanogenesis

The final stage in anaerobic digestion is the conversion of acetic acid to methane and carbon dioxide by a group of Archaea known as acetoclastic methanogens. The conversion to methane is the only strictly anaerobic step which results in the removal of chemical oxygen demand to the gas phase (see section 2.2.1).

2.2.1.6 Hydrogenotrophic methanogenesis

Hydrogenotrophic methanogenesis is the production of methane from dissolved hydrogen and carbon dioxide by a select group of slow-growing methanogens. This process can account for up to 30 % of the methane produced by anaerobic digestion of an organic waste.

2.2.1.7 Homoacetogenesis

Homoacetogenesis is the generation of acetic acid from dissolved hydrogen and carbon dioxide. Homoacetogens are one of the most versatile physiological groups among the anaerobic bacteria. They utilise and transform one-carbon compounds and can carry out incomplete oxidation of reduced fermentation products released by other fermenting bacteria. Homoacetogens can use various substrates sequentially or simultaneously and may constitute an energy link from hydrogen, via acetate to heterotrophic methanogens.

2.2.1.8 Sulphate reduction

The presence of any sulphate in the waste will result in sulphidogenesis, the generation of sulphide from sulphate. As with methanogenesis, this process can be either acetoclastic or hydrogenotrophic. Organisms reducing sulphur can obtain the electrons directly by oxidising organic acids, or by oxidising the hydrogen produced by acetogens. Additionally, organic acids are used as a carbon source, and as a result, organisms reducing sulphur compete with the majority of groups in anaerobic digestion (Kalyuzhnyi and Fedorovich, 1998). Further complicating the effect on anaerobic systems, the reduced product, sulphide, is inhibitory to different extent for almost all microbial groups. Reduced sulphide has a similar acid-base system to the inorganic carbon system. Hydrogen sulphide is also a gas phase component, with a relatively high solubility.

2.2.1.9 Denitrification

Dissimilatory nitrate reduction, a form of anaerobic respiration, is the reduction of nitrate (NO_3^-) to nitrogen oxides (such as NO_2^-, NO, N_2O) and N_2. Denitrification has a higher yield per unit of substrate consumed than methanogenesis and competes for the same substrate (acetate). Denitrification intermediates have also been found to inhibit methanogenesis. This results in a decrease in methane production and an increase in alkalinity. In an overall methanogenic system, nitrate reduction can have significant impact on both the carbon and electron flow, on microbial competition and inhibition, and on gas composition (Batstone et al., 2002).

2.2.2 Interaction of sub-processes in anaerobic digestion

Redox potential and acidity/alkalinity of the liquid phase are determined by intermediates and by-products of anaerobic digestion, H_2, CO_2 and VFA, and affect the available energy of many of the sub-processes (See e.g. Smith and Van Ness, 1987 for calculations of Gibbs free energy of reaction under different redox/pH conditions). Methanogenesis is particularly vulnerable to low pH conditions and quickly becomes inhibited if the pH drops below a value of 6.5. The overall anaerobic digestion process is therefore precariously balanced between acid producing acido/acetogenesis and acid consuming methanogenesis; any event that causes an increase in acid production rate (e.g. high organic load) or low rate of acid removal (due to e.g. a decrease in buffering and therefore pH) can cause souring where low pH causes complete inhibition of methanogenesis and the ability of the system to remove produced acid fails. Although it is possible to recover from souring, the overall rate of anaerobic digestion/stabilisation decreases considerably. Other anaerobic digestion products and intermediates (e.g. H_2, NH_3) can also cause inhibition of different sub-processes, with implications on micro-organism selection and overall rate of stabilisation. For a comprehensive presentation of this aspect of anaerobic digestion, see e.g. Batstone et al. (2002) or Remigi and Foxon (2004).

2.2.3 Stoichiometry of anaerobic digestion

An understanding of the stoichiometry of the individual processes is necessary in order to construct mass balance relationships between substrates and products, cell yields and to calculate pH changes.

2.2.3.1 Stoichiometry of dissimilatory (energy generating) processes

The biochemical pathways of anaerobic digestion are complex (as depicted in Figure 2.6) and moreover change to favour the most energetically favourable route according to operating conditions, available substrate/intermediate concentrations and micro-organisms. Standard stoichiometry to describe the sub-processes in anaerobic digestion, expressed in Peterson Matrix form are defined in the Anaerobic Digestion Model No. 1 (Batstone et al., 2002). These are necessarily a simplification of the complexity observed in actual systems, but are currently regarded as the highest order of complexity appropriate for simulating real systems with the available understanding and feasible measurements of the process. The Anaerobic Digestion Model No. 1 is a good starting point for modelling any anaerobic digestion process.

2.2.3.2 Stoichiometry of assimilatory (growth) processes

The amount of growth that is associated with the consumption of a unit of substrate is described by the yield coefficient. Each micro-organism that catalyses a process in anaerobic digestion has an independent yield coefficient determined by the free energy of reaction associated with the conversions it mediates. However, different sub-processes with similar functions tend to have similar yields. Generally, the yield of methane generating micro-organisms is lower than that of acid producing organisms.

Yields are reported in terms of COD(biomass)/mg COD(substrate) or mgVSS/mgCOD (substrate). Henze et al. (1997) report acidogenic micro-organism yields in the range of 0.2 to 0.3 mg COD/mg COD) or approximately 0.15 to 0.20 mgVSS/mgCOD, while maximum yields for methanogenesis are 0.04 to 0.05 mgCOD/mgCOD or 0.03 to 0.04 mgVSS/mgCOD. The overall maximum yield is therefore between 0.25 and 0.35 mg COD/mgCOD, although in most cases, substrate limitations result in a considerably lower yield of between 0.05 and 0.1 mgCOD/mgCOD (Henze et al., 1997).

2.2.3.3 Reaction rates in anaerobic digestion

Each of the 5 sub-processes described in Figure 2.6 proceed at different rates, depending on operating conditions and substrate concentrations. The overall rate of stabilisation therefore will be limited by the slowest, or rate limiting step. The rate limiting step will be different in different systems, and may even change from one process to another with time within a system. Extracellular process kinetics tend to be slow, and are generally poorly characterised. For this reason, disintegration and hydrolysis are often lumped in a single process with first order or surface saturation-type kinetics. Acid and methane producing steps usually exhibit a Monod-type relationship between reaction rate and substrate concentration, although methane production is generally slower than acid production. As a general rule, when the primary substrate is

soluble or labile, the rate-limiting step will be methanogenesis, while extracellular processes will dominate the overall kinetics of digestion of particulate or refractory substrates. In all processes, adverse conditions (e.g. low pH for methanogenesis or high dissolved H_2 concentration for acid production) will slow or halt the reaction in question.

2.2.4 Factors effecting the rate and extent of anaerobic digestion

Overall environmental conditions such as pH value, temperature, essential trace nutrients and toxicants can play a major role in modifying the individual rate equations.

2.2.4.1 Temperature

For cryophilic (0 to 25ºC) and mesophilic (20 to 40ºC) temperature ranges, the change in reaction rates of anaerobic processes with temperature can be described by an Arrhenius type exponential equation (Equation 2-1):

$$\mu_{\max}(T) = \mu_{\max}(20°)e^{\kappa(T-20)} \qquad \textbf{Equation 2-1}$$

Each subprocess will have different temperature coefficients (κ)

Anaerobic digestion between 40ºC and 50ºC is unstable and prone to failure. At temperatures above 50ºC, thermophilic micro-organisms operate at higher rates than their mesophilic counterparts, but little or no activity occurs above 70ºC.

2.2.4.2 pH

A pH range of 6 to 8 is generally considered acceptable (Henze et al., 1997), although the effect of pH is different for each of the subprocesses. Methanogenesis is particularly sensitive to pH values, exhibiting a rapid decrease in maximum reaction rate when the pH drops below a value of 6.5, or exceeds 8.5. Low pH values therefore have the ability to cause reactor failure by inhibiting removal of acid by methanogenesis, and thereby causing accumulation of volatile acids and therefore further inhibition.

2.2.4.3 Nutrients

The nutrient requirements of anaerobic digestion are relatively small since nutrient requirements are essentially linked to growth and anaerobic processes are characterised by low growth yields. As with all biological processes, nitrogen, phosphorus, sulphur and iron are required for growth, as well as a host of other micro-nutrients that are required in very small amounts. These nutrients are found in sufficient quantity in faecal material to supply anaerobic micro-organism requirements for complete stabilisation of the biodegradable material.

2.2.4.4 Toxicants

Anaerobic processes show similar patterns of inhibition to aerobic processes, and therefore cannot be regarded as being inherently sensitive to inhibition. As with all inhibition effects, the slowest processes will succumb first, and in the case of anaerobic

digestion, this is usually methanogenesis. The failure of methanogenesis quickly results in acid accumulation and failure of digestion, and therefore the overall sensitivity of anaerobic processes to toxicants is higher than aerobic processes (Henze et al., 1997).

2.2.5 Physico-chemical processes in anaerobic digestion

The physicochemical system can be defined as non-biologically mediated processes that commonly occur in anaerobic reactors. There are three broad types, listed below:

- *Liquid-liquid processes*: this mainly refers to ion association and dissociation with hydrogen and hydroxide ions. There are a number of important compounds, which have dissociation constants close to the operating pH of anaerobic systems.

- *Gas-liquid processes* (i.e., gas liquid transfer of carbon dioxide, methane, hydrogen, hydrogen sulphide and nitrogen gases).

- *Liquid-solid processes* (i.e., precipitation/solubilisation of ions).

Important physico-chemical subsystems include inorganic carbon, organic acids dissociation, sulphate, sulphite, sulphide, ammonia/ammonium, oxidised nitrogen, phosphate, and the gas liquid interaction hydrogen and methane gases.

2.2.6 Research on the performance of the ABR

This section provides a brief review of research on the ABR that may be relevant to designing systems for and interpreting data from an ABR treating domestic wastewater.

2.2.6.1 Start-up

One of the major problems associated with anaerobic treatment systems is the start-up procedure. The overall objective of start-up is the development of the most appropriate microbial culture for the waste stream to be treated. Initial loading rates should be low so that the slow growing micro-organisms are not over-loaded and both gas and liquid up-flow velocities should be low to facilitate flocculent and granular sludge growth. The recommended initial loading rate is ca. 1.2 kg COD/m³.d (Speece, 1996). Barber and Stuckey (1997) showed that by starting with a long hydraulic retention time (HRT) (80 h) and gradually reducing it, in a stepwise fashion, whilst keeping the substrate concentration constant, greater reactor stability is maintained, with superior performance in comparison to a reactor started up with a constant and low hydraulic retention time coupled to a stepwise increase in substrate concentration. This assessment was based on improved solids accumulation, promotion of methanogenic populations and faster recovery to hydraulic shocks.

2.2.6.2 Residence time distribution

Grobicki and Stuckey (1992) conducted a series of residence time distribution studies in the ABR. They found that the ABR could be characterised as a series of continuous stirred tank reactors (CSTRs) and that there were low levels of dead space (8 to 18 %

hydraulic dead space) in comparison with other anaerobic reactor designs. Investigations of the hydrodynamics to date have not taken into account various other factors which may be influential, such as: biogas mixing effects, viscosity changes due to extracellular polymer production and biomass particle size.

2.2.6.3 Response to hydraulic and organic shock loads

The ABR has been shown to tolerate hydraulic and organic shock loads. To a steady-state reactor, with an hydraulic retention time of 20 h and an organic loading rate of 4.8 kg COD/m³ (synthetic carbohydrate / sucrose protein feed), Grobicki (1989) introduced a hydraulic shock by decreasing the hydraulic retention time to 1 h, for a period of 3 h. The reactor returned to its previous COD removal efficiency of > 95 % within 24 h of resuming normal operating conditions. Less than 15 % of the active biomass was lost. In a similar experiment, the organic loading rate was increased to 20 kg COD/m³ and, under these conditions a COD removal efficiency of 72 % was still achieved.

2.2.6.4 Low-strength applications

Several authors have treated low-strength wastewaters effectively in the ABR (Barber and Stuckey, 1999). Dilute wastewaters inherently provide a low mass transfer driving force between the biomass and substrate, subsequently reducing biomass activities according to Monod kinetics. As a result, treatment of low-strength wastewaters has been found to encourage the dominance of scavenging micro-organisms, such as *Methanosaeta* species (Polpraset et al., 1992). Biomass retention is significantly enhanced due to lower gas production rates, suggesting that low hydraulic retention times are feasible during low-strength treatment. Witthauer and Stuckey (1982) (cited in Barber and Stuckey, 1999) observed that biogas mixing was greatly reduced and this resulted in minimal biomass/substrate mass transfer. The authors suggested that when treating dilute wastewaters, baffled reactors should be started-up with relatively high biomass concentrations in order to obtain a sufficiently high sludge blanket and better gas mixing.

2.2.6.5 Effect of temperature

Nachaiyasit and Stuckey (1997) did an extensive study on operation of the ABR at low temperatures. Generally, biochemical reactions double in relative activity for every 10 C° increase in temperature, however, these authors found no significant reduction in overall COD removal efficiency when the temperature of an ABR was dropped from 35 °C to 25 °C. Further reduction in temperature, to 15 °C, resulted in a 20 % decrease in COD removal. Changes in performance were gradual which is advantageous since this slow response would inherently provide improved protection to temperature shocks, in comparison to other reactor systems.

2.2.6.6 Sulphate reduction

Fox and Venkatasubbiah (1996) investigated the efficiency of sulphate reduction in the ABR. Reactor profiles showed that sulphate was almost completely reduced to sulphide within the first compartment and a concomitant increase in sulphide levels along the

reactor indicated that sulphate redirected electron equivalents to hydrogen sulphide production in preference to methane production. Increasing sulphate concentrations with glucose and isopropanol as a labile substrate (increase the COD:SO_4 ratio) showed an inhibition of sulphate reduction caused by elevated sulphide concentrations. Volatile fatty acids concentrations as high as 4 500 mg/ℓ were observed during inhibition.

2.2.6.7 Phase separation

The most significant advantage of the ABR is reported to be its ability to separate acidogenesis and methanogenesis longitudinally down the reactor, allowing the reactor to behave as a two-phase system without the associated control problems and high costs (Weiland and Rozzi, 1991). However evidence of this phase separation is limited in low strength applications (Hassouna and Stuckey, 2005).

2.2.6.8 Recovery after inactive period

Manariotis et al. (2002) used a 14.7 ℓ, three-chamber anaerobic baffled reactor (ABR) to evaluate the treatment of low-strength synthetic wastewater (COD of 300 to 400 mg/ℓ) and assess process reactivation after a prolonged period of inactivity. The reactor was inoculated with anaerobic seed and start-up was immediate. At 26 °C and hydraulic retention times of 24 and 12 h, COD removal averaged 87.2 and 91.0%, respectively, and biogas yield for CH_4 was 0.184 and 0.102 m^3 CH_4/kg COD removed, respectively. The ABR was reactivated after two years without feeding. Response was prompt and removal averaged 85.3% even during the initial 10 d period.

2.2.6.9 Comparison between septic tanks and baffled reactors

Wanasen (2003) undertook a laboratory-scale comparison between a conventional septic tank design, and septic tanks modified with 1 and 2 internal baffles to create a 2 and 3 compartment ABR. At a hydraulic retention time of 48 hours, the baffled septic tanks had approximately the same removal efficiencies (in terms of COD, BOD, TS, and TSS) as the septic tank. However, when operated with a hydraulic retention time of 24 h, the removal efficiency in the conventional septic tank was reduced by up to two-fold compared to the baffled reactors. The three-baffled septic tank removal efficiencies were 10 to 15% higher than observed in the conventional septic tank.

A total solids mass balance was undertaken which clearly showed that the baffled septic tanks retain much more solids than the conventional septic tank; 45 to 55% of solids are retained by the baffled tanks at an hydraulic retention time of 48 h, while only 30% was retained in the conventional septic tank. With a hydraulic retention time of 24 h, and higher TS loading rates, the three baffled septic tank was able to retain around 65% of the solids, the two-baffled septic tank retained about 40% of the solids, and the septic tank retained only about 15% of the solids.

2.2.6.10Modelling of biomass bed expansion during intermittent flow operation

Garuti et al. (2004) performed experiments on a 24.2 m^3 2-compartment hybrid ABR supported by laboratory-scale biomass transport experiments on a 9.4 ℓ UASB reactor. Measurements of TSS concentrations at two heights on each of the ABR compartments and in the effluent of the UASB were obtained, and sludge bed height in the UASB was measured visually. A mathematical model of sludge bed expansion was developed by considering the sludge column to be divided into 6 height zones and modelling the TSS dynamics in each zone. Predictions of sludge bed height with up-flow velocity dynamics were obtained, and it was concluded that short burst of flow at high flow rates resulted in better overall sludge retention than longer periods of flow at a lower flow rate, (but overall equal average hydraulic load).

2.2.7 Full-scale ABR installations

This section describes full-scale applications of the ABR treating domestic wastewater.

2.2.7.1 Tenjo, Colombia

Two 8-compartment ABRs were constructed in a Columbian town, Tenjo (population <2 500) to treat a combined stream consisting of industrial dairy waste and domestic wastewater (Orozco, 1997, cited in Barber and Stuckey, 1999). The two 197 m^3 reactors removed an average of 70% of COD and 80% of suspended solids from the wastewater over a two month period at an organic loading rate of 0.85 kg/m^3d and a design up-flow velocity of 3.00 m/h.

2.2.7.2 Biancolina, Italy

A hybrid *an*aerobic-*an*oxic-*ox*ic (ANANOX®) system was implemented at Biancolina WWTP near Bologna, Italy. This plant serves the small village of Biancolina (ca. 350 p.e.). The ANANOX® system consists of a two compartment ABR with a third anoxic compartment and a fourth compartment which operates as a sludge trap. Effluent from this unit passes into an aeration tank and then to a settling tank. A portion of the nitrified and therefore nitrate-bearing supernatant from the settling tank is returned to the anoxic compartment of the baffled reactor. Each of the ABR compartments have dimensions 2.80 m × 1.42 m × 2.05 m i.e. with a compartment volume of 8.15 m^3. Overflow between the compartments is carried by 6 PVC pipes directed to distribute the flow evenly over the bottom of the subsequent compartment. Waste anaerobic sludge is withdrawn from the bottom of the compartments and discharged to a thickening tank. Approximately 12.5 m^3 of screened degritted wastewater was fed to the plant daily (Garuti et al., 2001).

Biomass concentration in the anaerobic compartments was maintained at low values to prevent biomass washout. Feed with an average COD concentration of ca. 600 mg/ℓ was supplied intermittently to the ABR giving a maximum up-flow velocity of around 2.5 m/h. Total COD and TSS removal across the ABR was 31.2% and 45% respectively at the end of a 4 month test period. The ABR in this system is a pre-treatment device and was not designed to achieve complete COD removal. The effluent from the entire ANANOX® plant showed 95% COD removal (Garuti et al. 2001).

2.2.7.3 DEWATS system

DEWATS (DEcentralised WAstewater Treatment Systems) consist of hybrid anaerobic/aerobic systems for community based sanitation. The actual configuration of the system varies according to wastewater quality and effluent quality requirements as well as locally available materials of construction. These systems should be easily managed and maintained under local conditions, and operate without energy input (BORDA, 2005). Four treatment steps are included:

- Sedimentation and primary treatment
- Secondary anaerobic treatment in fixed bed filters or *baffled septic tanks*
- Secondary and tertiary aerobic/anaerobic treatment in constructed wetlands (subsurface)
- Secondary and tertiary aerobic/anaerobic treatment in ponds

By 2003, 120 000 DEWATS units had been implemented in China, as well as many more in India and the Philippines (Panzerbieter et al., 2005).

A baffled reactor is considered a suitable secondary treatment for *all kinds of wastewater* but preferably those with a high fraction of settlable solids and a small COD/BOD ratio (Sasse, 1998.). The DEWATS handbook (Sasse, 1998) reports 70 to 90% BOD removal in anaerobic filters or baffled reactors in a DEWATS system. The baffled reactors implemented in DEWATS systems have a minimum of 4 compartments and are designed to have an up-flow velocity not exceeding 2 m/h. The recommended organic loading rate is less than 3.0 kg COD/m^3.d. Hydraulic retention time should not be less than 8 h. A settling compartment is implemented at the beginning of the DEWATS baffled reactor, with a submerged outlet to the next compartment so that scum is retained. These units are reported to require 3 months *maturation* (start-up period) and desludging at similar intervals to septic tanks. Sasse (1998) describes the baffled reactor as *poorly known and little researched* and that the microbial dynamics are not well understood. The baffled reactor is usually followed by horizontal filters with constructed wetlands for pathogen and nitrogen removal.

3 PILOT ABR DESIGN, CONSTRUCTION AND INSTALLATIONS

This project undertook an experimental study using a 3000ℓ pilot-scale ABR treating wastewater at Umbilo and Kingsburgh WWTPs. This chapter presents the development of the design of the pilot ABR, the construction of the reactor, and details of the installation configurations during experimentation.

3.1 PILOT ABR DESIGN

The design of the pilot ABR was the basis of an MSc Eng study undertaken by Ms Dama in the School of Chemical Engineering, University of Natal. The starting point for the design of pilot reactor design was laboratory-scale ABRs that had been used for co-digesting toxic and high strength effluents in WRC project K5/853. These reactors consisted of 8 compartments, and a total working volume of 10 ℓ. Flow between compartments was through a slot in each alternate baffle. Figure 3.1 is a photograph of the laboratory-scale reactors.

Figure 3.1: 10 ℓ Perspex laboratory-scale ABRs showing inlet, internal baffles, gas vents and sampling ports

As there was no experimental basis for deciding whether the design should be changed, two design parameters were investigated using *computational fluid dynamics* (CFD) modelling techniques to identify improvements for the pilot reactor design.

3.1.1 Computational fluid dynamics

The CFD program, FLUENT, was used to investigate the effect of the position of the baffle in the compartment and the width of the slot between the compartments on fluid dynamics within the compartments. Since the reactor is symmetrical along the longitudinal axis and all eight compartments are identical, it was sufficient to model only half of a single compartment since the results would apply equally to all other half compartments. This model did not take into account the effect of solids or gas bubbles

on fluid dynamics, and the system was modelled as a single phase. The lamina phase model was selected with fluid properties equal to those of water. The gas-liquid surface was modelled as a frictionless surface, and gas-liquid interface effects were ignored.

3.1.1.1 Baffle spacing

Two baffle-spacing scenarios were investigated. The first considered a compartment with the baffle in the centre of the compartment, while the second placed the baffle such that the up-flow-to-down-flow area ratio was 2:1 (Figure 3.2). One of the main features of the ABR is retention of solids in the system. In order to achieve this, low up-flow velocities need to be maintained. As expected, a greater surface area for the up-flow region resulted in lower up-flow velocities. However, increasing the up-flow surface area also resulted in greater volume of dead-space.

The velocity vector profiles along a transverse plane for the two baffle positions are presented in Figure 3.2. The magnitude of the velocity is indicated by the length of the velocity vector i.e., the longer the arrow, the greater the velocity. A uniform distribution of flow was attained with configuration B. Increasing the up-flow area resulted in a further increase in channelling and dead-space in the up-flow region.

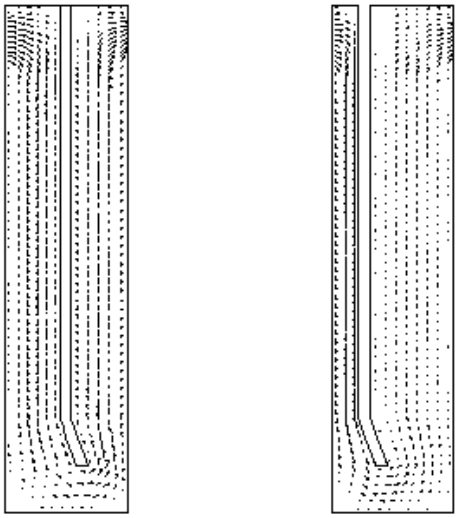
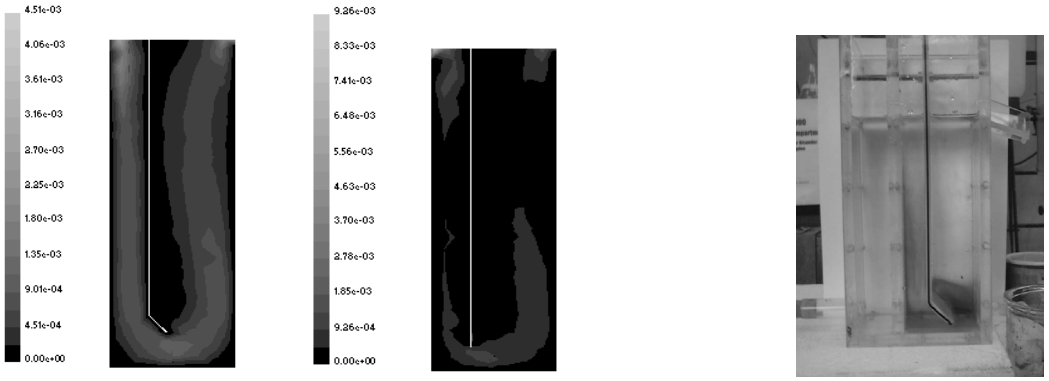


Figure 3.2: Velocity vector profiles obtained for a 20 h HRT using CFD software *FLUENT* for hanging baffle positioning. Profiles for 1:1 (left) and 2:1 (right) up-flow-to-down-flow area ratios are shown.



(a) (b) (c)

Figure 3.3: Longitudinal section through an ABR compartment illustrating the CFD velocity contours for the two different baffle configurations: (a) angled baffle, (b) straight baffle. Darker colours represent low flow rates. (c) Laboratory verification of CFD results using a dye tracer

A CFD model showing the effect of angling the bottom of the hanging baffle was attempted. Figure 3.3 shows the flow contours around an angled and straight baffle. It was found that the angled baffle resulted in more even flow distribution and reduced dead-space. CFD tests were visually reproduced using a single compartment laboratory-scale ABR and dye in water.

3.1.2 Construction of reactor

The pilot ABR was designed to have a total working volume of 3 000 ℓ. The hanging baffles were attached to the top of the reactor to separate the gas pockets between the compartments. The heights of the standing baffles were reduced across the reactor to facilitate an ease of flow through the reactor. A diagram of the pilot-scale reactor is shown in Figure 3.4.

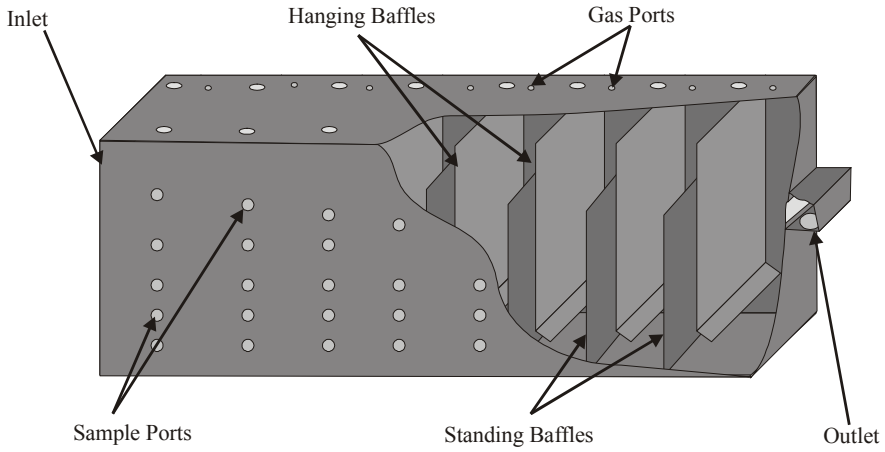


Figure 3.4: Diagram of the pilot-scale ABR with a cut-away to give an indication of the baffle configuration.

3.1.3 Materials of construction

The pilot reactor was built as a trial reactor with an intended life-span of one to two years. Mild steel was selected as the material of construction since this was the most cost effective material in which it was possible to construct a reactor with many sampling points.

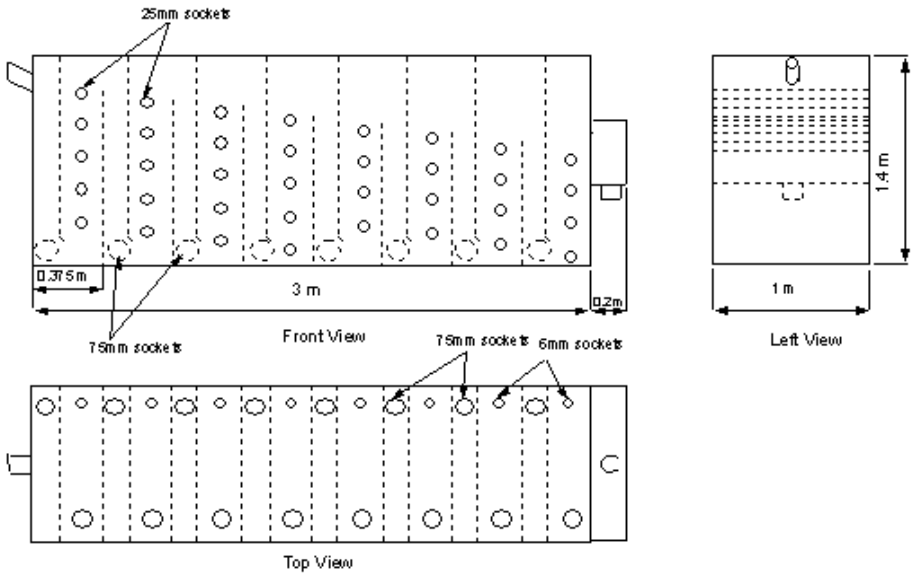


Figure 3.5: Orthographic projection of the pilot-scale ABR

The sheets were laser cut to the specifications presented in Figure 3.5 and welded together to form gas-tight compartments.

Several 25 mm sockets were added for sampling purposes. Galvanised ball valves were attached to the top and bottom socket of each compartment for sampling. Galvanised

plugs were used to plug the other sockets. A 75 mm socket was added at the bottom of each compartment to facilitate emptying of the compartment. These sockets were plugged using galvanised 75 mm plugs. 75 mm sockets were supplied on the top of the reactor for sampling. These were fitted with PVC plugs. 6 mm sockets above the up-flow area of each compartment were supplied for venting and collecting biogas.

3.1.4 Construction of feed box

Wastewater feed was pumped to the reactor by a feed splitter box. The splitter box was divided into 3 chambers with the aid of baffle plates (Figure 3.6). The effluent was pumped into the middle chamber. Weirs were cut into the baffle plates to divide the flow such that 90 % of the flow entered the return chamber and 10 % entered the feed chamber. A 100 mm pipe leads from the return chamber back into the channel. The feed chamber contained 3 outlets. A butterfly control valve (FC1) was fitted on the lowest outlet. This valve was used to control flow to the reactor by opening when the feed rate was too high, emptying the contents of the feed chamber. When FC1 closed, the level in the feed chamber rose until wastewater overflowed through the feed pipe into the ABR. A third outlet on the feed chamber was supplied to collect overflow in the event of a blockage to the ABR feed line.

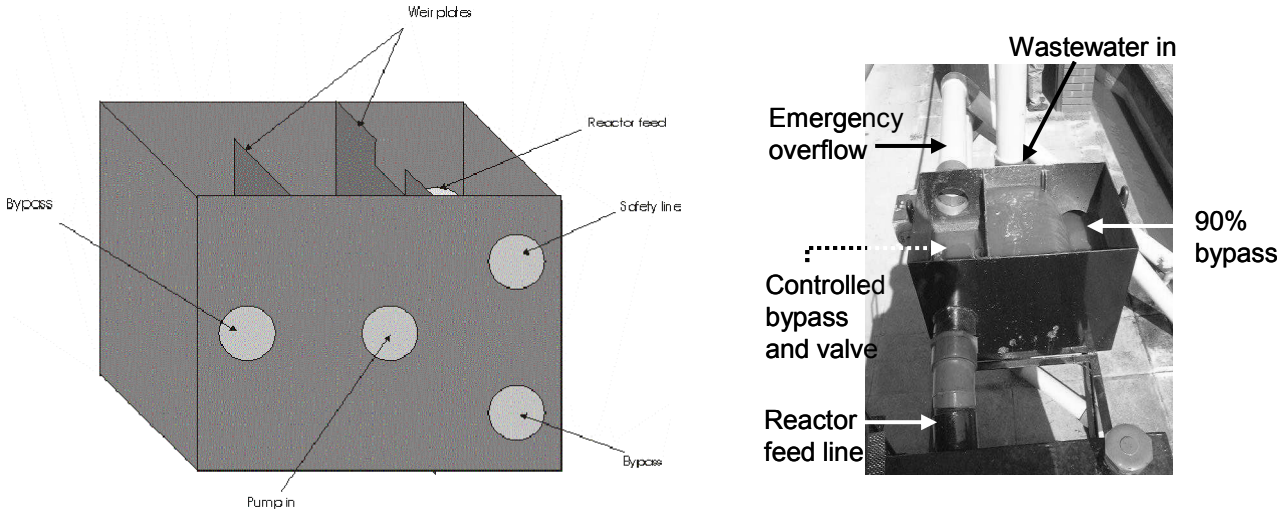


Figure 3.6: Schematic diagram of the feed splitter box installed at the inlet of the pilot ABR (left); and reverse view of the splitter box installed on the ABR (right)

3.1.5 Auxiliary equipment

A field ABR installation would be supplied with raw wastewater under gravity feed at varying flows and loads. However, the pilot ABR study aimed to investigate microbiological and chemical performance of the ABR configuration when fed domestic wastewater under controlled feed conditions. All of the additional equipment used in the pilot ABR installations was there for the purpose of sampling wastewater from a much

larger flow than could be handled by the ABR, and feeding it to the reactor in a controlled and quantifiable manner. These included:

- A submersible pump to deliver municipal wastewater to the pilot ABR.
- A pneumatic valve to control air supply to the by-pass valve.
- A compressor to supply air to the pneumatic valve.
- A magnetic flow meter (FI1) to record and transmit effluent flow rate and cumulative flow.
- A programmable logic controller (PLC) to capture flow rate data, calculate feeding/by-passing requirements and control the by-pass valve.
- A timer control switch to control the by-pass valve when the PLC was off-line.
- A PVC unit housing a microfilter was attached to the last compartment in 2004 (See Figure 3.8).

3.1.6 Principle of flow control

The reactor outlet passed through a magnetic flowmeter (FI1) which produced a signal that was recorded by a programmable logic controller (PLC). A number of different control algorithms were implemented to achieve a fixed and relatively steady flowrate. The measured flow at the outlet was used to increase or decrease the flow at the inlet by adjusting the timing of the bypass valve (FC1) opening.

During the experimental studies, there were three control regimes vis. timer control, bang-bang control and Proportional and Integral (PI) control.

3.1.6.1 Timer control

Before the PLC was correctly programmed a timer switch was used to open and close the by-pass valve for fixed times in a fixed control cycle. The timer control system had no mechanism for adapting when the pump delivery rate changed. Pump delivery was erratic due to the heterogeneous nature of the wastewater, particularly the presence of rags that would jam or slow the pump impellor. There was thus little control over the amount of wastewater delivered to the reactor.

3.1.6.2 On-off control to flow setpoint

A bang-bang control algorithm was implemented on the PLC. This aimed to control the flow rate to *not* exceed a specified flow rate. This target flow was determined as the flow required to achieve a target hydraulic retention time. This method of control did not allow compensation for periods of high or low flow, and no record was made by the PLC of the actual amount of flow through the reactor. Therefore, the average hydraulic

retention time cannot be accurately calculated when bang-bang flow control was implemented.

3.1.6.3 PI control to hydraulic retention time

A Proportional-Integral (PI) controller was programmed into the PLC using a time-slicing algorithm where, for a fixed cycle of 1 min, the PLC calculated the fraction of that minute that the bypass valve should be closed in order to achieve a target flow rate and target average hydraulic retention time. This control regime allowed less variable flow rates to the reactor than had been experienced using timer or bang-bang control, and ensured that the overall flow through the reactor was known. This program also included high and low flow warnings and emergency shut-down loops in the event of excessively high flows through the reactor being recorded.

3.2 INSTALLATIONS

The reactor was initially commissioned at the Umbilo Wastewater Treatment Plant (WWTP). This sewage works was chosen because it is situated close to the University of KwaZulu-Natal and has a well equipped laboratory where routine analyses can be carried out. Figure 3.7 shows the installation of the pilot ABR at the head of works at Umbilo WWTP. The compressor that supplied air to the valve was housed on top of the reactor. The submersible pump was lowered into the influent channel next to the reactor at the head of the works. The PLC was housed in an enclosure in a control room seen in the right of the right-hand picture in Figure 3.7.

Figure 3.7: Photographs of the front and back of the pilot ABR installed at Umbilo WWTP

The pilot ABR It was seeded with 10 ℓ of anaerobic sludge from the Umbilo anaerobic digesters and filled with screened and degritted wastewater from the inflow channel.

Weekly grab samples of the reactor influent and effluent and from the top and bottom of each reactor compartment were analysed for pH, chemical oxygen demand (COD),

alkalinity, total solids (TS) and % ash. Sampling and analyses were performed by Municipal staff at Umbilo WWTP.

Umbilo WWTP treats a combined industrial and domestic wastewater, where the industrial component arises mainly from nearby textile industries. The wastewater therefore often contains dye effluent and rags. The rags regularly caused obstructions in the pump impellor chamber resulting in regular no-flow periods. This problem was reduced slightly by building a triangular flow dispersing frame around the pump inlet, and surrounding the entire pump in chicken mesh. This reduced the incidence of pump blocking, but was unable to prevent strings and cords from entering and getting entangled in the pump.

The pilot ABR was moved to Kingsburgh WWTP in January 2002. Operation of the pilot ABR at Kingsburgh WWTP formed the basis of an MSc Eng Thesis (Mtembu, 2005). During operation at Umbilo WWTP, the ABR performance was not as good as expected in terms of COD removal. At the time, there was a concern that dye effluent in the feed to the reactor may have caused inhibition of anaerobic digestion and low activity. Kingsburgh WWTP treats a wastewater that has no formal industrial effluent component. It serves a community of about 350 000 population equivalents from middle-income suburbs. Although this wastewater was not considered to be the same as wastewater emanating from a low-income community, it was believed that a better understanding of the functioning of the ABR in sanitation would be obtained without complications from trade effluents.

Prior to removal from Umbilo WWTP, a week of sludge settling time without feed, was allowed to concentrate anaerobic sludge in the bottom of the reactor. The liquid fraction in each compartment above the bottom sample/drain valve was drained away, leaving 200 mm of sludge in the bottom of each compartment the reactor. This sludge was then available as a seed for treatment of Kingsburgh WWTP wastewater.

The PLC enclosure had been housed in a small building near the reactor at Umbilo WWTP. There was no similar sheltered position near the ABR installation at Kingsburgh WWTP and therefore, the PLC was mounted to the side of the reactor. However, the enclosure had been supplied with glands on the top that were not watertight, and rain was able to get into the enclosure, causing electrical damage to several of the components in the enclosure. The enclosure was replaced with a substantially larger and better sealing enclosure that was mounted to the reactor.

Although the incidences of rags blocking the pump were reduced during operation at Kingsburgh WWTP, string, rubber and hair was regularly found their way into the pump, causing damage. Several means of eliminating these from the pump were attempted, but by far the most successful measure was installing the pump in a laundry basket (Figure 3.8 – right). Incidences of pump blockages reduced by more than half as a result of the laundry basket system. During the time the ABR was operated at Kingsburgh, at least 5 laundry baskets were used in this way!

Figure 3.8 Installation of the ABR at Kingsburgh WWTP. The outlet end of the pilot ABR showing the membrane unit attached to the last compartment (left); and a laundry basket housing the submersible pump in a wastewater sump near the feed end of the ABR (right)

In the first year of operation at Kingsburgh WWTP (2002), there were a number of occasions when the outlet pipe from the ABR blocked, causing a build-up of liquid in the reactor. When the blockages were removed, a large amount of backed-up liquid would flow out, carrying large amounts of sludge. In this way, considerable amounts of sludge were lost during operation in 2002. The cause of the blockages was found to be tiny cones (about 15 mm diameter) from conifers growing next to the installation. These would jam just before the flow-meter on the outlet line. To remedy this, a lid was built for the splitter box to prevent ingress of the cones, and the effluent line was opened and installed with a screen before the flow meter (Figure 3.9).

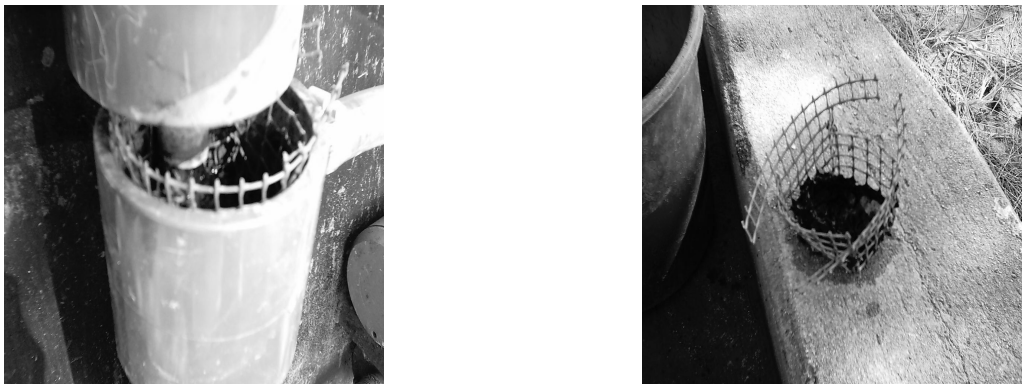


Figure 3.9: Modified outlet of the ABR showing mesh for preventing coarse solids entering the flow meter.

An A4 Kubota membrane module was donated to the project by AQUATOR, and was used to simulate performance of a membrane filtration step in the last compartment of the ABR. The membrane unit housing is shown in Figure 3.8 attached to the bottom of

compartment 8 of the pilot ABR at Kingsburgh. This unit will remove virtually all solids and pathogens, as well as a considerable amount of COD from the effluent. Preliminary trials were undertaken in the laboratory and at Kingsburgh WWTP.

4 RESULTS AND DISCUSSION OF PILOT ABR OPERATION

In this chapter, chemical and pathogen indicator organism results from the 4 operating periods, one at Umbilo Wastewater Treatment Plant and three at Kingsburgh Wastewater Treatment Plant (WWTP) are presented.

4.1 RESULTS FROM OPERATION AT UMBILO WASTEWATER TREATMENT PLANT

The pilot reactor was operated at Umbilo WWTP for a total of 409 d from 18 July 2000 to 31 August 2001. No record of the amount of flow treated during this time was kept. For the first 228 d, the flow to the reactor was under timer control (See Chapter 3). This resulted in a variable and unpredictable flow. The timer control was set to achieve a target hydraulic retention time of 60 h for the first 126 d. On day 127 (22 November 2000), the timer was adjusted to achieve a target hydraulic retention time of 32 h. The timer settings were changed again on day 205 (8 February 2001) to achieve a 20 h target hydraulic retention time.

On day 228 (3 March 2001), the programmable logic controller (PLC) was brought online. The control algorithm aimed to control the flow rate to not exceed a specified flow rate. This target flow was determined as the flow required to achieve a target hydraulic retention time of 20 h. Since no measurement of the total flow was recorded, it is not possible to say what the *actual* mean hydraulic retention time was in this period. However, it is reasonable to say that it was *greater than* 20 h since the flow rate was not allowed to exceed the target flow rate. The uncertainty regarding the actual flow rates and mean hydraulic retention time places a limit on the amount of quantitative information that can be extracted from the data presented.

During operation at Umbilo WWTP, weekly grab samples of the reactor influent and effluent and from the top and bottom of each reactor compartment were analysed for pH, chemical oxygen demand (COD), alkalinity, total solids (TS) and % ash. Volatile solids can be calculated from the last two measurements. The influent and effluent samples were also tested for ammonia and phosphorus. Physical measurements of the height of the sludge bed were performed on two occasions.

4.1.1 Total Solids Concentration

Figure 4.1 shows values of total solids measured at the bottom sampling valve of each compartment, located 200 mm above the floor of the ABR. Samples drawn from above the fluidised sludge bed contain less than 10 000 mg/ℓ while those that are drawn from within the sludge bed vary between 10 000 and 70 000 mg/ℓ The actual height of the sludge bed is not known for most of this operating period. At start-up, there was essentially no biomass in any of the compartments except compartment 1 since this compartment was seeded with a few bucketfuls of anaerobic digester sludge. Samples drawn from the bottom sample valve of each compartment showed low (<10 000 mg/ℓ) solids concentrations, indicating that the sludge bed had not risen above 200 mm.

From Figure 4.1, it is possible to see exactly when the sludge bed had built up above the bottom sample valve. For compartments 1 to 8 respectively, this occurred on days 45, 129, 157, 213, 220, 241, 262 and 290. Compartments 4 to 8 only developed 200 mm sludge beds in the 20 h hydraulic retention time operating period. All compartments had achieved 200 mm sludge beds by day 300.

4.1.2 COD

COD concentration was measured using the open reflux method according to Standard Methods (APHA, 1998). The average COD of the screened wastewater fed to the ABR was 712 ± 243 mgCOD/ℓ (n = 265). Figure 4.2 shows the measured COD in grab samples of the influent and effluent of the pilot ABR. Initially, effluent COD was measured to be near 600 mgCOD/ℓ, but dropped steadily to a value of 121 mgCOD/ℓ on day 129. The flow rate was increased to achieve a 32 h hydraulic retention time, and an immediate increase in effluent COD was observed. Few data were obtained for the 32 h hydraulic retention time. The average effluent COD in this period was 170 ±77 mgCOD/ℓ (n = 8). The flow rate was increased again on day 205 resulting in a clear increase in effluent COD value to a maximum measured value of 564 mgCOD/ℓ on day 206. By day 234, the COD value had decreased to 165 mgCOD/ℓ. For the remainder of the operating period at a target hydraulic retention time of 20 h, the average effluent COD value was 243±68 mgCOD/ℓ (n = 20). This can be interpreted as an average COD reduction of 66%.

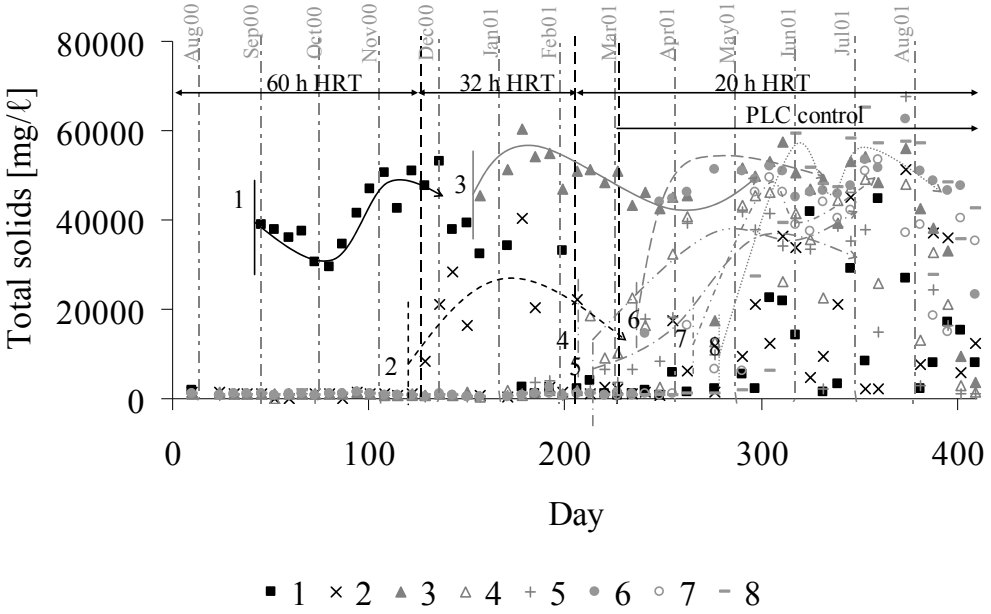


Figure 4.1: Total solids concentration measured 200 mm above the bottom of each compartment, with hand drawn trends to show the appearance of the sludge level above the sampling point. The numbers 1 to 8 represent compartments 1 to 8.

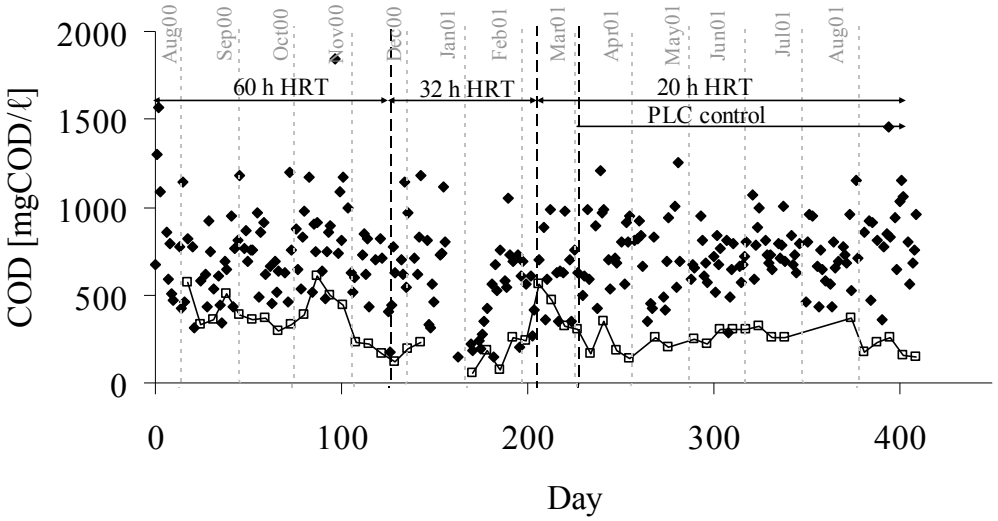


Figure 4.2: Total COD concentrations taken from the influent and effluent of the pilot ABR treating 50:50 industrial : domestic wastewater at Umbilo WWTP. Data for 60 h, 32 h and 20 h target HRT (timer and PLC control) are shown.

4.1.3 pH

pH measurements were performed on samples drawn from the influent, effluent, and the top of each compartment. Figure 4.3 presents the pH measurements of the influent and effluent in the three operating periods.

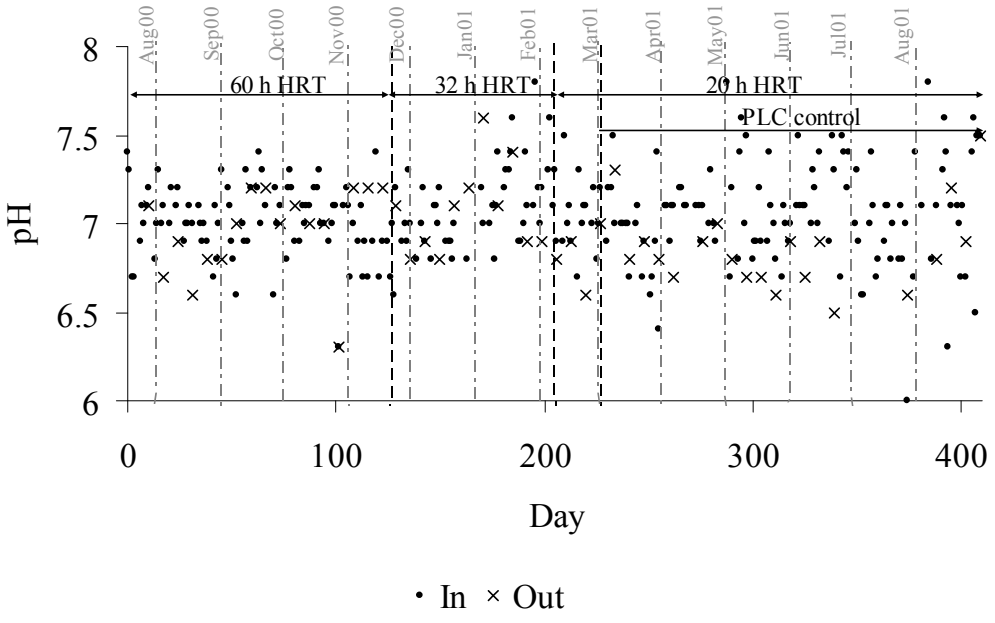


Figure 4.3: Influent and Effluent pH measurements from the pilot ABR treating 50:50 industrial : domestic wastewater at Umbilo WWTP. Data for

60 h, 32 h and 20 h target HRT (timer and PLC control) are shown. Points (●) indicate influent values and crosses (×) are effluent values.

Figure 4.3 shows no obvious trend of pH value with respect to time of the effluent. A low effluent pH value on day 101 corresponds to a low influent measurement on the same day, but other low influent measurements are not matched by similar low effluent measurements. This is not unexpected since influent and effluent samples are obtained simultaneously, but the effluent characteristics arise from an influent delivered to the reactor one retention time (e.g. 20 h) before the effluent sample was obtained. Therefore it is not appropriate to match individual measurements from the influent and effluent. Average effects should rather be considered.

Several anaerobic processes, but especially methanogenesis, are strongly inhibited by low pH values (below pH 6.5) (Batstone et al., 2002, section 2.2.4.2). Most pH measurements of influent, effluent and the compartments during operation of the ABR at Umbilo WWTP have values in the range 6 to 7, and therefore, the rate of methanogenesis can be expected to be sensitive to variations in pH value. Since methanogenesis is the step that actually removes COD from wastewater (see section 2.2.1), pH profile in the reactor can provide some clues to the overall status of anaerobic digestion. A conventional term defining extent of methanogenesis inhibition as a result of low pH values (Batstone et al., 2002) was calculated for each available pH measurement in each compartment and each day (Equation 4-1). A value of 0 indicates complete inhibition, while 1 indicates no inhibition. These inhibition terms were averaged for each compartment, influent and effluent for each operating period and are presented in (Figure 4.4).

$$\mathrm{I} = e^{-3\left(\frac{pH-pH_{LL}}{pH_{UL}-pH_{LL}}\right)^2}\Big|_{pH<pH_{UL}} \quad or\ 1\Big|_{pH\geq pH_{UL}} \qquad \textbf{Equation 4-1}$$

The corresponding pH values that would exert the averaged inhibition factor are also presented. (These numbers are not an average pH – a meaningless measure - but represent average conditions from the point of view of methanogenesis inhibition in each compartment).

The scales of inhibition and pH have been selected so that the inhibition (I) and corresponding pH curves coincide for higher values of pH and I. It can be seen that the inhibition value decreases at a faster rate than the pH value for lower values of pH, showing the accelerating effect of low pH on inhibition.

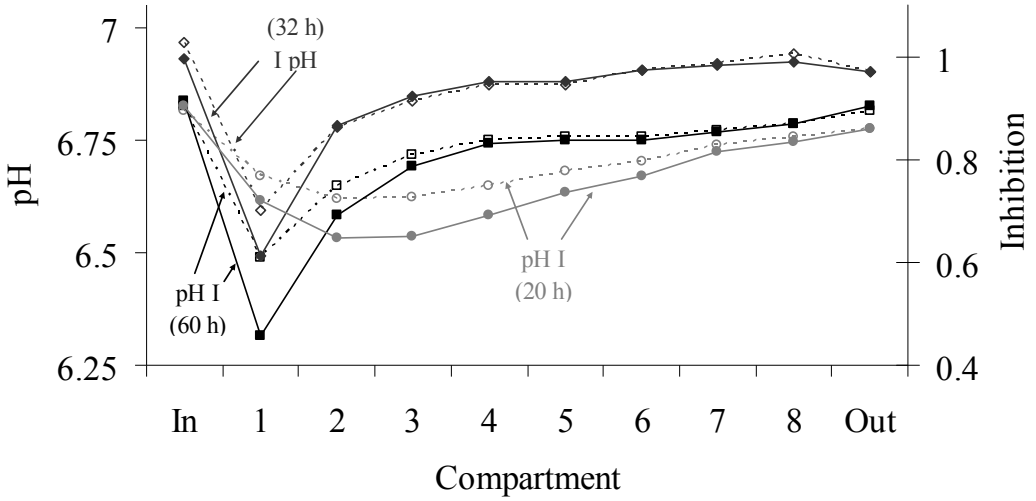


Figure 4.4: Average inhibition (1 = not inhibited, 0 = completely inhibited) and corresponding pH value calculated for each compartment, from the pilot ABR treating 50:50 industrial : domestic wastewater at Umbilo WWTP. Filled points (-■- -♦- -●-) represent *inhibition* values, while open points (··□·· ··◊·· ··○··) show corresponding *pH values* for 60 h, 32 h and 20 h respectively.

The compartment-by-compartment pH profiles indicate that acidogenesis (section 2.2.1.3) dominates in the earlier compartments causing increase in acid concentration and lower pH values and that methanogenesis removes accumulated acid in later compartments. Increase in pH is achieved by two routes: firstly acid produced by the acidogenesis reactions is consumed by methanogens, and hence an increase between earlier (acidogenesis dominated) and later compartments will be observed. Secondly, anaerobic digestion results in the production of alkalinity which increases pH.

In all cases, the effluent value is on average slightly lower than the influent value. This is unusual for anaerobic digestion applications where effluent pH values tend to exceed influent values (Speece, 1996). This situation usually implies poorly buffered and unstable digestion. In dilute wastewater treatment, relatively low alkalinity generation potential and poor buffering can be shown to be the cause of this phenomenon. The validity of this result is confirmed by predictions of pH and alkalinity concentration using mass balance considerations that predict low effluent pH values for low strength anaerobic digestion (Sötemann et al., 2005, Section 8.3.4 and 8.3.5).

Initial examination of the data presented in Figure 4.4 suggests that methanogenesis inhibition increases with hydraulic loading rate (i.e. decrease in target hydraulic retention time), but although this may be true in steady-state cases, it is not the cause of the difference in methanogenesis inhibition shown here. These data must be interpreted in conjunction with Figure 4.1 and Figure 4.2. During the first two operating periods, there was little sludge and therefore little anaerobic activity in all but the first two compartments. Hence inhibition by acidogenesis products is small since the overall

amount of acidogenesis is small. Significant amounts of solids and therefore biomass, and associated activity only develop in compartments 4 onwards in the 20 h target hydraulic retention time operating period. The 60 h and 32 h target hydraulic retention time operating periods show similar inhibition profiles, despite the big change in retention time. The big dip in the average inhibition term and corresponding pH value is due to most anaerobic activity occurring in compartment 1 and 2 with acid production and methanogenesis inhibition in compartment 1 and methanogenesis in compartment 2. Little digestion and therefore COD reduction of any kind occurs in the subsequent compartments due to low biomass concentration.

The 20 h target hydraulic retention time operating period shows a different trend. The methanogenesis inhibition term decreases in compartments 1 and 2 and begins to recover gradually over the subsequent compartments. This is attributed to growing amounts of sludge in all compartments resulting in acid production occurring in several compartments (not just compartment 1). Although average conditions for methanogenesis are worse than in the previous operating periods, the overall extent of treatment (fraction of biologically available COD removed) is greater since a greater amount of COD is removed.

4.1.4 Alkalinity

Bicarbonate alkalinity, measured in units of mg$CaCO_3$/ℓ was performed by acid titration using HCl.

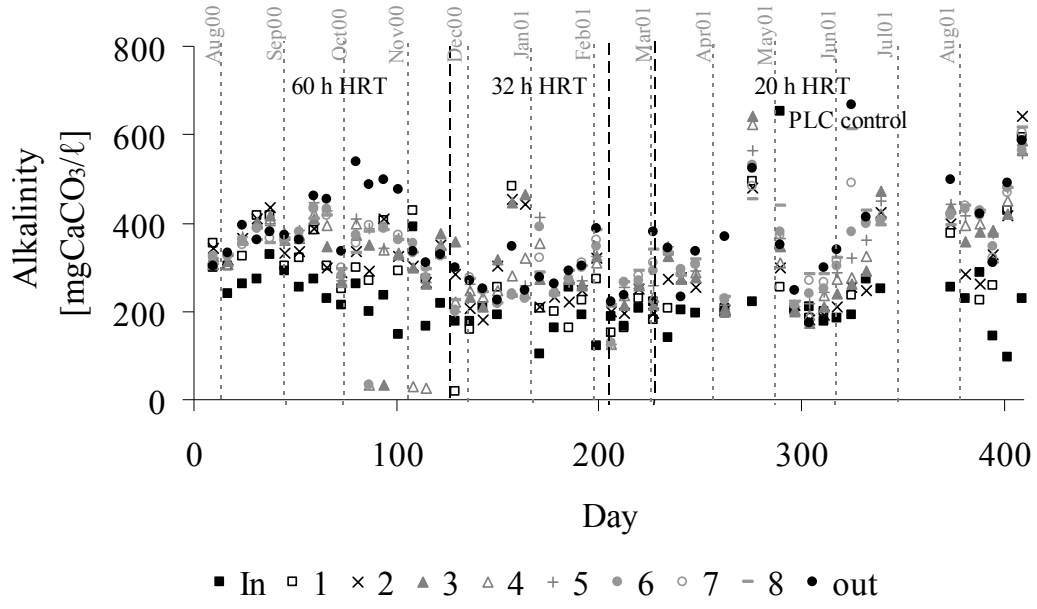


Figure 4.5: Influent, effluent, and individual compartment alkalinity measurements from the pilot ABR treating 50:50 industrial : domestic wastewater at Umbilo WWTP. Data for 60 h, 32 h and 20 h target HRT (timer and PLC control) are shown.

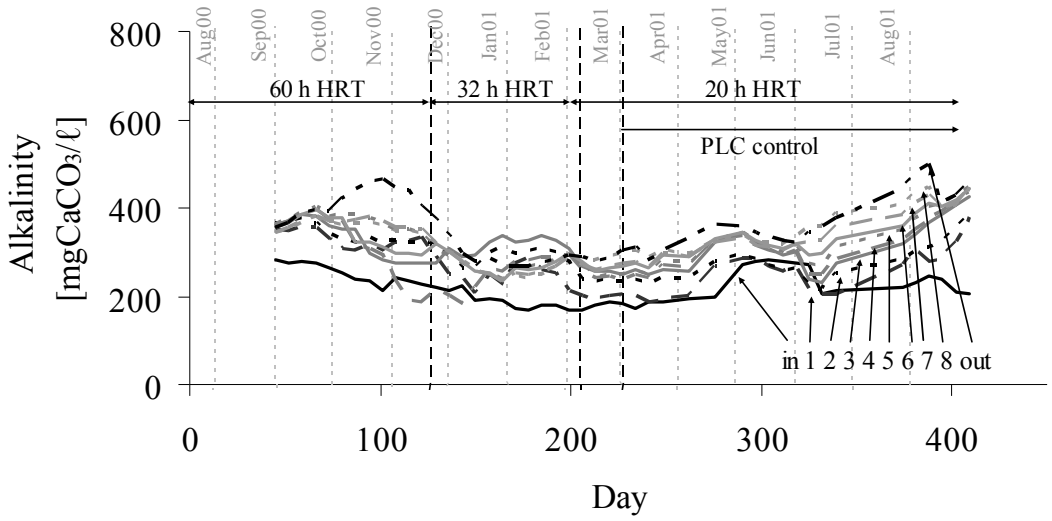


Figure 4.6: Six-period moving averages for influent, effluent, and individual compartment alkalinity measurements from the pilot ABR treating 50:50 industrial : domestic wastewater at Umbilo WWTP. Data for 60 h, 32 h and 20 h target HRT (timer and PLC control) are shown. Averages are plotted on the mid-point of the moving period for each calculation.

Figure 4.5 shows actual measured values of alkalinity in each compartment, giving an indication of the amount of scatter that was observed. Figure 4.6 presents the same data as 6-period moving averages so that overall trends may more easily be identified.

In general, there is an increase in alkalinity value from one compartment to the next, with effluent values in each operating period significantly higher than influent values. Figure 4.6 shows a seasonal variation of influent alkalinity with summer values (November to April) on average lower than winter values. Alkalinity shows a relatively constant increase between influent and effluent in the first two operating periods, and for the first half of the third operating period. From May 2001, there is an increase in the amount of alkalinity generated in each compartment, which implies that a larger extent of treatment is being obtained, since alkalinity is produced by anaerobic digestion (Speece, 1996).

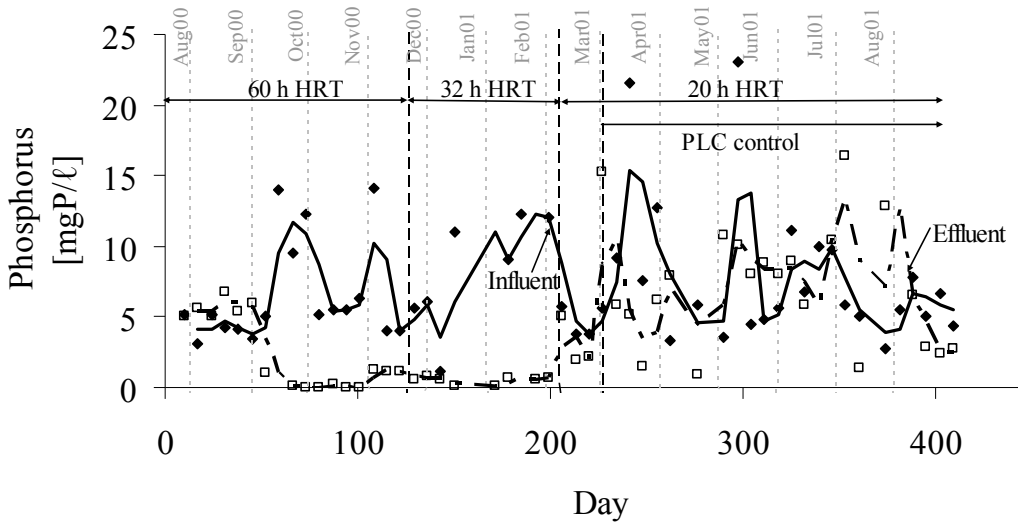


Figure 4.7: Total influent and soluble effluent phosphorus concentrations of the pilot ABR treating 50:50 industrial : domestic wastewater at Umbilo WWTP. Data for 60 h, 32 h and 20 h target HRT (timer and PLC control) inlet (□) and effluent (♦) concentrations are shown, with two-period moving average lines to assist in identification of trends.

Few effluent COD data are available from this period (Figure 4.2) to support this observation, however, the last 5 data points seem to suggest that the effluent COD values have a decreasing trend. Better treatment may be a result of greater biomass concentration which is shown in Section 4.1.1, Figure 4.1 above.

4.1.5 Phosphorus

Figure 4.7 presents influent and effluent phosphorus concentrations for the three operating periods. Measurement of total phosphorus, (achieved by pre-digesting samples) of the influent and soluble phosphate in the effluent was made.

Before day 50 effluent phosphorus concentrations are significantly lower than influent concentrations. This may be due to the low biomass concentration during the early days of operation resulting in little liberation of phosphorus from organically bound forms. This should not be regarded as phosphorus removal since a mass balance cannot be obtained from the two dissimilar measurements (soluble and total phosphorus). A further suggestion for the low effluent values has been adsorption or co-precipitation of phosphorus species on iron oxides originating from the walls and splitter box of the pilot reactor. No such apparent phosphorus reduction can be expected under ordinary circumstances from an anaerobic system (Speece, 1996).

4.1.6 Ammonia

Figure 4.8 presents free and saline (NH_3+NH_4^+) ammonia concentrations in the influent and effluent of the pilot ABR.

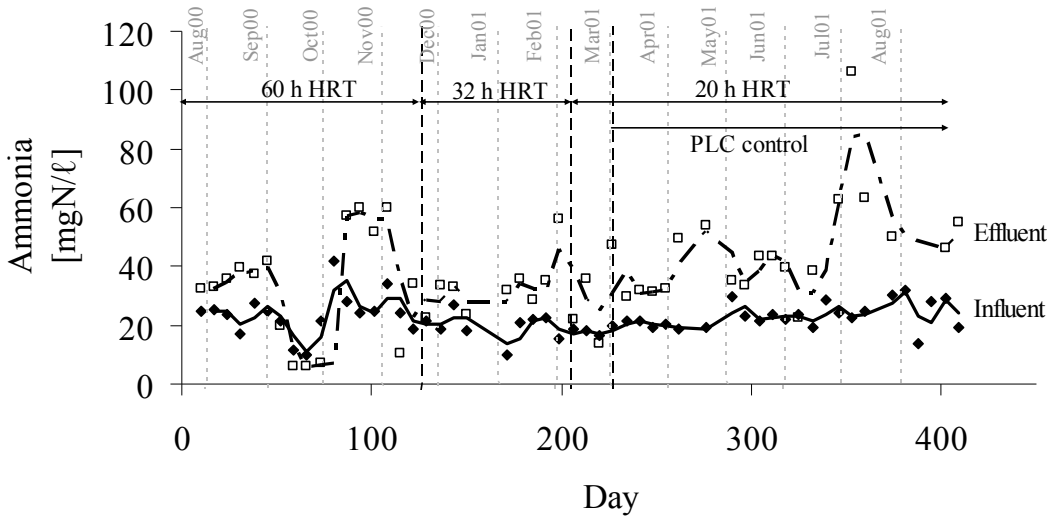


Figure 4.8: Influent and effluent free and saline ammonia concentrations of the pilot ABR treating 50:50 industrial : domestic wastewater at Umbilo WWTP. Data for 60 h, 32 h and 20 h target HRT (timer and PLC control) inlet (□) and effluent (♦) concentrations are shown, with two-period moving average lines to assist in identification of trends.

A net increase in free and saline ammonia is seen due to digestion in the ABR as a result of the liberation of organically bound nitrogen during anaerobic digestion (Speece, 1996). The difference between inlet and outlet values appears to be increasing after day 300, which, along with the increase in alkalinity production (Figure 4.6), and increasing compartment solids concentrations (Figure 4.1) implies an increase in the extent of treatment obtained.

4.1.7 Pathogen indicator organisms

On 23 April 2001, samples of influent and effluent were tested for total *Ascaris* spp., viable *Ascaris* spp., *E. Coli* and Total Coliforms. On 3 July 2001, samples of influent and effluent, and samples from each of the eight compartments were analyses for *E. Coli*, total coliforms, *Pseudomonas* spp., *Salmonella* spp., *Vibrio* spp., and *Shigella* spp.

Table 4.1: Pathogen indicator organisms detected in the influent and effluent of the pilot ABR treating 50:50 industrial : domestic wastewater at Umbilo WWTP. Data are single measurements or averages of two measurements (coliforms only) on grab samples obtained on 23 April 2001 and 3 July 2001 during the 20 h target HRT operating period under PLC control.

Pathogen indicator organism	Influent	Effluent
Total coliforms (cfu./100 mℓ)	> 4 000 000	46 500
E. coli (cfu./100 mℓ)	> 4 000 000	3 500
Total Ascaris spp. (/100 mℓ)	232	298

Viable Ascaris spp.(/100 mℓ)	83	5
Pseudomonas spp. (/100 mℓ)	0	1
Salmonella spp.(/100 m/ℓ)	0	0
Vibrio spp. (/100 mℓ)	0	0
Shigella spp. (/100 mℓ)	0	0

All analyses were performed by Durban Metro Wastewater (now eThekwini Water Services) Laboratories in Prior Road, Durban. Table 4.1 presents average results for the two sampling days. No standard deviations are calculated due to insufficient data.

A reduction of the order of 2 to 3 log units is obtained for coliforms, and viable *Ascaris* spp. reduced

Table 4.2: Summary of characteristics of the pilot ABR treating 50:50 industrial : domestic wastewater at Umbilo WWTP. Averages and standard deviations are presented for all measurements except pH value, for which median value is reported.

		Average/ Median	Std Deviation	Number of observations	Min.	Max.
COD [mgCOD/ℓ]	In	712	243	265	151	1 845
	Out (60 h HRT)	379	124	16	166	612
	Out (32 h HRT)	170	77	8	55	255
	Out (20 h HRT)	272	101	24	137	564
Alkalinity [mgCaCO3/ℓ]	In	215	52	271	66	424
	Out (60 h HRT)	396	74	17	303	540
	Out (32 h HRT)	286	47	11	225	387
	Out (20 h HRT)	371	129	20	172	666
NH_x [mgN/ℓ]	In	23	5	271	3	40
	Out (60 h HRT)	33	19	16	6	60
	Out (32 h HRT)	33	9	11	22	56
	Out (20 h HRT)	44	19	21	14	106
PO_4 [mgP/ℓ]	In	6.3	3.0	96	1.1	18.0
	Out (60 h HRT)	2.4	2.6	16	0.0	6.8
	Out (32 h HRT)	1.1	1.5	10	0.1	5.1
	Out (20 h HRT)	7.0	4.4	23	0.9	16.4
Total solids [mgTS/ℓ]	In	1 256	1 086	52	505	8 645
	Out (60 h HRT)	2 177	1 893	16	405	8 453

	Out (32 h HRT)	1 080	580	10	387	2 191
	Out (20 h HRT)	13 782	16 320	24	22	55 874
Volatile solids [mgVS/ℓ]	In	655	891	49	145	6 657
	Out (60 h HRT)	1 210	1 283	16	231	5 579
	Out (32 h HRT)	371	254	10	90	789
	Out (20 h HRT)	10 297	11 518	19	10	37 436
pH (median value reported)	In	7.0		272	6.0	9.2
	Out (60 h HRT)	7.0		17	6.3	7.2
	Out (32 h HRT)	7.1		11	6.8	7.6
	Out (20 h HRT)	6.8		24	6.5	7.5
TKN [mgN/ℓ]	In	42	10	21	21	68

by an order of magnitude as a result of treatment in the ABR. Virtually no *pseudomonas, Salmonella, vibrio or shigella* spp. were detected in either the influent or effluent of the reactor. These results have not been statistically verified. Pathogen indicator organisms that decrease as a result of treatment in the ABR show that some improvement in microbial quality is achieved, although the numbers seen here are not expected to be representative of pathogen indicator loads that will be obtained on community waters and therefore, effluent quality is expected to be different to effluent in a community installation. These data however indicate that microbial quality will be a limiting factor in the reuse/discharge potential of the treated effluent.

4.1.8 Summary of results from operation at Umbilo WWTP

Table 4.2 presents a summary of all influent and effluent measurements averaged for operation of the pilot ABR at Umbilo WWTP. During the three operating periods (60 h, 32 h and 20 h target hydraulic retention times), COD reduction of between 330 and 580 mgCOD/ℓ (48 to 81%) was achieved (calculated from the average influent COD and high and low effluent COD values).

Build-up of biomass after day 300 (2001) caused the change in ammonia and alkalinity measurements between compartments to increase. Ammonia and alkalinity are by-products of anaerobic digestion; therefore it is inferred that the extent of wastewater treatment achieved at this time was increasing. Clearly, the degree of digestion achieved increased with increase in amount of biomass. Theoretically, this should be matched by decreasing effluent COD values. The COD data for this period are noisy, as a result of the wide range of input COD concentrations, but do not refute this interpretation.

Operating pH values below 7 indicate that the system is relatively poorly buffered and that inhibition of methanogenesis occurred to a significant extent in early compartments. The shape of the compartment-by-compartment pH profile has been shown to be a useful indicator of that state of anaerobic digestion in the reactor as a result of the distributed nature of the different steps in digestion across compartments, when interpreted in conjunction with other data.

Microbiological steady state was not achieved in any of the operating periods due to the very slow build-up of sludge in compartments, indicating the importance of adequate seeding. However anaerobic treatment of raw wastewater was seen to occur almost from the beginning of operation.

4.2 RESULTS FROM OPERATION AT KINGSBURGH WWTP

The pilot ABR was moved to Kingsburgh WWTP at the beginning of 2002. This section describes results obtained during operation of the pilot ABR at Kingsburgh WWTP.

4.2.1 Incidents, down-time and flow rate

Three operating periods were attempted in the period January 2002 to January 2005. In the first period from 2 July 2002 to 20 November 2002, the hydraulic retention was set to 20 h, using a proportional-integral (PI) control algorithm implemented in the PLC. A total of 350 000 ℓ of wastewater was treated at an average of 2 800 ℓ/d in 127 days. Incidents that caused high flow with sludge loss, or down time were due to electrical and mechanical problems with the pump, compressor and pneumatic valve. A graph of down-time and performance-affecting incidents in 2002 is presented in Figure 4.9. A souring incident occurred on day 126 (8 November 2002).

In the second operating period at Kingsburgh WWTP (17 February 2003 to 24 June 2003), the pilot ABR was operated with a target hydraulic retention time of between 20 and 24 h, with an average of 22 h being achieved. A total flow of 353 000 ℓ of Kingsburgh WWTP influent wastewater was treated. Considerably less down time, or performance affecting incidents occurred during this operating period as a result of improvements to the control algorithm. Incidents, down time and volume of wastewater treated in 2003 are presented in Figure 4.10.

An attempt to increase flow rate (decrease hydraulic retention time) in the second part of 2003 was unsuccessful as a result of mechanical problems. Insufficient data were obtained for the purposes of analysing reactor performance at higher flow rates, and consequently, are not included in this chapter.

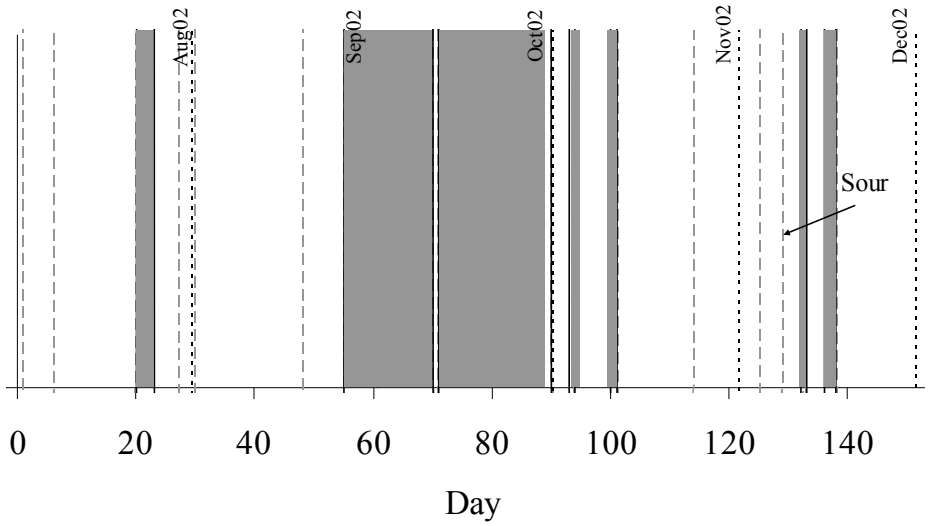


Figure 4.9: Incidents and down time during the 2002 operating period at Kingsburgh WWTP (2 July 2002 to 20 November 2002). Dashed lines (----) indicate potentially performance affecting incidents such as sludge washout, and shaded rectangles indicate reactor down time. A "souring" incident on day 126 is indicated by an arrow.

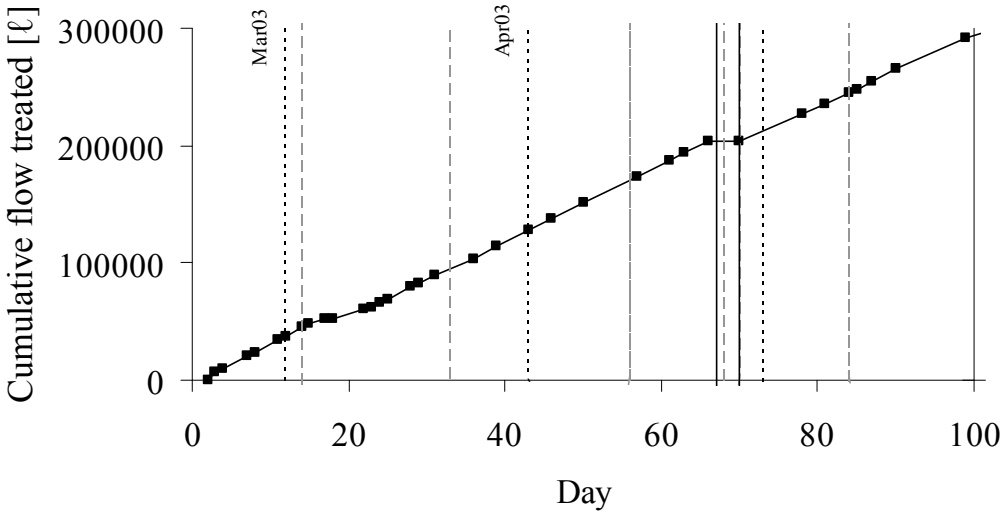


Figure 4.10: Incidents, down time and cumulative flow treated during the 2003 operating period at Kingsburgh WWTP (17 February 2003 to 24 June 2003). Dotted lines (····) indicate potentially performance affecting incidents such as sludge washout, and shaded rectangles indicate reactor down time.

In 2004, a third successful period of operation at Kingsburgh ensued from 7 April 2004 to 8 October 2004, with a target hydraulic retention time of between 40 and 44 h. In this time, 293 000 ℓ of Kingsburgh WWTP influent wastewater was treated. Incidents, down

time and volume of wastewater treated in 2004 are presented in Figure 4.11. Operation can be divided into 3 good operating periods with mean hydraulic residence times of 40.6 h; 44.2 h and 42.3 h (1.2 ℓ/min; 1.1 ℓ/min and 1.2 ℓ/min average flow rate) respectively.

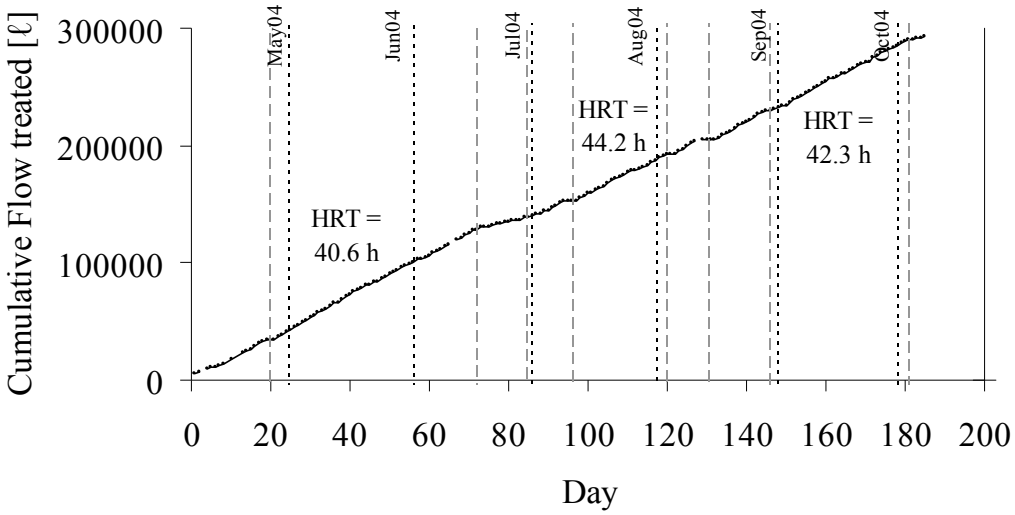


Figure 4.11: Incidents and cumulative flow treated during the 2004 operating period at Kingsburgh WWTP (7 April 2004 to 8 October 2004). Dotted lines (····) indicate potentially performance affecting incidents such as sludge washout. No significant periods of down time were experienced.

4.2.2 Solids Level

Figure 4.12, Figure 4.13 and Figure 4.14 show the settled sludge bed height in each compartment for the 2002, 2003 and 2004 operating periods respectively. Sludge bed height data were obtained using a core sampler (Appendix A1) and recording the height of the sludge bed after 5 min settling time. All three graphs are plotted against a height axis with a maximum of 1.2 m (the maximum height of the liquid within the first compartment). Settled sludge bed height is not an absolute indication of the amount of sludge in a compartment since sludge density can change significantly according to extent of granulation/dispersion, pH, redox potential, operating conditions and inert content (Speece 1996). However, it provides a good visual indication of how the amount of sludge in each compartment varies with time, and how the sludge load varies from one compartment to the next.

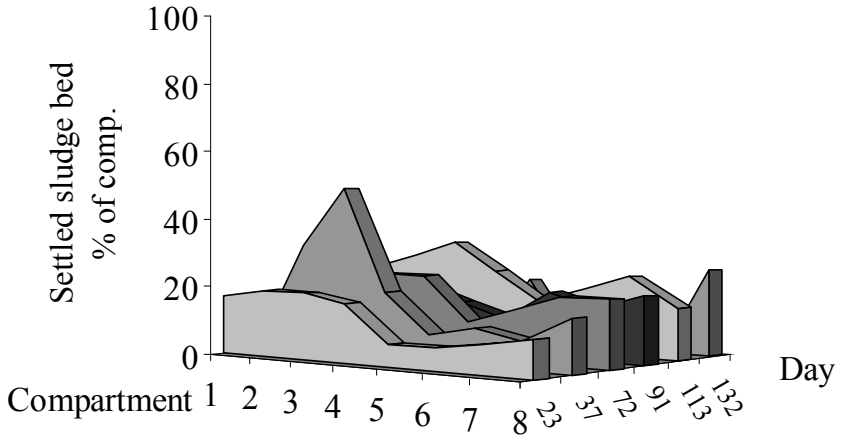


Figure 4.12: Settled sludge bed height in ABR compartments for 7 measurements from day 23 to day 132 during the 2002 operating period at Kingsburgh WWTP.

There is a clear difference in the height of the beds between the different operating periods, with the 2002 operating period (Figure 4.12) being characterised by significantly lower sludge beds than in 2003 and 2004. There is a slight increase in the height of the settled sludge bed with time during the first operating period. Greatest sludge bed height was usually found in compartment 2, although the difference between compartments was generally not great. There were many sludge washout incidents during the 2002 operating period, resulting in a flow-dominated distribution of sludge as opposed to a growth-dominated distribution.

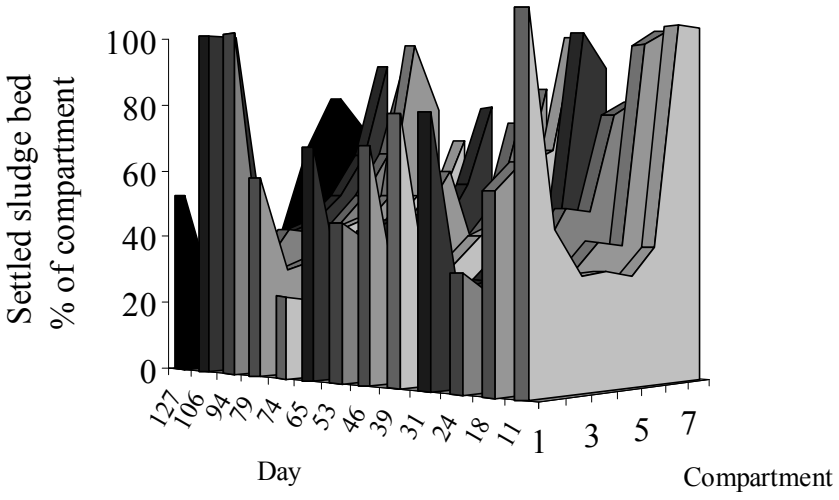


Figure 4.13: Settled sludge bed height in ABR compartments for 13 measurements from day 11 to day 127 during the 2003 operating period at Kingsburgh WWTP.

In 2003, there were few sludge washout incidents, and there is a much less homogenous distribution of sludge between compartments (Figure 4.13). The average sludge bed height is more than double that observed in 2002, and the highest sludge bed was usually in compartment 1. There is a dip in compartment sludge bed height from compartment 3 to 6, with high sludge beds seen in compartments 7 and 8, giving a bow shaped appearance to the profiles. Some degree of sludge granulation was observed in all compartments during this period, but the granule size decreased in progression from compartment 3 to 6. Consequently, the high sludge beds in compartments 7 and 8 are expected to be due to lower density sludge, rather than a higher sludge mass load than in compartments 3 to 6. A gradual increase in the overall sludge bed height with time is observed in the 2003 operating period.

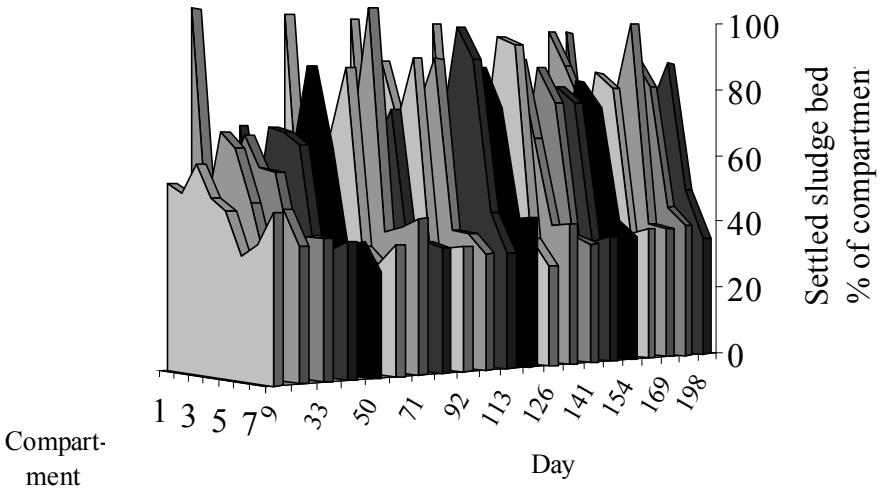


Figure 4.14: Settled sludge bed height in ABR compartments for 21 measurements from day 9 to day 198 during the 2004 operating period at Kingsburgh WWTP.

In the 2004 operating period (Figure 4.14), the average height of the sludge beds increased relative to the previous period, and the shape of the profiles also changed, with dominance in the first compartments. This is due to both the evolution of a granular sludge able to treat dilute wastewater, resulting in better settling in latter compartments, and a longer hydraulic retention time in 2004 than in 2003 resulting in less sludge washout and better sludge retention characteristics. It is difficult to quantify the temporal variation of the sludge bed heights due to the imprecise and subjective nature of the sampling technique, but there appears to be a net increase in sludge bed height until around day 50, while the ABR adapted to the new flow conditions.

4.2.3 COD

Figure 4.15, Figure 4.16 and Figure 4.17 show the influent and effluent COD concentrations measured during operation of the pilot ABR at Kingsburgh WWTP during the operating periods in 2002, 2003 and 2004 respectively. In all cases, black squares (■) indicate weekly measurements of influent COD made by the project team, while

open circles (○) in Figure 4.16 and Figure 4.17 (2003 and 2004) are similar measurements performed on a daily basis by the municipality. There is no significant difference between the measurements of influent COD undertaken by the team and the municipality, and the mean influent COD concentration did not change significantly between operating periods. Effluent samples were in some cases filtered through 0.45 µm acetate filters and analysed for COD as an indication of the amount of *soluble* COD present in the effluent.

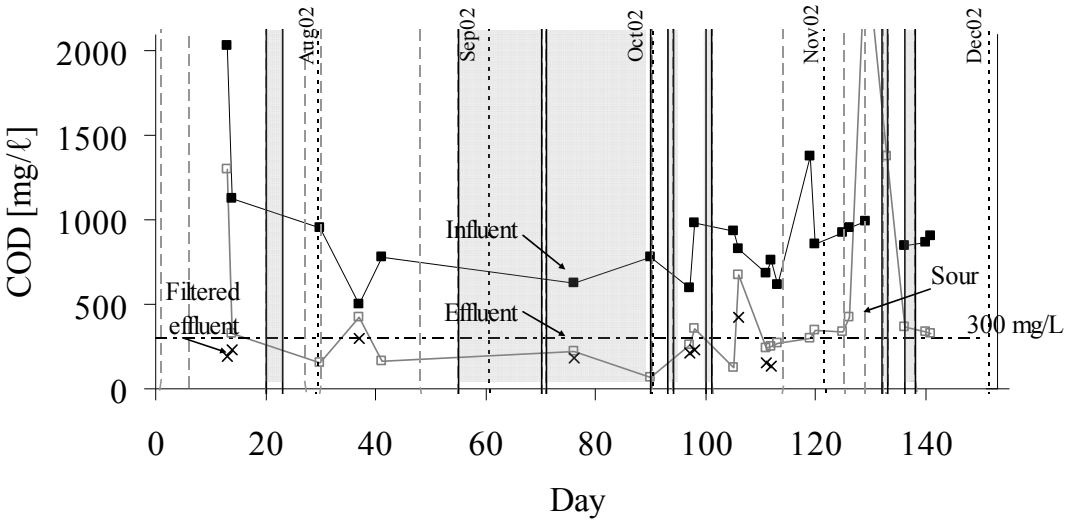


Figure 4.15: Influent and Effluent COD concentrations of the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2002 operating period (estimated mean hydraulic retention time =20 h. Influent (■), effluent (□) and 0.45 µm filtered effluent (×) measurements are shown. The black dash-dot (-•-•) line indicates a COD value of 300 mg/ℓ.

In 2002, effluent COD values were around 300 mg/ℓ, with 20% to 50% of the measured COD being associated with suspended. The ABR consistently removed 500 to 600 mg COD/ℓ except during the souring incident. The lowest effluent COD measurement was 64 mg COD/ℓ. The souring incident on day 126 (2002) (Figure 4.15) resulted in a large spike in the outlet COD concentration. Souring is a result of acidogenesis rate exceeding the rate of methanogenesis; volatile fatty acids accumulate, thereby lowering the pH and inhibiting methanogenesis. If no methanogenesis occurs, no COD removal will occur, and effluent COD values will be high.

In 2003 (Figure 4.16) the measured pilot ABR effluent COD was consistently below 200 mgCOD/ℓ, with the exception of an unexplained high effluent COD value obtained around day 50 (8 April 2003), when the effluent COD value was above 400 mgCOD/ℓ. Although the operating hydraulic retention time should have been similar during this period to the previous (2002) period, the COD removing performance of the reactor was significantly better. This is attributed to more stable flow conditions, and fewer sludge loss incidents than were observed in the 2002 period. A more concentrated biomass,

better suited to compartment conditions was able to develop. This is corroborated by the higher sludge levels seen in 2003 as compared to 2002 (Section 4.2.2).

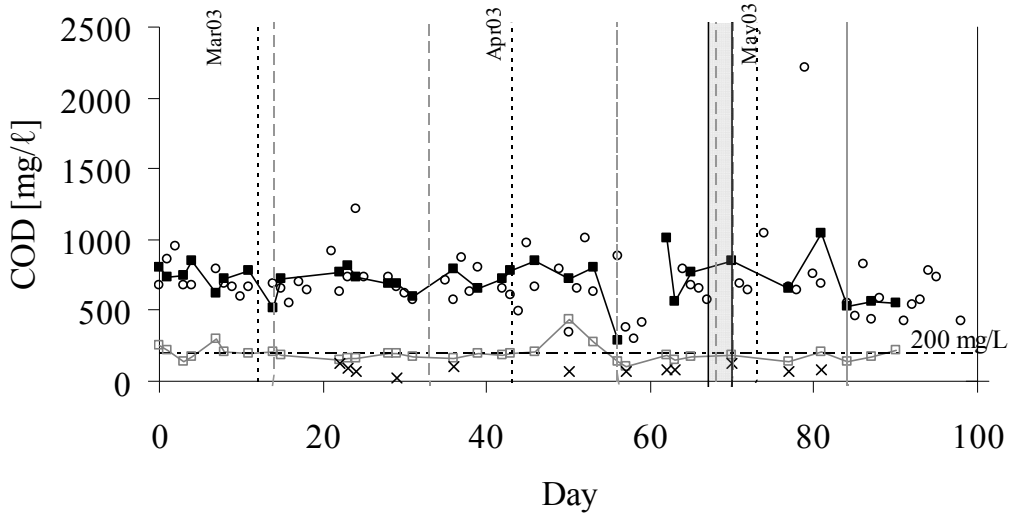


Figure 4.16: Influent and Effluent COD concentrations from the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2003 operating period (Mean hydraulic retention time =22 h). Project team measured influent (■), municipality measured influent (○), effluent (□) and 0.45 µm filtered effluent (×) measurements are shown. The black dash-dot (-·-·) line indicates a COD value of 200 mg/ℓ.

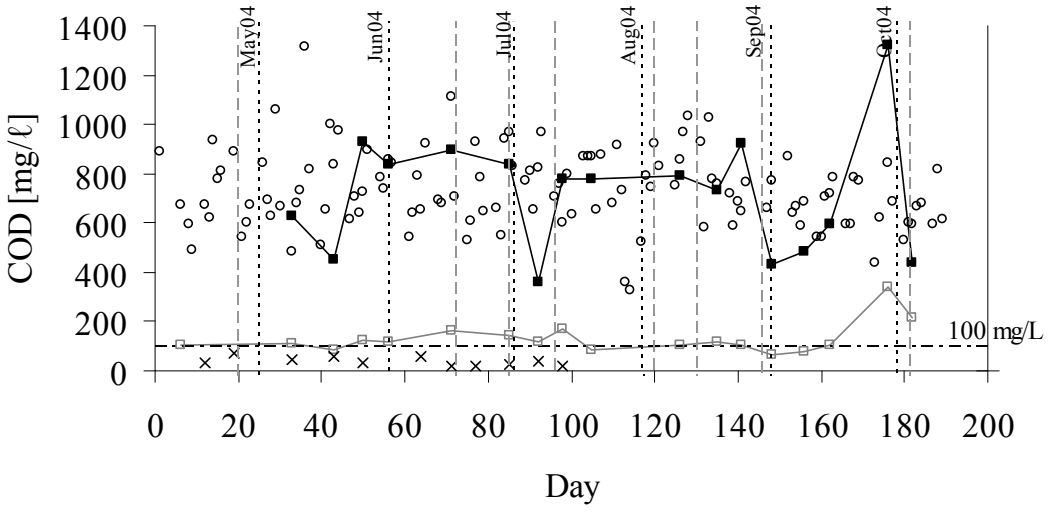


Figure 4.17: Influent and Effluent COD concentrations of the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2004 operating period (Mean hydraulic retention time between 40 and

44 h). Project team measured influent (■), municipality measured influent (○), effluent (□) and 0.45 µm filtered effluent (×) measurements are shown. The black dash-dot (-·-·) line indicates a COD value of 100 mg/ℓ.

In 2004, even lower effluent COD values were obtained, with a mean effluent COD value of 130 mg/ℓ, and with values regularly dipping below 100 mg/ℓ (Figure 4.17). The longer retention time of the 2004 period (42 h) as compared to the 2003 period (22 h) has resulted in a greater *extent of removal* (amount of biodegradable COD converted to methane). Since no reliable gas measurements were made, this cannot be confirmed by mass balance, but this interpretation is borne out by all available data. Low pH conditions after day 170 (2004) resulted in high effluent COD conditions, probably as a result of methanogenesis inhibition.

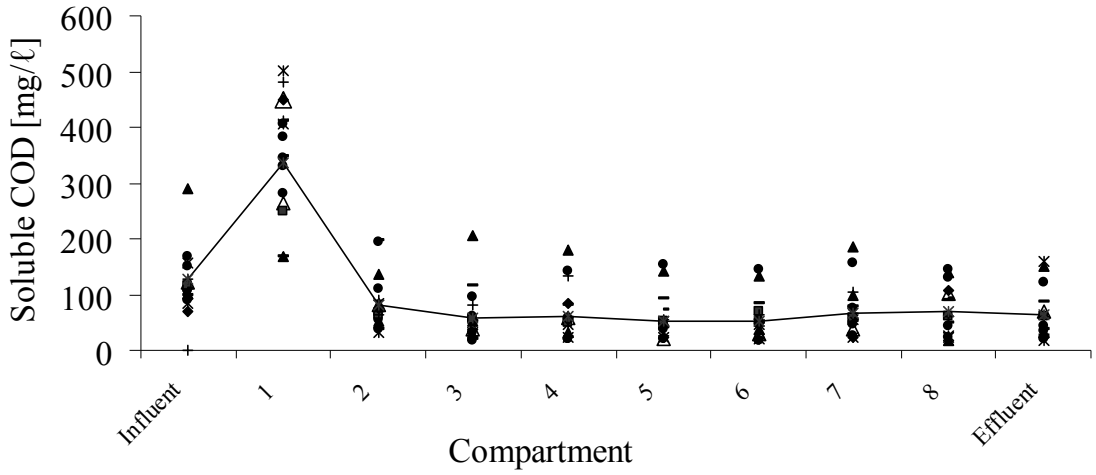


Figure 4.18: Pilot ABR compartment soluble COD concentrations obtained while treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2004 operating period (hydraulic retention times between 40 and 44 h).

Figure 4.18 shows a number of soluble COD profiles measured during the 2004 operating period. In all data sets, a similar pattern is seen: soluble COD concentration increases between the influent and compartment 1 as a result of hydrolysis and acidogenesis in compartment 1; acid production causes a shift of COD from the particulate to the soluble phase, with a corresponding dip in pH value. Methanogenesis in this compartment is unable to remove all the acid that is produced. Hence the pH remains lower and the soluble COD is higher in compartment 1 than the feed. In subsequent compartments, the pH value recovers slightly, and hydrolysis becomes the rate limiting step; i.e. remaining particulate COD is hydrolysed slowly to soluble COD and acid, which then undergoes methanogenesis at the rate at which it is produced. Consequently, a roughly constant pH value and soluble COD concentration is observed from compartments 2 to 8, and in the effluent.

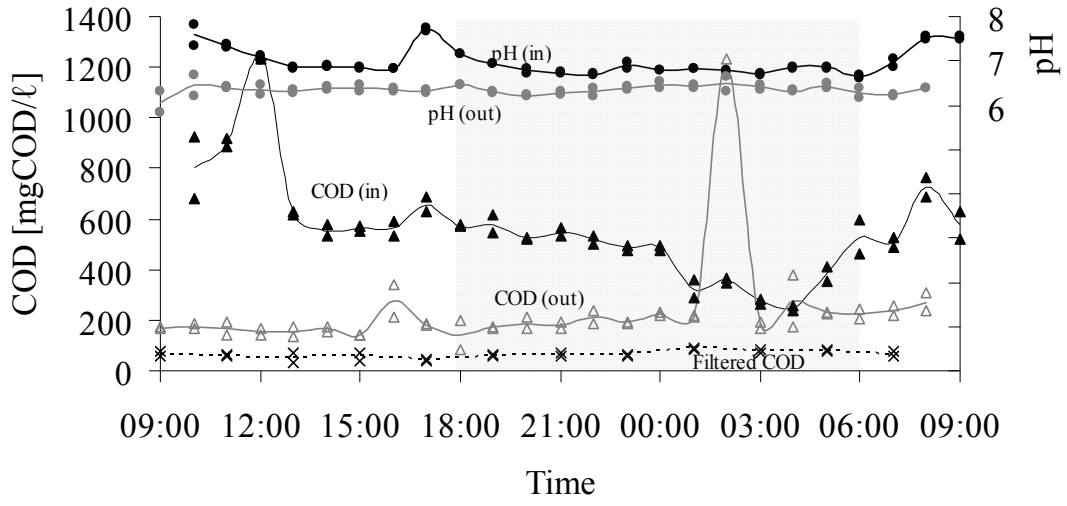


Figure 4.19: Diurnal variation of influent and effluent COD concentration and pH value for hourly samples obtained over 24 h periods in May 2003 from the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP (Mean hydraulic retention time = 22 h)).

Figure 4.19 shows the diurnal variation of influent and effluent COD for the 24 h sampling campaign undertaken in May 2003 (second operating period). The effluent curve has been transposed 22 h back in time, corresponding with the 22 h hydraulic retention time of this operating period, so that ABR effluent characteristics could be directly compared with incoming wastewater characteristics. The influent COD profile shows a sinusoidal variation in load, with highest COD concentrations observed between 10h00 and 14h00, and lowest COD loads between 01h00 and 05h00. There are no corresponding peaks or troughs on the effluent COD profile. The effluent COD concentration remained around 200 mg/ℓ, while filtered COD values were fairly constant at 64 ± 6 mgCOD/ℓ. At 02h00, the effluent COD sample exhibited a sudden spike in COD concentration, caused by a sudden and brief expulsion of sludge. Filtered COD values for these samples remained low. The incident is ascribed to a *burping* phenomenon, whereby gas production, or other fluid effects cause mixing of the sludge in the last compartment, with a short term overflow of the sludge to the effluent. The fact that there is no increase in filtered COD measurement implies a physical effect rather than a biochemical one: Upsets in the anaerobic digestion are most commonly associated with inhibited methanogenesis, and corresponding increase in acid and soluble COD concentration, which is clearly not the case in this event. Similar unexpected sludge overflow incidents were (visually) observed from time to time in all operating periods.

4.2.4 pH

Speece (1996) states that the *proper pH* for anaerobic digestion *must range between approximately 6.5 to 8.2*. The pH values measured in the pilot ABR regularly dropped below 6.5, indicating that the system is poorly buffered in the treatment of dilute wastewaters. This is a function of the low influent alkalinity concentration and low

alkalinity generation potential of the dilute wastewater. As a result, reactor operation is susceptible to souring by shock organic loads or dips in influent alkalinity. (See also Sections 4.1.3 and 2.2.2).

Figure 4.20 shows typical pH profiles across compartments for the different operating periods at Kingsburgh WWTP. The measured pH *values* depend on the alkalinity and pH of the influent specific to the measurements, but the overall *shape* of the profiles is due to the flow and biomass characteristics at the time of measurement. In 2002, the biggest dip in pH is seen in compartment 2, since this is where the greatest amount of sludge is found, and the greatest amount of acidogenesis occurs (Figure 4.12). In 2003 and 2004, the pH dip occurs between the influent and compartment 1, since hydrolysis dominates in compartment 1. In 2002, the pH value never seems to increase after the initial dip. This is attributed to frequent washout events resulting in very small and unstable methanogenic populations. 2003 data shows a decrease in pH value after compartment 3, which may be due to increasingly poor retention of methanogens in later compartments (Figure 4.13).

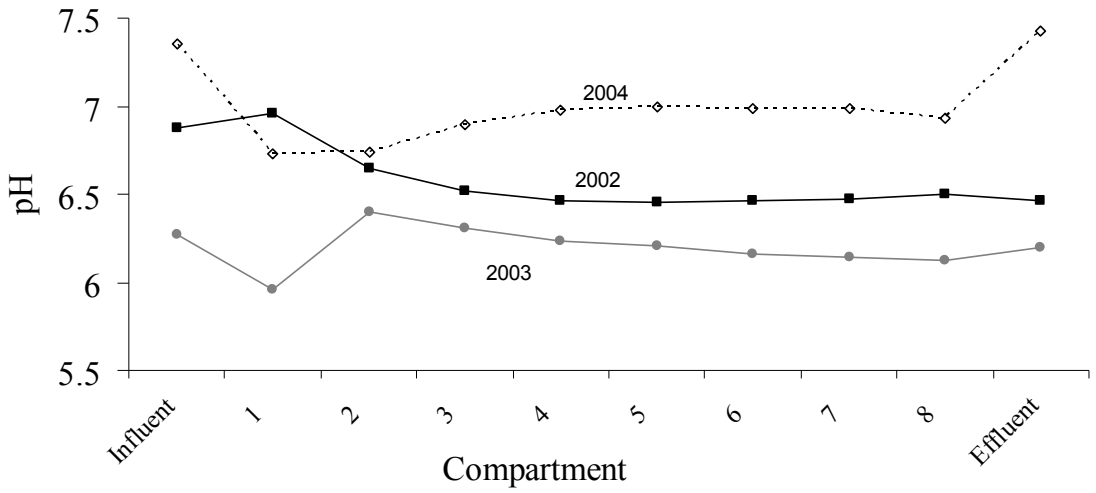


Figure 4.20: Typical pH profiles in the ABR compartments obtained in the 2002 (-■-), 2003 (-●-) and 2004 (··◊··) operating periods at Kingsburgh WWTP treating a relatively dilute (approximately 700 mgCOD/ℓ) wastewater from a middle-income suburb

In 2004, the pH increases continuously after compartment 1 indicating the presence of a stable methanogenic population that removes volatile acids by conversion to methane. The increase between compartment 8 and the effluent in 2003 and 2004 data is a result of aeration of the effluent and subsequent release of CO_2 gas to the atmosphere at the effluent screen. (Effluent screens were installed between the 2002 and 2003 operating periods, section 3.2).

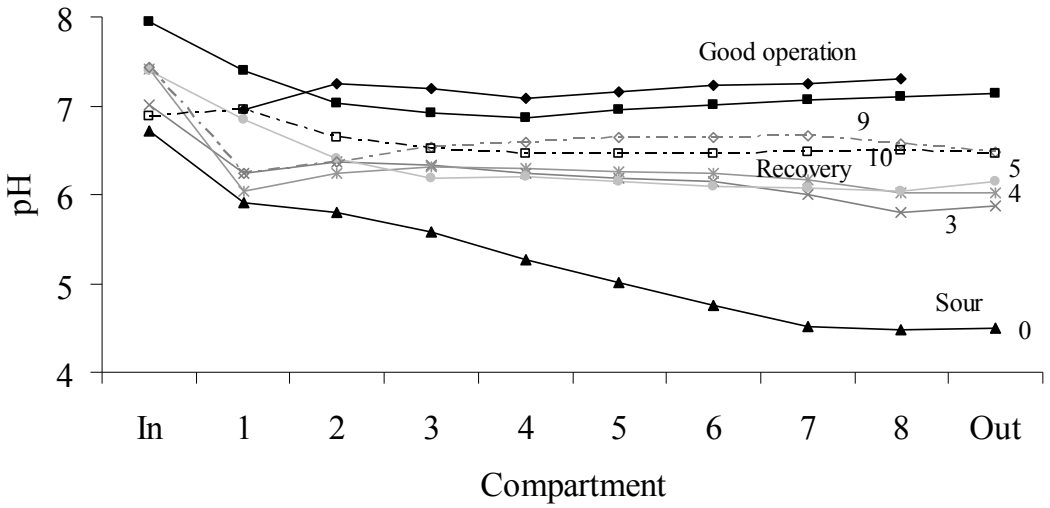


Figure 4.21: pH profiles in the ABR compartments showing good operation (♦ and■), pH profile shortly after souring (▲), labelled 0, and then profiles 3 days after souring (×), 4 days(☼), 5 days(●), 9 days (◊) and 10 days (□)

Figure 4.21 shows a series of pH profiles on different days during the 2002 operating period around a souring incident that may have been caused by an influx of low pH, high COD wastewater resulting from illegal dumping of septic tank contents into the Kingsburgh incoming wastewater. pH values during normal operation, souring and 3, 4, 5, 9 and 10 days after souring are shown. Sour anaerobic conditions result in pH values around 4.5, the pK_a value of acetic acid. It is expected that souring will occur in compartment 1 first, and be propagated to subsequent compartments. Measurements on the day of souring (0) were taken at around 13h00. Illegal dumping is reported to occur between 20h00 and 04h00 suggesting that between 9 and 17 h had passed between souring and pH measurement.

Data for the day of souring (0) shows that pH values as low as 4.5 are only seen in compartments 7 and 8, indicating that the first 6 compartments have already begun to recover. The time for influent flow to reach compartment 7 is 6/8 x 20 h (for a 20 h hydraulic retention time) = 15 h. Consequently, it is supposed that a high COD load was delivered to compartment 1 before 22h00, and by the time the reactor was sampled, the first 6 compartments had already begun to recover. Three days after souring the reactor had essentially recovered, with low (<6) pH values still observed in the later compartments. Ten days after souring, complete recovery was observed.

Operational problems in the recovery period resulted in little flow during this time, which will have accelerated recovery. However, the immediate increase in pH value in the early compartments on the day of souring (with normal flow) implies that rapid recovery is also possible under continuous flow conditions. It is hypothesised that the mechanism of the rapid recovery observed was based on the pseudo-plug-flow nature of the baffled reactor. Acid residues and untreated organics originating from the shock load are

washed out of a compartment at a much faster rate than would be the case in a mixed system while sufficient biomass is retained. Therefore, rapid recovery is seen to occur sequentially in each compartment. This is clear in Figure 4.21 where considerable recovery is seen only a few hours after the event in the early compartments.

Figure 4.19 shows influent and effluent pH values measured in duplicate on an hourly basis over a 24 h period. Effluent values are transposed back a period equivalent to the mean hydraulic residence time of the system so that influent and effluent characteristics can be directly compared. Influent pH values varied between 6.5 and 7.5. Effluent values showed smaller variations, and were consistently lower than influent values.

4.2.5 Alkalinity

Figure 4.23 and Figure 4.24 show influent and effluent alkalinity concentrations for the 2002 and 2003 operating periods respectively. Only a few data points are available for the 2004 operating period; therefore averages for 2004 are reported in Table 4.5.

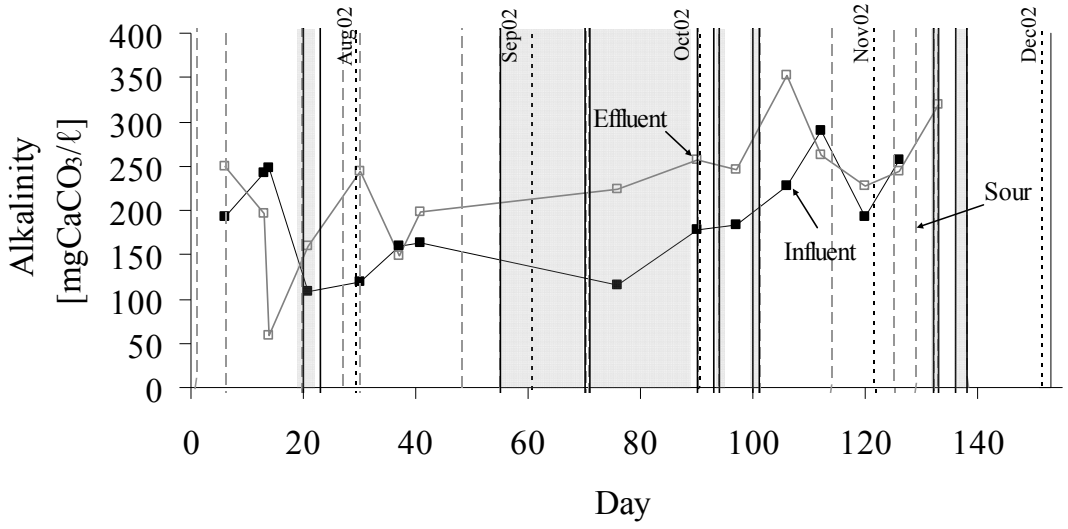


Figure 4.22: Influent and Effluent alkalinity concentrations from the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2002 operating period (Mean hydraulic retention time approximately 20 h). Project team measured influent (■) and effluent (□) values are shown.

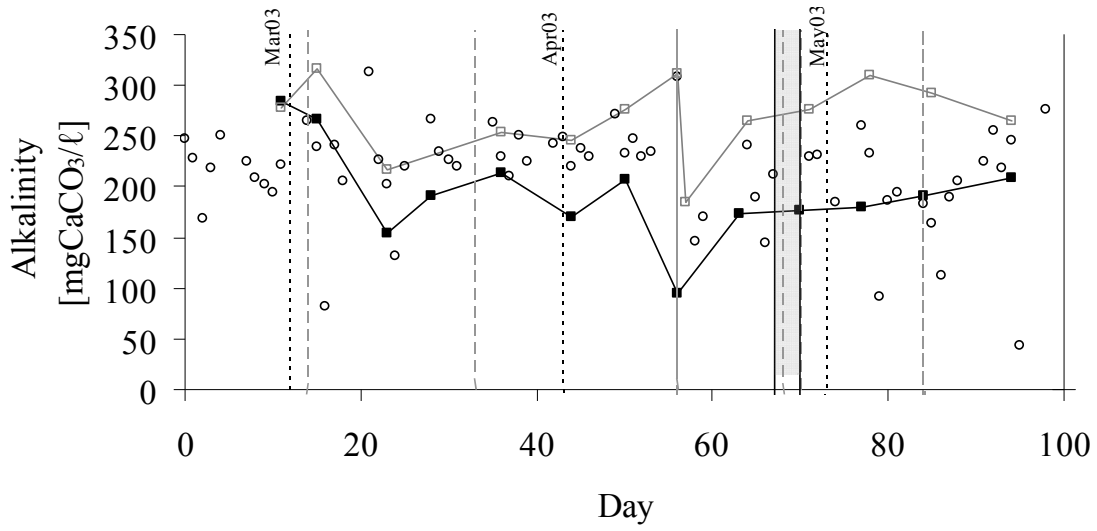


Figure 4.23: Influent and Effluent alkalinity concentrations of the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2003 operating period (Mean hydraulic retention time approximately 22 h). Project team measured influent (■), municipality measure influent (○) and project team measured effluent (□) values are shown.

WWTP in municipal laboratories. Relatively long periods between sampling and analysis occur in the Municipality measured values due to the collection of composite samples; biodegradable COD degrades, generating alkalinity during the collection phase. In comparison, the project team obtained grab samples that were refrigerated or analysed within an hour of sampling. In both Figure 4.23 and Figure 4.24, effluent alkalinity values were above influent alkalinity values since alkalinity is generated in anaerobic digestion by the partial conversion of COD to bicarbonate.

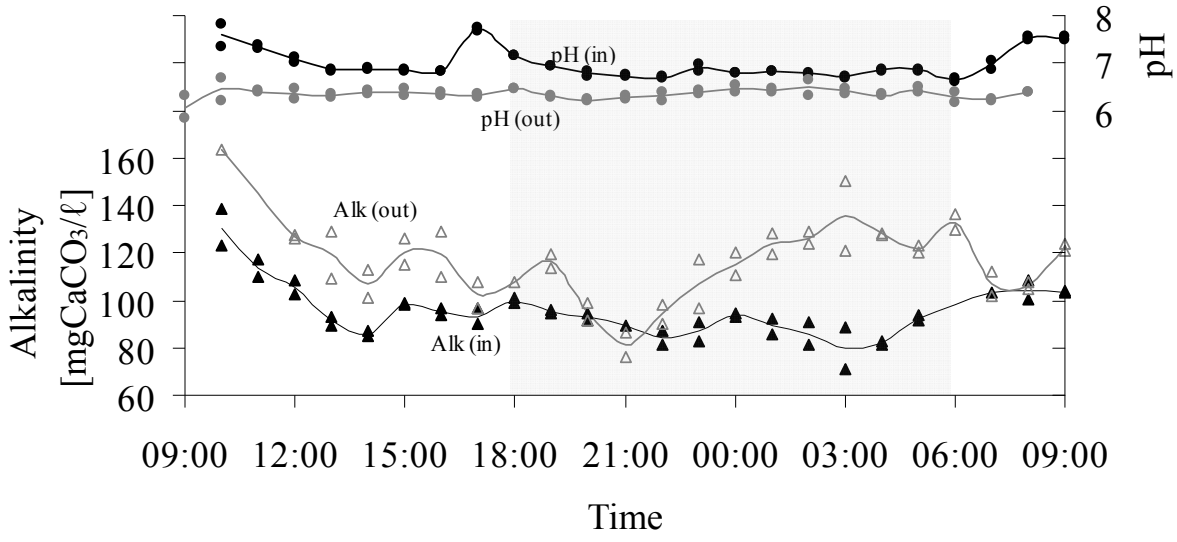


Figure 4.24: Diurnal variation of influent and effluent alkalinity concentration and pH value for hourly samples obtained over 24 h periods in May 2003 from the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP (Mean hydraulic retention time =22 h)).

Project team measurements of alkalinity are slightly lower than those routinely undertaken for the Speece (1996) recommends that alkalinity concentration in the operation of anaerobic digesters is maintained at sufficiently high concentrations to provide a *reserve alkalinity* that is available to neutralise additional acids produced by fermentation. The reserve alkalinity ensures that the operating pH value does not drop below 6.2 to 6.5 since metabolic rates may be adversely affected below these values. It is clear that the pilot ABR treating a relatively dilute (700 mgCOD/ℓ) wastewater is operating without any reserve alkalinity since pH values are lower than these recommended targets. It is therefore probable that poor buffering as a result of low alkalinity, and consequently low pH values, causes non-optimal conditions for microbial activity, resulting in less than maximum treatment rates being achieved.

4.2.6 Phosphorus

No phosphate measurements were performed in the 2002 and 2003 operating periods. Spectrometric phosphate measurements were obtained on influent and effluent samples in 2004. Mean values and standard deviations are presented in Table 4.5. A small but significant decrease in phosphate was observed but, since anaerobic digestion has no mechanism for the removal of significant amounts of phosphate, the apparent removal is expected to be a sampling phenomenon related to the small number of samples analysed (n=7).

4.2.7 Ammonia and Total Kjeldahl Nitrogen

Anaerobic digestion liberates organically bound nitrogen in the feed material as free and saline ammonia ($NH_4^+ + NH_3$), resulting in a net increase in ammonia concentration. In

the 2003 operating period, 7 *ad hoc* samples were obtained from inlet and outlet of the pilot ABR, and analysed for ammonia. As expected, a statistically significant increase in mean ammonia concentration was observed (Student's t-test, P = 0.001).

Total Kjeldahl Nitrogen (TKN) measures the sum of organically bound nitrogen and free and saline ammonia. In a steady-state anaerobic digestion system, (i.e. no net accumulation) the TKN concentration should not change between the inlet and outlet since no TKN exits in the gas stream. During 2004, 8 *ad hoc* measurements of TKN were made of influent and effluent samples, yielding average TKN concentrations of 44.6 and 37.1 mgN/ℓ respectively. This indicates a statistically significant reduction of mean TKN between influent and effluent (Student's t-test, P=0.001). although the reduction is slight, and further, its representivity is limited by the small sample size (n=8), it implies that chemical steady state has not been achieved in the period studied, since removal of TKN could be understood to mean that nitrogen is accumulating via sludge growth. This interpretation is corroborated by increasing total solids concentrations reported in Section 4.2.11.

4.2.8 Sulphate

No sulphate measurements were performed in the 2002 and 2003 operating periods. Spectrometric sulphate measurements were obtained on influent and effluent samples during the 2004 operating period. A statistically significant decrease (Student's t-test, P = 0.001) from 4.5 mgSO_4^{2-}/ℓ in the influent to 0.4 mgSO_4^{2-}/ℓ in the effluent was observed. Mean values and standard deviations are presented in Table 4.5. It is not expected that an average removal of 4 mgSO_4^{2-}/ℓ will support a large sulphate reducing population.

4.2.9 Volatile Fatty Acids

Several attempts were made to measure volatile fatty acids in the influent and compartments of the pilot ABR in the 2003 and 2004 operating periods. In 2003, a GC method in which VFAs were extracted in diethyl ether was used, and acetic, propionic and butyric acids were detected in the first 4 compartments. However, no reliable calibration of the method was obtained, and therefore concentrations cannot be reported. In the 2004 period, an HPLC method for determining VFA was implemented. Only a few data sets are available from this exercise. The 5-point titration method for determining alkalinity and VFA of Moosbrugger et al., (1992) was also generated 3 data sets. There did not appear to be any particular correlation between the two measurements. However, it was noted that the influent wastewater to the pilot ABR usually contained some acetate (between 0 and 80 mg CH_3COOH/ℓ), and that considerably greater concentrations of acetate were measured in compartment 1 (between 64 and 407 mg CH_3COOH/ℓ). No propionate was detected in the feed, while ca. 70 mg CH_3CH_2COOH/ℓ was measured in compartment 1 for two sets of data.

4.2.10 Solids concentrations

There are two causes of increase in solids load in each compartment. The first is the build-up of biomass due to growth on substrate originating from the reactor feed; the

second is an overflow of the sludge from one compartment to the next caused by either entrainment in the liquid flow, or growth in the previous compartment displacing extra sludge over the intermediate standing baffle. The last mechanism is illustrated in Figure 4.25, which shows the total solids concentration in compartments 4, 5 and 6 during the 2004 operating period. Compartment 5 shows an increase in mean solids concentration between the commencement of sampling and around day 71. In Figure 4.26, this corresponds to a *settled* sludge bed height of 100% of the total compartment contents height (i.e. the compartment is carrying a maximum load of sludge). This is followed by a dramatic increase in both total solids concentration and sludge bed height in compartment 6 between day 98 and 113 (2004), as a result of sludge being literally displaced from compartment 5 to compartment 6.

There is a considerable amount of inaccuracy in both the solids levels and solids concentration data as a result of the difficulty in obtaining representative samples. However, the trends, however imprecise, clearly suggest that compartment filling and sludge overflow is occurring. The implication is that there is a maximum concentration of solids that a compartment can hold before the mechanism by which sludge passes from the one compartment to the next changes from entrainment to displacement. A second implication is that if, in an even longer term experiment, similar patterns are seen in compartments 7 and 8, there will reach a point when sludge will wash out of the reactor in significant concentrations as a result of displacement. When this occurs, desludging would be required. Total solids data from compartments 7 and 8 seem to show an increasing trend after Day 113 (2004), although it is not a consistent increase, but there is no similar increase in sludge bed height (Figure 4.27). This is attributed to the changing nature of the sludge from a dispersed sludge blanket to the formation of granules.

Unless a steady-state point exists within the reactor at which the decay rate of sludge equals the rate at which sludge is created through growth, there will inevitably come a time when the last compartments fill with sludge and regular or continuous loss of unacceptable amounts of solids will occur. However, as the settling properties and density of sludge is still changing in the data presented here, it is not possible to predict when this will occur.

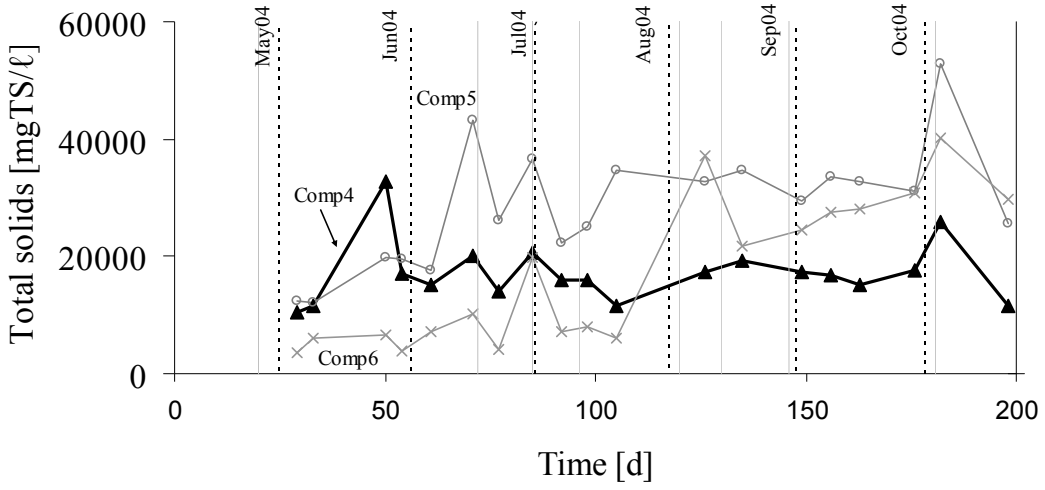


Figure 4.25: Compartments 4 (-▲-), 5 (-○-) and 6 (-×-) total solids (TS) concentrations from the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2004 operating period (Mean hydraulic retention time approximately 42 h), showing filling of compartments 5 and 6 as a result of growth and sludge carry-over

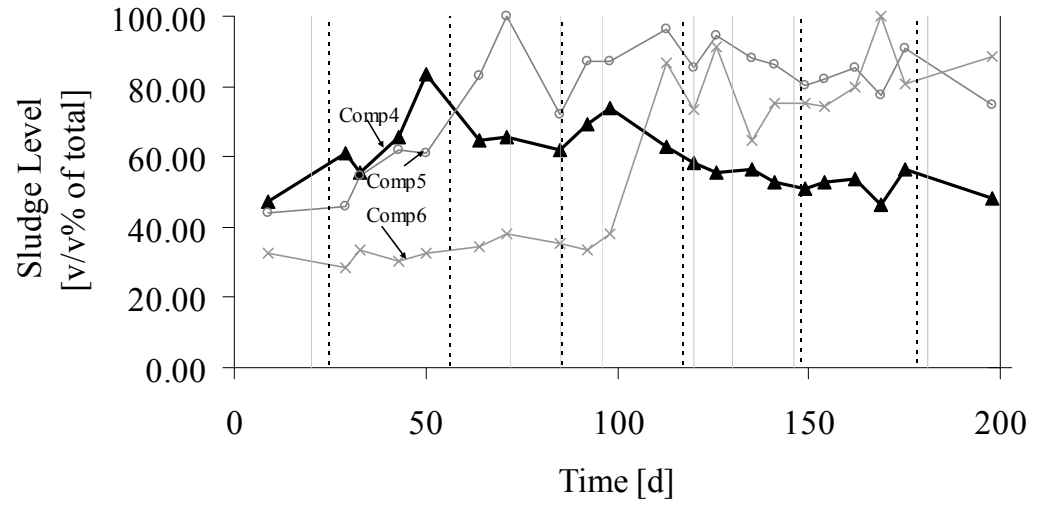


Figure 4.26: Compartments 4 (-▲-), 5 (-○-) and 6 (-×-) settled sludge bed height (as V/V% of compartment height) from the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2004 operating period (Mean hydraulic retention time approximately 42 h)), showing compartments 5 and 6 approach to maximum sludge bed height.

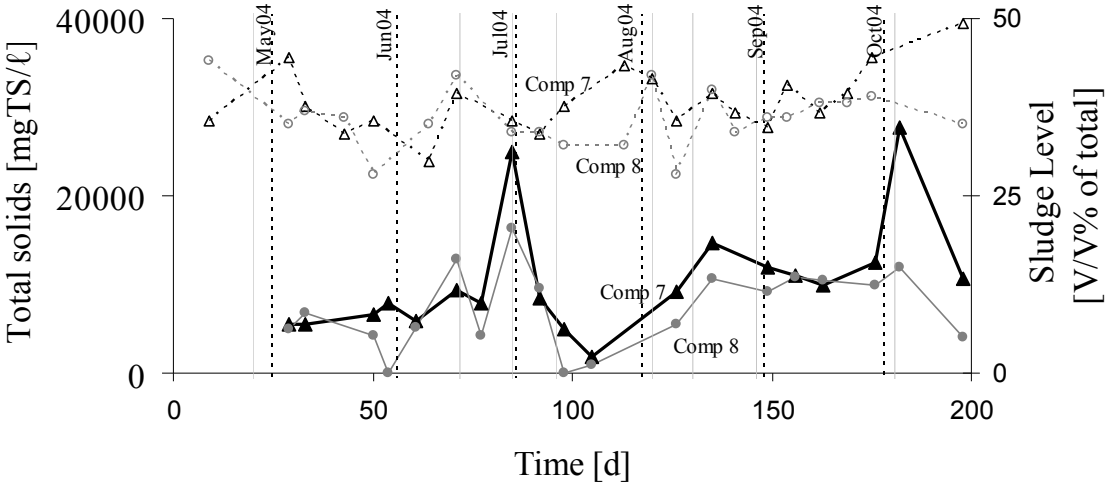


Figure 4.27: Total solids and sludge bed height (as V/V% of compartment height) of compartments 7 and 8 from the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2004 operating period (Mean hydraulic retention time approximately 42 h). Compartment 7 TS (-▲-), compartment 8 TS (-●-), compartment 7 sludge level (··Δ··) and compartment 8 sludge level (··○··) are shown.

4.2.11 Pathogen indicator organisms

In the 2004 operating period at Kingsburgh WWTP, measurements of *Escherichia coli* (*E. coli*), total coliforms, coliphages and *Ascaris* spp. were made on samples obtained from the influent and effluent of the pilot ABR.

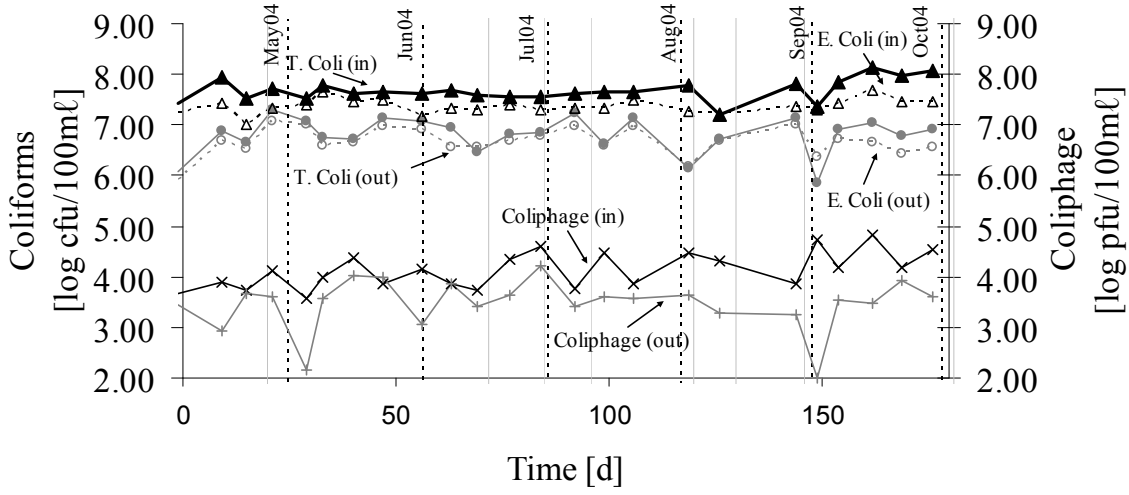


Figure 4.28: Pathogen indicator organisms in the influent and effluent of the pilot ABR treating a middle-income domestic wastewater at Kingsburgh WWTP during the 2004 operating period (Mean hydraulic retention

time approximately 42 h). Influent and effluent total coliforms (-▲- and -●-), *E. Coli* (··△·· and ··○··) and coliphage (-×- and -+-) are shown.

4.2.11.1E. coli

A comparison of inlet and outlet measurements for is shown in Figure 4.28. The number of *E. coli* fed into the reactor ranged between 1 x 10^7 and 5 x 10^7 cfu/100 mℓ. The reduction of *E. coli* from inlet to outlet was significant (Student's t-test, p = 0.000) but variable throughout the study period. Average effluent measurements ranged from 7 x 10^5 to 1 x 10^7 cfu/100 mℓ with an average reduction of 76 % (1.87 log).

4.2.11.2Total coliforms

A comparison of inlet and outlet measurements for total coliforms is shown in Figure 4.28. The number of total coliforms fed into the ABR from the inlet pipe ranged between 2 x 10^7 and 1 x 10^8 cfu/100 mℓ. The removal efficiency of total coliforms (83 %; 1.9 log) by the ABR was marginally higher than the removal of *E. Coli*. However, average total coliform count in the effluent was consistently higher than the *E. coli* count and ranged between 1 x 10^6 and 2 x 10^7 cfu/100 mℓ. The reduction of total coliforms from inlet to outlet was statistically significant (Student's t-test, p = 0.000).

4.2.11.3Coliphages

A comparison of inlet and outlet measurements for coliphages is shown in Figure 4.28. The reduction of coliphages in the digester was variable and, though significant (Student's t-test, p = 0.000), not as high as that of the coliform groups. Coliphages in the reactor ranged between 4 x 10^3 and 7 x 10^4 pfu/100 mℓ in the influent and 1 x 10^2 and 2 x 10^4 pfu/100 mℓ in the effluent. A mean removal efficiency of 64 % (1.8 log) was achieved.

4.2.11.4Ascaris spp.

Total helminth eggs were counted in samples of influent and effluent. Eggs counted were not assessed for viability. The concentration of eggs, assumed to be of *Ascaris* spp. varied weekly and eggs were observed in all influent samples analysed. The number of eggs in the influent was high (347 to 1 253 eggs/ℓ) with an average concentration of 772 eggs/ℓ (with a standard deviation of 340 eggs/ℓ) for the study period. The average egg concentration of the effluent followed a similar pattern to the influent, with an average egg concentration of 17 eggs/ℓ (with a standard deviation of 15 eggs/ℓ) observed after the digestion process. An average removal of 98% was achieved during the study period.

4.2.11.5Summary of pathogen indicator organism study, 2004

Reduction of coliform groups, coliphages and *Ascaris* eggs were variable throughout the study period. Significant reductions were obtained for all indicator organisms. The reduction of *E. coli* and total coliforms followed similar trends, but coliphage and parasite reduction did not correlate well with coliform reduction. The removal of total

coliforms was consistently higher than that of *E. Coli.* The reduction of coliphages was less than the coliform groups. The results imply that viruses have a marginally greater survival through the digestion process and that microbial indicators do not show similar survival rates. From all indicators tested, the greatest reductions were observed for *Ascaris* eggs. The performance of the ABR in removing helminth eggs is probably attributed to eggs having a larger mean residence time within the reactor due to sedimentation.

Although the reductions of the various indicators were significant, none of the microbial and parasitic parameters met the requirements for discharge, either to water resources or irrigation agriculture. It is therefore likely that the effluent may harbour a wide range of microbial pathogens and parasites, which may present a potential health risk to humans and water supplies. It is recommended that a post-treatment option be considered as an integral part of the technology.

4.2.12 Summary of operation at Kingsburgh Wastewater Treatment Plant

Table 4.3, Table 4.4, Table 4.5 and Table 4.6 present a summaries of all influent and effluent measurements averaged for operation of the pilot ABR at Kingsburgh WWTP for the 2002, 2003 and 2004 operating periods. As was observed for operation while installed at Umbilo WWTP, interpretation of the data hinges on understanding the dynamics of the sludge in each compartment. Section 4.2.1 shows that the average load of sludge in the pilot ABR increased with each subsequent operating period, indicating either that microbial steady state was not achieved during any of the operations or that inert solid material was accumulating in the compartments. It is also suggested that the distribution of the sludge between the compartments differs according to dominating operational aspects (e.g. in the 2002 period, which was characterised by repeated high flow incidents, sludge distribution was flow-dominated, while it appeared to be growth dominated in 2003 and 2004).

Higher sludge loads and fewer washout incidents in 2003 as opposed to 2002 resulted in better overall COD removal, although the hydraulic retention time was roughly the same for the two operating periods. Longer hydraulic retention times in 2004 resulted in even better COD removal due to extended wastewater-sludge contact time, and higher compartment sludge concentrations.

The mechanism of sludge build-up has been shown to be a combination of growth and sludge carry-over between compartments. In Section 4.2.11, it is seen that the sludge bed in a compartment can build up until the entire compartment is full. In the event of this occurring in the last compartment, considerable carry-out of sludge will occur, and desludging in all compartments, but especially the last, may be required. However, this did not occur in the 5 years of operation of the pilot plant. It was further observed that the nature of the sludge was still changing at the end of the experimental study, in that the initially dispersed sludge appeared to be forming granules, and that the degree of granulation and size of granules appeared to be increasing. Therefore an accurate prediction of the increase in sludge bed height with time/flow rate (and therefore desludging requirements) is not possible since sludge bed density at future times will probably be different to those observed during the experimental studies.

The profile of pH values in the ABR also provides some clues as to how the treatment process is progressing. In Section 4.2.4 analysis of the shape of the pH value profile in each operating period was used to understand the relationship between acidogenic and methanogenic processes in the ABR. In a poorly buffered application, the shape of the pH profile can therefore give an indication of how close the overall process is to failure.

Anaerobic digestion of domestic wastewater in the ABR occurs without any *reserve alkalinity*, causing operating pH values to regularly drop below 6.5. Significant inhibition of methanogenesis will therefore occur. Since hydrolysis has been identified as the rate limiting process in all but the first compartment, it is not expected that methanogenesis inhibition is reducing the overall COD removal (except possibly during the 2002 operating period). However, inhibition will reduce the methanogenic growth rate and therefore increase the risk of methanogen washout. This compromises the ability of the ABR to withstand, and recover from, shock loads, either hydraulic or organic, and therefore lessens the advantage of installing an ABR over simpler technology, such as a septic tank.

Anaerobic treatment of domestic wastewater in the ABR caused a net increase in alkalinity and ammonia concentrations, and a slight decrease in TKN was observed in the last operating period (2004). A small amount of sulphate in the feed stream was removed by the ABR. Nitrate present in the influent will be completely removed in the first few compartments of the system. Phosphate in the influent stream is not expected to change as a result of treatment in the ABR. ABR effluent therefore contains increased concentrations of alkalinity and ammonia and similar concentrations of phosphate compared to the influent wastewater. Negligible sulphate and nitrate occur in the effluent.

Significant removal of pathogen indicator organisms was observed in the 2004 operating period. However, effluent coliform, coliphage and *Ascaris* spp. concentrations in the effluent are sufficiently high that the effluent should be considered a risk to human health.

Table 4.3: Influent and effluent characteristics, 2002. Summary of data from the pilot ABR treating middle-income domestic wastewater at Kingsburgh WWTP during the 2002 operating period. Calculations of averages and standard deviations are presented for all measurements except pH value, for which median value is reported.

		Average/ median	Std Deviation	No of observations	Min.	Max.
COD	In	906	317	22	500	2 037
[mg COD/ℓ]	Out	299	131	20	64	674
Soluble COD						
[mg COD/ℓ]	Out	204	53	8	132	298
Alkalinity	In	191	56	14	109	289
[mg$CaCO_3$/ℓ]	Out	226	70	15	59	353
Total solids	In	808	135	15	570	1 052
[mgTS/ℓ]	Out	475	138	15	310	820
Volatile solids	In	557	93	11	417	705
[mgVS/ℓ]	Out	268	172	11	117	605
pH	In	7.4		7	6.7	7.9
[Median value]	Out	6.2		7	4.5	7.1
	Mean HRT:	**20 h**	**Total flow treated:**	**349 820 ℓ**		

Table 4.4: Influent and effluent characteristics, 2003. Summary of data from the pilot ABR treating middle-income domestic wastewater at Kingsburgh WWTP during the 2003 operating period. Calculations of averages and standard deviations are presented for all measurements except pH value, for which median value is reported.

		Average/ Median	Std Deviation	Number of observations	Min.	Max.
COD	In	651	190	56	249	1239
[mg COD/ℓ]	Out	212	143	57	107	1202
Soluble COD		127		2	113	141
[mg COD/ℓ]	Out	71	21	26	27	121
Alkalinity	In	193	48	13	95	285
[mg$CaCO_3$/ℓ]	Out	268	38	13	185	316
Ammonia	In	25	5	7	20	34
[mgN/ℓ]	Out	34	3	7	30	39
PO_4	In	2.6	2.3	4	0.4	4.7
[mgP/ℓ]	Out	5.5	0.5	5	4.7	6.0
Total solids	In	480	188	14	253	965
[mgTS/ℓ]	Out	225	96	14	80	390
Volatile solids	In	306	105	14	125	538
[mgVS/ℓ]	Out	127	79	14	5	290
pH	In	7.0		9	6.3	7.5
[Median value]	Out	6.5		9	6.2	6.7
	Mean HRT:	**22 h**	**Total flow treated:**	**352 658 ℓ**		

Table 4.5: Influent and effluent chemical characteristics, 2004. Summary of data from the pilot ABR treating middle-income domestic wastewater at Kingsburgh WWTP during the 2004 operating period. Calculations of averages and standard deviations are presented for all measurements except pH value, for which median value is reported.

		Average/ Median	Std Deviation	Number of observations	Min.	Max.
COD	In	688	204	202	246	1749
[mg COD/ℓ]	Out	130	64	18	62	339
Soluble COD	In	157	89	18	69	395
[mg COD/ℓ]	Out	104	110	19	18	427
Alkalinity	In	256	38	190	142	369
[mg$CaCO_3$/ℓ]	Out	246	53	4	168	286
Ammonia	In	40	11	189	11	95
[mgN/ℓ]	Out	51	23	10	20	90
TKN	In	45	3	8	40	51
[mgN/ℓ]	Out	37	4	8	32	45
PO_4	In	13.3	4.4	167	3.2	32.5
[mgP/ℓ]	Out	20.3	5.7	7	10.3	26.2
Total solids	In	701	186	15	416	1076
[mgTS/ℓ]	Out	368	114	13	135	556
pH	In	7.2		195	4.4	7.8
[Median value]	Out	6.5		6	6.2	7.4
VFA	In	33	34	4	0	79
[mgCH_3COOH/ℓ]	Out	nd				
Sulphate	In	4.5	1.3	5	2.4	5.8
[mgSO_4/ℓ]	Out	0.4	0.3	5	0.2	1.0
Sodium	In	150	119	5	87	362
[mgNa/ℓ]	Out	132	140	5	40	380
Potassium	In	21	4	6	16	26
[mgK/ℓ]	Out	25	5	6	19	31

Table 4.6: Influent and effluent microbial characteristics, 2004. Summary of data from the pilot ABR treating middle-income domestic wastewater at Kingsburgh WWTP during the 2004 operating period. Calculations of averages and standard deviations are presented for all measurements except pH value, for which median value is reported.

		Average/ Median	Std Deviation	Number of observations	Min.	Max.
Total Coliforms	In	7.3		25	7.0	7.7
[log(cfu/100mℓ)]	Out	6.6		25	5.8	7.1
E. Coli	In	7.7		25	7.2	8.1
[log(cfu/100mℓ)]	Out	6.8		25	5.9	7.3
Coliphage	In	4.1		24	3.6	4.8
[log(pfu/100mℓ)]	Out	3.5		24	2.0	4.2
***Ascaris* spp.**	In	772	341	13	347	1 500
[Number eggs/ℓ]	Out	17	15	13	2	56
	Mean HRT:	**22 h**	**Total flow treated:**	**352 658 ℓ**		

4.3 LIMITATIONS OF THE PILOT ABR STUDY

It proved impossible to obtain accurate measurements of gas production rates from the pilot ABR system as a result of the pressure buffering provided by the standing baffle system, in which gas production in the reactor displaces liquid within the reactor as well as in a liquid displacement gas measuring system. To overcome internal pressure buffering, all compartment gas production needs to be collected simultaneously to prevent the increased pressure in each compartment being redistributed to neighbouring compartments. This was not possible with the available equipment.

A reliable mass balance would increase confidence in the results of COD removal calculations. However, as many measurements of influent and effluent COD were made over the course of the project, and furthermore, other measurements support conclusions drawn from COD data, it is not believed that the lack of gas production data negatively impacts on the validity of the conclusions drawn from the pilot ABR experiments.

Similarly, in all of the operating periods, incomplete data sets were obtained for many components, including flow rate, total solids, volatile solids, alkalinity and VFA. However, all the data that has been obtained fits into a coherent explanation of ABR operation on domestic wastewater.

4.4 SUMMARY OF THE PILOT ABR STUDY

The 3 000 ℓ pilot ABR was operated over a 5 year period at Umbilo and Kingsburgh WWTP. During 2000 and 2001, the pilot ABR was operated at Umbilo WWTP on a feed of 50:50 industrial : domestic wastewater at target hydraulic retention times of 60 h, 32 h and 20 h. The reactor was initially seeded with a small amount of anaerobic digester sludge (approximately 10 ℓ) and consequently had a lengthy start-up period in which biomass built up in each of the compartments. Analysis of solids concentrations 200 mm above the bottom of the reactor in the first year of operation showed compartment sludges developing sequentially, i.e. the rate of accumulation in any compartment was faster than in the subsequent compartment.

Amounts of sludge in each compartment continued to change through most of the operating periods, and it cannot be stated with any certainty that a steady state with respect to solid load in each compartment was reached. The rate of sludge build-up was also dependent on wastewater feed flow rate. In the first operating periods at Umbilo WWTP, development of appreciable sludge levels only occurred after the target hydraulic retention time had been decreased to 20 h.

In 2002, the pilot ABR was moved to Kingsburgh WWTP where it operated on a feed of domestic wastewater from middle-income suburbs south of Durban. Three operating periods, in 2002, 2003 and 2004 were achieved. During operation at Kingsburgh WWTP, when fairly well established sludge blankets were present in all compartments, sludge levels were not high in the operating period characterised by repeated high flow incidents that resulted in sludge washout (2002). Sludge levels also seemed to be higher in the 2004 period (40 to 44 h hydraulic retention time) than the 2003 period (22 h hydraulic retention time). This could be due either to lower pseudo-steady-state sludge levels establishing at the higher washout rate of the 2003 period, or simply that sludge was still accumulating during the 2004 period.

Significant COD removal was observed in all operating periods, except immediately after commissioning. Fairly constant effluent COD concentrations were observed except during process upsets such as a souring incident in November 2002. Effluent COD concentration decreased with decreasing hydraulic retention time at Umbilo. This decrease is attributed to improving reactor performance as a result of establishing sludge populations, rather than a function of loading. Effluent COD concentrations decreased significantly when the hydraulic retention time was increased from 22 h to over 40 h between the 2003 and 2004 operating periods at Kingsburgh WWTP as a result of increased contact time in the reactor at the higher retention time.

During operation at Umbilo WWTP, higher pH values were observed than during operation at Kingsburgh WWTP. The reason for this difference is not clear, although it may be attributable to generally lower treatment rates during the Umbilo testing as a

result of low biomass populations resulting in low acidification rates, or some function of the semi-industrial nature of the Umbilo wastewater.

Alkalinity values during all of the experimental periods were low relative to standard anaerobic digestion applications. Consequently the pilot ABR was poorly buffered and therefore susceptible to pH inhibition. In general pH values in the ABR, especially during operation at Kingsburgh WWTP were low, and consequently reduced micro-organism activities, particularly of methanogens could be inferred. Alkalinity of the wastewater consistently increased as a result of anaerobic digestion in the pilot ABR, as expected.

The shape of the pH profile (i.e. the relative changes in pH value between compartments) showed different trends in all of the 4 operating periods. Examination of each of the profiles provided clues to the relative rates of acid producing and consuming processes in each of the compartments.

Results of analyses for VFA concentrations in samples were not conclusive. It appeared that after sampling, storage and pre-treatment VFA concentrations measured were not the same as those that existed in the ABR. However, it could be seen that some VFA were present in the influent to the ABR, and that the concentration in compartment 1 was much higher than the influent or subsequent compartments, supporting the hypothesis that acid accumulates in compartment 1.

Enumeration of pathogen indicator organisms (total coliforms, *E. coli*, coliphage and helminth eggs) in the influent and effluent of the pilot ABR in the 2004 operating period in each case showed significant removals. However significant counts of all indicator organisms were observed in all effluent samples indicating that further disinfection is required before ABR effluent can be reused.

5 MICROBIAL COMMUNITY CHARACTERISATION STUDIES OF THE ABR

Two studies of microbial community dynamics were undertaken during operation of the pilot anaerobic baffled reactor at Kingsburgh WWTP. The first study in 2003 studied the dynamics of a sample of micro-organism classes and genera using a number of molecular techniques. The second study was performed in 2004 using *Scanning Electron Microscopy* (SEM) to provide visual evidence to support FISH results, and to gain an understanding of the granulation process that appeared to be occurring in the pilot ABR.

5.1 CHARACTERISATION OF MICROBIAL COMMUNITIES USING MOLECULAR TECHNIQUES

This study formed the basis of an MTech dissertation at Durban Institute of Technology (Lalbahadur, 2004). The following results have been extracted from the dissertation and reanalysed in conjunction with other data available from the pilot ABR during this period.

5.1.1 Objective of this study

The microbial consortia in an anaerobic digester both catalyse biochemical processes of wastewater treatment and are generated through growth on the wastewater and its degradation products. In Chapter, physico-chemical data relating to the performance of the reactor in treating relatively dilute wastewater (COD ca. 700 mg/ℓ) were presented and interpreted to describe the microbial conditions in the system. A microbiological study was instigated to assist in the interpretation of the data, and in the development of a theory of the processes in an ABR. This study sought to:

- obtain a measure of biomass concentrations in the ABR at different periods during operation
- identify and quantify different types of micro-organisms present in different compartments of the ABR
- compare and correlate changes in the microbial population to physical and chemical changes in the ABR

5.1.2 Materials and methods

This study used DNA and RNA based techniques to identify and enumerate micro-organisms in the compartments of the pilot ABR operating at Kingsburgh WWTP on a feed of middle-income domestic municipal wastewater in 2003.

5.1.2.1 Sampling dates and procedure

Samples were obtained on 5 days during the 2003 operating period described in Section 4.2.1, namely days 36 (March 2003), 57 (April 2003), 85 (May 2003), 101 (May 2003) and 127 (June 2003).

Samples were obtained from each of the 8 compartments of the pilot ABR using the coring technique described in Appendix 2, mixed in a bucket and collected in 50 mℓ centrifuge tubes. Samples were sealed and transported on ice to the laboratories at the Centre for Water and Wastewater Research at Durban Institute of Technology.

5.1.2.2 Molecular techniques for identification and enumeration of micro-organisms

Three molecular techniques were used for the identification and enumeration of microbial consortia in the samples. The details of the methods for these techniques may be found in Lalbahadur (2004). The three methods are:

- Total cell counts, obtained by subjecting pretreated samples to 4'6-diamidino-2-phenylindole (DAPI) staining, using the membrane filtration method of Porter and Feig (1980).

- Fluorescent in situ hybridisation (FISH) studies, using general oligonucleotide probes for eubacteria and archaea and a suite of 12 genera- or family-specific probes according to Hicks et al. (1992) and Amann (1995).

The Polymerase Chain Reaction (PCR) technique (Jackson et al., 1991) to extract and amplify DNA from whole cells. Agarose gel electrophoresis was used to separate DNA fragments. DNA sequencing was performed by Inqaba Biotechnical Industries (South Africa) using a Spectrumedix SCE2410 genetic analysis system. Sequence results were determined using the Chromas Version 2.3 (Technelysium) sequence analysis programme.

5.1.3 Principle of FISH

FISH and DAPI techniques were used to obtain an indication of the relative numbers of different types of micro-organisms in the ABR compartments.

FISH is able to enumerate micro-organisms with specific genetic characteristics. A fluorescently labelled single oligonucleotide strand, the probe, is introduced into a morphologically intact cell; and binds specifically with a complementary site on ribosomal RNA (rRNA) if present in the cell. If not, the probe is washed away. The binding site is detected by epifluorescence microscopy (Hugenholtz et al., 2001); a single active cell contains many rRNA strands (the number depending on the activity of the cell) and therefore, many potential binding sites. As a result, an active cell will appear as a bright point of fluorescence when excited at the appropriate wave-length for the fluorescent label. In this way, a species- or genus-specific rRNA sequence can be detected, and micro-organisms present may be identified (Amann et al., 1995).

Probes vary in length between 15 and 25 nucleotides, and the order or *sequence* of the nucleotides is created to match that of 16S and 23S rRNA molecules in the target micro-organism. 16S rRNA sequences have been determined for a large number of bacterial species, and therefore, it is theoretically possible to obtain a comprehensive characterisation of the microbial community in a sample.

5.1.4 Principle of DAPI staining

FISH is almost always complemented by 4'6-diamidino-2-phenylindole (DAPI) staining. DAPI forms fluorescent complexes with DNA that fluoresce bright blue. Unbound DAPI, or DAPI bound to non-DNA material fluoresce pale yellow. DAPI does not differentiate between live, dying or dead organisms, as long as intact DNA is present (Porter and Feig, 1980), and therefore, under stressed conditions, where cell activities are generally low, it may result in cell counts in excess of those obtained by FISH techniques. The DAPI cell counts provide information on the total number of cells present, whereas FISH provides information on the number active cells of a specific target species.

5.1.5 Limitations of FISH

Although FISH techniques are widely regarded as providing accurate and reproducible results, a number of limitations exist which should be understood before interpreting FISH data:

- FISH probes may bind non-specifically, i.e. they bind to RNA with similar sequences to those targeted in the specific micro-organism or category of micro-organisms sought, leading to an overestimate in target cell numbers (Amann, 1995).

- Background fluorescence in the sample at the wavelength at which the probes fluoresce can cause an overestimate of cell numbers. This latter problem is more common when digital analysis tools are used to count cells, than in manual counting (Amann, 1995).

- Slow-growing, or starving cells have low cellular ribosome contents and therefore yield low signals that either may not be detected, or may be interpreted as belonging to background fluorescence.

- Certain micro-organisms have cell walls that have a low permeability to the probes, or rRNA that is not easily accessed by the probe due to cell morphology. In this case, underestimation of cell numbers is caused by failure of the probe to bind with rRNA present in target cells. (Schramm and Amman, 1999).

- When cells are aggregated in clusters or a biofilm, it can be difficult to distinguish individual cells, resulting in over- or underestimates of the actual cell number, depending on the bias of the counting system (Daims et al., 2001).

5.1.6 Details of FISH study

Table 5.1 lists the FISH probes used in this study.

Table 5.1: Name, specificity and 16S rRNA sequence of FISH probes used to enumerate micro-organisms in the 8 compartments of the pilot ABR

Probe	Specificity	Sequence (5'-3')
EUB338	Bacteria (16S, 338-355)	GCTGCCTCCCGTAGGAGT
ARC915	Archaea (16S, 915-934)	GTGCTCCCCCGCCAATTCCT

ALF1b	α-*Proteobacteria* (16S, 19-35)	CGTTCG(C/T)TCTGAGCCAG
BET42a	β-*Proteobacteria* (23S, 1027-1043)	GCCTTCCCACTTCGTTT
GAM42a	γ-*Proteobacteria* (23S, 1027-1043)	GCCTTCCCACATCGTTT
SRB385	δ-*Proteobacteria* (16S, 385-402)	CGGCGTCGCTGCGTCAGG
LGC354a	*Firmicutes* low G+C (16S, 354-371)	TGGAAGATTCCCTACTGC
HGC69a	High G+C (23S, 1901-1918)	TATAGTTACCACCGCCGT
CF319a	*Cytophaga-Flavobacterium* (16S, 354-371)	TGGTCCGTGTCTCAGTAC
BAC303	*Bacteriodes-Prevotella* (16S, 385-402)	CCAATGTGGGGGACCTT
DSB985	*Desulfobacteriaceae* (16S, 985-1004)	CACAGGATGTCAAACCCAG
DSV698	*Desulfovibrionaceae* (16S, 698-717)	GTTCCTCCAGATATCTACGG
MS821	Methanosarcina (16S, 821-844)	CGCCATGCCTGACACCTAGCGAGC
MX825	Methanosaeta (16S, 825-847)	TCGCACCGTGGCCGACACCTAGC

The first two probes referred to in Table 5.1 (EUB 338 and ARC915) are domain-specific for all eubacteria (bacteria) and archaea respectively. These two domains form the prokaryotes, small simple cells that do not contain a membrane-enclosed nucleus. (All other known cell types belong to the domain Eucarya, or Eukaryotic cells, which incorporates all animal cells, green plant cells, flagellates, ciliates, fungi and microsporidia, Bailey and Ollis, 1986). Archaea have different cell wall structures to eubacteria, and exhibit transcription and translation processes that have features that are different to bacteria, but show similarity to those of eukaryotes (Woese, 1987).

Figure 5.1 shows an unrooted[2] bacterial phylogenetic tree describing the divisions of bacteria. Bacterial groups probed for in this study include *Bacteriodes*, *Cytophaga-flexibacter*, α, β, γ and δ-*Proteobacteria*, low G+C Gram positive bacteria and high G+C Gram positive bacteria. Sulphur-reducing bacteria that belong to the δ-*Proteobacteria* group and are understood to be found in anaerobic digestion (*Desulfovibrionaceae* and *Desulfobacteriaceae*) were also investigated. Table 5.2 describes the functions of the bacteria probed.

[2] An *unrooted* phylogenetic tree is a group of phylogenetic trees where all the roots are depicted as being linked, although there is not necessarily any evidence that the evolution of these trees originating from a single root as depicted.

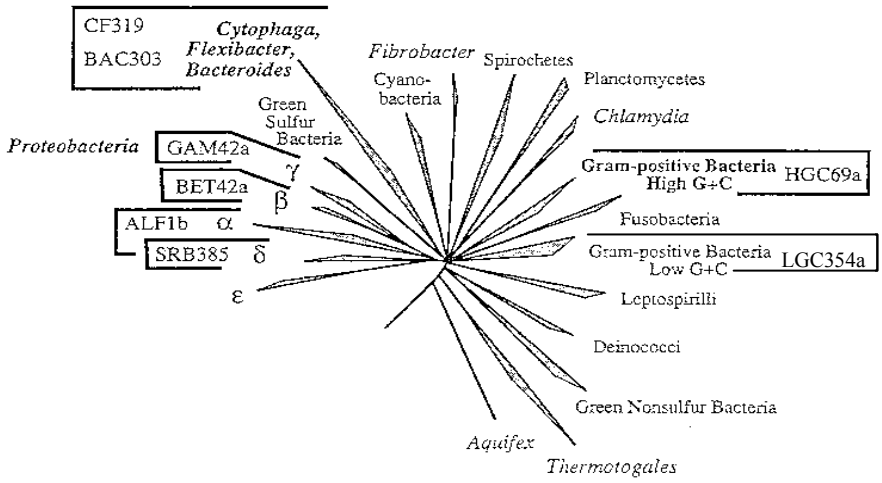


Figure 5.1: Unrooted bacterial phylogenetic tree showing divisions of bacteria, and bacterial groups probed in this study

Table 5.2: Function executed by micro-organisms enumerated by different probes

Probes	Specificity	Functionality
ALF1b	α-*Proteobacteria*	Fermentative acidogenic bacteria
BET42a	β-*Proteobacteria*	Aerobic bacteria
GAM42a	γ-*Proteobacteria*	Fermentative acidogenic bacteria
SRB385	δ-*Proteobacteria*	Sulphate reducing and acetogenic bacteria
LGC354a	*Firmicutes* low G+C	Hydrolytic and acidogenic bacteria
HGC69a	High G+C	Acetogenic bacteria
CF319a	*Cytophaga-Flavobacterium*	Hydrolytic bacteria
BAC303	*Bacteriodes-Prevotella*	Fermentative acidogenic bacteria
DSB985	*Desulfobacteriaceae*	Family of sulphate reducing bacteria
DSV698	*Desulfovibrionaceae*	Family of sulphate reducing bacteria
MS821	*Methanosarcina*	Coccoid methanogenic archaea
MX825	*Methanosaeta*	Filamentous scavenging methanogenic archaea

5.1.7 Results of FISH study

In this section, results of FISH and DAPI analyses are presented.

5.1.7.1 Total cell counts

Figure 5.2 presents total cell counts obtained from DAPI staining for the four sampling days. The sample for compartment 6 on day 127 was spilt, was not analysed. There is no clear trend across compartments or sampling days. However, on day 36, a complete set of compartment total solids concentration data is available, (Figure 5.3) and it can

be seen that the total solids and total cell counts plotted against compartment for that sampling day have the same shape, showing a significant correlation between the two measurements. It can therefore be assumed that the total cell count is dependent on the amount of solid retained in the compartment. This implies that the amount of micro-organism in a compartment is affected by solids retention characteristics, as well as growth kinetics, although only the later is usually considered in microbiological analysis.

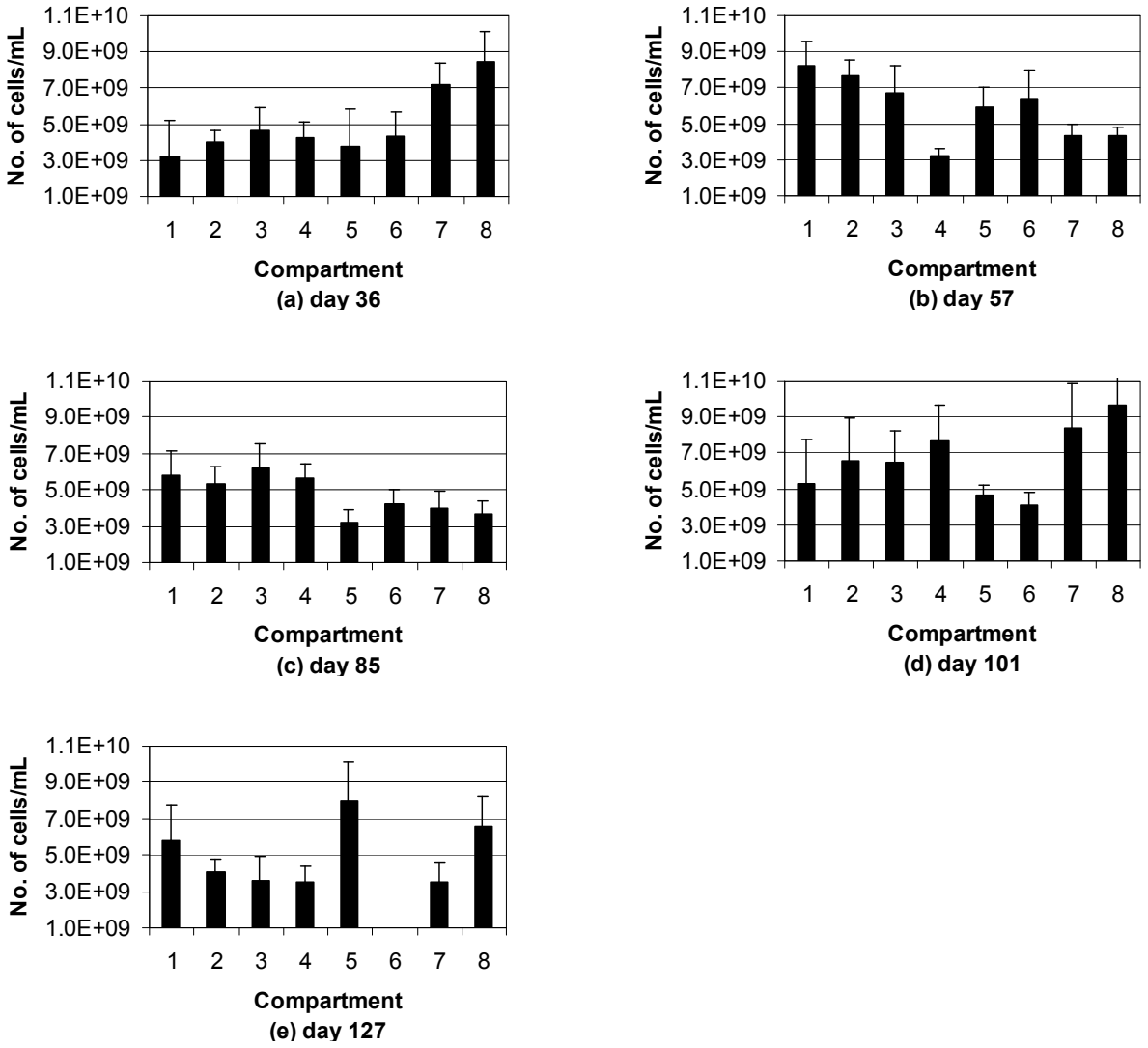


Figure 5.2: Total cell counts obtained by DAPI staining in each compartment for samples obtained on day 36, 57, 85, 101 and 127 respectively during the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb (each measurement calculated from counts from between 14 and 20 fields). The sample from compartment 6 on day 127 was lost

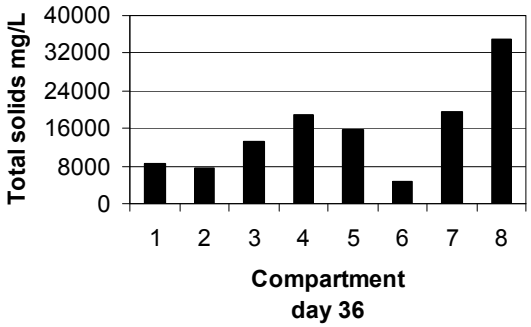


Figure 5.3: Total solids profile in pilot ABR on day 36 of the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb.

5.1.7.2 Eubacterial and Archaeal domain-specific probe counts

Domain-specific probes EUB338 and ARC915 were used to quantify micro-organisms in the domains Eubacteria and Archaea respectively. The EUB338 probe is complementary to the conserved 16S rRNA of most bacteria (Harmsen et al., 1996). This probe does not detect all Eubacteria: certain bacteria belonging to the *Cytophaga-Firmicutes-Bacteriodes* phylum, as well as *Spirochetes* and *Acidobacterium* are not detected by EUB338 (Daims et al., 1999). Hybridisations with these probes showed dominance of bacteria over archaea in all samples.

Eubacterial cells consisted on average of 47% of the total microbial population (Figure 5.4). Archaeal cells detected constituted approximately 4% of the microbial population (Figure 5.4).

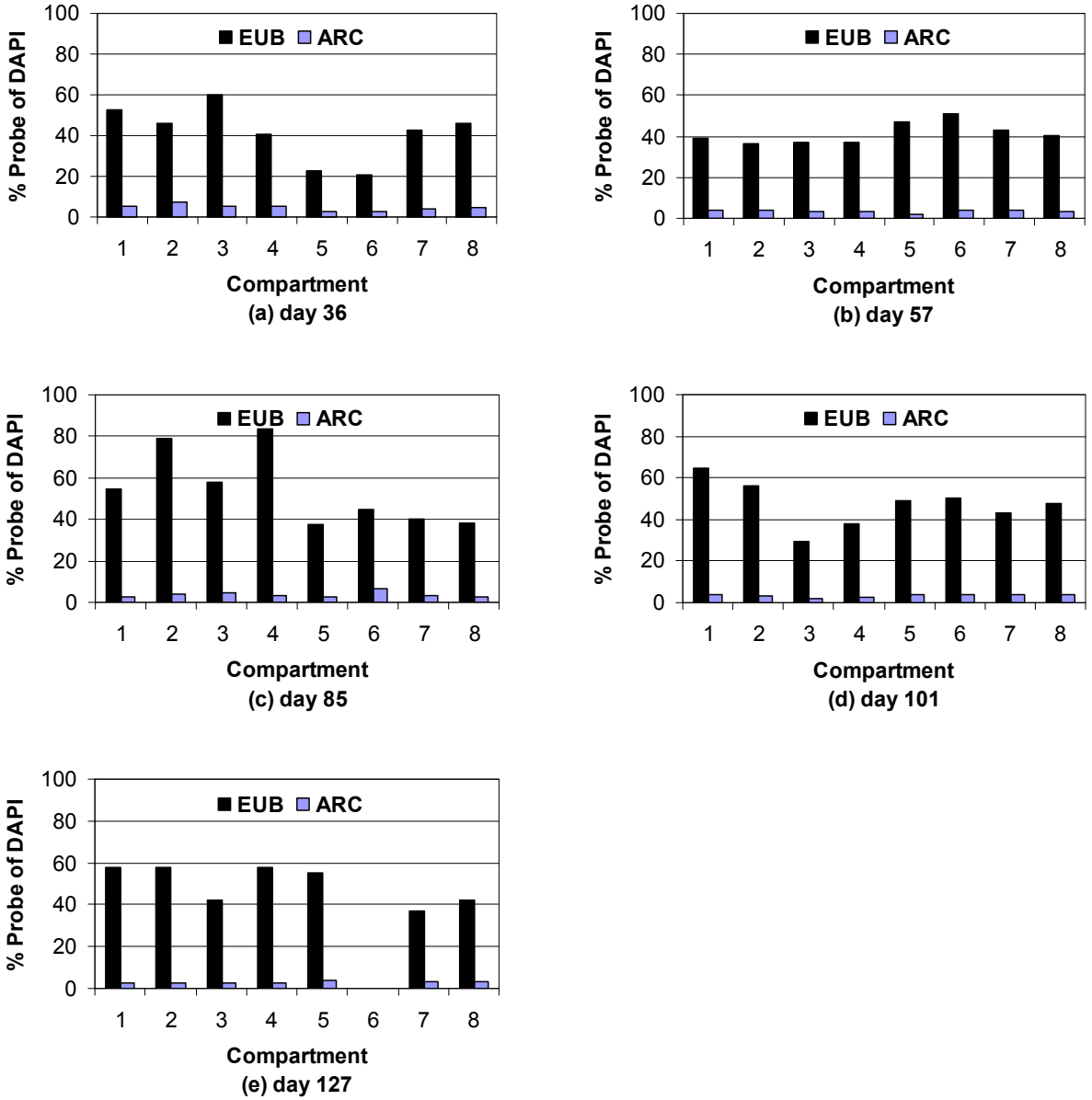


Figure 5.4: Domain-specific probe counts (Eubacteria and Archaea) as a fraction of total cell counts in each compartment for samples obtained on day 36, 57, 85, 101 and 127 respectively during the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb. The sample from compartment 6 on day 127 was lost.

In total, around half of the cells counted by DAPI staining were determined by domain-specific probing (Figure 5.4, Figure 5.5). It is possible that probe counts were low as a result of low amounts of cellular ribosomal RNA (rRNA) within intact cells, due to relatively low substrate concentrations for many of the micro-organisms. Low cellular rRNA is characteristic of slow-growing and starving cells (Amann et al., 1990). This is particularly true of acetoclastic methanogens, the category of Archaea responsible for methanogenesis, which are characterised by low growth rates, experience low acetate

concentrations in the ABR, and the cell walls of which often exhibit low permeability to probes. (Amann et al., 1990, Wagner et al, 2003).

Low recovery of total cell counts in the domain-specific counts was probably also be due to the presence of micro-organisms in the microbial community that were not probed. These include fermentative ciliates, protozoa and anaerobic fungi, as well as the few bacterial species not detected by the EUB338 probe.

Of the EUB338 detected cells, 54% on average for the five days sampled, were detected by the other 8 family-specific probes (Figure 5.6). Day 57 showed the highest number of eubacterial cells (average 76% for the eight compartments) being identified by the 8 probes. Day 127 showed the lowest number of bacterial groups identified (average of 36%). This indicates the presence of bacteria in the ABR which have not been accounted for by the probes.

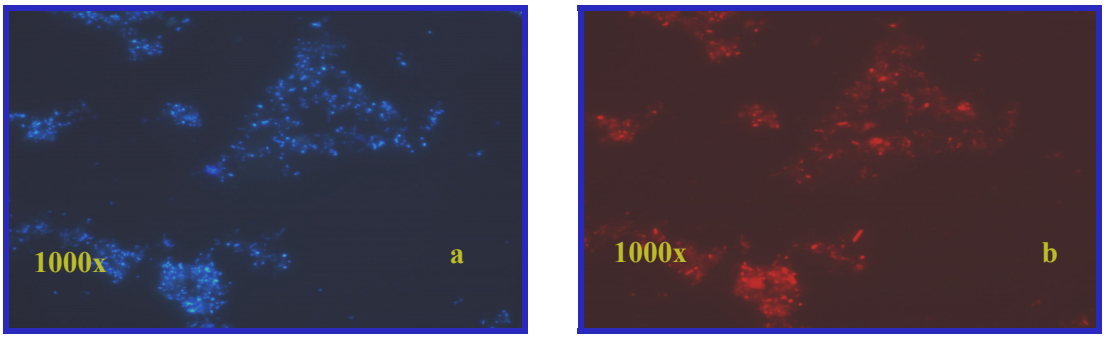


Figure 5.5: Images of the same field showing (a) DAPI stained and (b) EUB338 hybridised cells of Compartment 1 (Day 36).

5.1.7.3 Hydrolytic bacteria

(Section 2.2.1.1) Bacteria that effect hydrolysis are found mainly in the groups Low G+C Gram positive bacteria (LGC, detected by probe LGC354a) and *Cytophaga-Firmicutes* (probe CF319a). Some members of the LGC bacterial group are also responsible for acidogenesis; therefore these bacteria are shown in both hydrolytic and acidogenic categories. These bacteria execute hydrolysis of organic polymers (proteins, cellulose, lignin and lipids) into soluble monomers (amino acids, glucose, fatty acids and glycerol) (Bitton, 1994 and Guiot et al., 1992). Figure 5.7 shows number of cells detected by LGC354a and CF319a probes in each compartment on each sampling day.

There are no clear trends in the data presented here, although it appears that the number of cells detected by the LGC354a probe was in general greater than cells detected by the CF319a probe. The difference was less marked, and in some cases reversed in the later compartments as compared to earlier compartments. These data do not show a strong correlation with total cell count data.

5.1.7.4 Fermentative acidogenic bacteria

(Section 2.2.1.3) The fermentative acidogenic bacterial group consists of bacteria belonging to α and γ subclasses of *Proteobacteria* (detected by ALF1a and GAM1b probes respectively), low G+C Gram positive bacteria (detected by LGC354a probe) and *Bacteriodes* (BAC303 probe) classes. They use soluble monomers produced by the hydrolytic bacteria to form organic acids (acetic, propionic, formic and lactic), alcohols, ketones, carbon dioxide and hydrogen (Bitton, 1994 and Guiot et al., 1992).

Figure 5.8 shows the number of cells detected by ALF1a, GAM1b, LGC354a and BAC303 probes in each compartment on each sampling day.

For each compartment and each sampling day, there was a strong correlation among the different probe counts. LGC354a probe generally detected more cells than other group-specific probes. As with hydrolytic bacteria group-specific probe counts, there was no strong correlation between these data and total cell count data.

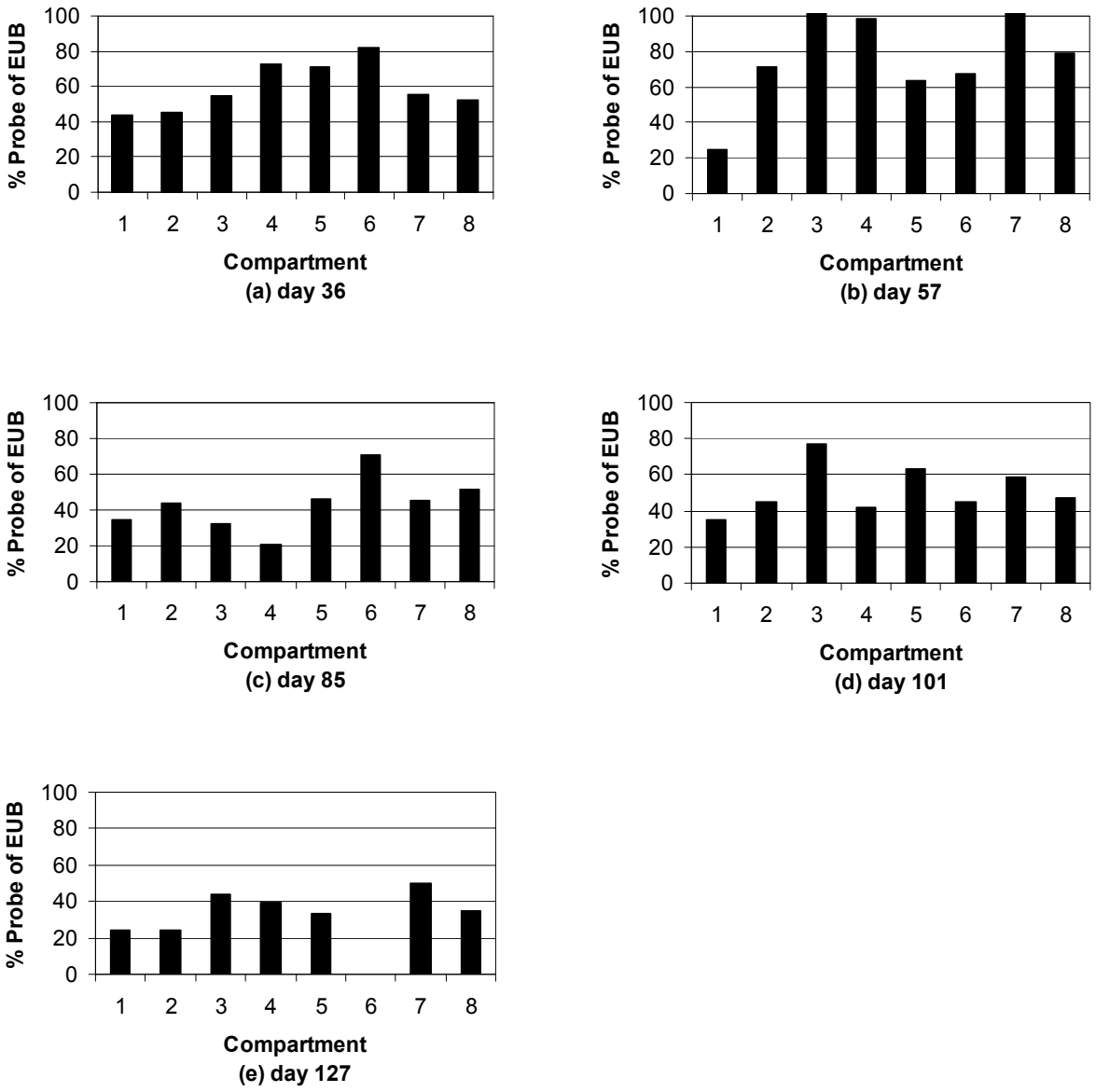


Figure 5.6: Group-specific probe counts as a fraction of EUB338 probe counts in each compartment for samples obtained on day 36, 57, 85, 101 and 127 respectively during the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb. The sample from compartment 6 on day 127 was lost.

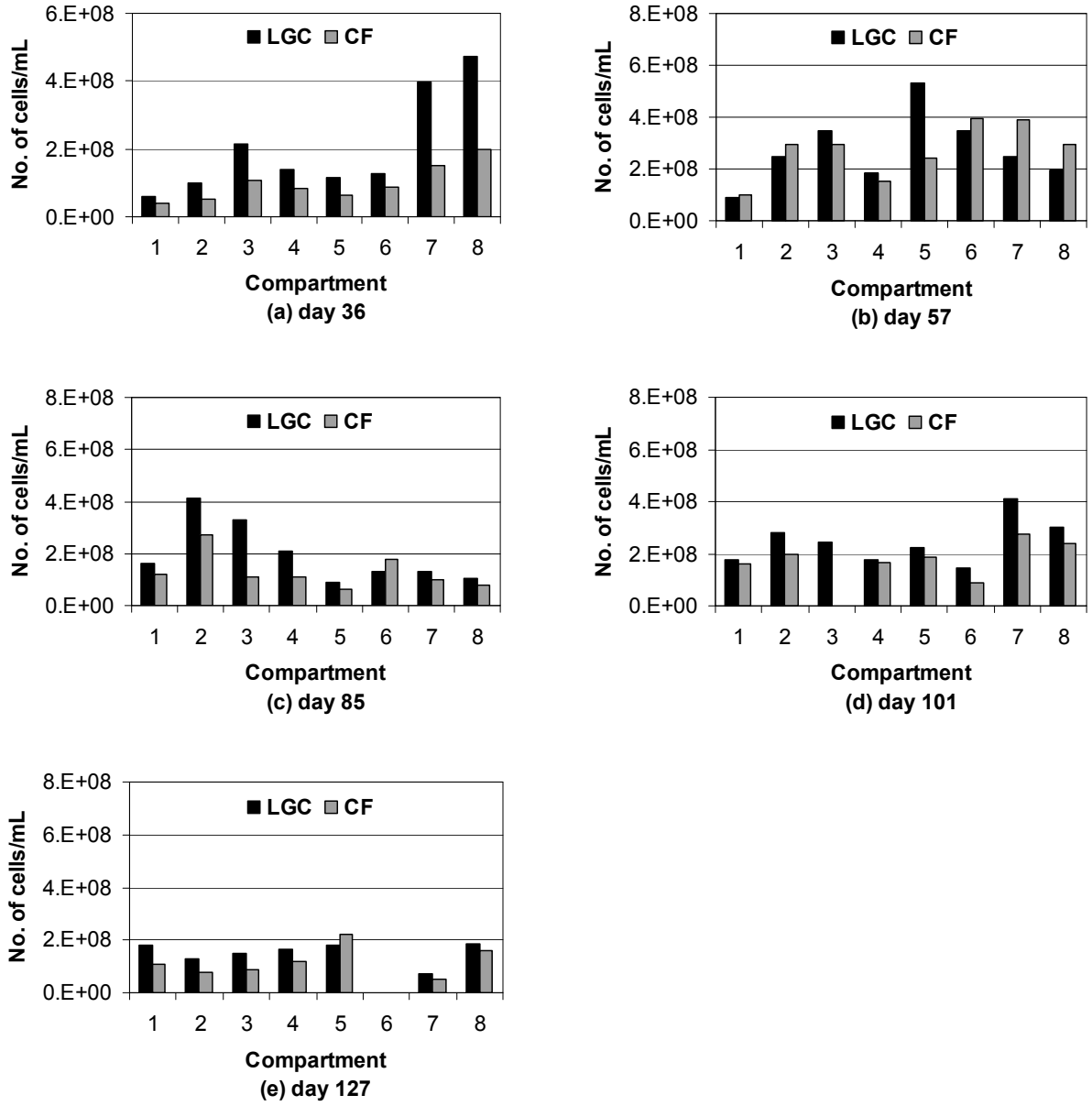


Figure 5.7: Hydrolytic Bacteria. Group-specific probes for Low G+C Gram positive bacteria (LGC, detected by probe LGC354a) and *Cytophaga-Firmicutes* (probe CF319a) in each compartment for samples obtained on day 36, 57, 85, 101 and 127 respectively during the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb. The sample from compartment 6 on day 127 was lost.

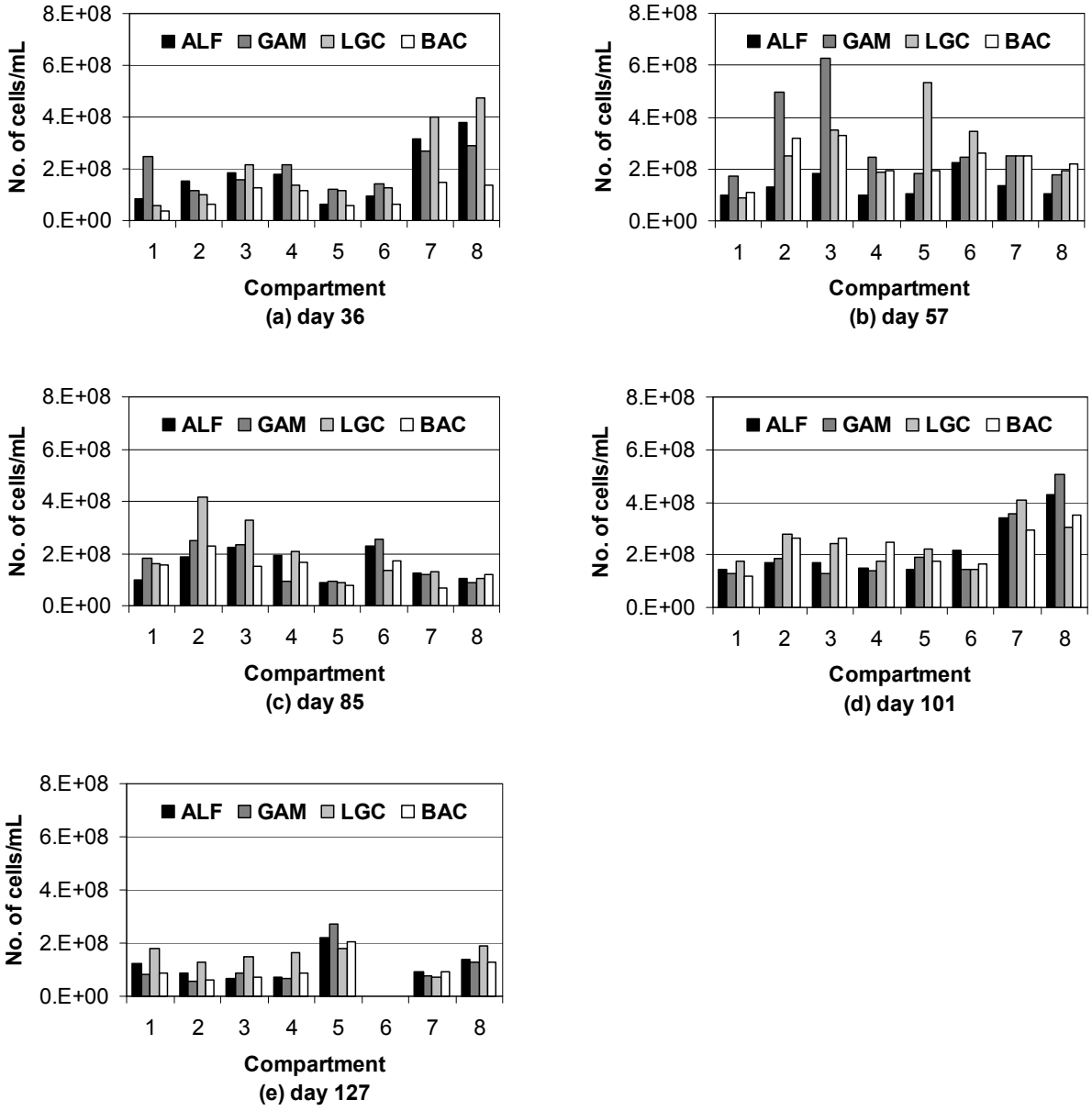


Figure 5.8: Acidogenic bacteria. Group-specific probes for α and γ subclasses of *Proteobacteria* (detected by ALF1a and GAM1b probes respectively), Low G+C Gram Positives (detected by LGC354a probe) and *Bacteriodes* (BAC303 probe) classes in each compartment for samples obtained on day 36, 57, 85, 101 and 127 respectively during the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb. The sample from compartment 6 on day 127 was lost.

5.1.7.5 Acetogenic bacteria

(Section 2.2.1.4) Acetogenic bacteria are found in the classes High G+C Gram positive bacteria (detected by HGC69a probe) and δ-*Proteobacteria*, (sulphate-reducing bacteria, detected by SRB385 probe). Obligate hydrogen-producing acetogens degrade

propionate, long chain VFAs and aromatic compounds to acetate, CO_2 and H_2. A minor group of hydrogen-consuming acetogens reduce CO_2, CO and methoxyl-groups of aromatic compounds to acetate and sometimes butyrate (Guiot et al., 1992).

Figure 5.9 shows number of cells detected by HGC69a and SRB385 probes in each compartment on each sampling day. Similar numbers of cells were detected by these two probes, with HGC dominating on day 36 and SRB dominating on day 127. The shapes of profiles of cell numbers versus compartment number for acidogenic (Figure 5.8) and acetogenic bacteria (Figure 5.9) were similar for each sampling day.

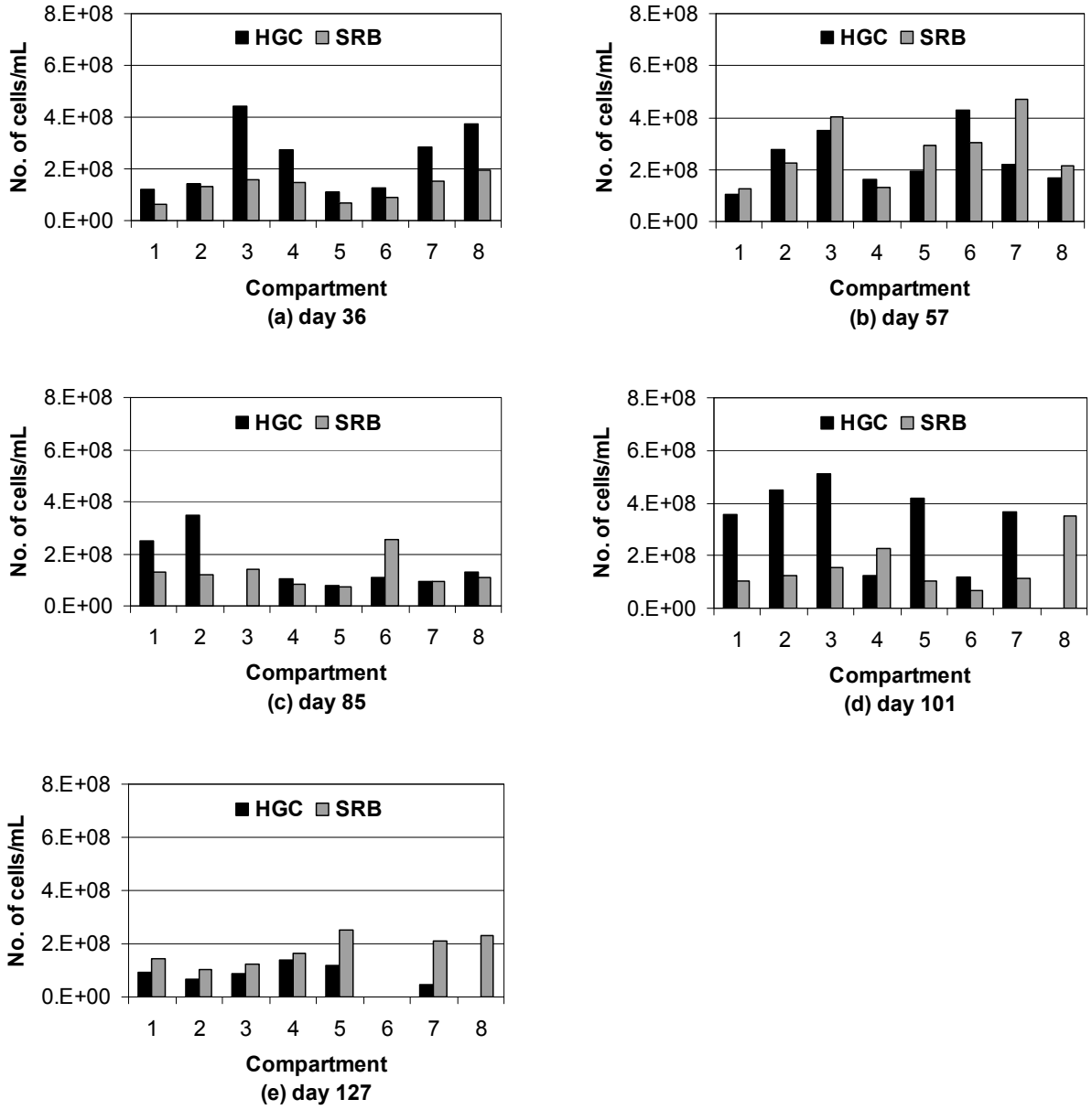


Figure 5.9: Acetogenic bacteria. Group-specific probes for high G+C Gram positive bacteria (detected by HGC69a probe) and δ-*Proteobacteria*

(SRB385 probe) classes in each compartment for samples obtained on day 36, 57, 85, 101 and 127 respectively during the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb. The sample from compartment 6 on day 127 was lost.

Sulphate-reducing bacteria (Section 2.2.1.8) grow syntrophically on lactate, ethanol, propionate and fumarate. Syntrophy with formate-utilising methanogens allows sulphate-reducing bacteria to metabolise without reducing sulphate. In this way, the sulphate-reducing bacteria can grow as proton-inducing acetogenic bacteria. This phenomenon is characteristic of *Desulfovibrio* and *Desulfobacterium* (Raskin et al, 1994a and Raskin et al., 1995). The SRB385 probe is reported to be phylogenetically inconsistent (Santegoeds et al., 1998). Therefore *Desulfovibrio* and *Desulfobacterium* genera were also specifically probed (using probes DSV698 and DSB985 respectively) in order that the dynamics of these micro-organisms could be better elucidated.

Figure 5.10 shows number of cells detected by DSV698 and DSB985 probes in each compartment on each sampling day. Numbers of cells detected by these two probes were virtually identical for each sample. For all of the sampling periods, the sum of DSV698 and DSB985 counts were often higher than total SRB numbers. No specific explanation for this is available, although it is noted that the two measurements are always within the same order of magnitude.

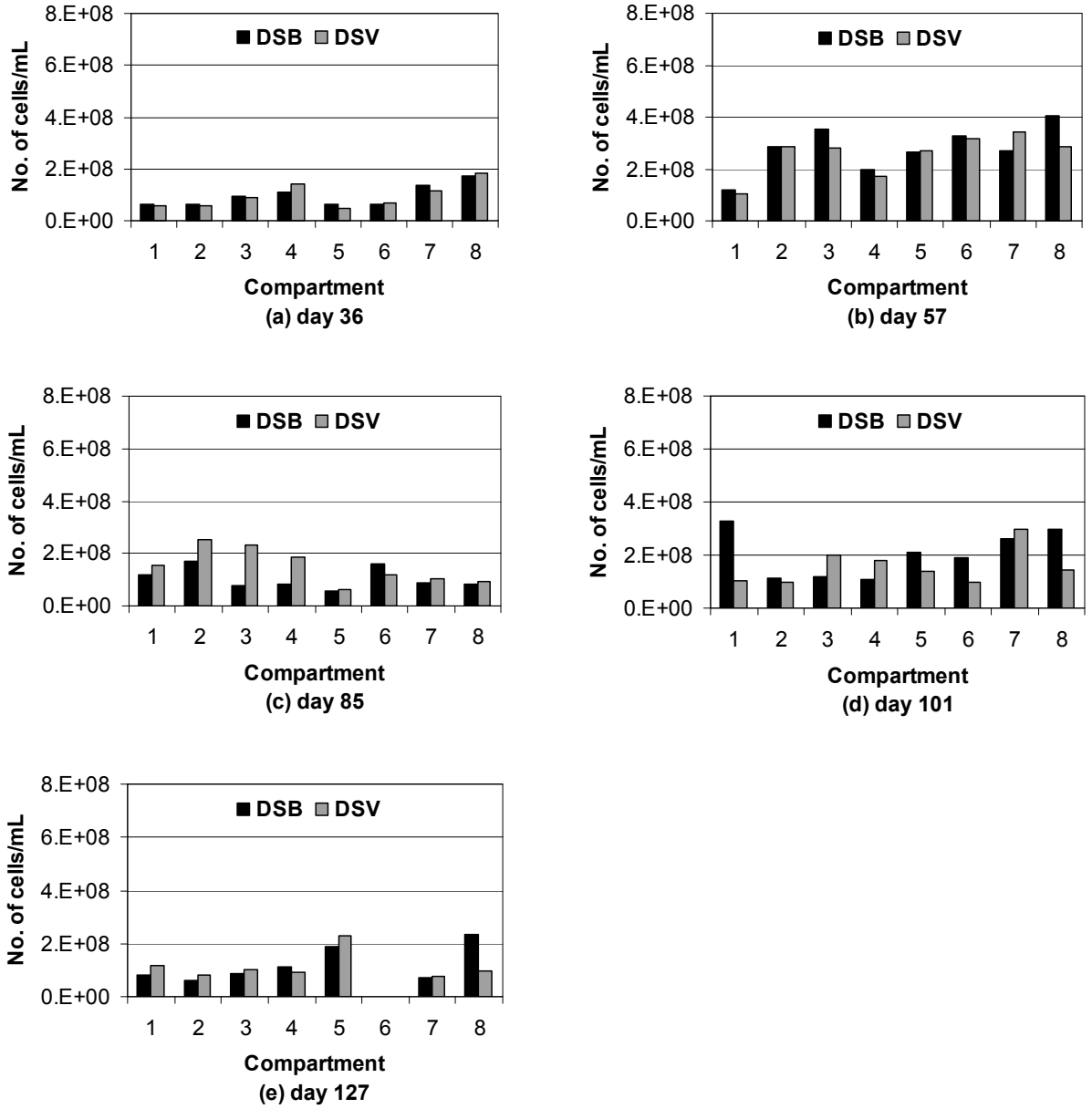


Figure 5.10: Sulphate-reducing bacteria. Genus-specific probes enumerating *Desulfovibrio* (detected by DSV698 probe) and *Desulfobacterium* (DSB985 probe) genera in each compartment for samples obtained on day 36, 57, 85, 101 and 127 respectively during the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb. The sample from compartment 6 on day 127 was lost.

5.1.7.6 Archaea

The most abundant group of *Archaea* involved in anaerobic digestion are methanogens. These are strict anaerobes that form methane as their major metabolic end product from CO_2, H_2, formate, methanol and acetate (Holland *et al*., 1987). Methanogens construct three different types of cell walls and generally differ from each other in shape,

16S rRNA sequence and other features (Prescott *et al*., 1999). They auto-fluoresce a greenish-blue when viewed under a fluorescent microscope.

Of the many methanogenic genera, only two are known to perform acetoclastic reactions, that is, the formation of methane from acetate (Rocheleau *et al*., 1999). These are the genus *Methanosarcina* and the genus *Methanosaeta*. Species belonging to the former genus have a higher maximum growth rate than the species belonging to the latter genus. *Methanosaeta* species, however, have a lower threshold for acetate than *Methanosarcina* species. Therefore *Methanosaeta* species will generally proliferate at low acetate conditions (Raskin *et al*., 1994a).

Along with inter-genus competition, methanogens have to compete with sulphate-reducing bacteria for common substrates including hydrogen and acetate. Both these groups catalyse the terminal stage of anaerobic digestion and are dependent on other micro-organisms to convert the complex organics to simpler compounds. In the presence of non-limiting levels of sulphate, sulphate-reducing bacteria generally out-compete methanogens. However, in the presence of low sulphate concentrations, methanogens are dominant (Raskin et al., 1996). In the absence of sulphate, certain sulphate-reducing bacteria e.g. *Desulfovibrio* spp. have been observed to grow together with methanogens, converting ethanol or lactate to acetate (Wu et al., 1991).

Archaea were detected in samples using domain-specific probe ARC915. Cell numbers hybridised by this probe are presented in Figure 5.11. These data accounted for an average of 4% of DAPI stained cells. This result is lower than was expected and could have a number of causes:

- Archaeal micro-organisms were not adequately detected due to low rates of hybridisation as a result of low cell wall permeability or some other limitation of the method.

- Archaeal micro-organisms were not detected due to low cellular rRNA content due to low activity as a result of low operating substrate concentrations.

- Very few archaeal micro-organisms were present in the reactor.

As the pilot ABR operating on medium strength wastewater (ca. 700 mgCOD/ℓ) generally experienced low concentrations of soluble organic material, especially methanogenesis precursors, it is probable that fairly low numbers of acetoclastic methanogens were present, and that these cells exhibited low activities in samples that were analysed.

The two acetoclastic methanogenic genera, *Methanosarcina* and *Methanosaeta* were individually probed using genus-specific probes MS821 and MX825. Data are presented in Figure 5.12.

No *Methanosaeta* (probe MX825) were detected in any samples, while *Methanosarcina* were detected in compartment 1 on all sampling days; compartments 2, 3 and 4 on day 36 and 57; and compartments 5 and 6 on day 57. When detected, *Methanosarcina* accounted for between 24% and 100% of ARC915. Clearly Archaea other than acetoclastic methanogens were present in the reactor. This was confirmed by DNA

sequencing, which detected methanogens other than *Methanosarcina*, and also did not detect *Methanosaeta.*

Theoretically, *Methanosaeta* should survive more easily than *Methanosarcina* as they exhibit higher growth rates at low substrate concentration. However, as is implied from the results of all the FISH work presented here, the concentration of micro-organisms is dominated by the flow conditions, particularly sludge washout and solids retention, and it is possible that the morphology of *Methanosaeta* (filamentous) renders them more susceptible to sludge washout. Further, samples were obtained from the middle of each compartment (Appendix 2) and therefore may not have been representative of micro-organism communities at the walls of the compartment. It is possible that filamentous *Methanosaeta* were retained by attachment to the walls, but were washed out of the bulk phase, and therefore were not found in the samples taken from the centre of the compartment.

No *Methanosarcina* were counted in the later compartments of the ABR. This is in direct contradiction to the original theory of operation that assumed that phase separation of micro-organisms would occur with acetoclastic methanogens dominating in later compartments. This result has been found in other studies; Langenhoff and Stuckey (2000) similarly found that no separation of phases occurred in an 8 compartment ABR treating low strength synthetic wastewater, as did Hassouna and Stuckey (unpublished).

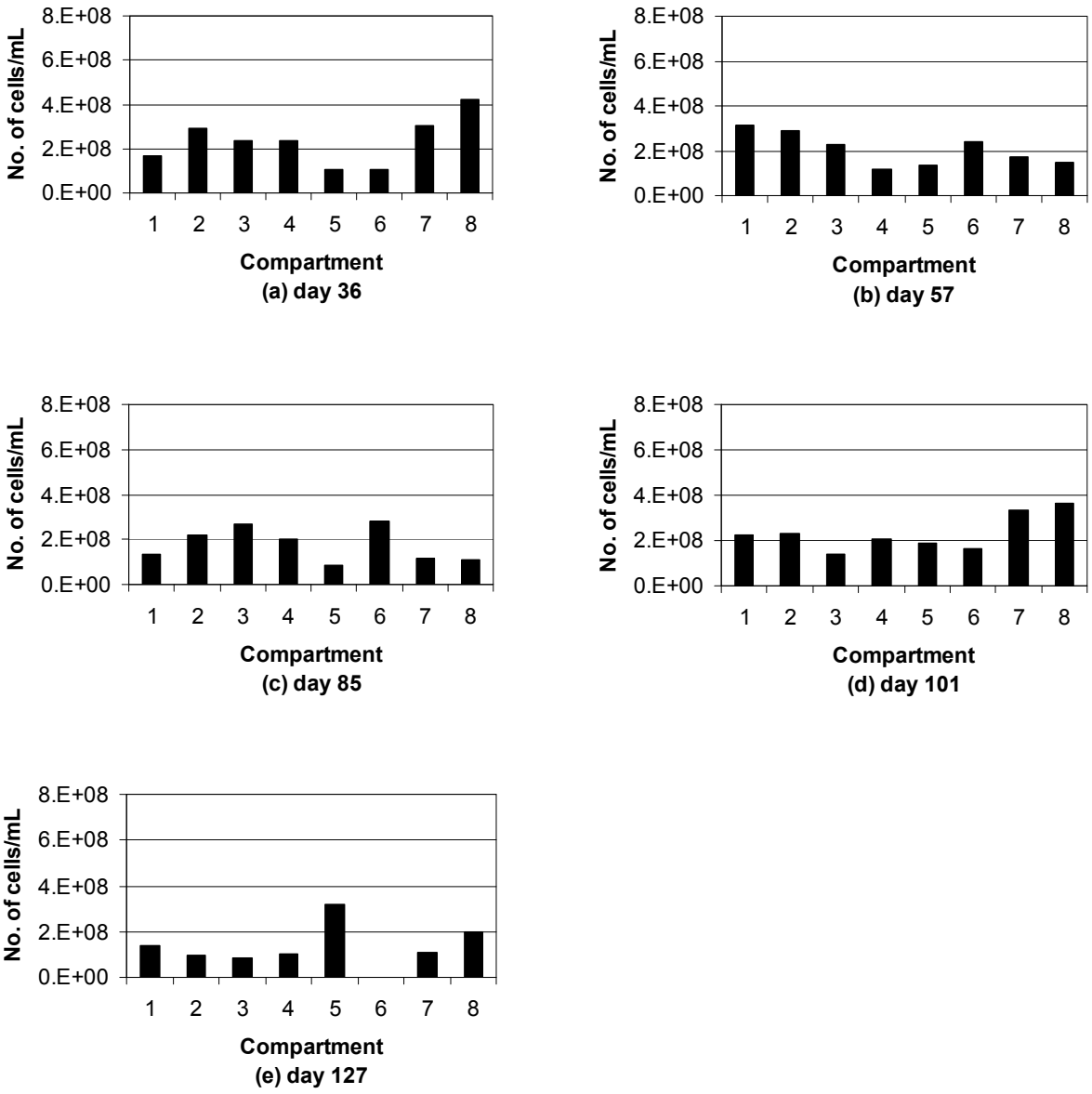


Figure 5.11: Domain-specific probe enumerating Archaea (detected by ARC915 probe) in each compartment for samples obtained on day 36, 57, 85, 101 and 127 respectively during the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb. The sample from compartment 6 on day 127 was lost.

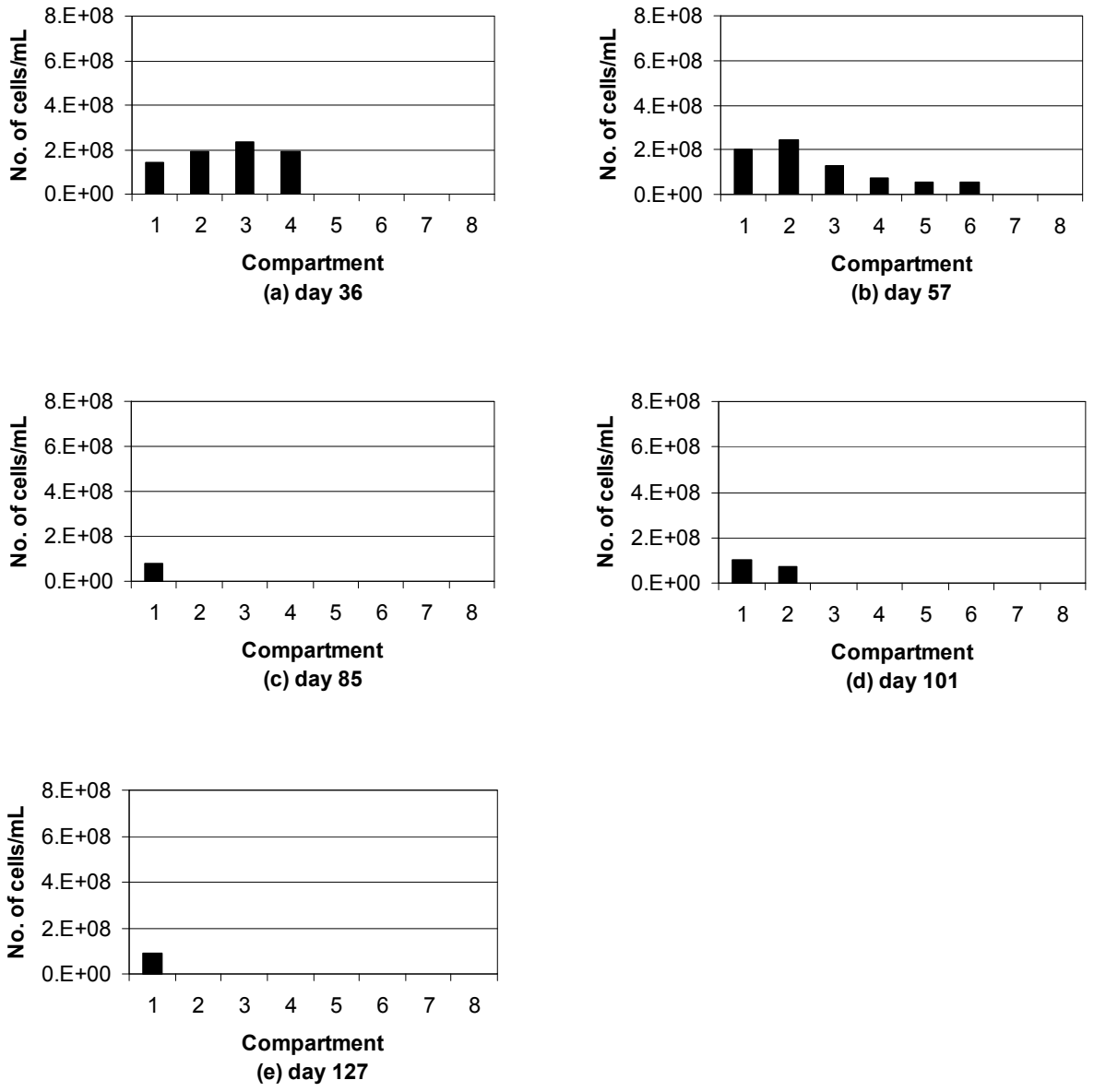


Figure 5.12: Genus-specific probe enumerating *Methanosarcina* (detected by MS821 probe) in each compartment for samples obtained on day 36, 57 , 85 and 127 respectively during the 2003 operating period (22 h HRT) at Kingsburgh WWTP, treating wastewater from a middle-income suburb. No *Methanosaeta* (probe MX825) were detected in any samples. The sample from compartment 6 on day 127 was lost.

5.1.8 DNA sequencing of samples from compartments of the pilot ABR

DNA sequencing of samples obtained from the compartments of the pilot ABR was performed. The details of the techniques used may be found in Lalbahadur (2004).

DNA sequencing of bacteria revealed the presence of LGC, HGC and δ-*Proteobacterial* classes of bacteria. DNA sequencing of archaea confirmed the presence of *Methanosarcina, Methanobacterium* spp. and *Methaococcus* spp. These results are qualitative, and give no indication of the number of the identified micro-organisms. However, they conclusively confirm the presence of the identified micro-organisms: the fact that *Methanosarcina* spp. were observed, but not *Methanosaeta* reaffirms that findings of the FISH study. Similarly, the identification of other methanogenic species is consistent with the conclusion that Archaea that belong to genera not specifically probed were present in the samples.

5.1.9 Summary of microbial community characterisation study

Interpretation of FISH data is limited by the lack of chemical data such as volatile fatty acid concentrations. However, considering the variable nature of the feed flow rate and composition experienced by the pilot ABR, it is expected that a general idea of steady-state component concentrations and the variability of these numbers under normal operation would provide better confirmation to the microbiological work than sampling-day-specific data since the populations would depend on the overall operating conditions more than the variations thereof. Operational events such as washout, no flow etc. are likely to have more significance in the interpretation of FISH results than exact chemical data.

The data presented in Sections 5.1.7.1 to 5.1.7.6 above indicate that the full range of micro-organisms that effect anaerobic digestion were present in the pilot ABR. It appeared that the variation in numbers of micro-organisms in each category was dominated by the variation in total solids and total cell counts, rather than some function of growth conditions. If this interpretation is correct, then it appears that the microbial community dynamics in an ABR treating domestic wastewater depend more on flow dynamics than on growth conditions. In other words, the composition of sludge (in terms of micro-organism functionality) in each compartment was similar, but the amount of sludge will depended on the specific settling properties of the sludge, and the history of hydraulic flow conditions in that compartment.

No spatial separation of micro-organisms with specific functionality (e.g. hydrolytic, acidogenic, acetogenic, methanogenic) was observed in the samples studied, although there was no mechanism for investigating the distribution of micro-organisms within a compartment between those in the bulk phase and those attached to the wall.

The presence of hydrolytic and acidogenic bacteria throughout the reactor indicates that hydrolysable material was present in all compartments, i.e. that initial breakdown of particulate and polymeric material were the rate-limiting steps in digestion of complex particulate wastewater. This implies that methanogenic micro-organisms scavenged volatile fatty acids as they are produced by hydrolytic and acidogenic micro-organisms. According to this hypothesis, low volatile fatty acids, and relatively low acetoclastic methanogenic populations may be expected, as appeared to be the case.

Surprisingly low numbers of Archaea, particularly acetoclastic methanogens, were obtained by FISH, although good COD removal was observed in operation. This may be attributable to low substrate concentrations resulting in low cellular rRNA concentrations

and therefore low counts, or association of acetoclastic methanogens with compartment walls. It is also noted that approximately half of the cells identified by DAPI staining were not counted by FISH techniques. It is possible that some of the difference between these two measurements could be accounted for by undetected Archaea. However, as there is no experimental indication as to what caused these anomalies, no certain explanation for low methanogen counts can be provided.

5.2 SCANNING ELECTRON MICROSCOPY STUDY OF MICROBIAL COMMUNITIES

This work was undertaken as part of a MSc research project in the School of Conservation and Biological Sciences at the University of KwaZulu-Natal in 2004. The aim of this investigation was to gain a deeper insight into the methanogenic population dynamics within compartments of the pilot ABR treating domestic sewage, since this aspect of microbial population dynamics was not fully understood from the previous study. Scanning electron microscopy (SEM) was used to measure the relative abundance of methanogenic bacteria. Although this technique does not allow direct identification and enumeration of methanogenic bacteria, certain methanogenic bacteria have distinct morphologies and can be tentatively identified.

The samples used in the SEM study were obtained from the pilot ABR operating at Kingsburgh WWTP on a feed of domestic wastewater from middle-income suburbs during 2004. During this period, the ABR operated at a hydraulic retention time of between 40 h and 44 h, approximately half the flow rate used in the 2003 operating period, during which the FISH study was undertaken. Therefore, it is expected that the microbial communities in the two studies would not necessarily be the same. However, since different techniques were used in each study, comparisons drawn must necessarily by qualitative.

5.2.1 Distribution of microbial communities within the ABR

Various bacterial morphologies were observed within the ABR, each having different distribution between the compartments. Table 5.3 lists micro-organisms that were observed in each compartment where the class or family may be reasonably identified from the cell morphology. Compartment 1 had the greatest variety of micro-organisms, predominated by clusters of rod-shaped bacteria and cocci of varying sizes and shapes (Figure 5.13). The prominence of these micro-organisms within compartment 1 implies that that they may be hydrolysing or acid-producing bacteria since the dominant processes expected in compartment 1 are hydrolysis and acidogenesis. Because these organisms could not be identified with any degree of confidence, they have been omitted from Table 5.3. Also excluded from Table 5.3, was a small cluster of two to four spherical cells closely resembling *Methanosarcina*-like organisms, which was only observed in one micrograph. Other morphotypes observed in compartment 1 included:

- Rods, which closely resemble *Syntrophomonas*-like species, but could also be a slender relative of *Desulfovibrios* or *Desulfomonas* species (Figure 5.14) (Harper and Pohland, 1997);

- Long, chain-forming rods resembling, either *Methanobacterium*, *Methanobrevibacter* or *Methanomicrobium* (LR in Figure 5.13);

- Long filamentous or chain-forming organisms consisting of 10 or more cells resembling the typical morphology of *Methanospirillum* species (FC in Figure 5.13);
- Large cocci (approximately 2-3 µm in diameter) and smaller cocci (approximately 0.5 to 1 µm in diameter) resembling *Methanococcus* or *Methanocorpusculum* species (SC and LC in Figure 5.13).

Table 5.3: Description and distribution of the most prevalent morphologies found in compartments of the ABR

Shape	**Size**	**Possible micro-organism**	**Possible primary substrates**	**Compartment location**
Long rods	2 µm	*Methanobacterium*	$H_2 + CO_2$, formate	1
		Methanobrevibacter	$H_2 + CO_2$, formate	
		Methanomicrobium	$H_2 + CO_2$, formate	
Long rods with slight curvature	1 to 2 µm	*Syntrophomonas*	Butyrate	1
Small cocci	0.5 to 1 µm	*Methanococcus*	$H_2 + CO_2$, formate	1, 2, 3, 5
Large cocci	2 µm	*Methanococcus*	$H_2 + CO_2$, formate	1 – 8
		Methanocorpusculum	$H_2 + CO_2$, formate	
Small chain-forming/filaments	1 µm each	*Methanospirillum*	$H_2 + CO_2$, formate	1 – 7
Bamboo-shaped rods and clusters	2 µm each	*Methanosaeta*	Acetate	2 – 8

The relative abundance of each of these organisms varied among compartments. For *Syntrophomonas*-like organisms, the number of observations made for this morphotype decreased markedly from compartment 1 onwards. A similar pattern was observed with respect to *Methanospirillum*-like organisms, with the number of observations decreasing from compartment 4 onwards. In general, the *concentration* of organisms observed in

samples from the different compartments appeared to decrease between the earlier and later compartments.

Two types of cocci were observed within the reactor: large cocci, with a diameter of between 2 and 3 µm; and smaller cocci, with a diameter between 0.5 - 1µm. Although these were not conclusively identified, these cocci resemble species of the *Methanococcus* genus. Variations in the relative abundance for the two types of cocci were evident. The number of observations of smaller cocci decreased from compartment 1 onwards, whereas for larger cocci, increased observations were made from compartment 5.

The sixth morphotype observed were bamboo-shaped rods, a typical characteristic of the acetoclastic methanogen, *Methanosaeta* (Figure 5.15). This morphotype was most prevalent in compartments 2 and 3, with decreased observations in the latter compartments. In compartments 2 and 3, *Methanosaeta*-like organisms were present almost exclusively as bacterial aggregates or granules. Within the granules, a complex matrix was observed with large numbers of *Methanosaeta*-like organisms embedded in extracellular polymers (Figure 5.16).

Observations from these micrographs suggest that phase separation was limited within the ABR, with the first three steps of anaerobic digestion confined mostly to compartment 1. Acidogenic-like bacteria were dominant in the first compartment, and thereafter, were not seen in such great abundance. The bacterial morphotype resembling a *Syntrophomonas* species, is likely to degrade butyrate into acetate and hydrogen. Hydrogen concentration would have been highest in the earlier compartments, and this probably explains the relatively high proportion of hydrogen-scavenging methanogens observed within these compartments (Boopathy and Tilche, 1992). Several authors (McInerney et al., 1979; Boone and Bryant, 1980; MacLeod et al., 1990) have noted the importance of such syntrophic associations as they maintain a low hydrogen partial pressure that is necessary for acetogenic substrate utilisation.

Acetate concentrations are typically highest in compartment 1, where *Methanosarcina*-like organisms were observed, although not in substantial numbers. SEM micrographs show the scavenging acetoclastic methanogen, *Methanosaeta*, predominating in the compartments thereafter. This distribution is in keeping with the expectation that *Methanosarcina* dominate at high substrate concentrations, while *Methanosaeta* grow more rapidly at lower acetate concentrations.

Identification of *Methanosaeta* in the SEM study challenges the conclusion of the FISH study that these Archaea were not present in the pilot ABR in the 2003 operating period. If *Methanosaeta* were present in the FISH samples, but bound in a polymeric phase, it is conceivable that they would not have been adequately detected by FISH. However, since operating conditions were different for the two studies, interpretation of one set of results based on the other should be practised with caution.

5.2.2 Granulation

Over the course of the pilot study, the physical appearance of the anaerobic sludge gradually changed, with the development of small granules being observed. With time, the size of these granules increased, particularly in the earlier compartments.

In SEM micrographs, granulation was observed in compartments in 2 and 3, and to a lesser extent, compartment 4. Such observations are contrary to expectation since significant granulation is not expected in low strength anaerobic treatment because soluble substrate concentrations are not high enough to exert sufficient concentration gradients to drive diffusion into and out of the granules. Advantages of granulation include high sludge density resulting in good sludge settling characteristics and enhanced substrate utilisation kinetics as a result of proximity of micro-organisms responsible for consecutive steps in the digestion process (Hulshoff Pol et al., 1986).

In SEM micrographs of granules observed in the pilot ABR, granule shape varied from oval to bowl-shaped with sizes varying from approximately 600 µm to 2 000 µm (Figure 5.17). On the surface of larger, well-developed oval granules, gas cavities 10 to 20 µm could be seen (Figure 5.18).

Granules had a two-layered structure, with a thin outer layer (2 µm thick) and central core. Close examination of the granule surface revealed the existence of numerous large and small cocci, resembling *Methanococcus*-like organisms, embedded in inert material (Figure 5.19). Also present to a lesser extent was the bacterial morphotype resembling *Methanospirillum*-like organisms and acidogenic-like bacteria. Bamboo-shaped rods, resembling *Methanosaeta*-like organisms, protrude from the inner layer to the surface of granules (Figure 5.19). Similar observations have been made from an up-flow sludge bed and filter reactor treating sucrose waste (MacLeod et al., 1990).

While the bacterial composition of the granules was mixed, the interior was predominated by large clusters of *Methanosaeta*-like organisms embedded in extracellular polymers. It is thought the extracellular polymers strengthen loosely adhered aggregates to form tight granules (Ross, 1984; Shen et al., 1993). The results suggest that *Methanosaeta* acts as a key structural element in the development of granules. Similar observations have been made by other researchers (MacLeod et al., 1990; Morgan et al., 1991; Banik et al., 1997). Furthermore, the arrangement of this bacterial group led to the formation of numerous gas cavities. It is thought these cavities are the site of vigorous gas production (Bochem et al., 1982), and are formed by the inactivation and autolysis of acidogenic bacteria through substrate diffusion limitation as the granule develops (Guiot et al., 1992). This idea is supported by observations of acidogenic bacterial morphotypes around clusters of *Methanosaeta*-like organisms, some of which, formed cavities within the granule (Figure 5.20). Contrary to MacLeod et al. (1990), these observations support the hypothesis that granule development occurs through a multinucleate approach.

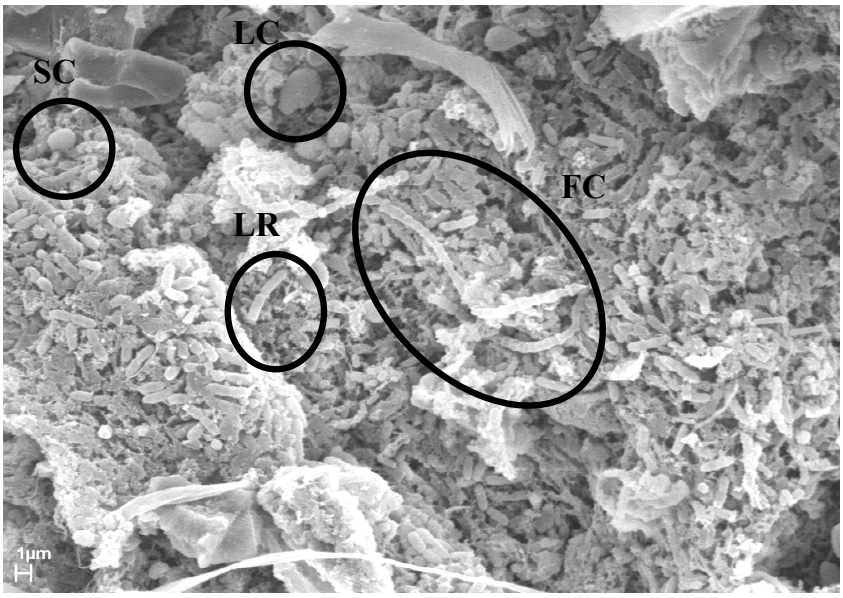


Figure 5.13: SEM micrograph of compartment 1, showing the wide diversity of bacteria found within this compartment. Various sizes of cocci (SC- small cocci, LC – large cocci), long rods (LR) and filamentous/chain-forming (FC) micro-organisms were found in close association with a predominate population of rods of varying size and curvature (See Table 5.3 for possible bacterium).

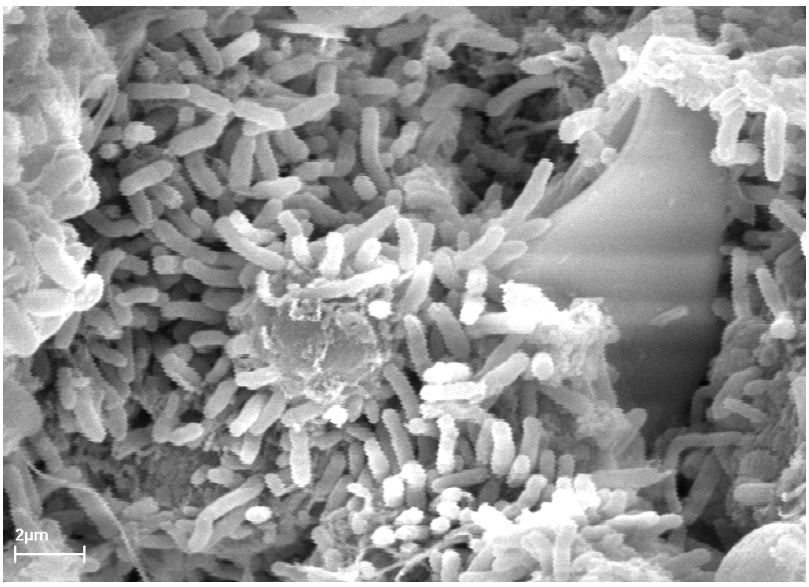


Figure 5.14: SEM micrograph of slender rod-shaped bacteria with gentle curves that closely resembles *Syntrophomona* species or a relative of the *Desulfovibrios* or *Desulfomonas* species

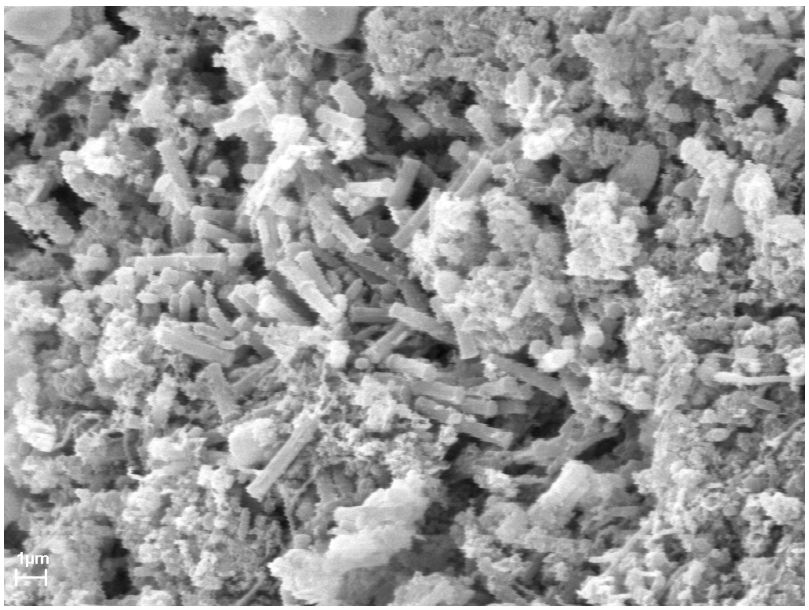


Figure 5.15: Bamboo-shaped bacteria that closely resemble the acetoclastic methanogen, *Methanosaeta*. This type of micro-organism was most prevalent in compartments 2 to 4, especially within bacterial aggregates or granules. Observations decreased in later compartments.

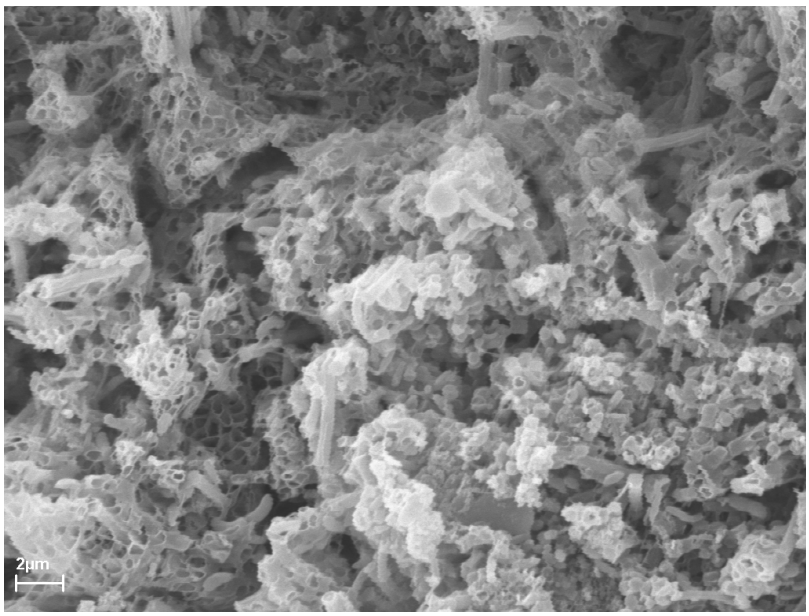


Figure 5.16: Complex network of *Methanosaeta*-like organisms embedded in extracellular polymer.

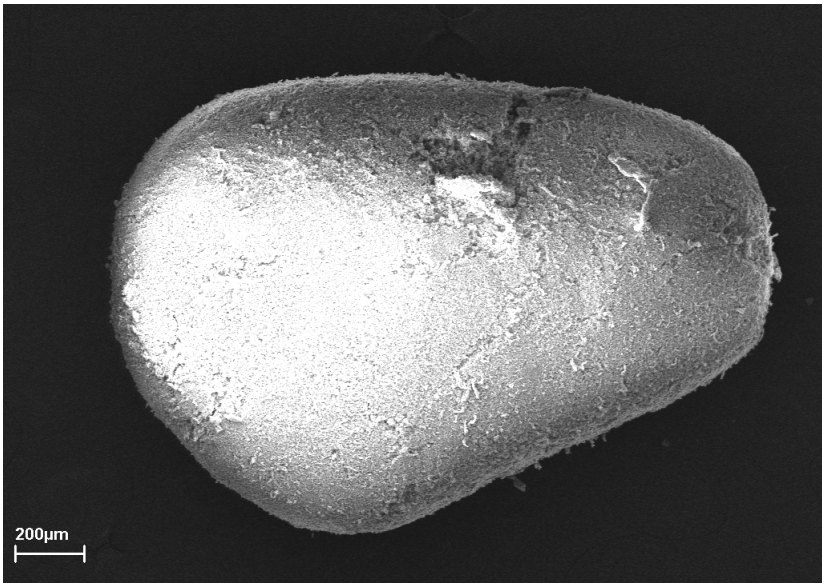


Figure 5.17: SEM micrograph of the surface topography of an entire granule from compartment 2.

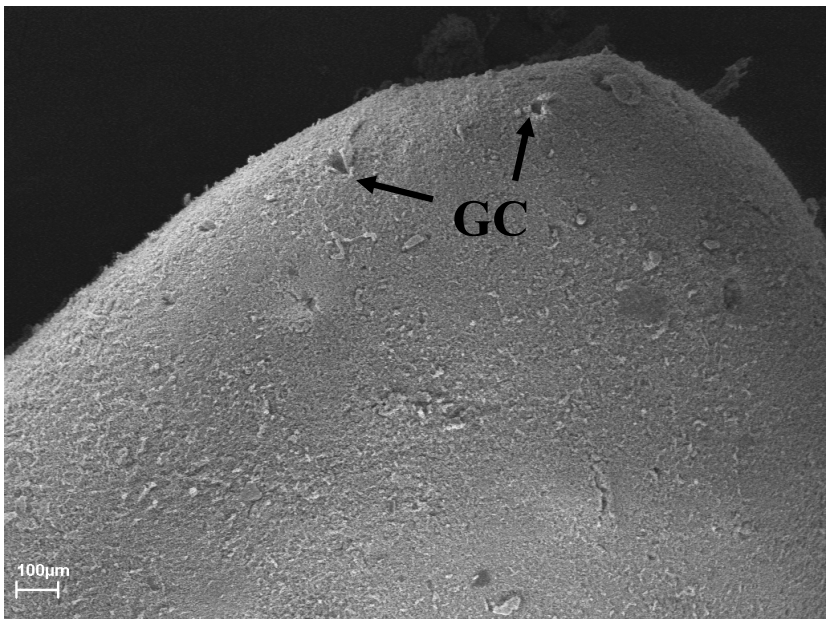


Figure 5.18: Surface of granule from compartment 3 showing the numerous gas cavities (GC) that cover the surface of the granule

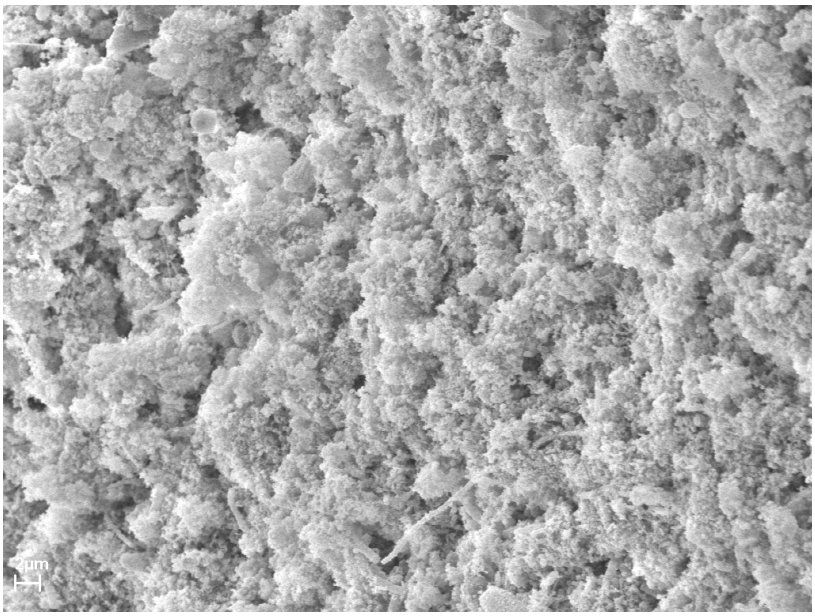


Figure 5.19: SEM micrograph of granule surface showing a wide diversity of bacterial morphotypes.

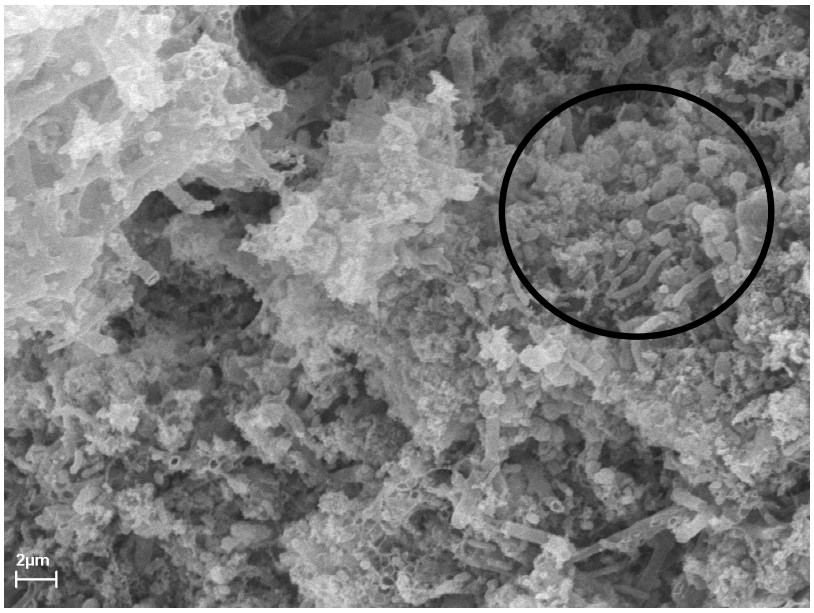


Figure 5.20: High magnification of the granule interior, revealing large clumps of cavity forming, *Methanosaeta*-like bacteria surrounded by acidogenic-like bacteria (circled area).

5.2.3 Summary of SEM study

Although conclusive identification of microbial species is not possible using SEM, this technique has the advantage of providing a visual indication of the relative abundance of different micro-organism types, using a tentative identification based on external morphological characteristics. There is therefore a risk of incorrectly identifying micro-organism types. However, provided the limitations of the technique are understood, it provides qualitative understanding of microbial population dynamics that cannot be gleaned from quantitative FISH data.

SEM micrographs of samples obtained during the 2004 (44 h hydraulic retention time) operating period indicate that

- there may be significant amounts of methanogenic micro-organisms throughout the ABR although *Methanosarcina*-like micro-organisms were only observed in compartment 1 while *Methanosaeta*-like micro-organisms were found through-out the rest of the reactor; and

- granulation occurs, forming complex consortia of acidogenic and methanogenic micro-organisms bound by extracellular polymeric substances.

Both of these findings imply that in the 2004 operating period, the pilot ABR had a well-established anaerobic sludge that exhibited a degree of adaptation to compartment conditions.

5.3 CONCLUSIONS FROM THE MICROBIAL COMMUNITY CHARACTERISATION STUDIES

Both the FISH / DNA sequencing study and the SEM study demonstrated that a diverse community of micro-organisms exist in the pilot ABR treating domestic wastewater. The FISH / DNA sequencing study positively identified and enumerated specific micro-organism types, while the SEM study provided insight into the mechanisms of anaerobic digestion and granule formation.

The two studies present conflicting evidence on the presence of acetoclastic methanogens, particularly those in the genus *Methanosaeta,* in the pilot ABR; The FISH study probed this genus but did not detect any, while micro-organisms with morphologies similar to *Methanosaeta* were observed in abundance in the SEM study. It is hypothesised that the binding of *Methanosaeta* in granules may have resulted in poor permeability to oligonucleotide probes in the FISH study, resulting in undetectable hybridisation. This hypothesis is supported by the fact that significant populations of acetoclastic methanogens would be required to achieve the COD removal obtained by treatment of domestic wastewater by the pilot ABR in these operating periods since acetoclastic methanogenesis is responsible for most of the conversion of COD to CH_4 gas in anaerobic digestion.

The FISH study concluded that little differentiation in population characteristics occurred among compartments. The SEM study was not able to quantify micro-organisms of different classes, but noted that *Methanosarcina*-like species were observed in the first compartment, but not in later compartments. From these observations, it would appear that phase separation, as originally expected, did not occur; i.e. spatial separation of hydrolysis, acidogenesis, acetogenesis and methanogenesis into different compartments was not seen in the ABR treating a relatively low strength (in terms of conventional anaerobic digestion applications), particulate wastewater.

The ratio of hydrolytic, acidogenic, and acetogenic micro-organisms to each other appeared to be relatively constant throughout the ABR, but a change in concentration and dominant genus of acetoclastic methanogens was observed, particularly between the first and subsequent compartments. It is hypothesised that hydrolysis was the overall rate-limiting step in treatment of domestic wastewater: hydrolysable material in the ABR feed was carried through the reactor, undergoing continuous hydrolysis from the surface of the waste material inwards. The exception to this theory is compartment 1 where acid production caused by readily hydrolysable material in the influent resulted in a decrease in pH value, that subsequently inhibited methanogenesis. Here, higher concentrations of soluble intermediates could be expected.

This hypothesis is borne out by the results of the chemical analyses reported in Chapter 4.

6 COMMUNITY WATER USE AND WASTEWATER GENERATION STUDY

This study formed the basis of an MSc research project in the School of Conservation and Biological Sciences, University of KwaZulu-Natal.

The literature contains much information on the characteristics of wastewater from middle-income suburbs. These data are used in the design of conventional wastewater treatment facilities. However, wastewater from poor and rural communities has not been thoroughly studied, and there is little information that can be used for predicting wastewater quality from low-income, rural or peri-urban communities.

It is expected that wastewater contaminant concentrations will be higher in low-income communities than in middle-income suburbs since the former often have limited access to water or are unable to pay for water in excess of the 200 ℓ/d that is supplied free by many municipalities.

The characteristics of wastewater generated by households in a community vary considerably depending on the type of house, number of occupants, age of occupants, plumbing fixtures and appliances used and bathing preferences. Other factors that have been found to be associated with wastewater characteristics include socio-economic status of the community, mode of water supply and geographical location of the community.

A study was undertaken to quantify water use and wastewater generation in a low-income peri-urban community and to characterise the wastewater in terms of chemical contaminants and pathogen indicator organisms. These data are intended to facilitate model-based predictions of the performance of an ABR or similar on-site or decentralised technology under conditions similar to those encountered in a South African low-income peri-urban community.

Wastewater of domestic origin is categorised as:

- Greywater: wastewater generated chiefly from bathing, washing and kitchen sink, but that does not contain excreted waste or
- Blackwater: wastewater generated primarily from the toilet

Contaminants in the water therefore arise from faeces, urine, soap, fats and grease, food, detergents and other household products

6.1 METHODOLOGY

This study had three components:

- Community water use habits were investigated by means of a household questionnaire. The questionnaire was designed to gather information about water use habits, daily quantity of water used and the daily amount of wastewater generated.

- A water meter data survey was undertaken in which eThekwini Municipality water meter records were studied to identify trends in water consumption in communities using semi-pressure (roof-tank) water delivery systems.

- Samples were obtained from sewers in the area in which the household questionnaire study was conducted. The samples were analysed for various chemical and microbial determinands.

6.2 COMMUNITY WATER USE QUESTIONNAIRE STUDY

In the eThekwini area, there are four levels of water supply. The lowest level of supply is a community stand pipe, which provides unlimited amounts of potable water to any member of a community, although they have to transport it home. The next level of supply, the most basic level of service is a 200 ℓ ground tank provided in the homeowner's yard that is filled with 200 ℓ of free basic water per day. The second level of service is a semi-pressure roof tank. In eThekwini Municipality, there are several designs of roof tank, with some allowing only 200 ℓ of water per day to a household, and others providing an unlimited low pressure supply of water, where the amount in excess of 200 ℓ is paid for by the home owner. The highest level of service is full pressure water supply.

6.2.1 Study area

This study was performed in the KwaMashu-Newlands Interface Housing Development, a low-income peri-urban community 20 km from the Durban CBD. Figure 6.1 shows the location of the region in eThekwini Municipality.

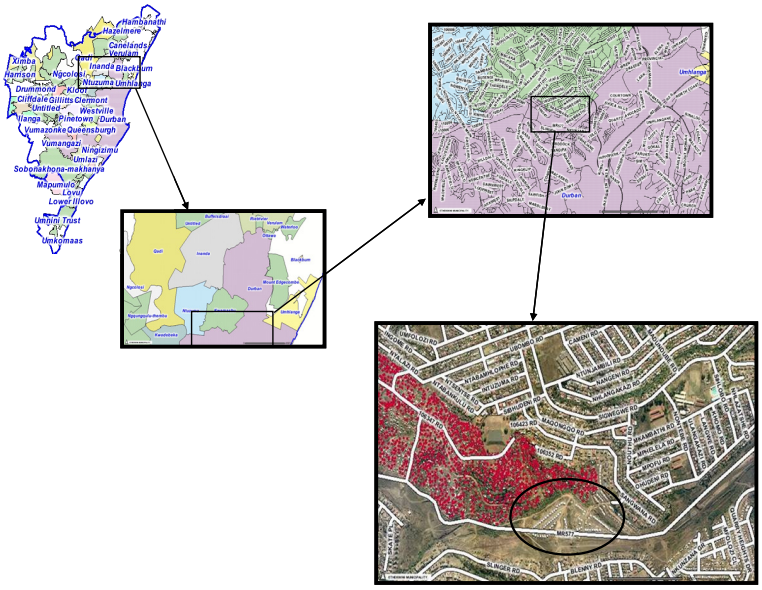


Figure 6.1: Maps showing location of study area, the Newlands-KwaMashu Interface housing development in eThekwini Municipality. The study area is shown by an oval in the bottom map.

The study area is a community that is being slowly transformed from an informal settlement to a subsidised housing estate. The oldest formal houses were less than two years old when the study was undertaken. At the time of the study, individual households did not have water meters and therefore were not billed for water consumption. Thus there were no municipal records of water consumption for the individual households studied. This raised the concern that householders would use more water than similar communities that were being metered, as householders had no financial incentive to reduce their water consumption.

This community had full waterborne sanitation. Each house was equipped with a flush toilet and all household wastewater was collected in a sewer and conveyed to the nearest municipal wastewater treatment facility.

Although the water and sanitation service levels and therefore water use practices of this community may be different to communities that could be serviced by a decentralised wastewater treatment system, it had the advantage of being accessible to the project team, both in terms of distance, and with the assistance, guidance and protection of the developer (Phakamisa) working in the community.

6.2.2 Methodology of questionnaire study

A questionnaire was administered to households within the study area. The questionnaire consisted of several sections each designed to gather information about water use habits, daily water use and daily wastewater generation. The survey was conducted verbally with the aid of a translator. A total of 81 households were interviewed.

Each householder was asked questions relating to the amount of water they *believed* that they used, and how much was used for specific daily functions. Where householders were unable to guess volumes of water used (which was true in most instances) estimates were made in terms of the numbers of 5 ℓ bucketfuls used for a task per day.

6.2.3 Results of questionnaire study

Table 6.1 presents the results of the questionnaire study.

The water use study of the KwaMashu-Newlands housing development showed that in 81 houses, the average number of occupants was 4.1; and the average potable water consumption was 342 ℓ/day per household or 83 ℓ/d per capita (Table 6.1). Of the various water use activities, bathing and personal hygiene account for the largest fraction of total water consumed (35 ℓ/d per capita). Food preparation makes up the smallest fraction of total water consumed (2.5 ℓ/d per capita).

The survey indicated that water use depends greatly on the number of occupants per household.

The data from this study imply that only one third of households (34 %) appeared to be using less than 200 ℓ/d of water. This amount would be supplied without charge by the municipality if the households were metered. It is conceivable that the installation of

water meters and billing for water consumption might result in changes in water use practice among residents of this community to reduce the average water consumption to near 200 ℓ/d per household.

Table 6.1: Results of questionnaire study determining total household and per capita water use and water use for different household activities.

	No. of houses	**Total (ℓ/d)**	**Household average (ℓ/d)**	***Per capita* average (ℓ/d)**
Water used for food production		830	10	2.5
Water used for washing clothes		7 780	96	23
Water used for bathing		11 800	146	35
Water used for flushing toilets		7 240	89	22
Total water used/d		27 700	342	83
No. of houses surveyed	81			
No. of occupants	333	(total)		
	4.1	(average)		
Household use >200 ℓ/d	55			
Household use <200 ℓ/d	26			

Theoretically, all water except that used for food preparation (including drinking) should be returned to the sewer since this community does not have an active water reuse culture. However, greywater return studies in other parts of the country suggest that wastewater return can range between 50 and 75% from communities of this nature.

6.3 WATER METERING DATA SURVEY

eThekwini Water Services has a *geographical information systems* (GIS) database which contains information about water and wastewater services, as well as geographical and demographic data and historical water consumption data for large portions of the municipal region. This study proposed to exploit the comprehensive nature of this data to find patterns in water use of low-income communities within the eThekwini Municipality.

6.3.1 Methodology of water metering data survey

The *geographical information systems* (GIS) databases were mined for water consumption data from water meter readings from communities serviced with semi-pressure (roof tank) water supply using ArcGIS™ software. Data was categorised into residential area and number of houses within an area. Figure 6.2 shows a map of the eThekwini Municipal area, indicating the 15 major low-income housing developments supplied by roof tank water systems. 3 of these areas (Durban, Pinetown and New Germany) are regarded as urban, whilst the other 11 are regarded as peri-urban.

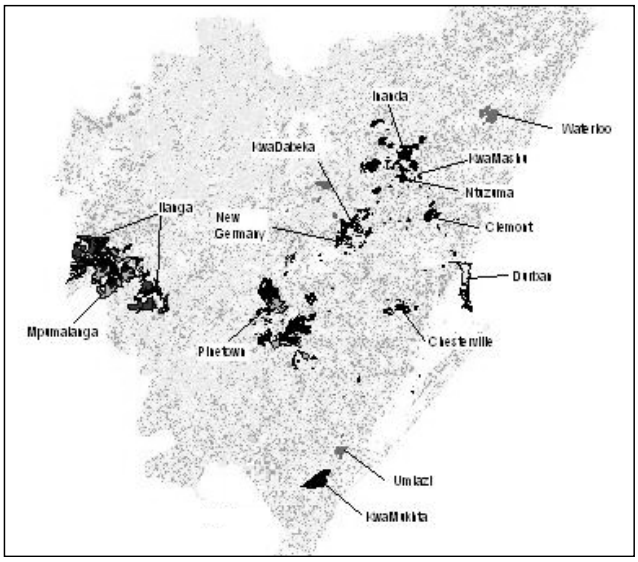


Figure 6.2: Map of the eThekwini Municipal Area, indicating the 15 major low-income housing developments supplied by roof tank water systems. 3 of these areas (Durban, Pinetown and New Germany) are regarded as urban, whilst the other 11 are regarded as peri-urban.

Records were not available for the Newlands-KwaMashu Interface housing development where the water use questionnaire study and wastewater characterisation studies were undertaken; however, a neighbouring area, Melkhout was supplied with metered roof-tanks. This community has similar house designs as the Newlands-KwaMashu community, and consists of 7 sections, numbered in the order in which they were built. It was hypothesised the geographical nearness of the two communities would equate to similar cultural and water use practices, and therefore that a study of water consumption data from in Melkhout would give an indication of the water use in Newlands-KwaMashu.

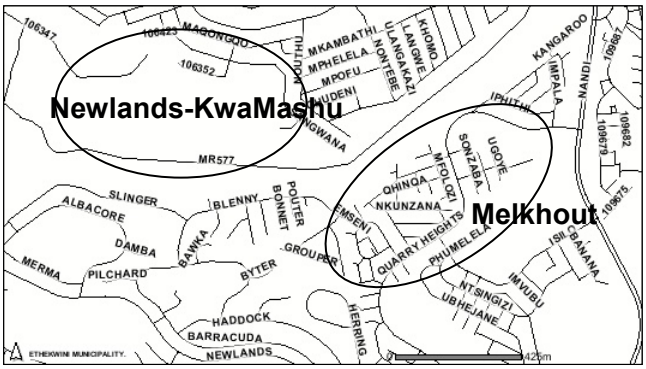


Figure 6.3: Map showing Newlands-KwaMashu Interface housing development project community and neighbouring Melkhout community.

6.3.2 Results of water metering data survey

Table 6.2 shows the water consumption in the 7 sections of the Melkhout community.

In the Melkhout community, average household water consumption in each of the 7 sections varied from 211 ℓ/d to 546 ℓ/d, with an overall average of 351 ℓ/d. The overall average is similar to the overall average of the Newlands-KwaMashu community household (342 ℓ/d). However, since the methods of determining these averages are not the same, it cannot be conclusively stated that water consumption in the two communities is similar.

Table 6.2: Water consumption for the 7 sections of the Melkhout housing development located opposite the KwaMashu-Newlands housing development. Both have similar dwelling and community structure, and are served by roof-tank water supply. Sections are numbered according to the order in which they were built.

	No. of houses	Total consumption (Kℓ/d)	Household average consumption (ℓ/d)
Section 1	111	27	211
Section 2	414	96	272
Section 3	342	117	461
Section 4	195	103	546
Section 5	133	45	384
Section 6	301	96	281
Section 7	318	80	300
Total	**1814**	**564**	**351**

Water consumption data for the 3 major urban housing developments (Durban, Pinetown and New Germany), and 11 peri-urban housing developments in eThekwini Municipality was analysed in the same way as for the Melkhout community, and results are presented in Table 6.3 and Table 6.4.

Table 6.3: GIS water consumption data for 3 major urban housing developments within the eThekwini Municipality, supplied with roof-tank water systems

Town	No. of houses	Total consumption (Kℓ/d)	Household average consumption (ℓ/d)
Durban	4669	1773	1 115
New Germany	527	125	892
Pinetown	4827	1304	976
Total	**10023**	**3202**	**994**

Table 6.3 and Table 6.4 show that there is a significant difference between average water consumption in established urban developments (994 /(d.household)) and in peri-urban developments (473 ℓ/(d.household).)This may be due to the differences in socio-economic status between these communities, for example unemployment rate, although the reasons for observed differences were not investigated in this study.

This study is continuing.

6.4 WASTEWATER CHARACTERISATION STUDY

Wastewater was sampled from sewers in the Newlands-KwaMashu interface housing development community at different locations, during different seasons and at different times of day. Samples were analysed for a range of physical, chemical and microbiological properties at the University of KwaZulu-Natal.

The purpose of this study was to obtain a characterisation of the wastewater from a low-income community for use in the design of a decentralised wastewater treatment system. To appropriately size a treatment system it is necessary to have an understanding of the average feed characteristics as well as the maximum and minimum conditions that can be expected. Since concentration data only describes conditions at a single point in time irrespective of the amount of the wastewater at that time, and therefore is not an indication of the contaminant load that a treatment system will have to handle, average feed conditions should be calculated on a flow-weighted basis.

The mass load of contaminants is defined as the product of a sample concentration and the wastewater flow in the sewer at the time of sampling. Mass loads from different sampling times should be averaged as an indication of the amount of contaminant that requires treating in a certain time period.

Table 6.4: GIS water consumption data for 11 peri-urban housing developments within the eThekwini Municipality all supplied with roof-tanked water systems

Community	No. of houses	Total consumption (Kℓ/d)	Household average consumption (ℓ/d)
Chesterville	759	263	346
Clermont	281	211	751
Ilanga	231	62	267
Inanda	5231	2039	509
KwaDabeka	1639	650	397
KwaMakhuta	349	143	831
KwaMashu	126	60	478
Mpumalanga	2956	1505	509
Ntuzuma	1383	701	507
Umlazi	1001	228	228
Waterloo	1559	585	375
Total	**15515**	**6447**	**473**

6.4.1 Methods

Winter and summer sampling campaigns were conducted. Three sewers each receiving input from approximately ten households within Section 1 of the housing development were sampled. Sample collection was conducted over three days (for each season) at five intervals per day (06h00, 09h00, 12h00, 15h00, 18h00).

Flow rate was measured on one day only by taking 5 sets of bulk water meter readings, with each set consisting of four readings at 20 min intervals, read from the bulk meter that serves Section 1 of the Newlands-KwaMashu Interface housing development.

The pH value of each sample was recorded at point of collection. Total and dissolved COD, total Kjeldahl nitrogen and total suspended solids were analysed according to Standard Methods (APHA, 1998). Total protein and total carbohydrate were analysed according to Raunkjaer et al., (1994).

Total coliform and *E. coli* were enumerated by standard membrane filtration. Filters were placed on Chromocult® coliform agar (Merck®). Coliphage were analysed by standard double layer plaque assay using *E. coli* (strain ATCC 13706) as host.

Only concentration data could be obtained from sewer samples; mass loads of contaminants in the sewer at each sampling time could not be calculated as there was no way of measuring flow rate in the sewers at the time of sampling.

6.4.2 Results of wastewater characterisation study

The average flow rate for five reading sets taken at different times over one day is reported in Table 6.5.

The flow data are also plotted below in Figure 6.4 (a). Highest flows are seen during the early morning (6.10 kℓ/h). Midday flow rate was similar although slightly less than the morning flow rate (6.02 kℓ/h). Flow rates measured at other times of day are lower (between 5.03 and 5.22 kℓ/h). These flows provide an indication of the variability of the water use, and therefore expected wastewater generation. These data are limited by the following:

These values pertain to potable water use, whereas this study is interested in wastewater flow values. As there is no means of quantifying wastewater return figures (fraction of potable water used that is returned in the sewer as wastewater), these data cannot be quantitatively used in wastewater characteristics calculations.

Only one set of data was obtained. There is no way of determining whether the values are representative or not.

The data describes water use by a whole section of the community. It is not known how many households are supplied by water that passes through this meter. Furthermore, wastewater samples were obtained from internal sewers in Section 1 of the community. Therefore, wastewater flow in the sewers sampled may have shown very different flow rate patterns to the overall potable water supply rate to the whole community.

Table 6.5: Average flow rate calculated from meter readings obtained from the bulk flow meter on the potable water line feeding Section 1 of the Newlands-KwaMashu interface housing development for different times of day. Data was collected on one day only.

Time of day	**Average flow rate kℓ/h**
Early morning	6.10
Mid morning	5.10
Mid day	6.02
Early afternoon	5.22
Late afternoon	5.03

The application of this data is limited; however, there are some clues that may be gleaned from it. Although some variation was observed, the magnitude of the variation was not large. It is understood that the community studied has a high unemployment rate, although no data for unemployment rates are available. Communities or suburbs where many inhabitants are at work during the day are expected to generate wastewater flows characterised by large differences in volume. In a community with

high unemployment rates, it is possible that water use will be spread more evenly throughout the day, and therefore less variation in wastewater flows could be expected.

Data for different measured wastewater characteristic were averaged for each sampling time: i.e. all samples measured at each of 06h00, 09h00 etc. were lumped to describe average measured wastewater characteristics at different times of day. These data are presented in Figure 6.4.

Most components measured in the wastewater showed a decreasing trend, from the early morning to the late afternoon, although the trend is not statistically significant as a result of large standard deviations calculated from the concentration data. COD concentrations are significantly higher in winter than in summer samples, as are measured pH values. This is attributed to the development of a biofilm in the sewers during the summer season which caused acidification of organic material in the wastewater. This resulted in a decreased pH value, which was measured at the sampling site. However organic acids are metabolised before samples can be analysed in the laboratory, resulting in the measurement of lower COD values.

Total coliforms and coliphage concentrations in the wastewater are higher in summer than in winter, while there is little difference in measured *E. coli* concentrations. The reasons for these differences are not certain.

It was considered inappropriate to attempt to calculate mass loads of contaminant from the concentration and flow rate data presented here since the relationship between the water flows measured and the wastewater sampled was extremely small. However, the wide variation of concentrations measured suggests that there is a good probability that the large confidence interval around the arithmetic means calculated from the data will overlap significantly with the confidence interval that would have been calculated for the flow-weighted average concentration, had appropriate wastewater flow rate data been available. The larger the number of samples obtained, and the larger the variation in the measurements, the greater the probability that the simple mean calculated from the concentrations will not be significantly different to the flow-weighted average. The number of measurements of each of the characteristics measured in this study was large, ranging from 45 to 137, it is therefore proposed that the actual average concentration (defined as the average mass load of the component, divided by the average flow) would fall in the same confidence interval as the mean measured concentration.

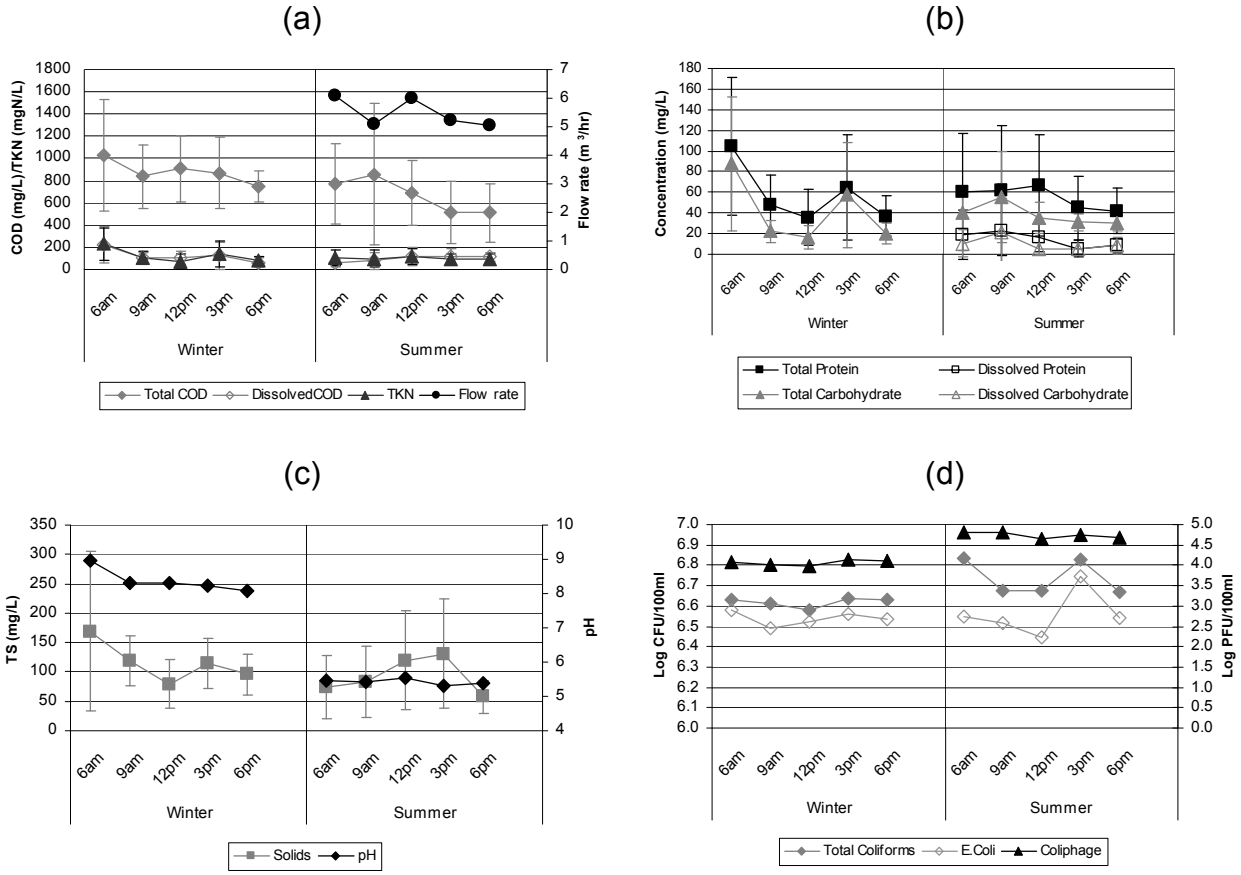


Figure 6.4: Measured wastewater characteristics for samples obtained from sewers in the Newlands-KwaMashu interface housing development averaged for sampling time in both the winter and summer studies (a) Total and dissolved COD, TKN and flow rate; (b) Total and dissolved protein, and total and dissolved carbohydrate; (c) Total solids and pH and; (d) Total coliforms, *E. Coli* and coliphage. Error bars show sample standard deviations.

This data is still has large uncertainties associated with it, and therefore, the arithmetic mean of concentrations is not an appropriate measure to use in design of a system to treat low-income community wastewater. The value of the 80th percentile concentration[1] was chosen as a representative measure of wastewater characteristics in design since this amount excludes extreme data values, but allows for *worse than average* characteristics for predicting wastewater treatment requirements.

Table 6.6 and Table 6.7 present data for different component concentrations measured in the wastewater samples from 3 sewers in the Newlands-KwaMashu interface housing development for three sampling days in each of winter and summer, and for five

[1] The lowest 80 % of measured component concentrations fall below the 80th percentile concentration

sampling times on each day. Averages for each sampling time are presented. Data are reported separately for the winter and summer studies. Table 6.8 presents overall values of mean, sample standard deviation (with n = 45 in all cases) and 80th percentile for all times, days and sewers, but separated into winter and summer studies. The 80th percentile value for pH was chosen on the lower side of the average (i.e. 80% of measurements were above this value) since low pH has a larger negative affect on anaerobic wastewater treatment than high pH.

A treatment system must be designed to be able to handle the worst case scenario; winter data show higher COD values, while summer data show lower pH values. Chapter 7 shows that the effect of the influent pH on anaerobic digestion is not great for the pH values measured, hence winter 80th percentile concentration data and pH values were chosen for designing a low-income community anaerobic wastewater treatment system.

By comparing Tables 6.6 to 6.8 with Tables 4.3 to 4.6. It is clear that the wastewater in the sewers in the low-income community studied is more concentrated in terms of COD and organic nitrogen than influent wastewater to Kingsburgh WWTP. This supports the theory that low-income communities will generate higher strength wastewater than is seen in middle-income suburbs.

6.5 CONCLUSIONS FROM THE COMMUNITY WATER USE AND WASTEWATER GENERATION STUDY

A community water use survey identified that low-income urban dwellers in general used significantly more water in their homes than their peri-urban counterparts.

A community wastewater characterisation study was unable to identify significant correlations between contaminant concentrations and time of day, sewer or day of the week for samples obtained from sewers in a low-income peri-urban community.

Low income community generated wastewater showed higher concentrations of COD and organic nitrogen then were measured in municipal wastewater from middle-income communities.

The community wastewater characterisation study obtained many measurements of concentration for a number of important wastewater characteristics in a low-income peri-urban community. Flow data were not obtained at the same time and therefore calculation of contaminant loads could not be performed. Furthermore, certain critical analyses were not performed, specifically VFA and alkalinity measurements. It is recommended that the study be repeated, with some means of calculating wastewater flow at the time of sampling so that contaminant loads through the sewer can be determined, and with measurements of alkalinity and VFA.

Table 6.6: Averages and standard deviation of wastewater characteristics measured at different times of day 3 sewers in the Newlands-KwaMashu Interface Housing Development during winter

		Time	06h00	09h00	12h00	15h00	18h00
		Units	Average	Average	Average	Average	Average
Winter	**Total COD**	mgCOD/ℓ	1 024	839	909	867	746
	Soluble COD	mgCOD/ℓ	222	105	105	123	62
	Total protein	mg/ℓ	105	48	36	65	37
	Total carbohydrate	mg/ℓ	87	22	17	57	20
	TKN	mgN/ℓ	230	105	75	140	79
	Total solids	mg/l	169	119	79	114	96
	T. Coli	log(cfu /100 mℓ)	6.63	6.61	6.58	6.64	6.63
	E. Coli	log(cfu /100 mℓ)	6.58	6.49	6.52	6.56	6.53
	Coliphage	log(pfu /100 mℓ)	4.06	4.01	3.97	4.14	4.12

Table 6.7: Averages and standard deviation of wastewater characteristics measured at different times of day 3 sewers in the Newlands-KwaMashu Interface Housing Development during summer

		Time	06h00	09h00	12h00	15h00	18h00
		Units	Average	Average	Average	Average	Average
Summer	**Total COD**	mgCOD/ℓ	773	856	690	515	508
	Soluble COD	mgCOD/ℓ	55	77	116	117	112
	Total protein	mg/ℓ	61	62	67	45	42
	Dissolved protein	mg/ℓ	19	23	16	6	8
	Total carbohydrate	mg/ℓ	40	55	36	31	31
	Dissolved carbohydrate	mg/ℓ	11	21	5	5	9
	TKN	mgN/ℓ	102	97	119	95	89
	Total solids	mg/ℓ	74	83	120	131	58
	T. Coli	log(cfu/100 mℓ)	6.83	6.52	6.44	6.75	6.54
	E. Coli	log(cfu/100 mℓ)	6.55	6.52	6.44	6.75	6.54
	Coliphage	log(pfu/100 mℓ)	4.80	4.81	4.66	4.73	4.68

Table 6.8: Overall averages, standard deviation and 80th percentile values of wastewater characteristics measured in the Newlands-KwaMashu Interface Housing Development during summer and winter studies

		Units	mean/ range	Standard deviation	80th percentile	No. of observations.
winter	**Total COD**	mgCOD/ℓ	877	37	1089	90
	Soluble COD	mgCOD/ℓ	124	9	169	135
	pH	-	7.76-9.56	NA	8.05	45
	Total protein	mg/ℓ	57	4	80	137
	Total carbohydrate	mg/ℓ	41	4	46	135
	TKN	mgN/ℓ	126	11	176	90
	Total solids	mg/ℓ	113	8	153	92
	T. Coli	log(cfu/100 mℓ)	6.62	NA	6.75	135
	E. Coli	log(cfu/100 mℓ)	6.54	NA	6.70	135
	Coliphage	log(pfu/100 mℓ)	4.06	NA	4.20	135
summer	**Total COD**	mgCOD/ℓ	676	42	907	89
	Soluble COD	mgCOD/ℓ	96	6	130	134
	pH	-	4.94–6.18	NA	5.30	45
	Total protein	mg/ℓ	59	5	94	87
	Dissolved protein	mg/ℓ	15	2	23	131
	Total carbohydrate	mg/ℓ	38	2	45	135
	Dissolved carbohydrate	mg/ℓ	10	2	15	134
	TKN	mgN/ℓ	100	7	162	90
	Total solids	mg/ℓ	91	8	150	92
	T. Coli	log(cfu/100 mℓ)	6.74	NA	7.11	135
	E. Coli	log(cfu/100 mℓ)	6.56	NA	6.97	135
	Coliphage	log(pfu/100 mℓ)	4.74	NA	4.97	135

7 MODELLING OF THE ABR

This chapter describes the results of mathematical modelling the microbiological processes occurring in an anaerobic baffled reactor. This study forms the basis of a PhD research project in the School of Chemical Engineering at the University of KwaZulu-Natal.

The modelling exercise serves to provide a basis for numerative predictions of the performance of an ABR to be determined under different operating conditions. In this chapter, preliminary results using a biochemical model and a steady-state model are presented and are used to predict the effect of different feed characteristics on an ABR.

7.1 MATHEMATICAL MODELLING

Mathematical modelling is faced with a compromise between detail and robustness: very detailed mechanistic models of many sub-processes may provide an accurate theoretical representation of the processes, but are unwieldy and difficult to calibrate. A common problem with biological models is poor practical (parameter) identifiability (Vanrolleghem et al., 1995; Dochain and Vanrolleghem, 2001): i.e. it is often not possible to identify independent parameter values for models with many parameters given the possible experimental measurements available. Thus a calibration that appears to reproduce experimental data accurately may only be a *local solution* or *local minimum* of the objective function for parameter values and unable to simulate experiments under slightly different operating conditions.

It is therefore necessary to select a less complicated model structure which leaves out highly specific sub-processes but is able to describe the general picture sufficiently well (Batstone et al. 2002).

Most modelling effort has been directed towards activated sludge processes, resulting in the proliferation of activated sludge models. The establishment of reference models has provided a common point of understanding for communication between scientists, engineers and water management professionals. Several *anaerobic digestion* models have been published in the literature. In 2001, a task group of the IWA published the first IWA reference model for anaerobic digestion, the Anaerobic Digestion Model No. 1 (ADM1) (Batstone et al., 2002). Despite the existence of a reference platform for anaerobic digestion modelling, it is widely recognised that there is a significant gap between routinely determined wastewater characteristics and the data requirements of current modelling practice, since easily measurable quantities generally do not resolve into the biological and chemical components that are required from a modelling perspective (Rozzi and Remigi, 2004). Hence, there is a need to develop a practical mapping between measurements and model inputs before anaerobic digestion modelling can be routinely applied.

There are two main objectives for modelling an anaerobic process with a complex configuration such as the ABR:

- Because the rates of anaerobic processes are so low, comprehensive experimentation is time-consuming and costly; a properly calibrated model could reduce experimental effort by extrapolating experimental data to predicting operating limits, and by directing experimental effort to where information to characterise the process is most lacking.

- The complex configuration provides a more stringent test of a model than a single mixed reactor, for which averaged descriptions of the processes involved are more likely to prove adequate.

The ABR therefore represents an opportunity to improve the models and thereby the understanding of anaerobic digestion.

7.2 SIEGRIST MODEL OF THE PILOT ABR

This section describes a modelling exercise using experimental data from the 2003 operating period of the pilot ABR treating middle-income domestic influent at the Kingsburgh WWTP. This work was presented in a paper at the 2004 Biennial WISA conference in Cape Town, entitled *Biochemical modelling of the anaerobic baffled reactor* by K.M. Foxon, C.J. Brouckaert, E.U. Remigi and C.A. Buckley (Foxon et al., 2004).

In this study, the following approach was used;

- An existing, but probably not completely appropriate model was applied to existing but probably inadequate experimental data.

- A preliminary round of data fitting was performed, adjusting as few model parameters as possible.

- Improvements were proposed for both the model structure and the experimental programme.

7.2.1 Siegrist model structure

A model of the ABR has been developed using the modelling platform WEST®. The configuration adopted was a series of 8 completely stirred tank reactors (CSTRs) in series, using feed characteristics determined from experimental data and recommendations from literature. The model was fitted to pH, COD and alkalinity data by manual tuning. Figure 7.1 shows the WEST® representation of the reactor configuration. Each element represents a constantly stirred tank reactor. Soluble components flow directly from one compartment / CSTR unit to the next, while a fixed fraction of total particulate components are retained in each unit.

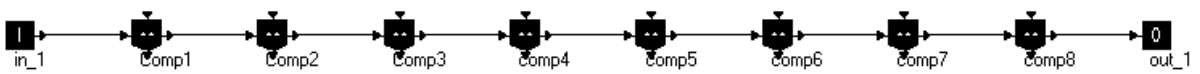


Figure 7.1: WEST® representation of the ABR flow configuration. Each element represents a constantly stirred tank reactor

At the beginning of the model building exercise, the resources available for inclusion in a model could be described as follows:

- Commonly measured quantities (pH, COD, alkalinity, free and saline ammonia)
- Further data sets obtained with the specific objective of identifying reactor characteristics related to the reactor configuration
- A modelling platform (WEST®) with built-in anaerobic biological reaction models (stoichiometry and rate data)

7.2.1.1 Selection of reaction model structure

At the time of this exercise, three reaction models were available in WEST®:

- The Lessard-Desjardins model uses one microbial population to hydrolyse biodegradable solids, generate volatile fatty acids and convert these to methane, with concomitant release of ammonia and phosphate. No alkalinity or pH effects are included in the model.
- The Siegrist model (Siegrist et al., 1993) describes anaerobic digestion of sewage sludge, and models degradation of a particulate biodegradable component to a combined sugar and amino acid component and a fatty acid (lipid) component. These in turn are degraded to propionate and acetate, and acetate and dissolved CO2 are converted into methane and CO2 gas by methanogenesis. Electron and carbon balances are maintained in each process by the generation of bicarbonate and CO2, and pH change is modelled by protolysis and deprotolysis of bicarbonate and hydrated CO2. Five distinct microbial populations catalyse these processes. The model is able to simulate variations in load on a full-scale digester, but exclusion of acid-base dynamics at low pH values (dissociation of volatile fatty acids) limits its ability to describe reactor failure.
- The Anaerobic Digestion Model No. 1 (ADM1) (Batstone et al., 2002) is similar to the Siegrist model, but includes particulate carbohydrates, proteins and lipid components, and butyric and valeric acids in addition to the Siegrist volatile fatty (organic) acids. Seven microbial populations are described. The pH changes are modelled through the acid-base reactions of all organic acids, inorganic carbon and ammonia. Detailed gas liquid interactions are described.

Anaerobic digestion is conventionally used for treating waste activated or primary sludge and other high strength (high COD) applications, which are characterised by very large generation of alkalinity (ca. 3 000 mg $CaCO_3/\ell$, Speece, 1996). Anaerobic digestion models have been developed with these applications in mind. In the present application, a (relatively) low strength domestic wastewater is being treated, and as a result, the process will not be as well buffered as in sludge digestion. With this difference in mind, it was assumed that a measure of the variation of pH would be necessary to be able to track the performance of the ABR and to accurately describe

inhibition effects on microbes from low pH due to poor buffering. Also, in previous work (Bell and Buckley, 2004, Barber and Stuckey, 1999) the ABR has been shown to establish differing microbial communities in each compartment as a result of changing concentration gradients of microbial process reactants. Consequently, it was decided that the failure to describe both microbial diversity and pH dynamics in the Lessard-Desjardins model made it inadequate for the purposes of this study.

While ADM1 is undoubtedly a more comprehensive representation of the biochemical pathways of anaerobic digestion than the Siegrist model, it was felt that insufficient data were available from the first round of experimentation to justify the implementation of such a complex reaction model structure as presented in ADM1. In modellers' terms, ADM1 was considered to be over-parameterised relative to the available data. Consequently, the Siegrist model was used as a starting point for modelling the ABR.

7.2.1.2 Compilation of available experimental data

The data available from the 2003 operating period of the pilot ABR at Kingsburgh WWTP was modified to match the input requirements of the Siegrist model. The assumptions that were chosen were:

- The concentrations of acetate, propionate and methane in the influent wastewater are negligible, so that the dissolved biodegradable fraction is made up of amino acids and sugars and long chain fatty acids.

- The concentrations of active biomass make up a negligible proportion of the incoming COD, so the biodegradable particulate fraction is all slowly biodegradable COD. However, the stable operation of the model requires some seeding of the biomass in the reactor from the feed, so arbitrary but low values were chosen.

Table 7.1: Feed COD fractionation implemented in the ABR model

Siegrist Model components	**Fraction of total COD %**
Inert dissolved COD	3.8
Inert particulate COD	13.0
Amino acids and sugars	18.2
Long chain fatty acids	1.8
acetate	0
propionate	0
methane	0
Active biomass (4 species)	0.2
slowly biodegradable COD	63.2

The soluble inert COD presented a small complication in that, in the Siegrist model, the *slowly biodegradable particulate COD* produces some soluble inert COD when it hydrolyses. It was assumed that this should be counted as part of the soluble inert COD

in the raw sewage characterisation, and that the *slowly biodegradable particulate COD* concentration should be adjusted to reflect only its biodegradable fraction.

This led to the speciation of incoming COD in terms of model components shown in Table 7.1. The carbonate system components were set according to the measured alkalinity and pH, and the NH_4^+ was set according to the measured free and saline ammonia.

Model inputs: Flow rate and temperature. The experimental data available were obtained for a reactor through-flow of ca. 2.3 ℓ/min. The reactor temperature was 22 ± 2 ºC.

Model outputs: Concentration data. Measurements of total and 0.45 µm filtrate COD of the feed and effluent were available. Average values for 0.45 µm filtrate COD in each compartment were determined, and one set of total COD concentration for a mixed core sample of each compartment was obtained. Filtrate COD measurements were assumed to be representative of the soluble fraction of COD, consisting of amino acids and sugars, fatty acids, acetate, propionate and soluble inert COD. Total COD was considered to be a measure of all components contributing to the total COD of a sample. Within compartments, these measurements were assumed to be largely attributable to the particulate COD (biomass, particulate biodegradable and inert COD). Total alkalinity measurements were interpreted as the bicarbonate concentration in solution, and dissolved CO_2 concentrations were calculated according to the equilibrium speciation at the reaction pH. Free and saline ammonia measurements were used to describe the free ammonium component of the model since at the reaction pH, the ammonia (NH_3) component contributes less than 0.2% of the total Free and saline measurement, which is well within experimental error.

Model outputs: Flow configuration related measurements. It was understood that the relationship between the up-flow velocity and the solids settling characteristics in the up-flow region of each compartment would be important in determining the amount of solids retained in each compartment, and consequently the absolute amount of biological activity that occurred. Consequently, batch settling tests to determine the fraction of solids retained at the operating up-flow velocity were performed (Mtembu, 2005). Results of these tests were not conclusive; considerable scatter in the data gave very large confidence intervals for the determination of the solids retained. However, it could be concluded that a greater fraction of total solids had settling velocities less than the average up-flow velocity (i.e. would be entrained in the flow and carried over) in the first two compartments than subsequent compartments. The implication is that the values for solids carry-over in the first two compartments were different from those in the subsequent compartments.

7.2.1.3 Model considerations

The Siegrist model default temperature is 35 ºC, whereas the ABR operating temperature was about 22 ºC. The model includes temperature corrections for the rates of all the processes apart from hydrolysis of particulate organics. A temperature correction following the recommendations of Batstone et al. (2002) was included for hydrolysis.

7.2.2 Siegrist model results and discussion

It was attempted to fit the model to the data without varying any model kinetic parameters (growth rate coefficients, half-saturation constants, yield coefficients) since none of the available experimental data was specific to individual biological processes. Parameters that could be manipulated were therefore limited to hydraulic parameters (sludge retention factor) and feed characteristics.

7.2.2.1 Model outputs

The figures below present the results obtained for soluble COD (i.e. the COD of 0.45 µm filtrate), ammonia, alkalinity and total COD. In each figure, the left (grey) bar of each pair indicates a model prediction, and the right (black) bar an experimental measurement.

In order to obtain reasonable correspondence between the model and experimental data for particulate COD concentration, the fraction of solids carried over in each compartment, a Siegrist model parameter $f_{X,out}$, was tuned for each compartment in sequence. Figure 7.2 shows the agreement between measured and predicted values. The model values for the influent and compartments are functions of model inputs only. However the particulate COD concentration in the effluent is calculated by the model, a model output, and similar orders of magnitude are observed between predicted and measured values. However, the model predicts that 30% more total COD, and 52% less particulate COD exits the system than was measured in the pilot rig. The values of solids carry-over, $f_{X,out}$ set in the tuning process are presented in Figure 7.3.

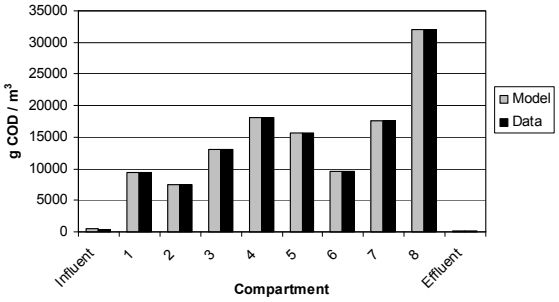


Figure 7.2: Particulate COD Profile

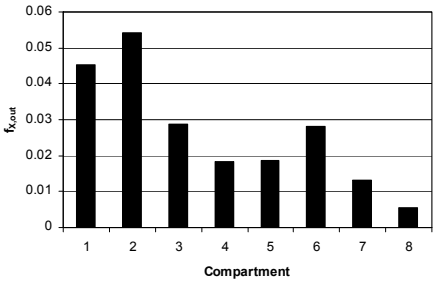


Figure 7.3: $f_{X,out}$ (solids carry-over fraction) in each

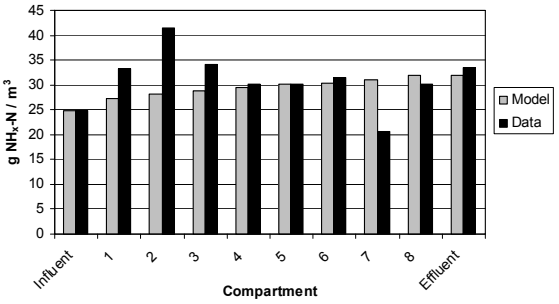


Figure 7.4: Free and saline ammonia

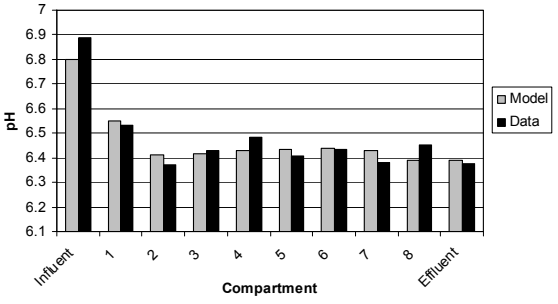


Figure 7.5: pH profile

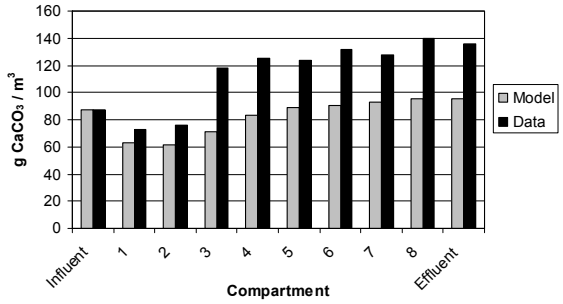


Figure 7.6: Alkalinity profile

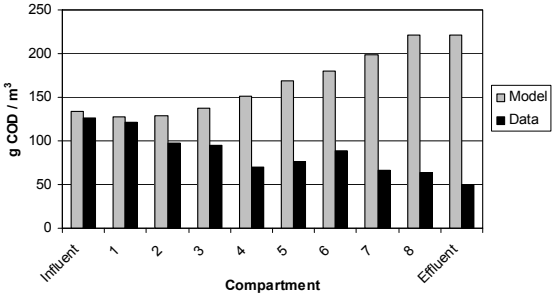


Figure 7.7: Soluble COD profile

Although no reliable experimental values for solids carry-over were available, the observation that solids settled more slowly in the first two compartments (Mtembu, 2005) than in subsequent compartments appears to be correct since the manually tuned values for $f_{X,out}$ show a greater fraction of solids carry-over to achieve similar compartment solids concentrations.

Figure 7.4 and Figure 7.5 show that the model free and saline ammonia and pH profiles are very similar to the experimental ones. It is interesting to note that the tuning of the solids carry-over fractions caused the modelled pH profile to simulate the experimental data better than any of many other exploratory attempts at tuning of biochemical parameters (not reported).

The modelled alkalinity profile (Figure 7.6) had a similar trend to experimental data, but in general predicted a lower alkalinity generation than was observed experimentally. This is probably due to the assumption that the influent VFA concentration was negligible (Section 7.1.1.2) since including approximately 30 mgVFA/ℓ (roughly the amount measured in the feed in the 2004 operating period) would result in an increase in alkalinity concentration to match the experimental data without significantly affecting other measurements.

Figure 7.7 shows the predicted soluble COD vs. experimental 0.45 µm filtrate COD. The trends diverge, with predicted concentrations increasing contrary to measurements, which show decreasing soluble COD concentrations. The three modelled processes responsible for removing soluble COD (amino acid and sugar fermentation, propionate consumption and acetoclastic methanogenesis), are subject to a product inhibition mechanism (pH, hydrogen or acetate inhibit certain processes at high concentrations in the Siegrist model). Both hydrogen ion and molecular hydrogen were found to be inhibiting in the model at levels of 60 % and between 6 and 50 % of the maximum rates respectively, and this caused an accumulation of modelled acidogenesis products, which was not seen experimentally.

The fact that soluble COD concentration decreases from one compartment to the next in the experimental data implies that hydrolysis processes (solubilisation processes) are the rate-limiting steps in the pilot-reactor, while increasing soluble COD concentrations in the model show that the modelled rate of complete anaerobic digestion is limited by some soluble product degradation process within the model. However, it is likely that manually increasing the soluble product removal rates will at best result in a constant

soluble COD profile as a result of soluble product generation by hydrolysis, unless there is a decreasing hydrolysis rate with residence time. This could be best achieved by having more than one particulate COD component, each with its own specific hydrolysis rate. This is intuitive since in later compartments, a considerable proportion of the particulate COD will have been generated by inactivated biomass, with lower hydrolysis rates than wastewater solids. The simplest scenario would be to model dead biomass as its own component. Extensive adaptations to the Siegrist model would be required to implement this change, but ADM1 has provision for this model structure. On an experimental level, it is necessary to have more specific measurements of soluble components, such as organic acids acetate and propionate, and gas production in order to be able to manipulate kinetic constants generally, but of hydrolysis processes in particular, in a meaningful way.

7.2.2.2 Feed speciation

The feed speciation adopted for this model was based on experimental measurements, and was chosen to match the recorded pH and COD data available. However, several assumptions had to be made to fix the mapping between available measurements and model inputs. To eliminate the assumptions, the following measurements are necessary:

- To fully describe the soluble COD portion of the feed, acetate and propionate should be measured, and if possible, some further measurement to quantify the proportions of amino acids and sugars, and fatty acids in the soluble portion of the feed.

- An investigation into the effect of active biomass concentration in the feed (seeding rate) is required to test whether or not the low values chosen in this exercise did in fact affect the steady-state biomass populations in the compartments. If the seeding rate is shown to be significant, then some measurement of this quantity must be attempted.

- Some measurement of the nitrogen content of the various soluble and particulate COD fractions should be made.

- The inert content of particulate and solid COD components should be determined.

Where possible, these measurements and determinations were implemented in the 2004 operating period.

7.2.3 Conclusions drawn from Siegrist model

In this study, an 8 compartment anaerobic baffled reactor was modelled as 8 CSTRs in series using WEST®. Existing experimental data from the pilot-scale reactor was manually fitted to a Siegrist model of anaerobic digestion that is already implemented in WEST®. An attempt to fit the existing data to the existing implementation of the Siegrist model revealed that the default model structure could not adequately describe digestion of a domestic wastewater since the model had originally been formulated for digestion of sewage sludge. Furthermore, it was observed that a critical factor in the ability of the model to realistically describe the biological processes in the ABR, was the appropriate characterisation of the feed, the experimental data for which were not available. A

mapping between the experimental data and required model inputs was developed and a minimal set of model parameters were altered, without altering the basic structure of the Siegrist model, such that a significant improvement in the agreement between model prediction, experimental observation and intuitive understanding of anaerobic digestion could be obtained.

It was then shown that reasonable model predictions could be obtained for pH, particulate COD, alkalinity and free and saline ammonia profiles within the reactor, but that the model could not describe the soluble COD profile without substantial modification.

It was concluded that the following change should be made to the model:

- One particulate biodegradable COD fraction is insufficient to represent decreasing average hydrolysis rates; a subdivision of particulate biodegradable COD is proposed. Since this change is possible in ADM1, it seems appropriate to adopt the ADM1 model structure in the next stage of modelling.

The following experimental measurements were found to be necessary to fully describe the model inputs:

- Particulate and soluble organic nitrogen in the feed
- Particulate and soluble inert COD in the feed
- Total gas production
- Organic acid concentration in the feed and in compartments
- Some measure of the biomass seeding rates, or indication of where the model is sensitive to the seeding rate

It is expected that implementing these changes will increase the plausibility of the model output, and that subsequent iterations will expose where rate data and model structure changes should be modified to improve model fit. This in turn will indicate what experimental measurements are necessary to verify the proposed model changes.

7.3 ADM1 MODEL OF ANAEROBIC DIGESTION

Modelling of the ABR using ADM1 in WEST® is the basis of co-operative research between the Pollution Research Group, University of KwaZulu-Natal and BIOMATH, Gent University (Belgium). At the time of writing this report, the ADM1 model of the ABR was still being developed. The remaining phases of this ABR modelling include hydraulic modelling in WEST, validation of the model using pilot ABR operating data, and prediction of ABR on community wastewater.

7.4 STEADY-STATE MODELLING

Steady-state models are based on the principle of the *rate-determining step*: in a steady-state system, the overall rate of treatment will depend on the slowest process

that occurs in the system. Provided the conditions of the system do not change such that another process becomes rate-limiting, a calibrated steady-state model will give a reasonably quick basis for designing a system and determining operating parameters, or estimating system performance under slightly different conditions.

Sötemann et al. (2005) have recently developed a steady-state model for anaerobic digestion of sewage sludges, based on the assumption that hydrolysis of macromolecules is the rate-limiting step in this process. This is a three step model consisting of (i) a kinetic part for determining COD removal and gas production, (ii) a stoichiometric part that calculates ammonia, alkalinity production and digester gas composition and (iii) a weak acid-base section that calculates the digester pH from the gas composition and alkalinity.

The chemical oxygen demand of the feed in the steady-state model is assumed to be a combination of particulate biodegradable organic materials with a known average elemental composition $C_XH_YO_ZN_A$, volatile fatty acids (represented by acetic acid) and a fraction of unbiodegradable material. All VFA is consumed in the process, and a portion of the remaining biodegradable organic material is converted to methane, CO_2, alkalinity and ammonia. The extent of biodegradation depends on the sludge age or length of contact time in the system.

7.4.1 Applicability of steady-state modelling to the ABR

The Siegrist modelling exercise and analysis of the experimental data obtained from the pilot ABR have indicated that the overall rate-limiting step in the anaerobic digestion of domestic wastewater is hydrolysis, and therefore it is appropriate to use a steady-state hydrolysis model to describe the overall digestion process in an ABR.

The Sötemann steady-state model bases its calculations on the sludge age for a completely stirred tank reactor (CSTR). The ABR however functions more like a plug flow reactor with sludge retention; consequently the methodology does not directly apply to the ABR case. In the Sötemann model, the sludge age in a CSTR is used for the calculation of the hydrolysis rate based on the sludge retention time in the system. This can be translated to the pseudo-plug-flow system as the hydraulic retention time since the contact time between waste components and biomass, i.e. the reaction time, is determined by the hydraulic retention time.

A more serious complication arises from the fact that a CSTR model assumes that the concentrations of all components in the gas, liquid and solid phases are identical throughout the reactor, and in the effluent. This is not true in the case of the ABR where distinct gradients of components are seen. A possible solution is to model the 8 compartment ABR as 8 CSTRs in series, (as in the Siegrist modelling exercise, see Figure 7.1) using the steady-state model, but it has been shown that although hydrolysis dominates the overall kinetics of anaerobic digestion in the ABR, acid consuming reactions may be rate limiting in the early compartments (shown by a lower pH and higher volatile acid concentration in compartments 1 and 2 relative to the reactor feed see, Chapter 4). The steady-state model as it stands could not be applied to these compartments.

Bearing in mind the above drawbacks, the steady-state model was applied to the overall pilot ABR operation since the available data more appropriately matched the level of input required for a steady-state model then for a detailed biochemical model. The difference between reactor conditions for the CSTR model and the pseudo-plug-flow conditions in the ABR should be constantly kept in mind when interpreting the results of the steady-state model.

7.4.2 Steady-state model structure for the ABR

A Monod relationship was used to describe the interaction between the residual particulate biodegradable concentration and the reaction rate. In an ABR configuration, this kinetic expression does not have a physical meaning on a microbiological scale, but rather, averages the amount of COD reduction that can occur within the retention time of the reactor. The Monod equation form is able to describe the non-linear relationship observed between retention time and experimentally obtained outlet COD concentration.

The equilibrium constants for carbonate chemistry used in the Sötemann model were adjusted to the lower temperature and lower TDS conditions prevailing in the pilot ABR.

7.4.3 Inputs into the steady-state model of the ABR

The average elemental composition of generic carbohydrate, lipid and protein compositions taken from Henze et al. (1992) were used to calculate the overall average elemental composition of the biodegradable organic material in the feed as follows:

Carbohydrate:	$C_{10}H_{18}O_9$	Composition = i (mol %)
Lipid:	$C_8H_6O_2$	Composition = j (mol %)
Protein:	$C_{14}H_{12}O_7N_2$	Composition = k (mol %)

Average composition of biodegradable organics is $C_XH_YO_ZN_A$

where $X=(i\cdot10 + j\cdot8 + k\cdot14)/100$

$Y=(i\cdot8 + j\cdot6 + k\cdot2)/100$

$Z=(i\cdot9 + j\cdot2 + k\cdot7)/100$

$A=(k\cdot2)/100$

Average wastewater compositions were calculated from all data collected during the two operating periods in which steady-state conditions were achieved, and are presented in Table 7.2.

Macroscopic data collected by both the project team and the municipality showed that the average composition of the wastewater was the same for the two operating periods.

The recommended values for average death rate and acidogen yield recommended by Sötemann et al (2005) were used in the ABR steady-state model. No precise measurement of the unbiodegradable fraction of influent COD was available, and consequently, the Sötemann et al. value of 0.08 (8% of total COD) was used. The results of kinetic calibration will depend strongly on how much biodegradable organic substrate has been utilised. If the residual substrate, the calculation of which is dependent on the value of the unbiodegradable fraction, is close to zero, the accuracy of the unbiodegradable fraction will have a strong effect on the models predictive power.

Table 7.2: Influent composition for model components for steady-state operating periods February to June 2003 and April to October 2004

Component	Unit	**22 h retention time (February to June 2003)**	**42 h retention time (April to October 2004)**
COD	mgCOD/ℓ	716	719
Alkalinity	mgCaCO3/ℓ	193	207
Ammonia	mgN/ℓ	35.0	39.9
pH	-	6.29	7.15
VFA	mgCOD/ℓ	33 (assumed to be the same as 2004)	35
Temperature	ºC	22	22
Protein	% of influent COD	12 (assumed to be the same as 2004)	12

7.4.4 Calibration of the ABR steady-state model: Kingsburgh data

The data presented in Table 7.2 was inputted into the steady-state model structure described by Sötemann. As no reliable measurements for carbohydrate and lipid composition were available, the carbohydrate fraction of biodegradable COD was regressed by matching the calculated alkalinity production to experimentally obtained values. The lipid fraction was calculated. Table 7.3 presents the model and experimental values obtained for the two operating periods. The reaction rates and effluent biodegradable COD values were used to obtain constants K_M (maximum reaction rate) and K_S (half saturation constant) for a Monod-type kinetic expression.

Good correlation was obtained between experimental and model values for effluent ammonia, alkalinity and pH for 2004 data, and for effluent ammonia for 2003 data. It was noted that the quality of data from 2004 was of a consistently higher standard than for 2003. Feed composition in terms of carbohydrate, lipid and protein was tuned using 2004 data, but attempted tuning using 2003 data did not result in a significant improvement in model prediction of effluent alkalinity and pH. The feed composition determined for 2004 data was therefore applied to both experimental periods since there was no reason to believe that this fractionation had changed substantially between operating periods.

Values for Monod constants describe an averaged performance of the ABR in the operating ranges tested. The half saturation constant K_S is lower than values obtained by Sötemann et al. (2005) for various data sets. The maximum specific reaction rate (K_M) value is significantly higher than any reported by Sötemann et al., and may be a function of the relatively high specific treatment rates obtained in the ABR as compared to UASB and CSTR type anaerobic reactors (Barber and Stuckey, 1999; Mudunge, 2001). However, the Monod constants do not have a direct meaning as in CSTR applications since there is a gradient of biodegradable organic COD between inlet and outlet, and therefore different kinetic expressions with different parameter values can be expected to govern the hydrolysis at different points in the reactor.

Table 7.3: Model outputs for two steady-state retention times showing experimental data and calculated kinetic and feed composition quantities used in the steady-state model. Values listed under experimental are averaged experimental data and are presented in ordinary text. Model values are either calculated outputs of the model (bold) or tuning parameters estimated to give a reasonable model fit to data (*italics*). Monod-type kinetics describing treatment rate vs. substrate concentration are also presented.

			22 h retention time (February to June 2003)		**42 h retention time (April to October 2004)**	
	Component	Unit	Experimental	Model	Experimental	Model
Effluent Characteristics	COD	mgCOD/ℓ	192	-	130	-
	Temperature	ºC	22	-	22	-
	Alkalinity	mgCaCO3/ℓ	268	248	266	267
	Ammonia	mgN/ℓ	33.2	32.8	51	49
	pH	-	6.3	5.84	5.98	5.87
Kinetic Calculations	rH	mgCOD/ℓ/d	-	578	-	354
	Effluent biodegradable COD	mgCOD/ℓ	-	75.3	-	11.8
	Acidogen biomass concentration	mgCOD/ℓ	-	59.9	-	62.8

	Unbiodegradable COD	% of influent COD	8	-	8	-
	VFA	% of influent COD	4.9	-	4.9	-
Feed composition	Protein	% of influent COD	(assumed same as 2004)		12	
	Lipid	% of influent COD			-	51
	Carbohydrate	% of influent COD			-	24
Kinetic constants	Monod Constants	-	K_M = *10.4 /d* K_S = *6.23 mgCOD/ℓ*			

The calibration of the kinetic model is the estimation of two Monod constants using only two data points. Therefore the prediction of reactor performance outside of the tested operating conditions will be strongly biased by any errors in the measurements used in the calibration.

The calculated values for the removal of biodegradable substrate (VFAs and hydrolysable material) are 88.6% and 99% for the 22 and 42 h retention times respectively using the unbiodegradable COD fraction of 0.08. Since the 42 h retention percent removal is so close to 100%, the relationship between retention time and substrate removal defined by the Monod kinetics determined in Table 7.3 will be dominated by the fact that modelled COD removal at retention times near to 42 h will not be very sensitive to changes in retention time.

These results represent a steady-state condition in the ABR. This implies that there is no net accumulation of sludge in the reactor, and that all produced sludge is leaving the reactor in the effluent, as is the case in a CSTR. In reality, a fair amount of the sludge accumulates in the compartments, and the effluent concentration should contain less biomass than predicted here.

This observation explains why the steady-state model predicted that such a large fraction of biodegradable COD was consumed by the ABR; In the mass balance calculations, the sludge growth required to achieve the observed COD removal was almost equal to the effluent COD concentration. Since there is no provision for accumulation of sludge in a steady-state model it must leave in the effluent, meaning that biodegradable COD must be completely consumed. However, it seems probable from the experimental data that sludge was accumulating in the ABR in the two operating periods used to calibrate the steady-state model, i.e. that steady state had not been achieved. The steady-state model principle can still be applied provided only a portion of the excess biomass is considered to be present in the effluent stream, i.e. allowing a sludge retention factor. Given the slow rate of sludge production compared to COD removal, an average sludge load could be calculated, as opposed to an increasing one, allowing the steady-state mass balance to hold.

Given the obvious drawbacks of applying a steady-state CSTR model to the ABR, it can nevertheless be seen that the steady-state model of Sötemann et al. is able to provide a reasonable estimation of the overall performance of the ABR, in terms of alkalinity and ammonia production under conditions where the overall treatment rate is governed by the hydrolysis step.

Only two operating points were used to calibrate the steady-state model, and there is at this stage no independent verification of the model results for domestic wastewater digestion at ambient temperatures. Therefore, the scope for predicting ABR performance outside of the conditions under which it was calibrated is limited. However, approximate reactor performance can be predicted by the model, with the following points to assist in interpreting results:

- The operating points studied experimentally were poorly buffered, and the relatively low strength of the feed to the process meant that little alkalinity was generated. As a result, low pH values dominated all compartments, and are also seen in the predicted effluent pH values calculated by the steady-state model. It is well known that depressed pH values can cause inhibition of methanogenesis in particular and anaerobic digestion processes in general (Speece, 1996, See section 2.2.4.2) It is concluded that in applications where the reactor feed has a significantly higher alkalinity and for higher strength wastewaters, pH values throughout the reactor will be higher, improving anaerobic activity relative to the Kingsburgh study, and therefore potentially resulting in better treatment rates than will be predicted using the Monod kinetics calculated from these two operating points. Conversely if even more poorly buffered or lower strength wastewaters are treated, the Monod kinetics calculated here may result in a higher estimation of treatment rates than can be experimentally obtained due to pH inhibition.

- The protein content of the feed contains the largest pool of organic nitrogen, and therefore will dominate the predicted ammonia production. Therefore, more experimental effort should be applied to the measurement of the protein fraction, or alternatively the organically bound nitrogen (Kjeldahl nitrogen) of the ABR feed for the accurate prediction of effluent ammonia concentrations.

- The net rate of treatment is dependent on the active biomass concentration within the reactor. The amount of biomass in turn is a complicated function of substrate availability, growth rate, and the flow rate in the reactor. The flow rate is particularly important since it determines the amount of biomass that is washed out of the reactor by entrainment in the liquid flow. The Monod kinetics calculated in this exercise apply specifically to the growth and flow conditions tested, and will have limited application outside of these conditions: significantly higher flow rates will result in lower biomass retention, or preferential retention of specific micro-organisms with faster settling rates, and different kinetic responses, and similarly, slower flow rates will also affect the kinetic behaviour of the biomass. While certain of these effects will have been captured in the determined Monod constants for the Kingsburgh case, this is by no means an exhaustive description of treatment rate under different flow conditions.

7.4.5 Predicting ABR performance for different feed characteristics

The steady-state model calibrated using the 22 and 42 h operating data was applied to a range of wastewater strengths and compositions to demonstrate the potential of the ABR for treating wastewater under different conditions. It is not possible to predict what the effects of changing the ABR dimensions on actual reactor performance using the steady-state model since the steady-state model has no input variable that could describe such changes. The only flow related variable is the average retention time R, which will determine the extent of treatment obtained. These results can be understood to apply to a reactor of similar proportions to the pilot ABR, but varying size, provided the hydraulic properties and overall retention time are similar to the pilot ABR. It must also be remembered that the effluent COD here is expected to consist mostly of wasted sludge and inert material, whereas in reality, some of the effluent COD would consist of biodegradable material, while the sludge continues to accumulate. Table 8.4 presents the wastewater characteristics used as inputs to the model, as well as the predicted effluent characteristics generated by the model.

The first case, Case A is regarded as a base case, and is similar to the characterisation used to model operation at Kingsburgh. In each subsequent case, only one value, or idea has been changed from case A, so that the individual effects of each variable can be observed. The input ammonia concentration has no effect on the outputs of the steady-state model, except for the output ammonia concentration which will change by precisely the same amount as the input ammonia. Consequently, a case investigating effect of input ammonia concentration is essentially trivial. Subsequent cases show the effect of wastewater strength (Case B1 and B2), retention time (Case C1 and C2), influent alkalinity (Case D1 and D2), influent pH (Case E1 and E2) and carbohydrate/lipid/protein compositions (Case F1 and F2).

Case A gives similar outputs to those seen in the 42 h operating period of the pilot ABR. From Case B, it can be seen that changes in the strength of the wastewater will result in proportional changes in the effluent COD, alkalinity and ammonia values. Increasing the output alkalinity in this way also causes a slight increase in outlet pH value.

Case C varied the retention time. Very little change was observed in any output values. However, this is a complicated function of the limited calibration, and does not match expectations of how the reactor would perform under these conditions. The relationship between COD removal and retention time was investigated mathematically, and it was found that COD removal is predicted to be essentially constant above 90% for retention times exceeding 23 h, but below 23 h, biomass is rapidly washed out, and total failure is predicted to occur at around 21.5 h. Although this type of relationship will exist for the ABR, washout is expected at retention times that are considerably lower than 20 h. The poor predictions are because the steady-state model describes operation for a CSTR, which shows higher, and more dramatic washout than the ABR. Therefore predictions of operating performance at retention times different to those investigated are not expected to be reliable.

Case D shows the effect of changed feed alkalinity. The calculated effluent pH is strongly affected by the feed alkalinity. The steady-state model is unable to model pH inhibition effects, and therefore, this does not appear from these data to be a significant

factor in the performance of the ABR; however, the increased pH value results in anaerobic digestion in a far more stable pH range. At higher pH values, the system is less susceptible to souring incidents, and better methanogenesis rates are obtained.

Case E shows that the feed pH value has very little effect on reactor performance.

Case F shows the effect of different carbohydrate, lipid and protein feed compositions on the ammonia and alkalinity generation during anaerobic digestion. The alkalinity generation potentials of the three components are significantly different, and are in the order: protein > lipid > carbohydrate. This means that, provided all other measurements are sufficiently accurate, a reasonable estimation of protein content can be obtained from the ammonia generated in digestion, while the relative ratios of lipid and carbohydrate can be estimated by regressing alkalinity production against alkalinity measurements. The steady-state model assumes that all three components are degraded to the same extent.

7.4.6 Conclusions drawn from the steady-state modelling of the ABR

The data obtained from 22 and 42 h steady-state operation were incorporated into a steady-state model modified by the differentiation of feed into carbohydrate, lipid and protein from the steady-state model presented by Sötemann et al. (2005). A good match between measured and calculated output conditions was obtained, despite the fundamental model structure being inappropriate for the plug flow-like behaviour of the ABR. The kinetic parameters obtained from the model are not expected to describe the reactor response well, particularly as only two operating points were used in the regression. Consequently, although the model was able to provide good insight into how changes in feed characteristics affect reactor performance, the prediction of performance at different retention times is probably not accurate.

A scenario analysis was performed in which the effect of organic strength, alkalinity, pH and composition of the wastewater, and retention time of the reactor were varied. It was seen that the feed alkalinity had the largest effect on calculated pH values, while organic strength and feed composition affected pH less. The calibration used indicated that retention time did not have a significant effect on the calculated pH values. It can be seen that for a constant feed composition, the pH values found in the reactor (under conditions where hydrolysis is the rate-limiting step) will be a function of alkalinity production defined by the extent of COD reduction.

It is concluded that for the hydrolysis-limited case, *the alkalinity, and alkalinity generation potential are the most important variables for maintaining reactor stability*. Furthermore, where low pH values may be resulting in pH inhibition of methanogenesis, increasing alkalinity will also result in improved COD reduction by causing an increase in the rate of methanogenesis.

Table 7.4: Wastewater strength and composition matrix and expected ABR effluent in each case predicted by steady-state anaerobic digestion model of Sötemann et al. 2005 calibrated using operating data from the pilot ABR located at Kingsburgh WWTW

Case	Model Inputs(Influent characteristics)									Predicted Outputs (Effluent characteristics)				
	R[1]	Total COD	VFA[2]	Inert	C[3]	L[3]	P[3]	Alk[3]	NH_3	pH	Total COD	Alk	NH_3	pH	CH_4
	[d]	mg/ℓ (COD)	% of inf. COD	% of inf. COD	% of inf. COD	% of inf. COD	% of inf. COD	mg/ℓ ($CaCO_3$)	mgN/ℓ	-	mg/ℓ (COD)	mg/ℓ ($CaCO_3$)	mgN/ℓ	-	mgCOD/ ℓ inf.
A	40	7004	5	8	25	50	12	200	35	7.0	1285	258	44	5.85	572
B1		1000									179	283	47	5.89	821
B2		500									93	241	41	5.83	407
C1	44										126	258	44	5.85	574
C2	36										130	258	44	5.85	570
D1								400			128	456	44	6.1	572

[1] Mean hydraulic retention time, approximating sludge age for the steady-state model

[2] Volatile fatty acid portion of feed COD, measured in terms of acetic acid COD equivalents

[3] C = Carbohydrate; L = Lipid; P = Protein; Alk = Alkalinity; inf. = influent

[4] Model inputs for cases B to F are the same as for the base case A, except where explicitly shown

[5] Numbers in italics are identical to values obtained for the base case.

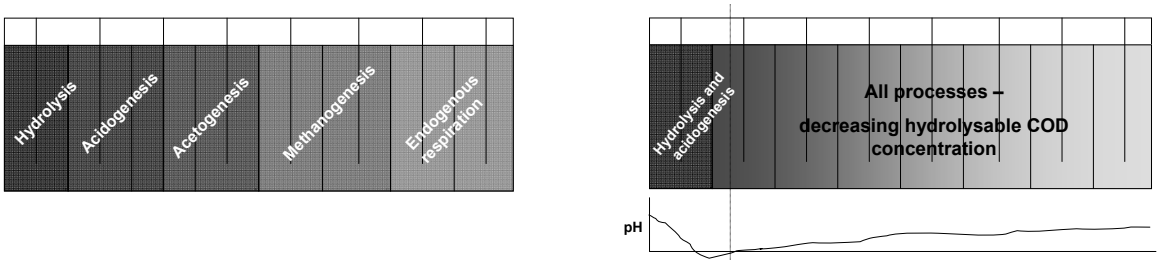
D2						100		128	158	44	5.64	572
E1							7.5	128	258	44	5.85	572
E2							6.5	128	258	44	5.85	572
F1		45	30	12				128	245	40	5.83	572
F2		30	12	45				128	308	58	5.93	572

8 DISCUSSION

This project has consisted of a number of phases that have been presented in Chapters 3 to 7, including design and construction of a pilot ABR (Chapter 3), operation and data analysis of the pilot reactor (Chapter 4), microbiological studies of the sludge from the pilot reactor compartments (Chapter 5), Community water use and wastewater characterisation studies (Chapter 6) and mathematical modelling of pilot ABR operation (Chapter 7). In addition to these studies, a number of smaller projects were undertaken to extend the project team's knowledge of issues relating to water and sanitation in low income South African communities. In this chapter, the collected knowledge from all of the phases of this project is gathered together to provide a basis for designing an ABR-centred sanitation system for decentralised community sanitation. In the first section, a hypothesis of anaerobic digestion of domestic wastewater in a baffled reactor configuration is presented, including a description of how selected factors affect the overall process. This is followed by a comparison between the performance of an ABR and a conventional septic tank (excluding soak-away). A brief description of the use of ABR effluent in irrigation follows. Finally, a proposal for an integrated system for sanitation in a small community is presented.

8.1 HOW DOES THE ABR WORK?

The initial concept of ABR operation was based on previous work using an ABR to treat soluble high strength or toxic industrial effluents (Bell and Buckley, 2003). For a high strength, soluble feed, the ABR has been shown to develop a thick granular sludge that exhibits considerable differentiation of microbial communities in the different compartments of the reactor. Furthermore, under these conditions, methanogenesis appears to be the rate limiting step of the anaerobic digestion process. It was hypothesised that an ABR treating domestic wastewater would exhibit similar properties. Figure 8.1 (a) is an example of how the sanitation ABR was expected to function.



(a) (b)

Figure 8.1: (a) Initial concept of ABR process showing spatial separation of anaerobic digestion sub-processes and (b) Evolved concept of hydrolysis-limited ABR operation showing acid production and decrease in pH value in the first compartment, and hydrolysis limited digestion through remainder of compartments.

However, it has been shown in Chapters 4 and 5 that the pilot ABR treating domestic wastewater from middle income origins did not develop populations that were markedly different from one compartment to the next. Specifically, methanogenic micro-organisms were found in the first compartment in larger concentrations than in later compartments. The combined physico-chemical, microbiological and modelling studies have indicated that the dynamics of digestion are somewhat different to the original concept. Figure 8.1(b) provides a visual representation of the new hypothesis of anaerobic digestion in an ABR treating domestic wastewater.

8.1.1 Hypothesis of digestion in an ABR

A hypothesis of the process interactions of anaerobic digestion in an ABR is presented, considering a range of mechanisms including feed solids retention, acid formation, distribution of microbial species and methane production.

8.1.1.1 Feed solids retention

Compartment 1 (and to a certain extent compartment 2) act as solids retention chambers. Solids and scum that enter the reactor with the influent wastewater are retained at the beginning of the reactor either by settling out in the bottom of compartment 1 or floating on the surface of the compartment. A small fraction of the solids in the influent will be entrained in the liquid flow and pass on to subsequent compartments. New solids entering compartment 1 are trapped in the existing sludge mass.

The high solids and scum concentration in compartment 1 equates to a high particulate substrate concentration. Micro-organisms proliferate in the feed solids as they provide both substrate and a support medium for retaining active biomass.

8.1.1.2 Acidification in the first compartment

Hydrolysis and acidification of feed solids in the early compartments occurs rapidly, resulting in acid production in excess of the methanogenic population's ability to metabolise it. Some accumulation of acid occurs, and a decrease in pH value is observed. Increased concentrations of soluble COD and VFAs relative to the feed and subsequent compartments are seen.

8.1.1.3 Mechanism of COD transport between compartments

Passage of substrate from one compartment to the next from compartment 2 onwards occurs as a result of entrainment, flow of soluble components with the flow, and to a certain extent, positive displacement, once compartments become full of sludge.

8.1.1.4 pH dynamics after compartment 1

Since a large amount of the undegraded solids remain in the earlier compartments, there is less hydrolysable material in compartment 2 onwards than in compartment 1, and the rate of acidification is therefore lower. Removal of acids by methanogenesis is able to occur at similar rates to acid production, so that acid concentrations decrease.

Furthermore alkalinity generation by anaerobic digestion increases the buffering capacity, and an overall rise in pH value is observed.

8.1.1.5 Distribution of microbial species in compartments

Since particulate and soluble organic substrate is transported from one compartment to the next, hydrolysis occurs in each compartment, thereby generating substrate for subsequent sub-processes of anaerobic digestion. Therefore, all micro-organisms types will be found in all compartments.

8.1.1.6 Mechanism and extent of degradation after compartment 1

The amount of treatment of entrained particulate or colloidal organic substrate will depend on the diameter of the particles or colloid; these are hydrolysed and acidified from the outside in. If the relative rate of movement of the hydrolysis front along the shortest route to the middle of the particle or colloid is slower than the relative movement of the particle or colloid through the reactor, than part of the particle or colloid will wash out of the ABR untreated. If not, the particle or colloid will be completely hydrolysed before exiting the reactor.

Acetogenic and methanogenic micro-organisms scavenge acidogenesis by-products as they are produced, resulting in a fairly constant and low steady-state concentrations of anaerobic digestion intermediates at all points within the reactor.

The *extent of treatment*, defined as the amount of COD converted to methane relative to the total *biodegradable* COD content of the feed wastewater, therefore increases as the time that the COD is retained in the reactor increases. For a specific lump of COD, this means that the extent of treatment of will increase as it progresses through the reactor. For a specific wastewater, the average extent of treatment for all organic components will increase when the hydraulic retention time increases.

8.1.1.7 Waste biomass

A fraction of the biomass generated by growth on the organic substrate will be entrained in the liquid flow and exit the reactor.

Biomass growing on the organic substrate will accumulate in the compartments at a rate equal to the difference in growth rate and biomass wash-out rates. Presuming that the biomass wash-out rate is not greater than biomass growth rate, the amount of biomass in each compartment will gradually increase until the compartment sludge overflows into the next compartment or the effluent.

8.1.1.8 Soluble inert

Any soluble inert material from originating from the feed, or by-products of anaerobic digestion will exit the ABR with the effluent.

8.1.1.9 Inert solids

Inert solids in the feed, or generated by the anaerobic digestion process that will either be entrained in the liquid flow and wash out of the reactor, or accumulate at the bottom of the compartments.

8.1.1.10Alkalinity and ammonia generation

Alkalinity and ammonia are generated by anaerobic digestion; therefore concentrations of these species will increase gradually through the reactor.

8.1.1.11Methane production

COD is converted to methane in all compartments of the ABR; therefore methane production will be observed in all compartments. After compartment 1, overall digestion will be limited by the rate of hydrolysis, and therefore methane production will occur at approximately the same rate as hydrolysis.

8.1.2 Factors affecting anaerobic digestion in a baffled reactor

Two factors have been identified as critical in determining how fast anaerobic digestion occurs in an ABR, the amount of solids retained (including biomass) and the alkalinity of the reaction medium.

8.1.2.1 Mechanism of solids retention

Analysis of analytical data from operation of the pilot ABR (Chapter 4) indicates that the major factor controlling the overall performance of the ABR is the solids retention in the compartments. Establishment of a concentrated and stable biomass population in each compartment is the key to achieving good COD reduction and maintaining balanced anaerobic digestion. The retention of solids in each compartment is as a result of settling of solids in the up-flow region of the compartment (Figure 8.2).

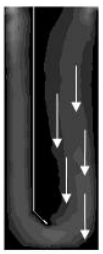


Figure 8.2: Fluid flow and solids settling in a compartment of an ABR. Regions of low flow are dark, while regions of higher flow are pale. Fluid flow is down on the left of the baffle, and up on the right. White arrows indicate solids settling

The amount of solids retention depends on the up-flow velocity, and the settling rate distribution of the solids. At any time in any compartment, the settling velocity distribution of the solids in that compartment will be less than the maximum sustained up-flow velocity that that compartment has experienced in its recent history. This is

because all solids with a settling velocity less than the maximum sustained up-flow velocity will have been carried into the next compartment. Peaks in up-flow velocity associated with sudden surges will not result in the loss of all solids with lower settling velocities being washed out since there is a time lapse between solids being entrained in the liquid flow and reaching the top of the standing baffle and thence being washed out. Therefore a characterisation of the settling velocity distribution of sludge from a compartment will only be valid for that compartment at the specific point in time that it was performed.

Factors affecting up-flow velocity

The average up-flow velocity in each compartment is a function of the *flow length* of the reactor i.e. the average distance that a particle must travel between inlet and outlet of the reactor, and the *hydraulic retention time*. Since wastewater is not generated at a fixed rate, there will be variations in inlet flow rate resulting in flow oscillations as well as occasional flow surges. The baffled design of the reactor will dampen oscillations and surges to a certain extent, but nevertheless, up-flow velocities will depend on the influent flow rate at any time.

- *Compartment dimensions:* For a fixed number of compartments, tall or long reactors will have a longer flow length than wide reactors. Similarly, for fixed external dimensions of a reactor, having a greater number of compartments will result in a longer reactor flow length than a few compartments. Therefore, to reduce up-flow velocity compartments should be as wide as is practical, and the minimum number of compartments required to ensure adequate contact between biomass and wastewater should be implemented.

- *Hydraulic retention time:* Obviously the larger the net flow to the reactor (i.e. the smaller the hydraulic retention time) the larger the up-flow velocity will be. For fixed reactor dimensions, by reducing the flow to the reactor, longer contact times will be achieved, and lower up-flow velocities and therefore better solids retention will be achieved.

- *Surge/cyclical flows:* The amplitude and period of in-flow oscillations for a fixed average hydraulic retention time will affect the overall solids retention in the reactor. Garuti et al. (2004) showed that for high flow rates in short bursts, the overall solids retention in an ABR was better than for lower flow rates but longer feeding periods (the two scenarios provide the same average flow to the ABR). In an ABR with a controlled dosing system, this implies that an increased number of short feeding cycles should be implemented than fewer long feeding cycles. In a gravity fed ABR where extreme oscillations of flow are expected, it may be necessary to implement a balancing tank up-stream of the reactor.

Factors affecting solids settling properties

The settling properties of the solids in the compartments of an ABR depend on the physical properties of the sludge, specifically shape and density. For a sanitation application, it is not envisaged that the operator of an ABR system would have much control over these sludge characteristics.

- *Granulation:* The extent of granulation, i.e. the relative proportion of micro-organisms bound in pellets or granules, and the size of these granules will be a big factor in determining the average settling characteristics of the sludge. Well-established reactors may exhibit a greater degree of granulation than new systems due to maturation and acclimation of the sludge. Granulation also depends on the relative availability of substrate, constrained by concentrations and diffusion rates.

- *Fixation:* many micro-organisms will attach to walls, flocs, films or other support media. Micro-organisms attached to a biofilm on the walls or baffles of the ABR will be retained under much higher up-flow velocity conditions than suspended micro-organisms.

- *Gas production:* The production of gas may have two significant effects on solids in the reactor. Firstly flotation of solids on gas bubbles will lead to increased solids entrainment, even when the actual solid density should result in settling. Secondly, gas production has been shown to have a mixing effect that hinders sedimentation of solids, and may result in breaking up of flocs and granules. This latter effect is not usually significant in low strength anaerobic digestion applications (Barber and Stuckey, 1999).

8.1.2.2 Effect of pH and alkalinity

Analysis of the pilot ABR revealed pH values in a range which could result in methanogen activity. Low pH values result from acid production that is not matched by acid removal in a system with little pH buffering capacity. There are several chemical buffering systems in wastewater treatment including $PO_4^{3-}/HPO_4^{2-}/H_2PO_4^{-}/H_3PO_4$ system equilibria, NH_3/NH_4^{+} equilibrium and $CO_3^{2-}/HCO_3^{-}/H_2CO_3$ equilibria. Of these systems, the last, the carbonate system has the largest effect on the total buffer capacity of the system (Speece, 1996). Measured bicarbonate alkalinity in the pilot ABR was low relative to conventional anaerobic digestion applications during all operating periods.

In order to prevent the pH value dropping to inhibitory levels, it is necessary to either eliminate VFA accumulation, or to increase the buffer capacity of the system. The steady-state mass balance calculations performed in Chapter 7 demonstrate that the steady-state effluent pH values, representing the maximum pH value achievable for good digestion of feed at the specified feed conditions, is low (near 6). This means that there is not enough alkalinity generation potential as a result of COD degradation in the feed to provide sufficient buffering to maintain the system at a pH value that is not inhibitory to methanogens.

In the case studies presented in Figure 7.4, the only parameter that can be varied to achieve a significant change in effluent pH value is the influent alkalinity. Influent alkalinity is a function of local water hardness, which in turn depends on the geology of the area that the water supply is drawn from, and to a lesser extent the potable water treatment process. It also depends on the amount of alkalinity added to the water by the user. None of these factors can be practically altered by the system operator. However, it may be feasible to achieve sufficient buffering in the ABR by controlled dosing of alkali.

A possible alkali dosing procedure would be addition of solid $CaCO_3$ in the first compartment. These could be supplied in a permeable bag such that dissolution and diffusion of alkalinity into the first compartment could proceed at a constant, slow rate. The bag could then be lifted out at intervals to ascertain how much $CaCO_3$ remains. Such a measure would have cost and maintenance implications. Further, testing would have to be undertaken to define dosing rates in terms of solid surface area that should be used, to ensure that the alkalinity and pH value do not increase too much if dissolution rates are too fast.

The benefits of alkalinity control would be:

- *Enhanced hydrolysis rates:* since the rate of hydrolysis has been found to be the determining factor in overall extent of treatment, enhancing hydrolysis rates should improve overall treatment rates;

- *Enhanced methanogenesis rates:* higher methanogenic rates would result in smaller pseudo-steady-state VFA concentrations, and more stable gas production;

- *Greater pH buffer capacity:* increasing the buffer capacity would increase the overall process stability to shock organic loads since there would be a greater ability to absorb excess VFA production. Greater acid accumulation during peak organic loading would be possible before pH values decrease sufficiently to result in methanogenesis inhibition;

- *Corrosion of reactor walls:* if concrete or cement is used in the construction of the ABR, low pH values would result in corrosion and dissolution of the structure. Increasing the alkalinity and therefore pH value could alter the requirements for materials that could be used in the construction of the ABR.

8.1.3 Benefits of ABR system in sanitation

Anaerobic treatment of domestic wastewater is regarded as many as being more sustainable than aerobic systems, due to the decreased aeration requirement (Jetten et al., 1997) as they are able to achieve good removal of biodegradable substrates, have acceptable efficiency levels, produce biomass as a reusable energy source, and have low running costs and environmental impact through not implementing forced aeration. The use of anaerobic treatment on a household or community level has the following benefits:

- It has low infrastructural cost relative to aerobic systems for the municipality

- It has low energy and operating cost relative to aerobic systems

- It is not dependent on energy supply and should be able to operate on low water supply.

- Nitrogen and phosphorus are conserved, and may potentially be used in agricultural initiatives.

- The system has the potential to be household or community maintained.

- There is a potential to reuse energy from biogas.

The advantages of the ABR configuration over other anaerobic digesters are related to its solids retention characteristics:

- High solids retention equates to high biomass retention. It is possible to maintain high biomass loads without having a separate solids retention system.

- Since biomass loads are maintained, high anaerobic treatment rates can be achieved, meaning that the overall treatment volume required for a certain application will be less than most other configurations.

Although all compartments have micro-organisms that are capable of performing the all of the sub-processes of anaerobic digestion, it is individual genera that are better suited to compartment conditions will establish in each compartment. For example, significant acetate concentrations only occur in compartment 1. In both the SEM study and the FISH study reported in Chapter 5 *Methanosarcina* were observed in compartment 1, decreased in subsequent compartments and were not observed from compartment 5 onwards. Conversely the SEM study found significant amounts of *Methanosaeta* in all compartments except the last. *Methanosaeta* will outcompete *Methanosarcina* at low acetate concentrations while the reverse is true when acetate is not in limiting supply.

A staged configuration provides a certain amount of protection against overloads and souring since flow velocities are dampened by the baffles. The baffled configuration has also been reported to provide higher stability than mixed configurations, and to recover from upsets more rapidly (Barber and Stuckey 1999).

8.1.4 Limitations of anaerobic digestion in sanitation

Even concentrated domestic wastewater is considered to be low strength in anaerobic digestion terms, as the overall alkalinity generating potential of the wastewater is low. Consequently, without additional alkalinity enhancement, poor pH buffering is likely to be observed, and therefore the system may be susceptible to souring.

Anaerobic digestion does not remove ammonia or phosphorus. In fact, ammonia may be generated as a result of biodegradation of organically bound nitrogen. It is therefore not able to achieve an effluent standard that is acceptable for discharge to surface or ground water. This necessitates an aerobic post-treatment step, or a disposal method that will not jeopardise surface or ground water resources, such as constructed wetlands, ponds or irrigation. Consequently, where there is insufficient absorption capacity in the surrounding soils for an expected wastewater flow.

Finally, but very importantly, anaerobic treatment alone is not able to reduce the pathogen load of the wastewater to a level where it is no longer considered to be a hazard to human health. In the pathogen deactivation study (Chapter 4) it was shown that up to 2 log removal of coliforms could be obtained, while effluent helminth egg observations reduced to 17 eggs/ℓ. However, these values remain above recommendations for reuse of effluent in irrigation and will constitute a risk of infection, although significantly reduced from that of the influent wastewater, to anyone who comes into contact with the effluent. This places severe restrictions on the reuse

potential of the wastewater without some additional polishing step. Appropriate means of disinfecting the water could be chlorination, membrane filtration or treatment in constructed wetlands.

8.1.5 Key parameters in ABR design

The design objective is to maximise the amount of contact time between suspended or dissolved contaminants and the biomass. This is achieved both by maximising the hydraulic retention time (the treatment time) and the load of biomass, determined by solids retention, within the constraints of space and capital cost. Solids retention is achieved by minimising the velocity of liquid on the up-flow side of each compartment since solids loss is through carryover of slow-settling solid particles when the up-flow velocity exceeds the particle settling velocity. Based on this design objective, the following key parameters in the design of an efficient ABR have been identified:

- Mean hydraulic retention time
- Number of compartments
- Design up-flow velocity
- Upflow-to-downflow area ratio
- Compartment length-to-width ratio
- Hanging baffle clearance
- Reserve Capacity

Each of these parameters is discussed in detail in the recommended design guidelines, Chapter 9.

8.2 BENEFITS OF THE ABR OVER A SEPTIC TANK

Anaerobic digestion in on-site or decentralised treatment of domestic wastewater is not a new concept. Many millions of people world-wide have used septic tank and soak-away systems successfully for years. To place this project into perspective, it is necessary to ask the question *what is the advantage of an ABR over a septic tank, or other on-site or decentralised system?*

8.2.1 Performance

The major advantage of an ABR is that it treats wastewater at a faster rate per unit volume than a septic tank. Depending on the system loading, this has two possible implications; either (i) the treated effluent quality will be better from an ABR than from a septic tank or (ii) the volume of an ABR required to treat a certain load of wastewater to a certain standard would be smaller than the equivalent septic tank. A cursory examination of the compiled septic tank performance data (Chapter 2) and the performance of the pilot ABR (Chapter 4) suggests that an ABR will remove at least 50% more COD than a septic tank at the same hydraulic retention time.

A septic tank retains suspended solids from wastewater in the scum and sludge layers and releases partially treated, clarified effluent to a secondary treatment system. Retained solids sediment, and undergo partial biological stabilisation by conversion to biogas. Very little treatment of soluble components occurs as the liquid phase bypasses the biomass on the bottom of the tank.

The ABR works under similar conditions to a septic tank but it *increases contact between biomass and wastewater* by forcing liquid to flow through biomass beds with each pass under the hanging baffles. In this way there is a biological filtering effect in which solid components are physically retained by settling, and liquid components are removed by adsorption and consumption. It is expected that the removal of pathogen indicator organisms will similarly be improved as a result of improved wastewater biomass contact, although there is insufficient data to support this theory.

The *separation of microbial consortia* in an ABR is well reported (Barber and Stuckey, 1999), although the evidence for this in low-strength applications is limited. Reported advantages of this phenomenon include increased disinfection rates as a result of low pH in early compartments of the ABR enhanced degradation rates through selection of microbial consortia optimised to compartment conditions, and increased resilience to organic shocks since micro-organisms in later compartments are shielded from the full impact of the shock. Based on the experience with the pilot ABR in which an unexpected organic overload caused the pilot ABR to go sour, it could be seen that the mechanism of recovery from the upset was similar to a plug-flow scenario. Acid residues and untreated organics originating from the shock load are washed out of a compartment at a much faster rate than would be the case in a mixed system while sufficient biomass is retained to achieve a full recovery soon after the event.

No such microbial separation exists in a septic tank system, and therefore treatment rates, resilience to and recovery from shock organic loads and pathogen deactivation will be less in a septic tank than in an ABR.

The baffled configuration of the ABR also has *hydraulic advantages* over the simpler construction of the septic tank in that it is able to slightly damp variations in flow. The biggest advantage of the baffles from a hydraulic perspective is that they prevent short-circuiting of liquid flow between the inlet and outlet, and bypassing of the sludge beds, as occurs in septic tanks. The disadvantage is that the addition of baffles significantly increases the mean flow path length between the inlet and outlet for a fixed hydraulic retention time, resulting in higher up-flow velocities, and therefore greater risk of biomass washout.

Finally, the application of septic tanks in high strength applications (such as systems that take the majority of their feed from a toilet) has been shown to be limited by the rate of accumulation of toilet paper or newspaper (Du Pisani., 2002). It is possible, although not proven, that this effect would be reduced in an ABR since forcing of liquid flow through the sludge bed prevents the development of quiescent inert layers. Beeharie (Ekama, private correspondence, 2004) has shown that toilet paper is biodegradable, and therefore it is suggested that with improved biological contact, it would exhibit a faster degradation rate, reducing blocking.

8.2.2 Management

The ABR is proposed as part of a decentralised system for treating domestic wastewater from a number of homes. There is a big difference between a single home unit and a community unit because in the former case, the ownership and responsibility of the system is clearly defined, while in communal system, there may be political and social complications that could lead to neglect or abuse of the system.

In a situation where the sanitation system is maintained and operated, or serviced by the local municipality, the ABR system has the advantage that fewer units need be installed than individual septic tanks, and therefore the overall maintenance requirements of supplying sanitation to a community may be less. However, as each unit can treat a higher flow than a septic tank, there is a greater potential for disaster in the case of a process or structural upset.

On a household level, an ABR should provide all the advantages of a septic tank, but require less space, and a deliver a reliable improved quality of effluent to the soak-away system. The latter should result in improved soak-away performance, with a lower incidence of soak-away problems such as seepage (effluent reaching the surface in the soak-away or a little distance away) or clogging with suspended solids, two of the common problems associated with soak-aways (Wright, 1999).

From a water and wastewater management perspective, the biggest advantage of the ABR over a septic tank is the improved and more reliable quality of the treated effluent. The effluent has a low COD content and is not offensive in appearance or smell. It has significant concentrations of nitrogen and phosphorus which may be used as fertiliser replacement in urban agriculture.

8.3 USE OF ABR EFFLUENT IN IRRIGATION

During the course of this project, several smaller projects were undertaken by small groups of undergraduate and postgraduate students. In most cases the scope of the projects was too small to obtain statistically significant data, but they contributed to the project team's understanding of some of the issues relating to the application of the ABR in community sanitation. One of these projects investigated the effect of using ABR effluent on the growth performance and microbial quality of food crops, compared to a positive control (nutrient solution) and negative control (tap water). A summary of this study is presented in Appendix A2.

Statistical significance of differences was rarely achieved, but there was a uniform trend with the negative control (tap water) yielding the poorest growth and the positive control (nutrient solution) yielding the best growth. Growth of plants irrigated with ABR effluent most closely resembled that of plants irrigated with nutrient solution, indicating that ABR effluent indeed appears to hold potential as a fertilising solution. Microbiological data was inconclusive, although plants watered with ABR effluent did not show significantly worse microbial contamination than plants irrigated with tap water and nutrient solution.

8.4 IMPLEMENTATION OF AN ABR-CENTRED SANITATION SYSTEM

This project has investigated the performance of a pilot-scale ABR treating domestic wastewater, and has developed a sound theory of how the baffled configuration affects process performance and microbial community dynamics. The application of this research is to develop a system that may be implemented in community sanitation. No community installations were tested during the course of the project, as it was only in the final stages of the project that a complete understanding of the system was developed.

The proposed water management concept has 6 stages: The first stage is wastewater generation stage, which is linked to potable water supply level and toilet superstructure design. The second stage is the blackwater collection system which in a community would be a condominium-type sewer, and in an institution, an appropriately designed toilet block. An appropriate greywater collection and reuse system is also required. The pre-treatment stage (stage 3) includes a solids trap and the ABR itself, and the fourth stage, a post-treatment step in the form of a constructed wetland, or membrane unit. The reuse of the effluent generated for irrigation purposes is the fifth stage of the process. Finally, it is necessary to have an inspection system involving both the users and the appropriate municipal authority to monitor system maintenance and effluent microbiological quality.

8.4.1 Stage 1: Wastewater generation

In domestic use, the wastewater with the highest COD and pathogen load originates from the toilet and the kitchen sink. If greywater is reused in a domestic context, i.e. only householders are exposed to their own greywater, the risk of the spread of disease as a result of greywater reuse is minimal. It is therefore only necessary to have an active treatment process for toilet and kitchen sink water. In a wastewater treatment and reuse context, separation of urine from faeces has no real advantage since the nutrients in the urine are eventually recovered, without requiring separate storage, dilution or handling.

The ABR sanitation concept will probably be targeted at households with semi-pressure rooftank water supply; lower levels of water supply are not sufficient to operate a wet-core (flushable) toilet, and the higher level, full mains pressure, may result in a wastewater that is too dilute, resulting in hydraulic overloading of the ABR. Full-pressure water users will in general be in a higher income bracket, where full water-borne sanitation can be afforded by the consumer, and therefore will be the only sanitation level acceptable to most people in this category.

The toilet superstructure and flushing procedure will require careful design since the amount of water used for flushing will affect the ability of small bore reticulation to transport toilet contents away. From a treatment perspective, low flush volumes will produce concentrated wastewater, resulting in efficient treatment, however, the smaller the flush volume, the bigger the risk of sewer blockage. In a situation where an effective greywater separation system is in place, a conventional toilet with a flush volume of 5 to 7 ℓ would be acceptable, with a 0.5 ℓ pour-flush arrangement for urine-only flushing. Where large amounts of greywater are likely to enter the treatment unit, it is recommended that a pour-flush toilet is installed.

8.4.2 Stage 2: Shallow sewer

A shallow sewer is a small bore sewer that connects a group of householders within a micro-catchment to a municipal sewer or local treatment process. Local studies (Eslick and Harrison, 2004) have shown that shallow sewer systems can work effectively, at considerably lower capital cost than conventional sewer systems, but that problems arose as a result of a difference between the level of service expected by the community, and that offered by the small bore sewer system. The system envisaged here would be on a considerably smaller scale with around 10 households connected to a treatment unit, but would nevertheless be subject to a wide range of difficulties at the user/technology interface.

8.4.3 Stage 3: Pre-treatment – screening and ABR unit

If a community owning an ABR system has a high level of commitment to maintaining the system, it may not be necessary to have a screening unit before the ABR. This would reduce the system maintenance requirements. However, in most instances, a screening and degritting unit would be required to limit the amount of non-biodegradable material entering the ABR.

8.4.4 Stage 4: Polishing step

After ABR treatment, the primary objective is to remove pathogens from the effluent so that it is safe to reuse. Both membrane filtration and the construction of a wetland would effectively remove pathogens from the effluent.

8.4.5 Stage 5: Effluent reuse

The effluent from the polishing step will be safe to use for irrigation, and should additionally provide a significant amount of fertilisation as a result of nutrients present in the effluent. However the release of the effluent must be to a sufficiently large area such that contamination of the water table or nearby water courses is avoided.

8.4.6 Stage 6: Monitoring and maintenance

As with all services, if no monitoring or maintenance of the system is undertaken, it is likely to fail as a result of mechanical failure (blockages, leaks, damage to reticulation and reactor) or user neglect or abuse. Furthermore, the implementation of a visible monitoring and maintenance programme will create an awareness of the system among users that it is susceptible to problems if not managed and cared for right from the toilet bowl to the secondary treatment process. A description of a maintenance programme is presented in the guidelines chapter (Chapter 9).

Individual homeowners are responsible for maintaining their own connection and reticulation on their property. Maintenance of the shallow sewer and screening unit would require a community elected or employed representative responsible for daily and weekly checks. Weekly removal of screenings, and inspections 3 or 4 times a year by the municipality will improve the chances of the system operating successfully.

8.4.7 Pitfalls of the proposed system

- Community education and participation are essential at each of the six stages in order that the system is not upset through negligence or abuse. Within the eThekwini Municipality, there has not been any indication that the kind of participation required will be forthcoming.

- A post treatment polishing step using either membrane filters or a constructed wetland will effectively eliminate pathogens. However people may be exposed to untreated wastewater through failure of the shallow sewer and at the screening unit.

- The application of this system is also severely limited by the availability of land below the treatment process which is available for agriculture, or in the absence of an active land-use programme, is able to absorb the nutrient load without contaminating the water table or nearby streams.

9 GUIDELINES FOR THE DESIGN, OPERATION AND MAINTENANCE OF AN ABR TREATING DOMESTIC WASTEWATER

WRC project K5/1248 *The anaerobic baffled reactor for sanitation in dense peri-urban communities* has characterised the performance of a 3 000 ℓ pilot ABR treating domestic wastewater from middle-income suburbs at a municipal wastewater treatment facility in terms of contaminant removal and microbial community dynamics. Dynamic and steady-state models have been developed and tuned to assist in developing a theory of how the process works. A study characterising wastewater from a low-income peri-urban community was undertaken to identify differences between wastewater characteristics in communities where the ABR could find an application, and the wastewater that the pilot ABR was tested on. This chapter draws together the findings of the different phases of this project to develop a set of guidelines for designing ABR systems and predicting ABR performance on different qualities of wastewater.

It must be emphasised that the effluent of an ABR **will not meet General or Special limits for nitrogen and phosphorus** for discharge to surface or ground water, and, untreated, contains **hazardous concentrations of pathogenic micro-organisms.** This means that a secondary polishing step must be implemented before effluent can be discharged or reused in agriculture.

9.1 INTRODUCTION

The guidelines presented in this chapter define a system that will perform under expected community conditions, according to the information and scientific backing available to the project team at the time of preparing this report. The following sizes and calculations are for a wastewater pre-treatment system for a group of 10 low-income houses, with a rooftank or similar level of potable water supply.

Table 9.1: Assumed treatment cluster characteristics for low-income community sanitation

Type of wastewater	**Concentrated**	**Dilute**
Greywater Separation?	Yes	No
Wastewater return	40 %	80 %
Average household water consumption per day	342 ℓ	342 ℓ
No. of inhabitants per household	4	4
Average consumption per individual per day	85.5 ℓ	85.5 ℓ
Wastewater production per individual per day	34.2 ℓ	68.4 ℓ
No. of households in treatment cluster	10	10
Population equivalents in treatment cluster	40	40
Wastewater flow per treatment cluster	1368 ℓ/d	2736 ℓ/d

The group of houses and the treatment unit is termed the *treatment cluster*. Table 9.1 presents values that have been adopted as representative of typical low-income households based on information supplied by the eThekwini Municipality in March 2005, and the results of the water use survey presented in Chapter 6. Two types of wastewater are considered: (i) in the first case, greywater originating from bathing, washing and cleaning is not discharged to the sewer, but reused and disposed of on-site in the garden or an agricultural enterprise; and (ii) greywater and blackwater are discharged to the sewer and treatment system. According to the national census carried out in 2001, the average household size in eThekwini is 4.00 persons per dwelling (Statistics South Africa, 2005). Wastewater return in the case of greywater separation and reuse (concentrated wastewater) was estimated to be 40%, and a value of 80% was applied for the case where greywater is not separated (dilute wastewater).

Wastewater characterisations for concentrated and dilute wastewater are presented in Table 9.2.

Values for dilute wastewater are based on the findings of the community wastewater characterisation study (see Chapter 6). The 80th percentile COD concentration for the winter wastewater characterisation study was used. Carbohydrate and protein fractions of the total COD were estimated from average carbohydrate and protein fractions calculated from the wastewater characterisation study data (Chapter 6). The lipid fraction was calculated by difference. Values for concentrated wastewater are based on 175% organic content of the diluted wastewater. Values for middle income homes are drawn from an extensive survey of wastewater characteristics in households that use a septic tank system in the USA (USEPA, 2002), while carbohydrate, protein and lipid fractions used are the same as those regressed from the steady-state modelling exercise (Chapter 7) on Kingsburgh WWTP wastewater.

Table 9.2: Wastewater characteristics for concentrated and dilute wastewater from low and middle income communities

	Unit	**Dilute low income community**	**Concentrated low income community**	**Dilute middle income cluster**
Flow Rate	ℓ/d	2736	1368	2337
Flow in PE[1]	ℓ/d per capita	68.4	34.2	260
COD	mgCOD/ℓ	1090	1908	510
TKN	mgN/ℓ	176	308	44
NH_4^++NH_3	mgN/ℓ	30	53	6
PO4	mgP/ℓ	25	44	8
Alkalinity	mgCaCO3/ℓ	250	300	250

[1] PE: population equivalents

T. Coliforms	cfu/100mℓ	107	107.3	106.8
TSS	mg/ℓ	153	267	575
pH	-	8	8	6.2
Protein	% of total COD	9	9	12
Carbohydrate	% of total COD	14	14	24
Lipid	% of total COD	66	66	51
VFA	% of total COD	3	3	3

9.2 PRINCIPLES OF DESIGN

In engineering terms, an ABR functions as a series of mixed reactors, in which the biological catalyst, the biomass in each compartment is retained when the liquid flow passes out of the compartment. The first one or two compartments have the added function of retaining solids originating from the feed.

9.2.1 Design objective

The design objective is to maximise the amount of contact time between suspended or dissolved contaminants and the biomass. This is achieved both by maximising the hydraulic retention time (the *treatment time*) and the load of biomass, determined by *solids retention*, within the constraints of space and capital cost. Solids retention is achieved by minimising the velocity of liquid on the up-flow side of each compartment since solids loss is through carryover of slow-settling solid particles when the up-flow velocity exceeds the particle settling velocity. Low up-flow velocity can be achieved by either selecting a reactor geometry that has a short flow path for a specified hydraulic retention time (e.g. a low, wide reactor, or few compartments), or by reducing the flow to a specific unit, i.e. increasing hydraulic retention time.

Clearly, from a mechanical perspective, a reactor with few, low, wide compartments would provide the best solids retention for a specific hydraulic retention time. However, from a biological perspective, it has been shown that biomass acclimation to compartment conditions results in increased treatment rates, and resilience to shock loads (Barber and Stuckey, 1999). The greater the number of compartments, the greater is the contact between contaminants and biomass, as a result of flow being repeatedly forced through biomass beds. However, more compartments for a fixed reactor size increases the flow length that wastewater must travel between inlet and outlet of the reactor, and hence increases the flow velocity and thereby biomass carry-over, resulting in a lower steady-state biomass concentration within the reactor, and therefore lower overall treatment rates.

A good design must find a compromise between solids retention and biomass performance.

9.2.2 Extent of treatment

There is a limit to the extent of treatment that can be obtained under any conditions by anaerobic digestion alone. The extent of treatment is defined as *the ratio between the COD removal obtained and the maximum possible removal*, which in turn is determined

by the amount of inert (non-biodegradable) COD in the influent. In reality, the maximum theoretically possible COD removal will never be achieved by anaerobic digestion alone because the treatment rate decreases as the substrate availability decreases according to Monod kinetics, and poorly biodegradable COD will usually not have sufficient contact time in an economically feasible design to be completely degraded.

Given the sensitivity of sanitation to health issues, the extent of treatment could also be defined in terms of pathogen indicator organism removal. As with COD reduction, complete removal of all pathogens cannot be achieved without a physical separation process, since encapsulation and cyst formation protect many micro-organisms from deactivation by anaerobic processes.

Nutrient reduction (nitrogen and phosphorus) cannot be achieved by anaerobic digestion and therefore is not considered to be part of a definition of extent of treatment.

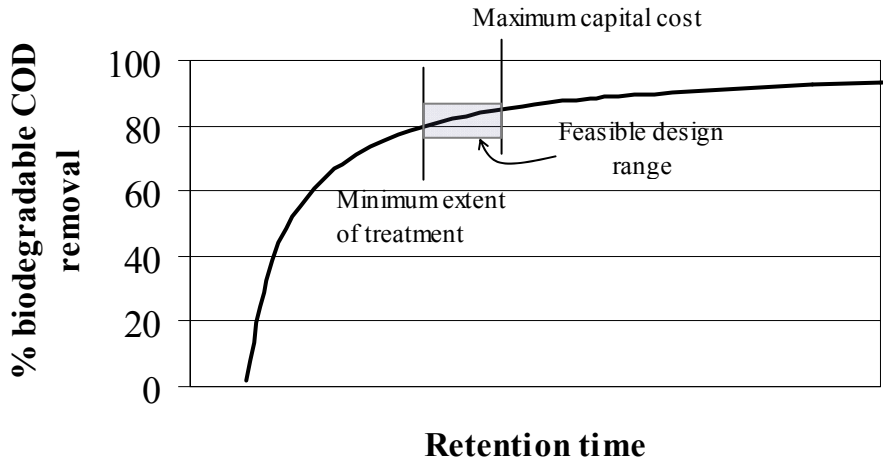


Figure 9.1: Relationship between % biodegradable COD removal (extent of treatment) and retention showing the feasible design range, based on the maximum capital cost and minimum acceptable extent of treatment

The relationship between hydraulic retention time and extent of treatment for a particular reactor geometry has an asymptotic form, as depicted in Figure 9.1. As the hydraulic retention time increases, so the extent of treatment increases. However, the capital cost of treatment also increases. It is therefore necessary to define a minimum acceptable extent of treatment and maximum cost of treatment, and select a reactor size that falls between the two. Figure 9.1 has no numbers shown for retention time since the relationship is not well characterised for the ABR, and depends on the influent COD concentration, alkalinity, and kinetic and stoichiometric constants.

A primary finding of this project is that some secondary treatment is required before disposal or reuse of the effluent from the ABR since it is unable to treat the wastewater to the required microbiological standards, by anaerobic digestion alone. In the interests of economy, a shorter treatment time (lower hydraulic retention time) then required for complete biodegradable COD reduction should be selected, with an accompanying

increase in effluent contaminant loads, since some polishing will occur in the secondary treatment process.

The characteristics of the secondary treatment process must also be considered when sizing the ABR since large amounts of suspended solids and biodegradable organic material in the effluent as a result of a lesser extent of treatment can seriously affect the functioning of the secondary treatment process.

9.3 CIVIL / MECHANICAL DESIGN

The physical design of the ABR is divided into those parameters that affect the up-flow velocity and solids retention (*design parameters*) and *secondary* design details.

9.3.1 Reactor design parameters

For a specific design flow (expected community wastewater generation) there are a number of key design parameters that must be selected to fix the overall design. These are:

- Mean hydraulic retention time
- Number of compartments
- Design up-flow velocity
- Up-flow-to-down-flow area ratio
- Compartment length-to-width ratio
- Hanging baffle clearance
- Reserve Capacity

All but the last of these parameters has a significant effect on the hydraulic and biological performance of the process. The following subsections expand on the effect of each of these parameters, summarise the experiences obtained with the pilot reactor regarding these parameters, and propose design values for the parameters based on these experiences.

9.3.1.1 Mean hydraulic retention time

As described in Section 9.2 (principles of design), the mean hydraulic retention time affects the *contact time* in which wastewater treatment may occur, and indirectly, the up-flow velocity, that controls solids/sludge retention. It is also the parameter that dictates the size of the reactor and therefore has a significant effect on the capital cost of the system.

This study has not quantitatively characterised the curves of Figure 9.1 for an ABR, but has indicated that good COD removal, and substantial pathogen removal can be obtained at all of the tested retention times. Microbiological studies have shown that

significantly more methanogenic micro-organisms were able to establish at longer retention times, and more granule formation was observed, indicating that better process performance and stability at longer retention times were as a result of fundamentally better microbiological conditions. It is proposed that the basic ABR as a pre-treatment device in a low-income community be designed with an average hydraulic retention time of 36 h, giving a functional residence time between 1 and 2 days. This is similar to the design residence time of septic tanks in a South African context, but the effluent quality from the ABR unit will be significantly better than an equivalent septic tank.

9.3.1.2 Number of compartments

The number of compartments affects the internal velocity of liquid within the reactor, and thereby the solids retention capability of each compartment. A large number of compartments results in development of highly efficient microbial populations that are optimally acclimatised to compartment conditions. Research has shown that these separated populations are resilient to organic shock loads, but higher internal liquid velocities means that they will be more susceptible to shock hydraulic loads. Fewer compartments for the same reactor size will result in less highly specialised communities, and less forced contact between liquid and sludge beds, but better sludge retention characteristics. The number of compartments will also affect the capital cost of the reactor.

Experiments performed with the pilot-scale ABR treating WWTP influent recorded the following observations:

- There was a significant build-up of organic solids in the first compartment that spilled over into the second compartment
- There was a decreasing trend in anaerobic granule size with from compartment to compartment, progressing from reactor inlet to outlet.
- Later compartments did not appear to show any particular specialisation in terms of microbial composition.
- High flow incidents caused carryover of biomass to later compartments

In conclusion, there were noticeable changes in microbial composition of the biomass between different compartments, but except for the solids retention in the first compartment, and later, solids over-flow to the second compartment, variations between any one compartment and the next were small. Furthermore, the process was susceptible to high flow incidents causing sludge carry-over between compartments. It was concluded that fewer compartments than the 8 of the pilot design could be implemented without seriously changing the microbial performance of the reactor.

The basic ABR design therefore has 5 compartments:

- Compartments 1 and 2 retain solids and accommodate accelerated hydrolysis through maintenance of large hydrolysing bacterial populations;

- Compartments 3, 4 and 5 perform anaerobic digestion of soluble and suspended components entrained in the liquid flow, generating a three-point gradient of organic material and selected micro-organisms. The last compartment (5) will have a higher predominance of scavenging micro-organisms than earlier compartments since the substrate concentrations will be lowest in the last compartment.

9.3.1.3 Design up-flow velocity

The design up-flow velocity affects the sludge retention characteristics since, for sludge with a distribution of settling velocities (determined under no-flow conditions), a greater percentage of the sludge will settle at a low velocity than at a high velocity. In other words, at higher velocities, there is a greater degree of entrainment and therefore sludge carryover. For a fixed compartment volume, the up-flow velocity can be set by fixing the compartment *height-to-up-flow area* ratio. A low, wide compartment will have a lower up-flow velocity than a tall narrow compartment with the same volume.

The 22 h and 44 h experiments reported in Chapter 4 had average up-flow velocities of 0.55 m/h and 0.27 m/h respectively. Based on the chemical and microbiological evidence, it was concluded that washout of methanogenic micro-organisms may have occurred during the 22 h experiment, and consequently 0.55 m/h is considered too high to support stable methanogenic growth. An up-flow velocity of 0.27 m/h is considered acceptable, and is divided by a factor of safety of 2 to give a design up-flow velocity of 0.14 m/h.

9.3.1.4 Up-flow-to-down-flow area ratio

The ratio of up-flow-to-down-flow area in a compartment (defined by the *position* of the hanging baffle in a compartment delineated by two standing baffles) will affect the fluid dynamics in the sludge bed since a ratio other than 1:1 will result in a change in average liquid velocity on both the up and down-flow sides.

The computational fluid dynamics study presented in Chapter 3 showed that an up-flow-to-down-flow volume ratio of 2:1 in each compartment resulted in less flow channelling on the up-flow side than for a 1:1 ratio. Further increasing the difference between the two volumes is not advised as the high velocity on the down-flow side could cause sludge mixing, and uneven flow through the sludge bed, both undesirable effects. An up-flow-to-down-flow volume ratio of 2:1 is therefore recommended for the on-site ABR design.

9.3.1.5 Compartment length-to-width ratio

The length and breadth of the reactor will have no average effect on the up-flow velocity, although dead zones and channelling will occur if compartments are either too wide, or too long. The pilot reactor was built with a compartment length-to-width ratio of 1:3.75. No problems with channelling and dead zones were observed. It is therefore recommended that length-to-width ratios of between 1:3 and 1:4 are employed, depending on available space at the installation site.

9.3.1.6 Hanging baffle clearance

The gap between the bottom of the hanging baffle and the bottom of the reactor, the hanging baffle clearance, must be sufficiently large to prevent the occurrence of blockages by the sludge bed, but not so large that liquid flow bypasses the bottom of the sludge bed, causing dead volume on the floor of the ABR. Since the biomass flocs and granules observed during operation of the pilot ABR never exceed a few centimetres at most, blockages would have to be caused by build up of inert material. A clearance roughly equivalent to the height of two beer cans was selected, i.e. 200 mm. This is 29% of the height of the standing baffle.

9.3.1.7 Reserve capacity

The USEPA (2002) recommends that septic tanks for domestic use are built with a reserve capacity above the outlet of 1 to 2 days flow so that blockages at the outlet do not cause an immediate back-flow at the inlet. Using this rule of thumb, total volume of the unit should be double the working volume for a 36 h retention time design.

9.4 SECONDARY DESIGN DETAILS

The basic civil design of the ABR is fully defined by adopting the design parameters outlined in the previous sections. The following points define optional variations on the basic design, and where applicable, reasons for exclusion from the basic design.

9.4.1.1 Relative compartment sizes

The amount of solids accumulation that occurs in the early compartments will depend on the characteristics of the wastewater being treated. If a large amount of inert solid is expected to be fed to the reactor, with the accompanying risk of blockage, it may be appropriate to design the first compartment to be larger than subsequent compartments, or to replace the hanging baffle with another design that is less likely to cause blockage (or remove the first hanging baffle completely). No problems with blockage were observed in the pilot ABR, as inert solids had largely been excluded before reaching the feed box. In the absence of any strong reason for variations, it is recommended that all compartments have the same dimensions, provided that it is possible to remove solids from the first compartment in the case of blockage, or excessive solids build-up. However, relative increases to the size of the first compartment, or minor alterations to the baffle structure of the first compartment should not affect the process performance.

9.4.1.2 Hanging baffle design

The hanging baffles of the pilot ABR were built with an inclined bottom edge to improve flow dynamics in the sludge bed. However, this design will be expensive to implement in cement or brick structures, and increase the cost of a pre-moulded HDPE structure. For the low flows envisaged, it is not expected that the increased cost of the inclined edge design would be justified by a significant improvement in overall performance, and therefore, a simple flat baffle is recommended.

9.4.1.3 Solids separation at the outlet

To reduce effluent turbidity, and improve solids retention in the last compartment, the following options could be considered:

- Convert the last compartment to an anaerobic membrane bioreactor, by inserting an appropriate number of membrane plates into the last compartment, and withdrawing effluent as membrane filtrate. A preliminary study on membrane treatment of the ABR effluent has shown a flux of 1 ℓ/(m2.h) using Kubota membranes to be achievable. The design of secondary treatment using anaerobic membrane filtration is the subject of the Pollution Research Group, University of KwaZulu-Natal contribution to European Union project proposal 018480.

- Add packing media to the last compartment to promote biofilm growth and to provide a coarse filtration for the effluent. (A study by Tilche and Yang (1987) showed that the inclusion of packing material at the top of each compartment resulted in the achievement of significantly higher loading rates as a result of superior biomass retention).

9.4.1.4 Collection and use of biogas

A system for the collection and use of biogas has not been included in these design guidelines, although the energy generation potential of the biogas has been shown. Since the biogas is a potentially explosive hazard and an environmental burden, while simultaneously being an easily harnessed energy source, a biogas system should be included with the ABR design, and should be designed according to appropriate design standards.

9.4.2 Construction of peripheral features

9.4.2.1 Screening / degritting and inert solids management

Individual installations will be accompanied by different levels of community education and management of the system. Ideally, the community would be educated to eliminate inert solids from the sanitation system, however, in all probability, no reliance can be safely placed on users to prevent inert material entering the reticulation. It may therefore be necessary to place a screening/degritting unit before the reactor. This will require a daily maintenance programme.

Alternatively, the first compartment can be built with increased volume and an altered hanging weir to retain inert solids. A regular programme of grit build-up testing and degritting would be required.

9.4.2.2 Inlet

The reactor inlet design will affect the biological/hydraulic performance of the system in a number of ways:

- The size of the inlet will determine the velocity of the feed, which affects the flow dynamics in the first compartment; a high velocity will cause stirring and may cause

channelling of flow through the first compartment with associated entrainment, and poor treatment, of both solid and soluble components;

- A small inlet is susceptible to blockage;

- An inlet that allows air ingress to the reactor will affect the biochemistry of the first compartment to a small extent by allowing a small aerated zone at the top of compartment 1. This is unlikely to result in significant changes since the high bacterial load in compartment 1 will remove oxygen efficiently below the initial air-liquid contact layer. There will be insufficient aeration to result in significant nitrification.

Ideally, the flow should enter the reactor over an inlet weir. This could be built in conjunction with a screening unit. Failing this, a manifold with three inlet connections to the reactor will improve flow distribution in the first compartment. The pilot ABR was constructed with a single inlet and no problems were reported. However, the pilot ABR feed was thoroughly macerated, with little inert material or grit, while a community wastewater is likely to be more problematic to handle.

9.4.2.3 Outlet and outlet isolation valve

The outlet should be designed to prevent egress of biogas, for safety reasons, as well as to reduce odours, and to retain the gas as a source of energy. Standard septic tank outlet designs may be used at the outlet of the ABR.

Outlet blockages are rare in a well designed ABR since dense solids settle out in the first compartment, and solids less dense than water float in the first compartment. Only solids that have a density similar to water are likely to cause a problem, as was seen with the Kingsburgh installation of the pilot ABR, where a certain type of seed caused blockages in the outlet flow meter. However, it is unlikely that a solid that is sufficiently large to cause a blockage in a full scale installation will find its way through the ABR. Nevertheless, should an outlet blockage occur, the liquid in the reactor will build up above the standing baffle height, and may cause back-up of influent in the inlet and reticulation before the ABR. There are two risks associated with this instance: firstly, the risk of over-flow or leaking of raw sewage poses a serious health and environmental hazard. Secondly, when the blockage at the outlet is cleared, the flow out of the reactor will be sudden and fast as the pressure of the backed-up liquid forces flow out of the reactor. This may result in the sudden release of a large volume of poorly treated wastewater to the secondary treatment process, or the environment, with a host of consequences. A significant amount of biomass may be washed out of the system as well. As it takes a long time to re-establish a healthy anaerobic microbial community, this may result in a long recovery period for the system with further implications for the secondary treatment process and the receiving environment.

It is therefore recommended that an *isolation valve* be installed on the outlet valve so that backed-up liquid in the reactor may be released in a controlled fashion.

9.4.2.4 Desludging hatch

Ideally, it should be possible to sample each compartment, but a gas seal is also desirable, and therefore it is not practical to have desludging hatches on each compartment. It is necessary to have a desludging hatch on compartment 1 so that grit can be removed as needed, since most inert solids accumulation will occur in compartment 1.

9.4.2.5 Compartment head space

A design in which the hanging baffles extend to the roof and provide a complete gas seal between the head spaces of adjacent compartments Figure 9.2-a. is beneficial to the microbiological process in that syntrophic bacteria in later compartments are shielded from high levels of hydrogen gas produced by acidogenesis in the earlier compartments. However constructing gas tight compartments will significantly increase the cost of the reactor. It is therefore recommended that, where possible, the first 2 compartments and last 3 compartments share head spaces (Figure 9.2 -b), which are isolated from each other. Where this is impractical because of cost, or material of construction, a single, open headspace for all compartments may be constructed, although poorer performance may be obtained (Figure 9.2-c). Note that, depending on the inlet and outlet design, compartment 1 and 5 hanging baffles should extend to the top to prevent gas escaping via the inlet and outlet.

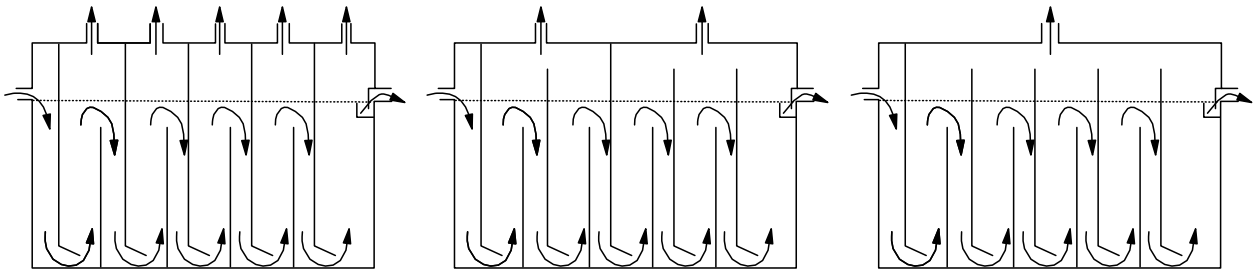


Figure 9.2: Headspace configurations for the ABR: (a) All compartments have separate headspace (b) Two headspace areas exist: one for compartments 1 and 2, and another for compartments 3,4 and 5 (c) One headspace for all 5 compartments.

9.4.3 Gas vents

The gas from each compartment should be collected via gas vents and a gas collection system. The design of the gas collection system should be undertaken to the appropriate safety standards, with the hazardous nature of biogas in mind. There should be at least one gas vent per biogas headspace area.

9.4.4 Sampling/dosing ports

Except for experimental purposes, it should not be necessary to take samples from the reactor itself. It is therefore recommended that samples be collected from the effluent

for analysis. Similarly, it should not be necessary to dose any chemicals to the reactor, but should any dosing be required, addition to the first compartment would be adequate. Sampling and dosing ports would not therefore be necessary.

9.4.5 Calculation of on-site ABR dimensions

$$\text{Process flow rate} = 2736\ \ell/\text{d} = 0.114\ \text{m}^3/\text{h}$$

$$\text{Hydraulic Retention time} = 36\ \text{h}$$

$$\text{Required working volume} = 0.114\ \text{m}^3/\text{h} \times 36\ \text{h} = 4.104\ \text{m}^3$$

$$\text{Number of compartments} = 5$$

$$\text{Comp. working volume} = \frac{4.104\ \text{m}^3}{5} = 0.821\ \text{m}^3$$

$$\text{Design upflow velocity} = 0.14\ \text{m/h}$$

$$\text{Comp. upflow area} = \frac{0.114\ \text{m}^3/\text{h}}{0.14\ \text{m.h}^{-1}} = 0.814\ \text{m}^2$$

$$\text{Upflow} - \text{to} - \text{downflow ratio} = 2:1$$

$$\text{Comp. downflow area} = \frac{0.814\ \text{m}^2}{2} = 0.407\ \text{m}^2$$

$$\text{Total comp. area} = 0.814\ \text{m}^2 + 0.407\ \text{m}^2 = 1.221\ \text{m}^2$$

$$\text{Comp. working height} = \frac{0.8208\ \text{m}^3}{1.221\ \text{m}^2} = 0.672\ \text{m}$$

$$\text{Width} - \text{to} - \text{length ratio} = 3.75:1$$

$$\text{Comp. length} = \sqrt{\frac{\text{Volume}}{\text{Height} \cdot 3.75}} = \sqrt{\frac{0.821\ \text{m}^3}{0.672\ \text{m} \cdot 3.75}} = 0.5707\ \text{m}$$

$$\text{Comp. width} = 0.5707\ \text{m} \times 3.75 = 2.104\ \text{m}$$

$$\text{Calculated Reactor width} = 2.104\ \text{m}$$

$$\text{Design Reactor width} = 2.1\ \text{m}$$

$$\text{Calculated Reactor length} = 5 \times 0.5707\text{m} = 2.85\ \text{m}$$

$$\text{Design Reactor length} = 2.9\ \text{m}$$

$$\text{Reserve capacity} = 1 \times \text{working volume}$$

$$\text{Design Reactor height} = 1.4\ \text{m}$$

$$\text{Standing baffle height} = 0.7\ \text{m}$$

$$\text{Hanging baffle clearance} = 0.2\ \text{m}$$

$$\text{Standing baffles at } i \times 0.55\ \text{m from inlet}$$

$$\text{where } i \in [1;4];\ \textit{i.e.}\ \ 0.55\ \text{m} ; 1.10\ \text{m} ; 1.65\ \text{m} ; 2.2\ \text{m}$$

$$\text{Hanging baffles at} \left(i + \frac{1}{3} \right) \times 0.55\ \text{m from inlet}$$

$$\text{where } i \in [0;4];\ \textit{i.e.}\ \ 0.18\ \text{m} ; 0.73\ \text{m} ; 1.28\ \text{m} ; 1.83\ \text{m} ; 2.38\ \text{m}$$

This calculation defines a reactor with external dimensions of 2.9 m × 2.1 m × 1.4 m. This is similar to the pilot reactor in length and height, but 40 % wider, and with fewer compartments. The flow velocity will be as little as a quarter of that generated in the pilot ABR, and consequently, the solids retention can be expected to be far better. This design also has the advantage of being approximately the right size to fit on the back of a truck.

9.5 FINAL RECOMMENDED DESIGN

Figure 9.3 presents the final recommended baffle design for an ABR for on-site domestic wastewater pre-treatment, drawn to scale. Full height hanging baffles in compartment 1 and 3 and reduced height hanging baffles in compartments 2, 4 and 5 are shown. A gas-tight outlet configuration is shown. Inlet, desludging hatch and gas vents are not shown. Liquid level during normal operation is at the height of the standing baffles and outlet port.

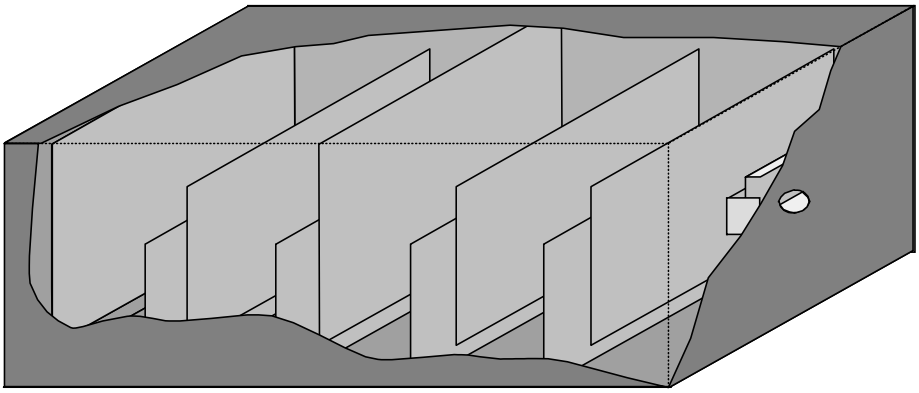


Figure 9.3: Recommended baffle design for an on-site ABR for pre-treatment of domestic wastewater.

9.6 PROCESS (MICROBIOLOGICAL/BIOCHEMICAL)

This section presents a description of how the design ABR is expected to perform on a number of different wastewater types, and methods of characterising the reactor performance

9.6.1 Expected design ABR performance

Wastewater characteristics presented in Table 9.2 were used in the steady-state model calibrated in Chapter 7, and expected reactor performance on three different wastewaters are presented in Table 9.3

Table 9.3: Expected design ABR performance on a dilute, low income community generated wastewater. (No greywater recycling). Predictions of effluent characteristics from the ABR primary treatment are shown.

	Unit	**Influent**	**ABR effluent**
Flow Rate	ℓ/d	2736	
Flow in PE	ℓ/d per capita	68.4	

Extent of treatment	% biodegradable COD removal		99
COD	mgCOD/ℓ	1090	200
$NH_4^+ + NH_3$	mgN/ℓ	30	48
PO_4	mgP/ℓ	25	25
Alkalinity	mgCaCO3/ℓ	250	341
Total Coliforms	log(cfu/100mℓ)	7	5.5
TSS	mg/ℓ	1500	750
pH	-	8	5.97

Note that the number presented for *% biodegradable COD removal* represents the percentage of biodegradable material that entered the reactor that is converted. Values of >98% are observed, but the outlet COD is still above 100 mgCOD/ℓ. The residual COD is made up of produced biomass and inert material originating from the feed

These results represent a steady-state condition in the ABR. This implies that there is no net accumulation of sludge in the reactor, and that all produced sludge is leaving the reactor in the effluent, as is the case in a CSTR. In reality, a fair amount of the sludge accumulates in the compartments, and the effluent concentration should contain less biomass than predicted here. The steady-state model requires some further alterations and calibration before it can be meaningfully used for reactor performance. However, given that the mechanism of COD removal is incorrectly modelled (all biodegradable COD is consumed and only waste anaerobic sludge contributes to effluent COD) the effluent COD values predicted are what could be expected intuitively from an ABR pre-treatment step in a sanitation system.

Table 9.4: Expected design ABR performance on a concentrated, low income community generated wastewater. (Greywater recycling reduces hydraulic load). Predictions of effluent characteristics from the ABR primary treatment are shown.

	Unit	**Influent**	**ABR effluent**
Flow Rate	ℓ/d	1368	
Flow in PE	ℓ/d per capita	34	
Extent of treatment	% biodegradable COD removal		99.8
COD	mgCOD/ℓ	1908	328
$NH_4^+ + NH_3$	mgN/ℓ	53	86
PO_4	mgP/ℓ	44	44
Alkalinity	mgCaCO3/ℓ	300	463

Total Coliforms	Log(cfu/100mℓ)	7	6
TSS	mg/ℓ	2600	800
pH	-	8	6.1

Table 9.5: Expected design ABR performance on a dilute, middle income generated wastewater for a cluster of 3 homes. (No greywater recycling). Predictions of effluent characteristics from the ABR primary treatment are shown.

	Unit	**Influent**	**ABR effluent**
Flow Rate	ℓ/d	2337	
Flow in PE	ℓ/d per capita	260	
Extent of treatment	% biodegradable COD removal		98.
COD	mgCOD/ℓ	510	95
$NH_4^+ + NH_3$	mgN/ℓ	6	13
PO_4	mgP/ℓ	8	8
Alkalinity	mgCaCO3/ℓ	250	286
Total Coliforms	Log(cfu/100mℓ)	6.8	6
TSS	mg/ℓ	575	288
pH	-	6.2	5.89

9.6.2 Calculation of actual ABR installation performance

In order to determine how well the ABR is performing, analytical measurements may be made. They are summarised in section 9.5.2.1 to section 9.5.2.4.

9.6.2.1 Effluent COD

Effluent COD is a measure of the amount of organic material that is untreated at the outlet of the reactor, and an important indicator of the acceptability of the wastewater for secondary treatment and reuse. Depending on the type of secondary treatment or disposal, values below 200 mgCOD/L indicate that the reactor is operating well.

9.6.2.2 % COD removal

% COD removal is calculated as follows:

$$\%COD\ removal = \frac{Inlet\ COD - Outlet\ COD}{Inlet\ COD} \cdot 100\%$$

This is not the same as the extent of treatment, since a portion of the inlet COD is not biodegradable, and therefore 100% COD removal can never be achieved by anaerobic digestion alone.

9.6.2.3 Effluent pH

Effluent pH provides an indication of the microbiological conditions in the ABR. Effluent values above pH 7 indicate that the process is stable, and that there is no pH inhibition of micro-organisms, and no immediate risk of failure. pH values between 6 and 7 indicate that although the process is functioning effectively, there is a significant risk of souring, if some change occurs, e.g. high organic load, influx of inhibitory substance or dramatic change in temperature. Values below 6 indicate that the process is potentially unstable, and some intervention such as reducing the flow (if possible) or dosing with alkaline may be advisable. Below pH 6, pH values can drop to a minimum of 4.5, indicating souring of the anaerobic digestion.

9.6.2.4 Effluent alkalinity

Effluent Alkalinity may be measured, but as the effluent alkalinity is strongly dependent on the influent alkalinity, it may be difficult to interpret this measurement. In general, effluent alkalinity should be above 250 mg$CaCO_3$/ℓ, although values above 400 mg$CaCO_3$/ℓ indicate that sufficient buffering is available for anaerobic digestion.

9.7 OPERATIONAL

This section outlines procedures, checks and remedies to ensure the continuing successful operation of an ABR system.

9.7.1 Start-up

The start-up period for any anaerobic process is very sensitive since the biomass populations are generally not well established, and probably not very concentrated, and therefore very susceptible to washout, and souring. Experience with the pilot ABR showed that not only was initial seeding of the reactor necessary, it was also necessary to start up with a significant amount of biomass, and a low flow rate. In a real application, there will be very little control over the flow rate to the unit, unless individual households are connected to the ABR one at a time. It is therefore critical that there is a significant amount of biomass present right from the beginning. It is recommended that between ***one quarter*** and ***one third*** of the working volume of the reactor be filled with anaerobic digester sludge from e.g. a local WWTP. It is also feasible to seed the ABR with septic tank sludge, provided the sludge does not have an excessively high concentration of inert solids. Too much seeding material will result in a relatively short operating period before desludging is required. Too little seeding will result in a lengthy start-up period characterised by poor quality effluent.

The start-up period should be regarded as at least 3 months, and the performance of the reactor should be carefully monitored during this time. There will be a period of adjustment in which anaerobic biomass that is not active, or has poor settling

characteristics is washed out of the system. There may also be a period in which insufficient treatment of wastewater is obtained while the biomass acclimatises to operating conditions, resulting in high COD and pathogen concentrations in the effluent. The effect of this on a secondary process must be considered.

9.7.2 Maintenance

While the ABR system for sanitation is supposed to be a low maintenance option, as with all technologies, if there is no programme of monitoring and maintenance, the risk of failure is significant. Furthermore, the implementation of a visible monitoring and maintenance programme will create an awareness among users of the system that it is susceptible to problems if not managed and cared for right from the toilet bowl to the secondary treatment process.

9.7.2.1 Solids screens

If solids screens are included in the ABR installation, a programme for daily raking of screens and, collection of screenings and weekly removal of screenings must be in place. The screening unit is liable to collect grit, both from the wastewater, and from the surroundings of the screening unit, and regular degritting/grit level monitoring should be undertaken.

9.7.2.2 Influent flow manifold/inlet weir

The most probable point likely to experience blockages is the inlet to the reactor. If the installation is built with an inlet weir, rags and tissue are likely to collect on the lip of the weir, resulting in uneven flow into the reactor, and ultimately blockage. If a single inlet pipe, or a manifold and multiple inlet pipes are built, then ells or constrictions in the reticulation may similarly experience blockage. It must be possible to rod out pipes and weirs where necessary, and a check of these points should be included in routine monitoring on a weekly basis.

9.7.2.3 Grit build-up

Thick sludge filling the first compartment is not an indication that desludging is required, since the sludge consists of hydrolysing particulate organics, which generate the substrate for subsequent steps in the anaerobic digestion, and a high load of micro-organisms, which constitute the process catalyst. Desludging is only required if either (i) there is a significant amount of inert material in the first compartment (e.g. sand, hair, twigs, sanitary towels, nappies) or (ii) there is a blockage.

Grit build-up will be difficult to detect when there is a thick layer of sludge in compartment 1. A simple method of detecting grit is to insert a rod to the bottom of the compartment, and make sweeping movements around the bottom of the compartment. It may be so thick that it will hold the rod upright without other support. However, active biomass and hydrolysing solids will tend to be fairly homogenous. Grit and undesirable solid material will tend to have a different density to the sludge, and will cause a variable and gravelly friction when *stirred* with the rod. This gravelly layer should not constitute more than half of the down-flow working volume (about 0.35 m).

9.7.2.4 Analyses

The analyses required will depend considerably on the requirements of the secondary treatment system. A regular check of outlet pH will provide information on the stability of the ABR process. A measurement of outlet COD should be performed to monitor the effectiveness of the process. pH checks can be performed weekly using universal indicator paper, and COD measurements may be performed monthly or bi-monthly. Samples for pH measurement should not be taken and analysed later in a laboratory, as anaerobic samples have a tendency to continue digesting in batch mode, causing a significant increase in pH between sampling and measurement.

9.7.2.5 Outlet

The outlet should be routinely checked to ensure that there is flow out of the reactor. No flow at the outlet could indicate either that there is a blockage at the outlet, or a leak in the reactor itself. The outlet check should be performed at least weekly. However, if there is a blockage, it should quickly be noticed by back-up of liquid in the screening unit if there is one, or poor drainage rates in toilets and kitchen sinks in homes, if there is no screening unit.

9.7.2.6 Biogas collection

The design for biogas collection and use must include guidelines for inspection and maintenance.

9.7.2.7 Maintenance checklist

Every installation should have a maintenance checklist, either in the hands of the users, or the local authority, depending on where the responsibility for the maintenance of the system resides. It is recommended that there are three levels of checks that are undertaken that is, daily checks, weekly checks and quarterly checks. These have been described in the sections above and are summarised in Table 9.6.

Table 9.6: Maintenance checklist for an on-site ABR pre-treating domestic wastewater.

Daily Checks	Weekly Checks	Quarterly checks
Rake screens	Remove screenings	Check grit level in compartment 1
Collect screenings	Check inlet for blockages	Measure outlet COD concentration
	Check outlet for blockages	Inspect biogas system
	Check outlet pH	
	Check grit in screenings unit	

9.7.2.8 Operator training

Any person who is *employed* to operate and maintain a sanitation system must be trained in the safe handling of hazardous biological agents, and supplied with the appropriate protective wear and equipment in terms of the Regulation of Hazardous Biological Agents Act.

9.7.2.9 User education

The single most important factor in the operation of an ABR for pre-treatment of domestic wastewater is that it is cared for by the people who use it. Although active maintenance may not be performed by the users themselves, they must be aware of what the system can tolerate, so that inappropriate materials and substances are not put into the system. As with septic tanks, the system will fail very quickly if overloaded with grit and non-degradable matter, or subjected to slugs of inhibitory substance from disinfectants and detergents. Considerable improvements in reactor performance can also be achieved if users are taught to recycle water other than toilet and kitchen sink water. The user education is therefore as important as the appropriate design of the system, and a programme should be developed for ensuring that an awareness of the needs of the system is maintained within the user community.

9.7.3 Troubleshooting

There are certain problems that can be expected to occur from time to time. This section provides guidelines for dealing with them.

9.7.3.1 Outlet blockages

In the case of a blockage at the outlet, the reserve capacity in the headspace of the reactor will begin to fill up. Before the blockage is removed, the outlet isolation valve should be closed. Once the blockage is cleared, the valve should be opened slowly to release a flow not exceeding 3 times the design flow rate i.e. around 8 ℓ/min.

9.7.3.2 Low alkalinity feed

If the feed wastewater is identified as having a very low alkalinity, regular measurement of the outlet pH value should be made. If the outlet pH value is routinely below 6.5, a programme of alkali dosing may be appropriate.

9.7.3.3 Recovering from a souring incident

Should an ABR *go sour*, the system can recover fairly quickly if the ABR is isolated, sufficient alkali is added to bring the reactor pH value above 6, the system for a few days. However, in most cases, it will not be possible to stop flow to the reactor, and therefore, a daily addition of alkali to the first compartment, until the pre-dose pH value is above 6 will assist a rapid recovery of the process.

An investigation into the cause of souring should be undertaken to identify whether there was a once-off problem that is unlikely to re-occur or a chronic problem that requires alteration to the system.

Possible causes of *once-off* problems include:

- A slug of inhibitory substance (poison, certain disinfectants and detergents)
- A high organic load associated with a party
- A sudden drop in temperature
- Possible causes of chronic problems are:
- Low wastewater alkalinity
- Regular abuse by users

Increase in the number of users as a result of additional homes being connected to the system, extensions to homes, or increase in the number of people per household

9.8 EFFLUENT MANAGEMENT

It has been shown that the effluent from an ABR will have significant amounts of nitrogen, phosphorus and pathogen indicator organisms. An effluent disposal plan must be in place before an ABR can be installed for decentralised treatment of community wastewater.

It was not within the scope of this project to undertake an exhaustive study of post-treatment and reuse options. However, a preliminary investigation into post-treatment options has revealed the following possibilities:

- Treatment of anaerobic effluent in constructed wetlands or sub-surface filters has been successfully implemented to achieve nitrogen and pathogen removal. Depending on the system used, some phosphorus removal can be obtained (Sasse, 1998).

- The possibility of installing a membrane filtration unit in the last compartment of an ABR is the subject of the University of KwaZulu-Natal contribution to European Union project proposal 018480 Membrane bioreactor technology (MBR) with an EU perspective for advanced municipal wastewater treatment strategies for the 21st century. The hybrid ABR/MBR (membrane bioreactor) system would produce a low COD, zero solid, disinfected effluent with some nutrients that would be an ideal fertilising liquid for urban agricultural initiatives.

- Hybrid post-treatment steps are available for removing nitrogen from the effluent. However, these would significantly increase the cost of the system, and the amount of area that is required for treatment.

- If there is sufficient area, and the geo-hydrology of the area in which the system to be installed is appropriate, a large scale soak-away or evapo-transpiration area can be installed using designs employed in septic tank systems.

It is clear that, in a low-income sanitation application, the ABR can only be considered a suitable option if the whole effluent can be safely treated in a soak-away or if it can be sufficiently disinfected to be used in a well defined agricultural enterprise.

10 CONCLUSIONS AND RECOMMENDATIONS

This project was undertaken to determine the appropriateness of an anaerobic baffled reactor in treatment of domestic wastewater in low-income communities. A pilot ABR was built and operated at two municipal WWTPs and operation in terms of chemical and microbial performance was characterised under a number of different operating conditions. A study was performed in which water use patterns and wastewater characteristics in a low-income community were measured. These data were incorporated in a model to predict the performance of the ABR would perform in a low-income community. Based on experiences with pilot ABR, a series of design, operating and maintenance guidelines were developed for future installations.

10.1 CONCLUSIONS

The conclusions of each of the phases of this project are presented:

10.1.1 Conclusion from a review of sanitation in South Africa

A number of sanitation options are available for low-income communities. The most economically and environmentally sustainable of these appear to be dry sanitation systems. A few systems are available for water-borne on-site or decentralised sanitation which make use of a septic tank or variation to pre-treat wastewater. However, the application of these in low-income communities is limited due to space considerations and process stability. There is therefore a gap in the technology available for water-borne sanitation in low-income communities.

10.1.2 Conclusions from operation of a pilot-scale ABR at municipal wastewater

- The pilot ABR operated fairly smoothly, showing good biological activity in all of the operating periods. Almost all the problems associated with operation of the system were related to the feeding system and peripheral equipment required to sample wastewater from a much larger flow. These included pump blockages, wear and tear on the compressor and pneumatic valve, limitations of the programmable logic controller (PLC) algorithm and blockages of the effluent pipe at the magnetic flow meter. In a community installation, none of these problems will occur since the ABR unit would be gravity fed, and would treat the entire wastewater flow generated.

- Fairly stable operation of the pilot ABR was obtained in all operating periods despite occasional variations in flow and load, and biomass washout incidents. Only one incidence of anaerobic digestion failure occurred during the 5 years of operation, and this was traced to illegal dumping of septic tank sludge in the influent to the WWTP, resulting in a shock organic load to the ABR.

- The pilot ABR showed rapid recovery after failure due to an organic overload. Physical separation of compartments results in a pseudo-plug-flow configuration with sludge retention in compartments. This exhibits rapid washout of acids and excess substrate from the organic overload, resulting in a more rapid normalisation of

conditions than would be observed in a completely mixed system, without substantial biomass loss.

- The pilot ABR exhibited COD removal in all operating periods, even during start-up where biomass concentrations in the reactor were very low. At a hydraulic retention time of 22 h, effluent COD was found to be ca. 190 mgCOD/ℓ and at a retention time of 40 to 44 h, this reduced further to ca. 130 mgCOD/ℓ.

- No nutrient removal is obtained in an anaerobic treatment system. Ammonia concentrations in the pilot ABR increases as a result of liberation of organically bound nitrogen during digestion of complex organic material, and phosphorus concentrations were largely unaffected. Small sulphate concentrations in the influent were reduced by sulphate reducing bacteria to H_2S. Alkalinity increased due to generation of bicarbonate and carbonate during digestion. The presence of significant concentrations of nutrient in ABR effluent mean that effluent cannot be discharged to surface or groundwater, or be reused in such a way that may lead to contamination of surface or groundwater.

- Significant removal was obtained for all pathogen indicator organisms tested vis. total coliforms, *Escherichia Coli*, coliphages and *Ascaris* eggs. However the effluent still contained unacceptably high concentrations of all of these indicators; Total coliforms and *E. coli* exhibited effluent concentrations in excess of 106 cfu/100mℓ and coliphages were greater than 103 pfu/100mℓ. Numbers of *Ascaris* eggs in effluent samples varied substantially with a mean concentration of 17 eggs/ℓ. These numbers indicate that ABR effluent should be considered a health hazard, and cannot be reused without further disinfection.

- The pilot ABR was initially seeded with only 10 ℓ of anaerobic digester sludge, and therefore took more than a year to develop stable biomass loads. Seeding of an ABR was identified as the critical factor in reducing the length of the system start-up period.

10.1.3 Conclusions from microbiological studies of the pilot ABR

- A diverse community of micro-organisms exist in the pilot ABR treating domestic wastewater but population dynamics differed from those that have been reported for high strength soluble COD applications.

- Fluorescent in-situ hybridisation techniques were unable to provide a satisfactory characterisation of methanogenic populations in the pilot reactor. This result was attributed to either low RNA activity in these micro-organisms, or binding of methanogens in granules rendering them inaccessible to oligonucleotide probes.

- Scanning electron microscopy showed that there was a difference in spatial distribution of acetoclastic methanogenic populations in the pilot ABR, with scavenging *Methanosaeta* spp. dominating in all compartments except compartment 1.

- Granulation of anaerobic sludge was observed, and a detailed examination of the structure of anaerobic granules was made. Observations indicated that the mechanism of granulation in the treatment of low-strength domestic wastewater differs from the mechanism seen in other applications.

10.1.4 Conclusions relating to the mechanism of anaerobic digestion in the pilot ABR

- Two distinct functions were seen in the pilot ABR. In the first compartment, hydrolysis and acidification of influent wastewater components dominated, resulting in an accumulation of volatile fatty acids measured as acetic acid and soluble COD, and a depression in pH, indicating that methanogenesis was the rate-limiting step.

- In all but the first compartment, the rate-limiting step was observed to be hydrolysis. Concentration of hydrolysable COD decreases from one compartment to the next with other concentrations remaining fairly constant. Little difference is seen in relative abundance of microbial populations responsible for undertaking different sub-processes of anaerobic digestion from one compartment to the next.

- A constant slow accumulation of sludge was observed in the pilot ABR. This implies that the system will eventually require desludging, although this point was not reached after 5 years of operation and more than 1 000 000 ℓ of wastewater treated.

10.1.5 Conclusions from the community water use and wastewater generation study

- A community water use survey identified that low-income urban dwellers in general used significantly more water in their homes than their peri-urban counterparts.

- A community wastewater characterisation study was unable to identify significant correlations between contaminant concentrations and time of day, sewer or day of the week for samples obtained from sewers in a low-income peri-urban community.

- Low income community generated wastewater showed higher concentrations of COD and organic nitrogen then were measured in municipal wastewater from middle-income communities.

10.1.6 Conclusions relating to modelling of the pilot ABR

- The Siegrist et al. (1993) model of anaerobic digestion was found to be inappropriate for modelling anaerobic digestion of domestic wastewater because it considered a single particulate substrate with fixed composition that was unable to imitate substrate dynamics in the compartments of the ABR.

- The Sötemann et al. (2005) steady-state model of anaerobic digestion was limited in its ability to describe ABR performance since all produced biomass is removed in its effluent, while in reality, excess biomass is accumulated in the reactor.

- The steady-state model was used to predict the effect of different wastewater characteristics on ABR effluent characteristics. The influent parameter that was seen

to have the biggest impact on effluent pH, and thereby inhibition of anaerobic processes and overall treatment rate was the influent alkalinity concentration. This implies that operation can be most effectively enhanced by ensuring there is sufficient pH buffering capacity in the reactor by appropriate dosing of alkali in the influent or first compartment.

10.1.7 Conclusions relating to the development of design guidelines for an ABR treating community wastewater

- The objectives of an ABR design are to maximise solids retention, and maximise contact time of wastewater with the biomass. A number of key parameters were identified to fix the dimensions of the ABR design, and a description of how the design objectives were affected by each of these parameters was supplied. The key design parameters are:
    - Mean hydraulic retention time
    - Number of compartments
    - Design up-flow velocity
    - Up-flow-to-down-flow area ratio
    - Compartment length-to-width ratio
    - Hanging baffle clearance
    - Reserve capacity
- An ABR unit should be appropriately seeded to ensure rapid start-up, but sufficiently long desludging intervals. It was recommended that between one third and one half of the ABR working volume should be filled with anaerobic digester sludge or septic tank sludge before commencing operation.
- Poor maintenance of a community ABR is more likely to cause system failure than biological failure of the anaerobic digestion process. A list of maintenance requirements was compiled that included routine checks of inlet, outlet and biogas collection and disposal/conversion system, solids screening, chemical analyses and determination of grit build-up.
- A tentative prediction of ABR performance on three different qualities of wastewater was reported. Reuse of greywater would result in a lower flow, and slightly lower load of contaminant to the unit. The ABR effluent characteristics for the more concentrated wastewater feed would be worse than those of the less dilute feed, but the overall load of contaminants leaving the ABR would be less. Improvements in pH value are seen for treatment of the more concentrated wastewater, which would result in improved treatment rates, although this effect could not be seen in these predictions.

10.1.8 Conclusion relating to ABR effluent management

- It has been shown that the effluent from an ABR will have significant amounts of nitrogen, phosphorus and pathogen indicator organisms. An effluent disposal plan must be in place before an ABR can be installed for decentralised treatment of community wastewater.

- Possible options for an integrated post-treatment step include

    - A hybrid ABR-membrane bioreactor system with integrated membrane units for disinfection and COD removal

    - A constructed wetland for nutrient and pathogen removal

    - An appropriately sized and located soak-away or evapo-transpiration area.

Each of these options will be constrained by the topography and geo-hydrology of the area in which the unit is to be installed.

10.1.9 Overall conclusions

In summary, the ABR was found to be a robust treatment system, with biological and hydraulic advantages over septic tank systems, and with considerably reduced installation, operation and maintenance costs compared to aerobic or centralised systems. It also provides an option for communities with dry sanitation that aspire to waterborne sanitation.

However, the ABR was not able to treat wastewater to an acceptable chemical and microbiological standard alone. There must be some post-treatment step and appropriate reuse or discharge method implemented with the ABR as an integrated sanitation system, since unpolished ABR effluent is not fit for discharge to surface or groundwater or for direct use in agriculture.

As with septic tank systems, the ABR has no intrinsic mechanism for managing build-up of inert solids. Therefore an installation treating domestic wastewater must include a screening and grit removal pre-treatment step, or a maintenance plan for regular degritting of the first compartment should be in place. A key factor in the management of inert solids in the ABR is to educate system users to avoid disposing of unsuitable substances into the wastewater treatment system.

10.2 RECOMMENDATIONS

There are many factors relating to the implementation of a decentralised wastewater treatment system that have not been addressed directly in this project. However, it is believed that a sufficient understanding of the process mechanisms of the ABR have been gained in this project to consider the technology ripe for application in certain situations.

10.2.1 Recommendations relating to the application of ABR technology in sanitation

- The ABR is able to provide better and more efficient treatment of wastewater than a septic tank. Therefore it is recommended that an ABR system can be used in any situation that is considered appropriate for a septic tank.

- Further research into post-treatment options is required for implementation of an ABR in a community setting where water cannot be disposed of in a soak-away.

- The application of an ABR in an institutional setting such as for schools, clinics or community toilet blocks should be thoroughly investigated.

10.2.2 Recommendations relating to research carried out in this project

- The pilot ABR was thoroughly tested on feeds of purely domestic and semi-industrial wastewater (Chapter 4). However, the domestic wastewater originated in middle-income formal suburbs, and therefore ABR performance in a community application could not be directly predicted from experimental results of this study. A further study of ABR performance when treating community wastewater will close this knowledge gap.

- Although the pilot ABR showed stable operation in spite of varying flows and loads (Chapter 4), and (during some periods) regular biomass washout incidents, the effect of regular surge flows and diurnal variation in flow and load was not tested. In a community situation, it can be expected that the flow rate and load will exhibit considerable peaks and troughs that may compromise the ability of the reactor to function in the same manner as was observed with the pilot ABR. It is recommended that the effect of sustained diurnal flow oscillations over a long period be investigated.

- Microbial community studies (Chapter 5) provided a great deal of insight into the microbiology of the ABR, however there were certain inconsistencies, in findings between the FISH and SEM study. Further work specifically required to understand the dynamics of methanogenic genera, and to elucidate the mechanism of granule formation.

- The community wastewater characterisation study obtained many measurements of concentration for a number of important wastewater characteristics in a low-income peri-urban community. Flow data was not obtained at the same time and therefore calculation of contaminant loads could not be performed. Furthermore, certain critical analyses were not performed, specifically VFA and alkalinity measurements. It is recommended that the study is repeated, with some means of calculating wastewater flow at the time of sampling so that contaminant loads through the sewer can be determined, and with measurements of alkalinity and VFA.

- The steady-state model of anaerobic digestion has the potential to provide useful predictions with little modelling effort. It is not ideally suited to ABR operation since a steady but very slow accumulation of biomass is expected, which violates the

assumption of steady state. The current steady-state model assumes that the small sludge production exits the ABR with the effluent. In order to maintain the effluent COD concentrations at the levels observed experimentally, the model compensates by calculating improbably high reaction rates. A simple adaptation of the steady-state model which includes only a portion of the sludge production in the effluent, but assumes that the overall sludge load is constant would result in improved predictions of reaction rate and improved ability to interpolate reactor performance between operating points.

- The groundwork for dynamically modelling the ABR in WEST ® has been laid using a Siegrist model of anaerobic digestion. This model was used to interrogate experimental data to gain an understanding of process dynamics, and to identify the way forward in creating a verified model of the ABR for design purposes. The next step is to implement the ABR model in a more complex mathematical model structure. A simplification of the Anaerobic Digestion Model No. 1 is proposed.

- The primary limitation of the ABR for community sanitation is that the effluent cannot be discharged to ground or surface water untreated, and has pathogen loads that are too high for direct use in agriculture. It is necessary to develop an integrated system which defines and tests appropriate post-treatment methods for specific reuse/discharge applications (e.g. an anaerobic membrane bioreactor).

- Experimentation with the pilot ABR showed that the biological processes were fairly robust, and that the ABR is (biologically) appropriate for treating domestic wastewater. However, the ABR must fit into an overall system that has community support, and where all the peripheral components including pipe-work, screening and biogas management function correctly and safely. Development of the technological and social aspects of this system is a critical further step in this research.

- This study has produced a set of guidelines for the design, operation and maintenance of an ABR as the primary step in the treatment of domestic wastewater. The recommended design must be tested in the field, and appropriate experiences captured in the guidelines document.

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APPENDIX A1: METHODS OF SAMPLING AND ANALYSIS

This project spanned five years of experimentation. During this time, many people were involved in sampling and analysis, and a number of different techniques were used.

1 SAMPLING

Samples of inlet and outlet concentrations were obtained from the feed splitter box and the outlet pipe just before treated effluent was discharged back to the wastewater channel. During experimentation at Umbilo WWTP, samples were obtained from the sample valves supplied on the side of the reactor. The initial 100 mℓ drawn from each valve was discarded and the subsequent volume collected and stored for analysis.

The relative amounts of sludge and liquid in each compartment were measured using a *sampling stick* or *core sampler* (Figure A1.1). This consisted of a Perspex tube with a 50 mm internal diameter, calibrated for height in metres, and fitted with a rubber bung attached to a steel rod. The rubber bung and rod was loosened from the outer tube and dropped into the ABR via the 75 mm port on the top of the compartment to be sampled.. The Perspex rod was then dropped over the steel rod to land on the bung, capturing a *core* sample that would be withdrawn from the reactor. Initial sludge and liquid levels were recorded. A 5 min settling time was allowed before *settled* sludge levels were measured.

Figure A1. 1: Core Sampler filled with compartment 1 sludge (left) and compartment 8 sludge and supernatant (right)

During the Kingsburgh experimentation, samples of compartment contents were not obtained from the valves on the side of the reactor as it was believed that conditions near the wall of the reactor did not represent bulk conditions. Compartment samples were obtained using the core sampler. Once sludge levels had been recorded, the core sampler was balanced in a bucket and the bung worked loose so that the core sample

flushed out into the bucket. Bucket contents were vigorously stirred and a sample withdrawn for storage and analysis.

2 SAMPLE STORAGE AND PREPARATION

Wherever possible, samples were transported immediately to a laboratory for analysis. Samples were stored in a cold room at the University of KwaZulu-Natal (Temperature varying between 4 and 10 ºC) or a refrigerator at Durban Institute of Technology. Where appropriate, samples were coarse filtered through Whatman No. 1 filter paper and micro-filtered through 0.45 µm acetate filter cartridges on site to reduce biological activity during transport and storage. For VFA measurements, samples were acidified using concentrated HCl. Samples for unstable analytes were transported in a cooler box filled with ice or ice-bricks.

3 ANALYTICAL METHODS

Where possible all analyses were conducted according to Standard Methods (APHA, 1998).

3.1 COD

Influent and effluent total COD concentrations were measured by the open reflux method; filtered or soluble COD concentrations were obtained by filtering samples through 0.45µm acetate filters and using the titrimetric closed reflux COD method (APHA, 1998).

3.2 Alkalinity

Alkalinity was determined by potentiometric titration using HCl to an end-point pH value of 5.3. In 2004, alkalinity was determined using the 5-point titration method of Moosbrugger et al. (1992).

3.3 Volatile Fatty Acids

Two methods were employed to measure VFA in samples:

Method 1-HPLC: Small samples (5 mℓ) were obtained from the influent and compartments 1 to 4 inclusive, and filtered through 0.45µm acetate filter cartridges on-site. These samples were transported on ice. A sample volume of 1 mℓ was passed through solid phase extraction cation exchange cartridges to extract organic acids, and eluted with a sodium carbonate solution. Pretreated samples were analysed using high performance liquid chromatography for acetic, propionic, butyric, iso-butyric, valeric and iso-valeric acids.

Method 2-Titrimetric: VFA were determined as acetic acid in samples that were analysed titrimetrically for alkalinity according to the Moosbrugger method (see Alkalinity above).

3.4 Sulphate

Sulphate measurements were obtained on influent and effluent samples spectrophotometrically.

3.5 Phosphate

Phosphate measurements were obtained on influent and effluent samples spectrophotometrically.

3.6 Carbohydrate and protein concentrations

Total carbohydrates were measured according to the method of Dreywood (1946) with minor modifications (Raunkjaer et al., 1994) using an anthrone reagent. Protein estimation was carried out using the Lowry method (Lowry et al., 1951).

3.7 Enumeration of total coliforms and *Escherichia coli*

Total coliforms and *E. coli* were simultaneously determined by the membrane filtration technique according to Standard Methods (APHA, 1998). Coliforms were enumerated as colony forming units (cfu) per 100 mℓ.

Samples were diluted (1: 10 000) and 10 mℓ volumes were filtered through a gridded 0.45 µm membrane filter (Schleicher and Schuell). Sterile phosphate buffer dilutions were done as controls at the beginning and at the end of filtrations.

Filters were aseptically placed on Chromocult Coliform Agar (Merck), and incubated at 35ºC for 18 – 24 h.

E. coli colonies appeared as dark-blue to violet colonies and total coliforms appear as salmon to red colonies. The absence of growth in controls indicated the sterility of the dilution water and filtration apparatus.

3.8 Enumeration of coliphages

Virus identification and isolation is difficult and expensive, and beyond the scope of most laboratories. For this reason, coliphages are routinely used as viral indicators. This technique involves enumerating the bacteriophage of host culture *E. coli* (ATCC 13706) using the double layer technique. Bacteriophages cause lysis on a lawn of *E. coli* host cells, forming clear plaques and were enumerated as plaque forming units (pfu) per 100mℓ.

3.9 Enumeration of helminth eggs

This was limited to a single helminth genus, namely *Ascaris*. Raw wastewater (1ℓ) and effluent (10ℓ) was collected on a weekly basis and allowed to sediment for 18 h. The supernatant of samples were discarded and the remaining sediments were centrifuged at 1000 g for 15 min. The centrifuged supernatant was discarded and the enumeration of parasite eggs realised according to the modified Bailenger method (Ayres and Mara, 1996).

3.10 Scanning Electron Microscopy

Sludge samples from each compartment of the pilot-scale ABR were obtained during stable operation and prepared for SEM. Each sample was centrifuged for 5 min and the supernatant removed. Samples were washed three times in 0.1 M phosphate buffer at pH 7.2. Washed samples were decanted and fixed in 10% paraformaldehyde in 0.1M phosphate buffer. Samples were fixed for 16 h, decanted and washed three times with 0.1M phosphate buffer, and post-fixed with 1% osmium tetraoxide for 1 h at room temperature. Fixed samples were then repeatedly rinsed with distilled water to remove excess fixative, and dehydrated in a graded alcohol series (25, 50, 75 and 100%) of 10 min each. Samples were placed on Nucleopore filters (0.20 μm) and further dehydrated in a critical point drier (CPD).

Fixed samples were mounted on aluminium stubs, and sputter-coated with gold. The SEM graphs were taken on a Cleo 1450 instrument.

APPENDIX A2: ABR EFFLUENT IRRIGATION STUDY

Results from other sections of the project have indicated that ABR effluent, at its present microbiological quality, is not suitable for irrigation of food crops. However, the high nutrient levels of the effluent suggest that it holds potential as a fertilising solution, if the microbial quality can be improved. On-site sanitation in poor peri-urban communities could then be linked to initiatives to improve food security. A small-scale irrigation trial was therefore conducted, focussing on plant growth rather than on microbial quality (which was acknowledged at the outset as being unsuitable at this stage).

Vegetable crops selected were peppers, spinach and maize. Three irrigation treatments were evaluated: tap water (as negative control, containing no nutrients), a commercially available hydroponics nutrient solution (as positive control, containing a balance of all the nutrients required for optimum plant growth), and ABR effluent (treatment). Ten plants of similar size and vigour were assigned to each experimental group - i.e. 10 plants per crop type, per irrigation treatment, yielding 30 plants per crop type and 90 plants in total. Plants were watered once weekly with 500 mℓ of the respective irrigation treatments. In addition, all plants were watered with tap water every two days to prevent dehydration damage influencing growth. Growth was monitored over a period of 7 weeks. The growth measures monitored included height, number of leaves, stem diameter, number of fruits and fresh weight of fruit (where appropriate), mean leaf length and width.

To obtain approximate estimates of the microbiological quality of the crops at the end of the experimental period, all leaves of each plant were swabbed with sterile gauze swabs, using one swab for each plant. Each swab was placed into sterile physiological saline solution in sterile glass bottles, the contents of which were agitated, filtered through 0.45µm membrane filters and incubated on Chromocult coliform agar for 18 h at 37°C. Total coliforms and *E. Coli* colonies were counted.

Detailed results are not presented here since trends were similar throughout. Statistical significance of differences was rarely achieved, but there was a uniform trend with the negative control (tap water) yielding the poorest growth and the positive control (nutrient solution) yielding the best growth. Growth of plants irrigated with ABR effluent most closely resembled that of plants irrigated with nutrient solution, indicating that ABR effluent indeed appears to hold potential as a fertilising solution. To obtain statistically significant results, considerably larger numbers of plants would need to be grown over a longer time period, since results from field trials are known to show high variability. Should this be deemed desirable, additional resources would need to be committed to such a study.

The limited microbiological analyses conducted on the crops at the end of the experimental period showed high variability of microbial counts, with no significant differences or consistent trends among either the crops or the irrigation treatments. Total coliforms varied between 0 cfu/plant and approximately 10 000 cfu/plant, with geometric mean counts typically being several thousand cfu/plant. *E. coli* counts varied between 1 cfu/plant and approximately 100 cfu/plant, with varying geometric means.

The reasons for these uninformative microbial data are difficult to discern from such a limited trial. To obtain reliable data, a more extensive experiment would be needed, with more frequent sampling and more rigorous expression of the results (cfu/plant allows for a great deal of variability in the unit of expression, which may have contributed to the lack of clarity in the results). Furthermore, since irrigation water was applied to the soil rather than directly to the plant, a study of microbial quality of the soil before, during and at varying intervals after irrigation may be useful. However, it is questionable whether the investment of resources required to conduct rigorous microbial studies related to irrigation can be justified until the microbial quality of the ABR effluent can be improved to meet irrigation limits.

APPENDIX A3: CAPACITY BUILDING REPORT

The following was proposed for capacity building on the project:

While the focus of the project is technical, the outcome of the project is community upliftment and during the project there will be strong capacity building elements. The different partners each have specific roles to play in this regard. This project will provide for capacity building at Natal Technikon and ML Sultan Technikon by their involvement in microbial population characterisation and reactor design / operation respectively. At these institutions both staff and students will be employed on the project. Furthermore a black in-service trainee from ML Sultan Technikon will be recruited to monitor the plants, both at the sewage works and in the designated peri urban area. Ideally, the student should be familiar with the area of operation.

In total, 26 students have been involved in this project, of which two are black, 13 are Indian, and 11 are white. Seventeen are female.

At the beginning of this project the lead organisation, the Pollution Research Group was affiliated to the University of Natal, Durban (UND). At the beginning of 2004, the University of Natal and the University of Durban-Westville merged to form the University of KwaZulu-Natal (UKZN).

One PhD, one MSc Eng and one MTech degree have been awarded for research on this project. One further PhD, two further MSc Eng and two further MSc dissertations are still to be submitted at the writing of this report. Three postgraduate students from institutions other than UKZN / UND have been awarded degrees, where the research component has been undertaken in conjunction with this project. Two honours projects have been completed within the project. A further 6 UND and UKZN undergraduate, and 1 external undergraduate research project have also been completed.

Table A3. 1: Students and assistant researchers involved in K5/1248

Category	Personnel	Year	Race	Gender
Post graduate students (UND/UKZN)	Mrs J Bell (PhD student, graduated 2002)	2001	W	F
	Ms P Dama (MScEng student)	2001, 2002	I	F
	Mrs K Foxon (MScEng, PhD student)	2001, 2002, 2003	W	F
	Mr Z Mtembu (MScEng student, graduated 2006)	2002, 2003	B	M
	Mr S Pillay (MSc student)	2003, 2004, 2005	I	M
	Ms. N Arjun (MSc student)	2004, 2005	I	F
	Mr R Stone (MSc Eng Student)	2004, 2005	W	M
Post graduate students (Other)	Ms D Mueller (MSc student, WAREM Water and resource Management University, Stuttgard	2001	W	F
	Ms M. Ondracek (Tech. Uni. Denmark)	2002	W	F
	Mr S Wiwe (Tech Uni. Denmark)	2002	W	M
	Ms T Lalbahadur (Durban Institute of Tech., graduated 2005)	2003, 2004	I	F
	Ms K Hudson (University of Witwatersrand)	2003, 2004, 2005	W	F
Undergraduate students (UND)	Ms N Arjun (SLES, UND, Honours)	2003	I	F
	Mr JP Joubert (SLES, UND, Honours)	2003	W	M
	Ms N McKay (SLES, UND, Third year)	2003	W	F
	Mr A Smith (SLES, UND, Third year)	2003	W	M
	Ms T Badat (SLES, UKZN, Third year)	2004	I	F
	Ms V Moodley (SLES, UKZN Third year)	2004	I	F
	Ms K Arumugam (Chem Eng, UKZN, 4th year)	2004	I	F
	Ms H Khan (Chem Eng, UKZN, 4th year)	2004	I	F
Undergraduate students (Other)	Ms S Spagnol (INSA, Toulouse, France, 5th year)	2003	W	F
Research assistants	Mr K Govender	2001	I	M
	Ms D Moodley	2004	I	F
In-service trainees	Mr Z Mtembu	2001	B	M
Vacation	Mr M Moodley	2001	I	M

students	Ms D Adari	2001	I	F
	Mr D Mzulwini	2001	B	M

APPENDIX A4: TECHNOLOGY TRANSFER

This section contains a report on technology transfer items arising from this project. These include publications, international collaboration, local collaboration, technical visits and courses and workshops.

1 PUBLICATIONS

1.1 Journal Articles

Dama P., Bell J., Foxon K., Brouckaert C. Huang T., Buckley C. Naidoo V., and Stuckey D. (2002) Pilot-scale Study of an Anaerobic Baffled Reactor for the Treatment of Domestic Wastewater *Wat. Sci. Technol.* **46** (9), pp. 263-270

Foxon K.M., Brouckaert C.J., Remigi E. and Buckley C.A. (2004) The anaerobic baffled reactor: An appropriate technology for onsite sanitation. Water SA, **30**(5) pp.44-50 (special edition).

Lalbahadur T., Pillay S., Rodda N., Smith M., Buckley C. Holder, F., Bux F. and Foxon K. (2005) Microbiological Studies of an Anaerobic Baffled Reactor: Microbial Community Characterisation and deactivation of health-related indicator bacteria *Wat. Sci. Technol* (accepted)

1.2 Conference Proceedings

Dama P., Govender K., Huang T., Foxon K.M., Bell J., Brouckaert C.J., Buckley C.A, Naidoo, V. and Stuckey D. (2001). Flow Patterns in an Anaerobic Baffled Reactor. *Proceedings Part 1. 9th World Congress on Anaerobic Digestion*, Antwerp, Belgium, 2-5 September 2001, pp. 793 - 798.

Bell, J., Dama, P., Govender, K.M., Buckley, C.A. and Stuckey, D.C. (2001) Performance characterisation and microbial populations associated with the start-up of a laboratory-scale and a pilot-scale anaerobic baffled reactor. *Proceedings Part 2. 9th World Congress on Anaerobic Digestion*, Antwerp, Belgium, 2-5 September 2001, pp. 389-392 (poster).

Dama P, Bell J, Naidoo V, Foxon KM, Brouckaert CJ, Buckley CA and Stuckey D (2001). The Anaerobic Baffled Reactor for the Treatment of Domestic Wastewater in Dense Peri-Urban Communities. *Proceedings Part 2. 9th World Congress on Anaerobic Digestion*, Antwerp, Belgium, 2-5 September 2001, pp. 385-388 (poster).

Dama P., Bell J., Foxon K.M., Brouckaert C.J., Huang T, Buckley C.A., Naidoo V. and Stuckey D. (2001) Pilot-scale study of an anaerobic baffled reactor for the treatment of domestic wastewater. *Proceedings. IWA Conference on Water and Wastewater Management in Developing Countries*, Kuala Lumpur, Malaysia, 29-31 October 2001.

Dama, P., Bell J., Naidoo, V., Foxon, K.M., Brouckaert, C.J., Buckley, C.A. and Stuckey, D. (2001) The anaerobic baffled reactor for the treatment of domestic wastewater in dense peri-urban communities. *Proceedings. IWA Conference on Water and*

Wastewater Management in Developing Countries, Kuala Lumpur, Malaysia, 29-31 October 2001.

Foxon K.M., Dama P., Bell J. and Buckley C.A. (2001) Design considerations for the implementation of an anaerobic baffled reactor in low-income settlements in KwaZulu-Natal. *Proceedings. IWA Conference on Water and Wastewater Management in Developing Countries*, Kuala Lumpur, Malaysia, 29-31 October 2001.

Dama P., Bell J., Foxon K.M., Brouckaert C.J., Huang T., Buckley C.A., Naidoo V. and Stuckey D. (2002) Pilot-scale study of an anaerobic baffled reactor for the treatment of domestic wastewater. *Proceedings. 2002 Water Institute of Southern Africa (WISA) Biennial Conference and Exhibition*, Durban, South Africa, 19-23 May 2002 (poster).

Foxon, K.M., Dama, P. and Buckley C.A. (2002). Application of aquatic modelling of the design of an anaerobic baffled reactor for peri-urban sanitation. *Proceedings. 2002 Water Institute of Southern Africa (WISA) Biennial Conference and Exhibition*, Durban, South Africa, 19-23 May 2002 (poster).

Foxon, K.M., Brouckaert, C.J. and Buckley, C.A. (2003) The application of an Anaerobic Baffled Reactor (ABR) to sanitation in peri-urban communities. *Proceedings. WWWest 2003 meeting of WEST Scientific Support Centres,* 14 - 16 April 2003.

N. Arjun, M. Smith, N. Rodda and C. Buckley (2003) Determination of quality of drinking water from standpipes and ground tanks in the Cato Manor peri-urban settlement 16th SASM-KZN Symposium, Durban 2003

S. Pillay, N. Rodda, M. Smith, K. Foxon, C. Buckley (2003) Pathogen Deactivation and Microbial Community Analysis of an Anaerobic Baffled Reactor. *Proceedings 16th Annual Symposium of the South African Society for Microbiology –KZN*, 2003.

Arjun, N., Smith, M., Rodda, N. and Buckley, C.A. (2003) Water quality in peri-urban communities: interaction of water supply devices and water use behaviours for users supplied by communal standpipes and household ground tanks in Cato Manor (Durban). *Proceedings. International Water Association Symposium on Health-Related Water Microbiology*, Cape Town, South Africa, 14-19 September 2003. (poster).

Mackay, N., Smith, A., Rodda, N. and Buckley, C.A. (2003) Feasibility study of the use of effluent from an Anaerobic Baffled Reactor (ABR) for irrigation in a peri-urban community (Cato Manor, Durban) *Proceedings. International Water Association Symposium on Health-Related Water Microbiology*, Cape Town, South Africa, 14-19 September 2003. (poster).

Lalbahadur T., Pillay S., Rodda N., Smith M., Buckley C. Holder, F., Bux F. and Foxon K. (2004) Microbiological Studies of an Anaerobic Baffled Reactor : Microbial Community Characterisation Deactivation of Health-related Indicator Bacteria *Proceedings*: 1st International Conference on Onsite Wastewater Treatment and Recycle, Esplanade Hotel Fremantle, Western Australia, 11-13 February 2004.

Foxon K., Mtembu D., Pillay S., Rodda N., Smith M. and Buckley C. (2004) The Anaerobic Baffled Reactor (ABR): An Appropriate Technology for On-site Sanitation

Proceedings: 1st International Conference on Onsite Wastewater Treatment and Recycle, Esplanade Hotel Fremantle, Western Australia, 11-13 February 2004

Foxon, K.M. Brouckaert, C.J. Remigi, E.U. and Buckley, C.A. (2004) Biochemical modelling of the anaerobic baffled reactor. *Proceedings. 2004 Water Institute of Southern Africa (WISA) Biennial Conference and Exhibition*, Cape Town, South Africa, 2-6 May 2004.

Foxon K.M., Brouckaert C.J., Remigi E. and Buckley C.A. (2004) The anaerobic baffled reactor: An appropriate technology for onsite sanitation. *Proceedings. 2004 Water Institute of Southern Africa (WISA) Biennial Conference and Exhibition*, Cape Town, South Africa, 2-6 May 2004.

Arjun N., Joubert J-P, Smith M., Rodda N. and Buckley C. (2004) Water Quality at Point for Users Supplied by Standpipes and Groundtanks in a Peri-urban Community *Proceedings: Sp*ecialist Conference on Water and Wastewater Management for Developing Countries (WAMDEC 2004) : Innovative And Sustainable Approaches For Serving The Poor, Elephant Hills Hotel, Victoria Falls, Zimbabwe, 28-30 July 2004

Pillay S., Foxon K., Smith M., Rodda N., and Buckley C. (2004) Treatment performance and effluent quality from an anaerobic baffled reactor treating domestic wastewaters : Implications for water reuse *Proceedings: Sp*ecialist Conference on Water and Wastewater Management for Developing Countries (WAMDEC 2004) : Innovative And Sustainable Approaches For Serving The Poor, Elephant Hills Hotel, Victoria Falls, Zimbabwe, 28-30 July 2004

Remigi, E.U. and Foxon K.M. (2004) Modelling of anaerobic digestion processes: A review. *Proceedings. Research and Innovation in Wastewater Treatment*. Workshop dedicated to the memory of Prof. Alberto Rozzi. Milan – Bologna, 24 – 26 November 2004, Politecnico di Milano, pp. 92 – 112.

Pillay S, Foxon K., Rodda N., Smith M. and Buckley C., The use of effluent from an anaerobic baffled reactor (ABR) for irrigation in a peri-urban community. *Proceedings*: Third International Conference on Ecological Sanitation, International Convention Centre, Durban, South Africa, 23-27 May 2005. (Paper)

Foxon K. and Buckley C., The ABR Concept: A holistic approach to managing wastewater from low-income communities and institutions. *Proceedings*: Third International Conference on Ecological Sanitation, International Convention Centre, Durban, South Africa, 23-27 May 2005. (Paper)

Joubert J., Arjun N., Rodda N., Smith M. and Buckley C. Water use practices and point-of-use water quality for users supplied with drinking water by standpipes or groundtanks. *Proceedings*: Third International Conference on Ecological Sanitation, International Convention Centre, Durban, South Africa, 23-27 May 2005. (Poster)

Arjun N., Foxon K., Smith M., Rodda N., and Buckley C. (2005) Characterisation of the wastewater produced by a developing community. *Proceedings*: Third International Conference on Ecological Sanitation, International Convention Centre, Durban, South Africa, 23-27 May 2005. (Poster)

Foxon, K.M., Remigi, E.U., Pillay, S., Brouckaert, C.J., Rodda, N, Pfaff, W. and Buckley, C.A. (2005) Management of sanitation residues. *Proceedings. IWA / WISA Conference: Management of Residues Emanating from Water and Wastewater Treatment*. Johannesburg, South Africa, 9 to 12 August 2005.

1.3 Theses

Bell J. (2002) *Treatment of dye wastewaters in the anaerobic baffled reactor and characterisation of the associated microbial populations* PhD Thesis, University of Natal. (Work in this thesis formed part of WRC project K5/853)

Mtembu Z. (2005) *The anaerobic baffled reactor for sanitation in dense peri-urban settlements.* MScEng Thesis, University of KwaZulu-Natal

Lalbahadur T. (2004) *Characterisation of the microbial communities present in an anaerobic baffled reactor utilising molecular techniques*. MTech Thesis, Durban Institute of Technology

2 INTERNATIONAL COLLABORATION

This project served as the basis for technical and academic collaboration between the Pollution Research Group, University of Natal / KwaZulu-Natal and international universities.

2.1 Collaboration with Gent University and HEMMIS

A research project entitled *Modelling and control of biological systems - Application to the Anaerobic Baffled Reactor (ABR)* was funded by the National Research Foundation (NRF) and the Ministry of the Flemish Community, Science Innovation and Media Department (Flanders-RSA Cooperative Research Agreement) The project leaders: were Prof. Chris Buckley (Pollution Research Group), Prof. George Ekama (University of Cape Town) and Dr. Ir. Peter Vanrolleghem (Gent University). This project ran from 2002 to December 2005 and was concerned with biochemical modelling of the ABR.

In January 2003, two researchers, Mr. Chris Brouckaert from Pollution Research Group and Mr. Sven Sötemann from University of Cape Town visited Gent University for a course on the application of the WEST software in modelling of wastewater systems. In April 2003, Mr. Chris Brouckaert returned to Gent to present the preliminary results of the ABR model to the 2003 WWWest meeting (see section 1.2, Foxon et al., 2003). In October 2003, a PhD student from Gent, Mr. Stijn van Hulle visited University of Cape Town and Pollution Research Group to share his expertise in the use of WEST and general biochemical modelling techniques. Mrs. K. Foxon travelled to Ghent University, Belgium in June/July 2004 to further develop skills in biological modelling and parameter identification techniques. Dr Usama Zaher from Gent University visited Durban in August 2005 to continue work on anaerobic digestion modelling.

2.2 Collaboration with Norwegian Technical University (NTNU)

In April 2003, Prof. Tor-Ove Leiknes from the Department of Hydraulic and Environmental Engineering visited the Pollution Research Group to discuss possible

collaboration of membrane bioreactor projects. The outcome of this meeting was a joint proposal to the European Commission 6th Framework Programme on Research, Technological Development and Demonstration for Specific Targeted Research or Innovation Projects (STREP) on development of hybrid membrane bioreactor processes for wastewater reuse through novel and enhanced treatment technologies, for which membrane filters for the ABR is to form a part. The proposal was accepted in April 2005 and will be funded from 2006.

2.3 Collaboration with Politecnico di Milano, Italy

In August 2001, Priyal Dama and Valerie Naidoo from Pollution Research Group visited the research facilities of Professor Rozzi of Politecnico di Milano in Italy from 27 to 31 August 2001 where they were shown the use of the latest ANITA biosensor built by Dr Rozzi's research group. They were also given software for the ANITA biosensor, which was then being used by Pollution Research Group to determine anaerobic activity in the various compartments of the anaerobic baffled reactor. Dr Enrico Remigi from Politecnico di Milano visited Pollution Research Group in 2002 to provide expertise in the use of the ANITA biosensor, and was subsequently awarded a post-doctoral position at the University of KwaZulu-Natal until September 2005. Dr. Remigi has contributed significantly to the Pollution Research Group's expertise in anaerobic digestion experimentation and modelling, has written several scientific papers, and has been involved in the supervision of post-graduate students at the University of KwaZulu-Natal.

3 LOCAL COLLABORATION

3.1 Centre for Water and Wastewater Research, DIT.

A Durban Institute of Technology (DIT) MTech student, Ms. Tharnija Lalbahadur studied the microbial characterisation of ABR sludge and population dynamics between compartments under the direction of Mr. Faizal Bux and Ms. Francisca Holder of Centre for Water and Wastewater Research, DIT. Ms. Lalbahadur graduated in 2004.

3.2 eThekwini Wastewater

eThekwini Wastewater have been actively involved in directing this research and have assisted by housing the pilot ABR, supplying WWTP data and performing some of the chemical and microbial analyses in the project.

3.3 Vela VKE Consulting

The project team was involved in transferring skills relating to the design and monitoring of ABRs to Vela VKE consulting.

4 TECHNICAL VISITS

4.1 ENEA, Bologna, Italy

While in Europe to attend the 2001 Anaerobic Digestion conference and visit Prof. Rozzi's group in Milan Valerie Naidoo and Priyal Dama of the Pollution Research Group

also visited the Energy and Environment Division of the ENEA Institute in Bologna, Italy and had the opportunity to observe the full-scale ANANOX system in operation. This system uses an anaerobic baffled reactor as the first biological treatment process and couples it to an anoxic reactor and a sludge trap.

5 COURSES AND WORKSHOPS

5.1 Aquatic modelling in postgraduate and undergraduate courses

Experience gained from the aquatic modelling exercise and the use of the AQUASIM software has been incorporated into a number of postgraduate modules taught by the Pollution Research Group. These include Process Principles for Environmental Engineers, Biological Wastewater Treatment and Applied Biochemical Engineering.

5.2 WISA conference workshops

Workshops at the 2002 and 2004 Biennial Water Institute of Southern Africa (WISA) conferences in Durban and Cape Town were presented on Aquatic Modelling. In both of these courses, representatives of Hemmis, Belgium were invited to present a section on the use of the WEST® modelling platform in simulation and control of wastewater treatment processes.